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List of Abbreviations and Acronyms

ACZA	ammoniacal copper zinc arsenate
ALC	aquatic life criteria
BLM	Biotic Ligand Model
BMP	best management practice
CCA	chromated copper arsenate
CCC	criterion chronic concentration
CEQ	Council on Environmental Quality
CFR	Code of Federal Regulations
CMC	criterion maximum concentration
Corps, the	U.S. Army Corps of Engineers
DNA	deoxyribonucleic acid
Ecology	Washington State Department of Ecology
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
GIS	geographic information system
HCP	Habitat Conservation Plan
HPA	Hydraulic Project Approval
HPDD	High-pressure directional drilling
HPMS	Hydraulic Project Management System
ITP	Incidental Take Permit
LWD	large woody debris
MHHW	mean higher high water
MLLW	mean lower low water
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
OHWL	ordinary high water line
PAH	polycyclic aromatic hydrocarbon
PAR	photosynthetically active radiation
PEC	probable effects concentration
RCW	Revised Code of Washington
RMS	root mean square
SEL	sound exposure level
SSC	suspended sediment concentration
SWD	small woody debris
TEC	threshold effects concentration
TRA	Tidal Reference Area
TSS	total suspended solids
USACE	U.S. Army Corps of Engineers
USC	United States Code
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
WAC	Washington Administrative Code
WDFW	Washington Department of Fish and Wildlife
WDNR	Washington Department of Natural Resources
WRIA	Water Resource Inventory Area
WSDOT	Washington State Department of Transportation
USFWS	U.S. Fish and Wildlife Service

List of Units of Measure

C	Celsius
cfs	cubic feet per second
cm	centimeter
cm/sec	centimeters per second
dB	decibels
dB_{peak}	peak decibels during each pulse (either maximum or minimum)
dB_{SEL}	decibels sound exposure level
dB_{RMS}	decibels root mean square – square root of sound energy divided by impulse duration
F	Fahrenheit
Hz	hertz
JTU	Jackson turbidity unit
m	meter
m^3	cubic meter
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
mm	millimeter
mm/hr	millimeters per hour
NTU	nephelometric turbidity unit
ppb	parts per billion
$\mu g/cm^2/mm$	micrograms per square centimeter per millimeter
µg/L	micrograms per liter
$\mu M/m^2/sec$	micro-moles per square meter per second

Note: In general, English measurement units (e.g., feet, inches, miles) are used in this white paper; when the source material expresses a value in metric units, that measurement is also provided in parentheses. However, measurements that by convention are typically made only in metric units are reported in those units (e.g., mg/L, $\mu M/m^2/sec$). Temperatures are reported in both Fahrenheit and Celsius, regardless of the scale used in the source material.

Executive Summary

The Revised Code of Washington (RCW) directs the Washington Department of Fish and Wildlife (WDFW) to “preserve, protect, perpetuate, and manage” the fish and wildlife species of the state as its paramount responsibility (RCW 77.04.012). Under RCW 77.55, any construction or work that uses, diverts, obstructs, or changes the natural bed or flow of state waters requires a Hydraulic Project Approval (HPA) issued by WDFW. The purpose of the HPA program is to ensure that hydraulic projects are completed in a manner that prevents damage to public fish and shellfish resources and their habitats. To ensure that the HPA program complies with the Endangered Species Act (ESA), WDFW is developing a programmatic multispecies Habitat Conservation Plan (HCP) to obtain Incidental Take Permits from the U.S. Fish and Wildlife Service (USFWS) and the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service (also known as NOAA Fisheries), in accordance with Section 10 of the ESA. For WDFW, the objective is to avoid and/or minimize the incidental take of those aquatic species potentially considered for coverage under the HCP (referred to in this white paper as “HCP species”) resulting from activities conducted under an HPA.

The HCP will address the impacts, potential for take, and mitigation measures for effects on HCP species from hydraulic projects that require HPAs. WDFW’s intent is to build the scientific foundation for the effort to prepare an HCP for hydraulic projects that receive HPAs. To accomplish this, WDFW is compiling the best available scientific information related to the impacts, potential for incidental “take” of species that may be covered in the HCP (as defined in the ESA), adequacy of existing rules (Washington Administrative Code [WAC] 220-110), and possible management directives and mitigation measures to avoid and/or minimize potential take to the maximum extent practicable. As the HPA authority covers all waters of the state, this white paper considers hydraulic project impacts in both freshwater and marine environments.

The objectives of this white paper are:

- To compile and synthesize the best available scientific information related to the potential human impacts on HCP species, their habitats, and associated ecological processes resulting from the construction, maintenance, repair, replacement, modification, and removal of HPA-permitted projects.
- To use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely to result from the construction and repair of HPA-permitted projects.
- To assess the extent to which current HPA rules address the potential impacts on covered species, their habitats, and ecological processes.
- To identify appropriate and practicable measures, including policy directives, conservation measures, and best management practices (BMPs), to avoid, minimize, or mitigate the risk of incidental take of HCP species.

This white paper is a consolidation of a suite of white papers prepared to establish the scientific basis for the HCP and assist WDFW decision-making on what specific HPA activities should be covered by the HCP. The original white papers covered the following activities:

- Water crossings (bridges, culverts, conduits)
- Fish passage (fish ladders, culverts, weirs, roughened channels, trap and haul)
- Flow control structures (dams, weirs, dikes, levees, tide gates, intakes, outfalls)
- Bank protection/stabilization (bulkheads, retaining walls, revetments, toe protection, beach nourishment, subsurface drainage, biotechnical bank protection, bank reshaping or regrading, soil reinforcement, coir and straw logs, integrated approaches)
- Shoreline modifications (groins, jetties, breakwaters)
- Channel modifications (dredging, gravel mining and bar scalping, sediment capping, channel creation and alignment.)
- Habitat modification (beaver dam removal, large woody debris manipulations, spawning substrate augmentation, riparian planting, wetland creation/restoration, enhancement, beach nourishment/contouring, reef creation, eelgrass planting/restoration/enhancement, in-channel and off-channel habitat modifications
- Overwater structures (docks, floats, piers, ramps, wharfs, pilings and non-structural pilings)
- Marinas and Terminals
- Fish screens (in-channel, off channel).

The literature review conducted for the original white papers identified seven mechanisms of impact that could potentially affect the HCP species. These mechanisms of impact have direct and indirect effects that can be temporary, short-term effects or permanent, long-term effects. The mechanisms of impact are:

- Construction and maintenance activities
- Facility operation and vessel activity
- Hydraulic and geomorphic modifications
- Water quality modifications
- Riparian vegetation modifications
- Aquatic vegetation modifications
- Ecosystem fragmentation.

Key elements of the white paper are to:

- Specify objectives. (Section 2).
- Identify methods used to find the pertinent literature. (Section 3).
- Describe the potentially covered activities in detail (Section 4).
- Identify the distribution of the 52 HCP species (i.e., whether they use fresh water, marine water, or both) and their habitat requirements (Section 5).
- Present a conceptual framework for assessing impacts (Section 6).

- Discuss the potential direct and indirect impacts on the HCP species and their habitats due to exposure to the mechanisms of impact (Section 7).
- Identify cumulative impacts (Section 8).
- Based on the distribution information, identify the risk of "take" associated with each of these impacts mechanisms (Section 9).
- Identify data gaps (Section 10).
- Identify habitat protection, conservation, and mitigation strategies that could avoid or minimize the identified potential impacts (Section 11).

1 Introduction

The Revised Code of Washington (RCW) directs the Washington Department of Fish and Wildlife (WDFW) to “preserve, protect, perpetuate, and manage” the fish and wildlife species of the state as its paramount responsibility (RCW 77.04.012). Under RCW 77.55, any construction or work that uses, diverts, obstructs, or changes the natural bed or flow of state waters requires a Hydraulic Project Approval (HPA) issued by WDFW. The purpose of the HPA program is to ensure that these activities are completed in a manner that prevents damage to public fish and shellfish resources and their habitats.

Because several fish species in the state are listed as threatened or endangered under the federal Endangered Species Act (ESA), many of the activities requiring an HPA may also require approvals from the National Oceanic and Atmospheric Administration Fisheries Service (known as NOAA Fisheries) and the U.S. Fish and Wildlife Service (USFWS) (collectively known as the Services). Such approvals can be in the form of an ESA Section 7 Incidental Take Statement or an ESA Section 10 Incidental Take Permit (ITP). As authorized in Section 10, ITPs may be issued for otherwise lawful activities that could result in the “take” of ESA-listed species or their habitats. To “take” means to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect or to attempt to engage in any such conduct (16 United States Code 1532(19)).

WDFW is developing a programmatic multispecies Habitat Conservation Plan (HCP) to obtain a Section 10 ITP. The ITP will ensure that the HPA program complies with the ESA, and will facilitate ESA compliance for citizens conducting work under an HPA.

For WDFW, the benefits of an HCP and ITP are to contribute to the long-term conservation of both listed and unlisted species through the minimization and mitigation of impacts on those species and their habitats, while ensuring that WDFW can legally proceed with the issuance of HPAs that might otherwise result in the incidental “take” of ESA-listed species.

The HCP will identify the impacts of HPA-permitted projects on those aquatic species considered for coverage, the potential for take, and conservation measures for avoiding, minimizing, and mitigating, to the maximum extent practicable, the impacts of the permitted take on the potentially covered species. The Services must find in their biological opinion that any permitted incidental take will not jeopardize the continued existence of the species, ore result in the destruction or adverse modification of designated critical habitat (i.e., the taking will not appreciably reduce the likelihood of survival and recovery of the species in the wild), before they can issue ITPs.

In 2006 – 2008, WDFW worked with contractors to develop a suite of white papers. The white papers compile the best available scientific information for the potential effects of up to 21 types of HPA projects on 52 potentially covered species of fish and shellfish. This compilation includes information from white papers developed in 2006 and 2007. Material developed in 2008 has not yet been peer reviewed (as of April 2009) and is not included here.

The white papers consider hydraulic project impacts in both freshwater and marine environments. Species considered for coverage under the HCP (referred to in this white paper as "HCP species") are listed in Table 1-1. WDFW intends to apply this best available scientific information to protect these species during all of the phases of projects that require a hydraulic permit, including construction, maintenance, repair, operation, replacement, modification, and removal.

In addition to establishing the scientific basis for the HCP, the white papers describe potential take mechanisms. They identify what avoidance, minimization and mitigation measures could address the potential effects of hydraulic projects. They are intended to assist WDFW decision-making regarding what specific HPA activities should be covered by the HCP.

Table 1-1. The 52 HCP species addressed in this white paper.

Common Name	Scientific Name	Status[a]	Habitat
Chinook salmon	*Oncorhynchus tshawytscha*	FE/FT/SC	Freshwater, Estuarine, Marine
Coho salmon	*Oncorhynchus kisutch*	FT/FSC	Freshwater, Estuarine, Marine
Chum salmon	*Oncorhynchus keta*	FT/SC	Freshwater, Estuarine, Marine
Pink salmon	*Oncorhynchus gorbuscha*	SPHS	Freshwater, Estuarine, Marine
Sockeye salmon	*Oncorhynchus nerka*	FE/FT/SC	Freshwater, Estuarine, Marine
Steelhead	*Oncorhynchus mykiss*	FE/FT/SC	Freshwater, Estuarine, Marine
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	FSC	Freshwater, Estuarine, Marine
Redband trout	*Oncorhynchus mykiss*	FSC	Freshwater
Westslope cutthroat trout	*Oncorhynchus clarki lewisii*	FSC	Freshwater
Bull trout	*Salvelinus confluentus*	FT/SC	Freshwater, Estuarine
Dolly Varden	*Salvelinus malma*	FP	Freshwater, Estuarine
Pygmy whitefish	*Prosopiim coulteri*	FSC/SS	Freshwater
Olympic mudminnow	*Novumbra hubbsi*	SS	Freshwater
Lake chub	*Couesius plumbeus*	SC	Freshwater
Leopard dace	*Rhinichthys falcatus*	SC	Freshwater
Margined sculpin	*Cottus marginatus*	FSC/SS	Freshwater
Mountain sucker	*Catostomus platyrynchus*	SC	Freshwater
Umatilla dace	*Rhinichthys umatilla*	SC	Freshwater
Pacific lamprey	*Lampetra tridentata*	FSC	Freshwater, Estuarine, Marine
River lamprey	*Lampetra ayresi*	FSC/SC	Freshwater, Estuarine, Marine
Western brook lamprey	*Lampetra richardsoni*	FSC	Freshwater
Green sturgeon	*Acipenser medirostris*	FT/FSC/SPHS	Freshwater, Estuarine, Marine
White sturgeon	*Acipenser transmontanus*	SPHS	Freshwater, Estuarine, Marine
Eulachon	*Thaleichthys pacificus*	FC/SC	Freshwater, Estuarine, Marine
Longfin smelt	*Spirinchus thaleichthys*	SPHS	Freshwater, Estuarine, Marine
Pacific sand lance	*Ammodytes hexapterus*	SPHS	Marine & Estuarine

Common Name	Scientific Name	Status[a]	Habitat
Surf smelt	*Hypomesus pretiosus*	SPHS	Marine & Estuarine
Pacific herring	*Clupea harengus pallasi*	FC/SC	Marine & Estuarine
Lingcod	*Ophiodon elongatus*	SPHS	Marine & Estuarine
Pacific cod	*Gadus macrocephalus*	FSC/SC	Marine (occ. Estuarine)
Pacific hake	*Merluccius productus*	FSC/SC	Marine & Estuarine
Walleye pollock	*Theragra chalcogramma*	FSC/SC	Marine (occ. Estuarine)
Black rockfish	*Sebastes melanops*	SC	Marine & Estuarine
Bocaccio rockfish	*Sebastes paucispinis*	SC	Marine & Estuarine
Brown rockfish	*Sebastes auriculatus*	SC	Marine & Estuarine
Canary rockfish	*Sebastes pinniger*	SC	Marine & Estuarine
China rockfish	*Sebastes nebulosis*	SC	Marine & Estuarine
Copper rockfish	*Sebastes caurinus*	FSC/SC	Marine & Estuarine
Greenstriped rockfish	*Sebastes elongates*	SC	Marine & Estuarine
Quillback rockfish	*Sebastes maliger*	FSC/SC	Marine & Estuarine
Redstripe rockfish	*Sebastes proriger*	SC	Marine & Estuarine
Tiger rockfish	*Sebastes nigrocinctus*	SC	Marine & Estuarine
Widow rockfish	*Sebastes entomelas*	SC	Marine & Estuarine
Yelloweye rockfish	*Sebastes ruberrimus*	SC	Marine & Estuarine
Yellowtail rockfish	*Sebastes flavidus*	SC	Marine & Estuarine
Olympia oyster	*Ostrea lurida*	SPHS	Marine & Estuarine
Northern abalone	*Haliotis kamtschatkana*	FSC/SC	Marine
Newcomb’s littorine snail	*Algamorda subrotundata*	FSC/SC	Marine
Giant Columbia River limpet	*Fisherola nuttalli*	SC	Freshwater
Great Columbia River spire snail	*Fluminicola columbiana*	FSC/SC	Freshwater
California floater (mussel)	*Anodonta californiensis*	FSC/SC	Freshwater
Western ridged mussel	*Gonidea angulata*	None	Freshwater

Notes: For the purpose of this white paper, some of the HCP species have been grouped when appropriate (each group is separated by a gray line).

[a] Status:

FE=Federal Endangered	FSC = Federal Species of Concern
FP=Federal Proposed	SC = State Candidate
FT = Federal Threatened	SS = State Sensitive
FC = Federal Candidate	SPHS = State Priority Habitat Species

2 Objectives

The objectives of the white papers are:

- To compile and synthesize the best available scientific information related to the potential human impacts on HCP species, their habitats, and associated ecological processes resulting from the construction, maintenance, repair, operation, replacement, modification, or removal of HPA-permitted activities.
- To use this scientific information to estimate the circumstances, mechanisms, and risks of incidental take potentially or likely resulting from HPA-permitted activities.
- To identify appropriate and practicable measures, including policy directives, conservation measures, and best management practices (BMPs), to avoid and/or minimize the risks of incidental take of HCP species.

3 Methods

White papers that are included in this compilation were written over the course of two years by several authors at four consulting firms. The white papers are:

> Draft Bank Protection/Stabilization White Paper, prepared for Washington Department of Fish and Wildlife by Anchor Environmental, L.L.C. in association with R2 Resource Consultants and Jones & Stokes Associates (2006).
>
> Overwater Structures and Non-structural Piling White Paper, prepared for Washington Department of Fish and Wildlife by Jones and Stokes Associates, in association with Anchor Environmental, L.L.C. and R2 Resource Consultants (2006).
>
> Water Crossings White Paper, prepared for Washington Department of Fish and Wildlife by Jones & Stokes Associates, in association with Anchor Environmental, L.L.C. and R2 Resource Consultants (2006).
>
> Shoreline Modifications White Paper, prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc. (2007).
>
> Marinas and Shipping/Ferry Terminals White Paper, prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc. (2007).
>
> Fish Passage White Paper, prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc. in consultation with Kozmo Ken Bates (Working draft 2008, not to be cited).
>
> Fish Screens White Paper, prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc. (Working draft March 2008, not to be cited).
>
> Channel Modifications White Paper, prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc. (2007).
>
> Flow Control Structures White paper, prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc. (Working draft December 2007, not to be cited).
>
> Habitat Modifications White Paper, prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc. (Working draft December 2007).

One additional white paper was prepared in 2006: Small Scale Mineral Prospecting, prepared for Washington Department of Fish and Wildlife by R2 Resource Consultants, in association with Anchor Environmental LLC (2006). This paper was not included in this compilation, because WDFW is undertaking revisions to the mineral prospecting rules separately. However, WDFW expects to incorporate changes from that rulemaking into the hydraulic project approval Habitat Conservation Plan.

Each of the white papers prepared in 2006 and 2007 included a description of the methods used to complete the white papers. The level of detail included in each method section varied, though more specific information was provided for papers prepared in 2007.

The methods section for white papers prepared in 2006 (Overwater Structures and Non-Structural Piling, Bank Protection/Stabilization, and Water Crossings) identified five principle tasks that were performed in preparing the white papers:

1. Existing WDFW rules and guidance were reviewed to identify current knowledge and practices relevant to the analysis of the impacts to potentially covered species associated with HCP-permitted activities.

2. A literature review was conducted to compile information reflecting the current state of knowledge regarding potential impacts associated with HCP-permitted activities and the potential to affect potentially covered species.

3. The compiled documents were reviewed to determine which potential pathways of impact were addressed in each document. The vast majority of collected documents considered impacts to salmonids or to physical habitat features, although documents that identified impacts to potentially covered species and their habitats other than salmonids were also identified and evaluated during the literature review.

4. Impact mechanism analyses were prepared for each of the principal impact pathways and for each principal type of HCP-permitted activity.

5. A draft version of this white paper was prepared and reviewed by technical specialists on the consultant team, then submitted to WDFW for comments. The white paper was amended based on the comments provided by WDFW and the white paper was finalized.

The methods section of white papers prepared in 2007 (Marinas and Terminals, Shoreline Modifications, Fish Passage, Fish Screens, Habitat Modifications, Channel Modifications, and Flow Control Structures) stated that information presented in these white papers is based primarily on the compilation and synthesis of the best available scientific information related to human impacts on HCP species, their habitats, and associated ecological processes. The methods used included the acquisition of existing literature, followed by an analysis of impacts based on a review of the literature.

In addition, the methods section of the marinas and terminals white paper specifically stated that best professional judgment was used to draw inferences from other pertinent, similar, or related studies and data sources where specific information was lacking.

WDFW staff subsequently organized, condensed, and edited the information from white papers prepared in 2006 and 2007, and included information resulting from peer reviews.

Each of the white papers listed databases and/or specific references that were consulted. The following tables compile that information.

Table 3-1: Resources used to develop white papers in 2006

Information Source	**2006 white paper**		
	Bank protection	overwater structures	water crossings
Relevant previous white papers prepared for WDFW (exact titles unspecified)	x	x	x
Relevant Washington Administrative Codes (WAC) (Sections unspecified in Methods, but discussed elsewhere in papers)	x		
Integrated Streambank Protection Guidelines (Cramer et al., 2003)	x		x
Stream Habitat Restoration Guidelines (Saldi-Caromile et al., 2004)	x		x
Alternative Mitigation Policy Guidance Interagency Implementation Agreement (Ecology, 2000).	x		
Design of Road Culverts for Fish Passage (Bates 2003)			x
Copies of HPAs provided by WDFW (citations unspecified in Methods)	x	x	x
Biological opinions prepared by NOAA Fisheries and USFWS, addressing various (unspecified) projects in Washington and Oregon	x	x	x
Keyword search of BIOSYS database	x		
Keyword search of Agricola database	x		
Internet (unspecified resources)	x	x	x
Google Scholar® searches	x		
Google searches	x	x	x
Other literature databases (not specifically identified in Methods)		x	x

The methodology sections for the 2006 white papers provided some additional detail:

- The principal keyword search strategy was to look for documents linking terms describing the species (i.e., common and scientific names of potentially covered species) with terms describing HPA-permitted structures or pathways of impact associated with the construction and presence of such structures.

- Additionally, some documents were identified by reviewing the bibliographies contained in documents identified through the preceding searches.

- Documents located during the literature review were in turn used in Internet searches (mostly conducted using the Google® search tool) to locate additional relevant literature addressing specific impact pathways.

Table 3-2: Resources used to develop Channel Modifications, Fish Passage, Fish Screen, Flow Control and Shoreline Modification white papers in 2007

Database	2007 White Paper					
	channel modifications	fish passage	fish screens	flow control	Habitat modifications	shoreline modifications
Thomson Scientific Web of Science (2007) • has electronic access to more than 8,500 scientific journals encompassing all fields of environmental science. • yielded several hundred relevant publications, most published within the last 10 years.	x	x	x	x	x	x
Thomson Scientific Web of Science for individual species • A keyword search of the scientific name and/or common name for each species in Table 1-1 was conducted. • For those species where the search returned more than 1,000 references, a few recent citations were selected for inclusion. Species in this category were the five salmon species (sockeye, chum, pink, coho, and Chinook), steelhead, and coastal cutthroat trout. • For the remaining species, every reference in the search result was reviewed for the relevance of species-specific information to be included in this white paper. • For several species, searches for scientific names and common names returned no references. These species included the margined sculpin, giant Columbia River limpet, great Columbia spire snail, western ridged mussel, river lamprey, longfin smelt, Newcomb's littorine snail, and many of the rockfish species.		x	x	x		
Previous white papers (exact titles unspecified)	x	x	x	x	x	x
Puget Sound-Georgia Basin Research Conferences conference proceedings (2001, 2003, 2005, 2007)	x	x	x	x	x	x
Summit system of libraries searched for theses. • Summit is a library catalog that combines information from Pacific Northwest academic libraries, including the Orbis and Cascade systems, into a single database available at URL = http://summit.orbiscascade.org/	x	x	x	x	x	x
University of Washington School of Aquatic and Fisheries Sciences, Fisheries Research Institute Reports (UW-FRI) database	x	x	x	x	x	x

• includes more than 500 report pertaining to research conducted by Fisheries Research Institute personnel from 1973 to the present.						
Personal collections of Herrera's (the consulting firm that prepared these white papers) staff including "consultant reports, textbooks, etc."	x	x	x	x	x	x

Table 3-3: Resources used to develop Marinas and Terminals white paper in 2007

Database
UW Library catalog • available at http://catalog.lib.washington.edu/search~/
University of Washington School of Aquatic and Fisheries Sciences, Fisheries Research Institute Reports (UW-FRI) database • includes more than 500 report pertaining to research conducted by Fisheries Research Institute personnel from 1973 to the present. • has unlimited Internet access. • http://www.fish.washington.edu/Publications/frireps.html.
Personal collections of Herrera's (the consulting firm that prepared these white papers) staff including "consultant reports, textbooks, etc."
Best professional judgement
NOAA regional library
Northwest Fishery Science Center (NWFSC)
Aquatic Sciences and Fisheries Abstracts (ASFA) • The ASFA database has limited online membership but was accessed through the UW library system. • The ASFA database includes literature dating back to 1982 covering the science, technology, and management of marine and freshwater environments. • It includes 5,000 international sources in the form of primary journals, source documents, books, monographic series, conference proceedings, and technical research reports.
National Technical Information Service (NTIS) • has unlimited Internet access.
UW Urban Water Resource Management database • has unlimited Internet access. • http://depts.washington.edu/cuwrm/.
Seattle Aquarium Salmon Information Center database • has unlimited Internet access.
UW library catalog • has unlimited Internet access.
USDOE Energy Citation Database • has unlimited Internet access • http://www.osti.gov/energycitations/.

The methodology sections for the 2007 white papers provided some additional detail:

To identify data gaps and evaluate the state of scientific knowledge applicable to the potential impacts of HPA-permitted projects on the HCP species and their habitats, the acquired literature was examined to assess the broader issue of how these species use aquatic habitats and how HPA-permitted projects and their construction alter habitat functions.

Existing literature reviews, peer-reviewed journal articles, books, theses/dissertations, and technical reports were reviewed for information specific to aquatic species and their interaction with HPA-permitted projects. Through this process, a collection of information was assembled on the life history, habitat uses, and the potential impacts that these structures pose to HCP species.

Reference material from each of the above databases was compiled in an Endnote personal reference database (Endnote version X). Reference types collected and entered into the database included journal articles, reports, web pages, conference proceedings, theses, statutes, books, and book sections. Each entry in the database included descriptive information, including author(s), year, title, volume, pages, publisher, etc. Whenever an electronic copy of the reference material was available, a link between the reference entry and a PDF copy of the reference material was included in the database. If an electronic (.PDF) copy of a reference was not available, a hardcopy of the material was kept on file. All reference materials cited in the literature review were either linked to the reference database or retained in an associated file as a hardcopy.

Endnote X is the industry standard software for organizing bibliographic information. It features a fully searchable and field sortable database that can contain an unlimited number of references. Reference information is entered into the database either by direct import from online databases or by manually entering the reference information into reference type templates. Once all the references were entered, the database was used for organizational and archival purposes.

4 Hydraulic Project Descriptions

4.1 General Descriptions

Riverine environments are those where flow is dominantly unidirectional, significant, and confined in a single channel or a set of intersecting channels.

Lacustrine environments are those freshwater bodies surrounded by land. Because impoundments behind dams have little significant flow, impoundments are included as a type of lacustrine environment.

Marine environments are those where physical processes are dominated by tides and/or waves and salinity is at least occasionally important. Marine environments include deltas and confined estuarine embayments (e.g., Willapa Bay, Commencement Bay).

These definitions mean that riverine environments may be tidally modulated, and marine environments may have varying salinity.

Hydraulic projects in the following categories are discussed in this chapter:

- Bank protection/stabilization (bulkheads, retaining walls, revetments, toe protection, beach nourishment, subsurface drainage, biotechnical bank protection, bank reshaping or regrading, soil reinforcement, coir and straw logs, integrated approaches)
- Shoreline modifications (groins, jetties, breakwaters)
- Overwater structures (docks, floats, piers, ramps, wharfs, pilings and non-structural pilings, and combined uses of these structures in marinas and terminals)
- Habitat modification (beaver dam removal, large woody debris manipulations, spawning substrate augmentation, riparian planting, wetland creation/restoration/enhancement, beach nourishment/contouring, reef creation, eelgrass planting/restoration/enhancement, in-channel and off-channel habitat modifications
- Channel modifications (dredging, gravel mining and bar scalping, sediment capping, channel creation and alignment)
- Water crossings (bridges, culverts, conduits)
- Fish passage (fish ladders, culverts, weirs, roughened channels, trap and haul)
- Fish screens (in-channel, off channel).
- Flow control structures (dams, weirs, dikes, levees, tide gates, intakes, outfalls).

4.2 Statutes and Rules Regulating Hydraulic Project Approval Permits

WDFW is charged by state law to preserve, protect, and perpetuate all fish and shellfish resources of the state. WDFW regulates construction that may affect fish and shellfish in accordance with the Hydraulic Code set forth in RCW 77.55. RCW 77.55.011(7) defines a hydraulic project as “the construction or performance of work that will use, divert, obstruct, or change the natural flow or bed of any of the salt or freshwaters of the state.”

The Hydraulic Code Rules establish "regulations for the construction of hydraulic projects or performance of other work that will use, divert, obstruct, or change the natural flow or bed of any of the salt or fresh waters of the state (WAC 220-110)." While WDFW exercises its authority by granting permits (known as hydraulic project approvals or HPA permits) prior to construction, in issuing the permit, it takes into account the likely ongoing effects of the project.

The following tables summarize the subsections of WAC 220-110 and of RCW 77.55 that may apply to various types of hydraulic projects. The tables are meant to be inclusive; some subsections are cited because they may be related to the HPA activity, even if the activity is not called out specifically in the citation.

Table 4-1: Freshwater hydraulic project provisions in Washington Administrative Code (WAC)

WAC – Freshwater Provisions		**White Paper Title (year)**									
WAC section number	WAC section title	bank protection (06)	water crossing structures (06)	overwater structures (06)	shoreline modifications (07)	fish passage structures (07)	marinas and terminals (07)	channel modifications (07)	habitat modifications (07)	fish screens (07)	flow control structures (07)
220-110-050	Bank protection	x	x	x	x	x	x	x	x	x	x
220-110-060	Construction of freshwater docks, piers, and floats and the driving or removal of piling		x	x			x				
220-110-070	Water crossing structures		x			x				x	x
220-110-080	Channel change/ realignment	x	x		x	x		x	x	x	x
220-110-100	Conduit crossing		x								x
220-110-120	Temporary bypass culvert, flume or channel	x	x		x	x		x	x	x	x
220-110-130	Dredging in freshwater areas	x	x		x	x	x	x	x	x	x
220-110-140	Gravel removal		x		x	x		x	x	x	x
220-110-150	Large woody material removal or repositioning	x			x	x	x	x	x	x	x
220-110-160	Felling and yarding of timber								x		
220-110-170	Outfall structures				x		x		x		x
220-110-180	Pond construction							x	x	x	x
220-110-190	Water diversions					x				x	x

WAC – Freshwater Provisions		**White Paper Title (year)**									
WAC section number	WAC section title	bank protection (06)	water crossing structures (06)	overwater structures (06)	shoreline modifications (07)	fish passage structures (07)	marinas and terminals (07)	channel modifications (07)	habitat modifications (07)	fish screens (07)	flow control structures (07)
220-110-200 through 220-110-209	Mineral prospecting[1]							x	x	x	
220-110-223	Freshwater lake bulkheads	x			x	x	x	x	x	x	x
220-110-224	Freshwater boat hoists, ramps, and launches	x		x	x	x	x	x	x		

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[1] The sections of WAC 220-110 pertaining to prospecting are currently under revision. As of April 2009, it is anticipated that the pertinent sections will become 220-110-200 through 220-110-206

Table 4-2: Saltwater hydraulic project provisions in Washington Administrative Code (WAC)

WAC – Saltwater Provisions		**White Paper Title (year)**									
WAC section number	WAC section title	bank protection (06)	water crossing structures (06)	overwater structures (06)	shoreline modifications (07)	fish passage structures (07)	marinas and terminals (07)	channel modifications (07)	habitat modifications (07)	fish screens (07)	flow control structures (07)
220-110-250	Saltwater habitats of special concern	x	x	x	x	x	x		x	x	x
220-110-270	Common saltwater technical provisions	x			x	x	x		x	x	x
220-110-271	Prohibited work times in saltwater areas	x	x	x	x	x	x		x	x	x
220-110-280	Bulkhead and bank protection in saltwater areas (nonsingle family residence)	x			x	x	x		x		x
220-110-285	Single-family residence bulkheads in saltwater areas	x			x	x			x		x
220-110-290	Saltwater boat ramps and launches	x			x	x	x		x		
220-110-300	Saltwater piers, pilings, docks, floats, rafts, ramps, boathouses, houseboats, and associated moorings	x		x	x	x	x		x		
220-110-310	Utility lines	x	x		x	x			x	x	x
220-110-320	Dredging in saltwater areas				x	x	x	x	x	x	x
220-110-330	Marinas in saltwater areas	x		x	x	x	x		x		x

Table 4-3: Aquatic plant control provisions in Washington Administrative Code (WAC)

WAC – Aquatic Plant Control Provisions		**White Paper Title (year)**									
WAC section number	WAC section title	bank protection (06)	water crossing structures (06)	overwater structures (06)	shoreline modifications (07)	fish passage structures (07)	marinas and terminals (07)	channel modifications (07)	habitat modifications (07)	fish screens (07)	flow control structures (07)
220-110-331	Aquatic plant removal and control technical provisions								x		
220-110-332	Hand removal or control								x		
220-110-333	Bottom barriers or screens								x		
220-110-334	Weed rolling								x		
220-110-335	Mechanical harvesting and cutting								x		
220-110-336	Rotovation							x	x		
220-110-337	Aquatic plant dredging							x	x		
220-110-338	Water level manipulation								x		x

In addition, the following sections are generally applicable to HPA permits:

220-110-020, Definitions
220-110-030, Procedures
220-110-032, Modification of technical provisions
220-110-035, Miscellaneous hydraulic projects – permit requirements and exemptions
220-110-040, Freshwater technical provisions
220-110-230, Saltwater technical provisions

Table 4-4: Revised Code of Washington (RCW) sections pertaining to hydraulic projects

RCW 77.55		**White Paper Title (year)**									
RCW section number	RCW section title	bank protection (06)	water crossing structures (06)	overwater structures (06)	shoreline modifications (07)	fish passage structures (07)	marinas and terminals (07)	channel modifications (07)	habitat modifications (07)	fish screens (07)	flow control structures (07)
77.55.011	Definitions	x	x	x	x	x	x	x	x	x	x
77.55.021	Permit	x	x	x	x	x	x	x	x	x	x
77.55.051	Spartina/purple loosestrife – removal or control							x	x		
77.55.081	Removal or control of aquatic noxious weeds – rules – pamphlet								x		
77.55.091	Small scale prospecting and mining - rules							X			
77.55.141	Marine beachfront protective bulkheads or rockwalls	X									
77.55.151	Marina or marine terminal			x			x				
77.55.161	Stormwater discharges										X
77.55.171	Watershed restoration projects - Permit processing								X		
77.55.181	Fish habitat enhancement project - Permit review and approval process	x	x		x	x		x	x		
77.55.191	Columbia River anadromous fish sanctuary - Restrictions									x	x

RCW 77.55		**White Paper Title (year)**									
RCW section number	RCW section title	bank protection (06)	water crossing structures (06)	overwater structures (06)	shoreline modifications (07)	fish passage structures (07)	marinas and terminals (07)	channel modifications (07)	habitat modifications (07)	fish screens (07)	flow control structures (07)
77.55.221	Flood damage repair and reduction activities - five year maintenance permit agreements	x	x					x	x		
77.55.231	Conditions imposed upon a permit - reasonably related to a projects	x	x	x	x	x	x	x	x	x	x
77.55.241	Off-site mitigation	x	x	x	x	x	x	x	x	x	x
77.55.261	Placement of woody debris as condition of permit	x	x	x	x	x	x	x	x	x	x
77.55.271	Sediment dredging or capping actions – dredging of existing channels and berthing areas – mitigation not required						x	x	x		
77.55.281	Fishways on certain agricultural drainage facilities					x					x

4.3 Federal regulations that may also apply.

Many activities that are permitted by the HPA program may also require a permit from the U.S. Army Corps of Engineers authorizing the placement of fill in waters of the United States (known as a Section 404 permit, referring to Section 404 of the federal Clean Water Act) or the placement of structures in navigable waters (known as a Section 10 permit, referring to Section 10 of the federal Rivers and Harbors Act).

In many cases, a Corps Nationwide Permit containing standard conditions applies. However, on September 26, 2006, the Corps proposed revision of the Nationwide Permit system; therefore, it is not practical for this analysis to make assumptions about future permit conditions that might be imposed by the Corps for projects authorized under the Nationwide Permit system.

All projects authorized under Corps permits are subject to additional conditions, some of which may be derived pursuant to interagency consultation with the federal agencies as provided for under Section 7 of the ESA.

4.4 Hydraulic Project Descriptions

4.4.1 Bank Protection and Stabilization

WDFW defines[2] bank protection structures as "permanent or temporary structures constructed parallel to and immediately adjacent to the shoreline and landward of the shoreline for the purpose of protecting or stabilizing the bank (e.g., bulkheads, retaining walls, etc.)." This category of activities is distinguished from shoreline modifications, which are structures constructed perpendicular or nearly perpendicular to the shoreline that extend into the water (e.g., jetties, groins, breakwaters, and bank barbs).

Particularly useful technical details for bank protection and stabilization techniques are presented in the *Integrated Streambank Protection Guidelines* (Cramer et al. 2003) and *Stream Management* (Fischenich and Allen 2000). The HCP-covered bank protection activities would include an array of these techniques as discussed below. This list is not intended to be exhaustive, but addresses many of the most commonly applied techniques. Photographs and designs of these techniques are available in Cramer et al. (2003).

Bank protection methods are described using the categories of hard approaches and soft approaches, following Williams and Thom (2001). Hard approaches armor the bank with material intended to resist shear forces experienced at the project site, such as riprap, concrete, or timber bulkheads that would prevent erosion of the bank. Soft approaches attempt to mimic natural processes with the use of biotechnical methods such as live plantings, rootwads, and large woody debris (LWD); soft approaches are used where shear forces are relatively low. Many projects integrate both hard and soft approaches, as described in the *Integrated Streambank Protection Guidelines* (Cramer et al. 2003).

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[2] The definition of bank protection structures was provided to the authors of the white paper by WDFW in Appendix B of Exhibit B of the Request for Proposal for this project, RFP No. 06-0005.

4.4.1.1 Hard Approaches

4.4.1.1.1 Vertical Retaining Walls

Vertical or near-vertical walls along banks and shores have many names, including bulkheads, seawalls, and cribwalls. These features may contain various materials, including concrete, metal, wood, and rock (Zelo et al. 2000). They consist of a vertical wall constructed of sheets of material or of piling with horizontal lagging and backfill.

Vertical walls are typically applied as bank protection on very steep slopes, in instances when landowners want to increase level property adjacent to the water, when floodplain encroachment has occurred, and/or when a near-vertical structure is necessary due to space limitations or to protect an eroding streambank. Issues of material choice, material preservation, and design for these walls typically depend on the project site and habitat concerns.

Cribwalls can be useful in stabilizing steep slopes where a near-vertical structure is required to protect an eroding streambank. Cribwalls are built as log-cabin-shaped structures parallel to the bank to deflect erosive currents away from the bank.

4.4.1.1.2 Revetments

"Revetment" is a generic term for sloping structures placed parallel to the contours of a shoreline in order to absorb incoming energy from stream or wave flow and to protect the slope (Williams and Thom 2001). Most revetments are constructed of rock, but may include other material such as concrete or logs. Revetments include the following:

- **Riprap**: Riprap is large, angular rock used for bank protection. It is typically placed over a filter layer of gravel or synthetic filter fabric. Riprap is the most common material used for bank protection in the United States (Cramer et al. 2003). Recent concerns about reduced habitat value and geomorphic repercussions of riprap have spawned development of alternative techniques. The *Integrated Streambank Protection Guidelines* (Cramer et al. 2003) recommends that new riprap installations be built "only where bank failure would have intolerable consequences or where site conditions are extreme," such as in instances of massive bank failure.
- **Gabions**: Gabions are wire mesh baskets filled with soil or rock material that are used along a shoreline (Freeman and Fischenich 2000). Gabions are often used where available rock sizes for a bank protection project are too small to withstand erosive forces, as well as to achieve a smoother bankside appearance for aesthetic reasons. Vegetation may or may not be incorporated into the structure, depending on needs for long-term stability, weathering, and habitat considerations.
- **Concrete-filled Bags**: These bags are placed in bricklaying fashion on the bank and the concrete is allowed to cure to the shape of the bag.

- **Interlaced Concrete Forms**: These forms consist of flexible, interlocking matrices of concrete blocks of uniform size and weight connected by a series of cables.
- **Cellular Blocks**: These pre-cast concrete blocks are designed to be placed on a prepared bank in a manner that leaves many openings, allowing planted vegetation to grow from cavities.

4.4.1.1.3 Hardened Toes

Hardened toes function to prevent erosion by providing the foundation for upper-bank features such as reinforced soil lifts or vegetative plantings (Cramer et al. 2003). These toes feature angular rock components for roughness attributes. Large woody debris (LWD) may be incorporated into roughened-rock toes as a habitat feature and to provide additional roughness. Rock toes can be used where there is less risk to infrastructure and where habitat mitigation must be incorporated into the treatment. Roughened-rock toes can also be employed as a complementary toe treatment to other bank protection methods, for instance in concert with bioengineered bank protection measures.

4.4.1.1.4 Levees

Levees are not bank protection per se, but are earthen embankments built to provide flood protection from occasional high-water events. Levees are more stable than a continuous form of bank protection, such as revetments on bank curves greater than 30 degrees (Fischenich and Allen 2000). Because they direct flow, levees can cause channel changes and clearly become ineffective when overtopped with high water. Levees are discussed under Flow Control Structures.

4.4.1.2 Soft Approaches

4.4.1.2.1 Log/Rootwad Toes

Log and rootwad toes are added as a preventive measure to stem erosion at the toe of a bank or shore, providing the basis for upper-bank treatments such as reinforced soil or resloped banks (Cramer et al. 2003). Typically, these toes consist of logs installed parallel to the bank and backfilled with gravel and may contain additional LWD features or other rock protection. Log and rootwad toes provide erosion protection and are not intended to function as structural retaining walls. Their top elevation does not exceed the lower limit of vegetation on the bank. LWD for bank protection is intended to resist shear until such time as vegetation can be reestablished, after which it can rot out with little risk. Currently, the *Integrated Streambank Protection Guidelines* (Cramer et al. 2003) considers this technique experimental because so few log and rootwad toe structures have been installed and monitored. The technique is likely best used where there is less risk to infrastructure and when habitat mitigation is required, because habitat elements can be incorporated into the design.

4.4.1.2.2 Beach Nourishment

Beach nourishment in Washington State is most commonly employed using coarse gravel to combat shoreline erosion at relatively small sites (Williams and Thom 2001). This method entails placing fill material either as an independent activity or integrated with hard structures or bioengineered solutions. Fill is typically required to be similar in size to existing native material, be contaminant-free, and have low silt/clay components to preclude turbidity issues. Beach nourishment is considered a temporary fix to a sediment supply problem and is undertaken with an ongoing commitment for periodic maintenance (Williams and Thom 2001).

4.4.1.2.3 Subsurface Drainage Systems

Subsurface drainage systems are typically installed under or behind other bank treatments in order to decrease the saturation of soil and increase slope stability on side banks (Cramer et al. 2003). Techniques include chimney drains, collection drains, and gravel seams, which may be gravity-based or pumped systems.

4.4.1.2.4 Biotechnical Bank Protection

Soft approaches to bank protection include a suite of developing biotechnical protection methods in which natural materials are used or vegetation is planted to address slope stability. Current biotechnical bank protection methods are as described below and in much of the available literature (e.g., Allen and Leech 1997; Cramer et al. 2003; Zelo et al. 2000; Saldi-Caromile et al. 2004; Fischenich and Allen 2000; Williams and Thom 2001):

- **Riparian Plantings**: Plantings may be added to bank protection projects in an effort to stabilize banks by the establishment of root material at the shoreline. Native herbaceous cover and woody plants of various species are added, depending on the project goals.
- **Live Stakes or Poles**: Live stakes or poles are the simplest form of vegetation planting along a shoreline, consisting of stakes of live material inserted into the ground, providing reinforcement of surface soil layers. Sometimes a row of stakes or poles features a basket-like live brush mat called a wattle. The live cuttings eventually root and provide long-term reinforcement and may provide some control over internal seepage.
- **Brush Packing, Layering, and Mattressing**: Brush packing involves alternating layers of live branches and earth to fill localized slumps. The branches protrude beyond the face of the slope and reinforce the bank, while the stems provide frictional resistance to shallow slides. The live cuttings eventually root and provide long-term reinforcement. Brush layering involves alternating layers of brush packed materials across larger areas than in brush packing. Brush mattresses lie along the slope with their root ends in a trench at the toe of the slope, as opposed to being planted along the slope.
- **Live Fascines**: Live fascines are sausage-shaped bundle structures made from cuttings of living woody plant material that are placed in a shallow trench along a bank slope contour (Sotir and Fischenich 2001). The live cuttings eventually root and provide long-term reinforcement. The live fascine is constructed from the

elevation of baseflow along the face of an eroded streambank. Live fascines are used for bank and toe protection as well as improvement of erosion control, infiltration, and other riparian zone functions.

- **Roughness Trees/Tree Revetments:** Roughness trees, which are also called tree revetments, function to slow down the water velocity in an active channel and reduce hydraulic shear stress, helping sediments accumulate at the site and enabling the establishment of vegetation (Cramer et al. 2003). This process ultimately results in the protection of vulnerable or eroding banks. These revetments may be expanded by installation of LWD, which is often anchored for stability, in the channel or along the banks (Fischenich and Morrow 1999).

4.4.1.2.5 Bank Reshaping or Regrading

Bank reshaping or regrading is employed to stabilize an eroding streambank by reducing the angle of its slope without changing the location of the toe. This technique is almost always conducted along with other bank protection treatments and may include vegetated components and a new toe installation. Regrading is most often applied along vertical and/or eroding banks, but the ability to reshape banks may be limited where access is difficult for heavy equipment, and regrading may be unsuitable where mature riparian vegetation or infrastructure exists. The technique is not considered effective to prevent continuing erosion at a reach level because it does not address the actual mechanisms of failure.

4.4.1.2.6 Soil Reinforcement

Soil reinforcement refers to a system of soil layers or lifts encapsulated or otherwise reinforced with a combination of natural or synthetic materials and vegetation, sometimes in a terraced fashion. These systems are also known as fabric-encapsulated soil, fabric-wrapped soil, soil burritos, vegetated geogrids, or soil pillows. This technique is best used on eroding banks on small creeks, large rivers of lower gradients, and estuaries where a resilient and bioengineered or biotechnical treatment is needed and where a wide range of bank-failure mechanisms occurs, including toe erosion, mass wasting, and scour.

4.4.1.2.7 Coir and Straw Logs

Coir and straw logs are similar to soil reinforcement in that they provide a system of layered materials, typically with integrated vegetation. Coir logs are long, sausage-shaped bundles of coir (coconut fiber) or straw, bound together with additional coir or synthetic netting (Allen and Fischenich 1999). They may be planted with herbaceous or woody vegetation and function to provide temporary biodegradable protection to banks while the vegetation develops. In addition, they also encourage sediment retention during overbank flows.

4.4.1.3 Integrated Approaches

Integrated approaches to bank protection have been developed to incorporate some of the best attributes of both hard and soft approaches (Cramer et al. 2003). One important general goal of integrated bank protection is to use habitat features that can deteriorate and ultimately allow the bank to protect itself through maturation of the design. For example, woody toe protection will deteriorate as native vegetation matures and begins to provide support and structure to a bank. Further examples of these approaches include integrating vegetation, coir logs, and woody debris into gabion or riprap structures; integrating vegetation and woody debris into rock or log toes to create habitat structure at the bank; and integrating rock toes with biotechnical soil reinforcement for toe and bank stability. Many of the hard and soft approaches discussed above can be similarly combined to protect against bank erosion while allowing habitat-forming processes to occur.

4.4.2 Shoreline Modifications

Shoreline modifications are oriented perpendicular to the direction of dominant sediment transport, and usually perpendicular to the shoreline. They include jetties, groins, bank barbs, and breakwaters. This category of activities is distinguished from bank protection structures, which are parallel to and immediately adjacent to the shoreline and landward of the shoreline for the purpose of protecting or stabilizing the bank.

Shoreline modification structures include:

- A **jetty** is a structure constructed at navigational channels to prevent sand from depositing in the channel and to provide wave protection for vessels (Dean and Dalrymple 2002). **Weir jetties** are submerged at most water levels for some portion of their length, usually the landward-most end. These features allow the passage of sediment for localized deposition in some inactive portion of the navigational channel (Seabergh and Kraus 2003).

- A **breakwater** is a structure that is built seaward of the breaker line parallel to the shoreline to protect nearshore infrastructure and prevent shoreline erosion. Breakwaters are often used in series (Dean and Dalrymple 2002).

- **Groins and bank barbs** are fingerlike, bank-protection structures keyed into one bank and oriented obliquely to the flow. Groins and bank barbs are typically constructed in sets along the outside of a meander bend, with the primary function of redirecting flow and bed material away from the bank and toward the middle of the channel. These structures reduce near-bank velocities, increase centerline velocities, retard bank erosion, cause local bed scour around the groin tip, and trap fine sediment and debris between structures (Li et al. 1984).

    - Groins are vertical barriers extending perpendicularly from the shore/bank that impede the downdrift/downstream movement of sediment. Groins are typically exposed above high water and are designed to divert flow (and bed sediment) around the structure. The primary purpose of a groin is to store sediment on the bank adjacent to the structure and prevent erosion to the updrift shoreline of the groin. Groins can be installed in rivers, lakeshores, and marine shorelines (NRC 2007).

    - A bank barb is a specific type of groin. It is a low-elevation structure projecting from a stream bank and angled upstream to redirect flow away from the bank, thereby controlling bank erosion (WDFW 2003). Bank barbs are typically submerged at or below low water. Weir-type structures, like bank barbs, are intended to redirect flow toward the center of the channel using weir hydraulics over the structure. They are often used in series. Because submerged structures are relatively ineffectual as a result of the large suspended load in high-wave-energy environments, bank barbs are often not associated with marine and lacustrine environments.

- Shoreline structures that intercept and disrupt normal transport processes, but whose primary purpose is not shoreline protection, are also considered shoreline modifications. Examples include beach-access stairways and boat ramps that project beyond the ordinary high water mark (OHWM). Such structures are hereafter called **analogs**. Typically, these structures would affect the aquatic environment in a similar manner as groins if they do not influence the flow or exchange of groundwater or surface water with the main water body, or act similar to a jetty if they do.

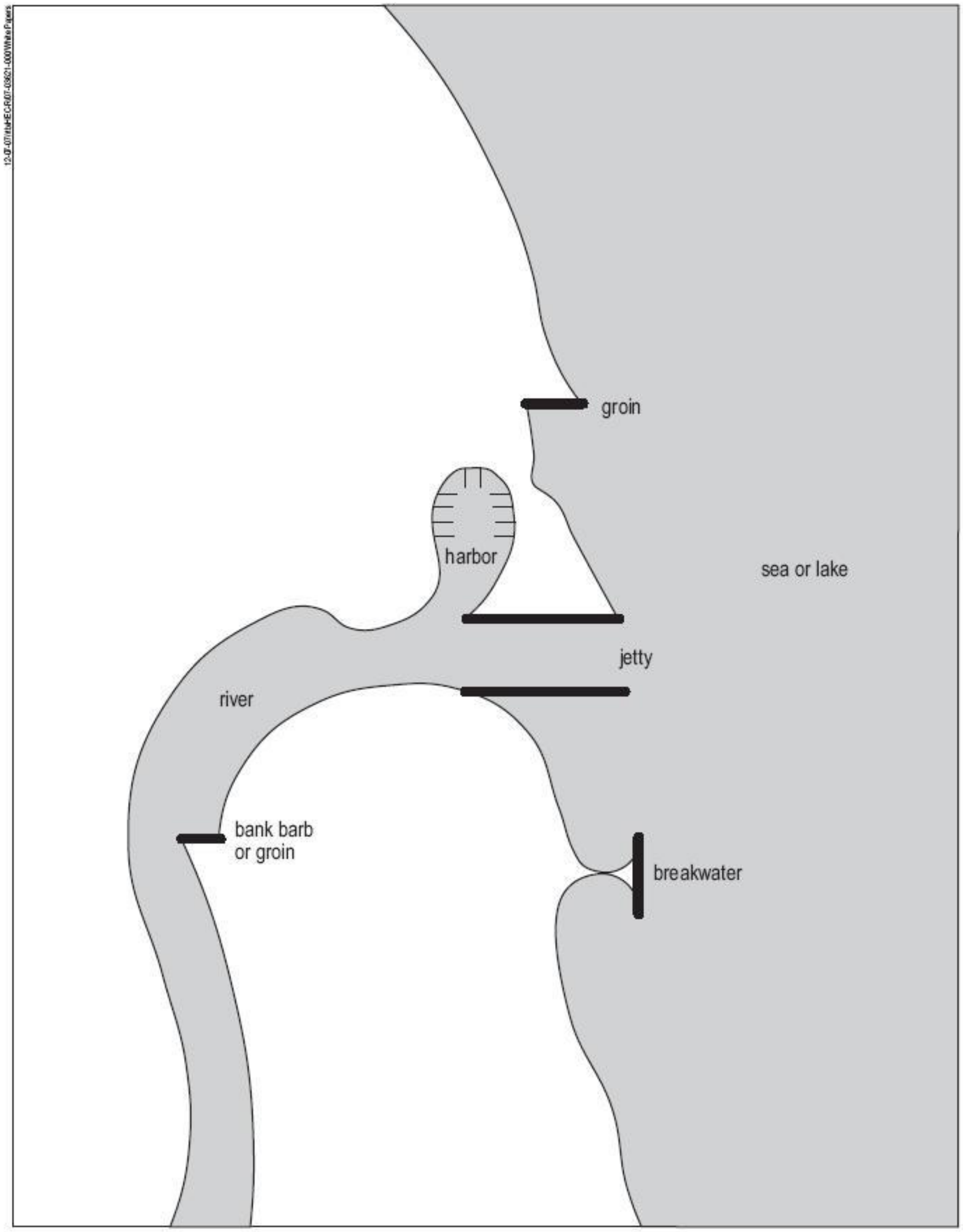

Figure 4-1. Location of Shoreline Structures

4.4.3 Overwater Structures

Overwater structures are simple structures located above and on the water to provide boat moorage and other sorts of access to the water. Single-family residential docks are a common example of overwater structures. Non-structural piling are discussed with overwater structures because of the overlap in potential impact mechanisms associated with the construction and presence of these structures. Larger and more complex structures, such as marinas and marine terminals, are discussed separately.

- A **pier** is an elevated and stationary walkway supported by piling that extends waterward of the shoreline.
- A **float** and a **dock** are walkways or other surfaces that float on the water.
- A **ramp** is a walkway connecting a pier or other shoreward structure to a float and providing access between the two.
- A **wharf** is an elevated and stationary structure oriented parallel to the shoreline, such that ships can lie alongside to load and unload cargo and passengers.
- A **piling** or pile is a pole, usually made of wood, steel or concrete, that is driven into the stream, lake, or ocean bed to support shoreline infrastructure. Non-structural pilings are individual pilings, including utility poles. Marinas and Terminals Project Description

A **marina** is a public or private facility providing vessel moorage space, fuel, or commercial services. Commercial services include but are not limited to overnight or live-aboard vessel accommodations (RCW 77.55.011(9)).

A **marine terminal** is a public or private commercial wharf and used, or intended to be used, as a port or facility for the storing, handling, transferring, or transporting of goods, passengers, and vehicles to and from vessels (RCW 77.55.011(10), with passengers added for purposes of this paper).

Marinas/terminals incorporate many individual components of overwater structures, including pilings and vessel access facilities. Marinas or terminals have a number of components, including:

- A **pier** is an elevated and stationary walkway supported by pilings that extends waterward of the shoreline.
- A **float** and a **dock** are walkways or other surfaces that float on the water.
- A **ramp** is a walkway connecting a pier or other shoreward structure to a float and providing access between the two.

- A **wharf** is an elevated and stationary structure oriented parallel to the shoreline, such that vessels can lie alongside to load and unload cargo and passengers.

- A **dolphin** is a buoy, pile, or group of piles used for mooring boats or group of piers used as a fender at a dock.

- A **fender panel** is a structure for protecting other nearby structures from collision with ships.

- A **piling** or **pile** is a pole, usually made of wood, steel or concrete, driven into the stream, lake, or ocean bed to support shoreline infrastructure. It includes both structural and nonstructural pilings. **Non-structural pilings** are individual pilings, including utility poles.

- An **access ramp or boat ramp** is "a uniformly sloping platform, walkway, or driveway common in the coastal environment as a launching area for small watercraft (Mulvihill et al. 1980). Ramps extend into the water at a slope of 12 percent to 15 percent (Mulvihill et al. 1980) and are typically oriented perpendicular to the shoreline. The design of ramp widths (3m to 15 m) varies with patterns and intensity of human-use needs, whereas lengths often depend on the slope of the shoreline and tidal amplitudes. Ramps extend from the terrestrial zone to below the low intertidal zone and are usually constructed in protected areas with access to fairly deep water close to shore. Construction materials commonly consist of gravel, concrete, or asphalt…" (Williams and Thom 2001).

- A **boat hoist** is a piece of equipment used to move boats from uplands to water. It may be free-standing or affixed to a dock or bulkhead.

- A **boat basin** is the area within a marina where multiple vessels are docked.

- A **pump-out station** is a facility used to remove sewage and/or gray water from the holding tanks of vessels. It is usually located on a dock within a marina for easy access by vessels.

- A **water intake site** is used to supply potable water to vessels. It is usually located on a dock within a marina or terminal for easy access by vessels.

- A **refueling facility** is used to supply fuel, such as marine grade gasoline or diesel fuel, to vessels. It is usually located on a dock within a marina or terminal for easy access by vessels.

Marinas/terminals also typically include considerable shoreline modification or bank protection structures in the form of breakwaters, wingwalls, bulkheads, seawalls, and nearshore buildings.

The area of potential alteration associated with a marina or terminal includes all the overwater structures and their associated dredged area; the area affected by changes in light regime from shading and artificial lighting; and the area affected by vessel propeller scour, vessel emissions and exhaust, and boat wakes. Furthermore, pollution from spillage and accidental discharges of toxins, waste, or stormwater may extend the area of alteration beyond the marina itself. The area of alteration and the effects on habitat-controlling factors may be limited, if enclosed by breakwaters (WDNR 2007).

4.4.4 Habitat Modifications

Habitat modifications are generally projects which are designed to improve habitat for fish and invertebrates. Although many of these activities may provide a net benefit to aquatic habitat and HCP-listed organisms, most are associated with short-term negative impacts during the construction phase. Beaver dam and large woody debris removal can be regarded as having longer-lasting negative impacts on the aquatic environment.

Habitat modifications have been widely applied in fluvial environments in the state of Washington over the past twenty years. The majority of these restoration measures are designed to create refugia within the river-floodplain system so that organisms can survive during extreme low flow and high flow conditions. In-channel habitat modification has been widely practiced but some research has indicated that these projects have a relatively high failure rate (Frissell and Nawa 1992; Roni et al. 2002) and practitioners and researchers have begun to recommend focusing on off-channel habitat creation and modification (Roni et al. 2002).

4.4.4.1 Beaver Dam Removal/Modifications

Beaver are ecosystem engineers that can drastically alter fluvial environments through the conversion of lotic habitat to lentic habitat (Naiman et al. 1986). The mosaic of connected aquatic habitat patches created by beaver are utilized by numerous aquatic species (Pollock et al. 2003) and beaver dam presence is widely believed to be a net benefit to aquatic organisms (Clifford et al. 1993; Ray et al. 2004; Sigourney et al. 2006). However, until relatively recently it was thought that beaver dams acted as barriers to fish migration and that beaver dam removal would benefit migratory species (Naiman et al. 1988). This, combined with concerns of flooding caused by beaver activity has lead to the modern day removal and modification of beaver dams.

Beaver dam removal is conducted using various methods. Most common is the removal of the dam with hand held tools. This removal method has the least impact on aquatic habitat because it generally is a more gradual process than the other methods. Other methods include using explosives and heavy machinery to remove dams. These methods are usually reserved for large dams and are associated with greater impacts on the aquatic ecosystem. To reduce impacts associated with dam removal but still control beaver-induced flooding, beaver dam modification has become a popular alternative to removal. The use of “beaver deceivers,” a trapezoidal device installed at the mouth of a culvert that would otherwise be dammed by beavers, can stabilize the pool volume and water level while conserving the ecological function of the dam and associated impoundment.

4.4.4.2 Large Woody Debris Placement/Movement/Removal

Large woody debris (LWD) placement, movement, and removal are the most common habitat modification activities in the state of Washington. Almost all LWD additions involve the use of natural materials taken from on-site riparian zones and off-site areas as well. The wood is often secured to the bank or bed of the channel using rebar, cable, or chains. Some wood additions involve burying the bole or root wad of the tree in the bank or bed, driving timber or steel piles, and layering rack member and key members through the structure. LWD additions range in scale from single logs to massive engineered logjams.

The removal of LWD from river channels began in earnest in 1829 with the launching of the first snagboats on the Ohio and Mississippi Rivers (Wohl 2004). Snags, or submerged large wood, were a menace to ferrying boats and as commerce along these riverine corridors grew so did the need for clear passage. The practice of “snagging” (also called de-snagging) a river did not however stop at lowland rivers. With clear passage up the mainstems of rivers loggers could more easily access heavily wooded tributaries. These smaller channels were also snagged and what wood was not removed by hand was later scoured out by log drives. Studies of paired streams, some with a history of log drives and some undisturbed, have shown that channels which have had a history of log drives contained 10 to 100 times fewer logs and significantly fewer pools (Wohl 2001; Napolitano 1998). With the realization that the habitat created by instream wood is vital for resident aquatic organisms, people began in earnest to put wood back into streams.

4.4.4.3 Spawning Substrate Augmentation

Gravel augmentation for the purpose of spawning habitat improvement has been promoted episodically by various government agencies since the 1960s (Bunte 2004). The two most common methods of spawning substrate augmentation are direct gravel placement and passive gravel placement.

Direct placement involves shaping the channel (by adding or removing gravel) to ensure that the 1.5-year recurrence interval flow fills the channel to its morphological bankfull stage. One form of this method involves providing bed material that is partially mobilized at bankfull flow and long-term gravel additions to match the post-restoration transport capacity. Another form of this method involves using a gravel size which will be minimally mobilized at bankfull discharges; this reduces the need for future augmentations but increases the probability that the gravels will become clogged with silt and organic material. Machinery must enter the channel and deposit gravel in various locations. This process involves front-end loaders, excavators, and/or bull dozers repeatedly entering and exiting the channel, usually along a constructed gravel ramp (Bunte 2004).

Passive gravel placement involves the dumping of gravels from a stream bank (Bunte 2004), supplying gravel to the channel at a logistically convenient location. The gravel is placed en-mass and mobilized by the stream during high-discharge events. As the gravel is entrained and deposited downstream, spawning habitat is created. This method is either conducted independently or in concert with direct gravel placement augmentations.

4.4.4.4 Riparian Planting/Restoration/Enhancement

Habitat modifications involving riparian vegetation involve a combination of invasive species removal and native species planting. Invasive species removal temporarily removes cover and destabilizes banks, while native species planting will increase cover and bank stability once plants have become established and grown. Riparian vegetation modifications generally do not involve potentially damaging work within the aquatic habitat. It is widely practiced in the state of Washington.

4.4.4.5 Wetland Creation/Restoration/Enhancement

Wetland creation, restoration and enhancement usually requires a considerable construction effort involving heavy machinery, the construction or alteration of weirs and other flow control structures, and/or planting. Wetland manipulations are conducted primarily by adjusting the hydroperiod of a parcel of land. This is done by either altering geomorphology in such a way as to increase or decrease the duration of wet or dry periods, or by adjusting the system hydrology. Initial construction phase activities are typically be associated with negative impacts on fish and invertebrates but, in time, the ecosystem functions provided by the wetland habitat can result in a net ecosystem benefit.

To control hydroperiod and create optimal habitat, wetland creation, restoration, and enhancement activities may in certain instances require partial wetland filling. Wetland filling is associated with localized habitat degradation, but if the project is properly designed a net ecosystem improvement can result.

- Wetland creation and restoration in riparian systems occurs via three primary methods:

- Floodplain activation through the removal of levees (Florsheim and Mount 2002). This type of wetland restoration usually involves regrading a hydraulically disconnected floodplain and subsequently removing portions of the levee. Consequently, the impact on the channel occurs only at the points where the levee is breached.

- Floodplain activation through base level or floodplain elevation alteration (which may require wetland filling) (Collins and Montgomery 2002).

- Channel realignment to connect a channelized reach to an active river-floodplain system (Whalen et al. 2002).

Wetland creation and restoration in estuarine systems typically involves the removal of dikes or levees, regrading to restore dendritic channels, and sometimes the installation of self-regulating tide gates.

Wetland enhancement typically involves removing noxious weeds, and replanting areas to improve native habitat and species diversity. Occasionally, enhancement includes changing the site’s water regime through excavation, construction of weirs, or removal of ditches and drains. Enhancement has historically focused on habitat, but other wetland functions can also be enhanced.

4.4.4.6 Beach Nourishment/Contouring

Beach nourishment, as it is practiced in Washington State, is primarily designed to restore a more natural (gradual) beach profile in response to either the loss of sediment supply (usually by dams or armored bluffs), increased wave energy associated with shoreline armoring, or to supplement and protect placed fill (Shipman 2001).

In Washington State, most beach nourishment occurs in sheltered, coarse-grained (i.e., gravel and cobble) settings (Shipman 2001). It also occurs in sandy, exposed environments, such as on the outer coast.

In sandy settings, a variety of mechanisms of nourishment have been used. One common method is to pump sandy material excavated from offshore the project site. This is most common where erosion problems are caused by the transport of sand offshore due to suspension from large waves (Dean and Dalrymple 2002). It is also common to use dredge spoils to nourish beaches. Jetty Island in Everett is an example of this type of nourished beach (Everett 2006). It is also conceivable that in Washington, where quarried sand is readily available, nourishment materials could originate from onshore, but past nourishment activities have primarily relied on dredged material as a nourishment material source (USACE 1983).

In coarse-grained environments, the most popular method is to use upland quarried alluvium. Delivery to the site is most often by truck, not barge (Shipman 2001). The material added can be from a range of sizes from pebble (0.1 inch in diameter) to cobble (several inches in diameter).

4.4.4.7 Reef Creation

Reef creation is probably the most well-studied fish habitat enhancement activity in the world (Baine 2001). Reefs have been created for habitat and recreational purposes for over seventy years in Washington State (NSC 2007). Early reefs were often harbor or coastal structures slated for demolition (NSC 2007). Wastes that would ordinarily be landfilled or incinerated (e.g., used tires) have also been used to create reefs (Hartwell et al. 1998). Derelict vessels have been used, though they are often stripped of potentially toxic or leachable materials before they are submerged at the project site (Baine 2001).

Often in the scientific literature, the terms reefs and breakwaters are used interchangeably (Pondella and Stephens 1994). Strictly speaking, there is no difference between these terms – both are structural elements seaward of the shoreline, and often permanently submerged. However, their purposes are distinctly different. Breakwaters are employed to protect the shoreline from wave energy (Dean and Dalrymple 2002), while reefs are placed to create habitat and to potentially increase the numbers of fish and invertebrates (West et al. 1994). As a result, a reef could be installed that has no effect on the wave environment.

4.4.4.8 Eelgrass and other Aquatic Vegetation Planting/Restoration/Enhancement

Eelgrass planting is a relatively new habitat enhancement activity. Although planting plans have been implemented for 25 years (Thom 1990), successful programs where eelgrass has been reintroduced to areas where it occurred prior to development, have only occurred recently (Thom et al. 2005). Successful programs must have an adaptive management plan, including the potential planting in successive years to ensure a viable stand (Thom et al. 2005). A number of mechanisms have been proposed to introduce new eelgrass. The most common is to manually plant shoots previously grown in a greenhouse (Fonseca et al. 1998). However, more recently broadcast seeding has been demonstrated to be successful when performed under certain conditions (Pickerell et al. 2005).

4.4.4.9 In-Channel and Off-Channel Habitat Modifications

In-channel and off-channel habitat modification is a broad set of activities which includes the placement of structures within the channel, bank protection measures, channel realignment, side channel creation/connection, and the creation or enhancement of backwater sloughs and pools. Such modifications are discussed in more detail in “roughened channels” under Fish Passage Projects. Artificial realignment and relocation of channels specifically designed to reconfigure the aquatic environment to promote human uses are discussed briefly under Channel Modifications.

The most common method of off-channel habitat rehabilitation is to work in a dry off-channel area and then connect the rehabilitated habitat to the main channel once the work is completed.

In-channel work can range from the placement of rock weirs which will have minimal impact on channel form, to complete channel realignment which will drastically alter the channel form. A goal of most in-channel habitat modification is to create habitat diversity and, in particular sheltered areas where organisms can reside during both high and low flow conditions. Many instream structures such as weirs are designed to locally elevate base-level and promote aggradation and pool formation. Weirs are also discussed under fish passage structures (as “weir-type fishways” and as “weirs”) and under flow control structures.

4.4.5 Channel Modifications

Hydraulic projects pertaining to channel modifications include dredging, gravel mining and bar scalping, sediment capping, and channel creation and alignment.

- **Dredging** includes the removal of substrate from riverine, lacustrine, and marine environments for purposes of improving vessel navigation, the maintenance of channels and sediment traps for flow conveyance and flood control, and hydraulic suction dredging to manage aquatic vegetation. Dredging may also be used in cleaning up contaminated sediments. Dredging related to mineral prospecting is addressed in a separate white paper (Anchor 2006).

- **Gravel mining and bar scalping** is the extraction of gravel resources from the active channel or floodplain by means of pit mining, bar scalping or “skimming,” bar excavation, gravel traps, or channel-wide instream gravel mining.

- **Sediment capping** refers to the placement of a subaqueous covering of clean material over contaminated sediments to isolate contaminants from riverine, lacustrine, and marine environments and biota.

- **Channel creation and alignment** includes the relocation, straightening, or meandering of an existing channel or the creation of a new channel where none existed before.

4.4.5.1 Dredging

Dredging is conducted for various purposes.

- Navigational and maintenance dredging is carried out in larger bodies of water (e.g., Columbia, Snake, and Cowlitz rivers; Puget Sound) and the embayments of the outer coast to allow the passage of deep-draft vessels in channels and marinas. Large-scale navigational dredging is generally conducted from a vessel or barge.

- Maintenance dredging to increase conveyance for flood and erosion control occurs primarily in relatively smaller channels at bridge and culvert crossings, along highways, in roadside and irrigation ditches, and in sediment traps constructed for this purpose. Small-scale navigational dredging may be performed from land using a clamshell bucket operated from a crane, a backhoe or excavator or a suction dredge.

- Dredging has been conducted to remove contaminated sediments. This type of dredging is often associated with sediment capping.

- Hydraulic suction dredging is used specifically for vegetation management in freshwater environments.

- The USACE (1983) describes two primary dredging techniques:

- **Mechanical dredges** include clamshell, dipper, and ladder dredges. Mechanical dredges remove material through direct force and are used both for new and maintenance projects. They can be used to remove loose or hard, compacted materials. Mechanical dredges result in more sediment resuspension when dredging occurs in fine, loose, or noncohesive substrates.

- **Hydraulic dredges** include cutterheads, dustpans, hoppers, hydraulic pipelines, plain suction, and sidecasters. Hydraulic dredges remove material in slurries and are generally used for maintenance projects. Hydraulic dredges are generally faster than mechanical dredges. Although hydraulic dredges create less resuspension of sediment than mechanical dredges, considerable resuspension can occur. Additionally, hydraulic dredging entrains considerably more water from the dredge site than mechanical dredging. Smaller, shallower dredging projects typically use different equipment than that required for deep water dredging projects.

4.4.5.2 Gravel Mining and Scalping

Gravel is extracted from riverine environments for use as base material in the construction of roads, highways, and railroads and as aggregate mix in the construction of roads and buildings. Gravel sources include active river sediments and glacial sediments deposited during the Pleistocene by meltwater streams. Sources are ideally located close to markets to reduce transportation costs and maximize profit. Extraction methods can include the following (Kondolf et al. 2002):

- **Dry-pit mining** is the excavation of gravel within the active channel on dry intermittent or ephemeral streams beds.

- **Wet-pit mining** is the excavation of gravel within the riverine floodplain below the groundwater table, requiring the use of a dragline or hydraulic excavator.

- **Bar scalping or “skimming”** is the extraction of gravel from the surface of gravel bars above the low-flow water level.

- **Bar excavation** involves pit excavation at the downstream end of the gravel bar for gravel extraction.

- **Gravel traps** are channel-spanning hydraulic controls that promote ponding and sediment deposition. The collected sediment is then extracted during low-flow conditions.

- **Channel-wide instream mining** occurs in rivers with variable flow regimes and involves the excavation of gravel across the entire active channel width during the dry season.

- **Floodplain and terrace-pit mining** is similar to wet-pit mining but includes dewatering of the pit to work in the dry.

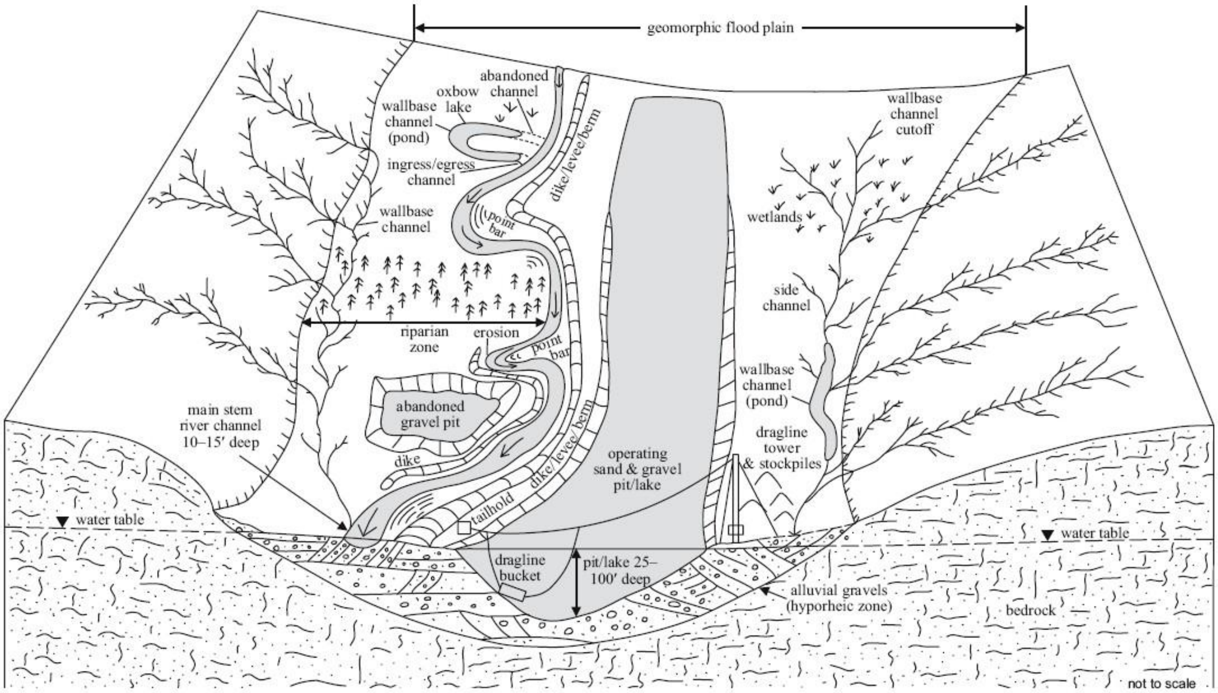

Figure 4-2. Schematic illustration of typical gravel extraction methods within the active floodplain (Norman et al. 1998).

4.4.5.3 Sediment Capping

Sediments contaminated with heavy metals, nutrients, polychlorinated biphenyls (PCBs), polyaromatic hydrocarbons (PAHs), and other organic pollutants are frequently present in urbanized marine and freshwater benthic habitat. In most situations, contaminant levels are sufficiently low such that a “natural recovery” (Garbaciak et al. 1998) or no-action alternative is the most effective form of remediation. However, when contaminant concentrations reach levels that require a more rapid solution, sediment capping or dredging may occur. Sediment capping is the placement of a contaminate-free isolating material over a contaminated sediment deposit (Palermo et al. 1998).

Sediment capping and dredging are frequently conducted in tandem. Sediment capping material is frequently dredged from an adjacent clean sediment source (USACE 1991a).

Two primary forms of sediment capping are practiced:

- The placement of a sediment cap over contaminated dredging spoils. Capping of dredge spoils is sometimes associated with the use of a confined aquatic disposal (CAD) cell. These cells are frequently constructed using piles and other treated wood products.

- The placement of a sediment cap over in-situ contaminated sediments.

The advantage of capping dredged sediments is that the practitioner has control over the physical location of the spoils and cap. A low-energy environment with low densities of aquatic life can be selected to minimize cap erosion and the impact on aquatic biota. However, the impacts associated with dredging can be significant and consequently, in-situ sediment capping (ISC) is frequently a preferred alternative. ISCs commonly have less of an environmental impact than dredging and capping, but the practice is associated with more uncertainty in terms of cap erosion and maintenance (Reible et al. 2003). ISC is a remediation technique that is becoming more common on a global scale (Palermo et al. 1998) but is still a relatively uncommon remediation technique in Washington State. Despite this, some of the most widely studied sediment capping projects in the world have occurred in Puget Sound.

A sediment cap most frequently consists of sand or silt (Palermo et al. 1998), but more recent efforts have focused on the use of active barrier systems (ABS), that incorporate various supportive materials (Jacobs and Forstner 1999; Murphy et al. 2006). The sorptive materials (e.g., activated carbon, zeolite, calcium carbonate) reduce contaminant leaching, while the sediment itself acts as a physical barrier and stabilizing force. Sediment caps can also include geotextiles, liners, or the addition of material such as organic carbon, to attenuate the flux of contaminants into the overlying water.

A sediment cap serves multiple functions including isolation of the contaminated sediment from benthic organisms, physical stabilization of the contaminated sediment, and prevention of contaminant leaching into the water column.

- Physical isolation of the contaminated sediment sufficient to reduce exposure due to direct contact and to reduce the ability of burrowing organisms to move contaminants to the cap surface

- Stabilization of contaminated sediment and erosion protection of sediment and the cap sufficient to reduce resuspension and the transport of contaminants into the water column

- Chemical isolation of contaminated sediment sufficient to reduce exposure from dissolved contaminants that may be transported into the water column.

- Sediment capping has been practiced in large rivers, lakes, estuaries, and coastal marine environments (Palermo et al. 1998; RETEC 2002). Capping has occurred most frequently in marine settings.

- There are six general techniques for constructing a sediment cap, but each technique can be placed in one of two categories (USACE 1991b):

Point dump methods include pipeline placement, hopper placement, and barge placement. All of these techniques entail releasing a large quantity of capping sediment near the water's surface. These techniques are economical and produce a well compacted cap.

Pump down techniques entail creating a sediment slurry and delivering it through a pipe to the surface of the contaminated sediment. Pump down methods include submerged diffusion, sand spreader placement, and gravity-fed downpiping (tremie).

- Tremie equipment consists of a large diameter vertical pipe through which capping material is gravity fed. This technique is similar to point dump techniques in that the velocity of the capping material is not controlled. Consequently, benthic displacement by capping material may become an issue (USACE 1991a).

- Submerged diffusion and sand spreader placement both control the capping material velocity through the use of pumps and diffusion techniques. These methods are characterized by a high degree of placement control and minimal displacement of the contaminated benthos (USACE 1991b).

HPA authority over capping is limited by the WAC. For instance, certain remedial actions conducted under a court order or performed by the Washington State Department of Ecology are exempt from the procedural requirements of the Hydraulic Code but must comply with the substantive provisions of the Hydraulic Code.

4.4.5.4 Channel Creation and Alignment

A primary purpose of channel creation and alignment activities is to relocate the alignment of a waterway away from an eroding bank. Relocation may be used where a significant building or road is directly threatened by erosion. Channel relocation is often a means to solve problems of channel encroachment and/or confinement, and to foster the development of a new, static channel with healthy riparian buffers. A channel can be entirely relocated to a new alignment, or just moved laterally within the existing alignment. Channel relocation permanently changes the location of the channel while preserving or recreating other characteristics, such as the overall channel profile, pattern, cross-section, and bed elevation. Channel relocation is a major undertaking involving the reconstruction of the channel bed, habitat features, channel banks, and floodplain. Channel creation and alignment conducted primarily for purposes of habitat restoration are discussed briefly as "in-channel and off-channel habitat modifications" under Habitat Modifications.

Channel creation and alignment occur primarily in riverine environments but may also include the creation of tidal channels in estuarine environments.

4.4.6 Water Crossings

Two closely-related analyses of "water crossings" and of "fish passage" were conducted in this series of white papers. The distinction has to do with the purpose of installing a particular structure.

Water crossings are defined by WDFW as "structures constructed to facilitate the movement of people, animals, or materials across water from bank to bank. These structures include bridges, culverts, fords, cable cars, tunnels, conduits (regardless of what the conduit is conducting), etc." Water crossings in the form of bridges, culverts, or conduits collectively represent 93 percent of all water crossing HPAs issued by WDFW between 1998 and 2006. Water crossings in the form of fords, cable cars (which would have effects similar to conduits that cross above waterbodies), and tunnels usable by humans (which would have effects similar to conduits placed in a tunnel) are not discussed separately.

- A **culvert** is used for conveying water through a fill. Fish passage aspects of culverts are discussed separately in the Fish Passage activity.

- A **bridge** is used for conveying goods or materials from one side of a water body to another.

- A **conduit** is used for conveying goods or materials. Examples of conduits authorized under HPAs include sewer lines, pipelines, and cables trenched or tunneled across streams and comparable structures trenched, tunneled, or laid on the bottom in marine waters. Conduits can also be attached to bridges, but there are no significant ESA-related risks associated with such installations, and conduit crossings attached to bridge structures do not require HPAs (WAC 220-110-100). Conduit approaches to bridges likely require HPAs.

4.4.7 Fish Passage

Two closely-related analyses of "water crossings" and of "fish passage" were conducted in this series of white papers. The distinction has to do with the purpose of installing a particular structure.

Fish passage structures are built to facilitate the passage of fish through or around a barrier. They are intended to restore upstream and downstream fish access to habitats that have become isolated by human activities (e.g., placement of culverts, dams, and other artificial obstructions). Fish passage facilities can be mitigation measures for adverse effects associated with flow control structures.

The baseline condition for assessing the effects of the fish passage facilities is the predeveloped state of the ecosystem (i.e., the natural stream channel with no dam, road crossing, or barrier), rather than comparison to the existing baseline (i.e., the ecosystem as modified by some anthropogenic passage barrier).

Current WDWF policy does not allow for the creation of passage around natural barriers, but some of the literature reviewed may include the effects of passage around natural obstructions (e.g., waterfalls, and changes in channel configuration imposed by landslides).

Five fish passage related subactivity types are currently recognized:

- A **culvert** is used for conveying waters of the state through fill. The "fish passage" discussion of the effects of culverts analyzes the effects of culvert removal, replacement, or retrofitting specifically for fish passage. Culverts that are meant to convey water and not fish are considered Water Crossings.

- **Fish ladders and fishways** are artificial structures that are used to provide passage through, over, and/or around artificial barriers (e.g., culverts, flumes, and/or dams). The artificial barriers themselves are discussed under Flow Control Structures.

- A **weir** is a low dam, usually with water flowing over the top. They can partially or fully span the channel. This discussion covers weirs used to prevent, facilitate, or manage the passage of fish, and includes weir-type fishways. Weirs for purposes of flow control and water diversion are discussed under Flow Control Structures.

- **Roughened channels** are intentional changes in channel configuration designed to facilitate the passage of adult and juvenile fish.

- **Trap-and-haul** operations involve three steps: (1) the capture of fish within some type of permanent structure, such as a weir or dam with an integrated trap structure, or using a temporarily placed trap device (e.g., a screw trap for capturing smolts); (2) the transport of the fish in a truck or barge to an upstream or downstream release point; and (3) release into the aquatic environment.

4.4.7.1 Culverts

To improve fish passage, culverts may be removed, replaced, or retrofitted. The distinctions between removal, replacement and retrofitting are:

- Removal – Complete removal of a culvert in conjunction with decommissioning of the roadway or flow control structure, or replacement of the culvert with a bridge.

- Replacement – Replacement of a culvert with a design that accommodates fish passage, using one of the approaches described below.

- Retrofit – Modification of an existing culvert with baffles, internal weirs, or similar structural elements to enhance fish passage.

Current WDFW guidance focuses on three culvert design options (Bates et al. 2003): the no-slope option, the hydraulic design option, and the stream-simulation option. The stream-simulation and no-slope options have emerged as the agency's preferred approaches for providing fish passage for most new culverts and culvert replacement projects. The hydraulic design option is not favored for removal and replacement but is applicable where an existing culvert is being retrofitted to improve fish passage. WAC 220-110-070 currently recognizes only the no-slope and the hydraulic design options.

4.4.7.1.1 No-Slope Option

The no-slope design option is employed in low-gradient channel environments, which allows for the culvert barrel or box to be placed at a zero slope. No-slope designs incorporate culverts of sufficient dimensions to support the accumulation of bedload within the structure at a natural channel slope, allowing the channel to maintain some degree of natural function. In ideal circumstances, channel morphological features such as gravel bars and a thalweg will form inside the culvert.

The no-slope option can be applied only to culvert replacements and new culvert installations; it is not applicable in retrofit scenarios. It is expected to provide unhindered passage for a broad range of aquatic species and life-history stages, provided that design objectives are met. Specifically, fish passage is expected to be provided when the culvert supports accumulation consistent with the natural upstream and downstream channel gradient, promoting the formation of natural channel features within the structure.

A no-slope culvert has the following characteristics:

- The culvert width is equal to or greater than the average channel bed width at the dimension where the culvert meets the streambed.
- The culvert is set at a flat gradient (i.e., zero slope).
- The downstream invert[3] is countersunk below the channel bed by a minimum of 20 percent of the culvert diameter or rise.
- The upstream invert is countersunk by a maximum of 40 percent of the culvert diameter or rise.
- There is adequate flood capacity.

The no-slope design option is usually applicable in the following situations:

- New and replacement culvert installations in low-complexity settings
- Low to moderate natural channel gradient (generally <3 percent slope)
- Site conditions that permit culvert width of at least 1.25 times the natural channel width upstream of the structure
- Shorter length culverts

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[3] A culvert "invert" is the bottom of the culvert. (Bob Barnard, personal communication December 8, 2008)

- Complex passage requirements for a range of species and life histories
- The likelihood of upstream headcutting can be avoided.

The upper limit for application of the no-slope option is at sites where the product of the channel slope and the culvert length does not exceed 20 percent of the culvert diameter or rise. The method can be applied with a certain degree of flexibility around these limits, provided the necessary hydraulic engineering expertise is available to account for the implications of constricting the upstream end of the culvert with the accreted bed or by installing a larger culvert. A typical no-slope option culvert configuration is shown in Figure 4-3.

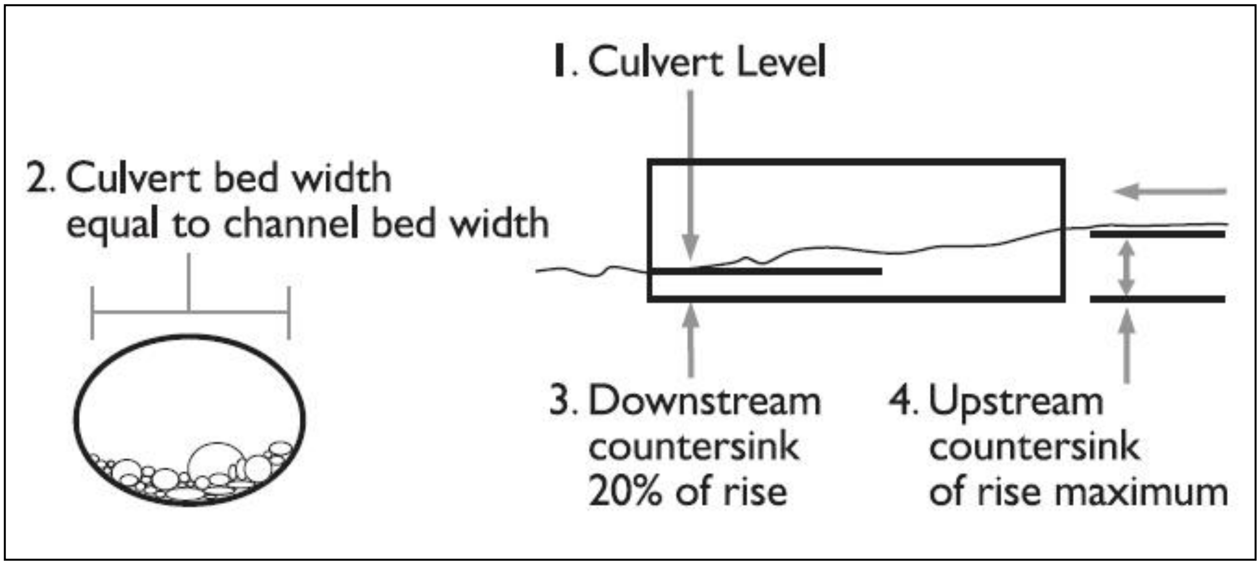


Figure 4-3. Profile and cross section for a typical no-slope culvert design (source: Bates et al. 2003).

4.4.7.1.2 Hydraulic Design Option

The hydraulic design option is used to design a culvert structure based on the swimming abilities of specific target fish species and age classes. The hydraulic design option can be applied to retrofits of existing culverts as well as to the design of new or replacement culverts, although the latter case is increasingly rare. Hydraulic design option culverts may employ features such as baffles, internal weirs, or other features that create the roughness necessary to promote fish passage.

Generally, the hydraulic design option may be employed in the following situations:

- New, replacement, and retrofit culvert installations
- Low to moderate culvert slope without baffles
- Moderate culvert slope with baffles (as retrofit)
- Target species have been identified for passage.

This design option requires a high degree of expertise in hydraulic engineering and hydrologic and geomorphic modeling capabilities, thorough understanding of the swimming performance and biological requirements of the target species, and site-specific survey information. Historically, this method was the standard approach used to design culverts for fish passage. It has become less favored, however, because of uncertainty related to fish passage performance, a limited range of applicable settings, and a number of ecological limitations. Specifically, the passage requirements of many target species are poorly understood, which contributes to design uncertainty. Even when the passage requirements of target species are adequately addressed, the structure may fail to provide passage for nontarget species. This may lead to a range of unforeseen ecological consequences. Finally, this type of structure may not provide adequate transport of sediment and organic material, contributing to broader effects on ecosystem function and declining performance over time.

Because of these limitations, the hydraulic design option is most commonly used for temporary retrofits of existing barrier culverts in circumstances where replacement or removal is not practicable in the immediate future. A typical hydraulic design option culvert schematic is shown in Figure 4-4.

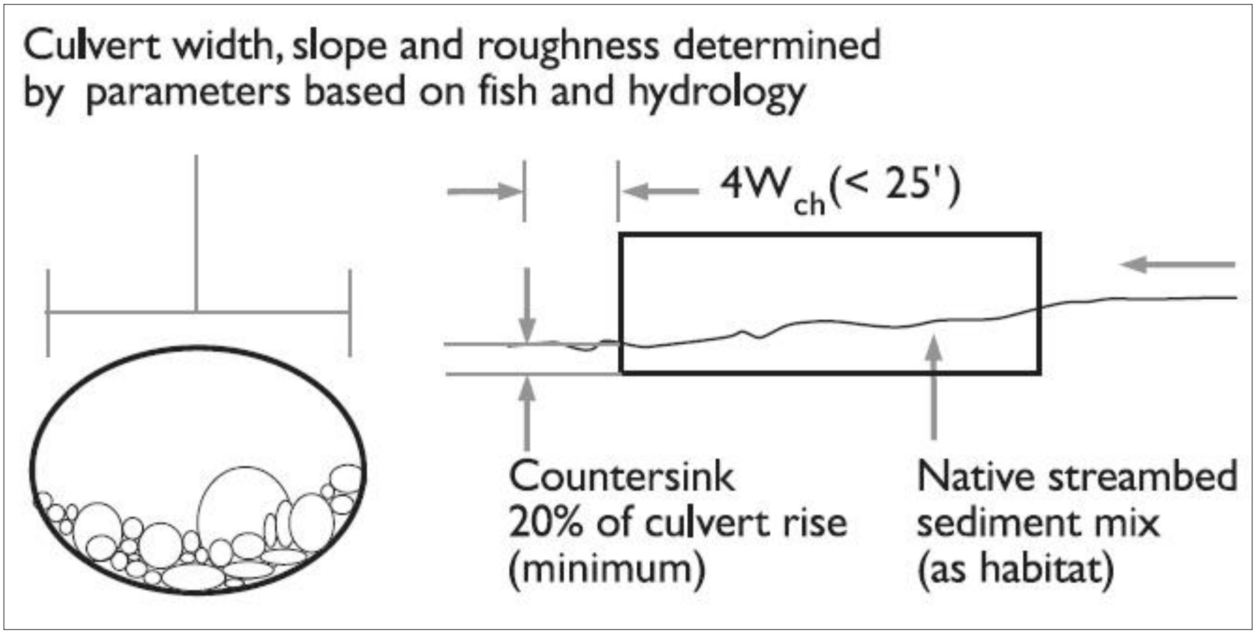


Figure 4-4. Profile and cross section for a typical hydraulic design option culvert, employing native sediment materials (source Bates et al. 2003).

4.4.7.1.3 Stream-Simulation Option

The stream-simulation option is similar to the no-slope option in that it attempts to mimic the natural streambed form to the greatest extent possible. Unlike the no-slope option, however, the culvert is installed at a slope matching or near the upstream channel gradient. This allows bed simulation to be employed over a broader range of gradients. The structure is placed at or near the natural channel slope and incorporates natural substrate features that mimic the streambed, provide for fish passage, and are transparent to the transport of sediment, wood, and organic debris.

Generally, the stream-simulation option is an appropriate method in the following circumstances (Bates et al. 2003):

- New and replacement-culvert installations.
- Complex settings, including sites with moderate to high natural channel gradient, sites requiring long culverts, or narrow stream valleys.
- Locations where passage is required for a broad range of aquatic species.
- Systems where passage must be provided for species with poorly understood requirements.
- Ecological connectivity (i.e., transparency to downstream transport of wood, sediment, and organic material) is required.

Culverts designed to simulate streambeds are sized wider than the channel width, and the bed inside the culvert is sloped at a similar or greater gradient than the upstream channel stream reach (i.e., no more than 125 percent of the upstream gradient). This type of culvert is filled with substrate material that emulates the natural channel, erodes and deforms similar to the natural channel, and is unlikely to change grade unless specifically designed to do so. This design method is intended to allow for minor adjustments in response to changes in upstream and downstream channel dynamics. The most basic stream simulation culvert is a bottomless culvert placed over a natural streambed. More complex designs may involve substrate intermixed with immobile bedform elements (e.g., boulders) to maintain bed conditions within the structure. Typical low-gradient and high-gradient stream-simulation schematics are shown in Figures 4-5 and 4-6.

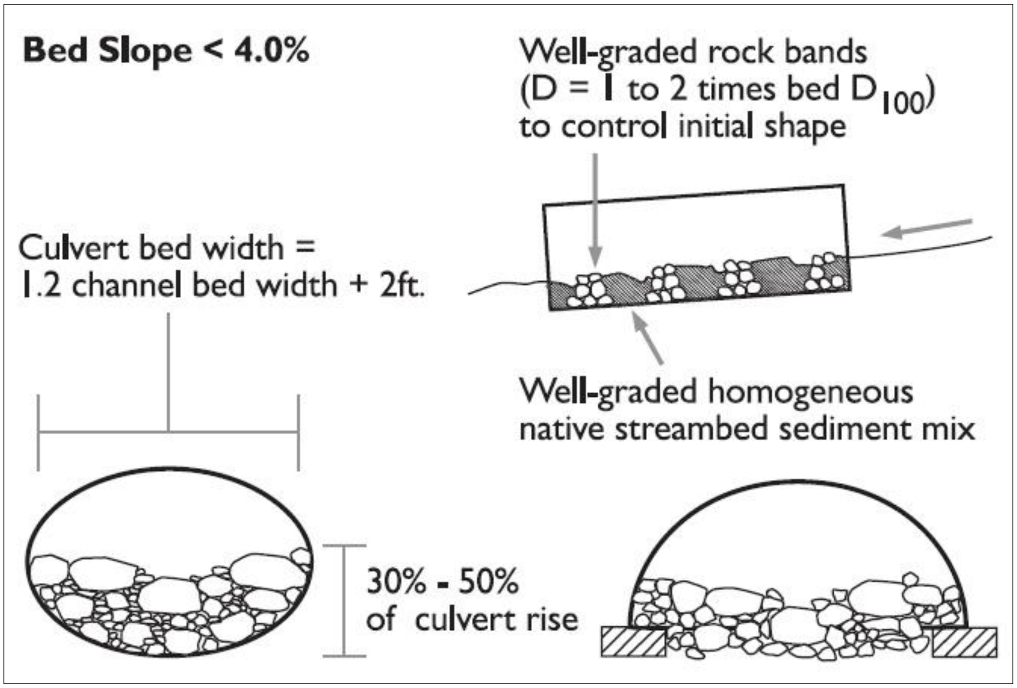


Figure 4-5. **Profile and cross sections for typical stream simulation option culverts for low to moderate gradient settings (less than 4 percent slope) (source: Bates et al. 2003).**

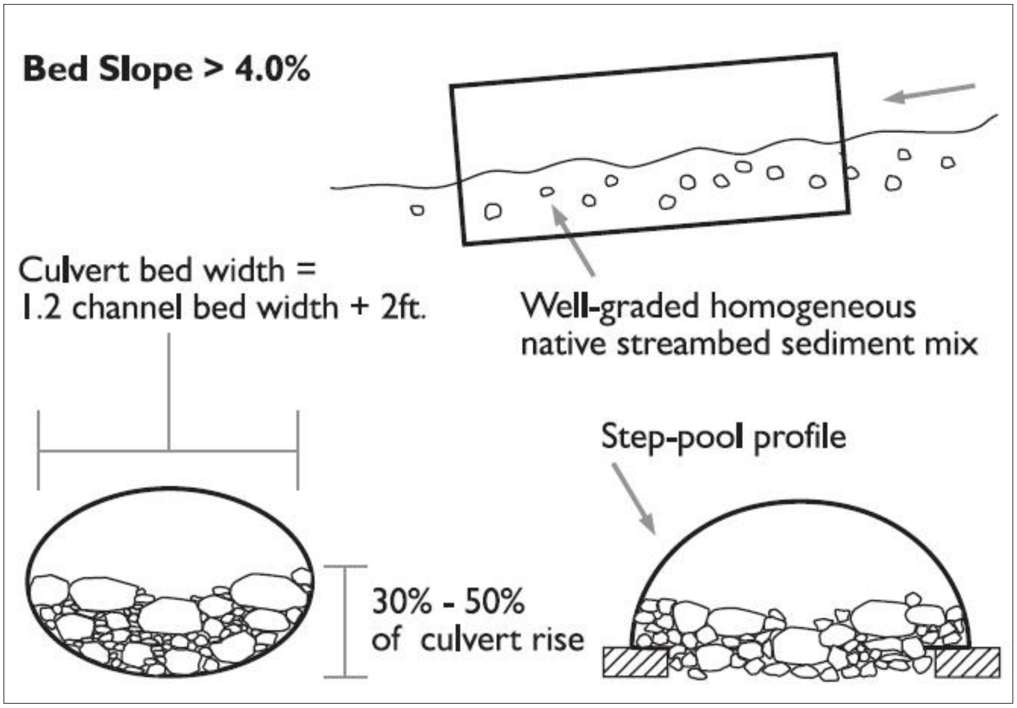


Figure 4-6. **Profile and cross sections for typical stream simulation option culverts for higher gradient settings (greater than 4 percent slope) (source: Bates et al. 2003).**

4.4.7.1.4 Gated Culverts

Gated culverts are culverts with a flap gate on one end that prevents the backflow of water through a channel modification, such as a dike or a levee. They are used to prevent landward inundation caused by high streamflows or tidal fluctuations, and are often employed in agricultural settings to aid in the draining and conversion of floodplains for human uses. Culverts of this type in tidally influenced environments are referred to as tide gates, which can affect both riverine and marine (i.e., estuarine) habitat types. In riverine environments, gated culverts are referred to as flap or flood gated culverts.

A gated culvert system is considered to be a flow control structure and is also discussed under that heading. New tide gates and flap gated culverts can be designed to allow for some degree of fish passage, and existing structures can be retrofitted for this purpose. For example, self-regulating tide gates (SRTs) include a flap gate fitted with a float system that allows the gate to remain open to backflow for specified periods, thereby providing improved fish passage in exchange for permitting some inundation of upstream lands.

4.4.7.2 Fish Ladders and Fishways

Fish ladders and fishways are artificial structures that are used to provide passage through, over, and/or around artificial barriers (e.g., culverts, flumes, and/or dams) or natural barriers (e.g., waterfalls) (although current WDWF policy does not allow for the creation of passage around natural barriers). Typically, fishways incorporate a sloping channel partitioned by internal weirs, baffles, or vanes with openings for fish to swim through. The sloping channels are designed to flatten hydraulic gradients and velocities to create conditions that target fish species can successfully navigate (Katopodis 1992). Examples of fishway designs include vertical slot, baffled, and weir type structures.

4.4.7.2.1 Vertical Slot Fishways

Vertical slot fishways incorporate a sloping channel partitioned by baffles spaced at regular intervals throughout the structure with passage through a vertical slot between the baffles. The hydraulic shadows behind the baffles provide refuge areas where organisms can rest before attempting to navigate the high-velocity flows in the slots between the baffles. A schematic of a typical vertical slot fishway design is shown in Figure 4-7.

A key advantage of this type of design is the ability to function over large variations in water levels. The design presents a number of disadvantages. This type of structure is limited to applications having slopes of 10 percent or less. The design also tends to produce uniform internal velocities between the baffles that may limit the passage of smaller or juvenile fish species. Finally, this design is also prone to sediment and debris accumulation, requiring routine maintenance.

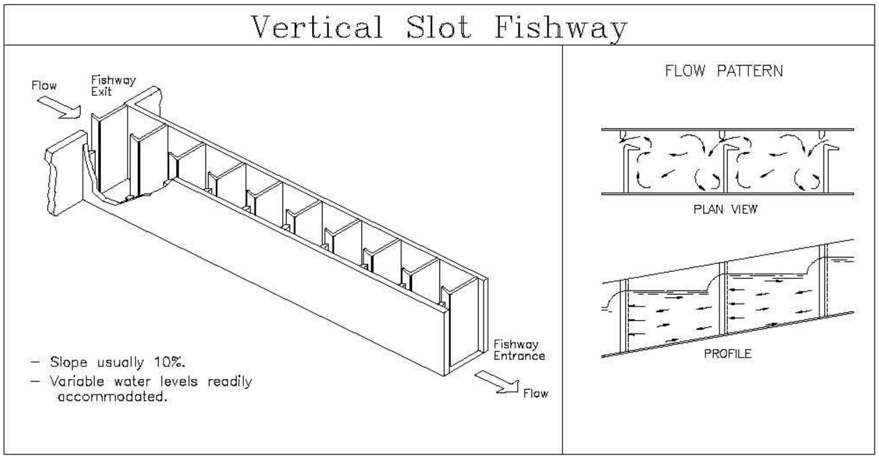


Figure 4-7. Typical vertical slot fishway (Source: Katopodis 1992).

4.4.7.2.2 Baffle Type Fishways

Baffle type fishways include a range of design types applied in a variety of settings. These include placement of baffles within existing culverts or other structures to address sheet flow (depth) and velocity barrier conditions, as well as structures designed to provide passage over man-made or natural vertical drop barriers. The Denil fishway is an example of the latter category. They incorporate a rectangular chute with a series of uniform, closely spaced baffles or vanes along the sides and bottom. Flow through this type of structure is turbulent with high energy dissipation, reducing the need for resting pools and similar features. A schematic of two typical baffle type fishway designs is shown in Figure 4-8.

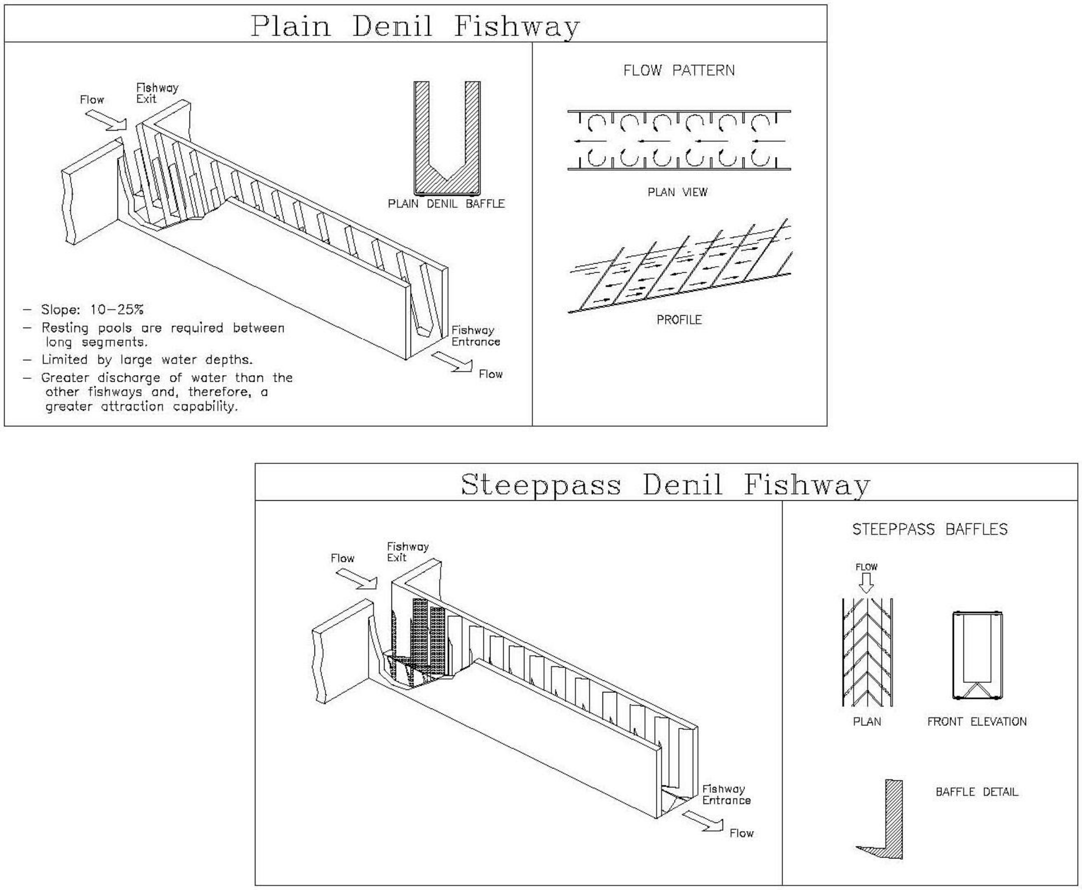


Figure 4-8. Typical baffle-type fishways (Source: Katopodis 1992).

The advantages provided by baffle type fishways vary depending on the environment in which they are employed. For example, barrier culverts are often retrofitted with baffles to create a fishway-like environment that improves passage conditions. However, this type of structure may not provide passage for the full range of species and life-history stages present under all relevant flow conditions. Moreover, retrofitted structures commonly require increased maintenance to maintain passage performance. As such, baffle retrofit projects are typically permitted only as an interim remedy until a long-term solution can be developed.

Alaskan steep pass fishways and similar designs are typically implemented in environments where artificial barriers such as dams and weirs present a vertical barrier to fish passage. This type of fishway can provide passage at gradients of up to 20 percent. Because they don’t require resting pools, these compact designs can be implemented in settings where space is limited. The relatively high flow capacity provides good attraction flows and creates turbulent conditions inside the structures that discourage sediment accumulation.

A key disadvantage of Alaskan steep pass and similar baffle type fishways is that they may not provide passage for a broad range of species and life-history stages. Because these structures create high flow velocities and turbulence, fish using them must swim constantly. This may hinder passage of smaller or juvenile fish with weaker swimming performance. This shortcoming may be overcome by the inclusion of resting pools, but these features would negate the advantages of compact size and low maintenance. Baffles and vanes commonly employed in this type of structure are also prone to sediment debris accumulation that can interfere with passage performance. In some cases, baffles can accumulate large debris that can overload and damage the structure. This leads to maintenance requirements and structural failure risk that negate to some degree the general benefit of limited coarse sediment accumulation.

4.4.7.2.3 Weir Type Fishways

Weir type fishways consist of a rectangular chute with weir-separated pools of uniform length arranged in a stepped pattern. This creates a long, sloping channel that gradually steps down the water level. This is the oldest type of fish ladder design. A schematic of a typical weir type fishway design is shown in Figure 4-9.

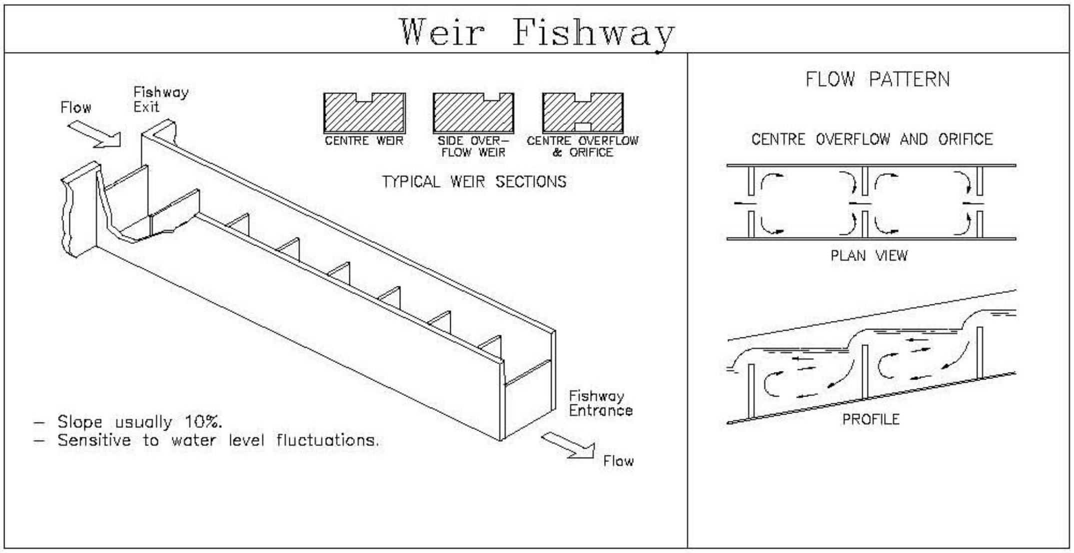


Figure 4-9. Typical weir-type fishway (Source: Katopodis 1992).

A weir type design requires fish to leap over the weir separating each step pool in the structure. However, an advantage of this type of structure is that the weirs can be configured with multiple notches and orifices to provide passage for fish of different sizes and life-history stages. The pools between weirs provide resting areas that enhance passage.

The weir type design presents several disadvantages. This structure is limited to sites permitting design slopes no greater than 10 percent. Fishway function is sensitive to water level fluctuations, and performance varies depending on flow rates. Flow occlusion may lead to dewatering and fish stranding. The pools between the weirs are prone to sediment aggradation, meaning that maintenance requirements are more extensive than for other fishway designs. In addition, stoplogs commonly used as design elements in weir type fishways have a tendency to become dislodged, requiring adjustment or replacement.

4.4.7.3 Roughened Channels

Roughened channels are specifically designed to facilitate the passage of adult and juvenile fish. They may be used to provide passage around abrupt hydraulic drops (i.e., falls or cascades) or uniform channels with high-velocity flows lacking refuge areas. The roughened channel typically moderates gradient and provides sufficient hydraulic complexity to allow for fish to pass the obstruction. Roughened channels have been seen as a more aesthetically pleasing and "natural" way to pass fish around a barrier than fishways, particularly in areas where land adjacent to a barrier is available and inexpensive. Generally speaking, the channel bottom is comprised of naturally occurring or processed quarry rock that is rounded and of sufficient size to ensure stability of the channel. Sometimes designers of these channels impose channel structure by designing the channel with a series of small steps or meanders, although this is not always the case.

Roughened channels can be broken into two broad categories:

- Roughened channels that exploit an existing channel or side channel.

  Advantages: Can utilize existing streamflow without diversion.

  Disadvantages: Large impact on existing ecologic communities.

- Roughened channels that are constructed through an upland area where no channel existed before.

  Advantages: Limited or nonexistent impacts on existing ecology.

  Disadvantages: More difficult to design; more prone to hydraulic and geomorphic effects that may lead to decreasing function over time.

Roughened channels are currently in the experimental stage of development. While conceptually simple, the design of a roughened channel is not as straightforward as one might expect. Natural channels are the result of extended periods of geomorphic evolution. The structure within the channel (riffle-pools, step-pools) is in dynamic equilibrium with material both underlying the channel and sediment supplied from upstream. Duplicating these relationships in an engineered or other design is extremely difficult. Also, natural channels often have riparian vegetation on the immediate edges of the channel, while newly constructed roughened channels must remove some vegetation in the vicinity of the channel to construct it. This may compromise the utility of the roughened channel for aquatic species. Roughened channels, particularly those sited in upland areas, can have longitudinal slopes that are out of equilibrium with respect to water flow and sediment supply. If out of balance with these factors, channels either erode or aggrade (accumulate) sediment such that they ultimately block fish passage and have negative impacts on the geomorphic character of adjacent water bodies. In addition, the coarse sizes of rock often used in roughened channel projects can initiate unnaturally high groundwater recharge, causing subsurface flow conditions during drier periods, limiting habitat capacity, presenting the potential for stranding and mortality of aquatic organisms, and posing barriers to fish passage.

4.4.7.4 Weirs

The term “weir” applies to a number of different structure types that are intended to serve a variety of purposes. Weirs include:

- Large channel-spanning structures, typically made of concrete, such as hatchery weirs. The effects of this type of weir are addressed under Flow Control Structures.

- Weirs installed in natural channels (e.g., a series of step pools formed by grade control structures composed of natural or man-made materials) to restore channel bed profiles and enhance passage.

- Weirs used to provide passage through barrier culverts. A series of weirs may be used to provide access into the culverts isolated by outfall drops, or to backwater culverts that present velocity or depth barriers.

- Fish passage control weirs constructed of natural or man-made materials intended to prevent upstream dispersal of invasive species or of hatchery fish that might produce detrimental effects should they spawn in the wild. These structures may integrate electrical barriers to increase the selectivity of fish passage management.

- Temporary or movable weirs, such as smolt panels, fence weirs, and similar structures intended to control upstream and downstream fish migrations, or to facilitate the counting of adult fish returning to spawning grounds.

4.4.7.5 Trap and Haul

Trap-and-haul activities are specialized operations involving the capture of fish for transport around fish passage barriers such as flow control structures. Trap-and-haul activities involving adult fish are often conducted at a dam, weir, or similar form of flow control structure that allows for the control of attraction flows used to direct fish into capture areas. Juvenile fish may be captured at similar structures, or may be captured using equipment and techniques (e.g., beach seining, rotary screw traps) that do not have a lasting physical effect on the environment.

Most of the relatively limited number of trap-and-haul facilities operating in Washington State are associated with a dam. Examples include trap and haul of sockeye salmon around barrier dams on the Baker River (Skagit River system) and downstream barge transport of migratory salmon smolts.

In one case in Washington, trap and haul activities are conducted at the base of a natural falls. Salmon, steelhead, and native char are released annually above Sunset Falls on the Skykomish River.

4.4.8 Fish Screens

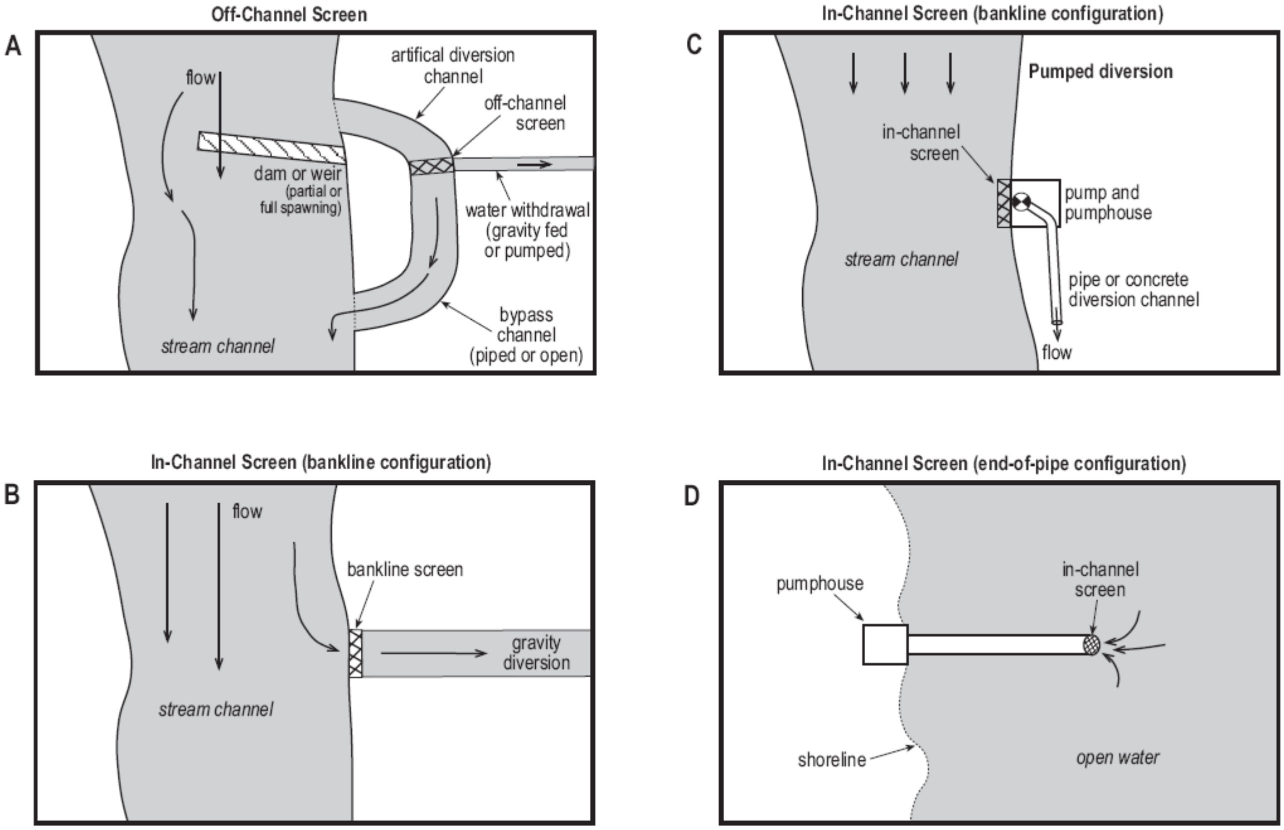
Fish screens range from small, temporary structures used on a seasonal basis to large, permanent structures associated with agricultural diversions or industrial or municipal water intakes. They are divided into in-channel screens and off-channel screens. Some screen designs can be used in either in-channel or off-channel configurations. Fish screens are continuously operating structures so long as the water intakes or diversions they are associated with are withdrawing water.

Summary descriptions of these types of screen designs are based on information in the most recent design guidance from WDFW and NMFS (NMFS 2004; WDFW 2001a).

In-channel screens are used as permanent, seasonal, and temporary screens in rivers, lakes, reservoirs, estuaries, and marine waters. These include “end-of-pipe” style screen systems and bankline screen designs.

Off-channel screens include both temporary and permanent screen systems located off of the main stream channel, adjacent to or downstream of the flow control structure providing the diversion, within artificially constructed irrigation canals or similar off-channel diversions.

Typical configurations for these two subactivity types are shown in Figure 4-10.



In riverine environments, fish screens are employed on both gravity-fed (A & B) and pumped (C) diversion systems. In marine environments, intake systems are commonly constructed on the shore and pipelines extend out into the open water (D). In lacustrine environments, several configurations are possible including those shown in C and D, with type D being favored in most circumstances.

Figure 4-10. Typical in-channel and off-channel screen configurations.

4.4.9.1 In-Channel Screens

In-channel screens are used in various environments, ranging from small lakes and streams to off-shore marine, riverine, or lacustrine environments. In-channel screens operate effectively only when fully submerged and intake flow is distributed over the entire surface, meaning that debris accumulation or partial exposure will reduce screen effectiveness. In smaller streams, lack of water depth necessary to fully submerge the screen and the intake system may also limit the effectiveness of the screen.

In-channel screens include both end-of-pipe configurations and bankline screens.

4.4.8.1.1 End-of-pipe Configurations

End-of-pipe screens, also referred to a pump screens or intake screens, are placed at the mouth of an intake pipe or outfall outlet to prevent the movement or entrainment of fish into or out of the intake or outfall.

End-of-pipe structures do not have a flow control device between the screen and the source body. The scale of end-of-pipe screens varies widely. On the small end of the scale, end-of-pipe structures can include small, relatively simple structures on temporary diversion pumps used for small water withdrawals, such as wire mesh screens on small temporary or permanent intake pipes for private water systems. On the large end of the scale, end-of-pipe structures can include elaborate screen systems on large water diversion structures such as hydropower penstocks; industrial, municipal or agricultural water intakes, or spillway outlets. Many different screen configurations are commercially available that are consistent with current screen guidance (WDFW 2000, 2001a).

Regardless of the nature of the screened structure, the predominant in-channel end-of-pipe screen is a barrier composed of an intake covered by perforated metal, wire, mesh, or some other permeable material. The designs are typically intended to limit organism entrainment and to diffuse the intake flow velocity to reduce the potential for impingement. However, in certain cases, screens that incorporate sharp metal grids, grinders, or similar features that are purposefully designed to entrain and kill fish or other organisms may be employed in the outlet structures of flow-controlled lakes or reservoirs to prevent the downstream dispersal of undesirable exotic species.

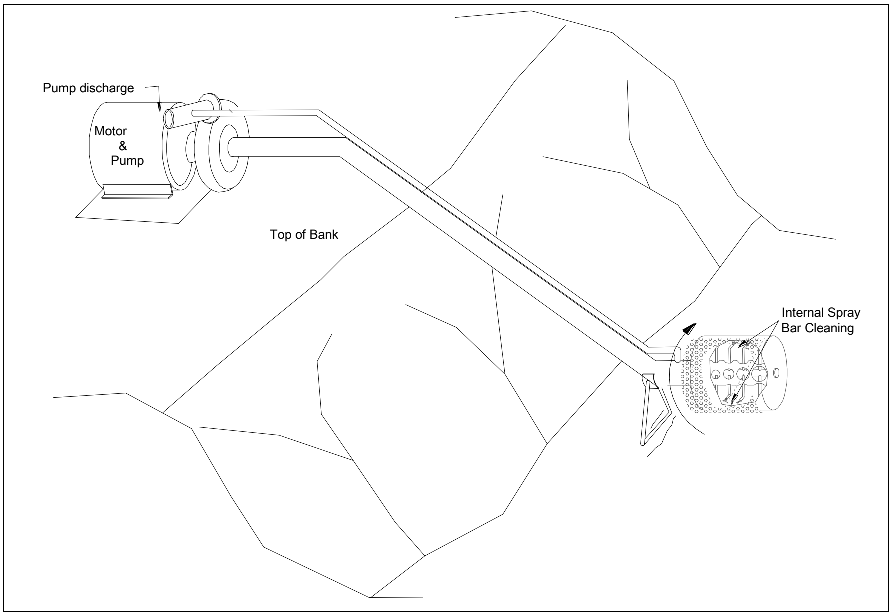
End-of-pipe screens are typically built in a chamber configuration, typically in a box or cylindrical shape, and attached to the end of a pipe. Smaller designs range in capacity from less than 1 cubic foot per second (cfs) intake capacity for small irrigation pumps, to larger designs for intakes with a capacity of 50 cfs or greater. Screen configurations vary depending on the application, with fixed drum and tee-screen designs being common. Removable pump screens associated with temporary diversion systems also fall into this category. Some models for small screens (up to 5 cfs) have extremely efficient water jet clearing systems. Small end-of-pipe screens are commonly used in conjunction with temporary diversion pumps. This type of system is in widespread use in Washington State.

Industrial or municipal water intake systems and power plant cooling water intakes commonly employ large end-of-pipe style screen systems. This type of system is used in association with high-capacity intake systems. They commonly incorporate an air-burst or water jet debris-clearing mechanism. Large end-of-pipe style screens are typically integrated into the mouth of the intake structure. This type of screen system is commonly used in lacustrine and marine environments. In these settings, the intake and screen system are usually located in deeper water away from nearshore areas used by sensitive organisms.

The advantages provided by end-of-pipe screens are that they are functional for both deep and shallow water intake systems. The disadvantages are primarily associated with the clearing of debris. End-of-pipe screens require sufficient ambient water velocity to carry debris away from the screen. Air burst clearing systems, the most common system used with in-channel screens, may not adequately remove debris accumulations, especially from the bottom of the screen. For HCP species, this is only problematic when debris accumulation decreases intake diffusion to the point that risk of impingement results. Otherwise, debris accumulation is a problem only for the water user.

Example schematics of end-of-pipe fish screens are shown in Figure 4-11.

a) Self-cleaning system, internal spray bar



b) Passive debris clearing system, T-screen

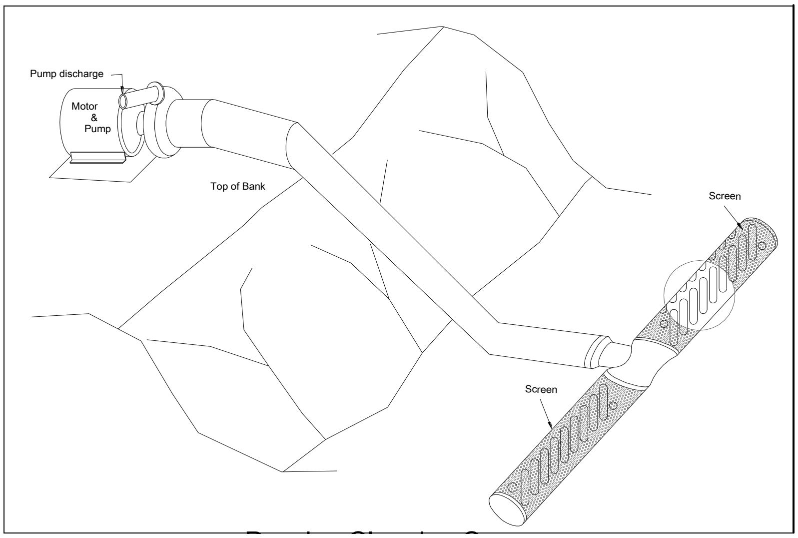


Figure 4-11. Typical end-of-pipe style fish screens with (a) self-cleaning and (b) passive debris clearing systems.

4.4.8.1.2 Bankline Configurations

Bankline screens are fixed or moving belt and panel type screen designs installed flush with the stream bank, providing a barrier between the diversion canal or gallery and the aquatic environment. Bankline screens differ from off-channel screens in that they lie between the source body and the diversion, without an intervening flow control structure such as a dam, weir, or artificial diversion channel.

Because of its location in the channel, construction and maintenance of this type of screen structure imposes a greater range of effects on the aquatic environment than a comparable off-channel screen system.

Screen designs used in bankline configurations include fixed and traveling screens, which can be installed in either a vertical or an inclined position. Fixed screens used in bankline configurations, whether installed vertically or inclined, typically have an air or water cleaning system on the back side of the screen to protect from in-stream debris. (Patrick Schille, WDFW, personal communication, March 2009). Examples of different types of fixed and traveling screens are discussed in more detail under off-channel screens in the following section.

Bankline screen systems do not always require an associated bypass channel. However, in certain circumstances, particularly when bankline screens are placed in sheltered embayments off the main channel, successful fish exclusion may require incorporation of pumped bypass systems. Sheltered embayments lack the necessary head loss to drive flow through a bypass, even if it is provided. In such cases, fish have no guidance away from the face of the screen, can become trapped within the screen chamber, and must be pumped or lifted into bypass systems and returned to the aquatic environment. However, bypass systems are more commonly associated with off-channel screen designs.

4.4.8.2 Off-Channel Screens

Off-channel screens are constructed in artificial diversions off the main stream channel. They are usually, but not always, integrated into or directly associated with a flow control structure such as a dam or a weir. They include an artificial bypass system, either a channel or a pipe, designed to return aquatic organisms and debris back to the main channel. These bypass systems must provide adequate sweeping flows to draw organisms safely past the screen, into the bypass and then discharge them safely downstream. Bypass systems must also pass debris without jamming.

In Washington State, the most common screen designs used in off-channel configurations include the following (WDFW 2001a, Schille 2008):

- Rotary drum screens
- Fixed plate screens (vertical and inclined designs)
- Vertical traveling screens (panel and belt types)
- Modular screens (rotating drum or vertical fixed plate).

4.4.8.2.1 Rotary Drum Screens

The rotary drum screen incorporates both screening and debris removal in a relatively simple configuration. Drum screens can be scaled to accommodate a variety of flows, and they are effective at avoiding impingement and entrainment of juvenile fish.

The rotary drum screen removes debris collected on its face through rotation, and the debris is washed off the screen on the downstream side. Screen rotation is achieved by an electric motor, paddle wheel, solar drive, or hydraulic motor. Its most common application is in open channel flow situations, such as irrigation ditches. Using single or multiple drum configurations, rotary drum screens can accommodate a range of diversion rates. In Washington State, they have been used to screen flows ranging from as low as a few cfs up to 3,000 cfs. Drum screens are typically used in conjunction with gravity diversion canals but can also be used to screen water drawn into a pumping gallery. A schematic of a typical rotary drum screen is shown in Figure 4-12.

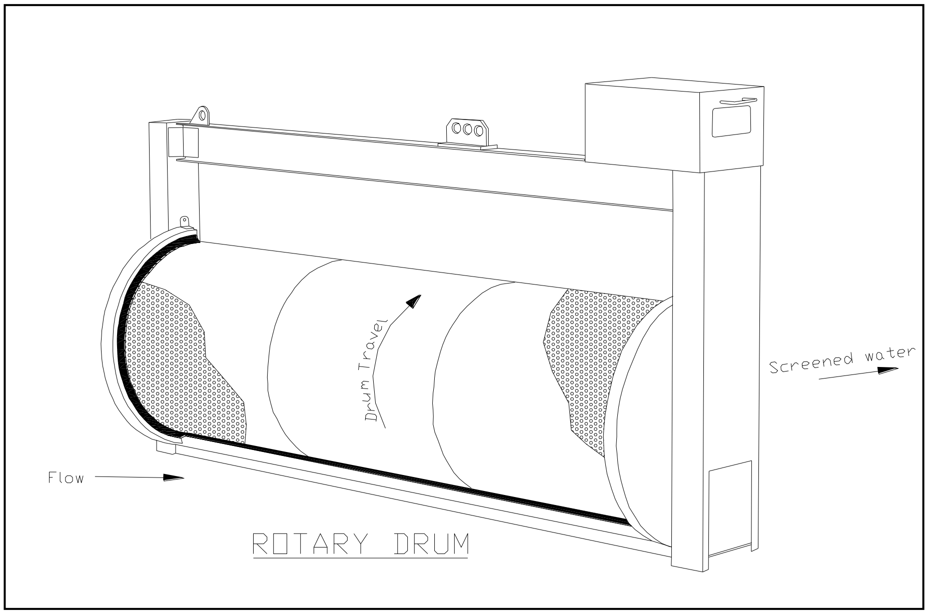


Figure 4-12. Rotary drum screen (Source: Schille 2008).

4.4.8.2.2 Fixed Plate Screens

Fixed plate screens include a variety of design types that are distinguished primarily by their orientation to flow. To suit site-specific design requirements, the screen can be oriented with a vertical, upward sloping, or downward sloping aspect relative to the direction of flow. This design is typically employed with gravity diversions, but it can also be used with pump intakes in certain configurations.

The vertical fixed plate screen, which is characterized by intake flow passing perpendicularly through a vertical screen surface, is commonly used for industrial, municipal, and agricultural water supply systems in the Pacific Northwest. This style can be used in either pump or gravity diversion intake configurations. The plate is commonly composed of punched metal or a profile bar, in either aluminum or stainless steel. Woven wire mesh is also used but is less typical, due to its tendency to accumulate debris that is difficult to clear. This design is relatively simple and tends to require less frequent maintenance because there are no moving parts or wear surfaces between the screen mesh and the structural frame. A major disadvantage to the design is that it does not passively clear accumulated debris readily. Typically, the design integrates a mechanical brush, hydraulic backspray, air burst, or some other type of debris-clearing system to overcome this limitation. A schematic of a vertical plate screen with a mechanical brush system is shown in Figure 4-13.

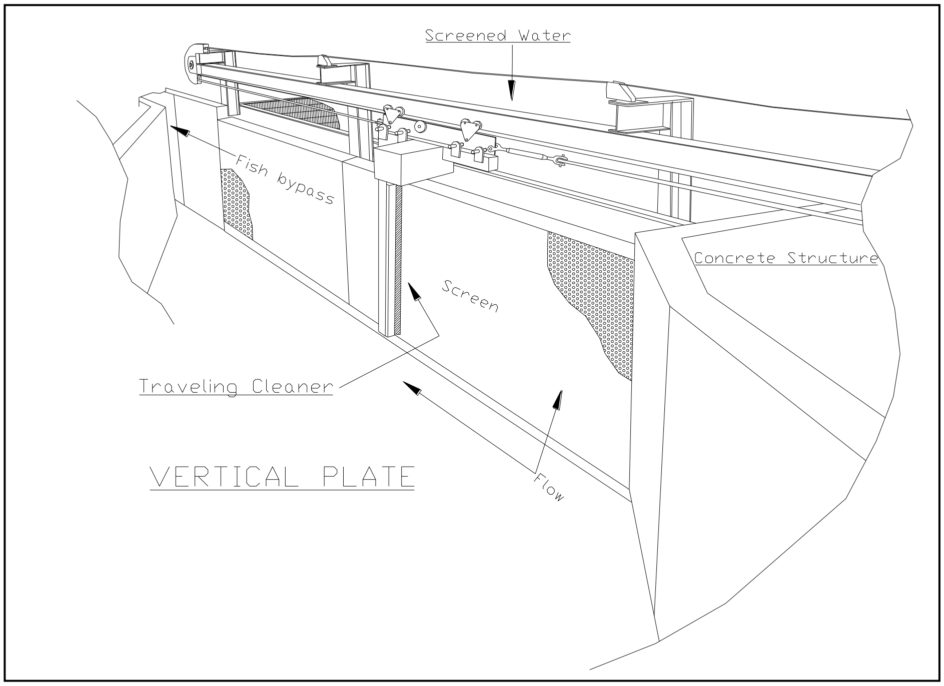


Figure 4-13. Vertical plate screen with a mechanical brush system (Source: Schille 2008).

Upward and downward sloping screens, much less commonly used, are characterized by diverted flow passing vertically through the inclined surface of the screen. They rely on large quantities of bypass flow to provide passive debris cleaning and avoid fish impingement. For this reason, they are used where the diversion rate is small, relative to total flow. Continuous streamflow across the surface of the screen sweeps fish and debris off the surface of the structure. These designs are typically used in conjunction with a gravity diversion. However, specific sloping screen designs may be paired with pumping galleries and incorporate a bypass channel to return water and fish back to the mainstem channel. Certain sloping screen designs, such as Eicher[4] screens, are used in hydropower systems to direct fish away from turbine intake systems.

Downward sloping screens can either be flat plate or contoured plate style designs (examples include the Coanda screen and the Farmers screen[5]). Water is directed from an impoundment created by a flow control structure (e.g., a small dam or weir) over the surface of the screen and into a bypass channel returning to the main channel. A portion of this flow passes through the screen and into the pump or gravity diversion. These designs are occasionally used in in-channel settings, but are most commonly used in off-channel configurations. An example schematic of a typical inclined plate screen is provided in Figure 4-14.

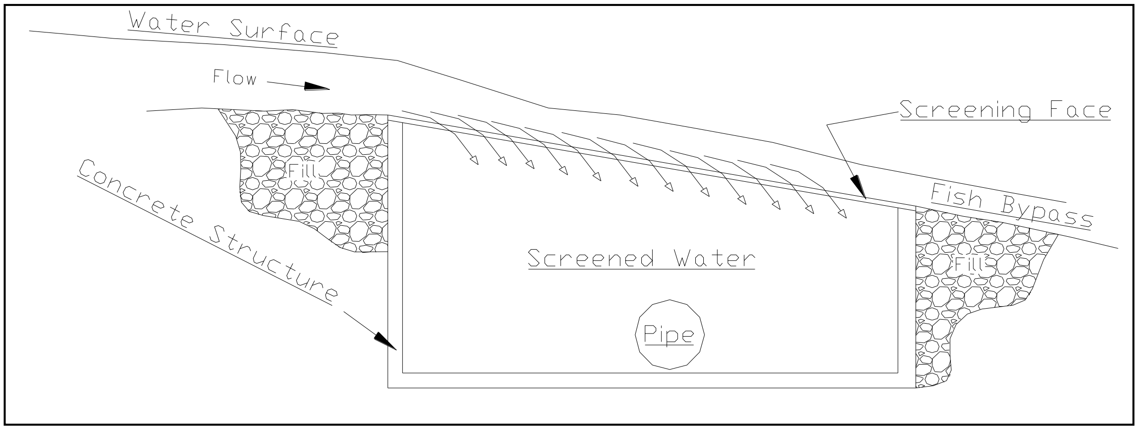


Figure 4-14. Typical inclined plate screen (Source: Schille 2008).

Upward sloping screens are quite similar in design except that their profile rises in the direction of the water flow. Excess water flowing over the top of the screen provides fish and debris bypass. Upward sloping screen designs do provide some degree of reliable passive debris clearance; however, they can become overwhelmed by large debris loads. This presents some risk of structural failure as the combined weight of water and debris may overcome the structural strength of the screen support frame. Active clearing systems are sometimes incorporated with these designs to reduce this risk.

The advantage of inclined plate screens is that there are no moving parts and they require no additional in-river diversion structures. Because this screen relies on passive hydraulics to clear debris and provide fish passage, it provides a reliability advantage over mechanical clearing systems. However, screen performance and reliability are highly dependent on precise flow control. Flow rates must be carefully balanced between the required rate of diversion and

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[4] Examples do not constitute a recommendation by the Washington Department of Fish and Wildlife.

[5] See previous footnote.

providing sufficient flow to clear debris and avoid fish impingement. Due to their sensitivity to debris and the need for consistent flow control to maintain performance, upward facing screen designs would typically not be permitted in Washington State (Schille 2008); however, the WAC does not specifically preclude their use, meaning there is some potential for such designs to be permitted in the future, and a number of legacy structures are in operation. Examples include Eicher screens integrated into hydropower dams and hatchery water system intakes. Because these structures may be maintained under existing or new HPAs, they are considered in this analysis. Inclined plate screen performance is sensitive to flow control, but they are less prone to debris accumulation and structural failure. Some newer downward facing screen designs, such as the contoured Coanda screen, are considered experimental and may be permitted in certain circumstances.

4.4.8.2.3 Vertical Traveling Screens

Vertical traveling screens are similar in concept to rotary drum screens in that the mesh of the screen cycles continuously to remove debris collecting on its face. Two design configurations are commonly used: panel-type screens, with individual mesh panels; and belt-type vertical traveling screens with a continuous mesh belt. Both types of screens are usually driven by electric motors and are commonly used in conjunction with pump diversions. A schematic of a vertical traveling screen design is shown in Figure 4-15.

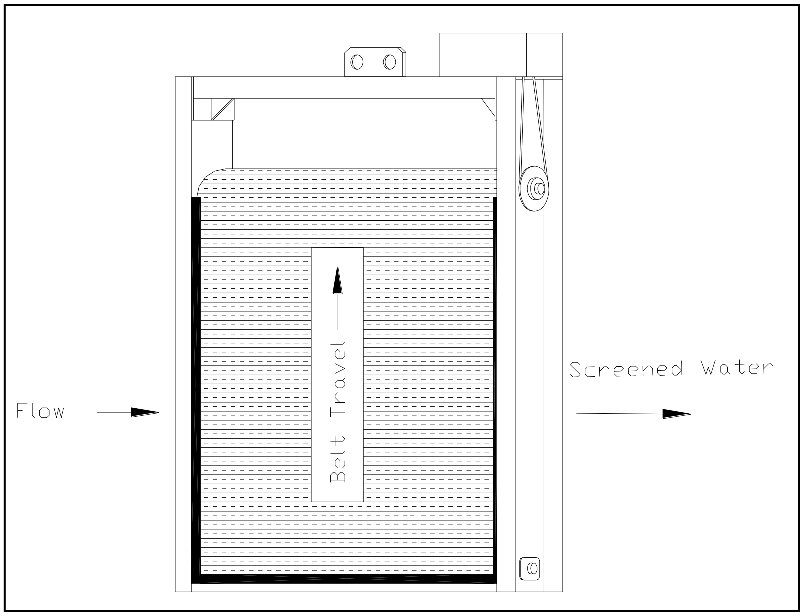


Figure 4-15. Vertical traveling screen (Source: Schille 2008).

The primary advantage of belt screens is they can be installed in deep water. The screen can be built to any length within the structural capacity of the frame and drive shafts. This design provides effective debris clearing across a range of water depths.

Other advantages of the vertical traveling screens are that they can be installed in bankline configuration (thereby requiring no bypass system); the associated foundation and frame are relatively compact; and they are self-clearing. Additional debris-clearing capacity can be added using jet spray or brush systems if needed.

Vertical traveling screens have historically been constructed with horizontal troughs, or ledges, built onto the face of the screen. The purpose of the troughs is to lift debris and fish with the screen as it rotates. A high-pressure spray bar, near the drive shaft, washes the debris and fish into a stationary trough on the deck of the structure. The debris can then be collected for removal. However, the troughs are problematic for fish protection. Fish entangled with debris and exposed to the spray bar prior to being deposited in the troughs may be injured or killed in the process. Once in the troughs, capture and removal may be difficult, increasing risk of stress and injury. Like upward facing plate screens, this type of screen system would typically not be permitted in Washington State today. However, the WAC currently does not preclude their use, meaning future permitted structures are possible, and a small number of legacy structures are in existence that may require permitting for future maintenance.

4.4.8.2.4 Modular Screen Systems

Modular screens are a recent addition to the suite of available screen design options (Schille 2008). Developed in the early 1990s by WDFW at their Yakima Screen Shop, various forms of modular screens are currently in wide use throughout the Pacific Northwest. The modular rotating drum and modular fixed plate systems are the most common forms. Originally designed for remote sites where conventional concrete construction was not feasible, modular screens can be assembled on site and installed in 1 or 2 days. They have proven to be an effective and inexpensive means for addressing numerous small, unscreened diversions. Schematics of the modular drum screen and the modular fixed plate screen are shown in Figures 4-16 and 4-17 respectively.

The modular drum screen is designed for diversions in the 2 to 6 cfs range. This type of system is typically employed in off-channel settings using a piped bypass system to channel fish back to their habitat. They are paddle wheel driven and can be fabricated to provide an angled orientation to flow. The plate screens were developed for diversions in the ½ to 3 cfs range and are used in both in-channel (i.e., bankline) and off-channel settings. The off-channel version uses rotating brushes driven by a paddle wheel to clear debris.

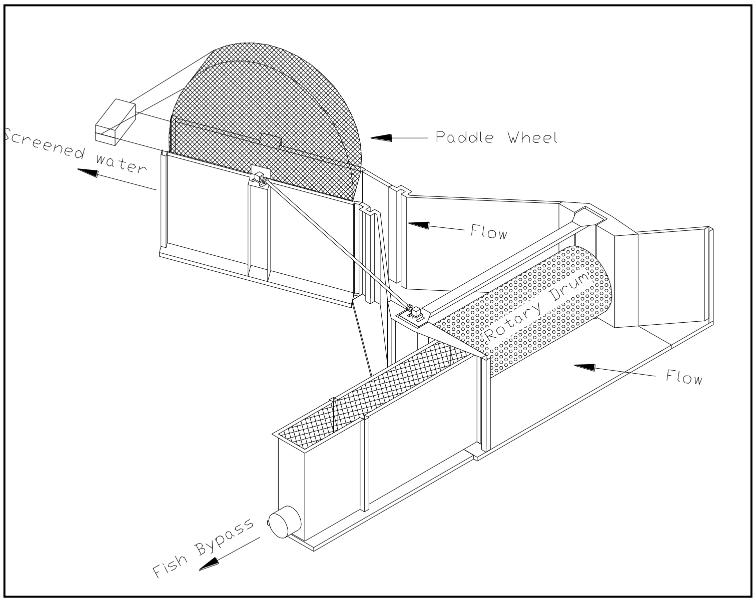


Figure 4-16. Modular drum screen (Source: Schille 2008).

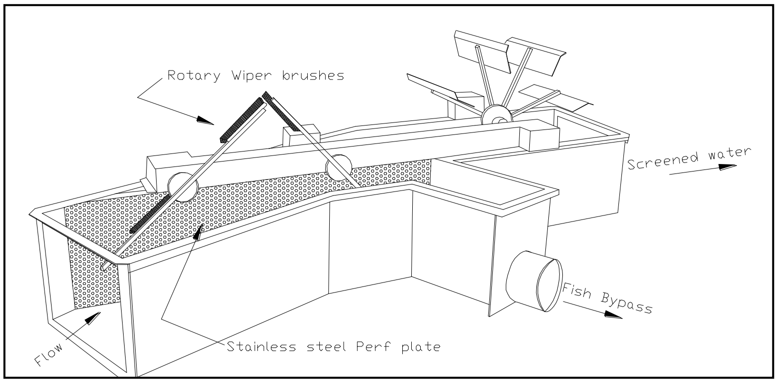


Figure 4-17. Modular fixed plate screen (Source: Schille 2008).

4.4.9

4.4.9.1 Additional Screening Systems

Other fish screen technologies are in use in Washington State that are not considered in detail. These include:

- Behavioral modification using environmental stimuli: Flashing strobe lights, underwater noise, or other forms of disturbance intended to induce avoidance of hazardous areas that are impractical to screen using traditional methods.

- Infiltration galleries: Intake pipes buried within the active channel that use the overlying alluvial bed material as a screen.

These types of screen designs represent a small proportion of the number of fish screen proposals submitted for approval under the HPA program. Behavioral modification using environmental stimuli is considered an experimental approach that is still in development. Infiltration galleries are infrequently used designs that represent a form of channel modification.

4.4.10 Flow Control Structures

Flow control structures include: dams, weirs, dikes and levees, outfalls, tide gates, and intakes and diversions.

Dams are structures built within a stream to control flow for flood control, divert flow for irrigation, or to utilize flow for generation of hydropower.

A **weir** is a low dam, usually with water flowing over the top. They are structures that can partially or fully span the channel for purposes of flow control and water diversion. Weirs are also used to prevent, facilitate, or manage the passage of fish, discussed under Fish Passage as “weir-type fishways” and as “weirs”.

Dikes and levees are built to maintain flows within a confined channel for flood control purposes, or are used to convert estuarine habitat into agricultural fields or freshwater habitat (e.g., used on WDFW lands and federal wildlife refuges to provide waterfowl habitat/hunting areas).

Outfalls move water from one place to another, typically into a body of water. They may convey irrigation water, stormwater, or other waste materials. Submerged outfalls open under water. They are most common in lakes and marine waters, often associated with municipal and industrial wastewater and stormwater discharges. Marine outfalls that emerge at intertidal elevations are also considered submerged outfalls, as they are submerged at least some of the time. Exposed outfalls open above water. They typically occur in riverine environments. Submerged and exposed outfalls are sometimes screened to prevent fish entering the outfall pipe and to prevent large debris from exiting the outfall.

Tide gates (also referred to as flood gates or gated culverts) are built to control tidal or floodwater inundation in low-lying areas. These structures are typically integrated into dikes and levees and are commonly used to drain river deltas and estuarine lowlands for conversion to agricultural or industrial uses. They allow water to drain from low-lying areas to marine or estuarine receiving waters while preventing the backflow of tidal or floodwater. In agricultural areas, tide gates prohibit salt water from entering croplands. In addition, tide gates lower the water table, pushing the anoxic layer deeper in the soil and promoting crop growth. Tide gates are commonly located at the mouths of streams or rivers where the estuary begins, or where tidal nonriverine channels drain ditches, fields, marshes, and small tributaries (Figure 4-18).

Tide gates come in many forms—from simple culverts through an earthen dike, to complex concrete structures that include deflecting walls and pilings both upstream and downstream of the structure (Figure 4-19). Associated with these structures are tide or flood boxes that restrict flow in one direction. Tide boxes can be either top-hinged or side-hinged and, depending on the type of gate, it will be open for shorter or longer times. The amount of time a gate is open is a function of the design, size, and weight of the tide box. The magnitude of tidal or floodwater fluctu

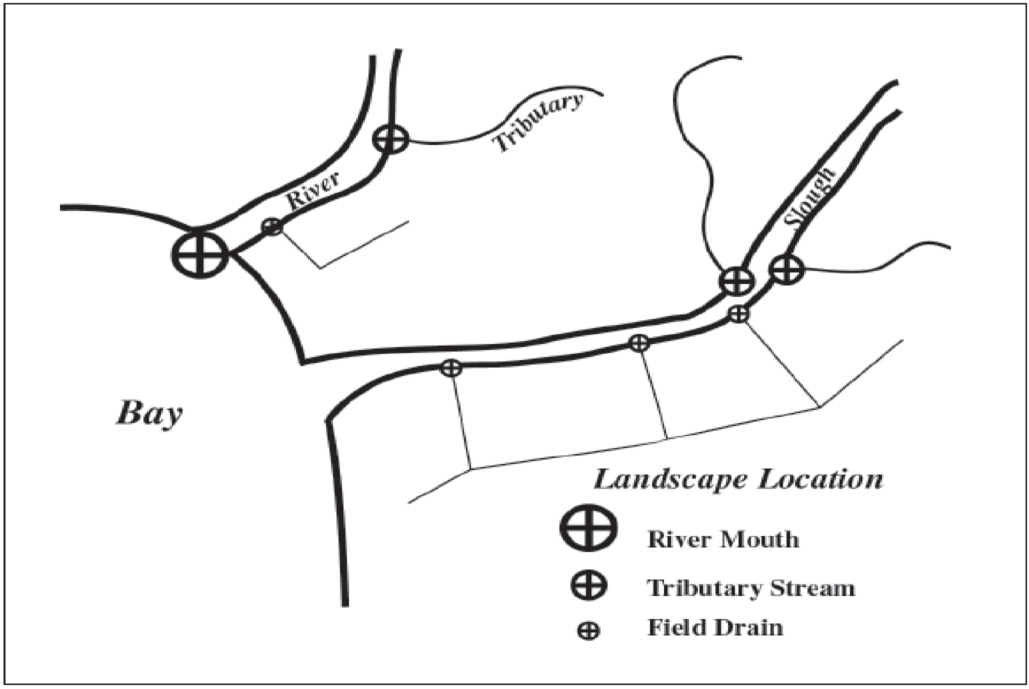


Figure 4-18. Common tide gate locations at the mouth of estuaries, tributary streams, and tidal nonriverine channels. Adopted from Giannico and Souder 2005.

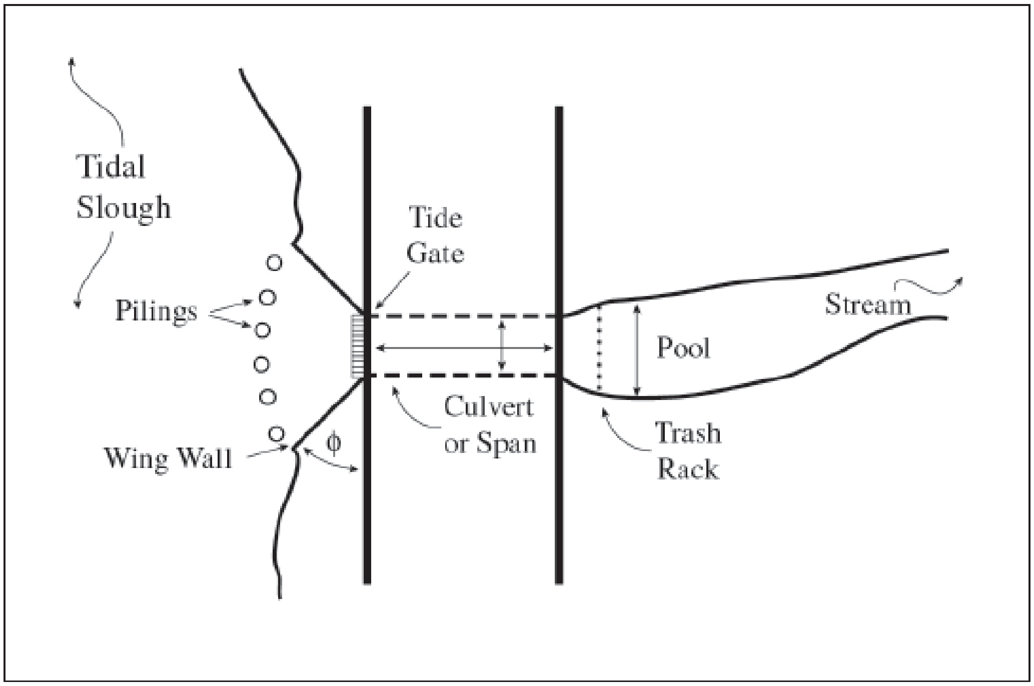


Figure 4-19. View of tide gate and supplemental features such as wing walls and pilings. Adopted from Giannico and Souder 2005.

Water intakes and diversion structures are used to divert water from a stream to another place, or to maintain water in an existing or new channel for flood control. Water diversion systems are built for a variety of reasons including, but not limited to: irrigation, domestic use, stock watering, hatcheries, power plants (hydropower, fossil fuel, and nuclear), water supply, general manufacturing, timber processing, and creation of fish habitat. Most diversion systems route water through a concrete channel and/or enclosed pipe.

Figure 4-20 illustrates general schematics of the most common forms of water diversion and intake systems. Diversion systems built in freshwater environments can either work by gravity or water pump, with gravity systems employed predominantly in riverine systems that provide the necessary head loss. For both, the diversion channel or pipe may run parallel or away from the stream channel.

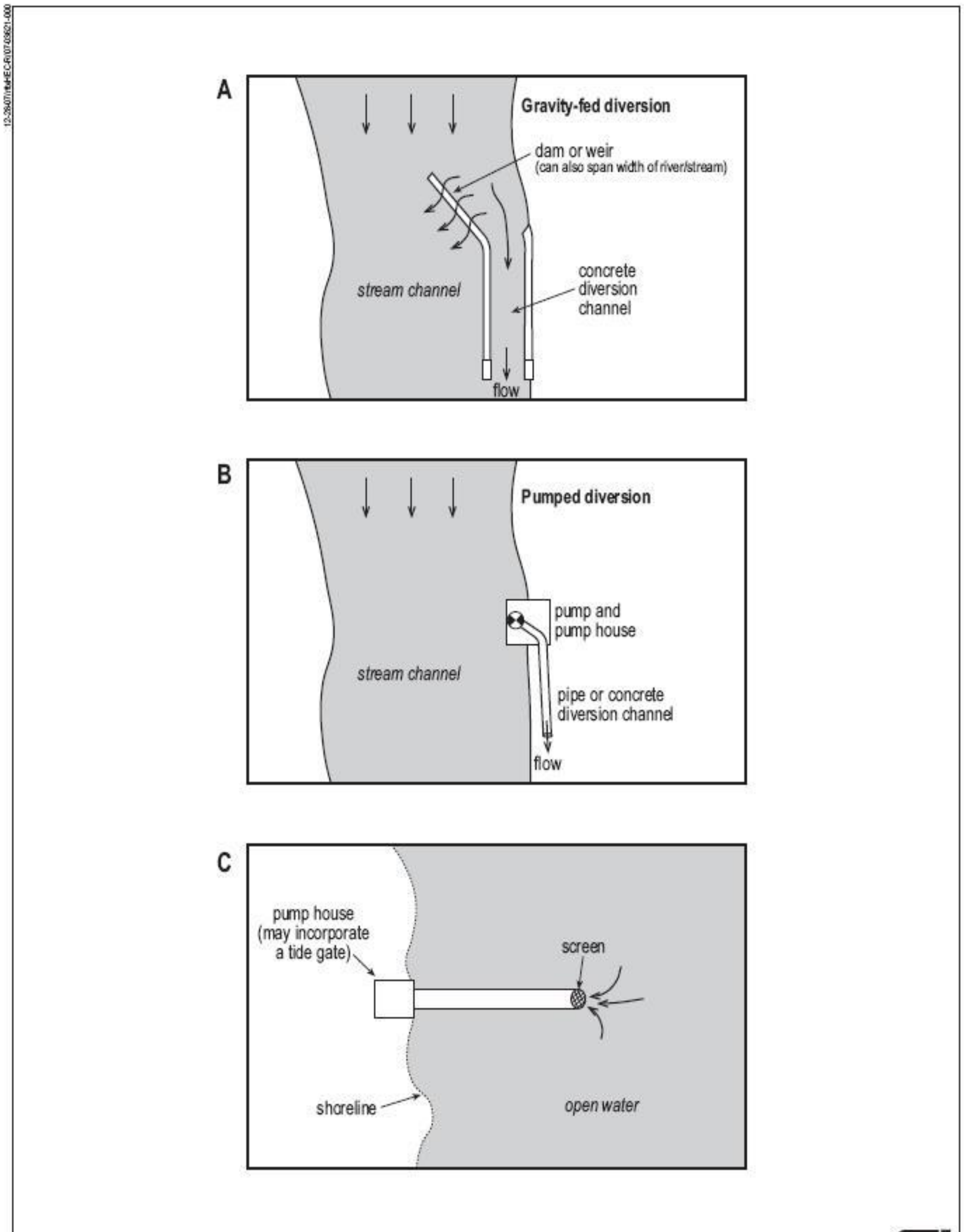
Gravity fed diversions usually include a dam or weir-like structure that partially or fully spans a river or stream channel. The flow control structure is used to create the hydraulic head necessary to divert the water out of the channel (Figure 4-20A). Construction of a gravity-fed diversion system includes the design and installation of flow control dam or weir and a diversion channel. This diversion channel is typically made of concrete but can also consist of a metal pipe. This type of diversion system typically includes a fish screen located at the downstream end of the diversion channel.

Pumped diversion systems typically take the form of a pump house and intake pipe or gallery with an associated concrete channel or a pipe used to transport the diverted water to its intended use (Figure 4-20B). This type of system is commonly located along the bank of a stream/river or lake. This type of system is used where water must be pumped up and out of the source body because the necessary hydraulic head for a gravity diversion is not available. This type of diversion system typically includes a fish screen at the pump intake.

Gravity-fed and pumped diversion systems can also be combined by having a pump station located at the end of the diversion channel. In these cases, fish screens are typically located at the pump intake.

In marine and lacustrine environments, water intake systems typically consist of an intake pipe with an associated pumping system. The intake may extend some distance into the water body while the pumping system is located onshore (Figure 4-20C). This type of intake system typically includes a fish screen located at the pipe’s mouth, although some configurations may incorporate additional internal screening mechanisms.

Intake systems fed by tidal exchange may also be employed in certain settings. This type of configuration may incorporate a tide gate on the shoreline to regulate intake flows. Construction of intake systems includes the design and installation of a pipeline and pump house and, potentially, a shoreline tide gate in marine environments. As with outfalls, the pipe associated with these diversion systems can be categorized as submerged intakes (typical in reservoirs, lakes, and marine environments), and exposed intakes (which are found in stream and river environments). Intakes are usually screened.



Note: In riverine environments, diversion systems can be gravity fed (A) or pumped (B). In marine environments, diversion systems are commonly constructed on the shore and pipelines extend out into the open water (C). In lacustrine environments, several configurations are possible including those shown in B and C.

Figure 4-20. Types of diversion systems.

5 Potentially Covered Species Habitat Use

This white paper identifies what is known about how activities associated with the construction, operation and repair of HPA-permitted activities can pose risks of take for the 52 HCP species. To understand species-specific impacts, it is important to understand the geographic distribution and habitat use of each species. Table 5-1 lists the scientific name, Water Resource Inventory Area (WRIA) of occurrence, Tidal Reference Area of occurance, and the reproductive patterns and habitat requirements of each of these HCP species. Through the identification of species-specific habitat needs, the risk of take associated with each mechanism of impact related to HPA-permitted activities can be identified. Once the potential for take has been identified, it can be avoided. If unavoidable, the risk of take can be minimized by design and/or through conservation and protection measures.

Table 5-1. Range of occurrence of the HCP species and their habitat requirements.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Chinook salmon	*Oncorhynchus tshawytscha*	01–42, 44–50	All	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries recognizes eight evolutionarily significant units (ESUs) of Chinook salmon in Washington: (1) Upper Columbia River spring-run; (2) Snake River spring/summer run; (3) Snake River fall-run; (4) Puget Sound; (5) lower Columbia River; (6) Washington coast; (7) Mid-Columbia River spring-run; and (8) Upper Columbia River summer/fall-run. Chinook salmon exhibit one of two life-history types, or races: the stream-type and the ocean-type. Stream-type Chinook tend to spend 1 (or less frequently 2) years in freshwater environments as juveniles prior to migrating to salt water as smolts. Stream-type Chinook are much more dependent on freshwater stream ecosystems than ocean-type Chinook. Stream-type Chinook do not extensively rear in estuarine and marine nearshore environments; rather, they head offshore and begin their seaward migrations. Ocean-type Chinook enter salt water at one of three phases: immediate fry migration soon after yolk is absorbed, fry migration 60–150 days after emergence, and fingerling migrants that migrate in the late summer or fall of their first year. Ocean-type Chinook are highly dependent on estuarine habitats to complete their life history. Chinook generally feed on invertebrates but become more piscivorous with age. **Reproduction/Life History** Chinook runs are designated on the basis of adult migration timing: • Spring-run Chinook: Tend to enter fresh water as immature fish,

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
				migrate far upriver, and finally spawn in the late summer and early autumn. • Fall-run Chinook: Enter fresh water at an advanced stage of maturity, move rapidly to their spawning areas on the mainstem or lower tributaries of the rivers, and spawn within a few days or weeks of freshwater entry. • Spring Chinook: Spawning occurs from mid-July to mid-December, and incubation lasts approximately 1.5–7 months, depending on temperature. Emergence follows, 6–8 months from fertilization. • Fall Chinook: Spawning occurs from late October to early December, with incubation occurring for 1–6 months. Emergence follows, approximately 6 months after fertilization. (Healey 1991; Myers et al. 1998; WDNR 2006a; Wydoski and Whitney 2003)
Coho salmon	*Oncorhynchus kisutch*	01–42, 44–48, 50	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes four ESUs of coho salmon in Washington: (1) Lower Columbia River; (2) Southwest Washington; (3) Puget Sound and Strait of Georgia; and (4) Olympic Peninsula. This species is found in a broader diversity of habitats than any of the other native anadromous salmonids. Fry feed primarily on aquatic insects and prefer pools and undercut banks with woody debris; adults feed on herring and other forage fish. **Reproduction/Life History** Coho adults spawn from September to late January, generally in the upper watersheds in gravel free of heavy sedimentation. Developing young remain in gravel for up to 3 months after hatching. Fry emerge from early March to late

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				July. Coho rear in fresh water for 12–18 months before moving downstream to the ocean in the spring. Coho spend between 1 and 2 years in the ocean before returning to spawn. (Groot and Margolis 1991; Murphy and Meehan 1991; WDNR 2005, 2006a; Wydoski and Whitney 2003)
Chum salmon	*Oncorhynchus keta*	01, 03–05, 07–29	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes four ESUs of chum salmon in Washington: (1) Hood Canal summer run; (2) Columbia River; (3) Puget Sound/Strait of Georgia; and (4) Pacific Coast. Little is known about their ocean distribution; maturing individuals that return to Washington streams have primarily been found in the Gulf of Alaska. Chum migrate into rivers and streams of Washington coast, Hood Canal, Strait of Juan de Fuca, Puget Sound, and the Columbia River basin to spawn, but their range does not extend upstream above the Dalles Dam in the Columbia River. Fry feed on chironomid and mayfly larvae, as well as other aquatic insects, whereas juvenile fish in the estuary feed on copepods, tunicates, and euphausiids. **Reproduction/Life History** Chum salmon have three distinct run times: summer, fall and winter. Summer chum begin their upstream migration and spawn from mid-August through mid-October, with fry emergence ranging from the beginning of February through mid-April. Chum fry arrive in estuaries earlier than most salmon, and juvenile chum reside in estuaries longer than most other anadromous species. Chum salmon rear in the ocean for the majority of their adult lives. Fall chum adults enter the rivers from late October through November and spawn in November and December. Winter chum adults migrate upstream from December through January and spawn from January through February. Fall and winter chum fry emerge in March and April and quickly emigrate to the estuary. Chum salmon

Common Name	**Scientific Name**	**Water Resource Inventory Area[a]**	**Tidal Reference Area[b]**	**Habitat Requirements and Reproduction Timing**
				utilize the low-gradient (from 1–2 percent grade), sometimes tidally influenced lower reaches of streams for spawning. (Healey 1982; Johnson et al. 1997; Quinn 2005; Salo 1991; WDNR 2005, 2006a; Wydoski and Whitney 2003)
Pink salmon	*Oncorhynchus gorbuscha*	01, 03–05, 07, 09–11, 16–19, 21	1–13	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes two ESUs of pink salmon in Washington, neither of which is listed: (1) Odd-year; and (2) Even-year. Pink salmon are the most abundant species of salmon, with 13 stocks identified in Washington. They are the smallest of the Pacific salmon and mature and spawn on a 2-year cycle in Washington (primarily spawning during odd years). Adults are opportunistic feeders in marine habitat, foraging on a variety of forage fish, crustaceans, ichthyoplankton, and zooplankton. Juveniles primarily feed on small crustaceans such as euphausiids, amphipods, and cladocerans. **Reproduction/Life History** Pink salmon will spawn in rivers with substantial amounts of silt. Spawning occurs from August through October. Fry emerge from their redds in late February to early May, depending on water temperature, and migrate downstream to the estuary within 1 month. Juveniles remain in estuarine or nearshore waters for several months before moving offshore as they migrate to the Pacific Ocean, where they remain approximately 1 year until the next spawning cycle. (Hard et al. 1996; Heard 1991; WDNR 2005, 2006a)
Sockeye salmon	*Oncorhynchus nerka*	01, 03–05, 07–11, 16, 19–22, 25–33, 35–37, 40, 41, 44–50	5, 8, 14	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries recognizes seven ESUs of sockeye salmon in Washington:

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
				(1) Snake river; (2) Ozette Lake; (3) Baker River; (4) Okanogan River; (5) Quinault Lake; (6) Lake Pleasant; and (7) Lake Wenatchee. WDFW recognizes an additional sockeye salmon stock in the Big Bear Creek drainage of Lake Washington. Kokanee (landlocked sockeye) occur in many lakes, with the larger populations in Banks and Loon lakes in eastern Washington and Lake Whatcom and Lake Washington-Sammamish in western Washington. Juveniles feed on zooplankton, and adults primarily feed on fish, euphausiids, and copepods. **Reproduction/Life History** Spawn in shallow, gravelly habitat in rivers and lakes during August to October. Juvenile sockeye rear in lakes for 1–2 years before migrating to the ocean. Emergence occurs within 3–5 months. (Gustafson et al. 1997; Wydoski and Whitney 2003)
Steelhead	*Oncorhynchus mykiss*	01, 03–05, 07–12, 14, 15, 17–41, 44–50	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes 15 Distinct Population Segments (DPSs) of steelhead, seven of which occur in Washington. During their ocean phase, steelhead are generally found within 10 and 25 miles of the shore; steelhead remain in the marine environment 2–4 years before returning to fresh water to spawn. Most steelhead spawn at least twice in their lifetimes. Escape cover, such as logs, undercut banks, and deep pools, is important for adult and young steelhead in the freshwater systems. The coastal west-side streams typically support more winter steelhead populations.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				Reproduction A summer spawning run enters fresh water in August and September, and a winter run occurs from December through February. Summer steelhead usually spawn farther upstream than winter populations and dominate inland areas such as the Columbia Basin. Spawning occurs from March to April for both winter and summer run steelhead. After hatching and emergence (approximately 3 months), juveniles establish territories, feeding on microscopic aquatic organisms and then larger organisms such as isopods, amphipods, and aquatic and terrestrial insects. Steelhead rear in fresh water for up to 4 years before migrating to sea. (Busby et al. 1996; McKinnell et al. 1997; WDNR 2006a; Wydoski and Whitney 2003)
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	01–05, 07–30	All	**General Information (Habitats and Feeding/Life-history Types)** NOAA Fisheries has recognized three ESUs in Washington: (1) Puget Sound; (2) Olympic Peninsula; and (3) Southwestern Washington/Columbia River. USFWS has assumed sole jurisdiction for this species. No coastal cutthroat trout DPSs are listed under the ESA in Washington. Coastal cutthroat trout exhibit varied life-history forms including: • Resident (stays in streams after rearing in their natal streams) – Resident coastal cutthroat trout utilize small headwater streams for all of their life stages. • Fluvial (migrates to larger rivers after rearing in their natal streams). • Adfluvial (migrates to lakes after rearing in their natal streams). • Anadromous (utilizes estuaries and nearshore habitat but has been

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				caught offshore). Juveniles of all life forms feed primarily on aquatic invertebrates but are opportunistic feeders; adults tend to feed on smaller fish, amphibians, and crustaceans while foraging within the nearshore environment. **Reproduction/Life History** Coastal cutthroat trout are repeat spawners, and juveniles typically rear in the natal streams for up to 2 years. Spawning occurs from late December to February, with incubation lasting approximately 2–4 months. Emergence occurs after 4 months. (Johnson et al. 1999; Pauley et al. 1988; WDNR 2006a)
Redband trout	*Oncorhynchus mykiss gardnerii*	37–40, 45–49, 54–57	NA	**General Information (Habitats and Feeding)** Redband trout is a subspecies of rainbow trout found east of the Cascade Mountains, which prefer cool water that is less than 70^{o}F (21^{o}C), and occupy streams and lakes with high amounts of dissolved oxygen. Their food primarily consists of Daphnia and chironomids as well as fish eggs, fish, and insect larvae and pupae. **Reproduction/Life History** Spawn in streams with clean, small gravel from March through May. Incubation takes approximately 1–3 months, with emergence occurring between June and July. (USFS 2007)
Westslope cutthroat trout	*Oncorhynchus clarki lewisii*	37–39, 44–55, 58–62	NA	**General Information (Habitats and Feeding/Life-history Types)** Cutthroat trout tend to thrive in streams with extensive pool habitat and cover. The westslope is a subspecies of cutthroat trout with three possible life forms: • Adfluvial (migrates to lakes)

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				- Fluvial (migrates to larger rivers) - Resident (stays in streams). The headwater tributaries used by resident cutthroat are typically cold, nutrient-poor waters that result in slow growth. Fluvial and adfluvial forms can exhibit more growth due to warmer water temperatures and nutrient availability. Fry feed on zooplankton, and fingerlings feed on aquatic insect larvae. Adults feed on terrestrial and aquatic insects. **Reproduction/Life History** Spawning: all three life forms spawn in small gravel substrates of tributary streams in the spring (March to July) when water temperature is about 50°F (10°C); incubation occurs during April to August, and emergence occurs from May through August. Fry spend 1–4 years in their natal stream before migrating to their ultimate habitat. (Liknes and Graham 1988; Shepard et al. 1984; Wydoski and Whitney 2003)
Bull trout	*Salvelinus confluentus*	01, 03–05, 07–23, 26, 27, 29–41, 44–55, 57–62	All	**General Information (Habitats and Feeding/Life-History Types)** Widely distributed in Washington; exhibit four life-history types: - Resident (stays in streams after rearing in their natal streams) - Fluvial (migrates to larger rivers after rearing in their natal streams) - Adfluvial (migrates to lakes after rearing in their natal streams) - Anadromous (bull trout in the nearshore ecosystem rely on estuarine wetlands and favor irregular shorelines with unconsolidated substrates). Young of the year occupy side channels, with juveniles in pools, runs, and riffles; adults occupy deep pools. Juvenile diet includes larval and adult aquatic insects; subadults and adults primarily feed on fish. **Reproduction/Life History** The migratory forms of bull trout, such as anadromous, adfluvial, and fluvial,

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				move upstream by early fall to spawn in September and October (November at higher elevations). Although resident bull trout are already in stream habitats, they move upstream looking for suitable spawning habitat. They prefer clean, cold water (50°F [10°C]) for spawning. Colder water (36–39°F [2–4°C]) is required for incubation. Preferred spawning areas often include groundwater infiltration. Extended incubation periods (up to 220 days) make eggs and fry particularly susceptible to increases in fine sediments. Bull trout typically rear in natal streams for 2–4 years, although resident fish may remain in these streams for their entire lives; multiple life-history forms may occur in the same habitat environments. (Goetz et al. 2004; WDNR 2005, 2006a; Wydoski and Whitney 2003)
Dolly Varden	*Salvelinus malma*	01, 03, 05, 07, 17–22, 24	6–10, 14–17	**General Information (Habitats and Feeding/Life-History Types)** Species restricted to coastal areas and rivers that empty into them. Juveniles extensively use instream cover; while in the marine systems, they use beaches of sand and gravel. Prefer pool areas and cool temperatures. Feed opportunistically on aquatic insects, crustaceans, salmon eggs, and fish. Closely related to bull trout and exhibit the same life-history traits. Four life-history types occur: • Resident (stays in streams after rearing in their natal streams) • Fluvial (migrates to larger rivers after rearing in their natal streams) • Adfluvial (migrates to lakes after rearing in their natal streams) • Anadromous (migrates to marine waters after rearing in their natal streams). **Reproduction/Life History** Spawn and rear in streams from mid-September through November. Incubation lasts approximately 130 days. Juveniles can spend 2–4 years in their natal streams before migration to marine waters. (Leary and Allendorf 1997; WDNR 2005; Wydoski and Whitney 2003)

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pygmy whitefish	*Prosopium coulteri*	08, 19, 39, 47, 49, 53, 55, 58, 59, 62	NA	**General Information (Habitats and Feeding)** In Washington, pygmy whitefish occur at the extreme southern edge of their natural range; pygmy whitefish were once found in at least 15 Washington lakes but have a current distribution in only nine. They occur most often in deep, oligotrophic lakes with temperatures less than 50°F (10°C), where they feed on zooplankton, such as cladocerans, copepods, and midge larvae. **Reproduction/Life History** Pygmy whitefish spawn in streams or lakes from July through November. They prefer pools, shallow riffles, and pool tail-outs when spawning in streams. Lake spawning by pygmy whitefish occurs at night. Spawning occurs by scattering their eggs over coarse gravel. Incubation and emergence timing are unknown, but eggs are believed to hatch in the spring. (Hallock and Mongillo 1998; WDNR 2005, 2006a; Wydoski and Whitney 2003)
Olympic mudminnow	*Novumbra hubbsi*	08–24	NA	**General Information (Habitats and Feeding)** Occur in the southern and western lowlands of the Olympic Peninsula, the Chehalis River drainage, lower Deschutes River drainage, south Puget Sound lowlands west of the Nisqually River, and in King County. They are generally found in quiet water with mud substrate, preferring bogs and swamps with dense aquatic vegetation. Mudminnows feed on annelids, insects, and crustaceans. **Reproduction/Life History** Adults spawn from November through June (peaking in April and May). Females deposit eggs onto vegetation where fry remain firmly attached for approximately 1 week after hatching. Incubation lasts approximately 8-10 days. (Harris 1974; Mongillo and Hallock 1999; WDNR 2005, 2006a)
Lake chub	*Couesius plumbeus*	48, 61; other locations unknown	NA	**General Information (Habitats and Feeding)** Bottom dwellers inhabiting a variety of habitats in lakes and streams, but are known to prefer small, slow streams. In Washington, they are known only from

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				the northeastern part of the state (small streams and lakes in Okanogan and Stevens counties). Juveniles feed on zooplankton and phytoplankton, whereas adults primarily feed on insects. **Reproduction/Life History** Lake chub move into shallow areas on rocky and gravelly substrates in tributary streams of lakes or lakeshores during the spring to spawn when water temperatures are between 55 and 65°F (13 and 18°C). The eggs are broadcast over large rocks and then settle into the smaller substrate, hatching after approximately 10 days. (WDNR 2005; Wydoski and Whitney 2003)
Leopard dace	*Rhinichthys falcatus*	25–31, 37–41, 44–50	NA	**General Information (Habitats and Feeding)** In Washington, leopard dace inhabit the bottoms of streams and small to mid-sized rivers, specifically the Columbia, Snake, Yakima, and Simikameen Rivers, with velocities less than 1.6 ft/sec (0.5 m/sec); prefer gravel and small cobble substrate covered by fine sediment with summer water temperatures ranging between 59 and 64°F (15 and 18°C). Juveniles feed primarily on aquatic insects; adult leopard dace consume terrestrial insects. **Reproduction/Life History** Breeding habitat for dace generally consists of the gravel or cobble bottoms of shallow riffles; leopard dace breed in slower, deeper waters than the other dace species. The spawning period for dace is from May through July. The eggs adhere to rocky substrates. Fry hatch approximately 6–10 days after fertilization, and juveniles spend 1–3 months rearing in shallow, slow water. (WDNR 2005, 2006a; Wydoski and Whitney 2003)
Margined sculpin	*Cottus marginatus*	32, 35	NA	**General Information (Habitats and Feeding)** Endemic to southeastern Washington (smaller tributary streams of the Walla Walla and Tucannon River drainages) where habitat is in deeper pools and slow-moving glides in headwater tributaries with silt and small gravel substrate. They

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				prefer cool water less than 68°F (20°C) and avoid high-velocity areas. Food includes immature aquatic insects, invertebrates, small fish, and eggs. **Reproduction/Life History** Spawning occurs in May and June primarily under rocks, root wads, or logs. The female deposits a mass of adhesive eggs in the nest, which is guarded by the male. Incubation duration unknown. (Mongillo and Hallock 1998; WDNR 2005; Wydoski and Whitney 2003)
Mountain sucker	*Catostomus platyrynchus*	25–35, 37–41, 44–50	NA	**General Information (Habitats and Feeding)** Distribution restricted to Columbia River system. Found in clear, cold mountain streams less than 40 ft wide and in some lakes; prefer deep pools in summer with moderate current. Food consists of algae and diatoms. Juveniles prefer slower side channels or weedy backwaters. **Reproduction/Life History** Males reach sexual maturity in 2–3 years and females in 4 years. Spawning in June and July when water temperatures exceed 50°F (10°C). Spawning occurs in gravelly riffles of small streams when suckers move into those reaches to feed on algae. Spawning likely occurs at night when water temperatures are in a range of 51–66°F (10.5–19°C). Fertilized eggs fall into and adhere to the spaces between the gravel composite. Incubation period lasts approximatley 8-14 days. (Wydoski and Whitney 2003)
Umatilla dace	*Rhinichthys umatilla*	31, 36–41, 44–50, 59–61	NA	**General Information (Habitats and Feeding)** Umatilla dace are benthic fish found in relatively productive, low-elevation streams with clean substrates of rock, boulders, and cobbles in reaches where water velocity is less than 1.5 ft/sec (0.5 m/sec). Feeding is similar to that described for leopard dace. Juveniles occupy streams with cobble and rubble substrates, whereas adults occupy deeper water habitats.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				Reproduction/Life History Spawning behaviors are similar to those described for leopard dace, with spawning primarily occurring from early to mid-July. (WDNR 2005, 2006a; Wydoski and Whitney 2003)
Pacific lamprey	*Lampetra tridentata*	01, 03–05, 07–35, 37–40, 44–50	All	**General Information (Habitats and Feeding)** Found in most large coastal and Puget Sound rivers and Columbia, Snake, and Yakima river basins. The larvae are filter feeders, residing in mud substrates and feeding on algae and other organic matter for at least 5 years. **Reproduction/Life History** From July through October, maturing Pacific lamprey enter fresh water and gradually move upstream to spawn the following spring. The nest usually consists of a shallow depression built in gravel and rock substrates. Eggs hatch in 2–4 weeks, with newly hatched larvae remaining in the nest for 2–3 weeks before moving downstream as larvae (ammocoetes). Juveniles migrate to the Pacific Ocean 4–7 years after hatching and attach to fish in the ocean for 20–40 months before returning to rivers to spawn. (WDNR 2005; Wydoski and Whitney 2003)
River lamprey	*Lampetra ayresi*	01, 03, 05, 07–16, 20–40	1–9, 11–17	**General Information (Habitats and Feeding)** Detailed distribution records are not available for Washington, but they are known to inhabit coastal rivers, estuaries, and the Columbia River system. They have also been observed in Lake Washington and its tributaries. In the marine system, river lamprey inhabit nearshore areas. Adults are anadromous living in the marine system as parasites on fish. Adult river lamprey are believed to occupy deep portions of large river systems. The larvae feed on microscopic plants and animals. **Reproduction/Life History** Adults migrate back into fresh water in the fall. Spawning occurs in winter and

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				spring. Eggs hatch in 2–3 weeks after spawning. Juveniles are believed to migrate from their natal rivers to the Pacific Ocean several years after hatching; adults spend 10–16 weeks between May and September in the ocean before migrating to fresh water. (WDNR 2005; Wydoski and Whitney 2003)
Western brook lamprey	*Lampetra richardsoni*	01, 03, 05, 07–14, 16, 20–40	NA	**General Information (Habitats and Feeding)** Found in small coastal and Puget Sound rivers and lower Columbia and Yakima river basins; spends entire life in fresh water. Adults are found in cool water (52–64°F [11–17.8°C]) on pebble/rocky substrate. Larvae (ammocoetes) are filter feeders, consuming primarily diatoms. Adults do not feed and die within a month of spawning. **Reproduction/Life History** Spawning generally occurs from April through July, with adults creating nests in coarse gravel at the head of riffles. Eggs hatch after about 10 days in water between 50 and 60°F (10 and 16°C). Within 30 days of hatching, ammocoetes emerge from the nests and move to the stream margin, where they burrow into silty substrates. Larvae remain in the stream bottom—apparently moving little—for approximately 4–6 years. (Wydoski and Whitney 2003)
Green sturgeon	*Acipenser medirostris*	22, 24, 28	All	**General Information (Habitats and Feeding)** NOAA Fisheries recognizes two DPSs of green sturgeon, both of which can be found in Washington. The southern DPS is listed as threatened and the northern DPS is a species of concern. Habits and life history not well known. Washington waters with green sturgeon populations include the Columbia River, Willapa Bay, and Grays Harbor, in addition to marine waters. They spend much of their life in marine nearshore waters and estuaries feeding on fishes and invertebrates. **Reproduction/Life History** Spawning generally occurs in spring in deep, fast-flowing sections of rivers.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				Spawning habitat includes cobble or boulder substrates. Green sturgeon move upstream during spring to spawn and downstream during fall and winter. Large eggs sink to bottom. (Adams et al. 2002; Emmett et al. 1991; Kynard et al. 2005; Nakamoto and Kisanuki 1995; Wydoski and Whitney 2003)
White sturgeon	*Acipenser transmontanus*	01, 03, 05–22, 24–37, 40–42, 44–61	All	**General Information (Habitats and Feeding)** Found in marine waters and major rivers in Washington, including the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. In marine environments, adults and subadults use estuarine and marine nearshore habitats, including some movement into intertidal flats to feed at high tide. Some landlocked populations exist behind dams on the Columbia River. Juveniles feed on mysid shrimp and amphipods; large fish feed on variety of crustaceans, annelid worms, mollusks, and fish. **Reproduction/Life History** Spawn in deep, fast-flowing sections of rivers (prefer swift [2.6–9.2 ft/sec (0.8–2.8 m/sec)] and deep [13–66 ft (4–20 m)] water) on bedrock, cobble, or boulder substrates. Spawning occurs from April through July, with incubation lasting approximately 7 days and emergence following in another 7 days. (Emmett et al. 1991; WDNR 2005; Wydoski and Whitney 2003)
Eulachon	*Thaleichthys pacificus*	01–29 (mouths of major rivers)	14–17	**General Information (Habitats and Feeding)** Eulachon occur from northern California to southwestern Alaska in offshore marine waters. They are plankton-feeders, eating crustaceans such as copepods and euphausiids; larvae and post larvae eat phytoplankton and copepods. They are an important prey species for fish, marine mammals, and birds. **Reproduction/Life History** Spawn in tidal portions of rivers in spring when water temperature is 40–50°F (4–10°C), generally from March through May; use a variety of substrates, but sand and gravel are most common. Eggs stick to substrate and incubation ranges

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
				from 20–40 days (dependent on temperature). Larvae drift downstream to salt water where juveniles rear in nearshore marine areas. (Howell et al. 2001; Langer et al. 1977; Lewis et al. 2002; WDFW 2001; WDNR 2005; Willson et al. 2006)
Longfin smelt	*Spirinchus thaleichthys*	01–03, 05–17, 22 and 24	1–9, 15–17	**General Information (Habitats and Feeding)** Marine species that spawns in streams not far from marine waters. They are anadromous, with some populations in Lake Washington that spawn in tributaries, including the Cedar River. Juveniles use nearshore habitats and a variety of substrates; juveniles feed on zooplankton. Adults feed on copepods and euphausiids. Most adults die after spawning. **Reproduction** Spawn in coastal rivers from October through December. Lake Washington populations spawn from January through April. Eggs hatch in approximately 40 days and the larvae drift downstream to salt water. (Gotthardt 2006; WDNR 2005; Wydoski and Whitney 2003)
Pacific sand lance	*Ammodytes hexapterus*	NA	All	**General Information (Habitats and Feeding)** Widespread in Puget Sound, Strait of Juan de Fuca, and coastal estuaries. Schooling plankton feeders. Adults feed during the day and burrow into the sand at night. **Reproduction/Life History** Spawn on sand and beaches with gravel up to 1-inch in diameter at tidal elevations of +4–5 ft (+1.5 meters) to approximately the mean higher high water (MHHW) line from November through February. Emergence occurs from January to April. Larvae and young rear in bays and nearshore areas. (Garrison and Miller 1982; Nightingale and Simenstad 2001b; NRC 2001; Penttila 2000; Penttila 2001; WDFW 1997a)

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
Surf smelt	*Hypomesus pretiosus*	NA	All	**General Information (Habitats and Feeding)** Schooling plankton-feeding forage fish. They feed on a variety of zooplankton, planktonic crustaceans, and fish larvae. Adult surf smelt are pelagic but remain in nearshore habitats. Juveniles rear in nearshore areas, and adults form schools offshore; feed on planktonic organisms. Also an important forage fish. **Reproduction/Life History** Spawning occurs year-round in north Puget Sound, fall and winter in south Puget Sound, and summer along the coast. They spawn at the highest tides during high slack tide on coarse sand and pea gravel. Incubation is 2–5 weeks. Emergence varies with season: 27–56 days in winter, 11–16 days in summer. (Nightingale and Simenstad 2001b; NRC 2001; Penttila 2000; Penttila 2001; WDFW 1997c)
Pacific herring	*Clupea harengus pallasi*	NA	1, 2, 4, 5, 8–13, 16, 17	**General Information (Habitats and Feeding)** Eighteen separate stocks in Puget Sound. Widely distributed throughout Puget Sound and coastal wetlands and estuaries. Pacific herring adults feed on small fish, copepods, decapod crab larvae, and euphausiids. Juveniles feed primarily on euphausiids, copepods, and small crustacean larvae. Are also an important forage fish. **Reproduction/Life History** Utilize intertidal and subtidal habitats (between 0 and -40 ft [0 and -12.2 m] mean lower low water [MLLW]) for spawning and juvenile rearing; spawning also occurs above MLLW. Spawning occurs from late January to early April. Eggs are adhered to eelgrass, kelp, seaweed, and sometimes on pilings. Eggs hatch after approximately 10 days. Larvae are pelagic. (Nightingale and Simenstad 2001b; Penttila 2000; Simenstad et al. 1979; WDFW 1997b)
Lingcod	*Ophiodon*	NA	All	**General Information (Habitats and Feeding)**

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
	elongatus			The lingcod is a large top-level carnivore fish found throughout the West Coast of North America. Adult lingcod have a relatively small home range. Juveniles prefer sand habitats near the mouths of bays and estuaries, while adults prefer rocky substrates. Larvae and juveniles are generally found in upper 115 ft (35 m) of water. Adults prefer slopes of submerged banks with macrophytes and channels with swift currents. Larvae feed on copepods and amphipods; juveniles feed on small fishes; and adults on fish, squid, and octopi. **Reproduction/Life History** Spawn in shallow water and intertidal zone from January through late March. Egg masses adhere to rocks, and incubation is from February to June. Larvae spend 2 months in pelagic nearshore habitat. (Adams and Hardwick 1992; Emmett et al. 1991; Giorgi 1981; NMFS 1990; NRC 2001)
Pacific cod	*Gadus macrocephalus*	NA	All	**General Information (Habitats and Feeding)** Pacific cod are widely distributed in relatively shallow marine waters throughout the northern Pacific Ocean (Washington's inland marine waters are considered the southern limit of populations). Adults and large juveniles are found over clay, mud, and coarse gravel bottoms; juveniles use shallow vegetated habitats such as sand-eelgrass. Feed opportunistically on invertebrates (worms, crabs, shrimp) and fishes (sand lance, pollock, flatfishes). Larvae feed on copepods, amphipods, and mysids. **Reproduction/Life History** Broadcast spawners during late fall through early spring. Eggs sink and adhere to the substrate. Incubate for 1–4 weeks, and larvae spend several months in the water column. Juvenile cod metamorphose and settle to shallow vegetated habitats. (Albers and Anderson 1985; Bargmann 1980; Dunn and Matarese 1987; Garrison and Miller 1982; Hart 1973; Nightingale and Simenstad 2001b; NMFS 1990; NRC 2001)

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
Pacific hake	*Merluccius productus*	NA	All	**General Information (Habitats and Feeding)** Pacific hake are schooling fish. The coastal stock of hake is migratory; Puget Sound stocks reside in estuaries and rarely migrate. Larvae feed on calanoid copepods; juveniles and small adults feed on euphasiids; adults eat amphipods, squid, herring, and smelt. **Reproduction/Life History** Puget Sound spawning occurs from March through May at mid-water depths of 50–350 ft (15–90 m); may spawn more than once per season. Eggs and larvae are pelagic. (Bailey 1982; McFarlane and Beamish 1986; NMFS 1990; NRC 2001; Quirollo 1992)
Walleye pollock	*Theragra chalcogramma*	NA	All	**General Information (Habitats and Feeding)** Widespread species in northern Pacific. Washington is the southern end of their habitat. Larvae and small juveniles are found at 200-ft (60-m) depth; juveniles use nearshore habitats of a variety of substrates. Juveniles feed on small crustaceans, adults feed on copepods, euphausiids, and young pollock. **Reproduction/Life History** Broadcast spawning occurs from February through April. Eggs are suspended at depths ranging from 330–1,320 ft (100–400 m). Pelagic larvae settle near the bottom and migrate to inshore, shallow habitats for their first year. (Bailey et al. 1999; Garrison and Miller 1982; Livingston 1991; Miller et al. 1976; NRC 2001)
Black rockfish	*Sebastes melanops*	NA	All	**General Information (Habitats and Feeding)** Adults prefer deep and shallow rock substrates in summer, deeper water in winter. Kelp and eelgrass are preferred habitat for juveniles that feed on nekton and zooplankton. Adults feed on amphipods, crabs, copepods, and small fish.

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
				Reproduction/Life History Spawning occurs from February through April; ovoviviparous incubation as with other rockfish species. Larvae are planktonic for 3–6 months, where they are dispersed by currents, advection, and upwelling. They begin to reappear as young-of-the-year fish in shallow, nearshore waters. (Kramer and O'Connell 1995; WDNR 2006a)
Bocaccio rockfish	*Sebastes paucispinis*	NA	All	**General Information (Habitats and Feeding)** Adults semidemersal in shallow water over rocks with algae, eelgrass, and floating kelp. Larvae feed on diatoms; juveniles feed on copepods and euphausiids. **Reproduction/Life History** Ovoviviparous spawning occurs year-round, with incubation lasting 40–50 days. Larvae and juveniles are pelagic. (Garrison and Miller 1982; Hart 1973; Kramer and O'Connell 1995; MBC Applied Environmental Sciences 1987; NRC 2001; Sumida and Moser 1984)
Brown rockfish	*Sebastes auriculatus*	NA	All	**General Information (Habitats and Feeding)** Utilize shallow-water bays with natural and artificial reefs and rock piles; estuaries used as nurseries; can tolerate water temperatures to at least 71°F (22°C); eat small fishes, crabs, and isopods. **Reproduction/Life History** Spawning occurs from March through June. Larvae are released from the female into the pelagic environment in May and June (ovoviviparous incubation). Larvae live in the upper zooplankton layer for up to 1 month before they metamorphose into pelagic juveniles. The pelagic juveniles spend 3–6 months in the water column as plankton. They then settle in shallow water nearshore, later migrating to deeper water.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				(Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Stein and Hassler 1989)
Canary rockfish	*Sebastes pinniger*	NA	All	**General Information (Habitats and Feeding)** Adults use sharp drop-offs and pinnacles with hard bottoms; often associated with kelp beds; feed on krill and occasionally on fish. Adults are mostly found at depths of 260–660 ft (80–200 meters) (with two recorded at 2,750 ft [838 meters]), tending to collect in groups around pinnacles and similar high-relief rock formations, especially where the current is strong. Young canary rockfish live in relatively shallow water, moving to deeper water as they mature. Juveniles feed on small crustacea such as krill larvae (and eggs), copepods, and amphipods, while adults eat krill and small fish. **Reproduction/Life History** Spawning is ovoviviparous and occurs from January through March. Larvae and juveniles are pelagic. (Boehlert 1980; Boehlert and Kappenman 1980; Boehlert et al. 1989; Hart 1973; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Sampson 1996)
China rockfish	*Sebastes nebulosis*	NA	All	**General Information (Habitats and Feeding)** Occur inshore and on open coast in sheltered crevices. Feed on crustacea (brittle stars and crabs), octopi, and fish. Juveniles are pelagic, but the adults are sedentary associating with rocky reefs or cobble substrates. **Reproduction/Life History** Spawning occurs from January through July; ovoviviparous incubation as with other rockfish species. Individual China rockfish spawn once a year. Larvae settle out of the plankton between 1 and 2 months after release. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; Rosenthal et al. 1988)

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
Copper rockfish	*Sebastes caurinus*	NA	All	**General Information (Habitats and Feeding)** Occur both inshore and on open coast; adults prefer rocky areas in shallower water than other rockfish species. Juveniles use shallow and nearshore macrophytes and eelgrass habitat; feed on crustaceans, fish, and mollusks. **Reproduction/Life History** Spawning occurs from March through May, with ovoviviparous incubation from April to June. Larvae are pelagic in deeper water before moving inshore. Newly spawned fish begin settling near the surface around large algae canopies or eelgrass, when available, or closer to the bottom when lacking canopies. (Eschmeyer et al. 1983; Haldorson and Richards 1986; Kramer and O’Connell 1995; Matthews 1990; NRC 2001; Stein and Hassler 1989)
Greenstriped rockfish	*Sebastes elongates*	NA	All	**General Information (Habitats and Feeding)** Adults found in benthic and mid-water columns. They live at between 330 and 825 ft (100 and 250 m). As they age, greenstriped rockfish move to deeper water. They are solitary and are often found resting on the seafloor and living among cobble, rubble, or mud. Adults feed on euphausiids, small fish, and squid. **Reproduction/Life History** From 10,000 to over 200,000 eggs are produced by the females each season by ovoviviparous spawning. Greenstriped rockfish release one brood of larvae in Washington. Larval release varies, occurring generally from January through July, depending on geographic location. (Eschmeyer et al. 1983; Kramer and O’Connell 1995; Love et al. 1990; NRC 2001)
Quillback rockfish	*Sebastes maliger*	NA	All	**General Information (Habitats and Feeding)** Shallow-water benthic species in inlets near shallow rock piles and reefs. Juveniles use eelgrass, sand, and kelp beds. Feed on amphipods, crabs, and

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
				copepods. **Reproduction/Life History** Ovoviviparous spawning from April through July, with larval release from May to July. (Kramer and O'Connell 1995; WDNR 2006a)
Redstripe rockfish	*Sebastes proriger*	NA	All	**General Information (Habitats and Feeding)** Adults found from 330- to 1,000-ft (100- to 300-m) depths, and young often found in estuaries in high- and low-relief rocky areas. Juveniles feed on copepods and euphausiids; adults eat anchovies, herring, and squid. **Reproduction/Life History** Spawning is ovoviviparous, occurring from January through March. Larvae and juveniles are pelagic. (Garrison and Miller 1982; Hart 1973; Kendall and Lenarz 1986; Kramer and O'Connell 1995; NRC 2001; Starr et al. 1996)
Tiger rockfish	*Sebastes nigrocinctus*	NA	All	**General Information (Habitats and Feeding)** Semidemersal to demersal species occurring at depths ranging from shallows to 1,000 ft (305 m); larvae and juveniles occur near surface and range of depth; adults use rocky reefs, canyons, and headlands; generalized feeders on shrimp, crabs, and small fishes. **Reproduction/Life History** Ovoviviparous spawning peaks in May and June. Juveniles are pelagic. (Garrison and Miller 1982; Kramer and O'Connell 1995; Moulton 1977; NRC 2001; Rosenthal et al. 1988)
Widow	*Sebastes*	NA	All	**General Information (Habitats and Feeding)**

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
rockfish	*entomelas*			Adults found from 330- to 1,000-ft (100- to 300-m) depths near rocky banks, ridges, and seamounts; adults feed on pelagic crustaceans, Pacific hake, and squid; juveniles feed on copepods and euphausiids. **Reproduction /Life History** Ovoviviparous spawning occurs from October through December. One brood of 95,000 to 1,113,000 eggs are produced by female widows per year. The season of larval release occurs earlier in the southern parts of their range than in the northern regions, likely January through April in Washington waters. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Laroche and Richardson 1981; NMFS 1990; NRC 2001; Reilly et al. 1992)
Yelloweye rockfish	*Sebastes ruberrimus*	NA	All	**General Information (Habitats and Feeding)** Adults are found from depths of 80–1,800 ft (24–550 m), near reefs and cobble bottom. Juveniles prefer shallow, broken-bottom habitat. Juveniles often hide in rock crevices; adults are demersal and solitary, tending to remain localized and not making extensive migrations. Adults feed on other rockfish species, sand lance, herring, shrimp, rock crabs, and snails. **Reproduction/Life History** Ovoviviparous spawning in late fall or early winter, with the larvae released from May to July. (Eschmeyer et al. 1983; Hart 1973; Kramer and O'Connell 1995; NRC 2001; Rosenthal et al. 1988)
Yellowtail rockfish	*Sebastes flavidus*	NA	All	**General Information (Habitats and Feeding)** Adults found from 165- to 1,000-ft (50- to 300-m) depths; adults semipelagic or pelagic over steep-sloping shores and rocky reefs. Juveniles occur in nearshore areas. Adults are opportunistic feeders on pelagic animals including hake, herring, smelt, squid, krill, and euphausiids.

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				Reproduction/Life History Ovoviviparous spawning from October through December. Incubation is between January and March. Larvae and juveniles are pelagic swimmers. (Eschmeyer et al. 1983; Kramer and O'Connell 1995; Love et al. 1990; NRC 2001; O'Connell and Carlile 1993)
Olympia oyster	*Ostrea lurida*	NA	1–14, 17	**General Information (Habitats and Feeding)** Species found throughout the inland waters of Puget Sound, as well as in Willapa Bay and possibly Grays Harbor; also grown commercially in Puget Sound. They occupy nearshore ecosystem on mixed substrates with solid attachment surfaces and are found from 1 ft (0.3 m) above MLLW to 2 ft (0.6m) below MLLW. Intolerant of siltation. **Reproduction/Life History** Reproduce spring to fall when water temperatures are between 54 and 61^{o}F (12.5 and 16^{o}C) by broadcast spawning. After 8–12 days, larvae develop into free-swimming larvae. Larvae are free-swimming for 2–3 weeks before they settle onto hard substrate, such as oyster shells and rocks. (Baker 1995; Couch and Hassler 1990; West 1997)
Northern abalone	*Haliotis kamtschatkana*	NA	10	**General Information (Habitats and Feeding)** Also known as pinto abalone. Presence in Washington is limited to the Strait of Juan de Fuca and the San Juan Islands. Occupies bedrock and boulders from extreme low water to 100 ft (30 m) below MLLW; usually associated with kelp beds. The abalone is completely vegetarian and uses its radula to scrape pieces of algae from the surface of rocks. **Reproduction/Life History** Broadcast spawners that release pelagic gametes that develop into free-swimming larvae using cilia to propel themselves. After up to a week, the larvae settle to the bottom, shed their cilia, and start growing a shell to begin sedentary

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
				adult life on crustose coralline algae. (Gardner 1981; NMFS 2007a; WDNR 2006b; West 1997)
Newcomb's littorine snail	*Algamorda subrotundata*	NA	14–17	**General Information (Habitats and Feeding)** Found in Grays Harbor and Willapa Bay on Washington coast; current distribution uncertain. Algae feeder occupying narrow band in *Salicornia* salt marshes above MHHW and is not considered a true marine gastropod. **Reproduction/Life History** Broadcast spawning in salt marshes. Other reproductive information unknown. (Larsen et al. 1995)
Giant Columbia River limpet	*Fisherola nuttalli*	35, 36, 40, 45, 47–49	NA	**General Information (Habitats and Feeding)** Also known as the shortface lanx, it occupies fast-moving and well-oxygenated streams. It is found in the Hanford Reach segment of the Columbia River, Wenatchee, Deschutes (OR), Okanogan, Snake, and Methow rivers. Prefers shallow, rocky areas of cobble to boulder substrates and diatom-covered rocks, and feeds by grazing on algae attached to rocks. **Reproduction/Life History** Broadcast external fertilization. Reproduction timing is unknown. (Neitzel and Frest 1989; Neitzel and Frest 1990; Pacific Biodiversity Institute 2007)
Great Columbia River spire snail	*Fluminicola columbiana*	35, 45, 48, 49; other locations unknown	NA	**General Information (Habitats and Feeding)** Also known as the Columbia pebblesnail and ashy pebblesnail, its current range is restricted to rivers, streams, and creeks of the Columbia River basin. It requires clear, cold streams with highly oxygenated water and is generally found in shallow water (less than 5 inches [13 cm] deep) with permanent flow on cobble-boulder substrates. Spire snails live on and under rocks and vegetation in

Common Name	**Scientific Name**	**Water Resource Inventory Area**[a]	**Tidal Reference Area**[b]	**Habitat Requirements and Reproduction Timing**
				the slow to rapid currents of streams where they graze on algae and small crustaceans. **Reproduction/Life History** They are short-lived, usually reaching sexual maturity within a year, at which time they breed and die**.** Unknown reproduction timing. (Neitzel and Frest 1989; Neitzel and Frest 1990; Pacific Biodiversity Institute 2007)
California floater (mussel)	*Anodonta californiensis*	30, 36, 37, 40, 42, 47–49, 52–54, 58–61	NA	**General Information (Habitats and Feeding)** In Washington, it is known to occur in the Columbia and Okanogan rivers and several lakes. Freshwater filter feeder requiring clean, well-oxygenated water for survival that is declining throughout much of its historical range. California floater mussels are intolerant of habitats with shifting substrates, excessive water flow fluctuations, or seasonal hypoxia. **Reproduction/Life History** Spring spawning occurs after adults reach 6–12 years in age. Fertilization takes place within the brood chambers of the female mussel. Fertilized eggs develop into a parasitic stage called glochidia, which attach to species-specific host fish during metamorphosis. After reaching adequate size, juvenile mussels release from the host and attach to gravel and rocks. (Box et al. 2003; Frest and Johannes 1995; Larsen et al. 1995; Nedeau et al. 2005; Watters 1999; WDNR 2006b)
Western ridged mussel	*Gonidea angulata*	01, 03–05, 07–11, 13, 21–42, 44–55, 57–62	NA	**General Information (Habitats and Feeding)** Specific information on this species is generally lacking; reside on substrates ranging from firm mud with the presence of some sand, silt, or clay to coarse gravel in creeks, streams, and rivers. They require constant, well-oxygenated flow, and shallow water (<10 ft [3 m] depth). This species may tolerate seasonal turbidity but is absent from areas with continuous turbidity and is sensitive to

Common Name	Scientific Name	Water Resource Inventory Area[a]	Tidal Reference Area[b]	Habitat Requirements and Reproduction Timing
				water quality changes such as eutrophication or presence of heavy metals. **Reproduction/Life History** During breeding, males release sperm into the water and females must bring this into their shell for fertilization to occur. Larvae called glochidia are released by the female and attach to the gills of fish for 1–6 weeks; postlarval mussels hatch from cysts as free-living juveniles to settle and bury in the substrate. (COSEWIC 2003; WDNR 2006b)

Source: Modified from Jones & Stokes 2006.

a Water Resource Inventory Areas (WRIAs) are administration and planning boundaries for watershed areas, as established and managed by the Washington State Department of Ecology (Ecology). WRIA designations were formalized under WAC 173-500-040 and authorized under the Water Resources Act of 1971, Revised Code of Washington (RCW) 90.54. For WRIA boundary locations and related information, see URL = http://www.ecy.wa.gov/services/gis/maps/wria/wria.htm.

b Tidal Reference Areas as follows (from WAC 220-110-240): 1 = Shelton, 2 = Olympia, 3 = South Puget Sound, 4 = Tacoma, 5 = Seattle, 6 = Edmonds, 7 = Everett, 8 = Yokeko Point, 9 = Blaine, 10 = Port Townsend, 11 = Union, 12 = Seabeck, 13 = Bangor, 14 = Ocean Beaches, 15 = Westport, 16 = Aberdeen, 17 = Willapa Bay.

6 Conceptual Framework for Assessing Impacts

HPA-permitted activities can impact potentially covered species via a suite of potential mechanisms that affect organisms, their habitat, or critical ecological functions. Mechanisms of impact may be direct, such as causing bodily injury or mortality, or indirect, such as altering the habitats upon which these species depend for critical ecological functions (e.g., reproduction, rearing, migration, or refugia).

The identification of impact mechanisms associated with HPA-authorized activities that affect habitat was based on a model developed by Williams and Thom (2001). The conceptual model, presented below as Figure 6-1, provides a simple but effective characterization of the link between impacts and the ecological functions supported by the habitat.

Figure 6-1. Conceptual Framework for Assessment

The Williams and Thom model provides the framework for analysis based on the literature search; in this white paper, our goals for using this framework are to:

- Elucidate impacts associated with each HPA activity
- Determine how those impacts manifest themselves in impacts on habitat and habitat functions utilized by the species that will be addressed in the HCP
- Develop recommendations for impact avoidance, minimization, and mitigation measures that target the identified impacts.

The analysis begins with an impact. An **impact** is defined as an unnatural disturbance to habitat-controlling factors. The impact will exert varying degrees of effect on controlling factors within the ecosystem (Williams and Thom 2001). **Controlling factors** are physical processes or environmental conditions such as light, temperature, salinity, stream or wave energy, substrate, water quality parameters, flow conditions, littoral drift, or channel geomorphology. These controlling factors determine various aspects of habitat structure. **Habitat structures** are the physical attributes of a habitat, for example, substrate type, aquatic vegetation, riparian vegetation or channel width. Habitat structure is linked to habitat processes. **Habitat processes** are defined as the dynamic biogeochemical, biologic, and physical processes which occur within a given aquatic habitat (e.g., shading, cover, sediment trapping, primary production), which are linked to **ecological functions** such as refuge and prey production.

As Williams and Thom (2001) describe the conceptual model, "Controlling factors are physical processes or environmental conditions that control local habitat structure and composition (e.g., vegetation, substrate), including where habitat occurs and how much is present. In turn, habitat structure is linked to support [habitat] processes, such as shading or cover, which are linked to ecological functions. Thus, impacts that affect controlling factors within an ecosystem will be reflected in changes to habitat structure, and will ultimately be manifested as changes to functions supported by the habitat. The effect at the functional level depends upon the level of disturbance and the relative sensitivity of the habitat to the disturbance."

Two examples:

- The habitat structure provided by shoreline overhanging vegetation can provide shade for species using nearshore shallow water and upper beach habitats. This shade serves the ecological function of regulating temperature and supporting the food web through organic litter and insect input.
- The habitat structure of a riparian floodplain will engender increased autotrophy (a habitat process). Subsequently, increased autotrophy serves the ecological function of providing more food resources for fish and invertebrates.

These linkages form the **impact pathway.** Along the impact pathway, alterations to the environment associated with HPA-authorized activities can lead to impacts on the ecological function of the habitat used by HCP species. Impacts can often be viewed as stressors to the environment. Stressors primarily act at the level of the controlling factors. For example, shading from an overwater structure affects the controlling factor *light*, which in turn affects the growth and community structure of aquatic plants that comprise the habitat structure. In certain instances, stressors can act at the ecological function level. For example, overfishing can result in an altered number of prey resources associated with a habitat. **Mechanisms of impact** are the alterations to any of the conceptual framework components along the impact pathway that can result in an impact on ecological function and therefore on HCP species. For each type of HPA-authorized activity, several principal mechanisms of impact were identified for each subactivity type, from a geomorphological, engineering, hydrologic, and biological perspective.

This impact analysis helped to identify the direct and indirect impacts that could potentially affect federally listed species and those species that will be addressed in the HCP.

Table 6-1 identifies the mechanisms of impact that are known to be associated with various activities that are subject to HPA permits. By identifying these impacts and the nature of the risks these impacts exert on HCP species, measures can be implemented to avoid and, if avoidance is not possible, minimize harmful impacts on these species and the habitats that support their growth and survival.

Table 6-1. Mechanisms of impact associated with various HPA-permitted activities.

	Mechanism of impact	**Water crossings**	**Fish passage**	**Flow control structures**	**Bank protection**	**Shoreline modifications**	**Channel modifications**	**Habitat modifications**	**Overwater structures**	**Marinas and terminals**	**Fish screens**
Construction, Maintenance, Operations	Increased noise (pile driving, vessels)	x	x	x	x	x	x	x	x	x	x
	Filling and dredging	x	x	x	x	x	x		x	x	x
	Work area dewatering	x	x	x	x	x	x	x	x	x	x
	Grounding, anchoring, propwash						x		x	x	
	Ambient light modification						x		x	x	
	Entrainment, impingement, handling	x	x	x			x		x	x	x
Hydraulic and Geomorphic	Altered wave energy				x	x	x	x	x	x	
	Altered current, flow velocities	x	x	x	x	x	x		x	x	x
	Altered nearshore circulation				x	x	x	x	x	x	
	Altered groundwater/ surface water/ hyporheic exchange	x	x	x	x	x	x	x	x	x	
	Altered sediment supply, transport, littoral drift			x	x	x	x	x	x	x	

	Mechanism of impact	**Water crossings**	**Fish passage**	**Flow control structures**	**Bank protection**	**Shoreline modifications**	**Channel modifications**	**Habitat modifications**	**Overwater structures**	**Marinas and terminals**	**Fish screens**
	Altered substrate composition	x	x	x	x	x	x	x	x	x	x
	Altered channel processes and geometry	x	x	x	x	x		x	x	x	x
	Altered hydrologic regime from stormwater								x	x	
Riparian Vegetation	Altered shading, solar input, and air temperature	x	x	x	x	x	x	x	x	x	x
	Altered streambank and shoreline stability	x	x	x	x	x	x	x	x	x	x
	Altered allocthonous inputs, including large woody debris	x	x	x	x	x	x	x	x	x	x
	Altered groundwater/ surface water/ hyporheic exchange	x	x	x	x	x	x		x	x	x
	Altered habitat complexity, including from construction	x	x	x	x	x	x	x	x	x	x

	Mechanism of impact	**Water crossings**	**Fish passage**	**Flow control structures**	**Bank protection**	**Shoreline modifications**	**Channel modifications**	**Habitat modifications**	**Overwater structures**	**Marinas and terminals**	**Fish screens**
Aquatic Vegetation	Altered autochthonous production	x	x	x	x	x	x	x	x	x	
	Altered habitat complexity	x	x	x	x	x	x	x	x	x	
Water Quality (including construction , operation and maintenance impacts)	Altered temperature	x	x	x	x	x	x	x			
	Altered dissolved oxygen	x	x	x	x	x	x	x	x	x	
	Altered pH	x	x	x	x	x			x	x	x
	Altered salinity			x		x					
	Altered suspended solids and turbidity, including from construction	x	x	x	x	x	x	x	x	x	x
	Altered nutrient and pollutant loading: eutrophication	x	x	x	x	x	x	x	x	x	
	Altered pollutant loading to sediment and water column: organic contaminants including creosote				x	x			x	x	x
	Altered pollutant loading to sediment and water column: metals (ACZA/CCA)	x	x	x	x	x			x	x	

	Mechanism of impact	**Water crossings**	**Fish passage**	**Flow control structures**	**Bank protection**	**Shoreline modifications**	**Channel modifications**	**Habitat modifications**	**Overwater structures**	**Marinas and terminals**	**Fish screens**
Ecosystem Fragmentation	Altered longitudinal connectivity and fish passage	x	x	x			x	x			x
	Altered lateral connections between rivers and floodplains		x	x	x	x	x	x			
	Altered habitat complexity	x	x	x	x	x	x	x			x

List of subactivity types for activities included in table 6-1:

- Water crossings
  (bridges, culverts, conduits)
- Fish passage
  (fish ladders, culverts, weirs, roughened channels, trap and haul)
- Flow control structures
  (dams, weirs, dikes, levees, tide gates, intakes, outfalls)
- Bank protection/stabilization
  (bulkheads, retaining walls, revetments, toe protection, beach nourishment, subsurface drainage, biotechnical bank protection, bank reshaping or regrading, soil reinforcement, coir and straw logs, integrated approaches.)
- Shoreline modifications
  (groins, jetties, breakwaters)
- Channel modifications
  (dredging, gravel mining and bar scalping, sediment capping, channel creation and alignment)
- Habitat modification
  (beaver dam removal, large woody debris manipulations, spawning substrate augmentation, riparian planting, wetland creation/restoration, enhancement, beach nourishment/contouring, reef creation, eelgrass planting/restoration/enhancement, in-channel and off-channel habitat modifications)
- Overwater structures
  (docks, floats, piers, ramps, wharfs, pilings and non-structural pilings)
- Marinas and Terminals
- Fish screens
  (in-channel, off channel)

To further refine the analysis in each white paper, the exposure-response model (National Conservation Training Center 2004) was incorporated into the impact analysis. The exposure-response model evaluates the likelihood that adverse effects may occur as a result of species exposure to one or more stressors. This model takes into account the effects of exposure for all life-history forms likely to experience stressor exposure, and characterizes the resultant effects across all life-history stages.

The exposure-response model was incorporated as a series of matrices, presented in Appendix A, with results synthesized in Sections 7 and 9 (*Direct and Indirect Impacts* and *Potential Risk of Take*, respectively) of this white paper. In these species-specific exposure-response matrices, each mechanism and submechanism was initially examined and evaluated to:

- Identify and characterize specific impacts or stressors (i.e., nature and magnitude)
- Evaluate the potential for exposure (potential for species to be exposed = stressor timing/duration/frequency coincident with habitat use by the various life-history forms of the species in question)
- Identify the species' anticipated response to stressor based on the exposure parameters and life-history specific sensitivity
- Identify measures that could reduce exposure
- Identify performance standards if appropriate
- Characterize the resulting effects of specific impacts on the various species.

With regard to exposure, standard language was used to indicate when an impact occurs, for how long, and how frequently the stressor or impact occurs; definitions of these terms used in the analysis are listed in Table 6-2.

Based on life-history information, an analysis of potential exposure was completed for each species. This included an analysis of the direct and indirect impacts (associated with each of the impact mechanisms) on the different lifestages of each species and likely responses of the species to these stressors. Impact minimization measures to reduce or avoid submechanism impacts were identified. A final conclusion regarding overall effect of the submechanism/ stressor on the species is also presented in Appendix A. Where information was available, the cumulative effects associated with the major impact mechanisms were also identified (see Section 8 [*Cumulative Effects*]).

The information generated by the exposure-response analysis is used to summarize the overall risk of take associated with the impact mechanisms produced by each subactivity type. The summary risk of take analysis is presented in Section 9 (Potential Risk of Take), which presents the risk of take associated with each subactivity type using: (1) a narrative discussion of the risk of take associated with each subactivity type by the specific associated submechanism of impact; and (2) risk of take assessment matrices that rate the risk of take resulting from each subactivity by impact mechanism and environment type. The risk of take ratings presented in the text and matrices in Section 9 are based on the rating criteria defined in Table 6-3.

This method of risk of take analysis helped identify specific thresholds associated with each of the impacts beyond which take will occur. Summary tables were prepared to indicate the potential for take by species for each major impact mechanism (Section 9, *Potential Risk of Take*).

Based on the identification of impacts and risk of take analysis, additional recommendations (e.g., conservation, management, protection, BMPs) for minimizing or avoiding project impacts or risk of take were developed and presented in Section 11. [*Habitat Protection, Conservation, Mitigation, and Management Strategies*].)

Table 6-2. Definitions of terms used in the exposure-response analysis for this white paper.

Parameter	Description	Exposure	Definition
When	The timing during which stressor exposure occurs (e.g., time of day, season, associated with activity)	-	Defined flexibly as appropriate for each stressor.
Duration	The length of time the receptor is expected to be exposed to the stressor	Permanent	Stressor is permanent (e.g., conversion of habitat to built environment)
		Long-term	Stressor will last for greater than five years to decades (e.g., time required for complete riparian recovery)
		Intermediate-term	Stressor will last from 6 months to approximately 5 years (e.g., time required for beach substrate to recover from construction equipment)
		Short-term	Stressor will last from days to 6 months (e.g., time required for invertebrate community to recolonize following dewatering)
		Temporary	Stressor associated with transient action (e.g., pile driving noise)

Parameter	**Description**	**Exposure**	**Definition**
Frequency	The regularity with which stressor exposure is expected to occur and/or the time interval between exposure	Continuous	Stressor is ongoing and occurs constantly (e.g., permanent modification of habitat suitability)
		Intermittent	Stressor occurs routinely on a daily basis
		Daily	Stressor occurs once per day for extended periods (e.g., daytime structural shading)
		Common	Stressor occurs routinely (i.e., at least once per week or several times per month)
		Seasonal	Stressor occurs for extended periods during specific seasons (e.g., temperature effects occurring predominantly in winter and summer)
		Annual	Stressor occurs for annually for a short period of time
		Interannual–decadal	Stressor occurs infrequently (e.g., pile driving associated with project construction and maintenance)

Table 6-3. Definitions of the terminology used for risk of take determinations.

Risk of Take Code	Potential for Take	Definition
H	High	Stressor exposure is likely to occur with high likelihood of individual take in the form of direct mortality, injury, and/or direct or indirect effects on long-term survival, growth, and fitness potential due to long-term or permanent alteration of habitat capacity or characteristics. Likely to equate to a Likely to Adversely Affect (LTAA) finding.
M	Moderate	Stressor exposure is likely to occur causing take in the form of direct or indirect effects potentially leading to reductions in individual survival, growth, and fitness due to short-term to intermediate-term alteration of habitat characteristics. May equate to an LTAA or a Not Likely to Adversely Affect (NLTAA) finding depending on specific circumstances.
L	Low	Stressor exposure is likely to occur, causing take in the form of temporary disturbance and minor behavioral alteration. If that take is insignificant or discountable, it would equate to an NLTAA finding.
I	Insignificant	Stressor exposure may potentially occur, but the likelihood is discountable and/or the effects of stressor exposure are insignificant. Likely to equate to an NLTAA finding.
N	No Risk	No risk of take ratings apply to species with no likelihood of stressor exposure because they do not occur in habitats that are suitable for the subactivity type in question, or the impact mechanisms caused by the subactivity type will not produce environmental stressors.
?	Unknown	Unknown risk of take ratings apply to cases where insufficient data are available to determine the probability of exposure or to assess stressor response.

7 Direct and Indirect Effects

7.1 Aquatic Vegetation Modifications

7.1.1 Introduction

Aquatic plants are a key element of marine, estuarine, riverine, and lacustrine ecosystems. Aquatic vegetation can be broadly sorted into "marine" and "freshwater" categories. In this paper, the term "marine aquatic vegetation" refers to eelgrass (Zostera spp.); kelps (e.g., bull kelp [Nereocystis leutkeana]); other green, brown, and red macroalgae species; and intertidal wetland vascular plants (i.e., saltmarsh plants) that grow in the marine and estuarine habitats of the state. "Freshwater aquatic vegetation" is submerged and emergent plant material that is rooted below the ordinary high water line (OHWL) of freshwater bodies such as rivers, streams, lakes, ponds, and open-water wetlands.

Aquatic plants provide shelter, habitat and clinging substrate for a variety of aquatic invertebrate species, including insects and zooplankton (Petr 2000). Fish use aquatic plants for cover, and terrestrial wildlife species (in addition to potentially covered species) use emergent aquatic plants for food and habitat (Petr 2000). Aquatic plants provide energy to aquatic ecosystems through photosynthesis, nutrient cycling, and provide food for herbivores and detritivores (Petr 2000). Emergent aquatic vegetation can reduce wave-induced bank erosion (Coops et al. 1996). Aquatic vegetation can modify its physicochemical environment by slowing water velocity, trapping sediment, and altering temperature and water quality (Chambers et al. 1999).

The benefits of aquatic plants listed by Cowx and Welcomme 1998 *(1988?),* in Petr 2000 include:

1. Water purification, both direct (for example, by oxygenation and conversion of toxic ammonia to usable nitrates) and indirect (for example, by plants providing a huge surface area for microbes to do the same tasks).
2. Nutrient recycling, including nutrient removal during the growth season and return during senescence .
3. Physical link between water and air for many invertebrates, e.g., larvae and nymphs of caddis flies, mayflies, and chironomids, which are food for fish and have aquatic larval stages and aerial adults.
4. Refugia for zooplankton, which graze phytoplankton and keep water clear.
5. Cover for a large variety of invertebrates, many of which are food for fish.
6. Cover for fish, which varies as to value and type with the age and species of fish, as well as type of vegetation.
7. Spawning areas and sites of oviposition for many fish species, including Olympic mudminnow, a potentially covered species.
8. Food sources for herbivorous and detritivorous fish, indirect food sources from invertebrate prey living on vegetation surfaces.
9. Effects on flow patterns, i.e., accretion of sediments and deflection of flow, thus providing quiescent waters and faster shallows.

10. Creation of discrete habitat that is as functional as physical structure.

Alterations to aquatic vegetation by HPA-permitted projects may result in altered habitat complexity. This includes changes or reduction in three dimensional habitat structure, refuge and edge habitat, foraging opportunities, and altered autochthonous inputs (alteration of primary productivity, nutrient cycling, and conversion of dissolved organic material into biomass available for grazers, affecting food web productivity). However, HPA-permitted projects are not the only factor that affects the distribution of aquatic vegetation; it is also limited by the ecological conditions of the water body and the requirements of aquatic plant species (Chambers et al. 1999). For example, substrate type can play a role in the survivability of aquatic plants (Koch 2001).

7.1.2 Aquatic Vegetation and Altered Habitat Complexity

Changes to aquatic vegetation may cause alterations to habitat complexity through altered autochthonous (primary productivity and other in-water) inputs, changes in the food web, and changes in habitat structure. In some case, alterations have been linked to specific types of HPA-permitted projects. For both fish and invertebrates, altered habitat could reduce foraging opportunity as well as the availability of nutrients, resulting in decreased survival, growth, and fitness.

7.1.2.1 General Effects: All Environments

HPA-permitted activities may uproot or permanently displace aquatic vegetation during construction, maintenance, and operation of structures. Short-term construction-related impacts may temporarily modify vegetation. In-water ground disturbance has been used as a measure of habitat take in ESA biological opinions (NMFS 2006e). Vessels used during installation, operation, and/or maintenance of HPA-permitted structures may physically disturb submerged vegetation through increased velocity from propeller wash. Lagler et al. (1950, in Carrasquero 2001) reported that outboard motor use has been shown to clear a swath when used within 1 foot (30 centimeters [cm]) of aquatic vegetation. In addition, propeller use may entrain air bubbles and cause sediment suspension that result in a temporary reduction in light availability (Haas et al. 2002).

HPA-permitted structures can cause losses of aquatic vegetation by several pathways in addition to direct disturbance.

- Hydrologic and geomorphic alterations may induce changes in substrate composition and stability and alter habitat suitability for aquatic vegetation. HPA-permitted activities may increase wave reflection, causing unsuitable energy conditions for vegetation, and may flush out small substrate that supports plant growth. Increased flow velocities and substrate characteristics caused by hydraulic and geomorphic modifications can scour algae and damage macrophytes. Conversely, changes in substrate composition with an increase in fine sediment transport can bury aquatic vegetation. Dewatered channels may experience loss of aquatic vegetation, depending on the duration of dewatering and the vulnerability of the affected plants to desiccation and drought stress.

- Increased turbidity is known to compromise the survivability of submerged aquatic vegetation (Parkhill and Gulliver 2002; Terrados et al. 1998) such as eelgrass (Erftemeijer and Lewis 2006) because it limits the amount of sunlight the plants receive. Increased turbidity can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados et al. (1998) noted an approximate 50 percent decline in the number of seagrass species and a precipitous decline in seagrass biomass with a 15 percent increase in the clay content of the sediments.

- Numerous studies have shown that macrophytes and algae in both marine and freshwater environments reduce ambient concentrations of suspended sediment (Abdelrhman 2003; Moore 2004), nutrients (Moore 2004), and metals (Fritioff and Greger 2003). The processing and retention of sediment, nutrients, and pollutants in aquatic systems is accelerated by the presence of aquatic vegetation (Clarke 2002). Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve. Aquatic vegetation does more than reduce nutrient and sediment concentrations; the plants themselves can sequester harmful trace metal pollutants and are frequently planted in wetland treatment systems with that intended function. In a comparative study of heavy metal uptake in terrestrial, emergent, and submerged vegetation, Fritioff and Greger (2003) noted that submerged vegetation was efficient at removing zinc, copper, cadmium, and lead from influent stormwater.

- HPA-permitted activities may affect aquatic vegetation through changes to ambient light. The growth and survival of submerged aquatic plants (benthic and planktonic) are dependent on identified light levels, known as photosynthetically active radiation (PAR). Light levels falling below PAR are known to limit the photosynthesis for a suite of aquatic photosynthesizers, such as diatoms, epiphytes, eelgrass, and other autotrophs important to HCP species. On average in Puget Sound, instantaneous mid-day PAR greater than approximately 150 $\mu M/m^2/sec$ is required to maintain eelgrass growth. Instantaneous PAR of approximately 325 $\mu M/m^2/sec$ is required to support maximum densities (Thom et al. 1998). The light requirements of different species vary, but reduced light in the littoral zone of freshwater environments can potentially limit aquatic vegetation (Chambers et al. 1999). The availability of light is also a crucial parameter for seagrasses and other marine aquatic vegetation (Hall et al. 1999). The loss of vegetation from shade could pose an indirect effect on the HCP species that rely on the species supported by that vegetation. Light limitations can lead to a local reduction in autochthonous primary production (i.e., organic matter produced by aquatic plants within a water body) and a reduction in the other functions of aquatic vegetation, including cover, substrate for invertebrate species, and food for herbivores (Hruby et al. 1999).

Structures can limit ambient light availability. Many types of HPA-permitted structures, including overwater structures, marinas, terminals, bridges, and culverts have been shown to shade the area underneath, within, and adjacent to the structures. The orientation of the structures, density of the structure (solid or open), culvert height, height above water (for bridges, overwater structures, marinas and terminals), length and width of the structure, size and spacing of supports (piers or piles), water depth, and tidal range all affect the extent and degree of shading (Nightingale and Simenstad 2001b).

Effects on fish associated with decreased light penetration and decreased vegetation include reduced foraging success and altered migration timing due to a reduction in primary productivity and associated reductions in prey species. Decreased light penetration and shading impacts on vegetation could also result in fish expending increased energy or being increasingly exposed to predation because of loss of suitable habitat and cover.

- HPA-permitted activities may introduce noxious aquatic weeds, or may alter ecosystems in ways that sustain their growth.

7.1.2.2 Ecosystem-Specific Effects: Marine and Estuarine

Marine aquatic vegetation is a fundamental structural and ecological component of the nearshore ecosystem and substantially influences the physical and chemical properties of the nearshore environment (Nightingale and Simenstad 2001b).

The Washington State Hydraulic Code Rules (WAC 220-110-250) designate eelgrass, kelp, and intertidal vascular plants as saltwater habitats of special concern and require that hydraulic projects result in no net loss of these habitats. The hydraulic code rules require that overwater structures be designed or located to avoid shading or other impacts that could result in the loss of eelgrass and kelp habitat (WAC 220-110-240 through 330).

Eelgrass is associated with important rearing habitats for a suite of marine fishes, such as Pacific cod, Pacific salmon, rockfish, Pacific herring, walleye pollock, and rockfish (Gustafson et al. 2000; Murphy et al. 2000; Nightingale and Simenstad 2001a; Simenstad et al. 1999).

7.1.2.2.1 Vegetation and primary productivity

The basis for nearly all life in the sea is the photosynthetic activity of aquatic autotrophs such as algae, cyanobacteria, benthic microalgae, benthic macroalgae (kelps and seaweeds), and seed plants (such as seagrasses, mangroves, and salt-marsh plants) (Nybakken and Bertness 2005). These photosynthesizers rely on the availability of light for photosynthesis (Govindjee 1975). Marine aquatic vegetation forms an important component of the base of the aquatic (and terrestrial) food web (Seliskar and Gallagher 1983).

Vegetation growth, survival, and depth of water column penetration are directly related to light availability, with the maximum depth of plant survival increasing with increasing light penetration into the water column (Dennison 1987; Dennison et al. 1993; Kenworthy and Haunert 1991). The level of light penetration is dependent upon water depth, water clarity (dissolved particulates reflect, refract, absorb, and scatter incident radiation), and light absorption by plant material in the water column.

Because of its importance to habitat structure and food webs, eelgrass is one type of aquatic vegetation that has received a lot of attention in the Pacific Northwest. Eelgrass is considered a "saltwater habitats of special concern" (WAC 220-110-250).

Phillips (1984) and Wyllie-Echeverria and Phillips (1994) describe eelgrass ecology in the Pacific Northwest. Two species of eelgrass (*Zostera* spp.) grow in Washington State: the native eelgrass, *Zostera marina*, and the smaller Asian species, *Zostera japonica* (Wyllie-Echeverria and Phillips 1994). Typically, *Z. marina* grows at lower elevations than *Z. japonica* and may either form extensive beds covering many acres or exist in smaller patches (Phillips 1984). Native eelgrass distributions range from approximately +2 feet mean lower low water (MLLW) to -22 feet MLLW (PSAT 2001), although light penetration conditions in many portions of Puget Sound typically limit the lower elevation to the range of -8 feet to -12 feet MLLW. *Z. japonica* is generally found at higher elevations than *Z. marina* and typically grows in patches or a narrow fringe (Phillips 1984). Eelgrass typically grows in sand and mud substrates in sheltered or turbulent waters (Phillips 1984).

Many species of macroalgae also grow in the marine waters of Washington, generally attached to rocky substrates and always within the nearshore photic zone (Kozloff 1983). Macroalgae have a wider elevation range than eelgrass, and macroalgae such as rockweed (*Fucus gardneri*) can grow as high as mean higher high water (MHHW). At the other extreme, brown algae (kelp) may grow at elevations as low as -100 feet MLLW where the water is clear enough and the substrate supports algal attachment (WDNR 2004). However, in Puget Sound, the depth to which sufficient light penetrates to support plant growth (i.e., photic zone) is considered to be -33 feet (-10 m) MLLW (PSNERP 2003).

In the estuarine salt marshes of Washington, bulrushes (*Scirpus* spp.) and sedges (*Carex* spp.) are the most commonly found vegetation types (Kozloff 1993). Other common vegetation types include rushes (*Juncus* spp.), salt grass (*Distichlis spicata*), dune grass (*Leymus mollis*), and pickleweed (*Salicornia virginica*) (Seliskar and Gallagher 1983).

Eelgrass can retard current velocity at the sediment-water interface, allowing fine particulates to settle (Phillips 1984). This action can affect sediment dynamics and local sediment characteristics, favoring continued growth and survival of eelgrass (Phillips 1984). The vertical structure of kelp forests also affords some dissipation of wave energy (Jackson 1984), which can offer some shoreline protection for other sensitive shoreline habitats.

Simenstad et al. (1999) describe the potential effects of propeller wash on eelgrass. Flume studies have shown that current velocities of 1 to 1.5 knots (50 to 80 centimeters per second [cm/sec]) may be sufficient to cause sediment disturbance around eelgrass and that velocities of 3.5 knots (180 cm/sec) can cause severe erosion of eelgrass patch edges. However, eelgrass patches in Puget Sound thrive in currents of up to 3.9 knots (200 cm/sec) (Thom et al. 1996, in Nightingale and Simenstad 2001b). The effect of vessels used during installation of HPA-permitted structures on eelgrass and macroalgae depends on local current and sediment conditions, as well as on maximum current velocity at the sediment surface. In addition to the direct effects of propeller wash on submerged vegetation, propeller wash can entrain bubbles and suspend sediment, causing reduced light availability that can indirectly affect eelgrass and, to a lesser extent, macroalgae (Simenstad et al. 1999).

7.1.2.2.2 Food web interactions

Marine littoral vegetation is important for the colonization of organisms that are important prey resources for potentially covered species such as Newcomb's littorine snail, Pacific sand lance, Pacific herring, Pacific cod, northern abalone, surf smelt, steelhead and coastal cutthroat trout, salmon (pink, chum, coho, and Chinook), Olympia oyster, bull trout, Dolly Varden, rockfish, longfin smelt, eulachon; and walleye pollock (Busby et al. 1996; Chambers et al. 1999; Couch and Hassler 1989; Gardner 1981; Goetz et al. 2004; Johnson et al. 1999; Larsen et al. 1995; Matthews 1987; Myers et al. 1998; Norris 1991; NRS Canada 2004; Orth et al. 1984; Pauley et al. 1988; WDNR 2006a, 2006b; West 1997).

Eelgrass, macroalgae, and saltmarsh plants are very productive and support marine food webs through their plant biomass and detritus. (Phillips 1984; Seliskar and Gallagher 1983; Simenstad 1983). Eelgrass provides a necessary structural surface for a community of epibenthic organisms, making eelgrass communities one of the most productive ecotones in the Pacific Northwest (Ferraro and Cole 2007). These epibenthic prey assemblages of copepods, such as the harpacticoids, are known to feed on bacteria, epiphytes, plant detritus, and diatoms. They in turn are prey for juvenile salmon, shiner perch, and other species (Cordell 1986; Nightingale and Simenstad 2001b; Simenstad and Salo 1980; Simenstad et al. 1980, 1988; Thom et al. 1988 (1989?)).

Studies of eelgrass communities in Padilla Bay show that a specific group of copepods (*Harpacticus uniremis* and other copepods of the genera *Zaus* and *Tisbe*) is unique to the eelgrass epiphyte assemblage and is the principal prey of juvenile chum salmon, Pacific herring, Pacific sand lance, and surf smelt (Nightingale and Simenstad 2001b), with *Harpacticus* spp. less likely to be found in low light conditions and *Tisbe* spp. found in areas high in detritus, irrespective of light levels.

Pacific herring spawn on the blades of eelgrass and macroalgae (WDNR 2006a). Pacific herring are, in turn, a direct food source of larger predators, including adult Chinook salmon, bull trout (Nightingale and Simenstad 2001b), Pacific hake (Bailey 1982; NMFS 1990; Quirollo 1992; McFarlane and Beamish 1986, in NRC 2001), Pacific lamprey, rockfish (WDNR 2006a), and many other species (WDNR 2006a). Thus, a reduction in Pacific herring productivity could produce indirect adverse impacts on a number of additional potentially covered fish species.

In studies on outmigrating juvenile chum in Hood Canal, Simenstad and Salo (1980) found juvenile chum fry (1.2–1.8 in [30–45 mm]) feeding extensively upon small, densely distributed harpacticoid copepods, selecting the largest copepods available. Similarly, Miller et al. (1976) reported that juvenile chum fed predominantly on epibenthic harpacticoid copepods. As the fish grew in size, their diet content was composed of larger epibenthos and pelagic crustaceans. Consistent with other studies, the highest densities of harpacticoid copepods occurred in magnitudes 4–5 times higher in eelgrass stands than in sand habitat without eelgrass. Also, recent dietary investigations in central Puget Sound found that juvenile Chinook salmon fed extensively on a polychaete worm (*Platynereis bincanaliculata*) that builds tubes on eelgrass and macroalgae (Brennan et al. 2004).

In a study of the Drayton Harbor marina, Thom et al. (1988) reported that during the study period from September 1987 to October 1988, juvenile salmon density was by far the highest on April 29 at the eelgrass habitat site that was also found to support, by far, the highest salmon prey density and the highest epibenthos density on that date. Total fish density increased dramatically immediately following a peak in maximum epibenthos and the most rapid increase in Zostera biomass (Thom et al. 1988 *(1989?)*). The limitation of habitat for key prey resources likely affects migration patterns and the survival of many juvenile fish species. For smaller fish less than 1.97 in (50 mm) in length, residence times along particular shorelines are thought to be a function of prey abundance (Simenstad and Salo 1980).

The complex structure of eelgrass communities and their associated epifauna and epiflora are thought to limit the success of predators that typically associate and feed in unvegetated communities (Heck and Orth 1980, in Nightingale and Simenstad 2001b; Heck and Thoman 1984). Given the strong association of important fish prey resources with eelgrass, reductions in eelgrass extent or vigor may reduce prey resources for fish.

Northern abalone could be impacted by alterations to marine aquatic vegetation because of its typical association with kelp beds (Gardner 1981). Northern abalone typically cling to rocks in thick kelp beds (Pacific Biodiversity Institute 2006). Larger northern abalone feed on detached, drifting algae and their growth rate can be influenced by the amount of algae available (Jamieson 1999).

Juvenile Dungeness crab, an important salmonid prey species, show a preference for eelgrass compared to other benthic habitats; this is thought to be due in part to the abundance of food items in eelgrass habitat (Pauley et al. 1989).

7.1.2.2.3 Predator avoidance and refugia

Salt marshes provide important feeding opportunities and predator refuge for fishes. For example, juvenile chum and fall Chinook salmon have been observed to selectively forage for chironomid larvae and adults in a restored marsh, suggesting that restored salt marsh provides indirect benefits to fisheries by production of preferred prey items through detritus-based food chains (Shreffler et al. 1992). Seliskar and Gallagher (1983) identified seven of the potentially covered species as associated with marsh habitats: Chinook, chum, coho, sockeye, and pink salmon, longfin smelt, and surf smelt. In the Fraser River estuary, most of the fish populations are dependent upon a small array of benthic invertebrates, many of which are tidal marsh inhabitants (Northcote et al. 1979). Salt marshes are highly sensitive to human disturbance (Seliskar and Gallagher 1983). The destruction of salt marshes can significantly impact the refuge habitat available for fish and the important functions that salt marshes provide at the base of the food web. Levy and Northcote (1981) concluded that juvenile salmon use the entire length of tidal channels and, therefore, that bank protection structures along any part of a marsh area can significantly reduce the estuary's capacity as a rearing area.

Aquatic vegetation, in particular eelgrass, is important cover for juvenile fish and invertebrates (Phillips 1984). Bostrom and Mattila (1999) note that eelgrass plays a role in protecting invertebrates from both fish and avian predators. It is uncertain what role eelgrass plays in the protection of potentially covered invertebrate species, but the generality of the existing work in the field would suggest that a loss of eelgrass would result in increased predation of those species that co-occur with eelgrass.

7.1.2.2.4 Spawning substrate

Both eelgrass and macroalgae provide substrate for herring spawning (Bargmann 1998). Herring is a key species in the nutrient and energy dynamics of the Puget Sound environment, providing an important link between zooplankton and larger predators, including Chinook salmon, bull trout, and other salmonid species (Bargmann 1998).

7.1.2.2.5 Habitat structure

Eelgrass and macroalgae provide vertical structure in nearshore marine habitats and facilitate several important ecological functions. Nightingale and Simenstad (2001b) note that eelgrass provides shelter and influences the physical and chemical properties of the nearshore environment. The vertical structure off the seafloor (substrate) that marine aquatic vegetation provides is important habitat for fish and invertebrates, including salmon, forage fish, and juvenile rockfish (Phillips 1984). The vertical structure of marine aquatic vegetation can also trap and stabilize sediments, and vegetation that grows through the entire water column, such as bull kelp, can dissipate wave energy before it hits the shoreline (Jackson 1984.)

Blackmon et al. (2006) provides a synopsis of research on the use of seagrass and kelp habitats by fish, including many of the marine potentially covered species. This synopsis noted that forage fish and juvenile Pacific salmon species preferentially use eelgrass over other habitats. Juvenile salmon are found in kelp habitat as well (Blackmon et al. 2006). Rockfish (*Sebastes* sp.) produce planktonic larvae that settle in eelgrass, shallow kelp beds, and floating kelp mats (Blackmon et al. 2006). Juvenile rockfish occupy shallow vegetated habitats, especially areas with eelgrass and kelp, during the summer growing period (Byerly et al. [no date]; Murphy et al. 2000), likely due to the enhanced forage opportunities and refuge from predators that the vertical structure can provide. Juvenile Dungeness crab (young of the year) (a major prey species for some rearing salmonids) are more frequently found in eelgrass and *Ulva* beds than in other benthic habitats, and eelgrass beds are considered valuable nursery habitat for Dungeness crab. This is thought to be due in part to the abundance of food items in eelgrass habitat (Pauley et al. 1989).

Newcomb's littorine snail is found primarily in association with a narrow band of nearshore intertidal habitat that contains certain marsh plant species (Larsen et al. 1995). Because detailed reproductive and habitat needs of Newcomb's littorine snail are not known, it might be conservatively assumed that Newcomb's littorine snail is subject to habitat loss if marine aquatic vegetation, particularly marsh plant species, is displaced by bank protection structures.

7.1.2.2.6 Large Woody Debris and Aquatic Vegetation

There have been few studies of the influence of LWD in marine environments. Depositional coastal areas throughout Washington are replete with large and small wood. Much of this wood has originated from forest harvesting practices from the turn-of-the-century. Therefore, it is difficult to determine a "natural" concentration of LWD for coastal areas. It is generally thought that small quantities of LWD can increase nearshore habitat quality; however, no significant research has been conducted to verify this hypothesis. Maclennan (2005) studied wood dynamics in two sites in Washington: Elger Bay on Camano Island, and Sullivan-Minor marsh in Padilla Bay. She noted that drift logs, or LWD, tend to accumulate in the upper portions of the intertidal zone and that the wracked pieces in the seaward sections of the estuaries became mobilized during storms and high tides. The mobile logs raked the surface of the marsh and prevented the establishment of emergent vegetation. Meanwhile, stable logs in the higher elevation sections of the estuaries acted as substrate for upland vegetation which could not have otherwise become established (Maclennan 2005). Other studies have shown that these stable logs can act as habitat for macroinvertebrates including grass shrimp (Everett and Ruiz 1993), chironomids, other dipterans, talitrids, homopterans, coleopterans, and collembolans and that beaches with abundant wrack have higher taxa richness than equivalent artificially hardened beaches (Sobocinski 2003). Outside of these few studies, little research has measured the ecological importance of LWD for the formation and persistence of marine aquatic vegetation communities.

7.1.2.2.7 Water Quality

In tidal areas, seagrasses have been linked to improved water quality. As an example, Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve.

Increased turbidity is known to compromise the survivability of submerged aquatic vegetation (Parkhill and Gulliver 2002; Terrados et al. 1998) (Bash et al. 2001; Newcombe and Jensen 1996) such as eelgrass (Erftemeijer and Lewis 2006) because it limits the amount of sunlight the plants receive. It can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). Eelgrass is associated with important rearing habitats for a suite of marine fishes including Pacific cod, Pacific salmon, rockfish, Pacific herring, and walleye pollock (Nightingale and Simenstad 2001a; Simenstad et al. 1999). Increased turbidity can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados et al. (1998) noted an approximate 50 percent decline in the number of seagrass species and a precipitous decline in seagrass biomass with a 15 percent increase in the clay content of the sediments.

In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados et al. (1998) noted that seagrass species richness and community leaf biomass declined sharply with a 15 percent increase in clay content of the sediments. Numerous studies have shown increased biomass of invertebrate (Cardoso et al. 2007; Seitz et al. 2005) and vertebrate species (Ferraro and Cole 2007; Pihl et al. 2006) in association with seagrass presence; thus, sedimentation-related negative impacts on seagrass arising from the construction or presence of bank protection structures would likely affect the HCP species by decreasing available nearshore habitat.

7.1.2.3 Ecosystem-Specific Effects: Riverine

7.1.2.3.1 Vegetation and primary productivity

Riverine aquatic vegetation includes benthic algae (microscopic unicellular algae, forming thin layers or assemblages called periphyton). Macrophytes include angiosperms rooted in the stream bottom, along with mosses, and other bryophytes. These include many forms such as rooted plants with aerial leaves, floating attached plants with submerged roots, floating unattached plants, and rooted submerged plants (Murphy 1998).

Although aquatic primary production is sometimes underrated due to the small amount of algae and plants present in many streams, it is a basic energy source for freshwater ecosystems. A small algal biomass in a stream can support a much larger biomass of consumers due to the rapid turnover in biomass (Hershey and Lamberti 1992; McIntire 1973; Murphy 1998).

Light is a controlling factor of primary production, with increased light and nutrients stimulating primary production and increasing the production of invertebrates and fish, or decreased light reducing overall productivity (Murphy 1998). Although aquatic plants and algae are adapted to low light intensity, there is a critical light level at which respiration equals photosynthesis, known as the compensation point. Below the compensation point, such plants would eventually starve to death as they would respire food faster than they could produce it (Murphy 1998).

In a study of macrophyte impact on sediment and nutrient retention in Danish streams, Sand-Jensen (1998) reported that dense-stemmed macrophytes created conditions conducive to sediment deposition and that the sediments retained within the macrophyte stands were fine-grained and nutrient-rich. He noted that enrichment of sediment within macrophyte beds relative to the surrounding substratum was 0.1597 lb organic matter per ft^2 (780 g $/m^2$), 0.006 lb nitrogen per ft^2 (30 g/m^2) and 0.005 lb phosphorus per ft^2 (25 g per m^2). Therefore, any large-scale modification of aquatic vegetation will likely result in increased suspended sediments, increased nutrient loading, and changes in hyporheic exchange, all adversely affecting HCP species.

Freshwater macrophytes are also known to modify their physicochemical environment by slowing water flow, trapping sediments, and altering temperature and water chemistry profiles. Through the trapping of particles by plant fronds, they also change the nature of the surrounding sediments by increasing the organic content and capturing smaller grain size than substrate in uncolonized areas (Chambers et al. 1999). In addition, submerged aquatic vegetation has been shown to increase hyporheic exchange (White 1990), which in turn will promote nutrient cycling.

7.1.2.3.2 Food web interactions

Potentially covered species that depend on freshwater aquatic vegetation for one or more of their life-history stages include green sturgeon, white sturgeon, California floater and western ridged mussels, mountain sucker, lake chub, great Columbia River limpet, pygmy whitefish, leopard and Umatilla dace, Olympic mudminnow, bull trout, Dolly Varden, and Pacific salmon (Frest and Johannes 1995; Hallock and Mongillo 1998, 1999; Hughes and Peden 1989; Mongillo and Hallock 1998; Watters 1999). More specifically, adhesive eggs of the Olympic mudminnow rely on attachment to aquatic vegetation for egg and larval development (Coutant 2004).

Freshwater aquatic primary producers, such as benthic algae, macrophytes, and phytoplankton, play key roles in the trophic support of stream ecosystems. The uptake of carbon, nitrogen, and phosphorous by aquatic vegetation and conversion into biologically available biomass provides important nutrients to fish and invertebrate consumers. This aquatic primary production is the source of autochthonous (instream) organic matter and part of the source of allochthonous (terrestrial) matter in each stream reach.

A reduction in aquatic vegetation may alter nutrient loading within stream and river ecosystems. Modification or removal of aquatic vegetation will result in reduced autochthonous production, which provides important energy sources in aquatic food webs. Primary productivity in fresh water is reduced commensurate with the degree that shade reduces the light level of the aquatic environment.

Invertebrate grazing of vegetation by snails, caddisflies, isopods, minnows, and other organisms is an important pathway of energy flow. For stream herbivores, for example, benthic diatoms are the most nutritious and easily assimilated food source (Lamberti et al. 1989). The availability of algae regulates the distribution, abundance, and growth of invertebrate scrapers (Gregory 1983; Hawkins and Sedell 1981), an important food source for fish.

Although terrestrial and adult aquatic insects are important (Bjornn and Reiser 1991), juvenile salmon in streams have been found to be primarily supported by autochthonous organic matter (Bilby and Bisson 1992). Invertebrate scrapers and collector–gatherers are known to be most frequently eaten by salmonids (Bilby and Bisson 1992; Hawkins et al. 1983; Murphy and Meehan 1991).

The density of coho salmon fry in the summer has been found to be directly related to the abundance of algae. A high density of fry can result from smaller feeding territories (Dill et al. 1981) due to increased invertebrate prey (Hawkins et al. 1983; Murphy et al. 1981). Increases in vertebrate production have been found to occur primarily in the spring and early summer, coincident with the primary production cycle of benthic algae (Murphy 1998). Therefore, removal of, permanent disturbance, or light limitations to algal communities could have an adverse effect on local freshwater ecosystems and the HCP species that depend upon these ecosystems. In the case of coho salmon fry, the reduction in prey area (i.e., smaller feeding territories) results in a direct effect on the fitness, growth, and survival of the affected fry.

7.1.2.3.3 Predator avoidance and refugia

Aquatic vegetation has been found to reduce predation rates by providing cover refuge for prey fish (Gregory and Levings 1996).

The effects of aquatic vegetation removal on invertebrates are not well known. However, the California floater in the Eel River (California) is commonly associated with aquatic vegetation, which is used for protection from high flows (Howard and Cuffey 2003).

7.1.2.3.4 Habitat structure

Activity that mechanically removes or by other means affects aquatic vegetation may reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. Indirect impacts from the removal of aquatic vegetation may cause increased nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity.

In particular, the Olympic mudminnow may be most vulnerable to changes in aquatic vegetation because it requires areas with dense aquatic vegetation (Harris 1974) and has been shown to no longer occupy areas where vegetation was removed (Mongillo and Hallock 1999).

7.1.2.4 Ecosystem-Specific Effects: Lacustrine

7.1.2.4.1 Vegetation and primary productivity

Lacustrine aquatic vegetation includes algae and plants. Lakes host both floating and benthic algae (microscopic unicellular algae, forming thin layers or assemblages called periphyton). Lacustrine macrophytes include both angiosperms and bryophytes, in many forms such as rooted plants with aerial leaves, floating attached plants with submerged roots, floating unattached plants, and rooted submerged plants (Murphy 1998). Large woody debris acts as a substrate for periphyton growth and secondary production in lakes (Smokorowski et al. 2006.)

Impacts from shading aquatic vegetation in lakes are similar to those found in marine, estuarine, and riverine systems (Garrison et al. 2005; Jennings et al. 2003; White 1975).

The introduction of noxious weeds can be a concern in aquatic environments (Chambers et al. 1999; WNWCB 2006). In Washington, concern about the impacts of aquatic noxious weeds seems to be higher for lacustrine ecosystems than for either riverine or marine systems. Under the right conditions, noxious weeds can out-compete native vegetation and can reduce habitat quality for native fish species (Chambers et al. 1999).

For example, the Lake Washington shorelines have developed extensive beds of Eurasian milfoil since it was first observed in the lake in 1974 (WNWCB 2005). Eurasian milfoil can cause several adverse habitat conditions, including reduced dissolved oxygen and reduced access to habitat (Chambers et al. 1999). Interlake transfer from boats is thought to be the chief means by which Eurasian milfoil is spread (WNWCB 2005).

The role of emergent vegetation in the lacustrine nearshore on the productivity of habitat has not been explored in detail in Washington waters or with regard to HCP species.

7.1.2.4.2 Food web interactions

Potentially covered species that depend on freshwater aquatic vegetation for one or more of their life-history stages include white sturgeon, lake chub, pygmy whitefish, bull trout, and Chinook and sockeye salmon (Frest and Johannes 1995; Hughes and Peden 1989; Mongillo and Hallock 1998; Mongillo and Hallock 1999; Watters 1999).

Aquatic primary producers, such as benthic algae, macrophytes, and phytoplankton, play key roles in the trophic support of freshwater ecosystems. The uptake of carbon, nitrogen, and phosphorous by these plants provides important nutrients to fish and invertebrate consumers. This aquatic primary production is the source of autochthonous organic matter and part of the source of allochthonous matter within all lakes. Grazing of these primary producers by snails, caddisflies, isopods, minnows, and other grazers is an important pathway of energy flow. For herbivores, benthic diatoms are the most nutritious and easily assimilated food source (Lamberti et al. 1989). The availability of algae regulates the distribution, abundance, and growth of invertebrate scrapers (Hawkins and Sedell 1981), an important food source for fish. Although no studies were found that have linked the productivity of invertebrate communities with an increase in fish numbers, these observations may explain the differences in fish density along disturbed lakeshores (Jennings et al. 1999).

Juvenile salmonids, as drift-feeders, focus on food from autochthonous pathways. Invertebrate scrapers and collector-gatherers are known to be most frequently eaten by salmonids (Bilby and Bisson 1992; Hawkins et al. 1983; Murphy and Meehan 1991). Although no studies were found that have linked the productivity of invertebrate communities with an increase in fish numbers, these observations may explain the differences in fish density along disturbed lakeshores (Jennings et al. 1999).

Freshwater macrophytes are known to modify their physiochemical environment and contribute to habitat complexity by changing surface water patterns, slowing water flow, trapping sediments, and altering temperature and water chemistry profiles. Through the trapping of particles by plant fronds, they also change the nature of the surrounding sediments by increasing the organic content and capturing smaller grain size than substrate in uncolonized areas (Fonseca and Bell 1998, Carrasquero 2001, Chambers et al. 1999). In addition, submerged aquatic vegetation has been shown to increase hyporheic exchange, which in turn will promote nutrient cycling. For example, White (1990) found that dense vegetation hummocks promote upwelling of porewater into the rootmass, which provides nutrients that encourage and sustain vegetation growth. In these ways, aquatic vegetation can contribute to habitat complexity and food web productivity.

7.1.2.4.3 Predator avoidance and refugia

As with marine aquatic vegetation, lacustrine aquatic vegetation has been shown to dramatically enhance the density of benthic invertebrates in temperate lakes through the relative protection from predation it affords (Beckett et al. 1992). In fact, even invasive vegetation can provide the same functions without compromising invertebrate productivity (Gardner et al. 2001).

Reductions in lacustrine aquatic vegetation may affect potentially covered species by loss of protection from predators due to a loss of cover.

7.1.2.4.4 Habitat structure

As in riverine systems, activity that mechanically removes or by other means affects aquatic vegetation may reduce the sediment, nutrient, and pollutant retention and reduction capabilities of the system. Indirect impacts from the removal of aquatic vegetation may cause increased nutrient and pollutant loading to receiving waters, which could exacerbate eutrophic conditions and/or metals toxicity.

One potentially covered freshwater invertebrate species would be impacted by loss of habitat if freshwater aquatic vegetation were disturbed or removed; the larvae of the California floater mussel in Curlew Lake depend primarily on the Tui chub (*Gila bicolor*) as a host (Pacific Biodiversity Institute 2006), and juvenile Tui chub typically stay close to vegetation until they are longer than 0.5 inch (Wydoski and Whitney 2003).

7.1.3 Activity-Specific Effects

7.1.3.1 Bank Protection

Bank protection structures in marine waters have the potential to affect marine aquatic vegetation through direct or indirect disturbance and displacement. Saltmarsh vegetation growing along the upper intertidal shoreline fringe, the backshore, or in larger saltmarsh complexes is highly susceptible to disturbance through the potential hydraulic disconnection, burial, or conversion of habitat associated with bank protection structures. Marine aquatic vegetation may be uprooted or displaced as the vegetation itself may be removed for projects constructed below the ordinary high water line (OHWL). During construction, vegetation may be trampled or subject to spills from construction equipment in the project area or work corridor. Vegetation may also be disturbed as a result of vessel grounding or propeller wash (Lagler et al. 1950, in Carrasquero 2001; Haas et al. 2002), which can entrain air bubbles and introduce sediment suspension (Haas et al 2002). Because light availability is a fundamental requirement for eelgrass and macroalgae growth, turbid conditions limit their ability to thrive.

Bank protection projects can indirectly impact marine aquatic vegetation distributions through the sediment coarsening that occurs as a result of the altered wave regime or disruption of littoral drift (e.g., Johannessen et al. 2005). The sediment coarsening may result in substrates too large to support marine aquatic vegetation.

Bank protection activities in freshwater settings have the potential to affect freshwater aquatic vegetation through direct disturbance or indirect disturbance and displacement. Bank protection structures positioned waterward of the OHWL will encroach upon areas that support or potentially would support freshwater aquatic vegetation.

7.1.3.2 Conduits

A study of eelgrass recovery at four submarine cable landings in Skagit and San Juan counties demonstrated the disturbance that occurs with conduit (electrical transmission cable) installation and some factors affecting recovery (Jones & Stokes 2005; Wones and Cziesla 2004). This project disturbed eelgrass habitat along a cable route that ranged from 3 to 15 feet in width. Although the upper beaches of these cable landings returned to the pre-excavation appearance within a few weeks through wave action and littoral drift, the disturbance through eelgrass beds required one to four years to fully recover. The sites where eelgrass recovered most rapidly included a site with only a narrow band of eelgrass and a site with an extensive, robust eelgrass bed. Eelgrass was slowest to recover at the deepest extent of eelgrass growth, where light availability may limit the rate of eelgrass recruitment and growth (Jones & Stokes 2002, 2005).

7.1.3.3 Culverts

Impacts to habitats and species may occur through the loss of vegetation resulting from construction of new water crossings.

Maintenance or retrofitting of culverts is not expected to have any appreciable effect on aquatic vegetation. This is due to the fact that the footprint of the existing structure has already imposed its effects on the vegetation community. Therefore, most work within the footprint of an existing structure will not displace or affect vegetation. Some extended effects are possible, however, if removal or replacement results in hydraulic and geomorphic modifications that change habitat suitability.

7.1.3.4 Fishways

The potential for the fishway subactivity type projects to result in aquatic vegetation modification is generally more limited in comparison to other types of HPA-permitted activity types. This is due to the fact that the in-water footprint of fishway structures is typically small and, in the case of passage around man-made barriers, the fishway is integrated into the barrier structure, and the incremental effect on aquatic vegetation is negligible. Some extended effects on aquatic vegetation are possible, however, if the fishway design results in hydraulic and geomorphic modifications that change habitat suitability for vegetation. For example, exit flows from the fishway may lead to localized alteration of substrate conditions. In general, however, any effects associated with aquatic vegetation modifications are expected to be limited in extent and insignificant in terms of stressors imposed on the aquatic community.

7.1.3.5 Roughened Channels

It is useful to note that roughened channels are often implemented to aid passage of fishes in high-velocity channels. This type of environment is less than ideal for aquatic vegetation and typically does not support extensive aquatic vegetation communities. Therefore, the direct effects of construction are expected to be limited overall. In general, effects associated with aquatic vegetation modifications are expected to be limited in extent and insignificant in terms of stressors imposed on the aquatic community.

The in-water footprint of roughened channels is typically small or nonexistent (in the case of channels constructed through uplands). Some extended effects on aquatic vegetation are possible, however, if the roughened channel design results in hydraulic and geomorphic modifications that change habitat suitability for vegetation immediately upstream or downstream. For example, exit flows from the roughened channel may lead to localized alteration of substrate conditions. Construction-related water quality impacts in the form of elevated suspended sediments may also occur.

7.1.3.6 Overwater Structures

7.1.3.6.1 Sediment disturbance associated with overwater structures

Vessels associated with overwater structures may cause propeller wash, which may disturb, damage, or uproot aquatic vegetation. The effect of vessels associated with overwater structures on eelgrass and macroalgae depends on local current and sediment conditions, as well as on maximum current velocity at the sediment surface. Flume studies have shown that current velocities of 20 to 31 inches per second (50 to 80 centimeters per second [cm/sec]) may be sufficient to cause sediment disturbance around eelgrass and that velocities of 71 inches per second (180 cm/sec) can cause severe erosion of eelgrass patch edges. However, eelgrass patches in Puget Sound thrive in currents of up to 79 inches per second (200 cm/sec) (Thom et al. 1996, in Nightingale and Simenstad 2001b).

Thom et al. (1996), in studying the impacts of passenger-only ferries at the Vashon Island terminal, found that at 187 feet (57 meters) from the boat, it is likely that the propeller wash has little effect on existing eelgrass. Thom et al. (1996) also concluded that currents with a velocity above 2.46 feet/second (0.75 meters/second) damaged eelgrass by eroding away overlying sediment and that currents above 3.61 feet/second (1.1 meters/second) caused extensive damage to eelgrass rhizomes.

The vertical and horizontal distance at which current velocity may affect eelgrass depends on the size and shape of the propeller. The U.S. Army Corps of Engineers' Regional General Permit No. 6 prohibits the construction or installation of floats or float support pilings within a 4-foot depth elevation between the top of the float stopper and the elevation of the landward-most edge of a macroalgae bed or eelgrass (USACE 2005). This restriction applies to a zone 25 feet wide on both sides of the float projecting waterward horizontally from the float (USACE 2005).

In a study of 44 shallow, sheltered soft-bottom inlets on the Baltic Sea, Eriksson et al. (2004) found that recreational boating activities and ferry boat traffic significantly affected aquatic vegetation both in percent cover and species richness. These effects were largely attributed to water movement and prop wash generated by boating activity, and turbidity blocking light transmission to vegetation. These findings are consistent with numerous studies on the effects of disturbance on eelgrass beds (Sargent et al. 1995; Loflin 1995; Haas et al. 2002).

Studies in Florida related to the impacts of boating activity on seagrass indicate that the largest concentration of scarring occurs in waters less than 6.5 feet (2 meters) deep (Sargent et al. 1995, in Dawes et al. 2004). In Florida, many shallow flats and mud banks are severely eroded due to constant scarring, ship groundings, chronic wave action from boats, and water-current scouring (Kruer 1994, in Dawes et al. 2004). Removal of seagrass roots and rhizomes due to prop scarring also destabilizes sediments and resuspension occurs, thereby lowering water transparency and retarding seagrass regrowth into the scar (Durako et al. 1992, in Dawes et al. 2004).

Studies in Florida have also found that fragmentation of seagrass beds caused by propeller scarring did not appear to have any consistent effects on some animal populations over a one-year period, as long as the seagrass patch sizes were greater than 3 square feet (1 square meter) (Bell et al. 2002, in Dawes et al. 2004). The numbers of pinfish (*L. rhomboides*), pipefish (*Syngnathus scovelli*), and eight species of epibenthic shrimp were similar in moderately scarred (6 percent to 31 percent loss of the beds) and non-scarred seagrass beds in Tampa Bay (Dawes et al. 2004). The results of these studies suggest that propeller scars that fragment seagrass beds may enhance certain faunal development caused by edge effects along the cuts, as long as they are not too severe (Dawes et al. 2004). Nevertheless, a recent study of scarring in a *T. testudinum* bed in

Puerto Rico revealed a negative effect of scarring on crabs and molluscs up to 16 feet (5 m) from the scar. Also, shrimp species within the scar differed from those in the non-scarred seagrasses. Fish populations did not show an effect from the scarring (Dawes et al. 2004). Further studies are clearly needed to define the effects of moderate scarring compared to those of severe scarring on seagrass productivity (Dawes et al. 2004).

In addition to direct disturbance, propeller wash can entrain bubbles and suspend sediment, causing reduced light availability that can indirectly affect eelgrass and, to a lesser extent, macroalgae (Simenstad et al. 1999). Propeller wash has also been found to resuspend nutrients and contaminants, such as nitrogen and phosphorous, that can stimulate algal blooms as well as increase turbidity (Haas et al. 2002; Michelsen et al. 1999; Parametrix 1996; Thom et al. 1997; Thom and Shreffler 1996). The resulting increased algal growth may lead to eutrophication and reduction of dissolved oxygen levels due to respiration during desiccation of the algal material. The result of increased turbidity and lower dissolved oxygen has effects throughout the food web.

7.1.3.6.2 Effects on light availability associated with overwater structures, bridges and culverts

Many types of HPA-permitted structures, including overwater structures, marinas, terminals, bridges, and culverts cast shade and thus may potentially affect aquatic vegetation.

Where shading reduces photosynthetically active radiation (PAR) levels, eelgrass and macroalgae growth will be affected and may be impaired or prevented (Nightingale and Simenstad 2001b; Penttila and Doty 1990). Penttila and Doty (1990) found that fixed and floating docks and structures largely eliminate existing eelgrass and macroalgae, even when the structures are only partially shading. Such shading impacts to eelgrass can be seen to occur in as little as 18 days (Backman and Barilotti 1976, in Nightingale and Simenstad 2001b), although light reduction capacity varies depending on combinations of both structure design and environmental factors. For example, Penttila and Doty (1990) found no apparent eelgrass loss due to shading under a floating dock secured by anchors and chains. In that case, it was thought that, given the winds and current of the site, the degree of movement allowed by the anchor-chain system resulted in no area beneath the dock being continuously shaded, thereby reducing the effect of shade on the eelgrass bed.

Thom et al. (1998) analyzed the photosynthetically active radiation (PAR) levels at seven Washington State ferry terminal sites. and found that no eelgrass was found where instantaneous mid-day PAR levels were less than about 100 micro-moles of photons within the PAR range of wavelengths striking a square meter in one second (µM/m2/sec). They also found that the lowest eelgrass shoot densities were found where instantaneous mid-day PAR was less than 150 µM/m2/sec. Thom et al. (1998) found that maximum shoot densities required instantaneous PAR of 325 µM/m2/sec. PAR intensities less than about 300 µM/m2/sec can be limiting to eelgrass, whereas intertidal macroalgae may be limited by PAR less than 400 to 600 µM/m2/sec (Thom and Schreffler 1996, cited in Simenstad et al. 1999). Subtidal macroalgae can survive lower light levels and may only be limited by PAR less than 100 μM/m2/sec (Luning 1981, cited in Simenstad et al. 1999). Light availability observed at multiple terminals in Puget Sound ranges from 0 to almost 9 PAR units under the Kingston terminal, with Port Townsend, Clinton, and Vashon terminals rangiingfrom 0.5 to1 PAR, and light availability varying significantly at different points under the terminal (Simenstad et al.. 1999)

Many other studies have focused on light limitation effects under ferry terminals in Washington State (Backman and Barilotti 1976; Blanton et al. 2001; Bulthuis and Woelkerling 1983; Burdick and Short 1999; Fresh et al. 1995; Glasby 1999; Haas et al. 2002; Loflin 1995; Olson et al. 1997; Parametrix 1996; Penttila and Doty 1990; Reish 1961; Shafer 1999, 2002; Shreffler and Moursund 1999; Simenstad et al. 1988; Thom et al. 1996, 1997; Thom and Shreffler 1996; Visconty 1997). Some of these studies focused on prey resource availability for juvenile salmon at ferry and shipping terminals in Puget Sound and found that these structures negatively affect prey availability (Blanton et al. 2001; Haas et al. 2002). In a study comparing light levels under the Clinton, Bainbridge, and Southworth terminals, Blanton et al. (2001) found terminal orientation to the arc of the sun, terminal height and width, construction materials, and piling type to influence the shadow cast on the nearshore environment and its effect on the littoral vegetation. Similarly, in a study comparing the Clinton, Edmonds, and Port Townsend terminals, Thom and Shreffler (1996) found similar light limitation effects on littoral vegetation, and Haas et al. (2002) found effects of underwater light limitation under the Bainbridge, Clinton, and Southworth ferry terminals.

In a study on the influence of piers and bulkheads on the aquatic organisms in Lake Washington, White (1975) reported that light levels under piers were consistently lower, and this light reduction resulted in reduced phytoplankton production. In general, the larger the overwater structure, the larger the area of light limitation and reduction of phytoplankton production. Also, macrophytes were generally absent or sparse under piers. In the fall, grazing invertebrates were found outside of piers where macrophytes were abundant; in the spring, grazing invertebrates were found under piers where they could graze on periphyton during the spring (White 1975).

Studies conducted at piers on lakes in Wisconsin also report the loss of submerged lake vegetation due to dock shading (Garrison et al. 2005; Jennings et al. 2003). A dock study in Montauk, New York (Ludwig et al. 1997) reported the exclusion of eelgrass near a floating pier due to insufficient light in the float's impact zone. Burdick and Short (1999) found that floating docks severely impact eelgrass. Three of the four floating docks they studied had no rooted eelgrass under them. Increased dock height above the bottom was identified to be the most important dock characteristic correlating to eelgrass bed quality, and a similar effect would be expected for bridges. Burdick and Short (1999) also found light to be the most important variable affecting canopy structure (i.e., shoot density and height) and eelgrass bed quality.

7.1.3.7 Pilings

Pilings that support overwater structures such as bridges may also reduce eelgrass recruitment and survival through biotic interactions with the piling reef community

(Nightingale and Simenstad 2001b). Pilings in marine waters become encrusted with mussels and other sessile organisms. Shell material from these organisms is then deposited around the pilings over time, altering the local substrate (Nightingale and Simenstad 2001b). The piling reef habitat provides food for sea stars and the shell bottom is prime settling habitat for juvenile Dungeness crabs (Nightingale and Simenstad 2001b). The burrowing activities of large numbers of crabs can affect the establishment of eelgrass (Nightingale and Simenstad 2001b).

7.1.3.8 Dredging

Dredging operations directly entrain and eliminate aquatic vegetation from the dredged site. In both marine and freshwater environments, dredging generally removes or disturbs benthic vegetation. Altering the vegetation also potentially alters the distribution and abundance of prey resources, the availability of refugia, and water quality benefits provided by vegetation to various species. The time period between dredging activities will determine the capacity for recolonization of the submerged aquatic vegetation.

A special case of freshwater dredging is the use of a hydraulic suction dredge for the specific goal of vegetation management in freshwater environments. The suction dredge is typically used to remove invasive aquatic vegetation; however, the technique is not plant-specific and can entrain aquatic vegetation that is beneficial to HCP species.

A potential threat from dredging is the burial of aquatic vegetation in adjacent areas due to increased sedimentation (Wilber and Clarke 2001). Dredging operations broadcast fine sediments over a broad area, the extent of which depends on tidal currents and basin geometry (Hossain et al. 2004). Burial of the submerged aquatic vegetation community in marine environments can lead to decreased primary and secondary productivity, which in turn may affect overall food-web productivity.

Another threat to aquatic vegetation is sedimentation from open-sea or nearshore beach nourishment disposal sites. Although open-sea disposal has been demonstrated to be protective of aquatic vegetation when disposal rates are regulated to produce low rates (less than 1 inch) of sedimentation over the course of the project (Simonini et al. 2005), the extreme sensitivity of eelgrass to burial makes it vulnerable even in areas that are distant from the disposal site (Mills and Fonseca 2003).

Gravel mining and sediment capping may also result in the burial of aquatic vegetation.

7.1.3.9 Channel Creation and Realignment

Aquatic vegetation will initially be destroyed by channel creation and alignment activities. Studies of gravel augmentation show that this vegetation recovers quickly following minor disturbances (Merz et al. 2004). Elevated nutrient levels during the growing season will accelerate primary production and post-project vegetation recovery. Consequently, initial impacts on aquatic vegetation may only be ephemeral and thus the associated impact on HCP species will be minimal. The loss of aquatic vegetation as part of channel creation and alignment includes altered autochthonous production and altered habitat complexity.

7.1.3.10 Dams

Dams can cause losses of aquatic vegetation by several pathways. Increased velocities can scour algae downstream and damage macrophytes, reducing cover for fish. Second, changes in substrate composition with an increase in fine sediment transport can bury aquatic vegetation. Finally, modification may occur directly from construction and maintenance activities.

Dam removals have shown contrasting results with respect to impacts on aquatic vegetation. In one study, the removal of a dam resulted in increases in aquatic macrophytes leading to increased cover and habitat for fish (Hill et al. 1993). However, dam removal may kill off some vegetation from sediment released during the removal process and subsequently cause abrasion of roots and stems (Wood and Armitage 1997). Dam removals may increase the scour of algae and insects, thereby altering food web interactions and food quality, particularly if algae or leaf accumulations are buried (Doeg and Koehn 1994; Newcombe and MacDonald 1991; Wood and Armitage 1997).

7.1.3.11 Outfalls

Effluent flows from outfalls may cause scour and loss of aquatic vegetation. Increased nutrient loading from outfall effluent may result in an indirect effect on aquatic vegetation. Increased nutrient loading can stimulate primary productivity and lead to decreased dissolved oxygen. Eutrophication can lower dissolved oxygen concentrations and negatively affect HCP species.

7.1.3.12 Beaver Dam Removal

Macrophytes which populate many floodplain ponds are most abundant in the littoral zones (Ahearn et al. 2006). Consequently, when a beaver pond is dewatered much of the aquatic vegetation is left isolated in dry upland areas. The result is a substantial decrease in aquatic vegetation and the standing vegetation stock (Naiman et al. 1988). This, coupled with the loss of shallow water refugia, may represent the single greatest impact on fish and invertebrates associated with beaver extirpation and dam removal.

Beaver dam removal will decrease the amount of shallow water habitat available for macrophyte and algal production. The decrease in autogenic production will cause a shift in the form of the food resources available to primary consumers. Shredders and scrappers may replace grazers, and the changing macroinvertebrate population may have an overall impact on higher trophic level species (Winkelmann et al. 2007). Altered food web complexity may, as a result, affect foraging opportunities for HCP species that feed on aquatic macroinvertebrates in beaver ponds and similar environments during some phase of their life history.

7.1.3.13 Large Woody Debris Placement/Movement/Removal

Large woody debris acts as a substrate for periphyton growth and secondary production in channels (Atilla et al. 2003; Bowen et al. 1998; Hoffmann 2000; Warmke and Hering 2000) and lakes (Smokorowski et al. 2006), but few studies have shown that the addition of this substrate increases primary productivity in the system as a whole [see (Atilla et al. 2003)] for an exception in marine systems). However, LWD can function to induce floodplain and backwater connection with the main channel. By creating more connectivity with productive backwaters, the presence of wood within the channel can increase aquatic productivity and access to aquatic vegetation. Consequently, LWD removal may be associated with a decrease in access to these productive areas.

7.1.3.14 Spawning Substrate Augmentation

Aquatic vegetation will initially be destroyed in areas where gravel is placed. But, research has shown that this vegetation recovers quickly (Merz et al. 2004). Many augmentations occur below dams. Dams with hypolimnetic release points can elevate growing-season nutrient concentrations in downstream reaches (Ahearn et al. 2005). Elevated nutrient levels during the growing season will accelerate primary production and post-project vegetation recovery. Consequently, initial impacts on aquatic vegetation may only be ephemeral and thus the associated impact on HCP species will be minimal.

7.1.3.15 In-Channel/Off-Channel Habitat Creation/Modifications

The creation of shallow water edge habitat (a goal of some in-channel and off-channel habitat modifications) will generally lead to increased autochthonous production. In a study of rehabilitated incised streams in Denmark, Pederson et al. (2006) found that a re-profiling of the stream bank to create shallow water habitat resulted in increased macrophyte densities. The study concluded that shallow and wide banks allowed for increased autogeny and for a larger migration of macrophytic species from the stream banks into the streams, thereby enhancing species diversity within the stream channel. Rehabilitation of off-channel habitat will promote the exchange of aquatic vegetation between the channel and the floodplain. Schemel et al. (2004) noted a 2-fold increase in chlorophyll-a concentrations (a measure of algal biomass) in a floodplain in California versus the adjacent Sacramento River. Meanwhile, Ahearn et al. (2006) noted chlorophyll-a concentrations five times greater in a Cosumnes River, California floodplain than in an adjacent channel. These studies and others (Hein et al. 2004; Tockner et al. 1999) indicate that autochthonous production is elevated in floodplains relative to channels. This biomass is transported between backwater areas and the main channel during floods, and the resultant increase in food resources within the channel can bolster aquatic food webs and benefit the HCP species.

Freshwater aquatic vegetation provides shelter and clinging substrate for a variety of prey of the HCP species, including mollusks and many fishes (nonsalmonids) with a strong association with this vegetation (Petr 2000). Some of the HCP species which have been shown to commonly utilize vegetated habitat in off-channel areas include cutthroat trout, bull trout, sculpins, Dolly Varden, sockeye and coho salmon, and dace species (Pollock et al. 2003). In fluvial systems, slack water productive patches provide habitat diversity in systems that are otherwise dominated by high velocity, less-productive habitat (Johnston and Naiman 1990). This is also likely the case in estuarine/slough areas. The creation of off-channel habitat will increase access to aquatic vegetation and benefit many of the HCP species.

7.1.3.16 Wetland Creation/Restoration/Enhancement

Riparian and coastal wetlands provide extensive shallow water habitat where macrophyte and algal species may thrive. Due to shallow water depths, warm temperatures, and protection from the erosive power of flooding and wave action, wetlands are ideal habitat for aquatic vegetation. In turn, aquatic vegetation provides food resources and structural habitat for both fish and invertebrate species. Wetland creation, restoration, and enhancement will increase aquatic vegetation habitat and the ecosystem functions associated with it.

The Schemel (2004), Ahearn (2006), Hein et al. (2004), Tockner et al. (1999), Petr (2000), Pollock et al. (2003), and Johnson and Naiman (1990) studies discussed in the section on “In-Channel/Off-Channel Habitat Creation/Modifications” are also pertinent to aquatic vegetation in wetlands. As with off-channel habitat creation, autochthonous production in wetlands can bolster aquatic food webs and benefit many of the HCP species, especially in areas where aquatic productivity within the channel is low. Nutrient-poor, or oligotrophic systems are common in Washington State, especially since a major pathway of nutrient import, namely marine-derived nutrients from salmon spawning, has drastically decreased over the past 100 years (Naiman et al. 2002). Consequently, any habitat modification measure which increases productivity in waters connected to these oligotrophic systems will benefit aquatic species.

Riparian and estuarine wetlands are characterized by abundant macrophyte growth which has been shown to provide habitat for coho (Swales and Levings 1989), marine invertebrates (Seitz et al. 2005), and numerous other species. The creation, restoration, or enhancement of estuarine and riparian wetlands may increase the amount of, and improve access to, this habitat. The potential increase in habitat may alleviate any density dependent mortality which may be occurring within the system (Greene and Beechie 2004).

Organisms which can access floodplain wetland habitat may benefit from the increased productivity which characterizes those systems by taking advantage of improved foraging and hunting opportunities. However, organisms that remain within the channel may also benefit from riparian wetland productivity because the systems tend to be hydraulically linked and carbon export to the channel will occur during periods of high flow.

Wetlands have been called the kidney of the landscape (Mitsch and Gosselink 2000) because of their ability to sequester and transform nutrients and pollutants. If they are designed with a sufficiently high hydraulic residence time, wetlands can retain pollutants and reduce upland pollutant loadings, including sediment, nutrients, and toxic substances, to downstream aquatic resources. This in turn will benefit any HCP species which may reside within the downstream freshwater or marine systemHickey and Doran (2004) review buffer widths and include information that is also applicable to wetlands.

7.1.3.17 Beach Nourishment

The primary potential impact on aquatic vegetation from beach nourishment is burial. Eelgrass, the dominant seagrass in western Washington waters, is sensitive to large sedimentation rates (Mills and Fonseca 2003). Therefore, if there is eelgrass present near the activity site, it is possible that the activity will cause an aquatic vegetation loss. As eelgrass is a crucial component to the life history of several HCP species (Phillips 1984), these activities would likely limit or reverse gains in fish populations from the addition of loose, mobile foreshore materials.

7.1.3.18 Reef Creation/Restoration/Enhancement

Depending on the depth of placement, artificial reefs could bury or block light to aquatic vegetation. However, this would only occur in the footprint of the proposed reef. Therefore reefs should not be placed in seagrass meadows.

7.1.3.19 Eelgrass and Other Aquatic Vegetation Creation/Restoration/Enhancement

Aquatic vegetation planting is the least often used technique among all habitat modification subactivities. This rehabilitation technique has been applied with mixed results in Puget Sound (Thom et al. 2005), but it is an attractive alternative considering the relative lack of impacts as compared to other nearshore restoration techniques (e.g., beach nourishment).

Eelgrass is the dominant species of macrophyte in Puget Sound, and is also an important member of the nearshore ecosystem elsewhere in Washington marine waters (Phillips 1984). The effects of introducing eelgrass on other aquatic plant communities are unknown. However, it is clear that eelgrass provides important surface for epiphytes which serve as an important food source for juvenile salmonids and forage fish (Phillips 1984). There are numerous synergistic and competitive interactions between eelgrass and other aquatic flora and fauna (Nelson and Lee 2001). These interactions control the relative abundance of each particular species. As a result, when eelgrass in not present, the likelihood of phytoplankton blooms markedly increases (Hiratsuka et al. 2007).

7.1.3.20 Vessel Traffic

Grounding, anchoring, and/or prop wash can cause benthic disturbance and turbidity, eelgrass and macroalgae disturbance, and freshwater aquatic vegetation disturbance. These effects have been well documented (e.g., Thom et al. 1997 and Thom and Shreffler 1996).

Prop wash or waves produced by boats and personal watercraft can cause shoreline erosion (Gatto and Doe 1987; Mason et al. 1993; Hurst and Brebner 1969) and increase suspended sediments and turbidity (Hilton and Phillips 1982; Kennish 2002; Yousef et al. 1980; Yousef 1974). The effect of this increased turbidity may decrease light levels, which could potentially affect the growth rates of submerged vegetation, upon which most HCP species depend. Turbidity is also known to be associated with fish respiratory injury (Berg and Northcote 1985) and increased sediment deposition on downstream spawning habitat (Hartman et al. 1996).

7.2 Construction, Maintenance and Operations

7.2.1 Introduction

The construction, maintenance and operation activities mechanism of impact includes several submechanisms of impact, capturing a range of activities that are short-lived but intensive and are required to build facilities as well as to provide or maintain access to these facilities. The five submechanisms that have been identified for analysis in these white paper(s) include:

(1) Elevated underwater sound:
   - Pile driving sound
   - Non-pile driving sounds

(2) Channel/work area dewatering

(3) Navigation/maintenance, filling and dredging

(4) Grounding, Anchoring and or Prop Wash

(5) Ambient Light Modifications

These five sub-mechanisms are likely to affect HCP species as they occur within the water. However, activities occurring landward of water bodies can also affect HCP species. These activities include staging and equipment access, including the use of heavy equipment around the wetted perimeter in riparian (marine, lacustrine, and riverine environments) and floodplain areas.

7.2.2 Elevated Underwater Sound

Projects permitted under the WDFW HPA program can produce underwater noise through a variety of mechanisms. These mechanisms include construction-related noise impacts from impulsive sources (i.e., short duration, high intensity noise from sources such as pile driving or materials placement), as well as continuous noise sources (e.g., vessel or equipment operation). This section summarizes existing information on sources of underwater noise, how underwater noise is characterized, existing and proposed effects thresholds, and the magnitude of noise stressors associated with typical project construction and maintenance activities. This discussion is derived in part from a summary of current science on the subject developed by WSDOT (2006a).

7.2.2.1 Measurement of Underwater Sound

Units of measurement: dBpeak and dBrms

Underwater sound levels are measured with a hydrophone, or underwater microphone, which converts sound pressure to voltage, which is then converted back to pressure, expressed in pascals (Pa), pounds per square inch (psi), or decibel (dB) units. Derivatives of dB units are most commonly used to describe the magnitude of sound pressure produced by an underwater noise source, with the two most commonly used measurements being the *instantaneous peak sound pressure level* (dB_{PEAK}) and the *root mean square* (dB_{RMS}) pressure level during the impulse, referenced to 1 micropascal (re: 1µPa) (Urick 1983). The dB_{PEAK} measure represents the instantaneous maximum sound pressure observed during each pulse. The RMS level represents the square root of the total sound pressure energy divided by the impulse duration, which provides a measure of the total sound pressure level produced by an impulsive source. The majority of literature uses dB_{PEAK} re: 1µPa sound pressures to evaluate potential injury to fish.

Up until recently, the USFWS and NOAA Fisheries have used both dB_{PEAK} (for injury) and dB_{RMS} (for behavioral effects) re: 1µPa threshold values to evaluate adverse injury and disturbance effects on fish, marine mammals, and diving birds (Stadler 2007; Teachout 2007). dB_{RMS} values are used to define disturbance thresholds in fish species, meaning the sound pressure level at which fish noticeably alter their behavior in response to the stimulus (e.g., through avoidance or a “startle” response). dB_{PEAK} values are used to define injury thresholds in salmonids, meaning the sound pressure level at which injury from barotraumas may occur (i.e., physical damage to body tissues caused by a sharp pressure gradient between a gas or fluid-filled space inside the body and the surrounding gas or liquid).

Based on a new agreement in Principle for Interim Criteria for Injury to Fish from Pile Driving Memorandum, June 12, 2008, Federal Highways Administration (FHWA), lead by research performed by NOAA and USFWS have set new criteria for effects from pile driving. These new pile driving criteria are shown below in Table X1,***Interim Hydroacoustic Criteria for the Physical and Behavioral Effect From Pile Driving on Fish***. Based on this new criteria, with emphasis on the accumulated sound exposure level (SEL) (SEL provides a measure of total sound pressure exposure and is expressed as dBre: $1uPa^2$/second), effects from repeated pile strikes, it is likely the current findings for the effects of pile driving in the OWS/Marinas white papers does not provide accurate measurement of impact to fish life.

As noted in the agreement: “The agreed upon criteria indentify sound pressure levels of 206 dB peak and 187 dB accumulated sound exposure level (SEL) for all listed fish except those that are less than 2 grams.” Fish less than 2 grams “the criteria for the accumulated SEL will be 183 dB.”

The criteria used for the onset of physical injury and adverse behavioral effects are in the table below. The onset of physical injury uses dual criteria - peak pressure(dBpeak) and sound exposure level (SEL). The onset of physical injury is expected if either of these criteria are exceeded. The criteria for accumulated sound exposure level (SEL) is based upon the mass of the fishes under consideration. If fishes smaller than 2 grams are present, the more conservative SEL (183 dB (re: $1uPa^2$*sec) criteria may be required.

Table 7-1: Interim Hydroacoustic Criteria for the Physical and Behavioral Effect From Pile Driving on Fish. June 11, 2008.

Effect	Metric	Fish mass (grams)	Theshold
Onset of physical injury	Peak pressure	N/A	206 dB (re: 1 μPa)
	Accumulated Sound Exposure Level (SEL)	≥ 2 g	187 dB (re: 1μPa^2•sec)
		< 2 g	183 dB (re: 1μPa^2•sec)
Adverse behavioral effects	Root Mean Square Pressure (RMS)	N/A	150 dB (re: 1 μPa)

Sound behaves in much the same way in air and in water, attenuating gradually over distance as the receptor moves away from the sound source. However, underwater sound exhibits a range of behaviors in response to environmental variables (Urick 1983). For example, sound waves bend upward when propagated upstream into currents and downward when propagated downstream in the direction of currents. Sound waves will also bend toward colder, denser water. Haloclines and other forms of stratification can also influence how sound travels. Sound shadows created by bottom topography and intervening land masses or artificial structures can, under certain circumstances, block the transmission of underwater sound waves. In freshwater systems, sound propagation is often influenced by depth and channel morphology. Underwater sound does not transmit as effectively when water depths are less than 3 feet due to the amplitude of the sound pressure wave (Urick 1983). Because underwater sound does not travel around obstructions, bends in a river or large changes in gradient will truncate sound propagation. This will limit the physical extent of sound related impacts.

Underwater sound attenuation, or transmission loss, is the reduction of the intensity of the acoustic pressure wave as it propagates, or spreads, outward from a source. Propagation can be categorized using two models, spherical spreading and cylindrical spreading. Spherical (free-field) spreading occurs when the source is free to expand with no refraction or reflection from boundaries (e.g., the bottom or the water surface). Cylindrical spreading applies when sound energy spreads outward in a cylindrical fashion bounded by the sediment and water surface. Because neither model applies perfectly in any given situation, most experts agree that a combination of the two best describes sound propagation in real-world conditions (Vagle 2003).

Currently, USFWS and NOAA Fisheries are using a practical spreading loss calculation, which accommodates this view (Stadler 2007; Teachout 2007). This formula accommodates some of the complexity of underwater sound behavior, but it does not account for a number of other factors that can significantly affect sound propagation. For example, decreasing temperature with depth can create significant shadow zones where actual sound pressure levels can be as much as 30 dB lower than calculated because sound bends toward the colder, denser water (Urick 1983). Haloclines, current mixing, water depth, acoustic wavelength, sound flanking (i.e., sound transmission through bottom sediments), and the reflective properties of the surface and the bottom can all influence sound propagation in ways that are difficult to predict.

Given these complexities, characterizing underwater sound propagation inherently involves a large amount of uncertainty. An alternative calculation approach, known as the Nedwell model (not used by USFWS or NOAA Fisheries), indirectly accounts for some of these factors. Nedwell and Edwards (2002) and Nedwell et al. (2003) measured underwater sound levels associated with pile driving close to and at distance from the source in a number of projects in English rivers. They found that the standard geometric transmission loss formula used in the practical spreading loss model did not fit well to the data, most likely because it does not account for the aforementioned factors that affect sound propagation. They developed an alternative model based on a manufactured formula that produced the best fit to sound attenuation rates measured in the field. This model thereby accounts for uncharacterized site-specific factors that affect sound attenuation, but does not explicitly identify each factor or its specific effects. Because there is considerable uncertainty regarding how to model the many factors affecting underwater sound propagation, and this would require site specific information that cannot practically be obtained in many instances, the Services (i.e., USFWS and NOAA Fisheries) use the more conservative practical spreading loss model in ESA consultations (Stadler 2007; Teachout 2007).

Though installation of piles “in the dry” might or might not require an HPA, it is worthwhile to note potential impacts of pile installations adjacent to water bodies. It is often assumed that installation of piles “in the dry” will result in minimal, or undetectable, sound production levels (SPL) in the water. Monitoring data from impact installation indicates that SPLs in the adjacent waterbody can be significantly elevated (Battelle Marine Sciences Laboratory, 2004; Reyff, 2006.). Hydroacoustic monitoring during impact installation of 48-inch steel piles that were 5 m from a river in California detected SPLs as high as 201 dB_{peak} and 188 dB_{rms} at 10 meters from the pile (Reyff 2006). As sound pressure travels through the substrate, its waveform might be altered, resulting in longer (and therefore less damaging) rise times, but this has not been adequately investigated. Also, during monitoring of vibratory installation of piles adjacent to a river, Reyff (2006) noted that there was clearly noticeable vibration in the river. Project-Related Sound Sources

The underwater sound produced by an HPA permitted project, either during construction or operation, is defined by the magnitude and duration of underwater sound above ambient sound levels. The action area for underwater sound effects in ESA consultations is defined by the distance required to attenuate construction sound levels to ambient levels, as calculated using the practical spreading loss calculation or other appropriate formula provided in evolving guidance from USFWS and NOAA Fisheries on this subject.

Although there are many sources of sound in the underwater environment, the following are typical sources of underwater sound associated with in-water construction:

- Ambient sound levels
- Project construction and maintenance: equipment operation and materials placement
- Project operation: vessel operation, equipment operation.

7.2.2.2 Ambient Sound Levels

Ambient underwater sound levels serve as the baseline for measuring the disturbance created by project construction or maintenance. Both natural environmental sound sources and mechanical or human-generated sound contribute to the ambient or baseline sound conditions within and surrounding a project site. Therefore, these sound measurements, particularly those recorded in the vicinity of ferry terminals and other high-activity locations, are indicative of the level of sound levels that could be produced by project construction and operation.

Ambient sound levels have been measured in several different marine environments on the West Coast and are variable depending on a number of factors, such as site bathymetry and human activity. For example:

- Measured ambient levels in Puget Sound are typically around 130 dBpeak (Laughlin 2005).

- Ambient levels at the Mukilteo ferry terminal reached approximately 145 dBpeak in the absence of ferry traffic (WSDOT 2006a).

- Ambient underwater sound levels measured in the vicinity of the Friday Harbor ferry terminal project ranged between 131 and 136 dBpeak (WSDOT 2005).

- Carlson et al. (2005) measured the underwater baseline for the Hood Canal and found it to range from 115 to 135 dBRMS.

- Heathershaw et al. (2001) reported open-ocean ambient sound levels to be between 74 and 100 dBpeak off the coast of central California.

These ambient sound levels are typical conditions, and typical conditions can be punctuated by atypical natural events. For example, lightning strikes can produce underwater sound levels as high as 260 dBpeak in the immediate vicinity (Urick 1983).

Limited data are available on ambient sound levels in freshwater environments, but it is reasonable to conclude that they vary considerably based on available information. High-gradient rivers, fast-flowing rivers, and large rivers and lakes with significant human activity are likely to produce more sound than lakes and slow-flowing rivers in more natural environments.

Burgess and Blackwell (2003) measured ambient sounds in the Duwamish River in Seattle, Washington, (averaged over 20 seconds to 5 minutes) and found the sound to vary between 110 and 130 dB continuous sound exposure level (SEL) (SEL provides a measure of total sound pressure exposure and is expressed as dB re: 1µPa^2/second). Amoser and Ladich (2005) measured ambient sound levels in the mainstem Danube River, a smaller, fast-flowing tributary stream, a small lake, and a quiet river backwater. The river and stream represented fast-flowing habitats, the lake and backwater quiet, slow-flowing habitats. Sound behavior was complex. They found that ambient sound levels ranged from as low as 60 to as high as 120 dBpeak in the fast-flowing habitats, depending on the sound frequency (lower frequency sound was typically louder). Ambient sound in the slackwater habitats was considerably lower, ranging from 40 to 80 dBpeak across the frequency range (again with lower frequency sounds being loudest).

7.2.2.3 Materials Placement (Pile Driving) Sound Levels

Sources of underwater sound resulting from materials placement during HPA permitted projects have received little direct study. Of the potential sources of construction-related sound, pile driving has received the most scrutiny because it produces the highest intensity stressors capable of causing sound-related injury. Other sources of underwater sound, such as dumping of large rock or underwater tool use, have received less study. Therefore, available data on sound levels associated with pile driving are presented here as a basis for comparison.

Two major types of pile driving hammers are in common use, impact hammers and vibratory hammers.

- Impact Hammer: There are four kinds of impact hammers: diesel, air or steam driven, hydraulic, and drop hammer (typically used for smaller timber piles). Impact hammers produce sharp sound pressure waves with rapid rise times, the equivalent of a punch versus a push in comparison to vibratory hammers. The sharp sound pressure waves associated with impact hammers represent a rapid change in water pressure level, with greater potential to cause injury or mortality in fish and invertebrates.

- Vibratory Hammer. Vibratory hammers produce a more rounded sound pressure wave with a slower rise time. Because the more rounded sound pressure wave produced by vibratory hammers produces a slower increase in pressure, the potential for injury and mortality is reduced. (Note that while vibratory hammers are often used to drive piles to depth, load-bearing piles must be “proofed” with some form of impact hammer to establish structural integrity.) The changes in pressure waveform generated by these different types of hammers are pictured in Figure 7-1.

Piling composition also influences the nature and magnitude of underwater sound produced during pile driving. Driven piles are typically composed of one of three basic material types: timber, concrete, or steel (although other special materials such as plastic may be used). Steel piles are often used as casings for pouring concrete piles. Sound levels associated with each of these types of piles are summarized in Table 7-1.

Reference sound levels are denoted in both dB_{PEAK} and dB_{RMS} values, at the specified measurement reference distance.

Table 7-2. Reference sound levels by structure type.

Material Type and Size	Impact Hammer Type	Reference Sound Levels[a]		Environment Type	Source
		dB_{PEAK}	dB_{RMS}		
12-inch timber	Drop	177 @ 10 m	165 @ 10 m	Marine	(Illingworth and Rodkin 2001)
24-inch concrete piles	Unspecified	188 @ 10 m	173 @ 10 m	Unspecified	[DesJardin 2003, personal communication cited by WSDOT (2006a)], (Hastings and Popper 2005)
Steel H-piles	Diesel	190 @10 m	175 @ 10 m	Marine	(Hastings and Popper 2005; Illingworth and Rodkin 2001)
12-inch steel piles	Diesel	190 @10 m	190 @ 10 m	Marine	(Illingworth and Rodkin 2001)
14-inch steel piles	Hydraulic	195 @ 30 m;	180 @ 30 m	Marine	(Reyff et al. 2003)
16-inch steel piles	Diesel	198 @ 10 m	187 @ 9 m	Freshwater	(Laughlin 2004)
24-inch steel piles	Diesel	217 @10 m	203 @ 10 m	Unspecified	(WSDOT 2006a)
24-inch steel piles	Diesel	217 @ 10 m	203 @ 10 m	Unspecified	(Hastings and Popper 2005)
30-inch steel piles	Diesel	208 @ 10 m	192 @10 m	Marine	(Hastings and Popper 2005)
66-inch steel piles	Hydraulic	210 @ 10 m	195 @ 10 m	Marine	(Reyff et al. 2003)
96-inch steel piles	Hydraulic	220 @ 10 m	205 @ 10 m	Marine	(Reyff et al. 2003)
126-inch steel piles	Hydraulic	191 @ 11 m	180-206 @ 11 m	Marine	(Reyff et al. 2003)
150-inch steel piles	Hydraulic	200 @ 100 m	185 @ 100 m	Marine	(Reyff et al. 2003)

[a] Metric distances are listed as they were provided in the source material; 9 m = 29.5 ft; 10 m = 32.8 ft; 11 m = 36 ft; 30 m = 98 ft; 100 m = 328 ft.

All sound pressure values in units re: 1µPa.

7.2.2.4 Pile Driving Noise Impacts to Fish

Hastings and Popper (2005) recently performed a comprehensive literature review to evaluate the current best available science regarding noise thresholds at which fish would be injured by the percussive sound generated by pile driving. Much of the information presented below has been extracted from that review.

Most fish sense sounds, vibrations, and other displacements of water in their environment through their inner ear and with the lateral line running the length of each side of the fish and on the head. The lateral line is a mechano-sensory system that plays an indirect role in hearing through its sensitivity to pressure changes at close range. The hearing organs and lateral line system are collectively referred to as the acoustico-lateralis system. The hearing thresholds of different fish species vary depending on the structure and sensitivity of this system.

Anatomical variations of the inner ear, swim bladder, esophagus, lateral line, and other structures determine how fish hear and feel sound pressure (Hastings and Popper 2005). All fish fall into two hearing categories: “hearing generalists” such as salmon and trout, and “hearing specialists” such as herring and eulachon (Hastings and Popper 2005).

Hearing specialists have particular adaptations that enhance their hearing bandwidth and sensitivity (Hastings and Popper 2005). Hearing specialists found on the Pacific coast include the sardine and related Clupeiforms such as herring, shad, menhaden, and anchovy (Hastings and Popper 2005).

The majority of fish on the Pacific coast are hearing generalists and do not have specialized hearing capabilities apart from their swim bladder, inner ear, and lateral line (Hastings and Popper 2005). Hearing generalists sense sound directly through the inner ear, and some use the inner ear coupled with the swim bladder to sense additional energy (Hastings and Popper 2005).

Both hearing generalists and hearing specialists are found in many taxonomic groups (Hastings and Popper 2005). Ideally, fish should be compared based on biomechanical properties of their swim bladder and any other internal gas-filled chamber, hearing capabilities, and aspects of their behavior (Hastings and Popper 2005). However, when such data are not available, it is probably more appropriate to extrapolate between species that have somewhat similar auditory structures or pressure-detecting mechanisms (most notably the swim bladder) and species of similar size, mass, and anatomical variety (Hastings and Popper 2005). This would enable at least a first-order approximation of extrapolation to fishes such as salmonids and other teleost fishes that presumably do not have hearing specialization (e.g., rockfish). The results are less easily extrapolated to teleosts without a swim bladder, such as sand lance and lingcod, and to fish with very different ear structures, such as lamprey and sturgeon (Hastings and Popper 2005).

Table 7-3 outlines the known and presumed hearing categories of potentially covered fish species.

Table 7-3: Hearing Categories for Potentially Covered Fish Species

Common Name (Scientific Name)	**Hearing Category**	**Notes and/or References**
Trout and salmon (Salvelinus, Onchorynchus spp.)	Generalist	Popper and Carlson 1998
Sturgeon (Acipenser spp.)	Undetermined	Popper (2005) states that sturgeon can detect an extremely wide range of sounds, and several studies have found that some sturgeon produce sounds that may be used to facilitate breeding. However, further studies are necessary to determine how sturgeon vocalize, what levels of sound are produced in the natural environment, and how their vocalizations are used in their behavior.
Eulachon (Thaleichthys pacificus)	Specialist	Blaxter et al. 1981, in Scholik and Yan 2001a
Rockfish (Sebastes spp.)	Generalist	Hastings and Popper 2005
Lake chub (Couesius plumbeus)	Specialist	Hastings and Popper 2005; Popper et al. 2005
Dace (Rhinicthys spp.)	Unknown/ Presumed	Not a member of a family or grouping identified as containing hearing specialists (Fay and Popper 1999)

Common Name (Scientific Name)	Hearing Category	Notes and/or References
	Generalist	
Lingcod (Ophiodon elongates)	Generalist	Does not have a swim bladder, which is generally an indication of poor hearing (Moyle and Cech 2004; Kapoor and Khanna 2004)
Surf smelt (Hypomesus pretiosus)	Generalist	Included in the taxonomic order Salmoniformes – hearing generalists (Hastings and Popper 2005)
Lamprey (Lampetra spp.)	Generalist	Popper 2005
Margined sculpin (Cottus marginatus)	Generalist	Closely related to the bullhead (Cottus scorpius), which is identified as a generalist (Fay and Popper 1999); also not a member of a family or grouping identified as containing hearing specialists (Fay and Popper 1999)
Mountain sucker (Catostomus platyrhynchus)	Unknown/ Presumed Specialist	Catostomus spp. are known to have weberian ossicles to assist with hearing (Krumholz 1943)
Olympic mudminnow (Novumbra hubbsi)	Unknown/ Presumed Specialist	May have weberian ossicles to assist with hearing (Moyle and Cech 2004). Many closely related fish (minnows, pikeminnow cyprinids) are specialists (Scholik and Yan 2001b; Popper 2005).
Pacific cod (Gadus macrocephalus)	Generalist	Gadus sp. more sensitive than most generalists (Astrup and Mohl 1998, in Scholik and Yan 2002; Hastings and Popper 2005)
Pacific hake (Merluccius productus)	Unknown/ Presumed Generalist	Not a member of a family or grouping identified as hearing specialists (Fay and Popper 1999)
Pacific herring (Clupea harengus pallasi)	Specialist	Hastings and Popper 2005
Pacific sand lance (Ammodytes hexapterus)	Generalist	Does not have a swim bladder, which is generally an indication of poor hearing (Moyle and Cech 2004; Kapoor and Khanna 2004)
Pygmy whitefish (Prosopium coulteri)	Generalist	Of the order Salmoniformes – hearing generalists (Hastings and Popper 2005)
Walleye pollock (Theragra chalcogramma)	Unknown/ Presumed Generalist	Not a member of a family or grouping identified as containing hearing specialists (Fay and Popper 1999)

Physical impacts to fish from intense noises may include temporary hearing loss (referred to as temporary threshold shift), permanent hearing loss (referred to as permanent threshold shift), damage or rupture to gas organs such as the swim bladder and the surrounding tissues, rupture of capillaries in the skin, neurotrauma, and eye hemorrhage (Popper and Fay 1973, 1993, Hastings and Popper 2005). The more serious of these impacts could cause instantaneous death or later death from injuries (e.g., breakdown of tissues in some organs) (NMFS 2003a).

7.2.2.4.1 Lethal Physiological Effects to Fish

- In general, injury and mortality effects from underwater sound are caused by rapid pressure changes, especially on gas-filled spaces in the body. Rapid volume changes of the swim bladder may cause it to tear, resulting in a loss of hearing sensitivity and hydrostatic control. Intense noise may also damage the tissue in hearing organs, as well as the heart, kidneys, and other highly vascular tissue.

Susceptibility to injury is variable and depends on species-specific physiology, auditory injury, and auditory thresholds (Popper and Fay 1973, 1993). While species-specific data are limited, the available information indicates variable effects related to physiology, size, and age, as well as the intensity, wavelength, and duration of sound exposure.

- A study by Abbot (Abbott, R.R., E. Bing-Sawyer, and R. Blizard, 2002) on caged fish demonstrated that energy accumulates over multiple pile driving strikes. This is demonstrated by the fact that fish that received exposure to multiple strikes had extreme internal injuries (in some cases their internal organs were homogenized). Abbott's work also demonstrated that fish with serious internal injuries might not appear harmed to observers (Abbott et al. 2002).

- Impacts on Eggs and Larvae**.** Although it is possible that some (but not all) fish species would swim away from a sound source, thereby decreasing exposure to sound, larvae and eggs are often at the mercy of currents, move slowly, or are sedentary (Hastings and Popper 2005). Data on the effects of sound on developing eggs and larvae are limited, although in a study by Banner and Hyatt (1973), increased mortality was found in eggs and embryos of sheepshead minnow (Cyprinodon variegates) exposed to broadband noise (100 to 1,000 hertz) that was about 15 dB above the ambient sound level. Hatched fry of sheepshead minnow and fry of longnose killifish (Fundulus similes) were not affected in this study. Jensen (2003) noted possible effects of sound impacts on the development of salmonid eggs.

- Susceptibility to injury may also be life-history specific. Banner and Hyatt (1973) demonstrated increased mortality of sheepshead minnow eggs and embryos when exposed to broadband noise approximately 15 dB above the ambient sound level. However, hatched sheepshead minnow fry were unaffected by the same exposure.

Although hearing loss is not a lethal effect of pile driving, the subsequent long-term effects, such as predator/prey detection may result in mortality.

7.2.2.4.2 Sub-Lethal Physiological Effects to Fish

- High-intensity sounds can also permanently damage fish hearing (Cox et al. 1987; Enger 1981; Popper and Clarke 1976). Hardyniec and Skeen (2005) and Popper et al. (2005) exposed three species of fish to high-intensity percussive sounds from a seismic air gun at sound levels ranging between 205 and 209 dB_{Peak}, intending to mimic exposure to pile driving. Subject species included a hearing generalist (broad whitefish), a hearing specialist (lake chub), and a species that is intermediate in hearing (northern pike). They found that the broad whitefish suffered no significant effects from noise exposure, the lake chub demonstrated a pronounced temporary threshold shift in hearing sensitivity (i.e., hearing loss), and the northern pike showed a significant temporary hearing loss but less than that of the lake chub. The hearing sensitivities of lake chub and northern pike returned to their respective normal thresholds after 18 to 24 hours.

- Enger (1981) found that pulsed sound at 180 dB was sufficient to damage the hearing organs of codfish (genus *Gadus*), resulting in permanent hearing loss.

- Hastings (1995) found that goldfish exposed to continuous tones of 189, 192, and 204 dB_{peak} at 250 Hz for 1 hour suffered permanent damage to auditory sensory cells.

- Injury effects may also vary depending on noise frequency and duration. Hastings et al. (1996) found destruction of sensory cells in the inner ears of oscars 4 days after exposure to continuous sound for 1 hour at 180 dB_{peak} at 300 Hz. In contrast, when the two groups of the same species were exposed to continuous and impulsive sound at 180 dB_{peak} at 60 Hz for 1 hour, and to impulsive sound at 180 dB_{peak} at 300 Hz repeatedly over 1 hour, they showed no apparent injury.

7.2.2.4.3 Behavioral Effects to Fish

Behavioral responses to sound stimuli are well established in the literature for many fish species. As noted under the *Lethal Physiological Effects* and *Sub-Lethal Effects* section, effected hearing ability of fish may result in long-term mortality. These types of effects on behavior are summarized below:

- Masking of existing ambient noise reducing the ability of fish to sense predators or prey.

- These activities may also have indirect effects such as reducing the foraging success of these fish by affecting the distribution or viability of potential prey species.

- Moore and Newman (1956) reported that the classic fright response of salmonids to instantaneous sound stimuli was the "startle" or "start" behavior, where a fish rapidly darts away from the noise source.

- Knudsen et al. (1992) found that in response to low-frequency (10 Hz range) sound, salmonids 1.6–2.4 in (40–60 mm) in length exhibited an initial startle response followed by habituation, while higher frequency sound caused no response even at high intensity.

- In a study of the effects of observed pile driving activities on the behavior and distribution of juvenile pink and chum salmon, Feist et al. (1992) found that pile-driving operations were associated with changes in the distribution and behavior of fish schools in the vicinity. Fish schools were two-fold more abundant during normal construction days in comparison to periods when pile driving took place.

- Blaxter et al. (1981) found Atlantic herring to exhibit an avoidance response to both continuous pulsed sound stimuli with habituation to more continuous stimuli occurring over time, and Schwarz and Greer (1984) found similar responses on the part of Pacific herring.

- Sound has also been shown to affect growth rates, fat stores, and reproduction (Banner and Hyatt 1973; Meier and Horseman 1977).

- Prolonged underwater noise can also reduce the sensitivity of fish to underwater noise stimuli, with potentially important effects on survival, growth, and fitness. The fish auditory system is likely one of the most important mechanisms fish use to detect and respond to prey, predators, and social interaction (Amoser and Ladich 2005; Fay 1988; Hawkins 1986; Kalmijn 1988; Myrberg 1972; Myrberg and Riggio 1985; Nelson 1965; Nelson et al. 1969; Richard 1968; Scholik and Yan 2001; Scholik and Yan 2002; Wisby et al. 1964). Scholik and Yan (2001) studied the auditory responses of the cyprinid fathead minnow to underwater noise levels typical of human-related activities (e.g., a 50 horsepower outboard motor). They found that prolonged exposure decreased noise sensitivity, increasing the threshold level required to elicit a disturbance response for as long as 14 days after the exposure. Amoser and Ladich (2005) reported similar findings in common carp in the Danube River, noting that auditory ability in this hearing specialist species was measurably masked in environments with higher background noise. They reported similar but far less pronounced responses in hearing generalist species such as perch. These data suggest that elevated ambient noise levels have the potential to impair hearing ability in a variety of fish species, which may in turn adversely affect the ability to detect prey and avoid predators, but that this effect is variable depending on the specific sensitivity of the species in question.

- Feist et al. (1992) similarly theorized that it was possible that auditory masking and habituation to loud continuous noise from machinery may decrease the ability of salmonids to detect approaching predators.

7.2.2.5 Pile Driving Noise Impacts to Invertebrates

Although studies of noise impacts on invertebrates have consistently shown that very high sound pressure levels (in excess of 217 dB) can cause serious injury, the information is sparse, is poorly reported, and was obtained without due experimental rigor (Turnpenny et al. 1994). The studies reported in Turnpenny et al. (1994) exposed mussels, periwinkles, amphipods, squid, scallops, and sea urchins to high airgun and slow-rise-time sounds at between 217 dB and 260 dB. Mussels, periwinkles, and amphipods showed no detectable effect at 229 dB (Kosheleva 1992, in Turnpenny et al. 1994), although one Iceland scallop suffered a split shell after being exposed to 217 dB from a single airgun strike (Matishov 1992, in Turnpenny et al. 1994), suggesting the potential for serious injury when percussive underwater noise exceeds these levels.

7.2.2.6 Vessel/Equipment Operation and Materials Placement (Non-Pile Driving) Sound Levels

In comparison to pile driving, data on sound levels produced by placement of other construction-related materials are limited. For example, measured sound levels associated with work on the Friday Harbor ferry terminal ranged between 133 dBpeak and 140 dBpeak, excluding pile driving. These sound levels were slightly higher than ambient levels, which include routine vessel traffic (WSDOT 2005). Nedwell et al. (1993) measured noise produced by underwater construction tools such as drills, grinders, and impact wrenches at 3.28 ft (1 m) from the source. When corrected for a reference distance 32.8 ft (10 m) from the source using the practical spreading loss model, the noise associated with these sources ranged from approximately 120 to 165 dBpeak.

These data suggest that sound associated with in-water tool use, placement of large rock and similar material, vessel operation, and in-water operation of heavy machinery, generally produce substantially lower sound levels than those associated with pile driving. However, other construction-related noises may generate continuous noise for longer periods, with the effect of elevating ambient noise levels or masking ambient noises in the aquatic environment that fish would ordinarily use to identify prey and predators.

This effect may be of particular concern for projects that result in changes in vessel operation or equipment use that change ambient noise levels for longer periods (e.g., days to years). For example, vessel operation can significantly influence ambient noise levels. Large vessel engines can produce underwater sound up to 198 dB, and depth sounders can produce noise in excess of 180 dB (Buck 1995; Heathershaw et al. 2001).

Hazelwood and Connelly (2005) monitored fishing vessel noise over a broad octave range from 10 Hz–40 kHz and documented noise levels ranging from 140–185 dB_{peak}, with the loudest noise occurring at the lower end of the octave range.

Commercial sonar devices operating in a frequency range of 15–200 kHz can produce underwater noise ranging from 150–215 dB at maximum levels (Stocker 2002).

Equipment and vessels necessary to dig trenches, place riprap, support equipment over water, and perform other activities associated with the construction of overwater structures also produce underwater noise. Construction equipment tends to produce the same type of slow-rise-time noise as do motor boats and ship engines.

- Jones and Stokes (2006) estimated that noise produced by a rather large ocean-cable-installation vessel is about 154 dB_{RMS}.

- JASCO (2005) estimated that noise produced by a rock-dumping vessel is approximately 177 dB (neither peak nor RMS identified) at 3.28 feet (1 m)

- Richardson et al. (1995, in Jones and Stokes 2006) estimated that an equipment support vessel produces noise levels of 152 dB_{peak} at 3.28 feet (1 m).

Sounds of this amplitude may affect the behavior or physiology of fishes, depending on their hearing sensitivity and proximity to the sound.

7.2.2.6.1 Effects on Fish

Operational activities are expected to produce intermittent, continuous noise from facilities and vessels for the life of the facility. In general, noise levels produced by small to moderate sized vessel operations are relatively low in comparison to those levels shown to cause injury in construction projects. Responses to these effects may range from minor changes in behavior, to increased predation risk or lowered foraging efficiency, to potential injury.

7.2.2.6.2 Effects on Invertebrates

No research has been identified regarding the effects of lower intensity continuous underwater noise on invertebrates. However, operational noise is typically associated with sound pressures well below levels that have been observed to cause injury in shellfish, suggesting that HCP invertebrate species might not be subject to these effects. Because HCP invertebrates with the potential for stressor exposure are either filter feeders or grazers and are essentially non-motile, these species are unlikely to be subject to auditory masking effects that would limit the ability to sense predators and prey. Some potential may exist for disturbance-induced interruption of feeding behavior, but more research on this subject is necessary to determine this definitively.

7.2.3 Channel/Work Area Dewatering

7.2.3.1 Impacts to Fish.

7.2.3.1.1 Fish Removal and Exclusion

In many cases, construction of HPA-permitted projects may require the exclusion of streamflows or even the dewatering of the work area to protect aquatic life and/or provide a suitable environment for construction. Channel dewatering occurs primarily in freshwater streams and is typically associated with the need to work “in the dry” during installation, construction, or replacement of culverts and bridges, or other in-water type activities. To reduce stranding, fish removal and exclusion from the construction zone is usually part of channel dewatering activities. This is typically accomplished through passive methods, such as the volitional movement of fish from the construction area during its slow dewatering, or through active methods, such as the use of hand nets, beach seines, or electrofishing equipment to capture and move fish from the construction area that will be dewatered (NMFS 2006). These activities have the potential to cause direct and indirect effects on HCP species. Fish exclusion and dewatering involve the placement of barriers (e.g., block nets, temporary berms, cofferdams) around a work area and the capture and removal of fish and other aquatic life within the work area. Electrofishing is a common practice used for fish capture in freshwater environments, as is the use of minnow traps, hand nets, beach seines, and other net-based capture methods.

Because electrofishing is ineffective in brackish or salt water, net-based capture methods are used in these environment types.

7.2.3.1.2 Bypass System

Use of a flow bypass system is a common means of creating exclusion areas via dewatering and flow reduction. This type of activity has the potential to entrain fish within the flow bypass system. If pumps are used to bypass water around a work site or to dewater residual pools within a portion of the dewatered channel, the hose or pipe pulling water from the channel is typically fitted with a protective screen to prevent entrainment of aquatic life into the intake hose/pipe of the pump. Such measures are required for all pumped diversions (WAC 220-110-190), and specific criteria for screens, including approach velocity, mesh site, and screen location, have been developed by NMFS (2008) and WDFW (1998).

Installation of a flow bypass system typically requires in-water work, which can disturb substrates and bank material and cause an increase in turbidity levels. Once the system is installed, operation of a flow bypass system generally will not result in disturbance to the streambed or cause an elevation in turbidity levels, unless the discharge at the outlet results in scouring of substrate material or erosion of streambanks. Energy dissipaters are generally required to preclude such scouring from occuring. Diversion outfalls require temporary erosion-protection measures to prevent scour at the point of return flow from the diversion channel or pipe (WDFW 2004). Removal of the flow bypass also requires in-water work and results in some disturbance to the streambed and banks as the cofferdam is removed and flow is returned to the channel. Generally, the downstream cofferdam is removed first to allow backwatering of a portion of the channel that was dewatered. Then the upstream cofferdam is removed, and flow is slowly returned to the channel to minimize resuspension of fine sediments and increases in turbidity.

7.2.3.1.3 Passive Capture

Passive capture of fish typically involves installing an upstream block net and a cofferdam and slowly dewatering the construction area. This type of passive fish removal eliminates the need to capture and handle some fish. Less commonly, active methods of fish removal may be used, such as the use of a beach seine to “herd” fish downstream to a point beyond the construction area and/or the use of electrofishing equipment to remove fish.

7.2.3.1.4 Partial Dewatering

Partial dewatering is a technique used to reduce the volume of water in the work area to make capture methods more efficient. In riverine habitats, this method is used to move fish out of affected habitats to reduce the number of individuals exposed to capture and handling stress and potential injury and mortality. NOAA Fisheries has estimated that 50–75 percent of fish in an affected reach will volitionally move out of an affected reach when flows are reduced by 80 percent (NMFS 2006). However, volitional movement will lead to concentration of fish in unaffected habitats, increasing competition for available space and resources.

The following sections describe the physical impacts of channel dewatering on potentially covered fish species.

- Fish that remain in a dewatered reach during construction may encounter lethal conditions. Fish left in the exclusion area would potentially be directly exposed to stranding, dessication and asphyxiation during dewatering or, if left inundated, to mechanical injury and/or high-intensity noise, turbidity, and other pollutants. Trampling, higher-than-normal temperatures, and increased vulnerability to predators could also kill or injure fish. Many species of fish, such as salmonids and larval lamprey, are highly cryptic and can avoid being detected even when using multiple-pass electrofishing because they hide in large interstices or are buried in sediments (Peterson et al. 2005; Peterson et al. 2004; Wydoski and Whitney 2003). Therefore, they face a higher likelihood of exposure to stranding or entrainment in dewatering pumps, which would be expected to lead to mortality. In freshwater environments, examples of species and life-history stages that are sensitive to dewatering impacts include incubating salmonid eggs and alevins; lamprey ammocoetes; and the adhesive eggs of eulachon, sturgeon, and other species.

- NOAA Fisheries has estimated incidental take resulting from dewatering and fish handling associated with stream crossing projects. In calculating incidental take from these activities, the agency applied an estimated stranding rate of 8 percent for ESA-listed salmonids (which equates to 8 percent mortality) (NMFS 2006), based on an expected 45 percent capture efficiency using three pass electrofishing (Peterson et al. 2004), and assuming a 25 percent injury rate. If bank protection projects require dewatering similar streams, then a similar level of take could be expected.

- Fish removal efforts such as beach seining and electrofishing could inadvertently result in fish mortality. The amount of unintentional mortality (and non-lethal injury) attributed to seining would vary widely depending on the seine used, the ambient conditions, and the expertise of the field crew (NMFS 2006). Professional experience has shown that beach seining in areas of dense aquatic vegetation or in muddy areas could also result in significant mortality of seined fish that become trapped in a mass of vegetation or mud.

- Electrofishing could also kill both juvenile and adult fish if improperly conducted. Mortality could result from direct trauma or from indirect factors (e.g., as a result of disease or subsequent fungal attack due to scale loss).

- There generally would be fewer adverse impacts associated with seining compared to electrofishing, and first using a seine to remove fish would minimize the adverse effects of electrofishing (NMFS 2006).

- The act of capture and handling demonstrably increases physiological stress in fishes (Frisch and Anderson 2000). Primary contributing factors to handling-induced stress and death include exposure to large changes in water temperatures and dissolved oxygen conditions (caused by large differences among the capture, holding, and release environments); duration of time held out of the water; and physical trauma (e.g., due to net abrasion, squeezing, accidental dropping). Even in the absence of injury, stress induced by capture and handling can have a lingering effect on survival and productivity. One study found that stress from handling impaired the salmonids' ability to evade predators for up to 24 hours following release and caused other forms of mortality (Olla et al. 1995). Capture and handling could also reduce fish access to prey.

- Beach seining could affect fish in several ways, including stress, scale loss, physical damage, suffocation, and desiccation. Anesthetics such as tricaine methane sulfonate (also known as MS-222) and clove oil are often used to sedate fish to facilitate easier fish handling and reduce fish stress.

- Electrofishing could also result in sublethal effects, such as spinal injury (NMFS 2006; Snyder 2003). The following excerpt from NMFS (2006) concisely describes the state of the knowledge pertaining to electrofishing impacts:

> Most of the studies on the effects of electrofishing have been conducted on adult fish greater than 12 inches in length (Dalbey et al. 1996). The relatively few studies that have been conducted on juvenile salmonids indicate that spinal injury rates are substantially lower than they are for large fish. Smaller fish intercept a smaller head-to-tail potential than larger fish (Sharber and Carothers 1988) and may therefore be subject to lower injury rates (e.g., Dalbey et al. 1996, Thompson et al. 1997). McMichael et al. (1998) found a 5.1 percent injury rate for juvenile middle Columbia River steelhead captured by electrofishing in the Yakima River subbasin while Ainslie et al. (1998) reported injury rates of 15% for direct current applications on juvenile rainbow trout. The incidence and severity of electrofishing damage is partly related to the type of equipment used and the waveform produced (Dalbey et al. 1996, Dwyer and White 1997, Sharber and Carothers 1988). Continuous direct current or low-frequency (equal or less than 30 Hz) pulsed direct current have been recommended for electrofishing (Dalbey et al. 1996, Fredenberg 1992) because lower spinal injury rates, particularly in salmonids, occur with these waveforms (Ainslie et al. 1998, Dalbey et al. 1996, Fredenberg 1992). Only a few recent studies have examined the long-term effects of electrofishing on salmonid survival and growth (Ainslie et al. 1998, Dalbey et al. 1996). These studies indicate that although some of the fish suffer spinal injury, few die as a result. However, severely injured fish grow at slower rates and sometimes they show no growth at all (Dalbey et al. 1996).

In the absence of additional supporting information, it is reasonable to conclude that these same effects would affect many of the HCP fish species, but this conservative assumption may not be universally accurate. Studies of the effects of electrofishing on other fish species are more limited, but available data indicate that at least some HCP species may be less sensitive to injury-related effects. Holliman et al. (2003) subjected a threatened cyprinid (minnow) species to electrofishing techniques in the laboratory and found that the typical current and voltage parameters used to minimize adverse effects on salmonid species produced no evidence of injury. This suggests that other cyprinids (such as leopard and spotted dace, lake chub, and suckers) may also be less sensitive.

Electrofishing-related injury rates are variable, reflecting a range of factors from fish size and sensitivity, individual site conditions (e.g., water conductivity, visibility, etc.), to crew experience and the type of equipment used, with the equipment type being a particularly important factor (Dalbey et al. 1996; Dwyer and White 1997; Sharber and Carothers 1988).

- Channel dewatering decreases benthic prey availability for young salmonid life stages and other species that feed upon benthic prey in the area near the dewatered zone. Bell (1991) reported that the permanent wetted area of a channel is the governing factor in food production for salmonids because aquatic food supplies do not shift in streams as water levels rise or fall. The loss of prey is generally temporary, and as flow is returned to the dewatered portion of the channel, benthic macroinvertebrates from outside the dewatered area and those that sought refuge in the hyporheic zone recolonize the previously dewatered channel. The amount of time necessary for the benthic macroinvertebrate community to recolonize a dewatered reach will depend upon the size and duration of dewatering, the size and life cycles of the benthic macroinvertebrate community in nearby areas, and the season of disturbance (NMFS 2001b, 2005).

- Another potential impact is related to displacement of fish, either naturally to avoid high turbidity or as a result of fish removal. This will increase the density of fish in the area being utilized, and increase competition for food and space. In addition, dewatering, or water diversions may also effect migration patterns of fish.

- Dewatering and diversion may also disrupt migration patterns to various fish species and life history strategies.

7.2.3.2 Impacts to Invertebrates

Typically, potentially covered benthic invertebrate species are not removed during channel dewatering and so would be subject to injury or mortality. Loss of macroinvertebrates can result from excavation, installation of bank protection structures, and placement of associated fill material.

HCP invertebrate species demonstrate different sensitivity to the effects of dewatering and relocation than fish, with many species being relatively insensitive to the effects of handling, at least during adult life-history stages. For example, Krueger et al. (2007) studied the effects of suction dredge entrainment on adult western ridged and western pearlshell mussels in the Similkameen River (Washington) and found no evidence of mortality or significant injury. Suction dredge entrainment is expected to be a more traumatic stressor than removal and relocation by hand. These findings suggest that careful handling would be unlikely to cause injury. However, the authors cautioned that these findings were limited to adult mussels, and the potential for injury and mortality in juveniles remains unknown.

Mussels provide a good example of potentially covered invertebrate species that may be affected by desiccation, as they exhibit sensitivities related to periodicity of inundation as well as temperature. Although no studies were located that specifically examined the impacts of construction-related dewatering, several studies have examined the influence of dam operations on freshwater mussel habitats, providing insight on the potential impacts from construction dewatering (summarized in Watters 1999). Depending on the use of the dam, water levels may fluctuate at regular intervals (for hydroelectric purposes) or random intervals (for flood control). In some areas, water levels may become shallow enough that thermal buffering is lost, allowing extreme temperatures to occur (Watters 1999). Blinn et al. (1995, in Watters 1999) reported that substrate subjected to 2- to 12-hour exposures to air required more than four months for mussels to regain a biomass similar to that in unexposed habitat. Federally endangered mussel species were reported by Neck and Howells (1994, in Watters 1999) as casualties of scheduled dewatering processes, and Riggs and Webb (1956) reported that several thousand mussels died in the tailwaters of Lake Texoma, an impoundment of the Red River formed by Denison Dam, when water levels dropped, in turn allowing water temperatures to become excessively warm (greater than 79 degrees Fahrenheit [F], 26 degrees Celsius [C]).

Combined with desiccation, exposure to cold air may be equally lethal to mussels. Nagel (1987, in Watters 1999) suggested that mussels would be more sensitive to cold water during frosts than to warm water during temporary droughts. Blinn et al. (1995) showed that a single overnight exposure to subzero temperatures resulted in at least a 90 percent loss of invertebrate biomass, and Valovirta (1990) reported that mussels were killed when water froze to the river bottom.

The sensitivity of other HCP invertebrate species, such as giant Columbia River limpet and great Columbia River spire snail, is somewhat less certain. Adults may be easily removed and relocated during dewatering, but juveniles and eggs may be difficult to locate and remove effectively. This suggests the potential for mortality from stranding.

While handling-related injury and mortality are relatively unlikely, relocation may lead to significant nonlethal effects. For example, scattering of closely packed groups of adult mussels may affect reproductive success. Because female freshwater mussels filter male gametes from the water column, successful fertilization is density dependent (Downing et al. 1993).

Failure to locate and remove small or cryptic invertebrate species or life-history stages may result in stranding or concentrated exposure to other stressors within the exclusion area. Stranding caused by operational water level fluctuations was associated with mass mortality of California floater and western ridged mussels in Snake River reservoir impoundments (Nedeau et al. 2005).

7.2.4 Navigation or Maintenance Filling and Dredging

Navigation or maintenance dredging is by far the most frequent form of dredging in Washington State. This type of dredging can convert intertidal habitat to subtidal habitat and shallower subtidal habitats to deeper subtidal habitats through periodic deepening to remove accumulated sediments that impede navigation to and from marinas/terminals. There are several different means by which dredging affects fish and invertebrates, the most significant being alteration of bathymetry, removal of aquatic vegetation, entrainment of benthic organisms, and turbidity and resuspension of contaminated sediments. These stressors are discussed below.

7.2.4.1 Altered Bathymetry and Substrate Composition

Large channel deepening projects can markedly alter ecological relationships through the change of freshwater inflow, tidal circulation, estuarine flushing, and freshwater and saltwater mixing. Miller et al. (1990) reported that only through comprehensive areal surveys over a minimum of four seasons before dredging, with follow-up surveys after dredging, could impacts of channel deepening on aquatic resources be determined. In a comparison between dredged and undredged areas in the Port of Everett’s public marina, Pentec (1991) found catches of fish to be higher in the dredged area before dredging than after dredging. Catches decreased from about 90 fish per tow to about 3 fish per tow and from eight species to five species.

Depending on site characteristics, maintenance dredging may occur annually or at intervals of 10 years or longer. These different dredging timelines represent different disturbance regimes both in terms of the ability of the benthos to recolonize prior to redisturbance and the magnitude of benthic productivity affected by dredging. In a literature review report on dredge and disposal effects, Morton (1977) reported the range of effects on invertebrate communities to be from negligible to severe, with impacts ranging from short to long term. In general, this literature review found that short-term, small-scale dredging and dredge disposal projects affected benthic communities less than long-term, large-scale projects. This is likely due to the fact that benthic communities are more likely (and quicker) to recover from short-term, less intense, small-scale disturbances than from large-scale and intense disturbances over long time periods (Guerra-García et al. 2003; Dernie et al 2002). For example, in experiments conducted in sheltered sand flats, the benthic community recovered from lower intensity disturbance (i.e., sediment removal to a depth of 3.9 inches [10 cm]) within 64 days, whereas recovery from higher intensity disturbance (i.e., sediment removal to a 7.9-inch [20-cm] depth) required 208 days postdisturbance (Dernie et al 2002).

In a study to evaluate the effects of dredged material disposal on biological communities, Hinton et al. (1992) reported a significant increase in benthic invertebrate densities at a disposal site between June 1989 (predisposal) and June 1990 (postdisposal). Recolonization could have occurred by invertebrates burrowing up through newly deposited sediments or recruitment from surrounding areas (Richardson et al. 1977).

Dredging is often required during marina and terminal projects as a component of facility development, as well as during routine maintenance to maintain navigability. In marine environments dredging converts intertidal into subtidal habitats, affecting the plant and animal assemblages that are uniquely adapted to the particular light, current, and substrate regimes of intertidal areas. By altering bathymetry and bottom substrates, such conversions are described as producing a habitat “trade-off” of intertidal and shallow-subtidal communities for deeper, subtidal communities. In lacustrine environments, dredging converts shallow-water littoral habitats into deeper water environments and may create a steeper bathymetric transition. This change in habitat characteristics may change the size and species distribution of fish in the localized environment, altering predator/prey dynamics. The effects of dredging on riverine environments are more complex still, because localized alteration of channel morphology can lead to dynamic shifts in channel form as the system adjusts to the changed conditions. These effects can extend a considerable distance beyond the bounds of the original dredging project.

Dredging activities result in short-term direct effects, including entrainment and potential mortality; periodic removal of potentially suitable habitats for fish and invertebrates; alteration of water circulation and subsequent nutrient, prey, and habitat availability; and increased turbidity and potential resuspension of contaminants. In addition, long-term and food web indirect effects can occur, such as reconfiguration of the benthos and the availability of nutrient and prey resources. Resulting impacts, include mortality, injury, decreased foraging opportunity, decreased growth and fitness, and physiological and behavioral responses. Deposition of dredge spoils can bury existing habitats and benthic organisms, resulting in a similar suite of impacts. For invertebrates at dredge disposal sites, research has shown potential increases in densities.

7.2.4.2 Effects of Entrainment and Burial on Fish

Entrainment occurs when an organism is trapped in the uptake of sediments and water being removed by dredging machinery (Reine and Clark 1998). Demersal fish, such as sand lance, sculpins, and pricklebacks, likely have the highest rates of entrainment as they reside on or in the bottom substrates, with life-history strategies of burrowing or hiding in the bottom substrate. This is also true in freshwater environments. For example, lamprey ammocoetes likely have a high risk of vulnerability to dredging due to the lengthy residence time in freshwater sediments in their early life-history stages. In general, larval fish that have little or no swimming capacity to avoid direct dredge impacts are also at significant risk of entrainment in dredge sites. Of particular concern for the purpose of this analysis are the HCP groundfish (lingcod, rockfish, Pacific cod, pollock, hake) and the forage fishes (herring, sand lance, and surf smelt), all of which have larval or juvenile life-history stages with low motility. The juvenile life-history stage of the groundfish species typically rear in shallow nearshore habitats, where dredging is likely to occur. Due to their demersal nature and limited motility, they face a higher risk of dredging entrainment.

Larger fish may also be susceptible to entrainment. Armstrong et al. (1982) found that larger fish were not necessarily able to avoid the hopper dredge, with the largest specimen being a 9.2-in (234-mm) tomcod. Tests of excluders mounted on the draghead of a hopper dredge showed that 66 percent fewer fishes (mostly flatfish and gunnels in the study) could be saved from entrainment through use of the device (Shaw 1996).

Buell (1992) found entrainment of juvenile white sturgeon (11.8–19.6 in [300–500 mm]) at a rate of 0.015 fish/cy. In another study, juvenile salmonids and eulachons were the dominant entrained taxa due to the dredge location in a constricted waterway, making it more difficult for salmonids to avoid the dredge operation (McGraw and Armstrong 1990; Larson and Moehl 1990).

Entrained bivalve larvae, such as larval oysters, are assumed to suffer 100 percent mortality by sediment smothering, anoxia, starvation, or desiccation even without direct mechanical impacts from pumping. However, the population-level effects of these stressors may be relatively limited. For example, concern for oyster larvae entrainment in Chesapeake Bay resulted in the development of a population model using conservative temporal and spatial distributions (Lunz 1985). The model predicted that entrainment would have minimal negative effect on the population, with the calculated mortality rate ranging between 0.005 and 0.3 percent of larval abundance. Lunz (1985) concluded that this represented no significant impact as the dredge entrained only a small fraction of the total water volume flowing past the dredge. Many species, particularly marine fish and invertebrates, have planktonic larval life-history stages that suffer naturally high mortality rates (in some cases exceeding 99 percent). Therefore, the potential mortality from entrainment is relatively insignificant in comparison (Lunz 1985).

7.2.4.3 Effects of Entrainment and Burial on Invertebrates

Benthic infauna are particularly vulnerable to being entrained by dredging uptake, but mobile epibenthic and demersal organisms such as burrowing shrimp, crabs, and fish also can be susceptible to entrainment. Entrainment rates are usually described by the number of organisms entrained per cubic yard (cy) of sediment dredged (Armstrong et al. 1982).

Because they are nonmotile, HCP invertebrate species are less able to avoid exposure to burial and entrainment-related stressors. Although some specifics on the effects of burial are known for marine invertebrate species (Hinchey et al. 2006), data on the tolerance limits of HCP freshwater mollusks with respect to burial are more limited. However, sufficient data are available on both marine and freshwater species to draw some conclusions about the effects of burial.

- Stress or mortality resulting from partial and complete burial of various mollusk species has been addressed empirically (Hinchey et al. 2006). Results of these studies indicate that species-specific responses vary as a function of motility, living position, and inferred physiological tolerance of anoxic conditions. Mechanical and physiological adaptations contribute to this tolerance.

- Olympia oysters have been shown to be intolerant of siltation and do best in the absence of fine-grained materials (WDNR 2006b). Thus, it can be inferred that burial of these organisms would lead to mortality.

- Increased fine sediment deposition has been shown to adversely affect estuarine mollusk species with low motility (Hinchey et al. 2006).

- Limpets in intertidal habitat are affected by burial and interference with feeding activity. In a field study in the United Kingdom, grazing by limpets was decreased by 35 percent after the addition of fine sediments, to as little as 0.04 in (1 mm) thick (50 mg/m^2), with mortality and inhibition of feeding at higher levels of fine sediment (200 mg/m^2) (Airoldi and Hawkins 2007). The mechanism of effect is postulated to be the clogging of filtering organs by fine sediments.

- Burial with fine sediments has been associated with high mortality levels in freshwater mollusk species. Mussel mortality rates exceeding 90 percent have been observed following burial with silt (Ellis 1942), and burial with fines has been implicated in large-scale mortality of western pearlshell mussels in the Salmon River in Idaho (Vannote and Minshall 1982).

- In a survey of native freshwater mussels in the United States and Canada, it was concluded that declines in populations were caused by habitat destruction, dams, siltation, and channel modifications, with siltation a significant issue in some areas (Williams et al. 1993).

- Burial with coarse sediment appears to be less problematic, provided that the stressor is short term in duration. Krueger et al. (2007) studied the effects of burial on western ridged and western pearlshell mussel species in the Similkameen River in Washington State. Interestingly, they found that mussels buried under less than 40 cm (15 inches) of coarse sediment (gravel and cobble) were able to extricate themselves. Test subjects buried at or beyond this depth suffered only a 10 percent mortality rate over the 6-week period. However, none of these individuals were able to extricate themselves. This suggests that burial in coarse sediments caused by bedload scouring could lead to high rates of delayed mortality from starvation and other effects.

- Krueger et al. (2007) also studied the effects of suction dredge entrainment on these two species of mussels. The test subjects entrained through the dredge showed no evidence of mortality or significant injury. This suggests that freshwater mollusk species may be relatively insensitive to entrainment-related effects. This is intuitively logical, as these species occur in environments where mobilization of coarse bedload is common. This suggests the likelihood of evolutionary adaptation to protect against mechanical injury from bedload mobility. However, the authors cautioned that their findings were applicable only to the adult life-history stages studied. The sensitivity of juvenile mussel species to entrainment remains unknown. This uncertainty would be expected to extend to the juvenile life-history stages of other HCP invertebrate species as well.

Mollusk larvae and juveniles are expected to be highly sensitive to the effects of entrainment and burial and are assumed to suffer high mortality from mechanical injury, smothering, anoxia, starvation, or desiccation. However, in the case of freshwater mussels, stressor exposure would have to be extensive to result in significant population-level effects. As an example, the issue of larval oyster mortality caused by dredge entrainment was studied in detail Chesapeake Bay. Lunz (1985) concluded that even if entrained larvae suffered 100 percent mortality, the absolute effects would be relatively limited because the dredge would entrain only a small fraction of larvae in the vicinity. The estimated mortality rate for oyster larvae ranged between 0.005 and 0.3 percent of total abundance. These effects are insignificant in comparison to natural mortality rates. Many species, particularly marine fish and invertebrates, have planktonic larval life-history stages that suffer naturally high mortality rates (in some cases exceeding 99 percent) (Lunz 1985). Therefore, it is likely that larval mortality from burial and/or entrainment is relatively insignificant when viewed from the perspective of natural population dynamics. Moreover, in the case of freshwater mussels, the potential for adverse effects is further limited by the fact that the parasitic glochidia life-history stage resides in the gills of host-fish where stressor exposure is less likely to occur.

The other freshwater mollusks, great Columbia River spire snail and giant Columbia River limpet, hatch from the egg fully formed. Therefore, these species would be expected to have a higher level of sensitivity to the effects of burial and entrainment.

7.2.5 Ambient Light Modifications

Along marine, riverine, and lake shorelines, marinas (as a collection of individual piers) and shipping or ferry terminals are known to affect light availability and the aquatic habitats upon which HCP species depend. A considerable body of literature provides evidence that shading from these structures can reduce ambient daytime aquatic light availability to levels below the light threshold levels required for aquatic plant photosynthesis and fish feeding and movement. Effects of reduced light availability on plants is discussed under Aquatic Vegetation Modifications. These facilities can also alter ambient nighttime light through the use of artificial light. In the case of terminals that berth large vessels, documented shade casting includes the reflective effects of sediment resuspension and bubbles generated by high propulsion prop wash in shallow environments (Blanton et al. 2001; Haas et al. 2002; Thom et al. 1996).

7.2.5.1 Fish Vision

Light perception by fish is dependent upon the light transmission qualities of the water environment coupled with the spectral qualities of the fish retinal visual pigments (Ali 1959, 1975; Brett and Groot 1963; Fields 1966; Hoar 1951; Hoar et al. 1957; McDonald 1960; McFarland and Munz 1975; Mork and Gulbrandsen 1994; Nemeth 1989).

Habitat and genetics determine the light absorption capacities of fish visual pigments. Capacities differ across the solar spectral compositions specific to the habitats upon which these species depend for growth and survival (Browman et al. 1993; Coughlin and Hawryshyn 1993; Hawryshyn and Harosi 1993, Novales-Flamarique and Hawryshyn 1996; Wald et al. 1957).

Light is received by the fish retina. This light reception triggers physiologic responses. The visual cell layers consist of two types of photoreceptors, rods, and cones. These retinal pigments have different light thresholds and respond to light and dark with changes in their relative positions. When the light intensity is above the retinal pigment and cone thresholds, the eye assumes the light-adapted state. When the light intensity falls below threshold values, the cones expand away, and the eye assumes a dark-adapted state (Ali 1959). In freshwater laboratory studies, Ali (1959) found that when the light drops below particular thresholds, the school disbands and feeding by visual means ceases, with the extent of expansion and elongation dependent upon ambient conditions (Ali 1975).

The time period for such physiologic changes in response to light variations varies across species and lifestages. At the juvenile stage, the time required for light-adapted chum and pink salmon fry to fully adapt to dark conditions was found to range from 30 to 40 minutes. However, the time required for dark-adapted fry to adapt to increased light conditions was found to range from 20 to 25 minutes (Ali 1959; Brett and Ali 1958; Protasov 1970). During these transition periods, the juvenile chum’s visual acuity ranges from periods of blindness to a slightly diminished capacity, depending upon the magnitude of light intensity contrasts. As the animals become older, the time required for light adaptation generally shortens. The time necessary to adapt to the dark, on the other hand, tends to increase with age. The progression of retinal changes from one state to another is influenced by the intensity of the introduced light and the intensity of light to which the fish have been previously exposed (Ali, 1962, 1975; Fields 1966; Protasov 1970; Puckett and Anderson 1987). It is the contrasts in light levels that determine the changes the eye undergoes and the speed of transition from one state to another. Fish previously exposed to higher light intensities become dark-adapted more slowly than those previously exposed to lower light intensities (Ali 1962). A review of the literature covering juvenile salmon behavioral responses to ambient and artificial light also revealed species-specific behavioral differences. Species that occupy and defend stream territories, such as coho, tend to be quiescent at night, while species that disperse to estuaries, such as Chinook, pink, and chum, typically school, show nocturnal activity, and demonstrate an aversion to light (Godin 1982; Hoar 1951).

The teleost fishes, a classification that includes all HCP fish species with the exception of the lampreys and the sturgeons, depend on sight for feeding, prey capture, and schooling. For these fishes, sight is the primary sensory organ used for spatial orientation, prey capture, schooling, predator avoidance, and migration. As juveniles, they utilize nearshore or shallow water habitats and share a sensitivity to ultraviolet wavelengths reflected in shallow-water habitats (Britt 2001, Tribble 2000, both in Nightingale and Simenstad 2001b). By interfering with sight, modification of the underwater light environment may affect these fundamental activities. Shade can affect fish and invertebrates by disrupting normal migration patterns, reducing the ability to avoid predators, capture prey and reducing available refuge (Ali 1962, 1975; Britt 2001; Fields 1966; Hoar et al. 1957; Johnson et al. 1998; McDonald 1960; Mork and Gulbrandsen 1994; Nightingale and Simenstad 2001a; Tribble 2000).

Juvenile and larval fish are primarily visual feeders, with starvation being the major cause of larval mortality in marine fish populations. Survival has been found to be linked to the ability to locate and capture prey and avoid predation (Britt 2001). This ability depends on sufficient light. Tribble (2000) found the swimming and feeding behavior of juvenile and larval sand lance to be reduced with low-light levels. Similar to other juvenile fishes with cone-based vision, the retinal cells of larval sand lance exhibit limited visual acuity in low-light environments. Their visual acuity increases with growth, with an eventual development of rod vision that provides them with vision in light-limited environments. Rods appear to develop at 0.94-in (24-mm) fork length, and full adult visual acuity develops at 1.38-in (35mm) fork length. This visual development prepares them for transition to deeper waters.

Tribble (2000) reports that the visual development of Pacific sand lance reflects the respective habitats they occupy given their size. At 1.97 in (50 mm) in length, they begin to move into deeper pelagic waters where the light environment changes, and their light requirements for prey capture change in response to the light wavelengths characteristic of that habitat. Many juvenile fishes using nearshore habitats, such as the Pacific sand lance (Tribble 2000), salmonids (Ali 1959), and lingcod (Britt 2001), share this sensitivity to ultraviolet (UV) wavelengths reflected in shallow nearshore marine habitats. Similar to salmonids, yellow perch and sand lance have been found to lose UV sensitivity with growth. Browman et al. (1993) reports this loss of UV sensitivities to be size-related rather than age-dependent and to likely correlate with the time that such fishes move from shallow to deeper water habitats and move from feeding on small crustaceans and other zooplankton to larger food items. As zooplankton reflect short wavelength light, such as UV, this provides an advantage for juvenile fishes with UV sensitivity feeding upon zooplankton in shallow nearshore waters. The ability of zooplankton to reflect UV is likely due to high concentrations of amino acids that protect them from the damaging effects of UV radiation.

Figure 7-1 depicts light conditions related to juvenile salmon behavior such as schooling, predator avoidance, feeding, and migratory behavior.

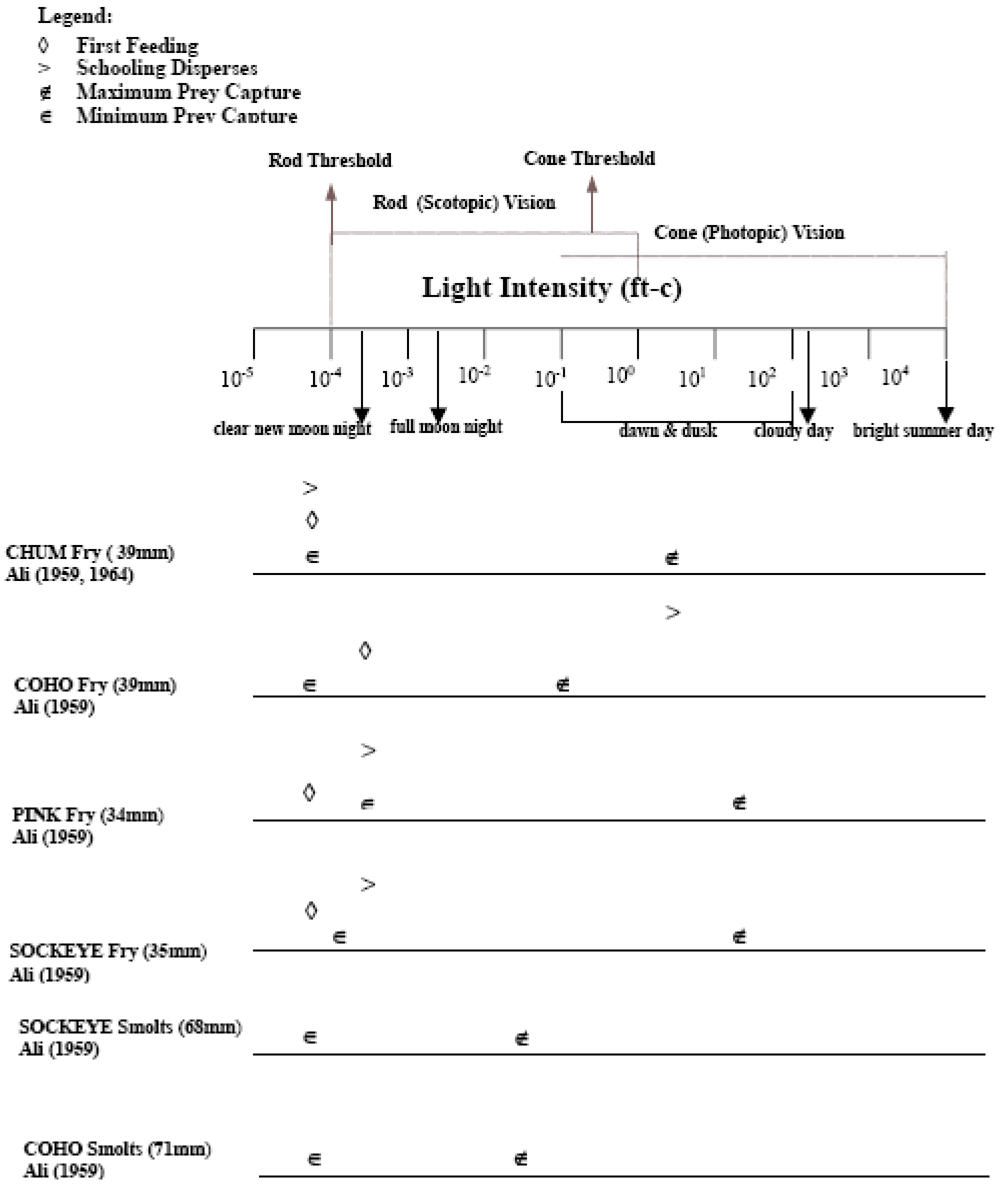


Figure 7-1: Juvenile Salmon Behavior Patterns Related to Light Intensity
Source: Nightingale and Simenstad 2001b

Tribble (2000, in Nightingale and Simenstad 2001b) found the swimming and feeding behavior of juvenile and larval sand lance (Ammodytes hexapterus) to be reduced with low light levels. Similar to other juvenile fishes with cone-based vision, the retinal cells of larval sand lance fall in the violet to green range, with limited visual acuity in low-light environments. Their visual acuity increases with growth as their cone pigments shift from violet to blue sensitivity. Tribble (2000, in Nightingale and Simenstad 2001b) reports that sand lance visual development reflects the habitats they occupy at given total lengths. Rods appear to develop when the fish reach approximately 1 inch (24 millimeters [mm]), and full adult visual acuity develops at 1.4 inches (35 mm). At approximately 2 inches (50 mm) in size, the fish will begin to move into deeper pelagic waters, where the light environment changes, and their light requirements for prey capture change in response to the light wavelengths characteristic of that habitat. At this point they will largely depart from the range of water depths where they may be affected by overwater structures. A similar change in visual sensitivity has been observed in yellow perch. Brownan and Hawryshyn (1994, in Nightingale and Simenstad 2001b) report this loss of ultraviolet sensitivity to be size-dependent rather than age-dependent and to likely correlates with the time when fishes move from shallow to deeper water. These results suggest that shading effects attributable to overwater structures predominantly affect smaller fish, and that "shading" as an impact includes the loss of both visual and ultraviolet wavelengths of light.

7.2.5.2 Daytime Shading and Fish Behavior

In response to daytime shading, fish potentially modify migration direction or behavior, resulting in increased energy expense. Shading can also reduce foraging success and increase potential exposure to predation. In addition, shading can modify species assemblages to a degree that available habitat is rendered unsuitable for native fish or invertebrate species. For invertebrates, shading can alter the suitability of habitat and reduce foraging opportunity as well as the availability of nutrients, resulting in decreased survival, growth, and fitness.

7.2.5.2.1 Shade in Freshwater Systems

Shade cast by overwater structures in the freshwater environment can be used by some fish as cover and can increase predation on juvenile salmonids (Tabor et al. 1998). Indeed, in freshwater environments of western Washington, largemouth and smallmouth bass are common predators of juvenile salmonids, and several authors have documented the use of overwater structures by bass (Carrasquero 2001; Kahler et al. 2000, Tabor et al, 1998, Stein 1970, Helfman 1979 both in Carrasquero 2001). Carrasquero's (2001) review found that the attraction of fish to floating or overhanging objects is linked to the shade produced by the objects, and Kahler et al. (2000) suggests that piers, piles, boatlifts, and moored boats provide cover, shade, and focal points that benefit exotic predators of juvenile salmon, such as smallmouth and largemouth bass. An alternative explanation of fish attraction to on-water and overwater structures in fresh water was presented by Fresh (pers. comm., in Carrasquero 2001), who explains that both the structures and the shade they cast may provide fishes with physical reference points for orientation.

Interactions of smallmouth bass and juvenile salmonids depend on timing of salmonid outmigration, salmonid species, and residence of the juvenile salmonids and found studies that suggest the attraction of predatory fish (including largemouth bass) to floating or overhanging objects is linked to the shade produced by the objects rather than to the tactile stimulus and that the larger the floating object, the greater the shaded area, and thus the greater the number of fish attracted to such objects (Carrasquero 2001). This assumption suggests that shading from overwater structures alters fish distribution and aggregation in fresh water. In addition, Kahler et al. (2000) states that shading from overwater structures may reduce the abundance of prey organisms available to juvenile salmonids and forage fish by reducing aquatic vegetation and phytoplankton abundance.

In freshwater environments of Western Washington, largemouth bass and smallmouth bass are common predators of juvenile salmonids, and several authors have documented the use of overwater structures by bass in Western Washington waters. Stein (1970, in Carrasquero 2001) examined the types of cover used by largemouth bass in Lake Washington and found that they prefer areas of heavy log and brush cover over other habitat types (including docks). However, largemouth bass are commonly found under docks in early spring and are thought to be present there until late summer (Stein 1970, in Carrasquero 2001).

Interactions between smallmouth bass and juvenile salmonids depend on factors such as the timing of salmonid outmigration, salmonid species, and residence time of juvenile salmonids in lentic (still-water) or lotic (flowing) environments (Fayram and Sibley 2000, in Carrasquero 2001; Gray et al. 1984; Gray and Rondorf 1986; Pflug and Pauley 1984; Poe et al. 1991; Shively et al. 1991; Tabor et al. 1993; Tabor et al. 2000; Warner 1972;).

Carrasquero (2001) presents the following observations and inferences of predator/prey aggregations in freshwater environments under and around structures:

- Different fish species respond differently to the shade produced by overwater structures.

- Smallmouth bass and largemouth bass have a strong affinity to structures, including piers, docks, and associated pilings.

- Bass have been observed foraging and spawning in the vicinity of docks, piers, and pilings; where vegetation is lacking, largemouth bass seek other forms of structures, such as dock pilings.

- Smallmouth bass are opportunistic predators that consume prey items as they are encountered and are major predators of juvenile salmonids.

- Fish, particularly largemouth bass, seem to be attracted to the shade produced by floats, rather than their physical structure. In contrast, smallmouth bass do not seem to be attracted to the shade produced by such structures.

- In reservoir systems of Eastern Washington, juvenile salmonid predation is specific to the behavior and distribution of each salmonid species and its predator. The behavior and distribution of predator and prey species reportedly depend on temperature, the degree of shore-zone development, the slope and substrate of the shoreline, and the presence of man-made in-water structures.

7.2.5.2.2 Shade In Marine Systems

In marine environments, shading also influences prey abundance and prey capture. Haas et al. (2002) found that densities and assemblages of important epibenthic prey organisms were reduced under large overwater structures. In New York Harbor, Able et al. (1998) found juvenile fish abundance to be reduced under piers when compared to open water or areas with only piles but no overwater structure. This is likely due to both limitations in prey abundance and prey capture under structures. In a New York study of pier impacts on fish growth and prey resource abundance, Duffy-Anderson and Able (1999) compared growth rates of caged juvenile fish under municipal piers to those of fish caged at pier edges and in open water beyond piers. Those fishes caged under the piers showed periods of starvation, which could potentially make these individuals more vulnerable to predation, physiological stress, and disease. Along the pier edge, they found growth rate variability to be extremely high and likely related to light levels. They concluded that light availability is likely an important component of feeding success. They concluded that large piers do not appear to be suitable habitat for some species of juvenile fishes and that increased sunlight enhances growth.

The addition of floating piers is also known to affect nearshore ecology by shifting population structures to non-native species as a result of shading. In southern California, Reish (1961) observed a succession of attached organisms occurring on marina floats with an apparent climax community of the *Mytilus* mussel and *Ulva* algae after the floats were in the water for 6 months.

In abundance, *Ulva* spp., an opportunistic green macroalgae, is known to reduce light and oxygen and create an anoxic environment (Hull 1987; Hernandez et al. 1997). Through shading, the algae *Ulva* is capable of triggering habitat shifts resulting in declines of eelgrass and concomitant increases in *Ulva* (Wilson and Atkinson 1995; Wilson 1993). The Puget Sound Expedition, a survey of nonindigenous species, sampled dock-fouling organisms on floats at 26 marinas throughout the entire Puget Sound region and identified 39 nonindigenous species (Cohen et al. 1998).

In the marine nearshore, daytime light reduction caused by shading under overwater structures could cause migrating juveniles to move into deeper waters, increasing the risk of predation by larger predators that occupy pelagic waters (Heiser and Finn 1981, Pentec 1977, in Nightingale and Simenstad 2001b). Predation mortality may increase through altering predator detection and reducing refugia provided by the schooling behavior of juvenile salmonids (Pentec 1997, in Nightingale and Simenstad 2001b).

Based on a combination of light measurements, visual fish survey, and acoustic tagging and telemetry fish tracking undertaken over a 7-week period between April 20 and June 3, 2005, Southard et al. (2006) found under-terminal light levels at the Anacortes, Bainbridge, Clinton, Edmonds, Fauntleroy, Kingston, Mukilteo, Port Townsend, Southworth, and Vashon terminals to deter or delay juvenile salmon movement along the nearshore. This effect was found to be dependent upon nearshore morphology, tidal level, and terminal design features affecting light availability.

Behaviors important to the growth and survival of fishes, such as migration, schooling, and feeding, are known to be altered by changes in light availability. For example, abrupt transitions from light to dark can cause juvenile Chinook salmon to alter their migration pathway from the nearshore (shallow water) to deeper water or avoid an overwater structure altogether (Tabor et al. 2004.) Some salmonids commence or terminate these behaviors in response to specific light levels or thresholds. In a snorkel and beach seine survey of Seattle marine shorelines, Toft et al. (2004) reported that juvenile salmon avoided swimming beneath overwater structures, while other animals (such as crabs and sculpin) were found in these under-dock habitats. Large groups of juvenile salmonids were found in the vicinity of overwater structure sites; however, most juvenile salmonids were observed at the edge of the overwater structure or farther away, with only one school observed underneath a structure. Similarly, only one Pacific sand lance was observed under an overwater structure, with most being along the periphery or in the general vicinity of the overwater structures. In general, most fish were not observed underneath overwater structures. This study suggests that the under-pier environment, in particular shading effects, could affect the behavior and movement of salmon along the nearshore area (Toft et al. 2004; Simenstad et al. 1999; Able et al. 1998).

Although it is believed that predation risks are elevated when fish move into deeper waters around piers, the actual potential for increased predation due to aggregating predators under structures in marine environments is uncertain (Weitkamp 1981; Taylor and Wiley 1997, in Nightingale and Simenstad 2001b). Taylor and Wiley (1997) found no aggregation of avian predators and Weitkamp (1981) reported no aggregation of aquatic predators during the peak juvenile chum outmigration. Consistent with these findings, Penttila and Aguero (1978, in Nightingale and Simenstad 2001b) found no empirical evidence of predation among the marina floats in Birch Bay, but instead found evidence of competition among fish species for mutually preferred prey resources (i.e., the calanoid and harpacticoid copepods). Fresh and Cardwell (1978, in Nightingale and Simenstad 2001b) list 17 potential predators of juvenile salmon in the southern Puget Sound region and find that only three (maturing Chinook, copper rockfish, and staghorn sculpins) prey extensively on nearshore fishes. Their analysis of food habits found only staghorn sculpins with juvenile salmon in their stomachs, and there was no evidence that staghorn sculpins were in greater abundance under structures than elsewhere in the study area. Additionally, Ratte (1985, in Nightingale and Simenstad 2001b) found sea perch and pile perch, which do not prey on salmonids, to be the most abundant fish species under docks. Nightingale and Simenstad (2001b) and Southard et al. (2006) summarize these and additional studies that pertain to fish behavior, including migration, distribution, and predator/prey relationships potentially associated with overwater structures in marine areas of Puget Sound.

7.2.5.3 Nighttime Artificial Lighting

Artificial night-light-induced changes to ambient nighttime conditions appear to affect fish migration behavior and place some species at risk of increased predation (Fields 1966; Johnson et al. 1998; Prinslow et al. 1979; Ratte and Salo 1985;Weitkamp and Campbell 1980; Weitkamp and Schadt 1982;). Prinslow et al. (1979) reported changes to fish assemblages and predation rates during a study of the effects of high-intensity security lights on a naval base (Bangor) in Puget Sound's Hood Canal. At that site, the level of intensity of artificial night lighting appeared to influence the behavior of fishes, with significantly greater light intensities (200–400 lux) attracting aggregations of juvenile chum and other small fishes. This aggradation suggested a potential to delay chum outmigration through the canal. Spiny dogfish, a Puget Sound shark, also appeared to be attracted to security lighting, likely due to the illumination of aggregating prey. Although herring and sand lance were not the subject of the study, Prinslow et al. (1979) reported potential exposure of herring and Pacific sand lance to predation due to the effects of the security lighting. Prinslow et al. (1979) suggested that based on study observations, the continuous use of high-intensity security lighting at the Bangor wharves could contribute to increased predation of HCP species.

Impacts to fish from artificial lighting are often the result of changes in nighttime behaviors such as migration, activity, and location (Nightingale and Simenstad 2001b) and potentially in schooling behavior in juvenile salmonids (Ali 1959, 1962, in Simenstad et al. 1999). Therefore, behavioral differences between species at differing life stages, life histories, and behaviors specific to the local environment must be considered when evaluating potential impacts from artificial light. For instance, different species of salmonids have different nighttime behaviors. Species that occupy and defend stream territories, such as coho salmon and steelhead trout, tend to be quiescent at night (Simenstad et al. 1999), while species that disperse to lakes and estuaries as juveniles, such as sockeye, Chinook, pink, and chum salmon, typically school and show nocturnal activity (Godin 1982, Hoar 1951, both in Nightingale and Simenstad 2001b). Behavioral differences in salmonid responses to artificial lighting have been observed by several authors. Ocean-type juvenile salmon, such as chum and summer and fall run Chinook, are attracted to lights at night (Simenstad et al. 1999). Pucket and Anderson (1988, in Simenstad et al. 1999) and Nemeth (1989, in Simenstad et al. 1999) found that different species of salmon react differently to strobe lights; Mork and Gulbrandsen (1994, in Simenstad et al. 1999) found differing activity levels in reaction to lights at surface and bottom depths in different species of salmon, trout, and char. Fields (1966, in Simenstad et al. 1999) found that spring migrant juvenile salmon were more repulsed by bright lights than were later migrants. Behavior patterns of different salmon species related to different light intensities and other details of artificial light impacts to juvenile salmonids are reviewed by Simenstad et al. (1999).

Impacts to fish also depend on the fish's ability to adapt to dark or lighted conditions and the intensity and type of light. Ali (1959, in Simenstad et al. 1999) found that the eyes of sockeye fry and smolts and coho smolts adapt to light more slowly than do the eyes of coho, Chinook, and pink fry. Other studies by Ali (1959, 1962, in Simenstad et al. 1999) reveal the threshold light intensities for different behaviors of juvenile salmon.

Artificial lighting may be used during the construction of overwater structures, and some kinds of structures also require nighttime lighting for security or operations. Nighttime artificial lighting has been shown to change fish species assemblages by:

- Attracting fish to lighted areas (Prinslow et al. 1979, in Nightingale and Simenstad 2001b; Simenstad et al. 1999; Nightingale and Simenstad 2001b)

- Delaying salmonid migrations (McDonald 1960, in Tabor et al. 1998; Prinslow et al. 1979, in Nightingale and Simenstad 2001b; Tabor et al. 1998)
- Increasing the risk of predation (Kahler et al. 2000; Tabor et al. 1998)
- Altering predator avoidance and detection (Tabor et al. 1998)
- Increasing prey capture success for some species of fish (Nightingale and Simenstad 2001b; Prinslow et al. 1979, in Nightingale and Simenstad 2001b).

Similarly, in a study of lighted and nonlighted areas along the Cedar River in the City of Renton, Washington, Tabor et al. (2001) found increased nighttime lighting intensities to have a profound effect on the behavior of salmon fry. Results indicated that increased levels of nighttime artificial light intensity, measured at lighted building and bridge sites, appeared to cause sockeye fry to delay migration and move to the low-velocity and lighted shoreline habitats, where they were found to be more vulnerable to increased predation. Even small increases in light intensity levels appeared to affect fry behavior. Tabor found nightly downstream migration of sockeye fry to be initiated after light intensity was less than 1 lux. However, with the addition of 32 lux, migration almost completely stopped. Given such changes to the habitat, Tabor et al. (1998) reported that a reduction in the intensity of artificial night lighting could benefit these sockeye salmon. In a study comparing urban and rural nighttime light regimes for lake environments, Moore et al. (2006) found the relative intensity of illumination to increase along the suburban-to-urban gradient, under both clear and cloudy conditions, with the nighttime surface light intensity for urban lakes ranging from 7 to 48 times the light intensity for lakes in rural environments. An effect of the higher nighttime light intensities found in urban environments was the suppression of vertical migration of zooplankton in urban lakes (Moore et al. 2006). Nighttime light intensities have also been found to affect fish foraging, schooling, spawning, and vertical movement in the pelagic zone (Blaxter 1975; Gliwicz 1986; Robertson et al. 1988; Luecke and Wurtsbaugh 1993; Appenzeller and Legget 1995; Contor and Griffith 1995).

A number of studies have shown that fish respond quite differently to various lighting types, such as flickering strobe, mercury, or halogen light sources (Fields and Finger 1954; Hoar et al. 1957; Fields 1966; Prinslow et al. 1979; Puckett and Anderson 1987; Nemeth 1989; Johnson et al. 1998). In Washington State, fish responses to increased nighttime underwater light intensities have been found to pose potentially significant population effects including changes in light-mediated predation rates on fish, reduction in prey capture efficiency by increased fish avoidance behavior, and slowing of migratory behavior (Prinslow et al. 1979; Tabor et al. 1998, 2001).

The few studies that have examined predation rates on juvenile salmonids under varying light intensities have generally shown that within the natural range of light intensities (e.g., overcast skies, moonless nights, clear nights, moonlit nights), predation increases with increasing light (Ginetz and Larkin 1976, Mace 1983, Patten 1971, all in Tabor et al. 1998); however, this occurrence cannot be extrapolated to determine impacts of artificial night lighting and for all species and life stages of fish. Ali (1959, in Simenstad et al. 1999) found that the maximum prey capture success for coho fry and sockeye and coho smolts was at light intensities equivalent to levels found at dawn or dusk, whereas maximum prey capture success for sockeye and pink fry was found to be equivalent to a cloudy day. Tabor et al. (1998) showed that under freshwater laboratory conditions, sculpin capture success of sockeye fry decreased with increased light. The authors also found that sculpin can capture sockeye fry even in complete darkness. Although sculpin success at capturing sockeye decreased with increasing light in a circular tank, the increased light slowed emigration of sockeye fry in a simulated stream, and predation increased under the lighted conditions due to the slower migration rate. The light may have also caused the fry to migrate in areas of lower water velocity and closer to the bottom, leaving them more susceptible to predation by sculpin (Tabor et al. 1998). Grebes, blue herons, and other birds have been observed feeding at night on the Cedar River delta in an area lit by Boeing Company facilities (Warner, pers. comm., in Kahler et al. 2000), and Tabor (pers. comm., in Kahler et al. 2000) observed grebes foraging under lights at night on Lake Washington. Finally, Kahler et al. (2000) suggests that lighting attached to piers in Lake Washington where bass congregate may benefit bass by extending the duration of predation because it allows the visual predators to forage at night.

Studies examining the use of artificial light for guiding salmonids safely through migration barriers, such as hydroelectric dams, have found measurable differences in different species’ responses to both the quantity and quality of the light stimulus. For example, Puckett and Anderson (1987) found juvenile salmon to be attracted to incandescent light when encountering a decrease in ambient light intensity. In the case of steelhead, Puckett and Anderson (1987) found the fish to initially avoid the mercury light and then to swim toward the light, likely following adaptation.

Nighttime lighting can result in altered migration behavior and timing (interruption or stalling as a result of attraction to light sources) as well as increased predation (as a result of aggregation). Subsequently, fish survival is reduced.

7.3 Ecosystem Fragmentation

Ecosystem fragmentation refers to the disruption of ecological processes by reducing the connectivity between different components of the ecosystem. The mechanism of ecosystem fragmentation differs from other mechanisms because it combines the elements of physical features, time, and location to represent how potentially covered species would be impacted by loss of ability to access these features.

HPA-permitted structures can be categorized as contributing to ecosystem fragmentation by:

- Altering habitat complexity
- Altering migration patterns/ presenting barriers to fish passage and dispersal
- Altering lateral connections between rivers and floodplains

7.3.1 Altered Habitat Complexity

7.3.1.1 General Alterations: All Environments

Human modifications of the environment, including activities permitted through the HPA program, can simplify and fragment habitat. Loss of habitat complexity can contribute directly to decreased growth, survival, and population productivity of HCP species. Studies have indicated that decreased habitat complexity negatively affects the survival and growth of aquatic organisms. Reduced shelter availability will increase predation and is not energetically favorable for fishes.

- In a recent study by Finstad et al. (2007), it was found that juvenile Atlantic salmon exhibit accelerated mass loss rates with decreasing access to shelter, indicating that the juvenile fish had to expend greater energy when there was no available shelter.

- In another study by Babbitt and Tanner (1998), tadpole survival was 32 percent greater under high than under low cover, suggesting that increased cover decreased predator foraging efficiency. Although the prey in this study were not HCP species, the effect of cover on predation rates can be extrapolated to HCP species that utilize vegetated cover during early lifestages.

- Limited habitat availability may lead to density-dependent mortality for those species that cannot find unoccupied cover and may be exposed to increased predation or high-energy environments (Forrester and Steele 2004).

7.3.1.2 Ecosystem-Specific Effects: Marine and Estuarine

Lagoons, sometimes called pocket estuaries, provide important rearing habitat for juvenile salmonids, including Chinook (Busby and Barnhart 1995) and coho (Minakawa and Kraft 2005) as well as Pacific herring (Saiki and Martin 2001). Access to high-quality lagoon habitat has been shown to be the critical path in the restoration of Chinook in the Skagit River system (Beamer et al. 2005).

Lagoons have declined both in terms of size and number due to human modifications to lowland coastal areas in Puget Sound (Beamer et al. 2005). Lagoons can be lost due to changes in the tidal prism (Sherwood et al. 1990). The primary impact of a loss of lagoon habitat would be to expose juvenile salmonids and forage fish to increased risk of predation (Hood 2006; Wagner and Austin 1999).

Shallow water marginal habitats in the nearshore environment are similarly important to a variety of salmon species (Fresh 2006). Loss or fragmentation of these habitats can have significant effects on the survival, growth, and fitness of dependent species.

The loss of shallow water and lagoon habitats can affect invertebrate species. The Olympia oyster uses lagoons, therefore the species could be susceptible to losses due to changes in tidal prism (Baker 1995).

7.3.1.3 Ecosystem-Specific Effects: Riverine

An indirect impact from the loss of decreased habitat complexity is an increase in nutrient loading to downstream receiving waters. Channel complexity promotes the retention of water and organic material.

Ecological connectivity is essential between riverine and riparian ecosystems (Stanford and Ward 1993).

Side channel habitat provides low energy refugia for fishes and is largely viewed as a net benefit to stream organisms (Jungwirth et al. 1993). Rock weirs create low velocity zones in the lee of the structures and accelerate flow between the rocks. These variable habitats are utilized by different species during various lifestages. For instance, juvenile Chinook salmon have been shown to take refuge in low velocity habitat behind rock weirs, while steelhead parr utilize deep water habitat between the rocks during summer low flow periods (Fuller 1990).

Ecosystem fragmentation can lead to changes in genetic diversity. A study of the genetic variation within European grayling populations from the Skjern River (Denmark) showed that present-day grayling differed from historic stocks due to the drift of larvae downstream and restricted migration upstream (Meldgaard et al. 2003).

Fishes' requirements for structural diversity and complexity within habitats may differ by life stage. Li et al. (1984) documented lower larval and juvenile fish densities and richness along revetted versus natural shorelines in the Willamette River, but higher adult abundances. Species captured by Li et al. (1984) included Chinook salmon, speckled dace (*Rhinichthys osculus*), torrent sculpin (*Cottus rhotheus*), and largemouth bass (*Micropterus salmoides*).

7.3.1.4 Ecosystem-Specific Effects: Lacustrine

Diversity and the interconnectedness of different shoreline types have been found to be more crucial than any particular substrate or vegetation type on lake shorelines for a variety of species (Pratt and Smokorowski 2003). This may explain the dramatic differences observed in ecosystem health when modified and unmodified shorelines are compared (Roth et al. 2007; Scheuerell and Schindler 2004).

Altering the wave energy reaching the shoreline can lead to a loss or fragmentation of existing spawning habitat for sockeye salmon. Sockeye salmon are the primary HCP species potentially affected by lakeshore modifications. Lake shorelines represent crucial spawning habitat for sockeye (Burgner 1991; Scheuerell and Schindler 2004). Juvenile Chinook salmon also use lacustrine littoral zones (Sergeant and Beauchamp 2006). Sergeant and Beauchamp (2006) have shown that although substrate preferences for juvenile Chinook are weak, they prefer fine substrates with cover.

No studies showing a relationship between effects on invertebrates and loss of nearshore habitat in lacustrine ecosystems were found.

7.3.2 Altered Migration Patterns/ Barriers to Fish Passage and Dispersal

7.3.2.1 General Effects: All Environments

Access to habitat is one of the ecological functions that is important to potentially covered species and that may be impaired to some degree by HPA-permitted structures.

Any structure that unintentionally selects against population diversity is likely to be detrimental to its long-term viability (McElhany et al. 2000; Thompson 1991).

7.3.2.2 Ecosystem-Specific Effects: Marine and Estuarine

Loss of access to estuarine and floodplain rearing habitats has been broadly implicated as a contributing factor in the decline of anadromous salmonids in the Pacific Northwest, particularly those species with demonstrated dependence on these habitat types (such as coho, Chinook, and chum salmon) (Beechie et al. 1994; Giannico and Souder 2004; Gregory and Bisson 1997). Loss of access to floodplain and estuarine rearing affects growth and fitness and, in the case of estuarine rearing, limits the potential residence time in brackish water habitats that facilitate the physiological transition from fresh to marine water during smoltification. These factors can influence survival.

7.3.2.3 Ecosystem-Specific Effects: Riverine

Alteration of longitudinal connectivity and resulting changes to both upstream and downstream habitat complexity will impact HCP species because many of the HCP fish species require a range of habitat types throughout their life histories. Potentially covered species must be able to access habitats with velocities favorable to their physiological needs as a species or a life stage. For example, such species as mountain sucker and lamprey require slow water as a general habitat need, while juvenile salmonids require slow-moving water for cover and energy refuge. Many species find the quiescent habitats they need in off-mainstem channels, backwaters, tidal sloughs, and shallow water areas.

Most fish must be able to move freely upstream and downstream during both juvenile and adult stages (Sargeant et al. 2004). Adult salmon returning to their spawning streams must have unobstructed upstream migration corridors. Specific studies include:

- Juvenile salmon rearing in rivers have been found to move both upstream and downstream to utilize rearing habitat, even in streams where spawning has not been documented (Kahler and Quinn 1998).
- Steelhead trout were found to use main channel and side tributaries equally (Bramblett et al. 2002), and coho salmon were observed more often in tributaries (Bramblett et al. 2002).
- Radio-tagged Pacific lamprey in the John Day River (Oregon) were observed using the lateral margins of riffles and glides, and used boulders for cover (Robinson and Bayer 2005).
- In Montana, mountain suckers were shown to prefer riffles (Wydoski and Wydoski 2002).

Little is known about the effects of longitudinal connectivity on invertebrate species. However, freshwater mussel larvae rely on attachment to host fish (Nedeau et al. 2005), so reduction in fish populations may have an indirect effect on mussels such as the California floater and western ridged mussel. In fact, several studies summarized by Watters (1999) showed that the loss of fish hosts has been linked to the decline of native mussel species.

Loss of habitat access may also restrict the dispersal of HCP freshwater invertebrate species. This can occur in two ways. First, the structure may restrict the distribution of host fish, affecting the dispersal of parasitic larvae (Vaughan 2002; Watters 1996). Second, the structure may restrict upstream movement of mussels and snails capable of crawling along the stream bottom (Vaughan 2002).

The potentially covered freshwater mussels migrate through their range as larvae attached to fish gill membranes (Brim Box et al. 2004), and loss of habitat access for the host fish would also lead to a reduction of the mussels' range. Among the 35 fish that Brim Box et al. (2004) found to be carrying mussel larvae, 34 were speckled dace and one was redside shiner (*Richardsonius balteatus*). Brim Box et al. (2004) found no mussel larvae attached to the small numbers of smallmouth bass, northern pikeminnow (*Pyctocheilus oregonensis*), and largescale sucker (*Catostomus macrocheilus*) that they inspected. Species such as California floater mussels may face demographic risks from the unintentional limitation of fish passage because the full range of their host fish species is poorly understood, meaning that the migration behavior and swimming requirements of these species may not be well accounted for. Likely host-fish species include native minnows (cyprinids) and non-native mosquito fish (Nedeau et al. 2005).

7.3.3 Altered upstream transport of organic material

Alteration of fish migration patterns can lead to the alteration of upstream transport of organic material, particularly marine-derived nutrients associated with the carcasses of anadromous fish species. Numerous studies have documented the contribution of marine-derived nutrients provided by anadromous fish on food web productivity (Brock et al. 2007; Chaloner et al. 2007; Chaloner et al. 2002; Chaloner and Wipfli 2002; Gross et al. 1998; Hicks et al. 2005; Lessard and Merritt 2006; MacAvoy et al. 2000; Merz and Moyle 2006; Minakawa et al. 2002; Mitchell and Lamberti 2005; Moore et al. 2007; Nagasaka et al. 2006; Scheuerell et al. 2005; Schindler et al. 2005; Yanai and Kochi 2005; Zhang et al. 2003). Reductions in the delivery of marine-derived nutrients can affect food web productivity in ways that are detrimental to the growth, fitness, and productivity of juvenile salmonids as well as other native fish species (Bilby et al. 1998; Heintz et al. 2004; MacAvoy et al. 2000).

Marine-derived nutrients distribute broadly in aquatic ecosystems (Cederholm et al. 1989), demonstrably affecting algal growth and nutrient export (Brock et al. 2007; Chaloner et al. 2007; Chaloner et al. 2002; Mitchell and Lamberti 2005; Moore et al. 2007; Schindler et al. 2005; Yanai and Kochi 2005). Several studies have examined the influence of marine-derived nutrients on the productivity of riparian vegetation, which in turn affects habitat structure (Bartz and Naiman 2005; Ben-David et al. 1998; Helfield and Naiman 2001, 2002; Merz and Moyle 2006; Nagasaka et al. 2006; Naiman et al. 2002; Scheuerell et al. 2005).

Structures that improve passage of native fish species will likely produce beneficial effects on food web productivity and habitat structure, with attendant benefits on HCP species dependent on these habitats. Conversely, reduction in upstream transport of allochthonous nutrients will detrimentally affect the productivity of native fish populations. This inference is supported by research documenting the uptake of marine-derived nutrients by juvenile salmonids and other native fish species (Bilby et al. 1998; Heintz et al. 2004; MacAvoy et al. 2000).

While no data were identified regarding the direct influence of marine-derived nutrients on HCP invertebrate species, changes in ecosystem productivity resulting from decreased fish passage could affect the growth and fitness of these species, through changes to the forage base for filter feeding and grazing invertebrate species. Because the availability of marine-derived nutrients affects the productivity of host fish populations (Bilby et al. 1998), the dispersal of freshwater mussels, which are dependent on fish to transport their parasitic larvae upstream against the current, will be reduced where fish productivity has been adversely affected.

7.3.4 Altered connections between channels and floodplains

Potentially covered species require habitats that offer varying depths, velocities, and cover for refuge from their predators and from needlessly expending energy. Floodplain habitats provide these diverse habitat conditions. A reduction in the amount of side channel and floodplain can impact species that rely on any floodplain-associated habitats such as wetlands, beaver ponds, bogs, and off-channels.

Floodplains are habitat-rich areas with pools and side channels that are important nurseries for aquatic organisms (Bednarek 2001). Oxbow lakes are important habitat for juvenile fish (Penczak et al. 2003). Floodplains require periodic inundation. Even low-occurrence flooding (e.g., a 2-year flood) is important to sustain floodplain habitat diversity and to support aquatic species (Thoms 2003). The loss of LWD may also contribute to floodplain-channel connectivity losses. Floodplain–channel connection augments allochthonous carbon budgets in restored channels and engages habitat that would otherwise be inaccessible.

Riparian wetlands provide food resources for adjacent, less productive aquatic ecosystems and function as habitat for fish and invertebrates. Shallow flooded riparian areas are some of the most productive habitat within a watershed (Bunn et al. 2003; Junk et al. 1989; Schemel et al. 2004; Sommer et al. 2005; Sommer et al. 2001). Consequently, access to and resource transport through this zone is essential for the healthy functioning of river-floodplain systems. Research has indicated that riparian wetlands are nutrient sinks and carbon sources for adjacent channels (Tockner et al. 1999; Valett et al. 2005). In this fashion, wetlands function as zones of transformation, converting inorganic nutrients to organic forms which can be consumed and transferred up the food chain. Organic material is exported from the floodplain in four primary forms, coarse woody debris, coarse particulate organic matter, dissolved organic matter, and suspended algal biomass. Suspended algal biomass represents one of the most important forms of carbon export from floodplains because of the high nutrient content of the algal cells (Muller-Solger et al. 2002). Studies have indicated that when hydraulic residence time on floodplains is in the range of 2 days (Ahearn et al. 2006), algal biomass concentrations may begin to increase, reaching a maximum level at approximately 10 days (Hein et al. 2004). Although site specific, these studies indicate that riparian wetlands with extended hydraulic residence times will produce high levels of algal biomass. This algal biomass is an important food resource for organisms in both the wetland itself and the connected channel or open water habitat.

Studies summarizing the use of floodplains and riparian wetlands by fish and shellfish include:

- Floodplain connectivity creates forage and refuge habitat for several species of fish (Feyrer et al. 2006; Henning 2004).

- Chinook that rear on floodplains have been shown to grow faster than those rearing in adjacent channels (Sommer et al. 2001).

- Hayman et al. (1996) found sub-yearling Chinook in higher densities at backwater and off-channel habitat than in mainstem edge habitat of the Skagit River. This use of habitat implies that access to all stream reaches, even those absent of adult salmon, is needed by juvenile salmon.

- In a 2004 study of the Sacramento splittail (Ribeiro et al. 2004), fishes rearing in floodplain habitat were healthier and larger than fish from the same cohort that did not rear in this type of environment.

- Floodplain areas are important rearing habitats for coho (Beechie et al. 1994; Swales and Levings 1989). Juvenile coho have also been shown to use floodplain habitats when they are available, even in locations where they have disconnected in the past (Henning et al. 2006; Morley et al. 2005).

- Swales and Levings (1989) found that off-channel habitat in the Coldwater River, British Columbia were vital rearing areas for coho, while juvenile Chinook, steelhead, and Dolly Varden were most abundant in floodplain ponds.

- In a survey of habitat use in Southeast Alaska, Dolly Varden were observed using step-pools (Bryant et al. 2007) and were more common in tributaries compared to the main channel (Bramblett et al. 2002).

- Steelhead use both main channel and side channel habitat (Bramblett et al. 2002).

- Egg dispersal into newly inundated habitat has been hypothesized to increase the number of Columbia River white sturgeon. Larval white sturgeon have been shown to disperse to flooded riparian habitats for early rearing. Fragmentation of these habitats may be a factor in the decreased productivity of this species (Coutant 2004).

- Green sturgeon tagged on the Rogue River (Oregon) showed use of off-channel coves (Erickson et al. 2002).

- Side channels create refugia for juvenile fish (Jungwirth et al. 1993).

- Off-channel areas offer spawning habitat for sockeye salmon (Hall and Wissmar 2004).

- Floodplain ponds and backwater sloughs create zones of high retention and productivity that provide vital rearing habitat (Hall and Wissmar 2004; Sommer et al. 2005).

- Floodplain ponds and backwater sloughs also contain important sources of organic material for the channel (Tockner et al. 1999). The loss of connectivity between the channel and these habitats can result in a decrease in organic matter recruitment (Tockner et al. 1999; Valett et al. 2005).

- Whether for rearing or spawning, the viability of wetland habitats hinges upon connectivity with the adjacent open water system. Too much connectivity, and productivity of the wetland decreases (Tockner et al. 1999). Too little connectivity and the stranding of organisms may become a problem (Henning et al. 2006; Sommer et al. 2005).

- Loss of connectivity can reduce access to valuable foraging and rearing habitat (Henning et al. 2006).

- It has been estimated that in the Skagit Valley, 41 percent of sloughs and 31 percent of small tributary habitat has been lost to channelization for agricultural purposes (Beechie et al. 1994).

- Where the interaction of LWD with the channel is affected by channel incision, the ability of debris jams to promote access to floodplain habitats may be diminished.

While these potential effects are of concern, care should be taken not to assume that they are universal in extent or severity. In many instances, it may be appropriate to allow a stream system to return to a natural equilibrium gradient through channel incision. For example, many culverts occur in smaller stream systems with naturally limited floodplains and off-channel habitat. Regrading this type of channel and placing grade control structures will produce extensive short-term adverse effects on aquatic habitat conditions and HCP species occurring in these environments. These effects may significantly outweigh those that result from allowing a natural channel response to proceed, while the intermediate- to long-term benefits of both approaches are comparable. In such cases, the cost and the short-term effects of channel regrading would not be justified, unless property or infrastructure faces unacceptable risk. Given these complexities, it is desirable to include a licensed geologist and a qualified aquatic biologist in the design process to avoid unnecessary actions and/or undesirable outcomes.

7.3.5 Activity-Specific Effects

7.3.5.1 Jetties

A number of processes associated with the installation of a jetty could lead to a loss or fragmentation of existing habitat. The construction and maintenance of a jetty can limit the accessibility of fish through changes in both the structure of the shoreline and its characteristics (linear and impermeable versus undulating and covered in large woody debris). This can take the form of complete removal of habitats, such as the elimination of lagoon habitats due to changes in alongshore sediment transport patterns or the tidal prism. However, habitat accessibility can also be compromised if characteristics, such as having different hypsometry (distribution of depths) or elimination of riparian vegetation and large woody debris, make the shoreline inhospitable to the HCP species.

These include a loss of accessibility to coastal lagoons through a reduction of the tidal prism, a loss of substrate and appropriate depth necessary for spawning, and a change in the abundance of predators and prey.

Jetties and other hard shoreline structures have the potential to both attract and deter use of the nearshore by fish, birds, and people. Jetties have been shown to be a locus for shorebird activity, mostly as a result of jetties acting as concentrating mechanisms for passively drifting horseshoe crab eggs (Botton et al. 1994). Shorebirds are similarly attracted to Pacific herring spawn in natural settings on the western shore of British Columbia (Rodway et al. 2003). The concentrations of spawn and shorebirds, however, were associated with hard-substrate outcrops, similar to artificial hard points commonly associated with jetties. Although these concentrations can occur in the absence of man-made structures, the unnatural concentration of spawn on jetties would artificially enhance these processes. The degree to which the concentration of spawn and shorebirds affects the mortality of spawn and the dwindling numbers of Pacific herring or other planktonic breeders is unknown (Rodway et al. 2003). The complex interaction between people, birds, and fish can lead to a net loss of HCP species (Roby et al. 2002).

Jetties have been shown to concentrate juvenile Chinook salmon in some areas due to strong changes in salinity (Yates 2001). Similar features (groins) have been shown to concentrate chum and coho salmon (Miller et al. 2001), while protruding shoreline structures (i.e., analogous to jetties) have been shown to concentrate juvenile salmonids and all of the HCP forage fish species (Toft et al. 2007). These concentrations likely do not reflect an increase in production of these fishes, but rather a concentration due to bottlenecks in access to nearshore habitat. Because these concentration mechanisms are exploited by commercial fishermen (Creque et al. 2006; Miller et al. 2001), the risk of increased catch due to jetty installation likely has a negative effect on salmonid survival and navigation near such activities.

Jetties may pose lingering ecological effects when they are not designed properly to account for movement of sediment and wood alongshore. In the construction and maintenance of jetties, LWD is often removed from the surrounding jetty site and nearby adjacent navigation channels. In addition, altered wave energy and nearshore circulation can cause a change in the transport energy such that less wrack accumulates on beaches adjacent to the jetty.

7.3.5.2 Breakwaters

Breakwaters have not been shown to disconnect nearshore habitats, alter tidal prisms, or significantly influence nearshore stratification in the same way that jetties do. Therefore, breakwaters do not present the same degree of risk of disconnection to crucial habitats (e.g., lagoons) as jetties. The interaction between fishes near man-made structures is more complex than in natural settings and has only been studied in relation to low-crested structures (i.e., breakwaters) and artificial reefs.

As offshore hard structures, breakwaters do, however, have a significant impact on the types of ecologic assemblages found in their vicinity (Perez-Ruzafa et al. 2006). They also have a tendency to increase the numbers of piscivores, which, depending on the species of predator and prey could have a negative effect on HCP species (Pondella et al. 2002). As a result, breakwaters could alter predator–prey abundance and compromise certain habitats for HCP species.

Breakwaters and other artificial reefs have been examined in terms of their productivity for Washington fishes and invertebrates (Buckley and Hueckel 1985). When properly designed, submerged breakwaters generally increase the numbers of fish, including rockfish and lingcod (West et al. 1994). However, there is significant debate in the scientific community as to whether they increase the total productivity of the nearshore or simply attract those species from surrounding waters [i.e., “attraction versus production” (Pickering and Whitmarsh 1997)].

In some locales, submerged breakwaters have been seen as an ecological loss because the goals of constructing a breakwater are different from those of an artificial reef (Bulleri and Chapman 2004). This is particularly true if those breakwaters are fished or if the species of concern are eaten by the fishes attracted to the reef (Pondella et al. 2002). Because the literature is somewhat contradictory, it is important to understand the three factors that have emerged as crucial variables in the determination of the ecological success of a breakwater:

- *Species and diet of attracted fishes*. If the species attracted to and potentially enhanced by the breakwater structures are HCP species [e.g., bocaccio rockfish (Love et al. 2006)], the installation could be considered to have a net ecological benefit. It is also possible, however, that the introduction of predatory fishes could shift biomass away from smaller fishes (Pondella et al. 2002). If these smaller fishes are HCP species (e.g., Pacific herring), the project could present a net loss.

- *Commercial fishing*. If fishing is allowed on the breakwater, the concentration of HCP species can result in a loss in the number of total fish, while protection from fishing has the potential to increase the total number (Guidetti, Verginella et al. 2005).

- *Geometric complexity of structure*. If the structure is simple (with vertical walls), attraction of fish will be restricted and possibly limiting (Bulleri and Chapman 2004; West et al. 1994). However, if the breakwater provides sheltering sites, its installation may be of net ecological benefit [particularly for rockfishes (Love and York 2006)].

Substantial work has demonstrated that, at least for rockfishes, submerged emplacements of rock and other hard substrate can be used as a habitat enhancement measure (West et al. 1994). Although artificial reefs designed for habitat are usually placed in deeper water, several studies have shown that rock placed in shallower waters, typical of submerged breakwaters (Dean and Dalrymple 2002), can have the same benefits (Pondella and Stephens 1994).

Breakwaters change the distribution of wave energy, substrate, and wrack material along the shoreline. Large wood will most likely preferentially accumulate in the lee of breakwaters, similar to the substrate changes induced by these structures (Bowman and Pranzini 2003; Thomalla and Vincent 2003). This would cause the shoreline to be discontinuously covered by LWD, potentially compromising natural migration corridors to HCP fish species that use littoral zones.

Breakwaters restrict deposition and erosion in specific geometries useful for human activities. However, this shoreline geomorphology can be poorly suited for any or all life stages of fish and invertebrates. Breakwaters may pose lingering ecological effects when they are not designed properly to account for channel movement or to allow sediment and wood transport. Breakwaters may interfere with littoral migration of fish and invertebrate species or life-history stages. In such instances, the costs of replacing a newly constructed structure may create a strong incentive against the additional investment required to address the problem correctly. This may delay the actions necessary to provide appropriate migration for fish, perhaps as long as the design-life of the underperforming structure.

7.3.5.2.1 Breakwaters in Lacustrine Environments

Most of the work that has been performed on breakwaters in lacustrine environments has been conducted in the Great Lakes (Fitzsimons 1996; Marsden and Chotkowski 2001; Olyphant and Bennett 1994), which are substantially larger (and therefore subject to much larger waves) than any lakes in Washington State.

The primary differences between breakwaters in lacustrine environments and those in marine environments are the following:

- *Short-term stable water levels*. Lakes, although subject to long-term water level variability, are not generally subject to tides. As a result, the size of the breakwater that may be required is significantly smaller and can be placed much closer to shore. This would mean the area of alteration associated with breakwaters would be generally smaller in lakes than in marine waters.

- *Species of interest*. Although some species use both lacustrine and marine shorelines (e.g., salmonids), most fish and invertebrates use only one or the other, with the freshwater-only species having limited scientific study. Further, because studies of the specific impacts on invertebrates in lakes has been extremely limited, there is a data gap in terms of understanding the specific effects of breakwaters on those freshwater species.

- *Invasive species*. Freshwater breakwaters have been shown to harbor invasive species infestations capable of the wholesale disruption of lacustrine nearshore ecosystems (Marsden and Chotkowski 2001). In particular, zebra mussels present an ongoing and serious threat to Washington freshwaters (WDFW 2004).

7.3.5.3 Groins and Bank Barbs

By forcing flow into narrow corridors, groins can interrupt floodplain-channel connection and thereby cause the loss of habitat area and productivity.

Groins simplify stream structure and focus flow energy into limited areas, causing LWD to be flushed downstream.

Certain types of shoreline protection measures, like groins and bank barbs, may pose lingering ecological effects when they are not designed properly to account for channel movement or to allow sediment and wood transport. Groins and bank barbs may not provide adequate passage for all fish and invertebrate species or life-history stages. In such instances, the costs of replacing a newly constructed structure may create a strong incentive against the additional investment required to address the problem correctly. This may delay the actions necessary to provide appropriate fish passage, perhaps as long as the design-life of the underperforming structure.

7.3.5.4 Bank Protection and Stabilization

Bank protection can restrict natural geomorphic processes along the marine and riverine shorelines and can impose significant impacts on fish and invertebrates. By altering flow patterns, bank protection can interrupt floodplain-channel connection and thereby cause the loss of habitat area and productivity. Aquatic species must be able to access habitat with water depths favorable to their habitat needs. When favorable depths are lost, species are cut off from habitats they require. Isolation of species from shallow-water habitats can occur when armoring is placed waterward of the OHWL or into the intertidal zone of marine habitats, and the area immediately adjacent to the structure exists at a deeper point than nearshore species can inhabit, forage, or find refuge. The deep water condition can be exacerbated if armoring further causes scour of the bank, erosion of sediment, and lowering of shore elevation.

When bank protection degrades and reduces access to off channel refugia, such as off-mainstem channels, backwaters, tidal sloughs, and shallow water habitats (Beamer et al. 2005), species can either be displaced or be unable to seek refuge and must expend excess energy to maintain position and avoid being flushed out of preferred habitats. Bank protection structures that alter the flow regime, and thus depth and velocity, can impact fish habitat accessibility where water velocity exceeds their swimming ability or creates areas without adequate water depth or sufficient refuge areas.

Bank protection structures that alter depths and velocities in the marine intertidal zone can limit habitat access for prey of potentially covered species; in particular, forage fish such as surf smelt and sand lance spawn on fine-grained substrate in the upper intertidal zone, which may be locally reduced along armored shorelines. Further, Pacific herring spawn on submerged eelgrass and macroalgae in shallow-water areas that may also be reduced along armored shorelines (Thom and Shreffler 1994). Although erosion associated with hardening of the shoreline is often ephemeral, when the sediment supply is not maintained (Finlayson 2006), it presents the possibility of the elimination of elevations necessary for the proper wave and sediment transport conditions for the survival of surf smelt and sand lance larvae, concomitant with a loss of appropriate spawning substrate. When a shoreline is hardened, erosion down to an underlying coarse cobble lag (i.e., a deposit of coarse cobbles [1–5 inches in diameter] left over from erosion in the geologic past) is possible (Finlayson 2006; Herrera 2005). The result is a nearly vertical shoreline at the location of the hardened structure, often colocated with or near the ordinary high water mark (OHWM).

Several studies have looked at the way fish interact with armored shorelines:

- Heiser and Finn (1970) found that juvenile salmonids avoided deep-water areas along bulkheaded shorelines.
- In marine areas, altered water depths along the marine shallow-water migration corridor preclude their use by migrating juvenile salmon (Thom and Shreffler 1994).
- Knudsen and Dilley (1987) found that abundance of juvenile salmonids was reduced by bank reinforcement activities due to a loss of structural diversity and that these reductions were correlated with the severity of habitat alteration, the size of the stream, and the size of the fish.
- Jennings et al. (1999) found that even within armoring types, those that provided higher structural diversity exhibited higher species richness.
- Many freshwater studies have documented that fish species richness and abundance are negatively correlated with bulkheads in general (see review by Kahler et al. 2000).
- Lange (1999) found that the presence of bank bulkheads was negatively correlated to fish abundance and species richness in Lake Simcoe, Ontario.
- Juvenile fall Chinook in the Columbia and Snake rivers were found to avoid riprap shorelines (Key et al. 1994; Garland and Tiffan 1999).

Newcomb’s littorine snail requires a narrow band of nearshore intertidal habitat that contains certain marsh plant species (pickleweed, *Salicornia virginica*; Larsen et al. 1995), and inadequate water depth or velocity conditions caused by bank armoring that threaten this plant would also impact the snail. Other potentially covered invertebrates are typically found in deeper-water areas (snails) or in mudflat habitats (oysters), where access to habitat would not be limited by bank protection structures.

7.3.5.5 Dams

Dams affect HCP species through

- Altered longitudinal connectivity

- Altered river-floodplain interactions
- Altered groundwater-surface water interactions
- Altered community composition.

Upstream–downstream connectivity is the most obvious alteration, with a reduction in sediment and food resources, increased potential for predation, and altered temperatures. Because flow variations are dampened by dams, connectivity of the river with its floodplain is reduced. This can result in reduced nutrient cycling, loss of favorable habitat, increased predation, and altered temperatures. Altered hyporheic exchange has similar effects with changes in temperature, nutrient cycling, and loss of refugia for invertebrates. Furthermore, these alterations to ecosystems contribute to changes in species interactions and diversity.

7.3.5.5.1 Altered Longitudinal Connectivity

Dams block the migration of aquatic species, which is one of their most detrimental impacts on riverine ecosystems. Longitudinal connectivity is vital to ecosystem health, as organisms rely on up–down transport in riverine systems in search of optimal conditions for feeding, cover from predators, spawning, and food resources. Depending on the size of the dam, upstream migrations of fish can be completely blocked. In other cases, fish passage structures may allow some individuals to migrate upstream. Although fish passage facilities are constructed at some dams to allow longitudinal migration, delays in passage are common. Indirect effects include increased exposure to predation.

Studies documenting the effects of dams on fish and shellfish populations include:

- Studies have shown that Atlantic salmon (Chanseau et al. 1999) and shrimp (Greathouse et al. 2006) have difficulties migrating over dams and weirs.
- In the Fox River, Illinois, freshwater mussel distribution was limited upstream of a dam site (Tiemann et al. 2007).
- In the Pacific Northwest, salmon, sturgeon, bull trout, and lamprey populations have suffered declines from decreased migration in the presence of dams (Hicks et al. 1991; Jager 2006; Moser et al. 2002; Neraas and Spruell 2001).
- Cumulative effects of dams have been documented as well. In the Columbia River, only 3 percent of tagged Pacific lampreys were able to reach the upper part of the river after passing over three dams (Moser et al. 2002). With each dam passed, the number of successful individuals decreased. In addition, lamprey were observed attempting passage multiple times (Moser et al. 2002).
- Migration delays can result in increased risk of predation or poaching while fish wait for an opportunity to pass a given structure (Bednarek 2001).

- Some fish undergo physiological changes if spawning migrations are delayed. American shad have been known to reabsorb their gonads when returning to the ocean, without releasing eggs and sperm (Dadswell 1996).

- Anadromous salmonids, which begin to decline physically upon entry to fresh water, may experience decreased spawning fitness as a result of delayed passage, including potential mortality before spawning is completed (Caudill et al. 2007).

- Delayed migration of adult Chinook on the Columbia River has been documented at the Bonneville Dam since the 1950s (Schoning and Johnson 1956).

- Juvenile migration downstream can be delayed as free-flowing reaches are transformed into slow-moving/slackwater reservoirs (Peven 1987).

- Caudill et al. (2007) examined the relationship between delayed migration, survival, and spawning productivity by using radiotelemetry to track the behavior and fate of Chinook salmon and steelhead navigating fish passage structures on Columbia River dams. Statistically correcting for other sources of mortality, they found a distinct inverse relationship between the time required for individual fish to transit fish ladders bypassing Columbia and Snake River dams, and survival to reach spawning grounds. While the drivers of this inverse relationship are complex, energy expenditure and stress associated with navigating the structures are primary contributing factors. In combination with other stressors imposed by fish passage structures (e.g., prolonged exposure to elevated water temperatures, increased harvest, and predation pressure), the effects of migration delay induced by fish passage structures appear to be cumulatively significant (Caudill et al. 2007).

- Upstream of a dam, habitat structure is altered from a free-flowing (lotic) habitat to a lake-like (lentic) habitat. In this case, lake-adapted organisms dominate and, in some cases, encourage salmon predation (Wik 1995).

- Downstream of a dam, habitat types (e.g., pools, riffles) can be altered as a result of the effects of changes in flow regime, sediment transport, and substrate composition on channel. After several decades of sediment trapping, the Warche River in Belgium showed significant losses of pool and riffle habitat downstream of a dam (Assani and Petit 2004).

- The effects of restricted access caused by dams and weirs have been broadly implicated in population declines of freshwater fish species from around the world (Northcote 1998).

In addition to their direct effects on fish and shellfish, dams alter LWD dynamics by physically blocking the downstream transport of LWD. Dams reduce LWD inputs through changes in river–floodplain connectivity.

7.3.5.5.2 Altered River–Floodplain Connectivity

Dams (as well as diversions) reduce the occurrence of inundation flows and can alter floodplain ecology, contributing to mortality and poor health of aquatic organisms (Kingsford 2000).

7.3.5.5.3 Altered Community Composition

Dams can result in alterations to natural species composition and diversity.

- Species composition can be altered in the presence of dams as invasive species become more dominant than native species as a result of dramatic changes in the riverine ecosystem (Moyle 1976).

- The losses of native fauna as a result of dams can dramatically alter basal food resources and assemblages of invertebrate competitors and prey (Greathouse et al. 2006).

- Changes in temperature associated with dam impoundments can cause shifts from resident cool water species to warm water species.

- For mussel species, it has been shown that changes in temperature and increased siltation from dams have reduced the number of native species and increased the number of invasive species nationwide (Watters 1999).

- In Japan, Katano et al. (2006) showed that the number of species, total diversity, and biomass of fish were lower above three small-scale dams (4.9–12.8 ft [1.5–3.9 m] in height) compared to downstream reaches. Food webs above dams were simpler relative to downstream communities.

- In New Zealand, species richness of different fish and crayfish were lower in areas upstream of dams (Joy and Death 2001).

- Food web structure and functional feeding groups of macroinvertebrates were changed above and below dams on the Grande River, Argentina (Vallania and Corigliano 2007). The authors found that collector-filterers, scrapers, and predators increased downstream, whereas the collector-gatherers and shredders decreased relative to upstream reaches. A similar result was observed in a series of mountain streams in Spain (Camargo et al. 2005).

- Alterations to species composition and diversity from dams are not limited to freshwater environments. The lack of freshwater inputs from reduced flow regimes can affect species in estuarine and marine environments as well (Drinkwater and Frank 1994).

- On the Olympic Peninsula, the presence of dams on the Elwha River has reduced sediment transport to coastal environments, causing a shift to invasive kelp and barnacles in the receiving water body (DOI 1995). This reduction of sediment contributed to a loss of estuaries that serve as nurseries for fish and shrimp because sediment bars separating brackish water from the ocean are no longer present (DOI 1995).

7.3.5.6 Dikes and levees

The main ecological connection altered in the presence of a dike or levee is the river–floodplain connection. In addition to floodplain habitat loss, changes in hydrology and geomorphology caused by dikes and levees could potentially lead to an increase in the invasion of exotic species. However, little information is available on this topic and represents a potential data gap.

- After construction of a dike or levee, floodplain, lagoon, and tidal marsh habitats are lost (Hood 2004).

- After dikes were removed on the Salmon River (Oregon), juvenile fall Chinook salmon were observed using many regions of the restored tidal marsh (Bottom et al. 2005).

- A loss of floodplain habitat in China from the construction of dikes and levees resulted in a 74 percent reduction in habitat and caused declines in many types of plant species (Liu et al. 2004).

7.3.5.7 Dredging

In marine and estuarine environments, dredging operations can limit the access of fish through changes in both the structure and characteristics of shorelines (e.g., linear and barren versus undulating and covered in large woody debris). This can take the form of complete removal of habitats, such as the elimination of marine lagoon habitats due to changes in alongshore sediment transport patterns or the tidal prism.

In riverine environments, dredging reduces structural complexity, and alters channel and floodplain form. Dredging can change how water, organisms, food resources, sediments and LWD are transported or retained. Dredging in rivers to improve navigation, and in smaller channels to increase conveyance, lowers the bed elevation. Dredging therefore has the potential to decrease lateral connectivity with side-channel, slough, and floodplain ponds, leading to habitat loss and fragmentation.

In lacustrine environments, dredging can alter the wave energy reaching the shoreline and thereby alter the local recruitment and transport of sedimen, fragmenting the nearshore ecosystem. Juvenile Chinook salmon use lacustrine littoral zones (Sergeant and Beauchamp 2006). Sergeant and Beauchamp (2006) have shown that although substrate preferences for juvenile Chinook are weak, they prefer fine substrates with cover. Because the wave energy caused by bathymetric changes due to dredging could change the local shoreline substrate, it possibly also changes both the distribution and continuity of spawning and rearing areas, increasing the potential for predation.

7.3.5.8 Gravel mining and scalping

Gravel mining can cause channel incision by the removal of gravel from the sediment budget and from the incorporation of excavation pits into the channel. Hydromodifications of riverine environments by gravel mining could result in a loss or fragmentation of existing habitat through alterations in the longitudinal connectivity and river-floodplain connectivity. Pit capture[1] can also result in the fragmentation and loss of spawning and rearing habitats (Norman et al. 1998). Levees constructed to isolate pits from the active channel reduce habitat complexity and dynamic channel migration.

Captured pits become lakes within the river and transform lotic environments into lentic environments. The incorporation of off-channel pits by river capture exposes juvenile salmonids to heavy predation by exotic, warm-water fish. For example, McMichael et al. (1999) showed that predation on juvenile salmonids by predaceous warm-water fishes in the Lower Yakima River is substantial. Smallmouth bass were estimated to consume about 0.5 million salmon smolts per year, resulting in an annual loss of about 1,350 adult salmon.

7.3.5.9 Sediment Capping

Sediment capping projects can limit fish movement and accessibility to various habitats through changes in both the structure of the shoreline and its characteristics. If a Confined Aquatic Disposal (CAD) cell is used, food web alterations may be induced by the attraction of piscivorous fish to the hard habitat of the CAD structure. Sediment capping may also result in alteration of nearshore circulation patterns.

7.3.5.10 Beach Nourishment

The ability of beach nourishment to reconnect or disconnect pre-existing shoreline communities depends on the nature of the shoreline adjacent to the site. If the substrate is significantly different than the shoreline adjacent to it, or if added sediment buries aquatic vegetation, the activity may fragment the alongshore transit of HCP species (Beamer et al. 2005). This is why some researchers have recommended that the substrate used in beach nourishment activities be as close to the adjacent shorelines as possible (Speybroeck et al. 2006). Conflicting recommendations calling for the installation of a coarser substrate have been based primarily on engineering grounds (NRC 2007).

It has been well documented that beach nourishment projects can negatively impact the prevalence of invertebrates on the shoreline (Peterson et al. 2006; Peterson et al. 2000; Rakocinski et al. 1996). Furthermore, it has been shown that these reductions can also impact invertebrate predators (Peterson et al. 2006). However, invertebrate communities respond quickly to disturbance and may, in certain situations, rebound quickly from these impacts (Dernie et al. 2002). The impact of these disruptions of food sources on HCP fish is currently a data gap.

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[1] National Marine Fisheries Service (NMFS) National Gravel Extraction Guidance (05-06-27) defines “pit capture” as “active channel migration into floodplain (gravel) pits.”

7.3.5.11 Reef Creation, Restoration or Enhancement

It is clear that more fish and invertebrates are present at reefs than in open water (Pickering and Whitmarsh 1997). The physical presence of a solid surface enables a greater diversity of organisms to grow and be protected. However, a remaining question from the hundreds of studies of artificial reefs around the world is whether reefs produce more biomass than would exist without the reef or whether they simply concentrate the existing biomass (Pickering and Whitmarsh 1997). In other words, do reefs produce or attract targeted species? Regardless of whether absolute production has increased, it has been suggested that artificial reefs can harbor significant fractions of total fish populations, particularly where natural environments have been declining (Love et al. 2006).

Rockfish are opportunistic piscivores (Yang et al. 2006). Although there have been no studies documenting rockfish preying on juvenile salmon, it is certainly possible given an increase in rockfish numbers near shore. Predation on salmonids by rockfish is currently a data gap and should be considered when siting an artificial reef in marine waters.

As newly available habitat, reefs have a tendency to attract invasive species. Similar hard structures have been the site of invasive communities, both in fresh water and salt water (Marsden and Chotkowski 2001; Wasson et al. 2005). The threat appears to be somewhat correlated to the degree of human alteration of the ecosystem (Wasson et al. 2005). This implies that estuaries and lakes are more prone to invasive species than exposed marine settings. In the case of freshwater invaders, there are species that can disrupt if not eliminate resident fish populations (Marsden and Chotkowski 2001).

Artificial reefs have the capability to connect existing rocky habitats, leveraging the gain from their installation (Thompson et al. 2002).

7.3.5.12 Channel Creation and Alignment

Channel creation and alignment can change how water, organisms, and food resources are transferred through the mosaic of habitat patches that constitute the river-floodplain system. Channel creation and alignment may alter longitudinal connectivity, alter river-floodplain connectivity, and alter hyporheic flow and exchange.

7.3.5.13 Roughened channels

In general, roughened channel projects fall into two categories: reconfiguration of an existing channel or side-channel to provide fish passage, or the creation of an entirely new channel through adjacent uplands or floodplain.

Roughened channels are intended to improve fish passage by promoting connectivity of habitats along the river. However, roughened channels that were improperly designed for their ecologic or geomorphic context, or because of decreased passage performance over time, may result in ecosystem fragmentation through the following mechanisms:

- Roughened channels may impose intentional or unintentional passage barriers, or passability may decrease over time due to design failure or improper maintenance.

- Effects on fish passage may in turn alter the upstream transport of nutrients from distant sources, particularly marine-derived nutrients, affecting ecosystem productivity.

- Unintentional loss of flow can cause organic material to accumulate in the channel bed, only to be released in a large pulse when flow is restored.

- Undersized roughened channels may flood lands never flooded in the past and produce large quantities of fine-grained sediment.

Many studies have identified fish passage structures that fail to provide adequate fish passage for the species they are intended to benefit, or that unintentionally limit passage of nontarget species (Agostinho et al. 2007; Boggs et al. 2004; Bunt et al. 1999; Caudill et al. 2007; Moser et al. 2000; Moser et al. 2002; Naughton et al. 2007)

7.3.5.14 Tide gates

Tide gates may cause ecosystem fragmentation from a loss of longitudinal connectivity. This alteration, combined with altered flow velocities when the gate is open, can block the migration of fishes and invertebrates (Giannico and Souder 2005). Blue crab showed increased difficulty navigating high flows through a tide gate in North Carolina (Rulifson and Wall 2006). Two factors may influence the extent that a tide gate blocks fish passage: the length of time the gate is open, and how wide it opens.

Tide gates may:

- cause a reduction in channel–floodplain connectivity as a result of altered channel geometry and flow regime,

- cause a loss of lagoon habitat,

- alter wave energy, current velocities, and nearshore circulation.

- alter species composition and diversity. For example, Danish wetlands subjected to the influence of tide gates experienced declines in bird species diversity, declines in benthivore abundance, and increases in herbivore abundance (Holm and Clausen 2006). In addition, macrophyte biomass increased, but sea grass diversity decreased. The authors attributed many of these changes in plant communities to altered salinity levels.

- contribute to the loss of large woody debris. Little information is available on LWD dynamics as they are affected by tide gates and represents a potential data gap.

- Gated culverts (tide gates) are also known to restrict fish passage to lateral riverine and estuarine habitats, even when designed according to current engineering standards (Novak and Goodell 2006). Gated culverts that are designed or retrofitted to promote fish passage (e.g., self-regulating tide gates) are likely to cause at least some degree of barrier condition (Novak and Goodell 2006). Because this type of culvert is typically found in estuarine environments, gated culverts have the potential to affect HCP species that use estuarine floodplain habitats, such as juvenile anadromous salmonids.

The degree of these impacts would depend on the volume of the tide gate discharge, magnitude of tidal changes, and local mixing.

7.3.5.15 Fish Passage Structures – General

All fish passage structures, with the exception of culvert removal, have some potential to impose barriers to fish passage when compared to the aquatic ecosystem in its natural condition (i.e., before the presence of man-made passage barriers). Providing passage for all HCP species during all pertinent life-history stages is a guiding principle governing the design of fish passage structures, and it is a difficult challenge.

Fish passage structures may provide less effective passage over time depending on how appropriate the design is for its ecological context and how frequently the structure is maintained.

Upstream migration and other movements within freshwater rearing habitats are important factors to consider when designing structures that will allow fish passage. In comparison to the natural stream baseline, fish passage projects may lead to detrimental effects on native fish populations by affecting their ability to migrate between important habitats. Juvenile salmonids are seasonally migratory, moving between refuge and rearing habitats (Bolton et al. 2002; Kahler and Quinn 1998; Kahler et al. 2001). Juveniles may cover considerable distances to occupy available rearing habitats, indicating that this dispersal mechanism is important to survival (Bolton et al. 2002). Even in the absence of well-defined migratory behavior, the ability to move between different habitat types is nonetheless important for many resident fish species (Rodriguez 2002). The ecological implications of decreased habitat access are potentially significant. Fish passage structures that unintentionally block access to summer and winter rearing habitats may be key factors limiting juvenile survival, growth, and fitness.

Even when a passage barrier project is intended to protect native fish populations, it could have detrimental effects. For example, projects that prevent brook trout invasions of headwater stream populations may unintentionally fragment genetic exchange between resident and adfluvial populations of bull trout. Maintaining this type of genetic exchange within bull trout metapopulations is considered essential for the long-term conservation of the species (Reiman and McIntyre 1993).

Fish passage structures can affect passage success based on life-history stage (i.e., by size). As juvenile salmonids migrate downstream on many larger river systems, they must travel past large dams where they are susceptible to injury and mortality if forced to travel through power turbines. Many experimental fish passage structures have been employed to direct downstream migrations through less injurious pathways, with varying degrees of success.

The interplay between fish swimming performance and hydraulic conditions in fish passage structures becomes increasingly complex with smaller target species. In their review of multiple sources of research on juvenile salmon passage, Kahler and Quinn (1998) noted that numerous *in situ* observations have demonstrated that extrapolation of adult salmon swimming performance curves to juveniles underpredicted the ability of smaller fish to navigate velocity barriers in fish passage structures. Based on postulations in the sources they reviewed, they concluded that complex hydraulic conditions within fish passage structures created low-velocity zones used by juvenile fish to transit the structures. Research conducted by WDFW supports this hypothesis. Kahler and Quinn (1998) suggested that design guidance for velocity limits based purely on swimming performance and mean channel hydraulics is likely to be conservative. However, there are many other uncertainties in the design guidance that are not compensated for by these unintentionally conservative assumptions.

Most regional studies on fish passage performance have focused on larger juvenile salmon (e.g., in the 4–5 inch [100–125 millimeter] range), which may not be fully representative of the requirements of smaller juveniles. While the preponderance of research on swimming performance suggests that ability is essentially constant relative to size across the majority of fish species (Katopodis 1992), this relationship may break down when the complexities of low-velocity pathways and individual adaptive ability are considered.

Fish passage structures may unintentionally select for fish of different size classes or run timing. For example, a culvert that is retrofitted for fish passage may provide adequate passage to salmonids during moderate streamflow conditions, but may be impassable during average high and low flows. In effect, the barrier may unintentionally select against individuals with run timing in the late summer low-flow period and during late fall when streamflows increase to high levels, truncating the genetic diversity of the stock. Similarly, a passage structure may prove impassable to fish above or below a certain size, selecting for smaller or larger individuals. Of particular concern, it is increasingly clear that juvenile salmonids are migratory when quite small and that design criteria in existing regulations may not adequately protect juveniles of smaller size. For example, the Washington Administrative Code (WAC 220-110-070) limits the vertical jumps formed by water-crossing structures to no more than 0.8 feet (9.6 inches) to provide for juvenile fish passage. However, research on juvenile salmonid jumping ability indicates that a vertical drop of this size would effectively limit passage of juvenile salmonids 3 inches in length or smaller. Pearson et al. (2005) found that the proportion of juvenile salmon able to navigate vertical jumps decreased steadily as jump height exceeded 2.5 times body length, with effectively no fish able to navigate jumps in excess of three times body length. However, a number of complicating factors must also be considered, such as weir crest shape, ambient approach velocity, shape of the nappe[2], and downstream approach conditions that can affect this threshold limit.

The ability of fish species to navigate fish passage structures may be over- or underestimated if their swimming and jumping abilities are not appropriately considered.

- In a study of the effects of weir vertical drop heights on the migration behavior of two native diadromous fish species in New Zealand, the common bully and the inanga, Baker (2003) found that juvenile inaga and all life-history stages of common bully were unable to navigate vertical drops of 4 inches (10 cm), and drops of 8 inches (20 cm) were barriers to adult inaga.

- Vertical drops of 12 cm have also been shown to limit passage of juvenile salmonids (Pearson et al. 2005).

- Adult Atlantic salmon in the Pau River (France) were able to pass over weirs between 59.1 inches (1.5 m) and 98.4 inches (2.5 m) in height (Chanseau et al. 1999), drop heights that would completely block passage for many fish species.

Fish passage barriers have been shown to differentially affect passage by sex. Brown et al. (2002) studied migratory Chinook salmon passage of natural waterfalls and found that males were more successful than females at navigating energetically demanding barriers. Man-made fish passage barriers and their remedies could conceivably have similar effects. Alteration of male to female sex ratios can have significant demographic consequences, potentially affecting the viability of the affected population (McElhany et al. 2000; Thompson 1991). However, differential male to female dispersal around barriers may be an evolved strategy to avoid inbreeding and genetic introgression in salmonids (Hutchings and Gerber 2002). This suggests that sexual selection effects should not be assumed without population and site-specific research.

Improperly conceived or maintained fish passage structures may delay migration and/or may be physically taxing to navigate. In the case of adult salmonids, the cost of delayed migration and energy expenditure can have a demonstrable effect on survival, as well as spawning productivity. Caudill et al. (2007) examined the relationship between delayed migration, survival, and spawning productivity by using radiotelemetry to track the behavior and fate of Chinook salmon and steelhead navigating fish passage structures. Statistically correcting for other sources of mortality, they found a distinct inverse relationship between the time required for individual fish to transit fish ladders bypassing

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[2] Nappe – n. A sheet of water flowing over a dam or similar structure. From http://www.answers.com/topic/nappe. Also a geological formation, a section of a cone (in mathematics), and a French culinary term for lightly coating food in a sauce.

Columbia and Snake River dams, and survival to reach spawning grounds. While the drivers of this inverse relationship are complex, energy expenditure and stress associated with navigating the structures are primary contributing factors. (However, extant natural barriers eliminated by impoundments are also energetically demanding (Brown et al. 2002)). In combination with other stressors imposed by fish passage structures (e.g., prolonged exposure to elevated water temperatures, increased harvest, and predation pressure), the effects of migration delay induced by fish passage structures appear to be cumulatively significant (Caudill et al. 2007).

Fish passage structures that primarily consider the passage of one species or class of species (e.g., culvert retrofits designed to pass salmonids) may unintentionally limit the passage of other important species. Species selection can alter species composition and community relationships upstream of the passage barrier, with important implications for conservation of individual species and biodiversity (Agostinho et al. 2007).

There are several examples of species-specific selectivity in the available literature.

- In the Columbia River system, fish ladders designed primarily to provide salmonid passage around mainstem dams perform relatively poorly for passage of Pacific lamprey (Moser et al. 2002). Loss of habitat access is a key factor implicated in the decline of native Pacific and river lamprey populations in the Pacific Northwest.

- Sturgeon passage may be enhanced by the inclusion of rapid velocity segments in fishways and other passage structures (Cheong et al. 2006; Webber et al. 2007), but these conditions may impede passage of other species with different swimming abilities.

- Fishways and fish ladders around dams in biodiverse tropical rivers show a strong tendency toward species selection, allowing passage of certain migratory species while truncating the distribution of relatively weak swimming species that nonetheless depend on seasonal dispersal mechanisms (Agostinho et al. 2007).

7.3.5.15.1 Culverts

Culverts can induce ecosystem fragmentation through effects on fish passage; the interruption of natural channel migration processes and channel configuration; altered transport of sediments, organic material, and large woody debris; and in some cases the creation of new habitat types that are inconsistent with the natural channel form.

Alterations to existing culverts fall into three categories:

> “Culvert removal” means complete removal of the culvert in conjunction with decommissioning of the roadway or flow control structure, or replacement of the culvert with a bridge.

> "Culvert replacement" means taking a culvert out of a stream, and replacing it with a design that accomodates fish passage.
>
> "Culvert retrofit" means modifying an existing culvert with baffles, internal weirs, or similar structural elements to enhance fish passage.

Addressing passage barriers at culverts to restore access to fragmented habitat is recognized as a priority issue in salmon habitat restoration strategy (Roni et al. 2002). From the perspective of existing conditions, improving fish passage at culverts must generally be viewed as having beneficial effects on ecosystem fragmentation. When comparing the effects of culverts to a natural stream baseline, only culvert removal projects have little to no potential to produce adverse ecosystem fragmentation effects. Even the most carefully designed culvert replacement or retrofitting project can produce some forms of ecosystem fragmentation.

- The culvert may not provide full fish passage, unintentionally imposing specific types of barriers.

- Fish passage may decrease over time due to design limitations.

- Improperly maintained culverts may become blocked by debris.

- Headcuts[3] induced or reinitiated by culvert removal or replacement can result in changes in channel geometry, such as incision, that affect habitat complexity. Headcut migration may cause channel incision and lateral and longitudinal habitat fragmentation. Such changes in channel morphology can impede fish passage.

- The culvert may not allow for the unrestricted downstream transport of woody debris and organic material, affecting habitat complexity and food web productivity in downstream reaches. Generally, this effect is associated with retrofitted culverts. Replacement culverts are generally less likely to produce ecosystem fragmentation effects because current design guidance favors approaches that mimic natural stream hydraulics ("stream simulation.")

- Headcut liberation and channel incision can also influence the recruitment, transport, and retention of sediment and LWD, leading to longer term impacts on habitat complexity. These impacts include altered storage and retention of sediment and particulate organic matter, as well as hydraulic simplification (which reduces the diversity of instream habitats available for fish). Channel incision may also alter the functional relationship between LWD and the stream channel. For example, spanning LWD that would be functional under natural channel conditions may not interact with surface water under the same range of

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[3] "A headcut, sometimes called a knickpoint, is a vertical or near-vertical drop or change in elevation of a stream channel, rill, or gully." http://www.ars.usda.gov/research/publications/publications.htm?SEQ_NO_115=158547

flow conditions in a degraded channel. This would reduce the influence that woody debris has on habitat conditions in the stream channel.

If a culvert is providing grade control, replacement or removal of the culvert may result in the following intermediate-term responses (Castro 2003):

- Headcut migration upstream and subsequent deepening of the stream channel.
- Relatively higher channel banks that may exceed critical height, resulting in slope failure and bank erosion.
- Addition of sediment to the stream system due to erosion of the channel boundary.
- Disconnection of floodplains from active stream channels.
- Prematurely dewatered or disconnected backwater habitat.
- Locally increased channel slope and loss of pool habitat.
- Drainage of shallow aquifers affecting riparian vegetation.
- Meander cut-offs due to knickpoint migration across a meander neck caused by an increased elevation drop between the old floodplain and active channel bed.
- Deposition of large masses of sediment causing localized channel braiding and instability of the stream banks.

Culvert retrofits are intended specifically to improve fish passage, thereby addressing one component of ecosystem fragmentation. However, retrofitting typically will not reduce the fragmentation of ecological processes induced by the structure, and may exacerbate certain problems. For example, placement of internal weirs and baffles will decrease the hydraulic capacity of the culvert, which can create backwater effects upstream of the structure and lead to sediment deposition. This can interrupt sediment transport processes, leading to some degree of sediment starvation downstream of the structure.

Culvert retrofits are more prone to produce fish passage related stressors than culvert replacement. Successfully retrofitting a culvert for fish passage requires knowledge of the swimming performance and passage requirements of the full range of HCP species likely to attempt to navigate the structure. It may be challenging to produce a design that provides for the needs of all species of interest, meaning that some degree of ecosystem fragmentation may occur. Moreover, retrofitted culverts typically require regular maintenance to maintain function. When they are improperly maintained, fish passage performance is likely to decline significantly.

Studies have shown that flow velocity affects fishes' ability to pass culverts.

- Juvenile salmonids have been shown to respond preferentially to different velocity conditions when traveling downstream through weirs (Kemp et al. 2006), suggesting that structures designed without proper consideration of attraction flows may be ineffective.

- Flow velocity, exposure distance to excess flow velocity without hydraulic refuge areas, and other factors, can influence the ability of juvenile salmonids to effectively pass through fish passage structures (Behlke 1991).

- In a laboratory environment, juvenile fish moving through culverts were observed using low-velocity pathways within turbulent boundary layers and behind baffles. The pathways selected differed between the baffled and unbaffled test environments, and also potentially differed depending on flow rates in the baffled environment (Pearson et al. 2006). In some instances, juvenile fish passed culverts at higher mean velocities in the cross-section than the physiological limits of swimming performance would suggest. This indicates that fish adaptively select low-velocity pathways to navigate through culverts where possible.

- Powers and Bates (1997) studied the passage of juvenile coho salmon through culverts composed of several different materials and found significant differences in the velocities that permitted passage. Specifically, they noted that the turbulent boundary layer created by corrugated metal pipes appeared to create a lower velocity zone that enhanced passage relative to culvert pipes with smooth interiors.

- The presence of lower velocity zones within culverts has been confirmed by subsequent research (Ead et al. 2000; Pearson et al. 2006). Pearson et al. (2006) confirmed that simple sized-based extrapolation of swimming performance underpredicted the ability of juvenile coho salmon to pass culverts. A key factor appeared to be the ability of smaller fish to utilize lower velocity zones in the hydraulically complex boundary layer at the culvert surface. They recommended additional studies to confirm whether the design and discharge parameters that provided successful passage for larger individuals are applicable to smaller juveniles (e.g., 1.1–.6 inches [30–40 millimeters]) that migrate during spring flow conditions. It is important to note, however, that this work was conducted in a laboratory setting. It is not clear how any information gleaned from these studies would apply to a range of culvert designs and settings in the field.

In certain cases, culvert replacement or removal can eliminate or alter distinct wetlands formed by road-impounded wetlands. For example, the Olympic mudminnow is restricted to slack-water habitats in slow-moving streams, ponds, and wetlands with several centimeters of soft sediment substrate (Mongillo and Hallock 1999). Culvert removal could alter flow, channel geometry, and substrate conditions, reverting the impounded reach to a coarse-grained, free flowing condition (Naiman et al. 1988). This would eliminate habitat for this species. In such special circumstances, it may be desirable to design a culvert replacement that maintains the impoundment habitat.

Culverts have been shown to affect passage of nonsalmonid species. Warren and Pardew (1998) evaluated the migration of 21 warm water fish species including centrarchids (sunfish), cyprinids (minnows and suckers), and fundulids (killifishes) through four types of culverts in western Arkansas streams. They found that regardless of design, culverts restricted the movement of each species studied by at least an order of magnitude relative to natural conditions. This suggests that even carefully designed fish passage structures have the potential to impose passage barriers on HCP fish species when the knowledge of their specific passage requirements is limited.

In addition to effects on aquatic species, roads and flow control structures also have the potential to fragment foraging, migratory, and dispersal corridors used by nontarget aquatic, semi-aquatic, and terrestrial species. Researchers in the United States and Europe are currently investigating the utility of different culvert designs to provide migratory corridors for terrestrial species (Bates et al. 2008). It is conceivable that ecosystem fragmentation effects imposed by culverts on terrestrial species could result in indirect effects on HCP species. However, insufficient research has been conducted on this subject to assess the nature and extent of any indirect effects that may occur.

7.3.5.15.2 Fish ladders/fishways

Fishways can result in ecosystem fragmentation through a number of pathways. Because these structures are intended to improve passage, an inherent objective of their design is to reduce ecosystem fragmentation by promoting connectivity of habitats along the river continuum. Improved fish passage promotes additional benefits from ecological connectivity.

Fishways may unintentionally result in ecosystem fragmentation by imposing intentional or unintentional passage barriers. Passability may decrease over time due to design failure or improper maintenance. Alterations to fish passage may in turn alter the upstream transport of nutrients from distant sources, particularly marine-derived nutrients (e.g., in the form of salmon carcasses), affecting ecosystem productivity.

Many studies have identified structures that fail to provide adequate fish passage for the species they are intended to benefit, or that unintentionally limit passage of nontarget species (Agostinho et al. 2007; Boggs et al. 2004; Bunt et al. 1999; Caudill et al. 2007; Moser et al. 2000; Moser et al. 2002; Naughton et al. 2007). In many cases, fishways have been designed and installed without a definition of specific performance objectives, so their performance is difficult to rate (Cada and Sale 1993).

A fishway may initially function appropriately but may lose effectiveness over time if the structure becomes compromised by changing channel conditions or improper maintenance. As an example, the fishway around the hatchery weir at the Quilcene

National Fish Hatchery (Big Quilcene River, Washington State) has become increasingly less effective at passing winter steelhead due to the combination of cobble aggradation and high-flow velocity at the upstream exit of the weir, which redirects fish along a path that results in a high rate of “fallback” (fallback occurs when fishway exits are located in areas where high current velocities wash fish back over the barrier the passage structure is intended to bypass).

Structure design may also lead to unnecessary stress and energy expenditure by fish. For example, high “fallback” rates have been observed at fishways at Columbia River dams. When passage over eight dams was considered, 15 to 22 percent of adult Chinook and 21 percent of adult steelhead fell back over at least one dam (Boggs et al. 2004). Available evidence suggests that the migration delay and energetic costs imposed by fallback could lead to decreased survival and spawning success (Caudill et al. 2007). These structures can be redesigned to decrease the likelihood of fallback.

7.3.5.15.3 Weirs

Fish passage weirs may unintentionally result in ecosystem fragmentation by imposing intentional or unintentional passage barriers. Passability may decrease over time due to design failure or improper maintenance. Alterations to fish passage may in turn alter the upstream transport of nutrients from distant sources, particularly marine-derived nutrients (e.g., in the form of salmon carcasses), affecting ecosystem productivity. Weirs may create impoundment conditions that may alter the downstream transport of woody debris and organic material, affecting habitat complexity and food web productivity in downstream reaches.

Temporary weirs are, by design, limited in effects. These structures are typically semipermiable to water, wood, sediment, and organic debris transport while in place, and transparent to these processes when not in use. The degree to which these structures alter ecological connectivity is expected to be minor in extent and limited to short-term construction and operational effects.

Permanent barrier weirs are expected to impose a broader suite of potential effects on HCP species.

Weirs and other structures intentionally designed to prevent or limit fish access have been broadly employed to restrict exotic species invasions. Competition for forage and habitat and genetic introgression and hybridization with non-native species have been demonstrated to adversely affect native salmonid populations (Reiman et al. 2006; Shephard et al. 2002; Utter 2001).

- Man-made barrier structures have been installed in many tributaries in the Laurentian Great Lakes to limit the distribution of sea lamprey (McLaughlin 2006), with the understanding that these barriers may unintentionally limit migration and dispersal of native fish species and invertebrates such as freshwater mussels that rely on fish for part of their life cycle.

- In the Western U.S., barrier weirs have been successfully employed to prevent habitat recolonization by brook trout following their eradication, supporting recovery of depressed westslope cutthroat and bull trout populations (Shephard et al. 2002).

Longitudinal connectivity is the most documented type of ecosystem fragmentation attributed to weirs in the literature.

- Weirs result in delayed migrations of fish that need to navigate over them (Chanseau et al. 1999).
- In Denmark, Atlantic salmon and brown trout losses increased due to delayed migrations and increased predation while these fish were trying to negotiate weirs (Aarestrup and Koed 2003).
- A similar result was observed with brown trout in the Bidasoa River in Spain (Gosset et al. 2006).
- In Australia, radio-tagged fish were removed from a river after passing over (and under) weirs and were placed back downstream of the structures. When faced with passing the weir a second time, few of the fish did, with most trying to avoid it altogether (O'Connor et al. 2006).
- Some studies have shown that restricted upstream movements are dependant on fish size, with larger fish able to pass small weirs more easily than smaller fish (Baker 2003; Winter and Van Densen 2001).

7.3.5.16 Fish Screens

Fish screens are intended to block the movement of fish and other organisms out of their habitat with water withdrawn from the system. (Water withdrawal itself is regulated by Ecology and is not subject to approval under the HPA program.) Fish screens are intended to prevent fish entrainment and loss caused by unscreened intakes and diversions associated with flow control structures such as dams and weirs. In this sense, they impose a passage barrier that should be considered beneficial. However, due to design limitations or improper maintenance and operations, fish screens can produce unintended adverse effects on fish passage. Fish screens can delay migration, exposing fish to increased stress and predation-related mortality; or may impose timing- or size-specific selection pressures on affected fish species.

Fish screens may impose unintended selection pressures by providing effective downstream passage of juveniles during only part of the downstream migration (Kiefer and Lockhart 1995). Kemp et al. (2006) found that flow velocity and depth strongly influenced the behavior of juvenile fish entering bypass systems at Snake River hydroelectric facilities. While this study did not explicitly evaluate the effects of screens, it nonetheless demonstrates the sensitivity of certain HCP species to parameters that are important in screen design. The potential for these parameters to influence species-specific screen performance is also a concern. Sweeping flows[4] that function well for salmonids may not be suitable for other fish species such as dace (Cyprinidae) or lamprey (Close et al. 1998).

To the extent that they affect upstream fish passage, fish screens may also have the unintended effect of restricting the dispersal of HCP freshwater invertebrate species. This can occur in two ways. First, the structure may restrict the distribution of host fish, affecting the dispersal of parasitic larvae (Vaughan 2002; Watters 1996). Second, attraction flows may draw mussels and snails capable of crawling along the stream bottom to bypass channel outlets that restrict their further upstream movement. For example, certain freshwater mussel species are known to move at least some distance upstream, using their muscular foot and byssal threads (Vaughan 2002). Species such as California floater mussels may face demographic risks from the unintentional limitations on distribution because the effects of fish screens on their host-fish species (minnows and other cyprinids) are less well understood.

7.3.5.16.1 In-Channel Fish Screens

Many HCP fish species, such as herring, rockfish, pollock, cod, eulachon and longfin smelt have planktonic larvae that are dependent on wave and current patterns for transport to and/or retention in productive rearing areas such as estuaries. Highly fecund species that produce spatially variable planktonic spawn rely on current-driven transport and retention mechanisms for reproductive productivity (Hernandez-Miranda et al. 2003; Rooper et al. 2006; Sinclair 1992).

In-channel intake systems employing bankline screens in sheltered alcoves or embayments (in riverine, marine, or lacustrine environments) commonly employ pumped bypass systems because there is insufficient hydraulic head to operate a gravity-driven bypass. Planktonic eggs and larvae, or weak swimming or behaviorally driven fish species, may be drawn into fish screens by the inflow and become trapped (Bates 2008). Screen systems may bypass planktonic or weak-swimming organisms but return them to locations where prevailing currents draw them back into the intake embayment. This is likely to lead to elevated mortality through predation, starvation, unfavorable water quality conditions, or a combination of these effects (Sinclair 1992).

Ecosystem fragmentation occurs when organisms are trapped in embayments by ineffective screening and bypass systems, limiting dispersal to favorable rearing areas. Fish species that migrate along nearshore marine and lacustrine environments, such as juvenile anadromous salmonids, that are drawn repeatedly into bypass systems may experience delayed migration, with attendant effects on survival, growth, and fitness.

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[4] "Sweeping velocity is the water velocity vector component parallel and ajacent to the screen face." National Marine Fisheries Service, Southwest Region. Fish Screening Criteria for Anadromous Salmonids. January 1997.

Bankline and other in-channel screen systems (e.g., Gunderboom[5] screens) attempt to limit this mortality by bypassing organisms back to the aquatic environment.

7.3.5.16.2 Off-Channel Fish Screens

Off-channel fish screens includes some unique design characteristics having their own potential to impose ecosystem fragmentation. Specifically, the additional increment of streamflows required for bypass system operation can modify channel and flow conditions in ways that fragment off-channel habitat. Moreover, bypass systems that discharge into blind side channels may create flow conditions that confuse migratory pathways.

The potential ecosystem fragmentation impact submechanisms potentially imposed by off-channel fish screens include the following:

- Passage and dispersal barriers: Bypass channel flows may attract upstream migrants, causing an unintentional migration delay. Sweeping flows in diversion channels may not be sufficient to draw downstream migrants into the bypass system, leading to unintentional delays in downstream migration or dispersal.

- Modified downstream transport of woody debris and organic material: Woody debris and organic material cleared from screen surfaces may not be returned to the aquatic ecosystem.

- Altered lateral habitat connectivity: Decreased flows within the bypassed reach may alter the connectivity to and availability of side-channel and off-channel habitats under lower flow conditions.

In certain circumstances, vegetation encroachment induced by bypass system operation may result in changes in channel form that can in turn fragment lateral habitat connectivity. Bypass systems that increase flow into existing natural side channels, or effectively create an artificial off-channel environment, can mitigate effects on habitat fragmentation in some cases. In highly hydromodified environments, this effect could be beneficial, increasing available habitat area and complexity.

While these effects are more commonly associated with structures that impose barriers to fish passage, fish screen structures may alter the downstream transport of wood and organic materials when measured against the natural stream condition or the environmental baseline. Certain off-channel screen designs, such as traveling belt screens, incorporate debris collection trays that isolate wood and organic material from the stream channel. Other off-channel screens may be prone to debris jams on the screen or in bypass systems that require manual clearing. These materials may be returned to the channel as an operational practice, or may be disposed of upland. In the latter case,

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[5] Examples do not constitute a recommendation from the Washington Department of Fish and Wildlife.

there would be an incremental decrease in the amount of wood and organic material available to downstream reaches.

In Washington State, a concerted effort to design and broadly implement diversion screening requirements started in the 1940s, and major upgrades started in the 1980s (McMichael et al. 2004, Schille personal communication March 2009). This cooperative program has promulgated research-based design guidance and monitoring criteria that are in broad use. While this program has produced fish screens that have undoubtedly reduced entrainment-related losses of anadromous and resident fishes, some of these screens have imposed unintentional barriers to fish passage (Carter et al. 2003; McMichael and Chamness 2001; Vucelick and McMichael 2003; Vucelick et al. 2004). These barriers can take the form of physical conditions that delay upstream or downstream migration, potentially coupled with conditions that increase predation risk, or that impede migration entirely during certain flow rates.

In the case of juvenile salmonids, downstream migration delays can occur when improperly designed screens may fail to provide sweeping flows adequate to draw fish into the bypass channel. For adult fish, upstream migration delay can be caused by false attraction to bypass outfalls or by locating the bypass discharge point in proximity to the diversion intake, causing fish disoriented by exiting the bypass system to enter and fall back through the bypass.

Migration delays and nonlethal stressors may increase predation exposure, resulting in increased mortality rates. For example, shear stresses associated with passage through dam bypass channels have been associated with temporary disorientation that leads to increased mortality rates (Cada et al. 1999; Mesa 1994). While it is unclear whether stresses occurring in bypass channels reach levels sufficient to increase predation vulnerability (Cada et al. 2003), WDFW guidance cites this potential as an important consideration in bypass channel design, noting that outlets should be located where conditions are unfavorable for predators to loiter (WDFW 2001a). Such steps may help to mitigate predation losses. For example, Mesa and Olson (1993) found that flow velocities in excess of 39–51 inches/second (100–130 centimeters per second [cm/s]) were likely to exceed the sustained swimming speed of predatory northern pikeminnow (referred to as squawfish by the authors), and cited this range of flow rates as useful guidance for locating bypass channel discharge points.

Fish screens can also unintentionally affect passage success based on life-history stage (i.e., by size). Because juvenile salmonids migrate downstream in many river systems, they must travel past numerous diversions with screens of various designs. Off-channel screens must provide sufficient sweeping flows to draw fish into bypass channels without significant migration delays. Suitable sweeping flows may vary by species and by size. For example, juvenile salmonids have been shown to respond preferentially to different velocity conditions when traveling downstream through weirs (Kemp et al. 2006). Attraction velocities must be balanced against other factors such as avoiding impingement while achieving the desired diversion rate. Design guidance focused on achieving this balance for juvenile salmon may be suitable for some other species, such as bull trout (Zydlewski and Johnson 2002), but may or may not provide adequate protection for other fish species with different swimming or biological requirements (Bestgen et al. 2004; Blackley 2004; Close et al. 1998; Moyle and Israel 2005; Peake 2004). A basic premise of screen and bypass design is that fish are actively migrating and seeking a downstream migration path (Bates 2008). This premise may be inappropriate for fish that are passively dispersing rather than migrating.

Upstream migration and other movements within freshwater rearing habitats are also recognized as important factors to consider when designing fish screens. Direct study and review of available research have demonstrated that juvenile salmonids (both anadromous and resident species) are seasonally migratory, moving between refuge and rearing habitats (Bolton et al. 2002; Kahler and Quinn 1998; Kahler et al. 2001). While fish screens are less of a factor than the flow control structures they are typically associated with, certain designs may nonetheless have undesirable effects on upstream passage. Specifically, dedicated bypass channel flows may unintentionally attract upstream migrants into impassable side channels (WDFW 2001a). For example, this may delay dispersal to habitats suitable for summer rearing. Proper design may avoid this unintended impact.

7.3.5.17 Outfalls

Ecosystem fragmentation would be minimally affected by outfalls. Habitat connectivity will likely remain intact for most outfalls, both submerged and exposed. If an outfall crosses a stream or river such that it interferes with downstream flow, then impacts related to upstream–downstream connectivity will be important. If an outfall is placed along the bank of a stream or river, it could potentially exclude access to side channel and floodplain habitat. When exposed outfalls are located such that they terminate at the riverbank and are located above the stream, their main impact is the result of water chemistry changes. Some studies have shown that species diversity and composition changes can be minimal above and below an outfall (Fries and Bowles 2002; Pillard 1996). These studies suggest that ecosystem fragmentation with respect to altered species composition and diversity is likely minimal.

Marine outfalls may cause ecosystem fragmentation as a result of altered wave energy, current velocities, and nearshore circulation. The degree of these impacts depends on the volume of discharge and local mixing and may result in impacts on some HCP species

7.3.5.18 Intakes and diversions

Intakes and diversions may alter flow, which can influence habitat connectivity and lead to habitat loss. Depending on the size of the diversion, changes in flow may be minimal or significant. Reduced discharges from diversions can reduce floodplain connectivity for those fish and invertebrates using these habitats (Kingsford 2000). Reduction in flow has also been shown to concentrate macroinvertebrates as a result of the reduction in available habitat, leading to increases in insect densities in the system (Dewson et al. 2007). Water diversions have been shown to alter temperature, which has been shown to change macroinvertebrate communities (Miller et al. 2007).

Intakes alter food webs and predator-prey interactions. Alteration of natural food webs could change species composition which, in turn, may allow the invasion of exotic species. Intakes and diversion can remove important resources from the aquatic ecosystem when they are entrained in the water column. Predator-prey relationships are altered when drifting insects and larvae are entrained in intake waters, effectively removing food resources from downstream organisms.

- Benstead et al. (1999) showed that entrainment of freshwater shrimps in Puerto Rico can vary from 34–62 percent of drifting larvae based on field data and a flow model using 30 years of discharge data.

- In Hawaii, McIntosh et al. (2002) studied the impacts of diversions on riffle macroinvertebrate communities. The authors collected larval populations upstream and downstream of diversions and showed that total density decreased by 54 percent, thereby affecting trophic interactions downstream.

7.3.5.19 Pipelines and Cables

Love and York (2005) examined pipelines serving oil wells in the Santa Barbara Channel in California. Along the pipeline, the diversity and number of both fish and invertebrates were much higher than on the adjacent seafloor. The hard substrate of the pipeline was colonized by numerous sessile and motile invertebrates, and fish densities were six to seven times greater than on the adjacent seafloor. Rockfish comprised 84 percent of the fish species and represented 22 species. Most were juveniles or represented species that are small at maturity.

Kogan et al. (2003) examined a submarine cable running from Half Moon Bay to Pioneer Seamount in California, a distance of 59 miles (95 kilometers). In the seven years the cable had been in operation, most of the cable had been buried by sediments along the continental shelf, where water depths range up to approximately 390 feet (120 meters [m]), except over areas of rocky substrate. Anemones, echinoderms, and sponges had colonized the cable and were conspicuous in areas where the cable provided the only available hard substrate. In three of nine survey locations, flatfish and rockfish congregated near the cable. The cable had no measurable effect on benthic infauna.

7.3.5.20 Beaver Dam Removal

The draining of beaver dam impoundments represents a significant modification of the aquatic environment which fragments ecological connectivity in the longitudinal, lateral, and vertical dimensions. These impact mechanisms have been demonstrated to produce a number of ecological stressors with the potential to impose a risk of take on HCP species.

7.3.5.20.1 Altered Longitudinal Connectivity

On initial consideration, breaching of beaver dams may appear to improve longitudinal connectivity in riverine systems. Beaver dams represent a potential barrier to fish passage as well as a zone of hydraulic complexity which sequesters sediment, wood, organic material, and water. However, beaver dams are typically semipermiable and do not pose total barriers to fish passage. Early research suggested that beaver dams may serve as barriers that are detrimental to resident species (Pollock et al. 2003). However, more recent research has indicated that longitudinal connectivity is only seasonally altered and that beaver activity is an overall net benefit to aquatic organisms (Pollock et al. 2003; Rolauffs et al. 2001; Sigourney et al. 2006). Beaver dams are usually built within free-flowing reaches so the presence of the dam itself creates pool habitat which is utilized by many of the HCP species, notably juvenile coho salmon (Pollock et al. 2004). The juveniles are present in beaver impoundments in part because the low energy environment of the impoundment provides refuge and foraging habitat (Rolauffs et al. 2001). Although beaver dams do create a change in head at the dam face which could impede fish passage, dam structures are complex and relatively permeable under a range of flow conditions (Naiman et al. 1986). Consequently, beaver dams do not present a barrier to passage comparable to man-made dams.

As a natural feature of the landscape, the hydraulic and structural complexity provided by beaver dams supports a broad array of species during different stages of their life history, including HCP species. The distribution of these features along a longitudinal gradient in riverine ecosystems is an important measure of ecological connectivity, particularly for species such as coho salmon that prefer slow water habitats like beaver ponds for rearing habitat. Altering the longitudinal connectivity of complex, diverse habitats in a riverine environment by draining beaver ponds represents a form of ecosystem fragmentation. Beaver dam removal or modification may result in increased longitudinal connectivity, but at the cost of reduced availability of suitable habitats for those species dependent on the impounded environment.

Beavers alter their environments by modifying riparian vegetation and the channel base level. These alterations lead to an increase in aquatic habitat patches (Johnston and Naiman 1990).

Much of the research concerning ecosystem fragmentation and impoundments has focused on the impacts of dam presence and not dam removal. The creation of a reservoir through damming turns the impounded section of the river into a slow-moving, lentic habitat and alters the species composition of the river (Grubbs and Taylor 2004). Lake-adapted species may begin to flourish and riverine biota may become more susceptible to displacement. For example, in the Snake River in Washington, the combination of a series of 4 large reservoirs has contributed to the increase of salmonid predator densities (Wik 1995). This same phenomenon has been noted in beaver ponds, where introduced predatory species have been shown to flourish (Pollock et al. 2003). Despite this, high salmonid abundance has been noted in beaver impoundments (Leidholtbruner et al. 1992; Pollock et al. 2003; Pollock et al. 2004; Ray et al. 2004; Sigourney et al. 2006). With the removal of a beaver dam, lentic habitat will be converted to lotic habitat and a corresponding shift in the community structure will ensue.

Although beaver impoundments have been shown to be home to over 80 North American fish species (Pollock et al. 2003), studies have shown that beaver dams can function as seasonal barriers to fish movement (Murphy et al. 1989; Schlosser 1995).

7.3.5.20.2 Altered Terrestrial and River-Floodplain Connectivity

There is unanimous agreement that beaver impoundments increase lateral terrestrial-aquatic connectivity. Beaver dams raise the water surface elevation within the channel and inundate adjacent floodplain habitat (Westbrook et al. 2006). Lateral connectivity of the channel with its floodplain is increased upstream and downstream of beaver dams (Westbrook et al. 2006). Beaver ponds create and also increase access to floodplain habitat. The removal of beaver impoundments will sever this connection and reduce the quality of habitat for many of the HCP species. Due to abundant food resources and habitat which is protected from the high velocities associated with flooding events, the inundated perimeters of beaver ponds are ideal habitat for many of the HCP species (Pollock et al. 2003).

Shallow flooded riparian areas are some of the most productive habitat within a watershed (Bunn et al. 2003; Junk et al. 1989; Schemel et al. 2004; Sommer et al. 2005; Sommer et al. 2001) and, consequently, beaver dam construction increases stream (Naiman et al. 1994) and riparian (Duke et al. 2007) productivity. Beaver impoundments flood adjacent riparian vegetation and create a hydraulic conduit for terrestrial organic matter to enter the channel. This additional carbon input combined with high retention caused by the structure of the dam and increased solar input from canopy loss, are the drivers of elevated productivity in beaver impoundments. With the removal of a beaver dam, the organic matter is exported and the terrestrial-aquatic linkage is weakened. This will lead to reduced food resources in the waterway and an associated impact on HCP species.

The draining of beaver dam impoundments eliminates open water habitats and causes the channel system to withdraw from riparian and floodplain areas. Depending on where the stream channel stabilizes in the impoundment area, riparian habitats may be separated from the channel by open ground. This effect fragments the channel from floodplain habitats, reducing the connectivity between terrestrial and aquatic habitats. The reduced availability of these productive habitats may limit survival, growth, and fitness of those species that utilize the affected riverine habitats. An additional effect of is the vulnerability of disturbed habitats to invasion by exotic plant species. Exposed impoundment beds are likely sites for colonization by invasive species. Once these species become established, they may create a barrier to riparian recovery and a dispersal source for additional colonization. Invasive species may reduce the suitability of floodplain and riparian habitat for refuge, food production, and other ecological functions. These effects would also be considered likely to limit the survival, growth, and fitness of species that utilize the affected riverine habitats.

Table 7-4 [adapted from (Pollock et al. 2003)] shows the relative abundance of several fish species that may occur in beaver impoundments in Washington.

Table 7-4. Fish species known to utilize Washington beaver pond habitat.

Species	Common Name	Abundance	Comment
Oncorhynchus clarkii	Cutthroat trout	C	HCP species
Oncorhynchus kisutch	Coho salmon	C	HCP species
Oncorhynchus nerka	Sockeye salmon	C	HCP species
Salvelinus malma	Dolly Varden char	C	HCP species
Oncorhynchus mykiss	Steelhead	U	HCP species
Oncorhynchus mykiss	Rainbow trout	U	HCP species
Oncorhynchus tshawytscha	Chinook salmon	U	HCP species
Oncorhynchus gorbuscha	Chum salmon	O	HCP species
Cottus spp.	Sculpins	LC	
Gasterosteus aculeatus	Threespine stickleback	LC	
Prosopium spp.	Whitefish	LC	
Lota lota	Burbot	C	
Prosopium williamsoni	Mountain whitefish	U	
Catostomus commersoni	White sucker	C	
Phoxinus eos	Northern redbelly dace	C	
Culaea inconstans	Brook stickleback	C	
Hybognathus hankinsoni	Brassy minnow	C	
Phoxinus neogaeus	Finescale dace	C	
Pungitius pungitius	Ninespine stickleback	C	
Notemigonus crysoleucas	Golden shiner	LC	
Margariscus margarita	Pearl dace	O	
Cyprinella lutrensis	Red shiner	U	Not native
Esox americanus	Redfin pickerel	C	Not native
Esox niger	Chain pickerel	C	Not native
Ameiurus melas	Black bullhead	C	Not native
Lepomis auritus	Redbreast sunfish	C	Not native
Lepomis cyanellus	Green sunfish	C	Not native
Lepomis gibbosus	Pumpkinseed sunfish	C	Not native
Lepomis gulosus	Warmouth	C	Not native
Lepomis marginatus	Dollar sunfish	C	Not native
Lepomis punctatus	Spotted sunfish	C	Not native
Phoxinus phoxinus	Minnow	C	Not native
Pimephales promelas	Fathead minnow	C	Not native
Salmo trutta	Brown trout	C	Not native
Salvelinus fontanilus	Brook trout	C	Not native
Esox lucius	Northern pike	LC	Not native
Micropterus salmoides	Largemouth bass	LC	Not native
Ameiurus natalis	Yellow bullhead	U	Not native
Ameiurus platycephalus	Flat bullhead	U	Not native
Ictalurus nebulosus	Brown bullhead	U	Not native
Lepomis humilis	Orangespotted sunfish	U	Not native
Lepomis macrochirus	Bluegill	O	Not native

Source: Pollock et al. 2003.
C = common, LC = locally common, U = uncommon, O = occasional.

7.3.5.20.3 Altered Fish and Invertebrate Habitat

The primary impact on fishes associated with beaver dam removal is loss of foraging, refuge, and rearing habitat. Secondary impacts result from elevated suspended sediments and related stressor conditions during and subsequent to dam removal. These ephemeral impacts are short-term to intermediate-term in duration, and the resulting effects on aquatic species (including HCP species) are predominantly due to changes in the availability of suitable habitat and foraging opportunities, and behavioral responses to stressor exposure (Thomson et al. 2005). Increased nutrient and pollutant export associated with a single beaver dam removal will most likely not affect fishes, but the cumulative impact of channel simplification from the pre-Columbian era until present day has had a profound effect on nutrient and pollutant spiraling in fluvial systems (Wohl 2004), and must be taken into account when assessing potential impacts associated with any channel simplification activity such as beaver dam removal.

No studies have addressed the impact of beaver dam removal on HCP invertebrate species. There have, however, been studies which have addressed macroinvertebrate response to the removal of man-made dams. Research has indicated that dam removal will result in a significant decrease in downstream macroinvertebrate density for at least 1 year following removal (Stanley et al. 2002; Thomson et al. 2005). Given the habitat preferences of the California floater, Western ridged mussel, and the Great Columbia River spire, these species may inhabit beaver impoundments and the reaches immediately downstream (WDNR 2006b). The removal of a beaver impoundment would strand upstream invertebrates and potentially bury downstream individuals. Thus the construction phase and the sediment export following removal would be the primary impact mechanisms for invertebrates.

7.3.5.21 Spawning Substrate Augmentation

Spawning substrate augmentation leads to the creation of suitable spawning habitat. Suitable habitat has a variety of characteristics including desirable depths and velocities, and a high rate of hyporheic exchange. An ancillary benefit of spawning substrate augmentation is that connectivity with adjacent floodplain environments may be increased with an associated increase in bed elevation.

7.3.5.21.1 Altered Groundwater-Surface Water Interactions

The primary goal of gravel augmentation is to create suitable spawning habitat both in terms of depth and velocity as well as hyporheic exchange through the gravels. Hyporheic exchange brings oxygen-rich waters into the gravels which promotes spawner embryo survival (Heywood and Walling 2007). Spawning chum and Chinook salmon have been found to preferentially select spawning sites where water is either upwelling or downwelling (Geist et al. 2002) and where benthic dissolved oxygen is elevated (Geist 2000). Also, bull trout redd distribution and abundance have been found to be influenced by hyporheic and groundwater-surface water exchange (Baxter and Hauer 2000). This indicates that fish select for areas with high hyporheic exchange (Mull 2005). Gravel augmentation has been shown to increase hyporheic exchange (Merz and Setka 2004), and this helps to explain why fish are attracted to augmentation sites (Merz and Setka 2004; Mesick 2002).

An active hyporheos not only provides biogeochemical benefits to fish but also to macroinvertebrates. In a study of a gravel augmentation on the Mokelumne River, California, the standing crop, as measured by macroinvertebrate densities and dry biomass, was significantly higher in enhancement sites 12 weeks after augmentation than in unenhanced sites and remained so over the following 10 weeks (Merz and Chan 2005). Thus, it would appear that augmentation may not only provide spawning habitat but also food resources for emerging fry.

7.3.5.21.2 Altered Lateral Connections Between Rivers and Floodplains

Although the goal of spawning gravel enhancement is to improve benthic habitat, an ancillary benefit may be increased lateral connectivity with adjacent floodplains. Miller and Benda (2000) monitored debris flow sediment wave propagation in the South Fork of Gate Creek, Oregon and found that sediments cause vertical accretion of the valley floor due to increased overbank flooding depositing material across the floodplain. Gravel augmentation will be on a much smaller scale than a debris flow but the same processes may be applicable.

Spawning substrate augmentation may increase allochthonous carbon budgets in restored channels. Floodplain connectivity also creates fish forage and refuge habitat. Gravel augmentation below dams could be combined with flow management to induce periodic floodplain connection.

7.3.5.22 In-channel and off-channel habitat creation or modification

Habitat creation or modification is meant to ameliorate the impacts associated with ecosystem fragmentation.

7.3.5.22.1 Altered Lateral Connections Between Rivers and Floodplains

Depending on the project, in-channel habitat modifications may or may not increase river-floodplain connectivity. Projects that increase local base level through increased channel roughness or constriction may locally increase the potential for flooding (Young 1991) and thus increase aquatic terrestrial resource exchange and off-channel habitat accessibility. However, projects that do not alter channel hydraulics to such a degree will likely not increase river-floodplain connectivity.

Off-channel habitat modification will always be associated with increased lateral connectivity within the river corridor. Side-channel, slough, and floodplain pond creation and reconnection will promote the distribution of channel waters across the width of the valley floor and in the process impede flows (Wyzga 1996), increase organic matter recruitment (Tockner et al. 1999; Valett et al. 2005), and provide access to valuable foraging and rearing habitat (Henning et al. 2006). It has been estimated that in the Skagit Valley, 41 percent of sloughs and 31 percent of small tributary habitat has been lost to channelization for agricultural purposes (Beechie et al. 1994). This highlights the importance of rehabilitating off-channel habitat in Washington’s waterways. Side channels create refugia for juvenile fish (Jungwirth et al. 1993), while floodplain ponds and backwater sloughs create zones of high retention and productivity that are vital rearing habitat (Hall and Wissmar 2004; Sommer et al. 2005) and important sources of organic material for the channel (Tockner et al. 1999).

7.3.5.22.2 Altered Groundwater-Surface Water Interactions

Many in-channel and off-channel habitat modification projects result in an increase in flowpath complexity. Boulder placement creates divergent flow, rock weirs force water through pool-riffle transition zones, and side channels increase hyporheic exchange through bars and islands (Tonina and Buffington 2007).

Increased vertical exchange between surface and subsurface waters will benefit aquatic biota by increasing benthic dissolved oxygen levels and promoting solute uptake, filtration, and transformation. Studies have shown that the availability of dissolved oxygen to incubating salmonids embryos is dependent upon hyporheic exchange (Geist 2000; Greig et al. 2007) and that the occlusion of this exchange through siltation can lead to hypoxia within redds and decreased embryo survival (Heywood and Walling 2007). The hyporheic zone does more than promote oxygen exchange in subsurface sediments, it can also act as an effective filter and zone of biogeochemical transformations. Increased hyporheic exchange has been associated with nutrient uptake (Anbutsu et al. 2006; Sheibley et al. 2003) and transformation (Fernald et al. 2006; Lefebvre et al. 2005), and may attenuate the transport of dissolved and particulate metals (Gandy et al. 2007).

7.3.5.22.3 Altered Habitat Complexity

The increased habitat complexity which is a goal of most in-channel habitat modification will, in theory, have a positive impact on aquatic species. However, the success of an individual project can not be guaranteed and in-channel restorations to date have not been thoroughly monitored (Bernhardt et al. 2005) and have been applied with mixed success (Babcock 1986; Frissell and Nawa 1992; Moerke and Lamberti 2004; Roper et al. 1998). Consequently, the potential benefits of in-channel restoration may not outweigh the impacts associated with the construction phase and potential for project failure.

7.3.5.23 Riparian planting, restoration, or enhancement

A primary goal of riparian planting is to create mature forests where they have been removed and, to the extent possible, rehabilitate the complex dynamic between the channel and adjacent vegetation. Channel restoration through riparian planting develops on a time scale which is much longer than other restoration methods. However, once vegetation has become established, bank roughness will increase and flood water velocity will decrease. This will, in turn, promote overbank flooding of riparian areas and increase connection with floodplain environments. Mature riparian vegetation will increase allochthonous input and augment aquatic food webs (Wipfli 2005). It has been shown that in the Pacific Northwest the relationship between riparian vegetation, instream wood, and channel geomorphology is complex and essential to natural system functioning. Riparian forest structure and distribution is controlled by catastrophic processes such as flooding, landslides, and wind storms. The associated recruitment of felled trees alters channel form and redistributes flow paths, creating new zones of deposition and vegetation establishment (Fetherston et al. 1995). In this way the vegetation influences the channel while channel response influences the vegetation, resulting in a complex productive system in which native aquatic organisms thrive.

7.3.5.24 Wetland creation, restoration or enhancement

The creation, restoration or enhancement of wetlands is meant to promote terrestrial and aquatic connectivity. Wetlands are ecotones between terrestrial and aquatic environments (Mitsch and Gosselink 2000). Estuarine marshes moderate the connection of terrestrial resources to the open sea (Tanner et al. 2002). These areas are highly productive, providing a wide range of food sources to littoral fishes. The elimination of marshes decreases channel complexity and reduces the productivity of the nearshore zone (Ferraro and Cole 2007; Hood 2004).

Years of restoration efforts have focused on increasing spawning habitat in upland systems, yet the resultant increased populations will not thrive if there is not adequate rearing habitat to support the population. It has been suggested that density dependent mortality, that is the mortality of fishes due to too many individuals and not enough habitat, is a factor in both the Skagit and Duwamish Rivers (Greene and Beechie 2004). This indicates that estuarine wetland rehabilitation and the increased rearing habitat availability associated with it will be vital to the rehabilitation of degraded fisheries in the State.

Several researchers have pointed out that the complexity of undisturbed systems is not easily recovered from altered environments (Simenstad et al. 2006; Simenstad and Thom 1996; Thom et al. 2002; Williams and Orr 2002). Subsidence of pre-development wetland areas during times when land-use was intensive means that many restored areas are now too low to provide the proper physical conditions (Thom et al. 2002; Williams and Orr 2002). While this problem can be pronounced in settings where the sediment supply is limited, most geomorphic situations where estuarine marshes are found in Washington State exhibit large sedimentation rates [e.g., the Skagit River: (Hood 2006)] where this issue is either not applicable or ameliorated after only a few years (Thom et al. 2002).

7.3.5.25 Eelgrass and other aquatic vegetation restoration/ enhancement

Because of its modest hydrogeomorphic impact, eelgrass planting is not expected to fragment existing ecosystems. Eelgrass meadows are one of the most productive environments in the temperate coastal ocean (Ferraro and Cole 2007; Phillips 1984). These meadows are host to a wide variety of organisms and thereby have the capability to connect less productive environments.

7.3.5.26 Trap and haul

Trap-and-haul programs are somewhat effective approaches to providing fish passage around man-made barriers. Ideally, this approach would be used as an interim measure until volitional passage around the barrier can be established (e.g., by removal of the barrier or by construction of a fishway, ladder, or other type of passage structure). This approach has been recommended to support conservation of native salmonids.

Fish trap-and-haul activities have the potential to impose three specific forms of ecosystem fragmentation, all specifically related to the selective effects on fish that they intentionally or unintentionally impose:

- Passage barriers: Several forms of passage barriers may occur (e.g., operational challenges may result in failure to successfully pass fish in a given year), and unintentional selection pressures (e.g., size, run timing) may be imposed on the affected population.

- Modified upstream transport of allochthonous nutrients: Trap-and-haul programs will affect the upstream transport of nutrients by the affected organisms.

- Alteration of migratory patterns: Trap-and-haul operations may introduce fish into tributaries they did not originally occupy, or release fish upstream of natal streams, frustrating migration behavior by bypassing olfactory cues used in homing.

Alteration of migratory patterns is a unique impact imposed by the trap-and-haul subactivity type. While trap-and-haul programs are intended to mitigate or address passage barriers, in certain cases these programs may alter or frustrate the migratory behavior of the target species with unintended effects on their viability.

- Toutle River Chinook, coho, and steelhead were forced to alter their migratory pathway by the heavy load of ash delivered to the system by the 1980 eruption of Mount St. Helens. Subsequent to the eruption, a sediment retention structure was placed on the North Fork Toutle River to intercept mass wasting. WDFW (in cooperation with the U.S. Army Corps of Engineers) has operated a trap-and-haul program to pass adult Chinook, coho salmon, and steelhead around the structure, which is a complete barrier to passage. However, due to access issues, the fish must be released several miles upstream of former mainstem and existing tributary spawning and rearing areas. These populations adopted alternate spawning locations in other watersheds and recolonized their former habitats after it had recovered sufficiently.

- Lake Washington basin Chinook salmon and steelhead historically migrated into the Cedar River and other tributaries from Elliot Bay via the Black River by way of the Duwamish River. The creation of the Lake Washington Ship Canal created a new entrance to the Lake at Shilshole Bay and dewatered the original migratory path through the Black River. Within a period of approximately 5 years, the native salmonid populations had adapted to this alteration and today access the lake and its tributaries through the ship canal.

- Hinson et al. (2007) examined the migratory behavior of coho salmon and steelhead released in non-natal tributaries using radiotelemetry and found that the released fish did not migrate to their birth streams as anticipated. In effect, the release location has altered the migratory pathway for the affected stocks, increasing the travel distance to their habitats by several miles and requiring adaptation to new habitats.

- Schmetterling (2003) studied the behavior of tagged bull trout and westslope cutthroat trout transported above Milltown Dam on the Clark Fork River to determine the likelihood of reoccupation of historic habitats. All the test subjects migrated at least some distance upstream, and he found that some individuals moved upstream as far as 62 miles (100 kilometers) or more. Based on these findings, Schmetterling (2003) concluded that a trap-and-haul program could be used to initiate recovery of adfluvial populations while options for volitional passage were being decided.

The fish passage related effects are more difficult to assess and potentially more significant. Even the most carefully operated trap-and-haul program has the potential to introduce some type of selection pressure on the affected population, which could alter genetic diversity. Because the selection pressures imposed are associated with artificial environments, these changes are likely to be detrimental.

The extent to which migration delays caused by alteration of migratory pathways affect fish will depend in many respects on the specific life history and physiology of the species affected. Salmonid species that enter river systems sexually immature and spend lengthier periods in fresh water, such as spring- and summer-run Chinook and summer-run steelhead, are likely to be less prone to adverse effects than sympatric ocean type races. These types of salmonids enter their natal streams ready to spawn; therefore, their reproductive success is more sensitive to the effects of migration delay.

Consider that summer-run steelhead populations in the interior Columbia Basin migrate several hundred miles to reach natal streams. In doing so, individual fish from these populations may explore non-natal streams over periods lasting from days to weeks and may migrate upstream for considerable distances (tens of miles or more). For example, steelhead from the Clearwater River basin, Idaho, are commonly caught by anglers in the Deschutes River in Oregon 10 or more miles upstream of its confluence with the Columbia River. In contrast, ocean-type fall-run Chinook in Puget Sound enter their natal streams essentially ready to spawn, suggesting that delayed migration would impose greater consequences on reproductive success.

The effects of migratory corridor alteration on HCP invertebrate species are expected to be limited. Alteration of the migratory path of fish species targeted by trap-and-haul programs would have no discernable effects on marine invertebrate species. Effects on giant Columbia River limpet and great Columbia River spire snail, which are not directly linked to fish migration patterns by life history, would be similarly unaffected. In contrast, alteration of fish migratory pathways could alter the dispersal of the glochidia larvae of western ridged mussels, potentially introducing these species into new habitats. However, as most trap-and-haul programs are transporting fish within their original ranges, the extent of these effects is considered to be minimal.

7.4 Hydraulic and Geomorphic Modifications

Hydraulic and geomorphologic modifications associated with HPA projects occur in riverine, marine, and lacustrine environments. This section reviews what is known about the effects of these modifications on the movement of water (i.e., flow velocity, littoral currents) and the substrates in riverine, marine, and lacustrine environments, as well as the resultant impacts on HCP species. WDFW noted that miles of historical habitat have been permanently lost due to the placement of structures and fill, with commensurate permanent loss of riparian vegetation and large organic debris, as well as extensive intertidal habitat degradation from increased wave and current turbulence waterward of such structures (Canning and Shipman 1994).

7.4.1 Basic Hydrology, Hydraulic, and Geomorphic Influence on Habitat

Channel hydraulics refers to the flow of water in an open channel, such as a river, stream, or tidal channel, as well as the interactions between the flow and the channel boundaries. It also includes the concentrated flow of surface water across the land or the flow of water across a valley floodplain. It can also include the exchange of marine and fresh water in channels under tidal influence.

Water flowing in any open channel is subject to the external force of gravity that propels the water downslope as well as the friction between the water and channel boundaries that tends to resist the downslope movement (Leopold et al. 1964). Resistance to flow is caused by bed roughness, instream and bank vegetation, bank obstructions or irregularities, steps in the channel bed profile, and changes in channel alignment (Knighton 1998). All of these factors influence the hydraulic regime of a channel and dictate the channel morphology and the habitat characteristics of marine and freshwater ecosystems.

Activities that alter channel hydraulics can influence the channel morphology and in turn alter channel processes that create and sustain suitable habitats for fluvial and marine aquatic organisms. Conceptual models based on key relationships governing channel processes can be used to predict an array of possible channel responses to changes in sediment supply, transport capacity, and external influences such as changes in vegetation and woody debris loading (Abbe and Montgomery 2003; Brummer et al. 2006; Gilbert 1917; Lane 1955; Montgomery and Buffington 1998; Schumm 1971).

7.4.1.1 Freshwater Systems (rivers and streams)

Bolton and Shellberg (2001) provide a fundamental description of how water flows and activities in a channel affect flow.

> The amount of water passing a point on the stream channel during a given time is a function of velocity and cross-sectional area of the flowing water.

> $Q = AV$ (Equation 1)
> where Q is stream discharge (volume/time), A is cross-sectional area, and V is flow velocity. Equation 1 is a form of a mass-balance equation typically referred to by hydrologists as the Continuity Equation.
>
> If you confine the channel through various channelization activities, then the cross-sectional area decreases. If the channel must still carry the same flow or discharge (Q), then equation 1 shows that the flow velocity must increase. An increase in velocity results in an increase in the energy in the flow…(or) if you decrease the channel roughness or increase the channel slope, velocity increases. …an increase in velocity increases the energy of the flow and the amount of work that the water can do. This can lead to erosion of the channel bed and banks and transport of sediment downstream… Channel roughness is affected by substrate size, vegetation and large wood.

Miller et al. (2001) describe the basic concepts of geomorphic processes and their effect on riverine habitat.

> Streams are the arterial system of the land. The stream continuum begins with the smallest stream and ends at the ocean. Streams form a continuum of physical environments and associated aquatic and terrestrial plant and animal communities (Vannote et al. 1980). This continuum is a longitudinally connected part of the ecosystem in which downstream processes are linked to upstream processes.
>
> The characteristics of streams and streamflow in a particular watershed are defined by climatic parameters such as precipitation and temperature, as well as by physical factors such as topography, soils, geology, vegetation, and land use. The watershed provides two primary inputs that control channel form – water and sediment. These inputs ultimately drive fluvial processes and largely determine the nature of channel systems and channel process.
>
> Plants and animals have adapted to several distinct habitats that are characteristic of river corridors. These habitats can be subdivided into benthic, aquatic, and terrestrial zones (MacBroom 1998). The benthic zone consists of the streambed. Biota associated with the benthic zone are generally attached to or buried under the channel bed substrate. The aquatic zone is characterized by flowing water, and is associated with animals such as fish, (and) amphibians, and aquatic plants. Adjacent uplands make up the terrestrial zone, which is occupied by plants and animals that live on land that is rarely submerged for long periods of time.
>
> Fundamental fluvial processes include the downstream conveyance of water, sediment, nutrients, and organic matter. River geomorphology is also strongly affected by vegetation and geotechnical characteristics of channel boundary materials. The combination of factors associated with hydrology, climate, sediment transport, riparian vegetation, and boundary materials ultimately determines river channel form and process. The range of geomorphic processes

> that result include sediment entrainment, sediment deposition, floodplain inundation, recruitment of large woody debris, and creation and maintenance of riparian and aquatic habitat.
>
> Aquatic habitat is a product of fluvial processes, and diversity is a key component of productive stream habitat (Hill et al. 1991, Gore 1985, Poff et al. 1997). While geomorphologists may speak of channel forming flows in a relatively mechanical sense, biologists may view flow events in terms of their effects on aquatic habitat. Hydraulic forces differ both on a reach scale, locally (such as in the vicinity of a boulder or submerged log), and over the range of flows that a stream experiences. These forces create scour pools and transport, sort, and deposit coarse and fine bed materials, thus creating a diversity of bed forms and local substrate sizes (Lisle 1981). The resulting variety of depths, velocities, substrate types, and cover meets the needs of the various life stages of fish and other aquatic organisms (Gore 1985).

Additional useful sources of information on channel design include Bates (2003), Copeland et al. (2001), Papanicolaou and Maxwell (2000), and Watson et al. (1999).

7.4.1.2 Marine or Estuarine Systems

In marine environments, hydraulic and geomorphic processes also play important roles in creating and maintaining habitat for aquatic species. Shallow nearshore marine habitats, structured by tidal currents, wind, and input from terrestrial and freshwater sources, support spawning and larval settlement substrates as well as burrowing habitats for many of the HCP species (including juvenile salmon and rockfish species, cod, hake, Pacific herring, walleye pollock, Newcomb’s littorine snail, and the Olympia oyster) (Bargmann 1998; Couch and Hassler 1990; Healey 1982; Larsen et al. 1995; Penttila 2001; Simenstad et al. 1979). The controlling factors in these habitats depend upon bathymetry, substrates, circulation and mixing, and sediment transport. These underlying hydrogeomorphic variables regulate a phenomenon known as longshore transport, or littoral drift (Komar 1998). Littoral drift is an important controlling factor in the determination of habitat structure; it is the transport and deposition of sediment that supports aquatic plants. Key to understanding littoral drift is the concept of a drift cell (also known as drift sectors), which is a segment of shoreline along which the longshore transport moves sediment at noticeable rates. Each drift cell includes: (1) a sediment source, such as a feeder bluff; (2) a driftway along which these sediments move; and (3) an accretion terminal where the drift material is deposited. In this way, a drift cell allows the uninterrupted movement of beach materials (Terich and Schwartz 1990; Cox et al. 1994).

Wave action striking shorelines at an angle causes littoral currents that move parallel to shore (Cox et al. 1994). While littoral processes are most conspicuous in marine waters, they can occur along lakeshores as well, where fetch and wind speed combine to produce waves and subsequent long shore currents strong enough to move shoreline sediments.

Shoreline features, including artificial structures, affect the velocity and direction of shoreline currents and sediment transport.

7.4.2 Mechanisms of Impact from Hydraulic and Geomorphic Modifications

HPA-permitted activities may result in altering the following processes:

- Riverine Environments
  - Altered Flow Conditions
  - Altered Channel Geometry
  - Altered Sediment Supply (Transport)
  - Altered Substrate Composition and Stability
  - Altered Groundwater/ Surface Water Interactions

- Marine Environments
  - Altered Wave Energy
  - Altered Current Velocities
  - Altered Nearshore Circulation
  - Altered Sediment Supply
  - Altered Substrate Composition

- Lacustrine Environments
  - Lakes
    - Altered Nearshore Circulation
      - Groundwater Input
      - Short-Period Waves
      - Buffering Capacity (WQ?)
  - Reservoirs
    - Altered Nearshore Circulation

Each of these mechanisms of impact may significantly affect the distribution, health, and survival of potentially covered species through direct or indirect adverse impacts on the habitat or other ecological life stage requirements of a given species. HPA-permitted activities that impose adverse impacts may in turn affect the population dynamics of fish and aquatic invertebrate species, either locally or on a broader regional scale. The magnitude of the potential impacts will depend upon:

1. The size, duration, and frequency of the impact.
2. The vulnerability of the affected life-history stage.
3. The ability of the organism to avoid the impact.
4. The physiological, developmental, and behavioral impairments suffered by the organism.
5. Indirect mechanisms such as exposure to predation.

7.4.2.1 Riverine Environments

River hydrology includes the movement of water in the stream, the movement of hyporheic groundwater to the stream, and the movement of surface water across land to the stream. It also includes the tidal delta hydrology and the river's exchange of marine and fresh water. Changes to riverine hydrology that reduce or increase the flow of water to the river alter the suitability of habitats within the river. During low-flow periods, alterations to hydrology can result in previously wetted areas going dry, thereby eliminating habitat area for aquatic organisms. Hydrologic alterations that increase overland surface water flow can, on the other hand, increase flooding and substrate scour.

Rivers are naturally dynamic systems that adjust to tectonic, climatic, and environmental changes (Dollar 2000). Environmental changes can be either human-induced or natural. The environmental components that contribute to channel processes are influenced by local and basin-scale variations in sediment supply, transport capacity, and the effects of vegetation (Montgomery and Buffington 1998). River systems adjust to maintain a steady state, or dynamic equilibrium, between the driving mechanisms of flow and sediment transport and the resisting forces of bed and bank stability and resistance to flow (Soar and Thorne 2001).

The quantity, quality, and diversity of aquatic habitats are the products of the fundamental channel processes entailing the conveyance of water, sediment, nutrients, and organic matter (Miller et al. 2001). The hydraulic forces acting in a river carve channels; recruit LWD; create scour pools; and transport, sort, and deposit coarse and fine bed materials. Riverine hydraulics determine the nature, as well as the distribution and deposition of, sediments and other materials along the path of the river's unidirectional movement toward lower elevations.

The resulting variety of depths, velocities, substrate types, and cover provides habitat diversity and meets the needs of the various life stages of fish and other aquatic organisms (Gore 1985). Fishes and invertebrates depend upon the diversity of habitats created by hydraulic forces (Montgomery et al. 1999). HCP species such as sturgeon, char, bull trout, salmonids, and freshwater mussels, depend on particular riverine sediment types, hydraulics, and habitats for reproduction, growth, and survival. Alterations to river hydraulics that change the flow of water and the ability of the water to move sediments and nutrients can have direct and indirect effects on HCP species. If flows become too strong, reaches of rivers can be made impassable to various fish species or life-history stages, or unsuitable for invertebrates. Projects that alter riverine hydrology can also have direct and indirect effects on HCP species.

Channels are defined by the transport of water and sediment confined between identifiable banks (Dietrich and Dunne 1993). Natural stream channels show great variety, reflecting differences in channel processes, disturbance regimes, structural controls, and geologic history (Washington Forest Practices Board 1995). One of the channel classification schemes most widely employed in Washington distinguishes channels primarily according to their roughness characteristics and their sediment transport regime (Montgomery and Buffington 1993, 1997). Some channel types addressed in this classification, i.e., **bedrock** and **colluvial channels**, are of little concern here because they seldom provide significant habitat for potentially covered species and because bedrock channels are unlikely to experience appreciable process change due to placement of artificial structures. **Alluvial channels** (as opposed to channels incised into bedrock) have erodible bed and banks comprised of sediments. An alluvial stream adjusts the dimensions of its channel to the wide range of flows that mobilize its boundary sediments. The adjustments of a river system are made over a continuum of spatial and temporal scales that result in corresponding gain, loss, or redistribution of habitat features.

In alluvial channels, a wide variety of channel types may develop. Montgomery and Buffington (1993) recognize six such channel types:

- cascade,
- step-pool,
- plane bed,
- pool-riffle,
- braided, and
- regime.

They propose that these types are controlled primarily by channel gradient and also by sediment supply (the amount of material available for transport) and transport capacity (determined by shear stress, which is similar to stream power). The singular importance of LWD as a structural element is also recognized. Changes in channel gradient, sediment supply, and stream power, which can be altered by placement of instream structures, therefore have the potential to directly alter habitat conditions for potentially covered species.

The steepest channels described by Montgomery and Buffington (1993) are **cascade channels**. Because of their high gradient (typically steeper than 8 percent), these channels usually have high roughness caused by boulder or bedrock bedforms. They typically have high transport capacity, so little sediment is stored in the bed or banks. The most common disturbance is debris flow. Cascade channels are predominant in small mountain tributaries in Washington, where they are often seasonal, non-fish-bearing streams. Some cascade channels, however, occur lower in the stream system, commonly where a stream transits a layer of relatively erosion-resistant rock; in such areas, they may link lower-gradient reaches having greater habitat value.

Step-pool channels commonly have a lower gradient of about 3 to 8 percent (Montgomery and Buffington 1993; Papanicolaou and Maxwell 2000). Many perennial, fish-bearing streams in hilly and mountainous parts of Washington have a step-pool morphology. Step-pool channels commonly provide spawning habitat for resident salmonids, especially when lower-gradient habitats downstream are utilized by anadromous salmonids (Montgomery and Buffington 1993). Step-pool channels are highly sensitive to the amount of LWD in a stream and to the stream's sediment supply; if LWD is removed from a step-pool channel, the channel's sediment storage capacity is reduced, sediment is transported from the reach, and the channel commonly shifts to a plane bed or pool-riffle morphology (Montgomery and Buffington 1993). This is an adverse habitat change for organisms that require deep and persistent pools, for example as cover or habitat buffer during low-flow periods. Severe increases in sediment supply also tend to cause loss of pools, again by filling, but step-pool channels tend to be robust against such a change, because filling pools reduces channel roughness, in turn increasing transport capacity and allowing scour to reestablish the pools (Montgomery and Buffington 1993). However, the pool filling and subsequent scour associated with this equilibration process could be expected to have adverse impacts on stream organisms. More moderate changes in sediment supply would also be expected to alter these channels, primarily by causing a general coarsening or fining of bed material. Generally, step-pool channels have a high enough gradient and transport capacity that it should be feasible to place additional roughness elements, such as artificial structures that occupy a fraction of the channel, without substantially altering channel hydraulics and sediment transport.

At more moderate gradients (typically 1 to 3 percent), the principal channel types are **pool-riffle** and **plane-bedded channels**. These channel types are highly vulnerable to hydraulic or sediment source changes, because they have low to moderate transport capacity; thus, relatively small changes in channel morphology can cause changes in net sediment accumulation or export, with associated changes in grain size and bedform (Montgomery and Buffington 1993, pg. 50). Normally, plane-bed channels have well-defined bed and banks with a lack of bedforms. LWD plays a critical role in pool-riffle and plane-bed channels. Adding LWD to a system will often cause a plane-bed channel to become a pool-riffle channel, while removing LWD will often cause the reverse transformation (Montgomery and Buffington 1993, pp. 41, 53). In channels that lack the transport capacity to move boulders, LWD provides the principal sites for both scour (which forms pools) and sediment accumulation (which forms riffles). Artificial instream structures such as abutments and pilings are often local sites for scour in these channels. In larger rivers with plane-bed channels, significant scour can occur, particularly in response to channel structures such as LWD (Sedell et al. 1986; Collins et al. 2002). This has been described, for instance, as the historical condition on the South Fork Nooksack River (Maudlin et al. 2002; Sedell and Luchessa 1982) and the Willamette River (Sedell and Froggatt 1984) and in the general case for larger Western Washington rivers (Abbé and Montgomery 1996).

Plane-bed and pool-riffle channels display a characteristic sensitivity to changes in sediment supply. Increases in fine sediment supply commonly lead to embedding, a process whereby fine sediments are incorporated to the bed of the stream and remain there after they become armored by a relatively thin surficial layer of coarse sediment. Embedding gives the stream a relatively hard, impervious bed that provides a poor substrate for salmonid spawning, impairs hyporheic exchange, and provides poor habitat for benthic invertebrate infauna. Typically, several years of peak flow events are required after the fine sediment inputs have ended for the bed to be sufficiently reworked that embedding abates.

Inputs of coarse sediment initially have little effect on pool-riffle channels, but as the inputs increase, the pools are filled, the channel aggrades, and the bedform changes from pool-riffle to plane bed (Montgomery and Buffington 1993). Continuing aggradation leads to channel widening and bar development (Montgomery and Buffington 1993). With sufficiently large increases in coarse sediment supply, the channel may develop a **braided** form (Montgomery and Buffington 1993).

Plane-bed and pool-riffle channels are among the most important for salmonid spawning because they have a bed mobility and scour regime to which salmon are well adapted, providing spawning habitat for large numbers of fish (Montgomery et al. 1999). These channels are also a principal habitat for freshwater molluscs, such as the potentially covered mussels, limpets, and spire snails.

The lowest-gradient channels, having gradients of less than 1 percent, are **regime channels** (Montgomery and Buffington 1993). These channels are abundant on floodplains and in tidewater areas of Washington. Regime channels are normally transport-limited and commonly have sand or silt beds. They are highly vulnerable to changes in sediment supply, alteration of bank vegetation, and artificial changes in gradient (Montgomery and Buffington 1993). Coarse sediment tends to fill the channel because the stream lacks the transport capacity to move it through the system. Finer sediment will be exported, but slowly; in the meantime, the channel tends to become wider and shallower (Montgomery and Buffington 1993). Because the bed and banks are comprised of relatively fine sediment, the roots of vegetation are particularly important to maintaining bank integrity; the loss of riparian vegetation can trigger bank erosion, causing sediment inputs and channel widening/shallowing (Montgomery and Buffington 1993, p 53).

All channels occur within a landscape context. Principal elements include the floodplain or channel migration zone, which is the area directly influenced by the channel during geologic time frames, and confinement, which is determined by the channel's proximity to neighboring hillslopes. Mountain channels (cascade and step-pool channel types) in Washington are often closely confined with no definable floodplain, but most fish-bearing channels do have a floodplain. Important structural and functional elements of floodplains and channel migration zones are described by Bolton and Shelberg (2001) and include:

- Channel complexity in the form of secondary channels, bars, channel sinuosity, and the way in which these change during floods
- Riparian ecosystems, particularly forested riparian systems that act as LWD sources and are subject to successional changes
- Groundwater and hyporheic components

Placement of structures within or beneath the stream channel can have the following primary effects on the channel (Brookes 1988, in Bolton and Shellberg 2001):

- channel shortened by straightening;

- channel cross-sectional area reduced (by placing fill, pilings, and/or abutments in the channel);
- channel bed and/or banks replaced with non-erodible artificial materials; and
- the channel loses the ability to migrate over time.

Channel roughness elements affect stream velocity by increasing boundary shear stress, thereby increasing resistance to flow (Leopold et al. 1964). Structures can increase or decrease channel roughness in a variety of ways that alter habitat, such as changes in in-channel roughness elements, changes in channel perimeter roughness elements, or changes in the relationship between channel area and wetted perimeter. All materials in contact with the wetted channel constitute roughness elements. The principal in-channel roughness elements are artificial structures such as gratings or pilings, and natural structures such as large woody debris.

An example of roughness effects on channels was encountered at a highway bridge reconstruction investigated by Barks and Funkhouser (2002), using a two-dimensional flow model to estimate conditions during the 100-year flood. Barks and Funkhouser (2002) found that relocating a bridge abutment from an area of dense vegetation to an agricultural area predicted a 67 percent decrease in channel roughness and a 29 percent increase in flow velocity, with associated high risk of scour and channel destabilization.

Because flow velocity is proportional to the product of roughness and wetted perimeter (Leopold et al. 1964), changes in the length of the wetted perimeter can also alter stream power. Structures in the channel alter the wetted perimeter directly, such as when flow is confined by a pier, or indirectly, such as when erosion or deposition causes changes in channel geometry. Structures such as docks and piers tend to confine the channel within artificial bounds and thus generally cause locally reduced channel roughness, potentially causing scour at the structure, with corresponding deposition downstream. Sturm (2004), modeling scour at bridge abutments in sandy sediments, found that scour could be significant enough to alter channel geometry, producing large excavations near bridge abutments and causing reduced water depths and sediment deposition immediately upstream. Sturm (2004) also found that this effect could be exacerbated in higher flows. The fact that the investigated abutment supported a bridge is immaterial; the structure represented by the abutment could have supported any kind of overwater structure, such a as pier. They used the same model to show that planting trees and placing riprap in the area would alleviate the predicted flow increase and move the area of maximum flow back into the stream's thalweg (the line of steepest descent along the stream). This study identified some of the principal channel border roughness elements, such as sediment, vegetation, and artificial elements like riprap and bridge abutments. This study underscores the importance of using hydraulic modeling to avoid locally significant changes in channel structure.

Each of these effects constitutes an "impact," but collectively these impacts affect channels primarily by altering only one controlling factor: stream power, which is in turn determined by water surface slope, flow volume, and channel roughness (Dunne and Leopold 1978). Structures placed in the channel have the potential to alter each of the factors identified in the above list. Because the surface of a stream is roughly parallel to its bed (Dunne and Leopold 1978), water surface slope is mainly altered by changes in channel gradient.

Channels are dynamic landscape elements that integrate inputs from tributary channels and from valley and hillslope processes (Washington Forest Practices Board 1995). Thus, a structure placed in a channel is likely, over time, to experience the effects of altered stream power and an altered sediment transport regime caused by changes in the watershed upstream. For example, in areas subject to progressive urbanization, gradual increases in catchment impervious surface cause predictable hydrologic changes characterized by increased variance in the hydrograph (Booth et al. 2002). One consequence of this change is increased peak flows and correspondingly increased sediment transport capacity, which often cause streambank instability and channel downcutting (Dunne and Leopold 1978, pp. 693-695). The resulting increases in flow and sediment around and through in-water structures can exceed the structures' design capacity, leading to outcomes such as scour around abutments and pilings, ponding upstream of culverts, culvert flow velocities that constitute a fish passage barrier, or a host of culvert structural problems.

To summarize, the placement of artificial structures in channels can, through a variety of mechanisms, cause increased erosion at or upstream of the structure, increased deposition downstream, and increased sediment transport past the structure. This amounts to a change in structural elements of the channel that relate to habitat such as channel type, substrate size distribution, channel cross-section, channel migration, bed mobility, and bank structure.

7.4.2.1.1 Altered Flow Conditions

The placement of pilings, fill, or nonerodible materials associated with the construction, operation, and repair of HPA projects can alter channel hydraulics through changes in roughness, channel geometry, and flow velocity. These changes are interrelated and can act in concert to modify channel morphology and interrupt natural habitat-forming processes (Montgomery and Buffington 1998) and even create predatory fish habitat (see Carrasquero [2001] for a related literature review). Increased velocities can indirectly affect various species by causing bed scour at channel obstructions (such as man-made structures) and corresponding sediment deposition downstream (Richardson and Davis 2001). Alterations to channel hydraulics that change the ability of water to transport and deposit sediment and nutrients can modify or eradicate suitable habitats for various lifestages of HCP species. Altered channel hydraulics can cause changes in nutrient flow, prey resources, and foraging opportunities which can result in reduced growth, fitness, reproductive success, and survival for both fish and invertebrates.

Pilings act as cylindrical flow obstructions that add hydraulic roughness to a channel. Likewise, fill placed in a channel can obstruct flow and add hydraulic roughness. Because flow velocity in a channel is proportional to hydraulic radius and inversely proportional to roughness (Leopold et al. 1964), adding pilings or hardening the bank can alter the flow velocity, depth, and width of a channel relative to natural conditions. The net effect of artificial fill or pile groups is to confine the flow. Riprap substrates (and, presumably, any substrates permanently simplifying channel margins) reduce complexity and diversity along the channel margin, leading to increased water velocity (Cramer et al. 2003). Hardened banks that replace riparian vegetation can increase the flow velocity and potential for scour and substrate coarsening through a reduction in hydraulic roughness compared to vegetated conditions (Millar and Quick 1998). Hard approaches to armoring tend to transfer energy downstream of the protected shore, and an increase in bank erosion and/or a loss of habitat in an adjacent reach can be readily anticipated (Cramer et al. 2003).

The primary effects of flow confinement by artificial structures are increased velocity and bed scour around structures and corresponding sediment deposition downstream (Richardson and Davies 2001). Scour is potentially an issue in all channel types, although it is most often a concern in alluvial plane-bed and pool-riffle channels, which have a relatively mobile bed. The term "scour" is usually used to refer to flow-driven horizontal excavation of the streambed, but it can also occur laterally along stream margins and result in bank erosion. Scour in stream and river systems chiefly occurs in conjunction with high-flow events that account for the largest fraction of annual sediment transport. Bed scour into a substrate of mixed particle sizes (i.e., sand and gravel) can selectively remove finer sediment and cause the substrate to coarsen. Likewise, deposition of the finer sediment downstream can bury organisms and cause the substrate to become finer.

The increased velocities and bedload movement associated with HPA-permitted modifications in the watershed that can impact peak flows encountered during flood events (e.g., bank protection, logging, addition of impervious surfaces, removal of riparian vegetation) exacerbate the natural scour conditions that fish may be adapted to and therefore can reduce egg survival. Substrate scour can affect fish egg nests by dislodging eggs and transporting them downstream before they have incubated sufficiently.

In addition to the location of the egg deposits in the channel and the bedload movement associated with flows, the vulnerability of these egg deposits depends upon the depth to which they are deposited. Montgomery et al. (1996) measured both scour and egg pocket burial depths of chum salmon and determined that a small increase in scour would affect the integrity of a large proportion of redds. There is a growing body of evidence (e.g., Montgomery et al. 1996, Montgomery et al. 1999) that salmon are adapted to natural bedload movement conditions. For example, based on observations that chum salmon bury their eggs just below scour depths during bankfull flow, Montgomery et al. (1996) hypothesized that the depths to which salmon bury their eggs represent an adaptation to the depths of scour during typical winter storms.

Further, Montgomery et al. (1999) provides evidence that salmon spawning distributions and timing are adapted to basin-specific scour conditions. These adaptations can result in salmon eggs being vulnerable to increases in the frequency and size of bedload movement associated with bank armoring. Such vulnerabilities could presumably be more severe for smaller fish species that bury eggs (e.g., lamprey, Olympic mudminnow, and resident trout). Smaller fish tend to spawn in smaller substrates and bury eggs at shallower depths than salmon and therefore may be more likely to be dislodged during unnaturally high scour events.

Freshwater mussels are particularly vulnerable to scour because they are long-lived, sessile organisms. Mussels are commonly found on relatively coarse (gravel to boulder) substrates in microsites that constitute flow refugia with low risk of scour (Cuffey 2002; Brim Box et al. 2004).

Increased scour can also have effects on floodplain processes. The geometry of a deepened channel disconnects it from the floodplain by creating a perched floodplain, or terrace, high enough above the channel that it is either no longer or less frequently inundated by the current hydrologic regime (Cramer et al. 2003). This can lead to abandonment of side channels and ponds in the short term and to reduction or prevention of sediment and nutrient delivery to the floodplain in the long term (Naiman and Bilby 1998). In addition, the formation of the terrace disconnects that surface from the water table and affects the establishment and survival of riparian vegetation. Other effects include bank instability as a result of oversteepening, groundwater discharge, increased shear stress because of very high peak flows within the channel, and loss of wetland/floodplain habitat and backwater areas.

Fish and invertebrates inhabiting riverine environments require certain flow velocities for spawning, rearing, migration, and foraging. Increases in flow velocities could present potential barriers to fish migration or could exceed thresholds for certain life-history stages of some HCP species.

- Chinook salmon tolerate velocities up to 49.9 ft/sec (15.2 m/sec) (Johnson et al. 2003) during migration.
- Pacific lamprey seek out slower velocities (0–0.33 ft/sec) for rearing (Stone and Barndt 2005).
- Optimal velocities for spawning habitat for mountain suckers in Lost Creek, Utah, are 2.4–7.9 in/sec (0.06–0.2 m/sec) (Wydoski and Wydoski 2002).
- Spawning velocities for Columbia River white sturgeon are similarly low (~2.6 ft/sec [0.8 m/sec]) (Paragamian et al. 2001), although this species spawns successfully in areas with higher average velocities by using river bed dunes and similar features for hydraulic refuge (Young and Scarnecchia 2005).
- Leopard and Umatilla dace inhabit riverine environments where the velocities are less than 1.6 ft/sec (Wydoski and Whitney 2003). Exceeding this velocity would render habitat unsuitable for these species.

Other species and life stages that may continue to use the habitat would need to expend higher energetic outputs to maintain position. This could impact growth rates and predation risks. In the case of larval fish, a study of fish use along natural and channelized habitats in the Willamette River, Oregon, concluded that continuous revetments are not good larval fish habitat (Li et al. 1984, in Bolton and Shellberg 2001). The authors determined that the combination of proximity to fast water, steep bank slopes, greater water depth, and cooler temperatures does not provide suitable habitat for larval fish.

Higher bank slope and velocity would also impact substrate composition and distribution such that the benthic and epibenthic invertebrates that are important in the diets of many fish species may no longer be as abundant or available. A shift in invertebrate species composition and abundance that affects diets would further exacerbate the problems created by increased energetic demand. As described by Bolton and Shellberg (2001), velocity is one of the critical factors contributing to the presence and abundance of macroinvertebrate species. Many species require low turbulence habitat for substrate. Bank protection activities that include channelization disrupt invertebrate communities (Bolton and Shellberg 2001). Reductions in the availability of prey can reduce the carrying capacity of a river system.

Direct and indirect effects of altered flow velocities on invertebrates are not well understood and represent an area for further research. However, for the HCP invertebrate species that are filter feeders (e.g., California floater and western ridged mussel) or rely on stable substrate for habitat structure, altered sediment transport is likely more important than changes in flow velocities.

Flow velocities influence swimming activity and respiration in fish species. Increased flow velocities during water releases can force fish species to rest in areas of slower moving water to recover from increased activity. This behavior can result in unsuccessful recruitment from delayed migration upstream for anadromous species (e.g., salmonids, sturgeon, lamprey), or increased predation from remaining longer in slow pools downstream of weirs and high-velocity reaches.

The addition of impervious surfaces is known to affect the hydrologic regime through changes in the magnitude, volume, and timing of flows (Booth 1991; Konrad 2000). Hydrologic changes that affect the velocity and depth of flows are considered a hydraulic alteration.

Increased impervious surface area can have local effects on water quality and flow conditions in streams and rivers, as well as on the cumulative effects of urbanization within a watershed. In particular, reduced infiltration can alter stream hydrology such that peak flow levels are increased and base flow levels decreased. Changes in peak flow volumes and the rates at which peak flow levels rise and fall can lead to damaging changes in channel morphology and substrate composition. Decreased base flow levels in summer months can reduce the amount of suitable habitat area available for aquatic species, as well as lead to unfavorable changes in the water temperature regime. Stormwater runoff from impervious surfaces is also likely to carry a range of pollutants known to have detrimental effects on aquatic species, including PAHs, metals, and organic compounds including pesticides, herbicides, fertilizers, and other substances.

7.4.2.1.2 Altered Channel Geometry

The alteration of channel processes and morphology can impact fish and invertebrates through the reduction of habitat quantity, quality, and diversity. These impacts can range from subtle shifts in the distribution and abundance of species to complete dislocation of a species from a particular locale. The reproduction, growth, and survival of HCP species depend upon particular channel hydraulics to maintain their specific habitats. Alterations to channel geometry can result in reduced growth, fitness, reproductive success, and survival.

The range and magnitude of potential responses of the channel to hydraulic and geomorphic changes and how these responses are transmitted downstream in riverine environments depend on the channel type and location of the disturbed reach in the drainage basin (Montgomery and Buffington 1997; Schumm 1971). The availability of backwater areas and off-channel habitat can be reduced by bank protection structures. Habitat quantity and complexity will be reduced by the shortening of the river and narrowing of the river cross section. The reduction in the amount of side channel and floodplain areas can impact fish species that rely on any of the associated habitats, including wetlands, beaver ponds, bogs, and off-channels.

Vannote et al. (1980) proposed the river continuum concept to describe freshwater habitat and the importance of various physical, chemical, and biological processes. According to the river continuum concept, the distribution of stream characteristics reflects a headwater to mouth gradient of physical conditions that affect the biological components in a river including the location, type, and abundance of food resources with a given stream size. The ecological significance of a potential channel response to channel modification depends on the species of interest.

Alteration of channel geometry has both direct and indirect effects on fish and invertebrates. Indirect impacts arising from the alteration of channel geometry include the modification of natural sediment transport, a reduction in habitat connectivity, and a reduction in habitat complexity. Fish and invertebrates require certain widths and depths for habitat, spawning, and cover.

As a result of the loss of side channel and floodplain habitats during high-flow events, fish could be displaced downstream or would require higher energetic outputs to maintain position in the higher velocities. For territorial species or life stages (e.g., coho juveniles), the displacement would require the fish to locate and establish a new territory with suitable habitat conditions. Presumably, this could impact any fish that may have been occupying the new habitat and trigger its displacement.

- The loss of side channel and floodplain habitats reduces the availability of refuge habitat during high flows as well as summer rearing and overwintering habitats for juvenile salmonids and other small fish species.
- Juvenile coho salmon are particularly impacted by a reduction in off-channel habitats and beaver ponds, and numerous studies have documented their reliance

on those habitat types (e.g., Brown and Hartman 1988; Bustard and Narver 1975; Swales and Levings 1989). In Carnation Creek watershed (a drainage in Vancouver, British Columbia), between 15 and 25 percent of the total coho smolt yield was captured in off-channel sites (Brown and Hartman 1988, in Henning 2004).

- Chinook (Swales and Levings 1989), sockeye (Burgner 1991), chum (Salo 1991), and steelhead (Puget Sound Steelhead Biological Review Team 2005) all rely on off-channel habitats to a lesser extent, but would be impacted by the loss of habitat.
- Pink salmon rely very little on off-channel habitats (Heard 1991) and would therefore be least impacted by the reduction of such habitats.
- Among trout and char, coastal cutthroat utilize off-channel environments the most (Lister and Finnigan 1997) and would be the most likely to be impacted by the loss of habitat.
- The loss of side channel and floodplain habitat could also impact species such as lamprey and mountain suckers that rely on slow-moving backwater areas for habitat.
- Olympic mudminnows require access to floodplain wetlands and bogs. In an investigation of the role of regulated floodplain wetlands in the Chehalis River as rearing (i.e., feeding and refugia) habitat for fishes, Henning (2004) documented high fish utilization in seasonally flooded habitats. The study captured 19 different fish species, including juvenile salmonids, Olympic mudminnows, and Pacific lampreys. Based on the high number and frequency of catch, it appears that these seasonally flooded habitats are preferred habitats for Olympic mudminnows (Henning 2004).

7.4.2.1.3 Altered Sediment Supply

Channel morphology (i.e., width, depth, bed slope, substrate size, bed forms, and pattern) is influenced by both local and downstream variation in sediment input from watershed sources (sediment supply), the ability of the channel to transmit these loads to downstream reaches (transport capacity), and the effects of instream woody debris and bank vegetation on channel processes (Montgomery and Buffington 1998). The relationship between these controlling factors is illustrated in Figure 7-2.

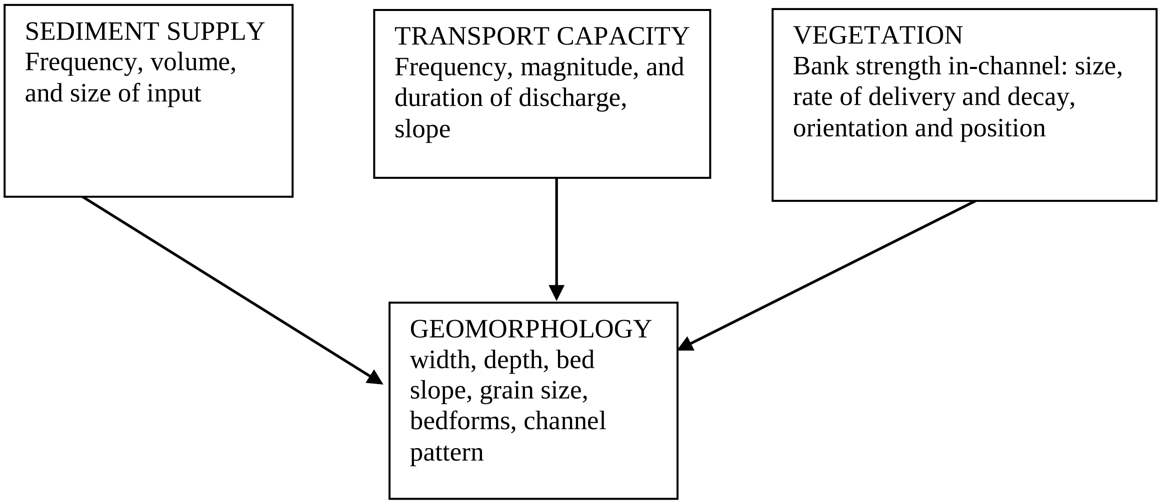

Figure 7-2. Riverine hydraulic controlling factors (adapted from Montgomery and Buffington 1998).

Because the rate and caliber of sediment supplied to a channel influences the substrate size (Dietrich et al. 1989), changes in sediment supply can alter the composition of substrate used by HCP species. Modifications affecting sediment supply may include increased scour of substrate and increased deposition of substrate. Scour and deposition may include impacts that can extend far beyond the project site.

Because HCP species depend on the presence or absence of particular substrate types to support important life-history functions, changes in sediment source contributions can have direct and indirect effects on those species. Fish and invertebrates require a range of substrate conditions in riverine environments for various life-history stages. These conditions rely on the replenishment of suitably sized substrates to offset natural sediment transport processes that remove sediment. In a study in California, the primary cause for the decline of salmon in the Sacramento River was linked to the loss of spawning gravels normally derived from bank erosion before riprap bank stabilization (Buer et al. 1984).

On the other hand, excessive sediment supply can affect fish and invertebrate species. Deposition effects depend on the particle size distribution and amount of sediment. For example, when sedimentation occurs, salmonids may be negatively affected in several ways: buried salmonid eggs may be smothered and suffocated; prey habitat may be displaced; future spawning habitat may be displaced (Spence et al. 1996; Wood and Armitage 1997), and juveniles and small fish may be prevented from using the interstices as refuge (Spence et al. 1996).

Channel incision, floodplain disconnectivity, and reduced lateral migration all contribute to a reduction in the recruitment of LWD, organic matter, and gravel. LWD is a major component of pool formation, channel braiding, cover, and habitat complexity (Bisson et al. 1987). Woodsmith and Buffington (1996) found that the number of pools in a channel system was highly correlated with the quantity of LWD. In contrast to areas where bank protection disconnects the river from the floodplain, inundation of floodplain areas recruits additional organic matter and nutrients that provide the base of a productive food web, which can result in high yields of fish (Bayley 1991, 1995). Gravel sources along river routes supply substrate for the continual natural replacement and transport downstream. In-channel gravel provides several functions for multiple trophic levels, including spawning substrate for fish, attachment points for sedentary invertebrates and aquatic vegetation, and habitat for epibenthic invertebrates.

7.4.2.1.4 Altered Substrate Composition and Stability

Alteration of the substrate composition through coarsening or fining of the bed can have direct and indirect effects on HCP species. The ecological effects of substrate coarsening and fining on salmonids and trout in riverine environments are well known.

Bed scour into a substrate of mixed particle sizes (i.e., sand and gravel) can selectively remove finer sediment and cause the substrate to coarsen. Likewise, deposition of the finer sediment downstream can bury organisms and cause the substrate to become finer. For blunt objects, the depth and extent of bed scour depend on the water depth, approach velocity, and shape and size of the obstruction (Richardson and Davis 2001). Increased bed scour and substrate coarsening are detrimental to habitat suitability. Scour and substrate coarsening are often accompanied by an increase in the interlocking strength of bed particles and the threshold force necessary for bed mobility, leading to bed armoring (Church et al. 1998; Konrad 2000; Lane 1955).

At the outset of spawning, adult fish winnow fine sediment from their gravel redds, mobilizing fine sediment into the water column and in the process coarsening the bed in the immediate vicinity of the spawning nest (Kondolf et al. 1993; Montgomery et al. 1999). However, if fine sediments are deposited again after redd construction, this material fills pore spaces between gravel particles in and over the redd.

Deposition of fine sediment may degrade instream spawning habitat and reduce survival from egg to emergence by smothering interstices (Chapman 1988; Phillips et al. 1975; Zimmermann and Lapointe 2005). Excessive deposition of fines can lead to substrate embeddedness, reduce the water circulation necessary to oxygenate the eggs, and reduce flushing of metabolic waste (Bjornn and Reiser 1991; Zimmermann and Lapointe 2005). Embryo mortality has been found to occur from poor water circulation and lack of oxygenation associated with the filling of intergravel pore spaces by fine sediment (Bennett et al. 2003 Chapman 1988; Cooper 1965; Lisle and Lewis 1992). The probability of pore space filling increases if the sediments are particularly fine, if the sediment amount is large, and if flows/currents are relatively low (Bjornn and Reiser 1991). For salmon, the process may be exacerbated by downwelling hyporheic flows, which often occur at salmonid spawning sites in Pacific Northwest rivers (Tonina and Buffington 2003, 2005). In a study of spawning chum salmon in low-gradient, gravel-bed channels of Washington and Alaska, Montgomery et al. (1996) found that minor increases in the depth of scour caused by bed fining and reduction in hydraulic roughness significantly reduced embryo survival. Dolly Varden prefer gravel as a spawning substrate (Kitano and Shimazaki 1995).

The amount of sediment does not need to be large to cause smothering effects. Although redds of large salmonids are usually buried beneath at least 6 inches (15 cm) and as much as 1 foot (30 cm) of gravel (Bjornn and Reiser 1991; DeVries 1997), near-surface deposits of fine sediment may be sufficient to reduce water flow through the redd and create a surface layer that physically prevents alevin emergence (Bjornn and Reiser 1991; Everest et al. 1987). Fines under approximately 0.03 inch (0.85 mm) in diameter have been shown to be particularly detrimental to salmon eggs through the associated decrease in dissolved oxygen (Chapman 1988). Research has documented significant declines in salmonid egg survival when the percentage of fine sediments under 0.03 inch (0.85 mm) in diameter reaches the range of 10 percent (Tappel and Bjornn 1983) to 13 percent (McHenry et al. 1984; see Chapman 1988).

Salmon require a range of sediment sizes, and spawning success depends on how well they can mobilize sediment with their tail to create a redd. Different species of salmon use gravels of different size and can effectively move only certain size classes of sediment (Kondolf 1997; Kondolf and Wolman 1993). During redd building, salmon avoid substrates larger than those they can mobilize (Kondolf and Wolman 1993; Kondolf al. 1993). This includes areas where erosion to bedrock has occurred. Field observations have shown that salmonids can build redds where the average substrate size (D_{50}) is up to 10 percent of average body length (Kondolf and Wolman 1993). Recommended average sizes for spawning gravels are listed in Table HG-1.

Table 7-5. Spawning gravel criteria for salmonids.

Gravel Bed Criteria	Small-Bodied Salmonids (<13.8 in) (<35 cm)	Large-Bodied Salmonids (>13.8 in) (>35 cm)
Dominant substrate particle size	0.3–2.5 in (8–64 mm)	0.6–5 in (16–128 mm)
Minimum gravel patch size	10.7 ft2 (1.0 m2)	21.5 ft2 (2.0 m2)

Note: Small-bodied salmonids include cutthroat trout. Large-bodied salmonids include coho and Chinook salmon and steelhead trout (after Schuett-Hames et al. 1996).

Embedding also reduces prey for foraging juvenile salmon by promoting a shift from epibenthic to benthic infaunal macroinvertebrates, which are not easily preyed upon by young salmonids (Bash et al. 2001; Suttle et al. 2004).

Although far less is known about the effects of changes in substrate on the life-history stages of other freshwater fish and invertebrates than on salmonids, a few studies have been done.

- White sturgeon prefer gravel and cobble substrate for spawning because their adhesive eggs are susceptible to burial by sand and silt-sized substrate (Paragamian et al. 2001).

- Mountain suckers in Lost Creek, Utah, showed a preference for spawning depths of 4.3–11.8 inches (11–30 cm) (Wydoski and Wydoski 2002).

- The deposition of fine sediment can adversely impact invertebrates (Wantzen 2006). Fine sediment particles may clog biological retention mechanisms such as the filtering nets of caddisfly larvae, or the filtering organs of mollusks.

- Overburden from increased deposition has been shown to adversely affect invertebrates having low motility (Hinchey et al. 2006).

- Sediment deposition can impair the growth and survival of filter-feeding organisms or organisms living on the substrate (Bash et al. 2001) by filling interstitial spaces needed for respiration and feeding.

- In freshwater mussels, Tucker and Theiling (1998) described a study in which fine sediment (silt) deposition of as little as 0.25 inch (6.35 mm) caused death in mussels. Siltation also is detrimental to young mussels and reduces their survival (Scruggs 1960, in Tucker and Theiling 1998). Juvenile survival (even of hardy species) may be reduced in silt-impacted mussel beds, which can limit recruitment of young in the entire bed (Tucker and Theiling 1998). While the exact mechanisms are not known, it is clear that siltation causes changes in water flow through the gravel and results in a shift in algal and microbial communities (Tucker and Theiling 1998).

- Different mussel species show varying responses to fine sediment inputs (Brim Box and Mossa 1999). Freshwater mussels are nearly sedentary filter feeders and occupy stable gravel substrate; therefore, they are sensitive to changes in channel hydraulics and sediment transport. Erosion of suitable substrate could dislodge the animals (Brim Box et al. 2004). McDowell (2001, in Brim Box et al. 2004) found that populations of western pearlshell (Margaritifera falcata), a freshwater mussel, were denser in reaches of the Middle Fork John Day River having no channel modification compared to modified reaches.

7.4.2.1.5 Altered Ground Water/Surface Water Interactions

The exchange of groundwater and stream flow through the hyporheic zone can provide several important ecological functions, including retention and storage of water, regulation of water releases to streams, promotion of habitat complexity, regulation of stream temperatures, refuge for fish eggs and invertebrates, and nutrient enrichment (Bolton and Shellberg 2001).

Hydraulic and geomorphic modifications can result in altered groundwater/surface water exchange through several pathways. Changes in channel form can affect the interaction between groundwater and surface water. Principally, channel aggradation or downcutting leads to altered surface water elevations, which affects the groundwater/surface elevation and groundwater flux to the channel. Bank erosion and substrate alterations can also alter these dynamics.

HPA-permitted structures that alter groundwater dynamics in riverine systems can directly affect fish and invertebrates in the short-term by influencing water quality and habitat suitability or availability. In the long-term, changes to groundwater exchange can generate indirect effects on fish and invertebrate species by affecting low flow conditions (i.e., increasing the magnitude of periods of drought resulting in reduced habitat availability and suitability, potential stranding or desiccation), and by affecting water quality through warmer stream temperatures and decreased organic and nutrient inputs.

The interplay between groundwater and surface water in the hyporheic zone has become increasingly recognized as a key process in the ecological functioning of riverine ecosystems. Changes in flow regime, sediment transport, and substrate composition all affect in-channel hyporheic exchange. In riverine environments, connectivity is generally expressed in three dimensions: longitudinally (upstream–downstream), laterally (channel–floodplain), and vertically (channel–hyporheic zone [the interface between surface and groundwater]) (Edwards 1998; Stanford and Ward 1992). The quality of habitat connectivity in one dimension may affect that in another dimension. For instance, the hyporheic zone serves as a medium across which dissolved organic matter and nutrients are exchanged between the riparian zone and surface water. A high level of substrate fines within the channel substrate may hinder the connection between surface and groundwater, limiting vertical and lateral connectivity (Edwards 1998; Pusch et al. 1998).

The presence of large woody debris in channels has been linked to increased hyporheic exchange (Mutz and Rohde 2003). The addition of LWD to channels has been shown in most cases to increase channel complexity. Log jams can cause stream flow to separate, (Abbe and Montgomery 1996), and part of the flow may be directed into the bed and banks of the channel. While a study by Sweka and Hartman (2006) found that large woody debris additions to eight Appalachian streams did not increase pool area, a number of other studies have shown that LWD presence increases pool frequency (Baillie and Davies 2002; Beechie and Sibley 1997) and area (Brooks et al. 2004; Cederholm et al. 1997; Hilderbrand et al. 1997). Increased pool density will be accompanied by an increase in pool-riffle transition zones. These areas are “hot-spots” of hyporheic exchange because head differential through the transition zone forces surface waters through the stream bed (Tonina and Buffington 2007). Consequently, through pool creation, LWD additions can increase hyporheic exchange rates. Conversely, the removal of LWD will decrease pool density (Ensign and Doyle 2005), act as a catalyst for incision (Diez et al. 2000), and thus reduce hyporheic exchange throughout the channel.

Lack of connectivity can degrade conditions for riparian zone vegetation, reducing LWD recruitment to the stream channel and subsequently limiting habitat-forming and maintaining processes and habitat complexity. Ecological connectivity is essential between riverine and riparian ecosystems (Kelsey and West 1998; Stanford and Ward

1992). Effects on ecological functions and freshwater aquatic species associated with degraded groundwater/surface water connectivity are well documented (Bilby and Bisson 1998; Hershey and Lamberti 1992; Karr 1991; Kelsey and West 1998; Montgomery et al. 1999; Naiman et al. 1992; Reiman and McIntyre 1993; Stanford and Ward 1992; Stanford et al. 1996).

Stream temperature is an important factor in determining the suitability of habitats for aquatic species. The interface between flow within the hyporheic zone and the stream channel is an important buffer for stream temperatures (Poole and Berman 2001a), so alteration of groundwater flow can affect stream temperature. The magnitude of the influence depends on many factors, such as stream channel flow patterns and depth of the aquifer (Poole and Berman 2001a).

The preferential selection of spawning substrates in groundwater upwelling zones is a common behavior among all HCP salmonid species (Baxter and Hauer 2000; Berman and Quinn 1991; Bjornn and Reiser 1991; Ebersole et al. 2003; Geist 2000; Geist and Dauble 1998; Geist et al. 2002; Greig et al. 2007; Zimmermann and Lapointe 2005). The disruption of flow through the hyporheic zone can affect fish spawning.

- In Montana, the distribution and abundance of bull trout is influenced by hyporheic and groundwater–surface water exchange (Baxter and Hauer 2000).
- Female bull trout tend to choose areas of groundwater discharge (i.e., cooler temperatures) for locating their spawning, and upwelling sites serve as important thermal refugia for all life-history stages (Baxter and McPhail 1999).
- Geist (2000a, 2000b) found that fall Chinook salmon chose spawning sites in the Hanford Reach of the Columbia River where groundwater was upwelling; where there was no upwelling, no spawning activity occurred. The dissolved oxygen content of upwelling groundwater was 9 mg/L, but only 7 mg/L or less where there was no hyporheic discharge (Geist 2000a, 2000b).

HPA-permitted activities that adversely affect groundwater upwelling may limit the availability and suitability of spawning and thermal refuge habitats.

Increased vertical exchange between surface and subsurface waters benefit aquatic biota by increasing benthic dissolved oxygen levels and promoting solute uptake, filtration, and transformation. Studies have shown that the availability of dissolved oxygen to incubating salmonid embryos is dependent upon hyporheic exchange (Geist 2000; Greig et al. 2007) and that the occlusion of this exchange through siltation can lead to hypoxia within redds and decreased embryo survival (Heywood and Walling 2007). The hyporheic zone does more than promote oxygen exchange in subsurface sediments, it can also act as an effective filter and zone of biogeochemical transformations.

Hyporheic exchange has been shown to influence water quality and food web productivity in flowing water ecosystems at multiple levels (Anbutsu et al. 2006; Ensign and Doyle 2005; Fernald et al. 2006; Jones et al. 1995; Lefebvre et al. 2005; Mulholland et al. 1997; Sheibley, Duff et al. 2003; Sheibley, Jackman et al. 2003; Tonina and Buffington 2003; Tonina and Buffington 2007; Triska et al. 1989; Valett et al. 2005). Increased hyporheic exchange has been associated with nutrient uptake (Anbutsu et al.

2006; Sheibley et al. 2003) and transformation (Fernald et al. 2006; Lefebvre et al. 2005), and may attenuate the transport of dissolved and particulate metals (Gandy et al. 2007). Elevated metals and nutrients can both have negative ramifications for fish and invertebrate health.

Any activity which impacts the functioning of the hyporheic zone, such as the removal of LWD, could impose an array of stressors on HCP species occurring in the affected environment through a number of related impact mechanisms. Hydraulic and geomorphic modifications that alter hyporheic zone functions are likely to impose some level of indirect effects on aquatic habitat conditions. By extension, this suggests the potential for adverse effects on HCP species dependent on these environments.

7.4.2.2 Marine Environments

7.4.2.2.1 Altered Wave Energy

The redistribution of wave energy can have a number of interrelated indirect and direct impacts on fish and invertebrates. Alterations to wave energy can cause changes in substrate and alter water column characteristics. Waves produce motions and induce transport both in the water column and near the seabed that are capable of transporting particulates large distances (Liang et al. 2007; McCool and Parsons 2004). Altering these mechanical processes alters transport rates (Liang et al. 2007; McCool and Parsons 2004). Wave action creates complex littoral habitat by removing fine or silty sediments (Beauchamp et al. 1994). Wave action may also be a source of desirable spawning substrate.

Wave energy is the dominant source of fluid mechanics in the nearshore area in most of Washington waters (Finlayson 2006), responsible for mixing the upper portion of the water column (Babanin 2006) and producing high shear stresses near the bed (Lamb et al. 2004). Shear stress is the force applied to the bed and also related to the intensity of the turbulence in the water column. Reduction in wave energy from natural levels lowers the near bed shear stress, resulting in the deposition of finer sediments (Miller et al. 1977). Considering the large volume of fine-grained sediment supplied to western Washington waters (Downing 1983), even areas that are not part of an active littoral cell can receive a large amount of fine sediment.

Attenuation of waves can increase water column stratification in marine waters and lead to dissolved oxygen reduction and temperature anomalies (Qiao et al. 2006). Surficial mixing and circulation also play an important role in primary productivity, particularly near large river mouths (e.g., Willapa Bay [Roegner et al. 2002]). Disruption of mixing and circulation may adversely affect primary productivity and marine species through the disruption of food web dynamics.

Changes in wave energy across substrates determine the size and distribution of sediments and associated detritus (Nightingale and Simenstad 2001b). Throughout Puget Sound, Hood Canal, and Washington’s coastal estuaries, variations in the interface between bottom slopes, wave energy, and sediments build beaches, nearshore substrates, and habitats unique to the climate, currents, and conditions of specific sites (Nightingale and Simenstad 2001b). Although specific characteristics of the factors at play vary with the geology of each region or subsystem, changing the type and distribution of sediment will generally alter key plant and animal assemblages (Nightingale and Simenstad 2001b).

Alterations in the natural distribution of wave energy can prove harmful to aquatic vegetation as well as the fish and invertebrates that use and consume them (Eriksson et al. 2004; Sandstrom et al. 2005). Wave energy plays a role in the distribution of aquatic vegetation used by salmonids and other nearshore fishes, particularly in energetic environments. High wave energies have been shown to inhibit the colonization and growth of some seagrasses (e.g., eelgrass) (Fonseca and Bell 1998); although in more recent studies in Puget Sound, no correlation was found between eelgrass prevalence and wave characteristics (Finlayson 2006). High wave energy can also dislodge kelp (Kawamata 2001).

The only direct impact of extreme wave energy would be on those invertebrates that cannot tolerate extremely high shear stresses or burial. If the shear stress exceeds the force securing invertebrates to the seabed, they become entrained into the water column and destroyed. Intense turbulence may also disrupt migration of fish. Experimental evidence of the mortality limits of large shear stresses on mollusks or other invertebrates is not available.

Increased wave energy may suspend and redistribute sediments, which may result in burial of invertebrates. Olympia oysters, the only marine HCP invertebrate species prone to this sort of burial, have been shown to be intolerant of siltation and do best in the absence of fine-grained materials (WDNR 2006b). The partial and complete burial of closely related estuarine mollusks has been addressed empirically (Hinchey et al. 2006). Results of these studies indicate that species-specific responses vary as a function of motility, living position, and inferred physiological tolerance of anoxic conditions. Mechanical and physiological adaptations contribute to this tolerance. Motile organisms are much more capable of surviving high sedimentation rates than sedentary ones such as the Olympia oyster. Survival of each species examined appeared to decrease exponentially with increasing overburden stress (i.e., depth of burial), with most species being killed once they were completely buried. Most shorelines in Washington do not experience the sedimentation rates that result in burial-related mortality. However, near river mouths, alterations in sedimentation rates are possible that would exceed the criteria for mortality established by Hinchey et al. (2006).

Wave and current interactions in shallow water (depths less than 3 feet) are particularly important to intertidal flora and fauna. For example, along the shallow edge of the tidal water, high suspended sediment concentrations may flow over a mudflat. This passage across the intertidal area potentially deposits large quantities of sediment and nutrients on upper mudflat areas, particularly at slack water (Christie and Dyner 1998, in Nightingale and Simenstad 2001b). These are part of the sedimentation and water transport processes that shape the geomorphology and consequently the plant and animal communities that rely on the shallow, soft sediment habitats of mud and sandflats (Nightingale and Simenstad 2001b).

Reducing wave energy from natural levels lowers the near bed shear stress, resulting in the deposition of finer sediments (Miller et al. 1977). Considering the large volume of fine-grained sediment supplied to western Washington waters (Downing 1983), even areas that are not part of an active littoral cell can receive a large amount of fine sediment. Deposition of large amounts of fine sediment can kill aquatic vegetation vital to nearshore HCP species. Recent work has shown that burying eelgrass at depths as little as 25 percent of the total plant height could decrease productivity and increase the mortality of eelgrass (Mills and Fonseca 2003). Eelgrass can also be discouraged from colonizing new areas with a high clay content as a result of recent sediment deposition (Koch 2001; Koch and Beer 2006).

7.4.2.2.2 Altered Current Velocities

In marine systems, reduced current velocities lead to the deposition of fine sediment (silt and clay) (Miller et al. 1977), particularly near major sources or sediment such as large rivers (Downing 1984). Altered sedimentation due to reduced current velocities could result in reduced spawning success, burial of organisms or habitats, and altered primary productivity. At the other extreme, strong currents can have significant impacts on both aquatic vegetation and the substrate it grows on or in. The relationship between flow velocity and a change is reflected through the boundary shear stress (Miller et al. 1977). Substrate and aquatic vegetation will be removed if the critical shear stress is exceeded.

The sensitivity of aquatic vegetation to altered current velocities is species-dependent and also dependent on other factors such as pollutant loading. Eelgrass and many other species of aquatic vegetation (e.g., bull kelp) require some water motion for survival (Fonseca and Bell 1998). It is likely that reduction in water velocity contributes to a lack of eelgrass.

In general, alterations in current velocities can contribute to modifications or removal of suitable habitats for fish in various lifestages. This alteration of habitat can inhibit the growth, survival, and fitness of various fish species. In addition, altered current velocities may also affect the exertion levels required for fish to move throughout the habitat. These changes could reduce the fitness required for migration or maintenance of normal behavioral functions or could result in direct mortality via direct burial and loss of suitable spawning or foraging habitat or indirect mortality resulting from impacts on prey species. Nearshore currents, even those in heavily altered environments, do not exceed the threshold for adult salmonid navigation, but high current velocities have been shown to exclude some small fishes from navigating nearshore waters (Michny and Deibel 1986; Schaffter et al. 1983). This would cause fragmentation of habitat for these species.

Alterations current velocity could alter transport and increase the mortality of planktonic spawn (e.g., rockfish). Altered currently velocity could directly impact those invertebrates that cannot tolerate extremely high shear stress or burial.

7.4.2.2.3 Altered Nearshore Circulation

Nearshore circulation is the flux of salt, water, and sediment associated with tidal and wave motion near the shoreline. In more exposed, sandy settings, such as the outer coast of Washington, nearshore circulation is dominated by the mechanics of wave breaking (Komar 1998). The effects of breaking waves are generally insignificant in Puget Sound (Finlayson 2006). In Puget Sound and near the mouth of large rivers such as the Columbia, tidal currents and freshwater input play a more important role in nearshore circulation. Tidal motions are rarely sufficient to mobilize material of gravel size or larger (Finlayson 2006), but they can mobilize fine sediments such as silt and clay, particularly in areas of high sediment supply (Nittrouer 1978).

Nearshore circulation patterns are a dominant characteristic that shapes the suitability of nearshore habitats for a range of HCP species. Alteration of nearshore circulation patterns can produce many of the same effects as alterations to wave energy or current velocities. Specifically, fish and invertebrate species that are planktonic breeders have been shown to produce spatially variable spawn that relies on the combination of wave motion, ambient currents, and circulation patterns for transport to and retention in productive nursery areas (Hernandez-Miranda et al. 2003; Rooper et al. 2006; Sinclair 1992). While studies on HCP species are lacking, virtually all of the purely marine HCP species such as herring, rockfish, pollock, and cod, have a planktonic egg and/or larval life-history stage dependent on wave and current patterns for transport to and/or retention in areas favorable for rearing. Developing larvae that are transported away from areas favorable for rearing before they are ready for life in open water face an increased risk of starvation and predation or, in the case of schooling pelagic species, may be permanently isolated from their spawning population (Sinclair 1992).

7.4.2.2.4 Altered Sediment Supply

Washington State contains thousands of miles of shorelines, including about 2,000 miles in Puget Sound alone. Much of this shoreline consists of poorly consolidated bluffs of glacial sediments faced with mixed sand and gravel, and some cobble. Erosion and occasional landslides on these bluffs provide the greater volume of sediment on Puget Sound shores compared with sediment delivered by rivers and streams (MacDonald et al. 1994). Local geomorphology, weather, fetch, and sediment sources determine the volume, timing, and direction of sediment transported past an individual beach. Shoreline sediment transport occurs along generally discrete segments ranging from a few hundred feet to several miles. These shoreline segments, called drift cells, include sediment source areas, sediment transport areas, and depositional areas. Sediment sources are the low and high bluffs that “feed” the beach with sand and gravel. Through littoral drift, sediments are transported along the shoreline. Actively eroding bluffs contribute to habitat conditions throughout the drift cell they support. The direction of drift within a drift cell may reverse between winter and summer as prevailing wind and wave directions change, causing sand to redistribute among beach areas (Cox et al. 1994).

Alteration of sediment transport patterns by HPA-permitted structures can present potential barriers to the natural processes that build spits and beaches and provide substrates required for plant propagation, fish and shellfish settlement and rearing, and forage fish spawning (Parametrix and Battelle 1996, Penttila 2000b, Thom et al. 1994, all in Nightingale and Simenstad 2001b; Thom and Shreffler 1996; Thom et al. 1996, 1997; Haas et al. 2002). Related wave energy and water transport alterations alter the size, distribution, and abundance of substrate and detrital materials required to maintain the nearshore food web (Thom et al. 1994). Pacific salmon, Pacific herring, surf smelt, sand lance, and a variety of other fish may be affected by habitat changes due to structures that affect littoral drift (Thom et al. 1994). Surf smelt, sand lance, and Pacific herring are important prey items for salmonids and other piscivorous species, therefore the impacts to these “forage fish” would extend up the food chain to other potentially covered species.

Alterations to littoral drift can also affect the beach profile (Thom et al. 1994). Changes in littoral drift that reduce sediment supply can make beach slopes steeper and increase erosional processes, especially in shorelines hardened by development resulting in a coarsening of the beach substrate, which can substantially interfere with the quality and quantity of intertidal forage fish spawning habitats (Thom et al. 1994).

Experimental investigations by Shteinman and Kamenir (1999, in Nightingale and Simenstad 2001b) demonstrate how the construction of jetties and other in-water structures can partially or completely disrupt the longshore transport process. In a natural hydraulic regime, size separation of sediments proceeds along the bottom slope with wave flow impact, and steep-sloped bottoms move larger sediments toward the shore, accumulating a thin nearshore strip along the shoreline. While smaller sediments were found to move toward deeper areas, where they accumulate or were further transported by currents, the opposite was found to occur on gentle bottom slopes, where smaller sediments accumulated near the shore and coarser sediments were moved toward the deeper areas.

One primary direct impact on fish and invertebrates from altered sediment supply is to alter the degree of turbidity in the nearshore environment (Au et al. 2004; Bash et al. 2001; Berry et al. 2003).

Benthic habitat may be impacted by alterations in natural sediment movement. For instance, a structure that interferes with littoral drift cells poses the risk of interference with the deposition of fine sediments to adjacent beaches that support beach spawning forage fish, such as surf smelt and sand lance (Nightingale and Simenstad 2001b). Limiting the fine sediments deposited to adjacent beaches also poses the risk of limiting the establishment of rooted vegetation, such as eelgrass, along submerged areas of adjacent shorelines and therefore the risk of reducing the available habitat for fish and shellfish species that rely on such vegetated habitats for spawning and rearing (Nightingale and Simenstad 2001b). The manner in which a structure is used by vessels determines additional effects of wave energy from vessel traffic and other effects such as vessel pollutant distribution or impacts to other adjacent shoreline structures (Nightingale and Simenstad 2001b).

Modifications to littoral drift have numerous indirect results, from substrate changes (Li and Komar 1992; Frihy and Komar 1993; El-Asmar and White 2002) to modifications in the distribution and delivery of groundwater to the coastal zone (Nakayama et al. 2007). The primary indirect impact of changing sediment supply by changing littoral drift, is to change the distribution of substrate within the littoral cell (Terich 1987). The loss of sediment to a drift cell results in a coarsening of the substrate, as fine-grained sediment is lost to deep portions of the basin by resuspension (Finlayson 2006) and not resupplied by freshly eroded bluff sediments. Because some drift cells can be extremely long (e.g., more than 20 miles long in the drift cell that extends between Seattle and Mukilteo on the northeastern shore of the main basin of Puget Sound [Terich 1987]), the effects of a modification can extend well beyond the primary activity area.

7.4.2.2.5 Altered Substrates

On the outer Washington coast, substrate is loose, deep, sandy, and unconsolidated. In these areas, increased or displaced wave energy associated with HPA-permitted structures creates wholesale erosion of the shoreline (Miller et al. 2001). In protected, previously glaciated areas, the basin topography is complex and the coarse nature of the substrate slows down erosion dramatically (Nordstrom 1992). In these locales, a lag deposit can easily form a near bedrock-like shoreline (e.g., Foulweather Beach [Finlayson 2006]).

HPA-permitted projects can alter substrate composition either directly, by purposely placing substrates that differ from those that occur naturally at a site, or indirectly, by altering wave and current energy, precluding the contribution of sediments from uplands, interfering with drift cell sediment transport and deposition, and introducing new substrates that result in shell-hash. Adding immobile substrate changes the mechanics of water motion on the shoreline, increasing wave reflection (Komar 1998; Finlayson 2006) and eliminating exchange of water into and out of the shoreline (Nakayama et al. 2007). The placement of structures can have the effect of increasing substrate scour or limiting deposition of sediments that provide suitable habitat for HCP species. Placement of fill associated with HPA-permitted structures alters the slope and depths of intertidal habitats.

Shoreline structures can modify species assemblages and habitats in the vicinity of the structures. Placement of riprap in the nearshore generally encourages a shift toward hard-substrate, often invasive, species (Wasson et al. 2005). These changes can directly affect the reproduction, growth, fitness, and survival of multiple life-history stages of HCP fish and invertebrate species, or result in indirect effects by affecting the viability and distribution of their prey species.

It is possible that coarser substrate could benefit some HCP species, particularly when the substrate is submerged and essentially acting as an artificial reef (Pondella and Stephens

1994). Placing rocky substrate in areas where it does not naturally occur can sometimes provide habitat for rockfish, a group of marine fish that are typically associated with hard, reef-like structure. Artificial reefs have been known to attract rockfish, but in the case of bank armoring, the potential benefits for rockfish are largely unknown. Some species of rockfish occur along shorelines and these could benefit, while other species typically do not occur along the immediate shoreline where bank protection structures would be placed.

Substrate is an important factor controlling the growth of aquatic vegetation in Puget Sound (Koch and Beer 2006). Placement of fill often results in a direct loss of vegetated shallow-water, nearshore habitat that juvenile salmonids use for rearing and migration. In general, the addition of immobile substrate decreases habitat suitability for juvenile salmonids and changes the character of the shoreline that was previously conducive to their use (e.g., Knudsen and Dilley 1987; Peters et al. 1998; Schaffter et al. 1983). While data indicate that habitat use of riprapped banks by yearling and older trout species may be equal to or higher than natural banks, use by subyearling trout, coho, and Chinook salmon is lower (Beamer and Garland et al. 2002; Hayman et al. 1996; Henderson 1998; Knudsen and Dilley 1987; Peters et al. 1998; Schmetterling et al. 2001; Weitkamp and Schadt 1982). In Elliott Bay, Toft et al. (2004) found similar densities of juvenile salmonids at sand/cobble beaches and riprap sites in settings where the riprap extended only into the upper intertidal zone. When riprap extended to the subtidal zone, higher densities of juvenile salmonids were found along riprap than at sand/cobble beaches. Toft et al. (2004) hypothesized that the shallow-water habitats preferred by juvenile salmonids were compressed along the highly modified shorelines with steep slopes; therefore, their snorkel observations were able to record all juvenile salmonids present. In comparison, at the sand/cobble beaches, the slopes were gentler, the zone of shallow water was much wider, and densities were therefore lower because the fish were more spread out.

Surf smelt and sand lance rely on substrates ranging in size from sand to gravel for spawning. Usual spawning substrates consist of fine gravel and coarse sand, typical of the pebble veneer found throughout Puget Sound (Finlayson 2006), with broken shells intermixed in some cases (Thom et al. 1994). Surf smelt are quite susceptible to the effects of alterations on shoreline processes (sediment supply, transport, and accretion) due to their reliance on specific beach profiles and substrate compositions for successful spawning (Penttila 1978). Surf smelt make no attempt to bury their demersal, adhesive eggs but rely on wave action to cover the eggs with a fine layer of substrate (Thom et al. 1994). Therefore, changing the wave environment may also change the survivability of surf smelt spawn or suitability of the site for future spawning habitat. The importance of substrate to spawning has also been empirically demonstrated in the closely related Japanese surf smelt (Hirose and Kawaguchi 1998).

Pacific sand lance spawn in the high intertidal zone, on substrates varying from sand to sandy gravel. Sand lance also rely on sandy substrates for burrowing at night. Like surf smelt, sand lance spawning is susceptible to the deleterious effects of littoral alterations because sand lance rely on a certain beach profile and specific substrate compositions (Penttila 1995).

Benthic communities, including invertebrate populations, are impacted by sediment alterations (Nightingale and Simenstad 2001b). For instance, the Olympia oyster is an epibenthic filter feeder found throughout the inland waters of Puget Sound, as well as in Willapa Bay and possibly Grays Harbor (WDNR 2006b). They occupy nearshore areas on mixed substrates with solid attachment surfaces and are found from 1 foot (0.3 meter [m]) above mean lower low water (MLLW) to 2 feet (0.6 m) below MLLW; the larvae settle onto hard substrate such as oyster shells and rocks (West 1997; Baker 1995, both in WDNR 2006b). Olympia oysters are intolerant of siltation and do best on firm substrates (WDNR 2006b).

Newcomb's littorine snail is found primarily in association with a narrow band of nearshore intertidal habitat that contains certain marsh plant species (Larsen et al. 1995). Newcomb's littorine snail may lose suitable habitat if large substrate is placed on top of substrate in the upper intertidal and supratidal areas that could otherwise support the snail's pickleweed (Salicornia virginica) vegetation habitat. Because detailed reproductive and habitat needs are not known, it might be conservatively assumed that Newcomb's littorine snail is also subject to smothering or substrate changes.

Mitigation may be available for the change from fine to coarse substrates (to some degree), as demonstrated by some projects that attempt to restore sand and gravel substrates to areas exhibiting large substrate. Monitoring in these projects has documented that epibenthic crustacean salmonid prey benefit from smaller substrate both in density and diversity of species (Parametrix 1985; Simenstad et al. 1991). Similarly, Thompson (1995) found an increase in hardshell clam abundance following beach graveling. [Peer review comment: Sobocinski (2003) and Sobocinski et al. (2004) would be better references for effects on invertebrates in Puget Sound, but these studies were not included.]

7.4.2.2.6 Altered Groundwater Input

Submarine groundwater discharge has been documented to play an important role in the circulation of fluids and nutrients on many coasts throughout the world (Gallardo and Marui 2006; Johannes 1980; Michael et al. 2005). Most work on the subject has focused on the nutrient load that these waters supply to the coastal ocean in sandy, exposed coastal environments (Gallardo and Marui 2006). The importance of groundwater seepage to the macroecology of the deep ocean (i.e., benthic environments) is well known (Kiel 2006). Both hydrothermal vents and cold seeps are known to be "hot spots" of biological activity, a direct result of groundwater discharge (Kelley et al. 2002; Kiel 2006). However, the direct impacts of submarine groundwater discharge on the nearshore environment are less clear. Solid concrete walls and steel piles that allow no groundwater penetration likely have increased impacts compared to more porous artificial substrates (e.g., riprap).

Several important effects have been documented in Puget Sound. For example, the lack of groundwater discharge can lead to increased substrate temperatures at comparable tidal elevations (Dale and Miller 2007; Rice 2006). Another loss of function is the removal of the seepage face at low tide (Gendron 2005). The correlation of the top of the seepage face to the landward limit of eelgrass beds has been anecdotally established in Puget Sound (Finlayson 2006). Although not demonstrated in a systematic study, the loss of the seepage face, as observed by Finlayson (2006) and Gendron (2005), would likely increase the risk of desiccation of aquatic plants. Desiccation has been found to be the dominant control on the growth of eelgrass (Zostera marina) in the Pacific Northwest (Boese et al. 2005).

Some species (such as the Olympia oyster) are known to take advantage of freshwater seeps along marine shorelines (West 1997; Couch and Hassler 1990). For species that are reliant upon freshwater seeps in the marine environment, groundwater impacts could potentially pose direct effects; however, the direct effect of submarine groundwater discharge on fish and invertebrates in nearshore areas is unclear (Simmons 1992).

7.4.2.3 Lacustrine Environments

The hydraulic and geomorphic modifications in lakes, natural or man-made, have the same six mechanisms of impact as the marine environment (i.e., altered wave energy, altered current velocities, altered nearshore circulation, a loss of groundwater input, altered sediment supply, and altered substrate composition), albeit on a different suite of species.

The impacts of HPA projects in lacustrine environments bear some similarity to impacts on marine environments. In both environments wave energy, and sediment recruitment and transport are altered. However, in lakes, these impacts are often exacerbated by differences in human-controlled water-level variability (in the case of reservoirs) and natural lake limnology (Wilcox et al. 2002). This inherent variability makes the differences between natural lakes and reservoirs less pronounced with respect to nearshore processes. However, there are other geomorphic differences with pronounced effects on habitat.

7.4.2.3.1 Lakes

Systematic studies of impacts on the habitat in the lakes in western Washington are extremely limited (Jones & Stokes 2006). Some analysis of habitat types and species distribution has been prepared as part of the development of shoreline master programs, but these only catalog species and activity types and do not provide information about their relation to one another.

Fish respond to habitat characteristics resulting from the association of shoreline and riparian zone modification. In a study of Wisconsin lakes, the habitat characteristics most influenced by this association were depth, substrate size and embeddedness, and amount of woody vegetation and macrophytes (Jennings et al. 1999). Species richness was greatest where there was complexity in this suite of factors.

Habitat in lacustrine environments is impacted by large, long-term, water-level fluctuations. These can be related to natural hydrologic changes; or, as is often true on Washington's largest lakes (e.g., Lake Washington), these fluctuations can be produced by human manipulation of inlets and outlets. The effects of such manipulations can manifest in a manner similar to natural changes and may complicate any assessment of impacts arising from human activities (Wilcox et al. 2002).

The physical processes discussed in depth under the Marine Environments section are considered to be relevant in lakes, recognizing that some differences occur (mostly apparent from previous work performed in the Great Lakes). The most important hydraulic and geomorphic differences between marine and lacustrine environments are in nearshore circulation, groundwater input, and short-period waves.

7.4.2.3.1.1 Nearshore circulation.

While wave energy in lakes is small relative to most marine beaches, wind plays an important role in driving the circulation (Rao and Schwab 2007). Unlike in the marine environment, where salinity is typically the most important water column constituent, temperature is the dominant factor in maintaining stratification in lakes. The absence of tides means that water level in lakes on the time scale of hours to days is stable, and any terraces that are formed are much more pronounced and discrete. Stratification and isolation of low dissolved oxygen zones are more easily achieved near lakeshores than marine shorelines, affecting all lake-dwelling HCP species that are sensitive to low dissolved oxygen.

7.4.2.3.1.2 Groundwater input.

Because lakes are fundamentally more connected to upland environments, the limiting nutrients in a lake are different than in a marine setting. However, just as in marine environments, benthic productivity and diversity have been linked to groundwater effluent (Hagerthey and Kerfoot 2005; Hunt et al. 2006). Unlike marine environments, lacustrine seeps have high productivity but low species diversity (Hagerthey and Kerfoot 2005; Hunt et al. 2006). Therefore, lacustrine deepwater species such as pygmy whitefish are less likely to be affected by groundwater alteration than marine pelagic species (e.g., rockfish) to the same alterations.

7.4.2.3.1.3 Short-period waves.

Because lakes are confined, all of their natural wave energy is generated from local winds. This makes all of the waves fetch-limited (Komar 1998). Fetch-limited waves have extremely short periods and small wave heights, compared to their open, marine counterparts. In this sense, lacustrine littoral processes are more similar to those found in Puget Sound (Finlayson 2006). Therefore, alterations to shorelines will not be felt as far from project activities as if they were to occur in the marine environment. The size of area affected by lakeshore development has relevance for sockeye spawning habitat (WDNR 2006a).

While littoral processes are most conspicuous in marine waters, they can occur along lake shores as well, where fetch and wind speed combine to produce waves and subsequent longshore currents strong enough to move shoreline sediments.

7.4.2.3.2 Reservoirs

Human-operated reservoirs present special issues. Reservoirs are morphologically, biologically, and hydrologically dissimilar from natural lakes. Morphologically, lakes are often deepest near the middle, whilst reservoirs are typically deepest at the downstream end. This difference has implications for current strength and direction. The plan view of reservoirs can be quite variable, depending on the degree of confinement, but the length of the shoreline is often longer than that of a natural lake. Also, the extent of shoreline development is much greater than in natural lakes because annual drawdown exposes a larger area to shore processes by expanding the area alongshore exposed to wave breaking (Baxter 1977). The location and nature of depositional forms are highly variable with reservoir morphometry, incoming sediment load, and reservoir operation. Reservoirs are also subject to density or turbidity currents resulting from differences in temperature or sediment concentration between inflows and reservoir waters (Snyder et al. 2006). Mixing zones between the water sources influence the usage of reservoir areas by fishes.

Reservoir environments can lack natural habitat due to loss of riparian forest because of flooding, siltation of rocky shorelines, and a paucity of aquatic vegetation resulting from fluctuating water levels (Prince and Maughan 1978). Dependent on reservoir operations, drawdown and filling cycles can re-entrain silty deposits in littoral areas. When jetties, barbs, or breakwaters are constructed, the combined footprint of fill materials and pilings obliterates physical habitat and can exacerbate the degradation of littoral areas.

7.4.2.3.2.1 Nearshore Circulation

The presence of structures such as marinas that disrupt either the movement of fishes within the littoral zone or nearshore circulation may add to the inherent temperature stressor present in a reservoir. Littoral zones separated from the larger reservoir body may become significantly warmer and exhibit larger diel temperature fluctuations (Kahler et al. 2000). Similarly, structures that extend into the mixing zone may also present a physical barrier to the movement of fishes in and out of these zones. The presence of a jetty was found to restrict circulation between a discharge stream and receiving water (Altayaran and Madany 1992).

7.4.3 Activity-Specific Effects

7.4.3.1 Overwater Structures: Docks, Piers, Marinas and Shipping Terminals

Impacts on fish species associated with marina/terminal structures include decreased growth and survival, decreased developmental and migratory fitness, and direct mortality. Migration timing may also be affected for some fish species, ultimately affecting reproductive success. Marinas have been found to attract large populations of juvenile salmon and baitfish and provide permanent habitat for a variety of other fish (Cardwell et al. 1978; Heiser and Finn 1970; Penttila and Aguero 1978; Thom et al. 1988; Weitkamp and Schadt 1982). This attraction is likely due to the low hydraulic energy similarities between a marina environment and a natural embayment (Cardwell and Koons 1981).

Increased impervious surfaces associated with marinas and terminals are unlikely to produce damaging effects on peak and base flow conditions of the adjacent water bodies. Marinas and terminals are typically developed on larger rivers, lakes, and marine waters. Such water bodies are considered insensitive to the relatively small increase in impervious surface area and to the effects of flow perturbation imposed by impervious surfaces. This exemption applies in ESA consultations as well (WSDOT 2006d).

Depending on the geomorphology, current transport processes, and climatic conditions of a specific area, overwater structures have the potential to alter important habitat-building processes (Nightingale and Simenstad 2001b) such as sedmentation and water transport.

One of the most profound changes produced by marinas is to change the shoreline from a dynamic, loose surface to a rigid, immobile one. Although there are distinct differences between artificial substrates placed in marina construction, over time they all behave like bedrock shorelines similar to extremely coarse-clastic beaches in Puget Sound (Finlayson 2006). The primary difference between these installations is whether they permit exchange of groundwater with the sea. Submarine groundwater discharge has been documented to play an important role in the circulation of fluid and nutrients on many coasts throughout the world (Gallardo and Marui 2006; Johannes 1980; Michael et al. 2005;). When marinas are installed, the substructure that interrupts the free exchange of groundwater between the sea and the uplands has been shown to have adverse effects on nearshore ecosystems (Nakayama et al. 2007).

7.4.3.1.1 Impacts to Littoral Drift

In-water structures such as piers and pilings have the potential to block or divert littoral currents. Alteration of littoral currents can cause sediment deposition and reduce beach nourishment down-current from the structure (Thom et al. 1994). Changes in beach nourishment and sediment deposition can in turn alter benthic and epibenthic communities, as well as bank erosion rates (Thom et al. 1994). The significance of these effects depends on the location and orientation of the structures (Thom et al. 1994). Closely spaced pilings can collect sediment along the up-current side (Nightingale and Simenstad 2001b), but widely spaced pilings allow currents to flow freely and sediment transport is essentially unaffected (Nightingale and Simenstad 2001b). For pile groupings, the magnitude of bed scour depends on the pile diameter, the spacing between piles, the number of pile rows and their staggering, and the alignment of pile rows relative to the principal direction of flow (Salim and Jones 1999; Smith 1999). Ratte and Salo (1985) and Penttila and Doty (1990) found that pilings associated with shoreline structures changed the flow of water around the pilings and over the substrate, thereby altering the bathymetry of the substrate and the flow of water in the immediate area.

Open pile structures tend to interfere less with sediment transport. Structures located in low-energy areas that block littoral drift tend to fill in with sediment and require maintenance dredging.

Marinas are specifically designed to diminish ambient wave energy and current velocity so that maritime activities can be conducted. In the process of creating a shoreline that suits this purpose, ambient waves are reflected (Wurjanto and Kobayashi 1993), diffracted (Melo and Guza 1991), and refracted (Komar 1998). Vessel traffic associated with the addition of a marina can interact with these artificial boundaries, causing a significant increase in wave energy even in some places inside the marina (Tarela and Menendez 2002; Isaacson et al. 1996).

Numerous studies have been performed that have attempted to manipulate the incoming wave energy to reduce reflected, refracted, and diffracted wave trains from entering the port or marina; in fact, there are entire journals dedicated to this topic (e.g., Journal of Waterway, Port, Coastal and Ocean Engineering). These alterations typically result in the construction of a series of jetties, groins, and breakwaters. Regardless of the nature of the alterations, the modified relationship between topography and wave energy results in a shoreline that is out of equilibrium with natural shoreline processes (Komar 1998). As a result, wave energy artificially accumulates in some areas and is diminished in others. For example, due to reduced wave energy, marinas are likely to experience accumulations of fine sediments in excess of levels that existed prior to the modification of the site.

Pilings, navigation dredging, and prop wash associated with the construction, operation, and repair of marinas/terminals alter both the bathymetry and littoral drift of the area around and under such structures, both in exposed (Komar 1998) and sheltered settings (NRC 2001).

7.4.3.1.2 Substrate Alteration: Shell-Hash

Pilings can alter adjacent substrates, with increased shell-hash deposition from piling communities and changes to substrate bathymetry (Haas et al. 2002; Shreffler and Moursund 1999; Blanton et al. 2001). Pilings provide surface area for encrusting communities of mussels and other sessile organisms such as seastars that prey upon the shellfish attached to the dock. The resulting shell-hash accumulated at the base of the piling alters adjacent substrates and changes the substrate bathymetry (Blanton et al. 2001; Haas et al. 2002; Parametrix 1996; Penttila and Doty 1990; Southard et al. 2006). These changes in substrate type can also change the nature of the flora and fauna at a given site. In the case of pilings, native dominant communities typically associated with sand, gravel, mud, sand, and seagrass substrates are replaced by those communities associated with shell-hash substrates. Shell-hash is a prime settling habitat for Dungeness crab. Both crab and seastar foraging activity can disrupt eelgrass and retard recruitment. Crabs burrowing into the substrate to avoid predation may significantly inhibit eelgrass recruitment (Thom and Shreffler 1996). Such disturbance of seagrass meadows by animal foraging is also reported elsewhere (Baldwin and Lovvorn 1994; Camp et al. 1973; Orth 1975; Williams 1994).

7.4.3.1.3 Effects on wave direction and intensity

Overwater structures and piling can affect wave direction and intensity. The effects of piers and pilings on wave action depend on spacing, orientation, and number of pilings, as well as depth and proximity to shore (Fresh 1998, in Nightingale and Simenstad 2001b; Nightingale and Simenstad 2001b). Widely spaced piles in deep water have relatively little effect, as waves refract around them (Nightingale and Simenstad 2001b). In contrast, a series of pilings can reflect waves, resulting in reduced littoral currents (Nightingale and Simenstad 2001b). Floating structures can also attenuate waves and alter the intensity of wave action that cause and maintain littoral drift (Nightingale and Simenstad 2001b). The effectiveness of a floating structure as a wave attenuator depends on the shape, dimensions, and orientation of the structure (Cox et al. 1994).

Overwater structures normally have little capacity to alter channel gradient.

7.4.3.1.4 Vessel Wake and Prop Wash

Wakes from large commercial vessels have profound effect on shallow water habitats. Wave energy striking the beach/bank causes redistribution/suspension of sediments, bank erosion, displacement of shoreline vegetation and wood debris, and disruption to flora and associated fauna. While this has not been studied extensively, there is information available from the studies of ferry wakes in Rich Passage. The frequency of ferry traffic, sometimes every half hour throughout the day, makes prop wash effects at ferry terminals an exception to other docks. Prop wash and benthic disturbance by ferries are well documented for ferry terminals (Blanton et al. 2001; Francisco 1995; Haas et al. 2002; Michelsen et al. 1999; Olson et al. 1997; Parametrix 1996; Shreffler and Gardiner 1999; Thom et al. 1997; Thom and Shreffler 1996). Carrasquero (2001) and Kahler et al. (2000) provide a review of what is known about shoreline and overwater structure impacts in freshwater environments.

Prop wash and waves are also known to be a primary cause of shoreline erosion (Gatto and Doe 1987; Mason et al. 1993). The number of boats in a given area has been correlated with wave height (Bhowmilk et al. 1991), with areas of high boat traffic exhibiting increased levels of shoreline erosion. Although it is difficult to quantify boat wake contributions to shoreline erosion, boat traffic has been found to contribute up to 50 percent of the factors responsible for shoreline erosion in small rivers less than 2,000 feet wide (Hurst and Brebner 1969). Sutherland and Ogle (1975) found prop wash and increased turbidity from jet boats to decrease salmon egg survival by 40 percent. In addition to turbidity, direct contact with spawning substrate can cause mortality.

7.4.3.2 Navigation/Maintenance Dredging

Dredging may occur in navigation channels, in marinas, or near terminals. Navigation or maintenance dredging is by far the most frequent form of dredging in Washington State.

Dredging in marine environments converts intertidal habitat to subtidal habitat and shallower subtidal habitats to deeper subtidal habitats. Dredging affects the plant and animal assemblages that are uniquely adapted to the particular light, current, and substrate regimes of intertidal areas. By altering bathymetry and bottom substrates, such conversions produce a “trade-off” of intertidal and shallow-subtidal communities for deeper, subtidal communities.

In lacustrine environments, dredging converts shallow-water littoral habitats into deeper water environments and may create a steeper bathymetric transition. This change in habitat characteristics may change the size and species distribution of fish in the localized environment, altering predator/prey dynamics.

The effects of dredging on riverine environments are more complex still, because localized alteration of channel morphology can lead to dynamic shifts in channel form as the system adjusts to the changed conditions. These effects can extend a considerable distance beyond the bounds of the original dredging project.

Construction and maintenance of shipping access to marinas have been shown to both increase (da Silva and Duck 2001) and decrease (Sherwood et al. 1990) tidal prisms, depending on the characteristics of the tides and freshwater input and the nature and geometry of the alterations. The reduction of the tidal prism, as documented on the Columbia River (Sherwood et al. 1990), can eliminate entire habitats from being exposed to tidal action. In addition to stranding areas from marine influence, reduction in tidal motions can increase stratification and limit the vertical mobility of nutrients and dissolved oxygen (Mickett et al. 2004). Recent work has shown that there is a complex interplay among these phenomena and the primary productivity of nearshore waters; however, more dramatic consequences could occur in naturally mixing-limited waters of Puget Sound. Aside from the obvious impacts on inundation of adjacent landowners, increasing tidal prisms can expose aquatic species (both fish and invertebrates) to polluted sediments, such as those found at Superfund sites, potentially resulting in long-term contaminant related impacts.

There are several different means by which dredging affects fish and invertebrates, the most significant being alteration of bathymetry and substrate composition.

Large channel deepening projects can markedly alter ecological relationships through the change of freshwater inflow, tidal circulation, estuarine flushing, and freshwater and saltwater mixing. Miller et al. (1990) reported that only through comprehensive areal surveys over a minimum of four seasons before dredging, with follow-up surveys after dredging, could impacts of channel deepening on aquatic resources be determined. In a comparison between dredged and undredged areas in the Port of Everett’s public marina, Pentec (1991) found catches of fish to be higher in the dredged area before dredging than after dredging. Catches decreased from about 90 fish per tow to about 3 fish per tow and from eight species to five species.

Depending on site characteristics, maintenance dredging may occur annually or at intervals of 10 years or longer. These different dredging timelines represent different disturbance regimes both in terms of the ability of the benthos to recolonize prior to redisturbance and the magnitude of benthic productivity affected by dredging. In a literature review report on dredge and disposal effects, Morton (1977) reported the range of effects on invertebrate communities to be from negligible to severe, with impacts ranging from short to long term. In experiments conducted in sheltered sand flats, the benthic community recovered from lower intensity disturbance (i.e., sediment removal to a depth of 3.9 inches [10 cm]) within 64 days, whereas recovery from higher intensity disturbance (i.e., sediment removal to a 7.9-inch [20-cm] depth) required 208 days postdisturbance (Dernie et al 2002).

In a study to evaluate the effects of dredged material disposal on biological communities, Hinton et al. (1992) reported a significant increase in benthic invertebrate densities at a disposal site between June 1989 (predisposal) and June 1990 (postdisposal). Recolonization could have occurred by invertebrates burrowing up through newly deposited sediments or recruitment from surrounding areas (Richardson et al. 1977).

7.4.3.3 Bank Protection and Shoreline Modifications

In marine, riverine, and lacustrine systems, a reason for installing many bank protection structures and shoreline modifications is to alter hydraulic and/or geomorphic processes. Bank protection structures such as bulkheads and revetments are constructed parallel to the shore. Shoreline modifications such as jetties, groins, and breakwaters, project out from the shore.

Structures built to prevent bank erosion can alter the contribution of sediment to the aquatic environment. Structures that are constructed to protect upland properties from erosion can entrain fine sediments during construction, modify the substrate available to species for spawning and rearing by blocking the contribution of sediments and LWD to the shoreline from the uplands or from upstream areas (NMFS 2003), and increase the scouring of substrates. This scouring action can affect downstream or downcurrent habitats by transporting and depositing fine sediments, thereby compromising spawning habitat, burying potentially covered species, or increasing embeddedness of occupied habitats. It can also dramatically modify the types and abundance of substrates available to support aquatic vegetation.

7.4.3.3.1 Bank Protection in Riverine Systems

The intent of adding non-erodible substrate to a riverine system (e.g., riprap) is to stabilize channels and limit natural fluvial processes. The anthropogenic alteration of the river environment through the addition of bank protection or shoreline modification structures can disrupt the balance of the channel processes that form and maintain habitats throughout a river system (Fischenich and Allen 2000). Such structures have direct effects on river processes because they modify river channels and are designed to limit or prevent natural channel processes along the length of the structure. The disruption of channel processes is the most significant mechanism of impact generated by bank protection projects. Bank protection structures in or adjacent to channels can produce the following alterations to the channel processes and morphology:

- Channel straightening and shortening

- Channel narrowing
- Reduced habitat complexity
- Channel incision/increased scour
- Substrate coarsening
- Channel braiding/increased deposition
- Decreased floodplain connectivity
- Decreased channel migration and side channel creation
- Reduced LWD and organic material recruitment
- Reduced gravel recruitment
- Disrupted flow through the hyporheic zone[1]

7.4.3.3.1.1 Changes in Channel Hydraulics and Geomorphology

Bank protection structures, particularly those that are designed for flood control tend to straighten and shorten channels (Brookes 1988, in Bolton and Shellberg 2001). If a bank protection structure is placed below the ordinary high water level (OHWL), the channel is effectively narrowed or constrained, and disconnected from the floodplain. These types of changes to the channel result in reduced habitat complexity, especially when the removal of logs or snags will coincide with the placement of the structure (Bolton and Shellberg 2001). For example, in the Skagit River, a comparison of protected conditions to natural riverbank conditions showed that habitat complexity and off-channel refugia were higher along natural banks (Hayman et al. 1996). River sections with extensive bank protection structures generally tend to create primarily glide habitat with poorly sorted substrates (Bolton and Shellberg 2001).

An associated outcome of the disconnected floodplain is the limitation of lateral channel migration. The lateral migration of rivers, as well as riparian succession, is a necessary process for the maintenance of appropriate energy levels in a system, and thus promotes habitat diversity (Fischenich 2001). Reduction in channel migration tends to limit the creation of complex main channel and side channel habitats (Beamer et al. 2005). If a bank protection structure is installed when the channel alignment is unstable, the structure will attempt to maintain that alignment (Saldi-Caromile et al. 2004), which may reduce the structure’s effectiveness.

Bank protection structures that constrict the channel generally lead to greater increases in velocities along the length of the structure compared to structures that do not constrict the channel (Fischenich 2001). Channel constriction can lead to incision or downcutting of the channel as erosion occurs across the entire channel bed at the constriction (Cramer et al. 2003). The intrinsic ability of flow to transport sediment increases in a deepened channel, which can result in a coarsening of substrates within and downstream of a constricted section (Naiman and Bilby 1998). Such increases usually have no effect on

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[1] Hyporheic zone is a broad term that defines the “saturated interstitial areas beneath the stream bed and into stream banks that contain some proportion of channel water or that have been altered by channel water infiltration (advection)” (White 1993, in Bolton and Shellberg 2001).

the average cross-sectional velocity; rather, there is a redistribution of velocities, such that higher velocities occur adjacent to the structure (Fischenich 2001).

Channel incision also occurs if the bank protection structure or material (e.g., riprap) reduces channel roughness and generates an increase in water velocity and turbulence near the bank protection structure (Fischenich 2001; Miller et al. 2001). The increased scour and channel incision usually occurs along the toe of the structure and/or immediately downstream (Fischenich 2003), and may extend into the stream approximately two to three times the scour depth (Fischenich 2001). Scour may occur as a short-term or long-term outcome of having a bank protection structure in place, but the impacts tend to persist over an extended period of time (Fischenich 2001). Hardened banks that replace riparian vegetation can increase the flow velocity and potential for scour and substrate coarsening through a reduction in hydraulic roughness compared to vegetated conditions (Millar and Quick 1998). Because of their stability and low hydraulic roughness, hardened banks can act as natural attractors for channels and result in a static channel form lacking habitat diversity (Dykaar and Wigington 2000).

These impacts to channel processes often occur in areas beyond the immediate extent of a bank protection structure. The type and extent of the alterations depend upon the geomorphic and hydrologic setting of the river (Bolton and Shellberg 2001). For example, an alluvial river system with a channel bed and banks comprised of sediments will more easily incise and scour than a channel over bedrock.

Additional sediment movement associated with increased scour and channel incision can result in increased volumes of sediment deposited at some distance downstream. The downstream river setting, including slope, floodplain width, and flow volume, as well as the volume of bedload material transported downstream, influences where the material is deposited and what impacts it may have on habitat and species. Similarly, areas upstream of bank protection structures may also encounter sediment deposition if associated channel narrowing backs up water to some extent. Such sediment deposition could contribute to upstream river instability, which could threaten land, including the parcels with bank protection.

Bank armoring with non-erodible substrate can coarsen the bed by directly adding material coarser than the ambient bed and through the attendant effects of channel homogenization. Substrates larger than those occurring naturally are often placed in or along water bodies as part of bank protection projects. Placement of large rock that remains stationary (i.e., is non-erodible) during high flows is more often a component of hard bank protection techniques than soft or integrated techniques. The size of the material placed, the substrate covered, and other environmental conditions determine the degree to which substrate-dependent functions are impacted. Because potentially covered species depend upon aquatic substrates for life history and habitat functions, impacts to substrates ultimately affect the species' distribution and ability to grow and survive. Available studies on the impacts of adding non-erodible substrates are primarily focused on the effects of riprap on salmonids.

7.4.3.3.1.2 Changes in Habitat

The addition of large, angular rock to banks is known to affect salmonid habitat and abundance. Knudsen and Dilley (1987) found that abundance of juvenile salmonids was reduced by bank reinforcement activities due to a loss of structural diversity and that these reductions were correlated with the severity of habitat alteration, the size of the stream, and the size of the fish. In a study from California, the primary cause for the decline of salmon in the Sacramento River was linked to the loss of spawning gravels normally derived from bank erosion before riprap bank stabilization (Buer et al. 1984). A comparative study in several western Washington streams found that salmon abundance was less along banks modified with riprap compared to natural banks containing vegetation and woody debris (Peters et al. 1998). Studies comparing the abundance of fish in areas of different size riprap correlate greater fish densities with larger rock (Beamer and Henderson 1998; Lister et al. 1995; Garland et al. 2002). Lister et al. (1995) found that juvenile salmonid densities were greater along banks with riprap greater than 1 foot (30 cm) median diameter compared to natural banks composed of cobble–boulder material presumably due to the cover provided by the relatively larger interstitial spaces created by the coarser bank protection. Indirect effects on fish from bank hardening (i.e., loss of temperature moderation and potential cover) can occur due to the replacement of riparian vegetation with rock (Chapman and Knudsen 1980).

The addition of large substrate for bank protection would generally negatively impact habitat for cold-water species that use shallow margin habitats for feeding and refuge (Fischenich 2003), but would positively impact species that are associated with rock structure and interstitial spaces. Generally, species benefiting from the placement of rock may be non-native species that are piscivorous (e.g., brook trout) (Schmetterling et al. 2001).

In general, the addition of artificial substrates will decrease habitat suitability for juvenile salmonids and will change the character of the shoreline that was previously conducive to their use (Knudsen and Dilley 1987; Li et al. 1984; Peters et al. 1998; Schaeffter et al. 1983, in USFWS 2000), whereas for fish found in the interstices or relying on prey found there (e.g., sculpin), artificial substrates can increase habitat availability and usage (Li et al. 1984). While data indicate habitat use of riprapped banks by yearling and older trout species may be equal to or higher than natural banks, use by sub-yearling trout, coho, and Chinook salmon is lower (Beamer and Henderson 1998; Garland et al. 2002; Hayman et al. 1996; Knudsen and Dilley 1987; Peters et al. 1998; Schmetterling et al. 2001; Weitkamp and Schadt 1982). Knudsen and Dilley (1987) found that abundance of juvenile salmonids was reduced by bank reinforcement activities due to a loss of structural diversity and that these reductions were correlated with the severity of habitat alteration, the size of the stream, and the size of the fish. Size of material is also relevant, as greater fish densities have been generally correlated with larger rock (Beamer and Garland et al. 2002; Henderson 1998; Lister et al. 1995). Lister et al. (1995) found that salmonid densities were greater along banks with riprap greater than 1 foot (30 cm) median diameter compared to natural banks composed of cobble-boulder material.

Kahler et al. (2000) noted that bulkheads that are nearly vertical and constructed of large boulders with large interstitial spaces can provide concealment to piscivores. No studies documenting the occurrence of increased predation of juvenile salmonids in riprap areas were identified. However, a study of fish diets in the Willamette River (Portland, Oregon) found that smallmouth bass (Micropterus dolomieui), yellow perch (Perca flavescens), and other centrarchids captured at riprap locations (although not likely to occupy interstitial spaces) were more likely to have fish in their stomachs than the same species captured along natural shorelines (Vile et al. 2004). Sculpins are piscivores that occupy interstitial spaces and, when larger than approximately 2 inches (50 millimeters [mm]), can prey upon juvenile salmonids (Tabor et al. 1998). Based on the Tabor et al. (1998) observation that more and larger sculpin were found in locations with larger substrates, Kahler et al. (2000) infers that increased predation to juvenile salmonids may occur in those areas.

These patterns in juvenile salmonid habitat use are generally attributed to the impacts of the bank protection material on localized hydraulics, substrate, and available food and cover for fish at stream sites where hard bank protection materials are used. Rock riprap can disrupt flows, reduce food delivery, and create difficult swimming for small fish (Michny and Deibel 1986; Schaffter et al. 1983). In addition, riprap shorelines will be less likely than natural shorelines to retain wood at the bank for increased habitat structure (Schmetterling et al. 2001). Several researchers (Beamer and Henderson 1998; Michny and Deibel 1986; Peters et al. 1998; Schaffter et al. 1983) found that where large, complex wood deposits have been either maintained or incorporated into riprap, fish densities were higher than densities at sites without such structures. The mechanisms affecting why yearling salmonids occur in higher numbers in riprap areas are not well understood.

Little has been documented regarding impacts to invertebrates from bank protection and shoreline stabilization projects. The addition of riprap usually results in an increase in macroinvertebrate biomass and density of those species using interstices and hard substrates (Fischenich 2003). The Western ridged mussel lives in small substrates that would be less available in areas where bank protection structures add large substrate.

7.4.3.3.1.3 Altered Groundwater/Surface Water Exchange

Bank protection structures can disrupt exchange of groundwater and surface water in the hyporheic zone by creating a physical barrier (Fischenich 2003). Only some sorts of bank protection require the use of structures such as pilings or other impermeable structures that impede the exchange of hyporheic water with main river channels. When such structures are necessary, ecological impacts associated with the loss of exchange of hyporheic flow occur.

7.4.3.3.2 Bank Protection in Marine Systems

Bank armoring is sometimes placed in marine environments to alter wave energy that would otherwise cause erosion of a stretch of shoreline. However, it has also been observed that bank protection can, in the long run, increase or displace erosion. Bank protection may alter nearshore circulation in ways that are similar to (but generally less pronounced than) alterations caused by shoreline modifications such as jetties, breakwaters, groins and bank barbs. Bank protection structures and other artificial shoreline features can affect littoral drift through their influence on sediment supply and sediment transport.

7.4.3.3.2.1 Changes in Hydraulics and Geomorphology

Bank protection structures may decrease complexity of the shoreline by altering wave action in the littoral zone. Wave action creates complex littoral habitat by removing fine or silty sediments (Beauchamp et al. 1994). At marine shorelines, bulkheads have been shown to sort and coarsen existing substrate by increasing turbulence, wave reflection, and scour in front of the structure (e.g., Williams and Thom 2001). This often leads to a need for further supplemental armoring of foreshore and adjacent beach areas (Cox et al. 1994), often occurring in the form of additional riprap at the toe of the bulkhead.

Revetments tend to have slightly reduced impacts due to altered wave energy relative to vertical bulkheads because of the materials used and their configuration. Revetments are generally constructed of non-erosive material (e.g., riprap or quarry spall) that varies in size depending on water levels and wave energy of a specific site and are usually built to a slope of 1.5 or 2 horizontal units to every 1 vertical unit (Williams and Thom 2001). Because they slope, revetments can partially attenuate wave energy (the remaining energy is reflected) and water can filter through the rock material in the swash zone, protecting the underlying beach sediment. Although revetments can attenuate wave energy, sediment supply is still isolated from the littoral drift system, and the material used in the revetment replaces or covers naturally occurring substrates. However, revetments generally occupy a much larger footprint on the beach/shoreline than vertical structures.

Both during and after construction, bank protection projects have the potential to directly or indirectly modify substrate conditions. In marine environments, bank protection can cut off naturally eroding uplands (feeder bluffs) from beaches, potentially resulting in changes in substrate, size, composition, distribution of aquatic vegetation, and beach angle. A structure such as a bulkhead, if functioning correctly, prevents potential bank and bluff material from supplying the aquatic system (Johannessen et al. 2005). Along the Puget Sound shoreline, this disconnection or impoundment of natural sediment sources is possibly the most significant impact of shoreline protection measures (MacDonald et al. 1994).

Studies on impacts from bank protection structures have quantitatively measured conditions in front of a bulkhead and at adjacent un-bulkheaded shores and have shown that in front of a bulkhead, the suspended sediment volume and littoral drift rate all increased substantially compared to unarmored shores, which resulted in beach scouring and lowering along the armored shores studied.(Miles et al. 2001). Hard shoreline structures in the wave zone reflect wave energy with little attenuation of power (Miles et al. 2001). Bank protection structures that inhibit the erosion of feeder bluffs or transport of sediment stored high on the beach would cause erosion of material on the beach at the face of the structure and from the beach downdrift of the structure. As a result, beaches located in front of, and downdrift from, shoreline armoring can experience coarsening of the substrate, beach lowering, and beach narrowing (Anchor Environmental et al. 2002; Dean 1986; Everts 1985; Galster and Schwartz 1990; Johannessen et al. 2005; MacDonald et al. 1994; Zelo et al. 2000). The negative impact of sediment impoundment is most pronounced when armoring occurs along actively eroding bluffs, because these areas supply beach substrate throughout the length of the drift cells they support (MacDonald et al. 1994).

Silvester (1977, in Gabriel and Terich 2005) found that the presence of seawalls doubled the littoral energy applied to the sediment, which led to increased scour downdrift. As a result, more small sediment (e.g., sand and gravel) is entrained and moved than would occur along a natural shoreline that attenuates wave energy. This scouring impact is generally greater in vertical structures, such as bulkheads, compared to artificially or naturally sloped beaches (Zelo et al. 2000). Vertical structures also tend to focus wave energy on adjacent beach and backshore areas, which could contribute to erosion in areas downdrift of the bulkhead (MacDonald et al. 1994). Shoreline hardening manifests itself by a loss of the pebble veneer that is common throughout much of Puget Sound (Finlayson 2006). This process is similar to what has occurred on the urbanized shorelines throughout the Great Lakes (Chrzastowski and Thompson 1994).

One example of the impacts of bank protection on sediment supply and transport conditions is Seahurst Park in central Puget Sound (Burien, Washington). At Seahurst Park, the placement of bank protection structures in the 1970s resulted in dramatic changes to the habitat conditions in the park and reduced the amount of sand and gravel available throughout the 11-mile-long drift cell. The park shoreline was armored using a combination of stacked gabions, vertical concrete bulkhead, and riprap. A survey conducted in 2001 demonstrated that since shoreline armoring, beach elevations in the park have dropped approximately 3 to 4 feet. Further, the former sand, gravel, and small cobble beach now consists of larger substrates because the bank protection structures caused an increase in the erosive energy of waves moving sediment offshore and disconnection of the beach from primary sediment sources (bluffs) (Anchor Environmental et al. 2002).

Soft shore protection structures tend to absorb and attenuate wave energy better than hard structures by mimicking natural processes (Johannessen et al. 2005). Soft shore protection structures that maintain more natural slopes and materials that can be reshaped (e.g., an enhanced gravel berm) can absorb incoming wave water and attenuate the energy before the water percolates out gradually.

There are certain situations in which bank protection structures, particularly soft-shore techniques, can benefit habitat conditions by limiting sediment introduction. These benefits occur in settings where there is an overabundance of sediments and/or the sediment sources being disconnected are particularly fine sediments.

7.4.3.3.2.2 Changes in Habitat

Damage to surf smelt spawning areas has been documented in the presence of bulkheads in Hood Canal (Herrera 2005; Penttila 1978, in Thom et al. 1994).

Ahn and Choi (1998) found that in the presence of a new seawall, sediment grain size became significantly coarser and some shifts in dominance of abundant species occurred, including a tenfold increase in total abundance and biomass of the surf clam (Mactra veneriformes).

An active debate exists in the scientific community as to whether protective structures associated with marinas are as productive and diverse as natural hard-rock shorelines, particularly in the Adriatic Sea west of Italy (Bacchiocchi and Airoldi 2003; Bulleri et al. 2006; Guidetti et al. 2005). These studies in the Adriatic Sea have shown that maritime structures caused elimination of mobile, sandy habitats; weighted abundances in piscivores and urchins; and decreased abundances of native species that prefer more mobile substrates (Guidetti et al. 2005). Although species distributions are clearly different in Italy than in Washington State, the steep, paraglacial landscape and relatively short period and locally generated waves make hydraulic and geomorphic variables essentially identical (Finlayson 2006).

7.4.3.3.3 Bank Protection in Lacustrine Systems

Bank protection projects have the potential to directly or indirectly modify substrate conditions both during and after construction. In lake environments, waves striking shorelines at an angle transports sediment parallel to shore in the direction of the prevailing wind (Jacobsen and Schwartz 1981). Bank protection structures can impact sediment transport through changes in wave energy reflection and attenuation.

In both natural lacustrine systems and in reservoirs, bank protection can remove physical habitat and can exacerbate the degradation of littoral areas. Wave action may be a source of desirable spawning substrate. Kokanee salmon were observed to prefer spawning locations characterized by wave action, steep slopes, and an abundance of small, loose particles in Flaming Gorge Reservoir, Wyoming (Gipson and Hubert 1993). Lorang et al. (1993) observed that docks and seawalls intercepted transported gravels in Flathead Lake, Montana, as regulated lake levels rose and fell from early spring to late summer.

7.4.3.3.4 Dikes and Levees

Dikes and levees alter the hydraulic and geomorphic properties of the environment where they are located. In a riverine system, dikes and levees reduce a river’s connection with its floodplain and increase peak flows (Liu et al. 2004). This can lead to habitat isolation and strand fish in isolated pools without connection to the mainstem, and prevent access to low velocity refuge areas (Bolton and Shellberg 2001).

Some bank protection structures, especially levees, are designed to increase flood capacity in a more vertical than horizontal configuration, so the flow confined between the levees during high flows tends to be deeper and faster than if the floodplain could be accessed. Higher velocities and deeper water compared to conditions prior to construction tends to lead to increased erosion downstream (Bolton and Shellberg 2001). Bank protection structures intended to address bank erosion at the point of installation often result in the long-term reverse effect of increasing scour via alterations to hydraulics. Levees typically confine river flows to straightened channels, reducing channel sinuosity and altering channel geometry and sediment transport.

In tidal marshes, impacts are similar and include changes in channel geometry, sediment transport, and flow regime. In addition, due to their proximity to tidal areas, dikes located in nearshore sloughs and estuaries can lead to changes in wave energy, current velocities, and nearshore circulation. In a study of the Skagit River delta, dikes caused a reduction in tidal flushing, which increased sedimentation within the tidal area and reduced channel sinuosity (Hood 2004). Furthermore, loss of floodplain area to dikes prevents flood energy dissipation over the marsh surface, causing the mean channel width to increase and sinuosity to decrease (Hood 2004).

A disconnected floodplain and single stream channel are often goals of bank protection, despite the fact that an active floodplain connection plays a critical role in the dynamic equilibrium of rivers. Bank protection structures typically restrict the inundation of the floodplain. In the case of levees, which are designed and built for the purpose of increasing the flow capacity of a channel as a means of flood control (Bolton and Shellberg 2001), the disconnection of the floodplain is often perceived as the proper alternative to maintain the safety of life and property. The disconnection of the floodplain results in more isolation of side channels and wetlands (Bolton and Shellberg 2001).

7.4.3.3.5 Groins and Bank Barbs

A primary purpose of groins and bank barbs is to store sediment along the shoreline and prevent shoreline erosion. Groins are common in marine, lacustrine, and riverine environments. They are finger-like, vertical barriers extending from the shore/bank and oriented obliquely to the flow. They are often placed in series. They impede the downdrift/downstream movement of sediment. In rivers, groins and bank barbs are typically constructed in sets along the outside of a meander bend, with the primary function of redirecting flow and bed material away from the bank and toward the middle of the channel. In marine systems, they are constructed to encourage sediment deposition at specific locations. Structures built primarily for other purposes, such as boat ramps and beach access staircases, may also function like a groin (WDFW 2003).

In riverine sytems, flow velocity in a channel is proportional to the hydraulic radius and inversely proportional to roughness (Leopold et al. 1964). Bank barbs are intended to redirect flow toward the center of the channel using weir hydraulics over the structure. In contrast, groins are typically exposed above high water and are designed to divert flow (and bed sediment) around the structure. Both classes of structures reduce near-bank velocities, increase centerline velocities, retard bank erosion, cause local bed scour around the groin tip, and trap fine sediment and debris between structures on the downstream side of the structures (Lagassee et al., 2001; Li et al., 1984). Bed scour into a substrate of mixed particle sizes (i.e., sand and gravel) can selectively remove finer sediment and cause the substrate to coarsen. The depth and extent of bed scour depends on the water depth, approach velocity, and shape and size of the obstruction (Richardson and Davies 2001).

Because the rate and caliber of sediment supplied to a channel can influence the substrate size (Dietrich et al. 1989), changes in sediment supply can alter the composition of substrate used by HCP species. Groins and bank barbs can reduce the supply of suitably sized substrates for spawning fish and invertebrates by limiting natural processes of channel migration and bank erosion. Deposition of the finer sediment downstream can bury organisms and alter substrates.

Groins may interrupt surf-zone generated alongshore currents and circulation. Because groins rarely protrude into depths significantly (more than 10 feet) below mean lower low water (MLLW), they do not play an important role in tidal and estuarine water circulation. However, they may alter the movement and deposition of sediment.

Groins and bank barbs are usually constructed of placed rock or riprap, instead of sheet pilings or other impermeable structures that impede the exchange of hyporheic water. However, if a groin or groin-like structure uses sheet piles or other significant, impermeable, embedded elements (e.g., isolating more than 10 lineal feet along the shoreline from groundwater influence), it may alter hyporheic flow and affect water temperature.

Marine shorelines that have been modified by human activities tend to have less LWD and driftwood than unmodified beaches (Herrera 2005; Higgins et al. 2005). In particular, jetties and groins redistribute LWD such that it concentrates in certain areas and is absent in others (Miller et al. 2001).

7.4.3.3.6 Jetties

Jetties alter both the bathymetry and littoral drift of the area around and under such structures both in exposed (Komar 1998) and sheltered settings (NRC 2007).

Jetties are designed to limit deposition in a navigable channel and to provide wave protection for vessels (Dean and Dalrymple 2002). As a part of jetty installation, substrate can be placed that is completely artificial (Komar 1998).

Jetty installation in rivers is extremely rare because one of the main purposes of a jetty is to obstruct littoral transport, which does not occur on most rivers. In rivers, transport is not confined to the shoreline, and areas near the bank are generally areas of deposition (Chow 1959).

Jetties restrict natural geomorphic processes along the shoreline and often fix the location of estuarine exchange (e.g., at the Columbia River mouth). These geomorphic changes will persist for the design-life of the structure and can impose significant impacts on fish and invertebrates. Jetties may prohibit migration of fish and invertebrate species or life-history stages. The costs of replacing a newly constructed structure may create a strong incentive against the additional investment required to address the problem correctly. This may delay the actions necessary to protect the fish migration corridor, perhaps as long as the design-life of the underperforming structure.

7.4.3.3.6.1 Altered Sediment Supply and Deposition

Jetties are designed to prevent sediment from depositing in a navigational channel. The principal effect of a jetty is to obstruct natural littoral transport, thus starving the downdrift shoreline (Dean and Dalrymple 2002). Jetties have even initiated shoreline instability on adjacent shorelines (Dias and Neal 1992) and redistributed turbidity in their vicinity (Sukhodolov et al. 2004). Alteration of sediment transport patterns can present potential barriers to the natural processes that build spits and beaches and provide substrates required for plant propagation, fish and shellfish settlement and rearing, and forage fish spawning (Haas et al. 2002; Penttila 2000; Thom and Shreffler 1996; Thom et al. 1994). In addition, a considerable amount will often deposit on the updrift side of a jetty. This causes the shoreline to protrude into the water body, distorting sediment transport farther up the shoreline (Komar 1998).

Weir jetties are submerged at most water levels for some portion of their length, usually the landward-most end. These features allow the passage of sediment for localized deposition in some inactive portion of the navigational channel (Seabergh and Kraus 2003). Weir jetties have a tendency to alleviate some of the geomorphic and hydraulic modifications associated with jetties; however, they do initiate change in the substrate and tend to produce geomorphic disturbance (Ranasinghe and Turner 2006).

By examining habitats used by juvenile salmonids, Fresh (2006) concluded that the conversion of sandy, mobile substrates, such as those on natal deltas, would produce a greater impact on salmonid rearing than those on naturally immobile shorelines. Because Puget Sound shorelines are diverse in terms of sediment mobility (Finlayson 2006), the effect on juvenile salmonids from shoreline hardening is highly site specific and could be small in places where the shoreline is naturally immobile. Unfortunately, jetties are often located near river mouths (and deltas) where the transition from mobile, sandy substrate to an immobile, rocky substrate will be most detrimental to juvenile salmonids.

7.4.3.3.6.2 Altered Wave Energy

Jetties are generally constructed out of rock or poured concrete. Therefore, they result in and cause the retention of wave energy in the surrounding area (Komar 1998). In addition, ship traffic associated with the addition of a pair of jetties can interact with these artificial boundaries, causing a significant increase in wave energy in between the two jetties (Melo and Guza 1991). The modified relationship between topography and wave energy caused by a jetty results in a shoreline that is out of equilibrium with natural shoreline processes (Komar 1998). The effects are generally independent of jetty design (i.e., weir jetties are as prone to these effects as exposed jetties), although some best management practices (BMPs) can reduce these effects. Wave energy artificially accumulates in some areas and is diminished in others.

Jetties may decrease complexity of the shoreline by deflecting wave action from the littoral zone. Wave action creates complex littoral habitat by removing fine or silty sediments (Beauchamp et al. 1994). Wave action may also be a source of desirable spawning substrate.

Jetties have been shown to both increase (da Silva and Duck 2001) and decrease (Sherwood et al. 1990) tidal prisms, depending on the characteristics of the tides and freshwater input and the nature and geometry of the alterations.

7.4.3.3.6.3 Altered Groundwater/Surface Water Exchange

Jetties change the shoreline from a dynamic, loose surface to a rigid, immobile one along their length. Although there are distinct differences between artificial substrates used in jetty construction, they all behave over time like bedrock shorelines, similar to extremely coarse-clastic beaches in Puget Sound (Finlayson 2006). The primary difference between these installments is whether they permit the exchange of groundwater with the marine system. In the construction of a jetty, it is common for pilings to be placed near the shoreline to ensure that the landward end of the jetty remains intact. In these cases, groundwater connections with the sea are interrupted. Submarine groundwater discharge has been documented to play an important role in the circulation of fluids and nutrients on many coasts throughout the world (Gallardo and Marui 2006; Johannes 1980; Michael et al. 2005). Solid concrete walls and steel pilings allow no flow-through and likely have additional impact as compared to other artificial substrates (e.g., riprap) (Nakayama et al. 2007). Sheet pilings could interrupt the free exchange of groundwater between the sea and the uplands. If this occurs, deleterious effects on nearshore ecosystems are likely (Nakayama et al. 2007).

Dumped rock or riprap jetties that do not have sheet piles associated with them do not impede or eliminate the exchange of groundwater with supratidal areas. Therefore, these types of jetties or their analogs do not exhibit groundwater impacts.

7.4.3.3.7 Breakwaters

7.4.3.3.7.1 Marine Breakwaters

Breakwaters modify the wave environment in the nearshore. This redistribution of wave energy can have a number of interrelated indirect and direct impacts on fish and invertebrates, and these may be grouped into two categories: those that relate to changes in substrate, and those that change water column characteristics. Reduction in wave energy from natural levels lowers near bed shear stress, resulting in the deposition of finer sediments (Miller et al. 1977).

Breakwaters are generally constructed out of placed rock, parallel to the shoreline, and are specifically designed to reduce wave energy between them and the shoreline (Dean and Dalrymple 2002). Thus, they diminish wave energy shoreward of the structure while wave energy is generally increased offshore (Dean and Dalrymple 2002). The patterns of wave energy produced by emergent and submerged breakwaters are different (Ranasinghe et al. 2006). Breakwaters are often used in series to protect a shoreline from erosion, and sometimes to enhance a beach nourishment project (Dean and Dalrymple 2002). They are typically used on sandy, open coastlines (Dean and Dalrymple 2002), although recent work has shown that they are equally effective at shoreline protection in coarse-clastic environments (King et al. 2000) more typical of Puget Sound (Finlayson 2006).

Breakwaters create a new shoreline that is rigid and immobile along its length. Many different materials have been used to construct breakwaters including riprap, reinforced concrete, pre-formed concrete elements like dolos, and timber structures (NRC 2007). Regardless of the material used, the addition of immobile substrate affects fishes and invertebrates (USFWS 2000). These impacts are generally most pronounced if the structure has a vertical wall, rather than a steep slope (Bulleri and Chapman 2004).

Although breakwaters are designed to protect areas from wave energy and therefore initiate deposition, they have been shown to induce scour on the seaward side of their ends (Sumer et al. 2005). This is primarily associated with artificial rip currents developed in these areas. However, there have been no experimental studies that have documented impacts on forage fish spawning areas.

Breakwaters are not designed to alter nearshore current velocities; however, there is evidence that they can unintentionally cause strong rip currents (Bellotti 2004; Dean and Dalrymple 2002). Also because they function essentially as a new obstacle to flow, they can also reduce velocities in other areas. The relationship between flow velocity and a change in substrate is related to the boundary shear stress (Miller et al. 1977). Substrate and aquatic vegetation are removed if a critical shear stress is exceeded. If the shear stress drops, anomalous deposition can occur.

Breakwaters alter nearshore circulation by modifying the transport processes associated with a variety of wave and wave-breaking mechanisms (Caceres et al. 2005).

Breakwaters have been shown to disrupt the littoral transport of sediments and subsequently cut off downdrift shorelines to a sediment supply (Bowman and Pranzini 2003; Sane et al. 2007; Thomalla and Vincent 2003). Reduction or elimination of the sediment supply can inhibit the proper functioning of spits and beaches and cause the elimination of substrates required for plant propagation, fish and shellfish settlement and rearing, and forage fish spawning (Haas et al. 2002; Parametrix 1996; Penttila 2000; Thom and Shreffler 1996; Thom et al. 1994).

In the construction of a breakwater, it is uncommon for structures (e.g., sheet pilings) to be placed that interrupt groundwater transfer between the sea and the shore. However, if the proposed breakwater has this type of design element, groundwater connections with the sea could be interrupted, and submarine groundwater discharge impacts would result.

7.4.3.3.7.2 Lacustrine Breakwaters

Breakwaters are generally installed on open, exposed coasts (Komar 1998) and are rarely used in lacustrine environments. Most of the work that has been performed on breakwaters in lacustrine environments has been conducted in the Great Lakes (Fitzsimons 1996; Marsden and Chotkowski 2001; Olyphant and Bennett 1994), which are substantially larger (and therefore subject to much larger waves) than any lakes in Washington State. Lakes are subject to long-term water level variability but not to tides. As a result, the size of the breakwater that may be required is significantly smaller and can be placed much closer to shore. This would mean the area of alteration associated with breakwaters would be generally smaller in lakes than in marine waters.

7.4.3.3.7.3 Riverine Breakwaters

Permanent breakwaters are built to protect the shoreline from wave energy (Dean and Dalrymple 2002). Nearly all rivers in Washington State are too small (both in width and depth) and fast moving to have shorelines where waves significantly influence the mobility of the shoreline substrate. Only the Columbia River is generally considered large and deep enough to produce wave heights significantly affecting the substrate and erodibility of its banks. Floating, temporary breakwaters, rather than permanent structures, are used in Washington riverine systems, and would be considered overwater structures .

Generally, water crossings are unlikely to result in significant direct alteration of the hyporheic and groundwater environments because they impact short stream reaches. However, if the water crossing contributes to an indirect effect, e.g., by facilitating land use change or floodplain development, then those indirect impacts could have a more substantial impact.

The principal effects of channel confinement on groundwater and hyporheic function are identified by Bolton and Shelberg (2001). Effects likely to be observed in connection with water crossing structures include:

- Changes in hyporheic chemistry attributable to altered vegetation in the riparian areas (most likely as a result of the impacts of road approaches, which impact a substantially larger area than bridges and culverts *per se*)
- Reduced hyporheic exchange due to increased sedimentation from various causes likely to be a minor effect for most water crossings)
- Lower base flows attributed to reduced recharge from a straightened channel (likely to be a minor effect for water crossings, which straighten short, isolated sections of channel)

7.4.3.4 Culverts

7.4.3.4.1 New Culvert Placement

The improper matching of culverts to local hydraulic and geomorphic conditions can result in a variety of channel responses, some of which create barriers to fish passage (e.g., outfall drops caused by localized scour), and others that modify habitat conditions (e.g., the creation of road-impounded wetlands).

Because the surface of a stream is roughly parallel to its bed (Dunne and Leopold 1978), water surface slope is mainly altered by changes in channel gradient. A culvert or other non-erodible artificial streambed has a fixed gradient, which may or may not be consistent with channel gradient when the culvert is installed and may later be inconsistent if gradient changes due to other factors. Bates (2003) provides extensive discussion of the role of culvert gradient in determining channel response, particularly for steeper channels and retrofit situations where the culvert gradient is steeper than that of the associated channel.

Culverts “lock” a stream channel by fixing it within artificial bounds set by the culvert walls. Closed culverts can lock the channel in the vertical dimension by imposing a fixed base level. This locking prevents the channel from adjusting to flow and sediment supply variability by altering its cross section and gradient; consequently, adjustment occurs by altering channel hydraulics, potentially destabilizing the channel. This effect is most pronounced in the immediate vicinity of the culvert and results in relatively frequent disturbance of in-channel habitat in the affected area.

In freshwater systems, the most common and pervasive substrate modification is the placement of pipe (as opposed to bottomless) culverts. Such culverts may acquire a veneer of bed material but usually are bedded by whatever material the culvert is made from, usually metal, plastic, or concrete. Culverts often have a small diameter compared to the functional channel width upstream and downstream. Culverts, because they closely confine the channel within a pipe, have some specific impacts on channel hydraulics that are most apparent in step-pool, pool-riffle and plane bed channels, where the stream commonly shows a highly variable capacity to transport its sediment load.

The impacts of culverts identified by Bates (2003) include:

- Channel realignment that eliminates natural features such as meanders, spawning riffles, and other diversity in the channel.
- Shortened channels that carry flows at higher velocity, causing streambed instability and downstream scour and bank erosion.
- Sediment mobilization that can smother redds downstream.
- Changes in stream base level that can destabilize the channel and cause reduced hydrology in floodplain water bodies.
- Upstream bed and bank instability if the culvert is undersized, which causes the repeated formation and draining of an upstream backwater pool.
- Blocking the downstream movement of coarse sediment such as boulders and LWD.

- Spawning gravels replaced with culvert pipe.
- Rearing habitat replaced with culvert pipe.
- No streambanks inside the culvert pipe.
- No riparian inputs of leaf litter or terrestrial insects along the culvert pipe.
- No pool, riffle, or hyporheic (in-gravel) habitat within the culvert pipe.
- Few or no benthic invertebrates in the culvert.
- No plants growing on substrate within the culvert, because it's dark.
- Culvert may contribute to loss of off-channel habitat.

When culverts are not designed appropriately for their hydraulic and geomorphic context, the culvert may fail to meet the dual objectives of providing fish passage while adequately conveying flood flows. Culverts that produce high exit velocities may scour the channel at the outlet, leading to an enlarging outfall drop that creates a fish passage barrier over time. Culvert designs that fail to address sediment transport requirements may aggrade over time, creating a barrier condition and reducing the hydraulic capacity of the structure, leading to flooding. Roads have commonly been placed at the edge of river valleys, perpendicular to stream channels draining onto the valley floor. Channels in these settings are naturally depositional, requiring the channel to migrate in response. Culvert designs that fail to recognize these characteristics are likely to aggrade and fail over time.

7.4.3.4.2 Culvert Removal, Retrofitting, and Replacement

Many existing culverts have altered the process of channel migration and evolution, as well as the transport of sediment and woody debris, particularly in cases where barrier conditions are created. Alterations of these physical processes are commonly associated with changes in channel gradient and morphology upstream and downstream of the culvert. Culvert removal or replacement with stream simulation either partially or fully eliminates this restriction, allowing the channel to adjust to a new equilibrium condition. The intent of the stream-simulation approach is to provide a culvert configuration that allows for natural channel processes to operate to the greatest extent possible.

The intent of culvert removal is to restore and reconnect the natural hydraulic and geomorphic processes, reducing or eliminating ecosystem fragmentation. Current culvert replacement guidance favors approaches that at least partially restore these processes (e.g., the stream-simulation and no-slope approaches). These approaches are generally expected to produce a net benefit, particularly when the existing structure is a complete barrier to fish passage. In certain cases, however, removal of the culvert or replacement with a structure that reconnects natural geomorphic processes can lead to broader hydraulic and geomorphic consequences, such as headcut migration or alterations to road-impounded wetlands.

Retrofitting existing culverts is not expected alter existing hydraulic and geomorphic effects in most cases, as this option will maintain the existing structure and not significantly perturb the current channel geometry. However, the placement of internal weirs or baffles can decrease flow capacity, which may impose backwater effects upstream of the structure, leading to potential sediment aggradation, bar formation, and changes to flood elevations. These perturbations can also promote debris accumulation, increasing the risk of structural failure. Depending on the amount of material captured, the natural sediment transport rate, and the maintenance frequency and methods used, this could result in effects on substrate composition in downstream reaches

7.4.3.4.2.1 Headcut Migration

Culvert removal or replacement can reinitiate headcuts that have been arrested by the existing structure, allowing these headcuts to continue to migrate upstream.

Bed scour occurs at culvert outfalls, initially the result of high flow velocities exiting the structure, and then by the impinging jet produced downstream of a sudden drop in channel elevation (Jia et al. 2001). As the water jet penetrates the pool and reaches its bottom, the jet divides into two jets parallel to the bed and in opposite (upstream and downstream) directions (Flores-Cervantes et al. 2006). In homogeneous soils, upstream migration of the scour hole occurs as the upstream jet scours the headcut face, and as the downstream jet removes this sediment and sediment delivered from upstream. Flores-Cervantes et al. (2006) showed that plunge pool erosion varies with the headcut height, flow rate into the pool, and soil properties. The formation of a scour pool at a culvert outfall sets up the condition for headcut or knickpoint propagation upstream if the culvert is removed.

In general, headcut migration will occur when erosion of the headcut face by the upstream jet is faster than the erosion of the bed at the top of the headcut (Flores-Cervantes et al. 2006), and when there is sufficient transport capacity downstream to remove the eroded sediment from the plunge pool (Jia et al. 2001). The distance a headcut propagates upstream will depend on how these conditions change with headcut migration and whether the headcut encounters resistant materials.

Headcuts are most often caused by downstream perturbations and include changes to processes related to hydrology and hydraulics, interruption of sediment transport, hardened bank stabilization or confinement modifications, or the lack of large woody debris that contributes to channel stability. In many cases, arrested headcuts are the cause of outfall drop formation at the mouth of the culvert that leads to a barrier condition. The outfall drop can become quite large in some cases, creating a large change in gradient across the structure. Culvert removal or replacement will likely reinitiate the arrested headcut and cause channel incision, bank instability, and bedload mobility, with a number of detrimental changes in habitat conditions in upstream reaches. Based on experience in Washington State, the potential for headcut migration is a factor that must be considered in 50 percent or more of culvert removal or replacement projects (Bates 2007). While headcut migration can be avoided in many cases by employing appropriate channel modifications, these measures are not always practicable or desirable due to cost, concerns about private property access, and the fact that instream structures interfere with natural geomorphic recovery after the culvert is removed.

Over time, headcut migration would be expected to return the channel gradient and floodplain connectivity to an equilibrium condition, provided that other factors occur (principally, that LWD of sufficient size to trap and retain sediments is available for recruitment).

7.4.3.4.2.2 Channel Incision

Culvert removal or replacement can change channel incision. Channel downcutting associated with headcut migration can cause a range of habitat-related effects.

Channel incision decreases the channel gradient and destabilizes the banks (Kondolf et al. 2002; Sandecki 1989). Bank erosion can increase the local supply of fine sediment and result in channel instability (Sear 1995), leading to increased bedload mobility and ongoing water quality effects in the form of sedimentation.

Lowering of surface water elevations can disconnect side channel and off-channel habitats, as well as reduce the frequency and extent of floodplain inundation. These forms of fragmentation can substantially reduce the extent and productivity of aquatic habitats. Channel incision can result in the loss of floodplain and channel complexity through the fragmentation of off-channel habitats, and can adversely affect riparian vegetation (Castro 2003, Kondolf et al. 2002). Decreased lateral connectivity with side-channel, slough, and floodplain ponds can have a range of effects on HCP species. Side channels create refugia for juvenile fish (Jungwirth et al. 1993), while floodplain ponds and backwater sloughs create zones of high retention and productivity that provide vital rearing habitat (Hall and Wissmar 2004; Sommer et al. 2005) and important sources of organic material for the channel (Tockner et al. 1999). The loss of connectivity between the river and these habitats can result in a decrease in organic matter recruitment (Tockner et al. 1999; Valett et al. 2005) and reduced access to valuable foraging and rearing habitats (Henning et al. 2006).

When channel incision exposes underlying bedrock, it can significantly reduce the productivity and quality of aquatic habitat for a range of fish species, particularly salmonids dependent on alluvial bedded systems for spawning habitat and forage (Kauffman et al. 1993). Depending on the underlying geology, bedrock exposure can accelerate weathering and erosion in lower gradient systems (Stock et al. 2005).

Channel incision can lead to temporary simplification of channel form, creating relatively uniform hydraulic and geomorphic conditions over extended lengths of channel. This reduction in habitat complexity can have a range of adverse effects on HCP species.

7.4.3.4.2.3 Alterations to Road-Impounded Wetlands

Culvert removal or replacement may dewater or otherwise alter road-impounded wetlands, leading to hydraulic and geomorphic changes and potentially a shift to wetland type habitat. Removal or replacement of the culvert can lead to reestablishment of natural geomorphic processes, with a range of effects on instream habitat conditions. The potential dewatering of road-impounded wetlands is a factor for consideration in a relatively low number of cases, estimated to be less than 5 percent of all culvert projects (Bates 2007). The cases where potentially significant hydraulic and geomorphic effects are likely to occur represent a small component of this total (Barnard 2002).

The quality of wetland habitats produced by road-impounded wetlands can vary (Barnard 2002). In most cases, these wetlands are of marginal habitat value, and the importance of restoring natural stream processes is overriding. In rare circumstances, however, high-value habitats may have developed that are occupied by species of interest. In cases where a significant change in hydraulic gradient is induced by the barrier, deposition of fine substrates will occur upstream of the culvert, and the interception of these sediments will cause some degree of sediment coarsening in downstream reaches. Because road-impounded wetlands can raise surface water levels, they may inundate adjacent floodplains more often, creating wetland conditions (Hammerson 1994). Larger impoundments with increased floodplain connectivity are also likely to accumulate organic material, increasing the size of the sediment wedge behind the barrier.

These perturbations and the related ecological stressors they impose range in severity depending on the size of the road-impounded wetland, the volume and characteristics of impounded sediments, and the equilibrium gradient of the restored channel.

7.4.3.5 Fish Passage Structures (Fish Passage Weirs and Roughened Channels)

Fish passage projects can alter flow conditions in the vicinity of the structure by altering channel morphology and hydraulics. Changes in flow velocities may significantly alter sediment transport. The presence of a fish passage structure may accelerate or slow streamflow in different portions of its zone of influence. For example, if a permanent weir installed to prevent upstream dispersal of invasive species creates an impoundment, altered flow velocities in the impoundment will cause increased sediment deposition. In contrast, a structure such as a roughened channel may increase flow velocities in slackwater areas to moderate flows elsewhere. Increased velocities can scour bed material and benthic organisms (Camargo and Voelz 1998).

Depending on configuration, fish passage structures may also change channel geometry. For example, an impoundment formed by a permanent barrier weir may cause upstream channels to widen, and downstream channels will likely become narrower. Because flow velocity in a channel is proportional to the hydraulic radius (the cross-sectional area of the channel divided by the wetted perimeter) and inversely proportional to roughness (Leopold et al. 1964), changes in flow velocity will ultimately change the channel geometry. Altered depth and width downstream of a fish passage structure may disconnect the river from its floodplain and side channel habitats, potentially reducing habitat accessibility.

Flow through fish passage structures will commonly increase local velocities and turbulence downstream of the structures, making fish passage difficult (Baker 2003). While fish passage structures are intended to provide passage benefits, when compared to the natural stream baseline, effects on HCP species may occur. For example, lampreys have been observed migrating over weirs, with short bursts of movement followed by extended resting periods (Quintella et al. 2004). The sea lampreys seemed affected by increasing fatigue, which the authors attributed to initiating a new burst of movement without fully recovering from the previous exertion.

The effects of fish passage structures on sediment composition and stability may range from relatively benign in the case of roughened channels, to more extensive in the case of barrier weirs that create impoundments and interrupt sediment transport.

Permanent fish passage weirs are typically designed to not interrupt the transport of sediment, LWD, and organic material. They are likely to affect reach-level sediment sorting, without necessarily having a broad effect on sediment transport and, by extension, sediment composition.

In some instances, increased velocities associated with weirs can indirectly affect HCP species by causing local bed scour around structures and result in a corresponding deposition of sediment downstream. Bed scour into a substrate of mixed particle sizes (e.g., sand and gravel) can selectively remove finer sediment and cause the substrate to coarsen. Likewise, increased deposition of the finer sediment downstream can bury organisms and result in finer substrate.

7.4.3.6 Bridges

Bridges can fix a stream channel within artificial bounds set by the bridge abutments and approach fills located on the floodplain. This prevents the channel from adjusting to flow and sediment supply variability by altering its cross section and gradient; consequently, adjustment occurs by altering channel hydraulics, potentially destabilizing the channel. This effect is most pronounced in the immediate vicinity of the locking structure and results in relatively frequent disturbance of in-channel habitat in the affected area.

7.4.3.6.1 Altered Wave Energy

Bridges located in the wave zone have the potential to affect wave direction and intensity. The literature on potential impacts is focused on overwater structures such as piers, but the physical processes involved are common to piling- and abutment-supported bridges as well.

7.4.3.6.2 Altered Groundwater/Surface Water Interactions

Generally, bridges are unlikely to result in significant direct alteration of the hyporheic and groundwater environments because they impact short stream reaches. However, if the water crossing contributes to an indirect effect, e.g., by facilitating land use change or floodplain development, then those indirect impacts could have a more substantial impact.

The principal effects of channel confinement on groundwater and hyporheic function are identified by Bolton and Shelberg (2001). Effects likely to be observed in connection with bridges include:

- Changes in hyporheic chemistry attributable to altered vegetation in the riparian areas (most likely as a result of the impacts of road approaches, which impact a substantially larger area than bridges themselves)
- Lower base flows attributed to reduced recharge from a straightened channel (likely to be a minor effect for bridges, which straighten short, isolated sections of channel)

7.4.3.7 Conduits

A pipe or other conduit trenched across the bottom of a stream, although customarily placed below the depth of scour, may function as a gradient control structure if subsequent downcutting occurs. Shallowly buried conduits can impose a fixed vertical base level on a stream channel. This prevents the channel from adjusting to flow and sediment supply variability by altering its gradient; consequently, adjustment occurs by altering channel hydraulics, potentially destabilizing the channel. This effect is most pronounced in the immediate vicinity of the consduit.

7.4.3.8 Dams

The presence of a dam dramatically influences the hydraulic and geomorphic properties of a riverine system. If a dam is located close to coastal and estuarine areas, it may affect the hydraulics of nearshore environments as well. These modifications include altered wave energy, altered current velocities, and altered nearshore circulation.

7.4.3.8.1 Altered Water Flow

Dams tend to reduce peak flows and increase base flows (Magilligan and Nislow 2005), especially for systems where dams are used for hydropower generation. Flow variability is changed from a natural fluctuation to one based on human needs. The changes in flow variability translate into changes in daily high and low water, which can alter flooding and inundation of side channels and floodplains, thereby affecting habitat connectivity.

Dams causes change in channel width and depth. Upstream of a dam, both depth and width increase; downstream, the average depth and width decrease (Tiemann et al. 2004). Because flow velocity in a channel is proportional to the hydraulic radius (the cross-sectional area of the channel divided by the wetted perimeter) and inversely proportional to roughness (Leopold et al. 1964), changes in flow velocity ultimately change the channel geometry. Altered depth and width downstream of a dam disconnect the river from its floodplain and side channel habitats, potentially reducing habitat accessibility and increasing the stranding of aquatic species.

Changes in flow variability can contribute to changes in species composition. High flows, which can displace organisms downstream, help maintain biodiversity through natural flow variability. When stable flows persist in the presence of a dam, organisms adapted to stable flows dominate and diversity will be reduced (Bednarek 2001). Dam removals have been shown to increase species diversity by restoring the natural flow variability. A dam removal on the Chipola River in Florida increased fish diversity downstream from 34 to 61 species (Hill et al. 1993).

Intermittent flooding and draining are needed for the regeneration of riparian forests. In the presence of dams, a loss of flooding reduces forest productivity, suppresses tree growth, and increases tree mortality (Kozlowski 2002). In addition, upstream flooding from reservoir inundation kills trees and seed sources, resulting in inadequate seed supplies for downstream forests (Kozlowski 2002).

Alteration of flow can have impacts far downstream. Reduced freshwater flows can affect tidal mixing and translate into impacts on marine species. Migration patterns, spawning habitat, and species diversity for adult and larval stages of fish and invertebrates are affected by the presence of dams upstream (Drinkwater and Frank 1994). Changes in tidal surges will particularly impact weak swimming fish or early life-history stages that rely on swimming with tidal flows during migration upstream or downstream during spring high flows (Dadswell 1996; Oullet and Dodson 1985).

Inherent in altered flow variability is the change in flow velocities. During times of water release from a dam, velocities downstream can become quite large; however, when water is held back, velocities downstream are depressed.

Flow velocities influence swimming activity and respiration in fish species. High flows below Hells Canyon Dam on the Snake River caused increased swimming activity and subsequently higher O_2 consumption, leading to suppressed movement in white sturgeon (Geist et al. 2005). The study suggested that high flows and velocities, even of short duration, can restrict the movement of juvenile white sturgeon; however, these increases may not cause an increase in energy expenditure due to the adaptation of white sturgeon to high-flow environments. For other HCP species that prefer slower velocities (e.g., Pacific lamprey) high velocities caused by dam releases may be more prohibitive.

Increased flow velocities during water releases can also cause fish species to rest in areas of slower moving water in order to recover from increased activity. This behavior can result in unsuccessful recruitment from delayed migration upstream for anadromous species (e.g., salmonids, sturgeon, lamprey), or increased predation from holding in slow pools downstream of dams and high-velocity reaches.

7.4.3.8.2 Altered Sediment Supply, Transport and Deposition

Changes in flow velocities may also significantly alter sediment transport. The presence of a dam slows river water upstream, causing increased sedimentation in the impoundment behind the dam. Downstream, increased velocities from water releases can scour bed material and benthic organisms (Camargo and Voelz 1998). Altered sediment transport can increase erosion downstream, widen the channel, and reduce channel roughness (Assani and Petit 2004).

Dams modify the sediment available to species for spawning and rearing by blocking the contribution of sediments from upland or upstream source areas.

Several studies have documented how dam-created reservoirs act as sediment sinks (Ahearn et al. 2005; Teodoru and Wehrli 2005). As fine particles settle out above dams, they can fill in cobble and boulder habitat and raise (aggrade) the stream bed (Bednarek 2001). As water velocities slow upstream of dams and as water enters the impounded area, sediment settles out, causing sedimentation upstream of a dam and "clean water," that has little to no suspended sediments downstream (Assani and Petit 2004; Kondolf 1997). Kondolf (1997) describes clean water as sediment starved; there is the potential to scour and erode downstream environments as the stream tries to regain sediment equilibrium. Increased erosion and incision downstream can lower groundwater tables and affect riparian vegetation through reduced access to water (Gillilan and Brown 1997). If erosion is extremely high, incision down to bedrock can occur and effectively reduce hyporheic and groundwater–surface water interactions (Assani and Petit 2004). Increased erosion can cause bank failures, resulting in large sediment inputs and a loss of riparian vegetation (Dietrich et al. 1989; Kondolf 1997; Sear 1995).

The reduction in suspended sediment (and turbidity) directly downstream of a dam can also influence predation of those species waiting to pass over dam structures. Experiments have shown that white sturgeon larvae predation by prickly sculpin increased in the presence of low-turbidity water (Gadomski and Parsley 2005). This suggests that some species use sediment as cover to some extent.

Impacts from altered sediment transport are not limited to the riverine environment; depending on the location of the dam and the river system, impacts on coastal ecosystems are also possible. The reduction of sediment supply to estuarine and coastal environments will change habitat quality and cause erosion of beaches that rely on sediment from rivers. For example, the lack of sediment supply from two large dams on the Elwha River, Washington, has contributed to a loss of beach and coastline habitat (DOI 1995).

Increased velocities associated with dams can indirectly affect HCP species by causing local bed scour around structures and with a corresponding deposition of sediment downstream. Bed scour into a substrate of mixed particle sizes (e.g., sand and gravel) can selectively remove finer sediment and cause the substrate to coarsen. Likewise, deposition of the finer sediment downstream can bury organisms and cause the substrate to become finer.

A high level of substrate fines in channel substrate from a dam may hinder the connection between surface and groundwater, limiting vertical and lateral connectivity between these two habitat types (Edwards 1998; Pusch et al. 1998). This lack of connectivity can degrade conditions for riparian zone vegetation, reducing LWD recruitment to the stream channel and subsequently limiting habitat-forming and maintaining processes. Effects on ecological functions and freshwater aquatic species associated with degraded connectivity between different riverine habitat elements are well documented (Bilby and Bisson 1998; Hershey and Lamberti 1992; Karr 1991; Kelsey and West 1998; Montgomery et al. 1999; Naiman et al. 1992; Reiman and McIntyre 1993; Stanford and Ward 1992; Stanford et al. 1996).

7.4.3.8.3 Dam removal

Dam removal alters sediment transport in a river. Because sediment is trapped upstream of a dam, removal of the dam will increase sediment downstream. Although the potential for increased suspended sediment downstream from dam removal is highly likely, the effects are often short term. The impact depends on the type of removal, time of the year, length of time the dam was present, flow rates, and flow velocities (Bednarek 2001). Studies have shown that sediment pulses from dam removal can migrate through a system in days to weeks to years (Bednarek 2001); in some cases, sediment releases are similar to a periodic storm event (Winter 1990). Dam removal is one possibility for restoring natural sediment transport in a riverine system.

Doyle et al. (2002) and Doyle et al. (2003) demonstrated that channel evolution after small dam removal follows the classic model of incision and widening that is induced by base-level lowering. Accumulated sediments erode rapidly and are transported to lower gradient, downstream reaches where aggradation is likely to occur. Water depths and flow conditions within the former impoundment will change, and the wetted perimeter will decrease. Bank stability within the former impoundment declines until the channel adjusts and vegetation becomes established (Bednarek 2001; Doyle et al. 2002, 2003; Pollock et al. 2003). Bank failure induces channel-widening and bed-aggradation processes that lead to an eventual dynamic equilibrium in the longitudinal profile of the channel (Schumm et al. 1984).

Downstream channel geometry will be only temporarily affected by the removal of small impoundments (Pollock et al. 2004). Deposited sediment will be transported to downstream low-energy environments (e.g., pools, channel margins) but will likely be entrained and exported farther downstream in subsequent flooding events. Upstream channel geometry will change more dramatically. The main channel in the upstream reach responds by narrowing. Channel narrowing may limit access to shallow water habitat and decrease the surface area exposed to solar radiation (Margolis, Raesly et al. 2001).

A study of the effects of removal of two small dams in Wisconsin found insignificant sediment export, attributed to the small impoundment size and relatively high thalweg velocities that limited sediment accumulation prior to removal (Orr et al. 2006).

7.4.3.8.4 Effects of dams on invertebrates

Altered flow variability can dewater floodplain habitat and strand fish and invertebrate species. As an example of effects on invertebrates, a drawdown of the Lower Granite

Reservoir on the lower Snake River killed many California floaters, western floaters, and western ridged mussels. Freshwater mussels are known to migrate to avoid receding waters and can be vulnerable to predators during this time. If dewatering occurs for long time periods, mussels may bury themselves during dewatering, but there is a risk of mortality if waters do not return to normal levels before the mussels overheat (Nedeau et al. 2005).

In a survey of native freshwater mussels in the United States and Canada, Williams et al. (1993) concluded that declines in populations were caused by habitat destruction, dams, siltation, and channel modifications Watters (1999) summarized the effects of impoundments on mussel species in the United States, and deposition of silt within and downstream of impoundments has been linked to extinction of several mussel species nationwide.

7.4.3.9 Weirs

The hydraulic and geomorphic impacts of weirs on HCP species are similar to those of dams. Flow over weirs increases turbulence below structures and increases local velocities, making fish passage difficult (Baker 2003). Sea lampreys have been observed migrating over weirs with short bursts of movement following by extended resting periods (Quintella et al. 2004). The sea lampreys seemed affected by increasing fatigue, which the authors attributed to initiating a new burst of movement without fully recovering from the previous efforts.

Weirs drop channel elevation, which can alter channel slopes. Abrupt changes in slope can alter sediment transport and represent migration barriers for fish. In a study of fall heights from weirs on movements with the common bully (*Gobiomorphus cotidianus*) and adult and juvenile inanga (*Galaxias maculatus*), Baker (2003) showed that both species were restricted by falls of 0.4 inches (10 cm), and the passage of adult inanga was restricted by falls of 0.8 inches (20 cm). Atlantic salmon in the Pau River (France) were able to pass over weirs of 59.1 inches (1.5 m) in height but had difficulty passing weirs of 98.4 inches (2.5 m) in height (Chanseau et al. 1999).

7.4.3.10 Outfalls

The hydraulic and geomorphic impacts of outfalls are diverse. Outfall design and effluent characteristics play an important role in the degree of impact on fish and invertebrates. Well-designed outfalls that discharge small flow rates of effluent with similar constituents (i.e., temperature, salinity, turbidity and density) as the receiving water do not have significant hydraulic and geomorphic impacts.

7.4.3.10.1 Submerged Outfalls

Given the sensitive nature of the sediment supply along the shorelines of Puget Sound, structures that span the beach foreshore, which is the zone of maximum sediment transport, may have significant effects (Finlayson 2006). Outfalls that are submerged below the water surface, but elevated above the natural grade, have the potential to act as groins, interrupting the natural flow of sediment along the shoreline (Herrera 2006b). If submerged outfall plumbing protrudes above grade and above the closure depth[2], such an interruption of longshore transport has the potential to be significant. If the outfall protrudes above grade but below closure depth, the effects will be minimal.

Hydraulic impacts of submerged outfalls are related primarily to the flow rate and the physical and chemical properties of the effluent. Typically, submerged outfall outlets are located below the closure depth and below significant light penetration, such that aquatic vegetation and fish use are limited. In these situations, hydraulic modifications likely have a minimal effect on fish and invertebrates. However, if the effluent is of a different density than the ambient water, stratification of the basin can occur, which can have severe water quality impacts, most notably through eutrophication and benthic anoxia (Fischer et al. 1979).

To prevent the deposition of debris in the outfall and the diffuser ports, minimum velocities are often required (Fischer et al. 1979). Large velocities can alter nearshore circulation patterns by mixing otherwise distinct water masses, even if outfalls are sited in deep waters (Fischer et al. 1979). Scour can also occur as a result of large discharge velocities (Rice and Kadavy 1994).

If the outfall outlet is located above the closure depth, significant impacts on local geomorphology can occur, including changing substrate, changing nearshore circulation patterns, and possibly excluding fish from key habitats with high velocities. High velocities (or changes in nearshore circulation produced by them) could also remove aquatic vegetation. Because many of the HCP species use surface waters preferentially to deeper water, the impact on fish and invertebrates would be greater the shallower the outfall outlet. The precise distribution of velocities and their change from preconstruction conditions would need to be determined with a hydraulic numerical model.

If outfalls or outfall pipes protrude above grade, alterations in local wave energy can occur. As hard points along the shoreline, outfall structures can result in the retention of wave energy in the surrounding area (Komar 1998). Regardless of the nature of the alterations, the modified relationship between topography and wave energy results in a shoreline that is out of equilibrium with natural shoreline processes (Komar 1998). As a result, wave energy artificially accumulates in some areas and is diminished in others. This redistribution of wave energy can have a number of interrelated indirect and direct effects on fish and invertebrates, including changes in substrate and changes in water column characteristics.

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[2] Closure depth is “the depth beyond which no significant longshore or cross-shore transports take place due to littoral transport processes. The closure depth can thus be defined as the depth at the seaward boundary of the littoral zone.” (Mangor, Karsten. 2004. “Shoreline Management Guidelines”. DHI Water and Environment, 294pp.)

7.4.3.10.2 Exposed Outfalls

Outfalls can alter the composition of bed and bank materials by virtue of adding material coarser than the ambient bed or by adding flow and coarsening the existing sediments. If the outfall extends into the channel, it can deflect high-velocity flows to the center of the channel and induce flow separation. Outfalls can also initiate the deposition of fine sediments leeward of the protruding structure. Protruding outfalls can reduce the local supply of coarse sediment by deflecting bed sediment from the riverbank to the center of the channel. Because the rate and caliber of sediment supplied to a channel can influence the substrate size (Dietrich et al. 1989), changes in sediment supply can alter the composition of substrate used by HCP species.

The most important hydraulic and geomorphic effect associated with exposed outfalls is the ability for the outfall to create a scour pool at its outlet. Increased velocities associated with flow constrictions created by protruding outfalls can indirectly affect HCP species by causing local bed scour around structures and corresponding sediment deposition downstream (Richardson and Davis 2001). In addition, high-velocity effluent can initiate bed scour, causing the selective removal of finer sediment, coarsening the substrate. Likewise, deposition of the finer materials originating from the outfall downstream can bury organisms and cause the substrate to become finer.

Often in the outfall design, riprap or other immobile surfaces are added to prevent erosion at the outlet. This protective material often protrudes into the channel, floodplain, or sea floor. These modifications potentially have a significant impact on the substrate surrounding the outfall outlet. They can:

- reduce hydraulic roughness,
- increase velocity and bed scour,
- reduce the supply of suitably sized substrates for spawning fish and invertebrates by limiting natural processes of channel migration and bank erosion.

Exposed outfalls can protrude into a stream or river channel and intercept the flow of sediment downstream. In this case, the outfall behaves like a groin and can disrupt the substrate. Protruding, exposed outfalls can alter the velocity field in riverine environments by redirecting flow away from the banks and toward the center of the channel, just as groins can do (Lagasse et al. 2001). The formation of flow-separation eddies adjacent to these structures results in areas of relatively low velocity in these areas and along the protected bank (Lagasse et al. 2001). The net effect is to confine the flow, contributing to increased velocity and bed scour. If outfalls do not protrude and their effluent exits at a small velocity, their impact on the flow regime is negligible.

Placement of outfalls above grade eliminates the potential to maintain riparian vegetation. This can increase the flow velocity and increase the potential for scour and substrate coarsening through a reduction in hydraulic roughness compared to vegetated conditions (Millar and Quick 1998).

Most outfalls do not require the use of sheet pilings or other impermeable structures that impede the exchange of hyporheic water with main river channels. However, if an outfall is placed parallel to a river or stream channel and it is sufficiently large, the outfall pipe itself has the potential to disrupt or eliminate hyporheic exchange, reduce lateral habitat connectivity, and alter stream temperatures buffered by groundwater inputs.

7.4.3.11 Intakes and Diversions

Intakes and diversions may involve a number of hydraulic and geomorphic modifications. As with outfalls, the design of pipes or diversion channels transporting water into upland infrastructure can interfere with the transport of sediment, if those pipes are exposed above grade. Typically, this results in coarsening and erosion of the substrate in the lee of the pipe, as well as deposition and fining on the upstream or updrift side of the pipe. These impacts can usually be avoided by constructing the pipe below grade.

Other hydraulic and geomorphic modifications are related to altered flow regimes and changes in channel geometry. Diversions reduce flows downstream, which can lead to habitat loss (Kingsford 2000) and changes in channel width, depth, and velocity (Dewson et al. 2007).

Unique to intake structures, inflowing water can attract fish toward the intake structure. All intakes should be screened in some manner to exclude fish. Unscreened intakes represent a severe hazard to all fish and their larvae; entrainment by an unscreened intake can cause mortality to all life stages of fish that inhabit areas near intakes (Newbold and Iovanna 2007). The area of influence of an intake is highly site- and design-dependent (Edinger and Kolluru 2000). To identify the area of influence of the intake, flow near any proposed unscreened intake should be investigated with a suitable hydraulic model.

The primary effect on invertebrates is through displacement of natural substrates. The emplacement of hard surfaces, either from the intake itself or piping connecting it to upland infrastructure, presents a surface on which invasive species can colonize. In the Great Lakes, extensive colonization by zebra mussels has completely clogged intake pipes (Ram et al. 1992).

7.4.3.12 Tide Gates

Tide gates regulate movement of water, sediments, and organic material between river–floodplain and marine–estuarine wetland environments. The presence of tide gates impacts hydraulic and geomorphic processes in a number of ways. Tide gates alter tidal exchange by preventing free movement of saline and fresh waters in estuarine settings. Channel geometry can be changed through restriction of freshwater flow through the tide gate. Substrate composition can be altered through changes in flow regime, similar to changes imposed by dams. The duration of inundation and water depths above tide gates do not resemble natural conditions. The hydraulic and geomorphic modifications that occur will likely adversely affect HCP species.

Hydraulic impacts extend both upstream and downstream from a tide gate, potentially affecting a range of habitats. Flow rates and flow paths are altered in the presence of a tide gate (Vandenavyle and Maynard 1994). Tide gates alter natural tidal flushing by restricting tidal flows for an unnaturally long time. In some cases, tide gates can be closed for more than 50 percent of the day (Giannico and Souder 2005).

Tide gates may alter channel geometry in several ways. When tide gates are open, high velocities through the tide gate may increase scour downstream, creating a scour pool (Giannico and Souder 2005; Zhang et al. 2000). These increased velocities are a function of the upstream–downstream differences in hydraulic head. Scour can alter the depth and width of the channel and marsh and potentially lead to habitat loss and fragmentation, as well as a loss of desirable depths if scour pools become large.

When the tide gate is closed, water velocity upstream of the gate slows, upstream water begins to pool, increasing the channel width and depth. Sedimentation increases landward of the structure due to slower velocities. The lack of two-way tidal flushing in the presence of tide gates also increases sedimentation (Anisfeld et al. 1999). Sedimentation can gradually convert aquatic habitats to terrestrial habitats as distributary channels and other features fill with sediment.

7.4.3.13 Beaver Dams

Beaver impoundments can increase vertical connectivity in riverine environments. Vertical connectivity, as defined by Ward (1989), is a measure of the exchange between groundwater and surface water through the bed and banks of the channel (i.e., hyporheic exchange). It has recently been quantitatively shown that beaver dam presence can raise local groundwater tables and thereby promote hyporheic exchange with the channel (Westbrook et al. 2006).

By creating a head differential across the structure of the dam, beaver activity directs water into the benthos. These waters either move through interflow or shallow groundwater routes to the floodplain and channel below the dam. This vertical connectivity between surface and groundwaters is associated with a number of important ecological processes, including the biogeochemical processing of nutrients and pollutants, and the creation of zones of upwelling that are preferential spawning habitats for salmonids and other species. In a study of small man-made dams in the black Prairie region of Texas it was found that the result of upstream impoundment was to increase riparian vegetation production by increasing hyporheic exchange through the riparian corridor (Duke et al. 2007). In a separate study in the Rocky Mountains, researchers showed that beaver dams and ponds increased water retention during flooding events and attenuated the wet season flows. Instead of efficient routing through the channel which was observed below the impoundments, in areas affected by beaver the flow routing was more complex and the local groundwater level was elevated during both low and high flows (Westbrook et al. 2006).

Increased hyporheic exchange can be beneficial to salmonids because the eggs of these species require well oxygenated gravels for proper egg development (Ecology 2002; Groot and Margolis 1991), and hyporheic exchange promotes increased oxygen levels in the benthos (Greig et al. 2007). Additionally, increased hyporheic exchange has been associated with nutrient uptake (Sheibley et al. 2003) and may attenuate the transport of dissolved and particulate metals (Gandy et al. 2007).

There is little if any experimental research on the impact of beaver dam removal on aquatic species. Instead, the impact must be inferred from studies which have assessed the benefits of beaver impoundments and other studies that have addressed the ramifications of small man-made dam removals. The removal of beaver dams can impact the HCP species which utilize beaver pond habitat during their life history.

7.5 Riparian Vegetation and Large Woody Debris Modifications

Riparian zones form the transition between terrestrial and aquatic systems. Riparian/shoreline vegetation is an important component of freshwater, estuarine, and marine systems. It provides shade, streambank and shoreline stability, and allochthonous inputs. Riparian vegetation also influences groundwater conveyance and storage, and the condition and complexity of aquatic habitats (Knutson and Naef 1997; Murphy and Meehan 1991). Removal or disturbance of riparian/shoreline vegetation during construction or other activities permitted under HPAs can have several potential impacts to habitat and species in each of these systems, including:

1. Altered shading and altered air and water temperature regime *(caused by alteration of vegetation canopy cover and insulating effect of boundary layer condition created by riparian forest).*
2. Altered streambank/shoreline stability *(caused by degradation of riparian vegetation, loss of vegetative cover as well as root cohesion, and reduced resistance to erosive forces.)*
3. Altered allochthonous input *(caused by reduced inputs of leaf litter, woody debris, and terrestrial insects and other biota associated with riparian vegetation.)*
4. Altered groundwater/surface water interactions *(due to the influence of altered riparian vegetation on hyporheic zone function.)*
5. Altered water quality *(due to alterations to temperature and dissolved oxygen, and effects from altered sedimentation associated with altered stability.)*
6. Altered habitat conditions *(due to loss of large woody debris (LWD), recruitment sources and reduced bank stability leading to simplification of complex bank habitat.)*

These potential impact mechanisms and related ecological stressors are discussed below for marine and estuarine, riverine, and lacustrine environments. In general, the specific effects of different kinds of HPA-permitted projects on riparian vegetation are not discussed in detail. Research on the effects of alterations to riparian vegetation has concentrated on the impact mechanism rather than the activity that sets the impact mechanism in motion.

There are some general principles reflecting the expected impacts associated with HPA-permitted projects that alter riparian vegetation. The size and configuration of a project influences how much riparian vegetation is removed or disturberd. Projects onshore and parallel to the shoreline have the greatest chance of significantly impacting riparian vegetation. For example, installing some kinds of bank armoring may require removing riparian vegetation along the entire length of the project. Other bank armoring designs may require planting riparian vegetation. On the other hand, water crossing structures such as culverts or conduits may be installed or replaced with minimal disturbance to riparian vegetation. The magnitude of effect of other projects, such as overwater structures, marinas, terminals, bridges, and shoreline modifications, is likely to fall somewhere between the effects of installing hard bank armoring and conduit installation.

7.5.1 Altered Shading, Solar Input, and Ambient Air Temperature

7.5.1.1 General Effects: All Environments

Riparian vegetation provides shade, resulting in cooler water than areas without riparian vegetation. Cool water holds more dissolved oxygen than warm water. Many potentially covered fish and aquatic invertebrate species require cool and well-oxygenated water. Because fish are ectothermic (cold-blooded), their survival is dependent upon external water temperatures, and they will experience adverse health effects when exposed to temperatures outside their optimal range (USEPA 2003). Invertebrates, which are also cold-blooded, have a similar temperature dependence.

Direct effects of removing riparian vegetation include temperatures outside the optimal growth range for fish and invertebrate species. Increases in temperature can also result in mortality. An altered temperature regime may also directly affect fish by presenting seasonal thermal barriers inhibiting migration and access to habitat and prey.

Indirect effects of temperature change can include the alteration of food webs. Invertebrate species with parasitic life-history stages may be indirectly affected by effects to host fish species. Increased water temperatures may indirectly impact potentially covered species is by allowing expanded distributions and/or increased activity of warm-water piscivorous fish that may be potential predators to potentially covered species.

Knutson and Naef (1997) identified the following mechanisms for deleterious effects on fish and other aquatic organisms from changes in water temperature and dissolved oxygen associated with the removal of riparian vegetation:

- Inhibiting growth and altering metabolism
- Amplifying effects of toxic substances
- Increasing susceptibility to disease and pathogens
- Increasing potential risk of eutrophication through increased growth of bacteria and algae.

7.5.1.2 Ecosystem-Specific Effects: Marine and Estuarine

The influence of shade on nearshore water quality parameters such as temperature is not well established in marine environments. In general, seasonal air temperature conditions, winds, currents, stratification, and tidal exchange play more dominant roles in determining marine water temperatures (Brennan and Culverwell 2004). In marine and estuarine waters, shoreline vegetation is not likely to have much influence on marine water temperatures (Lemieux et al. 2004). In general, given the limited capacity for shade to influence water temperatures in the nearshore marine environment, the direct and indirect effects of this impact mechanism on most fish species are likely to be negligible. An identified data gap in the nearshore environment is the effect of the loss of shade for predation avoidance and other factors influencing fish survival.

However, shade may strongly influence temperatures in specific habitat types under specific circumstances, such as the upper intertidal zone, tidal pools, pocket estuaries, and other habitat types that become temporarily isolated or exposed by tidal dynamics. These systems can experience increased variability in temperature and microclimate conditions in the absence of protective shading. Microclimatic conditions in the upper intertidal zone are demonstrably influenced by riparian vegetation.

Temperature plays an important role in determining the distribution, abundance, and species' survival in the upper intertidal zone. Loss of riparian shade is correlated with increased substrate temperatures and reduced humidity, which in turn are indicative of increased desiccation stress (Rice 2006). Rice (2006) compared microclimate parameters at a bulkheaded Puget Sound beach with no overhanging riparian vegetation to those at an adjacent unmodified site with extensive riparian vegetation. He documented significant differences in light intensity, air temperature, substrate temperature, and humidity levels at the modified site. Differences in peak substrate temperatures were particularly striking, averaging nearly 20^{o}F (11^{o}C) higher at the modified site: 81°F (27.3°C) versus 61.7°F (16.5°C), respectively. This is a significant finding because temperatures and desiccation are major limiting factors for upper intertidal organisms including HCP forage fish species, specifically sand lance and surf smelt (Brennan 2004; Brennan and Culverwell 2004).

Although the influence and importance of shade derived from shoreline vegetation in the Puget Sound nearshore ecosystem is not well understood, it is recognized as a limiting factor to be considered and has prompted investigations to determine direct linkages between riparian vegetation and marine organisms. One such link is the relationship between shade and surf smelt. Surf smelt spawn at the highest tide lines at high slack tide near the water's edge on coarse sand or pea gravel. They incubate for approximately two to four weeks before hatching. Embryo development is temperature dependent, with marine riparian vegetation serving to maintain lower temperatures for fish that spawn in the summer during high-temperature periods (Penttila 2000). On the basis of a comparison of adjacent shaded and unshaded spawning sites sampled in northern Puget Sound, Penttila (2001) and Lemieux et al. (2004) found significantly higher egg mortality on unshaded beaches compared to those sites with intact overhanging riparian vegetation.

Sand lance, another potentially covered species that spawns in the upper intertidal zone, would presumably encounter similar impacts on egg survival when riparian vegetation is removed. The hypothesized mechanism causing the observed higher rate of mortality was increased egg desiccation due to longer periods of direct sun exposure at sites with insufficient riparian vegetation to provide shade and other favorable microclimate conditions. The findings of Rice (2006) comparing differences in microclimate conditions and surf smelt spawn survival on shaded versus unshaded beaches strongly support this hypothesis.

Anthropogenic changes in shoreline microclimate will change the intertidal incubating environment, potentially altering developmental rates or increasing physiological stress in fish embryos (Rice 2006). Considering the influences of temperature, moisture, and exposure on the diversity, distribution, and abundance of organisms that use upper intertidal zones, additional benefits of natural shading likely will be discovered as further investigations continue (Brennan and Culverwell 2004).

Invertebrates present in habitats that are temporarily isolated or exposed by tidal dynamics could potentially be directly affected by loss of shade and microclimatic effects resulting from riparian vegetation modification. In general, however, because shading from riparian vegetation does not likely affect water temperatures in the marine environment, impacts on aquatic invertebrates associated with this mechanism are anticipated to be negligible. For example, Olympia oysters are potentially directly affected by microclimate effects stemming from loss of shade due to riparian modification. However, this species tends to occur in the lower intertidal zone, below mean lower low water (MLLW) (Dethier 2006). As such, oyster colonies are not as influenced by riparian shade and are inundated for longer periods of time, meaning that the influence of riparian zone conditions on temperature and desiccation is far more limited.

7.5.1.3 Ecosystem-Specific Effects: Riverine

In riverine environments, removal of riparian vegetation as part of HPA-permitted projects affects water temperature through a number of mechanisms. The dominant mechanism is the effect of shading on solar radiation exposure.

Exposure to the sun’s energy (due to a lack of riparian vegetation) causes an increase in water temperatures in summer when solar radiation exposure and ambient air temperatures are highest. The influence of shade on water temperature generally diminishes as the size of the stream increases because of the proportionally reduced area in which riparian vegetation can insulate against solar radiation and trap air next to the water surface (Knutson and Naef 1997; Quinn 2005; Poole and Berman 2001; Murphy and Meehan 1991, Bolton and Shellberg 2001. Increased stream temperatures can also cause a concomitant decrease in dissolved oxygen levels, an additional stressor with additive deleterious effects.

Riparian vegetation removal and alteration can cause surface waters to gain or lose heat more rapidly both by affecting local air temperatures and by increasing incident radiation and heat loss. (Quinn 2005; Bolton and Shellberg 2001; Poole and Berman 2001; Knutson and Naef 1997; Murphy and Meehan 1991). In winter, loss or degradation of the insulating capacity of riparian vegetation can decrease water temperatures and increase the incidence of ice scour.

In addition to the effects of shading, a broad array of research indicates that alterations of riparian vegetation can strongly affect temperatures even when adequate stream shading is still provided. Riparian vegetation restricts air movement, providing an insulating effect that regulates ambient air temperatures. Alterations of the riparian buffer width and vegetation composition can degrade this insulating effect, leading to greater variability in ambient air temperatures that in turn influence water temperatures (AFS and SER 2000; Bartholow 2002; Barton et al. 1985; Beschta 1991, 1997; Beschta et al. 1988;

Beschta and Taylor 1988; Brosofske et al. 1997; Brown 1970; Chen et al. 1992, 1993, 1995; Chen et al. 1999; Johnson and Jones 2000; Macdonald et al. 2003; May 2003; Murphy and Meehan 1991; Spence et al. 1996; Sridhar et al. 2004; Sullivan et al. 1990; Theurer et al. 1984; USFS et al. 1993). For example, Chen et al. (1995) found that maximum air temperatures at the margins of old-growth forest stands are elevated 3–29°F (2–16°C) relative to interior temperatures. Riparian buffer widths of 100–300 ft may be necessary to provide full ambient temperature regulation (AFS and SER 2000; Brosofske et al. 1997).

Numerous studies have documented the deleterious effects on fish species of changes in the temperature regime in freshwater stream systems (Poole et al. 2001; Sullivan et al. 2000). Water temperatures significantly affect the distribution, health, and survival of fish, especially salmonids. Alteration of the temperature regime in riverine systems due to the alternation of riparian vegetation is a well-documented stressor on native fish populations. Because fish are ectothermic (cold-blooded), their survival is dependent upon external water temperatures, and they will experience adverse health effects when exposed to temperatures outside their optimal range (USEPA 2003).

As stream temperatures rise, dissolved oxygen content decreases. Salmon, trout and other cold water fish, and many aquatic invertebrates require cool and well-oxygenated water, with a preferred temperature range of 40 to 58° Fahrenheit (F) (5.5 to 14.4° Celsius [C]), and dissolved oxygen levels of greater than 5 parts per million *(***need citation).* Selong et al. (2001) found that for age-0 bull trout, the upper lethal temperature is 70°F (20.9°C), and optimal growth occurred at 56°F (13.2°C); feeding declined significantly above 61°F (16°C). Also, a laboratory study showed that Dolly Varden displayed decreased appetite above 61°F (16°C), and temperatures were lethal above 68°F (20°C) (Takami et al. 1997).

It is useful to note that the effects of altered stream temperatures are not uniformly negative in all cases. For example, in light-limited streams, selective thinning of forests can have a positive effect on fish. In northern California, cutthroat and rainbow trout responded positively to increased light from riparian thinning through increased primary productivity that stimulated the food web (Wilzbach et al. 2005).

Several potentially covered invertebrate species would be impacted by the water conditions that could accompany freshwater riparian vegetation removal. In fresh water, the California floater and Western ridged mussels and the giant Columbia River limpet are all intolerant of low oxygen and high temperature conditions (WDNR 2006b) that can occur along shorelines lacking vegetation, although species profiles identified for this paper did not provide thresholds or ranges.

7.5.1.4 Ecosystem-Specific Effects: Lacustrine

Still-water systems, such as lakes or ponds, are subject to the same effects as those found in riverine and marine systems. As with both marine and riverine ecosystems, modifications to riparian vegetation in lacustrine systems can alter shading, solar input, and air temperature; alter allochthonous inputs; alter groundwater – surface water exchange, and alter habitat complexity. The dominant effect of vegetation removal is the loss of shading and increase in solar radiation exposure. These effects are generally less pronounced than those found in riverine systems, however, because lacustrine systems have a greater amount of unshaded surface area, large water volumes, and are often seasonally stratified, which can restrict water circulation. Little research has been conducted on the effect of riparian vegetation on lake temperatures. Lake surface area is usually dominated by shade-free open water zones, and thus the shaded margin does not have a large impact on overall lake temperatures (Lauck et al. 2005). As a result, riparian rehabilitation activities are not expected to alter lake temperatures. In lakes, water temperatures generally change gradually through the year with the seasons and show less change from night to day. However, loss of shading of nearshore littoral habitats can result in changes to water temperature of a sufficient magnitude to create thermal barriers for various fish species (Rice 2006; Carrasquero 2001), perhaps leading to changes in species composition.

7.5.2 Altered Streambank/Shoreline Stability

7.5.2.1 General Effects: All Environments

The root structure of riparian/shoreline vegetation resists the shear stresses created by moving water and thus retards bank cutting by streams, stabilizes streambanks and shorelines, maintains undercut banks along stream margins, and inhibits sediment from entering streams or marine shorelines by dissipating the erosive energy of flood waters, wind, and rain (Knutson and Naef 1997, Levings and Jamieson 2001; Brennan and Culverwell 2004, Waters 1995). Much of the scientific literature discusses the potential impacts of increased sediment as it relates to salmonids (Quinn 2005; Waters 1995; Furniss et al. 1991).

The bank instability that can result from the removal of riparian vegetation associated with bank protection structures can impact covered invertebrates as well as covered fish by elevating suspended solids concentrations and increasing the volume of fine sediments deposited.

The indirect effects on fish and invertebrates as a result of reduced shoreline stability relate to the increased probability of slope failures. Although slope failures occur naturally and are an important process that maintains proper substrate of adjacent beaches (Finlayson 2006), the immediate and unnatural impacts of riparian vegetation removal can adversely affect HCP species. These effects result from the increased turbidity of adjacent waters (Herrera 2006) and the potential burial of invertebrates.

7.5.2.2 Ecosystem-Specific Effects: Marine and Estuarine

Marine riparian vegetation clearly plays a role in stabilizing marine shorelines, particularly bluffs and steep slopes (Brennan and Culverwell 2004; Lemieux et al. 2004; Desbonnet et al. 1995; Myers 1993), but the specific mechanisms are not as well understood as they are in freshwater environments. The extent to which vegetation affects beach and slope stability varies depending on shoreline characteristics and the types of vegetation present (Lemieux et al. 2004; Myers 1993).

For marine shorelines, and particularly those in areas with steep and eroding bluffs, native vegetation is usually the best tool for keeping the bluff intact and/or minimizing erosion (Brennan and Culverwell 2004). On steeper slopes, marine riparian vegetation helps to bind the soils. Disturbing the face or toe of a bluff or bank may cause destabilization, slides and cave-ins (Clark et al. 1980, in Brennan and Culverwell 2004). Removal of the vegetation that helps to stabilize the face, or excavation along the face, increases the chance of slumping, which results in imperiled structures, lost land, a disruption to the ecological edge-zone, and increased sedimentation to the aquatic environment (Brennan and Culverwell 2004). On shorelines with shallower slopes, marine riparian vegetation dissipates wave energy, reducing erosion and promoting the accumulation of sediments.

Sedimentation and siltation impacts resulting from destabilized shorelines and bluffs can alter the ability of marine shorelines to support eelgrass beds (Finlayson 2006) and other marine littoral vegetation (Clark et al. 1980). This reduces habitat complexity, alters potential allochthonous inputs, and reduces the potential prey base available to the HCP species by inhibiting colonization by organisms upon which fish and invertebrates depend and altering the epibenthic assemblages upon which numerous fish species are dependent.

7.5.2.3 Ecosystem-Specific Effects: Riverine

In riverine environments, bank cohesion plays an important role in regulating channel width and substrate. The removal of riparian trees and understory can dramatically alter stream bank stability and the filtering of sediments from overland flow (Kondolf and Curry 1986; Shields 1991; Shields and Gray 1992; Simon 1994; Simon and Hupp 1992; Waters 1995), increasing erosion via wind, rain and current, and increasing inputs of fine sediment (Bolton and Shellberg 2001, Waters 1995).

By dissipating erosive energy, the root structure of riparian/shoreline vegetation resists the shear stresses created by flowing water and thus retards bank cutting by streams, stabilizes streambanks and shorelines, maintains undercut banks along stream margins, and inhibits sediment from entering streams (Knutson and Naef 1997, Levings and Jamieson 2001, Brennan and Culverwell 2004).

Increased delivery of coarse and fine-grained sediment to a river can affect water quality and habitat conditions. Slumping of unstable banks can bury eggs and larval fish, as well as HCP invertebrates. Although some specifics are known for marine invertebrate species (Hinchey et al. 2006), it is unknown what tolerance limits the HCP freshwater mollusks may have with respect to burial.

In addition to burial, increased sedimentation from destabilized banks can reduce or alter invertebrate habitat so that growth and fitness are reduced (or that the distribution of invertebrate species changes), and indirectly can trigger effects associated with food web alterations.

Much of the scientific literature discusses the potential impacts of increased sediment as it relates to salmonids. Increased sedimentation to streams or lakes can significantly affect the spawning success of salmonids (Quinn 2005; Waters 1995; Furniss et al. 1991). Increased sedimentation may also affect other HCP fish species, directly through gill damage or indirectly through impacts on their habitat.

7.5.2.4 Ecosystem-Specific Effects: Lacustrine

Riparian vegetation in lacustrine systems plays a similar role to that in marine, estuarine, and riverine systems, stabilizing shorelines against the erosive forces of wind-driven waves and boat wakes (Carrasquero 2001). The loss of riparian and emergent vegetation promotes shoreline erosion, creating an erosive cycle that further increases vegetation loss, with a resultant adverse effect on nutrient cycles. For example, loss of emergent vegetation can promote erosive cycles that preclude the recovery and reestablishment of such vegetation. Decreased emergent vegetation density results in altered sediment transport patterns. If sediments are not replenished, additional emergent vegetation loss can result, leading to additional shoreline erosion (Rolletschek and Kühl 1997).

7.5.3 Altered Allochthonous Input Including Large Woody Debris

7.5.3.1 General Effects: All Environments

In lakes, estuaries, and marine environments, riparian vegetation inputs contribute to habitat complexity and organic matter retention. Adjacent riparian vegetation is one source of woody debris. Woody debris provides essential cover for fish and invertebrate species, protection from currents, and foraging opportunities (Quinn 2005; Naiman et al. 2000; Knutson and Naef 1997; Murphy and Meehan 1991).

Removal of riparian vegetation for HPA-permitted projects diminishes externally derived (allochthonous) input into the aquatic environment, which can affect the prey base available to fish, the forage detritus available for benthic macroinvertebrates, future large woody debris (LWD) recruitment, and aquatic habitat complexity, diminishing the quality and complexity of habitat and species diversity of fish and benthic macroinvertebrates (Murphy and Meehan 1991).

7.5.3.2 Ecosystem-Specific Effects: Marine and Estuarine

The importance of allochthonous inputs of litter to marine ecosystems is apparent (Lemieux et al. 2004; Brennan and Culverwell 2004), although not as well documented as the linkages established for freshwater systems. Marine riparian vegetation is a known source of organic matter, nutrients, and macroinvertebrate prey items, and the recruitment of these materials is diminished when riparian vegetation is removed or modified (Lemieux et al. 2004; Spence et al. 1996; Maser and Sedell 1994; Williams et al. 2001; Brennan et al. 2004). The reduction of allochthonous inputs from riparian vegetation would inhibit nutrient inputs that contribute to the productivity of the intertidal food web and associated prey resources or forage base.

Allochthonous inputs of organic material and large wood from marine riparian systems have demonstrable effects on nearshore habitat conditions. In estuaries and marine environments, woody debris increases habitat complexity, affording cover for fish, protection from currents, and foraging opportunities (Quinn 2005).Woody debris also provides foraging, refuge, spawning and attachment substrate for aquatic invertebrates and algae in the marine/estuarine environment (Brennan and Culverwell 2004).

Driftwood and other LWD help build and maintain beach habitat structure in marine environments. LWD functions for beach stability include its contribution to roughness and sediment trapping as well as to inputs of organic matter, moisture, and nutrients that assist in the establishment and maintenance of dune and marsh plants.

There has been little research on the importance of wood in supporting beach structure and connectivity between estuarine environments. However, many shorelines in the Puget Sound area contain considerable wracked wood in the supratidal zone (Sobocinski 2003) which may serve to reduce shoreline erosion and protect the sediments which are the foundation for the shallow water environment. Riparian vegetation can help accumulate beach wrack by providing a complex surface to collect and retain loose marine debris.

Herrera (2005) described how multiple layers of LWD along a shoreline could provide effective energy dissipation, decreasing the amount of wave reflection during high water levels by increasing the roughness of the shoreline and by decreasing its slope relative to a vertical bulkhead. Because LWD is used in some marine soft-shore armoring applications to attenuate wave energy and lessen the potential for erosion, it is assumed that naturally occurring LWD on beaches would do the same, but this has not been empirically tested.

Marine shorelines modified by human activities tend to have less large woody debris and driftwood than unmodified beaches (Herrera 2005; Higgins et al. 2005). MacDonald et al.(1994) reported that shoreline armoring limited driftwood accumulation on a beach. Higgins et al. (2005) suggested that the mechanisms for the apparent reduction in LWD appeared to be the removal of adjacent riparian vegetation during and following placement of bank protection; reduced shoreline roughness at armored sites which causes more LWD to be transported away; and limited upper intertidal and backshore areas that allow for LWD deposition above tidal elevations that are routinely inundated.

LWD accumulations on beaches have been anecdotally correlated with increased distribution and suitability of spawning substrate for forage fishes (Herrera 2005), suggesting that the effects of LWD on sediment transport and distribution are meaningful from an ecological perspective.

Although little evidence has shown the direct impacts of the loss of large woody debris on HCP species on marine shorelines, differences in the survivability and diversity of lower trophic species near accumulations of woody debris on marine shorelines has been documented (Storry et al. 2006). Because one of the sources of LWD on marine shorelines is riparian vegetation, the loss of riparian vegetation would represent a loss of productivity on the upper foreshore and may partially explain the dramatic difference in supratidal productivity between modified and unmodified shorelines (Sobocinski 2003).

Recent studies indicate that for those salmonids known to be most dependent upon shallow marine nearshore habitats (i.e., Chinook and chum salmon, coastal cutthroat trout), insects derived from the terrestrial environment comprise major portions of their diets (Wipfli 1997; Levings and Jamieson 2001; Brennan et al. 2004; Brennan and Culverwell 2004; Toft and Cordell 2006). Stomach analyses of juvenile salmon have shown significant numbers of terrestrial insects, with higher numbers of insects found in those samples along marine shorelines with intact marine riparian vegetation (Sobocinski 2003). Vegetation modifications that reduce the abundance of terrestrial insects and/or reduce the likelihood of insect recruitment to the marine environment (e.g., removal of overhanging branches) are likely to result in localized reductions in the prey base available for these species.

7.5.3.3 Ecosystem-Specific Effects: Riverine

Riparian detritus and other allochthonous materials are the primary source of organic fodder in headwater streams, forming the basis for the food web (MacBroom 1998). Proximity of riparian vegetation provides terrestrial macroinvertebrates, which supplement the diets of fishes, and detritus such as leaves and branches, which provide food sources for benthic macroinvertebrates (Knutson and Naef 1997; Murphy and Meehan 1991; Bilby and Bisson 1998; Cummins 1980 (1975?)). As rivers increase in order and grow in size, these materials are processed and recycled by an increasing diversity of organisms (Vannote et al. 1980).

Removal of freshwater riparian vegetation as part of HPA-permitted projects would decrease the input of allochthonous materials to the nearby aquatic environment and food web, and diminish the ability of the system to support higher trophic organisms, including most of the freshwater HCP species. When the nutrients provided by riparian vegetation are altered, the net primary production of organic matter is affected, and the abundance and availability of prey resources for HCP species could be reduced (Bisson and Bilby 1998). Without allochthonous inputs, the forage detritus available for benthic macroinvertebrates is compromised, also diminishing habitat and species diversity of these prey items (Murphy and Meehan 1991). The grazing of organic material by macroinvertebrates such as caddisflies, stoneflies, and mayflies, is known to support the HCP salmon species (Hawkins et al. 1982; Murphy and Meehan 1991; Bilby and Bisson 1992), bull trout (Wydoski and Whitney 2003; Goetz et al. 2004), and sturgeon (Wydoski and Whitney 2003; Adams et al. 2002; Emmett et al. 1991). Although terrestrial and adult aquatic insects are important (Bjornn and Reiser 1991), juvenile salmon in streams have been found to be primarily supported by autochthonous organic matter (Bilby and Bisson 1992).

LWD promotes river-floodplain connectivity and thus the export of coarse particulate organic matter (CPOM) to the channel (Junk et al. 1989; Tockner et al. 1999). Floodplains have been shown to produce nutrient-rich organic matter (i.e., more live algae than decaying organic matter) which can serve as an import food resource for zooplankton (Muller-Solger et al. 2002; Schemel et al. 2004) and thus bolster aquatic food webs. Floodplains have been shown to act as nutrient sinks and carbon sources for adjacent channels (Tockner et al. 1999; Valett et al. 2005). The literature indicates that any activity which promotes the reconnection of once severed floodplain-channel pathways can be expected to have a net benefit for aquatic organisms (Crain et al. 2004; Schemel et al. 2004; Tockner et al. 1999). This is partially due to the fact that floodplains are vital for the transfer of energy (in the form of carbon) from terrestrial ecosystems to aquatic, especially in lowland river systems (Junk et al. 1989; Thoms 2003).

In streams, large woody debris (LWD) from riparian vegetation influences channel morphology and habitat complexity, retains organic matter, regulates the storage and transport of sediment, creates and maintains hydraulic complexity that contributes to fish habitat, and provides essential cover for fish (Knutson and Naef 1997; Murphy and Meehan 1991; Naiman et al. 2002; Quinn 2005). The removal of freshwater riparian vegetation limits the future input of woody debris as a habitat structure element and can limit habitat complexity, foraging opportunities, and predator avoidance (Schmetterling et al. 2001; Spence et al. 1996; Quinn 2005). Juvenile salmonid abundance in rivers in winter, particularly juvenile coho salmon abundance, is positively correlated to abundance of LWD (Hicks et al. 1991).

Riparian vegetation and LWD are important for bank-side habitat, providing shade, lower temperatures, and cover from predation. Radio-tagged cutthroat trout have been observed using pools associated with LWD for cover (Harvey et al. 1999). The use of submerged riparian vegetation during early development has been hypothesized to increase Columbia River white sturgeon recruitment (Coutant 2004).

Within channels, approximately 70 percent of structural diversity is derived from root wads, trees, and limbs that fall into the stream as a result of bank undercutting, mass slope movement, normal tree mortality, or windthrow (Knutson and Naef 1997). In small streams, LWD is a major factor influencing pool formation in plane–bed and step–pool channels. Bilby (1984) and Sedell et al. (1985) found that approximately 80 percent of the pools in several small streams in southwest Washington and Idaho were associated with wood. In larger streams, the position of LWD strongly influences the size and location of pools (Naiman et al. 2002). In larger streams, LWD is typically oriented downstream due to powerful streamflow, which favors the formation of backwater pools along margins of the mainstem (Naiman et al. 2002). The hydraulic complexity created by LWD encourages the capture and sequestration of other allochthonous inputs, making these materials more available to the food chain through grazing and decomposition (Knutson and Naef 1997; Murphy and Meehan 1991; Naiman et al. 2002; Quinn 2005)

Woody debris increases the hydraulic roughness and impounds sediment behind logjams (Buffington and Montgomery 1999; Gippel et al. 1996; Harvey et al. 1987; Keller and Swanson 1979; Manga and Kirchner 2000; Montgomery et al. 1996; Shields and Gippel 1995). Local aggradation behind stable logjams can raise the bed-surface elevation of channels and increase the potential for floodplain inundation and lateral channel migration (Abbe and Montgomery 2003). Consequently, changes in wood loading and the age structure and composition of riparian forests can dramatically influence channel dynamics and the potential for lateral channel migration in unconfined river systems (Brummer et al. 2006).

Studies have shown that channels with LWD retain more bedload (Faustini and Jones 2003) and particulate organic matter (POM) (Cordova et al. 2007; Diez et al. 2000) than similar reaches without wood. Depending on channel form and wood size and orientation, LWD-induced pools and bars can occur upstream, downstream, and/or lateral to wood structures (Abbe and Montgomery 2003; Gurnell and Petts 2006; Kail 2003).

The presence of LWD within channels has been shown to promote floodplain connection during storm flow conditions by increasing flow resistance within the channel (Dudley et al. 1998). Increased channel roughness promotes backwater conditions which locally connect floodplain and channel habitat. LWD promotes floodplain connectivity by diverting flow into side channels (Abbe and Montgomery 1996; Brummer et al. 2006). Side channel and floodplain habitat that are connected to the mainstem create more accessible habitat for foraging and rearing, altering the composition and/or abundance of accessible food sources for HCP species.

The addition of wood to a channel will, depending upon the size, quantity, orientation, and channel dimensions, create a complex depositional environment. Studies by Stewart and Martin (2005) and Baillie and Davies (2002) have shown that LWD promotes in-channel sediment storage as the logs deflect flow and increase channel roughness. Large woody debris promotes heterogeneity in channel form by creating flow divergence and changing local base-level (Latterell et al. 2006). These processes lead to sediment deposition in both upstream pools and downstream eddies (Beechie and Sibley 1997). Both flow diversion and base-level increase will lead to increased floodplain connection and side channel activation/formation (lateral connectivity). Additionally, complex flow patterns will lead to increased flow through gravel bars, channel embankments, and riffles (vertical connectivity). LWD removal adversely affects ecosystem connectivity in the fluvial environment by reversing these effects. LWD removal has been shown to promote scour (Diez et al. 2000) leading to a drop in water surface elevation and lateral disconnection. LWD removal will also reduce flow resistance and simplify flow paths (Curran and Wohl 2003) which could lead to reduced hyporheic exchange.

LWD increases the abundance of and access to floodplain habitat (Young 1991). LWD-induced floodplain-channel connection is expected to augment allochthonous carbon budgets in restored channels. These improved conditions will in turn be beneficial to HCP species that occur in riverine environments affected by LWD, especially those which favor floodplain habitat (e.g. coho, sockeye, and Chinook salmon). Conversely, the removal of LWD may limit access to floodplain habitat is expected to decrease allochthonous carbon inputs and the storage and cycling of nutrient inputs to the stream channel, and impact those species which utilize these areas for foraging, rearing, and refuge.

Fish rely on habitat complexity for cover and refuge (Cederholm et al. 1997; Everett and Ruiz 1993; Harvey et al. 1999). Specifically:

- Bryant et al. (2007) have documented fish species such as Dolly Varden, coho, steelhead, and cutthroat trout utilizing complex habitats with LWD.

- In a study of Smith Creek in northwest California, Harvey et al. (1999) found that tagged adult coastal cutthroat trout moved more frequently from pools without LWD than from pools with LWD. They hypothesized that the habitat created by LWD attracts fish, and once fish establish territory within the desirable habitat, they remain there longer. Fausch et al. (1995) and others have criticized studies such as Harvey et al. (1999) because it is difficult to determine if increased abundance in treatment sites is due to increased populations or simply just concentrations of fishes that would have thrived equally well in other habitat.

- A study by Cederholm et al. (1997) on a tributary of the Chehalis River found that increasing habitat complexity by adding LWD caused an increase in winter populations of juvenile coho salmon and age-0 steelhead.

While no studies of LWD and HCP invertebrates were found, other studies have shown varying effects of LWD on invertebrates in rivers.

- LWD can serve as a substrate for algal growth (Bowen et al. 1998).

- In the Ohe River, Germany, Hoffman (2000) showed a intimate connection between all lifestages of the lepidostomatid caddisfly, *Lasiocephala basalis* and LWD.

- Hilderbrand et al. (1997) noted no change in macroinvertebrate populations following a wood addition experiment in Virginia.

- Spanhoff et al. (2006) noted a net negative impact of wood addition on stream macroinvertebrates.

- LWD can serve as macroinvertebrate habitat (Rolauffs et al. 2001; Warmke and Hering 2000)

7.5.3.4 Ecosystem-Specific Effects: Lacustrine

Lacustrine systems are dependent on allochthonous inputs from riparian systems, receiving this material directly through litter fall and windblown detritus from their own riparian areas, as well as allochthonous material transported into the system by rivers and streams. The importance of riparian vegetation to the lacustrine nearshore area has been documented by Scheuerell and Schindler (2004).

Several studies of lakes in Washington State have demonstrated a loss of both productivity and diversity in their food webs in the presence of human development (Scheuerell and Schindler 2004). Studies have identified large woody debris and emergent and riparian vegetation as crucial to the maintenance of lake productivity and food webs (Roth et al. 2007; Smokorowski et al. 2006). In areas of intense shoreline development where riparian vegetation has been thinned or removed, littoral coarse woody debris is greatly diminished or entirely absent (Francis and Schindler 2006).

The loss of productivity due to the lack or removal of LWD, in conjunction with other human activities (e.g., fishing), can initiate a collapse of piscivorous fish species in freshwater lakes (Roth et al. 2007). It can also reduce the amount of periphyton available for other lower trophic fishes and remove substrate suitable for HCP species (Smokorowski et al. 2006).

Coarse woody debris has been shown to be an essential habitat component for sockeye salmon in lakes and is important for the overall health of lacustrine shoreline ecosystems (Naiman et al. 2000). Sockeye salmon are the primary HCP species potentially affected by shoreline modifications along lakeshores. Lake shorelines represent crucial spawning habitat for sockeye (Burgner 1991; Scheuerell and Schindler 2004). Tabor et al. (2004, 2006) have documented the importance of small woody debris as habitat structure in Lake Washington, where it may provide important periodic refuge from predators for juvenile Chinook salmon.

Woody debris along lakeshores provides important habitat for prey species (Sass et al. 2006) and invertebrate populations (Bowen et al. 1998). In a study of the effects of woody debris on the aquatic food web of Little Rock Lake, Wisconsin, Sass et al. (2006) removed more than 75 percent of the woody debris from a treatment section of the lake. They found that within the treatment section, increased largemouth bass predation caused a decrease in yellow perch abundance; subsequently, bass diets shifted to terrestrial prey and bass growth rates decreased. This study clearly indicates that the cover provided by woody debris plays an important role in lacustrine food webs and that alterations to this habitat will impact resident aquatic species.

7.5.4 Altered Groundwater/Surface Water Interactions

7.5.4.1 General Effects: All Environments

Alteration or removal of riparian vegetation would appreciably change the interface between plants, soil, and water on and near the bank surface. Riparian vegetation acts as a filter for groundwater, removing sediments and taking up nutrients (Knutson and Naef 1997).

Indirect impacts on HCP species from the reduction of groundwater–surface water interactions include increased nutrient loading, altered temperatures, and habitat changes. Any activity that affects riparian or nearshore areas will degrade the buffering capability of the terrestrial–aquatic ecotone. Numerous studies have shown that wide stream buffers are effective at attenuating nutrients (Feller 2005; Mayer et al. 2005), herbicides (Gay et al. 2006), and sediment loading (Jackson et al. 2001). Riparian vegetation retards overland flow, promotes infiltration, and assimilates shallow groundwater nutrients. When this vegetation is removed through any HPA-permitted activity, nutrients and pollutants will be more efficiently transported from upland sources to downgradient water bodies. Forested buffers can effectively remove nutrients in shallow groundwater. In a study of a forested buffer in Alabama, a 33-ft (10-m) buffer reduced the groundwater nitrate concentration by 61 percent (Schoonover and Williard 2003). In a subsequent study of a forested wetland buffer, a buffer averaging 125 ft (38 m) wide reduced the nitrate concentration by 78 percent and total phosphorus by 66 percent (Vellidis et al. 2003).

7.5.4.2 Ecosystem-Specific Effects: Marine and Estuarine

While less well studied, similar effects on groundwater – surface water exchange are likely to occur in marine systems as in riverine systems.

Bank armoring that requires the removal of riparian vegetation may lead to localized increases in substrate temperature due to the loss of cool groundwater flow (Penttila 2001). Riparian vegetation modifications can alter soil characteristics, infiltration rates, and groundwater transport that provides freshwater seepage to the intertidal zone. Alteration of these functions can directly reduce or eliminate rearing habitat for Olympia oysters and other marine organisms that use freshwater seeps along marine shorelines. These changes could also inhibit the suitability of marine shorelines for supporting eelgrass beds (Finlayson 2006). By affecting eelgrass habitat, these alterations could generate indirect food web effects for both invertebrates and fish.

7.5.4.3 Ecosystem-Specific Effects: Riverine

Riparian vegetation, in conjunction with upland vegetation, moderates stream flow by intercepting rainfall, contributing to water infiltration, and using water via evapotranspiration (Knutson and Naef 1997). Plant roots increase soil porosity, and vegetation helps to trap water flowing on the surface, thereby aiding in infiltration (Knutson and Naef 1997). Water stored in the soil is later released to streams through subsurface flows. Through these processes, riparian and upland vegetation help to moderate storm-related flows and reduce the magnitude of peak flows and the frequency of flooding (Knutson and Naef 1997). Riparian vegetation, the litter layer, and silty soils absorb and store water during wet periods and release it slowly over a period of months, maintaining stream flows during rainless periods (Knutson and Naef 1997).

The interface between flow within the hyporheic zone[1] and the stream channel is an important buffer for stream temperatures, so alteration of groundwater flow can affect stream temperature as well (Poole and Berman 2001). The magnitude of the influence depends on many factors, such as stream channel pattern, structure of the alluvial aquifer, and variability in the stream hydrograph (Poole and Berman 2001).

7.5.5 Altered Water Quality

7.5.5.1 General Effects: All Environments

Alterations to the water quality parameters of temperature and dissolved oxygen are discussed as a result of alterations to shading. See Altered Shading, Solar Input, and Ambient Air Temperature. Alterations to turbidity are discussed under Altered Streambank/Shoreline Stability.

Riparian zones are an ecotone between open water and terrestrial environments. Riparian vegetation retards overland flow, promotes infiltration, and assimilates shallow groundwater nutrients. When this vegetation is removed through any HPA-permitted activity, nutrients and pollutants will be more efficiently transported from upland sources to downgradient water bodies. Activities that affect riparian areas will also degrade the buffering capability of the terrestrial–aquatic ecotone. Once convoluted flow paths become simplified, the transfer of sediment, nutrient, and pollutants from terrestrial to aquatic systems increases. In theory, the restoration of riparian areas will decrease pollutant loading to the channel and benefit aquatic organisms.

Considerable research regarding riparian buffers and pollutant loading has been conducted and although the results have been mixed there has been a general trend observable in the data. Numerous studies have shown that wide stream buffers are effective at attenuating nutrients (Feller 2005; Mayer et al. 2005), herbicides (Gay et al. 2006), and sediment loading (Jackson et al. 2001).

Riparian buffer widths play an important role in determining the efficacy of a buffer at removing pollutants. Reviews of the literature have indicated that the most effective buffers are greater than 98 ft (30 m) in width (Hickey and Doran 2004) and this applies to all streams, including first order tributaries (Mayer et al. 2005). Riparian buffers which meet these criteria have been shown to remove as much as 95 percent of the influent total phosphorus (Hickey and Doran 2004). In lowland eutrophic systems this would serve as a net benefit to the aquatic ecosystem, especially in nutrient impacted lakes. In a study of a forested buffer in Alabama, a 33-ft (10-m) buffer reduced the groundwater nitrate concentration by 61 percent (Schoonover and Williard 2003). In a subsequent study of a forested wetland buffer, a buffer averaging 125 ft (38 m) in width reduced the nitrate concentration by 78 percent and total phosphorus by 66 percent (Vellidis et al. 2003).

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[1] The zone of hydrologic interchange between groundwater and surface water in stream channels.

7.5.5.2 Ecosystem-Specific Effects: Riverine

Reduced riparian buffer width can lead to decreased buffering capacity and an associated increase in pollutant and nutrient loading.

Dramatic increases in sediment and organic matter export occur immediately following removal or disturbance of LWD. These short-term direct impacts affect water quality parameters such as turbidity and dissolved oxygen levels, which in turn affect the viability of multiple lifestages of fish. These sedimentation impacts could also affect invertebrate distribution and fitness.

7.5.6 Altered Habitat Conditions

7.5.6.1 General Effects: All Environments

The removal of riparian/shoreline vegetation limits the future input of woody debris as a habitat structure element, and can limit habitat complexity, foraging opportunities, and predator avoidance in marine, estuarine, riverine, and lacustrine environments (Quinn 2005; Schmetterling et al. 2001; Spence et al. 1996).

Upland development has greatly reduced the amount of vegetation and nutrients available to the marine system. Such modifications have resulted in decreased abundance and taxa richness in both benthic and infaunal invertebrate and insect assemblages (Brennan and Culverwell 2004).

7.5.6.2 Ecosystem-Specific Effects: Marine and Estuarine

In marine environments, driftwood and/or large woody debris (LWD) from the riparian zone contributes to build and maintain beach habitat structure. Documented LWD functions for beach stability include its contribution to roughness and sediment trapping (Gonor et al. 1988; Brennan and Culverwell 2004) and to inputs of organic matter, moisture, and nutrients that assist in the establishment and maintenance of dune and marsh plants (Williams and Thom 2001). Eilers (1975) found that piles of downed trees in the Nehalem salt marsh (Oregon) trapped enough sediment to support vegetation, wherein marsh islands that trapped sedge seeds provided an elevated substrate for less salt-tolerant vegetation. Herrera (2005) suggested that driftwood at the top of the beach may also slow littoral drift and erosion by reducing wave energy and wave reflection energy and by creating pockets where larger sediment will accumulate. It has been suggested that estuarine wood can affect water flow and subsequent formation of bars and mudbanks (Gonor et al. 1988). The beneficial habitat structure functions of LWD along marine shorelines may be maximized if trees that fall perpendicular to beaches typically remain in place, as in the case of a study that found fallen trees tend to stay in place along Thurston County shorelines (Herrera 2005). The perpendicular alignment of LWD across the beach provides the LWD structure for the widest possible portion of the aquatic habitat, thus maximizing the potential area for sediment trapping and organic matter contributions.

The effects of modification of riparian vegetation on the structural habitat of fish have not been as well studied in marine systems as in freshwater environments. Sobocinski (2003) reports that food sources used by salmonids were directly related to the structural complexity provided by natural LWD-laden shorelines. Therefore, it is expected that the loss of such complexity will compromise salmonid food sources.

It is uncertain what role structural complexity plays in the life-history cycle of HCP invertebrate species, if any. In marine areas, Newcomb's littorine snail is found primarily in association with a narrow band of nearshore intertidal habitat that contains certain marsh plant species (Larsen et al. 1995); because detailed reproductive and habitat needs are not known, it might be conservatively assumed that Newcomb's littorine snail is subject to habitat loss or direct desiccation if riparian aquatic vegetation is impacted.

The removal or reduction of LWD in nearshore areas introduces the potential for direct impacts on fish and invertebrates via altered habitat suitability by reducing potential cover and stability of existing habitats. Indirect effects on habitat could include the reduced potential for development of future habitat.

7.5.6.3 Ecosystem-Specific Effects: Riverine

In riverine systems, riparian vegetation contributes to habitat complexity by maintaining bank stability, providing bank structure, and providing overhanging vegetation. Alterations of riparian vegetation are likely to affect habitat complexity in ways that are detrimental to HCP species. The quantity of woody debris in channels in the Pacific Coastal ecoregion has decreased over time as a result of various land use practices, including the removal of wood from rivers for navigation and fish passage, splash damming, and clearing of riparian trees (Bisson and Bilby 1998; Murphy and Meehan 1991; Jennings et al. 2003; Schindler et al. 2000; Abbe and Montgomery 2003; Montgomery et al. 2003; Collins et al. 2003).

A key mechanism by which riparian vegetation contributes significantly to habitat complexity in freshwater environments is by the input of woody debris (Naiman et al. 2002). LWD is a primary determinant of channel form in streams, creating pools and waterfalls and affecting channel width and depth (Bilby and Ward 1991; Bisson et al. 1987; Keller and Swanson 1979; Montgomery and Buffington 1993). ; Woody debris input in streams is important in controlling channel morphology, regulating the storage and transport of sediment and particulate organic matter, and creating and maintaining fish habitat (Abbe and Montgomery 2003; Collins et al. 2003; Jennings et al. 2003; Montgomery et al. 2003; Murphy and Meehan 1991; Naiman et al. 2002). Within streams, approximately 70 percent of structural diversity is derived from root wads, trees, and limbs that fall into the stream as a result of bank undercutting, mass slope movement, normal tree mortality, or windthrow (Knutson and Naef 1997). Additionally, juvenile salmonid abundance in winter, particularly juvenile coho salmon, is positively correlated to abundance of LWD (Hicks et al. 1991). In a study of Smith Creek in northwest California, Harvey et al. (1999) found that tagged adult coastal cutthroat trout moved more frequently from pools without large woody debris than from pools with large woody debris. They hypothesized that the habitat created by wood attracts fish, and once fish establish territory within the desirable habitat, they remain there longer. A study by Cederholm et al. (1997) in a tributary of the Chehalis River found that wood additions caused an increase in winter populations of juvenile coho salmon and age-0 steelhead populations. A study by Everett and Ruiz (1993) also found that fish use complex environments and the structure of the LWD itself for cover and refuge.

In small streams, LWD is a major factor influencing pool formation in plane-bed and step-pool channels. Bilby (1984, in Naiman et al. 2002) and Sedell et al. (1985, in Naiman et al. 2002) found that approximately 80 percent of the pools in several small streams in southwest Washington and Idaho are associated with wood.

The position of LWD strongly influences the size and location of pools in larger streams, as well (Naiman et al. 2002). In larger streams, LWD is typically oriented downstream due to powerful streamflow, which favors formation of backwater pools along margins of the mainstem (Naiman et al. 2002).

The hydraulic complexity created by LWD encourages the capture and sequestration of other allochthonous inputs, making these materials more available to the food web through grazing and decomposition (Quinn 2005; Naiman et al. 2002; Knutson and Naef 1997; Murphy and Meehan 1991).

Sediment accumulated by woody debris in streams provides substrate for the establishment of early successional plant species that can mature into riparian vegetation. In the Pacific Northwest, LWD in riparian areas provides an important germination site for several conifer species. *(need reference.)*

Channel complexity promotes the retention of water and organic material. This retentiveness plays an important role in the fate of nutrients in the stream channel. In a study by Mulholland et al. (1985), it was suggested that leaf litter in streams promotes nutrient retention as the leaf pack acts as a substrate for nutrient-hungry microbes. Using solute injection techniques, Valett et al. (2002) found that phosphorus uptake in channels with high LWD volumes, frequent debris dams, and fine-grained sediments was significantly greater than in channels in younger forests without these characteristics. Corroborating this finding, Ensign and Doyle (2005) conducted phosphorus injections in streams both before and after the removal of LWD and coarse-particulate organic matter (CPOM) in the channels and found that phosphate uptake decreased by up to 88 percent after LWD removal. These studies show that channel complexity increases water retention and, through CPOM and LWD retention, provides a substrate for biofilm growth.

Numerous studies have reported fish use of complex cover composed of riparian vegetation, such as branches, root wads, and woody debris, which provides increased and diverse surface area and interstices for prey colonization and protection from predation. Particulate organic matter accumulated by LWD is an important food source for many stream-dwelling invertebrates. Addition of wood to channels causes increased abundance of macroinvertebrates and changes the species composition in that stream. Pools formed by LWD in streams are an important habitat for many species of stream fishes.

The use of submerged riparian vegetation during early development has been hypothesized to increase Columbia River white sturgeon recruitment (Coutant 2004). In another study, radio-tagged cutthroat trout were observed using pools associated with LWD for cover (Harvey et al. 1999). In addition, undercut banks are stabilized by riparian vegetation roots, which reduce the effects of erosion from streamflow. These undercut banks provide shade and lower temperatures, which are important for fish habitat and cover from predation (Angermeier and Karr 1984; Heggenes and Borgstrom 1988; Rowe et al. 2004). Therefore, significant reduction in riparian vegetation will affect species composition.

Riparian vegetation modifications can impose a number of stressors on fish and aquatic invertebrate species, with potentially detrimental effects on survival, growth, and fitness. For example, riparian modification can cause hydraulic and geomorphic changes in the stream channel due to reduced LWD recruitment and increased bank erosion.

Long-term indirect effects on fish and invertebrates associated with altered habitat complexity include a reduction or modification of prey base and related changes or reduction in foraging success, modified distribution and/or diversity of species within riverine systems, and decreased habitat suitability for various lifestages, resulting in decreased viability and reproductive success.

7.5.6.4 Ecosystem-Specific Effects: Lacustrine

In lakes, large woody debris provides cover and foraging opportunities for fish (Quinn 2005).

Altered habitat complexity in the form of a reduction of shoreline LWD and overhanging vegetation has been linked to removal due to human development (Francis and Schindler 2006). Loss of LWD has a number of effects on the health of the lake food web. Fish also respond to habitat characteristics resulting from the association of shoreline and riparian zone modification. In a study of Wisconsin lakes, the habitat characteristics most influenced by this association were depth, substrate size and embeddedness, and amount of woody vegetation and macrophytes (Jennings et al. 1999).

Tabor et al. (2004, 2006) have documented the importance of small woody debris as habitat structure in Lake Washington, where it may provide important periodic refuge from predators for juvenile Chinook salmon.

While the importance of riparian vegetation to the lacustrine nearshore area has been documented (Scheuerell and Schindler 2004), the role of emergent vegetation on the productivity of habitat has not been explored in detail in Washington waters or with regard to HCP species.

7.5.7 Activity-Specific Effects

7.5.7.1 Culverts

Culvert retrofit projects are likely to have little or no effect on riparian vegetation, as construction and maintenance work will be implemented from the existing roadway and will take place within the footprint of the existing structure. Removal and replacement projects may have more broad-reaching effects on riparian vegetation. In many cases, the work can be implemented from the existing road prism or channel modification, requiring little riparian disturbance. However, in some cases culvert removal or replacement may cause hydraulic and geomorphic modifications that affect riparian conditions in both upstream and downstream reaches more broadly. This is particularly likely to be true in situations where road-impounded wetlands are created as a result of backwater conditions. Draining of these impoundments may alter the relationship between the stream and riparian vegetation as the channel adjusts to the new gradient condition.

7.5.7.2 Water-Crossing Structures

Streambanks and shorelines associated with water crossing structures may be managed so as to impair riparian/shoreline vegetation function. For example, Corps regulations restrict the growth of woody vegetation on dikes, and roads located close to the shoreline may cross or truncate the width of the riparian zone. Such impacts may be attributable to water crossing structures if the dike or road would not have been built (or would not still be maintained) but for the water crossing.

7.5.7.3 Bank Protection

Bank protection structures such as bulkheads may cause a physical barrier between the bank and hyporheic flow and prevent exchange between the bank and aquatic ecosystem.

Marine shorelines that have been modified by human activities tend to have less LWD and driftwood than unarmored beaches (Higgins et al. 2005; Herrera 2005). MacDonald et al. (1994) also reported that shoreline armoring limited driftwood accumulation on a beach. Higgins et al. (2005) suggested that the mechanisms for the apparent reduction in LWD appear to be the removal of adjacent riparian vegetation during and following placement of the bank protection; reduced shoreline roughness at armored sites, which causes more LWD to be transported away; and limited upper intertidal and backshore areas that allow for LWD deposition above tidal elevations that are routinely inundated.

Because LWD is used in some marine soft-shore armoring instances to attenuate wave energy and lessen the potential for erosion, it is assumed that naturally occurring LWD on beaches would do the same, but this has not been empirically tested. Herrera (2005) describes how multiple layers of LWD along a shoreline could provide effective energy dissipation, decreasing the amount of wave reflection during high water levels, by increasing the roughness of the shoreline and by decreasing its slope relative to a vertical bulkhead.

7.5.7.4 Logging

Logging practices in northwestern Montana have been shown to decrease habitat complexity through the reduction of LWD inputs (Hauer et al. 1999).

7.5.7.5 Groins

Channel complexity promotes the retention of organic material. Groins typically simplify channel structure, causing a drop in complexity. Decreased nutrient retention will affect both local waterways and downstream receiving waters. Local waterways will be affected through the associated reduction in primary production, and receiving waters (which are primarily located in more nutrient-impacted lowland areas) will be affected through additional nutrient loading, which may lead to eutrophication.

7.5.7.6 Roughened channels

Construction of new roughened channels will have extensive effects on terrestrial vegetation. In contrast, alteration of an existing channel may require alteration of both riparian and aquatic vegetation for construction purposes. On this basis, the effects of riparian vegetation modification on HCP species as a result of roughened channel creation are expected to be relatively extensive.

7.5.7.7 Dredging

Impacts on habitat complexity from riparian vegetation loss and/or removal will be variable and will depend on the type and extent of dredging activities. Reductions in large woody debris loading to a riverine system from the loss of riparian functions (i.e., recruitment) may contribute to a loss of habitat complexity. Processes that homogenize habitat also diminish ecological health and habitat diversity (Jacobson 2006). The loss of large woody debris may result in the loss of pools and alterations of channel geometry.

Instream large woody debris may be disturbed or removed from the channel as part of dredging operations. In addition, the potential recruitment of large woody debris may be adversely impacted as a result of vegetation removal from channel banks and bars during construction access and staging.

7.5.7.8 Gravel mining

Gravel mining activities alter the distribution of riparian vegetation both directly through the removal or disturbance of riparian vegetation during mining activities and indirectly through the conversion of riparian areas to open pits. The loss of riparian vegetation caused by gravel mining can reduce the supply and retention of LWD.

7.5.7.9 Fish screens

Bankline in-channel screens and all off-channel fish screens pose some potential for riparian vegetation modification due to the fact that these types of screens may require a bypass system. Bypass systems typically take the form of a pipe or a constructed channel to return water to the stream system. The extent of riparian modification associated with bypass systems varies widely depending on the type of system and its extent. Some screen systems may employ a piped bypass that returns flow to the stream almost directly downstream of the diversion, requiring little additional riparian modification. Even longer bypass pipe systems installed by hand labor may have only minor effects on riparian vegetation. In contrast, due to the size of the diversion and local topography, some screen systems may employ constructed bypass channels of considerable length. In such cases, development of these channels may require extensive riparian modification. Assessing the extent of likely effects requires consideration of the scale and design of the screen system in question.

7.5.7.10 Tide Gates

Flow alterations from tide gates may redistribute LWD such that it concentrates in certain areas and is absent in others (Miller et al. 2001).

7.5.7.11 Beaver Dam Removal

Riparian and bank vegetation has been shown to act as a filter for polluted waters from upgradient sources moving through overland flow, interflow, and shallow groundwater pathways (Hickey and Doran 2004). The removal of beaver dams may be associated with reduced buffering capacity in downstream riparian zones. Reduced buffering capacity may lead to increased nutrient and pollutant loading to the channel.

The creation of a beaver dam impacts the riparian zone upstream of the dam. Inundation and clearing of vegetation by beaver results in the conversion of the terrestrial riparian zone into an aquatic system with minimal cover. In this way the riparian buffer upstream of the dam can be reduced by beaver activity. Beaver ponds create riparian wetlands. It has been shown that impoundments can increase riparian vegetation downstream of the dam by augmenting floodplain groundwater (Duke et al. 2007). Because beaver-created wetlands have rich organic soils, there is the potential for changes in nutrient levels to occur in the littoral areas of the pond (Cooper 1990).

Beaver dam removal and impoundment dewatering results in the creation of a stream reach which initially has little riparian vegetation. Until riparian vegetation becomes established after dewatering, there will be reduced vegetation adjacent to the channel and thus decreased direct delivery of organic material to the channel.

Nutrient-rich sediments from the benthos of the former impoundment (Ahearn and Dahlgren 2005) may be rapidly colonized with wetland vegetation and invasive species. The establishment of mature riparian stands may take years to decades (Auble et al. 2007), but dense stands of rapidly growing native and invasive plants such as *Typha* spp. (cattails) may provide channel shading in the short term. The removal of beaver dams will weaken the terrestrial-aquatic linkage and initially reduce allochthonous input from the riparian zone. Additionally, bank stability will be degraded until the channel adjusts and vegetation begins to grow and stabilize the banks. Riparian buffering capacity may increase as the former impoundment will serve as additional buffer area between the channel and upland pollution sources. It can be assumed that the removal of beaver dams which impound a large cross section of the floodplain will impact downstream riparian vegetation by altering hyporheic flow and thus will also reduce downstream allochthonous inputs.

Beaver actively recruit wood from the adjacent floodplain to augment their dams and build lodges (Collen and Gibson 2001). Beaver extirpation would eliminate this pathway for wood recruitment. The wood imported to the channel by beaver activity can serve as shelter for fishes (Cederholm et al. 1997), substrate for algal growth (Atilla et al. 2003), and habitat for macroinvertebrates (Clifford et al. 1993; Rolauffs et al. 2001).

Rolauffs et al. (2001) found that macroinvertebrate emergence was 3.2 and 5.5 times higher from a beaver dam than from the adjacent brook and pond areas, respectively. This indicates that not only are beaver ponds important for fluvial ecosystem functioning, but that the associated dam structures provide additional habitat benefits. Removal of beaver dams will, of course, eliminate those habitats created by both the pond and the dam.

While some beaver ponds have been shown to raise stream temperatures (Margolis, Raesly et al. 2001; Mcrae and Edwards 1994), a literature review by Pollock et al. (2003) noted that other studies have indicated that beaver ponds do not significantly affect temperature.

7.5.7.12 In-Channel/Off-Channel Habitat Creation/Modifications

The creation of off-channel habitat and the placement of in-channel structure that promote river-floodplain connectivity will likely increase the export of coarse particulate organic matter (CPOM) from the floodplain to the channel (Junk et al. 1989; Tockner et al. 1999). The literature indicates that any activity which promotes the reconnection of once-severed floodplain-channel pathways can be expected to have a net benefit for aquatic organisms (Crain et al. 2004; Schemel et al. 2004; Tockner et al. 1999). This is partially due to the fact that floodplains and riparian areas are vital for the transfer of energy (in the form of carbon from leaf litter and other organic material) from terrestrial to aquatic ecosystems, especially in lowland river systems (Junk et al. 1989; Thoms 2003). However, it should be noted that if instream structures fail and act to catalyze channel incision, the result could be decreased connectivity with the riparian zone and decreased system productivity.

7.5.7.13 Riparian Planting/Restoration/Enhancement

One of the most common restoration practices is the restoration and planting of riparian environments. In some instances, invasive plants are removed from the riparian area and replaced with native vegetation. Increased shading caused by riparian canopy cover has both positive (LeBlanc and Brown 2000) and negative (Hetrick et al. 1998) ramifications for aquatic biota.

Invasive plant removal may initially be associated with reduced allochthonous inputs, but as the planted riparian community matures it is anticipated that allochthonous inputs would increase above pre-project levels (Bennett 2007). Certain macroinvertebrate functional feeding groups rely on organic input from riparian vegetation as a food source and/or food substrate (Parkyn et al. 2005). Increased allochthonous input would then conceivably increase macroinvertebrate populations and increase the foundation of the food web for fishes. Indeed, in a study of 36 sites with riparian buffers and 12 reference sites, Teels et al. (2006) observed increased macroinvertebrate density and diversity response at sites with highly disturbed local conditions prior to buffer establishment.

Increased shading from riparian plantings can decrease stream temperature by between 3.6 and 7.2^{o}F (2 and 4^{o}C) (Ebersole et al. 2003), which will benefit cold water species (Opperman and Merenlender 2004); conversely shading will decrease autotrophic production within the channel and possibly impact macroinvertebrate populations.

There is a growing body of literature indicating that riparian vegetation <u>removal</u> increases stream productivity. Canopy removal studies have indicated that macroinvertebrate populations increase after riparian vegetation is removed (Hetrick et al. 1998; Wipfli 1997). In headwater systems (which are generally oligotrophic), this increase in productivity is seen as a benefit to stream biota. Hetrick et al. (1998) studied riparian canopy removal in Eleven Creek, Alaska, and concluded that:

> Based on higher abundance of aerial invertebrates above the water surface and increased standing crop of benthic invertebrates that we observed in open- versus closed-canopy sections of Eleven Creek, it appears that canopy removal has the potential to increase the carrying capacity of juvenile coho salmon in streams where populations are food limited**.**

Findings such as these indicate that riparian plantings in headwater food limited reaches may be a detriment to juvenile fishes within the channel. In a similar study, Fuchs et al. (2003) found that streams flowing through newly logged forests in British Columbia (where the riparian zones were harvested within 5 years of the study) had nearly twice the macroinvertebrate biomass as those in unlogged or older logged sites, and a higher chlorophyll a concentration. In a study of open canopy and closed canopy patches by Zalewski et al. (1998), it was found that macroinvertebrate density was highest in mixed cover reaches which received both incident light and allochthonous input. Fish biomass followed the same trend, being lowest in heavily shaded areas and in open channels without riparian vegetation, but highest in ecotones of intermediate complexity.

Riparian buffer studies indicate that there are complex interactions between the channel and the riparian zone and that the ideal riparian environment may be a mix of closed canopy and open canopy patches (Zalewski et al. 1998). Riparian patch dynamics are vital to a healthy stream ecosystem. Riparian canopy provides shade and litter input, while open canopy patches are characterized by increased solar radiation, warm waters, and autochthonous production. A goal of riparian planting should be to increase the frequency of these habitat patches within degraded reaches with little canopy cover, while acknowledging that the increase in shading associated with riparian vegetation may locally decrease productivity within that patch.

There has been little research regarding the impact of riparian shading in marine systems. However, one study which entailed sampling at sites at northern Bellingham Bay, Dugualla Bay, northern Camano Island, and the west shore of Port Susan found that surf smelt eggs deposited in unshaded beaches had nearly twice the mortality of eggs deposited in shaded beaches. This study indicates that thermal cooling from riparian vegetation is important in marine as well as freshwater environments (Rice 2006). It has also been shown that juvenile salmonids prefer estuarine habitat with overhanging vegetation (Quinones and Mulligan 2005). The cover and shading provided by the vegetation apparently creates conditions favorable for salmonids. These studies indicate that riparian vegetation is important for fish species in estuarine as well as freshwater habitat.

7.5.7.14 Wetland Creation/Restoration/Enhancement

Wetland creation, restoration, or enhancement can have a wide variety of impacts on riparian vegetation and the ecologic functioning of the riparian zone. Increases of the wetted perimeter of the channel during flooding can enhance the recruitment of terrestrial organic material. The creation, restoration, and enhancement of wetlands in both coastal and riparian environments can produce a biogeochemical buffer between terrestrial pollution sources and aquatic environments.

Estuarine wetland enhancement will likely increase the productivity and connectivity between shallow water and open water habitat. The transfer of energy (in the form of carbon) between these habitats is complex. Initially it was assumed that shallow water habitat exported detrital carbon (from riparian vegetation) to open water systems thus forming the base of the detrital food web. However, later research indicated that this transfer was highly variable from one estuary to the next (Stevens et al. 2006). Recent research has indicated that a pathway for energy transfer between these systems may be found through trophic pathways. Piscivorous fishes which reside in open water habitat frequently feed on small fishes which reside in shallow water habitat. The result is a transfer of biomass which can amount to 2 percent of shallow habitat primary productivity (Stevens et al. 2006). Restored estuarine wetlands would, in theory, increase productivity and thus augment this trophic transfer of energy.

7.5.7.15 Beach Nourishment

As beach nourishment is primarily an activity that is staged offshore, there are limited riparian impacts. If an offshore sediment source is used, there may be no riparian impacts. In certain circumstances, beach nourishment can actually rebuild a riparian corridor lost to shoreline armoring (Nordstrom 2005). However, care should be used when significantly changing the shape of the shoreline. The shoreline in most locales in western Washington is shaped by extreme events (Finlayson 2006), and much of the fill material may be eroded extremely quickly during such events (Seymour et al. 2005). Provided construction impacts are small, beach nourishment would most likely result in a potential gain in riparian vegetation.

7.6 Water Quality Modifications

Water quality is an important habitat feature in marine, estuarine, riverine, and lacustrine environments. The possible mechanisms of impact to water quality from HPA-permitted structures include:

1. Alterations to temperature
2. Alterations to dissolved oxygen
3. Alterations to pH
4. Alterations to salinity
5. Alterations to suspended sediment concentrations and turbidity
6. Alterations to nutrient and pollutant loading

Each of the water quality parameters discussed can significantly affect the distribution, health, and survival of potentially covered species. Salmon, trout and other cold-water fish, and many aquatic invertebrates require cool, clean, and well-oxygenated water. HPA-permitted activities that impair these conditions may produce behaviors (e.g., avoidance of otherwise preferred location or increased feeding to meet increased metabolic demand) or physiological responses that reduce the organism's ability to survive and grow. The magnitude of the potential impacts will depend upon:

1. The size, duration, and frequency of the impact
2. The vulnerability of the affected life-history stage
3. The inability of the organism to avoid the impact
4. The physiological, developmental, and behavioral impairments suffered by the organism
5. Indirect mechanisms such as exposure to predation

7.6.1 Alterations to Temperature

Temperature is a primary metric of aquatic ecosystem health, as aquatic organisms have adapted to live within specific thermal regimes. Alterations to these thermal regimes occur at the detriment of local organisms.

7.6.1.1 General Effects: All Environments

Thermal stress can occur through multiple direct and indirect pathways in fish and invertebrates. These include direct mortality, altered migration and distribution, increased susceptibility to disease and toxicity, and altered development, spawning, and swimming speeds (Sullivan et al. 2000). Motile organisms have the ability to avoid or evacuate those areas of extreme temperature, but even then the stress induced from periodic exposure and resulting habitat avoidance can affect organism health and contribute to mortality (Groberg et al. 1978). Each of the HCP species is ectothermic (cold-blooded); consequently, temperature is a resource that organisms use for energetic means. With organism metabolism dependent on water temperature, thermal regime may be the single-most important habitat feature controlling aquatic organisms. Generally speaking, however, temperature alterations and resultant habitat changes are less pronounced in marine ecosystems than they are in freshwater ecosystems.

Much of the research identified pertaining to temperature effects on fish addresses salmonids in rivers. Considerably less research exists defining thermal criteria for invertebrates, and that has been done mostly on marine species.

A substantial amount of information is available regarding tolerances of HCP species (particularly salmonids) to thermal stress. These effects have been documented in many studies, which have been well summarized in meta-analyses. In the case of salmonids, summary analyses have been used as the basis for developing thermal tolerance ranges and threshold criteria for regulatory purposes. Poole et al. (2001) provides a useful example.

Reducing riparian shade allows an increase of exposure to solar radiation that may lead to an increase of water temperature (Fischenich 2003). Correlated with increased water temperature are reduced levels of dissolved oxygen and potential for stressors on aquatic organisms, especially juvenile salmon (Ecology 2000).

7.6.1.2 Ecosystem-Specific Effects: Marine and Estuarine

The small amount of research defining thermal criteria for invertebrates is for marine species. In general, an altered temperature regime will result in blocked migrations, increasing the chances of infection, deformities in developing eggs, stress, and mortality of several HCP species.

Gagnaire et al. (2006) noted that elevated temperatures caused blood cell mortality in Pacific oysters but not until temperatures exceeded 104^{o}F (40^{o}C), which is unlikely in Washington, even in altered settings. In studies on northern abalone, optimal growth rates were found between 44.6 and 62.6^{o}F (7 and 17^{o}C) (Hoshikawa et al. 1998), with significant mortality at 32.9^{o}F (0.5^{o}C) and 79.7^{o}F (26.5^{o}C) (Paul and Paul 1998).

7.6.1.3 Ecosystem-Specific Effects: Riverine

Water temperature is strongly dependent on mixing in rivers and streams (Fischer et al. 1979). Stratification within rivers can reduce both habitat complexity and connectivity; stratified waters can lead to elevated surface temperatures, particularly during the summer months (Fischer et al. 1979).

Temperatures have been shown to regulate nutrient cycling processes in streams (Sheibley, Duff, et al. 2003; Sheibley, Jackman, et al. 2003). In these studies, the authors showed through modeling, field monitoring, and laboratory experiments that coupled nitrification-denitrificaiton reactions were controlled by stream temperature (Sheibley, Duff, et al. 2003; Sheibley, Jackman, et al. 2003). In winter, nitrification-denitirfication reactions were suppressed and more nitrogen from groundwater discharge entered the stream channel. In summer, nitrification-denitrification reactions were more efficient, and very little nitrogen from groundwater discharge was observed in the surface water. Therefore, temperature alterations may also affect the nutrient concentration in rivers.

An altered temperature regime can shift species composition from cool water to warm water species (Bednarek 2001).

The majority of research on temperature impacts on aquatic species has focused on salmonids, and much of that has emphasized temperature effects on salmonid life stages in rivers.

Different species of salmonids have evolved to use different thermal regimes. For instance, it has been found that coho egg, alevin, and fry development is most rapid at 39°F (4°C), while alevin and fry of pink and chum salmon develop fastest at 46°F (8°C) (Beacham and Murray 1990). Despite these differences, the majority of salmonids prefer the same temperature ranges during most life-history stages. The primary exception to this is that char (bull trout and Dolly Varden) require lower temperatures for optimal incubation, growth, and spawning (Richter and Kolmes 2005). An optimal temperature matrix is presented in Table 7-6. Each group of species has a different range of optimal temperatures at each life-history stage. These same temperature ranges have been adopted by Ecology and incorporated into the state water quality standards (WAC 173-201A, Finalyson 2006). Table 7-7 presents highest 7-day average maximum thresholds as promulgated in the state standards.

Table 7-6 indicates that there are water quality thresholds for different life-history stages which are considerably lower than the lethal limit. Elevated water temperatures can impair adult migration and spawning. Thermal barriers to migration can isolate extensive areas of potentially suitable spawning habitat and contribute to prespawning mortality.

Table 7-6. Estimates of thermal conditions known to support various life-history stages and biological functions of bull trout (a species extremely intolerant of warm water) and anadromous (ocean-reared) salmon.

Life-History Stage or Biological Function	Anadromous Salmon	Bull Trout
Temperature of common summer habitat use	10–17°C (50–63°F)	6–12°C (43–54°F)
Lethal temperatures (1-week exposure)	Adults: >21–22°C (70–72°F)	—
	Juveniles: >23–24°C (73–75°F)	Juveniles: 22–23°C (72–73°F)
Adult migration	Blocked: >21–22°C (70–72°F)	Cued:10–13°C 50–55°F)
Swimming speed	Reduced: >20°C (68°F)	—
	Optimal: 15–19°C (59–66°F)	—
Gamete viability during holding	Reduced: >13–16°C (55–61°F)	—
Disease rates	Severe: >18–20°C (64–68°F)	—
	Elevated: 14–17°C (57–63°F)	—
	Minimized: <12–13°C (54–55°F)	—
Spawning	Initiated: 7–14°C (45–57°F)	Initiated:<9°C (48°F)
Egg incubation	Optimal: 6–10°C (43–50°F)	Optimal: 2-6°C (36–43°F)
Optimal growth	Unlimited food: 13–19°C (55–66°F)	Unlimited food: 12–16°C (54–61°F)
	Limited food: 10–16°C (50–61°F)	Limited food: 8–12°C (46–54°F)
Smoltification	Suppressed: >11–15°C (52–59°F)	—

Source: Poole et al. 2001

Note: These numbers do not represent rigid thresholds, but rather represent temperatures above which adverse effects are more likely to occur. In the interest of simplicity, important differences between various species of anadromous salmon are not reflected in this table, and requirements for other salmonids are not listed. Likewise, important differences in how temperatures are expressed are not included (e.g., instantaneous maximums, daily averages, etc.).

Table 7-7. Aquatic life temperature criteria in fresh water.

Category	Highest 7-DADMax
Char spawning	9°C (48.2°F)
Char spawning and rearing	12°C (53.6°F)
Salmon and trout spawning habitat	13°C (55.4°F)
Core summer salmonid habitat	16°C (60.8°F)
Salmonid spawning, rearing, and migration	17.5°C (63.5°F)
Salmonid rearing and migration **Only**	17.5°C (63.5°F)
Nonanadromous interior redband trout	18°C (64.4°F)
Indigenous warm water species	20°C (68°F)

Source: (WAC 173-201A 2006) Table 200(1)(c)

Note: Water temperature is measured by the 7-day average of the daily maximum temperatures (7-DADMax). Table 200(1)(c) lists the temperature criteria for each of the aquatic life use categories.

Fish are susceptible to a number of sublethal effects related to temperature. For instance, elevated but sublethal temperatures during smolting may result in desmoltification, altered emigration timing, and emigration barriers.

- Adult migration blockages occur consistently when temperatures exceed 70–72°F (21–22°C) (Poole and Berman 2001a).

- Temperatures that impair smolting are above a range of between 52 and 59°F (11 and 15°C) (Poole and Berman 2001a; Wedemeyer et al. 1980).

- Temperatures in this range have been shown to reduce the activity of gill ATPase (McCullough et al. 2001), an enzyme that prepares juvenile fish for osmoregulation in saline waters (Beeman et al. 1994).

- Temperature-induced decreased gill ATPase has been correlated with loss of migratory behavior in numerous salmonid species (Ban 2006; Marine and Cech 2004; McCormick et al. 1999) and constitutes a significant impairment to juvenile survival.

- If salmon are exposed to temperatures above 57°F (14°C) during spawning, gametes can be severely affected, resulting in reduced fertilization rates and embryo survival (Flett et al. 1996).

- Ideal temperatures for salmonid spawning are in the range of 44–57°F (7–14°C) (Brannon et al. 2004; McCullough et al. 2001).

The interface between flow within the hyporheic zone and the stream channel is an important buffer for stream temperature (Poole and Berman 2001a); therefore, the alteration of groundwater flow or hyporheic exchange can affect stream temperature. The magnitude of the influence depends on many factors, such as stream channel pattern and depth of the aquifer (Poole and Berman 2001a). Stream temperature has been shown to be an important factor in determining the suitability of habitats for aquatic species. Activities that adversely affect groundwater upwelling may limit the availability and suitability of spawning and thermal refuge habitats.

For example,

- in Montana, the distribution and abundance of bull trout is influenced by hyporheic and groundwater–surface water exchange (Baxter and Hauer 2000).

- Female bull trout tend to choose areas of groundwater discharge (i.e., cooler temperatures) for locating their spawning, and upwelling sites serve as important thermal refugia for all life-history stages (Baxter and McPhail 1999).

- The preferential selection of spawning substrates in groundwater upwelling zones is a common behavior among all HCP salmonid species (Baxter and Hauer 2000; Berman and Quinn 1991; Bjornn and Reiser 1991; Ebersole et al. 2003; Geist

2000; Geist and Dauble 1998; Geist et al. 2002; Greig et al. 2007; Zimmermann and Lapointe 2005).

Temperatures outside the optimal growth range can lead to decreased growth and competitive ability with sympatric species having broader temperature tolerances. Temperatures can also cause behaviors that limit population productivity.

For example,

- Selong et al. (2001) found that bull trout tend to avoid otherwise suitable habitats when water temperatures exceeded optimal growth ranges, and postulated that other species of salmonids would demonstrate similar behavior.

- McMahon et al. (2007) found that temperature-mediated competition in lower elevation reaches was one of several factors that gave introduced brook trout a competitive advantage over native bull trout, which tended to retreat to higher elevation refugia.

Elevated temperature regimes also affect salmonid species by altering behavior and reducing resistance to disease and toxic substances. Studies have indicated that under chronic thermal exposure conditions, aquatic organisms' susceptibility to toxic substances may increase. In nearshore areas where temperature (as well as pollutant levels) may be elevated, the combined effect of thermal and water pollution may be a primary driver of salmonid decline.

- Because elevated temperatures increase metabolic processes, gill ventilation also rises proportionately (Heath and Hughes 1973).

- Black et al. (1991) showed that an increase in water flow over the gills, which results from increased gill ventilation at increased temperature, resulted in rapid uptake of toxicants, including metals and organic chemicals, via the gills.

- Salmonids also become more susceptible to infectious disease at elevated temperatures (57–68^{o}F [14–20^{o}C]) because immune systems are compromised (Harrahy et al. 2001), while bacterial and viral activity is accelerated (Tops et al. 2006).

Additional studies, mainly in the laboratory, have developed limits for other HCP fish species.

- Rainbow trout mortality occurred at temperatures of 67.8 to 73.0^{o}F (19.9 to 22.8^{o}C) (Wagner et al. 1997).

- In developing white sturgeon, temperatures above 71.6^{o}F (22^{o}C) can cause deformities. White sturgen develop best between 59 and 66^{o}F (15 and 19^{o}C) (Mayfield and Cech 2004). Furthermore, elevated temperatures can make white sturgeon more susceptible to infection from viruses (Watson et al. 1998).

- In developing green sturgeon embryos, temperatures between 73 and 79^{o}F (23 and 26^{o}C) can cause complete mortality, with upper limits for survival around 62.6–64.4^{o}F (17–18^{o}C) (Van Eenennaam et al. 2005).

- Dolly Varden show decreased appetite above 60.8^{o}F (16^{o}C) and lethal temperatures are observed above 68^{o}F (20^{o}C) (Takami et al. 1997).

- Early lifestages of Pacific lamprey and western brook lamprey showed that growth and development in these species effectively ceased at experimental temperatures of 40.7^{o}F (4.85^{o}C) and 40.9^{o}F (4.97^{o}C), respectively, with survival greatest for both at 64^{o}F (18^{o}C) and lowest at 71.6^{o}F (22^{o}C). Ammocoete exposure to a treatment temperature of 71.6^{o}F (22^{o}C) resulted in a high rate of developmental abnormalities (Meeuwig et al. 2005).

Considerably less research exists defining thermal criteria for freshwater mollusk species, which are the most likely HCP invertebrate species to be affected by temperature-related stressors. However, Vaughn and Taylor (1999) reviewed several studies of freshwater mussel populations in river systems affected by dams and found multiple instances of decreased population persistence and abundance in reaches downstream of dams. They postulated that because abundance increased as temperature conditions moderated in downstream reaches, sensitivity to altered temperature regime was a primary factor controlling distribution. These findings imply that alteration of water temperature regime, in combination with other stressors, may adversely affect HCP invertebrate species.

7.6.1.4 Ecosystem-Specific Effects: Lacustrine

No studies were identified that specifically looked at temperature effects on HCP species in lakes, but some studies of fish in reservoirs include:

- Bull trout in reservoirs behave like adfluvial fish and prefer littoral areas (Block 1955; Chisholm et al. 1989.

- Bull trout have very low upper thermal limits, and the formation of thermal barriers may prevent their movement (Selong et al. 2001).

- Rinne et al. (1981) observed that “fishes introduced into western reservoirs are intrinsically shallow-water, littoral inhabitants,” and any structure in this zone may introduce a physical or thermal stressor.

- Reservoirs are subject to density or turbidity currents resulting from differences in temperature or sediment concentrations between inflows and reservoir waters (Snyder et al. 2006).

- Mixing zones and shallow water littoral habitat were preferred by the razorback sucker in Lake Powell, Utah (Karp and Mueller 2002); this study found that these fish primarily use shallow, vegetated habitats in side canyons, but these areas represent less than 1 percent of the available habitat in Lake Powell. However, temperature gradients in reservoirs can fragment these habitats.

- Mueller et al. (2000) found that temperature gradients caused the razorback sucker to abandon preferred inshore habitat in Lake Mohave, Arizona.

7.6.2 Alterations to Dissolved Oxygen

7.6.2.1 General Effects: All Environments

Dissolved oxygen (DO) content is critical to the growth and survival of the 52 HCP species. The amount of oxygen dissolved in water is dependent on temperature, physical mixing, respiration, photosynthesis, and, to a lesser degree, atmospheric pressure. These parameters can vary diurnally and seasonally and depend on processes such as daytime photosynthesis that inputs dissolved oxygen and night-time plant respiration that deplete dissolved oxygen levels. Dissolved oxygen concentration is temperature dependent; as temperatures rise, the gas-absorbing capacity of water decreases and dissolved oxygen saturation level decreases.

Reduced dissolved oxygen levels can be due to several factors, including:

- increased temperature (Snoeyink and Jenkins 1980),
- organic or nutrient loading (Ahearn et al. 2006),
- increased benthic sedimentation (Welch et al. 1998),
- chemical weathering of iron and other minerals (Schlesinger 1997),
- increased loading of carbon and the associated increase in biochemical oxygen demand (BOD). Increased BOD is frequently associated with eutrophication brought about by increased nutrient loading and solar radiation. Nutrient cycling has been closely linked to nearshore stratification and circulation (Roegner et al. 2002).
- oxidation reactions occuring as a result of resuspension of large quantities of anoxic sediments (Nightingale and Simenstad 2001a).

Juvenile salmon are highly sensitive to reductions in dissolved oxygen concentrations (USFWS 1986) and so are probably among the more vulnerable potentially covered species with regard to dissolved oxygen impairments. Salmon generally require dissolved oxygen levels of greater than 6 ppm for optimal survival and growth, with lethal one-day minimum concentrations of around 3.9 ppm (Ecology 2002).

Different organisms at different life-history stages require different levels of dissolved oxygen to thrive. Tolerance for low oxygen levels varies across other species as well. For example, pygmy whitefish can withstand dissolved oxygen conditions below 5 ppm (Zemlak and McPhail 2006).

The effects of low oxygen concentrations on invertebrates have been documented in both fresh water and marine environments. Little consensus exists concerning low dissolved oxygen criteria for macroinvertebrates, and tolerances to hypoxic conditions are taxonomically specific. Many invertebrates are adapted to live in benthic low-energy environments where dissolved oxygen concentrations are naturally low; consequently, these organisms can withstand hypoxic conditions.

- Hirudinea (leeches), Decapoda (crustaceans), and many aquatic insects tolerate dissolved oxygen levels below 1.0 ppm (Hart and Fuller 1974; Nebeker et al. 1992).

- Chen et al. (2001) found that freshwater mussels (Unionidae) showed a wide range of tolerance for low DO levels depending on the types of habitats they inhabit. As would be expected, they found that species that inhabit slack water and warm water environments show greater tolerance for low DO levels, while species that are found in flowing water and cold water environments were far more sensitive.

- Other aquatic invertebrate species (e.g., Ephemeroptera, Plecoptera, Trichoptera) also show variable sensitivity depending on the environments to which they are adapted.

- In general, organisms adapted to colder flowing water environments where DO levels are naturally high are expected to have lower tolerance for DO depletion (Nebeker 1972).

Increased vertical exchange between surface and subsurface waters will benefit aquatic biota by increasing benthic dissolved oxygen levels and promoting solute uptake, filtration, and transformation. Studies have shown that the availability of dissolved oxygen to incubating salmonid embryos is dependent upon hyporheic exchange (Geist 2000; Greig et al. 2007) and that the occlusion of this exchange through siltation can lead to hypoxia within redds and decreased embryo survival (Heywood and Walling 2007). The hyporheic zone does more than promote oxygen exchange in subsurface sediments, it can also act as an effective filter and zone of biogeochemical transformations. Increased hyporheic exchange has been associated with nutrient uptake (Anbutsu et al. 2006; Sheibley et al. 2003) and transformation (Fernald et al. 2006; Lefebvre et al. 2005), and may attenuate the transport of dissolved and particulate metals (Gandy et al. 2007). Elevated metals and nutrients can both have negative ramifications for fish and invertebrate health.

7.6.2.2 Ecosystem-Specific Effects: Marine and Estuarine

A literature review by Gray et al. (2002) found that in marine environments, invertebrates were not affected by low dissolved oxygen until concentrations fell below 1–2 ppm. Benthic dissolved oxygen levels can seasonally drop below this threshold in productive systems that receive high biochemical oxygen demand (BOD) loadings. For instance, depressed benthic dissolved oxygen levels in Hood Canal, Washington, have been associated with spot shrimp decline (Peterson and Amiotte 2006). This dissolved oxygen decline in turn has been linked to BOD loadings from leaking, improperly sited or improperly functioning onsite wastewater systems. These conditions in Puget Sound highlight the importance of reducing anthropogenically generated BOD.

7.6.2.3 Ecosystem-Specific Effects: Riverine and Lacustrine

Table 7-8 lists the minimum recommended dissolved oxygen concentrations for salmonids and stream-dwelling macroinvertebrates (Ecology 2002). The dissolved oxygen thresholds presented in this table were derived from more than 100 studies representing over 40 years of research.

Table 7-8. Summary of recommended dissolved oxygen levels for full protection (approximately less than 1 percent lethality, 5 percent reduction in growth, and 7 percent reduction in swim speed) of salmonid species and associated macroinvertebrates.

Life-history Stage or Activity	Oxygen Concentration (ppm)	Intended Application Conditions
Incubation through emergence	>9.0–11.5 *(30 to 90-DADMin)* and No measurable change when waters are above 52°F (11°C) (weekly average) during incubation.	Applies throughout the period from spawning through emergence Assumes 1-3 ppm will be lost between the water column and the incubating eggs
Growth of juvenile fish	>8.0–8.5 *(30-DADMin)* >5.0-6.0 *(1-DMin)*	In areas and at times where incubation is not occurring
Swimming performance	>8.0-9.0 *(1-DMin)*	Year-round in all salmonid waters
Avoidance	>5.0-6.0 *(1-DMin)*	Year-round in all salmonid waters
Acute lethality	>3.9 *(1-DMin)* >4.6 *(7 to 30-DADMin)*	Year-round in all salmonid waters
Macroinvertebrates *(stream insects)*	>8.5-9.0 *(1-DMin or 1-DAve)*	Mountainous headwater streams
	>7.5-8.0 *(1-DMin or 1-DAve)*	Mid-elevation spawning streams
	>5.5-6.0 *(1-DMin or 1-DAve)*	Low-elevation streams, lakes, and nonsalmonid waters
Synergistic effect protection	>8.5 *(1-DAve)*	Year-round in all salmonid waters to minimize synergistic effect with toxic substances

Source: Ecology 2002.

Notes:
1-DMin = annual lowest single daily minimum oxygen concentration.
1-DAve = annual lowest single daily average concentration.
7-, 30-, or 90-DADMin = lowest 7-, 30-, or 90-day average of daily minimum concentrations during incubation period, respectively.

It should be noted that recommendations are presented in Table 7-8 for dissolved oxygen thresholds in categories other than lethality. Fish are motile organisms and, where possible, will avoid dissolved oxygen levels that would cause direct mortality. However, this avoidance behavior in and of itself can affect fishes. Stanley and Wilson (2004) found that fish aggregate above the seasonal hypoxic benthic foraging habitat in the Gulf of Mexico, while Eby et al. (2005) found that fish in the Neuse River estuary (North Carolina) were restricted by hypoxic zones to shallow, oxygenated areas, where in the early part of the summer about one-third fewer prey resources were available. Studies such as these reveal how dissolved oxygen can change fish distributions relative to habitat and potentially exclude fishes from reaching foraging and rearing areas. Sublethal dissolved oxygen levels can also cause increased susceptibility to infection (Welker et al. 2007) and reduced swim speeds (Ecology 2002), both of which may cause indirect impacts on HCP fish species.

In freshwater environments, eggs can become oxygen-starved through the deposition of suspended fines on spawning gravels. Fine sediment fills interstitial spaces in stream bed gravels and lowers oxygen exchange through the hyporheos. This process could potentially impact many of the HCP species, such as Pacific salmon, sturgeon, pygmy whitefish, and dace species (Bash et al. 2001; Chapman 1988; Hallock and Mongillo 1998; Nightingale and Simenstad 2001a; Pitt et al. 1995; Quinn and Peterson 1994; Sigler 1988; Welch and Lindell 1992; Wildish and Power 1985; Williamson 1985; Wydoski and Whitney 2003).

Kaller and Kelso (2007) found benthic macroinvertebrate density, including mollusks, greatest in low dissolved oxygen areas of a Louisiana wetland.

7.6.3 Alterations to pH

The pH of fresh and salt water normally ranges from 6.5–8.5 (Schlesinger 1997).

Little information was identified regarding pH requirements of the potentially covered species. In general, fish species tend to have very narrow ranges of pH preference, and levels outside of this range will impact their health. Alterations in pH can also affect invertebrates, although no studies of pH tolerance among the potentially covered invertebrate species were identified.

7.6.3.1 General Effects: All Environments

Structures constructed in aquatic settings can adversely impact the pH of surrounding water via contact between water and uncured concrete (Ecology 1999). Best management practices found in Ecology's "Stormwater Management Manual for Western Washington" (Ecology 2005) require that concrete be cured before coming into contact with the adjacent water body.

Altered pH resulting from concrete would likely be most significant for construction of large projects in areas with poor water circulation.

7.6.3.2 Ecosystem-Specific Effects: Marine and Estuarine

In marine environments, the buffering capacity of seawater is such that the impacts on pH from concrete placement and other leachable building materials are expected to be small (Webster and Loehr 1996.)

Department of Ecology water quality regulations (WAC 173-201A-210 (1) (f)) sets pH criteria between 7.0 and 8.5 in marine waters.

7.6.3.3 Ecosystem-Specific Effects: Riverine and Lacustrine

In Washington, the surface water quality standards require pH to be between 6.5 and 8.5 in fresh water (WAC 173-201A-200 (1) (g)).

When uncured concrete comes in contact with fresh water, some or all of it dissolves and increases the pH (high alkalinity) (DFO 2006). For example, when Portland cement, an active ingredient in concrete, contacts water it dissolves and produces a pH of up to 12 at 77 degrees F (25 degrees Celsius), which is far outside the livable range for all of the HCP species (Ecology 1999, DFO 2006). The potential for impacts from elevated pH is greatest during construction when concrete wash-off and slurries come into contact with water (Dooley et al. 1999). Once construction is complete, concrete may still affect the surrounding environment. Curing concrete surfaces can exhibit pH values as high as 13 during the 3 to 6 months it takes for concrete to cure underwater (Dooley et al. 1999). This elevated pH prevents attached macroalgae growth during this period.

The effects of high pH on fish may include death; damage to outer surfaces such as gills, eyes, and skin; and an inability to dispose of metabolic wastes (DFO 2007). In a rainbow trout toxicity study, a pH above 8.4 caused an increase in glucose and cortisol levels, and a pH above 9.3 caused mortality (Wagner et al. 1997).

Elevated pH has been shown to increase ammonia toxicity in fish because the organisms have difficulty excreting ammonia waste through their gills when ambient conditions are characterized by elevated ammonia and pH. It has been shown that at ambient ammonia concentrations of 5.0 ppm, mortality of tambaqui (*Colosoma macropomum*; also known as pacu), increased from 0 to 15 to 100 percent at a pH of 7, 8, and 9, respectively (de Croux et al. 2004). Consequently, if ammonia concentrations are elevated, the toxicity may be compounded by elevated pH.

Studies have shown that low pH can also affect fish.

- In white sturgeon, decreased sperm motility was observed when fish were exposed to pH levels below 7.5 (Ingermann et al. 2002).

- An investigation of landlocked sockeye salmon in Japan, brown trout (*Salmo trutta*), and Japanese char (*Salvelinus leucomaenis*) found that spawning activities and upstream migration were significantly inhibited in weakly acidic water of pH 5.8 to 6.4 (Ikuta et al. 2003). The authors further noted that landlocked sockeye salmon were the most sensitive of the three species.

- Researchers on Atlantic salmon (*Salmo salar*) report that smolts are the life stage most sensitive to low pH (Staurnes et al. 1995). Staurnes et al. (1995) reports that to be protective of Atlantic salmon, the Norwegian water quality criteria for pH during the smolting season (February 1 to July 1) is 6.5 compared to 6.2 during the balance of the year.

- An investigation of brook trout (*Salvelinus fontinalis*), a non-native char, exposure to extremely low pH revealed that survival time was directly related to fish size and inversely related to temperature (Robinson et al. 1976). The authors also concluded that the tolerance to low pH had a genetic component (i.e., some fish populations are more predisposed to tolerate low pH than others).

The majority of research on the effect of pH on invertebrates is related to the impact of acidification on abundance and diversity. There is little research on the impact of elevated pH on invertebrates. In a study of the freshwater Malaysian prawn, Cheng and Chen (2000) noted a 38 percent decrease in haemocyte (invertebrate blood cell) count when pH dropped below 5 or rose above 9. In another study, Bowman and Bailey (1998) found that zebra mussels have an upper pH tolerance limit of 9.3 through 9.6. From these studies, it can be assumed that pH levels that exceed a pH of between 9 and 10 will have a negative impact on invertebrate HCP species. Curing concrete can exceed this pH threshold and thus there is the potential for impact on local invertebrate communities

7.6.4 Alterations to Salinity

Changes in salinity can result in delayed migration, increased predation, and mortality of developing eggs and larvae.

Salinity gradients are particularly important for anadromous species because of the physiological adjustment necessary to transition from fresh water to salt water and vice versa. Juvenile salmon need a gradual change in salinity as they undergo the physiological changes needed to migrate into salt water (Groot and Margolis 1991). An extended transition zone of increasing salinity can function as an area of physiological refuge as the body adapts. The tendency for Chinook and chum salmon fry to occupy lower salinity habitats, such as marsh channels, or freshwater regions after arriving at the estuary is hypothesized to be in part due to a need to acclimate to saline water over an extended period of time (Aitken 1998; Fresh and Averill 2005).

An abbreviated salinity transition area can affect anadromous species' acclimation to the new environments, thus making them vulnerable to predation, and may alter foraging patterns. When faced with abrupt changes in salinity, migrating fish slow down and predation could increase. For example, it can take 2–3 days (or much longer) for Atlantic salmon to reorient themselves after sudden salinity changes (Russel et al. 1998).

Several studies have shown that altered salinity can influence spawning and egg development in fish species.

In a study of Puget Sound lingcod, Cook et al. (2005) showed that the optimal salinity range was 20–30 ppt for incubation of eggs, and deformities were observed at both 15 and 35 ppt.

- For Pacific herring, the optimum range for development and fertilization was in the range of 4–8 ppt salinity (Griffin et al. 1998).
- Snake River cutthroat trout show significant mortality at 18 ppt, while a southern Bonneville stock showed higher tolerance and no mortality until 22 ppt (Wagner et al. 2001).
- Striped bass have shown a preference for low salinity (0.5 ppt or less) for spawning. In the Savannah River estuary (Georgia), striped bass have shown recruitment failure because eggs were in areas of higher salinity from tide gate operations (Vandenavyle and Maynard 1994). In laboratory experiments, striped bass eggs died within 24 hours at salinities greater than 18 ppt, and larvae exposed to salinities of 15 ppt and higher exhibited stunted growth and lower survival (Winger and Lasier 1994).

No studies pertaining to potential effects of altered salinity on invertebrates were found.

7.6.5 Alterations to Suspended Sediment Concentrations and Turbidity

In general, the response of aquatic biota to elevated suspended solid concentrations is highly variable and dependent upon life-history stage, species, background suspended solids concentrations, and ambient water quality.

A distinction can be made between “suspended sediments” and “turbidity.” The International Standards Organization (ISO) defines turbidity as the “reduction of transparency of a liquid caused by the presence of undissolved matter” (Lawler 2005), as measured by turbidimetry or nephelometry. Turbidity can be caused by a wide range of suspended particles of varying origin and composition. These include inorganic materials like silt and clay, and organic materials such as tannins, algae, plankton, micro-organisms and other organic matter. The term “suspended sediments’ refers to inorganic particulate materials in the water column. Suspended sediments can range in size from fine clay to boulders, but the term applies most commonly to suspended fines (i.e., sand size or finer material).

Suspended sediments are generally measured and reported in one of three ways: as turbidity, as total suspended solids (TSS), or as water clarity (Bash et al. 2001). These three measurement methods are not always well correlated and may yield different results for any single sample (Duchrow and Everhart 1971). Because suspended sediments are a component of turbidity, turbidity is commonly used as a surrogate measure. However, the accuracy of the results is dependent on establishing a clear correlation between turbidity and suspended sediment concentrations to account for the influence of organic materials. This correlation is site specific, given the highly variable nature of organic and inorganic material likely to occur in a given setting.

Turbidity measurements reflect the optical or refractory characteristics of the material suspended in the water. Turbidity is caused by a mixture of water molecules, dissolved substances, and suspended matter. The ability of a particle to scatter light depends on the size, shape, and relative refractive index of the particular particle and the light wavelength. Turbidity is not only a measure of the amount of sediments that may be suspended in the water but also the clarity of the water. Turbidity is reported in nephelometric turbidity units (NTUs), measured using a nephelometer, or in Jackson turbidity units (JTUs), measured using an older tool called a Jackson candle turbidimeter. NTUs and JTUs are roughly equivalent at higher values but measurement of JTUs below 25 relies on human judgment (USEPA 1999). NTUs are now the preferred turbidity unit (USEPA 1999).

Turbidity has a direct impact on biota because reduced water clarity occludes photosynthetically active radiation (Govindjee 1975; Luning 1981; Olson et al. 1996; Simenstad et al. 1999; Sheldon and Boylen 1977; Strickland 1958; Thom and Shreffler 1996), as well as limits vision-based feeding opportunities for predators (Aksnes and Utne 1997). In all types of aquatic habitats, alterations to water clarity have been found to alter predator and prey assemblages and behavior (Bash et al. 2001; Williams and Ruckelshaus 1993).

Total suspended solids (TSS) are a measure of the mass of solids (particles greater than 0.45 microns) for a given volume of water. Suspended solids consist of organic and inorganic particulates that can include bound or sorbed nutrients, metals, and organic chemicals. TSS concentration is measured by filtering the sample, weighing the dried, filtered residue, and reporting TSS as weight of dried residue per volume of water sample. Older literature sometimes refers to TSS as suspended sediment concentration. TSS and suspended sediment concentration are equivalent (Bash et al. 2001).

Water clarity is a measure of sight distance through water and is affected by both suspended and dissolved loads.

The size, concentration, and chemical composition of suspended sediments can affect biota through

- benthic smothering (Terrados et al. 1998),
- gill trauma (Au et al. 2004),
- contamination with toxic substances (Malins et al. 1984),
- the suitability of spawning beds (Heywood and Walling 2007),
- prey resource availability (Mazur and Beauchamp 2003), and
- fish physiology (Berry et al. 2003).

7.6.5.1 General Effects: All Environments

Determining background levels of suspended solids or turbidity is a difficult process confounded by the inconsistency in measurement methods and natural environmental variation in factors contributing to turbidity levels (Bash et al. 2001). Background levels of turbidity and suspended solids in the Pacific Northwest differ across the various landscapes. Background turbidity is dependent upon the geologic material and weathering processes, and the geomorphology that determines the velocity of water transport for watersheds and basins across the Northwest (Bash et al. 2001, Welch et al. 1998). Turbidity often varies temporally with variations in precipitation, runoff, and discharge regimes as erosion and transport of suspended material varies. Turbidity may also vary spatially between watersheds or within watersheds as geology and water velocity vary. Widespread, continuous sampling would be required to determine a reasonable estimate of natural background turbidity levels (Bash et al. 2001).

The effects of turbidity have been documented in a number of studies on fish (e.g., Bash et al. 2001; Hasler et al. 1987; Lloyd 1987; Martens and Servizi 1993; Newcombe and Jensen 1996; Newcombe and MacDonald 1991; Salo et al. 1980; Sigler et al. 1984;) as well as on invertebrate species (e.g., Cake 1983; Mulholland 1984; Widdows et al. 1979). Of all the taxonomic groups, fish (particularly salmon) have received the most attention from researchers studying the effects of suspended solids on aquatic resources.

Although the physics of turbidity generation can be calculated, adequate data do not exist to quantify the biological response in terms of threshold sediment dosages and exposure durations that can be tolerated by various marine and estuarine organisms. Numerical modeling simulations of dredging-related suspended-sediment plume dynamics are currently being developed under the USACE’s Dredging Operations and Environmental Research Program. Present data indicate that responses to suspended sediments are highly species-specific, with some species having lethal effects at several hundred parts per million (ppm) in 24 hours and others having no effect at concentrations above 10,000 ppm for 7 days. Studies on east coast species have identified lethal concentration levels and Newcombe and Jensen (1996) have developed a predictive model for defining lethal and sublethal fish injury threshold levels for suspended solid concentrations. However, threshold studies for the temporary impacts of suspended sediment levels specific to aquatic environments in the Northwest are lacking.

Suspended solids and the turbidity that high concentrations of suspended solids can produce are natural features of many aquatic systems. The range of potential impacts associated with elevated suspended solids includes some beneficial impacts. A broad range of research has demonstrated that suspended sediment and elevated turbidity can have a broad range of adverse effects on aquatic organisms, ranging from minor, short-term behavioral alterations, to effects on food web productivity and forage success that influence survival, growth, and fitness, to direct injury and mortality (Henley et al. 2000). As would be expected, these effects are complex and variable depending on the magnitude of the sediment impact in question relative to natural background conditions and the specific sensitivity of the organisms exposed to the stressor.

Recent studies have shown that the size and shape of suspended sediments and the duration of exposure are important factors in determining the extent of adverse effects of increased turbidity on salmonids (Martens and Servizi 1993; Newcombe and MacDonald 1991; Northcote and Larkin 1989; Servizi and Martens 1987, 1991).

In addition to size and shape, the concentration of suspended sediments would determine the severity of the responses elicited in aquatic organisms. Effects on aquatic organisms will differ based on their developmental stage. Suspended sediments may affect salmonids by altering their physiology, behavior, and habitat, all of which may lead to physiological stress and reduced survival rates. For example, high levels of suspended solids may be fatal to salmonids due to, for example, gill trauma, osmoregulation impairment, and changes in blood chemistry. Lower levels of suspended solids and turbidity may cause chronic sublethal effects, such as loss or reduction of foraging capability, reduced growth, reduced resistance to disease, increased stress, and interference with cues necessary for orientation in homing and migration (Bash et al. 2001; Lloyd 1987; Newcombe and MacDonald 1991).

Newcombe and MacDonald (1991) identified the effects of suspended solids on salmonids as:

> (1) **lethal effects** that can cause overall population declines with long-term effects;
> (2) **sublethal effects**, such as tissue injury or physiologic alterations to an animal, with effects that may not lead to immediate death but may produce mortalities and population declines over time; and
> (3) **behavior effects** that alter animal behavior and have the potential of immediate death or population decline and mortality over time. Although these effects can be chronic and may not lead to immediate death, they may produce mortalities and population declines over time.

Bash et al. (2001) group the effects of turbidity on salmonids as:

> (1) **physiological effects** that include gill trauma, osmoregulation, blood chemistry changes, and reproduction and growth effects;
> (2) **behavioral effects** that include avoidance, territoriality, foraging, predation, and homing/migration effects; and
> (3) **habitat effects** that include reduced spawning habitat due to increased deposition of suspended fines to stream beds, which are known to fill in interstitial spaces in stream bed gravels and lower the suitability of stream bed spawning and egg and larval rearing habitat, and effects on hyporheic upwelling that reduce the levels of dissolved oxygen in the gravels.

These effects hold for all salmonid species and many others, such as sturgeon, pygmy whitefish, and dace species (Bash et al. 2001; Chapman 1988; Hallock and Mongillo 1998; Nightingale and Simenstad 2001a; Pitt et al. 1995; Quinn and Peterson 1994; Salo et al. 1980; Sigler 1990; Welch et al. 1998; Wildish and Power 1985; Williamson 1985; Wydoski and Whitney 2003).

Although juveniles of many fish species thrive in rivers and estuaries with naturally high concentrations of suspended solids, studies have shown that suspended solids concentration, the duration of exposure, the frequency of exposure, water temperature, and the size of the suspended particles can be important factors in assessing risks posed to salmonid populations (McLeay et al. 1987; Newcombe and MacDonald 1991; Servizi and Martens 1992, in Bash et al., 2001). Many species have adapted to living in high suspended sediment conditions (Lake and Hinch 1999) and the impact of suspended sediment on fish physiology may be ameliorated by reduced predation pressure, as has been shown for emigrating Pacific salmon in the clear water Harrison and turbid Fraser Rivers of British Columbia (Gregory and Levings 1998).

It can be concluded that activities that allow significant increases in suspended sediment have a high risk of causing incidental take to potentially covered species exposed to this condition. The risk of take increases in proportion to:

- The magnitude and duration of the impact
- The vulnerability of the affected life-history stage
- The inability of the organism to avoid the impact through avoidance behavior
- The physiological, developmental, and behavioral impairments
- Indirect mechanisms such as exposure to predation.

Several NMFS biological opinions on bridges, water and gas line crossings, culverts and marinas have been reviewed for their conclusions on potential water quality impacts to listed fish species. In all cases, sediment- and turbidity-related impacts comprised the overwhelming majority of discussion on water quality effects. In most cases, the magnitude, frequency, and duration of sediment pulses are expected to be similar to naturally occurring conditions during natural fluctuations in flow conditions, and few salmonids are predicted to be present during in-water work windows; therefore, NMFS concluded that potential increases in turbidity would have negligible impacts on salmonids and their habitats (NMFS 2006a; NMFS 2006f; NMFS 2006h; NMFS 2006i; NMFS 2006j; NMFS 2006k; NMFS 2006m; NMFS 2006n). However, NMFS found that elevated turbidity can cause direct mortality (NMFS 2006g), while sublethal threats include harassment, as feeding patterns may be affected and fish are likely to avoid areas of increased turbidity (NMFS 2006d).

7.6.5.1.1 Suspended solids and mortality

Direct mortality may result from suspended solids depending upon the concentrations encountered, the duration of exposure, the size and shape of the particles, as well as other environmental stressors (e.g., high water temperatures or low dissolved oxygen).

Fish mortality could result from the damage to gills caused by the abrasive properties of suspended solids. As sediment begins to accumulate in the gill filaments, fish excessively open and close their gills to expunge the silt. If irritation continues, mucus is produced to protect the gill surface, which may impede the circulation of water over gills and interfere with fish respiration (Berg 1982, in Bash et al. 2001).

Servizi and others have investigated effects on juvenile salmon exposed to Fraser River sediments. Servizi and Martens 1987 (in Bash et al. 2001) demonstrated increased lethality of solids with increasing particle size, specifically for particles described as angular to subangular. The authors reported that juvenile sockeye salmon had a 96-hour LC_{50} (the concentration that is lethal to 50 percent of a sample population) of 17,600 ppm. Fine sediments (0 to 740 micrometers) lodged in gills and caused gill trauma at 3,148 milligrams per liter (mg/L) or 20 percent of the 96-hour LC_{50} value. Servizi and Martens (1991) exposed juvenile coho salmon to natural Fraser River suspended solids and found a 96-hour LC_{50} of 22,700 ppm (Servizi and Gordon 1990). Using the identical apparatus and sediment source, juvenile sockeye salmon had a 96 hour LC_{50} of 17,600 ppm (Servizi and Martens 1987), and an LC_{50} of 31,000 ppm for juvenile Chinook salmon (Servizi and Gordon 1990).

Although juveniles of many fish species thrive in rivers and estuaries with naturally high concentrations of suspended solids, studies have shown that suspended solids concentration as well as the duration of exposure can be important factors in assessing risks posed to salmonid populations (McLeay et al. 1987; Servizi and Martens 1987, 1992; Northcote and Larkin 1989; Newcombe and MacDonald 1991). McLeay et al. (1987) found 20 percent mortality of Arctic grayling at a concentration of 100,000 ppm.

Studies on salmonids exposed to volcanic ash attributed acute mortality in suspended sediment mixtures to reduced oxygen uptake (Newcomb and Flagg 1983; Noggle 1978).

For white sturgeon, laboratory studies have shown that the survival of developing embryos was reduced to 5 percent in the presence of 0.19–0.8 in (5–20 mm) thick layers of sediment compared to over 80 percent survival in controls (Kock et al. 2006).

Thresholds for lethal effects on clams and eastern oysters have been reported, with negative impacts on eastern oyster egg development occurring at 188 ppm of silt (Cake 1983) compared to a 1,000 ppm threshold for hard clam eggs (Mulholland 1984). For clams and oysters, there appears to be a break point at 750 mg/L between chronic and acute impacts of suspended sediment (Nightingale and Simenstad 2001a). At levels below 750 mg/L, development continues for both clams and oysters, but at levels above 750 mg/L that last for 10 to 12 days, effects become lethal (Nightingale and Simenstad 2001a).

The direct impacts to invertebrates could include clogging of filtration mechanisms, thereby interfering with ingestion and respiration; abrasion; and in extreme cases, smothering and burial resulting in mortality (Berry et al. 2003).

Burial of invertebrate species which have limited motility can lead to organism mortality as a direct effect from increased suspended sediments. Burial of invertebrate species will occur most frequently during the construction phase of a project.

- Limpets in intertidal habitat are affected by smothering and interference with feeding activity. In a field study in the UK, grazing by limpets (*Patella vulgata*) was decreased by 35 percent after the addition of fine sediments, to as little as

0.04 in (1 mm) thick (equivalent to 1.02 x 10^{-5} lb/ft^2 [50 mg/m^2]) with mortality and inhibition of feeding at higher levels of fine sediment (4.09 x 10^{-5} lb/ft^2 200 mg/m^2]) (Airoldi and Hawkins 2007).

- The burial of mollusks and related stress or mortality resulting from partial and complete burial have been addressed empirically (Hinchey et al. 2006).

- Olympia oysters have been shown to be intolerant of siltation and do best in the absence of fine-grained materials (WDNR 2006b).

Results of these studies indicate that species-specific responses vary as a function of motility, living position, and inferred physiological tolerance of anoxic conditions. Mechanical and physiological adaptations contribute to this tolerance.

7.6.5.1.2 Suspended solids and sublethal physiological effects

The non-lethal impacts of elevated suspended solid concentrations on fish could include reduction in feeding rates and physiological responses such as gill trauma, altered osmoregulation, altered blood chemistry, reproduction, and growth. Most research has entailed laboratory studies.

Stress response is a result of the combination of duration, frequency, and magnitude of exposure and other environmental factors. Stress responses vary between salmonid species and life stages.

Suspended sediment levels associated with injury or mortality are typically quite high. Lake and Hinch (1999) found concentrations in excess of 40,000 ppm suspended solids to elicit stress responses (e.g., decreased leukocrit, indicating reduced immunity response), which correlate to occurences of gill damage in juvenile coho salmon. Angular sediments, as opposed to rounder sediments, are associated with higher fish stress responses at lower sediment concentrations. This may be due to irritation caused by angular sediments that result in increased mucus production and decreased oxygen transfer. Although the causes of mortality were not clear, Lake and Hinch (1999) found that mortality occurred at concentrations of 100 parts per thousand (ppt), with no differences found in mortality rates in natural or more angular anthropogenically derived sediments. Suspended solids concentrations this high would likely only be associated with the most extreme construction-related impacts. However, other studies have shown lethal effects at much lower concentrations in salmonids, indicating that the issue is complex, and a precautionary approach to sediment impacts is desirable to limit the potential for adverse effects.

Servizi and Martens (1992) characterized suspended solids concentration, duration and frequency of exposure, water temperature, and size of suspended particles as synergistic factors affecting the physiological response in salmonids. That is, the combination of factors will elicit a greater total effect than would be expected by the “sum” of the individual effects.

Newcombe and MacDonald (1991, in NMFS 2004b) identify exposure duration as the critical determinant of the occurrence and magnitude of physical or behavioral effects for salmonids. This finding is supported by the fact that salmonids have evolved in systems that periodically experience short-term pulses (days to weeks) of high suspended solids loads, often associated with flood events, and are adapted to such short-term, high-pulse exposures. The timing of exposure to suspended sediment is also very important, as it may affect different life-history stages in different ways (Berry et al. 2003).

Smaller increases in suspended solids concentrations that occur over an extended period of time may produce similar impacts to greater suspended solids concentrations encountered during a shorter time period (Newcombe and Jensen 1996). Newcombe and Jensen (1996) documented differences in the onset of non-lethal impacts among juvenile and adult salmonids, egg and larval salmonids, adult non-salmonid estuarine species, and adult non-salmonid freshwater species. Juvenile and adult salmonids exhibited widely variable impact thresholds both in terms of duration of exposure and concentration of exposure. Juvenile salmonids exhibited more impacts related to very short-term exposure (1 hour) than the other species groups.

In reviewing the information presented by Newcombe and Jensen (1996), as well as additional, more recent information, U.S. Environmental Protection Agency (USEPA) scientists concluded that (with the possible exception of salmonids) insufficient information exists to confidently establish a dose response model at this time (Berry et al. 2003). However, the USEPA scientists conclude that with additional research it may be possible to develop national dose response criteria for suspended solids. Berry et al. (2003) provides a tabular summary of the widely variable dose response data for many species.

Studies on a variety of fishes, including sockeye and Chinook (Newcomb and Flagg 1983), coho, four-spine stickleback, cunner, and sheepshead minnow (Noggle 1978), attribute chronic and acute impacts from high suspended solids to reduced oxygen uptake (Wilber and Clarke 2001).

Gills may be irritated by abrasive suspended sediments. Several laboratory studies have shown gill trauma and increased coughing frequency with increased turbidity. Fish must keep their gills clear for oxygen exchange. In the presence of high loadings of suspended solids, they engage a cough reflex to perform that function. Due to increased metabolic oxygen demand with increased temperatures and the need to keep pathways free of sediments for oxygen uptake, increased temperature and reduced oxygen levels combine to reduce the ability of fish to cough and maintain ventilation rates. The stress induced by these conditions can lead to compromised immune defenses and reduced growth rates (Au et al. 2004). Sigler et al. (1984) noted reduced growth rates in juvenile steelhead and coho salmon at suspended solids concentrations as low as 100 ppm, while Servizi and Martens (1992) noted increased cough frequency in juvenile coho at concentrations of approximately 240 ppm. Such cumulative stressors are thought to be likely contributors to mortality when exposure to high suspended sediment levels occurs for extended durations (Servizi and Martens 1991).

Other studies have shown impacts on osmoregulation during smolting in association with increases in suspended sediment (Bash et al. 2001).

Suspended sediments affect light transmission in water. Light transmission is thought to play a role in the development of visual acuity in fish (Nightingale and Simenstad 2001a). Visual acuity adjustment in estuarine waters is part of the smolting process of salmonids (Beatty 1965; Folmar and Dickhoff 1981). Similar visual development has been reported in juveniles of other species, such as sand lance, kelp greenling, and lingcod (Britt 2001). Evidence of similar development among sand lance has also been reported by Tribble (2000).

Elevated concentrations of suspended solids could have a wide range of impacts on both pelagic and benthic invertebrates (Cordone and Kelly 1961; Peddicord 1980; Waters 1995; Wilber and Clarke 2001, in Berry et al. 2003). The limited mobility of many invertebrates would prevent them from escaping even temporary pulses of increased suspended sediment loads.

Negative impacts on eastern oyster (Crassostrea virginica) egg development have been shown to occur at 188 ppm total suspended solids (Cake 1983). Hardshell clam eggs appear to be more resilient, with egg development affected only after total suspended solids concentrations exceeded 1,000 ppm (Mulholland 1984). Mulholland (1984) showed that suspended solids concentrations of <750 ppm allowed for continued larval development but higher concentrations for durations of 10–12 days showed lethal effects for both clams and oysters. Comparable impacts could be expected in other benthic bivalves such as the California floater, Western ridged mussel, and Olympia oyster.

Evidence of physiological responses among shellfish to increased turbidity appears to be ambiguous.

- It has been hypothesized that at lower turbidity levels, resuspended chlorophyll may act as a food supplement enhancing growth, while at higher levels, planktonic food resources are diluted to the point of inhibiting growth (Nightingale and Simenstad 2001a).

- For bivalves, when suspended solids concentrations rise above their filtering capacities, their food becomes diluted (Widdows et al. 1979).

- In environments with high algal concentrations, Clarke and Wilber (2000) reported that the addition of silt, in relatively low concentrations, showed increased growth of mussels (Kiorboe et al. 1981), surf clams (Mohlenberg and Kiorboe 1981), and eastern oysters (Urban and Langdon 1984).

- Bricelj and Malouf (1984), found that hardshell clams decreased their algal ingestion with increased sediment loads, and no growth rate differences were observed between clams exposed to algal diets alone and clams with added sediment loads (Bricelj et al. 1984).

- Urban and Kirchman (1992) reported similarly ambiguous results concerning suspended clay. Suspended clay (20 ppm) interfered with juvenile eastern oyster ingestion of algae, but it did not reduce the overall amount of algae ingested.

- Grant et al. (1990) found that the summer growth of European oysters was enhanced at low levels of sediment resuspension and inhibited with increased deposition. It was hypothesized that the chlorophyll in suspended solids may act as a food supplement that could enhance growth, but higher levels may dilute planktonic food resources, thereby, suppressing food ingestion.

- Changes in behavior in response to sediment loads were also noted for soft-shelled clams under sediment loads of 100–200 ppm, with changes in their siphon and mantles over time (Grant and Thorpe 1991).

It appears likely that shellfish are generally less vulnerable to acute effects of suspended sediment than are fish, but have some risk from chronic exposure. Collectively, these studies show no clear pattern of sublethal effects from elevated concentrations of suspended solids (and thereby turbidity) that could be generally applied across aquatic mollusks. The uncertainty is further complicated by the fact that many of the HCP invertebrate species are poorly studied. This indicates the need for directed studies on the sensitivity of these species before effects thresholds can be set. In the absence of this information, however, it is useful to consider that HCP invertebrates are all bottom-dwelling mollusks that have evolved to live in dynamic environments under conditions of variable turbidity. Therefore, sensitivity to turbidity-related stressors would be expected to occur only when conditions exceed the range of natural variability occurring in their native habitats. The rate of sediment deposition is important: benthic invertebrates are adapted to moderate sediment movement and deposition, but not to extremely high rates. There is a risk that potentially covered shellfish species could experience some level of incidental take due to increased suspended sediments.

7.6.5.1.3 Suspended solids and behavioral effects

Fish in systems that naturally produce periods of elevated suspended solids concentrations can encounter prolonged periods in these conditions. Many fish species thrive in rivers and estuaries with naturally high concentrations of suspended sediments. For example, some of the largest salmon-producing river systems are turbid (see Gregory 1993) and juvenile salmon occupy turbid areas for significant portions of their early life (Levy and Northcote 1982; Simenstad et al. 1982, in Gregory 1993). Fishes' history of exposure to turbidity can affect their response to it.

It is currently unknown what behavioral mechanisms are triggered as various fish species encounter patches of increased turbidity, such as dredging plumes. Also unknown is what threshold of turbidity might be a cue to fish to avoid light-reducing turbidity.

Studies agree that turbidity may affect some aspects of salmonid behavior, but differ in their conclusions whether turbidity affects salmonids' homing ability. Field studies have indicated that while increased turbidity may delay salmonid migration, it does not seem to alter homing ability (Bash et al. 2001). However, Sigler et al. (1984) reported that suspended sediments have been shown to affect fish behavior such as homing, avoidance responses, territoriality, and feeding.

The final phase of salmon homing migration requires olfactory cues (Hasler et al. 1987; Hasler and Scholz 1983). In studies of returning Chinook spawners, Whitman et al. (1982) found that suspended ash at concentrations of 650 ppm did not influence homing performance. Preference experiments indicated that Chinook, when given the choice, preferred clean home water to municipal drinking water, with the presence of ash reducing the preference for home water. It was concluded that fish could recognize home water despite the ash suspension and that any reduced home-water preference was due to ash avoidance (Whitman et al. 1982).

Aksnes and Utne (1997), Mazur and Beauchamp (2003), and Vogel and Beauchamp (1999) all report that suspended solids at sublethal concentrations have been shown to affect fish functions such as avoidance responses, territoriality, feeding, and homing behavior.

Simenstad (1990) identifies the behavioral effects that would affect migrating fishes, such as reduced foraging success, increased risk of predation, and migration delay to be highly dependent upon the duration of exposure. The primary determinant of risk level is likely to lie in the spatial and temporal overlap between the area of elevated turbidity, the degree of turbidity elevation, the occurrence of fish, and the options available to the fish relative to carrying out the critical function of their present life-history stage.

Indirect effects of suspended solids on fish through alteration of their food source have been documented. Suttle et al. (2004) observed that prey species available to steelhead trout decreased with increasing fine-sediment concentration. With increasing fine sediments, gravel substrates became increasingly embedded, fewer epibenthic prey were available and macroinvertebrates in the stream shifted to burrowing taxa that were unavailable to trout as a food source.

If turbidity increases in the shallow nearshore area, the impacts on fish would likely be to the juvenile life-history stage. The results of Britt (2001), Britt et al. (2001), Tribble (2000), and Ali (1975) indicate the importance of light transmission to the fitness and survival of larval and juvenile estuarine fish.

Turbid water can provide a form of cover from predators, such as fish or birds, that need to see their prey (Cyrus and Blaber 1987; Gregory 1993). Experiments have shown that white sturgeon larvae predation by prickly sculpin increased in the presence of low-turbidity water (Gadomski and Parsley 2005). Several researchers have documented that turbidity can reduce predation pressure on young salmonids by providing protective cover that enables them to avoid detection or capture by predators (Gregory 1993; Gregory and Levings 1996; Gregory and Levings 1998).

Turbidity may trigger a predation cover response for salmonids. The studies of Gregory and Northcote (1993) demonstrate that at particular levels of increased turbidity, juvenile salmon actually increase their feeding rates, while at certain threshold levels (such as >200 ppm) they demonstrated pronounced behavioral changes in prey reaction and predator avoidance. It is not known what behavioral mechanisms are triggered when various fish species encounter patches of increased turbidity in their otherwise naturally turbid waters to which they are accustomed.

Bash et al. (2001) exhaustively reviewed 40 years of research on the physiological and behavioral effects of turbidity and suspended solids on salmonids. This review found both laboratory and field studies that show salmonids generally avoid areas of increased turbidity. Salmonids’ avoidance of turbid waters may be one of the most important effects of suspended solids (Birtwell et al. 1984; Bisson and Bilby 1982). Salmonids have been observed to move laterally and downstream to avoid turbid plumes (Lloyd 1987; McLeay et al. 1984, 1987; Servizi and Martens 1991; Sigler et al. 1984). Moderate turbidity levels (11 to 49 NTUs) were shown to cause juvenile steelhead and coho to leave rearing areas (Sigler et al. 1984).

Consistent with their early reliance on nearshore estuarine habitats, which have relatively high turbidity levels compared to pelagic or freshwater habitats, juvenile chum salmon are classified by Nightingale and Simenstad (2001a) as “turbidity tolerant compared to other fishes.” In a study of dredging impacts in Hood Canal, Salo et al. (1979) found that juvenile chum showed avoidance reactions to low levels of turbidity ranging from 2 to 10 ppm above ambient concentrations. However, in related laboratory tests, Salo et al. (1980) found that avoidance was not shown until a concentration of 182 ppm was reached. These behavioral thresholds vary across species and life-history stages. The size of the turbidity plume may be important; turbidity plumes that do not extend from bank to bank would not be expected to significantly impact the behavior of migrating salmonids, as the fish are able to avoid the areas of high turbidity (Nightingale and Simenstad 2001a).

Recent literature maintains that suspended solids are important to fish as visual feeders, and for young fish with limited prey capture aptitude. Visual feeders would generally experience reductions in feeding rates or success at elevated turbidity levels (Boehlert and Morgan 1985; Johnson and Wildish 1982; Vinyard and O’Brien 1976; all in Berry et al. 2003; Breitburg 1988; Rowe and Dean 1998). However, the amount that turbid conditions would modify feeding would be affected by various factors, including species’ visual acuity, target prey type, and adaptation to turbid habitats.

The effects of turbidity on larval fish feeding are not well understood. Larval fish typically have short reactive distances and require high prey densities. Larval salmonids, in particular, have little or no swimming capability, are visual feeders, and undergo high mortality rates due to starvation (Nightingale and Simenstad 2001a). Increased turbidity and reduced water clarity could negatively impact the already limited prey-catching ability of larval fish (Nightingale and Simenstad 2001a).

Laboratory studies have shown alterations in social interactions and decreased territoriality in response to increases in turbidity. It has been suggested that decreased territoriality and a breakdown in social structure can lead to secondary effects such as altered feeding and growth rates, which may in turn lead to increased mortality. In studies of coho behavior in the presence of short-term pulses of suspended solids, Berg and Northcote (1985) found that salmonid behavior is disrupted by elevated turbidity levels, as evidenced by changes in territorial, gill flaring, and feeding behaviors. At turbidity levels of between 30 and 60 nephelometric turbidity units (NTUs), social organization broke down, gill flaring occurred more frequently, and only after a return to a turbidity of 1–20 NTUs was the social organization re-established. Similarly, feeding success was also found to be linked to turbidity levels, with higher turbidity levels reducing prey capture success.

Studies on other species of fish have shown that increased turbidity affects other fishes' behavior in ways similar to its effects on salmonids.

- Wildish and Power (1985) reported avoidance of suspended sediments by smelt (*Osmerus mordax)* at concentrations of approximately 20 mg/L. They reiterated that, in a previous study, performed in 1981 with different protocol and analytical methods, Johnson and Wildish had observed Atlantic herring *(Clupea harengus harengus)* avoiding suspended sediments at approximately 10 mg/L.

- Herring and American shad (*Alosa sapidissima*) exhibited changes in depth preferences in the presence of turbid conditions (Johnson and Wildish 1982; Dadswell et al.1983, both in Berry et al. 2003).

- Striped bass (*Morone saxatilis*) larvae observed feeding under turbid conditions had varying success rates with different prey items (Breitburg 1988).

- In Midwestern U.S. prairie fishes, Bonner and Wilde (2002) found that elevated turbidity had less effect on prey consumption by chub species that are adapted to highly turbid habitats than on shiner species characteristic of less-turbid habitats.

- In an investigation of the effects of turbidity on juvenile marine species (several mullet and perch-like species) in southeastern Africa, Cyrus and Blaber (1987) concluded that some species appeared to prefer turbid (10 to 80 nephelometric turbidity units [NTUs]) over clear water (less than 10 NTUs).

Increased suspended sediment has also been associated with behavioral changes among shellfish. Changes in invertebrates' behavior in response to turbidity would primarily be related to light attenuation that could lead to changes in feeding efficiency and behavior (i.e., drift and avoidance) and alteration of habitat that would result from changes in substrate composition, which would affect the distribution of infaunal and epibenthic species (Donahue and Irvine 2003; Waters 1995; Zweig and Rabeni 2001, in Berry et al. 2003). Soft-shelled clams *(Mya arenaria)* at suspended sediment concentrations of 100 to 200 mg/L showed reduced valve gape and retracted siphons and mantles (Grant and

Thorpe 1991). Berry et al. (2003) provides a tabular summary of the widely variable dose response data for many species of invertebrates.

7.6.5.2 Ecosystem-Specific Effects: Marine and Estuarine

In tidal areas, seagrasses have been linked to improved water quality. As an example, Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve.

Increased turbidity is known to compromise the survivability of submerged aquatic vegetation (Parkhill and Gulliver 2002; Terrados et al. 1998) such as eelgrass (Erftemeijer and Lewis 2006) because it limits the amount of sunlight the plants receive. Eelgrass is associated with important rearing habitats for a suite of marine fishes, like Pacific cod, Pacific salmon, rockfish, Pacific herring, walleye pollock, and rockfish (Gustafson et al. 2000; Murphy et al. 2000; Nightingale and Simenstad 2001a; Simenstad et al. 1999).

Increased turbidity can also bury the plants if sediment in suspension settles out (Mills and Fonseca 2003). In a study of the impact of sedimentation on seagrass in southeast Asia, Terrados et al. (1998) noted an approximate 50 percent decline in the number of seagrass species and a precipitous decline in seagrass biomass with a 15 percent increase in the clay content of the sediments.

High turbidity and the resulting excessive siltation in nearshore marine habitats are known to decrease the suitability of larval settling habitat for the northern or Pinto abalone (NMFS 2007a). High sediment loads are also known to decrease survival rates of the abalone by impeding respiration and feeding efficiency. High turbidity levels due to high nutrient loads are also known to have an effect on the Pinto abalone by creating dense filamentous algae blooms that these shellfish may not be able to consume and that may cover important food resources. Impacts on kelp beds, which are important rearing habitat for abalone, limit the growth and survival of these shellfish in the marine nearshore environment.

Estuarine habitat loss and pollution are considered the greatest threats Newcomb's littorine snail, which uses nearshore ecosystems and coastal waters. This snail uses the narrow strip of land supporting pickleweed. Changes to the marsh in the form of effluent and waste that stem from turbidity or lack of water clarity have been known to destroy habitat and nearly extirpate populations in California, Oregon, and Grays Harbor (Larsen et al. 1995). Destruction or modification of tidelands and tidal wetlands poses a significant impact on this species.

The planktonic larvae of the nearshore marine Olympia oyster, found in lower intertidal areas at 1 to 2 ft elevation or in tidal channels, require a firm substrate, such as rock or shell. This species is particularly intolerant of siltation and grows best on firm substrates with substantial water flow (West 1997; Couch and Hassler 1990).

7.6.5.3 Ecosystem-Specific Effects: Riverine

No general discussions of turbidity in riverine systems were reviewed. Several specific examples using project-specific information to support the federal Services' biological opinions exist.

One example of thresholds developed and approved by the federal agencies is found in a biological opinion for an intensive 0.2-mile project that entailed "rebuilding" a severely eroded bank on the Stillaguamish River, Washington. This calculation depended upon site-specific information and was clearly intended for project-specific use. Details are provided as an example.

USFWS and NOAA Fisheries calculated suspended solid concentrations and periods of exposure that would result in adverse impacts to bull trout and Chinook salmon (NMFS and USFWS 2005). The calculation depended upon the ratio of turbidity (measured in NTUs) to suspended solids (measured in mg/L), an estimate of the length of time that sediments would be suspended, and a USFWS draft guidance document[1]. The federal agencies determined that adverse effects to bull trout and Chinook salmon would occur in the following circumstances:

1. When background NTU levels are exceeded by 96 NTUs at any point in time
2. When background NTU levels are exceeded by 35 NTUs for more than 1 hour cumulatively over a workday
3. When background NTU levels are exceeded by 13 NTUs for more than 3 hours cumulatively over a workday

To assess the potential downstream extent of these effects, USFWS reviewed its monitoring database and found that for construction activities involving cofferdam removal, bank stabilization, and river scour protection, the state water quality standards were not met in some cases until more than 600 feet downstream. USFWS identified another bank protection project in its database in which peak turbidity levels of more than 130 NTUs over background were detected 4,300 feet downstream of the work area (the farthest point downstream at which monitoring occurred) in a plume that lasted over 5 hours. USFWS determined that the plume persisted at an intensity and duration sufficient to adversely affect salmonids for several miles. Based on known extent, duration, and intensity of sediment plumes from previous instream work, the scale and methods of the proposed project, and the characteristics of the river in the action area, the federal agencies anticipated that turbidity levels that result in adverse effects to bull trout and Chinook salmon were reasonably certain to occur as far downstream as 3.3 miles (NMFS and USFWS 2005). For the specific bank protection project under review, the federal agencies concluded that the adverse impacts would extend downstream more than 16 times the length of the project.

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[1] Based on nine years of water quality data in the river, the ratio was determined to be 1.0 NTU:4.2 mg/L suspended solids. The length of time was estimated to be during daylight hours for six weeks. The USFWS guidance document identified (*Sediment Biological Review,* draft May 2005) was not available for use in this paper.

No studies specifically discussing impacts from altered suspended sediments and turbidity on freshwater covered invertebrates were identified.

Western ridged mussels are filter feeders that require constant water flow. They typically reside in stable, nonshifting habitats and are absent from areas with continuous turbidity or high nutrient content. Like the California floater mussel, increased suspended solids and sedimentation impede their ability to feed and can smother them (Watters 1999).

7.6.5.4 Ecosystem-Specific Effects: Lacustrine

In general, impacts of water quality modifications in lacustrine systems may be expected to be greater than those in either marine or riverine systems, because circulation in a lake is generally much more limited than in a river, estuary, or marine area. Thus, alterations in suspended solids, turbidity, nutrients, and pollutant loading could be expected to have the largest impact in lacustrine systems.

High sediment loads may decrease survival rates by impeding respiration and feeding efficiency. Suitable habitat for the freshwater bivalve California floater (which occurs in both lakes and rivers) is characterized by low turbidity levels. Limiting factors identified by Larsen et al. (1995) include sediment, debris, siltation, or bedload movement that is known to smother or crush juvenile clams and cover and kill adults. Similarly, WDNR (2005a) reported that increased suspended solids and sediment loads may impede floater feeding and cause mortality through smothering (Watters 1999).

7.6.6 Alterations to Nutrient and Pollutant Loading

Nutrients of concern in Washington waters include phosphates and nitrates. Pollutants of concern include polycyclic aromatic hydrocarbons (PAHs) polychlorinated biphenyls (PCBs), and metals such as copper (Cu), chromium (Cr), arsenic (As), and zinc (Zn). The chemicals can be found in water and in sediment.

Numerous studies have shown that fishes and invertebrates exposed to contaminants may bioaccumulate and concentrate trace pollutants to levels deemed harmful. Bioaccumulation occurs when contaminants are passed between two or more trophic levels. Invertebrates (in particular burrowing and attached organisms) are significantly affected by the contaminants associated with treated wood. Additionally, the trophic transfer of metals and hydrocarbons may adversely affect fishes. Studies in the Pacific Northwest by Stein et al. (1995) and Johnson et al. (2007) have indicated that PCB and PAH concentrations in juvenile Chinook salmon tissue are highest in industrial areas (e.g., Duwamish estuary, Columbia River). Activities that produce discharges containing high levels of sulfites and toxins have been known to threaten Olympia oyster populations.

Stormwater has been recognized as a potentially major source of pollution that could affect HCP-covered species. Sources of stormwater pollutants have been reviewed and summarized in numerous reports (Barber et al. 2006; Barrett et al. 1995; Yonge et al. 2002; Young et al. 1996). Sources of stormwater pollution can be classified into three general categories: atmospheric deposition, vehicles (including fuels and exhaust emissions), and direct and indirect deposition and application (Table WQ-4).

Atmospheric deposition refers to substances that are deposited on land surfaces from the air. The atmospheric deposition can contain pollutants such as nutrients, particulates, PAHs, PCBs, and heavy metals. Incomplete combustion of fossil fuels contributes nutrients and PAHs in deposition materials. PCBs primarily originate from historic usage of these compounds in industrial applications. Most pollutants associated with automobiles originate from engine wear and exhaust, lubricants, rusting, and tire wear. Brake pad wear is a source of copper and zinc, which are the metals most commonly found in highway runoff; tires contain zinc; some older brakes contain lead; and wheel-balance weights are made primarily of lead. The application of fertilizers, herbicides, and pesticides), roadway/parking lot maintenance (e.g., deicing and road repairs), and animal wastes can also contribute pollutants

Table 7-9. General source categories of roadway pollutants.

Source Category	Pollutants
Atmospheric deposition	Particulates, nitrogen, phosphorus, metals, polycyclic aromatic hydrocarbons (PAHs), and polychlorinated biphenyls (PCBs)
Vehicles	Particulates, rubber, asbestos, metals, sulfates, bromide, petroleum, and PAHs
Direct and indirect deposition and application	Particulates, nitrogen, phosphorus, metals, sodium, chloride, sulfates, petroleum, pesticides, and pathogens

Increased runoff and loadings of the pollutants noted above may degrade sediment and ambient water quality in the immediate vicinity of the facility and affect the aquatic food web. Food web impacts associated with stormwater include increased suspended solids, resuspension of contaminated sediments, and the introduction of toxic substances.

Stormwater impacts are mitigated by regulations promulgated by the Washington State Department of Ecology (Ecology) under the federal Clean Water Act (33 USC §§ 1251-1387). The Ecology regulations are subject to USEPA review and Section 7 requirements of the ESA (16 USC 1531-1544).

7.6.6.1 General Effects: All Environments

7.6.6.1.1 Eutrophication

Eutrophication, "the process by which a body of water becomes enriched in dissolved nutrients that stimulate the growth of aquatic plant life usually resulting in the depletion of dissolved oxygen" (Merriam-Webster on-line dictionary at http://www.merriam-webster.com/dictionary/eutrophication) can be a result of HPA-permitted activities.

Eutrophication occurs when limits to vegetative growth are reduced. In Washington, the primary limiting nutrient in fresh water is phosphorus. Abundant iron in freshwater systems binds with phosphorus (P) and reduces P availability for biotic assimilation. When nutrient limitations are eliminated, vegetative growth increases. Vegetative growth accelerates carbon fixation; the additional carbon loading to the aquatic system increases respiration as heterotrophs use carbon for energy. Through the process of carbon oxidation, oxygen is converted to CO_2, and ambient dissolved oxygen levels decrease. In eutrophic systems, nighttime respiration drives down dissolved oxygen to levels that would adversely affect many of the HCP species. Eutrophication-induced hypoxia is a nationwide problem (Scavia and Bricker 2006).

Eutrophication of receiving waters across Washington State has been identified as a major source of environmental degradation (Nelson et al. 2003; Pickett 1997). Activity that decreases in-channel processing of nutrients can contribute to the increased export of nutrients to downstream receiving waters, potentially affecting many of the HCP species.

In Washington, low dissolved oxygen episodes in Hood Canal have resulted in widespread fish and invertebrate kills (Peterson and Amiotte 2006). These low dissolved oxygen episodes have been linked to excess carbon loading due to nutrient enrichment. Resultant algal blooms may impact dissolved oxygen levels and, if certain species flourish, contribute to paralytic shellfish poisoning (Horner 1998).

When riparian canopies are opened, increased photosynthetic active radiation reaches the channel, temperatures increase, and nutrient loading increases. These alterations have been shown to increase macroinvertebrate abundance and biomass as well as algal biomass (Fuchs et al. 2003; Hetrick et al. 1998). However, the cumulative effect of increased nutrient loading will contribute to eutrophication in downstream receiving waters.

7.6.6.1.2 Metal Toxicity

HPA-permitted activities could introduce metals into water bodies directly (for example, by allowing metal-treated wood for in-water structures) or indirectly (for example, by supporting infrastructure such as roads and stormwater drainage that introduce metals from upland activities.) In urban environments, metals loading to local waters from anthropogenic sources is a major pathway for aquatic habitat degradation.

Current Washington practices do not permit new installation of creosote-treated wood, turning instead to ammoniacal copper zinc arsenate (ACZA) or chromated copper arsenate (CCA Type C) treatements (Poston 2001). These water-soluble treatments are used to protect wood from wood-boring organisms and fungi. They may also pose a threat to water quality through the potential to leach toxic chemicals into surrounding water (Poston 2001).

Metals from treated wood can contaminate sediment and affect benthic communities, which limits food resources for fish and exposes fish to metals contamination through the consumption of contaminated prey (Stratus 2005b). Site-specific sediment conditions such as particle size and organic content can dramatically influence metals toxicity, making sediment toxicity difficult to predict (Stratus 2005b).

The primary metals of concern in the surface waters of Washington are copper, zinc, arsenic, lead, and nickel (Embrey and Moran 2006). Metals above threshold concentrations act as carcinogens, mutagens, and teratogens in fish and invertebrates (Wohl 2004). Additionally, the sublethal effects of copper toxicity have been extensively studied, with reported effects including impaired predator avoidance and homing behavior (Baldwin et al. 2003).

Stratus (2005b) reviewed and evaluated models developed to predict the leaching of metals from treated wood. Stratus (2005b) found the most important factors affecting the leaching rates of metals from treated wood to be:

1. the metal being considered (Cu, Cr, As, or Zn);
2. post-treatment procedures used to fix the treatment chemical and remove excess treatment solution;
3. duration of post treatment exposure of water;
4. loading or retention of treatment solution in the wood;
5. ambient water quality conditions (including salinity, pH, and temperature);
6. current speed; and
7. wood surface physical features (including surface area-to-volume ratio).

The Stratus (2005b) review concluded that the chemical processes associated with the chemical fixing process are complex and poorly understood (Lebow and Tippie 2001). Lebow and Tippie (2001) and Lebow et al. (2004) found that water-repellent stain, latex paint, or oil-based paint greatly reduced arsenic, chromium, and copper leaching rates. Stratus (2005b) compared the applicability of laboratory studies to field conditions and concluded that much higher leaching rates are likely to occur in the field than what is observed in laboratories. A study on the leaching rate of arsenic from CCA Type C-treated lumber under simulated precipitation showed leaching rates of 0.0143, 0.0079, and 0.0062 micrograms per square centimeter per millimeter ($\mu g/cm^2/mm$) of simulated rainfall for the 0.1, 0.33 and 1.0 inch/hour (2.5, 8.0, and 25.4 mm/hour) rainfall rates, respectively (Lebow et al. 2004). This same study also found little reduction in arsenic leaching rates with the application of a water repellent (Lebow et al. 2004). In some cases, leaching rates seemed to increase with water repellent application.

The majority of leaching takes place within a few months of initial immersion. However, leaching may not be the primary pathway for contaminant transfer into local food webs. WDNR guidance provides specific measures to avoid the pulsed release of contaminants from treated materials during their removal from the environment (WDNR 2005d).

Metals are widely known to adversely affect fish species. Increased levels of copper and cadmium have been shown to cause mortality and lower growth rates in bull trout (Hansen, Welsh, Lipton and Cacela 2002; Hansen, Welsh, Lipton and Suedkamp 2002). Some species are more tolerant than others; for example, bull trout are more tolerant of zinc and copper compared to rainbow trout in laboratory studies (Hansen, Welsh, Lipton, Cacela et al. 2002). Dissolved copper, even at low concentrations, is a neurotoxin and damages the sensory capabilities of juvenile salmonids (Hecht et al. 2007). These effects can manifest over a period of minutes or hours and persist for weeks. In addition, copper can affect avoidance behavior; benchmarks developed by NOAA Fisheries showed that a range of 0.18–2.1 parts per billion (ppb) dissolved copper above background levels (for ambient waters below 3 ppb) were a cause for concern (Hecht et al. 2007).

Invertebrates may be exposed to increased metals because many metals adsorb onto sediment particles. Those invertebrates that reside in sediment and filter feed (e.g., California floater, Olympia oyster) are susceptible to increased metal loading and biomagnification in tissues.

In both fresh and salt water, invertebrates are the species most sensitive to copper, chromium VI, zinc, and arsenic (Stratus 2005b). A study by Brooks (2004) on the Olympic Peninsula found insignificant increases in arsenic, copper, and zinc in sediments and water at three out of four pier sampling sites and minimal uptake by shellfish. Weiss et al. (1993), however, found that oysters growing on CCA-treated wood piles had higher metals concentrations and a greater incidence of histopathological lesions compared to oysters collected from nearby rocks. In a subsequent study, Weis and Weis (1996) fed snails algae grown on CCA-treated docks. The snails in turn suffered mortality. Finally, Weis and Weis (1994) found significantly lower biomass and diversity of sessile epifaunal communities on treated wood panels compared to untreated panels. Studies such as these indicate that the primary trophic pathway for contaminants from treated wood is through invertebrates and algae either growing on or attached to treated wood.

The U.S. Environmental Protection Agency (USEPA) has established aquatic life criteria (ALC) (i.e., concentration criteria) for the constituent metals that may leach from ACZA- or CCA Type C-treated wood (USEPA 2002, in Stratus 2005b). The ALC have been established for criterion maximum concentrations (CMCs) for acute exposure and criterion chronic concentrations (CCCs) for chronic exposure for both salt and fresh water (Table WQ-5).

Table 7-10: USEPA water quality criteria for the protection of aquatic life (“aquatic life criteria”) for water soluble chemicals used in treating wood.

Chemical	Freshwater CMC (ppb)	Freshwater CCC (ppb)	Saltwater CMC (ppb)	Saltwater CCC (ppb)
Arsenic	340	150	69	36
Copper[e]	7.0[a]	5.0[a]	4.8	3.1
Copper (2003)	BLM[b]	BLM[b]	3.1	1.9
Chromium III	323	42	None (850)[c]	None (88)[d]

Chemical	Freshwater CMC (ppb)	Freshwater CCC (ppb)	Saltwater CMC (ppb)	Saltwater CCC (ppb)
Chromium VI	16	11	1,100	50
Zinc	65[a]	65[a]	90	81

Source: USEPA 2002, except as noted, as taken from Stratus 2005b.
CMC = criterion maximum concentration.
CCC = criterion chronic concentration.
ppb = parts per billion.

[a] Criteria are hardness-dependent. Criteria values calculated using site-specific hardness based on the equations presented in USEPA (2002). Hardness-dependent criteria values are presented for a hardness of 50 ppm (as CaCO3).

[b] Criteria developed using site-specific chemistry and the Biotic Ligand Model (BLM).

[c] No saltwater CMC. As a proxy, we report the lowest reported LC_{50} from the USEPA database (Lussier et al. 1985) divided by a factor of two. See text for additional details.

[d] No saltwater CCC. As a proxy, we report the lowest reported chronic value from the USEPA database (Lussier et al. 1985) divided by a factor of two. See text for additional details.

From draft aquatic life criteria guidance on copper provided by USEPA in 2003 that relies on the BLM for calculating freshwater criteria based on site-specific water chemistry.

These aquatic life criteria (ALC) appear to be appropriate for acute lethal impacts of copper and chromium VI (Stratus 2005b), but avoidance responses and olfactory neurotoxicity may occur in salmonids at sublethal copper concentrations, even with brief exposure (Hansen et al. 1999a, 1999b; Baldwin et al. 2003; Sandahl et al. 2004; all in Stratus 2005b), and there may be a risk of bioaccumulated toxicity in salmonid prey species at the chronic chromium VI criterion (Stratus 2005b).

There does not appear to be a pattern of sensitivity among species with respect to chromium III, but the ALC, although established only for fresh water, appears to be protective of fish, particularly salmonids (Stratus 2005b). If chromium III toxicity is related to salinity (similar to chromium VI and copper), then the application of the freshwater criteria to salt water would include a margin of safety. The ALC for zinc are water hardness-dependent and do not appear to be protective of salmonids in fresh water of low hardness (30 ppm) (Hansen et al. 2002, in Stratus 2005b); however, the zinc ALC for salt water are likely protective of salmonids (Stratus 2005b).

Avoidance behavior has also been observed among salmonids at zinc concentrations below or slightly above the ALC (Sprague 1964, 1968; Black and Birge 1980, all in Stratus 2005b). The ALC for arsenic are likely to be protective of salmonids (Stratus 2005b). Overall, the ALC are suitable for assessing the impacts of ACZA and CCA Type C-treated wood on water quality and the potential risk to HCP species (Stratus 2005b).

Poston (2001) reviews approximately 20 years of research on treated wood with findings pertinent to metal-treated woods summarized below:

- Metals will not degrade but may mineralize or become physically or chemically sequestered as they are likely incorporated into sediment. However, long-term accumulation of metals at the bases of pilings has not been reported. The risk of sediment resuspension during the removal of pilings is not well understood at this time.

- The sediment content of fines and organic carbon plays a key role in the fate of metals contaminants in the sediment. The function of acid volatile sulfides in the bioavailability of metals contaminants is not understood at this time, but acid volatile sulfides likely also play a role in toxicity. Metals contamination of sediments appears to be localized, while sediment disturbance will likely transport and redistribute metals, possibly diluting the contamination.

- Impacts from CCA Type C- and ACZA-treated wood (primarily the leaching of copper) pose the greatest risk of sediment contamination and direct impacts to organisms that directly colonize treated-wood structures. For immersed structures, the period of greatest risk is the few days to weeks immediately following installation; the period of risk related to stormwater runoff from above-water structures is longer and less predictable.

- In studies that evaluated effects in the environment, no adverse biological impacts were reported from sediment toxicity and no community changes were observed. Filter-feeding oysters exhibiting copper accumulation above background levels showed no biological impacts.

- The sediment characteristics of percent fines and organic carbon play key roles in the fate of metals contaminants in the sediment. The function of acid volatile sulfides in the bioavailability of metals contaminants is not understood at this time, but acid volatile sulfides likely play a role in toxicity. Metals contamination of sediments appears to be localized, while sediment disturbance will likely transport and redistribute metals, possibly diluting the contamination.

7.6.6.1.3 Organic contaminants

Creosote, because it is used to treat wood, is a common source of organic contaminants in marine and riverine systems. Creosote, a distillate of coal tar, includes PAHs (which comprise 85–90 percent of the mass of creosote), alkyl-PAHs, tar acids, phenolics, tarbases/N-heterocyclics (quiolines and carbazoles), S-heterocyclics (thiophenes), O-heterocyclics/furans (dibenzofuran), and aromatic amines (such as aniline). Creosote and other wood preservative products used in in-water structures pose water quality and sediment contamination risks associated with contaminant leaching.

Other petroleum-based contaminants, such as fuel, oil, and some hydraulic fluids also contain PAHs, which could be acutely toxic to salmonids at high levels of exposure and could also cause chronic lethal and sublethal effects on aquatic organisms (Hatch and Burton 1999).

Once contaminants are present in the system, processes that pose risks of contaminant transport include natural and anthropogenic aquatic disturbances such as storms, spills, bioturbation by animals, vessel prop wash, and dredging-related activities. When a stormwater plume eventually mixes and disperses along the seafloor, it results in the deposition of sediments with accumulations of stormwater inputs, including polycyclic aromatic hydrocarbons (PAHs), dichloro-diphenyl-trichloroethane (DDT), and polychlorinated biphenyls (PCBs). The potential result is an alteration to the seafloor biology. Those organisms residing within or upon the substrates that are less mobile, such as mollusks, may receive these accumulated stormwater inputs over long periods of time (Bay et al. 1999).

The greatest impacts of creosote-treated wood are on those benthic and burrowing organisms present on the treated wood structures. Creosote can also directly affect fish and invertebrates species that are associated with treated-wood piles in both marine and freshwater environments. For example, Pacific herring can spawn on creosote-treated wood piles, thereby becoming exposed to a number of chemical pollutants contained in creosote (Vines et al. 2000).

Habitat areas of lower pH and reduced water circulation are at a greater risk of contamination. Metals from creosote-treated wood generally become incorporated into the local sediments and are usually undetectable in ambient waters. An important consideration in the analysis of thresholds for biological effects of PAH is potential additive effects. Additive effects of chemicals could have greater detrimental impacts on species than what has been shown to occur by analyzing the effects of chemicals independent of one another.

The current state of knowledge on the biological effects of creosote-treated wood routes of exposure have been summarized in three major literature reviews: Meador et al. (1995) addressed the bioaccumulation of polycyclic aromatic hydrocarbons (PAHs) in marine fishes and invertebrates; Poston (2001) reviewed treated wood impacts on aquatic environments; and two Stratus documents (2005a, b) presented what is known about the impacts of creosote, CCA, and ACZA treated wood products. The major routes of exposure for marine animals were found to be through the uptake of waterborne chemicals, including the interstitial water of sediments and through trophic transfer; while the direct uptake of sediment-bound chemicals appeared to be negligible (Meador et al. 1995).

PCBs and PAHs impact fish species through multiple pathways. One of these pathways which has been studied in the Pacific Northwest is immunosuppression.

- McCain et al. (1990) reported that juvenile Chinook salmon from the Duwamish estuary are exposed to elevated levels of both PCBs and PAHs.

- Arkoosh et al. (2001) found that the immune response in juvenile Chinook salmon from the Duwamish was lower than that of cohorts from a nonurban estuary and those from the hatcheries that released into those systems.

- In a subsequent study, Jacobson et al. (2003) exposed juvenile Chinook to 20 percent of the LD_{50} for Aroclor 1254 (a common PCB) and to bacterial exposure. They found that fish exposed to both bacterial and contaminant stressors had a greater negative effect on salmon health than either stressor alone.

- In a laboratory study of the effects of PCB levels on juvenile Chinook, (Battelle 2003), no significant effect on growth, or immunocompetence was found.

- In contrast, a field study examining histopathology of liver tissue in English sole in Puget Sound found evidence of higher levels of liver lesions in fish from contaminated areas (Myers et al. 1998).

- Chinook exposed to elevated PCB levels in the Duwamish estuary have shown reduced growth rates (Varanasi et al. 1993).

If Chinook, and presumably other fish species, are in the immediate vicinity of a sediment capping project, the increased PCB levels may contribute to immune and metabolic impacts. The effects of elevated PAH levels on fish has been previously addressed by a number of studies and are presented in Table 7-10, as adapted from Anchor (2006) and Stratus (2005).

Stratus (2005a) reported that a number of jurisdictions have recently put prohibitions in place on the use of creosote-treated wood. However, existing structures made from treated wood could have effects on covered species.

Stratus (2005a) evaluated results from laboratory tests on the leaching of PAHs from creosote-treated pilings.

- Leaching rates in both fresh and salt water increased with higher water temperatures.

- In a study of aging effects on leaching (Ingram 1982), it was found that 12 years of field installation in seawater appeared to have reduced leaching rates by about 25 percent.

- Kang et al. (2003) determined leach rates in fresh water for two flow rates (0.5 and 1.3 in/sec [1.2 cm/sec and 3.3 cm/sec]). The 1.3 in/sec (3.3 cm/sec) flow rate was associated with double the leaching of the 0.5 in/sec (1.2 cm/sec) flow rate.

- PAH leaching rates also seem to increase with temperature, although water circulation appears to have a much greater effect on leaching rates than does water temperature, with the greatest leaching rates occurring in warm, turbulent water (Xiao et al. 2002).

- PAH leaching rates seem to vary with wood species (Cooper 1991; Rao and Kuppusamy 1992), decreasing as wood density increases as found in studies comparing loblolly pine and Douglas fir (Miller 1972, in Cooper 1991).

- PAH leaching rates also increase as treated wood surface area to volume ratios increase (Colley and Burch 196;, Gjovik 1977; Miller 1977; Stasse and Rogers 1965; all in Cooper 1991).

Several models have been developed to estimate PAH leaching rates from creosote-treated wood (Brooks 2004; Poston et al. 1996; Xiao et al. 2002). The models attempt to describe complex interactions and generally rely heavily on site-specific data and assumptions (Stratus 2005a). Evaluations of the CREOSS model (Brooks 2004) and the box plume model (Poston et al. 1996) have shown that although they may not fully explain transient concentrations, such as those immediately following installation or severe disturbance such as abrasion, they are helpful in qualitatively describing the effect of many factors, such as salinity, temperature, wood density, water circulation, surface area to volume ratio, wood grain direction, time from treatment, and whether the wood was treated using BMPs to reduce leaching rate (Stratus 2005b).

Poston (2001) reviewed approximately 20 years of research on treated wood with findings pertinent to creosote-treated wood summarized below:

- Creosote-treated wood poses a much greater risk to water quality from trace metals and polycyclic aromatic hydrocarbons (PAHs) in the immediate surrounding water over a relatively short period of time; toxic lighter-weight PAHs escape the wood, volatilize, and degrade rapidly, while higher-weight PAHs contribute to more chronic contamination as they incorporate into sediment.

- The greatest risk from creosote-treated wood in aquatic applications is to benthic organisms and organisms that directly colonize treated wood structures.

- Temporal and spatial impacts of creosote-treated wood on aquatic environments appear to be much greater than those of ACZA- or CCA-treated wood.

- The vast majority of research discussed in this review investigated the impacts of relatively small applications (<100 pilings) of treated wood. More investigation is needed into the potential impacts of larger projects.

- Impacts of treated wood projects alone may be difficult to assess in settings complicated by other ecological stressors. Therefore, applying the precautionary principle, cumulative impacts that include a proposed treated-wood project should be evaluated against cumulative impacts without the project.

- While the majority of leaching takes place within a few months of immersion, PAHs may continue to diffuse from creosote-treated wood for the life of the product. Diffusion from creosote-treated wood products that have been treated to fix or remove excess preservative may not be as great as previous studies have indicated. PAH releases from wood products may also reach equilibrium with PAH degradation in aerobic sediments over time; however, this may not be true for anaerobic sediments, where PAHs would likely persist for longer periods of time.

- Removal of creosote-treated wood structures may resuspend sediments contaminated with PAHs. Although no data were located regarding this, field data indicate higher degrees of PAH contamination in sediments immediately adjacent to creosote-treated structures. Special care must also be taken when removing creosote-treated material to avoid pulsed release of contaminants to the environment (Poston 2001; WDNR 2005d).

- PAH contamination from both immersed and above-water structures appears to diminish with distance from the structure and, although PAHs are relatively mobile, PAH contamination of sediments is unpredictable in relation to water currents.

- Areas with less water circulation and lower pH are at greater risk for contamination, because leaching is faster and dilution occurs more slowly.

In addition to chemicals diffusing out of treated wood directly into the water, treated wood can weep chemicals, for example PAHs, when the wood is warmed by sunlight (Brooks 2000).

Table 7-11 summarizes several studies on biological effects thresholds for PAHs in surface water (from Stratus 2005a) for both fishes and invertebrates. No research which directly addresses PAH or PCB impacts on HCP invertebrate species was found.

Table 7-11: Effects Thresholds for PAHs in Surface Water

Organism	Exposure Source	Toxicity Endpoint	Concentration in µg/L	Citation
Pacific herring	PAHs leaching from ~ 40-year-old pilings	LC50 for hatching success	50	Vines et al. 2000
Pacific herring	PAHs leaching from ~ 40-year-old pilings	Significant reduction in hatching success and increased abnormalities in surviving larvae	3	Vines et al. 2000
Trout	Commercial creosote added to microcosms	LOEC for immune effects	0.6	Karrow et al. 1999
Mysid, *Mysidopsis bahia*	Elizabeth River, Virginia, sediment extracts	24-hour LC50	180	Padma et al. 1999
Amphipod, *Rhepoxynius abronius*	Eagle Harbor, Washington, sediment extracts	96-hour LC50	100	Swartz et al. 1989
Zooplankton	PAHs leaching from pilings placed in microcosms	NOEC for communities	11.1	Sibley et al. 2004
Zooplankton	Commercial creosote added to microcosms	NOEC for communities	3.7	Sibley et al. 2001
Zooplankton	Commercial creosote added to microcosms	EC50 for abundance	2.9	Sibley et al. 2001

EC_{50} = Exposure concentration of a material that has a defined effect on 50 percent of the test population.
LC_{50} = Lethal concentration of a chemical within a medium that kills 50 percent of a sample population.
LOEC = Lowest observable effects concentration
NOEC = No observable effects concentration
µg/L = micrograms per liter
Source: Stratus 2005a

In addition:

- a literature review by Fuchsman et al. (2006) reported that 50 percent lethal concentrations for Aroclor 1254 (a PCB) ranged from 6.1 ppb for grass shrimp to 20,000 ppb for hydra over a 96 hour exposure period.

- Misitano et al. (1994) exposed larval surf smelt to Puget Sound (Eagle Harbor) sediments with high concentrations of PAHs and found 100 percent mortality after 96 hours of exposure. After diluting the sediments and repeating the experiments, they found that the larvae that did not expire within 96 hours suffered from decreased growth rates.

Many pollutants can be found in contaminated sediment of historically industrial or highly urbanized areas. The number of potential contaminants associated with sediments is vast and highly dependant on site-specific conditions. Different contaminants have different biomagnification potential. Contaminants that can accumulate on sediments include pesticides, PCBs, endocrine disruptors, PAHs, metals, and nutrients (Bednarek 2001). These contaminants may lead to reproductive problems and abnormalities in many of the HCP species.

- In the Colorado River, Feist et al. (2005) showed that plasma androgens and gonad size in male white sturgeon were negatively correlated with total DDT, total pesticides, and PCBs.
- In a study of Columbia River white sturgeon, Burner and Rein (2002) measured the occurrence of physical deformities, which included an additional row of lateral scutes on both sides of the fish and misshapened fins. Although they could not show a clear causal relationship, the authors inferred that these deformities might be the result of organics in the sediments, which are known to be harmful to aquatic organisms.
- Studies in the Pacific Northwest by Stein et al. (1995) and Johnson et al. (2007) have indicated that PCB and PAH concentrations in juvenile Chinook salmon tissue are highest in industrial areas (e.g., the Duwamish estuary, Columbia River).

7.6.6.2 Ecosystem-Specific Effects: Marine and Estuarine

The Washington State Department of Ecology has established water quality standards for marine waters for several metals. These standards, issued in WAC 173-201a, are listed in Table 7-12.

Table 7-12. Water quality criteria for metals in marine waters of the state of Washington.

Constituent	Acute (ppb)	Chronic (ppb)
Arsenic	69	36
Copper	4.8	3.1
Lead	210	8.1
Nickel	74	8.2
Zinc	90	81

Source: WAC 173-201A.

Many studies have investigated thresholds for biological effects of PAH concentrations in marine sediment. Several effects thresholds have been determined using many years of NOAA Fisheries data on the effects of PAH-contaminated sediments on benthic fish in Puget Sound (Stratus 2005a). Thresholds for effect on English sole were determined at 230 ppb for proliferated liver lesions; 630 ppb for spawning inhibition, infertile eggs, and abnormal larvae; and 288 ppb for DNA damage, measured as PAH-DNA adducts (Johnson et al. 2002).

Weis et al. (1998, in Stratus 2005b) measured metals concentrations in sediments and marine polychaete worms and diversity, abundance, and biomass in the benthic invertebrate community near five CCA-treated wood bulkheads ranging from one to eight years in age. It was found that concentrations of copper and arsenic in sediments were generally elevated within 3.3 feet (1 m) but diminished to background levels by 9.8 feet (3 m) from the bulkheads. Polychaete worms collected within 3.3 feet (1 m) of a one-year-old treated wood structure contained elevated copper and arsenic concentrations, and benthic community effects on abundance and diversity were noted at all treated wood sites, diminishing with distance from the bulkheads. Effects were negligible at distances greater than 3.3 feet (1 m) from bulkheads.

Diffusible creosote-derived compounds from weathered creosote-treated pilings have been shown to affect the embryonic development in Pacific herring (Vines et al. 2000). Pacific herring have been shown to have reduced hatching success at PAH concentrations as little as 3 ppb, while 50 percent of the eggs in the same study were viable at concentrations of 100 ppb (Vines et al. 2000). If adult salmon feed on herring that have been exposed to creosote-derived compounds, it is feasible that these components could then affect salmon through food web interactions.

7.6.6.3 Ecosystem-Specific Effects: Riverine and Lacustrine

The Washington State Department of Ecology has established water quality standards for fresh waters for several metals. These standards, issued in WAC 173-201a, are listed in Table 7-13. Freshwater toxicity thresholds are hardness dependent and can vary widely depending on alkalinity. The standards presented here are based on median freshwater hardness concentrations estimated from an extensive 3-year data set (2001–2003) from the Green River watershed (Herrera 2007b).

Table 7-13. Water quality criteria for fresh waters of the state of Washington based on median hardness values.

Constituent	Acute (ppb)	Chronic (ppb)
Arsenic	360[a]	190[b]
Copper	7.0[a]	7.5[b]
Lead	22.9[a]	1.5[b]
Nickel	640[a]	104[b]
Zinc	51.6[a]	69.2[b]

[a] Criterion varies with hardness. Acute criterion is based on an estimated median storm flow hardness of 39.1 ppm as $CaCO_3$.

[b] Criterion varies with hardness. Chronic criterion is based on an estimated median base flow hardness of 61.5 ppm as $CaCO_3$.

Tables 7-14 and 7-15 present some of the threshold effects concentrations (TECs) and probable effects concentrations (PECs) for arsenic, chromium, copper, and zinc in sediment as reported in recent literature (Stratus 2005b) in fresh water. In general, concentrations below the TEC are not expected to cause impacts, while concentrations above the PEC are expected to cause frequent impacts.

Table 7-14. Threshold Effects Concentrations (TECs) for Freshwater Sediment

Name	Definition	Concentration (mg/kg dry wt)					Reference
		Basis	As	Cr	Cu	Zn	
Lowest effects level	Level that can be tolerated by the majority of benthic organisms	Field data on benthic communities	6	26	16	120	Persaud et al. 1991
Biological threshold effects level	Concentration that is rarely associated with adverse biological effects	Compiled results of modeling, laboratory, and field studies on aquatic invertebrates and fish	5.9	37.3	35.7	123	Smith et al. 1996
Minimal effects threshold	Concentration at which minimal effects are observed on benthic organisms	Field data on benthic communities	7	55	28	150	Environment Canada 1992
Effects range low[a]	Concentration below which adverse effects would rarely be observed	Field data on benthic communities and spiked laboratory toxicity test data	33	80	70	120	Long and Morgan 1991
Survival and growth threshold effects level	Concentration below which adverse effects on survival or growth are expected to occur only rarely	Laboratory toxicity tests on the amphipod Hyalella azteca using field-collected sediment	11	36	28	98	Ingersoll et al. 1996; USEPA 1996
Consensus threshold effects concentration	Concentration below which adverse effects are expected to occur only rarely	Geometric mean of above published effect concentrations	9.79	43.4	31.6	121	MacDonald et al. 2000a

Based on data from both freshwater and marine sites.
Source: Taken from Stratus 2005b
mg/kg = milligrams per kilogram
As = arsenic; Cr = chromium; Cu = copper; Zn = zinc

Table 7-15. Probable Effects Concentrations (PECs) for Freshwater Sediment

Name	Definition	Basis	Concentration (mg/kg dry wt)				Reference
			As	Cr	Cu	Zn	
Severe effects level	Level at which pronounced disturbance of the sediment-dwelling community can be expected	Field data on benthic communities	33	110	110	820	Persaud et al. 1991
Probable effects level	Concentration that is frequently associated with adverse effects	Compiled results of modeling, laboratory, and field studies on aquatic invertebrates and fish	17	90	197	315	Smith et al. 1996
Toxic effects threshold	Critical concentration above which major damage is done to benthic organisms	Field data on benthic communities	17	100	86	540	Environment Canada 1992
Effects range median[a]	Concentration above which effects were frequently or always observed or predicted among most species	Field data on benthic communities and spiked laboratory toxicity test data	85	145	390	270	Long and Morgan 1991
Probable effects level	Concentration above which adverse effects on survival or growth are expected to occur frequently	Laboratory toxicity tests on the amphipod Hyalella azteca using field-collected sediment	48	120	100	540	Ingersoll et al.1996; USEPA 1996
Consensus probable effects concentration	Concentration above which harmful effects on sediment-dwelling organisms are expected to occur frequently	Geometric mean of above published effects concentrations	33.0	111	149	459	MacDonald et al. 2000a

Based on data from both freshwater and marine sites
Source: Taken from Stratus 2005b
mg/kg = milligrams per kilogram
As = arsenic; Cr = chromium; Cu = copper; Zn = zinc

A report from the Jimmycomelately Piling Removal Monitoring Project (Gardiner 2006), indicated a strong correlation between creosote piles and associated concentrations of PAHs. This study provides evidence that PAHs can accumulate in sediments at levels likely to affect fish eggs and/or larvae.

Poston (2001) concluded that the risk of potential impacts to salmonids from direct exposure to PAHs or metals leached from treated wood is low. Riverine spawning substrates for salmonids do not typically facilitate the accumulation of PAHs or metals, and juvenile salmonids are not likely to encounter high concentrations of such contamination in larger waterways when they begin their open-water, marine lifestage. However, salmonids are potentially at some risk of exposure from consumption of contaminated prey.

Treated wood is not often used in riverine environments, particularly in new construction, but it can often be found in older structures and can be re-exposed during maintenance operations.

7.6.6.4 Ecosystem-Specific Effects: Lacustrine

In general, impacts of water quality modifications in lacustrine systems may be expected to be greater than those in either marine or riverine systems, because circulation in a lake is generally much more limited than in a river, estuary, or marine area. One recognition of this is the existing limitation in the Hydraulic Project Approval permits that states that the use of wood treated with creosote or pentachlorophenol is not allowed in lakes (WAC 220-110-060 (4), 220-110-170 (6), 220-110-223 (6), and 220-110-224 (2).

7.6.7 Activity-Specific Effects

7.6.7.1 Bank Protection

Bank protection structures at the mouths of rivers and streams entering the marine environment can contribute to the alteration of a natural salinity gradient. This alteration could occur through the shortening of a river through the lower reaches in which tidal water extends into the river or stream. Dredging activities that may accompany bank protection measures can exacerbate this impact.

Characteristic of many Puget Sound beaches is a continuous corridor of reduced salinity. The Puget Sound Nearshore Ecosystem Recovery Program (PSNERP) conceptual model and the regional nearshore Chinook recovery chapter (extension from Fresh and Averill (2005) suggest that bulkheading along marine shorelines can also disrupt the natural flow of freshwater from bluffs into beach seeps thereby fragmenting this corridor.

Construction of bank protection could disturb fine sediment on banks that could lead to increased suspended solids, as could alterations in sediment supply associated with the ongoing existence of bank protection structures.

Bank protection can alter nutrient and pollutant loading indirectly by allowing modifications to adjacent uplands that may result in more nutrients being introduced into marine waters. In particular, fertilizer and pesticide runoff from lawns adjacent to bank protecton have the potential to increase nutrient and pollutant loading.

7.6.7.2 Groins

Water temperature is strongly dependent on mixing in rivers and streams (Fischer et al. 1979). Placement of groins affects these mixing processes, often reducing mixing and increasing thermal stratification. Stratified waters can lead to elevated surface temperatures, particularly during the summer months (Fischer et al. 1979).

7.6.7.3 Jetties

Jetties have been documented to reduce the tidal prism and increase stratification on the Columbia River (Sherwood et al. 1990). This would lead to a reduction of mixing in the estuary, which causes artificial increases in the temperature of surface waters (Fischer et al. 1979). Jetties have the potential to isolate and concentrate biochemical oxygen demand (BOD) (Fischer et al. 1979), resulting in lower dissolved oxygen in the water column. This can occur during the summer, at the time that fishes are most sensitive to temperature stress. Subsequent alterations in nutrient loading can have a profound effect on nearshore productivity and diversity (Roegner et al. 2002).

7.6.7.4 Overwater Structures

Some overwater structures are supported by wood piles. Wood piles are also sometimes used to construct temporary trestles that support equipment during construction activities. Wood piles that have been chemically treated to resist rot and are in contact with water have the potential to leach chemical contaminants into the surrounding water (Poston 2001). In addition to this possible direct impact, indirect pathways of contamination also exist; for instance, stormwater runoff from surfaces elevated above the water body or splinters of treated material that are dislodged by activity above the water line and fall into the water body (Poston 2001). As piles, decking, and other supporting structures degrade or are abraded over time with operation of overwater structures, contaminants are released into the water.

7.6.7.5 Marinas and Terminals

Marinas and terminals are known to affect water quality parameters including temperature (by limiting circulation), dissolved oxygen, suspended sediments and turbidity, nutrients, and pollutants.

Marinas and terminals may introduce contaminants through many pathways:

- the use of treated wood products, including creosote-derived contaminants from treated wood (e.g., Poston 2001 and WDNR 2005d),
- vessel waste and ballast water discharges,
- vessel maintenance and operations-related oil and fuel spills,
- structural impacts on natural shoreline geomorphology and vegetation, including shoreline hardening,
- stormwater pollution,
- construction disturbance.

Shading can result in reduced primary production beneath docks and correspondingly lower dissolved oxygen levels. Marinas and terminals, through the discharge of wastes or disturbance to bottom sediments from large or multiple vessels, can increase carbon, nutrient, and sediment loading in their zone of influence, thereby affecting local dissolved oxygen levels. Depressed dissolved oxygen from reduced primary production combined with potential carbon loading from vessel and nearshore waste sources can lead to low benthic dissolved oxygen levels (McAllister et al. 1996) and high biochemical oxygen demand. Marinas often have breakwaters that reduce circulation, which can further concentrate effects on water quality.

It has also been hypothesized that resuspension of large quantities of anoxic sediments, as can occur with dredging operations associated with terminals and marinas, may reduce dissolved oxygen levels in surrounding water as a result of oxidation reactions (Nightingale and Simenstad 2001a). However, even with the potentially large amounts of resuspended, deep-water, anoxic sediments associated with dredging, little evidence supports the notion that associated dissolved oxygen reduction in surrounding water poses a risk to fish moving through the area (Nightingale and Simenstad 2001a).

There are many sources of suspended sediment associated with marinas and terminals, including disturbance of sediment during construction or operation, maintenance dredging, alterations to sediment supply caused by the presence of marina and terminal structures, and vessel prop wash.

Contaminated sediments are an issue with marinas and terminals when they are located near contaminated sites. Dredging and vessel activity can contribute to the resuspension of benthic materials, thereby increasing the availability of contaminated sediments for biotic assimilation. Potential toxic substances from resuspension and contaminated sediments include PAHs and PCBs (Bay et al. 1999). Contaminated sediments are of particular concern due to the risk of contaminant transport, and exposure posed to aquatic organisms through bioaccumulation and biomagnification in the marine food web. These risks can also be passed on to humans through the consumption of seafood.

Michelsen et al. (1999) found that prop wash from large vessels, such as ferries, have the capacity to resuspended and transport contaminants along the Seattle urban waterfront, which can increase the risk of exposure to various species.

Cardwell et al. (1980) and Crecelius et al. (1989a, 1989b) have documented water quality characteristics in marinas in the Puget Sound region. Numerous studies have identified contaminant loadings and biological effects on fish and other organisms in Puget Sound waterways (Arkoosh et al. 1991, 1994, 1998, 2001, 2004; Johnson and Landahl 1994, 1995; Johnson et al. 1993, 2007; Jones 1996; Loge et al. 2005; Myers et al. 2003; Sandhal et al. 2007; Stehr et al. 2000; Varanasi et al. 1992, 1993; Williams et al. 1998; Whyte et al. 2000). Studies demonstrate that contaminants introduced to the aquatic environment and ingested by aquatic organisms are incorporated into the food web and can ultimately interfere with animal reproductive viability and population sustainability (Johnson et al. 1993, 1995; Johnson and Landahl 1994; Jones 1996; Lee 1985; O’Neill et al. 1995; West 1995, 1997).

Large vessels (i.e., more than 82 ft [25 m] in length) are allowed to use tributyltin bottom paint, which is highly toxic to aquatic organisms. Studies have shown that tributyltin can biomagnify through algae, invertebrate, and vertebrate species (Mamelona and Pelletier 2003).

Nutrient and contaminant loading from vessel discharges, engine operation, prop scouring, bottom paint sloughing, boat wash-downs, haul-outs, boat scraping, painting, and maintenance activities pose risks such as sediment contamination and water quality degradation (Cardwell et al. 1980; Cardwell and Koons 1981; Eisler 1998; Hall and Anderson 1999; Krone et al. 1989a, 1898b; Waite et al. 1991.)

Additional potential sources of toxic contaminants in marinas and terminals include hydrocarbons (such as PAHs) from structures and pilings of creosote treated wood, leaking engines, spills, vessel maintenance, and operational discharges. ACZA or CCA treated wood may also introduce metals into the environment in the vicinity of marinas and terminals.

7.6.7.6 Vessel Traffic

Recreational boating can significantly increase turbidity (Hilton and Phillips 1982; Warrington 1999; Yousef et al. 1980; Yousef 1974). Some models predict a 44 percent increase in riverine turbidity due to recreational boating (Hilton and Phillips 1982). Prop wash or waves produced by boats and personal watercraft are also known to increase suspended sediments and turbidity through resuspension of shallow water sediments (Kennish 2002; Yousef et al. 1980; Yousef 1974). In some freshwater environments, such as lakes, turbidity can decline slowly, taking as much as 24 hours for water clarity to return to baseline levels (Yousef 1974; Yousef et al. 1980).

Vessel traffic can disturb and suspend sediment in the water column as a result of water currents moving under and around the vessel, pressure fluctuations as the vessel displaces water during movement, propeller wash, and waves generated by the bow and stern of a vessel that wash up on the bank (McAnally et al. 2004). Vessel traffic has been correlated with an increase in turbidity of up to 50 percent in shallow waters (average depth 10 feet [2.9 m]) (Anthony and Downing 2003). Correlations of vessel traffic with turbidity patterns and sediment particle settling velocities suggest that vessel traffic may increase turbidity levels on a daily as well as seasonal temporal scale (Garrad and Hey 1988). Recreational vessel traffic has been observed to induce levee erosion at rates of 0.0004 to 0.009 inch (0.01 to 0.22 mm) per boat pass (Bauer et al. 2002). Water depth appears to have less influence on vessel-induced turbidity than does vessel speed (Hill and Beachler 2002). Field measurements have shown that at very low speeds and very high speeds, planing hull vessels have little effect on turbidity, even in shallow water, but at transitional speeds, significant sediment resuspension can occur, even in relatively deep water (Hill and Beachler 2002).

The effect of this increased turbidity may lead to decreased light levels, which could potentially affect the growth rates of submerged vegetation, upon which most HCP species depend (at least during their juvenile stages). Turbidity is also known to be associated with fish respiratory injury (Berg and Northcote 1985). Another risk from increased turbidity is the potential of increasing fine sediment deposition to downstream spawning beds, resulting in a loss of suitable spawning habitat (Hartman et al. 1996) and reduced disease tolerance (Redding et al. 1987).

Grounding, anchoring, and/or prop wash can cause benthic disturbance and turbidity, eelgrass and macroalgae disturbance, and freshwater aquatic vegetation disturbance. These effects have been well documented (e.g., Thom et al. 1997 and Thom and Shreffler 1996).

Prop wash and waves are also known to be a primary cause of shoreline erosion (Gatto and Doe 1987; Mason et al. 1993). The number of boats in a given area has been correlated with wave height (Bhowmilk et al. 1991), with areas of high boat traffic exhibiting increased levels of shoreline erosion. Although it is difficult to quantify boat wake contributions to shoreline erosion, boat traffic has been found to contribute up to 50 percent of the factors responsible for shoreline erosion in small rivers less than 2,000 feet wide (Hurst and Brebner 1969).

Sutherland and Ogle (1975) found prop wash and increased turbidity from jet boats to decrease salmon egg survival by 40 percent. In addition to turbidity, direct contact with spawning substrate can cause mortality.

Boat prop wash has also been found to stimulate the resuspension of nutrients and contaminants that can stimulate algal blooms, such as nitrogen and phosphorous, as well as increased turbidity (Haas et al. 2002; Michelsen et al. 1999; Thom et al. 1997; Thom and Shreffler 1996; Parametrix 1996). The result of this sediment disturbance and increased turbidity is to decrease available light for aquatic plants. The resulting increased algal growth may also lead to eutrophication and reduction of dissolved oxygen levels due to respiration during desiccation of the algal material. The result of increased turbidity and lower dissolved oxygen has effects throughout the food web

7.6.7.7 Culverts and Bridges

In general, the most extensive water quality effects of culverts are expected to be associated with the initial construction-related effects of culvert removal and replacement, because earthwork, in-channel work, and materials placement is most extensive during construction. Culvert retrofits have less extensive initial construction-related impacts, but water quality modifications are expected to occur on a more frequent basis because the maintenance requirements for retrofitted structures are more extensive.

Virtually every culvert project will result in some release of suspended sediments. In-water construction projects involving mechanized equipment pose some risk of release of toxic substances. In contrast, other stressors such as altered dissolved oxygen (DO), altered pH, and altered nutrient cycling are only expected to occur in specific circumstances. Altered pH is likely to occur only in culvert replacement or retrofitting projects involving concrete poured and cured on site, in contact with water. Culvert removals are unlikely to cause this effect because they typically will not involve pouring of new concrete.

Culvert removal or replacement may lead to altered nutrient cycling and changes in DO levels, because these methods may in certain circumstances involve the dewatering of distinct road-impounded wetlands. Distinct road-impounded wetlands are created by road beds and culverts that interfere with the hydraulic and geomorphic continuity of the stream system and create a new habitat type (Barnard 2002). This type of habitat feature can sequester a large amount of sediments and organic material. Removal or replacement projects that release this sequestered material can cause nutrient and DO effects in downstream reaches. Culvert retrofits are unlikely to cause these stressors because the configuration and conveyance characteristics of the structure remain similar. Even where distinct road-impounded wetlands are present, retrofits are unlikely to cause the extensive hydraulic and geomorphic effects associated with altered nutrient cycling and DO conditions.

Stormwater generated by above-water portions of water-crossing structures such as culverts and bridges may adversely impact potentially covered species by introducing pollution to waterways. Bridges and culverts provide a surface on which pollutants can accumulate, and those pollutants can become mobile with stormwater runoff. In addition, water crossings may also be associated with a variety of adjacent land uses, including roads and parking lots, and may deliver stormwater from those adjacent land uses to waterways. These stormwater impacts are mitigated by regulations promulgated by Ecology under the federal Clean Water Act (33 United States Code [USC] §§ 1251-1387). The Ecology regulations are subject to USEPA review and Section 7 requirements of the ESA (16 USC 1531-1544). Generally, the federal Services have found that full compliance with applicable Ecology and Washington State Department of Transportation (WSDOT) stormwater treatment guidance is sufficient to support a determination that stormwater generated from a project will not result in incidental take of listed species. However, there are few data on the stormwater vulnerability of potentially covered species other than salmonids.

In the context of culverts and other water crossings, decreased DO levels are likely to occur only in specific and limited circumstances where the activity imposes additional nutrient loading on a system that is eutrophic or close to a eutrophic condition. For example, a culvert replacement that results in dewatering of an upstream impoundment may result in the rapid export of sequestered nutrients. In eutrophic conditions, this may lead to decreased DO levels. Restoration of fish passage may have the unintended consequence of depleting DO levels if large numbers of spawning salmon are allowed to access a system where eutrophic conditions are already occurring. Under most circumstances, these effects will be short term in duration.

For some projects, NMFS concluded that increased turbidity from water crossing projects could have adverse impacts to salmonids due to pulses of increased turbidity from in-water work during construction, as well residual post-construction turbidity pulses generated as the area restabilizes. These effects were evaluated as posing potential threats to juvenile salmonids (NMFS 2006k). NMFS found that elevated turbidity can cause direct mortality (NMFS 2006g), while sublethal threats include harassment, as feeding patterns may be affected and fish are likely to avoid areas of increased turbidity (NMFS 2006d).

7.6.7.8 Conduits

The most likely impact of conduits on water quality is an increase in suspended sediments during installation. Wet, open-trench installation of conduits (consisting of trenching, placing conduit, and backfilling in an inundated water body) typically produces the greatest sediment loads of any conduit installation method, with peak turbidity occurring during trench excavation and backfilling and a rapid return to background levels after backfilling and trench stabilization (Reid and Anderson 1998; Reid et al. 2004). Sediment disturbance can be further increased by instream operation of equipment or storage of excavated material within the floodplain (Reid et al. 2004). Wet, open trenching minimizes disturbance time, because the work can be performed quickly while not blocking fish passage (Reid et al. 2004), and isolating the trenching site by diverting flow around it using a temporary dam and pumps or a flume is effective at limiting turbidity increases and sediment impacts to downstream habitat (Reid et al. 2004).

Conduits are sometimes intalled by boring under water bodies. Attempts to bore beneath a water body using high-pressure directional drilling (HPDD) present a risk of sediment disturbance through hydraulic fracturing ("frac-outs") and the release of drilling muds to the surface. NMFS has concluded that, with the implementation of appropriate BMPs, the accidental release of drilling fluid during HPDD is unlikely to occur and so is not likely to adversely impact aquatic species or their habitats (NMFS 2006c).

7.6.7.9 Dams (Including Intakes and Diversions)

Dams alter the thermal regime of a river both by impounding water and through the release of water from the upstream reservoir. Because water above the dam is relatively stagnant compared to the flowing reach downstream, water in an impoundment will typically absorb heat and become stratified. Depending on where the water is released from the reservoir, it will either increase (upper water column reservoir releases) or decrease (lower water column reservoir releases) stream temperatures downstream. Flow over dams can result in supersaturated oxygen conditions.

Vaughn and Taylor (1999) reviewed several studies of freshwater mussel populations in river systems affected by dams and found multiple instances of decreased population persistence and abundance in reaches downstream of dams. They postulated that because abundance increased as temperature conditions moderated in downstream reaches, sensitivity to altered temperature regime was a primary factor controlling distribution.

Sediments accumulate behind dams as a result of lowered stream velocities, thereby allowing sediments to settle and deposit in the reservoir. Sediments trapped behind dams are usually fine, and small particles tend to adsorb contaminants (Murakami and Takeishi 1977). In areas where contamination from organics, pesticides, and metals occurs, these will adsorb to sediments and accumulate behind dams.

The release of contaminated sediments may occur during maintenance activities or during dam removal. Dredging and construction equipment activity can contribute to sediment resuspension, and increase the availability of contaminated sediments for biotic assimilation. Dams also can result release of toxic substances through accidental fuel and chemical spills. Another source of toxic chemicals is accidental spills from increased recreational vessel use encouraged by the creation of the impoundment upstream of a dam. The introduction of toxic substances from recreational uses induced by dam development is potentially important because these small chronic sources occur at a greater frequency than for infrequent construction and maintenance activities. Contaminated sediments may be transported and aquatic organisms' exposure increased through bioaccumulation and biomagnification in food webs.

When considering a dam removal, it is important to assess the condition of accumulated sediments before deconstruction in order to minimize the release of both contaminants and suspended sediments. For example, during an accidental dam breach in New York, contaminated sediments caused an increase in the levels of polychlorinated biphenyls (PCBs) downstream (Shuman 1995).

Dams can cause increased nutrient loading when the bottom layer of upstream reservoirs is released, because this water is often high in nutrients (Camargo et al. 2005; Palmer and Okeeffe 1990; Teodoru and Wehrli 2005).

7.6.7.10 Roughened Channels

Roughening of an existing channel by definition requires in-water work, likely affecting an extensive length of channel. This suggests the need for in-water equipment use and materials placement, dredging, and/or dewatering of the work area. In contrast, creation of an entirely new channel avoids the majority of these effects because the extent of in-water work is limited to the disturbance necessary to connect the new channel to existing flow at the upstream and downstream ends.

7.6.7.11 Weirs

Potential water quality modifications associated with temporary weirs are expected to vary, ranging from limited pulses of suspended sediments during installation and removal, to more extensive short-term effects during the construction of movable structures (e.g., from equipment operation and materials placement, curing of concrete). Following construction and installation, however, water quality effects of operations are expected to be negligible. Permanent barrier weirs would have more extensive impacts on water quality, the same as those impacts caused by dams.

In general, weirs have the potential to alter temperature, dissolved oxygen, suspended sediments (turbidity), pH levels, and nutrient and pollutant loading. Metal toxicity and altered salinity are not common in the presence of weirs.

Similar to a dam, water is slowed behind a weir, and pooled water increases in temperature and flows downstream. Temperature changes from a weir are likely smaller than for a dam because weirs generally have smaller impoundments and are often run-of-the-river structures. Similar to dams, flow over weirs can result in supersaturated dissolved oxygen concentrations due to high velocities (Baylar and Bagatur 2000).

During construction and maintenance activities, suspended sediments can increase, and pH can be altered from the use of concrete.

7.6.7.12 Dredging

Increased turbidity is the principal water quality modification associated with dredging (Figure 7-3).

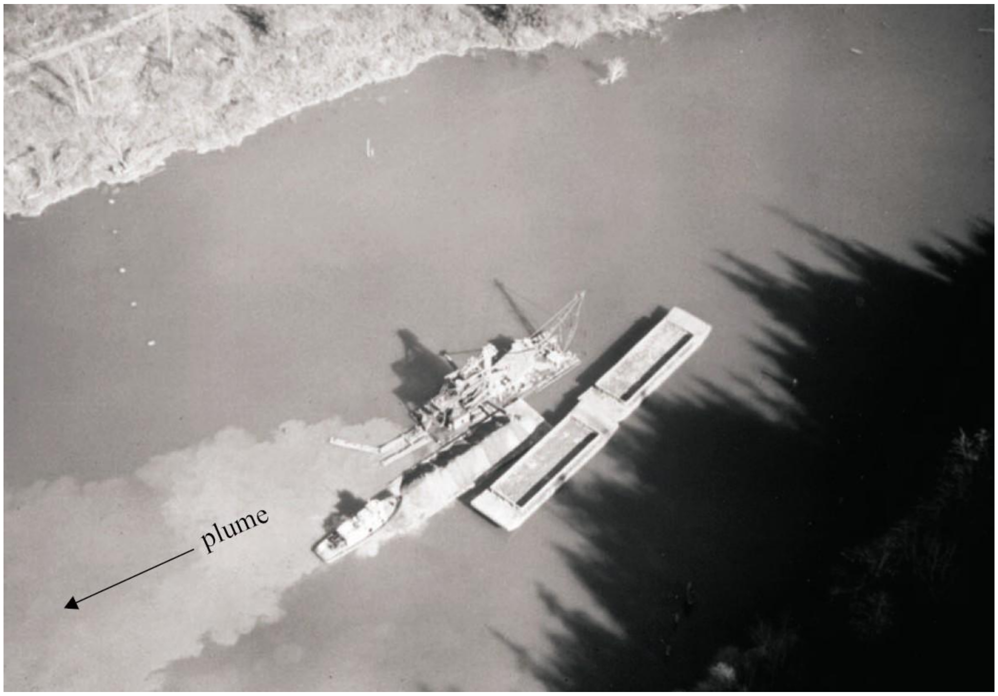

Figure 7-3. Turbidity plume associated with dredging. (Source Unattributed)

Dredging in marine and freshwater environments exposes fine sediments, and resuspends sediments which were intended to be removed but were “lost”. Disposal of dredged materials in water also results in resuspended sediments (USACE 1983). Inefficiencies in dredging can produce high levels of turbidity at the project site and also at some distance away from the project (Erftemeijer and Lewis 2006; Hossain et al. 2004; Perillo et al. 2005; USACE 1983, 2005). Dredging has the potential to release toxic compounds to the water column (Spadaro et al. 1993)

Hydraulic and geomorphic modifications associated with dredging can combine to increase stratification and reduce vertical mixing (Fischer et al. 1979).

In marine environments, dredging activities have been shown to alter tidal prisms and estuarine mixing (Hossain et al. 2004). These changes would alter the thermal profile of the estuary (Fischer et al. 1979) and could affect fish and invertebrates.

Dredging activities have been shown to alter the underlying physical processes (Perillo et al. 2005). It has also been linked directly to lowering dissolved oxygen levels caused by increased sedimentation (Hossain et al. 2004). There are numerous incidences of dredging affecting tidal circulation and estuarine stratification (Hossain et al. 2004; Perillo et al. 2005; Sherwood et al. 1990).

7.6.7.13 Gravel Mining

Gravel mining can alter water quality as a consequence of modifications to groundwater input, hyporheic flow, and through the surface and groundwater exchanges between the river and gravel pits. In-channel gravel mining activities such as bar scalping increase the suspension of fine sediment and fine organic material (Weigand 1991). The resulting changes in water temperature and turbidity can affect dissolved oxygen.

Instream gravel mining creates wide and shallow areas. The removal of riparian and bar vegetation during gravel mining activities reduces shading and thereby increases water temperatures, particularly on smaller rivers (Kondolf et al. 2002; Norman et al. 1998). Channel incision and floodplain disconnection can reduce the temperature-moderating effects of hyporheic zone interactions (Stanford and Ward 1993). Gravel pits convert formerly lotic (flowing) habitats into lentic (stillwater) habitats. Surface water temperatures in pits may rise during the spring and summer due to the longer periods of sunlight and the stagnant water (Norman et al. 1998). Off-channel pits that heat up in the summer provide habitat for warm-water fish that prey on juvenile salmonids. The warm, lentic waters from gravel pits following pit capture[2] may lower water quality by increasing the downstream water temperature of the receiving waters. In general, gravel mining activities that increase water temperature and suspended solids will tend to decrease the dissolved oxygen content. For example, captured gravel pits in the Naugatuck River, Connecticut function as lakes throughout most of the year with seasonally stagnant water and depressed dissolved oxygen levels (MacDonald 1988).

7.6.7.14 Sediment Capping

The degree to which water quality is impacted from a sediment capping project is dependent upon the scale and type of project. If a confined aquatic disposal (CAD) cell is used, bathymetry may be altered, with possible water quality impacts associated with changes in circulation. For both disposal in CADs and for in-situ capping, the primary water quality impact will be associated with increased suspended solids during construction of the cap. Water quality may be affected by the entrainment of contaminated sediments during the placement of the cap.

Monitoring of suspended solids concentrations during sediment cap emplacement has been conducted in only a few studies. In general, the suspended solids concentrations which occur during sediment capping would be lower than concentrations which occur during dredging. The sediment used in capping is usually a coarse sand, while dredged sediments can be of a finer grain and thus more easily entrained (Lyons et al. 2006). Hamblin et al. (2000) monitored benthic suspended solids concentration with an acoustic Doppler profiler in Hamilton Harbor, Lake Ontario. They found that maximum

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[2] National Marine Fisheries Service (NMFS) National Gravel Extraction Guidance (05-06-27) defines “pit capture” as “active channel migration into floodplain (gravel) pits.”

suspended solids concentrations reached 140 ppm shortly after each pass of the sediment barge. They also found that these concentrations quickly returned to background levels (10 ppm) until the next pass of the barge. This indicates that when coarse sand is used as a capping material, rapid particle settling prevents long duration increases in suspended solids. This same pattern of short-term increase in suspended solids was noted by Fredette et al. (2002) on the Palos Verdes Shelf, California. Consequently, it does not appear that elevated suspended solids during capping is an important impact mechanism.

There is the potential for the entrainment of contaminated sediments during cap placement. Capped sediments are most frequently contaminated with polychlorinated biphenyls **(**PCBs), polycyclic aromatic hydrocarbons (PAHs) and metals (Hull et al. 1999). Entrainment of these pollutants is a function of the capping technique with "pump down" techniques generating little disturbance and "point dump" techniques creating more (Palermo et al. 1998). To study how much contaminated sediment was entrained during point dump capping, Lyons et al. (2006) collected water samples during the Boston Harbor and Eagle Harbor capping projects. The sampling indicated that total PCB and total PAH concentrations spiked to levels as high as 84 ppb and 5.2 ppm, respectively. These pulses exceeded acute water quality standards (WAC 173-201A), but were of a relatively short duration. This, one of the only studies to monitor water quality during cap emplacement, indicates that techniques which minimize benthic disturbance should be used whenever possible. There are no available studies that have monitored ambient metals concentrations during placement of a cap.

7.6.7.15 Channel Creation and Alignment

Channel creation and alignment can alter water quality as a consequence of adjustments to a modified channel geometry, which can release fine sediment into the water. If those sediments are contaminated, that can also affect water quality. Channel creation and alignment activities can also alter water temperature and nutrient loading through changes in groundwater input, hyporheic flow, altered shading from riparian vegetation, and through the loss of instream woody debris. These changes in water quality can also affect the concentration of dissolved oxygen.

7.6.7.16 Fish Screens

Fish screens and related installation requirements vary considerably in scale. The water quality impacts are generally expected to be temporary perturbations associated with construction and maintenance activities, such as construction-related suspended sediments, or accidental releases of toxic substances from construction-related spills or equipment failures. The intensity of suspended sediment stressors will vary with the size and placement of the fish screens.

In-channel fish screens, because they involve in-water construction, have greater potential for specific water quality related stressors than off-channel fish screens. Constructing large, permanent in-channel screens potentially includes the placement of cofferdams to create in-water exclusion areas. The bed and bank disturbance associated with in-water construction produce far more suspended sediments than placement of a temporary pump intake and in-channel screen assembly associated with a temporary water withdrawal.

Off-channel screens typically require construction and maintenance of bypass channels and outfall structures within artificial channels. Off-channel screens are commonly constructed in the dry (Bates 2008; Schille 2008), presenting less potential for suspended sediment impacts during construction. Connection and watering of off-channel screen bypass channels and/or placement of erosion protection around outlets may present some potential for sediment impacts. In contrast, the performance of off-channel screens is dependent on effective flow control and debris clearance. If a screen clogs with debris and is overtopped, or high flows overwhelm channel and screen capacity, the entire system could fail, leading to extensive upland and bank erosion with direct delivery of elevated suspended sediments to the stream channel.

For both in-channel and off-channel fish screens, elevated suspended sediments and turbidity can occur as a result of construction, maintenance, and operation. For operations and maintenance, an effective screen necessitates a low water velocity, which causes deposition of material that might otherwise remain suspended or moving as bedload (Bates 2008). Once the material is deposited, it has to be dealt with as a maintenance issue. It might be dredged from the flowing water (in-channel screen) or from a dry canal (off-channel screen). It might also be sluiced downstream in either design. Material might accumulate during a high flow, but then be sluiced during a normal maintenance operation during a period of lower flow. This presents the potential to produce elevated suspended sediment levels during flow periods when the transport capacity is low, meaning that the effects are occurring during periods when suspended sediment levels are low under natural conditions.

7.6.7.17 Outfalls

The most significant impacts on HCP species from outfalls water quality modifications as a result of the presence of pollutants in the discharged effluent. Urban runoff, wastewater treatment plant effluent, and combined sewer overflows are the source of nutrients, sediment, metals, PAHs, and pesticides, all of which can change the chemistry and temperature of the receiving waters (Barber et al. 2006; Grapentine et al. 2004; Mulliss et al. 1997; Wenning et al. 1999). Examples of water quality impacts from outfalls include:

- Outfalls from fish hatcheries can increase suspended sediments in the receiving waters (Fries and Bowles 2002).

- Increased temperatures from a power plant outfall can affect migration patterns of stingrays at Seal Beach (California) (Vaudo and Lowe 2006).

- Nutrients in sewage outfalls can contribute to increases in the productivity of the receiving water (deBruyn et al. 2003).

- In Canada, hormonally active chemicals have been shown to accumulate in the local white sucker, disrupting reproductive activities in females (Hewitt et al. 2005).

- Stormwater flows can increase flows and erosion far downstream of the outfall structure, causing increased turbidity. If erosion is severe enough, it could result in bank failure and landslides, further affecting the stream with increased sediment loading (Williams and Thom 2001).

- Impacts on aquatic invertebrates from outfalls have also been documented. In the Clinch River (Virginia), effluent from a wastewater treatment plant that contained monochloramine and un-ionized ammonia from domestic effluent resulted in a 2.3-mile (3.7-km) reach below the outfall devoid of several freshwater mussels (*Unionidae*) (Goudreau et al. 1993).

7.6.7.18 Tide Gates

Tide gates can influence temperature, dissolved oxygen, pH, metal concentrations, sedimentation, and salinity.

Abrupt changes in temperature can form as a result of blocked tidal flushing and represent a thermal barrier for fish migration, similar to a dam (Giannico and Souder 2005).

Disruption of natural flow can cause stratification and depletion of oxygen, with the downstream side of the tide gate becoming anoxic at the bottom (Winn and Knott 1992). In Cape Cod (Massachusetts), periodic low oxygen levels can result in large fish kills (Portnoy 1991) in tidal marsh systems.

Altered tidal flows can affect dissolved oxygen concentrations through changes in soil chemistry. When tidal water is excluded, soils that would normally be kept under anaerobic conditions because they are inundated by tidal waters are exposed to the air and can become aerobic. Subsequently, the exclusion of salt water can lead to oxygen depletion in the water when organic matter in the soils begins to oxidize (Giannico and Souder 2005). Oxidation of peat soils can cause the level of a marsh to fall and to become compacted (Roman et al. 1984). Lowered dissolved oxygen concentrations will alter redox conditions of the soils, altering pH levels and increasing metal leaching from soils. Episodic acidification of estuarine waters from the drainage of sulfate floodplain sediments is common (Anisfeld and Benoit 1997; Johnston et al. 2005a; Sammut et al. 1996). Drainage promotes the oxidation and export of sulfuric acid, a lowering of pH levels, and can result in the release of iron, lead, aluminum, copper, silver, and cadmium (Giannico and Souder 2005). In some cases, lowering pH produces iron flocs that can precipitate out of solution and cover the benthos (Sammut et al. 1996) and kill marsh plants (Giannico and Souder 2005).

Tide gates alter natural flow regimes and change natural sedimentation patterns. In addition, high velocities through open flood gates increase erosion both up- and downstream, increasing turbidity in the downstream water.

As tide gates block the movement of salt water, they change salinity both upstream and downstream of the gate. In a natural estuarine system, salinity fluctuates daily and seasonally from tides (Giannico and Souder 2005), and the presence of a tide gate alters the natural flushing pattern. This alteration causes a displaced salt wedge to migrate upriver (Vandenavyle and Maynard 1994). Salt water is denser than fresh water, and when a tide gate is closed, salt water settles and will migrate upstream. In addition, because water that builds up behind a tide gate is usually fresh, this pulse of fresh water is released downstream, lowering salinity in the receiving water (Williams and Thom 2001). Altered salinities can cause marsh community shifts; when tide gates are present, salinity gradients are sharp and can delay the migration of fish (Pearlstine et al. 1993). Salinity is also altered in the groundwater environment. In Australia, saltwater seepage into the surrounding groundwater was observed. Depending on the soil properties, this seepage was less than 33 to more than 262 ft (10 to more than 80 m) from the impounded area (Johnston et al. 2005b). This saltwater intrusion could have devastating effects on riparian vegetation, leading to increased bank failures, increased temperatures, and reduced nutrient cycling.

7.6.7.19 Beaver Dam Removal

The primary water quality impacts associated with beaver dam removal are related to suspended solids and temperature. Elevated suspended solids concentrations may continue for years after dam removal (Ahearn and Dahlgren 2005) until the upstream channel has incised, widened, and stabilized (Stanley and Doyle 2002). A positive aspect of beaver dam removal is that there is the potential to decrease average stream temperatures through the altered reach, although this effect will be dependent on site specific conditions and should not be expected in all cases. While results from the literature are mixed with regards to the thermal impact of beaver impoundments, it can generally be assumed that warm water species will suffer most with the removal of beaver dams. Beaver dams serve a vital function as areas of hydraulic retention where organic materials and sediments accumulate. The transient storage provided by beaver dams serves to impede flows, increase the contact time between solutes and sediment and organisms, and promote carbon, nutrient, and pollutant retention (Ensign and Doyle 2005; Gandy et al. 2007; Naiman et al. 1988).

7.6.7.19.1 Altered Temperature Regime

Beaver ponds are generally wider and shallower than the associated stream channel and consequently receive more solar radiation and are more susceptible to elevated temperatures. If a reach is already thermally impacted, then the presence of a beaver dam could promote the elevation of stream temperatures above the acceptable threshold of between 48 and 68^{o}F (9 and 20^{o}C) depending upon the species and life-history stage affected (WAC 173-201A 2006). Beaver dam removal may lead to decreased stream temperatures which could benefit cold water species such as bull trout and Dolly Varden. However, it should also be recognized that slightly elevated temperatures may be beneficial for even cold water species as it can enhance growth rates and thus survival (McCullough et al. 2001). This type of response would be expected where coldwater fish species have evolved in those systems influenced by beaver activity.

7.6.7.19.2 Altered Suspended Solids

Beaver impoundments reduce stream velocities and act as sediment sinks (Naiman et al. 1994). Aquatic vegetation in beaver impoundments also slows stream velocities and contributes to sediment retention (Chambers et al. 1999). Beaver pond removal can contribute to suspended solids concentrations immediately after removal and in the long-term.

7.6.7.19.3 Altered Pollutant Loading

Human impact on stream channels is frequently manifested through geomorphic simplification and a decrease in transient storage of wood, sediment, nutrients, and other organic materials. Beaver dams serve to counteract geomorphic simplification and to provide this transient storage capacity as integral components of the natural landscape. The removal of beaver dams degrades transient storage capacity, potentially leading to degraded water quality in downstream aquatic ecosystems. Beaver impoundments have been shown to be effective sinks for nitrogen and phosphorus (Margolis, Castro et al. 2001; Naiman et al. 1994) and by increasing hyporheic exchange and transient storage they may also serve to sequester metals pollution from urban areas, particularly metals and other pollutants prone to sorption on fine sediments and organic material (Gandy et al. 2007).

7.6.7.20 Large Woody Debris Placement/Movement/Removal

7.6.7.20.1 Altered Suspended Solids

LWD placement, movement, and removal can all cause construction-related sediment releases. Movement and removal generally release more sediment than placement. Removal projects have a continuing negative impact on suspended solids concentrations; movement and placement projects reduce suspended solids concentrations in the long-term.

Through bank protection (Angradi et al. 2004), increased hyporheic flow (Mutz and Rohde 2003), and increased sedimentation in the channel (Gomi et al. 2001) and adjacent floodplain (Brunet et al. 1994), suspended solids concentrations may be reduced by the placement of LWD within the channel. There have been no studies which have directly measured the effect of LWD on suspended solids partly because of the difficulty of attributing variability in suspended solids concentrations to one facet of the watershed ecosystem. However, studies have indicated an increased deposition of fines associated with LWD (Gomi et al. 2001; Wallace et al. 2000; Wallace et al. 2001); thus, a decrease in suspended solids can be inferred.

LWD removal would have the opposite effect of LWD addition. Increased channel velocities (Curran and Wohl 2003; Ensign and Doyle 2005), would entrain more sediment and increase suspended solids concentrations, while reduced hyporheic exchange and depositional area would further contribute to elevated suspend solids. Additionally, LWD removal may destabilize banks (Diez et al. 2000) and thus increase source areas for suspended solids.

7.6.7.20.2 Altered Nutrient and Pollutant Loading

Large woody debris presence within streams increases channel roughness and complexity and consequently increases transient storage. Increased transient storage will likely increase nutrient uptake (Bukaveckas 2007; Ensign and Doyle 2005; Roberts et al. 2007; Valett et al. 2002) and sedimentation (Worman 1998). The retention of coarse particulate organic matter (CPOM) has been correlated with LWD presence in numerous systems. Jacobson et al. (1999) found that LWD trapped sediment and CPOM which was then incorporated into the benthic biomass, creating islands of organic matter in the channel that became focal points for decomposition and secondary production. This increase in biologic activity would likely be accompanied by an increase in pollutant and nutrient processing.

Channel complexity promotes the retention of water and organic material. Water retention from LWD and the presence of CPOM in the channel in turn plays an important role in the fate of nutrients in the stream channel. In a classic study by Mulholland et al. (1985) it was suggested that leaf litter in streams promotes nutrient retention as the leaf pack acts as a substrate for nutrient-hungry microbes. Using solute injection techniques Valett et al. (2002) found that phosphorus uptake in channels with high LWD volumes, frequent debris dams, and fine grained sediments was significantly greater than in channels in younger forests without these characteristics. Corroborating this finding, Ensign and Doyle (2005) conducted phosphorus injections in streams both before and after the removal of LWD and CPOM in the channels and found that phosphate uptake decreased by up to 88 percent after LWD removal. These studies show that LWD increases water retention and thereby contributes to higher nutrient retention in streams that have large volumes of LWD. Retention of CPOM and LWD provides a substrate for biofilm growth. Decreased nutrient retention affects both local waterways and downstream receiving waters. Local waterways are affected through the associated reduction in primary production, and receiving waters (which are primarily located in more nutrient-impacted lowland areas) are affected through additional nutrient loading, which may lead to eutrophication. Alteration of nutrient cycling is likely to affect food web complexity, which can have a range of effects on HCP fish and invertebrate species limiting to survival, growth, and fitness.

Marine and lacustrine waters are considered receiving waters and consequently there is limited transient processing of pollutants in these systems. Instead, pollutants are internally cycled within the water bodies until pollutants are metabolized, sequestered, or exported. No research to date has measured the effect of LWD on these internal cycling processes.

7.6.7.20.3 Altered Dissolved Oxygen

Increased nutrient loading to downstream areas can contribute to eutrophication. Eutrophication is characterized by elevated carbon fixation, and this excess carbon in the aquatic system contributes to elevated levels of respiration. In waters with minimal physical mixing, oxygen consumed through respiration is not readily replaced with oxygen from the atmosphere. The result is a decrease in ambient dissolved oxygen levels.

7.6.7.21 Spawning Substrate Modifications

7.6.7.21.1 Altered Suspended Solids

Although gravel placement can increase suspended solids during the construction phase, through bank and channel bed disturbance and through the wash-off of any fines associated with the gravels, placing gravel is not expected to alter suspended solids in the long term. After the construction phase, the presence of gravel within the channel will likely reduce suspended solids concentration though bank protection and filtration through the hyporheic zone. Because the majority of gravel augmentations occur below dams, suspended solids levels will likely not be elevated to levels which may harm fish or invertebrates. Dams tend to buffer variations in suspended solids concentrations relative to upstream conditions (Stanford and Ward 2001). Consequently, downstream reaches receive primarily waters with low concentrations of suspended solids.

7.6.7.21.2 Altered Dissolved Oxygen

The small amount of research that has been conducted concerning benthic dissolved oxygen levels and gravel augmentation indicates that spawning substrate augmentation leads to increased intergravel oxygenation (Merz and Setka 2004; Merz et al. 2004). Some studies have shown that salmon preferentially choose nesting sites with elevated benthic dissolved oxygen levels (Geist 2000) or elevated hyporheic exchange (Mull 2005). Elevated benthic dissolved oxygen concentration has been correlated with increase embryo survival (Heywood and Walling 2007). Benthic invertebrates which require elevated dissolved oxygen concentrations may also benefit from gravel augmentations.

7.6.7.21.3 Altered Nutrient and Pollutant Loading

Gravel augmentation can promote increased hyporheic exchange and thus increase the potential for nutrient retention and cycling within the channel. This in turn may reduce nutrient loading to sensitive downstream receiving waters. Bacteria, periphyton, and fungi within an active benthic zone take up, transform, and sequester solutes. Some studies have shown the hyporheic zone to function as a nitrate sink (Lefebvre et al. 2005; Sheibley et al. 2003) while others have indicated that the hyporheic zone is a source of nitrate (Fernald et al. 2006) and soluble reactive phosphorus (Fernald et al. 2006; Lefebvre et al. 2005). What seems evident from the literature is that the hyporheic zone is a dynamic system with biogeochemical properties that vary from site to site and season to season. Anbutsu et al. (2006) monitored interstitial water nutrient concentrations in a high residence time hyporheic zone and noted that the area was a sink for ammonium, nitrate, and phosphorus. Sheibley et al. (2003) noted that when the groundwater is enriched in ammonium, biogeochemical transformation within the hyporheic zone will convert the ammonium to nitrate and also convert a portion of this nitrate to nitrogen gas. And finally, Lefebrve et al. (2005) noted that organic material decomposition deep within the hyporheic zone can act to release ammonium and soluble reactive phosphorus while periphyton near the surface will retain nitrate.

7.6.7.22 In-Channel/Off-Channel Habitat Creation/Modifications

A primary goal of most in-channel and off-channel habitat modifications is to create habitat but many of the same projects may also have ancillary water quality benefits. These benefits, depending upon the nature of the project, include reduced stream temperatures and reduced pollutant loadings.

7.6.7.22.1 Altered Temperature Regime

The creation of in-channel structures and side-channel habitat has the potential to create cool water refugia which can be utilized by temperature sensitive species. An experimental shading study by Ebersole et al. (2003), concluded that shading from riparian vegetation can cool surface waters by 3.6–7.2^{o}F (2–4^{o}C). In-channel shading can come from structures that provide cover such as engineered logjams (Abbe and Montgomery 1996). In-channel structures can also direct flow into the hyporheic zone and thus potentially lower stream temperatures (Grant et al. 2006). In addition, side-channels are usually characterized by narrow stream widths and dense riparian vegetation. Consequently, these habitats tend to provide cool water refugia. Ebersole et al. (2003) observed this when analyzing where cold water patches naturally occur in the Grande Ronde basin in northeast Oregon; they found cool water patches associated with seeps, side channels, alcoves, and floodplain spring tributaries. The objective of many in-channel and off-channel stream rehabilitation projects is to mimic this thermal heterogeneity. However, Moerke and Lamberti (2004) found that of the 10 channel rehabilitation sites analyzed in streams across Indiana (primarily channel relocation and floodplain reconnection projects), the general trend was for reduced riparian vegetation along the restored reach. Consequently, the impact of in-channel and off-channel habitat modification on water quality will be project specific and depend on riparian cover and hyporheic exchange dynamics.

7.6.7.22.2 Altered Nutrient and Pollutant Loading

The effect of an in-channel or off-channel habitat modification project on pollutant loading is difficult to predict. Most projects aim to increase channel roughness and create more slack water habitat. Ecohydrologic theory dictates that these activities would increase transient storage and likely increase pollutant and nutrient retention. Studies by Vallett (2002) and Ensign and Doyle (2005) both indicate that nutrient retention is elevated in streams with more pools and fine benthic material (impounded within the pools). It can be inferred that projects which increase the retention of water and organic material will also increase pollutant retention. In eutrophic and urban systems these ecosystem alterations would benefit the HCP species.

7.6.7.22.3 Altered Suspended Solids

In-channel and off-channel habitat modification can increase suspended solids during the construction phase through bank, channel bed, and/or floodplain disturbance. If constructed correctly, after the construction phase and dependent on the project, total suspended solids concentrations will likely be reduced relative to preproject conditions. Floodplains have consistently been shown to be sediment sinks (Ahearn et al. 2006; Florsheim and Mount 2002; Tockner et al. 1999; Valett et al. 2005); thus, projects that enhance lateral connectivity can be expected to reduce suspended solids concentrations within the main channel. Likewise, in-channel projects that create complex flow paths and thus decrease mean velocities can be expected to increase within channel sediment retention and decrease suspended solids concentrations (Shields et al. 1995). Projects that enhance bank stabilization and protection will likely decrease sediment source areas and thus decrease in-channel suspended solids concentrations (Sear et al. 1994).

7.6.7.23 Riparian Planting/Restoration/Enhancement

7.6.7.23.1 Altered Temperature Regime

If clearing invasive species is part of the riparian management plan, the project may be associated with initial increases in instream temperature (Bennett 2007). However, once the riparian plantings mature, shading can be expected to decrease temperatures (LeBlanc and Brown 2000; Opperman and Merenlender 2004) by between 3.6 and 7.2^{o}F (2 and 4^{o}C) (Ebersole et al. 2003).

7.6.7.23.2 Altered Suspended Solids

During construction, planting riparian buffers may temporarily increase sediment to streams. If constructed correctly, riparian buffers are effective filters which can reduce sediment loading to adjacent aquatic environments. In a review of six studies, Hickey and Doran (2004) found that between 84 and 90 percent of influent total suspended solids can be removed by riparian buffers. Reduced sediment loading would benefit aquatic organisms that are sensitive to elevated turbidity.

7.6.7.23.3 Altered Pollutant Loading

A buffer of riparian vegetation, depending upon its geometry, preferential flow, and pollutant loading, may have a significant effect on pollutant attenuation through shallow groundwater and overland flow. Although there has been little research concerning urban runoff attenuation through riparian buffers, studies have examined filter strips along highways. Wu et al. (2003) found that highway filter strips can remove 60 percent of influent copper, while Barrett (2005) found that bio-filters remove 75 percent of influent zinc on average. Bio-filtration is becoming widely adopted for urban stormwater management, and the same principal should and will be applied to riparian buffers in the near future. Riparian planting in urban areas will benefit aquatic biota that have already shown signs of impairment due to urban pollution in western Washington (PSAT 2007).

7.6.7.24 Wetland Creation/Restoration/Enhancement

Riverine and estuarine wetlands are effective pollutant filters which have been shown in numerous studies to reduce metals (Sheoran and Sheoran 2006), nutrients (Vellidis et al. 2003), and sediment (Tockner et al. 1999) loading. Because wetlands are located between uplands and water bodies, they can intercept runoff from the land before it reaches open water. As runoff and surface water pass through these systems, wetlands remove or transform pollutants through physical, chemical, and biological processes. In aquatic systems that exhibit degraded water quality, the reduction of these pollutants through wetland creation, restoration or enhancement would benefit many of the HCP species.

7.6.7.24.1 Altered Suspended Solids

Riverine wetlands are areas of river channels that are occasionally-to-permanently flooded. These areas can be nonvegetated or vegetated by submersed and nonpersistent emergent aquatic plants. Estuarine wetlands are typically found on the deltas and in the lower reaches of most of the rivers in western Washington and are also nonvegetated or vegetated by submersed and nonpersistent emergent aquatic plants (Ecology 2005; USGS 1997). Emergent vegetation slows water velocities and decreases wind induced mixing. This results in quiescent waters and an associated settling of suspended particles (Kadlec and Knight 1996). Studies have indicated that riparian wetlands can decrease influent suspended solids loadings and concentrations by 90 percent or more (Michael 2003; Tockner et al. 1999). Wetland creation, restoration, or enhancement activities intended to reduce suspended solids would reduce suspended solids in adjacent waters and likely benefit HCP species that are sensitive to elevated turbidity. During the construction phase, wetland construction can temporarily increase suspended solids.

7.6.7.24.2 Altered Nutrient and Pollutant Loading

Historic draining and infilling of wetlands in the state of Washington (Ecology 2005; USGS 1997) has reduced the capacity of the landscape to process pollutants. This has resulted in increased pollutant loading and the degradation of aquatic ecosystems. Activities that result in the increase of wetland habitat will result in reduced pollutant loading to adjacent water bodies.

Wetlands are characterized by quiescent waters and thus sedimentation is a primary mechanism of pollutant treatment (Kadlec and Knight 1996). Wetlands have been effectively used in Washington to reduce nutrient loading from fish hatcheries (Michael 2003), and to reduce metals and nutrient loading from urban areas (Reinelt and Horner 1995). Wetland creation, restoration, and enhancement would increase the area of wetland where this treatment could be implemented.

7.6.7.25 Beach Nourishment

7.6.7.25.1 Suspended Solids and Turbidity

Beach nourishment activities can result in some temporary elevation of turbidity during construction and for some time after construction activity has ceased (Wilber et al. 2006). On open, sandy coasts, it has been proposed that coarser sediments than occur under natural conditions can be placed to counteract the problem of turbidity and nourishment loss (NRC 2007). However, placing coarser material than occurs naturally can have a detrimental effect on nearshore life, as those species that prefer naturally finer-grained substrates are lost (Peterson et al. 2006). For environments like the Puget Sound and large lakes (steep, coarse, quiescent shorelines), it is less clear whether the impact on aquatic organisms would be as significant due to the diminished importance of waves and wave-induced circulation at depth (Jackson et al. 2005).

7.6.7.25.2 Altered Pollutant Loading

Although there are strong political and economic temptations to use dredged materials for large beach nourishment projects (Dobkowski 1998; Yozzo et al. 2004), extreme caution should be exercised when these types of materials are used in beach nourishment projects. Fine-grained sediments found in subtidal areas preferentially adsorb pollutants that may have accumulated during historical times when controls on pollution were not as tight as today. There is a significant risk of remobilizing pollutants into the water column both during dredging operations and during sediment resuspension events during storms (Petersen et al. 1997). In fact, it is possible to pollute the water column with naturally occurring trace metals even when the sediments have not been previously contaminated (Saulnier and Mucci 2000).

7.6.7.26 Reef Creation/Restoration/Enhancement

Reef creation/restoration/enhancemnt can impact water quality because of the material used to construct the reef. If the material used leaches potentially harmful chemicals into the water column, losses of fish and invertebrates could result. Also, if the material used is toxic to species that burrow, species diversity on the reef will, at a minimum, be compromised.

Reef creation/restoration/enhancement can cause changes in nearshore circulation and wave energy. These effects will be negligible if the reef is placed entirely below the closure depth[3]. If the reef is placed above the influence of surface gravity waves, there will be water quality modifications associated with turbidity and nearshore circulation.

7.6.7.26.1 Altered Suspended Solids

As the shoreline adapts to the change in wave energy and nearshore circulation, there is a potential for an increase in suspended solids as sediment is resuspended and redistributed. Further, restricted circulation near the structure could initiate eutrophication and heighten concentrations of biological colloids.

7.6.7.26.2 Altered Nutrient and Pollutant Loading

Reefs have been constructed virtually out of every solid material known (Baine 2001). Therefore, it is impossible to cover all of the potential sources of contamination. However, concrete forms, used tires, and derelict vessels seem to be the most popular reef materials and these have been addressed specifically in the literature (Baine 2001). Even household refuse has been examined as an artificial reef material (Chapman and

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[3] "Closure depth" is "the depth beyond which no significant longshore or cross-shore transports take place due to littoral transport processes. The closure depth can thus be defined as the depth at the seaward boundary of the littoral zone." (Mangor, Karsten. 2004. "Shoreline Management Guidelines". DHI Water and Environment, 294pp.)

Clynick 2006). These authors are careful not to advocate ocean dumping of trash, but they consider the ecological consequences of removing existing trash piles. This extreme case is useful to identify the degree to which stability is a key component to artificial reef success.

Discarded tires are a common substrate for artificial reefs. Tires have been shown to be relatively benign from a water quality perspective (Hartwell et al. 1998); however, they can have a tendency to be mobilized during storms. As a result, they are not recommended for artificial reef creation.

Sinking of vessels for reefs can introduce pollutants to the water column. Vessels can contain many toxic compounds including many polycyclic aromatic hydrocarbons and heavy metals. Although these sources of pollutants are supposed to be removed before sinking a vessel for a reef, small amounts of toxic materials may be difficult to find and may present a risk to HCP species in the vicinity of the vessel placement.

The construction of freshwater reefs using concrete can affect the pH of surrounding waters if uncured cement is allowed to contact the receiving water body. Generally speaking, marine waters are sufficiently buffered to ameliorate effects from fresh concrete, but rivers and lakes are not buffered and can experience important pH effects (Webster and Loehr 1996). It should be noted that these impacts will only occur when fresh concrete is used in reef creation, and artificial reefs do not usually require concrete during construction.

There is potential for reefs to alter nearshore circulation so as to increase stratification and the potential for eutrophication (Fischer et al. 1979).

7.6.7.27 Eelgrass and Other Aquatic Vegetation Creation/Restoration/Enhancement

The processing and retention of sediment, nutrients, and pollutants in aquatic systems is accelerated by the presence of aquatic vegetation (Clarke 2002). Numerous studies have shown that macrophytes and algae in marine environments act to reduce ambient concentrations of suspended sediment (Abdelrhman 2003; Moore 2004), nutrients (Moore 2004), and metals (Fritioff and Greger 2003). Seagrasses have also been linked to improved water quality. As an example, Moore (2004) noted decreased nutrient concentrations and turbidity levels in seagrass beds relative to areas outside the beds along the littoral zone of the Chesapeake Bay National Estuarine Research Reserve. However, aquatic vegetation not only reduces nutrient and sediment concentrations, the plants themselves can sequester harmful trace metal pollutants and are frequently planted in wetland treatment systems with that intended function. In a comparative study of heavy metal uptake in terrestrial, emergent, and submerged vegetation, Fritioff and Greger (2003) noted that submerged vegetation was efficient at removing zinc, copper, cadmium, and lead from influent stormwater.

8 Cumulative Impacts

The cumulative impacts[1] evaluated for the purposes of consultation under ESA are those effects of “future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation” (50 Code of Federal Regulations 402.02). The “reasonably certain” language regarding activities is too restrictive to meet this paper’s objective of providing a general evaluation of cumulative effects of HPA-permitted activities. Therefore, a broader interpretation of cumulative effects is considered here. For the purposes of this paper, the cumulative impacts considered are the incremental impacts of individual projects considered in the context of other past, present, and reasonably foreseeable future actions.

Assessing cumulative impacts falls into the category of “an emerging science.” No sources were identified that established quantified thresholds. However, the literature search identified numerous planning efforts throughout the country where cumulative impacts are identified as a topic to be addressed.

The Council on Environmental Quality (CEQ 1997) presents a simple typology of cumulative impacts where cumulative effects arise from single or multiple actions and accumulate in an additive or interactive manner. This typology and bank protection examples are presented in Table 8-1.

Table 8-1. Types and Examples of Cumulative Impacts

Cumulative Impact Type (CEQ 1997)	Example
Type 1. Repeated “additive” (*or deletion*) effects from a single project	A single bulkhead disrupts sediment transport and after each storm event sediment is transported from the downdrift beach section without being replaced. The deletion of sediment accumulates incrementally over time.
Type 2. Stressors from a single source that interact with biota to have an “interactive” (nonlinear) net effect.	A single bulkhead disrupts sediment transport as illustrated above and reflects wave energy that scours sediment from the beachfront along the bulkhead. The beach becomes coarser due to scour and lack of sediment resupply. Intertidal habitat is altered and no longer available for the benthic fauna such as

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[1] Note to Reviewers: The content of this chapter is drawn from ten white papers prepared for WDFW in 2006 and 2007 on a variety of HPA-permitted activities. Because each of the original white papers discussed one category of activities, the discussion of “cumulative impacts” tends to be limited, emphasizing the effects of having several individual instances of a given activity in a particular area, (for example, the impacts of many bank armoring projects within a given reach.) The discussions do not necessarily discuss cumulative impacts in the broader sense, for example if bank armoring, overwater structures, water crossing structures, and habitat modifications were all permitted in a given area.

Cumulative Impact Type (CEQ 1997)	Example
	bivalves.
Type 3. Effects arising from multiple sources that affect environmental resources additively.	In addition to construction of a bulkhead, the riparian vegetation is removed. The bulkhead reduces the shore roughness and no longer retains LWD, and recruitment of LWD is lost due to clearing of the riparian vegetation. Shade provided by riparian vegetation is also lost, thereby increasing solar radiation and water temperature.
Type 4. Effects arising from multiple sources that affect environmental resources in an interactive (i.e., countervailing or synergistic) fashion.	Additional bulkheads are constructed due to concentration of wave energy from existing bulkhead or due to perceived threats increasing the length of protected shoreline. Effects accumulate in a linear manner to a threshold where habitat structure and composition are substantially changed, leading to an alteration of habitat processes and ultimately a shift in ecological function. This would be manifested in a reduction of habitat and loss of species richness.

This conceptual framework of cumulative impacts could be applied at a regional scale, where individual impacts could be quantified. However, due to the complexity of quantifying impacts and the lack of specific data, cumulative impacts are often assessed qualitatively. In the absence of a quantitative analysis of cumulative impacts, the following sections qualitatively describe the cumulative impacts of each impact mechanism.

Evidence increasingly indicates that the most devastating environmental effects are likely not the direct effects of a particular action, but the combination of individually minor effects of multiple actions over time (CEQ 1997).

Nightingale and Simenstad (2001b) specifically discussed the effects of overwater structures. However, much of their discussion is applicable to other types of HPA-permitted activities as well. They note that “The bathymetry of Washington's inland waters, that of a fjord surrounded by a narrow strip of shallow vegetated habitat, magnifies the need to protect the integrity and continuity of this limited area of nearshore habitat because of the concentrated zone of potential impact.” This is directly relevant to an ESA analysis, because it identifies the area where cumulative impacts will have a concentrated effect on habitat processes, structure and functions. In general, as the number of shoreline modification structures increases in a given area, impacts will accrue producing a net loss in vegetation production and a concomitant reduction in epibenthic and benthic nearshore habitat. The type and extent of each of these alterations depend on site-specific characteristics and structure types.

Nightingale and Simenstad (2001b) discuss cumulative effects on “rural and natural” as opposed to “urban industrialized” shorelines. For rural shorelines, the authors state that:

> The habitat value of an environment that directly supports the recruitment of fish and shellfish stocks is magnified by its overall importance in stock recruitment. Its value is intrinsic to its location but its loss to stocks and the larger ecosystem reaches beyond its specific location. In short, protection of habitats critical to important survival and recruitment needs of fish and shellfish magnify the importance of controlling any adverse effects to them. Economically, it is far less expensive and more productive to protect existing critically important habitat than to restore lost or degraded habitats. The factors controlling habitat characteristics and the biologic assemblages that have evolved are endemic to the geologic and biologic history specific to a geographic location and region. Perhaps more significantly, the linkages among these ecosystem components are not fully understood.

This finding is relevant to an ESA analysis because it identifies how cumulative impacts potentially impair habitat essential to reproduction and thus directly affect a species’ capacity to sustain and increase its numbers. Such impacts, if sufficiently severe, may jeopardize a species’ continued existence.

With regard to cumulative impacts along urban industrialized shorelines, Nightingale and Simenstad (2001b) identify three principal concerns:

- Reduced access to prey resources, compelling juvenile salmon to outmigrate farther and faster than they otherwise would, reducing their metabolic energy resources and potentially exposing them to other risks, such as predation. Although this finding is not directly transferable to other potentially covered species, it is plausible that they too would have to travel farther to access suitable habitat and would also suffer reduced metabolic energy resources and increased exposure to other stressors.
- Reduced autochthonous productivity due to limited light availability, an impact that could be reduced by incorporating design features to reduce shading.
- Landscape-scale effects (such as fragmentation) that could be minimized by landscape-scale habitat treatments, enhancing habitat in refuge areas such as beaches.

8.1 Cumulative Impacts Sorted by Mechanism of Impact

8.1.1 Cumulative Impacts Associated With Construction and Operations

8.1.1.1 Noise

Cumulative impacts associated with the construction activities of multiple projects could amplify the behavioral alterations or physical impacts that could occur as a result of individual projects. Cumulative noise impacts may result from the accumulation of exposure energy that fish receive from multiple pile drives (Popper et al. 2006), increased numbers of boats or boating use (Scholik and Yan 2001a), and increased use of construction equipment.

In speaking of cumulative noise impacts to marine mammals, Dr. Sylvia Earle, former chief scientist at NOAA, has stated that “each sound by itself is probably not a matter of much concern," but taken together, "the high level of [ocean] noise is bound to have a hard, sweeping impact on life in the sea" (Holing 1994, in Radle 2005). Applying this concept to the potentially covered species, the repeated occurrence of noise could prompt organisms to migrate away from an area. Conceivably, minor physical impacts associated with individual projects could become more severe if several projects in an area result in the same type of impact. Also, an organism or its habitat could be more vulnerable to physical damage due to the impacts of preceding activities.

Construction is only one of several sources of such noise; other major sources include large-vessel shipping traffic, military activities, and acoustic profiling for petrochemical and minerals exploration. However, the cumulative impacts of such noise sources on fish physiology and behavior are unknown at this time.

8.1.1.2 Artificial Light

Although it has been shown that juvenile salmonid migrations can be delayed by artificial light in freshwater and marine environments (McDonald 1960, in Tabor et al. 1998; Prinslow et al. 1979, in Nightingale and Simenstad 2001b; Tabor et al. 1998), the implications of this delay are not known. The cumulative impacts of increased artificial light in the aquatic environment have not been investigated. It has been suggested (and, in the case of sockeye fry and sculpin, shown [Tabor et al. 1998]) that rates of predation on juvenile fish increase under artificial light because of changes in migration patterns, congregation of predators, or increased opportunity time for predation. For some HPA-permitted activities, artificial lighting is temporary during the period of construction, but other facilities with installed artificial lighting will cumulatively add to light sources over water. It is unknown whether losses of threatened and endangered juvenile salmonids could occur due to regional-scale cumulative lighting impacts.

8.1.1.3 Vessel Traffic

Cumulative impacts of vessel activities are not well characterized. Some potential cumulative impacts include the following:

- Cumulative impacts from vessel activities have been reported with respect to turbidity. Vessel traffic may cause extended periods of elevated turbidity as boat traffic collectively churns the water, slowing the settling of suspended sediment (Garrad and Hey 1988).

- Successive passes by vessels may accelerate shoreline erosion; recreational vessel traffic has been observed to cause boat wake-induced levee erosion at rates of 0.0004 to 0.009 inch (0.01 mm to 0.22 mm) per boat pass (Bauer et al. 2002).

- Commercial shipping in the Northern Hemisphere has been implicated in a 10-fold to 100-fold increase in oceanic noise levels (Tyak 2000, in Scholik and Yan 2001a), and it has been shown that fish exhibit behavioral and physical responses to vessel noise. However, the cumulative impact of vessel noise on fish has not been specifically studied.

- Because marinas serve multiple vessels, the underwater noise generated by boating activities (i.e., outboard motors) is cumulatively higher than at a single dock. If the marina is surrounded by a breakwater, the noise effect may be limited to the area of the marina. However, as a hub for boats traveling in and out, it would add to the total noise levels in the surrounding area. For shipping and ferry terminals, the potential for increased ambient noise levels, benthic disturbance, ambient light modifications, and water quality degradation is even greater due to year-round boat traffic and the size of the vessels using these facilities.

- Vessel traffic is associated with grounding, anchoring, and prop wash. Boat use will likely increase as the state population increases, resulting in disturbance of bottom substrates and vegetation generated by propeller wash.
- Cumulative effects from vessel traffic would be expected based on projections of future growth. In 1980, total ridership of Washington State Ferries was 16.7 million; by 2002, it increased by 50 percent to 25.1 million. These volumes are projected to continue to increase to 43.4 million riders by 2020 (WSDOT 2006b). Recreational vessel numbers and commercial vessel traffic have also increased, and this trend is expected to continue.
- Vessel type affects the potential impacts generated by boat traffic. WSDOT is currently in the process of evaluating the feasibility of several different vessel types to replace existing ferries at the Port Townsend and Keystone ferry terminals (WSDOT 2007).
- Sandstrom et al. (2005) found that vessel activities had a profound effect on species composition of aquatic plants, and Eriksson et al. (2004) found similar effects on fish species in areas of marinas and ferry boat routes compared to those areas without such boating traffic.

8.1.1.4 Channel Dewatering

No studies examining the cumulative impacts of channel dewatering were found during the literature review. The following discussion is therefore based on the authors' professional experience.

Cumulative impacts of channel dewatering will most likely be associated with fish removal/exclusion methods, disturbance of the streambed, and modification of invertebrate habitat and consequent changes in species diversity. Although there are no available studies on the cumulative effects of temporary activities associated with channel dewatering, cumulative effects could result from the permitting of numerous dewatering activities within a watershed over a relatively short period of time.

The cumulative impacts on a particular species' population would depend on the number of concurrent projects at a watershed scale, as well as the population size of a given species. The cumulative impacts to fish populations resulting from multiple permitted activities within a watershed that require fish removal/exclusion could be measurable at the population scale depending on several factors, including watershed and population size. Fish removal/exclusion results in the capture and handling of fish, which can cause stress, harm, and mortality.

The threshold for watershed and population size and the number of activities that must occur within a particular watershed to have a measurable cumulative impact are not established in the literature.

Temporary losses of benthic macroinvertebrates are likely to occur as a result of dewatering associated with new construction or expansion of existing structures. Changes in the representative species assemblages as a result of changes in hydraulics and habitat conditions within affected reaches are also possible. The cumulative impacts of repeated channel dewatering efforts could lead to changes to benthic macroinvertebrate populations or species diversity that may lead to subsequent changes to fish populations or habitat occupancy.

Disturbance of the streambed associated with dewatering may result in temporary loss of habitat. The significance of the loss depends on the size of the watershed, the amount of habitat cumulatively lost, and the significance of the habitat lost to the population (i.e., spawning, rearing, or migration habitat). It seems unlikely that HPA-authorized activities would result in measurable cumulative effects except in the case of rare species where a single project might affect habitat critical to a large fraction of the watershed's population.

8.1.1.5 Dredging

Analysis of cumulative effects of landscape-scale bathymetry modifications and changes to habitat structure should include the overall scope of dredging activities undertaken in the region. Understanding the scope of current dredging activities requires a breakdown and comparison of the areal extent of maintenance dredging undertaken annually compared to new project dredging, as well as the extent to which this dredging alters the nature of existing habitats in marine, riverine, and lacustrine environments. An analysis of the scope and nature of current dredging activities can lay the groundwork for assessing the long-term, cumulative effects that dredging activities can pose on existing ecosystem dynamics and the effects such changes may have on a variety of species.

The scope of such an assessment will vary depending on the environment type. For example, in marine environments an assessment might focus on the areal extent of dredging activities within each of the oceanographically distinct basins in Puget Sound, differentiating habitat impacts in terms of the depth, substrate composition, and bathymetric profile of the affected areas. In lacustrine habitats, the analysis might have a similar focus but would be limited to the individual lake. In riverine environments, a watershed-scale approach or a more targeted approach differentiating estuarine impacts may be appropriate.

8.1.1.6 Accidents

One cause of cumulative impacts that is generally not addressed in the literature but that applies to HPA-permitted projects is accidents. Accidental chemical spills, accidental concrete spills, accidental erosion of material stockpiles, and various other kinds of accidents that occur during use of structures constructed under the HPA authority all constitute impacts that likely would not have occurred but for the issuance of an HPA. Such accidents can be predicted only in a statistical sense, and WDFW would likely not have legal liability for these accidents, but the impacts could still occur and therefore could affect populations of potentially covered species. This impact would be considered by the federal agencies in their decision to issue an Incidental Take Permit.

8.1.2 Hydraulic and Geomorphic Modifications

Generally, the question of cumulative impacts of channel hydraulic effects emerges as a data gap. The HPA program itself offers a means of collecting data to help measure these impacts, because WDFW has authority to require monitoring of authorized projects. To date, however, monitoring these types of effects has not been emphasized.

8.1.2.1 Urban Streams

Studies on the cumulative effects of increased impervious areas have focused on the effects of urbanization on the ecology of urban streams. The condition of urban streams is controlled by the altered timing and volume of water, sediment, nutrients, and contaminants resulting from the urbanized catchment (Bernhardt and Palmer 2007). The most noticeable determinant of channel change in urban watersheds is the increase in streamflow discharges (Booth and Henshaw 2001). Increased peak streamflows from urban development cause streams to incise deeper and wider channels (Booth 1990; Hammer 1972; Leopold et al. 2005). The consequence of this channelization is local bank failure, increase in sediment supply, and sediment deposition in lower gradient, downstream reaches (Booth 1990). Konrad (2000) examined urban watersheds in the Puget Lowland and found that urban development increased peak discharge magnitudes and decreased storm flow recession rates, causing “flashy” runoff conditions. Consequently, substrate reworking by flow was more frequent and extensive in urban streams than in suburban streams draining less-developed watersheds. Summer base flow was also suppressed relative to suburban streams. The ecological effects of urbanization on urban streams include species-poor assemblages of fish and invertebrates (Freeman and Schorr 2004).

8.1.2.2 Littoral Drift

Artificial structures that change longshore drift can alter organic and sediment deposition on beaches and therefore alter biotic assemblages (Thom et al. 1994). However, the overall cumulative impacts of changes in littoral drift due to artificial structures on the system as a whole cannot be predicted at this time (Thom et al. 1994).

8.1.2.3 Substrate Modification

Many HPA-permitted activities can result in substrate modification. The cumulative impacts of each component of substrate modification can lead to a reduction in the quantity and quality of habitat for potentially covered species. As noted by Quinn (2005), the incremental loss of spawning and rearing habitat has contributed to the declines in salmonid populations. Substrate modifications along marine shorelines have reduced the availability of suitable spawning habitat for surf smelt and sand lance. The cumulative impacts of these modifications are unknown; however, a crash in their populations could further impact salmonids and other piscivorous fish.

Among the potentially covered invertebrate species, Newcomb’s littorine snail is particularly vulnerable to cumulative impacts of substrate modifications given the species’ small geographic range (Grays Harbor and Willapa Bay) and specific habitat preference (*Salicornia virginica* marshes).

8.1.3 Shading

The studies reviewed evaluate cumulative effects of overwater structures, but the effects due to other structures that cast shade are likely similar. These studies suggest that the cumulative impacts of shading do not differ significantly from the direct and indirect impacts of single-structure shading, i.e., decreased primary productivity, loss of eelgrass beds with impacts to the associated food chain processes, and changes in the migration patterns of salmonids. There are data to suggest that the cumulative loss of habitat resulting from the shading of multiple structures can affect fish abundance and species richness within a region (Carrasquero 2001; Fayram 1996; Kalher et al. 2000; Williams and Thom 2001).

The cumulative impacts of even narrow residential piers can be detrimental in a freshwater environment (Carrasquero 2001). It has been suggested that the cumulative impact of an increase in the number of docks around the Lake Washington shoreline, where approximately 4 percent of shallow-water habitats are covered by overwater structures (Kalher et al. 2000), might have caused the observed decrease in freshwater survival of juvenile sockeye salmon (Fayram 1996). Although individual shoreline structures may not impose significant impacts on salmon species, populations, or stocks, the cumulative impacts of dense, contiguous shoreline modifications are likely contributors to the present decline of several Puget Sound salmon species and may inhibit the success of recovery actions (Williams and Thom 2001).

Fish feeding and migration abilities are closely linked to the ambient light environment. To the extent that under-dock environments block light transmission, they pose the risk of diminishing prey resources and triggering behavioral changes of HCP species. As these structures are typically in the shallow nearshore, the impacts on fish would likely be to the juvenile life-history stage. Following a study of ferry terminals in Puget Sound, Haas et al. (2002) reported that large overwater structures pose serious impacts on intertidal and subtidal nearshore habitats. These impacts include reduced benthic vegetation and decreased densities of epibenthic prey. Haas et al. (2002) concluded that the cumulative effects in densely population areas, such as Puget Sound, may be large. The extent to which this impact on prey availability is limiting to juvenile salmon is unknown. Haas et al. (2002) also identified extensive impacts of ferry terminals that pose habitat fragmentation effects.

8.1.4 Aquatic Vegetation

Aquatic vegetation is a fundamental structural component in marine, estuarine, and lake environments. Numerous species utilize the vegetation for cover, feeding, and spawning. The successive incremental losses of aquatic vegetation by multiple HPA-permitted projects could impact the species distributions and productivity. While aquatic vegetation may be resilient in recolonizing disturbed areas if suitable conditions are provided, the potential isolation of vegetation patches through the impacts of multiple projects could lead to the disappearance of the patch.

Existing structures will continue to modify ambient light conditions and subsequently aquatic vegetation via shading and turbidity. An increase in facilities or facility capacity and an overall increase in vessel traffic will likely magnify these impacts. Future construction of new facilities could result in the removal of existing aquatic vegetation, further affecting these resources.

8.1.4.1 Eelgrass and Macroalgae

The cumulative impact of structures that shade potential eelgrass habitat or otherwise inhibit growth would be a reduction in eelgrass coverage, as can be seen at individual structures (Nightingale and Simenstad 2001b). Large-scale eelgrass monitoring in the inland waters of Washington State (2000 through 2008 data set) indicates that the majority of eelgrass sites have no significant change, and that at sites with significant increases or decreases in eelgrass, the differences are small (Dowty et al., 2009). Preliminary data indicate downward trends may exist, but further analysis of data will be required to attribute significance to these results (personal communication, Dowty, 2009). However, because eelgrass coverage is affected by many variables in addition to the cumulative impacts of development, the results observed by Dowty et al. (2005) do not indicate a clear cause and effect of development on overall patterns of eelgrass coverage.

The real implications of cumulative changes in eelgrass distribution and cover are unclear, because it is not known how dependent many potentially covered species are on eelgrass. For instance, herring spawn on eelgrass, but there are extensive areas of eelgrass where no herring spawn, so changes in eelgrass cover alone would be a poor predictor of future herring spawning success. Similarly, young salmon forage extensively in eelgrass, but foraging habitat may not be a limiting factor for juvenile salmon in Puget Sound (Haas et al. 2002). Much human impact on eelgrass and macroalgae takes the form of habitat fragmentation, but although such fragmentation is in principle an adverse impact, it remains unclear just how that impact is delivered to affected species (Haas et al. 2002). Thus, our understanding of cumulative impacts on eelgrass and macroalgae is limited by major data gaps.

It has been documented that areas where eelgrass has been lost through direct disturbance or alteration of habitat conditions are sometimes colonized by other macroalgae species (Thom et al. 1994). This shift in aquatic vegetation would also be a shift in habitat structure, which could lead to a shift of fauna assemblages (Williams and Thom 2001). The shading of eelgrass beds that serve as important nursery habitat for many species can greatly affect numbers of marine biota within a region, including salmonids, crab, herring, and important epibenthic crustaceans. Pacific herring would be vulnerable to alterations in eelgrass distribution. Given the strong association of important fish prey resources with eelgrass, the shading out of eelgrass by numerous overwater structures poses a potential risk of reduced prey resources for fish, affecting fish populations.

8.1.4.2 Freshwater aquatic plants

Individual structures can reduce the overall coverage and density of freshwater aquatic plants in lakes and ponds with developed shorelines (Radomski and Goeman 2001). This could significantly affect the ecological functions of aquatic systems in the vicinity of HPA-permitted structures. For example, Radomski and Goeman (2001) found that because of reduced littoral vegetation, the most highly developed lakes are lacking in physical habitat structure compared to less developed lakes, which was reflected in a correlation between the occurrence of floating leaved and emergent plants and (warm-water) fish biomass.

8.1.5 Riparian Vegetation

Site-specific habitat functions are determined by whether an existing shoreline is in a relatively natural state or whether it is affected by urban development. Cumulative effects from additional HPA-permitted structures influence habitat functions. A natural environment that supports fish and shellfish spawning, rearing, and refugia is highly valuable from a biological perspective. Any alteration to that specific environment could influence the recruitment of fish and shellfish stocks in the larger ecosystem. As a result, the cumulative impact of structures along an ecologically intact shoreline could generate potentially significant cumulative effects.

In contrast, an urban, industrialized shoreline area may have, over a long period of time, lost its native vegetation and suffered major changes to its historical substrates. In that scenario, the addition of a new structure may pose a qualitatively different set of cumulative effects than the effects of the same new structure in a more natural environment.

Substantial loss and fragmentation of riparian habitat has occurred in the Puget Sound region over the last 100 years. Although empirical data are lacking to quantify the extent and quality of riparian habitat, existing data suggest that riparian areas within urbanized shoreline areas such as King County have been significantly altered (up to 100 percent) with upland development and increasing levels of urbanization (Brennan and Culverwell 2004). McClain et al. (1998) and Francis and Schnider (2006) described the process of urbanization in the Pacific Northwest as a trend that moves toward deforestation without replanting.

Although there have been numerous evaluations on the effects of large-scale removal of riparian habitat to aquatic habitats, few studies specifically address cumulative impacts from the localized removal of riparian and shoreline vegetation as part of the specific types of activities permitted by HPAs. It is expected that permitting multiple activities within a watershed can have cumulative impacts to riparian vegetation, including increased likelihood that the impacts will be measurable and thus more likely to have an adverse impact to aquatic species and habitat.

Such impacts may be more significant in smaller watersheds. The threshold at which a group of activities will have an adverse impact to aquatic species and habitat at the watershed scale cannot be quantified, because each watershed has unique characteristics, such as riparian/shoreline vegetation and the contribution such habitat makes to the quality of specific aquatic habitat.

Naiman et al. (2000) reports that although riparian communities are being managed for a wider-than-ever variety of ecological functions, riparian communities in heavily urbanized environments constrained by pavement are precluded from the full restoration of natural functions.

A major finding in a study of the cumulative effects of urbanization on 22 Puget Sound streams found that mature forested riparian corridors were effective in mitigating some of the cumulative effects of adjacent development. In riparian corridors found in highly urbanized areas, poor stream quality is common (May 1998).

8.1.6 Water Quality

8.1.6.1 Turbidity

The cumulative impacts of HPA-permitted projects on water quality appear to have more potential for significant impacts than the generally short-term impacts that may result from an individual project. When combined with the impacts of land uses, it is conceivable that species tolerances could be exceeded for temperature and dissolved oxygen, which would lead to mortality or displacement (avoidance).

Natural turbidity-causing events may vary greatly in magnitude and duration. Natural events are more likely to occur in an isolated fashion and affect different portions of the stream network at different times (Bash et al. 2001). This variation allows fish to use refuge areas that might otherwise be impacted by these events (Bash et al. 2001). Professional experience has shown that anthropogenic sediment disturbance is often different; such events are more likely to occur simultaneously in many scattered areas or in overlapping time frames across a watershed, causing secondary impacts and lingering effects with greater potential to affect larger portions of a stream network at any given time. In addition, anthropogenic disturbances may more frequently result in temporary barriers to fish movement, which could reduce the existence of or limit accessibility to refugia (Bash et al. 2001).

Turbidity impacts may not be the only source of stress to aquatic life in a system (Bash et al. 2001). The potential of an activity to increase turbidity should be evaluated in the context of other environmental stressors that may be present in the system (Bash et al. 2001), such as elevated water temperatures, excessive flow variation, reduced cover or reduced prey resources. It is also important to note that much of the research on turbidity impacts on salmonids has occurred in controlled laboratory settings and that extrapolation to complex natural systems may require consideration of other factors such as predator and prey abundances (Bash et al. 2001).

8.1.6.2 Altered Pollutant Loading

Water quality may be impacted by inputs of metals or organics associated with HPA-permitted activities. Much of the research has focused on smaller projects and little is known about the potential impacts of large projects (>100 pilings) involving the use of treated wood piles in aquatic settings (Poston 2001). It is conceivable that many smaller projects using ACZA- and CCA Type C-treated wood products, if close enough to one another both spatially (with respect to leachate dilution rates) and temporally (in terms of diminishing rates of leaching), could produce effects similar to those of larger projects (Poston 2001).

It is well known that PAHs and metals are significant components of urban stormwater. The risks of PAH and metals contamination from treated wood products should be considered in the context of background PAH and metals concentrations in the surrounding water and sediments, as well as in the context of potential PAH loads from other point and nonpoint sources, such as industrial outfalls and stormwater runoff (Menzie et al. 2002). This may be a difficult undertaking, given that few data are available on the background PAH and metals concentrations in most water bodies and their sediments (Poston 2001).

Studies have shown that marine areas with shoreline structures in areas with poor tidal exchange or freshwater areas with poor water circulation are characterized by higher concentrations of pathogens and PAHs than areas with elevated water circulation (Bordalo 2003), but there is no clear pattern indicating that shoreline modification structures consistently degrade water quality in every application.

8.1.7 Ecosystem Fragmentation

8.1.7.1 Habitat Loss

Disturbance of streambeds may result in loss of habitat. The significance of the loss depends on the size of the watershed, whether the loss is permanent or temporary, the amount of habitat cumulatively lost, and the significance of the habitat lost to the population (i.e., spawning, rearing, or migration habitat). The loss of streambed habitat could affect the prey base available for juvenile and adult resident fish species by reducing the abundance of benthic macroinvertebrates. Benthic macroinvertebrates, by definition, inhabit the stream bottom;

therefore, modification of the streambed will most likely have some effect on the benthic macroinvertebrate community (Waters 1995). Benthic macroinvertebrate populations generally recolonize disturbed areas quickly (within 45 days), but this recovery time may be extended when repeated disturbances occur (e.g., NMFS 2003).

Both permanent and temporary losses of benthic macroinvertebrates are likely to occur as a result of new construction or expansion of existing structures; changes in the representative species assemblages as a result of associated changes in hydraulics and habitat conditions within affected reaches are also likely. It is difficult to ascertain the cumulative impact of changes to benthic macroinvertebrate populations or species diversity and subsequent changes to fish populations or habitat occupancy that may result. Permanent loss of benthic macroinvertebrate numbers or a decrease in species diversity due to permanent loss of habitat will affect foraging opportunities for fish and could affect the population numbers within stream reaches; this may be measurable over time at the watershed scale depending on the size of the watershed and amount of habitat permanently lost.

8.1.7.2 Freshwater Habitat Accessibility

The cumulative impacts of reduced habitat accessibility can have significant impacts on the distributions of potentially covered species. The cumulative loss of access to floodplain and off-channel habitats can significantly reduce availability of required refuge, rearing, and spawning habitats. Such cumulative habitat accessibility losses would impact all freshwater species, but especially salmonids, lampreys, and Olympic mudminnow.

8.2 Cumulative Effects Sorted by Activity Type

8.2.1 Marinas, Terminals, Overwater Structures and Shoreline Modifications

As marina/terminal structures interact with other development in a given area, impacts accrue, producing a net loss in vegetation production and a concomitant reduction in epibenthic and benthic nearshore habitat. The type and extent of each of these alterations depends on specific site characteristics, structure types, design of the structures, and construction materials.

Marinas and large terminal structures produce cumulative effects by virtue of the fact that they contain multiple structures, operations and consequent impacts to shoreline processes functions. Marinas and terminals often include associated shoreline modifications such as breakwaters, jetties, or bank protection. Overwater structures (such as single family residential docks) may have direct and indirect localized effects on habitat, and multiple structures may produce cumulative effects similar to those from marinas and terminals. One study (Nightingale and Simenstad 2001b) specifically discusses the cumulative impacts of overwater structure construction.

8.2.1.1 Construction and Maintenance Activities

The WDNR Shorezone Inventory (2001) reports a total of 716 large marinas (i.e., over 100-foot slips) along Washington's marine shoreline. Several of the ferry terminals are currently slated for expansion or improvements, including: Anacortes, Bainbridge Island, Eagle Harbor, Edmonds, Mukilteo, Port Townsend, Keystone, and Seattle (Coleman Dock) (WSDOT 2007). It is also reasonable to assume that most of the other facilities in state waters will require at least some maintenance activities if not more substantial improvement activities in order to maintain operations.

8.2.1.2 Water Quality Modifications

Water quality impacts are dependent upon the level of use and design of the marina, terminal, or shoreline modification structure; the hydrography and geomorphology of the surroundings including the level of tidal exchange for structures located in marine areas; as well as proximity to other affected habitat.

Ferry terminals do not affect water circulation to the degree that marinas do, but both marinas and ferry terminals are associated with docks, shoreline protection structures, and elevated vessel activity. The cumulative impact of these facilities may be manifest through the increased occurrence and degree of usage of individual facilities. Increased dock usage levels can pose risks to water quality through the introduction of sloughing bottom paints, vessel engine exhausts, fuel spillage, overboard sewage discharge, paint and cleaning product contamination, and introduction of contaminants from automobile traffic and asphalted parking lots adjacent to a marina via stormwater (USEPA 2001).

8.2.1.3 Riparian Vegetation Modification

Marinas/terminals and associated shoreline modifications serve as transportation hubs and pathways for the movement of passengers and freight. As such, they require substantial impervious surface and encourage further shoreline development.

The Seattle-Tacoma area is an area of intense urbanization. The WDNR Shorezone Inventory (2001) reports that out of a total of 716 large marinas (i.e., over 100-foot slips) along Washington’s marine shoreline, 41 percent are in this Seattle-Tacoma area. In contrast, out of the total of 3,000 miles of marine shoreline in Washington State, the Seattle-Tacoma shorelines represent less than 5 percent. In this particular area, marinas and ferry and terminal areas are typically denuded of riparian and shoreline vegetation.

8.2.2 Bank Protection, Stabilization and Shoreline Modifications

Literature reviews conducted by Canning and Shipman (1994), MacDonald et al. (1994), and Zelo et al. (2000) conclude that shoreline armoring does have cumulative effects and that while impacts of individual structures may not be substantial, the aggregate of several structures may be significant where littoral sediment supplies, transport, and beach substrate are altered. Reynolds (1983, in MacDonald et al. 1994) concludes that the cumulative effect of structural response to beach erosion is the escalation of engineered structures and the consequent loss of beach. Silvester (1977, in Gabriel and Terich 2005) found that the littoral energy applied to the sediment doubled in the presence of seawalls, which lead to increased scour downdrift. In this way, the cumulative effect of an incremental increase of seawalls would not necessarily be a linear addition of effects but could be interactive and synergistic.

The cumulative impacts of bank protection structures are particularly important because:

1. The structures are often constructed to counteract or curtail natural habitat-forming processes.

2. The shorelines of Washington State's water bodies are often lined with numerous small parcels that individually may produce only minor impacts, but cumulatively may be significant.
3. As noted by Nightingale and Simenstad (2001b), the bathymetry of Washington's inland marine waters is that of a fjord surrounded by a narrow vegetated habitat, which essentially concentrates the zone of impact.

8.2.2.1 Channel Processes and Morphology

The fact that bank protection projects typically work in direct opposition to natural channel processes results in the potential for significant cumulative impacts. As evidenced by the listing of several salmon populations as threatened or endangered under the ESA, significant habitat alterations, including bank protection, can cumulatively generate lasting impacts that have great implications for population viability.

8.2.2.2 Hydraulic and Geomorphic Modifications

Numerous studies throughout the world have documented the cumulative hydraulic and geomorphic impact of shoreline hardening and maritime activities on the coastal ecological communities (Byrnes and Hiland 1995; Guidetti 2004; Meadows et al. 2005; Penland et al. 2005; Wijnberg 2002). Because of the nature of these studies, they have not focused strictly on one activity type. The primary impacts addressed by these studies include the disruption of littoral processes as well as hardening of the shoreline and consequent coarsening of the substrate, although other maritime activities likely play a role as well (e.g., fishing) (Blaber et al. 2000; Guidetti et al. 2005). Although the notion of cumulative environmental impacts has been hypothesized to be important in the marine environment in Washington State (e.g., in Puget Sound [Gelfenbaum et al. 2006]), there have been no systematic, peer-reviewed studies that have investigated the phenomenon in Washington waters. Despite this lack of local data, the sum of work performed outside of Washington State documents a general pattern of ecological change due to the construction of shoreline protection structures. In particular, the switch from biological communities preferring soft substrates and relatively quiescent conditions to those preferring higher wave-energies and harder substrates is almost always identified (Guidetti 2004; Guidetti et al. 2005; Meadows et al. 2005). For the outer coast of Washington, the coast of California provides a relevant analog of patterns of ecological changes due to the construction of shoreline protection structures. Although development has been more recent, there has been some documentation of the general hardening of shorelines in California. For instance, Wasson et al. (2005) described the increased prevalence of coarse substrate-dependent (invasive) communities on shoreline works.

Although many of these locales are superficially different from Washington State, some of these studies are particularly germane to anthropogenic environmental degradation. In particular, the paraglacial landscape of the Great Lakes and the Adriatic Sea provide similar templates to the geomorphic variables responsible for nearshore change in the Puget Sound, the Strait of Juan de Fuca, and the large lakes of western Washington (Finlayson 2006). These areas have also been developed for a much longer time (in the case of the Great Lakes, hundreds of years; in northern Italy, millennia), such that the cumulative effect has been made much clearer. For instance, Bearzi et al. (2004) documented the historical loss of marine mammals in the Adriatic and attributed the loss to human activities (in general).

Although there are no known studies specific to the cumulative effects of modifications associated with the construction and operation of marinas, terminals or shoreline modifications such as jetties or groins, a few studies have documented the cumulative effects of bank hardening on the riverine ecology of large navigable channels.

- Riprap stabilization of one 15.5-mile (25-km) reach of the Sacramento River was cited as the primary cause of salmon decline in this river due to the loss of spawning gravels previously supplied from bank erosion (Buer et al. 1984; Shields 1991).
- In a comprehensive study of the historical decline of coho salmon smolt production in the lower Skagit River, Washington, Beechie et al. (1994) found that hydraulic modification from the combined effects of levee construction, bank hardening, and dredging accounted for 73 percent of summer habitat losses and 91 percent of winter habitat losses.
- The cumulative effects of bank hardening and historical removal of riparian forests throughout the lower Skagit River, Washington, have prevented wood recruitment from the natural processes of channel migration, thereby reducing the delivery of large wood to the estuary (Collins 2000). The loss of this wood can disrupt food webs for juvenile salmonids in estuarine marshes.
- Bank stabilization along 25 percent of the 99-mile (160-km) Garrison Reach of the Missouri River in North Dakota nearly eliminated the positive effect of riparian forest on the density of instream woody debris (Angradi et al. 2004).

8.2.3 Water Crossings

No studies that specifically address the cumulative impacts of water crossing structures were located. However, general discussions of cumulative impacts on channel hydraulics and substrates are pertinent.

8.2.3.1 Channel Hydraulics

Bates (2003) cites the importance of proper structure siting and land use practices for minimizing the cumulative impacts of culverts. Bates (2003) recommends as most effective those solutions that avoid the need for a water crossing structure, and states that impacts can be minimized by "consolidating water crossings; employing full-floodplain spanning bridges, by simulating a natural channel through culverts; or removing water crossings."

Water crossings entail an element of risk that catastrophic failure may occur, with dire consequences for affected animals and habitat. Debris flows, dam-break floods, footing scour, and channel avulsions are all relatively common failure scenarios in Washington. Although such failures are not and cannot be authorized by issuance of an HPA, there is a calculable risk that any water crossing structure will fail within a given time frame. The incidence of such failures is presumably a function of the number of structures authorized and the flood event design standard used. In general, the larger and more robust the structure the more tolerant it is of large scale events. Many bridges and culverts were installed to pass the 25- or 50-year event (current standards require passing the 100-year event). Events larger than the design can result in simultaneous failure of many "underdesigned" facilities in the watershed. This constitutes a cumulative impact from the construction of water crossing structures. The impacts of such failures have been observed periodically in Washington in association with major weather/flooding events. Observed impacts include bank and channel erosion, sedimentation of stream gravels and pools, and loss of redds through scour or suffocation. These impacts are somewhat ameliorated by more long-term consequences of the event, which can include beneficial changes such as increased channel complexity, accumulation of debris jams, and introduction of spawning-size gravels. Data are not currently adequate to determine the full effect of such flood events on potentially covered species, and no literature addressing this risk and its magnitude within Washington State was found.

8.2.3.2 Substrate Modifications

No studies were found analyzing the cumulative impacts of substrate modifications in association with water crossings. However, since substrate modification largely consists of replacing habitat with nonhabitat in the form of fills, piers, piling, or culverts, it follows that cumulative impacts are roughly proportional, at a watershed scale, to the fraction of aquatic habitat lost to substrate modification.

8.2.4 Channel Modification

In general, as the number of channel modifications increases in a given area, impacts will accrue producing a net loss in riverine, lacustrine, and/or marine habitat. The type and extent of each of these alterations depends on specific site characteristics and the subactivity types.

8.2.4.1 Dredging

The cumulative effects of dredging have been documented by a number of studies (Byrnes et al. 2004; Cooper et al. 2007; Erftemeijer and Lewis 2006). These studies have documented that repeated dredging reduces the prevalence of seagrasses and macroinvertebrates; however, impacts on invertebrates conditioned for disturbance can respond quickly and recover significant populations of benthic invertebrates (Bolam and Rees 2003; Robinson et al. 2005). However, even in the most optimistic studies, the major cumulative impact of dredging is lower seabed productivity and diversity (Robinson et al. 2005).

Dredging from the lower Columbia River since at least 1904 has had cumulative effects on the sediment budget of the river and the littoral cell extending 160 km (100 miles) along the Pacific coast, from Point Grenville, Washington, to Tillamook Head, Oregon. The coast along this cell has experienced accelerated erosion, with recent coastal erosion in the Westport area alone costing $30 million in repairs (Kondolf et al. 2002).

8.2.4.2 Gravel Mining and Scalping

The greatest effects of instream gravel mining, bar scalping, and pit mining may be considered as cumulative because they may become obvious only over time and extend beyond the limits of the mining site itself (Kondolf 1997). Moreover, the effects of one mining activity may interact with nearby mining, yielding a net cumulative effect not apparent from a single mining action (Kondolf et al. 2002). Individually subtle effects of gravel mining can become more visible and serious through the propagation of channel incision upstream and downstream of such activities (often for distances of kilometers) on mainstem and tributaries and through the coalescing of incision effects.

Channel incision caused by the cumulative effects of gravel mining causes lowered alluvial groundwater tables, desiccation of riparian and floodplain vegetation, reduced channel-floodplain interactions, and the elimination of processes of channel migration and the consequent creation of habitat. Any extraction of gravel from the channel bed or floodplain interrupts sediment transport continuity and represents a net loss in the sediment transport budget, thereby inducing channel instability and reducing the volume of downstream bars (Dunne et al. 1981).

Because the direct and indirect effects of bar scalping are far-reaching, the cumulative effects of numerous bar-scalping operations can result in long-term habitat degradation. For example, Dunne et al. (1981) documented cases in which the current channel was abandoned and a former channel adopted following bar scalping. Bar scalping has also been shown to eliminate side channels, which are important habitats for juvenile salmonids (Pauley et al. 1989; Weigand 1991). Bar scalping on the Puyallup, Carbon, and White rivers from 1987 to 1988 reduced the mean side-channel riffle habitat area from 1350 to 930 cubic yards and mean side-channel glide and pool habitat area from 1550 to 0 yards at treatment sites, while the representative habitat areas increased or remained unchanged at control sites (Weigand 1991).

Small-scale extractions are often viewed as having only small, insignificant impacts. However, a small extraction on a small stream can take a large fraction of the annual sediment load, and multiple small extractions on a larger stream can add up to be equivalent to a large proportion of the total load. Even when the extractions are small, they can add up to have a significant cumulative effect on channel form, especially in small channels, where the sediment load would be naturally low (Kondolf et al. 2002).

8.2.4.3 Sediment Capping

Although numerous sediment capping projects are seldom performed in any one area, they are typically performed in marine and freshwater harbors that have been impacted by previous industrial activities. Sediment capping activities can therefore contribute to the cumulative effects of numerous, related types of industrial cleanup activities. These cumulative effects could include the loss of nearshore habitats, habitat fragmentation, and the displacement of endemic species as a result of large-scale modifications to substrate composition and bathymetry. Some of the cumulative effects of sediment capping can be observed from studies examining the cumulative effects of multiple beach nourishment projects. Beach nourishment involves the rapid deposition of large quantities of sand and because of this, the impacts associated with the work are similar to those impacts that are associated with sediment capping.

Peterson et al. (2006) documented the loss of benthic macroinvertebrates on a stretch of beach in North Carolina from a number of small beach nourishment projects. Several earlier studies have shown that invertebrates can be harmed by nourishment projects (Diaz et al. 2004; Peterson et al. 2006; Rakocinski et al. 1996), but Peterson et al. 2006 were the first to show that the cumulative damage could occur due to multiple ongoing projects, and overcome the rapid recolonization typical of invertebrates. This same process of reburial before invertebrate recolonization could occur if a sediment cap was successively maintained or if multiple caps were placed adjacent to one another.

8.2.4.4 Channel Creation and Alignment

Although numerous stream restoration and channel creation projects have been completed over the past decade, the cumulative effects of these projects has not been adequately assessed by the scientific community. In general, the cumulative effects of multiple channel creation and alignment projects that fall short of rehabilitating degraded conditions are likely to result in the loss of native habitat for many HCP species. For instance, the listing of several salmon populations as threatened or endangered under the Endangered Species Act has been linked to (among other things) the widespread loss of spawning and rearing habitat resulting from channel modifications throughout the region (Montgomery et al. 2003).

8.2.5 Fish Passage

Fish passage projects (with the exception of specific classes of weirs) are intended to improve the condition of fisheries resources by restoring fish passage to mitigate the effects of man-made perturbations on the environment. While the benefits of providing fish passage are clear and measurable, fish passage projects may produce unforeseen consequences.

The majority of the negative effects associated with fish passage activities occur as a result of two discrete impact mechanisms: construction and maintenance; and subsequent changes resulting in ecosystem fragmentation. Other impact mechanisms, such as hydraulic and geomorphic modifications and effects on aquatic and riparian vegetation, are expected to be minor in comparison. Construction-related effects are short term, while effects on ecosystem fragmentation are long term and more pervasive. Consequently, cumulative impacts associated with construction activities are unlikely to occur unless multiple projects are being constructed simultaneously and in proximity to each other. In contrast, the cumulative effects of altered fish passage and the upstream transport of allochthonous nutrients have significant potential for cumulative effects on ecosystem structure and function.

Restoration of access to historic habitats is widely recognized as a key element in strategies for the restoration of native aquatic fauna (Roni et al. 2002). However, it must also be recognized that fish passage projects may not equally restore full access to all migratory species that historically utilized the affected habitat. Projects intended to block upstream dispersal of non-native species may broadly affect the migration of nontarget species. The cumulative effects of these types of perturbations are twofold. First, altered passage conditions may impose selection pressures on HCP species, altering the genetic diversity of the affected population. Second, altering the range, abundance, and diversity of species able to access historic habitats is likely to alter the adaptive trajectory of the ecosystem in ways that are difficult to predict.

The cumulative effects of the fish passage projects are on balance expected to be beneficial to HCP species as a whole. However, some detrimental effects may occur as a result of the broad application of this activity type across the landscape due to the effects of stressors that are difficult to predict and/or assess.

8.2.6 Fish Screens

Fish screens are intended to minimize adverse effects from water withdrawals on aquatic species. Screening of diversion and intake structures has been broadly imposed as a matter of management policy across the landscape. This policy decision represents a defensibly precautionary approach to water resources management. While fish screens in many cases demonstrably reduce entrainment mortality, they may also impose unforeseen or unavoidable effects that must be considered.

The majority of the negative effects associated with fish screens occur as a result of construction and maintenance, and operations. Cumulative impacts by other impact mechanisms, such as hydraulic and geomorphic modifications, are expected to be minor in comparison. Construction-related effects are short term, while operational effects are long term but less intensive on an individual screen basis. Cumulative impacts associated with construction are unlikely to occur unless multiple projects are being constructed simultaneously and in proximity to each other.

Fish screens are a necessary impact minimization technology used to limit the effects of dams, diversions, and intake systems. When properly employed, they can reduce mortality caused by entrainment into intake and diversion systems. Such mortality can have significant implications for populations of many HCP species. From this standpoint, the positive impacts of fish screens outweigh the negatives. However, fish screens may impose some detrimental cumulative impacts when numerous screens are used across the landscape. The extent of these effects is difficult to predict and/or assess. Examples of potential cumulative impacts include:

- Delayed migration: Multiple off-channel screen systems arrayed along a stream corridor could conceivably significantly delay migration, presenting a number of adverse consequences. In the case of upstream migration, screens with accessible bypass channels and/or high-flow bypass discharges may cause confusion regarding the migratory corridor, slowing migration or attracting fish up blind channels. Upstream migrant juveniles may be repeatedly drawn into bypass systems and discharged downstream, slowing migration to desirable habitats. In the case of juvenile downstream migration, the bypass system must provide suitable sweeping flows to avoid fish delay at the bypass structure and loitering in the diversion.

- Delayed or modified dispersal: The dispersal of weak-swimming or planktonic fish and invertebrate larvae may be affected by the operation of fish screens. Organisms drawn into screen systems may be effectively bypassed and removed, but could be discharged to environments that are unfavorable for rearing, or dispersal to favorable habitats may be delayed by exposure to multiple screens.

- Nonlethal impingement, bypass entrainment: Juvenile fish may experience nonlethal impingement on in-channel and off-channel screen surfaces, followed by escape, or stress from entrainment through high velocity bypass systems and discharge to the stream channel. While the effects of temporary impingement or bypass entrainment from a single screen may be small, the combined effects of incremental migration delays, stress, and injuries from encountering many fish screens may be cumulatively significant.

- Effects of multiple screens on channel geometry and habitat complexity: In small streams, or in instances where bypass systems represent a significant component of stream length, off-channel screens incorporating bypass channels have the potential to exacerbate vegetation encroachment induced by changes in base flow conditions. This can in turn result in changes in channel geometry, flow velocity, substrate conditions, and resulting effects on habitat complexity in the affected bypass reach. Multiple off-channel screens distributed throughout a stream system present some potential for cumulative effects on channel form. These changes could have implications for the survival, growth, and fitness of HCP species.

Fish that are migratory or that are dependent on dispersal throughout the affected habitat types are most likely to experience cumulative impacts from fish screens. Anadromous and migrant resident salmonids are a prime example. The potential for entrainment-related losses of salmonids was a primary concern driving the widespread use of fish screens on agricultural diversions in the Columbia River basin and elsewhere. Most fish screens in Washington State are focused on avoiding adverse effects on salmon. Because of their migratory nature, however, salmon have the potential to be exposed to many fish screens throughout their life history. As such, they are likely to be exposed to impingement, migration delay, entrainment through bypass systems, and other related stressors several times. Individually, these stressors may not impose noticeable effects on survival, growth, and fitness, but the cumulative effects of multiple exposures could be significant.

Other HCP species potentially affected by the cumulative effects of fish screens include white sturgeon, mountain suckers, lamprey, and dace. Lamprey, suckers, and sturgeon are migratory species and are therefore potentially exposed to multiple fish screens during their life history. For lamprey, many screens designed to protect salmonids may not be adequately protective of weak-swimming amocoetes. Sturgeon larvae may depend on dispersal to nearshore and inundated riparian habitats for successful recruitment, exposing them to screen-related stressors. Fish screens may not provide adequate protection for these life-history stages. Dace, while not explicitly migratory, may depend on dispersal between suitable habitats to maintain population diversity. The cumulative effects of multiple fish screens could potentially limit the effectiveness of these dispersal mechanisms, affecting gene flow between populations and colonization of suitable habitats. Freshwater mussel species may be subject to cumulative indirect effects from cumulative effects on host fish distribution and abundance. Fish screens may block dispersal of some freshwater invertebrates.

In marine systems, fish screens may similarly help to limit entrainment-related losses. However, it is difficult to avoid entrainment of species with planktonic eggs and larvae, such as hake, cod, and Olympia oyster, when these life-history stages are present. These entrainment-related effects are more the result of intake operation than the effects of the screens, and better represent the cumulative effects of the flow control structure. However, these effects also reflect fish screen design limitations. Knowledge of planktonic egg and larval sensitivity to entrainment, and technologies suitable for limiting adverse effects, may not be available for all potentially affected HCP species. Currently available screening technologies are sensitive to biofouling and require consistent maintenance to remain effective.

This assessment of effects considers the effects of fish screens relative to a natural system baseline. The cumulative effects of fish screens are, on balance, likely to be of lesser magnitude than the impacts of multiple unscreened intakes and diversions. In a similar fashion, the cumulative effects of fish screens are likely to be small relative to the combined effects of multiple water withdrawals on habitat capacity and productivity.

8.2.7 Flow Control Structures

Flow control structures have cumulative effect ramifications. In general, as the number of flow control structures increases in a given area, impacts accrue that increase habitat loss, alter the flow regime, and shift the composition and diversity of species.

8.2.7.1 Dams

Cumulative effects from dams are well known. The presence of a dam alters stream temperatures, dissolved oxygen concentrations, nutrient loading, natural sediment transport, channel geometry, flow regime, habitat connectivity, and changes in species composition that result in cumulative impacts on HCP species. If only one of these impacts were realized, the impacts may be minor; however, taken in concert, these impacts can overwhelm some species and negatively affect their survival, growth, or fitness.

A series of dams on a given river or river system will compound difficulties for migrating species. For example, in a study on the Columbia River, only 3 percent of tagged Pacific lamprey reached the most upstream site of a series of 3 dams (Moser et al. 2002). However, 40–50 percent of them passed over the lower dams, indicating that as the number of structures increase, successful migration to the upper reaches of a watershed will decrease. In addition, declines in Columbia River salmon and steelhead were the result of cumulative impacts from nine hydropower dams on the mainstem, each contributing 2–20 percent of the overall loss (Williams and Thom 2001). From a geomorphic standpoint, a series of dams will compound sediment losses to downstream coastal systems, exacerbating beach loss and erosion. In terms of eutrophication, nutrient loading from several dams may lead to the development of low-oxygen zones in coastal areas.

In many cases, these cumulative impacts extend well beyond the location of the dam. For example, in the highly impounded Columbia River watershed, effects from dams high in the watershed will translate to the marine environment. On the Olympic Peninsula, the Elwha River dams are causing significant beach losses from sediment accumulation in reservoirs behind two large dams (DOI 1995).

8.2.7.2 Weirs

The cumulative effects from weirs on HCP species are similar to those described above for dams. However, these impacts are lessened due to the scale of weir projects and the fact that these are overflow structures with fewer impacts on the downstream water quality.

8.2.7.3 Dikes and Levees

Dikes and levees alter channel geometry, flow regime, and habitat connectivity, contributing to cumulative effects on HCP species. As with most flow control structures, the more levees constructed in a given area, the more fragmentation of the habitat will result. In addition, the presence of several dikes and levees in a watershed will compound the effects of flow changes downstream. For example, a given increase in flood flow from one channelized reach flowing into another such reach will increase the peak flood flows because there will be an increased amount of disconnected floodplain area. Normally, the floodplain would be able to absorb these flood flows and to minimize the downstream effects of peak flows.

8.2.7.4 Outfalls

Limited information is available regarding the cumulative impacts of hydraulic and geomorphic modifications associated with outfall structures. However, a string of poorly designed outfalls could easily starve a shoreline of sediment, just as groins have done in other parts of the world (Byrnes and Hiland 1995). If riparian vegetation is removed during the construction of an outfall, changes in temperature and solar input will be magnified as more such outfalls are placed within a watershed. Similarly, water quality degradation from a single outfall might be minimal;

however, the more outfalls that are located in a single stream reach, the more likely it is that impacts will occur on HCP species from metals toxicity, low oxygen, and exposure to organic pollutants.

8.2.7.5 Intakes and Diversions

As with outfalls, limited information is available regarding the cumulative impacts of hydraulic and geomorphic modifications associated with intakes and diversion infrastructure. Intakes have specific modifications that could have significant cumulative impacts. In particular, their design does not adequately account for the entrainment of spawn and drifting larvae along river system. This type of cumulative impact has been described in terms of large-scale hydropower planning in Europe (Larinier 1998). If riparian vegetation is removed during construction of an intake, changes in temperature and solar input will be magnified as more outfalls are placed within a watershed. In addition, as more diversions are located within a watershed, the more of an impact will occur on the downstream flow regime. An extreme situation could result in a completely dry channel from multiple diversions, which would make the river reach unusable for HCP species.

8.2.7.6 Tide Gates

Tide gates are often constructed in areas converted for agriculture. As a result, irrigation that routes diversions and runoff from fields through outfalls is likely. The cumulative effects from tide gates are similar to those for a dam. Because tide gates block migration and tidal flows, the more tide gates are present in a given area, the more impacts on HCP species would occur. These cumulative impacts translate to water quality modifications as well. Changes in salinity are a fundamental impact from the presence of a tide gate. The more tide gates there are in a system, the greater this impact will become. Changes in salinity are important to migration patterns and to provide suitable habitat for species that use these areas. In addition, metals toxicity from altered flow, oxidation of marsh soils, and changes in pH will be compounded if several tide gates are located within a given area.

Cumulative effects from saltwater intrusion into the riparian zone may also develop. In Australia, it was observed that saltwater seepage into the surrounding groundwater occurred. Depending on soil properties, this seepage was less than 33 ft to more than 262 ft (10 m to more than 80 m) from the impounded area (Johnston et al. 2005b). This saltwater intrusion could have a devastating effect on riparian vegetation, leading to increased bank failures, increased temperatures, and reduced nutrient cycling.

8.2.8 Habitat Modifications

Each of the habitat modifications has cumulative effect ramifications. All of the habitat modification subactivity types except beaver dam and large woody debris removal aim to restore habitat function to a condition which supports a sustainable, diverse, and abundant array of native flora and fauna. Consequently, the cumulative effect of these activities is to create diverse, productive, and connected habitat mosaics which bolster the HCP species and ameliorate human impact on the environment. The full potential of these habitat modifications may not be realized until the application of the activities becomes so wide spread as to minimize the existence of the degraded habitat which today serves to fragment aquatic ecosystems across the state.

The majority of the negative impacts associated with habitat modification activities occur during the construction phase. Because the construction phase is of a short duration, these impacts tend to be ephemeral. Consequently, cumulative impacts associated with construction phase activities are unlikely to occur unless multiple projects are being constructed simultaneously and in close proximity to each other. As this is an unlikely scenario, the cumulative impacts of construction-related activities are not discussed in this section.

8.2.8.1 Beaver Dam Removal/Modifications

Before European settlement in North America, beaver populations were estimated to be between 60 and 400 million individuals (Seton 1929 in Naiman et al. 1988). Today *Castor* spp. are estimated to number between 6 and 12 million (Ringelman 1991). This represents a significant reduction in the number of impoundments which serve as habitat for beaver. The reduction in hydraulic and resource retention provided by beaver impoundments has been partially counter-balanced by the impounding of the nation’s waterways for resource extraction and recreational purposes. Consequently, humans have unintentionally mitigated for a portion of the negative impact of beaver dam removal on carbon, nutrient, and water retention in watersheds.

The potential for cumulative impacts associated with beaver dam removal cannot be assessed without accounting for the cumulative impacts associated with the elimination of other barriers such as dams, diversions, and culverts. The combined effects associated with these activities will act to reduce system retentiveness and thus decrease secondary production. Additionally, a reduction in lentic habitat and access to floodplains for cover, rearing, holding, and foraging will impact numerous aquatic species. These cumulative impacts will be realized unless parallel habitat modification activities are enacted which increase retention, floodplain connection, and slack water habitat. Many of the activities discussed below will serve these functions.

8.2.8.2 Large Woody Debris Placement/Movement/Removal

The cumulative effects of reintroducing wood to rivers, streams, and shorelines is generally viewed as a positive step toward offsetting the habitat degradation resulting from the effects of historical logging, river snagging, and splash damming. Most riparian forests in Washington currently lack trees large enough to serve as key members in the formation of stable logjams (Beechie et al. 2001; Collins et al. 2002). Thus, engineered jams with large key members will serve a vital function as points of stability within fluvial systems. The cumulative effects of both wood reintroduction and the natural recovery of riparian forests include an increase in habitat diversity (Bryant and Sedell 1995; Warren and Kraft 2003), the reconnection of floodplain and off-channel habitats (Abbe and Montgomery 1996; Fetherston et al. 1995; Warren and Kraft 2003), the moderation of punctuated sediment inputs to river systems due to sediment retention (Massong and Montgomery 2000), and an increase in the frequency and spatial extent of habitat-forming channel migration (Brummer et al. 2006).

The increase in hydraulic roughness and resident time of water following the reintroduction of wood to rivers, streams, and shorelines can have positive cumulative effects on water quality and nutrient retention. Decomposition and grazing of coarse particulate organic matter trapped with sediment behind accumulations of woody debris has been found to increase the retention of dissolved organic carbon (Lampert 1978; Sinsabaugh et al. 1994). Organic material and sediment storage resulting from increased wood loading should also promote nutrient retention (Mulholland et al. 1985) and increased uptake of phosphorus (Ensign and Doyle 2005; Valett et al. 2002). The more convoluted flow paths and more organic fines in more numerous pools provided by wood will also create increased pollutant retention while increasing ecosystem productivity (Ensign and Doyle 2005). The result will be decreased pollutant loadings to downstream systems and increased stream carrying capacity.

The cumulative effects of large woody debris removal are well known because our present day waterways have been shaped by a legacy of large scale wood removal. The removal of LWD on the watershed scale disconnects channels from floodplains (Fetherston et al. 1995), promotes channel incision (Diez et al. 2000), reduces habitat complexity (Warren and Kraft 2003), and decreases organic matter retention and pollutant removal capacity (Ensign and Doyle 2005; Valett et al. 2002). If LWD removal cannot be avoided, then mitigation strategies should be employed to ensure that there is no net decrease of wood within the water body.

8.2.8.3 Spawning Substrate Augmentation

Spawning substrate augmentation is in most cases an ephemeral solution to a lasting problem. Degraded substrate in channels is usually associated with reduced sediment supply and/or flow alteration. Gravel augmentation does not address these issues but instead provides a remedy for the effect, while the cause (i.e., geomorphic and hydrologic processes) goes untreated. In this way, spawning substrate augmentation measures are by design short-lived. If the potential positive benefits of gravel augmentations are to be realized, then continual maintenance of the site is required. Maintenance may come in the form of passive or active gravel replenishment (Bunte 2004) and will be expensive, but the cumulative effects of continual replenishment (i.e., an active, well oxygenated, and dynamically stable riffle habitat) will be the only way to prolong the life of the project to a temporal scale that will benefit salmonid spawners and their off-spring through multiple life cycles. This suggests that isolated gravel replenishments which are not maintained may not meet the restoration goals and indeed, if improperly implemented, may cause more ecosystem harm than good.

8.2.8.4 In-Channel/Off-Channel Habitat Creation/Modifications

As with most fluvial restoration projects, the more widespread the application the more likely a measurable effect will be realized. One of the primary difficulties associated with assessing the impact of in-channel and off-channel habitat modification efforts is that the biotic response may be subtle and/or not measurable in the reach where the project was initiated. This helps explain the mixed results from numerous restoration monitoring efforts (Fausch et al. 1995; Larson et al. 2001; Pretty et al. 2003). However, as the number of successful in-channel and off-channel restoration projects increase, the likelihood of observing a measurable response also increases, (Korman and Higgins 1997). There are many factors which will determine the health of a fishery and many of those factors cannot be addressed on the reach scale. Consequently, the cumulative effect of restoration efforts in channels and floodplains will not be fully realized until whole watershed and marine life-stage problems are addressed.

8.2.8.5 Riparian Planting/Restoration/Enhancement

Riparian planting in highly degraded systems needs to be conducted within the context of larger watershed restoration efforts. Riparian rehabilitation efforts that create a corridor of improved habitat downstream of a degraded watershed may not ameliorate stream conditions (Teels et al. 2006). In a study of forest fragments in agricultural areas of the South Island, New Zealand, Harding et al. (2006) found that forest fragments of 5-7 ha, located in the lower reaches of the study catchment did not mitigate the negative effects of upstream agriculture on stream functioning. They concluded that fragment size (i.e., riparian forest length), riparian forest width and vegetation type, and fragment location in the catchment may have critical roles in enabling forest fragments to reset the negative impacts of agriculture. This study suggest that in highly impacted watersheds, the cumulative impact of multiple riparian planting projects is vital for the improvement of the stream and its biota and indeed, improvement may not be measurable until the cumulative effect of multiple projects is realized. However, in less impacted environments, riparian restoration may serve to create a continuous buffer between the uplands and fragile stream habitat. Many riparian planting impacts are subtle at the reach scale, but as riparian rehabilitation continues throughout a watershed the impacts will become more significant and measurable.

8.2.8.6 Wetland Creation/Restoration/Enhancement

Research has indicated that floodplain wetlands are most productive when hydraulic residence time on the floodplain is on the order of 2 to 10 days (Ahearn and Dahlgren 2005; Hein et al. 2004). Additionally, studies have indicated that when residence time on floodplains is below this threshold the floodplain becomes a net sink for algal biomass instead of a net source (Ahearn and Dahlgren 2005; Tockner et al. 1999). This suggests that small floodplain restorations may not increase food resources within the waterway and that restoration efforts should focus on large floodplains (or small floodplains which receive relatively low volumes of water). These studies also indicate that if small projects are constructed then the cumulative effect of numerous small projects is vital for optimal ecosystem functioning. Floodplain habitat has been reduced dramatically due to agricultural (Beechie et al. 1994) and urban development (USGS 1997). To restore the ecosystems services these habitats once provided is vital for the survival of native aquatic fauna including the 52 HCP species. The cumulative effect of numerous created or rehabilitated wetlands will be to restore this habitat on a scale that will measurably improve ecosystem health and watershed carrying capacity.

Coastal wetlands are the most common type of wetland in Washington (USGS 1997), but the areal extent and quality of these habitats have been impacted by anthropogenic activities. Coastal wetland rehabilitation and the increased rearing habitat availability associated with it will be vital to the rehabilitation of degraded fisheries in the state. The importance of this habitat for the restoration of the state’s fisheries came to light with the realization that density dependent mortality brought on by a limited availability of rearing habitat may be reducing the efficacy of other restoration efforts in upland waterways (Greene and Beechie 2004). Consequently, the cumulative effect of coastal wetland rehabilitation efforts may be not only to augment rearing habitat but also to improve the effectiveness of other restoration efforts which share the goal of increasing native fish populations.

8.2.8.7 Beach Nourishment/Contouring

Although there is limited information on the cumulative impacts of numerous small activities along a given long stretch of shoreline (Speybroeck et al. 2006), there has been recent work that has demonstrated the cumulative environmental impact of beach nourishment (Peterson et al. 2006). Peterson et al. (2006) documented the loss of benthic macroinvertebrates on a stretch of beach in North Carolina from a number of smaller nourishment projects. Several earlier studies have shown that invertebrates can be harmed by nourishment projects (Diaz et al. 2004; Peterson et al. 2000; Rakocinski et al. 1996), but Peterson et al. (2006) was the first show that cumulative damage could occur due to multiple ongoing projects, and could overcome the rapid recolonization typical of invertebrates. However, it is important to mention that these studies have been in open coast environments. These would be relevant to the outer coast or possibly the Strait of Juan de Fuca, but not within the confines of Puget Sound. No information exists regarding the cumulative impacts of beach nourishment on protected shorelines.

There is the potential that the cumulative effect of numerous augmentations of a sandy, pebbly nearshore typical in pre-development Puget Sound could bolster the populations of many HCP species, including forage fish and salmonids (Beamer et al. 2005). There is substantial anecdotal evidence that forage fishes will use placed materials for spawning (Penttila 2007). For example, a beach nourishment project in Silverdale Waterfront Park, Kitsap County, continues to be used by surf smelt. Further, shorelines that have been cut into man-made fill in Commencement Bay are also designated forage fish spawning areas (Penttila 2007). Consequently, the cumulative impacts of beach nourishment may be positive for some fish species, but more research is needed to inform future beach nourishment activities.

8.2.8.8 Reef Creation

There have not been enough artificial reefs created anywhere in the world to warrant a cumulative impact study. Given the limited number of HPAs issued and the relatively limited number of documented impacts of created reefs, it is unlikely that cumulative impacts of this subactivity are significant in Washington waters. However, if there were enough reefs created to generate a cumulative impact, it is likely that the nature of the impact would be an ecological shift from soft-substrate to hard-substrate organisms observed in the Adriatic associated with shoreline armoring (Guidetti 2004).

8.2.8.9 Eelgrass and Other Aquatic Vegetation Enhancement

Because there have been few eelgrass restoration projects in any environment, there have been no studies regarding the cumulative effects with regard to eelgrass restoration. However, based upon the importance of eelgrass to the life cycle of many HCP species, it is expected that if large-scale eelgrass planting were to occur, there would be substantial gains in several of the HCP species.

9 Potential Risk of Take

White papers prepared in 2006 (Bank Protection and Stabilization Structures, Overwater Structures and Non-Structural Piling, and Water Crossing Structures) and those prepared in 2007 (Channel Modifications, Fish Passage, Fish Screens, Flow Control Structures, Habitat Modifications, Marinas and Shipping Terminals, and Shoreline Modifications) used somewhat different methods and provided somewhat different levels of detail for estimating potential risk of take. Instead of revisiting the methodology and conclusions of the original white papers, this consolidation organizes the information to present general information, followed by information specific to a particular activity. It has been edited to minimize information that was repeated in several white papers. Unique tables have been retained from the original white papers. Specifically:

Section 9.1 consolidates the general discussion of the risk of take that was originally presented in the 2006 white papers.

Section 9.2 presents risk-of-take information specific to Bank Protection and Stabilization Structures, Overwater Structures and Non-Structural Piling, or Water Crossing Structures.

Section 9.3 consolidates the general discussion of the risk of take that was originally presented in the 2007 white papers.

Section 9.4 presents risk-of-take information specific to Channel Modifications, Fish Passage, Fish Screens, Flow Control Structures, Habitat Modifications, Marinas and Shipping Terminals, or Shoreline Modifications.

Discussions of “mechanisms of impact” are presented in the following order:

- Construction and Maintenance
- Operations
- Hydraulic and Geomorphic Modifications
- Ecosystem Fragmentation
- Riparian Vegetation Modifications
- Aquatic Vegetation Modifications
- Water Quality Modifications

However, not all of the original papers discussed all of the mechanisms of impact. If there was no discussion in the original paper, then that section is also missing from this consolidation.

9.1 General Risk of Take: 2006 White Papers (Bank Protection and Stabilization Structures, Overwater Structures and Non-Structural Piling, and Water Crossing Structures)

In its biological opinion for a bridge replacement on an Oregon river, NMFS (2006a) determined that the take caused by habitat-related effects of a project could not be accurately quantified (i.e., as a number of fish) because the relationship between habitat conditions and the distribution and abundance of those individuals in the action area was imprecise, and nearshore areas damaged by construction would require years to recover characteristics favorable for rearing and migration.

In such instances, NMFS uses the causal link established between the activity and the change in habitat conditions affecting the listed species to describe the extent of take as a numerical level of habitat disturbance, rather than stating an expected amount of take (50 Code of Federal Regulations 402.14(i)). NMFS (2006a) found that the best available indicators for the extent of take is the area of riparian habitat that will be permanently modified by the action, because it is directly proportional to long-term harm attributable to the project.

9.1.1 General Risk of Take from Construction, Operations, and Maintenance

9.1.1.1 Channel Dewatering

The primary risks of incidental take associated with channel dewatering result from the capture and handling of fish, the loss of small fish (particularly salmonid fry) that seek refuge in the substrate of the dewatered bed, and the use of pumped bypass systems. This conclusion is based on a review of several biological opinions.

Capture-related take, such as injury or mortality from electrofishing, varies from 2 percent (no distinction between injury and mortality) (NMFS 2006a) to 30 percent (25 percent injury and 5 percent mortality) (NMFS 2006b) of fish captured using electrofishing equipment. Some biological opinions did not distinguish between methods of capture (e.g., volitional movement of fish from the project site during slow dewatering, capture by seining or dip-netting, capture by electrofishing). One biological opinion estimated take due to stranding (i.e., fish not captured and removed and thus remaining in the work area to be dewatered) at 8 percent (NMFS 2006b). All such injury and mortality represent incidental take directly attributable to a project.

9.1.1.2 Noise

It is well established that impact pile driving can result in incidental take of fish. NMFS and USFWS biological opinions commonly identify such take and quantify it based on the area of habitat affected by sounds above the threshold levels and the duration of pile driving activities. However, the sound sensitivity of individual species is not well known. Species that lack internal gas-filled voids (such as swim bladders) appear to be less vulnerable to noise impacts than are fish, such as salmonids, that have gas-filled voids. For species without gas-filled voids, the risk of take is somewhat lower than it is for salmonids. Species-specific studies would be required to quantify the difference in risk.

Construction noise and activity associated with the La Conner Wharf and Float Project was thought to cause forage fish to temporarily leave the vicinity, which would temporarily reduce the prey base for Chinook and other fish species (NMFS 2005b); project effects on other predators, such as those eating young Chinook, were not addressed.

In the consultations reviewed, NMFS has not assigned quantifiable incidental take associated with construction noise other than pile driving.

9.1.1.3 Artificial Light

Incidental take for listed fish species as a result of artificial lighting has not been quantified in past biological opinions and corresponding incidental take statements. Studies indicate that artificial light has mixed effects; many of these effects are detrimental, and all of them represent a change from natural patterns of behavior. This suggests that although artificial light responses are unknown for most potentially covered species, there is a risk that nighttime illumination of the water surface may contribute to incidental take. Data are not adequate to define the magnitude of that risk; however, such impacts can generally be minimized.

9.1.1.4 Shading

Mechanisms of take related to shading include the following:

- The principal impact of shading is reduction in cover and productivity of underwater vegetation.

- Most studies of shading are focused on juvenile salmonids. However, available data on light sensitivity suggest that those impacts may reasonably be extrapolated to other small fishes, particularly nearshore marine species. For all other potentially covered species, almost nothing is known about sensitivity to shading.

- In freshwater environments that support significant bass populations, bass are effective, high-level predators that forage from under shade-producing structures.

- Migration of juvenile salmonids is sometimes impeded by shade-producing structures.

Shading from HPA-permitted structures could result in incidental take, if it is located where longshore movement of juvenile salmon might be affected. NMFS (2005b) identified incidental take of juvenile Puget Sound Chinook resulting from shading by a wharf and moorage float in Swinomish Slough, which may impede longshore movement during certain times of the day, and from a reduction in primary productivity and consequent reduction in food resources. Based on the shading footprint, the extent of take (identified as harm in this biological opinion) was determined to be any juvenile Puget Sound Chinook rearing and outmigrating within less than 1 acre around the structure.

Shade cast by HPA-permitted structures may also provide a site for predators to congregate. In a freshwater environment, NMFS (2006c) determined that the shading and structure resulting from the proposed expansion of a marina in the Columbia River will likely result in increased predation of listed juvenile salmon by a number of piscivorous fish species found in the area, although NMFS was unable to quantify the number of salmon expected to be killed.

9.1.1.5 Vessel Activities

Vessel activities may result in incidental take of potentially covered species via several mechanisms, including:

- Physical disturbance of sediment, organisms (Haas et al. 2002), and submerged vegetation through grounding or water turbulence caused by propeller wash, potentially resuspending sediment, physically dislodging vegetation and organisms, or damaging vegetation.

- Noise from vessel activity, which would most likely harm organisms by causing them to move from the affected area, potentially impairing foraging or reproductive activities or exposing them to increased risk of predation.

- Propeller wash-entrained air bubbles that combine with turbidity increases from disturbed sediment, with the potential consequences resulting from increased turbidity and from decreased light availability.

9.1.2 General Risk of Take from Hydraulic and Geomorphic Modifications

9.1.2.1 Channel Hydraulics

Impacts to potentially covered species may result when a vulnerable life-history stage of a species is exposed to an impact directly or indirectly caused by an HPA-approved structure. A direct impact arises when a structure alters the process of sediment transport, and an indirect impact arises when the change in sediment transport causes further habitat changes, such as bank erosion and loss of riparian vegetation.

Table 9-1. Potential Impacts of Changes in Stream Channel Hydraulics on Potentially Covered Species

Impact	Potentially Affected Species
No impact identified	Marine species or marine life stages of estuarine and anadromous species
Habitat destruction due to siting of structure	Species potentially occupying the affected stream
Embedding due to reduced sediment transport capacity or indirectly as a result of bank erosion	Species potentially occupying the affected streambed: gravel spawners and benthos
Scour due to locally increased transport capacity	Species potentially occupying the affected streambed: gravel spawners and benthos
Deposition downstream of scour areas	Species potentially occupying the affected streambed: gravel spawners and benthos
Loss of riparian vegetation due to bank erosion	Species potentially occupying the affected stream.

9.1.2.2 Habitat Loss

Habitat loss is the replacement of habitat with an artificial structure. Habitat loss includes temporary and permanent elements. Temporary habitat loss occurs when an area of habitat is inaccessible during or for a time following construction but becomes accessible within a reasonable time after construction, typically by the time work on the site concludes. Permanent habitat loss occurs when an area of habitat remains inaccessible for the service life of the structure or longer.

Permanent loss of channel habitat occurs when fill is placed in the channel or floodplain, usually in the form of fill intended to raise an area above the OHWL. Temporary channel habitat loss includes fill placement when it is not permanent, as well as channel dewatering resulting from the diversion of flow or flow exclusion via structures such as cofferdams. Habitat loss presents a high potential risk for incidental take; the risks are related to use of the habitat by potentially covered species, the area affected, the time frame during which the area is affected, and how potentially covered species respond to the loss or degradation of habitat.

The process of placing fill may cause harm to individual animals. However, in-water placement of fill generally requires isolating and dewatering the work site.

9.1.2.3 Embedding

Embedding gives the stream a relatively hard, impervious bed that provides a poor substrate for salmonid spawning, impairs hyporheic exchange, and provides poor habitat for benthic invertebrate infauna. Typically, several years of peak flow events are required after the fine sediment inputs have ended for the bed to be sufficiently reworked that embedding ceases.

Embedding is an issue principally in moderate-gradient channels that normally have a gravel or cobble bed, i.e., plane-bed and pool-riffle channels. Steeper channels have sufficient stream power that the “fines” consist of coarse sand and gravel, which do not substantially impair habitat quality. The less steep regime channels have fine-grained bed materials (generally defined as particles smaller than 0.04 inch [1 mm] in diameter) that are vulnerable to deposition rather than embedding. Embedding has a high risk of causing incidental take if it affects sediments used for spawning.

9.1.2.4 Scour

Scour is potentially an issue in all channel types, although it is most often a concern in plane-bed and pool-riffle channels, which have a relatively mobile bed. The term “scour” is usually used to refer to flow-driven excavation of the streambed, but it can also occur along stream margins and result in bank erosion. Scour that occurs in areas where it has previously been rare (for instance, due to the placement of HPA-permitted structures) may result in the loss of redds with eggs or of gravels containing fry or the benthic invertebrates that constitute part of the prey base for fish in the stream. Such scour events are particularly likely around hard structures placed in the channel, because shear stresses, and therefore energy available to mobilize sediments, are exceptionally high near such structures (Yager et al. 2004). The opposite effect is observed in the vicinity of aquatic vegetation (Bennett et al. 2002), raising the possibility that aquatic vegetation plantings may help to decrease scour around structures at some sites.

Scour can potentially result in incidental take via several mechanisms. Impacts to eggs and fry of potentially covered species (e.g. salmonids), or to sessile organisms such as mussels, constitute the potential for incidental take of animals. Impacts to the prey base can be interpreted as incidental take if the food supply is a limiting factor on fish productivity. The literature review did not specifically identify scour impacts on other potentially covered species, but such impacts are likely for sessile species and for species that spawn in benthic habitats.

9.1.2.5 Deposition

Deposition may occur in slackwater areas created upstream or downstream of an artificial structure, or it may occur farther downstream when sediment mobilized by scour is redeposited. Deposition can have a variety of effects, depending on the amount of sediment and its particle size distribution. Deposition of large quantities in a localized area results in the creation of bedforms. Deposition of somewhat smaller quantities that do not significantly modify bedforms may still result in burial of redds and benthic organisms such as mussels. Both coarse and fine sediment deposition can present potential for incidental take by burying animals living in the bed, such as eggs and alevins in redds and invertebrate infauna, and/or impairing habitat by reducing access to necessary resources such as prey and well-oxygenated water.

9.1.2.6 Littoral Drift

The littoral drift processes of wave action and littoral current affect benthic substrate and vegetation and therefore influence species assemblages (Thom et al. 1994). Primary productivity, organic matter flow, nutrient dynamics, benthic biota, and the entire local food web may also respond to alterations in littoral drift (Thom et al. 1994).

Pacific salmon, Pacific herring, surf smelt, sand lance, and a variety of other fish may be affected by habitat changes caused by structures that affect littoral drift (Thom et al. 1994). Altering substrate composition in surf smelt spawning areas can affect surf smelt spawning or reduce egg survival. One study found that suitable surf smelt spawning areas were adversely impacted by littoral drift alterations resulting from bulkheads along the Hood Canal (Penttila and Aguero 1978, in Thom et al. 1994). However, no studies were found identifying comparable changes in association with a water crossing structure; thus there are no data to identify the probability of incidental take via this mechanism.

Pacific sand lance spawn in the high intertidal zone on substrates varying from sand to sandy gravel. Sand lance also rely on sandy substrates for burrowing at night. Like surf smelt, sand lance are susceptible to deleterious effects of littoral alterations because they rely on a certain beach profile and specific substrate compositions.

Any species that depends on eelgrass, such as Pacific salmon or Pacific herring, is susceptible to changes in littoral drift. Benthic communities, including invertebrate populations, are impacted by sediment alterations (Nightingale and Simenstad 2001b). Impacts to littoral drift may change beach substrate characteristics and sediment deposition. Changes to these processes can alter benthic and epibenthic communities, fish spawning and rearing habitat, and vegetation (Thom et al. 1994).

Benthic communities, including invertebrate populations, are impacted by sediment alterations (Nightingale and Simenstad 2001b) caused by littoral drift. Local impacts to littoral drift can alter preferred substrate or smother oysters beneath silt.

9.1.2.7 Substrate Modifications

It appears that in marine environments, the primary direct impact of placing structures is to create hard substrates in settings where such substrates did not previously occur, increasing habitat diversity. This change would likely benefit rockfish and any other potentially covered species that use hard or rocky substrates. However, the indirect impact of increased shellhash deposition can harm productive natural habitat types, specifically eelgrass and macroalgae communities. In that case, the risk of incidental take will be the risk of adversely impacting eelgrass and macroalgae. In freshwater environments, the principal substrate modifications entail habitat loss due to placing fill within the channel or floodplain, and habitat modification by replacing native substrate with artificial structures.

9.1.2.8 Rapid Channel Change

Many streams in the Pacific Northwest are highly energetic and capable of rapid, sometimes dramatic changes in their channels. Examples of this include debris flows, dam-break floods, channel avulsions, and rapid channel migration. HPA-permitted activities can have an impact on rapid channel change.

Debris flows are commonly observed in areas that have experienced severe vegetation loss due to forest harvest, forest fire, or land clearance for development. Death and decay of tree roots on steep soils reduces soil cohesion, resulting in shallow-rapid landslides that usually occur during or shortly after severe rainfall events (Croft and Adams 1950, in Coho and Burges 1994). Shallow-rapid landslides commonly initiate on slopes steeper than the angle of repose (about 77 percent) and mix with streamflow in mountain channels to create debris flows that readily transit channels with gradients steeper than about 10 percent (Swanston 1991; Montgomery and Buffington 1993). Such flows entrain sediment and coarse wood that scour the stream channel, often to bedrock, devastating all habitats in the affected reach (Swanston 1991; Benda and Cundy 1990). Commonly, several years to a decade are required before riparian vegetation, fish populations, and water quality recover from the event. Debris flows can be regarded as a cumulative impact that may result from the placement of artificial structures in a channel.

Channel avulsions occur when a stream leaves its old channel and cuts a new one. It has been hypothesized that channel avulsion is the principal mode of channel migration in relatively high-gradient, sediment-rich rivers of Western Washington, such as the Nooksack, the Skykomish, the Green, the Nisqually, and the Queets (Latterell et al. 2006). Channel avulsion is also commonly observed in smaller mountain channels, where it can often be triggered by a debris flow; sediment and wood may fill the original channel and subsequent flows cut a new channel. Channel avulsion is also the dominant channel change process on alluvial fans, where channels are typically transport-limited and avulsion occurs in response to sediment aggradation within the channel. Channel avulsions typically are associated with severe deposition (amounting often to several meters of sediment) in the channel immediately upstream of the avulsion point and dewatering of the channel downstream to the point where the avulsed channel and the initial channel merge. Studies on the Queets River have found that the dewatered channel may be hundreds of meters long (Latterell et al. 2006). Channel avulsions on large rivers are usually not anthropogenic events or are only indirectly caused by human activity, but they may occur in unconfined reaches of smaller streams in response to a culvert becoming plugged by sediment and/or woody debris. Avulsions can be regarded as a cumulative impact that may result from placement of artificial structures in the channel.

Rapid channel migration occurs when bank cutting allows a channel to move laterally by a distance comparable to or greater than the initial channel width. Although the phenomenon has been observed on rivers in Washington, the literature does not contain examples of it happening in response to placement of an artificial structure in the channel.

9.1.3 General Risk of Take from Riparian Vegetation and Large Woody Debris Modifications

NMFS (2006a) found that the best available indicator for the extent of take is the area of riparian habitat that will be permanently modified by the action, because it is directly proportional to long-term harm attributable to the project. In another instance, NMFS (2006b) indicated that the risk of take associated with the removal or disturbance of riparian/shoreline vegetation should be described in terms of acres of riparian/shoreline or miles of stream affected.

9.1.4 General Risk of Take from Aquatic Vegetation Modifications

HPA-permitted structures can sometimes be sited to avoid eelgrass and macroalgae, but some structures must be sited within a narrowly defined area, and in some areas eelgrass and/or macroalgae are very common; thus some structures are likely to directly impact eelgrass and/or macroalgae.

Generally, the federal agencies have treated loss or reduced density of eelgrass as equivalent to loss of essential habitat for listed species known to occur in the area; as such, it constitutes a take of listed species such as salmon and bull trout. A similar perspective has been adopted by state jurisdictional agencies, including WDFW and the Washington Department of Natural Resources (WDNR).

Compensatory mitigation has been required, typically including consideration of temporal impacts related to the time between impact and full eelgrass recovery. Based on the regulatory background, the federal agencies are almost certain to evaluate eelgrass loss as resulting in incidental take of potentially covered species that use eelgrass. Those species include anadromous salmonids, anadromous and marine forage fishes, and certain larval pelagic fishes.

The federal agencies have generally not regarded impacts to macroalgae as amounting to incidental take. The macroalgae most critical to potentially covered species are kelps that chiefly occur in areas of rocky substrate, often in deep water.

Noxious aquatic weed introductions have a high probability of causing incidental take of ESA listed fish species, because noxious weeds can potentially out-compete native vegetation and alter water quality and food web interactions (WNWCB 2006). The impacts of noxious aquatic weeds are indirect, deriving mainly from their accidental introduction during the construction and use of artificial structures. There are no data that provide a basis for stating the likelihood that this impact might occur.

9.1.5 General Risk of Take from Water Quality Modifications

Incidental take risk associated with dissolved oxygen impacts is probably quite low.

Risk of incidental take of potentially covered species due to the use of treated wood appears to be related to factors that include proximity, dilution, and type of treatment. PAH releases from creosote pilings may pose risk of incidental take to some of the covered species, given that many types of organisms have significant PAH sensitivities at low exposure levels (Incardona et al. 2004; Incardona and Scholz 2006). Potentially vulnerable species include mollusks and mussels that may be sessile and juvenile fish that consume epibenthic prey inhabiting those sediments. ACZA-treated wood appears to be somewhat less harmful, with most impacts expected during initial leaching (up to 10 days [Poston 2001]), although recent investigations (Baldwin et al. 2003; Linbo et al. 2006) indicate that juvenile salmonids may have substantially higher sensitivities to dissolved copper (the primary active ingredient of ACZA) than previously suspected. That sensitivity includes an impaired sense of smell, with potential sublethal effects including reduced foraging efficiency and reduced predator avoidance ability.

Activities that allow significant increases in suspended sediment have a high risk of causing incidental take of potentially covered fish species exposed to this condition. Fine sediment deposition also poses an incidental take risk to invertebrates. The risk of take increases in proportion to:

- The magnitude and duration of the impact
- The vulnerability of the affected life-history stage
- The inability of the organism to avoid the impact through avoidance behavior
- The physiological, developmental, and behavioral impairments suffered by the fish
- Indirect mechanisms such as exposure to predation .

9.2 Activity-Specific Risk of Take: 2006 White Papers (Bank Protection and Stabilization Structures, Overwater Structures and Non-Structural Piling, and Water Crossing Structures)

9.2.1 Bank Protection and Stabilization Structures

Table 9-2 summarizes whether potentially covered species may be exposed to incidental take resulting from the impact mechanisms associated with bank protection and stabilization structures. Risk of take is rated as Y (yes; potential for take), N (no potential for take), or U (unknown potential for take). These ratings are based on general consideration of the species distribution (only in terms of fresh water versus marine), habitat use (e.g., movements into immediate shoreline areas during some life stage), habitat requirements (e.g., substrate preferences), prey resources (specifically related to habitat elements promoting their production), and water quality. The magnitude of the risk is highly dependent on how the impact is expressed. For species for which there is no potential for take, no additional conservation measures would be required apart from those currently employed. For species for which the potential for take is unknown, a lack of information on species life history or other data gaps preclude reaching a conclusion.

Table 9-2
Summary of Potential for Incidental Take of Potentially Covered Species

Common Name	Scientific Name	Impact Mechanisms of Bank Protection Projects							Comments
		Construction Activities	Channel Processes	Substrate Modifications	Habitat Accessibility	Aquatic Vegetation	Riparian Vegetation	Water Quality	
Green sturgeon	*Acipenser medirostris*	Y	Y	Y	N	N	N	Y	Most vulnerable to projects that limit availability of deep pools and lead to scour of substrate holding incubating eggs
White sturgeon	*Acipenser transmontanus*	Y	Y	Y	N	N	Y	Y	Most vulnerable to projects that limit availability of deep pools
Newcomb's littorine snail	*Algamorda subrotundata*	Y	N	Y	Y	N	Y	Y	Particularly vulnerable to projects that reduce *Salicornia virginica* habitat in Grays Harbor and Willapa Bay
Pacific sand lance	*Ammodytes hexapterus*	Y	N	Y	Y	N	Y	Y	Particularly vulnerable to marine projects that encroach intertidal zone or lead to reduction in availability of sand in upper intertidal
California floater mussel	*Anodonta californiensis*	Y	Y	Y	Y	U	U	Y	Particularly vulnerable to burial, substrate modifications, and water quality impairment
Mountain sucker	*Catostomus platyrhynchus*	Y	Y	U	Y	U	Y	Y	Most vulnerable to projects that reduce the availability/accessibility of side channel or backwater habitats
Pacific herring	*Clupea harengus pallasi*	Y	N	Y	Y	Y	Y	Y	Particularly vulnerable to projects that reduce availability of marine aquatic vegetation, especially eelgrass
Margined sculpin	*Cottus marginatus*	Y	Y	Y	Y	Y	Y	Y	Particularly vulnerable to projects that impair water quality or reduce availability of sand and gravel substrate
Lake chub	*Couesius plumbeus*	Y	Y	U	Y	Y	Y	Y	Particularly vulnerable to projects that impair water quality, reduce availability of gravel substrate, or reduce availability of terrestrial insects
Giant Columbia River limpet	*Fisherola nuttalli*	Y	Y	Y	N	U	U	Y	Particularly vulnerable to burial, substrate modifications, water quality impairment, and high flows

Common Name	**Scientific Name**	**Impact Mechanisms of Bank Protection Projects** **Construction Activities**	**Channel Processes**	**Substrate Modifications**	**Habitat Accessibility**	**Aquatic Vegetation**	**Riparian Vegetation**	**Water Quality**	**Comments**
Great Columbia River spire snail	*Fluminicola columbiana*	Y	Y	Y	N	U	U	Y	Particularly vulnerable to burial, substrate modifications, and water quality impairment
Pacific cod	*Gadus macrocephalus*	Y	N	Y	N	Y	N	Y	Most vulnerable to projects affecting lower intertidal zone and availability of sand habitats for juveniles
Western ridged mussel	*Gonidea angulata*	Y	Y	Y	Y	Y	Y	Y	Particularly vulnerable to burial, substrate modifications, and water quality impairment; also vulnerable if larva distribution on fishes is limited by habitat accessibility conditions
Northern abalone	*Haliotis kamtschatkana*	Y	N	N	N	Y	N	Y	Particularly vulnerable to burial, substrate modifications, and projects that reduce the availability of marine aquatic vegetation, especially kelp beds
Surf smelt	*Hypomesus pretiosus*	Y	N	Y	Y	N	Y	Y	Particularly vulnerable to marine projects that encroach intertidal zone or lead to reduction in availability of sand and gravel in upper intertidal
River lamprey	*Lampetra ayresi*	Y	Y	Y	Y	N	N	Y	Particularly vulnerable to projects that impair water quality or reduce the availability/accessibility of backwater habitats and other areas with mud/silt accumulations
Western brook lamprey	*Lampetra richardsoni*	Y	Y	Y	Y	N	N	Y	Particularly vulnerable to projects that impair water quality or reduce the availability/accessibility of backwater habitats and other areas with mud/silt accumulations
Pacific lamprey	*Lampetra tridentata*	Y	Y	Y	Y	N	N	Y	Particularly vulnerable to projects that impair water quality or reduce the availability/accessibility of backwater habitats and other areas with mud/silt accumulations; species is often concentrated in extremely high numbers, therefore short-term lethal conditions (e.g., chemical spills or extremely high suspended solids) can affect large portion of population
Pacific hake	*Merluccius productus*	Y	N	Y	N	Y	N	Y	Most vulnerable to projects affecting lower intertidal zone and availability of sand habitats for juveniles
Olympic mudminnow	*Novumbra hubbsi*	Y	Y	Y	Y	Y	Y	Y	Particularly vulnerable to projects that impair water quality or reduce the availability/accessibility of quiet water habitats, such as bogs or swamps, with mud and dense aquatic vegetation

Common Name	Scientific Name	Impact Mechanisms of Bank Protection Projects: Construction Activities	Channel Processes	Substrate Modifications	Habitat Accessibility	Aquatic Vegetation	Riparian Vegetation	Water Quality	Comments
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Pink salmon	*Oncorhynchus gorbuscha*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Chum salmon	*Oncorhynchus keta*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Coho salmon	*Oncorhynchus kisutch*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Redband trout	*Oncorhynchus mykiss gairdneri*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Steelhead	*Oncorhynchus mykiss*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Sockeye salmon	*Oncorhynchus nerka*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Chinook salmon	*Oncorhynchus tschawytscha*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Lingcod	*Ophiodon elongatus*	Y	N	Y	N	Y	N	Y	Most vulnerable to projects affecting lower intertidal zone and availability of sand habitats for juveniles
Olympia oyster	*Ostrea lurida*	Y	N	Y	Y	N	Y	Y	Particularly vulnerable to burial, substrate modifications, and water quality impairment
Pygmy whitefish	*Prosopium coulteri*	Y	Y	U	Y	U	U	Y	Most vulnerable to projects that impair water quality or reduce the availability/accessibility of shallow water and tributary streams
Leopard dace	*Rhinichthys falcatus*	Y	Y	U	Y	Y	Y	Y	Most vulnerable to projects that reduce the availability/accessibility of slow-moving shallow water, decrease habitat structure used for refuge, or reduce prey availability
Umatilla dace	*Rhinichthys umatilla*	Y	Y	U	Y	U	U	Y	Most vulnerable to projects that impair water quality; lack of information on food habits precludes evaluation of impacts to prey availability

Common Name	Scientific Name	Impact Mechanisms of Bank Protection Projects							Comments
		Construction Activities	Channel Processes	Substrate Modifications	Habitat Accessibility	Aquatic Vegetation	Riparian Vegetation	Water Quality	
Bull trout	*Salvelinus confluentus*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Dolly Varden	*Salvelinus malma*	Y	Y	Y	Y	Y	Y	Y	Potential vulnerability via all impact mechanisms
Brown rockfish	*Sebastes auriculatus*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Copper rockfish	*Sebastes caurinus*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Greenstriped rockfish	*Sebastes elongates*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Widow rockfish	*Sebastes entomelas*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Yellowtail rockfish	*Sebastes flavidus*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Quillback rockfish	*Sebastes maliger*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Black rockfish	*Sebastes melanops*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
China rockfish	*Sebastes nebulosus*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Tiger rockfish	*Sebastes nigrocinctus*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Bocaccio rockfish	*Sebastes paucispinis*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Canary rockfish	*Sebastes pinniger*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects, but often associated with kelp beds
Redstripe rockfish	*Sebastes proriger*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects
Yelloweye rockfish	*Sebastes ruberrimus*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects

Common Name	Scientific Name	Impact Mechanisms of Bank Protection Projects							Comments
		Construction Activities	Channel Processes	Substrate Modifications	Habitat Accessibility	Aquatic Vegetation	Riparian Vegetation	Water Quality	
Longfin smelt	*Spirinchus thaleichthys*	Y	Y	U	Y	N	N	Y	Most vulnerable to projects that impair water quality and access to streams
Eulachon	*Thaleichthys pacificus*	Y	Y	Y	Y	N	Y	Y	Most vulnerable to projects that impair water quality and availability of sandy habitats in marine, estuarine, and lower rivers
Walleye pollock	*Theragra chalcogramma*	Y	N	Y	N	Y	N	Y	Marine species not closely associated with immediate vicinity of bank protection projects

When evaluating risk of take for habitat-modifying projects, the federal agencies generally do not attempt to quantify the number of fish injured or killed because the relationship between habitat conditions and the distribution and abundance of those individuals in the action area cannot accurately be determined. Instead, the federal agencies tend to quantify the extent of anticipated take by measure of the amount of impacted habitat (e.g., length of streambank modified or area below the OHWL modified). In this way, every project had some level of take that was quantified only in terms of the physical size of the project. No explicit take thresholds (such as shoreline length) were identified during a review of bank protection-related biological opinions prepared by NOAA Fisheries and USFWS in recent years. However, it can be interpreted that by characterizing a project's incidental take based on project size, the federal agencies deem bank protection projects of any size as having some level of take. This approach provides the federal agencies with assurances that consultation with them will be re-initiated if a project is anticipated to expand in size and that such expansion cannot occur without additional consultation.

For the purposes of evaluating the risk of take, the potential impacts were divided into two categories: those associated with the installation of the bank protection structures and those associated with the existence of the structure once it is in place. Potential impacts associated with the construction of the bank protection structure are generally short term, e.g., elevated suspended solids and noise, although longer-term impacts can occur, e.g., lack of shade due to riparian vegetation removal. Many of the potential construction-related impacts can be avoided or minimized using BMPs or other conservation measures. The potential risk of take associated with construction activities will therefore be highly dependent upon the measures taken to avoid or minimize impacts. Little information is available on potential thresholds based on the available literature presented in Section 7, which almost exclusively focused on impacts to salmonids.

The presence of bank protection structures can generate lasting impacts that may have greater implications for species take, distribution, and population viability than any short-term construction-related impacts. These long-term impacts can vary greatly over time and are therefore less predictable and quantifiable. Bank protection projects for which the primary purpose and function is to prevent the habitat-forming and sustaining processes of water bodies, e.g., those projects focused on flood control and the protection of uplands, will generally have the most significant long-term impacts on the habitat and therefore the highest risk of take. However, project-specific details such as size, location (both in terms of species distributions and position/function within a reach), and technique all contribute significantly to the risk of take associated with a bank protection structure.

Many of the potential impacts associated with bank protection may be more evident in an evaluation of cumulative impacts than in a project-specific evaluation. For example, in rivers, bank protection structures generally limit or eliminate channel-forming and channel-sustaining processes along a finite portion of a water body and therefore incrementally diminish the water body's ability to naturally function. Neither a technique for evaluating the cumulative effects nor the outcome of such an evaluation was identified. The literature review did not identify information sources that would support a recommended threshold for the amount of shoreline with bank protection structures beyond which the degree of water body impairment becomes significant. The reasons for the lack of a threshold may include a lack of data as well as the existence of water body-specific conditions that would limit the applicability of a threshold to other systems. If such a technique were to be developed, then among the most significant water body-specific conditions that should be considered are spatial distribution of bank protection structures, spatial distribution of gravel sources, spatial distribution and width of floodplain, gradient, and flow.

In terms of the risk of take associated with different types of bank protection techniques, bank protection projects that incorporate natural features and/or allow for partial function of channel-forming and channel-maintaining processes would have a lower risk of take than techniques that stop the functions. Soft armoring techniques have a lower risk of take than hard armoring techniques. In situations where some hard armoring techniques are necessary to adequately protect a bank, then integrated techniques that incorporate hard and soft elements would produce an intermediate risk of take.

Activities that occur subsequently on land protected by bank protection structures can also contribute to the long-term risk of take. Bank protection structures can provide landowners with a false sense of safety, particularly regarding large floods and bluff erosion. As a result, upland structures are built closer to the shoreline or bluff than would occur otherwise and may be imperiled in the long term or may allow the landowner to aggressively maintain structures that significantly impact habitat for potentially covered species.

Bank protection projects can have beneficial impacts, and many bank protection projects are indeed designed as habitat restoration projects. For example, a bank protection project that addresses mass wasting and fine sediment contributions can be beneficial to habitats and species if properly designed. A distinguishing feature of beneficial bank protection projects is a project design that works with natural processes and that incorporates large wood to add habitat complexity to a reach. In river, stream, and estuarine environments, bank protection projects that allow continuation of full or partial function of the natural processes associated with floodplain connectivity, side channel formation, and sediment (gravel) source additions can provide beneficial outcomes. In marine and lake environments, bank protection structures that allow continuation of full or partial function of the natural littoral drift processes, including the sediment source entrainment and sediment transport, can provide beneficial outcomes. The placement of large wood in the channel (either random or designed) can add habitat complexity by creating habitats in areas where the natural processes, including LWD recruitment, have been altered. In fact, properly designed bank protection projects can re-establish natural processes, e.g., wood recruitment in pool-forming structures or littoral drift along marine shorelines. Along this same line of discussion, it should be noted that where bank protection projects are often needed is in highly modified (e.g., flow altered, channelized, armored, denuded) rivers and streams where, because of substantial capital improvements and infrastructure, it is unrealistic to expect that truly “natural river erosion/deposition processes“ will be restored. In these rivers and streams, properly designed bank protection projects may provide some of the better fish habitat opportunities in the reach.

A long-term perspective is necessary when considering the potential impacts of a bank protection project. Potential short-term benefits of a bank protection project may not outweigh its long-term impacts. The location of the stream channel and bank protection project with respect to the floodplain is an important determining factor of potential impacts. If the bank protection is located on the stream channel at the outer limits of the 100- or 500-year floodplain, the potential impacts are much different (generally much less) than if the same project were implemented on property located in the middle of a 1-mile-wide floodplain

9.2.1.1 Evaluation of Relative Risk of Take Associated with Bank Protection Structures

All bank protection activities have potential for some take, unless no potentially covered species occur in the project area, including the areas upstream and downstream (or updrift and downdrift) that may be impacted by the structure. Table 9-3 provides some general guidelines regarding the project elements that contribute to a bank protection project of "low," "moderate," or "high" risk of take. These general categorizations are based on the best professional judgment of the analysis team and require interpretation beyond the empirical data available in the literature. The categorizations are intended to be widely applicable to potentially covered species; however, it is possible that the categorizations will not be valid for all species, particularly those with lesser known habitat and ecological requirements. Since much of the literature is based on impacts to salmonids, the categorizations are perhaps most applicable to salmonids.

For a bank protection project to be of "low" risk, it must meet all applicable requirements in the low-risk category, i.e., no "moderate" or "high" risk aspects to the project. In addition, the "low"-risk conditions in the row labeled "Construction-Related Activities" must also be satisfied for a project to be of "low" risk. In general terms, activities in the low-risk category appear to be well suited for programmatic approval, whereas activities in the high-risk category would likely require consideration of project-specific elements (e.g., environmental setting, size, and installation technique) and present a clear need to implement conservation measures to reduce the risk of take. The appropriateness of programmatic approval of activities in the moderate-risk category is debatable and would depend in part on the use of conservation measures. The risk evaluation summarized in Table 9-3 assumes that potentially covered species are present when the described impact occurs; thus, impacts may be avoided by performing the activities when or where potentially covered species are absent.

Table 9-3
Evaluation of Relative Risk of Take Associated with Bank Protection Structures

Activity or Structure	Risk of Take			Rationale and Assumptions
	Low	Moderate	High	
Construction-Related Activities	• In areas inhabited by only migratory potentially covered species (e.g., anadromous species) and/or species that move between habitats with some predictability (e.g., spawning runs from lakes to streams), activities that occur within allowable work windows based on tributary-specific species presence and periodicity data that avoid working during periods of species presence • Activities that do not entail removing native riparian vegetation, LWD, or small woody debris (SWD) • Pile-driving activities with peak underwater sound <150 dB • Activities that avoid need for dewatering	• In areas inhabited by only migratory potentially covered species (e.g., anadromous species) and/or species that move between habitats with some predictability (e.g., spawning runs from lakes to streams), activities that occur within allowable work windows based on general species presence information (e.g., statewide species distribution maps) and periodicity data that attempt to avoid working during periods of species presence • Project areas where non-migratory potentially covered fish species presence is presumed, but not documented • Activities that minimize the removal of native riparian vegetation and that replant (including maintenance) the cleared area's native vegetation upon construction completion • Pile-driving activities with peak underwater sound between 150 and 180 dB	• Project areas where potentially covered invertebrate species presence is documented • Project areas where any potentially covered fish species presence is documented and the construction timing coincides with their presence • Activities that do not minimize the removal of native riparian vegetation and/or that do not replant (including maintenance) the cleared area's native vegetation upon construction completion • Pile-driving activities requiring hammer pile driving with peak underwater sound >180 dB • Activities that include dewatering a portion of channel and either do not remove species from area or do not implement BMPs to reduce introduction of suspended	For areas inhabited by potentially covered species during in-water construction, bank protection activities represent a high risk of take due to the various disturbances to aquatic habitats that typically occur during in-water work. Risk of take is low when the project completely avoids timing in-water construction during species presence or known sensitivity periods. Moderate risk is indicated when in-water work is completed mostly within these periods, but still maintains some in-water work outside the periods. For bank protection activities that permanently remove native riparian vegetation, risk of take is high because bank vegetation is closely linked to habitat quality and direct survival (most importantly, via water temperature control) for many potentially covered species. For pile-driving activities, risk of take for potentially covered fish is set as high for bank protection projects that produce underwater sound above the injury and disturbance threshold for threatened and endangered salmonids, >180dB. Risk of take is moderate for projects producing peak underwater sound between the 180 dB injury threshold and the 150 dB threshold for behavioral disturbance.

Activity or Structure	Risk of Take			Rationale and Assumptions
	Low	Moderate	High	
		• Activities that minimize the dewatered area and length of time, remove species from area, and implement BMPs to minimize the addition of suspended solids	solids	Activities producing peak underwater sound below 150 dB would be expected to exhibit a low risk of take for potentially covered fish. Because invertebrate sound studies are sparse, it is expected that these risk levels, which are set based on effects to fish, will adequately apply to invertebrate responses to construction-related sound. Activities that require dewatering may minimize the dewatered area and length of time of dewatering, remove species from the area, and implement BMPs to minimize the addition of suspended solids; however, under the take definition, these activities would still constitute take. Therefore, risk of take is high and severe for dewatering activities that do not minimize the dewatered area and length of time dewatered, and for those that do not remove species from the area, and that do not minimize suspended solids. Risk of take is moderate and less severe if these minimization measures are implemented. Risk of take is low when dewatering can be avoided.

Activity or Structure	Risk of Take			Rationale and Assumptions
	Low	Moderate	High	
Vertical Retaining Walls, Rock Revetments, and Rock Toes	• Reaches in all environments that are not sediment sources (i.e., not feeder bluffs) and in which the structure does not extend into intertidal zone or below OHWL	• Marine and estuarine reaches that do not contain sediment sources (i.e., not feeder bluffs) and in which the structure does not extend into intertidal zone, but forage fish spawning is known to occur • All environments in which rock toes support soft armoring approaches along remainder of bank	• Reaches in all environments that contain sediment sources (i.e., feeder bluffs) • Marine and estuarine reaches that do not contain sediment sources (i.e., not feeder bluffs) but in which the structure extends into intertidal zone • All environments along known spawning areas for potentially covered fish species • All environments along known areas that contain sessile potentially covered invertebrate species • All environments in which rock toes support upper bank rock or wall revetments	For vertical retaining walls, risk of take is high in marine environments where forage fish spawning could occur and salmonid migration occurs. Take risk is also high in other environments due to indirect effects because these structures isolate sediment supply, cause scour, reflect wave energy, and contribute to a loss of fine sediment, causing ensuing effects to biota and vegetation. For rock revetments, similar to vertical retaining walls, risk of take is high in marine environments potentially supporting forage fish spawning and salmonid migration due to indirect effects in reducing gravel recruitment and sediment transport and affecting shoreline currents. In addition, rock revetments can disrupt flows, reduce food delivery, and create difficult swimming for smaller fish. For rock toes, risk of take is moderate when toes support upper bank biostabilization structures, which function to improve overall habitat, but risk of take is high where rock toes are placed to support rock or wall revetments.

Activity or Structure	Risk of Take			Rationale and Assumptions
	Low	Moderate	High	
Levees		• Levee "setbacks" that increase the width of the channel, provide high flow refuge habitat, and incorporate LWD	• Levees other than those described as moderate risk	Risk of take is high for levees, except when the project is attempting habitat restoration by setting back existing levees or other bank protection structures. This is because levees limit channel hydraulics and sediment recruitment, sometimes isolating sediment supply to the substrate and transport of that sediment through the system. In addition, levees fragment ecosystem connectivity and limit habitat accessibility for many potentially covered species, depending on the habitat. For example, in an estuary, levees can isolate marsh areas and limit LWD distribution.
Log/Rootwad Toes	• All environments in which the toe is combined with other biotechnical bank approaches	• All environments in which the toe is combined with rock or concrete bank approaches		Risk of take is low for log and rootwad toes where they typically are used to support upper bank biostabilization structures. They also increase habitat complexity along the bank.

Activity or Structure	Risk of Take			Rationale and Assumptions
	Low	Moderate	High	
Beach Nourishment	• Marine and freshwater environments using pre-washed substrate in which turbidity increases are not likely to occur • Marine environments in which macroalgae or eelgrass is not covered • Freshwater environments in which aquatic vegetation is not covered • All environments in which material is placed above the OHWL or MHHW. • Marine and freshwater environments in which similarly sized materials as compared to an appropriate reference site are placed	• Marine environments in which turbidity increases are likely to occur • All environments in which material is placed below the OHWL or MHHW	• Marine environments in which macroalgae or eelgrass is covered • Freshwater environments in which aquatic vegetation is covered	Risk of take due to beach nourishment is low if material is pre-washed or of larger (pebble/gravel) size and not likely to increase turbidity on site, if existing eelgrass or macroalgae will not be disturbed. Risk of take is moderate for all environments in which beach nourishment occurs on the upper beach only, because this material may move down the beach and ultimately affect species occurring in lower elevations. Risk of take is moderate if material is fine/sand, if eelgrass, macroalgae, or aquatic vegetation will be disturbed, and/or if material is placed to a large extent below the OHWL or MHHW.
Avulsion Prevention	• All environments in which avulsion prevention elements involve natural logs, brush, rootwad structures			Risk of take due to avulsion prevention is low because these structures are typically natural logs, brush rootwads placed in the habitat, which increases habitat complexity and a host of other habitat functions.
Subsurface Drainage Systems	• All environments in which drainage system elements involve natural logs, brush, rootwad structures	• All environments in which drainage system elements involve synthetic pipes or installations		Similar to avulsion prevention techniques, risk of take due to subsurface drainage systems is low where these structures consist of natural materials that will eventually degrade and become part of the environment and long-term bank stability solution.

Activity or Structure	Risk of Take			Rationale and Assumptions
	Low	Moderate	High	
Biotechnical Bank Protection Techniques	• All environments			Risk of take due to biotechnical bank protection is low because these structures typically provide beneficial effects to aquatic species, such as increases of refugia and habitat structure along the bank or shoreline, detrital inputs, and vegetative cover.
Bank Reshaping or Regrading	• All environments in which no in-water work is used • All environments in which bank reshaping/regrading is combined with biotechnical toe	• All environments in which in-water work is used • All environments in which bank reshaping/regrading is combined with rock toe		Risk of take due to bank reshaping or regrading is moderate if in-water work is used, because of the high potential for turbidity increases during regrading/reshaping work. If work is completed in the dry, risk of take is low. If bank reshaping/regrading entails placing a rock toe, risk of take is higher than if a log or rootwad toe is used.
Soil Reinforcement	• All environments			Risk of take due to soil reinforcements is low because these elements are typically surrounded by fabric and do not entail placing exposed soil or sediment on the bank or shore.
Coir and Straw Logs	• All environments			Similar to soil reinforcement, risk of take due to coir and straw logs is low because these elements typically consist of natural, biodegradable fabric or material and do not entail placing exposed soil or sediment on the bank or shore.
Integrated Approaches	• See Vertical Retaining Walls, Rock Revetments, and Rock Toes; see Bank Reshaping or Regrading	• See Vertical Retaining Walls, Rock Revetments, and Rock Toes; see Bank Reshaping or Regrading	• See Vertical Retaining Walls, Rock Revetments, and Rock Toes; see Bank Reshaping or Regrading	See Vertical Retaining Walls, Rock Revetments, and Rock Toes; see Bank Reshaping or Regrading

9.2.2 Overwater Structures

Table 9-4 summarizes the risk that potentially covered species may suffer incidental take resulting from twelve impact mechanisms. The potential that a species may experience incidental take is characterized in Table 9-4 as Y (yes; potential for take), N (no potential for take), or U (unknown potential for take). The magnitude of the risk is highly dependent on how the impact is expressed, which in turn is highly dependent on the suite of conservation measures employed to minimize the risk of causing take. For species for which there is no potential for take, no additional precautions would be required apart from those currently employed. For species for which the potential for take is unknown, a lack of information on species life history or other data gaps preclude reaching a conclusion.

The following decision rules explain most of the content of Table 9-4:

- Marine species are not at risk of take due to impacts to channel hydraulics, or to freshwater aquatic vegetation.
- Species that spend all of their lives in freshwater are not at risk of take due to impacts to eelgrass and macroalgae.
- For most species except salmonids, the effects of noise, artificial light, shading, and vessel activities are largely unknown.

Table 9-4
Summary of Potential for Incidental Take of Potentially Covered Species

Common Name	**Scientific Name**	**Impact Mechanisms**											
		Shading	**Eelgrass and Macroalgae Modifications**	**Freshwater Aquatic Vegetation Modifications**	**Riparian and Shoreline Vegetation Modifications**	**Noise**	**Water Quality Modifications**	**Channel Hydraulic Modifications**	**Littoral Drift Modifications**	**Substrate Modifications**	**Channel Dewatering**	**Artificial Light**	**Vessel Activities**
Green sturgeon	*Acipenser medirostris*	U	U	Y	Y	U	Y	Y	Y	Y	Y	U	U
White sturgeon	*Acipenser transmontanus*	U	U	Y	Y	U	Y	Y	Y	Y	Y	U	U
Newcomb's littorine snail	*Algamorda subrotundata*	U	Y	N	Y	U	Y	N	Y	Y	N	U	U
Pacific sand lance	*Ammodytes hexapterus*	Y	Y	N	Y	U	Y	N	Y	Y	N	U	U
California floater mussel	*Anodonta californiensis*	U	N	Y	Y	U	Y	Y	Y	Y	Y	U	U
Mountain sucker	*Catostomus platyrhynchus*	U	N	U	Y	U	Y	Y	N	U	Y	U	U
Pacific herring	*Clupea*	U	Y	N	Y	U	Y	N	Y	Y	N	U	U

		Impact Mechanisms											
Common Name	**Scientific Name**	**Shading**	**Eelgrass and Macroalgae Modifications**	**Freshwater Aquatic Vegetation Modifications**	**Riparian and Shoreline Vegetation Modifications**	**Noise**	**Water Quality Modifications**	**Channel Hydraulic Modifications**	**Littoral Drift Modifications**	**Substrate Modifications**	**Channel Dewatering**	**Artificial Light**	**Vessel Activities**
	harengus pallasi												
Margined sculpin	*Cottus marginatus*	Y	N	Y	Y	U	U	Y	N	U	Y	U	U
Lake chub	*Couesius plumbeus*	U	N	Y	U	U	U	U	N	U	U	U	U
Giant Columbia River limpet	*Fisherola nuttalli*	U	N	U	U	U	Y	Y	N	Y	Y	U	U
Great Columbia River spire snail	*Fluminicola columbiana*	U	N	U	U	U	Y	Y	N	Y	Y	U	U
Pacific cod	*Gadus macrocephalus*	N	Y	N	N	U	Y	N	Y	Y	N	U	U
Western ridged mussel	*Gonidea angulata*	U	N	Y	Y	U	Y	Y	Y	Y	Y	U	U
Northern abalone	*Haliotis kamtschatkana*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Surf smelt	*Hypomesus pretiosus*	U	Y	N	Y	U	Y	N	Y	Y	N	U	U
River lamprey	*Lampetra ayresi*	U	N	N	Y	U	Y	Y	Y	Y	Y	U	U
Western brook lamprey	*Lampetra richardsoni*	U	N	N	Y	U	Y	Y	N	Y	Y	U	U
Pacific lamprey	*Lampetra tridentata*	U	N	N	Y	U	Y	Y	Y	Y	Y	U	U
Pacific hake	*Merluccius productus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Olympic mudminnow	*Novumbra hubbsi*	U	N	Y	Y	U	Y	Y	N	Y	Y	U	U
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U	U
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	Y	N	Y	Y	Y	Y	Y	N	Y	Y	U	U
Pink salmon	*Oncorhynchus gorbuscha*	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U
Chum salmon	*Oncorhynchus keta*	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U
Coho salmon	*Oncorhynchus kisutch*	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U

Common Name	Scientific Name	Impact Mechanisms											
		Shading	Eelgrass and Macroalgae Modifications	Freshwater Aquatic Vegetation Modifications	Riparian and Shoreline Vegetation Modifications	Noise	Water Quality Modifications	Channel Hydraulic Modifications	Littoral Drift Modifications	Substrate Modifications	Channel Dewatering	Artificial Light	Vessel Activities
Redband trout	*Oncorhynchus mykiss*	Y	N	Y	Y	Y	Y	Y	N	Y	Y	U	U
Steelhead	*Oncorhynchus mykiss*	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U
Sockeye salmon	*Oncorhynchus nerka*	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U
Chinook salmon	*Oncorhynchus tschawytscha*	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U
Lingcod	*Ophiodon elongatus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Olympia oyster	*Ostrea lurida*	Y	Y	N	Y	U	Y	N	Y	Y	N	U	U
Pygmy whitefish	*Prosopium coulteri*	U	N	U	U	Y	U	Y	N	U	Y	U	U
Leopard dace	*Rhinichthys falcatus*	U	N	U	U	U	U	Y	N	U	Y	U	U
Umatilla dace	*Rhinichthys Umatilla*	U	N	U	U	U	U	Y	N	U	Y	U	U
Bull trout	*Salvelinus confluentus*	U	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U
Dolly Varden	*Salvelinus malma*	U	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	U
Brown rockfish	*Sebastes auriculatus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Copper rockfish	*Sebastes caurinus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Greenstriped rockfish	*Sebastes elongates*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Widow rockfish	*Sebastes entomelas*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Yellowtail rockfish	*Sebastes flavidus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Quillback rockfish	*Sebastes maliger*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Black rockfish	*Sebastes melanops*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
China rockfish	*Sebastes nebulosus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Tiger rockfish	*Sebastes nigrocinctus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U

Common Name	Scientific Name	Impact Mechanisms											
		Shading	Eelgrass and Macroalgae Modifications	Freshwater Aquatic Vegetation Modifications	Riparian and Shoreline Vegetation Modifications	Noise	Water Quality Modifications	Channel Hydraulic Modifications	Littoral Drift Modifications	Substrate Modifications	Channel Dewatering	Artificial Light	Vessel Activities
Bocaccio rockfish	*Sebastes paucispinis*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Canary rockfish	*Sebastes pinniger*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Redstripe rockfish	*Sebastes proriger*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Yelloweye rockfish	*Sebastes ruberrimus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Longfin smelt	*Spirinchus thaleichthys*	U	Y	N	Y	Y	Y	Y	Y	U	Y	U	U
Eulachon	*Thaleichthys pacificus*	U	Y	N	Y	Y	Y	N	Y	Y	N	U	U
Walleye pollock	*Theragra chalcogramma*	U	Y	N	N	U	Y	N	Y	Y	N	U	U

Note: Species listed in alphabetical order by scientific name.

9.2.2.1 Hydraulic and Geomorphic Modifications

Non-structural piling and piling associated with other overwater structures (i.e., piers) could potentially cause scour in marine or estuarine areas with strong tidal currents, or riverine environments with strong currents.

As with scour, deposition impacts are most likely when an overwater structure and associated support structures and non-structural piling are installed and have not received proper hydraulic design. While significant amounts of deposition (i.e., amounts potentially causing measurable incidental take) are not likely to occur from the installation of an overwater structure or non-structural piling, some localized deposition may occur as a result of changes in hydraulics in the immediate vicinity of the structure. Potential impacts from deposition associated with installation of an overwater structure or non-structural piling would be localized and relatively minor with a low potential risk for take of the covered species.

9.2.2.1 Aquatic Vegetation Modifications: Eelgrass and Macroalgae

Overwater structures and non-structural piling can sometimes be sited to avoid eelgrass and macroalgae, but some structures must be sited within a narrowly defined area, and in some areas eelgrass and/or macroalgae are very common, thus some over water structures and/or non-structural piling are likely to directly impact eelgrass and/or macroalgae.

9.2.2.2 Risk Evaluation

Table 9-5 presents a summary of the incidental take risk analysis. This risk evaluation is at best a qualitative assessment and is based strongly on professional experience of the analysis team in the context of their work in ESA implementation. It assumes that potentially covered species are present when the described impact occurs; thus, impacts may be avoided by performing the activities when or where covered species are absent.

Table 9-5: Conclusions of the Risk Evaluation

Activity	Low Risk	Moderate Risk	High Risk
Freshwater structures per WAC 220-110-060	• Structures located in areas lacking submerged aquatic vegetation; • Structures causing little increased shading, either due to size or incorporation of grating or other light penetrating features • Pile-driving activities with peak sound <150 dB; • Structures in areas with little sediment transport; • Structures not increasing the volume of untreated stormwater; • Placing small areas of non-conforming substrate; • Activities avoiding the impacts potentially causing “moderate” or “high” risk.	• Structures removing riparian vegetation; • Structures that require removing LWD in lentic waters; • Pile-driving activities with peak sound between 150 and 180 dB; • Structures increasing the volume of untreated stormwater due to increased impervious surface; • Structures comprised of CCA- or ACZA-treated wood; • Structures that measurably alter channel hydraulics or littoral drift; • Structures causing nighttime illumination of the water surface.	• Structures in areas of submerged aquatic vegetation that are used by dependent species (e.g., Olympic mudminnow); • Structures that require removing LWD in lotic waters; • Pile-driving activities requiring hammer pile driving with peak sound >180 dB; • Structures that substantially alter channel hydraulics; • Placing large areas of non-conforming substrate; • Activities that require dewatering of the work area; • Activities requiring substantial in-water operation of mechanized equipment. • Structures in riverine environments that use creosote treated wood;
Saltwater structures per WAC 220-110-300	• Structures located in areas lacking submerged aquatic vegetation; • Structures causing low shade; • Pile-driving activities with peak sound <150 dB; • Structures in areas with little sediment transport; • Placing small areas of non-conforming substrate; • Activities avoiding the impacts potentially causing “moderate” or “high” risk.	• Structures removing riparian vegetation; • Pile-driving activities with peak sound between 150 and 180 dB; • Structures discharging stormwater; • Structures requiring CCA- or ACZA-treated wood; • Structures measurably altering littoral drift; • Structures causing nighttime illumination of the water surface.	• Structures located in areas of eelgrass or macroalgae; • Structures shading large areas; • Structures requiring hammer pile driving with peak sound >180 dB; • Structures that require creosote-treated wood; • Placing large areas of non-conforming substrate; • Activities that require dewatering of the work area; • Activities requiring substantial in-water operation of mechanized equipment.
Non-	• Pile-driving activities with	• Pile-driving activities with	• Piling located in areas of

structural or structural piling	peak sound <150 dB; • Structures that avoid the impacts potentially causing “moderate” or “high” risk.	peak sound between 150 and 180 dB • Structures requiring CCA- or ACZA-treated wood.	eelgrass or macroalgae; • Structures requiring hammer pile driving with peak sound >180 dB. • Structures requiring creosote-treated wood.

9.2.3 Water Crossing Structures

In Table 9-6, the potential that a species may experience incidental take is characterized as Y (yes; potential for take), N (no potential for take), or U (unknown potential for take). The magnitude of the risk is highly dependent on how the impact is expressed, which in turn is highly dependent on the suite of conservation measures employed to minimize the risk of causing take. For species for which there is no potential for take, no additional conservation measures would be required apart from those currently employed. For species for which the potential for take is unknown, a lack of information on species life history or other data gaps preclude reaching a conclusion. The “unknown” category may be the most problematic from the standpoint of ESA compliance, because we lack information needed for the federal agencies to determine whether incidental take would be likely to jeopardize continued existence of affected populations.

The following decision rules explain most of the content of Table 9-6:

- Marine species are not at risk of take due to impacts to channel hydraulics, substrate modification, or freshwater aquatic vegetation.
- Freshwater species are not at risk of take due to impacts to eelgrass and macroalgae.
- For most species except salmonids, the effects of noise, artificial light, shading, and vessel activities are largely unknown.

Table 9-6. Summary of Potential for Incidental Take of Potentially Covered Species

		Impact Mechanisms											
Common Name	**Scientific Name**	**Shading**	**Eelgrass and Macroalgae**	**Freshwater Aquatic Vegetation**	**Riparian and Shoreline Vegetation**	**Noise**	**Water Quality**	**Channel Hydraulic Effects**	**Littoral Drift**	**Substrate Modifications**	**Channel Dewatering**	**Artificial Light**	**Vessel Activities**
Green sturgeon	*Acipenser medirostris*	U	U	U	Y	U	Y	Y	Y	Y	Y	U	U
White sturgeon	*Acipenser transmontanus*	U	U	U	Y	U	Y	Y	Y	Y	Y	U	U
Newcomb's littorine snail	*Algamorda subrotundata*	U	Y	N	Y	U	Y	N	Y	Y	N	U	U
Pacific sand lance	*Ammodytes hexapterus*	Y	Y	N	Y	U	Y	N	Y	Y	N	U	U
California floater mussel	*Anodonta californiensis*	U	N	Y	Y	U	Y	Y	Y	Y	Y	U	U
Mountain sucker	*Catostomus platyrhynchus*	U	N	U	Y	U	Y	Y	N	U	Y	U	U
Pacific herring	*Clupea harengus pallasi*	U	Y	N	Y	U	Y	N	Y	Y	N	U	U
Margined sculpin	*Cottus marginatus*	Y	N	U	Y	U	U	Y	N	U	Y	U	U
Lake chub	*Couesius plumbeus*	U	N	U	U	U	U	U	N	U	U	U	U
Giant Columbia River limpet	*Fisherola nuttalli*	U	N	U	U	U	Y	Y	N	Y	Y	U	U
Great Columbia River spire snail	*Fluminicola columbiana*	U	N	U	U	U	Y	Y	N	Y	Y	U	U
Pacific cod	*Gadus macrocephalus*	N	Y	N	N	U	Y	N	Y	Y	N	U	U
Western ridged mussel	*Gonidea angulata*	U	N	Y	Y	U	Y	Y	Y	Y	Y	U	U
Northern abalone	*Haliotis kamtschatkana*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Surf smelt	*Hypomesus pretiosus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
River lamprey	*Lampetra ayresi*	U	N	N	Y	U	Y	Y	Y	Y	Y	U	U
Western brook lamprey	*Lampetra richardsoni*	U	N	N	Y	U	Y	Y	N	Y	Y	U	U
Pacific lamprey	*Lampetra tridentata*	U	N	N	Y	U	Y	Y	Y	Y	Y	U	U
Pacific hake	*Merluccius productus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Olympic mudminnow	*Novumbra hubbsi*	U	N	Y	Y	U	Y	Y	N	Y	Y	U	U
Coastal cutthroat trout	*Oncorhynchus clarki clarki*	Y	Y	U	Y	Y	Y	Y	Y	Y	Y	U	U

		Impact Mechanisms											
Common Name	**Scientific Name**	**Shading**	**Eelgrass and Macroalgae**	**Freshwater Aquatic Vegetation**	**Riparian and Shoreline Vegetation**	**Noise**	**Water Quality**	**Channel Hydraulic Effects**	**Littoral Drift**	**Substrate Modifications**	**Channel Dewatering**	**Artificial Light**	**Vessel Activities**
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	Y	N	U	Y	Y	Y	Y	N	Y	Y	U	U
Pink salmon	*Oncorhynchus gorbuscha*	Y	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Chum salmon	*Oncorhynchus keta*	Y	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Coho salmon	*Oncorhynchus kisutch*	Y	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Redband trout	*Oncorhynchus mykiss*	Y	N	U	Y	Y	Y	Y	N	Y	Y	U	U
Steelhead	*Oncorhynchus mykiss*	Y	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Sockeye salmon	*Oncorhynchus nerka*	Y	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Chinook salmon	*Oncorhynchus tschawytscha*	Y	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Lingcod	*Ophiodon elongatus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Olympia oyster	*Ostrea lurida*	Y	Y	N	Y	U	Y	N	Y	Y	N	U	U
Pygmy whitefish	*Prosopium coulteri*	U	N	U	U	Y	U	Y	N	U	Y	U	U
Leopard dace	*Rhinichthys falcatus*	U	N	U	U	U	U	Y	N	U	Y	U	U
Umatilla dace	*Rhinichthys Umatilla*	U	N	U	U	U	U	Y	N	U	Y	U	U
Bull trout	*Salvelinus confluentus*	U	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Dolly Varden	*Salvelinus malma*	U	Y	U	Y	Y	Y	Y	Y	Y	Y	Y	U
Brown rockfish	*Sebastes auriculatus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Copper rockfish	*Sebastes caurinus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Greenstriped rockfish	*Sebastes elongates*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Widow rockfish	*Sebastes entomelas*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Yellowtail rockfish	*Sebastes flavidus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Quillback rockfish	*Sebastes maliger*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Black rockfish	*Sebastes melanops*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
China rockfish	*Sebastes nebulosus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U

Common Name	Scientific Name	Impact Mechanisms											
		Shading	Eelgrass and Macroalgae	Freshwater Aquatic Vegetation	Riparian and Shoreline Vegetation	Noise	Water Quality	Channel Hydraulic Effects	Littoral Drift	Substrate Modifications	Channel Dewatering	Artificial Light	Vessel Activities
Tiger rockfish	*Sebastes nigrocinctus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Bocaccio rockfish	*Sebastes paucispinis*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Canary rockfish	*Sebastes pinniger*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Redstripe rockfish	*Sebastes proriger*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Yelloweye rockfish	*Sebastes ruberrimus*	U	Y	N	N	U	Y	N	Y	Y	N	U	U
Longfin smelt	*Spirinchus thaleichthys*	U	Y	N	Y	Y	Y	Y	Y	U	Y	U	U
Eulachon	*Thaleichthys pacificus*	U	Y	N	Y	Y	Y	N	Y	Y	N	U	U
Walleye pollock	*Theragra chalcogramma*	U	Y	N	N	U	Y	N	Y	Y	N	U	U

Note: Species listed in alphabetical order by scientific name.

9.2.3.1 Hydraulic and Geomorphic Modifications

Water crossing structures can interrupt hyporheic exchange and groundwater recharge by placing fill and/or impervious surface on previously pervious areas. This impact is particularly severe in the case of full culverts, where both the approach fill and the base of the culvert represent surfaces that impede or prevent infiltration. In bottomless culverts, approach fills impede infiltration, and in bridges, the impact is due to approach fills and areas occupied by pilings or piers. In all cases, though, the impact of impaired hyporheic and groundwater function is generally minor in comparison to the permanent habitat loss represented by the loss of stream channel and floodplain areas overlain by fills, piers, pilings, and culvert bottoms.

9.2.3.1.1 Embedding

Fine sediment inputs leading to embedding may occur in association with water crossing construction when a poorly designed structure causes locally increased deposition or locally increased erosion of fine sediments in the bed or banks that may be deposited in gravel-bedded streams farther downstream. Ditches and stormwater discharges associated with water crossing structures may also contribute fine sediment to the stream. Since water crossing structures often alter channel hydraulics but seldom cause persistent increases in fine sediment supply, the resulting impacts are normally local, occurring in the immediate vicinity of the structure or at a deposition site a short distance downstream. Significant incidental take may occur if the affected area includes spawning habitat.

9.2.3.1.2 Scour

Scour may be observed upstream or downstream of culverts, around bridge piers or pilings, or in places where hydraulic effects direct streamflow against the bank. Scour effects are normally local, occurring very near the water crossing structure, but the scoured sediments may be transported downstream to contribute to impacts such as embedding and deposition.

9.2.3.1.3 Deposition

When a conduit is installed, direct impacts on waters can often be minimized by high-pressure directional drilling (HPDD), a trenchless method of crossing a watercourse using subsurface drilling with a pressurized bore fluid lubricant system (Fisheries and Oceans Canada 2006). HPDD is used to install cables and pipelines for gas, water, telecommunications, fiber optics, power, sewer, oil, and water lines underneath watercourses. WAC 220-110-100 provides little protection against potential habitat impacts arising from boring of conduits, providing only that launch and receiving pits be isolated from the water body and that wastewater from the activity be routed to an area outside the ordinary high water line.

"Frac-outs" constitute a distinctive form of fine sediment deposition that sometimes occurs during HPDD operations. A frac-out is the escape of drilling mud into the environment as a result of a spill, tunnel collapse, or rupture of mud to the surface. Frac-outs are caused when excessive drilling pressure results in drilling fluid propagating vertically toward the surface. The principal constituent of the drilling fluid is clay, specifically bentonite, although a variety of secondary constituents may be added to the fluid.

The potential for frac-outs can be limited by careful monitoring, use of appropriate equipment, sufficient depth of conduit placement, appropriate boring pit location, and having response plans ready in the event that a frac-out occurs (NMFS 2005a, FERC 2005).

9.2.3.1.4 Substrate Modification

NMFS (2006a) assesses incidental take due to fill placement or culvert installation as proportional to the area of habitat lost.

Substrate modification due to conduit placement was largely not addressed in the reviewed literature.

9.2.3.1.5 Rapid Channel Change

A plugged culvert at a road fill can cause debris flow damming. Debris flows occur in response to natural causes as well as forest practices, and past experience has shown that some debris flows occur in every severe rainfall event that affects Washington. Such events, because of their burden of LWD and sediment, can easily exceed the calculated 100-year flow volume of the affected stream and thus have a high risk of plugging a culvert that is designed to pass a 100-year flow volume, resulting in a dam-break flood, a more severe debris flow, or a channel avulsion. Because debris flows can be expected to occur in vulnerable channels conveyed via culverts, debris flows can be regarded as a cumulative impact risk resulting from culvert installation. Debris flow or dam-break floods triggered by blockage and subsequent failure of a water crossing have the potential to result in incidental take of any potentially covered species and their habitats in the affected stream reach.

9.2.3.2 Altered Riparian and Shoreline Vegetation

Altering riparian and shoreline vegetation could result in a moderate to high risk of take.

9.2.3.1 Water quality

Turbidity may occur during construction due to accidental discharge of high pressure directional drilling (HPDD) fluids, disturbance of the streambed, or runoff from the work site into the stream, and turbidity may occur during operations if the water crossing structure channels flows to the stream. One of the highest-risk activities, with potential to cause mortality due to short-term acute turbidity exposure, is HPDD.

Fine sediment deposition also poses an incidental take risk to invertebrates.

Incidental take risk associated with dissolved oxygen impacts is probably quite low.

9.2.3.2 Conclusions of the Risk Evaluation

Table 9-7 summarizes the analysis of incidental take risk from water crossing structures. This risk evaluation is at best a qualitative assessment and is based strongly on the professional experience of the analysis team within the context of their work in ESA implementation. In general terms, activities in the low-risk category appear to be well suited for programmatic approval, whereas activities in the high-risk category would require consideration of project-specific elements (e.g., environmental setting, size, and installation technique) and present a clear need to implement conservation measures to reduce the risk of take. The appropriateness of programmatic approval of activities in the moderate-risk category is debatable and would depend in part on the use of conservation measures. The risk evaluation summarized in Table 9-7 assumes that potentially covered species or their habitat are present when the described impact occurs; thus, impacts and risk may be avoided by avoiding habitat for potentially covered species and may be minimized by performing the activities when potentially covered species are absent from the site.

Table 9-7: Conclusions of the Risk Evaluation

Activity	Low Risk	Moderate Risk	High Risk
Water crossing structures per WAC 220-110-070	• Work not requiring channel dewatering; • Work that does not alter channel form; • Structures in areas with little sediment transport; • Structures not requiring fill placement within the channel or floodplain; • Structures that do not use treated wood; • Structures that do not channel runoff to the water body; • Structures located in areas lacking submerged aquatic vegetation; • Structures that do not require removal of riparian vegetation; • Work that does not require production of in-water sound with peak levels more than 150 dB; • Structures that are built and operated without artificial illumination of the water surface; • Structures causing little increased shading of the water surface; • Activities avoiding the impacts potentially causing “moderate” or “high” risk.	• Work not requiring channel dewatering; • Projects that use hydraulic modeling to demonstrate minimal alteration of channel form and minimal modification of the floodway; • Structures requiring little or no fill placement within the channel; • Structures use treated wood; • Structures that channel runoff to the water body, when that runoff is treated in accordance with state and local stormwater treatment requirements; • Structures that have only temporary impacts to submerged aquatic vegetation; • Structures that have only temporary impacts to riparian vegetation; • Work that does not require production of in-water sound with peak levels more than 180 dB; • Structures that are designed to minimize artificial illumination of the water surface; • Structures that are designed to minimize shading of the water	• Work requiring channel dewatering; • Projects that do not use hydraulic modeling to demonstrate minimal alteration of channel form and minimal modification of the floodway, or for which hydraulic modeling does not show minimal alteration; • Structures requiring fill placement within the channel; • Structures that channel untreated runoff to the water body, • Structures in areas of submerged aquatic vegetation that are used by dependent species (e.g., Olympic mudminnow, freshwater mussels); • Structures that require removing LWD in lotic waters; • Structures permanently removing riparian vegetation; • Work that requires production of in-water sound with peak levels more than 180 dB; • Structures that fail to minimize artificial illumination of the water surface; • Structures that fail to minimize shading of the water surface;

Activity	Low Risk	Moderate Risk	High Risk
		surface; - Activities requiring vessel use; - Activities avoiding the impacts potentially causing "high" risk.	- Activities requiring in-water operation of mechanized equipment other than vessels.
Conduit crossings per WAC 220-110-100	- All provisions above, plus: - Work not requiring HPDD; - Work not requiring trenching "in the wet"	- All provisions above, plus: - Work requiring HPDD but potentially covered species not present; - Work requiring trenching "in the wet" but potentially covered species not present; - Absence of potentially covered species confirmed via survey by qualified biologist.	- All provisions above, plus: - Work requiring HPDD and potentially covered species may be present; - Work requiring trenching "in the wet" and potentially covered species may be present
Utility lines per WAC 220-110-310	- All provisions above; no additional provisions	- All provisions above; no additional provisions	- All provisions above; no additional provisions

9.3 General Risk of Take: 2007 White Papers (Channel Modifications, Fish Passage, Fish Screens, Flow Control Structures, Habitat Modifications, Marinas and Shipping Terminals, and Shoreline Modifications)

The risk of take is rated by impact mechanism for each species based on the assumptions presented in Table 9-8. (Also appears earlier in the consolidation as Table 6-3.)

Table 9-8. Definitions of the terminology used for risk of take determinations.

Risk of Take Code	Potential for Take	Definition
H	High	Stressor exposure is likely to occur with high likelihood of individual take in the form of direct mortality, injury, and/or direct or indirect effects on long-term survival, growth, and fitness potential due to long-term or permanent alteration of habitat capacity or characteristics. Likely to equate to a Likely to Adversely Affect (LTAA) finding.

M	Moderate	Stressor exposure is likely to occur, causing take in the form of direct or indirect effects potentially leading to reductions in individual survival, growth, and fitness due to short-term to intermediate-term alteration of habitat characteristics. May equate to an LTAA or a Not Likely to Adversely Affect (NLTAA) finding depending on specific circumstances.
L	Low	Stressor exposure is likely to occur, causing take in the form of temporary disturbance and minor behavioral alteration. If that take is insignificant or discountable, it would equate to an NLTAA finding.
I	Insignificant	Stressor exposure may potentially occur, but the likelihood is discountable and/or the effects of stressor exposure are insignificant. Likely to equate to an NLTAA finding.
N	No Risk	No risk of take ratings apply to species with no likelihood of stressor exposure because they do not occur in habitats that are suitable for the activity in question, or the impact mechanisms caused by the activity will not produce environmental stressors.
?	Unknown	Unknown risk of take ratings apply to cases where insufficient data are available to determine the probability of exposure or to assess stressor response.

Assessing risk of take assumes the following:

- HPA-permitted activities result in significant modification of the project site and the surrounding area, altering the environmental characteristics of the natural shoreline, bed, or water body.
- The impact mechanisms produced by development of the structures create environmental stressors.
- The risk of take resulting from stressor exposure will vary by species, depending on the nature of stressor exposure, as well as the sensitivity of the species and life-history stage exposed to the stressor.
- The magnitude, timing, duration, and frequency of each impact mechanism will vary widely with the project scale and location.
- The assessment of risk of take associated with each impact mechanism is broad and applies a “worst-case scenario” standard.
- This assessment is conditioned by the species occurrence and life-history specific uses of habitats where the particular type of structure is typically developed. A structure that would be built only in deep water would not affect species that occur only in shallow

water; a structure that would be built only in fresh water would not affect marine species or life-stages.

9.3.1 General Risk of Take from Construction, Operations, and Maintenance

9.3.1.1 Visual and Physical Disturbance

Visual disturbance and physical disturbance are expected to produce moderate risks of take for motile life-history stages due to temporary disturbance and displacement. Specifically for fish behavior, visual and physical disturbance can cause temporary avoidance and startle responses, compelling individuals to move out of the affected habitats or to assume a cryptic posture. Such disturbances will increase stress and exertion, may alter spawning and foraging behavior, or increase the risk of predation if fish are startled away from protective habitat. These effects may lead to decreased survival, growth, fitness, and spawning success, which equates to a moderate risk of take. Non-motile species or life-history stages are unable to escape or avoid physical disturbance. Therefore, they are at increased risk of mechanical injury from crushing or burial during construction, which constitutes a high risk of take.

9.3.1.2 Noise

Specific information on the risk of take associated with underwater noise is relatively limited for the majority of HCP species. For the purpose of ESA consultation, most available research has focused on the effects of pile driving related underwater noise on fish. This subject has received the most scrutiny because pile driving is a relatively common activity that produces noise stressors of sufficient magnitude to cause observed injury and mortality in fish by a number of mechanisms (e.g., cardiovascular and other tissue damage, hearing organ damage). A sufficient base of information has been assembled to establish effects thresholds for disturbance and injury in the HCP salmonid species.

Aside from pile-driving, noise produced by the in-water operation of heavy equipment is unlikely to exceed established injury thresholds. Noise related disturbance may occur in the form of acute spikes in underwater sound pressure levels from equipment impacts, and continuous noise created by vessel engines, generators, and pump or dredge operation.

Noise stressors produced by construction are likely to exceed levels sufficient to cause disturbance and behavioral modification, or to cause other physiological responses detrimental to survival, growth and fitness. Behavioral modification and habitat displacement from noise exposure may lead to increased exertion, alteration of feeding behavior, and increased predation exposure. Auditory masking effects caused by protracted alteration of the ambient noise environment (e.g., from extended vessel and motorized equipment operation) may affect their ability to detect predators and prey. Behavioral and auditory masking effects would generally be temporary to short-term in nature, lasting for the duration of the construction activity. Prolonged exposure to elevated ambient noise levels may also cause temporary changes in hearing sensitivity in certain fish species. These hearing threshold effects may last for some period after activities are completed (e.g., hours to days). Collectively, these effects may limit the survival, growth, and fitness of individuals exposed to these stressors. Because these stressors are short-term in nature, stressor exposure equates to a moderate risk of take.

9.3.1.3 Channel Dewatering

Temporary dewatering and flow bypass with fish removal and relocation from work areas may be required for some construction projects. Even when dewatering is not required for construction and maintenance, exclusion areas are often created around the work sites to contain sediments and other pollutants and to reduce the magnitude of stressor exposure. This construction and maintenance activity poses a relatively high risk of take. Well-designed protocols and trained personnel are necessary to avoid high levels of mortality. Even with appropriate protocols and experienced field crews, high levels of mortality can result. For example, NOAA Fisheries evaluated take associated with dewatering and fish handling in a recent biological opinion. They estimated that salmonid mortality rates in the range of 8 to as high as 20 percent may occur even when trained personnel are used, and have assumed an injury rate of 25 percent (NMFS 2006).

Mortality rates may be even higher in areas with complex substrate and bathymetry. During the egg, larval, or juvenile life-history stage of many species, individuals may be too small or too cryptic to collect and relocate effectively (e.g., juvenile salmonids hiding in cobble interstices, river lamprey ammocoetes buried in fine substrate, larval or juvenile dace). Mortality is the expected outcome for any individuals stranded within the exclusion area. Even in the absence of mortality, fish handling and relocation may result in stress and injury, as well as increased competition for forage and refuge in the relocation habitat. Moreover, the act of capture, handling, or forced behavioral modification of an ESA-listed species constitutes harassment, which is considered a form of take. Thus, the permitting of channel and work area dewatering poses a high risk of take of varying levels of severity depending on habitat and species-specific factors.

In addition to these effects, the act of dewatering a stream and redirecting flow may pose a barrier to fish migration. Delays in migration can lead to adverse effects on spawning fitness, can increase exposure to predation and poaching, and can deny juvenile fish access to rearing habitats during critical periods. These effects also constitute a moderate risk of take of HCP species with migratory life-history stages.

9.3.2 General Risk of Take from Hydraulic and Geomorphic Modifications

Flow regime, channel geometry, and substrate composition and stability are dominant factors determining aquatic habitat structure in riverine environments. Alteration of any of these habitat components can change the suitability of the habitat for various life-history stages of HCP species. These habitat alterations are essentially permanent and continuous, and can lead to changes in the productivity of the habitat for spawning, forage, rearing, and refuge. In a worst-case scenario, these effects are in turn likely to lead to reduced spawning success, as well as reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat. This equates to a high risk of take for species with exposure to these impact mechanisms.

9.3.2.1 Altered Hyporheic Exchange

Hyporheic exchange is an important component of ecosystem function (including water quality moderation) in riverine environments. Alterations to hyporheic exchange have the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species in the long-term, equating to a high risk of take.

Species with a high risk of take include those with life-history stages that are dependent on hyporheic exchange for its beneficial effects on water temperature and dissolved oxygen levels. For example, most salmonids preferentially spawn in areas with groundwater-induced upwelling, which promotes oxygenation of spawning gravels. Alteration of hyporheic exchange in environments suitable for spawning could potentially affect egg survival and reduce the availability of suitable spawning habitat, resulting in reduced spawning success. Similarly, groundwater inflow can provide important thermal refugia for migrating adult and rearing juvenile salmonids during periods with high water temperatures. A reduction in the amount of thermal refugia may negatively affect survival during these life-history stages. Similar effects would be expected for other coldwater fish species with low thermal tolerance thresholds, such as pygmy whitefish. More generally, hyporheic exchange also plays a key role in nutrient cycling and food web productivity in alluvial bed rivers. Activities resulting in significant alteration of hyporheic exchange could adversely affect food web productivity, limiting foraging opportunities for fish and invertebrate species dependent on these types of environments.

9.3.2.2 Altered Wave Energy, Altered Current Velocities, and Altered Nearshore Circulation Patterns

Wave energy, current velocities, and circulation patterns are all important determinants governing nearshore habitat characteristics in marine and lacustrine systems. These factors determine habitat suitability for a number of species-specific life-history processes. For example, wave energy conditions, currents, and circulation patterns will have a strong influence on nearshore water temperatures, shoreline stability, sorting and transport of sediments, and the accumulation of allochthonous and autochthonous materials. Many fish species selectively spawn in locations where current and circulation patterns promote the settling of planktonic larvae in favorable environments for rearing. Alteration of these patterns can cause larvae to be transported to unfavorable environments. Similarly, juvenile fish rearing in nearshore environments selectively choose environments with suitable wave energy and current conditions. These impact mechanisms can fundamentally alter habitat suitability for these uses, leading to decreased survival, growth, and fitness. This translates to a moderate to high risk of take for those HCP species that are dependent on these habitats during some phase of their life history.

9.3.2.3 Altered Sediment Supply, Substrate Composition and Stability

Sediment supply and substrate composition are fundamental components of the nearshore ecosystem structure in marine and lacustrine systems. Because substrate composition is an important determinant of community structure in the nearshore environment, these habitat changes can fundamentally alter community structure and habitat suitability for species dependent on the original habitat condition. This equates to a moderate-to-high risk of take for species that are dependent on these habitats due to effects on the survival, growth, and productivity of exposed life-history stages.

9.3.2.4 Altered River-Floodplain Connectivity

Lateral habitat connectivity is an important feature of riverine environments that contributes to their productivity. The implications of this degraded connectivity are significant for ecosystem productivity. A number of HCP species are dependent on off-channel and floodplain habitats during one or more life-history stages. Reduction in the availability of suitable habitat will lead to increased competition for available habitat, decreased growth and fitness, increased exposure to predation, and potentially decreased availability of suitable spawning sites. While these effects primarily concern fish, invertebrate species such as mussels could also be affected due to reduced productivity of host fish populations. The effects on survival, growth, fitness, and productivity caused by long-term alteration of environmental and habitat characteristics imposed by altered river-floodplain connectivity equates to a high risk of take.

9.3.2.5 Altered Freshwater Inputs/Altered Groundwater-Surface Water Exchange

Freshwater inputs to the marine nearshore environment are demonstrably linked to a number of important habitat parameters such as temperatures in forage fish spawning substrates, eelgrass distribution, and habitat selection by certain fish species. Hyporheic exchange is also an important component of ecosystem function in lacustrine and riverine environments. Alteration of groundwater inputs would be expected to cause a corresponding alteration in the distribution of desirable habitat features and availability, which has the potential to affect survival, growth, fitness, and (in some cases) the spawning productivity of a range of species. This equates to a risk of take ranging from low to high, depending on species-specific life-history characteristics and habitat requirements.

9.3.3 General Risk of Take from Ecosystem Fragmentation

Ecosystem fragmentation is an impact mechanism that incorporates the collective effects of habitat modification in the footprint of the structure, the resulting effects on the migration and dispersal of organisms, hydraulic modification, the transport, distribution, and biogeochemical processing of LWD, other organic material, nutrients, and pollutants, and the impact mechanisms imposed by hydraulic and geomorphic modifications.

Modification of downstream transport processes can lead to alteration in habitat complexity, changes in nutrient cycling, and subsequent hydraulic and geomorphic modifications. Each of these perturbations is associated with some risk of take. Given the long-term nature of these effects and the significance of altered ecosystem function, the risk of take is generally considered high.

Complex channels capture and retain sediment, which promotes the formation of pools and other hydraulically complex features. Hydraulic complexity in turn encourages the sorting and deposition of sediments and organic material in diverse patches, supporting food web productivity and providing spawning and rearing habitat for a diverse array of species. Diverse habitat patches support a biologically diverse community. Channel simplification or channel downcutting reduces the longitudinal distribution and frequency of these habitat patches across the riverine landscape, and can lead to fragmentation of floodplain and off-channel habitats from the riverine ecosystem. Reduction in habitat complexity leads to reduced food web productivity, as well as the reduced availability of habitats suitable for HCP species that occur in these environments. For example, side channel habitats are preferentially selected by various species of salmonids (e.g., sockeye salmon) for spawning. These habitats also provide key winter rearing and storm refuge habitats for coho salmon, steelhead, spring Chinook, native char (bull trout and Dolly Varden), and other species. Floodplain wetlands are also highly productive refuge habitats for a variety of species, such as coho salmon, during high winter flows. The reduction in suitable refuge and foraging habitat area caused by ecosystem fragmentation increases competition for remaining habitat, predation risk, and risk of displacement to habitats unfavorable for rearing. Because these effects are extensive and intermediate term to long term in nature, this equates to a high risk of take for HCP species.

9.3.4 General Risk of Take from Riparian Vegetation and Large Woody Debris Modifications

9.3.4.1 Altered Riparian Shading and Altered Ambient Air Temperature Regime

The risk of take from riparian vegetation removal varies depending on the nature of the project and the type of environment in which it is implemented.

The influence of riparian shading on water temperatures in the nearshore marine environment is limited in most circumstances. However, specific microhabitats (e.g., upper intertidal beaches used as spawning habitat by various fish species, and pocket estuaries that are isolated during tidal exchange) can experience significant changes in microclimatic conditions when riparian vegetation is altered. This equates to a moderate-to-high risk of take for those species with a demonstrable dependence on these habitats because the reduction in suitable habitat area caused by these impact mechanisms will lead to reduced survival, growth, and fitness, and these effects will be long term in nature.

Riparian shading in lacustrine environments can have a pronounced effect on nearshore water temperatures. The effect of riparian modification on the ambient air temperature regime is less clear and depends on a range of site-specific environmental factors. In general, water temperatures in lacustrine environments are predominantly driven by solar radiation exposure, seasonal stratification, turnover rate, and the temperature of source water. However, specific microhabitats such as shallow waters in protected embayments may be sensitive to temperature effects if shading and ambient air temperatures are altered by riparian modification. Such temperature effects may alter the suitability of these habitats for species that use them during some portion of their life history. These effects would be long term in duration and seasonal in frequency, meaning that these habitats may be unavailable or unsuitable for rearing for a significant segment of a population’s life history. This equates to a moderate-to-high risk of take for those species with a demonstrable dependence on these habitats because the reduction in suitable habitat area caused by these impact mechanisms will lead to reduced survival, growth, and fitness.

Removal of riparian vegetation can affect the temperatures of streams and smaller rivers, producing a range of potential effects on fish and wildlife species. In smaller streams, stream temperature effects may influence local habitat suitability and by extension affect the survival, growth, and fitness of exposed species and life-history stages.

In larger rivers, this effect will be far less pronounced. Water temperatures in larger rivers are less influenced by localized shading and ambient air temperature than by the combined effects of basin conditions in upstream areas of the watershed, hydromodification (e.g., dam and reservoir development), and other factors that influence water temperatures flowing through the affected area. The risk of take associated with altered temperatures is insignificant and the localized effects discountable in large rivers.

9.3.4.2 Altered Allochthonous Inputs

Riparian vegetation is an important source of nutrient input to the aquatic environment, strongly influencing the productivity of the aquatic food chain. Allochthonous nutrient inputs include sources such as insect-fall, leaf litter, and other organic debris, and LWD inputs that contribute both organic material and habitat complexity. These inputs clearly contribute to aquatic food web productivity in nearshore, lacustrine, and riverine environments. In riverine environments, the importance of allochthonous inputs to food web productivity decreases along a downstream gradient. As rivers grow in size, the contributions of autochthonous production and nutrient cycling to the food web increase. The science regarding the significance of allochthonous inputs in marine nearshore environments is relatively limited.

In smaller streams, allochthonous inputs are more important to food web productivity, while they provide a minor contribution in the lower reaches of large river systems. The loss of allochthonous production from a project that removes riparian vegetation or LWD near the mouth of a large river will produce related stressors of potentially far lower magnitude than a series of projects in a small, higher elevation stream. In smaller streams, a localized reduction in food web productivity might result, leading to decreased foraging opportunities, decreased overall habitat suitability, and decreased growth and fitness. This equates to a moderate risk of take for a range of HCP species that are dependent on riverine rearing conditions.

In marine, lacustrine, and riverine environments, LWD recruitment is an important contributor to habitat structure. Because removal of riparian vegetation and LWD has the potential to alter food web productivity and habitat complexity, it is likely to affect the survival, growth, and fitness of those species dependent on the nearshore environment for foraging and rearing during some portion of their life history. This equates to a moderate-to-high risk of take for those species with demonstrable dependence on these habitats.

9.3.4.3 Altered Habitat Complexity

In marine or lacustrine ecosystems, the physical structure of riparian vegetation, allochthonous inputs of LWD, shoreline stability, and effects on localized microhabitat conditions all contribute to habitat structure and complexity of the nearshore environment. Alteration of habitat complexity can have demonstrable effects on the productivity of aquatic species dependent on the nearshore environment, particularly fish species that spawn and rear in these areas, through effects on survival, growth, and fitness. These effects will be long-term. This equates to a moderate-to-high risk of take for species with demonstrable dependence on these habitats.

In riverine systems, modification of riparian vegetation alters habitat complexity primarily through the loss of undercut banks, root structure, and LWD inputs to the channel. The hydraulic and geomorphic effects of riparian vegetation modification can lead to further alterations in habitat complexity. Changes in flow and sediment transport conditions can lead to channel simplification and reduced availability of valuable habitat features, limiting the productive capacity of the affected habitat. Depending on the particular life history of the affected species, alteration in habitat complexity may limit the availability of suitable spawning, resting, and rearing habitat, and may alter foraging opportunities and predation exposure. In general, fish species that are dependent on habitats potentially affected through this mechanism of impact are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. This equates to a high risk of take for those HCP fish and invertebrate species occurring in riparian habitats.

9.3.4.4 Altered Shoreline and Bluff Stability

In riverine systems, removal of riparian vegetation can affect shoreline stability through the reduction in root cohesion and the loss of large woody debris (LWD) inputs that affect localized erosion and scour conditions. These effects may become pronounced in smaller stream systems where riparian modification effects are imposed over a considerable length of channel relative to the overall size of the stream. In the worst-case scenario, this type of riparian vegetation modification could result in decreased stream bank and shoreline stability, leading to erosion and elevated turbidity along the length of affected channel. These effects will be pronounced during seasonal high-flow conditions. Risk of take associated with this stressor varies depending on species-specific sensitivity to increased turbidity and dependence on the habitat structure provided by intact stream banks and shorelines. In general, more motile fish species experience only temporary behavioral alterations and low risk of take. In contrast, less motile fish life-history stages or sessile invertebrates could experience a high risk of take from decreased survival due to substrate sedimentation or mortality from smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success. These effects can become chronic and intermediate- to long-term in nature. Therefore, these effects equate to a high risk of take.

Modifications of marine or lacustrine riparian vegetation can lead to physical alteration of the shoreline and to bluff instability. In general, this would be expected to alter shoreline habitat conditions and habitat suitability for those species dependent on the nearshore environment during some portion of their life history. This equates to a moderate-to-high risk of take for those species with a demonstrable dependence on these habitats because the reduction in suitable habitat area caused by these impact mechanisms will lead to reduced survival, growth, and fitness.

9.3.4.5 Altered Groundwater-Surface Water Exchange

In the nearshore environment in both marine and lacustrine ecosystems, freshwater inputs are demonstrably linked to a number of important habitat parameters such as temperatures in forage fish spawning substrates, eelgrass distribution, and habitat selection by certain fish species (for instance, beach spawning sockeye salmon populations in lacustrine systems). Alteration of groundwater inputs would be expected to cause a corresponding alteration in the distribution of desirable habitat features and availability for species dependent on the nearshore environment. This equates to a high risk of take for species with demonstrable dependence on these habitats because the reduction in suitable habitat area caused by these impact mechanisms will lead to reduced survival, growth, and fitness, and these effects will be intermediate term to long term in duration.

In riverine systems, the influence of riparian vegetation on hyporheic exchange is well documented as an important component of ecosystem health. Alteration of riparian vegetation can lead to alteration of surface water and groundwater exchange, with important effects on the riverine ecosystem. For example, some salmonid populations that spawn in the mainstems of large river systems are dependent on groundwater inflow to maintain spawning habitat quality. For rearing salmonids and other temperature-sensitive species, groundwater inflow may provide thermal refuges important for survival during summer rearing periods. Hyporheic connectivity is also an important component of food web productivity. As such, this impact mechanism has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. Therefore, this mechanism is generally equated with a high risk of take for species exposed to this stressor, depending on species-specific life-history characteristics.

9.3.5 General Risk of Take from Aquatic Vegetation Modifications

Stressors imposed by aquatic vegetation modification occur through

- reduction in autochthonous productivity provided by the plant community; and
- the changes in habitat structure imposed by the removal of vegetation.

Autochthonous production by submerged aquatic vegetation is a source of primary and secondary production in the aquatic food web of the marine littoral zone. A diversity of species feed directly on live and fragmented submerged aquatic vegetation, forming the basis of the food web for a number of other species. Numerous species use submerged aquatic vegetation for cover and rearing during larval and juvenile life-history stages. Of specific interest, Pacific herring are (primarily) effectively obligate spawners on submerged aquatic vegetation in the low intertidal and subtidal zone.

The risk of take associated with alteration of aquatic vegetation varies depending on the environment type.

Aquatic vegetation plays a key role in the productivity of the nearshore marine ecosystem. Alterations of the submerged aquatic vegetation community through reduction in aerial extent or conversion to other habitat types (e.g., conversion of eelgrass habitat to algae and kelp) can reduce the productivity of these habitats, possibly affecting foraging opportunities for dependent life-history stages. This translates to a moderate to high risk of take for species dependent on these habitats through reduced survival, spawning success, or growth and fitness. In nearshore marine environments, submerged aquatic vegetation also provides habitat structure, creating vertical dimension and overhead cover. Alteration of habitat complexity can decrease the availability of suitable rearing habitat for species and life-history stages dependent on the nearshore environment, leading to increased predation risk and increased competition for suitable space, resulting in long-term effects on survival, growth, and fitness. This equates to a moderate to high risk of take for species dependent on aquatic vegetation functions in these environments. A high risk of take would only apply to species adapted to habitats with naturally abundant aquatic vegetation. Otherwise, only a moderate risk of take would be expected.

In most river systems in the Pacific Northwest, particularly in coldwater streams, aquatic vegetation plays a relatively small ecological role. Aside from native emergent vegetation confined to a relatively narrow range of depths, most aquatic vegetation species in rivers and lakes are invasive species. Therefore, changes in autochthonous production and habitat complexity imposed by alteration of aquatic vegetation may have relatively minor effects on the majority of HCP species occurring in riverine environments. The risk of take associated with altered autochthonous production and habitat complexity is expected to be low to moderate, except in specific cases where species are known to be dependent on aquatic vegetation (e.g., Olympic mudminnow), which would be associated with a high risk of take.

Vegetation plays a more significant role in lacustrine habitats, where emergent and submerged aquatic vegetation are often abundant in the photic zone and play a larger role in habitat structure and food web productivity.

9.3.6 General Risk of Take from Water Quality Modifications

Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects in exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of immobile lamprey ammocoetes buried in bottom substrates, fish exposed to accidental vessel spills in enclosed embayments). Even in the absence of mortality, exposure to a variety of contaminants can cause physiological injury and/or contaminant bioaccumulation, leading to decreased growth and fitness. Changes in nutrient loading may lead to detrimental changes in food web community structure, which may be limiting to growth and fitness.

9.4 Activity-Specific Risk of Take: 2007 White Papers (Channel Modifications, Fish Passage, Fish Screens, Flow Control Structures, Habitat Modifications, Marinas and Shipping Terminals, and Shoreline Modifications).

This section consolidates narrative and tabular summaries of the risk of take from white papers prepared in 2007: channel modifications, fish passage activities and structures, fish screens, flow control structures, habitat modifications, marinas and shipping terminals, and shoreline modifications. Each of these major categories is broken down into separate activity types. The risk-of-take summaries are organized by the type of activity or structure, the mechanism of impact, and the type of environment. In cases where the physical effects and related risk of take are similar between environment types, the risk of take discussion is grouped to avoid redundancy. A risk of take matrix for each type of activity, identifying the overall risk of take for each of the 52 HCP species (Tables 9-9 through 9-43) is presented at the end of each narrative. These matrices provide an individual risk of take for each species by impact mechanism category and environment type (i.e., riverine, marine and lacustrine). The matrices are derived from the impact mechanism and stressor specific risk of take ratings developed for the 52 HCP species in the exposure response matrices, which are presented in Appendix A to the original white papers. This risk of take assessment was developed based on the likelihood of exposure, for each of these 52 species, to the impact mechanisms and stressors imposed by each type of structure or activity as well as the sensitivity of exposed life-history stages to these stressors. The summary risk of take presented in the narrative and the matrices represents the greatest overall risk of take; a given activity could have a lower risk of take in some circumstances.

9.4.1 Channel Modifications: Dredging, Gravel Mining and Bar Scalping, Sediment Capping, and Channel Creation and Alignment

Channel modification activities are typically designed to promote human uses of the aquatic environment for purposes including navigation, flood control, pollution management, and landscape conversion. The resulting alteration of ecological process and imposition of stressors may persist well after project construction is completed. The magnitude of these stressors will vary depending on the scale of the project in question and the degree to which it modifies ecological conditions and processes.

In the original white papers, the risk of take for each type of channel modification was discussed separately. Those discussions concluded that, for the most part, the risks of take for each type of channel modification are quite similar to one another for the environments in which they occur. In this consolidation, the common risks are grouped. A risk of take matrix for each type of channel modification, identifying the overall risk of take for each of the 52 HCP species (Tables 9-9 through 9-12) is presented at the end of the narrative.

Of the four types of channel modification, **dredging** takes place in the widest variety of environments, including marine, lacustrine, large rivers and small streams. The nature and scale of an individual dredging activity can vary widely depending on its specific purpose and the environment in which it is implemented. A broad range of HCP species face the potential for stressor exposure from dredging. Species-specific risk of take ratings for dredging operations are presented by impact mechanism in Table 9-9. The species level risk of take ratings are conditioned based on the nature of the stressor exposure anticipated in each environment type.

Gravel mining and bar scalping is anticipated to occur only in alluvial bed rivers where the desirable substrate resources are abundant. Gravel mining and bar scalping operations are expected to impose impact mechanisms and related ecological stressors similar to those caused by dredging in riverine environments. Therefore, the risk of stressor exposure and resulting risk of take are considered to be essentially the same as those described for dredging activities in smaller riverine environments. Species specific risk of take ratings for gravel mining and scalping operations are presented by impact mechanism in Table 9-10.

Sediment caps are used primarily as a means of sequestering contaminated substrate material, isolating these materials from the aquatic environment and limiting potential exposure pathways for toxic substances. Predominantly employed in the marine environment, sediment caps are occasionally used in lacustrine environments and in riverine environments in depositional settings where scour of the cap is unlikely to occur. These environments are most commonly found in estuarine reaches, which for the purpose of this white paper are considered to be part of the marine environment. Species specific risk of take ratings for sediment cap development and maintenance are presented by impact mechanism in Table 9-11.

Artificial or realigned channels are extensive hydromodifications specifically designed to reconfigure the aquatic environment to promote human uses. (Channel realignment projects conducted for the primary purpose of habitat restoration are discussed under Habitat Modifications.) Extensive in size and pervasive in effect, artificial or realigned channels impose a number of ecological stressors on the environment through essentially permanent alteration of habitat and water quality conditions. These types of channel modifications are commonly accompanied by dike and levee development and may be maintained by maintenance dredging, structures, and activities that impose their own risk of take. HCP species occurring in environments modified by this type of project will typically experience a high risk of take from one or more impact mechanisms. Species specific risk of take ratings for channel creation or alignment are presented by impact mechanism in Table 9-12.

9.4.1.1 Construction, Operations, and Maintenance

The construction component of a channel modification activity is typically temporary to short-term in duration, lasting from days to weeks. Stressors associated with channel modification include visual, physical, and noise related disturbance from vessel and equipment operation. The risk of take associated with these stressors varies depending on the nature of the exposure and the sensitivity of species and life-history stages exposed. Motile species and life-stages may face a moderate risk of take resulting from behavioral avoidance, stress, and habitat displacement.

Construction-related stressors unique to one type of channel modification include the following.

- Dredging: Entrainment of organisms. Entrainment is the unintentional capture of organisms within the dredged material or the surrounding water column, and the unintentional removal of these organisms from the environment. Entrainment is a likely occurrence regardless of equipment type if non-motile species or life-history stages are present during dredging activities. Motile fish species and life-history stages are most likely able to avoid entrainment. Entrainment is likely to cause mortality through mechanical injury, smothering, or stranding. Species with one or more non-motile life-history stages (i.e., fish eggs and demersal or planktonic larvae and juveniles, as well as the HCP invertebrate species) are vulnerable to entrainment, and in environments suitable for dredging activities face a high risk of take from injury or mortality.

- Dredging: Reoccurrence. Dredging may recur at interannual to decadal frequencies. Larger projects (such as Columbia River navigation channel dredging) may extend over several months of continuous activity.

- Channel creation or realignment: Effects of connection to the existing channel. The construction and maintenance of artificial channels involves significant disturbance and alteration of stream banks and lacustrine and marine shorelines. Channel and bed disturbance may lead to behavioral and physiological stress on species or life-history stages exposed to the disturbance, or may limit the availability and suitability of habitats for sensitive life-history stages during critical periods. Non-motile species exposed to these stressors may face immediate effects on survival if occupied habitats are eliminated. Once the barriers isolating the newly excavated channel are breached, it will fill by drawing surface water from existing surface water, creating a potential dewatering and stranding hazard as well as potential entrainment into the new channel environment. In marine and larger lacustrine systems, the dewatering and stranding hazard is likely limited, because the volume of the new channel will be relatively insignificant. In contrast, in riverine environments the creation of the new channel may redirect the entire surface flow leading to dewatering of the existing channel. Aquatic species trapped in rapidly dewatering habitats face risk of mortality from stranding, particularly non-motile species and life-history stages. Motile species able to avoid stranding will be displaced from existing habitats and forced to relocate within disturbed and/or occupied habitat that may present limited foraging opportunities, which could limit survival, growth, and fitness. It is generally presumed that care will be taken during channel connection to dewater slowly, reducing stranding risk. Consistent with a worst-case scenario approach however, this activity must be associated with a high risk of take, particularly for non-motile species and life-history stages that may be exposed to this stressor. These effects would be equated with a moderate to high risk of take, depending on species specific sensitivity.

9.4.1.2 Hydraulic and Geomorphic Modifications

Channel modifications impose significant changes in the hydraulic and geomorphic characteristics of the project area and the surrounding environment. These modifications can in turn significantly alter the suitability of the affected habitats for HCP species. Dredging, sediment capping, gravel mining and bar scalping, and channel creation and realignment are expected to cause similar hydraulic and geomorphic modifications. These activities may range in scale from removing or placing a relatively small amount of sediment using equipment operating from a bank, to multi-year maintenance dredging projects on the Columbia River employing ships, barges, or other floating platforms. The risk of take ratings are therefore applicable across a broad range of environment types.

Channel modifications change flow regimen, channel geometry, and substrate composition. These alterations are likely to change local channel hydraulics and sediment transport and stability. The effects on survival, growth, fitness, and productivity caused by long-term alteration of environmental and habitat characteristics equates to a high risk of take.

Channel modifications can lead to the alteration of groundwater exchange in riverine environments through changes imposed on channel geometry. Increased flood conveyance may lead to reduced water surface elevations, and reduced connectivity between the river and the floodplain during peak flows. This is likely to lead to changes in hyporheic exchange with detrimental effects on ecological productivity.

Channel modifications in the marine environment will modify hydraulic and geomorphic conditions in and around the project area, resulting in the imposition of several impact mechanisms and related stressors. Risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on the affected nearshore environment.

Channel geometry and substrate composition and stability are dominant factors determining aquatic habitat structure in riverine environments. Alteration of any of these habitat components can change the suitability of the habitat for various life-history stages of HCP species. These habitat alterations are essentially permanent and continuous, and can lead to changes in the productivity of the habitat for spawning, forage, rearing, and refuge. In a worst-case scenario, these effects are in turn likely to lead to reduced spawning success, as well as reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat. This equates to a high risk of take for species with exposure to these impact mechanisms.

Artificial channels and channel realignments that are put in place to facilitate conversion of land to human uses (rather than those that are put in place for habitat restoration) alter flow conditions in riverine environments by simplifying the channel geometry, often by straightening the channel and changing (increasing) the stream gradient. Artificial channels are often created in conjunction with dikes and levees to accelerate the flow of water through the landscape, concentrating high flows in the stream channel, accelerating flow velocity and erosive forces. Substrate composition and stability in the channel change through the loss of sources of sediment and altered sediment transport capacity. The effects of artificial channels on HCP species are complex and variable, depending on the position of the hydromodification in the riverine environment and how the affected habitats are used by HCP species. Applying a worst-case scenario perspective, these pervasive long-term effects would be expected to reduce habitat suitability for species utilizing the affected environment, limiting individual survival, growth, and fitness and overall population productivity. This equates to a high risk of take.

Changed hydraulic, geomorphic, and riparian conditions imposed by channel modifications are likely to alter groundwater and surface water exchange in the project area and downstream. This hyporheic exchange is an important component of ecosystem function (including water quality moderation) in riverine environments. Therefore, this impact mechanism has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. Because this effect will be pervasive and essentially permanent, this mechanism is generally equated with a high risk of take for species exposed to this stressor, depending on species-specific, life-history characteristics. Species facing high risk of take include those with life-history stages that are dependent on hyporheic exchange for its beneficial effects on water temperature and dissolved oxygen levels.

9.4.1.3 Ecosystem Fragmentation

Depending on their siting and configuration, channel modifications can present a significant potential for habitat loss and fragmentation in marine, lacustrine, and riverine environments. Large projects have the most potential for adverse effects.

In estuarine environments, channel modification projects that remove shallow bars at the riverine and marine interface potential accelerate the flow of water from estuaries into open ocean waters. Altering bathymetry and flow conditions may in turn lead to changes in salinity, tidal exchange, and circulation patterns within the estuarine and nearshore environment, altering habitat conditions and potentially eliminating certain desirable habitat types.

Channel modification projects in nearshore environments may cause the conversion of shallow water to deeper water habitats, reducing the suitability of these habitats for certain species. For example, many salmonid species typically migrate as juveniles in shallow water along marine and lacustrine shorelines. The fragmentation of shallow water habitat along the shoreline may increase predation exposure and reduce foraging opportunities.

Channel modification projects can alter wave energy, current, and circulation patterns in the nearshore and offshore environment. Alteration of these habitat characteristics may render productive habitats less suitable for a given species or, in the case of organisms with a planktonic life-history stage, may hinder the dispersal and retention of eggs and larvae in areas suitable for rearing. Collectively, this can result in take through effects on survival, growth, and fitness of affected populations, which equates to a moderate risk of take for exposed species.

Channel modification projects reduce the structural complexity of instream habitat by changing channel geometry. They can simplify channel structure, disconnecting floodplain, off-channel, and terrestrial riparian habitats from the riverine ecosystem. They can alter longitudinal connectivity. They can influence the recruitment, transport, and retention of sediments, organic matter and nutrients and LWD. They can disconnect the channel from important sinks for pollutants. Reduction in habitat complexity leads to reduced food web productivity, and the reduced availability of habitats suitable for HCP species that occur in these environments. The reduction in suitable refuge and foraging habitat area caused by ecosystem fragmentation increases competition for remaining habitat, predation risk, and risk of displacement to habitats unfavorable for rearing. These effects are extensive and long lasting. Ecosystem fragmentation in riverine environments equates to a high risk of take.

The intended purpose of channel modification projects in smaller rivers and streams is often to improve flood conveyance capacity, limiting floodplain connectivity during high flow events. Localized changes in water surface elevation may lead to decreased inundation of off-channel and side channel habitats during high flow events. The effects on survival, growth, fitness, and productivity caused by the long-term alteration of environmental and habitat characteristics imposed by altered connectivity equates to a high risk of take.

9.4.1.4 Riparian Vegetation Modifications

In large bodies of water, channel modifications are expected to take place from floating platforms, barges/vessels, and/or existing overwater structures. Therefore, no modification of the riparian environment would be expected to occur and there will be no related risk of take.

On small to moderate sized streams and rivers that cannot practically be accessed from a floating dredge platform, channel modifications may result in riparian vegetation modification. Examples include:

- Modified stream channels in agricultural and urban settings that rapidly accumulate sediment and lose flood conveyance capacity. Some of these systems may incorporate sediment traps that are subject to routine maintenance dredging. Dredging activities in stream systems of this type can lead to extensive modification of riparian vegetation over a significant length of channel. Riparian recovery may be retarded if dredging activities occur at a high frequency (e.g., annually or biennially), meaning that the stressor exposure will occur over an extended duration.

- Building and maintaining dikes and levees along realigned channels may require removal of riparian vegetation.

- Gravel mining and bar scalping may require removal of riparian vegetation.

Loss of riparian shading can affect the temperatures of streams and smaller rivers, producing a range of potential effects on fish and wildlife species. In smaller stream systems, temperature effects of channel modifications can become pronounced. Increased stream temperatures can lead to a variety of unfavorable effects on HCP species occurring in these environment types. Due to their potential to occur over an extended duration, these effects are equated with a high risk of take. In higher order river environments, this effect is far less pronounced. Water temperatures in systems of this nature are less influenced by localized shading and ambient air temperature than by the combined effects of basin conditions in upstream areas.

Channel modifications that repeatedly alter an extensive length of riparian zone relative to channel size may lead to chronic reduction in allochthonous inputs. In such cases, a localized reduction in food web productivity might result, leading to decreased foraging opportunities, decreased overall habitat suitability, and decreased growth and fitness. This equates to a high risk of take for a range for species that are dependent on riverine rearing conditions. This impact is likely to be greater in small streams than in the lower reaches of large river systems, where allochthonous inputs provide a minor contribution to food web productivity.

The influence of riparian vegetation on hyporheic exchange is well documented as an important component of ecosystem health. Alteration of riparian vegetation can in turn lead to an alteration of surface water and groundwater exchange, with important effects on the riverine ecosystem, especially in smaller streams and rivers. Altered surface and groundwater exchange has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. Applying a worst-case scenario perspective, channel modifications that cause chronic degradation of riparian vegetation may permanently alter groundwater-surface water interactions. This level of stressor exposure would impose a high risk of take on those HCP species dependent on groundwater/surface water exchange.

9.4.1.5 Aquatic Vegetation Modifications

Channel modification projects may lead to the modification, loss, or burial of aquatic vegetation in the project footprint and within the zone of hydraulic and geomorphic effects imposed by the channel modification. The resulting risk of take associated with these stressors varies based on the sensitivity of the HCP species and the environment type in which stressor exposure occurs.

Channel modification activities in the marine environment are most likely to be permitted only if they can demonstrate that losses of aquatic vegetation will be substantially limited and mitigated. However, in a worst case scenario a project could result in the loss of a substantial amount of aquatic vegetation habitat with extensive localized losses of autochthonous productivity and habitat structure. Because local bathymetry and substrate conditions are usually altered in the process, reduced habitat suitability may limit the potential for natural recovery following project completion. Alteration of marine littoral vegetation may in some cases lead to localized shifts in food web productivity, possibly affecting foraging opportunities for dependent species and life-history stages. This translates to a high risk of take resulting from decreased growth and fitness.

The effects of channel modification on aquatic vegetation in lacustrine and riverine habitats varies considerably depending on the scale of the activity, the nature of the affected habitat, and the sensitivity of the species exposed to the resulting ecological stressors. Modification of the submerged aquatic vegetation community in lakes and rivers can lead to decreased primary and secondary productivity, which in turn may affect overall food web productivity. In systems where the aquatic vegetation community is an important component of food web productivity, this can lead to a high risk of take through long-term, indirect effects on foraging success, growth, and fitness of species and life-history stages that depend on forage in the nearshore environment. A high risk of take would only apply to those species adapted to habitats with naturally abundant aquatic vegetation. Otherwise, only a moderate risk of take would be expected.

Aquatic vegetation-related stressors unique to one type of channel modification include the following.

- Dredging is used to manage aquatic vegetation in lakes, particularly for controlling invasive species.

- Sediment caps may alter substrate conditions, reducing the suitability of the substrates for rooted vegetation or the availability of hard substrates for encrusting vegetation or kelp holdfasts.

9.4.1.6 Water Quality Modifications

Channel modification projects can result in an alteration of the temperature regime. In marine and lacustrine settings, temperature alterations occur primarily through changes in current circulation and stratification induced by wave energy, current circulation and vertical mixing, and other hydraulic and geomorphic effects. In riverine environments, channel modifications can alter temperatures through reduction in riparian shading and change in groundwater-surface water interactions. Effects will persist for the life of the structure. Alteration in temperature regime attributable to channel modification is unlikely to be of sufficient magnitude to cause acute mortality, but may cause increased stress leading to decreased survival, growth, and fitness. Motile species may also exhibit behavioral avoidance of affected areas, increasing competition for available suitable habitats with attendant effects on survival, growth, and fitness. Ultimately, the suitability of the habitat for a range of species may be affected. Applying a worst-case scenario perspective, these effects are likely to lead to a high risk of take because channel modifications can cause long-term changes in the ecological factors that contribute to temperature regime changes.

Channel modifications can lead to altered dissolved oxygen levels through changes in water temperature regime, hydraulic and geomorphic effects leading to altered nutrient cycling and eutrophication, and chemical weathering of substances in substrate exposed by dredging. These effects vary in duration from short-term to long-term and their magnitude is dependent on site specific conditions. Altered dissolved oxygen levels are unlikely to lead to acute mortality, but may cause increased stress leading to decreased survival, growth, and fitness. Motile species may also exhibit behavioral avoidance of affected areas, increasing competition for available suitable habitats with attendant effects on survival, growth, and fitness. Applying a worst-case scenario perspective, these effects are likely to lead to a high risk of take because channel modifications can cause long-term changes in the ecological factors that contribute to dissolved oxygen levels.

Channel modifications are likely to result in a short-term increase in suspended sediment levels in the aquatic environment. Subsequent geomorphic effects may lead to increased erosion or changes in wave energy that may cause chronic elevation in suspended sediment loading as the system adjusts to the new hydraulic and hydrologic regime imposed by changes in channel geometry or local bathymetry. Non-motile species or life-history stages exposed to pulses of high concentrations of suspended sediment may suffer direct mortality, injury, or extreme physiological stress from burial and smothering or gill irritation and injury, while motile species may be able to avoid these stressors. Chronic elevation in suspended sediment levels caused by hydraulic and geomorphic adjustments would be less likely to reach levels sufficient to cause direct mortality, but may be sufficient to affect growth and fitness over the intermediate-term by limiting ecological productivity and the ability to detect prey species. The long-term risk of take from changes in suspended sediment concentrations and turbidity caused by channel modifications will be variable depending on the specific site conditions. However, given the potential for short-term injury or mortality resulting from elevated suspended sediment levels, a high risk of take must be assumed for HCP species that occur in suitable environments.

Channel modifications can induce changes in nutrient and pollutant loading through a number of mechanisms. In the marine environment, channel modifications have been associated with changes in estuarine tidal dynamics, which affects the processing and distribution of nutrients and pollutants. The effects of channel modifications on marine aquatic vegetation can lead to changes in nutrient cycling and pollutant sequestration. Dredging and sediment capping in Puget Sound has often been associated with the resuspension of contaminated sediments, creating new exposure pathways for organisms in the water column. In riverine environments, fragmentation of floodplain habitats due to channel modifications may affect the riparian buffering capacity and limit the contribution of floodplain habitats to nutrient cycling, leading to detrimental changes in water quality.

Equipment operations present the potential for the introduction of toxic substances from accidental spills from equipment used during the activity. Because some contaminant exposure and changes in nutrient loading induced by channel modifications may be intermediate-term to long-term in duration, these stressors are equated with a high risk of take in riverine, marine, and lacustrine environment types.

Table 9-9. Species- and habitat-specific risk of take for mechanisms of impact associated with dredging

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	M	M	H	H	H	H	H	H	H	N	N	M	H	M	H	M	M
Coho salmon	H	M	M	H	H	H	H	H	H	H	N	N	M	H	M	H	M	M
Chum salmon	H	M	I	H	H	I	H	H	I	H	N	N	M	H	I	H	M	I
Pink salmon	H	M	I	H	H	I	H	H	I	H	N	N	M	H	I	H	M	I
Sockeye salmon	H	M	M	H	H	H	H	H	H	H	N	N	M	H	M	H	M	M
Steelhead	H	M	M	H	?	H	H	?	H	H	?	N	M	?	M	H	?	M
Coastal cutthroat trout	H	M	M	H	H	H	H	H	H	H	N	N	M	H	M	H	M	M
Redband trout	H	N	M	H	N	H	H	N	H	H	N	N	M	N	M	H	N	M
Westslope cutthroat trout	H	N	M	H	N	H	H	N	H	H	N	N	M	N	M	H	N	M
Bull trout	H	M	M	H	H	H	H	H	H	H	N	N	M	H	M	H	M	M
Dolly Varden	H	M	M	H	H	H	H	H	H	H	N	N	M	H	M	H	M	M
Pygmy whitefish	H	N	M	H	N	H	H	N	H	H	N	N	M	N	M	H	N	M
Olympic mudminnow	H	N	N	H	N	H	H	N	H	H	N	N	H	N	N	H	N	N
Lake chub	H	N	M	H	N	H	H	N	H	H	N	N	M	N	M	H	N	H
Leopard dace	H	N	M	H	N	H	H	N	H	H	N	N	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	H	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	M	H	N	H	H	N	H	H	N	N	M	N	M	H	N	H
Umatilla dace	H	N	M	H	N	H	H	N	H	H	N	N	M	N	M	H	N	H
Pacific lamprey	H	I	H	H	N	H	H	H	H	H	N	N	M	I	M	H	L	H
River lamprey	H	M	H	H	N	H	H	H	H	H	N	N	M	H	M	H	M	H
Western brook lamprey	H	N	H	H	N	H	H	N	H	H	N	N	M	N	M	H	N	H
Green sturgeon	N	M	N	N	N	N	N	?	N	N	N	N	N	?	N	N	L	N
White sturgeon	H	M	H	H	N	H	H	?	H	H	N	N	H	?	H	H	L	H
Eulachon	H	H	N	H	N	N	H	H	N	M	N	N	I	H	N	H	H	N
Longfin smelt	H	H	H	H	N	H	H	H	N	M	N	N	I	H	H	H	H	N
Pacific sand lance	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Lingcod	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Pacific cod	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Walleye pollock	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
China rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Northern abalone	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Newcomb's littorine snail	N	N	N	N	H	N	N	H	N	N	N	N	N	N	N	N	H	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	N	M	N	M	H	N	H
Western ridged mussel	H	N	H	H	N	H	H	N	H	H	N	N	M	N	M	H	N	H

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-10. Species- and habitat-specific risk of take for mechanisms of impacts associated with gravel mining and scalping.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Coho salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Chum salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Pink salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Sockeye salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Steelhead	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Coastal cutthroat trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Redband trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Westslope cutthroat trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Bull trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Dolly Varden	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Pygmy whitefish	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Olympic mudminnow	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Lake chub	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Leopard dace	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Umatilla dace	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Pacific lamprey	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
River lamprey	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Western brook lamprey	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Eulachon	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Longfin smelt	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb’s littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
California floater (mussel)	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-11. Species- and habitat-specific risk of take for mechanisms of impacts associated with sediment caps.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentations			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	N	M	M	N	H	H	H	H	H	N	N	N	N	H	M	N	M	M
Coho salmon	N	M	M	N	H	H	H	H	H	N	N	N	N	H	M	N	M	M
Chum salmon	N	M	I	N	H	I	H	H	I	N	N	N	N	H	I	N	M	I
Pink salmon	N	M	I	N	H	I	H	H	I	N	N	N	N	H	I	N	M	I
Sockeye salmon	N	M	M	N	H	H	H	H	H	N	N	N	N	H	M	N	M	M
Steelhead	N	M	M	N	?	H	H	?	H	N	?	N	N	?	M	N	?	M
Coastal cutthroat trout	N	M	M	N	H	H	H	H	H	N	N	N	N	H	M	N	M	M
Redband trout	N	N	M	N	N	H	H	N	H	N	N	N	N	N	M	N	N	M
Westslope cutthroat trout	N	N	M	N	N	H	H	N	H	N	N	N	N	N	M	N	N	M
Bull trout	N	M	M	N	H	H	H	H	H	N	N	N	N	H	M	N	M	M
Dolly Varden	N	M	M	N	H	H	H	H	H	N	N	N	N	H	M	N	M	M
Pygmy whitefish	N	N	M	N	N	H	N	N	N	N	N	N	N	N	M	N	N	M
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lake chub	N	N	M	N	N	H	N	N	H	N	N	N	N	N	M	N	N	H
Leopard dace	N	N	M	N	N	H	H	N	H	N	N	N	N	N	M	N	N	H
Margined sculpin	N	N	N	N	N	H	N	N	N	N	N	N	N	N	N	N	N	N
Mountain sucker	N	N	M	N	N	H	H	N	H	N	N	N	N	N	M	N	N	H
Umatilla dace	N	N	M	N	N	H	H	N	H	N	N	N	N	N	M	N	N	H
Pacific lamprey	N	I	H	N	N	H	H	H	H	N	N	N	N	I	M	N	L	H
River lamprey	N	M	H	N	N	H	H	H	H	N	N	N	N	H	M	N	M	H
Western brook lamprey	N	N	H	N	N	H	N	N	H	N	N	N	N	N	M	N	N	H
Green sturgeon	N	M	N	N	N	N	N	?	N	N	N	N	N	?	N	N	L	N
White sturgeon	N	M	H	N	N	H	H	?	H	N	N	N	N	?	H	N	L	H
Eulachon	N	H	N	N	N	N	H	H	N	N	N	N	N	H	N	N	H	N
Longfin smelt	N	H	H	N	N	H	H	H	H	N	N	N	N	H	H	N	H	N
Pacific sand lance	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Lingcod	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Pacific cod	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentations			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Walleye pollock	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
China rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Northern abalone	N	H	N	N	N	N	N	H	N	N	N	N	N	H	N	N	H	N
Newcomb's littorine snail	N	N	N	N	H	N	N	H	N	N	N	N	N	N	N	N	H	N
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
California floater (mussel)	N	N	H	N	N	H	H	N	H	N	N	N	N	N	M	N	N	H
Western ridged mussel	N	N	H	N	N	H	H	N	H	N	N	N	N	N	M	N	N	H

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-12. Species- and habitat-specific risk of take for mechanisms of impacts associated with channel creation and alignment.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Coho salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Chum salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Pink salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Sockeye salmon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Steelhead	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Coastal cutthroat trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Redband trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Westslope cutthroat trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Bull trout	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Dolly Varden	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Pygmy whitefish	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Olympic mudminnow	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Lake chub	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Leopard dace	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Umatilla dace	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Pacific lamprey	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
River lamprey	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Western brook lamprey	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Eulachon	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Longfin smelt	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

9.4.2 Fish Passage (Culverts, Fish Ladders/Fishways, Roughened Channels, Weirs, Trap and Haul)

Fish passage structures are usually intended to improve fish passage conditions relative to the existing state. While fish passage structures often do provide improvements in passage, for the purpose of assessing risk of take, in this analysis the baseline condition is the stream system in the absence of artificial structures. There are so many possible combinations of existing conditions and new fish passage structures (for example, installing a new structure where one doesn't exist, replacing a barrier culvert with a new culvert, replacing an old bridge with a new bridge, replacing a culvert with a bridge, etc.) that it would not have been possible to characterize the potential take associated with all of the possibilities. By comparing fish passage structures to a situation without a water crossing, the analysis evaluates the worst case scenario. The worst case analyses assume that the structure is likely to produce stressors of the greatest magnitude.

Culvert retrofits are not expected to cause riparian modification of any significance; therefore, the resulting stressors are expected to be minor and the risk of take low. **Culvert removal or replacement** requires significant in-water work and channel modification. Stressors associated with these activities would be associated with a high risk of take.

The impact mechanisms associated with **fish ladders, fishways,** and **weirs** produce a number of environmental stressors with the potential to impose risk of take of HCP species. The degree of risk associated with each impact mechanisms varies. Some mechanisms are expected to produce stressors with relatively low risk of take due to their limited extent and/or short-term nature. In contrast, some mechanisms may result in stressors with the potential to produce direct mortality or injury, or long-term modifications in habitat conditions detrimental to survival, growth, and fitness. These impact mechanisms would be associated with a high risk of take.

The impact mechanisms associated with the creation of **roughened channels** present several environmental stressors that lead to potential risk of take of HCP species. The degree of risk associated with each of these impact mechanisms varies, but roughened channels are generally associated with a relatively high risk of take in total.

The majority of **trap-and-haul** facilities are expected to be associated with a weir or some other type of flow control structure (with rare exceptions for trap-and-haul operations at natural barriers). The risk of take analysis for trap-and-haul operations considers only the effects of the operation itself, and not the effects of the barrier structures that necessitate the operation.

Species-specific risk of take ratings for fish passage structures are presented by impact mechanism and by ecosystem (marine, lacustrine, riverine) in Tables 9-13 through 9-17. Fish passage structures are almost always placed in riverine environments, so there is no risk of take in marine and lacustrine environments.

9.4.2.1 Construction and Maintenance

Culverts, fishways, fish ladders, and weirs are associated with a high risk of take due to the potential for direct injury or mortality from several possible impact mechanisms.

Roughened channel creation may require extensive in-channel work involving one or more impact mechanisms with the potential for direct injury or mortality. This equates to a high risk of take.

Trap-and-haul operations are associated with a high risk of take of target species because they involve the capture, handling, transport, and release of fish. These actions have the potential for direct or delayed mortality from stress or injury, even when the most thoughtful precautions are taken. The acts of capture and handling constitute take as defined for the purpose of Section 7 ESA consultations. The risk of take for nontarget species is generally considered to be low. Some potential for take exists via the introduction of toxic substances from accidental spills during operations. However, this potential is limited if proper BMPs are in place. Western ridged mussels, which are dependent on migratory salmonids that are typically the target species for trap-and-haul would be expected to incur a high risk of take due to the long-term indirect effects of fish passage operations on host-fish species.

9.4.2.1.1 Equipment Operation and Materials Placement

The risk of take resulting from construction and/or operation of fish passage structures varies by species depending on the species occurrence, the nature of stressor exposure, the sensitivity of the species, the life-history stage exposed to the stressor, and the habitats where fish passage facilities are typically developed. The magnitude, timing, duration, and frequency of each impact mechanism vary widely with project scale and location. The assessment of risk of take associated with each impact mechanism is broad and applies a “worst-case scenario” standard.

The construction of fish passage structures involves the use of heavy machinery and the placement of structural materials in and around the stream channel. Use of machinery (e.g., excavators) generates noise and visual and physical disturbance. At a minimum, underwater noise and visual and physical disturbance are likely to displace HCP fish species from occupied habitats, and otherwise modify behavior in ways that could affect survival, growth, and fitness. At worst, construction activities that produce intense underwater noise (e.g., installation of steel piles to support a fish ladder chute using an impact hammer) could lead to direct injury or mortality. For invertebrates, the risk of take could range from moderate (e.g., from displacement) to high (e.g., from crushing or other forms of mechanical injury).

9.4.2.1.2 Dewatering and Handling

Temporary dewatering and flow bypass with fish removal and relocation from work areas are common and necessary practices during construction and maintenance of fish passage structures. Even when dewatering is not required for construction and maintenance, exclusion areas are often created around the work sites to contain sediments and other pollutants as well as to reduce the magnitude of stressor exposure. This construction and maintenance activity poses a relatively high risk of take.

The act of dewatering the stream and redirecting flow may pose a barrier to fish migration. Delays in migration can lead to adverse effects on spawning fitness, can increase exposure to predation and poaching, and can deny juvenile fish access to rearing habitats during critical periods. These effects constitute a moderate risk of take of HCP species with migratory life-history stages.

9.4.2.1.3 Dredging and Fill

Dredging and fill activities associated with construction would ideally be conducted within a dewatered exclusion area to limit risk of take on HCP species. If this activity occurs in the open channel, it presents the potential for burial and entrainment. Each HCP species that occurs in freshwater environments where fish passage is likely to be implemented has at least one life-history stage with a high likelihood of suffering mortality or injury when exposed to either burial or entrainment. Therefore, dredging and fill activities must be associated with a high risk of take.

9.4.2.1 Hydraulic and Geomorphic Modifications

Hydraulic and geomorphic modifications associated with fish passage structures are expected to range considerably depending on specific circumstances. In general, however, fish passage structures are expected to have less extensive effects than activities such as the installation of large flow control structures. The construction and physical presence of fish passage structures can lead to alteration of physical habitat features. Because these structures are typically intended for long-term use, these habitat alterations are essentially permanent and continuous. If the effects are extensive, they can alter the productivity of the affected habitat for spawning, foraging, rearing, refuge, and other uses by HCP species. In a worst-case scenario, these effects in turn are likely to lead to reduced spawning success, as well as reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat. In cases where hydraulic and geomorphic modifications are extensive, a broad array of research has demonstrated that detrimental effects on survival, growth, and fitness are likely to occur for many of the HCP species that occur in riverine environments. In comparison to a water body with no structures present, this equates to a high risk of take.

Culverts are associated with a low risk of take for the majority of cases because the physical extent of hydraulic and geomorphic effects is expected to be limited. However, culverts that create upstream impoundments may cause more extensive hydraulic and geomorphic effects that are intermediate term in duration. These special cases are associated with a high risk of take due to the potential for direct mortality or injury in species reliant on the affected habitats. Culverts retrofitted with baffles or other internal structures to promote fish passage have reduced hydraulic capacity. This may in turn promote a backwater effect that leads to sediment deposition at the upstream end of the structure, creating flow conditions that limit fish passage. In some cases, existing culverts have arrested migrating headcuts, and removal or replacement will allow the nickpoint to continue migrating upstream causing channel downcutting. Existing undersized culverts can also cause upstream sediment aggradation that is subject to incision and downcutting when the culvert is removed. Channel downcutting from the migrating headcut can simplify channel geometry and influence the recruitment, transport, and retention of sediments and LWD. This type of channel simplification can affect habitat suitability for HCP species.

Fishways are associated with a low risk because the physical extent of hydraulic and geomorphic effects is expected to be limited.

Roughened channels are associated with a high risk of take. This conclusion is based on the specific design challenges, which create the potential for unexpected and potentially adverse hydraulic and geomorphic conditions to develop over time.

Most weirs are associated with a low risk of take because the physical extent of hydraulic and geomorphic effects is expected to be limited. Weirs constructed to manage (prevent) fish passage could have broad-reaching hydraulic and geomorphic effects, influencing habitat complexity both upstream and downstream of the structure. Weirs intended to block upstream passage of certain species may form impoundments that alter the transport of wood, sediment, and organic material. Temporary weirs would be expected to have negligible influence on groundwater/surface water interactions; therefore, the risk of take associated with this type of structure would likely be considered insignificant. In contrast, a structure such as a larger barrier weir may alter these interactions more extensively, leading to effects similar to those of a small dam.

9.4.2.1 Ecosystem Fragmentation

Compared to a water body with no structures present, fish passage projects have the potential to impose a number of barrier conditions that could potentially lead to take of HCP species. Specifically, fish passage structures or operations may fail to provide passage for all species as intended, may place unintended selection pressures on affected populations that limit or alter phenotypic diversity, or may become less effective at passing fish over time if improperly designed for the conditions present or if maintenance is neglected. Fish passage structures may limit the upstream movement of certain invertebrate species, or indirectly affect upstream dispersal through direct effects on the migration and productivity of host-fish populations. Limitations on fish passage may in turn result in long-term reductions in the abundance of migratory fish reaching areas upstream of the barrier. This may result in decreased food web productivity by reducing the delivery of nutrients derived from allochthonous sources. Given these potential ecosystem fragmentation effects, fish passage structures are considered to be associated with a high risk of take.

Culverts are associated with a high risk of take because even a well-designed structure may pose some risk of long-term ecosystem fragmentation in comparison to the natural system baseline. This may occur through effects on fish passage, or hydraulic and geomorphic effects. Culvert removal and replacement projects have the potential to alter lateral and longitudinal habitat connectivity in ways that can be detrimental to HCP species. Culverts have the potential to become significant barriers to the transport of LWD and sediment. If an improperly designed culvert results in the creation of an upstream impoundment, downstream transport of organic material may also be interrupted, altering nutrient cycling.

Fishways are generally expected to have limited effects on habitat complexity as a whole, which would be more than balanced by increased access to productive habitats. A fishway around a dam will have limited effects in comparison to the effects of the dam itself. The additional incremental effect of the structure will be slight, and the risk of take would be considered low. Fish ladders are associated with a high risk of take because they pose at least some risk of long-term ecosystem fragmentation in comparison to the natural system baseline.

Roughened channels are associated with a high risk of take because they pose at least some risk of long-term ecosystem fragmentation in comparison to a natural stream baseline.

The risk of weirs causing ecosystem fragmentation varies depending on the type of weir. Temporary weirs installed for fisheries management purposes are expected to produce only minor and temporary effects associated with a low risk of take. In contrast, permanent weirs intended to promote passage or to restrict passage of undesirable species are associated with a high risk of take because of the broad implications of unintended effects on movement of HCP species and on ecological processes.

Depending on specific configuration, trap-and-haul can impose a number of unintended effects related to ecosystem fragmentation. Imposing an artificial management regime during a critical phase in the life history of migratory fish species has the potential to create selection pressures that partially disconnect the adaptive capacity of the affected population from the natural environment. Alteration of migratory corridors by modifying release location may lead to decreased survival, fitness, and/or spawning productivity, potentially affecting long-term population viability. These effects would extend indirectly to freshwater mussels that are dependent on affected host-fish species. Any effects that reduce or modify the upstream transport of allochthonous nutrients may lead to altered food web productivity, an effect with broad consequences for all HCP species occurring in affected habitats. Given the range and breadth of these potential effects, as well as the typical longevity of trap-and-haul operations (which are usually associated with long-lived structures such as dams), ecosystem fragmentation must be associated with a high risk of take.

9.4.2.2 Riparian Vegetation Modifications

Riparian vegetation modification associated with fish passage is generally expected to be limited. In most cases, fish passage structures will be placed in areas that are already modified by human activities, and the incremental degradation associated with their construction will be insignificant. Most riparian vegetation modifications associated with fish passage structures is likely to be associated with construction impacts and therefore subject to restoration. This implies that any modest temperature effects would be intermediate term in nature. Therefore, the degree to which shade, solar exposure, and air temperature regime are affected is likely to be at best insignificant or at worst extremely small. The risk of take is expected to be low. In specific circumstances where more extensive and permanent vegetation modification occurs, a higher risk of take rating may be warranted. Examples of possible exceptions include roughened channel creation and the placement of fishways around natural passage barriers.

Culverts are associated with a low risk of take for the majority of cases because the physical extent of riparian vegetation modification is likely to be limited. However, in certain circumstances (i.e., where removal or replacement dewaters upstream impoundments), riparian vegetation effects may be more pronounced, resulting in a moderate risk of take due to their intermediate-term duration. The risk of take associated with altered allochthonous inputs is expected to be low.

Fishways are associated with a low risk of take because the physical extent of riparian vegetation modification is likely to be limited. However, in certain circumstances riparian vegetation effects may be more pronounced, resulting in a high risk of take.

Roughened channels are associated with a moderate risk of take because the physical extent of riparian vegetation modification associated with construction is likely to be relatively extensive in comparison to other fish passage structures. However, these effects are likely to be intermediate term in nature, as roughened channels lend themselves to riparian restoration.

Weirs are associated with a low risk of take because the physical extent of riparian vegetation modification is likely to be limited. However, in certain circumstances (i.e., permanent weirs installed to prevent upstream passage), riparian vegetation effects may be more pronounced, resulting in a high risk of take.

9.4.2.3 Aquatic Vegetation Modifications

The effects of fish passage structures on aquatic vegetation are generally expected to be limited because in-water footprints of most fish passage structures are usually relatively small. However, in specific circumstances, indirect effects due to changes in nutrient cycling may occur. Fish passage projects that result in a decrease in upstream transport of allochthonous nutrients may in turn limit habitat productivity and, by extension, aquatic vegetation growth. Alternatively, the increased delivery of allochthonous nutrients derived from marine or other productive downstream sources is likely to have the opposite effect. Given the potential for ill-conceived fish passage projects to increase ecosystem fragmentation, some effects on aquatic vegetation may occur.

Culverts and fishways are associated with a low risk of take for the majority of cases because the physical extent of aquatic vegetation modification is likely to be limited. However, in certain circumstances (i.e., where removal/replacement of a culvert dewaters upstream impoundments), aquatic vegetation effects may be more pronounced, resulting in a low to moderate risk of take (depending on species-specific reliance on aquatic vegetation) due to their intermediate-term duration.

Where aquatic vegetation is an important component of the riverine landscape, the physical extent of aquatic vegetation modification associated with roughened channel creation is likely to be relatively extensive in comparison to other fish passage structures. Because these effects are expected to be short term to intermediate term in nature, this impact mechanism imposes a moderate risk of take.

9.4.2.1 Water Quality Modifications

Fish passage structures have the potential to alter aquatic temperature regimes through alterations of riparian vegetation, reducing shading, altering ambient air temperatures, and altering groundwater/surface water interactions. The extent of effects on water temperature from removing riparian vegetation would be expected to be quite limited. The risk of take from altered temperature regime is expected to range from low to moderate depending on the specific type of structure and site-specific circumstances.

Construction of fish passage structures is likely to result in bank and channel disturbance through the use of heavy equipment, materials placement, dredging and fill, and rewatering of exclusion areas. This disturbance is in turn likely to produce a short-term increase in suspended sediment loading to riverine environments downstream of the structure. In certain cases, such as culvert removal or replacement that dewaters upstream impoundments, subsequent geomorphic effects may lead to ongoing bank and channel bed erosion, leading to a chronic elevation in suspended sediment load as the channel adjusts to the new hydraulic and hydrologic regime. The effects of elevated suspended sediments vary depending on the magnitude of the stressor and the sensitivity of the species or life-history stage exposed to the stressor. Given the potential for short-term injury or mortality resulting from elevated suspended sediment levels associated with construction, a high risk of take must be assumed for HCP species that occur in riverine habitat types where fish passage projects are likely to be implemented.

Generally, the direct effects of fish passage structures on dissolved oxygen conditions are not expected to be significant, and the risk of take associated with these effects is insignificant. Indirect effects on dissolved oxygen may occur as a result of improved ecosystem connectivity and hydraulic and geomorphic modifications. Increased upstream delivery of allochthonous nutrients, particularly large quantities of marine-derived nutrients in the form of salmon carcasses, has the potential to significantly increase ecosystem productivity. This in turn could increase biochemical oxygen demand resulting in decreased dissolved oxygen levels in certain cases. Restoration of fish passage to relatively unimpaired stream systems would generally not be expected to produce these conditions. However, if passage is restored to systems that are in a eutrophic state due to nutrient pollution from other sources, more extensive effects could occur. Fish passage work that results in dewatering of upstream impoundments (e.g., removing or replacing culverts) can result in the release of a pulse of sequestered nutrients when fine sediments in the impoundment bed are scoured. This is most likely to occur when large wetland areas are created by artificial barriers. A large pulse of nutrients could cause temporary eutrophication that, depending on the nature of the downstream environment, could cause a relatively rapid decrease in dissolved oxygen levels with the potential to adversely affect HCP species. Due to the short- to intermediate-term nature of these effects in freshwater environments, these effects are equated with a moderate to high risk of take for species occurring in the affected environment. Nonmotile species and life-history stages are most likely to experience high risk of take because they lack the capacity for avoidance.

The construction of fish passage structures can in some cases lead to the temporary alteration of pH levels. Many types of fish passage structures are constructed using concrete, a material that produces caustic leachate while curing. Concrete leachate released to surface waters from runoff or curing surfaces “in the wet” can increase pH levels well beyond levels capable of causing injury or mortality of all HCP species. This effect is typically short term in nature and moderates as the concrete cures, and is easily minimized using appropriate BMPs. However, due to the significant level of potential adverse effects, this stressor is equated with a high risk of take.

Construction of fish passage structures could introduce toxic substances into the aquatic environment through accidental spills from heavy equipment. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects on exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of nonmotile larvae to fuel spills). More commonly, intermittent low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation, leading to decreased survival, growth, and fitness. This presents a moderate risk of take to species potentially exposed to this stressor.

Culverts, fishways, and weirs are associated with a high risk of take due to the potential for short-term water quality impacts that can cause direct mortality or injury. Other mechanisms of impact associated with culverts, fishways, and weirs result in a moderate risk of take.

Certain types of weir structures may create impoundments that expand surface area, increasing solar radiation inputs and raising water temperatures.

Roughened channels require extensive in-channel work and have a large wetted area footprint, suggesting the potential for relatively extensive short-term water quality impacts in comparison to other fish passage types. Roughened channels can be associated with a high risk of take in some cases, as many water quality impacts have the potential to cause direct mortality or injury in sensitive species experiencing acute exposure. In many cases, however, a moderate risk of take is more appropriate because stressor exposure is more likely to result in nonlethal responses, and these stressors are typically short term in duration.

Table 9-13. Species- and habitat-specific risk of take for mechanisms of impact associated with culvert removal/replacement/retrofit.

	Construction & Maintenance Activities			Water Quality Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation		
Species	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Coho salmon	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Chum salmon	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Pink salmon	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Sockeye salmon	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Steelhead	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Coastal cutthroat trout	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Redband trout	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Westslope cutthroat trout	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Bull trout	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Dolly Varden	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Pygmy whitefish	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Olympic mudminnow	N	N	N	N	N	N	M	N	N	M	N	N	H	N	N	H	N	N
Lake chub	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Leopard dace	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Margined sculpin	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Mountain sucker	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Umatilla dace	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Pacific lamprey	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
River lamprey	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Western brook lamprey	H	N	N	H	N	N	M	N	N	L	N	N	H	N	N	H	N	N
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Longfin smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Eulachon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Species	Construction & Maintenance Activities			Water Quality Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	M	N	N	M	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	M	N	N	M	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	N	H	N	N	M	N	N	M	N	N	H	N	N	H	N	N
Western ridged mussel	H	N	N	H	N	N	M	N	N	M	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-14. Species- and habitat-specific risk of take for mechanisms of impact associated with fish ladders/fishways.

Species	Construction & Maintenance Activities			Water Quality Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Coho salmon	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Chum salmon	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Pink salmon	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Sockeye salmon	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Steelhead	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Coastal cutthroat trout	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Redband trout	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Westslope cutthroat trout	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Bull trout	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Dolly Varden	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Pygmy whitefish	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Olympic mudminnow	N	N	N	N	N	N	N	N	N	L	N	N	L	N	N	H	N	N
Lake chub	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Leopard dace	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Margined sculpin	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Mountain sucker	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Umatilla dace	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Pacific lamprey	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
River lamprey	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Western brook lamprey	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Longfin smelt	H	N	N	H	N	N	M	N	N	I	N	N	I	N	N	H	N	N
Eulachon	H	N	N	H	N	N	M	N	N	I	N	N	I	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

	Construction & Maintenance Activities			**Water Quality Modifications**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Hydraulic and Geomorphic Modifications**			**Ecosystem Fragmentation**		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
California floater (mussel)	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N
Western ridged mussel	H	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-15. Species- and habitat-specific risk of take for mechanisms of impact associated with roughened channels.

	Construction & Maintenance Activities			Water Quality Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation		
Species	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Coho salmon	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Chum salmon	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N	N
Pink salmon	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N	N
Sockeye salmon	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Steelhead	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Coastal cutthroat trout	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Redband trout	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Westslope cutthroat trout	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Bull trout	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Dolly Varden	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Pygmy whitefish	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Olympic mudminnow	N	N	N	N	N	N	N	N	N	M	N	N	H	N	N	H	N	N
Lake chub	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Leopard dace	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Margined sculpin	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Mountain sucker	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Umatilla dace	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Pacific lamprey	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
River lamprey	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Western brook lamprey	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Longfin smelt	H	N	N	H	N	N	M	N	N	M	N	N	H	N	N	H	N	N
Eulachon	H	N	N	H	N	N	M	N	N	M	N	N	H	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Species	Construction & Maintenance Activities			Water Quality Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N
Western ridged mussel	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-16. Species- and habitat-specific risk of take for mechanisms of impact associated with weirs.

	Construction & Maintenance Activities			Water Quality Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation		
Species	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Coho salmon	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Chum salmon	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Pink salmon	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Sockeye salmon	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Steelhead	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Coastal cutthroat trout	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Redband trout	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Westslope cutthroat trout	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Bull trout	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Dolly Varden	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Pygmy whitefish	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Olympic mudminnow	N	N	N	N	N	N	N	N	N	L	N	N	H	N	N	H	N	N
Lake chub	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Leopard dace	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Margined sculpin	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Mountain sucker	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Umatilla dace	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Pacific lamprey	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
River lamprey	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Western brook lamprey	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Longfin smelt	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N	H	N	N
Eulachon	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Species	Construction & Maintenance Activities			Water Quality Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb’s littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N
Western ridged mussel	H	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-17. Species- and habitat-specific risk of take for mechanisms of impact associated with trap-and-haul fish passage techniques.

	Operational Activities			Ecosystem Fragmentation		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	N	N	H	N	N
Coho salmon	H	N	N	H	N	N
Chum salmon	H	N	N	H	N	N
Pink salmon	H	N	N	H	N	N
Sockeye salmon	H	N	N	H	N	N
Steelhead	H	N	N	H	N	N
Coastal cutthroat trout	H	N	N	H	N	N
Redband trout	H	N	N	H	N	N
Westslope cutthroat trout	H	N	N	H	N	N
Bull trout	H	N	N	H	N	N
Dolly Varden	H	N	N	H	N	N
Pygmy whitefish	H	N	N	H	N	N
Olympic mudminnow	N	N	N	N	N	N
Lake chub	M	N	N	N	N	N
Leopard dace	M	N	N	N	N	N
Margined sculpin	M	N	N	N	N	N
Mountain sucker	H	N	N	H	N	N
Umatilla dace	M	N	N	N	N	N
Pacific lamprey	H	N	N	H	N	N
River lamprey	H	N	N	H	N	N
Western brook lamprey	M	N	N	N	N	N
Green sturgeon	N	N	N	N	N	N
White sturgeon	H	N	N	H	N	N
Longfin smelt	I	N	N	N	N	N
Eulachon	I	N	N	N	N	N
Pacific sand lance	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N
Walleye pollock	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N

Species	Operational Activities			Ecosystem Fragmentation		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Newcomb's littorine snail	N	N	N	N	N	N
Giant Columbia River limpet	N	N	N	H	N	N
Great Columbia River spire snail	N	N	N	H	N	N
California floater (mussel)	N	N	N	H	N	N
Western ridged mussel	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

9.4.3 Fish Screens

Fish screens are intended to protect against adverse effects on aquatic species caused by entrainment into or impingement on water intake or diversion systems. Current design guidance encourages the selection of screen designs that are appropriate for their ecological context. However, for the purpose of assessing risk of take, the baseline condition for this analysis is the stream system in the absence of artificial structures. Although fish screens provide an environmental benefit compared to unscreened water intakes or diversions, they present some risk of take when compared to a stream system with no structures.

In-channel screens in smaller streams and rivers are typically small, often temporary, end-of-pipe style structures. The impacts and resulting ecological stressors are relatively small in magnitude, and the risk of take associated with these types of structures will generally be quite low. In-channel screen designs employed in larger rivers, estuaries, large lakes and reservoirs, and the marine environment are commonly larger, permanent structures with greater potential for adverse effects, and therefore a greater risk of take. Bankline screens in marine and lacustrine systems, as well as larger rivers, may be located in embayments where they can impose ecosystem fragmentation effects. Bankline screens may employ pump or lift-driven bypass systems with additional potential for adverse effects.

Off-channel screens vary widely in scale, but are employed solely in riverine environments. Off-channel screens range from small, modular structures to large and complex systems. The impact mechanisms and resulting ecological stressors produced by small, modular screen systems installed by hand will be of lesser magnitude or intensity than those produced by large, permanent structures.

Species-specific risk of take ratings by impact mechanism are provided in Tables 9-18 and 9-19.

9.4.3.1 Construction and Maintenance

Construction and maintenance requirements for fish screens vary widely.

Temporary pump intake screens require little construction. They are simply placed in the source body with the intake pipe and anchored in place using some type of anchoring mechanism. They are commonly placed by hand, resulting in little disturbance of the stream bank or substrate. They are removed at the end of the use period. Screen placement and removal would be expected to result in minor visual and noise-related disturbance and minor pulses of suspended sediments, resulting in temporary behavior modification. Screen maintenance involves removal, cleaning, and replacement, resulting in similarly limited effects. This would equate to a low risk of take.

The worst-case scenario for in-channel screen construction would be associated with large, permanent end-of-pipe intake screens or bankline screen structures. These screen designs would likely require extensive in-water construction activity, potentially including dewatering and fish handling, pile driving (for cofferdam placement), and in-water use of heavy equipment. At best, underwater noise and visual and physical disturbance are likely to displace HCP fish species from occupied habitats, and to otherwise modify their behavior in ways that could affect survival, growth, and fitness. These short-term stressors would equate to a moderate risk of take. At worst, construction activities that produce intense underwater noise (e.g., installation of sheet piles for temporary construction cofferdams using an impact hammer) could lead to direct injury or mortality. This equates to a high risk of take. For invertebrate species, direct physical disturbance imposes some risk of take. Depending on the nature and severity of the disturbance, the risk of take could range from moderate (e.g., from displacement) to high (e.g., from crushing or other forms of mechanical injury). This is associated with a high risk of take due to the potential for direct injury or mortality from noise, visual disturbance, physical disturbance, dewatering, and fish handling.

Off-channel screens are constructed outside of the aquatic environment, either in an artificial diversion channel or entirely “in the dry”. When screen systems are constructed in the dry, the potential for construction-related disturbance and water quality impacts is considerably diminished. Even when placed in existing diversion channels, the structures can be placed behind splashboard dams or similar flow control structures avoiding the need for dewatering. In such cases, the need for in-water construction work in most circumstances would be limited to the connection of bypass channels to the aquatic ecosystem. Off-channel configuration also allows for relatively simple isolation of the structure as required for maintenance purposes. This in turn limits the potential for construction and maintenance related impacts on HCP species. Risk of take from off-channel screen construction and maintenance is expected to range from insignificant to low in the case of modular and smaller permanent screen systems. Screen construction and maintenance constructed “in the dry” are associated with an insignificant risk of take.

Temporary dewatering and flow bypass with fish removal and relocation from work areas are common and necessary practices during fish screen installation and possibly during maintenance. Even when dewatering is not required for construction and maintenance, exclusion areas are often created around the work sites to contain sediments and other pollutants as well as to reduce the magnitude of stressor exposure. This construction and maintenance activity may pose a relatively high risk of take.

Dewatering the stream and redirecting flow may pose a barrier to fish migration. Delays in migration can lead to adverse effects on spawning fitness, can increase exposure to predation and poaching, and can deny juvenile fish access to rearing habitats during critical periods. These effects constitute a moderate risk of take of HCP species with migratory life-history stages.

If dredging and fill associated with construction would ideally occur in the open channel, they present the potential for high risk of take from burial and entrainment. The sensitivity to these stressors generally varies by species and life-history stage. However, each HCP species that occurs in freshwater environments where fish screens are likely to be used has at least one life-history stage with a high likelihood of suffering mortality or injury when buried or entrained. Therefore, dredging and fill activities are considered to be associated with a high risk of take.

9.4.3.2 Operations

Correctly operating fish screens avoid and minimize adverse effects on aquatic species caused by water withdrawal or diversion. While the benefits of a correctly operating screen are fairly clear, the fact that these structures are continuously interacting with the aquatic environment indicates the potential for adverse effects on HCP species.

The risks and mechanisms of take associated with fish screen operation are variable depending on the type of screen design in question. Small, temporary screen structures employing passive debris clearing (e.g., T-screens on temporary pump intakes) or continuous active debris clearing (e.g., low velocity water jets or mechanical brushes) have minimal effect on the aquatic environment. Operational risk of take for these types of screens is generally considered to be low, providing that the structures are adequately maintained. Large in-channel fish screens pose additional risk of take from the operation of active debris-clearing and bypass systems.

Both in-channel and off-channel fish screens produce some noise, visual, and physical disturbance when in operation. Some in-channel screen designs, typically the larger systems associated with large industrial or agricultural water intake systems, incorporate hydraulic jet or air burst debris-clearing systems that are activated periodically. The related disturbance is intermittent in frequency and short-term in duration. Stressor response is expected to vary depending on the sensitivity of the species exposed, with the most extensive effects involving behavioral alteration and habitat avoidance. Under a worst-case scenario, the long-term operations of these types of systems would be associated with a high risk of take for HCP species that are sensitive to low-level disturbance (e.g., hearing specialists species such as suckers and dace), while species that are relatively insensitive (i.e., HCP invertebrates) would be expected to experience an insignificant risk of take.

Off-channel screens generally create disturbance that is more continuous in nature. For example, motorized rotating barrel screens or designs with mechanical debris-clearing systems produce continuous underwater noise, splashing, and visual disturbance during operation. The level of disturbance produced is generally expected to be limited to levels associated with behavioral avoidance, or potential habituation. Risk of take resulting from these stressors varies by species and life-history stage. Species such as HCP invertebrates that are insensitive to disturbance would be expected to face an insignificant risk of take. In contrast, fish species that become habituated to continuous disturbance may experience auditory masking effects that result in increased vulnerability to predation or reduced foraging success. These effects are associated with a high risk of take for hearing specialist species such as cyprinids (which include HCP dace, chub, and suckers). This risk is minimized by the fact that off-channel screens produce this stressor primarily in an artificially constructed environment (the diversion channel). This means that exposure would occur only for those species that are entrained into and occupy the diversion channel for extended periods. Hearing generalist species, such as salmonids, would be expected to be less sensitive to these effects. Because off-channel screens are configured to limit loitering by organisms drawn into the head ditch, exposure to these stressors will be limited and are therefore associated with a low risk of take.

Both in-channel and off-channel fish screens pose some unavoidable risk of entrainment or impingement of aquatic organisms when in operation. Risk of impingement is a function of screen design, operation, and maintenance, and the swimming ability of the HCP species in question. In general, this impact mechanism is associated with a high risk of take due to the potential for mortality and injury. It is necessary to qualify this risk against the level of take that would likely occur from unmitigated entrainment of organisms into unscreened intakes or diversions. A lessened probability of impingement or entrainment with fish screens is preferable to entrainment into unscreened diversions.

For certain species, specifically those with planktonic life-history stages, entrainment of free-floating eggs or larvae may simply be unavoidable if they are in the water column when an intake or diversion is in operation. HCP species that are likely to be in proximity to screens during their juvenile life-history stage, and/or are small in body size as adults have a high risk of impingement. The needs of weak-swimming organisms may not be fully accommodated by current screen design criteria. Given the potential for direct injury or mortality for small individuals, entrainment or impingement on fish screens must be equated with a high risk of take. Operational entrainment risk may also occur due to site-specific design limitations, or poor performance due to improper maintenance.

While a high risk of take rating is appropriate based on entrainment risk, the actual potential for population-level effects varies considerably by species, and should be considered when assessing impacts. For example, considerable numbers of Olympia oyster larvae may be entrained by a screened intake structure, but the resulting risk of take may be insignificant relative to natural larval mortality rates. In such a case, even though larval mortality may occur, the actual effect on population productivity would likely be insignificant. In contrast, the same intake may entrain larval lingcod at rates that greatly exceed natural mortality, suggesting the potential for significant population-level effects.

Fish species that come into proximity with fish screens only as large adults are less likely to experience impingement due to their stronger swimming ability. Research has demonstrated that many fish species, including HCP species such as bull trout, can withstand short periods of screen impingement with no apparent ill effects. Low-motility HCP invertebrate species (e.g., Olympia oyster, freshwater mussels) are unlikely to come into contact with fish screens as adults. Risk of take for these species/life-history stages from entrainment is rated as insignificant.

Bypass system operation can create circumstances take could occur. Organisms inhabiting or transiting bypass channels can become stranded when the intake and screen is shut off and the channel is dewatered. In the absence of flowing water, stranded organisms may be exposed to rapidly increasing or decreasing temperatures, creating the risk of injury or mortality from thermal stress, increased predation exposure, and lack of forage. This potential equates to a high risk of take, with the recognition that this risk can be limited through screen design and operation. Rapid dewatering of bypass channels that are recognized to provide habitat functions for aquatic species of interest is not permitted. Bypass flows are often maintained in these channels to support beneficial habitat functions.

9.4.3.3 Hydraulic and Geomorphic Modifications

The hydraulic and geomorphic effects of fish screens are expected to be relatively modest in comparison to the intake or diversion structure they are associated with, but some level of effect may result from fish screens themselves. The magnitude of hydraulic and geomorphic impacts, resulting stressors, and risk of take vary depending on the scale and placement of the screen in question.

In many cases, the design parameters of fish screens provide a means for controlling diversion flows, limiting diversion rates that exceed water rights. This provides a mechanism for preservation of base flows that may negate the influence of bypass system operation on base flow conditions.

Small end-of-pipe screens on temporary pump intakes are expected to have little if any measurable hydraulic and geomorphic effect in most settings. They have little potential to alter flow conditions, channel geometry, or substrate composition (Schille 2008). The resulting risk of take associated with this type of structure is expected to be insignificant.

Large permanent bankline or end-of-pipe screens may require placement of significant structures, with shoreline armoring and other forms of erosion protection. This presents the potential for a broader range of hydraulic and geomorphic effects and a greater risk of take. However, these requirements are considered to be components of the intake or diversion system with which the screen is associated. The related effects and resulting risk of take are therefore considered also to be the result of the intake or diversion, rather than of the screen.

Off-channel screens, which are typically intended for long-term use, can cause permanent and continuous changes in flow regime, channel geometry, and substrate composition and stability in the bypassed reach, especially if flow-mediated vegetation encroachment changes the trajectory of channel evolution. If these effects are extensive, they can alter the productivity of the affected habitat for spawning, foraging, rearing, refuge, and other uses by HCP species. In cases where hydraulic and geomorphic modifications are extensive, a broad array of research has demonstrated that detrimental effects on survival, growth, and fitness are likely to occur for many of the HCP species that occur in riverine environments. Effects of this nature equate to a high risk of take, with the recognition that the circumstances where this is likely to occur are rare.

9.4.3.1 Ecosystem Fragmentation

In-channel fish screens have the potential to produce ecosystem fragmentation effects in specific circumstances. Intakes employing bankline screens in marine environments, lakes, and large rivers are commonly located in embayments. Because there is little or no available hydraulic head to operate bypass systems in embayments, aquatic organisms drawn into the intake must be pumped or lifted into bypass systems. HCP species with planktonic eggs and larvae may be drawn into these embayments by the intake and either retained or bypassed by the screen. Bypass systems have their own inherent potential to cause injury and mortality. From a worst-case scenario perspective, this type of screen could also impose ecosystem fragmentation effects if organisms drawn into the embayment area cannot be effectively bypassed, or if they are repeatedly bypassed and drawn back into the intake system. These effects are associated with a high risk of take.

Off-channel fish screens have the potential to impose barrier conditions that could potentially lead to take of HCP species. Fish screens may unintentionally delay or otherwise hinder passage of migrants due to design limitations. A fish screen may delay or affect passage of only certain species, and may place unintended selection pressures on affected populations that limit or alter phenotypic diversity. Screens may entrain more organisms or create passage barriers over time, if improperly designed for the conditions or if maintenance is neglected.

Although the overall effects of fish screens on fish passage are relatively minor in comparison to the effects imposed by the flow control structures and channel modifications associated with water diversions and withdrawals, the long-term nature of fish screen effects is consistent with a high risk of take.

Fish screens could have an effect on HCP invertebrate species if the screens affect the migration and productivity of host-fish populations.

In rivers, limitations imposed by screens on upstream fish passage may result in long-term decreases in food web productivity through reduced delivery of nutrients derived from allochthonous sources. The overall extent of this effect due to fish screens is expected to be small relative to the related flow control structures. The risk of take associated with this impact mechanism is expected to be insignificant. Upstream transport of nutrients is not relevant in marine and lacustrine environments.

Fish screens designs that collect debris in troughs for disposal, or that divert water into bypass channels that require maintenance clearing, may modify the downstream transport of woody debris. The actual amount of wood and organic debris trapped on fish screens is not likely to represent a significant proportion of the natural flux. The incremental effect of the fish screen is likely to be minor in comparison to the flow control structure or channel modification associated with the water diversion. Because the extent of this effect on the environment is not quantified, the associated risk of take is unknown.

9.4.3.2 Riparian Vegetation Modifications

Installation of bankline in-channel screens and all off-channel screen types may result in some level of riparian vegetation modification to install the bypass system. The scale of the bypass system may range from a simple pipe with erosion protection at the outfall, to excavation of an artificial channel. The extent of effects on riparian vegetation, and the resulting risk of take, is expected to vary depending on the scale of the screen and bypass system in question. Piped diversion systems associated with modular off-channel screens on small diversions would not be expected to have extensive effects on riparian vegetation. The resulting risk of take associated with these designs would be expected to range from insignificant to low. Excavation of artificial bypass channels to support large off-channel or bankline screens would be expected to have potentially significant effects on riparian vegetation, resulting in a high risk of take.

9.4.3.1 Water Quality Modifications

Fish screen operation has a limited capacity to affect water temperatures through riparian vegetation modification. The extent of riparian vegetation modification associated with the fish screen structures is expected to be limited. Riparian modification associated with bypass channel creation should be considered a component of intake or diversion development and/or artificial channel creation instead. Piped bypass systems are more arguably attributable to the fish screen system, but the magnitude of riparian vegetation modification associated with these structures is expected to be limited. On this basis, the temperature effects resulting from this impact mechanism are expected to be similarly limited and the related risk of take insignificant relative to the effects of flow diversion.

Bypass channel operation that results in dewatering and stranding can result in increased water temperatures and decreased dissolved oxygen. In a worst case scenario, when operation of bypass systems (i.e., rapid dewatering) exposes organisms in bypass channels to stranding, the combination of higher temperatures and lower dissolved oxygen levels can increase the likelihood of injury or lethality. This is equated with a high risk of take. Rapid dewatering is not permitted in channels that are known to be used as rearing habitat by aquatic organisms. Fish screens are otherwise expected to have limited influence on dissolved oxygen, and the risk of take is insignificant.

Construction of fish screens may result in short-term impacts due to elevated suspended sediments through the use of heavy equipment, materials placement, dredging and fill, and rewatering of exclusion areas. Given the potential for short-term injury or mortality resulting from elevated suspended sediment levels associated with construction, a high risk of take must be assumed.

The construction of fish screen structures may in some cases lead to the temporary alteration of pH levels. Many fish screens are constructed using concrete, a material that produces caustic leachate while curing. This stressor is equated with a high risk of take.

Fish screens could introduce toxic substances into the aquatic environment through accidental spills from heavy equipment during construction and maintenance, and through failure of mechanical equipment (i.e., debris-clearing systems) during operations. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects on exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of nonmotile larvae to fuel spills). More commonly, intermittent low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation, leading to decreased survival, growth, and fitness. This presents a moderate risk of take.

Table 9-18. Species- and habitat-specific risk of take for mechanisms of impact associated with in-channel fish screens.

Species	Construction & Maintenance Activities			Operations			Water Quality Modifications			Riparian Vegetation Modifications			Hydraulic & Geomorphic Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	H	H	H	H	H	H	H	H	L	L	L	I	I	I
Coho salmon	H	H	H	H	L	H	H	L	H	L	U	L	I	I	I
Chum salmon	H	H	I	H	H	I	H	H	I	L	L	I	I	I	I
Pink salmon	H	H	I	H	H	I	H	H	I	L	L	I	I	I	I
Sockeye salmon	H	H	H	H	L	H	H	L	H	L	U	L	I	I	I
Steelhead	H	H	H	H	L	H	H	L	H	L	U	L	I	I	I
Coastal cutthroat trout	H	H	H	H	L	H	H	L	H	L	U	L	I	I	I
Redband trout	H	N	H	H	N	H	H	N	H	I	N	L	I	N	I
Westslope cutthroat trout	H	N	H	H	N	H	H	N	H	I	N	L	I	N	I
Bull trout	H	H	H	H	L	H	H	L	H	L	L	L	I	I	I
Dolly Varden	H	H	H	H	L	H	H	L	H	L	L	L	I	I	I
Pygmy whitefish	H	N	H	H	N	H	H	N	H	I	N	I	I	N	I
Olympic mudminnow	H	N	H	H	N	H	H	N	H	I	N	I	I	N	I
Lake chub	H	N	H	H	N	H	H	N	H	I	N	I	I	N	I
Leopard dace	H	N	H	H	N	H	H	N	H	L	N	L	I	N	I
Margined sculpin	H	N	H	H	N	H	H	N	H	I	N	I	I	N	I
Mountain sucker	H	N	H	H	N	H	H	N	H	L	N	L	I	N	I
Umatilla dace	H	N	H	H	N	H	H	N	H	L	N	L	I	N	I
Pacific lamprey	H	H	H	H	H	H	H	L	H	L	I	L	I	I	I
River lamprey	H	H	H	H	H	H	H	L	H	L	U	L	I	I	I
Western brook lamprey	H	N	H	H	N	H	H	N	H	I	N	I	I	N	I
Green sturgeon	N	H	N	N	L	N	N	L	N	N	I	N	N	I	N
White sturgeon	H	H	H	H	L	H	H	L	H	L	I	L	I	I	I
Longfin smelt	H	H	H	H	H	H	H	H	H	I	I	L	I	I	I
Eulachon	H	H	N	H	H	N	H	H	N	I	I	N	I	I	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Lingcod	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Pacific cod	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
China rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N

Species	**Construction & Maintenance Activities**			**Operations**			**Water Quality Modifications**			**Riparian Vegetation Modifications**			**Hydraulic & Geomorphic Modifications**		
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Copper rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Olympia oyster	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Northern abalone	N	H	N	N	H	N	N	H	N	N	L	N	N	I	N
Newcomb's littorine snail	N	H	N	N	N	N	N	L	N	N	L	N	N	I	N
Giant Columbia River limpet	H	N	H	I	N	I	M	N	N	L	N	N	I	N	N
Great Columbia River spire snail	H	N	H	I	N	I	M	N	N	L	N	N	I	N	N
California floater (mussel)	H	N	H	H	N	H	M	N	I	L	N	N	I	N	N
Western ridged mussel	H	N	H	H	N	H	M	N	M	L	N	N	I	N	I

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-19. Species- and habitat-specific risk of take for mechanisms of impact associated with off-channel fish screens.

Species	Construction & Maintenance Activities			Operations			Water Quality Modifications			Riparian Vegetation Modification			Hydraulic & Geomorphic Modifications			Ecosystem Fragmentation		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	L	L	L	H	H	H	H	H	H	L	L	L	H	I	I	I	I	I
Coho salmon	L	L	L	H	L	H	H	L	H	L	U	L	H	I	I	I	I	I
Chum salmon	L	L	I	H	H	I	H	H	I	L	L	I	H	I	I	I	I	I
Pink salmon	L	L	I	H	H	I	H	H	I	L	L	I	H	I	I	I	I	I
Sockeye salmon	L	L	L	H	L	H	H	L	H	L	U	L	H	I	I	I	I	I
Steelhead	L	L	L	H	L	H	H	L	H	L	U	L	H	I	I	I	I	I
Coastal cutthroat trout	L	L	L	H	L	H	H	L	H	L	U	L	H	I	I	I	I	I
Redband trout	L	N	L	H	N	H	H	N	H	I	N	L	H	N	I	I	N	I
Westslope cutthroat trout	L	N	L	H	N	H	H	N	H	I	N	L	H	N	I	I	N	I
Bull trout	L	L	L	H	L	H	H	L	H	L	L	L	H	I	I	I	I	I
Dolly Varden	L	L	L	H	L	H	H	L	H	L	L	L	H	I	I	I	I	I
Pygmy whitefish	L	N	L	H	N	H	H	N	H	I	N	I	H	N	I	I	N	I
Olympic mudminnow	L	N	L	H	N	H	H	N	H	I	N	I	H	N	I	I	N	N
Lake chub	L	N	L	H	N	H	H	N	H	I	N	I	H	N	I	I	N	I
Leopard dace	L	N	L	H	N	H	H	N	H	L	N	L	H	N	I	I	N	I
Margined sculpin	L	N	L	H	N	H	H	N	H	I	N	I	H	N	I	I	N	I
Mountain sucker	L	N	L	H	N	H	H	N	H	L	N	L	H	N	I	I	N	I
Umatilla dace	L	N	L	H	N	H	H	N	H	L	N	L	H	N	I	I	N	I
Pacific lamprey	L	L	L	H	I	H	H	L	H	L	I	L	H	I	I	I	I	I
River lamprey	L	L	L	H	H	H	H	L	H	L	U	L	H	I	I	I	I	I
Western brook lamprey	L	N	L	H	N	H	H	N	H	I	N	I	H	N	I	I	N	I
Green sturgeon	N	L	N	N	I	N	N	L	N	N	I	N	N	I	N	N	N	N
White sturgeon	L	L	L	H	I	H	H	L	H	L	I	L	I	I	I	N	N	N
Longfin smelt	L	L	L	H	H	H	H	H	H	I	I	L	H	I	I	I	I	I
Eulachon	L	L	N	H	H	N	H	H	N	I	I	N	H	I	N	I	I	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Species	**Construction & Maintenance Activities**			**Operations**			**Water Quality Modifications**			**Riparian Vegetation Modification**			**Hydraulic & Geomorphic Modifications**			**Ecosystem Fragmentation**		
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb’s littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	L	N	L	I	N	I	H	N	N	L	N	N	H	N	I	?	N	I
Great Columbia River spire snail	L	N	L	I	N	I	H	N	N	L	N	N	H	N	I	?	N	I
California floater (mussel)	L	N	L	H	N	H	H	N	H	L	N	N	H	N	I	I	N	I
Western ridged mussel	L	N	L	H	N	H	H	N	H	L	N	N	H	N	I	I	N	I

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take;
? = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

9.4.4 Flow Control Structures

Flow control projects are typically designed with the intent of withdrawing water and/or modifying the hydraulic and hydrologic characteristics to promote human uses of the aquatic environment and the surrounding landscape. These projects lead to a fundamental alteration of ecological processes. They impose a range of direct and indirect effects on the environment, resulting in an array of ecological stressors, during both the construction phase and over the course of operation. The magnitude of these stressors varies depending on the scale of the project in question and the degree to which it modifies ecological conditions and processes.

Flow control structures include the following:

Dams are a significant form of hydromodification that impose broad and pervasive effects on riverine environments. Dam projects range in scale from the relatively modest on small stream systems to immense projects on large river systems, such as the Mossy Rock Dam on the Cowlitz River or the Grand Coulee Dam on the Columbia River. Dams are channel spanning structures that create upstream impoundments. These structures impose stressors on aquatic organisms through a range of impact mechanisms and fundamentally alter the characteristics of riverine ecosystems, and in some cases lacustrine ecosystems (e.g., where dams are created at lake outlets). The hydrologic and water quality effects of dams can extend to marine ecosystems as well.

Weirs include both temporary and permanent structures constructed to control the movement of water, sediments, or organisms in riverine and floodplain environments. Flow control weirs create impoundments or divert streamflow and act similar to a dam. The risk of take analysis for weirs focuses on the worst-case scenario: permanent, typically concrete structures that span the entire channel and create a barrier to fish passage.

Dikes and levees are extensive hydromodifications designed to prevent flooding in low-lying landscapes, and to protect and promote human uses. By preventing regular tidal or floodwater inundation, these structures facilitate the conversion of wetland, floodplain, or estuarine habitats for terrestrial uses such as agriculture and development.

Outfalls discharge water or effluent.

Water diversion and water intake structures include a broad range of designs with purposes ranging from municipal and irrigation water diversions, to power plant and industrial water intakes, to hatchery water supply systems. Structure designs associated with these types of facilities can range from bankline intake systems oriented parallel to the shoreline in any environment type, to dam or weir type diversion structures in river systems oriented perpendicular to streamflow.

Tide gates and flood gates are structures designed to facilitate the flow of water out of floodplain, wetland, or estuarine habitats, as well as manage or prevent the reflooding of these lands by tidal fluctuations or flood flows. Tide gates and flood gates range in scale from simple, corrugated metal culverts with metal or fiberglass flap gates buried in dikes, to larger, more complex wood or concrete structures with mechanically controlled gates. They are typically incorporated into dikes and levees to promote the conversion of these habitat types into terrestrial or modified aquatic environment types for human uses. In some cases, tide gates are used to manage habitat conditions within an impounded area to support recreational fish and wildlife populations, but in many cases these structures are intended to facilitate the conversion of estuarine or floodplain wetlands to terrestrial habitats for agricultural or industrial uses.

Risk of take is rated for each species by impact mechanism and environment type (i.e., riverine, marine, and lacustrine) in Tables 9-20 through 9-25. The summary risk of take presented in the narrative and the matrices represents the greatest overall risk of take for the category.

9.4.4.1 Dams and Weirs

9.4.4.1.1 Construction and Maintenance

Construction, operation, and maintenance of dams involve a diverse array of activities that can impose a variety of environmental stressors on HCP species occurring in riverine and lacustrine environments. Construction and maintenance may include such activities as heavy equipment operation, materials placement, pile driving, and flow bypass and dewatering around work areas. The majority of construction and maintenance activities are temporary in nature, lasting from a few days to several weeks, depending on the size of the project and the nature of the activity. In the case of large dams, however, construction and maintenance activities may last for months or even years, with continuous activity occurring throughout. The risk of take associated with construction activity varies by impact mechanism and is dependent on the project-specific magnitude of that impact mechanism. Some mechanisms may produce a high risk of individual take due to their intensity, while others may result in a low risk of take due to their limited magnitude and duration.

Construction-related effects during dam removal must also be considered. Many of the activities associated with dam removal, such as equipment use, materials placement, and visual, noise, and physical disturbance, are similar to those imposed during construction. However, the dewatering of impoundments creates the potential for unique effects in the form of stranding in dewatered areas that must be considered when evaluating risk of take.

Applying a worst-case scenario perspective, the largest weirs may be comparable in scale to smaller dams, implying that the construction-related impacts would also be similar.

The construction, operation, and maintenance of dams will result in some alteration of the underwater noise environment. The nature of this habitat modification will vary depending on the phase of the project. During construction and maintenance, intense sources of underwater noise such as pile driving, materials placement, or in-water equipment operation may create short-term pulses of high intensity sound pressure. Auditory masking effects caused by continuous noise sources that alter the ambient noise level (e.g., from extended operation of construction and maintenance vessels, in-water equipment use, or spillway and turbine operation) may affect the ability of fish to detect predators and prey, affecting their survival, growth, and productivity.

The construction of dams requires the operation of heavy equipment and the placement of materials in and around aquatic habitats and adjacent terrestrial habitats, including riparian zones and floodplains. In-water use of equipment and the placement of materials impose stressors in the form of physical and visual disturbance. The magnitude of these stressors will vary widely, depending on the scale of the project in question and the specific construction measures used. Applying a worst-case-scenario perspective, the magnitude of these stressors can be significant.

Construction-related bank, channel, and shoreline disturbance could result in decreased stream bank and shoreline stability, as well as increased erosion and turbidity. Motile fish species would be expected to experience only temporary behavioral alteration and low risk of take. Less motile fish life-history stages or sessile invertebrates could experience a high risk of take due to mortality caused by smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success.

Temporary dewatering and flow bypass with fish removal and relocation from work areas is a common and necessary practice during dam construction and maintenance. Even when dewatering is not required for construction and maintenance, exclusion areas are often created around the work sites to contain sediments and other pollutants and to reduce the magnitude of stressor exposure. This construction and maintenance activity poses a relatively high risk of take, depending on habitat and species and life-history stage-specific factors.

Dewatering and redirecting flow may pose a barrier to fish migration. Delays in migration can lead to adverse effects on spawning fitness, can increase exposure to predation and poaching, and can deny juvenile fish access to rearing habitats during critical periods. These effects constitute a moderate risk of take of HCP species with migratory life-history stages.

Dewatering is also associated with dam removal. Once a dam is breached, the impoundment behind the structure will drain. Aquatic species in the impoundment trapped in rapidly dewatering habitats face risk of mortality from stranding, particularly non-motile species and life-history stages. Motile species able to avoid stranding will be displaced from existing habitats and forced to relocate within disturbed habitats that may present limited foraging opportunities, which could similarly limit survival, growth, and fitness. It is generally presumed that care will be taken during dam removal to dewater slowly, reducing stranding risk. Consistent with a worst-case scenario approach, however, this activity must be associated with a high risk of take, particularly for non-motile species and life-history stages.

9.4.4.1.2 Hydraulic and Geomorphic Modifications

Dams impose significant changes in the hydraulic and geomorphic characteristics of riverine and lacustrine environments, and can modify the characteristics and suitability of the affected habitats for HCP species adapted to riverine environments. The impact mechanisms associated with dams are complex, even before considering the complexity of the responses of HCP species to stressor exposure. Therefore, we view the risk of take in a holistic fashion. With the exception of altered flow regime, the mechanisms of impact, stressors, and related risk of take from hydraulic and geomorphic modifications associated with weir development are similar to those for dams.

Dams fundamentally alter flow regime, channel geometry, and substrate composition and stability by converting a flowing water environment upstream of the structure to a slack water impoundment, altering the hydrologic regime and interrupting the transport of wood, sediment, and organic material. Downstream of the structure, alteration of flow regime and reduced transport of LWD and sediment from upstream sources are likely to lead to changes in channel morphology, with detrimental effects on habitat structure. Operational water level fluctuations may also affect habitat productivity, creating risk of stranding for non-motile fish life-history stages and invertebrates, which is likely to lead to mortality. All of these effects that dams impose on ecosystem structure and function are interrelated, as is the risk of take. These effects alter habitat suitability for fish and invertebrate species adapted to the original environmental condition and affect the survival, growth, and fitness of many of the HCP species that occur in riverine environments. In some cases, these effects have been shown to limit productivity at the population level, depending on the nature of the facility and the species affected. The long-term alteration of flow regime, channel geometry, and substrate composition and stability equates to a high risk of take.

The effects that dams impose on the connectivity between surface water and groundwater are complex and change over time. Most dams are designed to be relatively impermeable at their base to prevent the loss of impounded water to groundwater. However, the large hydraulic head created by dams can, in some cases, increase groundwater exchange, resulting in increased hyporheic flow to downstream reaches. Over time, however, the accumulation of fine sediments in the impoundment decreases bed permeability and retards groundwater exchange. Changes in flow regime, sediment transport, and substrate composition will all affect in-channel hyporheic exchange as well. The effects on survival, growth, fitness, and productivity caused by long-term alteration of hyporheic exchange equate to a high risk of take.

9.4.4.1.3 Ecosystem Fragmentation

Ecosystem fragmentation is a significant and multifaceted component of the effects that dams impose on the aquatic environment. Weirs have similar effects, but to a lesser degree. Because weirs are not intended to create impoundments, the fragmentation of longitudinal connectivity associated with these structures is restricted to effects on the passage of fish and other organisms, as well as the downstream transport of LWD and organic material. Similarly, there is a lesser effect on community composition. The effects and related risks of take from altered longitudinal connectivity, altered river-flood plain connectivity, altered LWD transport, and altered groundwater-surface water interactions are otherwise similar.

The predominant effect of dams is the fragmentation of longitudinal connectivity of the river continuum. Dams interrupt the downstream transport of water, wood, sediment, and organic material, and, depending on design and scale, may also prevent the upstream and downstream movements of migratory fish and invertebrates. The impoundment also creates a lentic habitat that is discontinuous within the riverine landscape, capable of altering temperature, nutrient loading, and food web productivity. These changes to longitudinal connectivity equate to a high risk of take.

Dams can cause a significant alteration in the connectivity of the river system to floodplain and terrestrial habitats. In the impoundment, the channel, floodplain, and portions of the surrounding valley are inundated. Depending on site-specific topography, the natural gradient between the river and floodplain is replaced by a steeper ecological gradient between the new aquatic and surrounding terrestrial habitat. This gradient may be quite abrupt if impoundment management causes extreme water level fluctuations, creating simplified habitat conditions at the impoundment margin that are not suitable for rearing, spawning, refuge, or other important life-history requirements. In downstream habitats, changes in flow regime and sediment starvation may lead to channel degradation, causing fragmentation of the main channel from off-channel and floodplain habitats. The connectivity between river and floodplain habitats is reduced over a broad range of flow conditions. In smaller rivers and streams, dams also affect water temperature, with further effects on river–floodplain connectivity, decreasing the influence of stream shading and altered ambient temperatures in downstream reaches. A number of HCP species are dependent on off-channel and floodplain habitats during one or more life-history stage. A reduction in the availability of suitable habitat will lead to increased competition for the remaining available habitat, decreased growth and fitness, increased exposure to predation, and potentially decreased availability of suitable spawning sites. While these effects primarily concern fish, invertebrate species such as mussels would also be affected due to reduced productivity of host fish populations. These changes to river-floodplain connectivity equate to a high risk of take.

Dams interrupt the transport of LWD along the longitudinal gradient in riverine environments. Modification of the flow regime in downstream reaches and channel downcutting caused by sediment starvation may also lead to lateral river-floodplain fragmentation, which could limit the recruitment in downstream reaches, further starving the channel of LWD. The hydraulic and geomorphic effects of reduced LWD density in the channel network can lead to further alterations in habitat complexity. Reduced LWD presents a potential risk of take for a broad range of species dependent on riverine aquatic ecosystems through a variety of species-specific stressors. Depending on the particular life history of the affected species, alterations in habitat complexity may limit the availability of suitable spawning, resting, and rearing habitat, and may alter foraging opportunities and predation exposure. In general, fish species that are dependent on habitats potentially affected by changes to LWD are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. These changes equate to a high risk of take.

The conversion of riverine habitats from lotic to lentic environments upstream of dams, and alterations of flow and thermal regime both upstream and downstream of the structure can lead to changes in community composition within the riverine ecosystem. By creating lentic habitats and altering downstream habitat complexity and water quality conditions, dams may create suitable conditions for a range of species that would not otherwise be able to survive in the undisturbed system. For example, impoundments create warm water habitats that promote the growth of emergent vegetation, creating habitat conditions suitable for warm water fish (e.g., bass, perch, and sunfish) that would not normally survive in a flowing river with naturally cool temperatures. These species may compete with juvenile salmonids for food resources, or may prey on them directly, affecting their survival, growth, and productivity. By causing reductions in downstream habitat complexity and interrupting the transport of coarse particulate organic matter, dams may indirectly cause a shift in macroinvertebrate community structure, affecting food web diversity. This may in turn limit foraging opportunities for HCP species exposed to this stressor, affecting survival, growth, and fitness. The effects of altered community structure on HCP species are complex and variable depending on the nature of the changes and how these species interact with the altered environment. From an ecological perspective, alterations in community structure are generally viewed as negative overall, even though effects on individual species can be negative, positive, or neutral. Applying a worst-case scenario perspective, the effects must be viewed as negative because of the potential for adverse effects on survival, growth, and fitness of any native species within the affected environment. Because these effects are effectively permanent or at least long term on the scale of the life of the structure, they are equated with a high risk of take.

9.4.4.1.4 Riparian Vegetation Modifications

Dams alter the extent to which riparian vegetation influences temperature in riverine environments. By greatly expanding the surface area, impoundments limit the shading and ambient temperature buffering influence of the riparian zone upstream of the dam. In downstream reaches, alterations in riparian vegetation characteristics and channel morphology caused by the effects of dams can alter the influence of vegetation on stream temperatures, allochthonous inputs to the riverine ecosystem, and the influence of riparian vegetation on habitat complexity.

The mechanisms of impact, stressors, and related risk of take from riparian vegetation modifications associated with weir development are similar to those described for dams, but occur to a lesser degree.

Water temperatures in riverine systems suitable for dams are less influenced by localized shading and ambient air temperature than by the combined effects of basin conditions in upstream areas. The risk of take associated with temperature changes due to removal of riparian vegetation is variable, depending on the nature of the project and the type of environment in which it is implemented. Using the worst-case scenario perspective, the effects of altered stream temperatures must be equated with a high risk of take due to the long-term nature of the habitat alteration and the potential effects on survival, growth, and fitness of HCP species.

Dam projects may cause intermediate-term alteration of riparian conditions in downstream reaches when vegetation is removed. Once riparian vegetation is established adjacent to the modified channel bank, instability is likely to decrease, unless downcutting caused by sediment starvation leads to long-term instability. The risk of take from increased turbidity associated with riparian vegetation removal varies; motile fish experience only temporary behavioral alteration and a low risk of take. Less motile fish life-history stages or sessile invertebrates could experience a moderate to high risk of take from decreased survival due to substrate sedimentation and smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success.

Removing riparian vegetation for a dam, and the associated loss of allochthonous production near the mouth of a large river will produce related stressors of potentially lower magnitude than a dam on a small, higher elevation stream. On smaller streams, a localized reduction in food web productivity might result, leading to decreased foraging opportunities, decreased overall habitat suitability, and decreased growth and fitness. This equates to a moderate risk of take for a range of HCP species that are dependent on riverine rearing conditions.

Altered habitat complexity due to riparian vegetation removal equates to a moderate risk of take, which applies broadly across all exposed species.

9.4.4.1.5 Aquatic Vegetation Modifications

Dams and weirs can modify the aquatic vegetation community through the effects of the structure on hydraulic and geomorphic conditions in riverine ecosystems, through the alteration or elimination of vegetation in the construction footprint, through changes from a lotic to a lentic environment suitable for the establishment of emergent vegetation, and by providing colonization opportunities for invasive species. However, aquatic vegetation is a relatively minor component of the ecological structure of riverine and lacustrine systems in Washington State. Aside from native emergent vegetation confined to a relatively narrow range of depths, a large portion of aquatic vegetation species in rivers and lakes are invasive species. Moreover, once the channel has adjusted to the presence of the structure, the aquatic vegetation community would be expected to recover to some extent. The risk of take resulting from altered autochthonous production and altered habitat complexity is expected to be low to moderate depending on the species-specific sensitivity to these impacts.

9.4.4.1.6 Water Quality Modifications

Dams have significant and pervasive effects on water quality conditions. Dam construction is a large undertaking, involving a number of water quality effects such as increased sedimentation, alteration of pH, and the potential introduction of toxic substances to surface waters. Once in place, the ecological fragmentation imposed by the structure, changes in biogeochemical processes that occur within the impoundment, and the effects of hydraulic and geomorphic modification on downstream reaches can in turn result in a number of changes in water temperature and chemistry. Sources of water quality modification resulting from weir development are associated primarily with project construction and include increases in suspended sediments and turbidity, altered pH levels, and the introduction of toxic substances, and are similar to those described for dams.

Dams result in the long-term alteration of the aquatic temperature regime in riverine predominantly by converting riverine habitats to lacustrine environments exposed to increased insolation. Impoundments tend to stratify during summer months, significantly increasing water in the impoundment temperatures. Depending on how dams are constructed and operated, they can also significantly alter downstream temperatures. Dams that spill water from surface layers of the impoundment during summer months when the impoundment is stratified may cause significant increases in downstream temperatures. Dams that release flows drawn from deeper, cold water layers of the reservoir may create downstream temperatures that are significantly cooler than the natural temperature range. Temperature effects will persist for the life of the structure and have the potential to affect the survival, growth, and fitness of HCP species, equating to a high risk of take.

Dams can lead to alterations in the concentration of dissolved oxygen and other gases in surface waters through decreased dissolved oxygen concentrations caused by eutrophication in the impoundment and potentially surface waters downstream of the dam, and through supersaturation of dissolved gases (predominantly DO, but also nitrogen). If dissolved oxygen concentrations drop below optimal levels, fish will begin to exhibit stress and avoidance behavior. DO concentrations below tolerance thresholds, or depressed DO in combination with elevated water temperatures, may be sufficient to cause mortality, particularly for less-motile life-history stages. Gas supersaturation can occur from the extreme turbulence created by spillways and other dam structures. Sufficient exposure to supersaturated conditions can cause mortality under laboratory conditions, and gas bubble disease, which has been shown to cause injury to juvenile salmonids, is known to occur *in situ*. Less specific information is available regarding the effects of depressed DO levels on invertebrate HCP species. Mussels are known to be intolerant of low DO levels, while the sensitivity of other species is less certain. Given the predilection of all freshwater mollusk HCP species for flowing water environments, however, it is reasonable to conclude that these species are adapted to environments with relatively high natural DO levels. Therefore, depression of DO levels caused by eutrophication in impoundments would be considered a likely adverse effect. Both increased and decreased DO levels can lead to adverse effects on survival, growth, and fitness of fish populations exposed to these conditions. The collective effects of dams on dissolved oxygen conditions will last for the lifetime of the structure. Therefore, they must be equated with a high risk of take.

Dams can alter turbidity during construction and while the channel adjusts to the new hydraulic and hydrologic regime imposed by the hydromodification. Dams can lead to a reduction in natural suspended sediment loading downstream of the structure, because impoundments encourage settling of fine sediments transported from upstream. Eutrophication in impoundments may elevate turbidity levels in the impoundment, which would be transported to downstream reaches. On balance, the long-term risk of take from changes in suspended sediments and turbidity caused by dams will be variable depending on site-specific conditions. However, given the potential for short-term injury or mortality resulting from elevated suspended sediment levels associated with construction, a high risk of take must be assumed for HCP species that occur in suitable riverine and lacustrine environments.

Dams may provide a mechanism for the accumulation of contaminated sediments within the impoundment, due to their tendency to capture fine sediments and the tendency of certain contaminants to sorb to small organic and inorganic particles. In general, these sediments are sequestered and typically become capped as new layers of sediment recruitment are deposited in the impoundment. However, these sediments may be released into the environment during maintenance dredging, or during eventual dam removal. This could result in the release of large volumes of contaminated material over a relatively short period of time, in combination with high levels of suspended sediments overall. Beyond the effects of suspended sediment loading, exposure to toxic substances in contaminated sediments can lead to effects on the survival, growth, and fitness of exposed species. These effects would be expected to be short term and acute in duration and are therefore equated with a moderate risk of take.

Dams that are constructed of concrete can lead to the alteration of pH levels through concrete leachate released to surface waters from runoff or curing surfaces. This effect is typically short-term in nature and moderates as the concrete cures. If adequate procedures are not in place to protect against this water quality impact, this effect is equated with a high risk of take with potential exposure over a short-term period.

Within impoundments, conditions can be favorable for eutrophication, which can significantly alter pH and DO levels. CO_2 combines with water in solution to form carbonic acid, which measurably decreases pH. Photosynthesis by aquatic vegetation and phytoplankton leads to decreased CO_2 and increased DO during daylight hours, while respiration causes the opposite effect after dark. In eutrophic systems, phytoplankton blooms and subsequent die-offs of aquatic vegetation and plankton can cause a rapid spike in respiration, which rapidly depletes DO levels and increases CO_2. These changes can lead to pH fluctuations within the impoundment that may exceed effects thresholds for certain HCP species. In combination with depleted DO, elevated temperatures, and other water quality effects imposed by impoundments, this stressor could cause behavioral avoidance, increased stress and physiological injury, or even mortality to HCP species adapted to cold water and high DO environments with relatively stable pH conditions. In certain impoundment environments, altered pH conditions could occur chronically on a seasonal or annual basis over the life of the structure, and could be limiting to the survival, growth, fitness, and/or spawning productivity of HCP species living within or migrating through the affected environment. Therefore, these effects would be equated with a high risk of take.

Dam projects present multiple pathways for the introduction of a range of toxic substances to the aquatic environment, primarily through construction activities and, in some cases, the use of treated wood materials in the structure. Dams may also indirectly encourage pollutant and nutrient loading by supporting the development of additional infrastructure and expanded recreational vessel use in the impoundment. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects in exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of immobile lamprey ammocoetes buried in bottom substrates, fish exposed to accidental vessel spills in enclosed embayments). More commonly, chronic, low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation, leading to decreased growth and fitness. This presents a moderate risk of take to species potentially exposed to this stressor.

9.4.4.2 Dikes and Levees

Extensive in size and pervasive in effect, dikes and levees impose a number of ecological stressors on the environment through essentially permanent alteration of habitat and water quality conditions. HCP species occurring in environments modified by these types of structures will typically experience a high risk of take from one or more impact mechanisms.

9.4.4.2.1 Construction and Maintenance

The construction of dikes and levees uses heavy machinery, places extensive fill, and the removes riparian vegetation throughout the length of the project. Maintaining these structures includes similar activities, at a lesser magnitude and scale, at an annual to decadal frequency.

The operation of heavy construction equipment to build or maintain dikes and levees imposes stressors in the form of physical and visual disturbance of bank and channel habitat, and, potentially, increased underwater noise from in-water equipment use and materials placement. The magnitude of these stressors varies widely, depending on the scale of the project in question and the specific construction measures used. Applying a worst-case-scenario perspective, the magnitude of these stressors can be significant.

Bank, channel, and/or shoreline disturbance during the construction and maintenance of dikes causes short-term water quality impacts, as well as long-term (essentially permanent) modification of hydraulic and geomorphic conditions and ecosystem connectivity. The short-term water quality effects of channel and bed disturbance may lead to behavioral and physiological stress on species or life-history stages exposed to the disturbance, or may limit the availability and suitability of habitats for sensitive life-history stages during critical periods. Non-motile species exposed to these stressors may face immediate effects on survival if occupied habitats are eliminated, or may experience injury or mortality from related water quality effects. These effects would be equated with a moderate to high risk of take, depending on species-specific sensitivity.

The effects of temporary dewatering and flow bypass during construction and maintenance of dikes and levees are equated with a high risk of take.

9.4.4.2.2 Hydraulic and Geomorphic Modifications

Dikes and levees may cause a significant modification of hydraulic and geomorphic processes. These effects are effectively permanent, given the longevity of these structures and the tendency for valuable property improvements and infrastructure to develop landward of them.

Dikes and levees alter flow conditions in riverine environments by preventing the flooding of adjacent terrestrial and riparian habitats, concentrating high flows in the stream channel, accelerating flow velocity and erosive forces. Reduced floodplain storage of water in hydromodified areas may induce flooding in reaches upstream and downstream of the structure in areas where flooding otherwise would not occur. The effects of altered flow conditions on HCP species are complex and variable, depending on the position of the hydromodification in the riverine environment and how the affected habitats are used by HCP species. In a worst-case scenario, these pervasive, long-term effects would be expected to reduce habitat suitability for species utilizing the affected environment, limiting individual survival, growth, and fitness and overall population productivity. This equates to a high risk of take.

Dikes and levees change channel geometry and substrate composition and stability. They are often built in conjunction with channel straightening and simplification to accelerate the flow of water through the landscape, to facilitate the conversion of this land to human uses. Substrate composition and stability can be altered through the loss of sources of sediment recruitment and altered sediment transport capacity. These habitat alterations are essentially permanent and continuous, and can lead to changes in the productivity of the habitat for spawning, forage, rearing, and refuge. In a worst-case scenario, these effects can lead to reduced spawning success, reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat. This equates to a high risk of take.

Dikes and levees can alter groundwater and surface water exchange in the project area and downstream. This has the potential to affect juvenile and/or adult survival, growth, and fitness, and in some cases the spawning productivity of a range of species. Because this effect will be pervasive and essentially permanent, this mechanism is generally equated with a moderate to high risk of take for species exposed to this stressor, depending on species-specific life-history characteristics.

Hydraulic and geomorphic effects waterward of a dike or levee in a river delta or estuary can alter bathymetry, current patterns, circulation patterns, salinity, and tidal exchange, potentially altering desirable habitat types.

9.4.4.2.3 Ecosystem Fragmentation

Dikes and levees can fragment ecological connectivity between aquatic and terrestrial environments. By aiding the conversion of low-lying floodplain and wetland habitats to terrestrial uses, these structures sharpen the gradient between the aquatic and terrestrial landscape.

In riverine environments, dikes and levees reduce the structural complexity of instream habitat by changing the channel geometry and influencing the recruitment, transport, and retention of sediments and LWD. Such simplification reduces habitat complexity, leads to reduced food web productivity, and reduces availability of habitats suitable for HCP species. Because these effects are extensive and effectively permanent, this impact mechanisms equates to a high risk of take for HCP species.

Dikes and levees purposefully disconnect of floodplain and off-channel habitats from the riverine ecosystem. This disconnects the stream channel from important sources and sinks of organic matter, nutrients, and pollutants. Such disconnection may limit food web productivity, affecting the survival, growth, and fitness of any species dependent on the riverine environment for rearing. In addition, this loss of connectivity may limit the availability of important habitat types for HCP species. The reduction in suitable refuge and foraging habitat area increases competition for remaining habitat, predation risk, and risk of displacement to habitats unfavorable for rearing. Collectively, these long-term ecological stressors pose a high risk of take for HCP species that occur in the affected riverine environment.

Dikes and levees could potentially be built in lacustrine or marine environments, for example in river deltas. Such projects prevent access to habitats and facilitate their conversion for terrestrial uses. The associated risk of take is strongly linked to species-specific dependence on floodplain, nearshore, or estuarine environments. In the case of organisms with a planktonic life-history stage, the effects of dikes and levees may limit the dispersal and retention of eggs and larvae to areas suitable for rearing. Habitat fragmentation caused by dikes and levees in the lacustrine or marine environment would be expected to affect the survival, growth, and fitness of affected species, as well as the overall population productivity. These effects are associated with a high risk of take because they are essentially permanent.

9.4.4.2.4 Riparian Vegetation Modifications

Riparian vegetation is often removed to create dikes and levees. Once the structures are established, vegetation is often managed to prevent the degradation of structural integrity caused by root penetration. Using the worst-case scenario perspective, in riverine systems, the effects of altered stream temperatures, altered allochthonous inputs, and altered habitat complexity associated with removal of riparian vegetation are equated with a high risk of take due to the long-term nature of the habitat alteration and the potential effects on survival, growth, and fitness of HCP species.

In marine environments, altered allochthonous inputs and altered habitat complexity are likely to affect the survival, growth, and fitness of those HCP species dependent on the nearshore environment. This equates to a high risk of take for species with demonstrable dependence on these habitats because these effects will be long term in duration.

9.4.4.2.5 Aquatic Vegetation Modifications

Newcomb’s littorine snail is found only on *Salicornia* spp. (glasswort) in saltmarsh environments. Dike or levee projects that convert saltmarsh environments for terrestrial uses would effectively eliminate the only habitat used by this obligate species. This equates to a high risk of take, based on the dependence of the species on nearshore aquatic vegetation and the effectively permanent nature of the habitat modification.

9.4.4.2.6 Water Quality Modifications

Sources of water quality modification associated with dikes and levees include increased suspended sediments and the potential introduction of toxic substances during project construction, as well as the effects of riparian and hydraulic and geomorphic modification on stream temperatures, similar to the effects and risks of take associated with dams.

9.4.4.3 Outfalls

Outfalls are commonly relatively small in scale and have relatively limited physical effects on the aquatic environment in comparison to other types of flow control structures. However, outfalls are a significant source of potential take because they facilitate the delivery of nutrients and pollutants to surface waters.

9.4.4.3.1 Construction and Maintenance

The construction of outfalls typically involves disturbance of bank and shoreline habitat to place the outfall structure and related erosion protection at the outlet. In lacustrine and marine environments, outfall construction may extend through the littoral zone to place the outlet below the water surface, preventing beach erosion. Regardless of configuration, outfall construction involves the use of heavy equipment to place the structure.

Underwater noise effects would likely be insufficient to cause direct injury, meaning that stressor response would likely be limited to short-term disturbance and behavioral modification. Stressor exposure of this magnitude is equated with a low to moderate risk of take, depending on the size scale of the structure in question.

In a worst-case scenario, outfall construction may include in-water equipment use and material placement or significant disturbance of the bank/shoreline. These activities could result in potential injury or mortality of HCP species having sessile or non-motile life-history stages. These effects are equated with a high risk of take. Motile species or life-history stages would experience temporary disturbance and displacement, potentially affecting survival, growth, and productivity. These effects are equated with a moderate risk of take.

Outfall construction may require temporary dewatering and/or flow bypass during construction. Creation of exclusion areas, fish removal and relocation, and work area dewatering/flow bypass are all activities with the potential to cause injury or mortality to HCP species. These effects are equated with a high risk of take.

9.4.4.3.2 Hydraulic and Geomorphic Modifications

The effects of hydraulic and geomorphic modifications caused by outfalls in riverine environments are relatively limited because these structures are typically located on the stream bank and have a relatively small footprint. A broad array of riverine habitat types may be considered suitable for outfall projects. Therefore, effectively all riverine species and life-history stages could be exposed to stressors and experience a resulting risk of take due to hydraulic and geomorphic modification caused by outfalls.

Outfalls in rivers can alter hydraulic and geomorphic conditions through altered channel geometry, altered flow regime, and altered substrate composition, but because outfall size is typically relatively limited, the magnitude of the effects caused by individual outfall projects is not likely sufficient to affect HCP species survival, growth, and fitness at a large scale. Therefore, the resulting risk of take associated with these effects is likely to be moderate.

Outfalls in the marine environment are typically more extensive structurally than those in lacustrine and riverine environments. Marine outfalls typically extend from upland habitats through the littoral zone and discharge into subtidal habitats. These projects modify hydraulic and geomorphic conditions in the nearshore marine environment, resulting in the imposition of several impact mechanisms and related stressors. The risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on the nearshore environment.

Outfall structures that are exposed (whether by design or unintentionally) could potentially attenuate wave energy, alter localized circulation patterns, interrupt longshore sediment transport, alter sediment supply or alter substrate composition. This equates to a high risk of take for species that are dependent on nearshore habitats due to the long-term existence of outfall structures.

Outfalls change fresh water inputs to the nearshore marine environment, and may carry undesirable pollutants leading to degradation of water quality. The alteration in freshwater inputs imposed by outfalls is viewed to be an ecologically undesirable effect that is long term in duration, potentially leading to reduced survival, growth, and fitness. This equates to a high risk of take for species experiencing stressor exposure.

In lakes, the effects of outfalls on wave energy, current, and circulation patterns are equated with a high risk of take for species that are dependent on these habitats during some phase of their life history. Applying a worst-case scenario perspective, an exposed outfall could cause long-term alteration of substrate conditions in the vicinity of the structure. This equates to a high risk of take for species that are dependent on these habitats due to effects on the survival, growth, and productivity of exposed life-history stages given the long-term nature of stressor exposure.

9.4.4.3.3 Ecosystem Fragmentation

The degree to which outfalls cause ecosystem fragmentation in riverine environments is limited. Outfalls in riverine environments are typically located on the bank and discharge at the edge of the stream channel. If concentrated discharge of stormwater or effluents create a dilution zone with water quality conditions that are sufficiently unfavorable to cause avoidance behavior, and if this mixing zone extends across a majority of the channel, it could impose a barrier to fish passage. This would represent fragmentation of longitudinal connectivity. Depending on the duration and frequency of the effect, this could deny access to productive habitats, potentially limiting the survival, growth, fitness, and productivity of affected populations. Under a worst-case scenario, this effect would equate to a high risk of take.

The risk of take from ecosystem fragmentation caused by outfalls in marine or lacustrine environments ranges from insignificant (e.g., for buried outfall pipes with discharge points located far offshore) to high (e.g., for exposed outfalls or outfall pipes that create a perpendicular barrier and causing hydraulic and geomorphic modifications of the nearshore environment).

9.4.4.3.4 Riparian Vegetation Modifications

In general, outfalls would be expected to have a relatively limited effect on riparian vegetation because their onshore footprint is relatively small. However, should the structure impose extensive hydraulic and geomorphic effects that alter bank stability, effects on riparian vegetation could be more extensive. In general, outfall structures are not expected to be associated with bank erosion to a degree that would cause widespread losses of riparian vegetation; therefore, effects would be expected to be intermediate-term in nature as riparian vegetation adjusts to changing conditions. The risk of take associated with stressors resulting from this impact mechanism is expected to be moderate.

9.4.4.3.5 Aquatic Vegetation Modifications

The effects of outfalls on aquatic vegetation from project construction are expected to be relatively minor given the limited footprint of these structures. Over time, however, these structures may modify the aquatic vegetation through their effects on hydraulic and geomorphic processes, as well as on water quality conditions.

In lakes and rivers, modification of the submerged aquatic vegetation community would typically be limited to the footprint of the structure, and possibly the effects of effluent on vegetation growth. Assuming that effluent concentrations are managed properly, the effects of outfalls on autochthonous productivity and habitat structure would be expected to be minor, and are equated with an insignificant risk of take.

In marine systems, buried outfall pipes discharging offshore may have a limited effect on the aquatic vegetation community following recovery from construction impacts. In contrast, exposed outfall pipes may affect vegetation community structure through hydraulic and geomorphic effects imposed on the nearshore environment. Outfall discharges may cause alteration of the aquatic vegetation community through the introduction of toxics or through eutrophication induced by nutrient loading. Alterations of the submerged aquatic vegetation community through reduction in aerial extent or conversion to other habitat types (e.g., conversion of eelgrass habitat to algae and kelp) can reduce the productivity of these habitats for dependent life-history stages. Applying a worst-case scenario perspective, outfalls could result in the long-term alteration of the nearshore aquatic vegetation community through their effects on habitat structure and water quality. This equates to a high risk of take for species dependent on these habitats due to long-term effects on spawning productivity, as well as larval survival, growth, and fitness.

9.4.4.3.6 Water Quality Modifications

Outfalls deliver pollutants into surface waters. Stormwater and effluent discharges may contain a variety of toxic substances or other pollutants, including PAHs, metals, agricultural chemicals, and nutrients. Alteration of water quality conditions is associated with long-term detrimental effects on the survival, growth, and fitness of aquatic species exposed to the component stressors. Eutrophication caused by nutrient inputs may ultimately lead to decreased DO levels and altered pH conditions, also having potential effects on the survival, growth, and fitness of aquatic receptors. Exposure to these stressors is equated with a high risk of take based on the potential for long-term, chronic exposure.

9.4.4.4 Intakes and Diversions

For the purpose of assessing the risk of take, a worst-case scenario perspective is applied. In riverine environments, the worst-case scenario design is a cross-channel type diversion structure similar to a dam or a weir. In marine and lacustrine environments, the worst-case scenario design is a bankline structure similar in magnitude to large tide gates or similar structures.

9.4.4.5 Tide Gates and Flood Gates

For the purpose of assessing risk of take, a worst-case scenario perspective is taken.

9.4.4.5.1 Construction and Maintenance

Tide gate construction usually takes place in environments that are already highly modified by dikes and levees. Degraded channel and bank conditions may not present suitable habitat for HCP species and life-history stages that would otherwise occupy the affected environment. Therefore, while the risk of take ratings are representative of the effects of stressor exposure, the potential for stressor exposure is likely to be more limited than in more pristine environments. Tide gates present a smaller magnitude of risk due to the smaller size of the construction footprint.

Due to the potential for injury and mortality, the risk of take associated with underwater noise is rated as high for species with life-history stages that occur in environments suitable for tide gates. However, the potential for stressor exposure is more limited because tide gate construction would typically be expected to be more limited and to take less time than dam construction.

In a worst case scenario, tide gate construction and maintenance may involve in-water work, including equipment use and material placement. These activities could result in potential injury or mortality of HCP species occurring in the vicinity that have sessile or non-motile life-history stages. These effects are equated with a high risk of take. Motile species or those with motile life-history stages would experience temporary disturbance and displacement, potentially affecting survival, growth, and productivity. These effects are equated with a moderate risk of take.

In a worst case scenario, tide gate construction may require significant disturbance of the bank/shoreline and substrate, degrading habitat conditions in the affected habitat and resulting in the release of suspended sediments. These activities could result in potential injury or mortality of HCP species having sessile or non-motile life-history stages. These effects are equated with a high risk of take. Motile species or those with motile life-history stages would experience temporary disturbance and displacement, potentially affecting survival, growth, and productivity. These effects are equated with a moderate risk of take.

Creation of exclusion areas, fish removal and relocation, and work area dewatering/flow bypass are all activities with the potential to cause injury or mortality to HCP species, and are equated with a high risk of take.

9.4.4.5.2 Hydraulic and Geomorphic Modifications

Alteration of tidal and/or floodwater exchange is the primary way tide gates impose their effects on aquatic systems. Tide gates concentrate and thereby accelerate the rate at which floodwaters drain from inundated habitats. This change in flow regime may cause the displacement of small or relatively non-motile species adapted to slow-water environments. Accelerated flows draining the wetland and stream system caused by the installation of a dike and flood gate system could lead to the displacement of Olympic mud minnows, potentially to a riverine environment with unsuitable habitat conditions. In such special cases, mortality would be likely, and would be equated with a high risk of take, but this stressor would be considered a relatively minor component of the overall impacts of the conversion of floodplain wetland habitat into a managed terrestrial habitat.

Tide gates and flood gates alter channel geometry and alter substrate composition. The structure can force scouring, deposition, and simplification of channel structure by changing inundation frequency and flow velocities in channel networks landward of the structure. By encouraging sedimentation of the channel network over time, distributary channels and ponds gradually fill and become terrestrial habitat (or are converted to managed ditches that are dredged). This alters the habitat suitability and productivity for HCP species adapted to this type of environment, and these effects will be long term and progressive in nature. This is equated with a high risk of take, with this stressor considered to be one component of the broader risk of take resulting from the conversion of aquatic habitat into a managed terrestrial environment.

Waterward of the structure, high-velocity flows out of the tide gate can cause localized scour, mobilizing fine sediments and changing the bed composition. These effects would be limited in scale to a relatively small area, and would occur in an already-modified channel. The additive risk of take is considered to be moderate for HCP species with life-history stages that occur in the affected environment.

9.4.4.5.3 Ecosystem Fragmentation

The purpose of tide gates is to facilitate the flow of water out of floodplain, wetland, or estuarine habitats, while preventing these lands from being reflooded by tidal exchange or flood waters. The alteration and conversion of habitats to conditions that are poorly suited for HCP species are the ultimate results within the zone of effect of the structure, and are long-term in duration. The essentially permanent modification of high-value habitats to unsuitable conditions equates to a high risk of take for those species dependent on these habitats during some portion of their life history.

9.4.4.5.4 Riparian Vegetation Modifications

Tide gate construction may require the permanent alteration of riparian vegetation within the footprint of the structure, as well as additional temporary modification of the surrounding habitat during construction. Tide gates are typically developed in environments where riparian conditions have already been extensively modified for dike and levee development; therefore, the actual risk of take associated with this impact mechanism may be insignificant in comparison to that imposed by the dike or levee.

9.4.4.5.5 Aquatic Vegetation Modifications

Aquatic vegetation modifications associated with tide gates are equated with a high risk of take for those HCP species dependent on floodplain, wetland, and estuarine marsh habitats during some portion of their life history, particularly species such as Newcomb's littorine snail that are obligate occupants of emergent saltmarsh vegetation. The effects of tide gates and flood gates on aquatic vegetation are compounded by water quality related effects exacerbated by the exposure of anaerobic sediments in floodplain and estuarine environments.

9.4.4.5.6 Water Quality Modifications

Tide gates alter the ambient water temperature in aquatic environments landward of the structure by limiting the exchange and flushing effects of tidal inundation and floodwaters. These effects occur predominantly in tidally influenced areas where the flushing effects of tidal exchange normally occur on a daily basis. In such circumstances, aquatic habitats landward of the structure would be expected to experience elevated water temperatures, particularly during summer months. Organisms exposed to chronic elevations in water temperatures beyond tolerance thresholds would be expected to experience reduced survival, growth, and fitness. Due to the essentially permanent nature of these effects, this is equated with a high risk of take.

Tide gates alter the salinity of surface waters upstream of the structure by preventing the tidal inflow of marine water, resulting in conversion to freshwater habitat over time. This conversion from estuarine or marine to freshwater habitat represents a fundamental alteration in habitat suitability for species adapted to the original habitat conditions. Because these effects will persist for the life of the structure, they are associated with a high risk of take for HCP species that utilize environments suitable for tide gate development.

Alteration of flow regime and inundation frequency in saltmarsh and wetland environments has been demonstrated to cause depleted oxygen conditions as organic matter in anoxic soils becomes exposed and available for aerobic decomposition. These combined effects have been demonstrated in saltmarsh ecosystems regulated by tide gates to deplete DO concentrations below levels sufficient to cause direct mortality of fish. Even in the absence of mortality, stress from DO depletion in combination with increased water temperatures and poor habitat suitability may lead to decreased survival, growth, and fitness of HCP species occurring within the modified habitat. Freshwater wetland environments would be expected to experience similar effects, where the operative physical, biological, and chemical processes are similar. Due to their long-term and progressive nature, these effects are equated with a high risk of take for species occurring in the affected environment.

Some tide gate and flood gate structures are built using concrete, a material capable of causing acute changes in surface water pH if appropriate best management practices are not employed during construction. Once a tide gate or flood gate is in place, the alteration in inundation frequency describe above can lead to the exposure of anaerobic sediments to open air. Oxidation of sulfides released from anaerobic sediments can in turn rapidly reduce the pH of surface waters. This effect is well documented in the literature in natural systems, and may be compounded in environments that are undergoing a conversion to terrestrial habitat imposed by a dike/tide gate system. Rapid reductions in pH are capable of causing physiological stress, injury, and mortality in many fish and invertebrate species. Therefore, this is equated with a high risk of take.

Tide gate and flood gate construction may introduce toxic substances from accidental spills. Once a tide gate or flood gate is in place, the processes enabled when anaerobic sediments are exposed to oxidation can release potentially toxic substances into the aquatic environment. Decreased surface water pH and altered redox conditions in exposed soils can cause rapid leaching of toxic metals, including aluminum, cadmium, copper, and silver into the water column. Decreased pH can, in some cases, produce rapidly precipitating iron flocs capable of smothering wildlife and vegetation. Water quality modifications initially occur landward of the structure but can extend beyond the dike into the nearshore environment as the altered surface water drains during low tide or low streamflow conditions. These kinds of effects are well documented in natural systems and may be compounded in environments that are undergoing a relatively rapid conversion to terrestrial habitat imposed by a dike/tide gate system. Exposure to dissolved metals and floc precipitates can impose physiological stress, injury, and mortality on HCP species exposed to these stressors. These stressors may also weaken or kill aquatic vegetation, altering habitat structure and suitability for organisms dependent on these habitat types. Due to the potential for direct mortality and the intermediate to long-term nature of these effects, this is equated with a high risk of take.

Table 9-20. Species- and habitat-specific risk of take for mechanisms of impacts associated with dams.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Coho salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Chum salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Pink salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Sockeye salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Steelhead	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Coastal cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Bull trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Dolly Varden	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pacific lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
River lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Longfin smelt	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Eulachon	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

	Construction & Maintenance Activities			**Hydraulic and Geomorphic Modifications**			**Ecosystem Fragmentation**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb’s littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-21. Species- and habitat-specific risk of take for mechanisms of impacts associated with weirs.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	N	H	H	N	M	H	N	H	I	I	I	M	N	M	H	N	H
Coho salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Chum salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Pink salmon	H	N	I	H	N	I	H	N	I	H	N	I	M	N	I	H	N	I
Sockeye salmon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Steelhead	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Coastal cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Bull trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Dolly Varden	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pacific lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
River lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
White sturgeon	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Longfin smelt	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Eulachon	H	N	N	H	N	N	H	N	N	M	N	N	I	N	N	H	N	N
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb’s littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-22. Species- and habitat-specific risk of take for mechanisms of impacts associated with dikes and levees.

	Construction & Maintenance Activities			**Hydraulic and Geomorphic Modifications**			**Ecosystem Fragmentation**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Coho salmon	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Chum salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Pink salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Sockeye salmon	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Steelhead	H	M	H	H	?	M	H	?	H	H	?	M	M	?	M	H	M	H
Coastal cutthroat trout	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Bull trout	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Dolly Varden	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Pacific lamprey	H	I	H	H	I	M	H	I	H	H	I	M	M	I	M	H	I	H
River lamprey	H	H	H	H	H	M	H	H	H	H	H	M	M	H	M	H	H	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	M	M	N	M	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	M	H	?	H	H	?	M	M	?	M	H	?	H
Longfin smelt	H	H	N	H	H	N	H	N	N	M	I	N	I	?	?	H	H	N
Eulachon	H	H	N	H	H	N	H	N	N	M	I	N	I	?	N	H	H	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N

Species	**Construction & Maintenance Activities**			**Hydraulic and Geomorphic Modifications**			**Ecosystem Fragmentation**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**		
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Pacific cod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	N	N	N	I	N	N	I	N	N	H	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-23. Species- and habitat-specific risk of take for mechanisms of impacts associated with outfalls.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Coho salmon	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Chum salmon	H	H	I	M	H	I	H	H	I	I	I	I	I	H	I	H	H	I
Pink salmon	H	H	I	M	H	I	H	H	I	I	I	I	I	H	I	H	H	I
Sockeye salmon	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Steelhead	H	M	H	H	?	H	H	?	H	H	?	I	I	?	I	H	?	H
Coastal cutthroat trout	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Redband trout	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Westslope cutthroat trout	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Bull trout	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Dolly Varden	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Pygmy whitefish	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Olympic mudminnow	H	N	H	M	N	H	H	N	H	I	N	I	N	N	N	H	N	H
Lake chub	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Leopard dace	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Margined sculpin	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Mountain sucker	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Umatilla dace	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Pacific lamprey	H	I	H	M	I	H	H	I	H	I	I	I	I	I	I	H	I	H
River lamprey	H	H	H	M	H	H	H	H	H	I	I	I	I	H	I	H	H	H
Western brook lamprey	H	N	H	M	N	H	H	N	H	I	N	I	I	N	I	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	M	H	?	H	H	?	I	M	?	M	H	?	H
Longfin smelt	H	H	H	M	H	H	H	H	H	I	I	I	I	?	I	H	H	H
Eulachon	H	H	N	M	H	N	H	H	N	I	I	N	I	?	N	H	H	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Pacific cod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	N	N	N	I	N	N	I	N	N	H	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	I	N	N	H	N
Newcomb's littorine snail	N	H	N	N	H	N	N	N	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	H	N	N
Great Columbia River spire snail	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	H	N	N
California floater (mussel)	H	N	N	M	N	N	H	N	M	H	N	N	M	N	N	H	N	N
Western ridged mussel	H	N	N	M	N	N	H	N	M	H	N	N	M	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-24. Species- and habitat-specific risk of take for mechanisms of impacts associated with diversion structures and water intakes.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Chinook salmon	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Coho salmon	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Chum salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Pink salmon	H	H	I	H	H	I	H	H	I	H	H	I	M	M	I	H	H	I
Sockeye salmon	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Steelhead	H	M	H	H	?	H	H	?	H	H	?	I	M	?	M	H	?	H
Coastal cutthroat trout	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Redband trout	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Bull trout	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Dolly Varden	H	H	H	H	H	H	H	H	H	H	I	I	M	H	M	H	H	H
Pygmy whitefish	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Olympic mudminnow	N	N	N	H	N	H	H	N	H	N	N	N	N	N	N	N	N	N
Lake chub	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Leopard dace	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Margined sculpin	H	N	N	H	N	N	H	N	N	H	N	I	M	N	N	H	N	N
Mountain sucker	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Pacific lamprey	H	I	H	H	I	M	H	I	H	H	I	I	M	I	M	H	I	H
River lamprey	H	H	H	H	H	M	H	H	H	H	I	I	M	H	M	H	H	H
Western brook lamprey	H	N	H	H	N	M	H	N	H	H	N	I	M	N	M	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	M	H	?	H	H	?	I	M	?	M	H	?	H
Longfin smelt	H	H	N	H	H	N	H	N	N	M	I	N	I	?	N	H	I	N
Eulachon	H	H	N	H	H	N	H	N	N	H	I	N	I	?	N	H	I	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N

	Construction & Maintenance Activities			**Hydraulic and Geomorphic Modifications**			**Ecosystem Fragmentation**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**		
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Pacific cod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	N	N	N	I	N	N	I	N	N	H	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Newcomb’s littorine snail	N	H	N	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
Great Columbia River spire snail	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N
California floater (mussel)	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H	H	N	H
Western ridged mussel	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-25. **Species- and habitat-specific risk of take for mechanisms of impacts associated with tide gates.**

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Ecosystem Fragmentation			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications		
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine
Chinook salmon	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Coho salmon	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Chum salmon	H	H	I	H	H	I	H	H	H	I	I	I	M	H	I	H	H	H
Pink salmon	H	H	I	H	H	I	H	H	H	I	I	I	M	H	I	H	H	H
Sockeye salmon	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Steelhead	H	H	H	H	?	H	H	?	H	I	?	I	M	?	M	H	M	H
Coastal cutthroat trout	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Redband trout	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Westslope cutthroat trout	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Bull trout	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Dolly Varden	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Pygmy whitefish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Olympic mudminnow	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Leopard dace	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Mountain sucker	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Umatilla dace	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Pacific lamprey	H	I	H	H	I	H	H	I	H	I	I	I	M	I	M	H	M	H
River lamprey	H	H	H	H	H	H	H	H	H	I	I	I	M	H	M	H	H	H
Western brook lamprey	H	N	H	H	N	H	H	N	H	I	N	I	M	N	M	H	N	H
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N
White sturgeon	H	?	H	H	?	H	H	?	H	H	?	I	M	?	M	H	?	H
Longfin smelt	H	H	H	H	H	H	H	H	H	I	I	I	I	?	?	H	H	H
Eulachon	H	H	N	M	H	N	H	H	N	I	I	N	I	?	N	H	H	N
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Surf smelt	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific herring	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Lingcod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N

Species	**Construction & Maintenance Activities**			**Hydraulic and Geomorphic Modifications**			**Ecosystem Fragmentation**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**		
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**
Pacific cod	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Pacific hake	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Walleye pollock	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Black rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Brown rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Canary rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
China rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Copper rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Widow rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Olympia oyster	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	H	N
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
California floater (mussel)	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

9.4.5 Habitat Modifications

Habitat modification projects are typically designed with the intent of promoting improvements in habitat conditions for a range of species. Once construction is completed, habitat modifications will not generally impose stressors that result in potential take. This position is predicated on two key assumptions: (1) the project in question has been conceived and designed with proper consideration of the broader ecosystem context in which it will be implemented; and (2) the project is constructed properly and performs as expected.

One exception is beaver dam removal, which is typically intended to address problematic flooding caused by beaver dams and not necessarily to improve habitat conditions. Another exception is woody debris removal, often promoted for the purpose of fish passage, flood protection, and infrastructure protection. These activities are expected to impose stressors that lead to possible take of HCP species.

Construction-related impacts will impose stressors on HCP species that may occur in the affected environment. The magnitude, timing, duration, and frequency of each impact mechanism will vary widely with the project scale and location. The risk of take assessment applies a "worst-case scenario" standard. This assessment is conditioned by the species occurrence and life-history specific uses of habitats. For example, beaver dam removal and in-channel/off-channel habitat creation do not occur in marine and lacustrine environments. Therefore, species and species life-history stages that occur only in these environments will not be exposed to related impact mechanisms and stressors, and there is no resulting risk of take. In contrast, large woody debris removal/placement/modification can occur in any environment type. Therefore, the risk of take in this case must be considered more broadly.

Tables 9-26 through 9-34 identify the risk of take for each of the 52 HCP species by impact mechanism and environment type. The summary risk of take presented in the narrative and the matrices represents the greatest overall risk of take for the category.

9.4.5.1 Beaver Dam Removal/Modifications

The removal or modification of beaver dams in Washington State is not intended to improve habitat conditions; instead, the purpose is to address flooding caused by the beaver dam impoundment or to avoid the potential for catastrophic dam failure with the potential to threaten infrastructure, property, or public health in downstream areas. Beaver dams are a normal constituent of riverine environments in the Pacific Northwest, so removing or modifying them alters natural habitat forming process that HCP species occurring in these environments have adapted to throughout their evolutionary history. Therefore, beaver dam removal would be expected to impose a number of stressors on aquatic species occurring in the affected environment, resulting in a broad potential for risk of take. Beaver dams occur only in riverine environments and associated habitats, so the risk of take resulting from removal applies only to riverine environments.

9.4.5.1.1 Construction Activities

Human activity and equipment operation during dam removal imposes stressors in the form of visual and physical disturbance in the vicinity of the structure. Levels of underwater noise produced by beaver dam removal are uncertain, but given the scale of work and tools used in comparison to known reference values for underwater construction activities, noise levels would not be expected to exceed tolerance thresholds capable of causing injury. Disturbance related to construction would be expected to cause behavioral modification, increased stress, and displacement, and could affect survival, growth, and productivity. This equates to a moderate risk of take.

Once a beaver dam is breached, the impoundment behind the structure will drain. Aquatic species in the impoundment that are trapped in rapidly dewatering habitats face a risk of mortality from stranding, particularly non-motile species and life-history stages. Motile species able to avoid stranding will be rapidly displaced from existing habitats and forced to relocate within disturbed habitat that may present limited foraging opportunities, which could similarly limit survival as well as growth and productivity. These combined stressors equate to a high risk of take for species that utilize beaver dam impoundments.

9.4.5.1.2 Hydraulic and Geomorphic Modifications

Beaver dam removal substantially modifies hydraulic and geomorphic conditions both in the impoundment area and the downstream reach. Following dam removal or modification, open water impoundment and wetland areas upstream of beaver dams will be converted into flowing water environments with unstable channels forming in the impoundment bed. The stream channel in the former impoundment area will seek to find an equilibrium condition. The channel will erode to a stable gradient within the fine sediment bed, creating unstable vertical banks with little or no riparian vegetation to provide root cohesion. These banks will remain in an unstable condition until sufficient erosion and vegetation growth has occurred. This will limit the availability of underbank habitat, and contribute chronic, fine sediment loading to the channel. In systems where sediment loading exceeds transport capacity, the detrimental effects of increased fines on substrate composition may persist for some time. These conditions typically result in poor habitat suitability for HCP species occurring in riverine environments where beaver dam removal is likely to occur, resulting in conditions that are limiting to survival, growth, fitness, and spawning productivity. Species exposed to these stressors face a moderate risk of take.

Beaver dams play an active role in hyporheic exchange in riverine environments. The hydraulic head created by the impoundments has been shown to cause downwelling upstream of the structure, which emerges in downstream areas. This vertical connectivity between surface and groundwater is associated with a number of important ecological processes, including the biogeochemical processing of nutrients and pollutants, and the creation of zones of upwelling that are preferential spawning habitats for salmonids and other species. Consequently, any activity that disrupts vertical connectivity will disrupt these processes, reducing water quality and affecting the availability of suitable habitats. These effects will limit the survival, growth, fitness, and in some cases spawning success. This represents a moderate risk of take for species utilizing these habitats

9.4.5.1.3 Ecosystem Fragmentation

On initial consideration, breaching of beaver dams may appear to improve longitudinal connectivity in riverine systems. Beaver dams represent a potential barrier to fish passage as well as a zone of hydraulic complexity which sequesters sediment, wood, organic material, and water. However, beaver dams are typically semipermeable and do not pose total barriers to fish passage. As a natural feature of the landscape, the hydraulic and structural complexity provided by beaver dams supports a broad array of species during different stages of their life history, including HCP species. The distribution of these features along a longitudinal gradient in riverine ecosystems is an important measure of ecological connectivity, particularly for species such as coho salmon that prefer slow water habitats like beaver ponds for rearing habitat. Altering the longitudinal connectivity of complex, diverse habitats in a riverine environment by draining beaver ponds represents a form of ecosystem fragmentation. Reducing the total area of suitable habitat and increasing the distance between habitat patches limits the abundance and productivity of affected populations, which represents a moderate risk of take.

The draining of beaver dam impoundments eliminates open water habitats and causes the channel system to withdraw from riparian and floodplain areas. Depending on where the stream channel stabilizes in the impoundment area, riparian habitats may be separated from the channel by open ground. This effect fragments the channel from floodplain habitats, reducing the connectivity between terrestrial and aquatic habitats which are highly productive. The reduced availability of these productive habitats may limit survival, growth, and fitness of those species that utilize the affected riverine habitats.

An additional related effect is the vulnerability of disturbed habitats to invasion by exotic plant species. Exposed impoundment beds are likely sites for colonization by invasive species. Once these species become established, they may create a barrier to riparian recovery and a dispersal source for additional colonization. Invasive species may reduce the suitability of floodplain and riparian habitat for refuge, food production, and other ecological functions. These effects would also be considered likely to limit the survival, growth, and fitness of species that utilize the affected riverine habitats.

Collectively, these stressors would be expected to impose a moderate risk of take on those HCP species occurring in the affected area.

9.4.5.1.4 Riparian Vegetation Modifications

Removing beaver dams weakens terrestrial-aquatic linkages, reducing riparian influence on stream channels. Until riparian vegetation can establish after dewatering, there will be reduced vegetation adjacent to the channel and thus decreased direct delivery of organic material to the channel. Impoundments can increase riparian vegetation downstream of the dam by augmenting floodplain groundwater, so removing beaver dams which impound a large cross section of the floodplain can affect downstream riparian functions by altering hyporheic flow.

Fragmented connectivity between the active channel and the riparian zone and reduced riparian productivity in downstream habitats lead to reduction in allochthonous inputs of insects, leaf litter, and LWD. This would reduce habitat suitability and food web productivity, limiting the survival, growth, and fitness of species dependent on the affected environment. This equates to a moderate risk of take.

9.4.5.1.5 Aquatic Vegetation Modifications

Draining beaver dam impoundments converts slack water habitats into flowing water, reducing the amount of habitat suitable for aquatic vegetation. Reduced aquatic vegetation results in the loss of autochthonous production and habitat structure within the affected reach. While these are unique stressors, they are considered to be a component of the broader effects of conversion from slack water to flowing water habitats, and the resulting ecological fragmentation. Therefore, they impose a similar moderate risk of take.

9.4.5.1.6 Water Quality Modifications

The literature on beaver dams and their removal is equivocal with regard to the potential effects on stream temperatures. Beaver dam impoundments are typically shallow, open water habitats that expose greater surface area to solar radiation, and therefore could have higher ambient temperatures on average than open stream channels. Removal of beaver dams may result in reduced stream temperatures which could benefit certain species such as native char that are cold water dependent. However, beaver dam impoundments may also serve moderate water temperatures within optimal ranges for aquatic species that co-evolved with beavers in riverine environments. Applying a worst-case scenario perspective, the removal of or modification of beaver dams is expected to modify stream temperatures unfavorably for the HCP species occurring in these environments, cause avoidance behavior, and otherwise limit the survival, growth, and fitness of exposed species. This equates to a moderate risk of take.

Beaver dam removal or modification mobilizes fine sediments deposited in the impoundment. This will increase suspended sediment levels within the affected area immediately upon dam removal, and for an extended period afterwards as the channel within the former impoundment erodes to a stable configuration. Bank erosion within the impoundment will continue to contribute fine sediment loading during high flow events until riparian vegetation growth provides sufficient root cohesion for bank stability. Short-term increases in suspended sediment loading following beaver dam removal could potentially reach concentrations high enough to cause injury or mortality to sensitive species and life-history stages in downstream environments, which equates to a high risk of take. Chronic sediment loading over time would be expected to alter habitat suitability, affecting foraging opportunities and behavior. These effects are potentially limiting to survival, growth, and fitness, which equates to a moderate risk of take.

Beaver dam impoundments sequester a variety of nutrients and pollutants. Research has demonstrated that the biogeochemical processes that are active in beaver dam impoundments can trap pollutants and render them less toxic. Draining the impoundment removes some portion of this capacity and has been shown to result in the relatively rapid release and transport of stored pollutants and nutrients to downstream environments. A large pulse of nutrients could cause temporary eutrophication that, depending on the nature of the downstream environment, could cause a relatively rapid decrease in dissolved oxygen levels. Acute exposure to nutrients or pollutants has the potential to cause injury or mortality, which represents a high risk of take.

9.4.5.2 Large Woody Debris Placement/Removal/Modifications

LWD projects may involve (1) the placement or repositioning of LWD to improve habitat conditions and the functioning of ecological processes, or (2) the removal of LWD from the aquatic environment to facilitate human uses. This latter type of project occurs most often in riverine environments. In the marine environment, LWD removal from structures such as jetties and breakwaters may interfere with eventual deposition of LWD in the littoral environment.

If LWD placement projects are properly designed for their ecological context, and function as intended, the impact mechanisms associated with the project would not be expected to impose stressors on aquatic species once construction is complete. In contrast, LWD removal projects have been shown to detrimentally affect ecological conditions, resulting in an ongoing risk of take. To assess take from LWD placement and removal, we assumed worst-case scenarios. Because the construction impacts for LWD placement projects are more extensive than those for removal, risk of take from construction activities is based on the stressors imposed by LWD placement. For the remaining impact mechanisms, risk of take is rated based on the effects of LWD removal.

9.4.5.2.1 Construction Activities

Construction of LWD projects may involve driving pilings, heavy equipment operation and materials placement, and work area dewatering. The majority of these activities are temporary in nature, lasting from a few days to several weeks, depending on the size of the project. The risk of take associated with construction activity varies by impact mechanism and is dependent on the project-specific magnitude of that impact mechanism. The risk of take resulting from construction also varies by the type of environment, the life-history stages exposed, and the intent of the project. For example, an engineered logjam in a riverine setting may have significant construction-related impacts but will produce an array of beneficial changes in habitat conditions. The risk of take associated with the project would be limited to those individuals that are in the river during construction. The impact mechanisms associated with beneficial changes in habitat conditions are presumed not to impose stressors leading to risk of take. In contrast, the removal of LWD from a stream system (e.g., to protect infrastructure) would involve impact mechanisms that impose stressors during construction, as well as from adverse changes in habitat characteristics.

The operation of heavy construction equipment and the physical placement or removal of LWD and other related materials imposes stressors in the form of increased underwater noise, as well as physical and visual disturbance. The magnitude of these stressors varies widely, depending on the scale of the project in question and the specific construction measures used. Applying a "worst-case-scenario" perspective, the magnitude of these stressors can be significant. For example, many engineered logjam designs include placement of timber or in some cases steel piles using either impact or vibratory hammers. Sound pressure from pile driving has the potential to cause injury and mortality.

Construction-related bank, channel, and shoreline disturbance could result in localized decreased stream bank and shoreline stability, as well as increased erosion and turbidity. These effects could recur during seasonal high-flow conditions. The risk of take depends on species-specific sensitivity to increased turbidity. More motile fish species experience only temporary behavioral alteration and a low risk of take. Less motile fish life-history stages or sessile invertebrates could experience a high risk of take from decreased survival due to mortality from substrate sedimentation and smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success.

Temporary dewatering and fish handling pose a relatively high risk of take. Even with appropriate protocols and experienced field crews, high levels of mortality can result.

9.4.5.2.2 Hydraulic and Geomorphic Modifications

To assess take, we assumed that LWD placement projects are properly designed for the ecosystem context, and that the impact mechanisms imposed will result in beneficial changes in habitat conditions. Therefore, regardless of environment type, these impact mechanisms will produce no stressors and no resulting risk of take.

In contrast, we expect that hydraulic and geomorphic modification caused by LWD removal impose an array of impact mechanisms and related stressors. LWD removal projects in rivers often extensively modify the environment, imposing a number of stressors on those species that use these habitats. Risk of take depends on the size and scale of the project in question, and on species that use the area. The ratings represent the highest potential risk of take associated with LWD removal projects.

In rivers, LWD removal can change channel geometry, flow conditions, and substrate composition. Alteration of any of these habitat components can change the suitability of the habitat for various life-history stages of HCP species. These habitat alterations are essentially permanent and continuous, and can lead to changes in the productivity of the habitat for spawning, forage, rearing, and refuge. In a worst-case scenario, these effects are in turn likely to lead to reduced spawning success as well as reduced survival, growth, and fitness for species and life-history stages dependent on the affected habitat.

In the nearshore marine environment, although specific research data are lacking, anecdotal assessments suggest that LWD can modify local scale hydraulic and geomorphic conditions, affecting habitat structure and the quality and distribution of habitat patches. The risk of take resulting from removing LWD is strongly linked to species-specific dependence on the nearshore environment. Removing LWD can alter wave energy, altering water temperatures and the sorting and transport of sediments, and resulting in a moderate risk of take. Removing LWD can alter longshore transport of sediments, leading to localized alterations in substrate composition and stability, and resulting in a moderate risk of take.

In lacustrine environments, LWD can alter water temperatures, shoreline stability, and the accumulation of allochthonous and autochthonous materials, altering the suitability of nearshore habitats for those species dependent on these habitats, leading to decreased survival, growth, and fitness. This equates to a moderate risk of take for species that are dependent on these habitats during some phase of their life history. Removal of LWD can change the depositional environment by altering nearshore current and wave energy regimes, and by altering longshore sediment transport. This can lead to changes in substrate conditions that may be beneficial or detrimental to individual species. Because substrate composition is an important determinant of community structure in the lacustrine environment, these habitat changes can alter community structure and habitat suitability for those species dependent on the original habitat condition. This equates to a moderate risk of take for species that are dependent on these habitats due to effects on the survival, growth, and productivity of exposed life-history stages.

The hydraulic and geomorphic modifications caused by the removal of LWD from a stream channel can influence and alter groundwater and surface water exchange in the vicinity. This mechanism is generally equated with a moderate to low risk of take for species exposed to this stressor, depending on species-specific, life-history characteristics. Species with a moderate risk of take include those with life-history stages that are dependent on hyporheic exchange for its beneficial effects on water temperature and dissolved oxygen levels. Hyporheic exchange also plays a key role in nutrient cycling and food web productivity in alluvial bed rivers. Projects resulting in significant alteration of hyporheic exchange could adversely affect food web productivity, thereby limiting foraging opportunities for fish and invertebrate species dependent on these types of environments.

9.4.5.2.3 Ecosystem Fragmentation

In riverine systems, LWD removal can result in channel degradation and other forms of hydraulic and geomorphic modification leading to the disconnection of floodplain and off-channel habitats. This poses a moderate risk of take. The removal of LWD reduces the structural complexity of instream habitat, reducing the density and longitudinal distribution of habitat patches and leading to reduced food web productivity and the reduced availability of habitats suitable for those HCP species that occur in these environments. This impact mechanism equates to a moderate risk of take.

Marine and lacustrine environments are not as dominated as riverine environments by the longitudinal transport of water, sediment, and other materials, so the influence of LWD on ecological connectivity is less pronounced. LWD provides cover and organic substrate and has been shown to influence wave energy and sediment deposition in the surrounding environment, and to influence the stability of the boundary between the riparian and littoral zone. The removal of LWD may lead to simplification of the nearshore environment and reduced longshore connectivity of suitable habitat patches, and may alter connectivity along the gradient between the littoral and riparian environment. Reduced longshore connectivity of suitable habitats may lead to increased stress, increased predation risk, and reduced foraging opportunities for juvenile Chinook, chum, and pink salmon, and other species that utilize the nearshore environment during early life-history stages. Exposure to these stressors may limit survival, growth, and fitness, which would equate to a moderate risk of take. Fragmentation of riparian and littoral connectivity may equate to a similar level of risk. For example, LWD accumulations have been shown to promote littoral vegetation growth, and riparian vegetation has been demonstrated to influence incubation success in forage fishes. Alteration of the connectivity between the littoral and riparian zone could affect the suitability of habitats for species such as forage fish and Newcomb's littorine snail that are dependent on these fringing environments.

9.4.5.2.4 Riparian Vegetation Modifications

In riverine environments, LWD placement projects may present riparian impacts if excavation of the bank is necessary to anchor the foundation of an engineered LWD structure. Because restoration of the affected area is typically required as a condition of the HPA permit process, impacts on riparian vegetation are usually intermediate-term in their duration with riparian function returning as the replanted vegetation becomes established. The extent of riparian impacts associated with LWD placement is likely to be limited, and the duration over which the impact mechanism imposes stressors will depend on the time required for the riparian function to recover.

In many riverine projects, LWD removal projects take place from existing infrastructure, such as roadways and bridges, with the intent of providing protection of that infrastructure, and thus do not modify riparian vegetation. In other cases, LWD removal may require the disturbance of intact riparian vegetation to create a construction access point. Hydraulic and geomorphic effects caused by LWD removal may lead to fragmentation of riparian habitat from the aquatic environment, imposing a number of stressors on those HCP species that occur in the affected habitat. The longer term effects of removal projects are a primary consideration in the worst-case scenario based approach to assessing the risk of take. Fish species that are dependent on habitats altered by the removal of LWD are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. This equates to a moderate risk of take.

In marine environments, LWD placement projects most often take place on exposed beaches. The effects of construction activities during wood placement on riparian vegetation are typically limited. Usually only the construction access point is affected, and riparian disturbance may be further limited if an established access point is used. If existing access points are used, or the project is implemented from a barge or vessel, then the effects of the project on riparian vegetation from construction will be insignificant.

Removal of LWD in marine environments may expose the shoreline to increased wave action, leading to soil erosion and loss of riparian habitat. For many species, the risk of take associated with marine riparian impact mechanisms is unknown because the scientific understanding of the related ecological processes is in its infancy, and the extent to which many marine or anadromous species rely on the nearshore environment during their life history is unclear.

In lacustrine environments, LWD placement projects would not be expected to degrade riparian vegetation conditions outside of construction access points, and these effects would be expected to diminish over time as the site restoration matures. LWD removal projects could expose the shoreline to increased wave energy, encouraging cyclical shoreline erosion that chronically degrades riparian functions over longer time periods. This equates to a moderate risk of take for species with a demonstrable dependence on these habitats because the reduction in suitable habitat area because of reduced survival, growth, and fitness.

9.4.5.2.5 Aquatic Vegetation Modifications

During construction, aquatic vegetation in the footprint of LWD structures can be eradicated or buried by the placement of fill or structural material. After construction of a LWD structure, or the removal or repositioning of LWD, changes in wave energy, circulation patterns, flow and/or current velocities, and substrate composition can lead to adverse or beneficial alterations in aquatic vegetation.

In riverine systems, protected slow-water areas created by LWD placement projects may increase suitable habitat for emergent vegetation. The removal of LWD would be expected to reduce this area, resulting in the loss of aquatic vegetation functions.

9.4.5.2.6 Water Quality Modifications

LWD placement and removal projects have the potential to introduce toxic substances from accidental spills during the project construction phase. This presents a moderate risk of take.

LWD placement and removal projects can increase suspended solids during construction or from bank and channel bed instability caused by channel adjustment following LWD removal projects. The severity of individual stressor exposure will vary depending on the nature of the effect, its magnitude and duration, and the sensitivity of the species and life-history stage exposed. These stressors would induce a moderate risk of take.

In rivers, additions of large wood debris are generally expected to have limited effects on dissolved oxygen conditions. Decreased nutrient retention associated with LWD removal from riverine environments could theoretically impose some eutrophication-related effects on downstream habitats, but the scale of these effects is expected to be insignificant in all but the most extreme cases (e.g., LWD removal projects that cause dewatering of impounded or backwatered areas). LWD placement projects would be expected to increase sequestration of organic material, distributing nutrient cycling more broadly across the riverine landscape. Dissolved oxygen levels in marine or lacustrine environments are not driven by large woody debris. The risk of take associated with changes to dissolved oxygen caused by LWD projects is insignificant.

9.4.5.3 Spawning Substrate Augmentation

Spawning substrate augmentation projects are usually designed to mitigate the loss of spawning suitable substrate caused by hydromodification or other sources of environmental degradation. If these projects are designed properly and are implemented as intended, spawning substrate augmentation is expected to improve the functioning of ecological processes resulting in improved habitat conditions. Therefore, with the exception of construction activities and subsequent channel adjustments, the impact mechanisms associated with this type of project would not be expected to impose ecological stressors, and the related risk of take is limited.

9.4.5.3.1 Construction

Substrate augmentation projects require the use of heavy machinery to place gravel sized material either directly into the stream channel or along the channel bank to allow for passive distribution during flood conditions. Primary impact mechanisms associated with project construction include the in-water operation of heavy equipment and related noise, visual, and physical disturbance, and bank and channel disturbance from equipment use and materials placement. These disturbances equate to a moderate risk of take.

9.4.5.3.2 Hydraulic and Geomorphic Modifications

The expected effects of gravel augmentation on channel geometry include particle sorting that creates diverse substrate patches, creation of exposed bars, increased hydraulic complexity and shear zones, and creation of backwaters and other complex alluvial features. These morphologic changes have been observed to increase the quality, quantity, and diversity of both aquatic habitats and associated terrestrial habitats associated with the stream channel. Gravel augmentation can also have the undesirable effect of filling pools, decreasing the amount of pool habitat available. Properly implemented projects would not be expected to impose stressors on HCP species. Therefore, there is no anticipated risk of take.

Gravel augmentation can temporarily reduce bank instability as the channel adjusts to the presence of the new bed material and as the bed elevation rises. Increased bank stability will reduce sediment import into the channel and subsequent spawning gravel and organism burial, and there would be no related risk of take.

Properly implemented spawning gravel augmentation projects improve the composition and stability of spawning substrates. There is no associated risk of take resulting from sediment changes.

9.4.5.3.3 Ecosystem Fragmentation

Spawning gravel augmentation has the potential to raise the channel bed, affecting surface water elevations and, in turn, the frequency at which side channel, off-channel, and floodplain habitats are activated over a range of flow conditions. Properly implemented projects could lead to increased floodplain and side-channel connectivity in riverine environments. This beneficial result would not lead to a risk of take.

Passive augmentation projects often involve the piling of introduced substrate on bars or other channel features, allowing high flows to recruit the introduced material into the channel. Once sediments are entrained into the channel, temporary low flow barriers may occur under certain circumstances before they are fully distributed. In marine and lacustrine environments, substrate piles may be left for recruitment by wave action and longshore sediment transport. Depending on placement, these substrate piles may locally affect the availability of shallow water habitat until the pile has been fully dispersed and distributed. Therefore, this impact mechanism may result in a temporary reduction in the availability and/or accessibility of suitable habitats. This equates to a moderate risk of take for certain types of gravel augmentation projects.

9.4.5.3.4 Aquatic Vegetation Modifications

Spawning gravel augmentation potentially could result in burial or other physical damage to aquatic vegetation. This would impose a temporary reduction in autochthonous production and alteration of the habitat complexity associated with the vegetation itself. These effects may be short-term or long-term in nature, depending on the degree to which the augmentation project changes the existing substrate characteristics and the sensitivity of the local plant community to this change. From a worst-case scenario perspective, these impact mechanisms could limit the availability of foraging habitat, refuge, and cover, and limit food web productivity by reducing autochthonous production. These stressors would equate to a moderate risk of take for those species and life-history stages dependent on aquatic vegetation in the affected environment type.

9.4.5.3.5 Water Quality Modifications

Substrate augmentation projects temporarily increase suspended sediment loading, equated with a moderate risk of take. Once the project has stabilized, substrate augmentation would be expected to have either a neutral or a potentially beneficial effect on water quality conditions.

The increased hyporheic exchange promoted by substrate augmentation promotes the biogeochemical transformation of nutrients, metals, and other pollutants. Stressors related to pollutant exposure would remain unchanged or would be reduced by gravel augmentation projects; therefore, there is no associated risk of take.

The available research tends to indicate that spawning gravel augmentation increases intergravel DO levels resulting in an improvement in habitat conditions. There is no risk of take.

9.4.5.4 In-Channel/Off-Channel Habitat Creation/Modifications

In-channel and off-channel habitat creation or modification projects are intended to enhance or restore degraded habitat conditions. Properly designed properly and implemented, they improve the functioning of ecological processes, resulting in improved habitat conditions. With the exception of the short-term effects associated with construction activities, the impact mechanisms associated with this type of project would not be expected to impose ecological stressors. Therefore, there will be no associated risk of take once project construction is complete.

9.4.5.4.1 Construction Activities

Construction of in-channel/off-channel habitat creation/modification projects could cause disturbances due to noise, physical and visual disturbance, temporary disturbances to the bank, temporary dewatering and fish handling. Each of these is associated with a moderate risk of take until the system reaches a new equilibrium.

9.4.5.4.2 Hydraulic and Geomorphic Modifications

Hydraulic and geomorphic modifications caused by off-channel and side-channel habitat creation are anticipated to improve habitat complexity and increase habitat suitability. Therefore, this impact mechanism category is not expected to impose any stressors on HCP species and there is no related risk of take.

9.4.5.4.3 Ecosystem Fragmentation

Off-channel and side-channel habitat creation will result in increased ecological connectivity and complexity, which will increase the availability and suitability of habitats for HCP species. Therefore, this impact mechanism category is not expected to impose any stressors and there is no related risk of take.

9.4.5.4.4 Riparian Vegetation Modifications

Off-channel and side-channel habitat creation will effectively increase the amount of functional riparian habitat in connection with the active channel, thereby increasing allochthonous inputs, reducing solar radiation exposure and related effects on water temperature, and increasing the buffering capacity. This will increase the availability and suitability of habitats for HCP species. Therefore, this impact mechanism category is not expected to impose any stressors and there is no related risk of take.

9.4.5.4.5 Aquatic Vegetation Modifications

Off-channel and side-channel habitat creation will effectively increase the amount of habitat available for aquatic vegetation growth, thereby increasing autochthonous production, habitat complexity and community structure. This will increase the availability and suitability of habitats for HCP species. Therefore, this impact mechanism category is not expected to impose any stressors and there is no related risk of take.

9.4.5.4.6 Water Quality Modifications

Water quality modifications associated with in-channel and off-channel habitat creation that have the potential to impose stressors on HCP species will occur principally during project construction. The primary water quality related impact mechanism is increased suspended sediments caused by bank and channel disturbance, and the “first flush” effect when the dewatered project areas are first exposed to stream flows. Pollutant loading may also occur as a result of accidental spills from heavy equipment during construction. The related risk of take associated with these impact mechanisms is moderate.

As this type of project becomes functional, increased hyporheic exchange and storage of flood waters in off-channel habitats is likely to provide additional biogeochemical processing capacity that will aid in the sequestration and detoxification of certain forms of pollutants. This effect would be expected to provide beneficial improvements in water quality. This type of project is also expected to improve temperature conditions.

9.4.5.5 Riparian Planting/Restoration/Enhancement

Riparian planting, restoration, and enhancement projects are commonly implemented in conjunction with habitat restoration initiatives or as mitigation for a separate human induced source of habitat degradation. Riparian restoration occurs in riverine, marine, and lacustrine environments and is most typically implemented using manual labor or, in specific circumstances, light machinery. Riparian restoration usually requires only limited disturbance of the bank or shoreline and little or no disturbance of the aquatic environment itself. Once implemented, riparian enhancement projects will generally result in improved riparian function and the related impact mechanisms would not be expected to impose stressors on HCP species. Therefore, the overall risk of take associated is low and is primarily associated with construction for almost all of the HCP species. An exception includes the Newcomb's littorine snail because this species is actually dependent on littoral vegetation and is therefore potentially subject to direct disturbance or injury.

9.4.5.5.1 Construction Activities

Riparian planting may produce construction-related impacts in the form of visual and noise-related disturbance, as well as the disturbance of the stream bank or shoreline. The magnitude of this disturbance is minor in comparison to that produced by the construction of other types of habitat modifications. Because riparian planting takes place primarily out of the water and is short-term in duration, the extent of stressor exposure is limited to short-term behavioral alteration. This equates to a low risk of take for species present in the affected habitat when the activity takes place.

9.4.5.5.2 Hydraulic and Geomorphic Modifications

The immediate effect of riparian enhancement projects on hydraulic and geomorphic conditions is limited. Over time, vegetation growth will consolidate the stream bank or shoreline through root cohesion, thereby increasing stability. As vegetation matures, it will eventually provide a source of LWD recruitment that will have a broad beneficial influence on aquatic habitat. Therefore, riparian vegetation modification impact mechanisms are not expected to impose stressors on the HCP species and there is no related risk of take.

9.4.5.5.3 Ecosystem Fragmentation

The immediate benefits of riparian enhancement projects on ecosystem connectivity are limited, but over time mature vegetation will enhance connectivity by expanding the frequency and distribution of desirable habitat patches. Riparian vegetation modification impact mechanisms are not expected to impose stressors on HCP species, and there is no related risk of take in any environment type.

9.4.5.5.4 Riparian Vegetation Modifications

Riparian enhancement projects are specifically intended to modify the riparian environment for the purpose of providing habitat benefits. These projects are expected to lessen the magnitude of stressors imposed by degraded riparian conditions and will result in no related risk of take in any environment type.

9.4.5.5.5 Aquatic Vegetation Modifications

Riparian enhancement projects are not expected to cause adverse aquatic vegetation modification or to impose any stressors on HCP species. Therefore, there is no associated risk of take in any environment type.

9.4.5.5.6 Water Quality Modifications

Once established, riparian enhancement projects are expected to alter temperature conditions for the benefit of native aquatic species through increased shading and through buffering ambient air temperatures. In riverine environments, these effects will primarily take the form of moderated water temperatures. In both marine and lacustrine environments, increased shading will moderate water temperatures primarily in isolated nearshore shallow water environments. Altered ambient air temperatures and increased shading on marine shorelines will provide additional benefits for sand lance and surf smelt, HCP species that spawn in the upper intertidal zone. Collectively, this is expected to improve habitat suitability in all environment types. Therefore, it will not impose stressors on HCP species and there will be no resulting risk of take.

Riparian enhancement projects have some limited potential to increase sediment loading to the aquatic environment during and immediately following the construction phase. This may occur during manual reworking of the bank or shoreline environment for planting and soil amendment, and exposure to the first high-water or runoff events that follow project completion. In practice, the amount of sediment loading likely to result from riparian enhancement is low relative to that produced by other types of habitat projects because the extent of ground disturbance is generally more limited. With proper project design and BMP implementation, the short-term increase in sediment loading produced by riparian enhancement is not expected to exceed levels sufficient to adversely affect survival, growth, or fitness of HCP species. Therefore, this impact mechanism is equated with a low risk of take.

Once established, riparian enhancement projects are expected to slow the overland flow of stormwater, encouraging infiltration and vegetative filtering. The improved buffering and filtering capacity would be expected to reduce the delivery of pollutants to aquatic ecosystems, and decrease shoreline erosion that contributes to sediment loading. As such, this type of project will not directly produce any pollutant-related stressors, and will reduce the incidence and severity of pollutant loading from other sources. Therefore, no risk of take is anticipated.

9.4.5.6 Wetland Creation/Restoration/Enhancement

Wetland creation, restoration, and enhancement projects enhance or restore degraded habitat conditions. Under the presumption that these projects are designed properly for the surrounding ecological context and are implemented as intended, they would be expected to improve the functioning of ecological processes and to result in improved habitat conditions. Therefore, ecological stressors would only be expected to occur during the short-term period required for construction and the intermediate-term period required for vegetation and site hydrology to mature. The related risk of take resulting from wetland projects would be expected to diminish over time.

9.4.5.6.1 Construction Activities

Under a worst-case scenario, wetland construction effects from large-scale projects could occur within existing aquatic habitat, requiring fish exclusion and dewatering; use heavy machinery for clearing and grading to contour the project area for the desired hydrologic conditions; place LWD, rock, or other materials as habitat structure or components in water level control structures; breach existing hydromodifications to establish connectivity with surface waters; and require extensive revegetation.

Heavy equipment operation in and around riparian areas during wetland construction and the breaching of hydromodifications or other barriers to connect wetlands to surface waters have the potential to impose a number of stressors on the aquatic environment, and equate with a moderate risk of take. Bank, channel, and shoreline disturbance equates to a moderate risk of take. Dewatering and fish handling equates to a moderate risk of take.

9.4.5.6.2 Hydraulic and Geomorphic Modifications

Wetland creation and enhancement projects are typically designed specifically for local hydraulic and geomorphic conditions, often through the reconnection of fragmented floodplain, off-channel habitat, and estuarine habitats. These measures would be expected to improve habitat complexity and increase habitat suitability for a wide range of aquatic and terrestrial species. Therefore, this impact mechanism category is not expected to impose any stressors on HCP species and there is no related risk of take.

9.4.5.6.3 Ecosystem Fragmentation

Wetland creation and enhancement projects are designed to increase ecological connectivity and complexity, increasing the availability and suitability of habitats for HCP species. Therefore, this impact mechanism category is not expected to impose any stressors and there is no related risk of take.

9.4.5.6.4 Riparian Vegetation Modifications

Wetland creation and enhancement projects typically incorporate the preservation and restoration of riparian buffer vegetation, maintaining or increasing the availability and suitability of habitats for HCP species. Therefore, this impact mechanism category is not expected to impose any stressors and there is no related risk of take.

9.4.5.6.5 Aquatic Vegetation Modifications

Wetland creation and enhancement projects will, in most cases, increase the amount of habitat available for aquatic vegetation growth, thereby increasing autochthonous production, habitat complexity and community structure. This will increase the availability and suitability of habitats for HCP species. Therefore, this impact mechanism category is not expected to impose any stressors and there is no related risk of take.

9.4.5.6.6 Water Quality Modifications

The risk of take resulting from wetland creation and enhancement, from temporary increases in suspended sediment, is expected to be moderate.

9.4.5.7 Beach Nourishment/Contouring

Beach nourishment and contouring projects address degraded beach conditions, most often caused by shoreline modification or overwater structures. Under the presumption that these projects are designed properly for the surrounding ecological context and are implemented as intended, beach nourishment would be expected to improve the functioning of ecological processes and result in improved habitat conditions. After a short period of construction activities and subsequent channel adjustments, beach nourishment and contouring would not be expected to impose ecological stressors.

9.4.5.7.1 Construction Activities

Beach nourishment can result in the immediate burial of benthic organisms and aquatic vegetation and, if present, forage fish eggs and the non-motile larvae of certain fish species that are prevalent in the nearshore environment. Impacts on benthic organism diversity and abundance are typically temporary as these communities tend to recover from disturbance quickly. However, this impact mechanism could result in a short-term, localized reduction in foraging opportunities for those species dependent on these prey resources, potentially affecting growth and fitness. This equates to a moderate risk of take. In the case of non-motile HCP species or species life-history stages exposed to this stressor, there is a high likelihood of direct mortality or injury, which equates to a high risk of take.

9.4.5.7.2 Hydraulic and Geomorphic Modifications

Beach nourishment projects directly alter the hydraulic and geomorphic characteristics of the affected shoreline environment. Because they are typically intended to address beach degradation most often caused by shoreline modification projects, properly designed beach nourishment projects either directly or indirectly result in improved hydraulic and geomorphic conditions from a habitat perspective. On this basis, this impact mechanism would generally not be expected to impose stressors on aquatic organisms and there would be no related risk of take. In practice, however, current understanding of marine and lacustrine geomorphology is sufficiently limited to create design uncertainty in site-specific circumstances. On this basis, some risk of take may occur that is difficult to quantify, resulting in an uncertain risk of take.

9.4.5.7.3 Ecosystem Fragmentation

The ability of beach nourishment to reconnect or disconnect pre-existing shoreline communities depends on the nature of the shorelines adjacent to the activity site. If the substrate is significantly different than the shorelines adjacent to it, or if added sediment buries aquatic vegetation, the activity may fragment the alongshore transit of HCP species. Under the basic presumption that the project is properly designed and implemented, these forms of ecosystem fragmentation should not occur. In contrast, beach contouring can moderate the ecological gradient between the littoral and riparian zones, thereby improving ecological connectivity. Properly designed beach nourishment projects should not further degrade or may even improve this impact mechanism and would therefore not impose any related stressors. Accordingly, there will be no related risk of take from this impact mechanism.

9.4.5.7.4 Riparian Vegetation Modifications

Beach nourishment projects do not involve direct modification of the riparian environment, except where necessary to provide access for equipment and materials. In a worst-case scenario, limited riparian disturbance necessary for equipment and materials access may occur. For HCP species with limited dependence on marine or lacustrine riparian vegetation, the resultant effects of this limited disturbance are expected to be insignificant. Some HCP species (e.g., sand lance, surf smelt, Chinook salmon) inhabit littoral fringe areas during life-history stages that are more sensitive to stressor exposure. These species face a moderate risk from the limited and minor resultant effects.

Newcomb’s littorine snail is considered an exception. Because this species has a limited distribution and is entirely dependent on shoreline vegetation, any alteration of its habitat would be associated with a high risk of take.

Once established, beach nourishment projects are expected to produce beneficial changes in hydraulic and geomorphic conditions along the shoreline, thereby contributing to preservation and improvement of riparian conditions.

9.4.5.7.5 Aquatic Vegetation Modifications

Beach nourishment projects may in some cases alter aquatic vegetation, leading to localized shifts in food web productivity, possibly affecting foraging opportunities for dependent species and life-history stages. This equates to a moderate risk of take resulting from decreased growth and fitness. Alterations may reduce cover and rearing habitat, equating to a moderate risk of take.

9.4.5.7.6 Water Quality Modifications

Beach nourishment projects could temporarily increase suspended sediments. Motile species and life-history stages exposed to temporary sediment impacts at low occurrence frequency experience only temporary disturbance, behavioral alteration, and low risk of take. Sessile invertebrates or relatively immobile life-history stages may experience decreased survival and reduced foraging opportunities leading to a moderate to high risk of take. Sublethal levels of suspended sediments may affect the foraging success of planktonic herring larvae, leading to decreased foraging success and decreased survival, growth, and fitness.

Beach nourishment projects could introduce toxics through accidental spills from construction equipment. In extreme cases, these effects can include direct mortality. More commonly, chronic, low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation leading to decreased survival, growth, and fitness. This presents a moderate to high risk of take to species potentially exposed to this stressor, depending on life-history specific sensitivity.

9.4.5.8 Reef Creation/Restoration/Enhancement

Reef creation, restoration, or enhancement projects involve the placement of rock, wood, concrete, metal (e.g., sunken vessel hulls), or other materials on the bottom, creating three dimensional structure that attracts or encourages the settlement of fish, invertebrates, and aquatic vegetation. Ideally, these structures are intended to increase the availability of suitable habitat for fish and invertebrates, leading to increased abundance and productivity. However, the degree to which reefs provide this function versus merely concentrating existing populations without increasing abundance or productivity remains uncertain.

9.4.5.8.1 Construction Activities

Reef construction may result in visual, physical, and noise related disturbance and displacement, injury or mortality. Temporary disturbance and displacement and a decreased ability to sense predators and prey due to auditory masking effects equate to a moderate risk of take. Limited or non-motile species or life-history stages occurring in the project area during materials placement face a high risk of take from physical injury or mortality from burial and mechanical injury.

9.4.5.8.2 Hydraulic and Geomorphic Modifications

Artificial reefs that extend above the wave closure depth can significantly affect nearshore wave energy, current velocities, and circulation patterns, leading to decreased habitat availability, decreased survival, growth, and fitness, and a moderate risk of take. The physical alterations of the shoreline environment that accompany some reef creation projects can cause alterations in sediment supply and substrate conditions through alteration of longshore sediment transport. In conjunction with altered wave energy, this can lead to changes in substrate conditions. This equates to a moderate risk of take.

9.4.5.8.3 Ecosystem Fragmentation

Reefs created in nearshore habitats may alter habitat characteristics. Changes in foraging opportunities and increased predation risk due to increased cover and habitat for predatory fish species may lead to decreased survival, growth, and fitness, which equates to a moderate risk of take. Reefs constructed offshore and below the wave closure depth would be expected to have limited effects on the nearshore environment and would provide beneficial habitat conditions for a variety of HCP species including rockfish, lingcod, and northern abalone. These structures may present little or no risk of take from ecosystem fragmentation.

9.4.5.8.4 Aquatic Vegetation Modifications

Artificial reefs may displace aquatic vegetation, altering autochthonous inputs and habitat complexity/community structure, resulting in a moderate risk of take.

9.4.5.8.5 Water Quality Modifications

Reef creation projects may temporarily increase suspended solids. In general, motile species and life-history stages exposed to temporary sediment impacts at low occurrence frequency experience only temporary disturbance, behavioral alteration, and low risk of take. Sessile invertebrates or relatively immobile life-history stages exposed to increased suspended solids may experience decreased survival and reduced foraging opportunities leading to a moderate risk of take.

Reef creation projects may introduce toxic substances through accidental spills, through the presence of toxic substances in materials used to create the structure (e.g., decommissioned ships), or through resuspension of contaminated sediments during construction if these substances are present in the project area. This effect may continue for some time if the hydraulic effects of the structure induce scouring. In extreme cases, exposure to contaminants can result in direct mortality. More commonly, chronic, low-level exposure to a variety of contaminants can cause physiological injury and/or contaminant bioaccumulation leading to decreased survival, growth, and fitness. This presents a moderate risk of take.

9.4.5.9 Eelgrass and Other Aquatic Vegetation Creation/Enhancement/Restoration

Aquatic vegetation restoration has the least potential for take of any habitat modification. Assuming that the project has been conceived and designed properly for the ecosystem context, augmentation of eelgrass and other types of aquatic vegetation are expected to provide beneficial improvements habitat conditions.

9.4.5.9.1 Construction Activities

Eelgrass and aquatic vegetation enhancement projects are typically implemented by hand or by using nonpowered equipment. Construction-related effects would be low intensity physical and visual disturbance. Because planting success requires careful placement, sessile or non-motile organisms would be at relatively low risk of physical injury when carefully trained staff are used. Therefore, the stressors imposed by construction would be expected to result only in short-term disturbance and behavioral modification, which equates to a low risk of take.

9.4.5.9.2 Hydraulic and Geomorphic Modifications

Properly designed aquatic vegetation enhancement projects would not be expected to impose hydraulic and geomorphic stressors on the aquatic environment, and there is no associated risk of take.

9.4.5.9.3 Ecosystem Fragmentation

Properly designed aquatic vegetation enhancement projects would be expected to improve ecological connectivity by increasing the diversity of habitat patches and improving their distribution. There is no associated risk of take.

9.4.5.9.4 Aquatic Vegetation Modifications

The intention of aquatic vegetation projects is to enhance the ecological functions provided by aquatic vegetation. There is no associated risk of take.

9.4.5.9.5 Water Quality Modifications

Enhancement of eelgrass and other aquatic vegetation has essentially no potential for adverse effects on water quality, with the exception of minor effects during project construction. In the case of eelgrass enhancement, there are often effectively no discernable construction-related effects on water quality. Taking a worst-case scenario perspective, short-term increases in suspended sediment levels may occur during construction-related disturbance from vessel operation and manual or diver labor. This would be expected to result in a low risk of take, predominantly in the form of temporary behavioral effects, for a short-term period. Once vegetation has been established, chronic levels of suspended sediment should decrease as vegetation encourages the settling of fines. Increased dissolved oxygen levels and other beneficial water quality effects would be expected to develop once a successfully implemented eelgrass or aquatic vegetation enhancement project matures. Following project completion, no further risk of take would be expected.

Table 9-26. Species- and habitat-specific risk of take for mechanisms of impacts associated with beaver dam removal.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Chinook salmon are known to occur in environments where beaver dam removal or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Coho salmon	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Coho salmon are known to occur in environments where beaver dam removal or modification may occur, and preferentially select beaver impoundments for juvenile rearing habitat. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Chum salmon	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Chum salmon are known to spawn in environments where beaver dam removal or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Pink salmon	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Pink salmon are known to spawn in environments where beaver dam removal or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Sockeye salmon	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Sockeye salmon are known to spawn in environments where beaver dam removal or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Steelhead	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Steelhead are known to occur in environments where beaver dam removal or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Coastal cutthroat trout	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Coastal cutthroat trout are known to occur in environments where beaver dam removal or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Westslope cutthroat trout	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Westslope cutthroat and redband trout are known to occur in environments where beaver dam removal or modification may occur. Therefore, these species are vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Redband trout	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	
Bull trout	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Native char occur in rivers and streams where beaver are often abundant, indicating the potential for these species to be exposed to the effects of beaver dam removal or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Dolly Varden	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	
Pygmy whitefish	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	This species spawns in small, cold water tributary streams to rearing lakes. These habitats are potentially within the range of beaver distribution, indicating the potential for exposure to beaver dam removal projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Olympic mudminnow	H	N	N	H	N	N	M	N	N	H	N	N	M	N	N	H	N	N	Primary habitats are wetlands and small, slow-flowing streams, presumably including beaver pond habitats. Therefore, this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Margined sculpin	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages where beaver dam removal or modification has the potential to occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Mountain sucker	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	This species spawns in tributary habitats potentially suitable for beaver dam removal or modification projects. Therefore this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Lake chub	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats are potentially subject to beaver dam removal or modification projects. Therefore, this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Leopard dace	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east, including smaller rivers and stream systems. Therefore, this species occurs in habitats potentially subject to beaver dam removal or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Umatilla dace	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in habitats potentially subject to beaver dam removal or modification. Therefore, this

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																			species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Western brook lamprey	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Western brook lamprey spend their entire life history in habitats potentially affected by beaver dam removal. Due to its limited motility and dependence on small streams and similar habitats where beaver dams are prevalent, this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
River lamprey	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	Pacific and river lamprey are anadromous species that spawn in habitats potentially affected by beaver dam removal. Ammocoetes burrow into riverine sediments to rear for extended periods and are similarly vulnerable. Freshwater life-history stages of both species are potentially exposed to a range of impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Pacific lamprey	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Green sturgeon distribution in Washington State is restricted to marine waters; therefore, there is no potential for exposure to beaver dam modification and no related risk of take.
White sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The freshwater distribution of White sturgeon in Washington State is restricted to large river environments that are insensitive to the effects of beaver dam removal projects. Therefore there is no related risk of take.
Longfin smelt	H	N	N	H	N	N	H	N	N	N	N	N	H	N	N	N	N	N	The freshwater distribution of longfin smelt is limited to larger river environments that are insensitive to the effects of beaver dam removal, with the possible exception of the Lake Washington population. Spawning habitats for this population may be in river systems where beaver dam removal or modification could occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Eulachon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The freshwater distribution of this species in Washington State is limited to larger river environments that are insensitive to the effects of beaver dam removal. Therefore there is no related risk of take.
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species does not occur in environments where beaver dam removal or modification take place; therefore, there is no related risk of take.
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where beaver dam removal or modification take place; therefore, there is no related risk of take.
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where beaver dam removal or modification take place; therefore, there is no related risk of take.
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species do not occur in environments where beaver dam removal or modification take place; therefore, there is no related risk of take.
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species do not occur in environments where beaver dam removal or modification take place; therefore, there is no related risk of take.
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where beaver dam removal or modification take place; therefore, there is no related risk of take.
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where beaver dam removal or modification take place; therefore, there

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
																			is no related risk of take.
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where beaver dam removal or modification take place; therefore, there is no related risk of take.
Giant Columbia River limpet	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	M	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, indicating the potential for exposure to beaver dam removal projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Great Columbia River spire snail	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	M	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. This species is unlikely to be exposed to the effects of beaver dam removal projects and no effects are expected. In contrast, the great Columbia River spire snail inhabits smaller tributary streams to the Columbia River where exposure to the effects of beaver dam removal is likely. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
California floater (mussel)	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. The California floater mussel is known to occur in the Okanogan River basin (as well as fringing ponds of the Columbia River). The distribution of both species in small to moderate sized rivers and streams indicates the potential for exposure to beaver dam projects. These non-motile species are particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beaver dam removal.
Western ridged mussel	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	H	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-27. Species- and habitat-specific risk of take for mechanisms of impacts associated with large woody debris placement/removal/modification.

	Construction & Maintenance Activities			**Hydraulic and Geomorphic Modifications**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	H	H	H	H	M	M	N	N	N	M	M	M	M	M	M	M	M	M	Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats. Individuals occurring in spawning, incubation, rearing, and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Coho salmon	H	H	H	H	M	M	N	N	N	M	M	M	M	M	M	M	M	M	Coho salmon occur in riverine, lacustrine, and nearshore marine habitats. Individuals occurring in spawning, incubation, rearing, and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Chum salmon	H	H	I	H	M	I	N	N	I	L	M	I	M	M	I	L	M	I	Chum salmon in Washington State do not use lacustrine habitats and occur in this environment type infrequently. Therefore, the effects of stressor exposure in lacustrine environments are expected to be insignificant. Individuals occurring in spawning, incubation and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Pink salmon	H	H	I	H	M	I	N	N	I	L	M	I	M	M	I	L	M	I	Pink salmon in Washington State do not use lacustrine habitats and occur in this environment type infrequently. Therefore, the effects of stressor exposure in lacustrine environments are expected to be insignificant. Individuals occurring in spawning, incubation and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Sockeye salmon	H	H	H	H	M	M	N	N	N	M	M	M	M	M	M	L	M	M	Sockeye salmon occur in riverine, lacustrine, and nearshore marine habitats, and are particularly dependent on the latter two environment types. Individuals occurring in spawning, incubation, rearing and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Steelhead	H	?	H	H	?	M	M	M	M	M	?	M	M	?	M	M	?	M	Steelhead occur in riverine, lacustrine, and nearshore marine habitats and are particularly dependent on the latter two environment types. As juvenile steelhead are more typically found far from shore in the marine environment, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain. Individuals occurring in spawning, incubation, rearing, and migratory habitats in fresh water, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Coastal cutthroat trout	H	H	H	H	M	M	M	M	M	M	M	M	M	M	M	M	M	M	This species is prevalent in rivers, estuaries, and nearshore marine habitats, and also occurs at lesser frequencies in lacustrine habitats (e.g., Lake Washington). It is highly dependent on nearshore marine areas for foraging. Individuals occurring in spawning, incubation, rearing, and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Westslope cutthroat trout	H	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	N	M	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Individuals occurring in spawning, incubation, rearing, and migratory habitats in freshwater, and foraging and rearing habitats in lacustrine waters may be exposed to stressors resulting from LWD placement and removal projects.
Redband trout	H	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	N	M	
Bull trout	H	H	H	H	M	M	N	N	N	L	M	M	M	M	M	L	M	M	Native char occur in riverine, lacustrine, and nearshore marine habitats. Individuals occurring in spawning, incubation, rearing and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Dolly Varden	H	H	H	H	M	M	N	N	N	L	M	M	M	M	M	L	M	M	
Pygmy whitefish	H	N	H	H	N	M	M	N	M	N	N	M	M	N	M	M	N	M	Lakes and smaller lake tributaries are primary habitats used by pygmy whitefish. Individuals occurring in spawning, incubation, rearing and migratory habitats in freshwater, and juvenile migratory habitats in marine waters may be exposed to stressors resulting from LWD placement and removal projects.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. These habitats are not typically suited for LWD placement and removal projects, except in the context of wetland enhancement projects (which are addressed in Table 9-8). Outside of this context this species would not likely be exposed to this type of project and there would be no related risk of take.
Margined sculpin	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages. Individuals occurring in spawning, incubation, rearing, and migratory habitats may be exposed to stressors resulting from LWD placement and removal projects.
Mountain sucker	H	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	N	M	This species is commonly found in moderate to large rivers and lakes suitable for LWD placement and removal projects. Individuals occurring in spawning, incubation, rearing, and migratory habitats may be exposed to stressors resulting from LWD placement and removal projects.
Lake chub	H	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	N	M	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. Individuals occurring in spawning, incubation, rearing, and migratory habitats may be exposed to stressors

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																			resulting from LWD placement and removal projects.
Leopard dace	H	N	H	H	N	M	N	N	N	M	N	M	M	N	M	M	N	M	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. Therefore, this species occurs in habitats potentially suitable for LWD placement and removal projects at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Umatilla dace	H	N	H	H	N	M	N	N	N	M	N	M	M	N	M	M	N	M	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers (including reservoirs within the Columbia and Snake River systems). Therefore, this species occurs in habitats potentially suitable for LWD placement and removal projects at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Western brook lamprey	H	N	H	H	N	M	N	N	N	N	N	N	N	N	N	M	N	M	This species is characterized by isolated breeding populations favoring small streams and brooks. Therefore, this species occurs in habitats potentially suitable for LWD placement and removal projects at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
River lamprey	H	H	H	H	M	M	M	?	M	?	?	?	M	M	M	M	M	M	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of rivers to rear for extended periods, potentially years. The non-motile ammocoete life-history stage is more susceptible to acute transient water quality impacts and direct physical disturbance. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanisms affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults, which in turn equates to a moderate risk of take.
Pacific lamprey	H	L	H	H	L	M	M	N	M	?	N	?	M	L	M	M	L	M	Pacific lamprey are anadromous, with migratory corridors extending from marine waters to small tributary streams. Ammocoetes burrow into riverine sediments to rear for extended periods. The non-motile ammocoete life-history stage is more susceptible to acute transient water quality impacts and direct physical disturbance. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore less likely to be exposed to project-related stressors in the nearshore marine environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanisms affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This in turn equates to a moderate risk of take.
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	Green sturgeon distribution in Washington State is restricted to marine waters; therefore, there is no potential for exposure to LWD placement/removal projects in freshwater and marine environments and no related risk of take. Sensitivity to impact mechanisms resulting from this project type in marine environments is uncertain.
White sturgeon	H	?	H	H	?	M	M	?	M	M	?	M	M	?	M	M	?	M	The freshwater distribution of White sturgeon in Washington State is restricted to large river environments that are insensitive to the effects of LWD placement and removal projects. However, side channel and margin habitats in the Columbia River and lacustrine impoundments used for juvenile rearing may be suitable environments for this project type. Therefore some potential for stressor exposure exists. Sensitivity to impact mechanisms resulting from this project type in marine environments is uncertain.
Longfin smelt	H	I	M	H	I	M	M	I	M	N	I	M	M	I	M	M	I	M	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems potentially suitable for LWD placement or removal projects. Longfin smelt are also located in Lake Washington. Demersal adhesive eggs are vulnerable to acute transient water quality impacts and direct physical effects. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors.
Eulachon	H	I	N	H	I	N	M	I	N	N	I	N	M	I	N	M	I	N	

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Pacific sand lance	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	M	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure from marine LWD projects is high. Larvae of both species disperse in nearshore waters for early rearing. These beach-spawning species depend on a narrow range of substrate conditions for suitable spawning habitat, increasing sensitivity to hydraulic and geomorphic effects. Planktonic larvae are also dependent on nearshore current and circulation patterns for rearing survival. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Surf smelt	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	M	N	
Pacific herring	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	M	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure from hydraulic/geomorphic and aquatic vegetation modifications is high. Planktonic larvae disperse in nearshore waters for early rearing and are dependent on current and circulation patterns for survival, growth, and fitness. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lingcod	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to water quality related impact mechanisms from LWD placement and removal projects. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 and 150 m) and, therefore, have low exposure potential. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific hake	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. Therefore, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific cod	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Walleye pollock	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Brown rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. Therefore, rockfish can experience stressor exposure across all life-history stages. Juveniles disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile once they have settled, juveniles are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Copper rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Greenstriped rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Widow rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Yellowtail rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Quillback rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Black rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
China rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Tiger rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Bocaccio rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Canary rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Redstripe rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	

Species	**Construction & Maintenance Activities**			**Hydraulic and Geomorphic Modifications**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			**Comments**
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Yelloweye rockfish	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	
Olympia oyster	N	H	N	N	M	N	N	N	N	N	M	N	N	M	N	N	M	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile during much of its life-history, it is vulnerable to both short-term construction and water quality related impacts, as well as modification of hydraulic and geomorphic conditions in the nearshore environment. Modification of current, wave, and circulation patterns may also affect larval settlement, influencing survival during this life-history stage. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Northern abalone	N	I	N	N	I	N	N	I	N	N	I	N	N	I	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) in depth, but is not found in shallow water habitats where the construction and water quality-related effects of LWD projects are most pronounced.
Newcomb's littorine snail	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	M	N	The Newcomb's littorine snail inhabits *Salicornia* marshes on the littoral fringe. It is intolerant of extended submergence in both fresh and marine water; therefore, it not a true aquatic species. This species will be particularly vulnerable to LWD placement and removal projects in saltmarsh environments, particularly removal projects.
Giant Columbia River limpet	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, indicating the potential for exposure to spawning gravel augmentation projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from LWD placement and removal projects.
Great Columbia River spire snail	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. Exposure to the effects of spawning gravel augmentation is likely to occur in smaller river systems and streams in habitat by this species. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from LWD placement and removal projects.
California floater (mussel)	H	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	N	M	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake rivers and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. Therefore, both species may occur in habitats potentially suitable for LWD placement and removal projects. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Habitat accessibility modifications will not directly affect this species; however, indirect effects could occur through direct effects on host-fish.
Western ridged mussel	H	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-28. Species- and habitat-specific risk of take for mechanisms of impacts associated with spawning substrate augmentation.

	Construction & Maintenance Activities			**Hydraulic and Geomorphic Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	Chinook salmon are known to occur in environments where spawning gravel augmentation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Coho salmon	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	Coho salmon are known to occur in environments where spawning gravel augmentation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Chum salmon	H	N	N	N	N	N	I	N	N	M	N	N	M	N	N	Chum salmon are known to spawn in environments where spawning gravel augmentation or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Pink salmon	H	N	N	N	N	N	I	N	N	M	N	N	M	N	N	Pink salmon are known to spawn in environments where spawning gravel augmentation or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Sockeye salmon	H	N	N	N	N	N	I	N	N	M	N	N	M	N	N	Sockeye salmon are known to spawn in environments where spawning gravel augmentation or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Steelhead	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	Steelhead are known to occur in environments where spawning gravel augmentation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Coastal cutthroat trout	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	Coastal cutthroat trout are known to occur in environments where spawning gravel augmentation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Westslope cutthroat trout	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	Westslope cutthroat and redband trout are known to occur in environments where spawning gravel augmentation or modification may occur. Therefore, these species are vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Redband trout	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	
Bull trout	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	Native char occur in rivers and streams, indicating the potential for these species to be exposed to the effects of spawning gravel augmentation or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Dolly Varden	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	
Pygmy whitefish	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	This species spawns in small, cold water tributary streams to rearing lakes, indicating the potential for exposure to spawning gravel augmentation projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats used by this species include wetlands and small, slow-flowing streams, environments unsuitable for spawning gravel augmentation. Therefore there is no related risk of take.
Margined sculpin	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages where spawning gravel augmentation or modification has the potential to occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Mountain sucker	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	This species spawns in tributary habitats potentially suitable for spawning gravel augmentation projects. Therefore this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Lake chub	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats are potentially subject to spawning gravel augmentation or modification projects. Therefore, this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Leopard dace	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east, including smaller rivers and stream systems. Therefore, this species occurs in habitats potentially subject to spawning gravel augmentation or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Umatilla dace	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in habitats potentially subject to spawning gravel augmentation. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Western brook lamprey	H	N	N	N	N	N	?	N	N	M	N	N	M	N	N	Western brook lamprey spend their entire life history in habitats potentially affected by spawning gravel augmentation. Due to its limited motility and dependence on small streams and similar habitats this species is particularly vulnerable to the impact mechanisms, stressors,

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																and related risk of take resulting from spawning gravel augmentation.
River lamprey	H	N	N	N	N	N	?	N	N	M	N	N	M	N	N	Pacific and river lamprey are anadromous species that spawn in habitats potentially affected by spawning gravel augmentation. Ammocoetes burrow into riverine sediments to rear for extended periods and are similarly vulnerable. Freshwater life-history stages of both species are potentially exposed to a range of impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Pacific lamprey	H	N	N	N	N	N	?	N	N	M	N	N	M	N	N	
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Green sturgeon distribution in Washington State is restricted to marine waters; therefore, there is no potential for exposure to spawning gravel augmentation and no related risk of take.
White sturgeon	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	The freshwater distribution of white sturgeon in Washington State is restricted to large river environments that are potentially suitable for spawning gravel augmentation. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Longfin smelt	H	N	N	N	N	N	N	N	N	M	N	N	M	N	N	The freshwater distribution of longfin smelt may include river environments where spawning gravel augmentation may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Eulachon	H	N	N	N	N	N	N	N	N	M	N	N	M	N	N	The freshwater distribution of eulachon may include river environments where spawning gravel augmentation may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species does not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species do not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species do not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where spawning gravel augmentation takes place; therefore, there is no related risk of take.

Species	**Construction & Maintenance Activities**			**Hydraulic and Geomorphic Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			**Comments**
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Giant Columbia River limpet	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, indicating the potential for exposure to spawning gravel augmentation projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Great Columbia River spire snail	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. Exposure to the effects of spawning gravel augmentation is likely to occur in smaller river systems and streams in habitat by this species. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
California floater (mussel)	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. The California floater mussel is known to occur in the Okanogan River basin (as well as fringing ponds of the Columbia River). The distribution of both species in small to moderate sized rivers and streams indicates the potential for exposure to spawning gravel augmentation projects. These non-motile species are particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting from spawning gravel augmentation.
Western ridged mussel	H	N	N	N	N	N	M	N	N	M	N	N	M	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take? = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-29. Species- and habitat-specific risk of take for mechanisms of impacts associated with in-channel and off-channel habitat creation/modification.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Chinook salmon are known to occur in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coho salmon	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Coho salmon are known to occur in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Chum salmon	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Chum salmon are known to spawn in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pink salmon	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Pink salmon are known to spawn in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Sockeye salmon	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Sockeye salmon are known to spawn in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from in-channel/off-channel habitat creation.
Steelhead	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Steelhead are known to occur in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coastal cutthroat trout	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Coastal cutthroat trout are known to occur in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Westslope cutthroat trout	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Westslope cutthroat and redband trout are known to occur in environments where in-channel/off-channel habitat creation or modification may occur. Therefore, these species are vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Redband trout	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	
Bull trout	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Native char occur in rivers and streams, indicating the potential for these species to be exposed to the effects of in-channel/off-channel habitat creation or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Dolly Varden	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	
Pygmy whitefish	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	This species spawns in small, cold water tributary streams to rearing lakes, indicating the potential for exposure to in-channel/off-channel habitat creation projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																			habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats used by this species include wetlands and small, slow-flowing streams, environments unsuitable for in-channel/off-channel habitat creation. Therefore there is no related risk of take.
Margined sculpin	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages where in-channel/off-channel habitat creation or modification has the potential to occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Mountain sucker	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	This species spawns in tributary habitats potentially suitable for in-channel/off-channel habitat creation projects. Therefore this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Lake chub	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats are potentially subject to in-channel/off-channel habitat creation or modification projects. Therefore, this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Leopard dace	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east, including smaller rivers and stream systems. Therefore, this species occurs in habitats potentially subject to in-channel/off-channel habitat creation or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Umatilla dace	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in habitats potentially subject to in-channel/off-channel habitat creation. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Western brook lamprey	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Western brook lamprey spend their entire life history in habitats potentially affected by in-channel/off-channel habitat creation. Due to its limited motility and dependence on small streams and similar habitats this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
River lamprey	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	Pacific and river lamprey are anadromous species that spawn in habitats potentially affected by in-channel/off-channel habitat creation. Ammocoetes burrow into riverine sediments to rear for extended periods and are similarly vulnerable. Freshwater life-history stages of both species are potentially exposed to a range of impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific lamprey	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	
Green sturgeon	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Green sturgeon distribution in Washington State is restricted to marine waters; therefore, there is no potential for exposure to in-channel and off-channel habitat creation projects and no related risk of take.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
White sturgeon	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The freshwater distribution of white sturgeon in Washington State is restricted to large river environments that are potentially suitable for in-channel/off-channel habitat creation. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Longfin smelt	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The freshwater distribution of longfin smelt may include river environments where in-channel/off-channel habitat creation may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Eulachon	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The freshwater distribution of eulachon may include river environments where in-channel/off-channel habitat creation may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific sand lance	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species do not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.
Surf smelt	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Pacific herring	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.
Lingcod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.
Pacific hake	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species do not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.
Pacific cod	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Walleye pollock	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Brown rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	These marine species do not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.
Copper rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Greenstriped rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Widow rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Yellowtail rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Quillback rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Black rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
China rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Tiger rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Bocaccio rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Canary rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Redstripe rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Yelloweye rockfish	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where in-channel/off-channel habitat creation takes place; therefore, there is no related risk of take.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Giant Columbia River limpet	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, indicating the potential for exposure to in-channel/off-channel habitat creation projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Great Columbia River spire snail	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. Exposure to the effects of in-channel/off-channel habitat creation is likely to occur in smaller river systems and streams inhabited by this species. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
California floater (mussel)	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. The California floater mussel is known to occur in the Okanogan River basin (as well as fringing ponds of the Columbia River). The distribution of both species in small to moderate sized rivers and streams indicates the potential for exposure to this project type. These non-motile species are particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with in-channel/off-channel habitat creation. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Western ridged mussel	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-30. Species- and habitat-specific risk of take for mechanisms of impacts associated with riparian planting/restoration/enhancement.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Chinook salmon	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	Chinook salmon are known to occur in environments where riparian planting/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coho salmon	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	Coho salmon are known to occur in environments where riparian planting/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Chum salmon	L	L	I	N	N	I	N	N	I	N	N	I	L	L	I	N	N	I	Chum salmon are known to spawn in environments where riparian planting/restoration/enhancement may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pink salmon	L	L	I	N	N	I	N	N	I	N	N	I	L	L	I	N	N	I	Pink salmon are known to spawn in environments where riparian planting/restoration/enhancement may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Sockeye salmon	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	Sockeye salmon are known to spawn in environments where riparian planting/restoration/enhancement may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from riparian planting/restoration/enhancement.
Steelhead	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	Steelhead are known to occur in environments where riparian planting/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coastal cutthroat trout	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	Coastal cutthroat trout are known to occur in environments where riparian planting/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Westslope cutthroat trout	L	N	L	N	N	N	N	N	N	N	N	N	L	N	L	N	N	N	Westslope cutthroat and redband trout are known to occur in environments where riparian planting/restoration/enhancement may occur. Therefore, these species are vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Redband trout	L	N	L	N	N	N	N	N	N	N	N	N	L	N	L	N	N	N	
Bull trout	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	Native char occur in rivers and streams, indicating the potential for these species to be exposed to the effects of riparian planting/restoration/enhancement or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Dolly Varden	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	
Pygmy whitefish	L	N	L	N	N	N	N	N	N	N	N	N	L	N	L	N	N	N	This species spawns in small, cold water tributary streams to rearing lakes, indicating the potential for exposure to riparian planting/restoration/enhancement projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.

Species	**Construction & Maintenance Activities**			**Hydraulic and Geomorphic Modifications**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Olympic mudminnow	N	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	Primary habitats used by this species include ponds, wetlands and small, slow-flowing streams. For the purpose of this assessment, these habitats are considered lacustrine, and are potentially suitable for riparian planting/restoration/enhancement projects. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Margined sculpin	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages where riparian planting/restoration/enhancement has the potential to occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Mountain sucker	L	N	L	N	N	N	N	N	N	N	N	N	L	N	L	N	N	N	This species spawns in tributary habitats potentially suitable for riparian planting/restoration/enhancement projects. Therefore this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Lake chub	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats are potentially subject to riparian planting/restoration/enhancement projects. Therefore, this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Leopard dace	L	N	L	N	N	N	N	N	N	N	N	N	L	N	L	N	N	N	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east, including smaller rivers and stream systems. Therefore, this species occurs in habitats potentially subject to riparian planting/restoration/enhancement or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Umatilla dace	L	N	L	N	N	N	N	N	N	N	N	N	L	N	L	N	N	N	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in habitats potentially subject to riparian planting/restoration/enhancement. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Western brook lamprey	L	N	L	N	N	N	N	N	N	N	N	N	L	N	L	N	N	N	Western brook lamprey spend their entire life history in habitats potentially affected by riparian planting/restoration/enhancement. Due to its limited motility and dependence on small streams and similar habitats this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
River lamprey	L	L	L	N	N	N	N	N	N	N	N	N	L	L	L	N	N	N	Pacific and river lamprey are anadromous species that spawn in habitats potentially affected by riparian planting/restoration/enhancement. Ammocoetes burrow into riverine sediments to rear for extended periods and are similarly vulnerable. Freshwater life-history stages of both species are potentially exposed to a range of impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific lamprey	L	I	L	N	N	N	N	N	N	N	N	N	L	I	L	N	N	N	
Green sturgeon	N	L	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	Green sturgeon distribution in Washington State is restricted to marine waters, typically in offshore environments. Therefore this species has limited potential for stressor exposure to and related risk of take.
White sturgeon	L	L	L	N	?	N	N	?	N	N	?	N	L	?	L	N	?	N	The freshwater distribution of white sturgeon in Washington State is restricted to large river environments that are potentially suitable for riparian planting/restoration/enhancement. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Longfin smelt	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	The freshwater distribution of longfin smelt may include river environments where riparian planting/restoration/enhancement may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Eulachon	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	The freshwater distribution of eulachon may include river environments where riparian planting/restoration/enhancement may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific sand lance	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	These marine species use upper intertidal habitats subject to the effects of marine riparian planting/restoration/enhancement projects. Therefore, some exposure to short-term stressors may occur, resulting in risk of take.
Surf smelt	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Pacific herring	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	This marine species uses lower intertidal habitats subject to the effects of marine riparian planting/restoration/enhancement projects. Therefore, some exposure to short-term stressors may occur, resulting in risk of take.
Lingcod	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	This marine species uses nearshore habitats during larval rearing and may experience exposure to minor stressors resulting from marine riparian planting/restoration/enhancement projects.
Pacific hake	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	These marine species use nearshore habitats during larval rearing and may experience exposure to minor stressors resulting from marine riparian planting/restoration/enhancement projects.
Pacific cod	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Walleye pollock	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Brown rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	These marine species use nearshore habitats during larval rearing and may experience exposure to minor stressors resulting from marine riparian planting/restoration/enhancement projects.
Copper rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Greenstriped rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Widow rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Yellowtail rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Quillback rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Black rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
China rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Tiger rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Bocaccio rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Canary rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Redstripe rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Yelloweye rockfish	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	
Olympia oyster	N	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	This marine species uses nearshore habitats and may experience exposure to minor stressors resulting from marine riparian planting/restoration/enhancement projects.
Northern abalone	N	I	N	N	N	N	N	N	N	N	N	N	N	I	N	N	N	N	This marine species may occur in nearshore habitats and may experience exposure to minor stressors resulting from marine riparian planting/restoration/enhancement projects. However, distribution in deeper waters away from the shoreline limits the severity of stressor exposure to insignificant levels.
Newcomb's littorine snail	N	H	N	N	N	N	N	N	N	N	N	N	N	N	N	N	L	N	This marine species uses a specific type of littoral vegetation (*Salicornia* spp.) as its sole habitat and is limited in distribution to a few discrete locations in Washington State. Therefore, this species will be highly sensitive to adverse effects from riparian vegetation projects that affect its habitat.

Species	**Construction & Maintenance Activities**			**Hydraulic and Geomorphic Modifications**			**Riparian Vegetation Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			**Comments**
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Giant Columbia River limpet	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, indicating the potential for exposure to riparian planting/restoration/enhancement projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Great Columbia River spire snail	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. Exposure to the effects of riparian planting/restoration/enhancement is likely to occur in smaller river systems and streams inhabited by this species. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
California floater (mussel)	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. The California floater mussel is known to occur in the Okanogan River basin (as well as fringing ponds of the Columbia River). The distribution of both species in small to moderate sized rivers and streams indicates the potential for exposure to this project type. These non-motile species are particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with riparian planting/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Western ridged mussel	L	N	N	N	N	N	N	N	N	N	N	N	L	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-31. Species- and habitat-specific risk of take for mechanisms of impacts associated with wetland creation/restoration/enhancement.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	Chinook salmon are known to occur in environments where wetland creation/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coho salmon	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	Coho salmon are known to occur in environments where wetland creation/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Chum salmon	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	Chum salmon are known to spawn in environments where wetland creation/restoration/enhancement may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pink salmon	H	H	I	N	N	N	N	N	N	N	N	N	M	M	I	N	N	N	Pink salmon are known to spawn in environments where wetland creation/restoration/enhancement may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Sockeye salmon	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	Sockeye salmon are known to spawn in environments where wetland creation/restoration/enhancement may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from wetland creation/restoration/enhancement.
Steelhead	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	Steelhead are known to occur in environments where wetland creation/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coastal cutthroat trout	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	Coastal cutthroat trout are known to occur in environments where wetland creation/restoration/enhancement may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Westslope cutthroat trout	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	Westslope cutthroat and redband trout are known to occur in environments where wetland creation/restoration/enhancement may occur. Therefore, these species are vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Redband trout	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	
Bull trout	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	Native char occur in rivers and streams, indicating the potential for these species to be exposed to the effects of wetland creation/restoration/enhancement or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Dolly Varden	H	H	H	N	N	N	N	N	N	N	N	N	M	M	M	N	N	N	
Pygmy whitefish	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	This species spawns in small, cold water tributary streams to rearing lakes, indicating the potential for exposure to wetland creation/restoration/enhancement projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Olympic mudminnow	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	Primary habitats used by this species include ponds, wetlands and small, slow-flowing streams. For the purpose of this assessment, these habitats are considered lacustrine, and are potentially suitable for wetland creation/restoration/enhancement projects. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Margined sculpin	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages where wetland creation/restoration/enhancement has the potential to occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Mountain sucker	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	This species spawns in tributary habitats potentially suitable for wetland creation/restoration/enhancement projects. Therefore this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Lake chub	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats are potentially subject to wetland creation/restoration/enhancement projects. Therefore, this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Leopard dace	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east, including smaller rivers and stream systems. Therefore, this species occurs in habitats potentially subject to wetland creation/restoration/enhancement or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Umatilla dace	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in habitats potentially subject to wetland creation/restoration/enhancement. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Western brook lamprey	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	Western brook lamprey spend their entire life history in habitats potentially affected by wetland creation/restoration/enhancement. Due to its limited motility and dependence on small streams and similar habitats this species is particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
River lamprey	H	L	H	N	N	N	N	N	N	N	N	N	M	L	M	N	N	N	Pacific and river lamprey are anadromous species that spawn in habitats potentially affected by wetland creation/restoration/enhancement. Ammocoetes burrow into riverine sediments to rear for extended periods and are similarly vulnerable. Freshwater life-history stages of both species are potentially exposed to a range of impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific lamprey	H	I	H	N	N	N	N	N	N	N	N	N	M	I	M	N	N	N	
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	L	N	N	?	N	Green sturgeon distribution in Washington State is restricted to marine waters, typically in offshore environments. Therefore this species has limited potential for exposure to wetland enhancement projects in coastal environments, and similarly limited risk of take.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
White sturgeon	H	?	H	N	?	N	N	?	N	N	?	N	M	L	M	N	?	N	The freshwater distribution of white sturgeon in Washington State is restricted to large river environments that are potentially suitable for wetland creation/restoration/enhancement. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Longfin smelt	H	N	N	N	N	N	N	N	N	N	N	N	M	L	N	N	N	N	The freshwater distribution of longfin smelt may include river environments where wetland creation/restoration/enhancement may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Eulachon	H	N	N	N	N	N	N	N	N	N	N	N	M	L	N	N	N	N	The freshwater distribution of eulachon may include river environments where wetland creation/restoration/enhancement may be appropriate. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific sand lance	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	These marine species use upper intertidal habitats subject to the effects of estuarine and coastal marine wetland restoration/enhancement projects. Therefore, some exposure to short-term stressors may occur, resulting in risk of take.
Surf smelt	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Pacific herring	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	This marine species uses lower intertidal habitats subject to the effects of estuarine and coastal marine wetland restoration/enhancement projects. Therefore, some exposure to short-term stressors may occur, resulting in risk of take.
Lingcod	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	This marine species uses nearshore habitats during larval rearing and may experience exposure to minor stressors resulting from estuarine and coastal marine wetland restoration/enhancement projects.
Pacific hake	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	These marine species use nearshore habitats during larval rearing and may experience exposure to minor stressors resulting from estuarine and coastal marine wetland restoration/enhancement projects.
Pacific cod	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Walleye pollock	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Brown rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	These marine species use nearshore habitats during larval rearing and may experience exposure to minor stressors resulting from estuarine and coastal marine wetland restoration/enhancement projects.
Copper rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Greenstriped rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Widow rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Yellowtail rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Quillback rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Black rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
China rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Tiger rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Bocaccio rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Canary rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Redstripe rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Yelloweye rockfish	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	
Olympia oyster	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	This marine species uses nearshore habitats and may experience exposure to minor stressors resulting from estuarine and coastal marine wetland restoration/enhancement projects.
Northern abalone	N	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	This marine species may occur in nearshore habitats and may experience exposure to minor stressors resulting from estuarine and coastal marine wetland restoration/enhancement projects. However, distribution in deeper waters away from the shoreline

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
																			limits the severity of stressor exposure to insignificant levels.
Newcomb's littorine snail	N	H	N	N	N	N	N	H	N	N	N	N	N	N	N	N	N	N	This marine species uses a specific type of littoral vegetation (*Salicornia* spp.) as its sole habitat and is limited in distribution to a few discrete locations in Washington State. Therefore, this species will be highly sensitive to adverse effects from estuarine and coastal marine wetland restoration/enhancement projects that affect its habitat.
Giant Columbia River limpet	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, indicating the potential for exposure to wetland creation/restoration/enhancement projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Great Columbia River spire snail	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. Exposure to the effects of wetland creation/restoration/enhancement is likely to occur in smaller river systems and streams inhabited by this species. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
California floater (mussel)	H	N	H	N	N	N	N	N	N	N	N	N	M	N	M	N	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. The California floater mussel is known to occur in the Okanogan River basin (as well as fringing ponds of the Columbia River). The distribution of both species in small to moderate sized rivers and streams indicates the potential for exposure to this project type. These non-motile species are particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with wetland creation/restoration/enhancement. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Western ridged mussel	H	N	N	N	N	N	N	N	N	N	N	N	M	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-32. Species- and habitat-specific risk of take for mechanisms of impacts associated with beach nourishment/contouring.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	N	H	H	N	?	?	N	M	M	N	N	N	N	M	M	N	N	N	Chinook salmon are known to occur in lacustrine and marine environments where beach nourishment/contouring may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coho salmon	N	H	H	N	?	?	N	M	M	N	N	N	N	M	M	N	N	N	Coho salmon are known to occur in lacustrine and marine environments where beach nourishment/contouring may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Chum salmon	N	H	I	N	?	I	N	M	I	N	N	N	N	M	I	N	N	N	Chum salmon are known to occur in marine environments where beach nourishment/contouring may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pink salmon	N	H	I	N	?	I	N	M	I	N	N	N	N	M	I	N	N	N	Pink salmon are known to occur in marine environments where beach nourishment/contouring may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Sockeye salmon	N	H	H	N	?	?	N	I	M	N	N	N	N	M	M	N	N	N	Sockeye salmon are known to occur in lacustrine and marine environments where beach nourishment/contouring may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from beach nourishment/contouring.
Steelhead	N	H	H	N	?	?	N	I	M	N	N	N	N	M	M	N	N	N	Steelhead are known to occur in lacustrine and marine environments where beach nourishment/contouring may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Coastal cutthroat trout	N	H	H	N	?	?	N	M	M	N	N	N	N	M	M	N	N	N	Coastal cutthroat trout are known to occur in lacustrine and marine environments where beach nourishment/contouring may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Westslope cutthroat trout	N	N	H	N	N	?	N	N	M	N	N	N	N	N	M	N	N	N	Westslope cutthroat and redband trout are known to occur in lacustrine environments where beach nourishment/contouring may occur. Therefore, these species are vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Redband trout	N	N	H	N	N	?	N	N	M	N	N	N	N	N	M	N	N	N	
Bull trout	N	H	H	N	?	?	N	M	M	N	N	N	N	M	M	N	N	N	Native char occur in lacustrine and marine environments, indicating the potential for these species to be exposed to the effects of beach nourishment/contouring or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Dolly Varden	N	H	H	N	?	?	N	M	M	N	N	N	N	M	M	N	N	N	
Pygmy whitefish	N	N	H	N	N	?	N	N	I	N	N	N	N	N	M	N	N	N	This species rears in lakes, indicating the potential for exposure to lacustrine beach nourishment/contouring projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																			improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats used by this species include ponds, wetlands and small, slow-flowing streams. These habitats are unsuitable for beach nourishment/contouring projects. Therefore there will be no risk of take from this project type.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages. Therefore there is no potential for exposure to this type of project and no related risk of take.
Mountain sucker	N	N	H	N	N	?	N	N	M	N	N	N	N	N	M	N	N	N	This species rears in lacustrine habitats potentially suitable for beach nourishment/contouring projects. Therefore this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Lake chub	N	N	H	N	N	?	N	N	M	N	N	N	N	N	M	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. While unlikely, the lacustrine habitats used by this species could potentially be subject to beach nourishment/contouring projects. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Leopard dace	N	N	H	N	N	?	N	N	M	N	N	N	N	N	M	N	N	N	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east, including smaller rivers and stream systems. While exposure is generally unlikely, this species may occur in lacustrine impoundments potentially subject to beach nourishment/contouring or modification. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Umatilla dace	N	N	H	N	N	?	N	N	M	N	N	N	N	N	M	N	N	N	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in lacustrine impoundments potentially subject to beach nourishment/contouring projects. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Western brook lamprey spend their entire life history in small streams and rivers unsuitable for beach nourishment/contouring. Therefore there is no risk of stressor exposure and no related risk of take.
River lamprey	N	H	H	N	?	?	N	I	M	N	N	N	N	L	M	N	N	N	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of rivers to rear for extended periods, potentially years. The non-motile ammocoete life-history stage is more susceptible to acute transient water quality impacts and direct physical disturbance. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, meaning there is some potential for exposure to beach nourishment/contouring related stressors in the nearshore environment. Life-history stages exposed to lacustrine beach nourishment/contouring projects face a high risk of take during project construction, while exposure to marine projects produce lesser risk of take because the effects are avoidable. Once established, these projects should result in no risk of take.
Pacific lamprey	N	I	H	N	N	?	N	N	M	N	N	N	N	I	M	N	N	N	Pacific lamprey are anadromous, with migratory corridors extending from marine waters to small tributary streams. Ammocoetes burrow into riverine sediments to rear for extended periods. The non-motile ammocoete life-history stage is more susceptible to acute transient water quality impacts and direct physical disturbance during construction of lacustrine beach nourishment/contouring projects and face high risk of take. In marine waters, Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore face an insignificant potential for exposure to stressors from marine beach nourishment projects. Once established, these projects should result in no risk of take.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Green sturgeon	N	?	N	N	?	N	N	?	N	N	N	N	N	?	N	N	N	N	Green sturgeon distribution in Washington State is restricted to offshore marine waters. The potential for exposure to stressors resulting from beach nourishment/contouring is limited to avoidable disturbance and water quality effects. Risk of take is similarly low.
White sturgeon	N	L	H	N	I	?	N	I	M	N	N	N	N	L	M	N	N	N	The freshwater distribution of white sturgeon in Washington State includes lacustrine impoundments that are potentially suitable for beach nourishment/contouring. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Longfin smelt	N	H	H	N	I	?	N	I	M	N	N	N	N	M	M	N	N	N	The freshwater distribution of longfin smelt includes lacustrine environments where beach nourishment/contouring may be appropriate. The marine distribution of this species is primarily limited to offshore habitats so the risk of stressor exposure in these environments is insignificant. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting construction and temporary water quality impacts associated with beach nourishment/contouring in lacustrine environments. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Eulachon	N	H	N	N	I	N	N	I	N	N	N	N	N	M	N	N	N	N	Eulachon distribution in freshwater is limited to river environments unsuitable for beach nourishment/contouring projects. The marine distribution of this species is primarily limited to offshore habitats so the risk of stressor exposure in these environments is insignificant.
Pacific sand lance	N	H	N	N	?	N	N	M	N	N	N	N	N	H	N	N	N	N	These marine species are dependent on littoral beach habitats for spawning, which are directly affected by beach nourishment/contouring are highly likely to occur; therefore, the likelihood of stressor exposure and related risk of take during project construction is high. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Surf smelt	N	H	N	N	?	N	N	M	N	N	N	N	N	H	N	N	N	N	
Pacific herring	N	H	N	N	?	N	N	M	N	N	N	N	N	H	N	N	N	N	This marine species is dependent on littoral beach habitats for spawning which are directly affected by beach nourishment/contouring are likely to occur; therefore, the likelihood of stressor exposure and related risk of take during project construction is high. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Lingcod	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	This marine species occurs in nearshore habitats as rearing larvae and juveniles with limited motility. Therefore, these vulnerable life-history stages may be exposed to stressors from beach nourishment/contouring projects associated with construction and water quality impacts. Exposure to these short-term stressors presents risk of take. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific hake	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	These marine species occur in nearshore habitats as rearing larvae and juveniles with limited motility. Therefore, these vulnerable life-history stages may be exposed to stressors from beach nourishment/contouring projects associated with construction and water quality impacts. Exposure to these short-term stressors presents risk of take. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Pacific cod	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Walleye pollock	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Brown rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	These marine species occur in nearshore habitats as rearing juveniles with limited motility. Therefore, this vulnerable life history-stage may be exposed to stressors from beach nourishment/contouring projects associated with construction and water quality impacts. Exposure to these short-term stressors presents risk of take. Once established, these projects will improve habitat conditions and would not be expected to impose ecological stressors; therefore, there will be no related risk of take.
Copper rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Greenstriped rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Widow rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Yellowtail rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Quillback rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Black rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
China rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Tiger rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Bocaccio rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Canary rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Redstripe rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Yelloweye rockfish	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	
Olympia oyster	N	H	N	N	?	N	N	I	N	N	N	N	N	H	N	N	N	N	This marine species occurs in the shallow nearshore marine environments and is non-motile once settled. Therefore, this species could potentially be exposed to impact mechanisms and stressors from beach nourishment projects in the nearshore environment. Limited mobility increases sensitivity to construction and water quality related stressors. Once established, these projects should result in improved habitat conditions and will have no ongoing risk of take.
Northern abalone	N	H	N	N	?	N	N	I	N	N	N	N	N	M	N	N	N	N	This marine species occupies nearshore marine habitats covering a range of depths and is effectively non-motile. Therefore, this species could potentially be exposed to impact mechanisms and stressors from beach nourishment projects. Limited mobility increases sensitivity to construction and water quality related stressors. Once established, these projects should result in improved habitat conditions and will have no ongoing risk of take, with the exception of potential water quality impacts if reef materials include toxic substances with leaching potential.
Newcomb's littorine snail	N	N	N	N	?	N	N	H	N	N	N	N	N	N	N	N	N	N	Newcomb's littorine snail occurs solely in saltmarsh environments unsuitable for beach nourishment. Therefore, there is no potential for stressor exposure and no related risk of take.
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for beach nourishment projects. Therefore there is no potential for stressor exposure and no related risk of take
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are unsuitable for beach nourishment projects. Therefore there is no risk of stressor exposure and no related risk of take.
California floater (mussel)	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. The California floater mussel is known to occur in the Okanogan River basin (as well as fringing ponds of the Columbia River). The distribution of both species in small to moderate sized rivers and streams indicates no potential for exposure to beach nourishment projects. Therefore there is no risk of stressor exposure and no related risk of take.
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-33. Species- and habitat-specific risk of take for mechanisms of impacts associated with reef creation.

	Construction & Maintenance Activities			**Hydraulic and Geomorphic Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	Chinook salmon are known to occur in lacustrine and marine environments where reef creation may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Coho salmon	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	Coho salmon are known to occur in lacustrine and marine environments where reef creation or modification may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Chum salmon	N	H	I	N	M	I	N	M	I	N	M	I	N	M	I	Chum salmon are known to occur in marine environments where reef creation may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Pink salmon	N	H	I	N	M	I	N	M	I	N	M	I	N	M	I	Pink salmon are known to occur in marine environments where reef creation may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Sockeye salmon	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	Sockeye salmon are known to occur in lacustrine and marine environments where reef creation may occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Steelhead	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	Steelhead are known to occur in lacustrine and marine environments where reef creation may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Coastal cutthroat trout	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	Coastal cutthroat trout are known to occur in lacustrine and marine environments where reef creation may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Westslope cutthroat trout	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	Westslope cutthroat and redband trout are known to occur in lacustrine environments where reef creation may occur. Therefore, these species are vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Redband trout	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	
Bull trout	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	Native char occur in lacustrine and marine habitats where reef creation projects may occur. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Dolly Varden	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	
Pygmy whitefish	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species rears throughout their juvenile and adult life history in lakes, indicating the potential for exposure to reef creation projects in lacustrine environments. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats used by this species include wetlands and small, slow-flowing streams, environments unsuitable for reef creation. Therefore there is no related risk of take.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for reef creation; therefore, there is no risk of stressor exposure and no related risk of take.
Mountain sucker	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species occurs in lacustrine habitats potentially suitable for reef creation projects. Therefore there is some potential for stressor exposure and related risk of take.
Lake chub	N	N	H	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats are unsuitable for reef creation projects; therefore, there is no potential for stressor exposure and no related risk of take.
Leopard dace	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. Therefore, this species occurs in habitats potentially subject to reef creation or modification. Therefore, this species is vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Umatilla dace	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in lacustrine impoundments potentially subject to reef creation. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Western brook lamprey spend their entire life history in habitats potentially unsuitable for reef creation. Therefore there is no risk of stressor exposure and no related risk of take.
River lamprey	N	H	H	N	M	M	N	?	?	N	M	M	N	M	M	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of rivers to rear for extended periods, potentially years. The non-motile ammocoete life-history stage is more susceptible to acute transient water quality impacts and direct physical disturbance. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, meaning there is some potential for exposure to beach nourishment/contouring related stressors in the nearshore environment. Life-history stages exposed to lacustrine reef creation projects face a high risk of take during project construction, while exposure to marine projects produce lesser risk of take because the exposed life-history stages have higher motility. Once established, these projects should result in no risk of take.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Pacific lamprey	N	H	H	N	M	M	N	?	?	N	M	M	N	M	M	Pacific lamprey are anadromous, with migratory corridors extending from marine waters to small tributary streams. Ammocoetes burrow into riverine sediments to rear for extended periods. The non-motile ammocoete life-history stage is more susceptible to acute transient water quality impacts and direct physical disturbance during construction of lacustrine beach nourishment/contouring projects and face high risk of take. In marine waters, Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore face reduced potential for exposure to stressors from marine reef creation projects. Once established, these projects should result in no risk of take.
Green sturgeon	N	L	N	N	?	N	N	?	N	N	L	N	N	?	N	Green sturgeon distribution in Washington State is restricted to marine waters as foraging adults. Therefore, the potential for stressor exposure is limited to this large, mobile life-history stage.
White sturgeon	N	L	H	N	?	M	N	?	M	N	L	M	N	?	M	The freshwater distribution of white sturgeon in Washington State includes lacustrine environments that are potentially suitable for reef creation, as well as marine habitats where reef creation is likely to occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation. However, sensitivity to stressor exposure in the marine environment is lower because only large motile adults occur in this environment type.
Longfin smelt	M	H	H	N	M	M	N	N	N	N	M	M	N	?	M	The freshwater distribution of longfin smelt includes lacustrine environments (Lake Washington) where reef creation may be appropriate, and marine habitats where reef creation is likely to occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Eulachon	N	H	N	N	M	N	N	N	N	N	M	N	N	?	N	Eulachon occur in marine environments where reef creation is likely to occur. Therefore, this species is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Pacific sand lance	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	These marine species occur in marine environments where reef creation is likely to occur. Therefore, they are potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Surf smelt	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Pacific herring	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	This marine species occurs in marine environments where reef creation is likely to occur. Therefore, it is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Lingcod	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	This marine species occurs in marine environments where reef creation is likely to occur. Therefore, it is potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Pacific hake	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	These marine species occur in marine environments where reef creation is likely to occur. Therefore, they are potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Pacific cod	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Walleye pollock	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Brown rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	These marine species occur in marine environments where reef creation is likely to occur. Therefore, they are potentially vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Copper rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Greenstriped rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Widow rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Yellowtail rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Quillback rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Black rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
China rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Tiger rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Bocaccio rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Canary rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Redstripe rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Yelloweye rockfish	N	H	N	N	M	N	N	M	N	N	M	N	N	M	N	
Olympia oyster	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where reef creation takes place; therefore, there is no related risk of take.
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where reef creation takes place; therefore, there is no related risk of take.
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species does not occur in environments where reef creation takes place; therefore, there is no related risk of take.
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, indicating no potential for exposure to reef creation projects. Therefore, this species is not vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are unsuitable for reef creation projects; therefore, there is no potential for stressor exposure and no related risk of take.
California floater (mussel)	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	The California floater mussel is known to occur in the Okanogan River basin (as well as fringing ponds of the Columbia River). While unlikely, the distribution of this species in lacustrine impoundments presents the potential for exposure to reef creation projects. This non-motile species would be particularly vulnerable to the impact mechanisms, stressors, and related risk of take resulting from reef creation.
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. These environments are unsuitable for reef creation; therefore, there is no risk of stressor exposure and no related risk of take.

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-34. **Species- and habitat-specific risk of take for mechanisms of impacts associated with eelgrass and other aquatic vegetation creation/restoration/enhancement.**

	Construction & Maintenance Activities			**Hydraulic and Geomorphic Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	This species occurs in marine and lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Coho salmon	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	This species occurs in marine and lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Chum salmon	N	H	I	N	M	I	N	M	I	N	M	I	N	M	I	This species occurs in marine and lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Pink salmon	N	H	I	N	M	I	N	M	I	N	M	I	N	M	I	This species occurs in marine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Sockeye salmon	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	This species occurs in marine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Steelhead	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	This species occurs in marine and lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Coastal cutthroat trout	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	This species occurs in marine and lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Westslope cutthroat trout	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	These species occur in lacustrine habitats where aquatic vegetation restoration/enhancement projects may occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Redband trout	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	
Bull trout	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	These species occur in marine and lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Dolly Varden	N	H	H	N	M	M	N	M	M	N	M	M	N	M	M	
Pygmy whitefish	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species occurs in lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats used by this species include wetlands and small, slow-flowing streams, environments potentially suitable for emergent vegetation enhancement projects. Therefore some potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for aquatic vegetation enhancement projects; therefore, there is no risk of stressor exposure and no related risk of take.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Mountain sucker	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species occurs in lacustrine habitats where aquatic vegetation restoration/enhancement projects are likely to occur. Therefore the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Lake chub	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. Occurrence in lacustrine habitats potentially suitable for aquatic vegetation restoration/enhancement projects are likely to occur suggests the potential for stressor exposure exists. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Leopard dace	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. Lacustrine and riverine habitats in the Columbia River could be suitable environments for aquatic vegetation enhancement.
Umatilla dace	N	N	H	N	N	M	N	N	M	N	N	M	N	N	M	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers. Therefore, this species occurs in lacustrine impoundments potentially subject to reef creation. Lacustrine and riverine habitats in the Columbia and Snake rivers could be suitable environments for aquatic vegetation enhancement.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Western brook lamprey spend their entire life history in habitats potentially unsuitable for aquatic vegetation restoration and enhancement projects. Therefore there is no risk of stressor exposure and no related risk of take.
River lamprey	N	H	H	N	M	M	N	?	?	N	M	M	N	M	M	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of rivers to rear for extended periods, potentially years, indicating the potential for exposure to aquatic vegetation enhancement and restoration projects in lakes and estuaries. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, meaning there is some potential for exposure to aquatic vegetation restoration and enhancement projects in both environment types. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take.
Pacific lamprey	N	I	H	N	N	M	N	I	?	N	I	M	N	N	M	Pacific lamprey are anadromous, with migratory corridors extending from marine waters to small tributary streams. Ammocoetes burrow into riverine and lacustrine sediments to rear for extended periods, indicating the potential for exposure to aquatic vegetation enhancement and restoration projects in lakes. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established these projects would be expected to improve habitat conditions; therefore, there will be no ongoing risk of take. In marine waters, Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore face little potential for exposure to this type of project in the marine environment.
Green sturgeon	N	I	N	N	N	N	N	N	N	N	I	N	N	N	N	Green sturgeon distribution in Washington State is restricted to marine waters as foraging adults. Given the tendency for distribution in offshore waters, the potential for exposure to stressors from this project type is insignificant. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
White sturgeon	N	I	L	N	N	N	N	N	N	N	I	L	N	N	N	The freshwater distribution of white sturgeon in Washington State includes lacustrine environments that are potentially suitable for aquatic vegetation restoration and enhancement projects, as well as marine habitats where this project type is likely to occur. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Given the tendency for distribution in offshore waters, the potential for exposure to stressors from this project type is insignificant. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Longfin smelt	N	I	L	N	N	N	N	N	N	N	I	L	N	N	N	The freshwater distribution of longfin smelt includes lacustrine environments (Lake Washington) where aquatic vegetation restoration and enhancement may be appropriate. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Offshore distribution in marine waters suggests that the potential for stressor exposure and related risk of take from this project type are insignificant. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Eulachon	N	I	N	N	N	N	N	N	N	N	I	N	N	N	N	Eulachon occur in marine environments where reef creation is likely to occur. However, associated are limited in magnitude; therefore, the related risk of take is similarly limited. Offshore distribution in marine waters suggests that the potential for stressor exposure and related risk of take from this project type are insignificant. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Pacific sand lance	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	These marine species occur in marine environments where aquatic vegetation restoration and enhancement projects are likely to occur. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Surf smelt	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Pacific herring	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	This marine species occurs in marine environments where aquatic vegetation restoration and enhancement projects are likely to occur. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Lingcod	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	This marine species occurs in marine environments where aquatic vegetation restoration and enhancement projects are likely to occur. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Pacific hake	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	These marine species occur in marine environments where aquatic vegetation restoration and enhancement projects are likely to occur. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Pacific cod	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Walleye pollock	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Brown rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	These marine species occur in marine environments where aquatic vegetation restoration and enhancement projects are likely to occur. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Copper rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Greenstriped rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Widow rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Yellowtail rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Quillback rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Black rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
China rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Tiger rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Bocaccio rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Canary rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Redstripe rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Yelloweye rockfish	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	
Olympia oyster	N	L	N	N	N	N	N	N	N	N	L	N	N	N	N	This marine species occurs in marine environments where aquatic vegetation restoration and enhancement projects are likely to occur. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Northern abalone	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This marine species occurs in nearshore marine environments, however it is typically distributed in habitats unsuitable for aquatic vegetation restoration and enhancement projects. Therefore there is no potential for stressor exposure and no related risk of take.
Newcomb's littorine snail	N	H	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is dependent on saltmarsh vegetation as its sole habitat. Vegetation restoration and enhancement projects may occur in this environment type, suggesting the potential for direct physical injury or mortality during planting activities. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep. These habitats are unsuitable for aquatic vegetation restoration and enhancement projects. Therefore there is no potential for stressor exposure and no related risk of take.
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These habitats are unsuitable for aquatic vegetation restoration and enhancement projects. Therefore there is no potential for stressor exposure and no related risk of take.
California floater (mussel)	N	N	L	N	N	N	N	N	N	N	N	L	N	N	n	The California floater mussel is known to occur in the Okanogan River basin as well as fringing ponds of the Columbia River. The distribution of this species in lacustrine impoundments and fringing pond habitats presents the potential for exposure to aquatic vegetation restoration and enhancement projects. However, associated stressors are limited in magnitude; therefore, the related risk of take is similarly limited. Once established, these projects will result in improved habitat conditions and will produce no ongoing risk of take.
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The western ridged mussel is commonly found in small, clear water tributaries and streams. These habitats are unsuitable for aquatic vegetation restoration and enhancement projects. Therefore there is no potential for stressor exposure and no related risk of take.

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

9.4.6 Marinas and Terminals

Marinas and terminals are very similar to each other, with similar mechanisms of impacts, stressors, and potential risks of take, and so they are treated as one type of activity. Terminals would generally be expected to produce impact mechanisms of greater magnitude and frequency than marinas, but marinas and terminals can vary broadly in scale and activity frequency, and the related impact mechanisms will vary accordingly. A high-volume marine terminal frequented by cargo vessels produces larger and more frequent disturbances than a small recreational marina on a lake. A low-volume ferry terminal serving a lightly populated area will produce less operational and vessel-related disturbance than a large marina supporting a mix of commercial and recreational vessels. Therefore, it is not possible to estimate the risk of take based on facility type alone.

The matrices summarizing the potential risks of take associated with marinas/terminals discuss risks from each mechanism of impact in greater detail than for other types of activities. Tables 9-35 through 9-40 show potential risk of take associated with construction and maintenance, operations and vessel activities, water quality modifications, riparian vegetation modifications, aquatic vegetation modifications, and hydrologic-geomorphic modifications respectively.

9.4.6.1 Construction, Maintenance, and Operation Activities

Impact mechanisms imposed by marinas/terminals will vary in terms of magnitude and to a certain extent in the frequency of disturbance associated with construction and maintenance.

The potential for injury or mortality from pile driving varies depending on piling size and composition, pile driving methods, and site-specific environmental characteristics such as bathymetry, intervening land masses, and substrate composition.

Construction vessel operation results in increased ambient noise levels in and around the project vicinity, disturbance of substrates from anchors, shading cast by the vessels (if they stay in the same place over a longer period of time), grounding of construction vessels, and operational or accidental discharges. The overall risk of take associated construction vessels is considered moderate because of its limited duration and because of timing restrictions that will limit the duration of effects on many HCP species.

Work area dewatering poses a high risk of take of varying levels of severity depending on habitat and species-specific factors.

Marina/terminal development often involves dredging to establish and maintain approach and navigation channels. Dredging activities are typically temporary to short term in duration, lasting from days to weeks, and recur at interannual to decadal frequencies. Stressors associated with dredging include disturbance and the potential for direct injury or mortality from physical entrainment. Many juvenile and most adult fish are sufficiently mobile to avoid entrainment and injury. In combination with timing restrictions, this will limit exposure so that only moderate risk of take will result from disturbance and temporary or permanent displacement. In contrast, eggs, sessile invertebrates, and demersal or planktonic larvae are vulnerable to entrainment, and timing restrictions may not provide protection for all HCP species in all environments.

Once a marina or terminal is constructed, the operation of the facility and related vessel activities will impose a suite of ongoing impact mechanisms on the aquatic environment. Stressors associated with these impact mechanisms vary in nature and severity, but are similar in that they will be essentially permanent in duration and common to continuous in frequency.

Species occurring in larger rivers, estuaries of large rivers, and the marine environment are more likely to be exposed to larger, higher activity facilities. Species occurring only in lakes or smaller rivers will not receive the same type of exposure, as these environments are more suitable for smaller-scale facilities supporting predominantly recreational uses.

Grounding, anchoring, and prop wash are forms of direct disturbance from vessel activity associated with marinas/terminals. The risk of take for is variable, with likelihood of adverse effects dependent on project-specific considerations. In general, the risk of take from stressors associated with grounding, anchoring, and prop wash is low to moderate for species and life-history stages that do not utilize the affected habitat extensively and are mobile and can avoid the stressor with minor behavioral alteration. Species with less mobile life-history stages that are exposed to this stressor may experience a moderate to high potential for take.

Vessel maintenance and operational discharges may degrade water quality through the introduction of potentially toxic substances. The ratings of species-specific risk of take associated with discharges are based on a combination of the general effects of receptor exposure to toxic substances and the duration and frequency of potential exposure resulting from facility operation. Because the associated stressors are likely to occur at a greater frequency over the long term, vessel discharges are generally associated with a high risk of take.

Facility and vessel operation result in permanent alterations to ambient noise levels at frequencies ranging from intermittent to continuous depending on the type of facility involved. The risks of take associated with ongoing noise are greater than that associated with construction because of the longer duration and higher frequency of exposure. Shipping or ferry terminals frequented by large vessels capable of producing high levels of underwater noise would be expected to produce a higher level of risk of take, as the potential for auditory masking, hearing threshold effects, and avoidance behavior are greater. Large marinas frequented by numerous commercial and recreational vessels may also produce considerable ambient noise and related risk of take that are comparable to or exceed smaller shipping terminals. Smaller marinas serving recreational vessels may produce less pronounced effects on ambient noise levels overall, with seasonal peaks in activity punctuated by long periods of less activity. Under a "worst-case scenario" altered ambient noise equates to a high risk of take, with likelihood of adverse effects dependent on project-specific considerations.

Marinas/terminals alter ambient light conditions in the nearshore environment. Daytime shading produced by overwater structures and vessels and nighttime lighting both modify the ambient light environment, forcing behavioral adaptations by fish. Structural shading can also lead to alteration of submerged aquatic vegetation, producing additional impact mechanisms. In marine environments, the diffusion of small bubbles from cavitation and prop wash can also modify the ambient light environment by diminishing light penetration, again resulting in additional impact mechanisms caused by alteration of submerged aquatic vegetation. Risk of take associated with altered ambient light varies by species and environment. Fish species that are exposed to this stressor, particularly in lacustrine and nearshore marine environments, may alter their behavior, with variable effects on survival, growth, and fitness. The sensitivity of invertebrates to altered ambient light conditions is less understood. In a "worst-case scenario" species are generally likely to experience a high risk of take because the habitat alterations associated with altered ambient light conditions, and resulting effects on survival, growth, and fitness, are long term in nature.

9.4.6.1 Hydraulic and Geomorphic Modifications

Marina/terminal projects modify hydraulic and geomorphic conditions, resulting in the imposition of several impact mechanisms and related stressors. Risk of take resulting from these impact mechanisms is strongly linked to species-specific dependence on the nearshore environment.

Alterations to wave energy, current velocities, nearshore circulation patterns, sediment supply and transport, altered substrate composition, and altered freshwater inputs caused by marinas/terminals all equate to a high risk of take in both marine and lacustrine environments.

Altered shoreline and bluff stability can be variable depending on specific design elements of the marina/terminal. Most marinas/terminals armor the shoreline, increasing shoreline and bluff stability locally, as well as possibly decreased stability elsewhere through alteration of wave energy. In other cases, unmitigated vegetation alteration may decrease shoreline stability. Changes are associated with a high risk of take.

Permitting of marinas/terminals implicitly authorizes the development of some amount of associated impervious surface. Runoff from these surfaces that is not detained or infiltrated will alter peak flows entering the receiving body and, in theory, could result in localized alteration of hydraulic conditions. In reality, however, the larger water bodies suitable for marina and terminal development are insensitive to the relatively small amount of impervious surface area created by this type of facility. These types of water bodies are considered flow control exempt by the Washington State Departments of Ecology and Transportation (WSDOT 2006c), meaning that for regulatory purposes they are considered insensitive to the effects of flow perturbation imposed by impervious surfaces. Flow effects in flow control exempt water bodies are not considered a source of take for ESA consultation purposes (WSDOT 2006d), meaning that the risk of take is considered insignificant and discountable. Therefore, the risk of take resulting from this stressor will be insignificant.

9.4.6.2 Riparian Vegetation Modifications

The nature and scale of riparian vegetation modifications depend on the size and design of the individual project in combination with site-specific conditions. The majority of riparian vegetation modifications associated with marinas and terminals involves permanent conversion to an armored shoreline using bulkheads or some similar structure.

In marine and lacustrine environments, risk of take from marina/terminal projects' effects on riparian vegetation is strongly linked to species-specific dependence on the nearshore environment and riparian functions. For many species, the risk of take associated with marine riparian impact mechanisms is unknown because scientific understanding of the related ecological processes is in its infancy, and the extent to which many marine or anadromous species rely on the nearshore environment during their life history is unclear.

In riverine environments, marina/terminal projects are limited to the lower reaches of larger river systems in virtually all circumstances, meaning that they are located in a position on the river continuum where allochthonous inputs from riparian vegetation are less important to overall food web productivity. The loss of allochthonous production from riparian vegetation modification at the scale of a typical terminal or marina project is likely to have an insignificant effect on food web productivity and foraging opportunities. In a worst-case scenario, a large marina shipping terminal project could alter a large amount of riparian area, leading to a localized reduction in allochthonous inputs in a relatively enclosed circulation environment. However, these effects are not expected to be significant relative to the broader effects on habitat suitability imposed by the activity.

If riparian vegetation is removed and not replaced with armoring, bank stability may decrease. Such changes are associated with a high risk of take.

9.4.6.3 Aquatic Vegetation Modifications

Both the construction and operation of marinas/terminals can result in aquatic vegetation modifications. During construction, vegetation in the structural footprint of the project will be eradicated or buried by the placement of fill or structural material. After construction, vegetation growth and persistence can be affected by changes in ambient light conditions caused by vessel and structural shading.

In marine environments, changes in wave energy, flow and/or current velocities, and substrate composition can also lead to alteration of the vegetation community, shifts in the food web, and altered habitat complexity. This results in a high risk of take. In riverine and lacustrine environments, aquatic vegetation is a relatively minor component of the habitat structure. Aside from native emergent vegetation confined to a relatively narrow range of depths, the majority of aquatic vegetation species in lake systems are invasive exotic species. But for species that depend on native aquatic vegetation, alterations caused by marinas/terminals result in a high risk of take.

9.4.6.1 Water Quality Modifications

The size of the facility, its operation and maintenance requirements, and the intensity of vessel traffic determine stressor intensity from water quality modifications.

Increased suspended solids from marina or terminal operations and maintenance result in a low risk of take for motile species and a moderate risk of take for non-motile species.

Dredging, grounding and anchoring, pile driving, and other activities can result in the resuspension of previously contaminated sediments. Depending on the nature and concentration of the contaminant and the duration of exposure, the toxic substances in contaminated sediments can cause a range of adverse effects in exposed species. These effects may include physiological injury and/or contaminant bioaccumulation leading to decreased survival, growth, and fitness. This presents a moderate risk of take.

Construction and operation of marinas/terminals presents multiple pathways for the introduction of a range of toxic substances to the aquatic environment. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects in exposed species. In extreme cases, these effects can include direct mortality. More commonly, chronic, low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation leading to decreased survival, growth, and fitness. This presents a moderate risk of take.

Dissolved oxygen levels could be reduced near marinas/terminals. In extreme circumstances, nutrient-rich discharge from shipboard sanitary systems or ballast water may cause temporary or short-term decreases in dissolved oxygen levels. A large decrease in aquatic vegetation may limit photosynthetic production of oxygen, but the likelihood of this effect substantially decreasing dissolved oxygen levels is quite limited. In general, the likelihood of decreased dissolved oxygen occurring as a direct or indirect result of marina/terminal development is low. Fish species that are highly mobile will generally be able to avoid adverse effects, translating to a low risk of take. Sessile invertebrates and less mobile life-history stages could experience direct mortality as a result of low levels of dissolved oxygen, equating to moderate or even high risk of take depending on species-specific life history. However, because of the low likelihood of occurrence, the overall risk of take associated with this stressor is considered low for all species.

Curing concrete in water or operational discharges and accidental spills of acidic or caustic materials may lead to alteration of normal pH levels. In general, alterations to pH will be limited to low-frequency events that are temporary to short term in duration. Fish species that are highly mobile will generally be able to avoid adverse effects through behavioral avoidance, translating to a low risk of take. In contrast, sessile invertebrates and less mobile life-history stages could experience direct mortality as a result of exposure, equating to high risk of take.

Creosote-treated wood is expected to present a moderate risk of take, in part because it is no longer frequently installed. WACs 220-110-060 and -224 prohibit the use of creosote- and pentachlorophenol-treated wood in lakes; therefore, exposure to this stressor will not occur in most lacustrine habitats for new projects. There is some uncertainty about potential exposure in lacustrine environments because the applicability of this statute to reservoirs (which are functionally similar to lacustrine environments) is not clear. ACZA and CCA type C treated wood is expected to present a high risk of take.

Marinas/terminals have some amount of associated impervious surface. Runoff from these surfaces that is not detained and treated or infiltrated transports toxic substances and contaminated sediments to the aquatic environment, creating a new permanent stressor of temporary to short-term duration, occurring at common frequencies with seasonal peaks. Depending on the nature and concentration of the transported contaminants, stormwater-related toxic substances can cause a range of adverse effects on exposed species. In extreme cases, these effects can include direct mortality. More commonly, chronic, low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation leading to decreased survival, growth, and fitness. This presents a high risk of take.

Table 9-35. Species- and habitat-specific risk of take for mechanisms of impact associated with marina/terminal construction and maintenance activities.

	Pile Driving			Construction Vessel Operation			Channel/Work Area Dewatering			Navigation/ Maintenance Dredging			
Species	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Comments
Chinook salmon	H	H	H	M	M	M	H	H	H	M	M	M	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors.
Coho salmon	H	H	H	M	M	M	H	H	H	M	L	M	This species has a complex and variable life history depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure.
Chum salmon	H	H	I	M	M	I	H	H	I	M	M	I	Chum salmon in Washington State do not use lacustrine habitats suitable for marina/terminal development. Therefore, stressor exposure will not occur in lacustrine environments. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River) and may therefore be subject to temporary effects of maintenance dredging on spawning habitat, as well as juvenile and adult exposure during migration. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from marina/terminal development in these environments.
Pink salmon	H	H	I	M	M	I	H	H	I	M	M	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in lacustrine environments. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river systems potentially suitable for marina/terminal development. As such, this species may potentially experience related stressor exposure.
Sockeye salmon	H	H	H	M	M	M	H	H	H	M	L	M	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for marina development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence.
Steelhead	H	L	H	L	L	M	H	L	H	M	L	M	Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the risk of take associated with activities in these habitat types is unknown.
Coastal cutthroat trout	H	H	H	L	L	M	H	H	H	M	M	M	This species is prevalent in estuaries and large rivers and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for marina/terminal development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure.
Westslope cutthroat trout	I	N	H	I	N	M	I	N	H	I	N	M	These species occur primarily in coldwater streams and small to medium sized rivers, and in lakes. Occurrence in larger rivers suitable for marina/terminal development is unlikely.
Redband trout	I	N	H	I	N	M	I	N	H	I	N	M	
Bull trout	H	H	H	M	M	M	H	H	H	M	M	M	Spawning by these species occurs in habitats that are generally unsuitable for marina/terminal development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory corridors, as well as riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults.
Dolly Varden	H	H	H	M	M	M	H	H	H	M	M	M	
Pygmy whitefish	N	N	H	N	N	M	N	N	H	N	N	M	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for marina/terminal development; therefore, stressor exposure will only occur in lacustrine environments.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for marina/terminal development.
Margined sculpin	N	N	•	N	N	•	N	N	•	N	N	•	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for marina/terminal development.
Mountain sucker	H	N	H	L	N	M	H	N	H	M	N	M	This species is commonly found in large rivers and lakes suitable for marina and potentially terminal development.
Lake chub	I	N	I	N	N	I	I	N	I	I	N	I	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens Counties that are generally unsuitable for marina/terminal development. Therefore, the likelihood of stressor exposure is considered discountable.
Leopard dace	H	N	H	M	N	M	H	N	H	M	N	M	This species has been reported in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for marina/terminal development at sensitive life-history stages, including egg incubation.
Umatilla dace	H	N	H	M	N	M	H	N	H	M	N	M	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake Rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for marina/terminal development at sensitive life-history stages, including egg incubation.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for marina development. Therefore, marina/terminal development will have no-effect on this species.
River lamprey	H	H	H	?	?	?	H	H	H	H	M	H	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. In their saltwater phase, river lamprey remain close to shore for periods of 10 to 16 weeks from spring through fall. They are therefore susceptible to dredging and dewatering impacts. Sound sensitivity of primitive fishes such as lamprey is currently a data gap, so the potential effects of this stressor are unknown. This life-history makes this species particularly

Species	Pile Driving			Construction Vessel Operation			Channel/Work Area Dewatering			Navigation/ Maintenance Dredging			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
													sensitive to dredging and dewatering in lakes and rivers, as well as in the nearshore marine environment. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	H	H	H	?	I	?	H	I	H	H	L	H	Pacific lamprey are anadromous with migratory corridors that cross estuaries and mainstems of larger river systems suitable for marina/terminal development. Ammocoetes burrow into riverine sediments to rear for extended periods. They are therefore susceptible to dredging and dewatering impacts in freshwater environments. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6 to 40 months. Sound sensitivity of primitive fishes such as lamprey is currently a data gap, so the potential effects of this stressor are unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Green sturgeon	N	H	N	N	?	N	N	L	N	N	M	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, some populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore sensitive to dewatering, dredging, and other direct impacts. Green sturgeon fisheries occur in the Columbia River below Bonneville Dam, Willapa Bay, and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Sturgeon are wide ranging in marine waters. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats.
White sturgeon	H	H	H	L	?	L	H	L	H	M	M	M	
Longfin smelt	H	H	N	M	M	N	H	H	H	H	H	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems, which are preferred areas for marina development. Demersal adhesive eggs are vulnerable to short-term dewatering and dredging impacts. Adults, eggs, and larvae are vulnerable to impacts from pile driving. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Mature juveniles and adults are found in offshore environments.
Eulachon	H	H	N	M	M	N	H	H	N	H	H	N	
Pacific sand lance	N	H	N	N	M	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high.
Surf smelt	N	H	N	N	M	N	N	H	N	N	H	N	
Pacific herring	N	H	N	N	M	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure is high.
Lingcod	N	H	N	N	M	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are subject to impacts from dewatering and dredging. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 to 150 m) and therefore have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Low mobility larvae settle in nearshore areas, increasing risk of take from dredging and dewatering.
Pacific hake	N	H	N	N	M	N	N	H	N	N	H	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae and smaller juveniles of all three species may be at higher risk of dredging entrainment due to limited mobility.
Pacific cod	N	H	N	N	M	N	N	H	N	N	H	N	
Walleye pollock	N	H	N	N	M	N	N	H	N	N	H	N	
Brown rockfish	N	H	N	N	M	N	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Planktonic larvae and demersal juveniles are particularly vulnerable to dewatering and fish handling, as well as dredging activities.
Copper rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Black rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
China rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Redstripe rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	M	N	N	H	N	N	H	N	
Olympia oyster	N	?	N	N	M	N	N	H	N	N	H	N	Olympia oysters are found in intertidal and subtidal environments potentially subject to dredging and dewatering impacts. Exposure to these impact mechanisms could lead to direct mortality or injury. Sound sensitivity of this species is currently a data gap, and the effects of related stressors are unknown.
Northern abalone	N	?	N	N	M	N	N	H	N	N	H	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) depth. This distribution

Species	Pile Driving			Construction Vessel Operation			Channel/Work Area Dewatering			Navigation/ Maintenance Dredging			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
													increases risk of stressor exposure and potential for take from dewatering and dredging activities. The effect of underwater noise on mollusks is a data gap so the potential for take related to this stressor is unknown.
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	This species inhabits a narrow band of upper littoral zone habitat above MHHW and is therefore not exposed to stressors resulting from in-water construction activities.
Giant Columbia River limpet	?	N	N	?	N	?	?	N	N	I	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for marina/terminal development. As such, there is essentially no likelihood of stressor exposure and therefore no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. This distribution likely limits exposure to navigational dredging. Exposure to work area dewatering is possible, but sensitivity to this stressor is a data gap so the potential for take is unknown. The effects of underwater noise on mollusks are currently a data gap so the potential for take related to this stressor is unknown.
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	
California floater (mussel)	?	N	?	?	N	?	H	N	H	H	N	H	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake River and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats suitable for marina/terminal development. This distribution presents risk of stressor exposure and potential for take, particularly from dewatering and dredging activities. Exposure to dewatering can cause mortality in both species. The effect of underwater noise on mollusks is currently a data gap so the potential for take related to this stressor is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Western ridged mussel	?	N	N	?	N	N	H	N	N	H	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-36. Species- and habitat-specific risk of take for mechanisms of impact associated with marina/terminal facility operation and vessel activities.

Species	Grounding, Anchoring, and/or Prop Wash			Vessel Maintenance and Operational Discharges			Increased or Altered Ambient Noise Levels			Ambient Light Modifications			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	H	H	H	H	H	H	L	L	L	L	H	H	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Ambient light modification is a recognized stressor for this species in nearshore marine and lacustrine environments.
Coho salmon	H	H	H	H	M	H	L	L	L	L	?	H	This species has a complex and variable life history depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Ambient light modification is a likely source of risk of take for this species in nearshore lacustrine environments and may also pose risk of take in marine environments. However, as juvenile coho salmon are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain.
Chum salmon	H	H	I	H	H	I	L	L	I	L	H	I	Chum salmon in Washington State do not use lacustrine habitats suitable for marina/terminal development. Therefore, stressor exposure will not occur in lacustrine environments. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River) and may therefore be subject to facility and vessel operational effects dredging on spawning habitat, in addition to juvenile and adult exposure during migration. Juvenile chum salmon are dependent on nearshore marine habitats, and are therefore subject to stressor exposure from marina/terminal development in these environments. Ambient light modification is a recognized stressor for this species, resulting in a moderate risk of take from chronic behavioral alteration.
Pink salmon	H	H	I	H	H	I	L	L	I	L	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in lacustrine environments. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river systems potentially suitable for marina/terminal development. As such, this species may potentially experience related stressor exposure.
Sockeye salmon	H	L	H	H	H	H	L	L	L	L	?	H	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for marina development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence. Ambient light modification is a likely source of risk of take for this species in nearshore lacustrine environments and may also pose risk of take in marine environments. However, as juvenile sockeye salmon are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain.
Steelhead	H	L	H	H	H	H	L	L	L	L	?	H	Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the risk of take associated with activities in these habitat types is unknown. Ambient light modification is a potential source of take for this species in nearshore lacustrine environments and may also pose risk of take in marine environments. However, as juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain.
Coastal cutthroat trout	H	H	H	H	H	H	L	L	L	H	H	H	This species is prevalent in estuaries and large rivers and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for marina/terminal development. Migratory behavior and residence timing are variable. Spawning and juvenile rearing activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Ambient light modification is a likely source of risk of take for this species in nearshore marine environments, based on similar sensitivity of other salmonid species in these environments.
Westslope cutthroat trout	I	N	H	I	N	H	I	N	L	I	N	H	These species occur primarily in coldwater streams, small to medium-sized rivers, and in lakes. Occurrence in larger rivers suitable for marina/terminal development is unlikely. Ambient light modification is a likely source of risk of take for these species in nearshore lacustrine environments, based on similar sensitivity of other salmonid species in these environments.
Redband trout	I	N	H	I	N	H	I	N	L	I	N	H	
Bull trout	H	H	H	H	H	H	L	L	L	?	?	?	Spawning by these species occurs in habitats that are generally unsuitable for marina/terminal development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory corridors, as well as riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults. Sensitivity to this stressor in lacustrine environments is a data gap. However, char in lakes are typically found in deeper water.
Dolly Varden	H	H	H	H	H	H	L	L	L	?	?	?	
Pygmy whitefish	N	N	H	N	N	H	N	N	L	N	N	?	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for marina/terminal development; therefore, stressor exposure will only occur in lacustrine environments.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for marina/terminal development.

Species	Grounding, Anchoring, and/or Prop Wash			Vessel Maintenance and Operational Discharges			Increased or Altered Ambient Noise Levels			Ambient Light Modifications			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for marina/terminal development.
Mountain sucker	H	N	H	H	N	H	M	N	M	?	N	?	This species is commonly found in large rivers and lakes suitable for marina and potentially terminal development. Sensitivity of this species to ambient light modification is currently a data gap; therefore, the potential for take resulting from this stressor is unknown.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens Counties that are generally unsuitable for marina/terminal development.
Leopard dace	H	N	H	H	N	H	M	N	M	?	N	?	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for marina/terminal development at sensitive life-history stages, including egg incubation.
Umatilla dace	H	N	H	H	N	H	M	N	M	?	N	?	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake Rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for marina/terminal development at sensitive life-history stages, including egg incubation.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for marina development. Therefore, marina/terminal development will have no-effect on this species.
River lamprey	H	H	H	H	H	H	?	?	?	?	I	?	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. In their saltwater phase, river lamprey remain close to shore for periods of 10 to 16 weeks from spring through fall. They are therefore susceptible to injury or mortality from grounding, anchoring, and prop wash. Sensitivity to ambient noise and light modification in lamprey is currently a data gap so the potential effects of these stressors are unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	H	I	H	H	L	H	?	?	?	?	?	?	Pacific lamprey are anadromous, with migratory corridors that cross estuaries and mainstems of larger river systems suitable for marina/terminal development. Ammocoetes burrow into riverine sediments to rear for extended periods. They are therefore susceptible to injury or mortality from grounding, anchoring, and prop wash. Sensitivity to ambient noise and light modification in lamprey is currently a data gap so the potential effects of these stressors are unknown. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6 to 40 months. Sound sensitivity of primitive fishes such as lamprey is currently a data gap so the potential effects of this stressor are unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Green sturgeon	N	L	N	N	H	N	N	?	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, some populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore sensitive to grounding, anchoring, and other direct impacts. Individuals are occasionally caught incidentally in small coastal bays and the Puget Sound. Sturgeon are wide ranging in marine waters. Green sturgeon fisheries occur in the Columbia River below Bonneville Dam, Willapa Bay, and Grays Harbor. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats. Sensitivity to ambient noise and light modification in primitive fishes like sturgeon is currently a data gap so the potential effects of these stressors are unknown.
White sturgeon	H	L	H	H	H	H	?	?	?	?	?	?	
Longfin smelt	H	L	H	H	H	H	L	H	H	?	?	?	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems, which are preferred areas for marina development. Adults, eggs, and larvae are vulnerable to impacts from vessel anchoring and grounding, and other operational impacts. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Mature juveniles and adults occupy offshore environments and are therefore not at risk of take from marina/terminal related impact mechanisms until they return to nearshore and riverine environments for spawning. Smelt sensitivity to ambient light modification is a data gap; therefore, the risk of take from this stressor is uncertain.
Eulachon	H	L	N	H	H	N	L	H	N	?	?	N	
Pacific sand lance	N	H	N	N	H	N	N	L	N	N	?	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Smelt and sand lance sensitivity to ambient light modification is a data gap; therefore, the risk of take resulting from this stressor is uncertain.
Surf smelt	N	H	N	N	H	N	N	L	N	N	?	N	
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure is high. Sensitivity of spawning habitat and incubating eggs from vessel grounding, anchoring, and prop wash is high, meaning that there is high risk of take resulting from this stressor. Herring display demonstrable sensitivity to vessel noise, meaning that risk of take from ambient noise modification is likely.
Lingcod	N	H	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are subject to impacts from vessel ground, anchoring and prop wash, and other operational impact mechanisms. Adults may occur anywhere from the intertidal zone to depths

Species	Grounding, Anchoring, and/or Prop Wash			Vessel Maintenance and Operational Discharges			Increased or Altered Ambient Noise Levels			Ambient Light Modifications			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
													of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 to 150 m) and therefore have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Low-mobility larvae settle in nearshore areas, increasing risk of take from grounding, anchoring, and prop wash. Lingcod sensitivity to ambient light modification is a data gap, meaning the risk of take resulting from this stressor is unknown.
Pacific hake	N	H	N	N	H	N	N	H	N	N	?	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Low-mobility larvae settle in nearshore areas, increasing risk of take from grounding, anchoring, and prop wash. The sensitivity of these species to ambient light modification is a data gap, meaning the risk of take resulting from this stressor is unknown.
Pacific cod	N	H	N	N	H	N	N	H	N	N	?	N	
Walleye pollock	N	H	N	N	H	N	N	H	N	N	?	N	
Brown rockfish	N	H	N	N	H	N	N	H	N	N	?	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Demersal larvae are vulnerable to injury and mortality from grounding, anchoring, and prop wash. The sensitivity of these species to ambient light modification is a data gap, meaning the risk of take resulting from this stressor is unknown.
Copper rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Widow rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Black rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
China rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Canary rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	?	N	
Olympia oyster	N	H	N	N	H	N	N	H	N	N	?	N	This species occurs commonly in shallow nearshore habitats. This distribution increases risk of stressor exposure and potential for take from grounding and anchoring activities. The effect of underwater noise and ambient light modification on mollusks is a data gap; therefore, the related risk of take is unknown.
Northern abalone	N	H	N	N	H	N	N	H	N	N	?	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) depth. This distribution increases risk of stressor exposure and potential for take from grounding and anchoring activities. The effect of underwater noise and ambient light modification on mollusks is a data gap; therefore, the related risk of take is unknown.
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	This species inhabits a narrow band of upper littoral zone habitat above MHHW and is therefore not exposed to stressors resulting from marina/terminal operation.
Giant Columbia River limpet	H	N	N	H	N	N	?	N	?	?	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for marina/terminal development. As such, there is essentially no likelihood of stressor exposure and therefore no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. This distribution may increase exposure to anchoring and prop wash. The effect of ambient light and noise modification on mollusks is currently a data gap so the potential for take related to this stressor is unknown.
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	
California floater (mussel)	H	N	H	H	N	H	?	N	?	?	N	?	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake River and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats suitable for marina/terminal development. This distribution presents risk of stressor exposure and potential for take, particularly from grounding, anchoring, and prop wash. The effect of ambient light and noise modification on mollusks is currently a data gap so the potential for take related to this stressor is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Western ridged mussel	H	N	N	H	N	N	?	N	N	?	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.

Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-37. Species- and habitat-specific risk of take for mechanisms of impact associated with water quality modifications caused by marinas/terminals.

	Increased Suspended Solids			Resuspension of Contaminated Sediments			Introduction of Toxic Substances			Altered pH			Altered Dissolved Oxygen			Use of Creosote-Treated Wood			Use of ACZA and CCA Type C Treated Wood			Increased Stormwater and Nonpoint Source Pollution			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	M	M	M	M	M	M	M	M	M	L	L	L	L	L	L	M	H	N	M	M	M	M	M	M	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors.
Coho salmon	M	M	M	M	M	M	M	M	M	L	L	L	L	L	L	M	H	N	M	M	M	M	M	M	This species has a complex and variable life history depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure.
Chum salmon	M	M	I	M	M	I	M	M	I	L	L	I	L	L	I	M	H	N	M	M	I	M	M	I	Chum salmon in Washington State do not use lacustrine habitats suitable for marina/terminal development. Therefore, stressor exposure will not occur in lacustrine environments. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River). As such, in addition to migratory juveniles and adults, spawning habitats may therefore be exposed to water quality related stressors. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from marina/terminal development in these environments.
Pink salmon	M	M	I	M	M	I	M	M	I	L	L	I	L	L	I	M	H	N	M	M	I	M	M	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in lacustrine environments. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river systems potentially suitable for marina/terminal development. As such, this species may potentially experience related stressor exposure.
Sockeye salmon	M	M	M	M	M	M	M	M	M	L	L	M	L	L	L	M	H	N	M	M	M	M	M	M	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for marina development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence.
Steelhead	M	M	M	L	M	M	M	M	M	L	L	L	L	L	L	M	M	N	M	M	M	M	M	M	Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the risk of take associated with activities in these habitat types is unknown. As juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain.
Coastal cutthroat trout	M	M	M	M	M	M	M	M	M	L	L	L	L	L	L	M	M	N	M	M	M	M	M	M	This species is prevalent in estuaries and large rivers and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for marina/terminal development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure.
Westslope cutthroat trout	I	N	M	I	N	M	I	N	M	I	N	L	I	N	L	I	N	N	I	N	L	I	N	M	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Occurrence in larger rivers suitable for marina/terminal development is unlikely.
Redband trout	I	N	M	I	N	M	I	N	M	I	N	L	I	N	L	I	N	N	I	N	L	I	N	M	
Bull trout	M	M	M	L	M	M	M	M	M	L	L	L	L	L	L	M	M	N	M	M	M	M	M	M	Spawning by these species occurs in habitats that are generally unsuitable for marina/terminal development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory

Species	Increased Suspended Solids			Resuspension of Contaminated Sediments			Introduction of Toxic Substances			Altered pH			Altered Dissolved Oxygen			Use of Creosote-Treated Wood			Use of ACZA and CCA Type C Treated Wood			Increased Stormwater and Nonpoint Source Pollution			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Dolly Varden	M	M	M	L	M	M	M	M	M	L	L	L	L	L	L	M	M	N	M	M	M	M	M	M	corridors, as well as in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults. However, char in lakes are typically found in deeper water.
Pygmy whitefish	N	N	M	N	N	M	N	N	M	N	N	L	N	N	L	N	N	N	N	N	M	N	N	M	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for marina/terminal development; therefore, stressor exposure will only occur in lacustrine environments.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for marina/terminal development.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for marina/terminal development.
Mountain sucker	M	N	M	M	N	M	M	N	M	M	N	M	L	N	L	M	N	N	M	N	N	M	N	M	This species is commonly found in large rivers and lakes suitable for marina and potentially terminal development.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens Counties that are generally unsuitable for marina/terminal development.
Leopard dace	M	N	M	M	N	M	M	N	M	M	N	M	L	N	L	M	N	N	M	N	M	M	N	M	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for marina/terminal development at sensitive life-history stages, including egg incubation.
Umatilla dace	M	N	M	M	N	M	M	N	M	M	N	M	L	N	L	M	N	N	M	N	M	M	N	M	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake Rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for marina/terminal development at sensitive life-history stages, including egg incubation.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for marina development. Therefore, marina/terminal development will have no-effect on this species.
River lamprey	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	N	M	M	M	M	M	M	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. This nonmobile life-history stage is more susceptible to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. In their saltwater phase, river lamprey remain close to shore for periods of 10 to 16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	M	L	M	M	L	M	M	L	M	M	L	M	M	L	M	M	L	N	M	L	M	M	L	M	Pacific lamprey are anadromous with migratory corridors that cross estuaries and mainstems of larger river systems suitable for marina/terminal development. Ammocoetes burrow into riverine sediments to rear for extended periods. This nonmobile life-history stage is more susceptible to acute transient water quality impacts, such as reduced dissolved oxygen or altered pH. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6 to 40 months and are therefore less likely to be exposed to project-related stressors in the nearshore marine environment. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.

Species	Increased Suspended Solids			Resuspension of Contaminated Sediments			Introduction of Toxic Substances			Altered pH			Altered Dissolved Oxygen			Use of Creosote-Treated Wood			Use of ACZA and CCA Type C Treated Wood			Increased Stormwater and Nonpoint Source Pollution			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Green sturgeon	N	L	N	N	M	N	N	M	N	N	L	N	N	L	N	N	M	N	N	M	N	N	M	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered to be anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore potentially exposed to water quality related impact mechanisms from marinas/terminals. Their relative lack of mobility increases sensitivity to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Sturgeon are wide ranging in marine waters. Green sturgeon fisheries occur in Columbia River below Bonneville Dam, Willapa Bay, and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats.
White sturgeon	H	L	H	M	M	M	M	M	M	H	L	H	H	L	H	M	M	M	M	M	M	M	M	M	
Longfin smelt	H	L	H	M	M	M	M	M	M	H	L	H	M	L	M	M	M	M	M	M	N	M	M	M	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems, which are preferred areas for marina development. Demersal adhesive eggs are vulnerable to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors.
Eulachon	H	L	N	M	M	N	M	M	N	M	L	N	M	L	N	M	M	N	M	M	N	M	M	N	
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	M	N	N	M	N	N	M	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Larvae of both species disperse in nearshore waters for early rearing. Because they are essentially planktonic, larvae are vulnerable to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Larvae are also visual feeders. Increased turbidity can reduce foraging success, leading to decreased growth and productivity.
Surf smelt	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	M	N	N	M	N	N	M	N	
Pacific herring	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure is high. Larvae disperse in nearshore waters for early rearing. Because they are essentially planktonic, larvae are vulnerable to acute transient water quality impacts such as reduced dissolved or altered pH. Larvae are also visual feeders. Increased turbidity can reduce foraging success, leading to decreased growth and productivity.
Lingcod	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	M	N	N	M	N	N	M	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to water quality related impact mechanisms from marinas/terminals. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 to 150 m) and therefore have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Larvae are also visual feeders. Increased turbidity can reduce foraging success, leading to decreased growth and productivity.

Species	**Increased Suspended Solids**			**Resuspension of Contaminated Sediments**			**Introduction of Toxic Substances**			**Altered pH**			**Altered Dissolved Oxygen**			**Use of Creosote-Treated Wood**			**Use of ACZA and CCA Type C Treated Wood**			**Increased Stormwater and Nonpoint Source Pollution**			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Pacific hake	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	M	N	N	M	N	N	M	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Because they are demersal and relatively immobile, larvae are vulnerable to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Larvae are visual feeders. Increased turbidity can reduce foraging success, leading to decreased growth and productivity.
Pacific cod	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	M	N	N	M	N	N	M	N	
Walleye pollock	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	M	N	N	M	N	N	M	N	
Brown rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Because they are demersal and relatively immobile, larvae are vulnerable to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Larvae are also visual feeders. Increased turbidity can reduce foraging success, leading to decreased growth and productivity.
Copper rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Widow rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Black rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
China rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Canary rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	M	N	
Olympia oyster	N	H	N	N	H	N	N	H	N	N	M	N	N	L	N	N	M	N	N	M	N	N	M	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile at all live-history stages, it is vulnerable to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Increased turbidity may reduce foraging success of this filter feeding species, leading to decreased growth and productivity.
Northern abalone	N	H	N	N	H	N	N	H	N	N	M	N	N	L	N	N	M	N	N	M	N	N	M	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) depth. Because this species is sessile at all life-history stages, it is vulnerable to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Increased turbidity may affect algal growth, reducing available forage.
Newcomb's littorine snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species inhabits a narrow band of upper littoral zone habitat above MHHW and is therefore not directly exposed to water quality-related stressors resulting from marina/terminal operation.
Giant Columbia River limpet	M	N	M	M	N	M	H	N	H	H	N	H	L	N	L	M	N	M	M	N	M	M	N	M	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for marina/terminal development. As such, there is essentially no likelihood of stressor exposure and therefore no potential for take resulting

Species	Increased Suspended Solids			Resuspension of Contaminated Sediments			Introduction of Toxic Substances			Altered pH			Altered Dissolved Oxygen			Use of Creosote-Treated Wood			Use of ACZA and CCA Type C Treated Wood			Increased Stormwater and Nonpoint Source Pollution			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. The distribution of this species presents the possibility of stressor exposure. However, because it lives in lotic habitats, water quality effects will by nature be transitory, meaning that exposure to acute events will be temporary. Their sessile nature makes behavioral avoidance impossible, however, increasing the duration of acute exposure and potential for physiological injury.
California floater (mussel)	H	N	H	M	N	M	H	N	H	H	N	H	M	N	M	H	N	N	H	N	M	H	N	H	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake River and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats suitable for marina/terminal development. Because they occur primarily in lotic habitats, water quality effects will by nature be transitory, meaning that exposure to acute events will be temporary. Their sessile nature makes behavioral avoidance impossible, however, increasing the duration of acute exposure and potential for physiological injury. Toxicity of copper, ammonia, and chlorine has been demonstrated in closely related species. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Western ridged mussel	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	H	N	N	H	N	N	M	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-38. Species- and habitat-specific risk of take for mechanisms of impact associated with riparian vegetation modifications caused by marina/terminal development.

Species	Altered Riparian Shading and Ambient Air Temperature Regime			Altered Stream Bank and Shoreline Stability			Altered Allochthonous Inputs			Altered Habitat Complexity			Altered Surface Water-Groundwater Exchange, or Freshwater Input			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	I	H	H	H	H	H	I	H	H	H	H	H	H	H	H	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Marinas/terminals in riverine environments will be developed in habitats where modification of riparian vegetation will have little influence on water temperatures or food web productivity as a whole. Modification of riparian habitat will also most likely involve permanent conversion to an armored state, meaning that effects on bank stability will be minimal.
Coho salmon	I	H	H	H	H	H	I	H	H	H	H	H	H	H	H	This species has a complex and variable life history depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure.
Chum salmon	I	H	I	H	H	I	I	H	I	H	H	I	H	H	I	Chum salmon in Washington State do not use lacustrine habitats suitable for marina/terminal development. Therefore, stressor exposure will not occur in lacustrine environments. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River) and may therefore be subject to temporary effects of riparian modification on spawning habitat, in addition to juvenile and adult exposure during migration. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from marina/terminal development in these environments.
Pink salmon	I	H	I	H	H	I	I	H	I	H	H	I	H	H	I	Pink salmon in Washington State do not utilize lacustrine habitats. Therefore, stressor exposure will not occur in lacustrine environments. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river systems potentially suitable for marina/terminal development. As such, this species may potentially experience related stressor exposure.
Sockeye salmon	I	H	H	H	H	H	I	H	H	H	H	H	H	H	H	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for marina development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence.
Steelhead	I	?	H	H	?	H	I	?	H	H	?	H	H	L	H	Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the risk of take associated with activities in these habitat types is unknown.
Coastal cutthroat trout	I	H	H	H	H	H	I	H	H	H	H	H	H	H	H	This species is prevalent in estuaries and large rivers and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for marina/terminal development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure.
Westslope cutthroat trout	I	N	H	I	N	H	I	N	H	I	N	H	I	N	H	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Occurrence in larger rivers suitable for marina/terminal development is unlikely.
Redband trout	I	N	H	I	N	H	I	N	H	I	N	H	I	N	H	
Bull trout	I	H	H	H	H	H	I	H	H	H	H	H	H	?	H	These species spawn in habitats that are generally unsuitable for marina/terminal development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory corridors, as well as in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults.
Dolly Varden	I	H	H	H	H	H	I	H	H	H	H	H	H	?	H	
Pygmy whitefish	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for marina/terminal development; therefore, stressor exposure will only occur in lacustrine environments.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for marina/terminal development.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for marina/terminal development.
Mountain sucker	I	N	H	I	N	H	H	N	H	H	N	H	H	N	H	This species is commonly found in large rivers and lakes suitable for marina and potentially terminal development.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens Counties

Species	Altered Riparian Shading and Ambient Air Temperature Regime			Altered Stream Bank and Shoreline Stability			Altered Allochthonous Inputs			Altered Habitat Complexity			Altered Surface Water-Groundwater Exchange, or Freshwater Input			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
																that are generally unsuitable for marina/terminal development.
Leopard dace	I	N	H	H	N	H	I	N	H	H	N	H	?	N	?	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for marina/terminal development.
Umatilla dace	I	N	H	H	N	H	I	N	H	H	N	H	?	N	?	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake Rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for marina/terminal development.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for marina development. Therefore, marina/terminal development will have no-effect on this species.
River lamprey	I	?	?	H	H	H	I	?	?	H	H	H	?	?	?	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. They are therefore susceptible to changes in stream bank stability with the potential to affect bottom sediments. In their saltwater phase, river lamprey remain close to shore for periods of 10 to 16 weeks from spring through fall. The dependence of this species on riparian vegetation and freshwater inflow in lacustrine and marine environments is a data gap, so the potential risk of take associated with these stressors is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	I	I	?	H	I	H	I	I	H	H	I	H	?	?	?	Pacific lamprey are anadromous, with migratory corridors that cross estuaries and mainstems of larger river systems suitable for marina/terminal development. Ammocoetes burrow into riverine sediments to rear for extended periods. They are therefore susceptible to changes in stream bank stability with the potential to affect bottom sediments. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6 to 40 months. The dependence of this species on riparian vegetation and freshwater inflow in lacustrine and marine environments is a data gap, so the potential risk of take associated with these stressors is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Green sturgeon	N	I	N	N	H	N	N	I	N	N	L	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. Green sturgeon fisheries occur in Columbia River below Bonneville Dam, Willapa Bay, and Grays Harbor. Sturgeon eggs are demersal and adhesive. These life-history stages are therefore potentially exposed to riparian modification impact mechanisms. Adults are occasionally caught incidentally in small coastal bays and Puget Sound. Sturgeon are wide ranging in marine waters. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats.
White sturgeon	I	I	H	H	H	H	I	I	H	H	L	H	H	?	H	
Longfin smelt	I	I	N	H	I	H	I	I	H	H	I	H	?	?	?	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems, which are preferred areas for marina development. Demersal adhesive eggs are vulnerable to short-term dewatering and dredging impacts. Adults, eggs, and larvae may be exposed to riparian modification impact mechanisms in marine and riverine environments. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing, particularly suspended sediments from decreased bank stability, which may decrease foraging success for these visual feeders. Dependence on freshwater inflow is a data gap, so the related risk of take resulting from this stressor is unknown. Mature juveniles and adults are found in offshore environments and are not exposed to these stressors.
Eulachon	I	I	N	H	I	N	I	I	N	H	I	N	?	?	N	
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Egg survival is demonstrably affected by modification of riparian shading and ambient temperature regime, and by alteration of freshwater inflow. Larvae and juveniles are highly dependent on the habitat complexity and productivity of the nearshore environment for rearing, and changes in habitat complexity and allochthonous inputs affecting food web productivity are likely to affect growth and fitness. Changes in stream bank and shoreline stability may affect the suitability of spawning substrate, and increased suspended sediments may affect foraging success of visual feeding larvae.
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	

Species	Altered Riparian Shading and Ambient Air Temperature Regime			Altered Stream Bank and Shoreline Stability			Altered Allochthonous Inputs			Altered Habitat Complexity			Altered Surface Water-Groundwater Exchange, or Freshwater Input			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure is high. Larvae and juveniles are highly dependent on the habitat complexity and productivity of the nearshore environment for rearing, and changes in habitat complexity and allochthonous inputs affecting food web productivity are likely to affect growth and fitness. Changes in stream bank and shoreline stability may increase suspended sediments, affecting egg incubation and the foraging success of visual feeding larvae.
Lingcod	N	I	N	N	H	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are therefore potentially exposed to riparian modification impact mechanisms. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 to 150 m) and therefore have less exposure potential. Larvae and juveniles are highly dependent on the habitat complexity and productivity of the nearshore environment for rearing, and changes in habitat complexity and allochthonous inputs affecting food web productivity are likely to affect growth and fitness. Changes in stream bank and shoreline stability may increase suspended sediments, affecting egg incubation and the foraging success of visual feeding larvae.
Pacific hake	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	Hake, Pacific cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae and juveniles may experience stressor exposure. Larvae and juveniles are highly dependent on the habitat complexity and productivity of the nearshore environment for rearing, and changes in habitat complexity and allochthonous inputs affecting food web productivity are likely to affect growth and fitness. Changes in stream bank and shoreline stability may increase suspended sediments, affecting egg incubation and the foraging success of visual feeding larvae.
Pacific cod	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Walleye pollock	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Brown rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Larvae and juveniles are highly dependent on the habitat complexity and productivity of the nearshore environment for rearing, and changes in habitat complexity and allochthonous inputs affecting food web productivity are likely to affect growth and fitness. Changes in shoreline stability may increase suspended sediments, affecting egg incubation and the foraging success of visual feeding larvae.
Copper rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Greenstriped rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Widow rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Yellowtail rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Quillback rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Black rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
China rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Tiger rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Bocaccio rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Canary rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Redstripe rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Yelloweye rockfish	N	I	N	N	H	N	N	H	N	N	H	N	N	?	N	
Olympia oyster	N	L	N	N	H	N	N	L	N	N	H	N	N	H	N	While the influence of shading and buffer on lower intertidal zone is limited, Olympia oyster growth and fitness may benefit from thermal extremes in some cases. In contrast, sedimentation demonstrably affects survival, growth and fitness in this species. Dependence on allochthonous inputs is currently a data gap. Habitat complexity and groundwater inflow affect habitat suitability for larval settlement and development, as well as juvenile and adult survival.
Northern abalone	N	I	N	N	H	N	N	I	N	N	I	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) depth. Subtidal distribution generally limits exposure to riparian modification impact mechanisms and related stressors. For example, riparian shading will have effectively no influence on this species. In contrast, sedimentation resulting from decreased shoreline stability may extend into the subtidal zone, affecting foraging success. Exposure to other stressors resulting from these impact mechanisms is insignificant, given the subtidal distribution of this species. Therefore, these impact mechanisms are expected to have no effect.
Newcomb’s littorine snail	N	H	N	N	H	N	N	N	N	N	H	N	N	?	N	This species inhabits a narrow band of upper littoral zone vegetation above MHHW and is therefore directly exposed to riparian

Species	**Altered Riparian Shading and Ambient Air Temperature Regime**			**Altered Stream Bank and Shoreline Stability**			**Altered Allochthonous Inputs**			**Altered Habitat Complexity**			**Altered Surface Water-Groundwater Exchange, or Freshwater Input**			**Comments**
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
																vegetation modification where it is known to occur. Because this species is largely terrestrial, it is unaffected by alteration in allochthonous inputs and groundwater inputs.
Giant Columbia River limpet	I	N	I	I	N	I	L	N	L	H	N	H	?	N	?	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for marina/terminal development. As such, there is essentially no likelihood of stressor exposure and therefore no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. Dependence of this species on groundwater inputs is a data gap, so the risk of take from this stressor is unknown.
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
California floater (mussel)	I	N	I	H	N	H	I	N	I	H	N	H	H	N	H	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake River and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats suitable for marina/terminal development. The localized influence of riparian vegetation on temperature conditions in these larger river systems is limited. Dependence of these species on groundwater inputs is a data gap, so the risk of take from this stressor is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Western ridged mussel	I	N	N	H	N	N	I	N	N	H	N	N	H	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-39. Species- and habitat-specific risk of take for mechanisms of impact associated with aquatic vegetation modifications caused by marinas and terminal development and operation.

	Altered Autochthonous Production			Altered Habitat Complexity			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	L	H	H	L	H	H	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors.
Coho salmon	L	H	H	L	H	H	This species has a complex and variable life history depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure.
Chum salmon	I	H	I	I	H	I	Chum salmon in Washington State do not use lacustrine habitats suitable for marina/terminal development. Therefore, stressor exposure will not occur in lacustrine environments. Chum migrate through and in some cases may spawn in the lower reaches of large river environments and may therefore be exposed to aquatic vegetation modification impact mechanisms. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from marina/terminal development in these environments.
Pink salmon	I	H	I	I	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in lacustrine environments. This species is dependent on nearshore marine habitats for juvenile rearing, and migrates through the mainstems and estuaries of larger river systems potentially suitable for marina/terminal development. As such, this species may potentially experience related stressor exposure.
Sockeye salmon	I	?	H	L	?	H	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for marina development. Alteration of lacustrine aquatic vegetation may affect survival, growth and fitness of rearing juveniles. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor.
Steelhead	L	?	H	L	?	H	Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the risk of take associated with activities in these habitat types is unknown.
Coastal cutthroat trout	L	H	H	L	H	H	This species is prevalent in estuaries and large rivers, and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for marina/terminal development. Migratory behavior and residence timing are variable.
Westslope cutthroat trout	I	N	H	I	N	H	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Occurrence in larger rivers suitable for marina/terminal development is unlikely.
Redband trout	I	N	H	I	N	H	
Bull trout	L	H	H	L	H	H	Spawning by these species occurs in habitats that are generally unsuitable for marina/terminal development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory corridors, and in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults. Predominant riverine habitats do not support extensive aquatic vegetation.
Dolly Varden	L	H	H	L	H	H	
Pygmy whitefish	N	N	H	N	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for marina/terminal development, therefore stressor exposure will only occur in lacustrine environments.
Olympic mudminnow	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for marina/terminal development.
Margined sculpin	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for marina/terminal development.
Mountain sucker	M	N	M	H	N	H	This species is commonly found in large rivers and lakes suitable for marina and potentially terminal development.
Lake chub	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens Counties that are generally unsuitable for marina/terminal development.
Leopard dace	H	N	H	H	N	H	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for marina/terminal development.
Umatilla dace	H	N	H	H	N	H	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake Rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for marina/terminal development.
Western brook lamprey	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for marina development. Therefore, marina/terminal development will have no-effect on this species.
River lamprey	?	?	?	?	?	?	Dependence of this species on aquatic vegetation is a data gap; therefore, the risk of take associated with these stressors is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	?	?	?	?	?	?	Dependence of this species on aquatic vegetation is a data gap; therefore, the risk of take associated with these stressors is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Green sturgeon	N	?	N	N	?	N	Dependence on aquatic vegetation in freshwater environments and nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats.
White sturgeon	H	?	H	H	?	H	
Longfin smelt	I	I	H	I	I	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems, which are preferred areas for marina development. This species has limited freshwater residence time and is not dependent on aquatic vegetation during adult, egg, and larval life-history stages. Rearing larvae in nearshore marine areas may be dependent on habitat complexity and food web productivity.
Eulachon	I	I	N	I	I	N	
Pacific sand lance	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for

Species	Altered Autochthonous Production			Altered Habitat Complexity			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Surf smelt	N	H	N	N	H	N	spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Planktonic larvae rear in nearshore areas and are dependent on food web productivity and habitat complexity of these environments.
Pacific herring	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on aquatic vegetation in nearshore habitats for spawning, and egg incubation, meaning that the likelihood of stressor exposure is high. Planktonic larvae rear in nearshore areas and are dependent on food web productivity and habitat complexity of these environments.
Lingcod	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are therefore potentially exposed to aquatic vegetation modification impact mechanisms. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 to 150 m) and therefore have less exposure potential. Planktonic larvae and demersal juveniles rear in nearshore areas and are dependent on food web productivity and habitat complexity of these environments.
Pacific hake	N	H	N	N	H	N	Hake, cod, and Pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Planktonic larvae and demersal juveniles rear in nearshore areas and are dependent on food web productivity and habitat complexity of these environments.
Pacific cod	N	H	N	N	H	N	
Walleye pollock	N	H	N	N	H	N	
Brown rockfish	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Planktonic larvae and demersal juveniles rear in nearshore areas and are dependent on food web productivity and habitat complexity of these environments.
Copper rockfish	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	H	N	
Black rockfish	N	H	N	N	H	N	
China rockfish	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	H	N	
Redstripe rockfish	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	H	N	
Olympia oyster	N	H	N	N	H	N	Alteration of aquatic vegetation may affect the productivity of the nearshore food web, leading to reduced growth and fitness of larval, juvenile and adult Olympia oyster.
Northern abalone	N	?	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) depth. While this species feeds on intertidal and subtidal algal biomass and could be affected by altered autochthonous production, but the level of dependence is a data gap and effects are unknown. Alteration of habitat complexity may alter the suitability and productivity of larval settlement habitat, leading to effects on survival, growth, and fitness of this species.
Newcomb's littorine snail	N	N	N	N	N	N	This species inhabits a narrow band of upper littoral zone habitat above MHHW and is therefore not exposed to stressors resulting from these impact mechanisms.
Giant Columbia River limpet	H	N	H	?	N	?	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for marina/terminal development. As such, there is essentially no likelihood of stressor exposure and therefore no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. The dependence of this species on allochthonous inputs from riparian vegetation is unknown. However, being substrate feeding species dependent on functional nutrient cycling, activities that affect allochthonous production may cause at least some risk of take. The effect of diminished habitat complexity due to aquatic vegetation modification on this species is a data gap; therefore, the associated risk of take is unknown.
Great Columbia River spire snail	N	N	N	N	N	N	
California floater (mussel)	H	N	H	?	N	?	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake River and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats suitable for marina/terminal development. However, being filter feeding species dependent on functional nutrient cycling, activities that affect autochthonous production may cause at least some risk of take. The effect of diminished habitat complexity due to aquatic vegetation modification on this species is a data gap; therefore, the associated risk of take is unknown. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Western ridged mussel	H	N	N	?	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-40. Species- and habitat-specific risk of take for mechanisms of impact associated with hydraulic and geomorphic modifications caused by marinas/terminals.

	Altered Channel Geometry	Altered Flow Velocity	Altered Wave Velocities		Altered Current Velocities		Altered Nearshore Circulation Patterns		Altered Sediment Supply		Altered Substrate Composition			Altered Surface Water Groundwater Exchange, or Freshwater Input			Addition of Impervious Surface			
Species	**Riverine**	**Riverine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Chinook salmon	H	H	H	H	H	H	H	H	H	H	H	H	H	H	?	H	I	I	I	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Migrating adults and migrating and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Coho salmon	H	H	H	H	H	H	H	H	H	H	H	H	H	H	?	H	I	I	I	This species has a complex and variable life history depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for marina/terminal development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Migrating adults and migrating and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Chum salmon	H	H	H	I	H	I	H	I	H	I	H	H	I	H	?	I	I	I	I	Chum salmon in Washington State do not use lacustrine habitats suitable for marina/terminal development. Therefore, stressor exposure will not occur in lacustrine environments. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River) and may therefore be exposed to impact mechanisms from hydraulic and geomorphic modification during spawning as well as during juvenile and adult migration. Juvenile chum salmon are dependent on nearshore marine habitats, and are therefore subject to stressor exposure from marina/terminal development in these environments. Migrating adults and migrating and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Pink salmon	H	H	H	I	H	I	H	I	H	I	H	H	I	H	?	I	I	I	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in lacustrine environments. This species is dependent on nearshore marine habitats for juvenile rearing, and migrates through the mainstems and estuaries of larger river systems potentially suitable for marina/terminal development. As such, this species may potentially experience related stressor exposure. Migrating adults and migrating and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Sockeye salmon	H	H	H	H	H	H	H	H	H	H	H	H	H	H	?	H	I	I	I	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for marina development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor. Migrating adults and migrating and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all

Species	Altered Channel Geometry	Altered Flow Velocity	Altered Wave Velocities		Altered Current Velocities		Altered Nearshore Circulation Patterns		Altered Sediment Supply		Altered Substrate Composition			Altered Surface Water Groundwater Exchange, or Freshwater Input			Addition of Impervious Surface			Comments
	Riverine	Riverine	Marine	Lacustrine	Marine	Lacustrine	Marine	Lacustrine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																				environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Steelhead	H	H	?	H	?	H	?	H	?	H	H	?	H	H	?	H	I	I	I	Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the risk of take associated with activities in these environment types is unknown. Migrating adults and migrating and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Coastal cutthroat trout	H	H	H	H	H	H	H	H	H	H	H	H	H	H	?	H	I	I	I	This species is prevalent in estuaries and large rivers, and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for marina/terminal development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for terminal and marina development; therefore, eggs and alevins will not experience stressor exposure. Migrating adults and migrating and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Westslope cutthroat trout	I	I	N	H	N	H	N	H	N	H	H	N	H	H	N	H	I	N	I	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Occurrence in larger rivers suitable for marina/terminal development is unlikely.
Redband trout	I	I	N	H	N	H	N	H	N	H	H	N	H	H	N	H	I	N	I	These species occur primarily in coldwater streams, small to medium-sized rivers, and in lakes. Occurrence in larger rivers suitable for marina/terminal development is unlikely.
Bull trout	H	H	H	H	H	H	H	H	H	H	H	H	H	H	?	H	I	I	I	Spawning by these species occurs in habitats that are generally unsuitable for marina/terminal development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory corridors, as well as in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults. Migrating adults, migrating and rearing juveniles, and foraging adults in marine habitats are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased juvenile growth and fitness.
Dolly Varden	H	H	H	H	H	H	H	H	H	H	H	H	H	H	?	H	I	I	I	
Pygmy whitefish	N	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	H	N	I	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for marina/terminal development; therefore, stressor exposure will only occur in lacustrine environments. This species is sensitive to alteration of hydraulic and geomorphic conditions in lacustrine environments.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for marina/terminal development.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for marina/terminal development.
Mountain sucker	H	H	N	H	N	H	N	H	N	H	H	N	H	H	N	H	I	N	I	This species is commonly found in large rivers and lakes suitable for marina and potentially terminal development. Adults and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all

Species	Altered Channel Geometry	Altered Flow Velocity	Altered Wave Velocities		Altered Current Velocities		Altered Nearshore Circulation Patterns		Altered Sediment Supply		Altered Substrate Composition			Altered Surface Water Groundwater Exchange, or Freshwater Input			Addition of Impervious Surface			Comments
	Riverine	Riverine	Marine	Lacustrine	Marine	Lacustrine	Marine	Lacustrine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																				environment types, experiencing decreased survival, decreased spawning fitness, and decreased growth and fitness.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens Counties that are generally unsuitable for marina/terminal development.
Leopard dace	H	H	N	H	N	H	N	H	N	H	H	N	H	H	N	H	I	N	I	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for marina/terminal development. Adults and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased growth and fitness.
Umatilla dace	H	H	N	H	N	H	N	H	N	H	H	N	H	H	N	H	I	N	I	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake Rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for marina/terminal development. Adults and rearing juveniles are sensitive to alterations in hydraulic and geomorphic conditions in all environment types, experiencing decreased survival, decreased spawning fitness, and decreased growth and fitness.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for marina development. Therefore, marina/terminal development will have no-effect on this species.
River lamprey	H	H	H	H	H	H	H	H	H	H	H	H	H	H	H	H	I	I	I	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. In their saltwater phase, river lamprey remain close to shore for periods of 10 to 16 weeks from spring through fall. They are therefore susceptible to alteration of riverine, lacustrine, and nearshore marine environments caused by hydraulic and geomorphic modification. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	H	H	L	H	L	H	L	H	L	H	H	L	H	H	L	H	I	I	I	Pacific lamprey are anadromous, with migratory corridors that cross estuaries and mainstems of larger river systems suitable for marina/terminal development. Ammocoetes burrow into riverine sediments to rear for extended periods. They are therefore susceptible to hydraulic and geomorphic modifications in riverine and lacustrine environments. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6 to 40 months. While some exposure to nearshore habitat conditions altered by hydraulic and geomorphic modification is possible, the dependence on these habitats is low so the associated risk of take is also believed to be low. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.

Species	Altered Channel Geometry	Altered Flow Velocity	Altered Wave Velocities		Altered Current Velocities		Altered Nearshore Circulation Patterns		Altered Sediment Supply		Altered Substrate Composition			Altered Surface Water Groundwater Exchange, or Freshwater Input			Addition of Impervious Surface			Comments
	Riverine	Riverine	Marine	Lacustrine	Marine	Lacustrine	Marine	Lacustrine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Green sturgeon	N	N	?	N	?	N	?	N	?	N	N	?	N	N	?	N	N	I	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore sensitive to altered riverine and lacustrine habitat conditions caused by hydraulic and geomorphic modification. Green sturgeon fisheries occur in Columbia River below Bonneville Dam, Willapa Bay, and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and Puget Sound. Sturgeon are wide ranging in marine waters. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in these habitats.
White sturgeon	H	H	?	H	?	H	?	H	?	H	H	?	H	H	?	H	I	I	I	
Longfin smelt	H	H	H	H	H	H	H	H	I	H	H	I	N	?	?	?	I	I	N	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems, which are preferred areas for marina development. Demersal adhesive eggs are sensitive to altered riverine habitat conditions caused by hydraulic and geomorphic modification. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. These life-history requirements translate to risk of take resulting from riverine and marine habitat alteration caused by hydraulic and geomorphic modification. Mature juveniles and adults are found in offshore environments.
Eulachon	H	H	H	N	H	N	H	N	I	N	H	I	N	?	?	N	I	I	N	
Pacific sand lance	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Dependence on shoreline and nearshore habitats for spawning and rearing means that these species are sensitive to habitat alterations caused by hydraulic and geomorphic modification.
Surf smelt	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Pacific herring	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure is high. Dependence on nearshore habitats for spawning and rearing means that this species is sensitive to habitat alterations caused by hydraulic and geomorphic modification.
Lingcod	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are therefore exposed to impact mechanisms associated with hydraulic and geomorphic modification. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 to 150 m) and therefore have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Dependence on nearshore habitats for rearing means that this species is sensitive to habitat alterations caused by hydraulic and geomorphic modification.

	Altered Channel Geometry	**Altered Flow Velocity**	**Altered Wave Velocities**		**Altered Current Velocities**		**Altered Nearshore Circulation Patterns**		**Altered Sediment Supply**		**Altered Substrate Composition**			**Altered Surface Water Groundwater Exchange, or Freshwater Input**			**Addition of Impervious Surface**			
Species	**Riverine**	**Riverine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Pacific hake	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Dependence on nearshore habitats for rearing means that these species are sensitive to habitat alterations caused by hydraulic and geomorphic modification.
Pacific cod	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Walleye pollock	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Brown rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Dependence on nearshore habitats for rearing means that these species are sensitive to habitat alterations caused by hydraulic and geomorphic modification.
Copper rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Greenstriped rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Widow rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Yellowtail rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Quillback rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Black rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
China rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Tiger rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Bocaccio rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Canary rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Redstripe rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Yelloweye rockfish	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	
Olympia oyster	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	Dependence on nearshore habitats throughout this species' life history means that it is sensitive to habitat alterations caused by hydraulic and geomorphic modification. These impact mechanisms are likely to result in effects on survival, growth, and productivity across veliger, juvenile, and adult life-history stages.
Northern abalone	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) depth. Dependence on nearshore habitats throughout much of this species' life history means that it is sensitive to habitat alterations caused by hydraulic and geomorphic modification.
Newcomb's littorine snail	N	N	H	N	H	N	H	N	H	N	N	H	N	N	H	N	N	I	N	This species inhabits a narrow band of upper littoral zone habitat above MHHW. Hydraulic and geomorphic modification of nearshore habitats can result in alteration of upper intertidal habitat characteristics, leading to indirect risk of take on this species.
Giant Columbia River limpet	H	H	N	H	N	H	N	H	N	H	H	N	H	?	N	?	I	N	I	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for marina/terminal development. As such, there is essentially no likelihood of stressor exposure

Species	Altered Channel Geometry	Altered Flow Velocity	Altered Wave Velocities		Altered Current Velocities		Altered Nearshore Circulation Patterns		Altered Sediment Supply		Altered Substrate Composition			Altered Surface Water Groundwater Exchange, or Freshwater Input			Addition of Impervious Surface			Comments
	Riverine	**Riverine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	and therefore no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. This species is sensitive to alterations in hydraulic and geomorphic conditions in riverine and lacustrine habitats, leading to decreased survival, growth, and fitness.
California floater (mussel)	H	H	N	H	N	H	N	H	N	H	H	N	H	H	N	H	I	N	I	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake River and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats suitable for marina/terminal development. These species are sensitive to alterations in hydraulic and geomorphic conditions in riverine and lacustrine habitats, leading to decreased survival, growth, and fitness. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Western ridged mussel	H	H	N	N	N	N	N	N	N	N	M	N	N	M	N	N	I	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

9.4.7 Shoreline Modifications

Shoreline modifications include jetties, breakwaters, groins, and bank barbs.

Tables 9-41 through 9-43 identify the risk of take for each of the 52 HCP species by impact mechanism and environment type. The summary risk of take presented in the narrative and the matrices represents the greatest overall risk of take for the category.

9.4.7.1 Construction and Maintenance Activities

Construction and maintenance of shoreline modification projects involve a diverse array of activities, including driving pilings, placement of materials, construction vessel operation, maintenance dredging, and work area dewatering. The majority of these activities are temporary in nature, lasting from a few days to several weeks. Some mechanisms may produce a high risk of individual take due to their intensity, while others may result in a low risk of take due to their limited magnitude and duration.

The risk of take associated with construction activity is dependent on the scale of the project and the type of environment where it is implemented. For example, a large jetty project at a river mouth may have a significant impact on the nearshore and estuarine environment, but the risk of take associated with the structure would be limited to those species and individual life-history stages that occur in those habitat types. Breakwaters are typically developed in marine and lacustrine habitats and would only occur in the largest of rivers, specifically the Columbia River, with sufficient open water to allow the formation of wind-driven waves, or supporting vessel traffic producing wakes large enough to cause bank erosion. The distribution of these project types limits the potential for related stressor exposure to the species and life-history stages that occur in these environments. For example, bull trout would be exposed to jetty and breakwater related stressors during subadult and adult life-history stages, but the egg, alevin, and juvenile stages would not, as they occur in upriver environments that are not suitable for jetties or breakwaters.

In contrast, groins and bank barbs are commonly placed in marine, riverine, and lacustrine environments, and are often used in smaller streams and rivers, so the range of species and life-history stages exposed to stressors from these project types is much broader. Species such as the western brook lamprey and the Columbia River spire snail are limited in distribution to free-flowing rivers and streams inappropriate for jetties and breakwaters, so these species will be exposed to only to stressors related to groins or bank barbs.

9.4.7.1.1 Noise

Jetties, breakwaters, or groins may incorporate structural pilings. Project scale and location determine the piling material types and placement methods. The potential for injury or mortality from the noise generated by pile driving varies depending on piling size and composition, pile driving methods, and site-specific environmental characteristics such as bathymetry, intervening land masses, and substrate composition. Applying a worst-case scenario perspective, pile driving must be associated with a high risk of take due to the potential for injury or mortality for the majority of HCP species experiencing possible exposure. Equipment operation and materials placement results in increased ambient noise levels in and around the project vicinity, resulting in a moderate risk of take due to their short-term duration.

9.4.7.1.2 Channel/Work Area Dewatering

Channel or work area dewatering is often required for groin and bank barb construction. Dewatering is not commonly used in jetty and breakwater construction due to the large scale of these structures and the environments where they are typically constructed. Channel and work area dewatering poses a high risk of take.

9.4.7.1.3 Construction and Maintenance Dredging

Development of shoreline modification structures may involve dredging during construction and maintenance. Groins and bank barb structures often extend below the substrate surface, requiring dredging to excavate the foundation (although these activities are usually conducted within a dewatered exclusion area). Dredging activities are typically temporary to short-term in duration, lasting from days to weeks, with maintenance recurring at interannual to decadal frequencies. Stressors associated with dredging include direct disturbance and the potential for injury or mortality from physical entrainment. The potential for take associated with this stressor varies by species and life-history stage, ranging from a moderate risk of take (e.g., from limited exposure to disturbance and displacement) to a high risk (e.g., exposure to entrainment resulting in injury and/or mortality). Many juvenile and most adult fish are sufficiently motile to avoid entrainment and injury. In combination with timing restrictions, this will limit exposure so that only a low to moderate risk of take results from activity-related disturbance and temporary displacement. Fish eggs and demersal larvae and the HCP invertebrate species are effectively nonmotile and therefore are vulnerable to entrainment. These life stages and species would face a high risk of take.

Dredging causes increased suspended solids (turbidity), altered substrate composition, and changes in bathymetry that alter habitat suitability and potentially alter wave energy, current, and circulation patterns. In specific cases, dredging may also result in the resuspension of contaminated sediments. These stressors are associated with a moderate to high risk of take.

9.4.7.2 Hydraulic and Geomorphic Modifications

Shoreline modifications create structures perpendicular to the direction of water flow, inherently involving modification of the hydraulic and geomorphic conditions in the project vicinity, and the subsequent imposition of a number of impact mechanisms and related stressors on the aquatic environment. The nature and scale of hydraulic and geomorphic modification, and the associated risk of take, vary by project type and scale. Jetties, by virtue of their size and location, typically have the most significant effects. Impacts from breakwaters may manifest differently than those from jetties, because breakwaters are typically oriented parallel to the shore, while jetties are oriented perpendicularly. In the absence of other shoreline structures, breakwaters are less prone to interrupt alongshore drift, thereby having lesser effects on substrate conditions. Groins and bank barbs have effects similar to jetties, but more limited in scale because these structures are usually far less extensive and intrusive.

Jetties, breakwaters, and groins and bank barbs all could be built in marine and lacustrine environments, although few lakes in Washington are suitable locations for jetties. The associated risk of take is strongly linked to the potential distribution of the structures, the size and scale of the project, and species-specific dependence on the nearshore environment. Changes to wave energy, current velocities, circulation patterns, sediment supply, and substrate composition caused by shoreline modifications are associated with a high risk of take for species that are dependent on nearshore marine or lacustrine habitats during some phase of their life history. Alteration of groundwater inputs would be expected to cause a corresponding alteration in the distribution of desirable habitat features and availability for species dependent on the nearshore environment. This equates to a moderate to high risk of take for species with demonstrable dependence on these habitats because freshwater inputs will likely still occur; however, they will be modified, resulting in a potential reduction in suitable habitat area, which in turn will lead to reduced survival, growth, and fitness.

Breakwaters and groins and bank barbs could be located in riverine environments. Jetties, which are typically placed at river mouths where they enter the ocean or large lakes, are considered not to affect the riverine environment, with no risk of take. Breakwaters are most likely to be placed in large rivers where wind-driven waves or boat wakes are sufficiently large to warrant these structures to protect marinas, boat launches, or other infrastructure. Groins and bank barbs are used in a broad array of river environments, from small mountain streams to large river mainstems. Therefore, the range of HCP species and life-history stages that could be exposed to breakwater-related stressors in riverine environments is limited, whereas effectively all riverine species and life-history stages could be exposed to stressors resulting from groins and bank barbs. Changes to channel geometry, flow conditions, substrate composition, and groundwater-surface water exchange from shoreline modifications equate to a high risk of take for species with exposure to these impact mechanisms.

9.4.7.1 Ecosystem Fragmentation

9.4.7.1.1 Habitat Loss and Fragmentation

In marine and lacustrine environments, jetties and groins and bank barbs (depending on their scale and location) present significant potential for habitat loss and fragmentation. The magnitude of fragmentation and the related risk of take are driven by the scale of the project in question, with larger projects having the most potential for adverse effects.

By design, jetties are intended to accelerate the flow of water from river mouths into open ocean waters, thereby keeping shallow bar areas from forming. As a consequence, they can alter bathymetric, and circulation patterns in the nearshore environment. In estuaries, they can also alter salinity and tidal exchange. These changes can alter habitat conditions and potentially eliminate desirable habitat types. Habitats in the physical footprint of the structure are permanently lost as a result of construction. Due to their perpendicular orientation to the shore, jetties and groins and bank barbs present a physical barrier to the migration of many species. For example, many salmonid species typically migrate as juveniles in shallow water along the shoreline. These structures effectively force these individuals to migrate around the structure into deeper water where predation risk and foraging opportunities are less favorable to survival. Because jetties are typically larger in size, these effects are more pronounced. Because breakwaters are constructed offshore, typically parallel to the shoreline, they present less of a barrier to migration overall.

9.4.7.1.2 Altered Wave Energy, Current, and Circulation Patterns

In marine and lacustrine environments, jetties, breakwaters, groins, and bank barbs can alter wave energy, current, and circulation patterns in the nearshore and offshore environment. These effects can in many cases result in habitat fragmentation through various pathways. Alteration of these habitat characteristics may render productive habitats less suitable for a given species or, in the case of organisms with a planktonic life-history stage, may hinder the dispersal and retention of eggs and larvae in areas suitable for rearing. Collectively, this can result in take through long-term effects on survival, growth, and fitness of affected populations, which equates to a high risk of take for exposed species.

In riverine environments, particularly those with higher velocity flows, groins and bank barbs often cause localized changes in river geomorphology. In addition to the loss of habitat area within the structural footprint, these structures can concentrate and accelerate river flows, causing localized channel downcutting that can lead to a lowering of mean water surface and the consequent fragmentation of side channels and other floodplain habitats. This hydraulic and geomorphic effect is most prevalent in higher gradient reaches with sufficient velocity to transport bedload, and less prevalent in the lower gradient depositional reaches of large river mainstems. Therefore, this effect is not as likely to occur as a result of breakwater development. Many HCP species depend on floodplain habitats during one or more life-history stages, or depend on host species with these requirements. Loss of access to these habitat types represents take.

9.4.7.1.3 Loss of LWD Recruitment

In marine and lacustrine environments, placement of shoreline modification structures can alter the transport of drift wood to beaches. Many large jetties and breakwaters are intentionally cleared of driftwood accumulations for maintenance purposes, which may further limit the potential for recruitment to nearby beach areas. Groins and bank barbs may similarly alter the transport of woody material along the shoreline.

In riverine environments, placement of shoreline modifications, particularly groins and bank barbs, can alter the transport of LWD, limiting recruitment to downstream environments. The magnitude of this effect is expected to be less pronounced with breakwaters due to their orientation parallel to flow in riverine environments, as well as their typical location in higher order mainstem reaches.

Shorelines with limited LWD recruitment potential due to natural conditions or existing riparian vegetation modifications may become increasingly starved of LWD.

9.4.7.2 Riparian Vegetation Modifications

The development of shoreline modification projects in many cases involves the modification of riparian vegetation in the project area. Because jetties and groins and bank barbs are most typically oriented perpendicular to the shoreline, the extent of riparian impacts during construction and the amount of habitat permanently modified will be relatively minor in comparison to activities such as bank protection.

Breakwaters are not expected to result in any riparian vegetation related stressors, as these structures have no onshore component intersecting the riparian environment. Breakwaters are constructed primarily from barges or floating platforms accessed from established landings. Therefore, effectively no riparian vegetation modification is associated with breakwaters, and no risk of take is expected.

Since jetties are not built in rivers, they do not affect riverine riparian vegetation. In marine and lacustrine environments, the onshore component of jetties results in both short-term and effectively permanent modification of riparian vegetation. Because the onshore component of jetties is relatively small in comparison to the overall footprint of the structure, and the majority of these structures are away from shore and oriented perpendicular to the shoreline, the overall magnitude of riparian vegetation modification in most cases will be relatively limited. The related risk of take is expected to be low for most species due to the limited area affected.

Groins and bank barbs can occur in marine, lacustrine, or riverine environments, and so can affect riparian vegetation in each of these environments. The riparian footprint is typically limited, as groins and bank barbs are oriented perpendicular to the shoreline. However, because a groin or bank barb project often incorporates a series of several structures, the resulting short-term to intermediate-term construction impacts can be fairly extensive, affecting a larger riparian footprint. While the risk of take from groins and bank barbs resulting from riparian vegetation modification is low, the number of individuals affected may potentially be larger in cases where the affected riparian footprint is more extensive.

9.4.7.2.1 Altered Riparian Shading and Altered Ambient Air Temperature Regime

In marine and lacustrine environments, the risk of take from altered riparian shading is low. The perpendicular orientation of jetties, groins, and bank barbs to the shoreline reduces the effects of riparian vegetation modifications on shading. Jetties have no associated risk of take in riverine environments, because jetties are not built in rivers.

Breakwaters have no onshore components and therefore have no effects on riparian conditions, and they will impose no related risk of take from modifications to shading.

In larger river systems, altered temperatures due to changes in riparian vegetation may not be measurable, and the resulting risk of take discountable. In smaller streams, stream temperature effects related to groins and bank barbs may influence local habitat suitability and by, extension affect the survival, growth, and fitness of exposed species and life-history stages.

9.4.7.2.2 Altered Shoreline, Bluff, and Streambank Stability

Depending on site-specific conditions, modifications of marine and lacustrine riparian vegetation can lead to physical alteration of the shoreline and bluff instability. In the context of shoreline modification projects, this effect is expected to be small because the onshore footprint of these structures is limited. In addition, the structure itself will stabilize the shoreline where vegetation has been removed. However, unmitigated vegetation alteration may lead to localized decreases in shoreline stability and cyclical erosion. Where this impact mechanism occurs, it would be expected to alter shoreline habitat conditions and habitat suitability for species dependent on the nearshore environment during some portion of their life history. This equates to a low risk of take for species with demonstrable dependence on these habitats because the reduction in suitable habitat area caused by these impact mechanisms will lead to reduced survival, growth, and fitness.

No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component.

In riverine environments, groins and bank barbs are typically intended to increase local bank stability. In the worst-case scenario, however, riparian vegetation modification associated with a permitted project could result in decreased stream bank and shoreline stability, as well as increased erosion and turbidity. These effects are localized and predominant during seasonal high-flow conditions. The risk of take associated with this stressor varies depending on species-specific sensitivity to increased turbidity. In general, more motile fish species experience only temporary behavioral alteration and low risk of take. In contrast, less motile fish life-history stages or sessile invertebrates could experience a high risk of take from decreased survival due to substrate sedimentation and smothering, as well as decreased growth and fitness due to the effects of high turbidity on foraging success.

9.4.7.2.3 Altered Allochthonous Inputs

Because the footprint of jetties, groins, and bank barbs within marine and lacustrine riparian areas is limited, the extent of alterations to allochthonous inputs is likely to be low. This equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component.

In riverine environments, the impact from alterations to allochthonous inputs varies depending on the scale of the groin or barb project and its position in the watershed. Allochthonous inputs are more important to food web productivity in small streams, and less important in large rivers. A groin project near the mouth of a large river will produce lower-magnitude stressors related to allochthonous inputs than a series of bank barbs in a small, higher elevation stream. In smaller streams, a localized reduction in food web productivity might result, leading to decreased foraging opportunities, decreased overall habitat suitability, and decreased growth and fitness. This equates to a high risk of take for a range of HCP species that are dependent on riverine rearing conditions due to the long-term nature of the effect.

9.4.7.2.4 Altered Habitat Complexity

Because the footprint of jetties, groins, and bank barbs within the marine and lacustrine riparian area is limited, the extent of effects related to altered habitat complexity is likely to be low. This equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component.

In riverine environments, fish species that are dependent on habitats potentially affected by altered habitat complexity from groin and bank barb development are likely to experience decreased spawning success and/or decreased survival, growth, and fitness due to an overall reduction in suitable habitat area. This equates to a high risk of take, which applies broadly across all species exposed to the stressor.

9.4.7.2.5 Altered Freshwater Inputs

Because the footprint of jetties, groins, and bank barbs within the riparian area of marine, lacustrine, and riverine environments is limited, the extent of alterations to freshwater input due to removing riparian vegetation is likely to be low. This equates to a low risk of take for species with demonstrable dependence on these habitats. No associated risk of take is anticipated for breakwaters because these structures do not have an onshore component.

9.4.7.3 Aquatic Vegetation Modifications

Shoreline modification projects can result in aquatic vegetation modification through the alteration or elimination of vegetation in the construction footprint, as well as the subsequent effects of the structure on hydraulic and geomorphic conditions. During construction, vegetation in the structural footprint of the project can be eradicated or buried by the placement of fill or structural material. After construction, changes in wave energy, circulation patterns, flow and/or current velocities, and substrate composition can also alter the vegetation community. The nature and scale of aquatic vegetation modification are dependent on the size and design of the individual project in combination with site-specific conditions.

Submerged aquatic vegetation (including eelgrass, kelp, and other forms of marine algae) is an important component of the marine littoral ecosystem relied upon by many species during critical life-history stages.

Aquatic vegetation is a relatively minor component of the ecological structure of riverine and lacustrine systems in Washington State. Aside from native emergent vegetation confined to a relatively narrow range of depths, the majority of aquatic vegetation species in rivers and lakes are invasive species. Thus, the risk of take resulting from modifying freshwater aquatic vegetation is relatively minor in comparison to the marine environment. In riverine systems, protected slow-water areas created by groins and bank barbs may increase suitable habitat for emergent vegetation.

Alteration of aquatic vegetation imposes impact mechanisms on the nearshore environment in the form of changes in autochthonous production and altered habitat complexity.

9.4.7.3.1 Altered Autochthonous Production

Alteration of marine littoral vegetation caused by shoreline development projects may in some cases lead to localized shifts in food web productivity, possibly affecting foraging opportunities for dependent species and life-history stages. This equates to a high risk of take resulting from decreased growth and fitness.

Modification of the submerged aquatic vegetation community in lakes and rivers can lead to decreased primary and secondary productivity, which in turn may affect overall food web productivity in the nearshore environment. In systems where the aquatic vegetation community is an important component of food web productivity, this can lead to a high risk of take through indirect effects on foraging success, growth, and fitness of species and life-history stages that depend on forage in the nearshore environment.

9.4.7.3.2 Altered Habitat Complexity

In marine environments, alterations of the submerged aquatic vegetation community through reduction in aerial extent or conversion to other habitat types (e.g., conversion of eelgrass habitat to algae and kelp) can reduce the productivity of these habitats for dependent life-history stages. This equates to a high risk of take for species dependent on these habitats through reduced survival, spawning success, or growth and fitness.

Submerged aquatic vegetation provides habitat structure in lacustrine and riverine environments, creating vertical dimension and overhead cover. Alteration of habitat complexity can decrease the availability of suitable rearing habitat for species and life-history stages dependent on the nearshore environment, leading to increased predation risk and increased competition for suitable space, leading to effects on survival, growth, and fitness. This equates to a high risk of take for species dependent on aquatic vegetation functions in these environments.

9.4.7.4 Water Quality Modifications

The size of the shoreline modification structure, its construction and maintenance requirements, and the level of associated development and activity determine the extent of water quality modifications. To assess the risk of take associated with these facilities, a “worst-case scenario” approach is taken, with consideration of the scale of the structure and related water quality effect that a given species is likely exposed to in each environment.

9.4.7.4.1 Altered Temperature

Shoreline modifications have the potential to alter temperature conditions through the hydraulic and geomorphic mechanisms they impose. Shoreline modification structures can alter waves, currents, and circulation patterns in marine and lacustrine environments, leading to increased stratification. In riverine environments, groins and bank barbs can slow water flows in the lee of the structures, creating slow water areas prone to stratification and elevated temperature conditions.

These effects can be magnified when stratified areas experience decreased shading due to modification of shading riparian vegetation, which may occur in association with jetties, groins, and bank barbs. However, because these structures are typically oriented perpendicular to the shoreline and their onshore footprint is small, the extent of vegetation modification is usually limited, and these effects are small.

Modification of temperature conditions can change the suitability of nearshore habitats. This may in turn affect the survival, growth, and fitness of HCP species that use the affected habitats. Because these effects are essentially permanent, they must be associated with a high risk of take.

9.4.7.4.2 Suspended Solids and Turbidity

Increased suspended solids can result from several different impact mechanisms. The severity of this stressor varies depending on its magnitude, duration, and frequency, as well as the sensitivity of the species and life-history stage exposed.

9.4.7.4.3 Dissolved Oxygen

There are limited pathways through which shoreline modification projects can lead to alterations in surface water dissolved oxygen levels that are not implicitly addressed by other impact mechanisms. A primary area of concern related to the effects of shoreline modifications in marine and lacustrine environments is their potential to alter wave energy, current, and circulation patterns sufficiently to change stratification, isolating biochemical oxygen demand (BOD) and contributing to eutrophication. In extreme circumstances, this could lead to eutrophication-driven DO depletion in affected habitats. This effect equates to a high risk of take from changes in DO conditions in these environment types, due to the effectively permanent nature of the change in habitat conditions that shoreline modifications impose. These effects would not be anticipated in riverine environments due to the continuous, unidirectional flow path imposed by riverine environments.

Other potential causes of altered DO conditions include inputs of nutrient-rich discharge from construction vessel sanitary systems or ballast water that could cause temporary or short-term decreases in dissolved oxygen levels. A large decrease in aquatic vegetation may limit photosynthetic production of oxygen, but the likelihood of this effect substantially decreasing dissolved oxygen levels is quite limited. In general, the likelihood of this stressor occurring as a direct or indirect result of a shoreline modification project is low.

9.4.7.4.4 Nutrient and Pollutant Loading

Shoreline modification projects present multiple pathways for the introduction of a range of toxic substances to the aquatic environment, primarily through construction activities and, in some cases, the use of treated wood materials in the structure. Shoreline modification projects may also indirectly encourage pollutant and nutrient loading by supporting the development of additional infrastructure. Depending on the nature and concentration of the contaminant, toxic substance exposure can cause a range of adverse effects in exposed species. In extreme cases, these effects can include direct mortality (e.g., exposure of immobile rockfish larvae in the demersal microlayer). More commonly, chronic, low-level exposure to a variety of contaminants is likely to cause physiological injury and/or contaminant bioaccumulation, leading to decreased survival, growth, and fitness. This presents a moderate risk of take to species potentially exposed to this stressor.

9.4.7.4.5 Altered pH Levels

There are limited pathways through which shoreline modification projects can lead to alterations in surface water pH. A primary pathway is the in-water curing of concrete and discharge of concrete leachate to surface waters. Operational discharges and accidental spills of acidic or caustic materials may also lead to the alteration of normal pH levels. In general, this stressor is limited to low-frequency events that are temporary to short-term in duration. Fish species that are highly motile are generally able to avoid adverse effects through behavioral avoidance, equating to a low risk of take. In contrast, sessile invertebrates and less motile life-history stages could experience direct mortality as a result of exposure, equating to a high risk of take depending on species-specific life history.

9.4.7.4.6 Treated Wood Pollution

Creosote-treated wood was often used historically in shoreline modification projects and other structures in marine and freshwater environments. This substance is still permitted in some circumstances. Creosote is a wood preservative with a complex formula composed of more than 150 toxic chemical substances. The Hydraulic Code prohibits use of creosote- and pentachlorophenol-treated wood in lakes; therefore, exposure to this stressor exposure will not occur in most lacustrine habitats. There is some uncertainty about potential exposure in lacustrine environments because the applicability of this statute to reservoirs is not clear.

Prohibitions on the use of creosote, pentachlorophenol, and other wood preservatives have prompted the development of alternatives. ACZA and CCA type C are alternative wood preservatives that are less toxic than prohibited materials but are still effective against undesirable invertebrates. These substances, which slowly leach out of treated wood over time, are toxic to other forms of aquatic life than the intended target species and also have the potential to bioaccumulate.

These substances are expected to produce a moderate risk of take for species potentially exposed to this stressor. It is worthwhile to note, however, that this treated wood poses greater potential for chronic exposure as leaching of toxics occurs over extended periods.

Table 9-41. Species- and habitat-specific risk of take for mechanisms of impacts associated with jetties.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	This species has a complex and variable life history depending on race. In general, Chinook salmon occur in marine and lacustrine habitats suitable for jetty development and are thereby potentially exposed to stressors resulting from related impact mechanisms. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine or lacustrine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Coho salmon	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	This species has a complex and variable life history depending on race. In general, coho salmon occur in lacustrine and nearshore marine habitats suitable for jetty development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine or lacustrine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Chum salmon	N	H	I	N	H	I	N	L	I	N	H	I	N	H	I	N	H	I	Chum salmon in Washington State do not use lacustrine habitats suitable for jetty development. Therefore, likelihood of stressor exposure in lacustrine environments is considered discountable. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from jetty development in these environments. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Pink salmon	N	H	I	N	H	I	N	L	I	N	H	I	N	H	I	N	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, likelihood of stressor exposure in lacustrine environments is considered discountable. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river systems potentially suitable for jetty development. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Sockeye salmon	N	H	H	N	H	H	N	L	L	N	H	H	N	H	H	N	H	H	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for jetty development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridor. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Steelhead	N	H	H	N	?	H	N	?	L	N	?	L	N	?	H	N	?	H	Spawning activity typically occurs in habitats that are not suitable for jetty development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the level of take associated with activities in these habitat types is less certain but is conservatively presumed to occur. As juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain.
Coastal cutthroat trout	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	This species is prevalent in estuaries and large rivers (although it also occurs in Lake Washington) and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for jetty development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migration between marine or lacustrine and riverine habitats and adult foraging in the marine and estuarine environment. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Westslope cutthroat trout	N	N	H	N	N	H	N	NA	L	N	NA	H	N	NA	H	N	NA	H	These species occur primarily in coldwater streams and small to medium-sized rivers unsuitable for jetty development. Rearing juveniles and adults do occur in lacustrine environments, creating some potential for

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Redband trout	N	N	H	N	N	H	N	NA	L	N	NA	H	N	NA	H	N	NA	H	stressor exposure. As a consequence, there is effectively no risk of take in riverine environment types, while exposure in lacustrine environments may result in a moderate (from project effects on habitat quality and quantity) to high (from project construction) risk of take.
Bull trout	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	Spawning activity typically occurs in habitats that are not suitable for jetties; therefore, stressor exposure will only occur during migratory life-history stages at transitional locations between marine or lacustrine and riverine habitats. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment. Most effects would occur from development in nearshore marine migratory corridors, as well as lacustrine and marine foraging habitats used by mature juveniles and adults. However, bull trout in lakes are typically (but not exclusively) found in deeper water, limiting the potential for direct stressor exposure.
Dolly Varden	N	H	H	N	H	H	N	L	L	N	H	L	N	H	H	N	H	H	
Pygmy whitefish	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. Stressor exposure will only occur in lacustrine environments. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in the marine environment or lakes suitable for jetty development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for jetty development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Mountain sucker	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	This species is commonly found in large lakes potentially suitable for jetty development. Stressor exposure is likely to occur in these environments during the juvenile and adult life-history stages. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties that are unsuitable for jetty development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Leopard dace	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in reservoir habitats potentially suitable for jetty development at sensitive life-history stages, including egg incubation. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Umatilla dace	N	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for jetty development at sensitive life-history stages, including egg incubation. This species faces high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the lacustrine environment.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for jetty development. There is effectively no risk of take.
River lamprey	N	H	H	N	H	H	N	?	?	N	?	?	N	H	H	N	H	H	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. The ammocoete life-history stage is potentially exposed to a range of impact mechanisms resulting from jetty development in lacustrine environments. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Impact mechanism effects affecting the abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults.
Pacific lamprey	N	I	H	N	I	H	N	I	?	N	I	?	N	I	H	N	I	H	Pacific lamprey are anadromous with migratory corridors that cross estuaries and mainstems of larger river systems suitable for jetty development. Ammocoetes burrow into riverine sediments to rear for extended periods. The ammocoete life-history stage is more susceptible to a range of impact mechanisms resulting from jetty development in lacustrine environments. In the marine environment Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore less likely to be

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																			exposed to project-related stressors. Therefore, the moderate to high risk of take associated with structure-related habitat alteration and construction activities, respectively, applies primarily to lacustrine habitat. Impact mechanisms in marine and lacustrine environments that affect the abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This in turn equates to a moderate risk of take.
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered to be anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore potentially exposed to jetty-related impact mechanisms in lacustrine environments. Their relative lack of mobility increases sensitivity to a range of impact mechanisms. Sturgeon are wide ranging in marine waters. Green sturgeon occur in Washington State only as adults in marine waters, with fisheries occurring in the Willapa Bay, and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats.
White sturgeon	N	?	H	N	?	H	N	?	L	N	?	?	N	?	H	N	?	H	
Longfin smelt	N	H	H	N	H	H	N	L	?	N	I	?	N	H	H	N	H	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems. Planktonic larvae and juveniles of these species may also be vulnerable to jetty-related stressor exposure in the nearshore marine environment during early rearing. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors. Similar to other species' exposure profiles, life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. The Lake Washington population of longfin smelt rears and forages in the lacustrine environment throughout the larval, juvenile, and nonspawning adult portion of its life history and is subject to the effects of jetties in this water body.
Eulachon	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	H	N	
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Larvae of both species disperse in nearshore waters for early rearing. These beach-spawning species depend on a narrow range of substrate conditions for suitable spawning habitat, increasing sensitivity to hydraulic and geomorphic effects. Planktonic larvae are also dependent on nearshore current and circulation patterns for rearing survival. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure from hydraulic and geomorphic, and aquatic vegetation modifications is high. Planktonic larvae disperse in nearshore waters for early rearing and are dependent on current and circulation patterns for survival, growth, and fitness. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lingcod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to stressors resulting from jetty related impact mechanisms. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 and 150 m) and therefore have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific hake	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Pacific cod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Walleye pollock	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Brown rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Juveniles disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile once they have settled, juveniles are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Copper rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Black rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
China rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Redstripe rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Olympia oyster	N	H	N	N	H	N	N	H	N	N	I	N	N	H	N	N	I	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile during much of its life history, it is vulnerable to both short-term construction and water quality related impacts, as well as modification of hydraulic and geomorphic conditions in the nearshore environment. Modification of current, wave, and circulation patterns may also affect larval settlement, influencing survival during this life-history stage. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Northern abalone	N	H	N	N	H	N	N	I	N	N	I	N	N	H	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) in depth. Because this species has low mobility, it is more sensitive to a variety of impact mechanisms potentially resulting from jetty development, including construction and water quality effects. Being planktonic spawners, the species' spawning productivity is dependent on current and circulation patterns. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	I	N	N	M	N	N	?	N	The Newcomb's littorine snail inhabits *Salicornia* marshes on the littoral fringe. It is intolerant of extended submergence in both fresh and marine water; as such, it not a true aquatic species. Therefore, the potential for exposure to most stressors from jetty-related impact mechanisms is minimal. Exceptions include alteration of riparian vegetation affecting this vegetation community. Risk of take for this species is similarly limited, with the exception of a moderate risk of take resulting from potential effects on marine littoral vegetation, and low risk of take associated with behavioral avoidance of water quality degradation. It is important to note, however, that suitable habitats for these species do not typically occur in locations suitable for jetty development; therefore, the likelihood of stressor exposure in general is considered to be limited.
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for jetty development. As such, there is essentially no likelihood of stressor exposure and therefore no potential for take resulting from these activities. The giant Columbia River limpet is known to occur

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are likewise not suitable for jetty development. As such, there is effectively no risk of take resulting from jetty-related stressor exposure.
California floater (mussel)	N	N	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N	H	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake rivers and the mainstems of these systems in flowing water environments unsuitable for jetty development. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes, the latter being suitable for jetties. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Habitat accessibility modifications will not directly affect these species; however, indirect effects could occur through direct effects on host fish.
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-42. Species- and habitat-specific risk of take for mechanisms of impacts associated with breakwaters.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species has a complex and variable life history, depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for breakwater development and are thereby potentially exposed to stressors resulting from related impact mechanisms. Spawning activity typically occurs in habitats that are not suitable for breakwaters. Therefore, stressor exposure will only occur during migratory life-history stages in the lower reaches of large rivers, and lacustrine and marine environments. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Coho salmon	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species has a complex and variable life history, depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for breakwater development and may experience exposure to related stressors. Spawning activity typically occurs in habitats that are not suitable for breakwaters. Therefore, stressor exposure will only occur during migratory life-history stages in the lower reaches of large rivers, and lacustrine and marine environments. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.
Chum salmon	H	H	I	H	H	I	L	H	I	H	H	I	L	H	I	Chum salmon in Washington State do not use lacustrine habitats suitable for breakwater development. Therefore, stressor exposure will not occur in lacustrine environments. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River). As such, in addition to migratory juveniles and adults, spawning habitats may be exposed to stressors resulting from breakwater-related impact mechanisms. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from breakwater development in these environments.
Pink salmon	H	H	I	H	H	I	L	H	I	H	H	I	L	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in this environment type. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through the mainstems and estuaries of larger river environments potentially suitable for breakwater development. As such, this species may potentially be exposed to stressors resulting from related impact mechanisms. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Sockeye salmon	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species is highly dependent on lacustrine environments for juvenile rearing. Most spawning behavior occurs in smaller rivers and streams that are not suitable for breakwater development. However, some populations spawn in nearshore lacustrine habitats, creating increased risk of stressor exposure at sensitive egg and alevin life-history stages. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridors. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Steelhead	H	H	H	H	?	H	L	?	H	H	H	H	L	?	H	Spawning activity typically occurs in habitats that are not suitable for breakwater development; therefore, eggs and alevins will not experience stressor exposure. Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats, so the level of take associated with activities in these habitat types is less certain, but is conservatively presumed to occur. As juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Coastal cutthroat trout	H	H	H	H	H	H	L	H	H	H	H	H	L	H	H	This species is prevalent in estuaries and large rivers (although it also occurs in Lake Washington) and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for breakwater development. Migratory behavior and residence timing are variable. Spawning activity typically occurs in habitats that are not suitable for breakwaters; therefore, stressor exposure will only occur during juvenile rearing adult foraging. These life-history stages face high risk of take during the construction of these structure types, and moderate risk of take from the effects of the structure on the migratory environment.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Westslope cutthroat trout	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Occurrence in estuaries of larger rivers suitable for breakwater development is highly unlikely; therefore, the risk of take associated with these structures is considered discountable. Stressor exposure in lacustrine environments is possible, however. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Redband trout	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	
Bull trout	H	H	H	L	H	H	L	H	H	H	H	H	L	H	H	Spawning by these species occurs in habitats that are generally unsuitable for breakwater development. Therefore, spawning, egg incubation, and early rearing will not be directly affected by these activities. Most effects will occur from development in riverine migratory corridors, as well as in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults. However, bull trout in lakes are typically (but not exclusively) found in deeper water, limiting the potential for direct stressor exposure. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Dolly Varden	H	H	H	L	H	H	L	H	H	H	H	H	L	H	H	
Pygmy whitefish	N	N	H	N	N	H	N	N	H	N	N	H	N	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. Whitefish do not occur in larger rivers suitable for breakwater development; therefore, stressor exposure will only occur in lacustrine environments. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. Species does not occur in larger rivers or lakes suitable for breakwater development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Margined sculpin	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages unsuitable for breakwater development. Therefore, stressor exposure will not occur and there is effectively no risk of take.
Mountain sucker	H	N	H	H	N	H	H	N	H	H	N	H	L	N	H	This species is commonly found in large rivers and lakes suitable for breakwater development. Stressor exposure is likely to occur across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lake chub	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties that are unsuitable for breakwater development. Therefore, stressor exposure will not occur, and there is effectively no risk of take.
Leopard dace	H	N	H	H	N	H	H	N	H	H	N	H	L	N	L	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for breakwater development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Umatilla dace	H	N	H	H	N	H	H	N	H	H	N	H	L	N	L	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for breakwater development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Western brook lamprey	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	This species is characterized by isolated breeding populations favoring small streams and brooks, which are unsuitable environments for breakwater development. There is effectively no risk of take.
River lamprey	H	H	H	H	H	H	?	?	?	H	H	H	L	H	L	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries and lower reaches of larger rivers to rear for extended periods, potentially years. The ammocoete life-history stage is more susceptible to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanisms affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults, which in turn equates to a moderate

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																risk of take.
Pacific lamprey	H	I	H	H	I	H	?	I	?	H	I	H	L	I	L	Pacific lamprey are anadromous, with migratory corridors that cross estuaries and mainstems of larger river systems suitable for breakwater development. Ammocoetes burrow into riverine sediments to rear for extended periods. The ammocoete life-history stage is more susceptible to acute transient water quality impacts, such as reduced dissolved oxygen or altered pH. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore less likely to be exposed to project-related stressors in the nearshore marine environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanisms affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This in turn equates to a moderate risk of take.
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. These life-history stages are therefore potentially exposed to water quality related impact mechanisms from breakwaters. Their relative lack of mobility increases sensitivity to acute transient water quality impacts such as reduced dissolved oxygen or altered pH. Sturgeon are wide ranging in marine waters. Green sturgeon occur in Washington State only as adults in marine waters, with fisheries occurring in Willapa Bay and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats and the related risk of take.
White sturgeon	H	?	H	H	?	H	H	?	H	H	?	H	H	?	H	
Longfin smelt	H	H	H	H	H	H	I	I	H	H	H	H	H	H	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems potentially suitable for breakwater development. Demersal adhesive eggs are vulnerable to acute transient water quality impacts, such as reduced dissolved oxygen or altered pH. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors. The Lake Washington population of longfin smelt rears and forages in the lacustrine environment throughout the larval, juvenile, and nonspawning adult portion of its life history and is subject to the effects of breakwaters in this water body.
Eulachon	H	H	N	H	H	N	I	I	N	H	H	N	H	H	N	
Pacific sand lance	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Larvae of both species disperse in nearshore waters for early rearing. These beach-spawning species depend on a narrow range of substrate conditions for suitable spawning habitat, increasing sensitivity to hydraulic and geomorphic effects. Planktonic larvae are also dependent on nearshore current and circulation patterns for rearing survival. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Surf smelt	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Pacific herring	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure from hydraulic/geomorphic and aquatic vegetation modifications is high. Planktonic larvae disperse in nearshore waters for early rearing and are dependent on current and circulation patterns for survival, growth, and fitness. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lingcod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to water quality related impact mechanisms from breakwaters. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
																m), but are most prominent between 330 and 500 ft (100 and 150 m) and, therefore, have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific hake	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific cod	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Walleye pollock	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Brown rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Juveniles disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile once they have settled, juveniles are vulnerable to short-term construction and water quality related impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Copper rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Black rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
China rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Redstripe rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	H	N	N	H	N	N	H	N	N	H	N	
Olympia oyster	N	H	N	N	H	N	N	I	N	N	H	N	N	I	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile during much of its life-history, it is vulnerable to both short-term construction and water quality related impacts, as well as modification of hydraulic and geomorphic conditions in the nearshore environment. Modification of current, wave, and circulation patterns may also affect larval settlement, influencing survival during this life-history stage. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Northern abalone	N	H	N	N	H	N	N	I	N	N	H	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) in depth. Because this species has low mobility, it is more sensitive to a variety of impact mechanisms potentially resulting from breakwater development, including construction and water quality effects. Being planktonic spawners, spawning productivity is dependent on current and circulation patterns. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Newcomb's littorine snail	N	N	N	N	H	N	N	N	N	N	L	N	N	?	N	The Newcomb's littorine snail inhabits *Salicornia* marshes on the littoral fringe. It is intolerant of extended submergence in both fresh and marine water; as such, it not a true aquatic species. Therefore, the potential for exposure to most stressors from breakwater-related impact mechanisms is minimal, as these offshore structures have limited effects on littoral vegetation. This is particularly true for *Salicornia* marshes, which predominantly occur in low-energy environments less subject to the effects of breakwaters on wave energy.

Species	**Construction & Maintenance Activities**			**Hydraulic and Geomorphic Modifications**			**Aquatic Vegetation Modifications**			**Water Quality Modifications**			**Ecosystem Fragmentation**			**Comments**
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
																The only potential risk of take associated with breakwater development is from temporary water quality effects. This risk is rated as low.
Giant Columbia River limpet	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	The great Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for breakwater development. As such, there is essentially no likelihood of stressor exposure and, therefore, no potential for take resulting from these activities. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are generally not suitable for breakwater development. There is no risk of take for either species.
Great Columbia River spire snail	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	
California floater (mussel)	H	N	H	H	N	H	I	N	I	H	N	H	L	N	L	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake rivers. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. Only the latter species occurs in habitats suitable for breakwater development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Habitat accessibility modifications will not directly affect this species; however, indirect effects could occur through direct effects on host-fish.
Western ridged mussel	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

Table 9-43. Species- and habitat-specific risk of take for mechanisms of impacts associated with groins and bank barbs.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Chinook salmon	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species has a complex and variable life history, depending on race. In general, Chinook salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for groin or bank barb development and are thereby potentially exposed to stressors resulting from related impact mechanisms across all life-history stages. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Coho salmon	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species has a complex and variable life history, depending on race. In general, coho salmon occur in riverine, lacustrine, and nearshore marine habitats suitable for groin or bank barb development and may experience exposure to related stressors across all life-history stages. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Chum salmon	H	H	I	H	H	I	L	L	I	L	H	I	H	H	I	H	H	I	Chum salmon in Washington State do not use lacustrine habitats in any significant fashion. Therefore, the likelihood of stressor exposure in lacustrine environments is considered discountable. Chum may spawn in the lower reaches of large river environments (e.g., the Columbia River). As such, in addition to migratory juveniles and adults, spawning habitats may be exposed to stressors resulting from groin or bank barb related impact mechanisms. Juvenile chum salmon are dependent on nearshore marine habitats and are therefore subject to stressor exposure from groin or bank barb development in these environments. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pink salmon	H	H	I	H	H	I	L	L	I	L	H	I	H	H	I	H	H	I	Pink salmon in Washington State do not use lacustrine habitats. Therefore, stressor exposure will not occur in this environment type and the likelihood of stressor exposure in lacustrine environments is considered discountable. This species is dependent on nearshore marine habitats for juvenile rearing and migrates through and spawns in the mainstems and estuaries of larger river systems potentially suitable for groin or bank barb development. As such, this species may potentially be exposed to stressors resulting from related impact mechanisms. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Sockeye salmon	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species is highly dependent on lacustrine environments for juvenile rearing, and most spawning behavior occurs in smaller rivers and streams that are also suitable for groin or bank barb development. Lake spawning populations also face risk of stressor exposure at sensitive egg and alevin life-history stages in lacustrine environments. Migrating juveniles and adults may experience stressor exposure in larger rivers and reservoirs along their migratory corridors. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Avoidance of impacts on juvenile sockeye in lacustrine environments is difficult due to year-round residence. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Steelhead	H	?	H	H	?	H	L	?	L	L	?	H	H	?	H	H	?	H	Steelhead have a lesser but uncertain level of dependence on nearshore marine habitats; the level of take associated with activities in these habitat types is less certain, but is conservatively presumed to occur. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. As juvenile steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the risk of take in the marine environment is uncertain. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Coastal cutthroat trout	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	This species is prevalent in estuaries and large rivers and is highly dependent on nearshore marine areas for foraging. These habitats are suitable for groin or bank barb development. Migratory behavior and residence timing are variable. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.

	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			
Species	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Comments**
Westslope cutthroat trout	N	N	H	H	N	H	N	N	L	N	N	H	N	N	H	N	N	H	These species occur primarily in coldwater streams, small to medium-sized rivers, and lakes. Bank barb development in smaller streams may affect spawning adults, eggs, alevins, and rearing juveniles. Groin development in moderate-sized rivers may have similar effects across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Redband trout	N	N	H	H	N	H	N	N	L	N	N	H	N	N	H	N	N	H	
Bull trout	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	Most effects will occur from development in riverine migratory corridors, as well as in riverine, lacustrine, and marine foraging habitats used by mature juveniles and adults, which are all potential sites for groin or bank barb development. In lakes, however, char are typically found in deeper water, limiting the potential for direct stressor exposure. Bank barb development in smaller streams may affect spawning adults, eggs, and alevins. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Dolly Varden	H	H	H	H	H	H	L	L	L	L	H	H	H	H	H	H	H	H	
Pygmy whitefish	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	Lakes and smaller lake tributaries are primary habitats used by this species. These environments are suitable for groin or bank barb development, meaning that this species may be exposed to stressors from related impact mechanisms across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Olympic mudminnow	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Primary habitats are wetlands and small, slow-flowing streams. These environments are generally not suitable for groin or bank barb development. Therefore, there is essentially no likelihood of stressor exposure and effectively no risk of take.
Margined sculpin	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	Primary habitats are located in smaller tributary streams of the Walla Walla and Tucannon River drainages. While generally remote, these streams are potentially suitable for groin or bank barb development. Therefore, stressor exposure may occur across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Mountain sucker	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	This species is commonly found in large rivers and lakes suitable for groin or bank barb development. Stressor exposure is likely to occur across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lake chub	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	The known distribution of this species in Washington State is limited to small streams and lakes in Okanogan and Stevens counties. These habitats may be suitable for groin or bank barb development, presenting the potential for stressor exposure across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Leopard dace	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	This species has been reported to occur in the Columbia and Cowlitz River systems west of the Cascade Range, and in the Columbia River mainstem to the east. As such, this species occurs in habitats potentially suitable for groin or bank barb development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Umatilla dace	H	N	H	H	N	H	L	N	L	H	N	H	H	N	H	H	N	H	This species has been reported to occur in the Columbia, Yakima, Okanogan, Similkameen, Kettle, Colville, and Snake rivers (including reservoirs within the Columbia and Snake River systems). As such, this species occurs in habitats potentially suitable for groin or bank barb development at sensitive life-history stages, including egg incubation. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Western brook lamprey	H	N	I	H	N	I	L	N	I	H	N	I	H	N	I	H	N	I	This species is characterized by isolated breeding populations favoring small streams and brooks. This species is particularly vulnerable to impact mechanisms resulting from bank barb development, and experiences exposure to related stressors across all life-history stages. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Occurrence in lacustrine habitats is extremely rare; therefore, the likelihood of stressor exposure and the related potential for take in this environment type are considered discountable.
River lamprey	H	H	H	H	H	H	L	?	?	?	?	?	H	H	H	H	H	H	River lamprey are commonly found in nearshore areas of rivers and some lake systems. Lamprey ammocoetes burrow into sediments in quiet backwaters of lakes and nearshore areas of estuaries, and lower reaches of larger rivers to rear for extended periods, potentially years. This nonmobile life-history stage is more susceptible to acute construction-related impacts and longer term alteration of habitat suitability due to hydraulic and geomorphic modifications, as well as other changes in habitat complexity. In their saltwater phase, river lamprey remain close to shore for periods of 10–16 weeks from spring through fall, increasing exposure to stressors in the nearshore environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanism effects affecting the abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This equates to a moderate risk of take.
Pacific lamprey	H	I	H	H	I	H	L	I	?	?	I	?	H	I	H	H	I	H	Pacific lamprey are anadromous, with migratory corridors that cross estuaries and mainstems of larger river systems suitable for groin or bank barb development. Ammocoetes burrow into riverine sediments to rear for extended periods. This nonmobile life-history stage is more susceptible to acute construction-related impacts, and longer term alteration of habitat suitability due to hydraulic and geomorphic modifications and other changes in habitat complexity. Pacific lamprey occupy epipelagic habitats away from the nearshore environment for periods ranging from 6–40 months and are therefore less likely to be exposed to project-related stressors in the nearshore marine environment. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Impact mechanism effects affecting abundance of host fish may lead to indirect effects on growth and fitness of transforming adults and adults. This equates to a moderate risk of take.
Green sturgeon	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	N	?	N	In Washington, white sturgeon are found in the Columbia River, Snake River, Grays Harbor, Willapa Bay, Puget Sound, and Lake Washington. Although this species is considered anadromous, populations in the Columbia River may be reproducing successfully in some impoundments. Sturgeon eggs are demersal and adhesive. Larval sturgeon are essentially planktonic and rear in quiet backwaters of the large rivers and lakes where they are transported by currents following emergence. This less mobile life-history stage is more susceptible to acute construction-related impacts and longer term alteration of habitat suitability due to hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Sturgeon are wide ranging in marine waters. Green sturgeon fisheries occur in the Columbia River below Bonneville Dam, Willapa Bay, and Grays Harbor. Individuals are also occasionally caught incidentally in small coastal bays and the Puget Sound. Dependence on nearshore marine habitats is unknown, as is the potential for exposure to stressors occurring in nearshore habitats. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
White sturgeon	H	?	H	H	?	H	L	?	L	H	?	H	H	?	H	H	?	H	
Longfin smelt	H	H	H	H	H	H	L	I	L	I	I	H	H	H	H	H	H	H	Eulachon and longfin smelt spawn in the lower reaches of moderate to large river systems potentially suitable for groin or bank barb development. Spawning habitat suitability may be adversely affected by construction and longer term modifications of habitat suitability from hydraulic and geomorphic modifications or other changes in habitat complexity. Planktonic larvae and juveniles of these species may also be vulnerable to stressor exposure in the nearshore marine environment during early rearing. Mature juveniles and adults occupy offshore environments and are therefore at less risk of take from these stressors. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Eulachon	H	H	N	H	H	N	L	L	N	I	I	N	H	H	N	H	H	N	
Pacific sand lance	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Surf smelt and sand lance populations are widespread and ubiquitous in Puget Sound, the Strait of Juan de Fuca, and the coastal estuaries of Washington. They are dependent on shoreline habitats for spawning and are prevalent in the nearshore environment, meaning that the likelihood of stressor exposure is high. Larvae

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	Riverine	Marine	Lacustrine	
Surf smelt	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	of both species disperse in nearshore waters for early rearing. These beach-spawning species depend on a narrow range of substrate conditions for suitable spawning habitat, increasing sensitivity to hydraulic and geomorphic effects. Planktonic larvae are also dependent on nearshore current and circulation patterns for rearing survival. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific herring	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Pacific herring are common throughout the inland marine waters of Washington, particularly in protected bays and shorelines used for spawning. This species is dependent on nearshore habitats for spawning, egg incubation, and larval rearing, meaning that the likelihood of stressor exposure from hydraulic/geomorphic and aquatic vegetation modifications is high. Planktonic larvae disperse in nearshore waters for early rearing and are dependent on current and circulation patterns for survival, growth, and fitness. Planktonic life-history stages are also incapable of escaping acute water quality impacts. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Lingcod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Larval lingcod settle in nearshore habitats for juvenile rearing, favoring habitats with freshwater inflow and reduced salinities, and are potentially exposed to water quality related impact mechanisms from groins and bank barbs. Adults may occur anywhere from the intertidal zone to depths of approximately 1,560 ft (475 m), but are most prominent between 330 and 500 ft (100 and 150 m) and, therefore, have less exposure potential. Temporary disturbance while brooding may increase risk of egg predation. Larvae disperse and settle in nearshore waters for early rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction impacts and longer term impacts from hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Changes in wave energy, current, and circulation patterns may adversely affect larval settlement in areas favorable for development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific hake	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Hake, cod, and pollock spawn in nearshore areas and estuaries, and their planktonic larvae settle in nearshore areas for rearing. Larval Pacific cod settle in nearshore areas associated with eelgrass. Larval pollock settle in nearshore areas at depths as shallow as 33 ft (10 m) for juvenile rearing and are commonly associated with eelgrass algae. As such, spawning adults, eggs, larvae, and juveniles may experience stressor exposure. Larvae disperse and settle in nearshore waters for early rearing, and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction impacts and longer term impacts from hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Changes in wave energy, current, and circulation patterns may adversely affect larval settlement in areas favorable for development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Pacific cod	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Walleye pollock	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Brown rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	Rockfish are ovoviviparous species that release their planktonic larvae in open water, depending on favorable currents and circulation patterns to carry them into nearshore habitats where they settle for rearing as demersal juveniles. Many species remain in the vicinity of the nearshore environment as they grow into adulthood. As such, rockfish can experience stressor exposure across all life-history stages. Juveniles disperse and settle in nearshore waters for early rearing and are dependent on current, wave, and circulation patterns to ensure dispersal to environments favorable for rearing. Because they are demersal and relatively immobile, larvae are vulnerable to short-term construction impacts and longer term impacts from hydraulic and geomorphic modifications, as well as other changes in habitat complexity. Changes in wave energy, current, and circulation patterns may adversely affect larval settlement in areas favorable for development. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Copper rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Greenstriped rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Widow rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yellowtail rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Quillback rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Black rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
China rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Tiger rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Bocaccio rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Canary rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	

Species	Construction & Maintenance Activities			Hydraulic and Geomorphic Modifications			Riparian Vegetation Modifications			Aquatic Vegetation Modifications			Water Quality Modifications			Ecosystem Fragmentation			Comments
	Riverine	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	**Riverine**	**Marine**	**Lacustrine**	
Redstripe rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Yelloweye rockfish	N	H	N	N	H	N	N	L	N	N	H	N	N	H	N	N	H	N	
Olympia oyster	N	H	N	N	H	N	N	L	N	N	I	N	N	H	N	N	I	N	This species occurs commonly in shallow water nearshore habitats. This distribution increases risk of stressor exposure and potential for take resulting from water quality modification in the nearshore environment. Because this species is sessile during much of its life history, it is vulnerable to both short-term construction-related impacts, as well as modification of hydraulic and geomorphic conditions in the nearshore environment. Modification of current, wave, and circulation patterns may also affect larval settlement, influencing survival during this life-history stage. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Northern abalone	N	H	N	N	H	N	N	L	N	N	I	N	N	H	N	N	I	N	While increasingly rare due to depressed population status, this species occurs commonly in nearshore habitats less than 33 ft (10 m) in depth. Because this species has low mobility, it is more sensitive to a variety of impact mechanisms potentially resulting from development associated with groins and bank barbs, including construction and water quality effects. Being planktonic spawners, this species' spawning productivity is dependent on current and circulation patterns. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Newcomb's littorine snail	N	H	N	N	H	N	N	H	N	N	N	N	N	L	N	N	?	N	The Newcomb's littorine snail inhabits *Salicornia* marshes on the littoral fringe. It is intolerant of extended submergence in both fresh and marine water; as such, it not a true aquatic species and the potential for exposure to most stressors from groin or bank barb related impact mechanisms is minimal. Exceptions include alteration of riparian vegetation affecting this vegetation community in the direct footprint of these structures, as well as hydraulic and geomorphic modifications. Life-history stages exposed to construction activities face a high risk of take (from direct mortality or injury), while the effects of the structure on the environment are likely to result in a moderate to low risk of take.
Giant Columbia River limpet	H	N	N	H	M	N	L	N	N	L	N	N	H	N	N	?	N	N	The Columbia River spire snail is typically found in smaller streams in water less than 5 inches deep, environments unsuitable for development of groins and bank barbs. The giant Columbia River limpet is known to occur in the Hanford Reach of the Columbia River and other moderate to large river environments in the state, typically in shallow, flowing water environments with cobble and boulder substrates. These environments are suitable for groin or bank barb development so there is a risk of stressor exposure. These species are dependent on flowing water and therefore will not experience stressor exposure from related impact mechanisms in lacustrine environments. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take.
Great Columbia River spire snail	H	N	N	H	N	N	L	N	N	L	N	N	H	N	N	?	N	N	
California floater (mussel)	H	N	H	H	M	H	L	N	L	L	N	L	H	N	H	H	N	H	The western ridged mussel is predominantly found in the larger tributaries of the Columbia and Snake rivers and the mainstems of these systems. The California floater occurs in shallow muddy or sandy habitats in larger rivers, reservoirs, and lakes. As such, both species may occur in habitats potentially suitable for groin or bank barb development and have potential for stressor exposure from all related impact mechanisms. Life-history stages exposed to construction activities face a high risk of take, while the effects of the structure on the environment are likely to result in a moderate risk of take. Habitat accessibility modifications will not directly affect this species; however, indirect effects could occur through direct effects on host-fish. This equates to a moderate risk of take.
Western ridged mussel	H	N	N	H	M	N	L	N	N	L	N	N	H	N	N	H	N	N	

Risk of Take Ratings: **H** = High, **M** = Moderate; **L** = Low; **I** = Insignificant or Discountable; **N**= No Risk of Take; **?** = Unknown Risk of Take.
Shaded cells indicate environment types in which the species in question does not occur; therefore, there is no risk of take from the impact mechanism in question.

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10 Data Gaps

The activity-specific white papers prepared for WDFW in 2006 and 2007 identified data gaps in two ways:

- data gaps identified for specific "mechanisms of impact"(i.e., missing information on the impacts from construction and maintenance, hydraulic and geomorphic modifications, riparian vegetation modification, aquatic vegetation modification, water quality modifications, and ecosystem fragmentation on HCP species, and

- data gaps identified for specific activities and sub-activities.

In addition, there are several data gaps related to the general lack of knowledge of the more obscure HCP species. These gaps include:

- the extent of utilization of nearshore areas by species like green and white sturgeon and steelhead.

- the dependence of some species on both riparian and aquatic vegetation, such as Pacific lamprey and sturgeon.

- Survival and fitness following physical disturbance, handling, and relocation-related dispersal (e.g., to what degree does the unintentional dispersal of freshwater mussels limit density-dependent reproductive success).

10.1 Data Gaps Identified for Specific Mechanisms of Impact

10.1.1 Construction, Maintenance and Operations

10.1.1.1 Noise

Data on the effects of exposure to sound from pile driving on specific fish or invertebrates are few, and although the few studies completed provide some information about exposures to pile-driving sounds, there is little that can be definitively concluded (Hastings and Popper 2005). Hastings and Popper (2005) stress that because monitoring data show that sound pressure levels do not necessarily decrease monotonically with increasing distance from the pile, it is important that received sound levels be measured in future experiments to develop exposure metrics that correlate with mortality and different types of damage observed in fish exposed to pile driving. Hastings and Popper (2005) conclude that it is important to initiate experimental studies that start with basic questions about the effects on fishes from exposure to pile-driving sounds. Recommended studies from Hastings and Popper (2005) are presented in Table 10. Two data gaps are particularly significant: the cumulative impact of sound to fish and the effects of noise on the behavior of fish and the consequent impact to species survival and recovery.

Table 10-1. Research Questions on the Impact of Pile Driving on Fishes (Hastings and Popper 2005)

Project Title	Project Objectives	Significance	Relationship to Other Studies	Relationship to Pile Driving Needs
Characterize Pile Driving Sounds				
Define acoustic dose for exposure to pile driving sound	Develop ways to express exposure to pile driving sounds in terms of total energy received and the degree of temporal variation in the waveform, and to define the acoustic particle velocity within the sound field	This will provide a series of "standard" pile driving sounds in water and substrate for use as the stimuli with which to do studies on representative species	This study is fundamental to investigations of effects on fishes because it provides laboratory signals that would be representative of the range of pile driving stimuli in different locations	Without this standardization it will be impossible to generalize between studies done in different locales and with different piles
Structural acoustic analysis of piles	Develop structural acoustics models of piles to investigate how modifications to piles and hammering could alter the sounds and potentially incur less damage to animals	This could result in potential modifications to the structure, hammer, and/or process that could reshape the temporal characteristics of the pile driving stimulus without changing structural integrity	Would need to test modified sounds on animal models	This analysis will help provide ways to mitigate some effects of pile driving on aquatic organisms
Define characteristics of the underwater sound field	Develop underwater sound propagation model and integrate with pile structural acoustics models to estimate received levels of sound pressure and particle velocity in the vicinity of pile driving operations and verify with field measurements of underwater sound pressure measurements	This is the only way to define zones of impact on fishes because the sound energy received by a fish depends on not only the pile-driving source, but also the size, shape, and properties of the underwater environment.	Would be able to map the impact of pile driving sounds on the underwater environment based on results of tests of pile driving sounds on animal models	Received levels of sound pressure and acoustic particle velocity must be known underwater in the region surrounding the pile to calculate appropriate metrics related to observed effects and define the zone of impact
Characterize injury of fish exposed to pile driving sounds				
Hearing capabilities of Pacific Coast fishes	Determine hearing capabilities (using Auditory Brainstem Response [ABR]) of representative species. Determine in terms of both pressure and particle motion.	Useful for prediction of detection range of pile driving sounds and potential effects on hearing capabilities	Previous behavioral studies did not use any Pacific Coast fishes or elasmobranches	Studies would be on species that are particularly germane to those affected by pile driving

Project Title	Project Objectives	Significance	Relationship to Other Studies	Relationship to Pile Driving Needs
Mortality of fishes exposed to pile driving	Determination of short and long term effects on mortality of representative species as a result of pile driving. Measure pathology (using necropsy studies) of the effects on fishes of received sounds representative of different distances from the source	Provide baseline data on effects of pile driving and the effects of such signals of different levels and spectral components	Studies of this type have, heretofore, not be done under controlled situations	Provide mortality data as well as pathology as to the effects of pile driving and determination of the cause of immediate and long-term mortality
Effects of pile driving on non-auditory tissues	Using the precise same paradigm as for effects on the ear, examine other tissues using standard fish necropsy techniques to asses gross, cellular, and molecular damage to fish. Furthermore, determine stress effects on fish using appropriate stress measures (e.g., hormone levels). Do for representative species.	Provide insight into how the sounds affect fish, even when there is no immediate mortality	The only comparable data are from blasts, which suggest significantly different effects depending on fish size and species.	Direct measure of potential long-term damage to fishes.
Effects of pile driving on hearing capabilities	Determine temporary threshold shifts and permanent threshold shifts on representative species.	Provide insight into hearing loss and possible recovery as a result of different sound levels and sound types	No studies of this type have been done using pile-driving sounds	Data that will help understand the sound levels and other parameters that could result in the loss of the ability of different species types to detect sounds, and thus detect biologically critical signals
Effects of pile driving on fish eggs and larvae	Determine mortality, growth rates, and pathological changes in developing fishes of representative species with exposure at different times during the development cycle	Since eggs and larvae do not move from the sites of spawning, determine if long-term pile driving could affect fish populations	No studies done on any fish system are relevant to this investigation	If fish spawn in the vicinity of pile driving sites, or cannot be kept from spawning during pile driving operations, effects on eggs and larvae could be considerable

Project Title	Project Objectives	Significance	Relationship to Other Studies	Relationship to Pile Driving Needs
Behavioral responses of fish to pile driving	Observe, in large-scale cages, the short-term behavioral responses of representative species to pile driving sounds. Do fish attempt to swim from the source? Do they react to the sounds? Do they "freeze" in place?	In knowing behavioral responses, it may be possible to predict which species would remain in an area of pile driving vs. species that could be expected to leave the area after the initial pile driving activity.	None have been done to date.	This may help limit the number of species that would need to be "protected."
Long-term behavioral effects of pile driving on fish	Attempt to do field studies that would provide insight into movement patterns of fishes and normal behaviors and how these might be affected, in the long-term, by the presence of continuous pile driving.	While there may be few or no apparent effects on immediate behavior (e.g., rapid swimming), physiology (e.g., hearing, effects on other organs), or mortality, there may be longer-term behavioral effects such as those from continual sounds from pile driving preventing fish from reaching breeding sites, finding food, hearing and finding mates, etc. This could result in long-term effects on reproduction and population survival.	None have been done to date.	Pile driving may not have an immediate impact on fishes, but continual pile driving may have longer-term effects that could significantly alter fish populations in the areas in which pile driving takes place.
Effects of pile driving on the ear and lateral line	Determine morphological changes over time for representative species on sensory cells of the ear and lateral line, and whether such changes are reversible	If there is loss of sensory cells there is a loss in hearing ability or the ability of the lateral line to be used in hydrodynamic reception. If there is recovery of these cells, fishes may be able to survive (assuming they did not die prior to recovery).	A few studies suggest that exposure to high sound pressure levels will affect the sensory cells of the ear, but almost nothing is known about the lateral line. However, no studies were done with sounds comparable to those from pile driving	Loss of hearing capabilities, even for a short period of time, could dramatically affect survival of fishes.
Effects of multiple pile driving exposures on fish	For the appropriate experiments cited above, determine effects of multiple exposures, over time, of pile driving	Some fishes may stay in the pile driving area, or go between areas that have different time tables for pile driving. Thus, there may be multiple exposures over time	No data in the literature.	If fish remain in an area over time, there may be cumulative effects that need to be understood

Specific data gaps on the sound sensitivity of fish and shellfish were identified:

- The sound sensitivity of primitive fishes (such as lamprey) is currently unknown.
- Hearing capacities of other HCP species and the effects of increased underwater noise on hearing is unknown.
- Effects on heart, kidneys, and other highly vascular tissue due to construction are unknown.
- Although studies have identified elevated hearing thresholds in response to engine and other white noises for cyprinid fishes (which are hearing specialists), data are needed on hearing and on how fish react to temporary, chronic, and cumulative anthropogenic noises caused by vessels, construction, and other sources.
- Effects of underwater noise on mollusks in general are a data gap. The sound sensitivity of the Olympia oyster is currently a data gap, and the effects of related sound stressors are unknown.
- No research has been identified regarding the effects of lower intensity, continuous underwater noise on invertebrates. However, operational noise is typically associated with sound pressures well below levels that have been observed to cause injury in shellfish, suggesting that HCP invertebrate species would not be subject to these effects. Because HCP invertebrates with the potential for stressor exposure are either filter feeders or grazers and are essentially non-motile, these species are unlikely to be subject to auditory masking effects that would limit their ability to sense predators and prey. Some potential may exist for disturbance induced interruption of feeding behavior, but more research on this subject is necessary to definitively determine this and this subject is considered a data gap.
- It is important to develop information on ambient noise levels for particular areas, because ambient noise levels influence the area of effect (attention to ambient), and fish reaction to sound likely varies depending on the "loudness" of ambient conditions.

10.1.1.2 Vessel Activities and Nearshore Structures

Relatively little is known about the potential impacts of vessel activities on potentially covered species. Although some work has been done with respect to turbidity, much of the research to date has focused on freshwater environments.

- More work is needed with respect to impacts of smaller vessels on turbidity in estuarine and marine environments.
- Much work is also needed to assess the noise impacts of small vessels operating at varying speeds, so that noise levels specific to conditions created by a particular project can be estimated.

- Potential impacts of small vessels on eelgrass and aquatic vegetation are not well known, and more work is needed to support impacts to these resources.

- Haas et al. (2002) recommends determining thresholds of disturbance for epibenthic communities affected by varying degrees of vessel activity.

- No literature was identified describing the potential impacts of vessel activities with respect to artificial light.

- Additional data gaps include the effects of temporary shading associated with vessel operations during construction of overwater structures or installation of non-structural piling. However, in general vessels required for the construction of overwater structures and installation of non-structural piling operate during the approved in-water work window, which minimizes potential impacts associated with shading.

- Additional data gaps relate to the operation of commercial and recreational vessels which may be moored at an overwater structure or non-structural piling, and may occur at various times of year and therefore affect covered species.

- Information specific to facility operation and vessel activities is needed to address temporary, chronic, and cumulative impacts on HCP species.

Grounding, anchoring, and prop wash are forms of direct disturbance from vessel activity. Grounding, anchoring, and prop wash are likely to cause effectively permanent alteration of substrate characteristics and the aquatic vegetation community. While numerous studies have documented the effects of grounding, anchoring, and prop wash on habitat, direct assessments of the impacts on species have not been studied for most HCP species and remain a data gap. These include temporary, chronic, and cumulative impacts on HCP species in marine, riverine, and lacustrine environments.

Previous white papers prepared in conjunction with WDFW, WSDOT, and Ecology on overwater structures (Nightingale and Simenstad 2001a; Carrasquero 2001) identified significant gaps on the subject of ambient light modifications and effects on habitat along marine shorelines. Specifically, these gaps included further exploration to:

(1) determine the conditions for and the significance of avoidance of shoreline structures by migrating juvenile salmon;

(2) measure the effects of using artificial lights in under-pier environments to avoid interference with natural ambient light patterns in shallow nearshore habitats;

(3) further quantify the effects of overwater structures on salmonid prey resource abundance; and

(4) develop a scientifically based approach to determine cumulative impact thresholds.

Since 2001, Toft et al. (2004), studying fish distribution, abundance, and behavior at nearshore habitats, reported on fish behavior along the urban marine shorelines of Seattle. This observational work (with an emphasis on juvenile salmonids) has helped to identify fish behavioral responses to overwater structures on these urban shorelines. Haas et al. (2002) added information on the impacts of terminals and vessel activities on shading and the response of epifaunal biota to these changes. Southard et al. (2006) further studied the conditions and responses of juvenile salmon to ferry terminals. These studies have supported the previous findings of salmonid avoidance of docks identified in (Nightingale and Simenstad 2001a; Weitkamp and Schadt 1982; Pentec 1997; Shreffler and Moursund 1999; Simenstad et al. 1999).

The question of cumulative effects of HPA-permitted structures has yet to be addressed. There is still a need for a scientifically based cumulative assessment tool to guide the design and placement of structures. This assessment should include steps to:

> (1) develop a landscape-scale model of shoreline processes that create and maintain biological habitats;
>
> (2) develop assessment indices for identifying ecological responses to structures within the context of the model;
>
> (3) identify landscape-level subunits, such as shoreline drift cells (sectors); and
>
> (4) identify landscape elements in terms of connectivity and homogeneity using the fundamental definitions of corridors, matrices, patches, and other landscape attributes.

Although effects of HPA-permitted structures on the behavior of salmonids and their prey resources have been studied, similar effects on other HCP species can only be inferred from these findings, as the majority of these species have not been the focus of the studies to date.

10.1.1.3 Construction/Maintenance Dredging

There are numerous studies of impacts on aquatic species from dredging activities (Cooper et al. 2007; Erftemeijer and Lewis 2006; Newell et al. 2004). However, these impacts have been shown to be site- and species-specific (Byrnes et al. 2004), with “opportunistic” species (e.g., mollusks) being much less affected than those that have long life histories (e.g., rockfish) (Newell et al. 2004). Considering the diversity of environments present in Washington, a number of data gaps exist with respect to specific HCP species, particularly with the effects on rockfish from adjacent dredging operations. While dredging is already prohibited in rockfish nursery areas by WAC 220-110-320, adjacent areas potentially exposed to heightened turbidity are not covered by this legislation. Turbidity thresholds that have been used successfully in existing monitoring programs to protect aquatic species are unknown and are considered a data gap (Thorkilsen and Dynesen 2001).

Although the physics of turbidity generation can be calculated, adequate data do not exist to quantify the biological response in terms of threshold sediment dosages and exposure durations that can be tolerated by each of the HCP species. Numerical modeling simulations of dredging-related suspended sediment plume dynamics need to be correlated with field and laboratory studies to further identify information needs on each of the HCP species. In marine environments, existing data indicate that responses to suspended sediments are highly species-specific, with some species having lethal effects at several hundred parts per million (ppm) in 24 hours and others having no effect at concentrations above 10 parts per trillion (ppt) for 7 days. Studies on East Coast species have identified lethal suspended concentration levels, and Newcombe and Jensen (1996) developed a predictive model for defining lethal and sublethal fish injury threshold levels for suspended sediment concentrations. However, threshold studies (single-event as well as cumulatively) are lacking for the temporary impacts of suspended sediment levels specific to dredging in Pacific Northwest marine, lacustrine, and riverine environments (Nightingale and Simenstad 2001a).

The following information needs are also considered data gaps:

- Comprehensive data on the spatial and temporal distribution of spawning, rearing, and migration behaviors of HCP species to determine and assign dredging work windows on a site-specific basis have not been compiled.

- Cumulative thresholds associated with dredge-induced changes in salinity intrusion and other critical physicochemical processes in marine environments have not been identified.

- Recovery capability for HCP species that may be at risk of impacts from temporary exposure, chronic exposure, and cumulative thresholds associated with dredging in marine, lacustrine, and riverine environments in early life-history as well as adult stages are not fully understood for many HCP species.

- Recolonization capacities, after temporary, chronic, or cumulative thresholds are reached, of HCP species and the species endemic to those habitats (in marine, lacustrine, and riverine environments) that are important to their growth and survival are not yet understood.

- Temporary, chronic, and cumulative effects associated with nighttime lighting from dredge equipment (during construction as well as during operations following construction) have not been comprehensively investigated. The role of lighting in attracting predator species to affected sites is not fully understood.

- The magnitude and duration of noise associated with dredging operations have not been evaluated. Additional research on fish responses to noise is needed. This information is needed to evaluate potential noise impacts on HCP species.

- Fish behavior responses to dredging-related turbidity plumes of different extents are not yet understood.

10.1.1.4 Ambient Light Modifications and Artificial Light

Species-specific sensitivity to ambient light modification is a data gap for most HCP species.

- Ambient light modification is a likely stressor for many species in nearshore lacustrine environments and may also pose risk in marine environments. However, as juvenile sockeye salmon and steelhead are more typically found farther from shore, the effects of shading are less clear; therefore, the impact potential in the marine environment is uncertain.

- Information is needed on the temporary, chronic, and cumulative impacts of ambient light modification on HCP species in marine, riverine, and lacustrine environments.

- Extensive gaps exist in our understanding of how artificial light impacts aquatic organisms. Impacts to fish resulting from artificial light are often related to changes in nighttime behaviors such as migration, activity, location (Nightingale and Simenstad 2001b), and potentially schooling behavior in juvenile salmonids (Ali 1959, 1962, in Simenstad et al. 1999).

- Further studies on the qualitative effects of predator/prey relationships associated with artificial light, and investigations focused on the consequences of behavioral changes in aquatic organisms in a natural environment, are necessary to better understand the impacts associated with nighttime artificial light.

10.1.1.5 Channel Dewatering and Handling Fish and Shellfish

Few studies have compared the susceptibility of various fish and macroinvertebrate species to different types of handling techniques. More information comparing the susceptibility to injuries associated with these types of techniques is needed to identify potential take for these species. Training and minimum qualifications for personnel performing fish capture and handling (particularly electrofishing) are also needed to define standard protocols that would minimize risk of take.

Most of the studies on the effects of fish handling have been performed on electrofishing. Electrofishing effects have been conducted on adult fish greater than 12 inches in length (Dalbey et al. 1996). The relatively few studies that have been conducted on juvenile salmonids indicate that spinal injury rates are substantially lower than they are for large fish. Only a few recent studies have examined the long-term effects of electrofishing on salmonid survival and growth (e.g., Ainslie et al. 1998, Dalbey et al. 1996).

Little research has been conducted on the effects of dewatering and fish capture and handling on nonsalmonid HCP species. Injury frequencies reported for specific species are highly variable among and often within investigations and sometimes appear to be contradictory. Differences in rates and degree of injury, especially between investigations, are often difficult to attribute to species, fish size, fish condition, environment (including water conductivity and temperature), field intensity, or other current or field characteristics. Still, most existing data support Salmonidae as the fish taxon most susceptible to electrofishing injury (Snyder 2003).

More directed research is necessary to understand the risk of take resulting from channel dewatering and fish handling.

Exposure of the giant Columbia River limpet and great Columbia River spire snail to work area dewatering is possible, but sensitivity to this stressor is a data gap so the potential for take is unknown.

10.1.2 Hydraulic and Geomorphic Modifications Data Gaps

Finlayson (2006) identified five areas where additional research pertaining to physical nearshore processes is needed:

(1) characterizing the role of historical morphology;

(2) identifying tide-level controls on littoral phenomena;

(3) further development of existing littoral transport models;

(4) improved characterization of the role of extreme events in shaping low-energy, mixed-sediment beaches; and

(5) further testing and adaptation of numerical wave models for fetch-limited environments.

No research has been conducted to study submarine and intertidal groundwater n Puget Sound. It is clear from work elsewhere that such flows are crucial in sustaining nearshore ecosystems (Gallardo and Marui 2006); however, their role on the nearshore environment throughout Puget Sound is virtually unknown (Finlayson 2006).

10.1.2.1 Channel Hydraulics

Some studies that address the effects of HPA-permitted structures on habitat features such as scour or sediment composition, and studies that address the effects of changes in habitat features on potentially covered species exist. There are few case studies demonstrating quantitative impacts on animals or their habitat. The existing studies are often of limited use because they focus on “legacy” effects, i.e., impacts that occurred because of practices that are rarely, if ever, authorized under current regulations.

Nearly all studies that specifically look at impacts to potentially covered species address only impacts on salmonids listed under the ESA (i.e., Pacific salmon and bull trout). Some studies address effects on resident salmonids, sturgeon, lamprey, or mussels, but the literature is largely barren for all other potentially covered species. For many potentially covered species, the literature does not provide sufficient information to estimate how a given alteration in physical habitat might affect the species, because their life histories and habitat requirements are imperfectly understood. For such species, which include most potentially covered warm-water fish and invertebrate species (except mussels), this lack of information makes it difficult to estimate take potential.

10.1.2.2 Littoral Drift

Littoral drift cells can change over time with natural and human-caused alterations in shoreline configuration, sediment sources, and other variables. Mapped shoreline sediment sources and the location and direction of littoral currents and drift cells should be updated periodically to help users avoid adversely affecting important aquatic habitat characteristics and the potentially covered species that depend on them.

10.1.2.3 Substrate Modifications

The literature on substrate modifications is limited. A large data gap exists on the effects of substrate modifications in both freshwater and marine habitats on HCP species. There are two areas where research, particularly as it relates to the protection of aquatic resources and managing future development, needs to be performed:

- **Cutting-edge technique effectiveness assessments**. Current practical knowledge exists to protect shoreline areas through the implementation of innovative (cutting-edge) engineering alternatives. These cutting-edge alternatives can provide the desired degree of infrastructure protection while restoring physical processes, habitat features, and/or ecological functions. Unfortunately, while technically feasible, the effectiveness of these techniques has not been fully tested through the implementation of prototype projects.

- **Peer-reviewed monitoring of constructed restoration efforts**. Habitat restoration activities have been undertaken in many nearshore settings throughout western Washington over the last 30 years. While some have been large and monitored (Cheney et al. 1994; Carney et al. 2005), most have a tendency to be for small properties with no monitoring. Even when monitoring has been performed, it is generally used more as a design tool, rather than to evaluate the efficacy of the restoration activities to restore the targeted species (Carney et al. 2005).

Most studies of substrate changes have examined changes in a hydraulically active environment. Hydraulically passive environments are mainly deep marine and deep lake environments, where substrates are seldom altered except by point and linear structures such as pilings. Relevant studies focus on the marine environment. No data were identified as applicable to lake environments, where the potentially covered species include sturgeon and, to a lesser degree, suckers and mature salmonids. Conducting interviews and reviewing agency documents might provide further detail on the impacts of structures in hydraulically passive environments, but seems impracticable in view of the small risk of incidental take associated with such structures.

10.1.3 Riparian Vegetation Modifications Data Gaps

Most of our understanding of the role of riparian and streamside vegetation as a mediator of instream habitat condition has grown out of concern over its role in providing salmonid habitat. Although the reviewed literature addresses many ecosystem functions affected by riparian vegetation, such as shading, large woody debris recruitment, and allochthonous nutrient inputs, there is little discussion of how these changes may affect species other than salmonids. Knutsen and Naef (1997) indicate that nutrient inputs from riparian vegetation are important for suckers, whitefish and minnows, which feed directly on such detritus. Riparian habitat is also important for terrestrial wildlife.

10.1.3.1 Marine Riparian Vegetation Modifications

Although the functions of freshwater riparian vegetation have been identified for riverine systems, exploring and defining the functions of marine riparian vegetation are ongoing. There is reason to believe that marine riparian vegetation provides similar functions to riparian vegetation adjacent to freshwater habitats; however, the extent and nature of those functions are not fully understood (Desbonnet et al. 1995; NRC 2001; Brennan and Culverwell 2004). The following information needs are outstanding:

(1) understanding the specific nature and function of riparian habitat elements along marine shorelines;

(2) the dependence of HCP species on riparian marine and freshwater habitat functions;

(3) the dependence of HCP species on marine allochthonous inputs; and

(4) the cumulative and synergistic effects of riparian and shoreline removal.

10.1.4 Aquatic Vegetation Modifications Data Gaps

HCP species-specific data gaps with regard to aquatic vegetation include the following:

- Dependency of Pacific and river lamprey as well as northern abalone on aquatic vegetation.
- Effect of diminished habitat complexity due to aquatic vegetation modification on the giant Columbia River limpet and California floater (mussel).
- Of the potentially covered species, current data have shown a clear and consistent dependence on freshwater aquatic vegetation only for the Olympic mudminnow, although it is expected that freshwater aquatic vegetation is important for other potentially covered species as well, which is why this is identified as an important information gap.

10.1.4.1 Marine and Estuarine Aquatic Vegetation

Numerous significant data gaps preclude a clear understanding of how human activities cumulatively impact aquatic vegetation in marine and estuarine waters. Relatively little work has been done on macroalgae. For eelgrass, the following gaps are particularly significant (Jones and Stokes 2006):

- Factors governing the extent of eelgrass coverage, including local and large-scale changes in eelgrass coverage, are just beginning to be researched (Dowty et al. 2005).
- How large-scale changes in eelgrass cover resulting from HPA-permitted structures vary in conjunction with other large-scale changes, such as climate variability, has not been determined.
- More research is needed to determine the causes of local declines in eelgrass coverage observed in Washington State (Dowty et al. 2005).
- It is not known how strongly many potentially covered species depend on eelgrass. For instance, young salmon forage extensively in eelgrass, but foraging habitat may not be a limiting factor for juvenile salmon in Puget Sound (Haas et al. 2002).
- Much human impact on eelgrass and macroalgae takes the form of habitat fragmentation, but although such fragmentation is in principle an adverse impact, it remains unclear just how that impact is delivered to affected species (Haas et al. 2002).
- Understanding of the carrying capacity of Puget Sound for juvenile salmon, including food limitation thresholds is lacking.
- It is not known what the minimum patch size and connectivity elements needed for littoral vegetation to function as a prey source for HCP species are.

10.1.4.2 Freshwater Aquatic Vegetation

It is not known at what point the cumulative impact of HPA-permitted activities on aquatic vegetation becomes significant to most potentially covered freshwater species. Most of these species are thought to be affected by the loss of aquatic vegetation through indirect impact pathways that could vary from one location to another. To assess the relative merits of aquatic plant conservation and mitigation measures, the importance of aquatic vegetation in different systems and for all of the potentially covered species needs to be better understood.

10.1.5 Water Quality Modifications Data Gaps

There is still much work to be done to understand the impacts of suspended sediment and turbidity on potentially covered species. Most of the reviewed literature discussed impacts only with respect to salmonid species. Many of the studies were conducted in the laboratory in the absence of complex interactions that occur in natural systems. While the laboratory work is useful for describing interactions around which a study has been designed, additional field data would help to verify laboratory-derived conclusions. In addition, many data gaps that were identified by Bash et al. in 2001 still appear to be gaps. This includes:

- a lack of background water quality data for most waters in Washington,
- exposure thresholds for sublethal effects,
- the effects of short-term sediment pulses,
- species responses to varying sediment particle sizes and shapes, or patches of increased turbidity, including responses to reduced light,
- The effects of turbidity and suspended sediment on freshwater and marine HCP species and habitats,
- the effect of fine sediment deposition on hyporheic mechanisms, and
- how these affect habitat quality and quantity.

This information would help in estimating the potential impacts of aquatic projects by providing a more comprehensive impact analysis in the context of existing conditions and species response thresholds to suspended sediment exposure.

A particular data gap is how HCP species are affected by resuspension and transport of contaminated sediments (Michelsen et al. 1999). Many data gaps exist with respect to the potential for treated wood applied to aquatic settings to impact potentially covered species. Information is needed regarding creosote-treated wood effects on fish and shellfish in riverine and lacustrine environments. The chronic and cumulative effects of copper-treated wood in marine, riverine, and lacustrine environments are a data gap. Little work has been done to evaluate the potential impacts of treated wood applications in large projects on water quality and sediment and dose responses of potentially covered species to PAH and metals concentrations in water and sediment (Poston 2001). Poston (2001) reported a lack of knowledge on bioaccumulation and pathways of exposure of potentially covered species to PAHs and metals, as well as microbial and physical degradation processes of PAHs and metals. These processes are still not well described in the literature. Recent work has called into question the reduction in PAH leaching rates achieved by current BMPs for creosote treatment (Poston 2001). This information would allow for better estimates of take.

10.1.6 Ecosystem Fragmentation Data Gaps

10.1.6.1 Habitat Effects on Species

One of the biggest gaps in the literature is information that directly relates habitat changes to fish productivity (Bolton and Shellberg 2001). This information is difficult to collect because multiple factors may simultaneously influence the overall productivity and survival of fish species. However, this type of information is crucial to understand to minimize impacts on HCP species from HPA-permitted activities.

Although it is recognized that lost-opportunity impacts must be mitigated to achieve no loss of habitat (WDFW 2003), currently there are no tools for universal and consistent application of the concept.

10.2 Data Gaps Identified for Specific Activities and Sub-Activities

10.2.1 Bank Protection Data Gaps

There is an overall need for controlled, hypothesis-based studies directed at documenting and understanding the biological impacts of bank protection structures and activities to estuarine, marine, and freshwater ecosystems, particularly the effects associated with the structures both before and after impacts occur. Most current knowledge is based on anecdotal observations after the fact or those collected intuitively over time. Specific study needs include:

- Studies on the magnitude of the loss of salmonid food resources caused by bulkheading.
- Studies developing quantitative, comparative understanding of the effectiveness and habitat impacts of hard versus soft bank protection approaches/technologies.
- Information on long-term and cumulative habitat effects or relative benefits to biota for biotechnical approaches.
- Studies quantifying construction-related impacts related to specific bank protection activities (such as turbidity).
- Studies developing information on bank/shoreline morphology related to bank structures, such as:
    - Accurate estimates regarding the rate of marine beach erosion and accretion in the presence of bank structures, including both seasonal and long-term effects.
    - Effect of marine bulkheads specific to wave reflection and erosion of the upper beach.
    - Role of marine log structures in attenuation of energy at the shore.
    - Role of marine log bank protection structures in recruiting and retaining sediment and naturally occurring driftwood.

  - Differences in sediment transport at unarmored versus armored shorelines/banks and in areas with and without naturally occurring wood debris.

- Basic understanding of nearshore and bank ecosystem functions (e.g., roles of marine riparian vegetation, impact of LWD reductions ecosystem-wide); this will help to support the rationale for installing, leaving undisturbed, or enhancing certain existing natural shoreline features.

- More specific information on migration and movement requirements of non-salmonid potentially covered species related to banks and bank protection structures; most research has focused on salmonids in this regard.

- Studies investigating effects of bank protection on predation, feeding behavior, and prey production for covered species; very few studies document the links between specific bank protection types and behavior/diet of shoreline-associated species.

- Studies investigating linkages between bank protection project impacts and the context in the watershed and nearby upland systems.

- Information on how changes in habitat opportunity or capacity change with addition of bank protection and whether and how these affect biological resources on a landscape scale.

- Predictive cumulative impact tools that model the potential effect of armoring on specific sites as well as systems. A possible approach is to focus on floodplain disconnection by using historical aerial photography. Photo-interpretation of bank protection structure locations and corresponding side channel and high-flow channels at each time step could provide insight on the relationship between those parameters as well as stream length (as an indicator of amount of habitat available). Potentially, such an analysis could demonstrate whether disconnection of key sediment sources or river reaches had an inordinate impact on floodplain connectivity.

- Maps and updates based on existing databases and inventories that:

  - Illustrate historical and current channel and/or shoreline alignments

  - Determine/prioritize critical areas for protection or restoration

  - Identify ecosystems that are most at risk to cumulative impacts.

- Monitoring studies (short- and long-term) confirming that BMPs and conservation measures have had the desired effect

- Objective, post-project evaluations to maximize opportunities to learn from past experience and improve upon future design

- Summary/collection of information on process and outcome for use of adaptive management related to bank protection
- System for tracking and evaluating impacts on watershed level

Schmetterling et al. (2001) noted four areas where research on construction and maintenance of shoreline protection is lacking:

(1) quantifying the habitat availability and quality of riprap;

(2) correlation of the effects of riprap banks on salmonid density in the absence of other dependent variables such as diking, channelization, and watershed land use;

(3) comparative studies on the use of riprap and alternative “soft” techniques, such as the integration of natural materials; and

(4) the cumulative effects of numerous bank-hardening projects at the watershed level.

10.2.2 Channel Modifications Data Gaps

The cumulative impacts of multiple bank protection projects on channel processes and morphology is a significant data gap.

10.2.2.1 Dredging

Although the sources of turbidity generation due to dredging are well known, the connection of this source to a measurable biological response is a crucial data gap. Adequate data do not exist to quantify the biological response in terms of threshold sediment dosages and exposure durations that can be tolerated by various organisms. The existing data indicate that responses to suspended sediments are highly species-specific, with some species having lethal effects at several hundred parts per million (ppm) in 24 hours and others having no effect at concentrations above 10,000 ppm for 7 days. Studies on east coast species have identified lethal concentration levels, and Newcombe and Jensen (1996) have developed a predictive model for defining lethal and sublethal fish injury threshold levels for suspended solids concentrations in streams and estuaries. However, threshold studies for the temporary impacts of suspended sediment levels specific to aquatic environments in the Pacific Northwest are lacking. Additionally, although dredging of drainage channels for agriculture is widespread throughout the state, the extent of the impacts and effects on HCP species remains essentially undocumented.

Data regarding benthic recolonization after dredging is limited and only suggested one vector (i.e., invertebrates burrowing up through sediments) to explain significantly higher benthic invertebrate recolonization after dredge disposal. Additional studies have likely been completed to provide a clearer understanding of effects from dredge disposal.

10.2.2.2 Gravel Mining and Scalping

Although considerable advances have been made in the understanding of fluvial geomorphology and aquatic and riparian ecology in recent decades, there are still relatively few studies directly addressing the impacts of gravel mining in its various forms (e.g., wet and dry pit mining, bar scalping) and ecological restoration after mining. Most of the case studies of the geomorphic effects of mining have involved large extraction rates over a decade or more, resulting in large, measurable changes in channel form (Norman et al. 1998). Studies of long-term, indirect, and cumulative effects of mining up the food chain stand as a data gap. Such studies designed to measure mining-induced changes would require the collection of baseline data, which is seldom performed prior to gravel mining activities.

Food-web impacts of gravel mining are not well understood. Predation on juvenile salmon by introduced warm water species (such as those that thrive in the artificial habitats created by floodplain pits) has been documented in California, but no such studies are known to have been undertaken in Washington. The food-web implications of disrupting or eliminating shallow gravel riffle habitats, and reducing the abundance of large woody debris in the channel as a consequence of instream mining, have not been directly measured in the field.

10.2.2.3 Sediment Capping

When compared with dredging, sediment capping is a relatively new practice and consequently there are a number of data gaps concerning the impacts on HCP species. Nearly all of the literature regarding sediment capping is oriented toward the physical and biogeochemical properties of sediment caps and the impact of capping on benthic macroinvertebrates. Consequently, data gaps exist concerning the use of capped areas as fish nesting and foraging habitat. Despite this data gap, insight into how capped areas may be used by fish can be found by reviewing the information which exists regarding fish colonization of nourished beach habitat.

There is substantial anecdotal evidence that forage fishes use placed materials for spawning. For instance, a beach nourishment project in Silverdale Waterfront Park continues to be used by surf smelt. Shorelines cut into man-made fill in Commencement Bay have also been designated forage fish spawning areas (Penttila 2007). Developing this anecdotal evidence into a peer-reviewed article should be a target of future research considering the novelty and applicability of the work to both beach nourishment and sediment capping projects.

Additional studies regarding cap longevity would also be useful. As more sediment caps are applied and studied, our knowledge of how capping materials and techniques affect project longevity will improve. With this knowledge, a more accurate assessment of impact on biota can be made.

There is substantial evidence that invertebrate communities can rapidly recolonize an area after sediment capping, but all of this research has been conducted in marine waters. Capping of nutrient-laden sediments in small lakes practiced primarily in Japan (see Palermo et al. 1998) may have different impacts on invertebrates. Lateral recolonization, which has been found to be a factor in marine waters (Qian et al. 2003), may occur more rapidly near tributary inlets, but as of yet there is no evidence to indicate that recolonization would occur at different rates in marine, lacustrine, or riverine environments.

10.2.2.4 Channel Creation and Alignment

Restoring the ecological integrity of rivers, streams, and tidal channels requires an understanding of pre-disturbance conditions at the time of earliest Euro-American settlement in the mid-19th century. Unfortunately, documentation of these conditions is limited. Archival investigations, field studies, and geographic information systems and remote sensing analyses (such as those undertaken by the Puget Sound River History Project at the University of Washington) are needed to understand the historical landscape to address regional problems of resource management, restoration, and planning.

Straightening and dredging of drainage channels for agriculture is widespread throughout the state but remains essentially undocumented in its extent or impacts. The need for additional studies of this common practice stands as a data gap for channel creation and alignment.

10.2.3 Fish Passage Structures Data Gaps

Data gaps specific for fish passage structures include:

- Knowledge of the movement patterns of HCP species at different life-history stages relevant to the definition of design flows for fish passage.

- Understanding of the situational limits of different design approaches (e.g., stream simulation).

- Passage requirements of nonsalmonid HCP species and knowledge of the behavioral and physiological limits on the swimming ability of HCP species, sufficient to guide definition of hydraulic design criteria.

- Upstream movement requirements of HCP invertebrate species: The freshwater HCP invertebrate species vary in terms of the mechanisms they use to influence dispersal in flowing water environments appropriate for the fish passage activity type. Unionid mussels rely on host-fish species to disperse their parasitic larvae to upstream environments. These species have also been shown to disperse upstream for short distances by crawling along the bottom using their muscular foot and byssal thread attachments (Vaughan 2002). Other HCP invertebrate species, such as the giant Columbia River limpet and great Columbia River spire snail, crawl along hard substrates and are theoretically capable of navigating upstream for short distances. The degree to which fish passage subactivity types may help or hinder these dispersal mechanisms and the ramifications for population health are an area requiring additional study.

10.2.3.1 Culverts

In WAC 220-110-070 as well as in current design guidance documents (e.g., Bates et al. 2003), assumptions are made to define the period of year during which fish passage is required, based on the species that are expected to inhabit a stream. Many culverts present only a temporary barrier to fish passage or are barriers to juvenile and resident fish only. However, the significance of such barriers on fish movement in the field has not been thoroughly investigated, particularly where the occurrence and timing of fish movement are poorly understood. There are gaps in knowledge regarding the movement patterns of various salmonid species and life-history stages (particularly of resident and juvenile anadromous salmonids) in small stream channels. Nonsalmonid species are less well understood in many cases. Finally, there are key data gaps relative to the design requirements necessary for structures that maintain performance over time.

With regard to the migration requirements of fish, it is known that volitional movement can vary greatly among species, lifestages, habitats, seasons, and years (Gowan et al. 1994; Kahler and Quinn 1998; Kahler et al. 2001). Research on several critical fish passage related topics is currently in progress or has recently been completed. However, it may be difficult to translate this information into meaningful guidance because the research is typically focused on a single species and may not adequately reflect the requirements of a broad range of HCP species; therefore, a number of related uncertainties may remain.

The combined effects of culvert length, material selection, and the utility of baffles and similar elements lead to significant uncertainty when applied across a broad range of species. For example, the role of boundary layer turbulence is known to affect the ability of fish to pass through culverts, but the specifics of these effects are poorly understood and considered to be a data gap in an earlier white paper (Kahler and Quinn 1998). Subsequent to Kahler and Quinn’s (1998) review, research on the role of boundary layer turbulence has been studied by examining the swimming performance and behavior of juvenile salmon in test beds (Pearson et al. 2006). Research has demonstrated that juvenile salmonids are able to navigate culverts at higher average flow velocities than would be expected from standard swimming performance curves (Kahler and Quinn 1998; Pearson, Southard et al. 2006; Powers and Bates 1997). They do so by exploiting low-velocity zones in the turbulent boundary layer and other areas of hydraulic complexity. The ability of other fish species to similarly exploit these low-velocity zones in many cases is poorly understood. This creates the potential to over- or underestimate the passage requirements of nonsalmonid fish species.

Despite this directed research, several additional data gaps on issues relevant to design guidance remain (Bates et al. 2003; Pearson et al. 2006). Relatively few data are available on the passage requirements of smaller nonsalmonid fish species, such as dace and chub. While Katopodis (1992) has noted that swimming performance tends to be generally similar across species relative to size when grouped by swimming physiology, this may not fully account for the effects of hydraulic complexity in the passage environment. The flow conditions that constitute an appropriate upper design flow limit for juvenile fish passage are poorly understood for most HCP fish species, as well as other aquatic and semi-aquatic species. The movement of aquatic invertebrates and their passage requirements have received even less study (Vaughan 2002) and are a data gap for the HCP invertebrate species exposed to this activity type.

Current WDFW guidance emphasizes the use of “geomorphic designs” for new and replacement culverts (specifically, structures designed following the no-slope and the stream-simulation options). The intent of these designs is to produce a culvert that allows a broader range of geomorphic processes to function across a broad range of channel types. In the case of the no-slope option, the range of slope and sediment transport conditions over which it can provide effective fish passage remain uncertain. This is particularly true in higher gradient systems and systems with less-mobile bed conditions. It is generally intended for use in low-gradient systems with higher rates of sediment transport.

The development of stream simulation criteria and design procedures is recent. Most of the experience with the method is in mountainous streams. Uncertainties remain about the efficacy of specific criteria and design guidance across a broad range of channel types and hydro-geographic regions. Additional research would fine tune the criteria and guidance, broaden the application, and inform designers of appropriate criteria for unique situations.

Even fish passable culverts may impose ecosystem fragmentation effects on terrestrial and amphibian wildlife species. This may potentially result in indirect ecosystem-level effects on HCP species that are complex and difficult to predict. Even in the absence of complete understanding, it can generally be assumed that designs that promote more natural migration and dispersal behavior are desirable over those that produce barrier conditions. Ongoing research on this subject at the U.S. Forest Service may produce information that will improve guidance in the future (Bates et al. 2008).

10.2.3.2 Fish Ladders/Fishways

Additional study of the factors influencing passage of nonsalmonid HCP fish species is necessary to develop improved design criteria. For example, knowledge of juvenile fish jumping ability is necessary to design for the maximum allowable hydraulic jump (i.e., vertical drop) within a fishway. In this regard, recent research on juvenile coho salmon jumping ability has determined that jump heights exceeding 2.5 times the fish length block passage of a high percentage of individuals (Pearson et al. 2005). This information provides useful design guidance for salmonids, but these findings may not apply to nonsalmonid species. For example, recent research has documented low Pacific lamprey passage efficiency through fish ladders in the Columbia River system, but the specific fishway design factors that support or limit successful passage are unclear (Moser et al. 2002).

10.2.3.3 Roughened Channels

Roughened channels are outwardly simple structures, but in reality the design parameters required to construct a channel that will function as intended over time are demanding and complex. Improper design may lead to unintended perturbations in hydrologic, geomorphic, and riparian conditions that can cause a number of undesirable indirect effects on HCP species. Definitive design guidance for this type of structure is currently lacking and must be considered a data gap.

10.2.3.4 Weirs

Impacts from fish passage weirs are in many cases similar to those for small dams, which have been well documented and a topic of research for decades. In general, there are no major data gaps that exist. However, little research on the hyporheic zone has been conducted in highly altered and degraded fluvial systems (Bolton and Shellberg 2001). While the physical effects of weirs on the environment are well understood, there is a lack of information on the impacts from weirs on specific HCP species. When combined with limited understanding of species-specific migration behavior, a lack of knowledge of the physiological and behavioral passage limitations of all potentially affected HCP species presents the likelihood of unforeseen undesirable consequences. This further suggests that definitive guidance is lacking for the design of structures that function as intended across all species.

10.2.3.5 Trap and Haul

The effects of alteration of migratory corridors on subject fish species is an area of limited but increasing study. Alteration of migratory corridors may have unintended effects on homing selectivity that are undesirable for long-term evolutionary fitness; therefore, this is an area deserving of further study.

10.2.4 Fish Screens Data Gaps

Key data gaps remain in the following general areas:

1. Knowledge of the movement patterns of HCP species at different life-history stages relevant to the development of design and operational guidance for fish screens is a data gap. This is relevant to fish screen operation, but it encompasses an issue driven more so by water removal from the system. Essentially, fish screens can impose operational effects only when an intake or diversion is active, meaning that both fish screen design and managing the timing of water withdrawals are two available tools to limit adverse effects on HCP species. A better understanding of the range of species and life-history stages likely to occur, as well as the timing of their occurrence, is necessary to select the most appropriate screen design.

2. Knowledge of the behavioral and physiological limits on the swimming ability of HCP species, sufficient to guide definition of screen design criteria, particularly for nonsalmonid species is a data gap. Available data show that screen effectiveness may vary by species, depending on factors such as swimming physiology, behavior, sensitivity to bypass entrainment, the unintended effects of stimuli that might be used to guide them toward or away from intakes, and other factors. Fish screens designed to protect salmon may also be effective at protecting fish species with similar swimming physiology, but may not be protective for weaker-swimming species such as juvenile lamprey (Close et al. 1998). Most screen research in Washington State has focused on protection of salmonids, resulting in criteria that may not provide adequate protection for other native species. However, at least some research is available on the response of nonsalmonid

species to fish screens. Better understanding of the tradeoffs between screen function and species protection will allow for more effective design and operational guidance.

3. Useful design criteria across the range of environment types and conditions where screens are employed are lacking. While uniform design guidance would be desirable, the bulk of available research indicates that it is impractical to develop guidance applicable for all environments and uses. The factors that determine the most appropriate screen design for a given situation are highly dependent on both the type of withdrawal (intake or diversion) and site-specific conditions. For example, an effective screen design for an agricultural diversion must consider a number of competing factors, such as the diversion flow rate, flow conditions and variability of the source body, the expected volume of naturally transported debris that must be cleared or passed, and the swimming physiology and sensitivity of the full range of HCP species that occur in the affected environment. This presents a complex set of demands that are not easily addressed by uniform design guidance. This suggests a need for a broader set of assessment steps that can be used to develop site-appropriate designs.

4. Clear demonstrations that fish screens are an effective tool for protecting the productivity and diversity of HCP species (relative to other conservation measures) are lacking. It is not clear that fish screens provide a conservation benefit for all species and all circumstances. While the issue of fish entrainment in industrial and power plant water intake systems is effectively mitigated by fish screens (Goodyear 1977; Hadderingh 1979; Taft and Mussalli 1978; Travnichek et al. 1993), the effectiveness of off-channel screen designs has been less well studied in agricultural applications. Moyle and White (2002) and Moyle and Israel (2005) conducted a broad review of published literature and found that despite policy directives dictating the widespread implementation of fish screens on agricultural diversions, relatively few studies have attempted to evaluate their effectiveness at maintaining or increasing population abundance and productivity. The literature suggests that this lack of evaluation is typical throughout the western United States, despite millions of dollars spent annually on fish screen installation and maintenance (Moyle and Israel 2005). While it can be argued that these studies are unnecessary because the conservation benefits are clear, it may be useful to consider more directed study to identify and prioritize the diversions with the greatest impact on fish populations, and to determine which types of screens provide the best protection for HCP species likely to be exposed.

5. Effects of off-channel diversions when the area between the point of diversion and the screen are quickly dewatered at the end of irrigation season are lacking. The magnitude of the negative effects are unknown. Lamprey are known to use these areas, and may be impacted by sudden dewatering. Effects might be mitigated by an incremental reduction of flow over a number of days and salvage of fish from residual pools.

10.2.4.1 Fish Screen Operations – Entrainment and Impingement

Entrainment and impingement risk is a subject of continuous and ongoing research as fish screen design advances. Despite a large body of existing research, much of the information necessary to protect the broad range of HCP species potentially exposed to screens from these stressors remains unknown. For example, the bulk of available research has focused on fish with subcarangiform swimming physiology (side-to-side undulation of the posterior one-third to one-half of body length), a characteristic of most, but not all, HCP fish species. This research provides the primary base of information on swimming performance used to guide design. However, design criteria based on these data are not likely to provide adequate protection for weaker-swimming fish, specifically lampreys, with anguilliform swimming physiology (eel-like full body undulation). Even for well-understood species such as salmonids, several factors such as species, age class (i.e. size), condition, and water temperature can influence swimming performance in ways that are relevant to design. Sensitivity to injury or other adverse effects also varies between species. For example, Zydlewski and Johnson (2002) evaluated fish screens designed for anadromous salmon protection and found that while juvenile bull trout were frequently impinged on the screen, they were able to escape and were effectively passed downstream without apparent injury or adverse effects. In contrast, Swanson et al. (2005) and White et al. (2007) found that even limited screen contact caused stress and injury sufficient to lead to delayed mortality in delta smelt.

Many design criteria in common use today are based on untested theories (Bates 2008). Design criteria that should be subjected to further research and scrutiny include the following:

- The relationship between screen mesh size and approach and sweeping velocity for balancing debris-clearing effectiveness against impingement and entrainment risks

- The efficacy of widely used sweeping velocity parameters for guiding various HCP fish species and life-history stages across screens and into bypass systems

- Effects of nonuniform approach velocity on impingement risk

- Use of turbulence, light, sound, and other mechanisms to deter or guide fish

- Efficacy of various cleaning mechanisms relative to different types of debris (e.g., hydraulic eddy cleaners), and related risks to HCP species

- Investigation and development of new cleaning technologies, such as vortex separators, to continuously clear sediment from screen bays

- Optimization of bypass configuration for fish collection and flow management

- Appropriate bypass depths and velocity for fish protection and water management

- Screening designs for planktonic larval life stages including effects of impingement, handling, and release.

10.2.4.2 Fish Screen Design Effects on Hydraulic and Geomorphic Modifications

While fish screens are expected to have relatively modest hydraulic and geomorphic effects in comparison to the flow control structure (or other activity types) associated with the related water intake or diversion system, additional data and analysis describing the hydraulic processes affected by certain types of fish screens are desirable. Screen designs of potential concern include large, permanent in-channel structures capable of altering local hydraulic and geomorphic conditions in riverine, marine, and lacustrine environments. In certain circumstances, off-channel structures may also cause undesirable effects. Specifically, screens that require a significant component of remaining instream flow to operate a bypass system may cause hydraulic and geomorphic effects by encouraging vegetation encroachment. Additional research to identify the types of stream channels sensitive to these effects may be desirable.

10.2.4.3 In-Channel Screens

In-channel or end-of-pipe screen systems are relatively simple in design in comparison to off-channel structures, and their effects are more broadly understood. Data gaps related to this subactivity type primarily concern uncertainty about the presence of HCP species with sensitive life-history stages (e.g., small size, planktonic or weak swimming) that cannot be effectively protected by current fish screen designs. For example, flow and velocity requirements necessary to draw various life-history stages of salmon into bypass systems are not well known. Although current designs seem to be effective, they are not likely optimized for either fish passage or flow management because of a lack of empirical data.

This uncertainty can be addressed only by amassing available site-specific data or conducting the necessary research to understand the timing and distribution of sensitive life-history stages in relation to the desired operating parameters of the water intake system. This understanding can be used to set operational limits as necessary to overcome limitations in screen performance.

Guidance criteria for the siting, design, and operation of infiltration gallery screens are currently lacking. Additional research should be conducted to determine if this technology has practical utility and, if so, to identify appropriate uses and develop design criteria.

For additional information, see Rychetsy and Card (2000).

10.2.4.4 Off-Channel Screens

The off-channel screen subactivity type encompasses a number of screen designs that range from relatively simple to complex. The design requirements for these structures are highly site specific. Although generalized guidance can provide some basis for selecting an appropriate design, site-specific assessments and research are necessary to develop these designs fully.

A number of data gaps have been identified that–when addressed–could improve both the general guidance for species protection and an understanding of the limitations of certain screen designs. These include:

- ***Passage-related effects of fish screen designs:*** The potential for certain types of off-channel fish screens, specifically those with integrated bypass channels, to create attraction flows that unintentionally delay adult migration has been identified as an issue of concern from a design perspective by WDFW (2001a). However, empirical data necessary to provide clear design guidance on this subject are currently lacking. Similarly, screens with bypass channels must produce adequate sweeping flows to avoid delaying downstream migrant salmonids. While sweeping flow requirements are fairly well understood for salmonids and some other fish species, the needs of some HCP species (e.g., lamprey) appear to be less clear.

- ***Upstream movement requirements of HCP invertebrate species:*** The freshwater HCP invertebrate species vary in terms of the mechanisms they use to disperse in flowing water environments. The degree to which fish screens may help or hinder these dispersal mechanisms and the ramifications for population health are an area requiring additional study.

- ***The ecosystem fragmentation effects of screens:*** Certain off-channel fish screen designs may affect upstream and downstream fish passage by delaying migration, or imposing unintended selection pressures on affected populations. Water withdrawals may also affect the transport of organic material and woody debris. Fish screens with bypasses provide a conduit for some of the debris to return to the river. The extent of these effects, particularly the cumulative effects of multiple screens distributed across the landscape, are not clear. This is an area that could benefit from additional research. Given the site-specific nature of these effects, however, it may be difficult to produce results that lead to broadly applicable guidance.

10.2.5 Flow Control Structures Data Gaps

10.2.5.1 HCP Species-Specific Information

Besides the extensive research on salmonids, there is a general lack of information regarding the effects that flow control structures may have on most other HCP fish species. An exception is that several studies have been conducted examining sturgeon and dams. During a detailed literature review, little information on impacts on invertebrates was found.

10.2.5.2 Dams

Impacts from dams have been well documented. Dams have been a topic of research for decades. Minor data gaps are as follows:

- little research on the hyporheic zone has been conducted in highly altered and degraded fluvial systems (Bolton and Shellberg 2001). As the understanding of these processes increases, studies will likely begin to focus more on the effects of land use and other human activities on surface–groundwater interactions.

- The effects of dam removal on aquatic species, their habitats, and ecological processes represent a data gap. Although there have been several studies on the ecological impacts of dam removal (Bednarek 2001), there is a general lack of post-removal data to document these changes. More specifically, dam removal data to date have focused on smaller dams, so the actual impacts from a large dam removal are often inferred. The future removal of two dams on the Elwha River (Washington) represents an opportunity to study the impacts of a large-scale dam removal.

10.2.5.3 Weirs

There is a lack on information on the impacts from weirs on specific HCP species.

10.2.5.4 Dikes and Levees

As with dams, little research regarding the hyporheic zone has been conducted in systems supporting dikes and levees. As the understanding of these processes increases, studies will likely focus on the effects of land use and other human activities on surface–groundwater interactions. In addition, while a number of studies document changes to habitat after construction of dikes and levees on the landward side of the structure, more information is needed with respect to in-channel changes.

10.2.5.5 Outfalls

Limited information is available on the hydraulic and geomorphic modifications of outfalls and intakes and their direct impact on fish and invertebrates. Information on the effects of outfalls on riparian and aquatic vegetation and ecosystem fragmentation is scarce. In general, most studies of outfalls are related to water quality modifications, and these impacts are well documented.

10.2.5.6 Intakes and Diversions

Limited information is available on the hydraulic and geomorphic modifications of intakes and diversions and their direct impact on fish and invertebrates. Information on the effects of intakes on riparian and aquatic vegetation, hyporheic flows, and ecosystem fragmentation is scarce.

10.2.5.7 Tide Gates

In a review of tide gate operations in the Pacific Northwest, Giannico and Souder (2005) failed to find studies that examined the effect of tide gates on juvenile fishes, reporting that this represents a large data gap in our understanding of how these structures influence fish populations. Specific information is lacking on migration patterns of species that use habitats where tide gates occur (Giannico and Souder 2005). If detailed information on the behavior and movements of HCP species were better understood, then tide gates could be better designed to allow for increased fish passage.

There is a potential for a loss of LWD as the result of a tide gate. However, there is little information about how tide gates alter LWD transport and recruitment, which represents a potential data gap. In marine ecosystems in general, the influence of LWD on primary productivity is somewhat unclear.

10.2.6 Habitat Modification Projects Data Gaps

10.2.6.1 Beaver Dam Removal/Modifications

There has been considerable research regarding the use of beaver dam habitat and similar backwater habitat by fish and invertebrates. However, there are no studies which have specifically monitored fish and invertebrate populations both before and after a beaver dam removal. Impacts on these species must be inferred from studies which have monitored the impacts of man-made dam removal projects on aquatic species. The impact of nonlethal beaver management strategies on pond habitat and fish passage is even less well understood. It is assumed that *(removing?)* these structures would have little impact on ecosystem dynamics while simultaneously controlling upstream flooding, but there have been no studies to support this assumption. There have been no studies on the impact of dam removal (beaver or otherwise) on any of the HCP invertebrate species.

10.2.6.2 Large Woody Debris Placement/Movement/Removal

The impact of LWD placement and removal is a thoroughly researched topic but there are several data gaps that still exist. The largest data gap regarding LWD placement/movement/removal is that little research on LWD in marine environments has been conducted.

- There has been little research on the importance of wood in supporting beach structure and connectivity between estuarine environments.

- Many shorelines in the Puget Sound area contain considerable wracked wood in the supertidal zone (Sobocinski 2003) which may serve to reduce shoreline erosion and protect the sediments which are the foundation for the shallow water habitat (Herrera 2005). Additional research is needed before the impact associated with wood modification on beaches can be assessed.

- In marine ecosystems, the influence of LWD on primary productivity is somewhat unclear. Supratidal food web dynamics are likely driven by both terrestrial and marine processes including (but not limited to) marine deposition of large wood. LWD along with wrack material and other organic debris can be a source for the detritus-based nearshore food web. Colonization of wrack by scavengers, infauna, and ultimately bacteria and diatoms, is an important process in maintaining energy exchange between the terrestrial and marine systems (Sobocinski 2003).

10.2.6.3 Spawning Substrate Augmentation

The impact of augmented salmonid spawning gravels on channel geomorphology and ecosystem dynamics is poorly understood. There have been a limited number of studies regarding the ecological ramifications of spawning substrate augmentations, despite the fact that it is frequently done. There is no information regarding the impact that added gravel may have on riparian flooding. Increased flooding of riparian areas could potentially benefit stream biota by providing increased habitat access and exporting additional food resources to the channel. Conversely, increased flooding could endanger man-made structures within the floodplain and potentially import organic material which could degrade the permeability of the augmented gravels.

The goal of spawning substrate augmentation is to create quality salmonid spawning habitat which will result in an increase in redd density and fry productivity and survival. The research to date has reported conflicting results as to whether gravel augmentation actually increases redd density. Additional research is needed to verify that augmented sites have a higher carrying capacity than unaltered sites. Research to date has indicated that benthic dissolved oxygen levels are higher in augmented gravels and that egg survivorship is elevated in augmented gravels (Merz and Setka 2004). Thus, even if redd density does not increase in a restored site, fry productivity and survival may still be improved.

The potential benefit of spawning substrate augmentation for invertebrates is unclear. It is evident that initially organisms with limited motility will be buried and will perish, and it is also clear that benthic organisms have the ability to rapidly recolonize augmented reaches (Merz and Chan 2005). Increased benthic dissolved oxygen will, in theory, benefit mollusks but there have been no studies that have identified a net benefit to invertebrate populations after gravel augmentation.

10.2.6.4 In-Channel/Off-Channel Habitat Creation/Modifications

There has been a wealth of studies on the ramifications of in-channel habitat modification. Yet, due to the wide array of projects which are built in highly variable fluvial environments, the research results to date regarding the efficacy of such projects are mixed. The science of river restoration is constantly evolving and more studies are needed to quantify the effectiveness of projects built to the most rigorous and up-to-date standards. If indeed, these projects fail as frequently as projects built in years past (see Frissell and Nawa 1992; Merz and Chan 2005; Roni et al. 2002; Roper et al. 1998), then it can be assumed that channel rehabilitation efforts should be focused elsewhere (e.g. volume control, riparian restoration, off-channel habitat rehabilitation).

The most significant data gap related to in-channel and off-channel habitat modification is regarding the effect of the restoration on fish populations, because fish are mobile and are affected by impacts from outside the restored reach. Consequently, it is difficult to correlate alterations to a defined reach with fish population dynamics. Studies that monitor fish movement using different remote tracking technologies may be most useful in clarifying this issue. As tracking technologies improve, the question of habitat usage and restoration efficacy will be more definitively addressed.

10.2.6.5 Riparian Planting/Restoration/Enhancement

The research findings regarding fish response to riparian restoration are mixed. Increased riparian shading will reduce water temperatures (LeBlanc and Brown 2000; Opperman and Merenlender 2004) and this may benefit fish in thermally impacted reaches, but fish response to riparian vegetation planting/restoration has not been clearly defined. For instance, Bjornn et al. (1991) found that age-0 coho did not respond to either riparian vegetation removal or artificial cover creation in an Alaskan stream. Numerous researchers have monitored physical and macroinvertebrate response to riparian vegetation alteration (Fuchs et al. 2003; Sweeney et al. 2004; Teels et al. 2006; Wipfli 2005). These studies have found that partial riparian cover promotes the greatest macroinvertebrate abundance. It can be assumed that increased macroinvertebrate abundance will support a greater fish population, but more studies of fish response to riparian vegetation addition and removal are needed to definitively characterize the impact of riparian planting on fish.

There are no widely available studies regarding the impact of invasive vegetation removal on stream ecology. Vegetation removal can destabilize banks and increase stream temperatures. The removal of riparian invasives such as Himalayan blackberry is a common riparian restoration practice (Bennett 2007), and further studies are required to assess the impact of this and similar activities on both fish and invertebrates.

There is very limited information on marine riparian restoration efforts. Loss of riparian vegetation in marine environments has been identified as an important problem, but monitoring of marine riparian restoration efforts has not been done.

10.2.6.6 Wetland Creation/Restoration/Enhancement

The primary data gap regarding wetland creation is the impact of wetland creation on invertebrate species. It can be assumed that there will be no impact because in these types of restorations, habitat is not being degraded or eliminated but rather is being connected and/or augmented. Regardless, there is no available research to quantify the impact of riparian wetland creation/restoration on the HCP invertebrate species.

There is limited information as to what size riparian wetland would be most beneficial to the river-floodplain ecosystem. Most floodplain restorations are limited in scope and the resultant wetland is much less extensive then the natural historic wetland that once existed. Consequently, residence times are generally lower in restored versus natural riparian wetlands. There has been no research which has quantified how this lowered residence time may affect nutrient processing and carbon export from restored floodplains.

No studies have examined the impact of dike breaching on HCP fishes. Tidal flow over modified landscapes can produce fish traps, but this effect has not been described in the literature. The effects of nutrient loading from the inundation of former nutrient-rich (fertilized) agricultural lands have not been investigated.

10.2.6.7 Beach Nourishment/Contouring

There is substantial information about beach nourishment projects in exposed, sandy settings (Speybroeck et al. 2006). However, there is no peer-reviewed literature describing the restorative characteristics of beach nourishment projects for HCP fishes in coarse-clastic environments that are typical in Puget Sound (Shipman 2001). While there have been several reports of nourishment projects in Puget Sound (Gerstel and Brown 2006; Shipman 2001; Zelo et al. 2000), these “grey literature” reports have only cataloged the physical response of the beach to nourishment activities. These studies are useful for design purpose, but they have not addressed the core issue of whether nourished shorelines are actually used by species targeted by this activity (i.e., forage fishes).

Despite the lack of peer-reviewed work there is substantial anecdotal evidence that forage fishes do use placed materials for spawning. For instance, a beach nourishment project in Silverdale Waterfront Park, Kitsap County, continues to be used by surf smelt. Shorelines cut into man-made fill in Commencement Bay have also been designated forage fish spawning areas (Penttila 2007). Developing this anecdotal evidence into peer-reviewed articles should be the target of future research funding considering the novelty of the work and the large number of beach nourishment projects.

While there a number of peer-reviewed articles regarding the efficacy of beach nourishment in sandy settings (Speybroeck et al. 2006), none of these studies focus on the benefits or detriments to the HCP species. Many of the impacts that have been described in the literature relate to excess turbidity and deposition of fine sediments near nourishment project sites (Speybroeck et al. 2006; Wilber et al. 2003). It may be that these effects are less pronounced on the HCP species. The HCP species in the Pacific Northwest have adapted to environments where sediment concentrations are substantially higher than the tectonically benign east coast (Montgomery 2000).

There is essentially no information (peer-reviewed or otherwise) on the ecological impacts of beach nourishment on lakeshores. Only two peer-reviewed studies were found that examined the ecological impacts of freshwater beach nourishment. The results of these studies mirrored the work performed in the marine environment, with one study advocating nourishment for fish populations (Winfield 2004), while the other documented the loss of invertebrate species due to nourishment in the Great Lakes (Garza and Whitman 2004). Considering that numerous HPAs were authorized in 2006 alone for nourishment projects on lakeshores, more work is urgently needed on both the design parameters and ecological impacts of lakeshore nourishment projects.

10.2.6.8 Reef Creation

Much of the information collected in the Environmental Design of Low Crested Coastal Defense Structures (DELOS) program sponsored by the European Community is useful for identifying the impacts of reefs and the subsequent effects on fish and invertebrates. However, the relative lack of salmonids on the European continent limits the applicability of that work to the nearshore ecology of the Pacific Northwest.

The loss of fish that serve as food (forage fish) for salmonids has been studied, but relating the construction of an artificial reef to salmonid loss has not. Also, identifying the role that invasive species infestations (associated with the deployment of offshore rocky structures) can have on native species needs to be investigated (e.g., the abundance of urchins or macroalgae at the expense of salmonids).

The only research literature on freshwater reefs comes from the Great Lakes (Marsden and Chotkowski 2001; Meadows et al. 2005). However, it would be helpful to have a study similar to Marsden and Chotkowski (2001) to identify the ability of Washington freshwater reefs to attract invasive species.

There has been no work on the attraction of the HCP invertebrate species to artificial reefs.

10.2.6.9 Eelgrass and other Aquatic Vegetation Enhancement

Although eelgrass planting programs have been undertaken for nearly twenty years (Thom 1990), successful programs have only existed for a few years (Thom et al. 2005). Techniques and procedures for successful programs have been established (Thom et al. 2005); however, no program has yet to document the net gain to HCP species. The lack of data regarding the return of higher trophic species remains a large data gap.

10.2.7 Marinas and Terminals Data Gaps

In general, the thresholds for watershed and population size and the number of activities that must occur within a particular watershed to have a measurable cumulative impact are not yet established in the literature. These are needed to assess the effects of marinas/terminals on HCP species in a holistic approach.

In other regions of the United States, studies have documented the cumulative impacts on the nearshore environment (e.g., the Great Lakes [Meadows et al. 2005]. Impacts on numerous HCP species have been documented due to hydraulic and geomorphic modifications associated with marina development. Data and analysis describing the ecosystem processes affected by marina/terminal activities are needed.

Jones & Stokes (2006) reported that no data pertaining to substrate modification associated with marina/terminal structures were found on lake environments for HCP species.

10.2.7.1 Vessel Activities

Little is known about the impacts of marina/terminal vessel activities on HCP species. Although some work has examined the effects of vessel waves, sediment resuspension, and turbidity, these studies addressed salmonid species or cetaceans and not the other HCP species. Measurements incorporating the elements of repetitious exposure over time, effects resulting from numerous vessels, and large vessels idling and approaching and leaving terminal docks are needed to understand the potential effects on the HCP species occupying those habitats in marine, riverine, and lacustrine environments.

Given the large numbers of vessels typically associated with a marina and the large-sized vessels using terminals, the potential effects on fish and invertebrate growth, survival, and fitness in the vicinity of these maritime structures from these types of discharges could be significant. Information is needed on the temporary, chronic, and cumulative impacts of vessel operation, maintenance and discharges on HCP species in marine, riverine, and lacustrine environments.

No data that would allow quantification of the amount of habitat lost due to placement of footings located below the OHWL or MLLW associated with piers or ramps or temporarily disturbed each year as a result of the construction of overwater structures were identified. Such data would make it possible to improve estimates of take and cumulative impacts.

Recent work specific to identifying the impacts of marinas and terminals on migrating juvenile salmon along marine and lake shorelines has begun to address these information needs.

10.2.8 Overwater Structures Data Gaps

10.2.8.1 Shading

Significant gaps and uncertainties remain in knowledge about the impacts of overwater structures and shading on the aquatic environment and biota (Nightingale and Simenstad 2001b; Carrasquero 2001). Some of these gaps are basic to understanding the ecology and life history of potentially impacted species, such as those defining the extent and ecological dependence of shoreline habitat use by certain biota.

Since the publication of the two WDFW white papers cited above, a few studies have been completed regarding shoreline habitat use of aquatic biota.

- Toft et al. (2004) reported on fish distribution, abundance, and behavior in nearshore habitats along the marine shoreline of the City of Seattle.
- Tabor et al. (2006) studied nearshore habitat use by juvenile Chinook salmon in the Lake Washington basin.
- Southard et al. (2006) studied conditions for, and the significance of, avoidance of shoreline structures by migrating juvenile salmon in *Impacts of Ferry Terminals on Juvenile Salmon Movement along Puget Sound Shorelines*. This study supported other findings that identified shading of overwater structures as the mechanism for salmonid avoidance (Weitkamp 1982, Pentec 1997, in Nightingale and Simenstad 2001b; Shreffler and Moursund 1999) and recommended ways to minimize impacts of ferry terminals on juvenile salmonids.
- Haas et al. (2002) suggest that additional research is necessary to determine the thresholds at which epibenthic biota become affected by the shading of vegetation.

10.2.9 Shoreline Modification Data Gaps

10.2.9.1 Jetties

10.2.9.1.1 Marine Environments

Significant work has been performed on the various ways that shoreline modifications have altered the nearshore ecosystem, although much of this applies to shoreline hardening and human activities in general, not specific actions. Nearly all of the modifications associated with shoreline development can be attributed to jetties. Within this body of work, three areas have been identified that are relevant to the HPA process where data gaps remain:

- *Cutting-edge technique effectiveness assessments*. Current practical knowledge exists to protect navigational channels through the implementation of innovative engineering alternatives to jetties (e.g., engineered wood placement). These alternatives can provide the desired degree of infrastructure protection while restoring physical processes, habitat features, and/or ecological functions. The effectiveness of these techniques has not been fully tested through the implementation of prototype projects.

- *Peer-reviewed monitoring of constructed restoration efforts.* Habitat restoration activities have been undertaken in many nearshore settings throughout western Washington over the last 30 years. Some have been large and have included monitoring (Carney et al. 2005; Cheney et al. 1994), but most have a tendency to be carried out on small, private properties where no consistent monitoring of the project objectives has been calculated (Gerstel and Brown 2006). Even when monitoring has been performed, it has generally been used more as a design tool, rather than investigating the efficacy of the activities to restore the targeted species (Carney et al. 2005).

- *Cumulative impact studies*. Many other regions around the U.S. have established large interdisciplinary studies to document the cumulative impacts on the nearshore environment [e.g., the Great Lakes: (Meadows et al. 2005)]. Impacts on numerous HCP species have been documented due to hydraulic and geomorphic modifications associated with shoreline hardening in general.

10.2.9.1.2 Freshwater Environments

Because there are few jetties in fresh water, and likely few to be constructed due to the relatively weak demand for them, information on environmental impacts of jetties in fresh water on HCP species stands as a data gap.

10.2.9.2 Breakwaters

10.2.9.2.1 Marine Environments

There have been a large number of studies related to the hydrogeomorphic and ecologic impacts of breakwaters, mostly as a result of the Environmental Design of Low Crested Coastal Defense

Structures (DELOS) program sponsored by the European Community (Losada et al. 2005). The data collected as a part of the hydrogeomorphic portion of the program are relevant to projects in Washington State. The results that document the ecological transition from soft-substrate to hard-substrate communities are particularly relevant to rockfish and sculpin species, as well as the invertebrates. The relative lack of salmonids on the European continent limits the applicability of that work to the nearshore ecology of the Pacific Northwest.

The loss of fish that serve as food for the salmonids has been studied, but relating the construction of a single breakwater or artificial reef to salmonid loss has not. Identifying the role that invasive species infestations associated with the deployment offshore rocky structures can have on native species needs to be investigated (e.g., the abundance of urchins or macroalgae at the expense of salmonids).

10.2.9.2.2 Freshwater Environments

The only literature on freshwater breakwaters comes from the Great Lakes (Marsden and Chotkowski 2001; Meadows et al. 2005). Because there are few breakwaters in Washington in fresh water, and likely few to be constructed due to the relatively weak demand for them, information regarding any environmental impacts on HCP species stands as a data gap. It would be helpful to have a study like Marsden and Chotkowski (2001) to identify the ability of Washington freshwater breakwaters to encourage invasive species infestations.

10.2.9.3 Groins and Bank Barbs

10.2.9.3.1 Marine and Lacustrine Environments

There have been no systematic studies of the cumulative impact of groins, as they have not been built in large number for some time because of other legal restrictions (e.g., Clean Water Act, Coastal Zone Management Act, Shoreline Management Act, and city and county critical areas regulations).

Also absent are studies of the differing degree of impacts from different types of common modifications that would be equivalent to a groin or barb. Construction or repair of access stairways would clearly be less disruptive than construction of a groin that extended from above the ordinary high water mark (OHWM) to subtidal (in marine environments) or limnetic (in lacustrine environments) depths. However, there is essentially no peer-reviewed literature on the impact of these modifications on nearshore geomorphology, let alone nearshore ecology.

There is no literature that describes innovative engineering techniques related to alternative structures on the shoreline, most notably structural access to intertidal or lacustrine littoral areas of the shoreline.

10.2.9.3.2 Riverine Environments

In a literature review of the effects of riprap, by far the most common material used in groins and barbs, on salmonids in streams and rivers of the Western Unites States, Schmetterling et al. (2001) noted four areas where research on this subject is lacking:

> (1) quantifying the habitat availability and quality of riprap;
>
> (2) correlating the effects of riprap banks on salmonid density in the absence of other dependent variables such as diking, channelization, and watershed land use;
>
> (3) comparative studies on the use of riprap and alternative "soft" techniques such as the integration of natural materials; and
>
> (4) the cumulative and synergistic effects of numerous bank-hardening projects at the watershed level.

11 Habitat Protection, Conservation, And Mitigation Strategies

If the impacts described in Section 7 of this document occur within habitat used by a potentially covered species, the result may be incidental take of aquatic animals through either physical harm to the animals or reduced capacity of the habitat to serve essential life functions, such as reproduction, foraging, and migration. The ESA requires that such impacts be avoided or, if unavoidable, minimized to the maximum extent practicable. Measures for avoiding or minimizing the risk of incidental take are identified below. Mitigation measures to compensate for unavoidable take and management strategies are also provided.

It is difficult to programmatically quantify the risk of incidental take attributable to any structure that modifies hydraulics because of the great variety of site-specific factors at work. However, the reviews performed for these white papers indicate that habitat impacts are approximately defined by the **area of habitat affected**, the **number of species affected**, and the **importance of the habitat** to each species.

The **area of habitat affected** is the area of habitat destruction, which can be determined from project plans, plus the area of habitat subject to embedding, scour, or deposition, which can be determined via hydraulic modeling of the structure using a common sediment transport model (appropriate models are described by Miller et al. 2001). Impacts resulting from rare and unpredictable events such as debris flows may not have to be analyzed in an ESA context, but if necessary could be estimated within a cumulative effects context using landscape-scale studies such as published watershed analyses.

The **number of species** affected can be determined at the site scale via surveys or from an inventory database, such as the Streamnet database, the Priority Habitats and Species database, the distribution maps developed for the WDNR Aquatic Lands HCP effort, or the Forest Practices HCP (WDNR 2005c), Streamnet database, and/or the Priority Habitats and Species database. For certain species, these resources identify species use as well as presence, e.g., spawning, migration, or rearing habitat.

The **importance of a habitat** can be estimated by the principle of limiting factors: The resource that is most limiting to a population's growth will be the principal control on that population. For example, if the fish in a given stream are most limited by insufficient spawning habitat, then a project that destroys spawning habitat will result in greater harm than one that destroys an equivalent area of foraging habitat. Baseline data on limiting factors for some species are available from watershed councils and have been prepared for most WRIAs that contain habitat accessible to anadromous salmonids; a current inventory and summaries of limiting factors are available from the Washington State Conservation Commission website at http://salmon.scc.wa.gov. However, these summaries are rarely informative enough to make a determination about which habitat elements are directly limiting for fish production. For salmonids, quantitative analysis has estimated limiting factors for most streams in Washington using the Ecosystem Diagnosis and Treatment model; further information is available at http://www.mobrand.com/edt/.

WDFW might consider a requirement to assess take risk for each HPA. Estimates of area affected, species affected, and habitat importance would allow unprecedented quantification of habitat impacts on a statewide level and would provide an invaluable tool for adaptive management of the HPA program.

This analysis assumes that all activities and structures permitted under the HPA authority are fully compliant with applicable local, state, and federal regulations, particularly including the *Hydraulic Code Rules* (WAC 220-110).

Measures that could minimize impacts from artificial structures include finding an alternative to building the structure; siting the structure as far as possible outside of the active channel/water body; minimizing the structure's footprint; and generally designing the structure to have the least possible effect on channel hydraulics (Bates 2003).

Additional measures for further avoiding or minimizing the risk of incidental take are identified below. These measures include one that was not specified in any of the documents reviewed for this white paper: modifying in-water work windows to be protective of spawning and incubation by any potentially covered species that could be present in the area affected by a proposed project.

11.1 General Actions Applicable to All Activity Types

11.1.1 Information Gathering Recommendations

1. Establish and implement a plan to address data gaps identified in Section 10.

2. Develop additional information on many of the potentially covered species' life histories, habitat needs, and habitat tolerances.

3. Develop and apply a technique for evaluating cumulative impacts of HPA-permitted projects.

4. Track additional information in the HPMS database.
    - Size of structures
    - Specific type of structures
    - Monitoring requirements
    - Mitigation requirements
    - Summary of monitoring findings.

    This information would be useful for analyses at a variety of scales (e.g., basin, stream, region, state) and for WDFW biologists during their reviews of proposed bank protection projects.

5. Develop WDFW guidelines on a series of topics relevant to designing, constructing, and monitoring bank protection projects, including:
    - Beach nourishment
    - Riparian revegetation
    - Channel dewatering
    - Fish and invertebrate species presence
    - Fish and invertebrate removal

6. Update eelgrass/macroalgae guidelines, possibly to include
    - incorporating technology-based approaches (e.g., towed video with diver-based ground-truthing and density data gathering)
    - standardizing monitoring data delivery to facilitate its incorporation into a statewide database (similar to Ecology's SEDQUAL database).

11.1.2 Enforcement Recommendation

Commit to enforcing applicable regulations and providing sufficient staff to meet enforcement needs.

11.1.3 Education Recommendations

Education recommendations apply to information sharing within WDFW and education of the public, particularly local jurisdictions and shoreline landowners.

Within WDFW:

1. Educate staff through information- and monitoring data-sharing workshops for WDFW biologists.

2. Develop an improved system of using monitoring data and making it more widely available. Presumably the use of data could be improved at both the project-specific level (i.e., monitoring data reviewed and acted upon to ensure project compliance) and more generally (i.e., to guide subsequent proposal reviews).

3. Develop statewide clearinghouse for monitoring data, including aquatic and riparian vegetation, fish use, and physical habitat data.

4. Use statewide clearinghouse of eelgrass data to generate updated geographic information system (also known as GIS) layers.

5. Educate the public on shoreline components, habitat function, and species vulnerabilities.

6. Have staff available to assist in development of project monitoring plans and monitoring oversight, as necessary.

Public education:

1. Educate the public on shoreline components, habitat function, and species vulnerabilities. It is critical that decision makers and the general public be educated about the outcomes of their actions, especially those who have the greatest influence on outcomes (i.e., those who live, work, and play along our shorelines).

2. Have staff available to assist in development of project monitoring plans and monitoring oversight, as necessary.

11.1.4 Conservation Program Recommendations

1. Develop and implement conservation programs. Use ecological principles to guide actions and incorporate multiple functions and processes in developing goals and objectives for conservation actions.

2. Develop incentives for conservation programs. Land acquisition, tax incentives, regulatory incentives, and other measures have been used and should be considered in the development of conservation programs.

11.1.5 Construction Recommendations

11.1.5.1 Construction and Maintenance Best Management Practices

The U.S. Environmental Protection Agency (U.S. EPA) has released a recent publication relevant to the management of construction and maintenance related effects on water quality (U.S. EPA 2007). The report summarized best management practices (BMPs) that are relevant to the construction and maintenance of HPA-permitted activities. The recommended BMPs, which should be applied to hydromodificaton projects to reduce nonpoint source pollution, include:

- Stockpile fertile topsoil for later use for plants
- Use hand equipment rather than heavy equipment
- If using heavy equipment, use wide-track or rubberized tires
- Avoid instream work except as authorized by the local fishery and wildlife authority
- Stay 100 ft away from water when refueling or adding oil
- Avoid using wood treated with creosote or copper compounds
- Protect areas exposed during construction.

Other nonconstruction-related recommendations put forth by U.S. EPA (2007) include:

- Incorporating monitoring and maintenance of structures
- Using adaptive management
- Conducting a watershed assessment to determine project fate and effects
- Focusing on prevention rather than mitigation
- Emphasizing simple, low-tech, and low cost methods.

The National Marine Fisheries Service (NMFS 2001) says that temporary crossings placed in salmonid streams for water diversion during construction activities should meet all fish passage guidelines where fish are expected to be present during the construction window.

In the construction of many kinds of HPA-permitted structures, avoidance or minimization of impacts can be accomplished through proper site selection. For construction and maintenance activities, management strategies can be implemented to minimize underwater noise, project area dewatering, and navigational dredging impacts.

Construction activities should be timed to occur when sensitive life stages (e.g., spawning, incubation, emergence) of potentially covered species are less likely to be present (NMFS 2003a). To minimize effects to aquatic vegetation, they could also be timed to occur at times of the year when aquatic vegetation biomass is at a minimum.

11.1.5.2 Pile Driving

The intensity of underwater noise produced by pile driving varies considerably depending on site characteristics and the type of materials and methods employed. A desirable approach for avoiding underwater noise impacts from pile driving is to conduct this activity within a dewatered exclusion area. This measure may not be practicable in many circumstances. In such cases, a number of BMPs can be used to limit underwater noise impacts.

The following BMPs should be considered to minimize effects related to pile driving on HCP species:

- Use pile caps[1], if feasible and safe, to reduce the sound of pile driving below injury level (Laughlin 2006).

- Use vibratory hammers[2]; the low rise in sound over a longer period of time is less stressful to aquatic animals, and the sound is typically 10 to 20 dB lower than impact hammer pile driving (WSDOT 2006a).

- For projects with pile sizes less than 24 inches in diameter, use the smallest piling size practicable to lower sound pressure levels when driven.

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[1] Pile caps have been shown to effectively reduce underwater sound levels. Laughlin (2006) reduced sound levels by 27 dB with a wood pile cap when driving a 12-inch-diameter steel pile, which would reduce noise levels to below those established for injury (at 33 feet [10 meters]) by NMFS and USFWS. Conbest, Micarta, and Nylon pile caps have also been shown to reduce sound levels (Laughlin 2006).

[2] Under certain conditions, a vibratory hammer can be used to reduce noise impacts. Vibratory hammers vibrate the pile into the sediment by oscillating the pile into the substrate. The vibratory action of this hammer causes the sediment surrounding the pile to liquefy so that the pile can be driven (WSDOT 2006a). Peak sound levels for vibratory hammers can exceed 180 dB; however, the sound from these hammers has a relatively slow rise, produces sound energy that is spread out over time, and is generally 10 to 20 dB lower than pile driving using an impact hammer (WSDOT 2006a). However, it is frequently necessary to proof a piling driven with a vibratory hammer with an impact hammer to ensure the integrity of the piling.

- Use (untreated) wood or concrete piles where practicable, as these also induce lower sound pressure levels. Even though these materials are less strong, increasing the size of the structure would be considered less impactful as long as the structure does not become so large as to produce other hydrogeomorphic impacts (e.g., if the additional wood piles inhibit transport of sediment, water, or groundwater).

- Use air bubble curtains [3]to create a bubble screen (Reyff et al. 2003; Vagle 2003). (Dual layer air bubble curtain or similar[4] noise abatement technology.)

- Maintain the integrity of the air bubble curtain; no boat traffic or other structure or equipment should be allowed to penetrate the air curtain during pile driving activities.

- In marine environments, install geotubes during low tide to minimize the potential for entrapment and stranding of fish within the enclosed area.

- Use fabric barriers and/or cofferdams to create an additional interface to buffer sound transmission into the underwater environment (WSDOT 2006).

- Use helical piles where possible. These piles do not require vibration or hammering. The only noise produced is from the screwing action of the driller.

- To avoid attracting fishes with lights during nighttime pile driving operations, limit pile driving to daylight hours to the extent practicable

11.1.5.3 Channel Dewatering

- Develop guidelines for channel dewatering and stream bypasses. Adopt a protocol for review/approval of proposed dewatering and stream bypass plans. The isolation plan should include information on timing, channel dewatering, and bypass plans. The isolation method should be able to withstand any flows that are encountered during the

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[3] Proper design and implementation of a bubble curtain are key factors in the effectiveness of this strategy (WSDOT 2006a). Based on the literature, NMFS and USFWS usually assume there will be a 15 dB_{peak} and RMS reduction in sound levels when using a bubble curtain (WSDOT 2006a). For steel piling 14 inches or less in diameter, as well as concrete and wooden piling, such a reduction would reduce noise levels to below injury thresholds established by NMFS and USFWS at a distance of 33 feet (10 meters).

[4] Fabric barriers and cofferdams are also used to attenuate sound levels from pile driving by creating another interface through which sound travels. The concept is similar to that behind the use of bubble curtains (WSDOT 2006a).

isolaiton period, to avoid flooding and the possibility of fish reoccupying the area prior to dewatering.

- Adopt science-based protocols for fish removal and exclusion activities. An example protocol is provided by WSDOT (WSDOT 2006b). NMFS also provides electrofishing guidelines, which are in common use and are usually required as conditions of NMFS scientific take permits. Recommended guidance/protocols include those for:
    - Fish capture including seining and electrofishing.
    - Fish handling.
    - Tracking and reporting of number and species of fish captured, fish injured, injuries observed, and fish killed.
- Make sure qualified people are available who can perform fish removal, capture, handling, and exclusion.
    - Define the qualifications of a “qualified fish biologist” or “qualified personnel.” A qualified biologist needs to be on-site supervising and/or implementing the operation.
    - NMFS often requires a resume from the permittee prior to issuing a take permit.
    - Develop an appropriate training or qualification process for biologists.
    - Maintain a list of qualified fish biologists.
    - If electrofishing, at least two people (an operator and a netter) are required to safely and effectively capture the fish. In larger stream areas, two or more electrofishers operating simultaneously may be necessary to effectively capture all of the fish, as each electrofisher only has a limited range of effectiveness.
- A scientific collection permit from WDFW is required to capture fish.

For fish salvage/electrofishing operations:

- Require slow dewatering and passive fish removal from the dewatered area before initiating active fish-removal protocols. Fish removal by seining is recommended before resorting to electrofishing, which carries a greater risk of mortality (NMFS 2006). Seining alone is not as effective at removing fish as electrofishing, and is likely to miss fish during a salvage operation. Such fish would die when the stream is dewatered. Initially seining and then electrofishing is a more effective way of safely removing fish from the work isolation area.

- Pay attention to timing and conditions during the operation. Perform work during low-flow or dry conditions, and/or during dry weather. With less water in the channel, there are likely to be fewer fish affected by channel dewatering. However, electrofishing may have impacts if sensitive life stages of fish are present, for example adults that are migrating into the system to spawn, or when the eggs and alevin are still in the gravel. Also, during lower flows the water temperature often is elevated. Electrofishing should not be performed when temperatures exceed 64 F or 18 C, as it reduces the oxygen content of the water, affects the conductivity of the water (influencing the effect of the electric current), and fish are often already stressed, which could lead to mortality during electrofishing and handling.

- When electrofishing, use the minimum voltage and duty cycle necessary to effectively capture the fish. Use the lowest power output that provides for effective electrofishing (sufficiently large field for taxis and narcosis). This will be influenced by the conductivity and the temperature of the water, as well as the size of fish expected to be encountered. Fish should recover quickly (within a minute) and should not show any external signs of injury, such as branding or deformation (Snyder 2003). Use the least damaging current available. Most electrofishers now use a pulsed direct current, where the “duty cycle” or pulse length and frequency can be adjusted to minimize impacts on fish. Do not use electrofishers that use alternating current (Snyder 2003).

- Watch for the occurrence of brands (i.e., burn-type marks caused by electrofishing) and extended tetany (tonic spasm of muscles), which indicate harmful effects are still a problem, even when using currents designed to be less harmful (Snyder 2003).

- Backpack electrofishers generally have a circular anode and a cable cathode. Boat-based electrofishers use spherical anodes, but under most circumstances a boat shocker would not be used in a fish salvage operation. The size of anode that is used must be appropriate to the size of the stream. Personal communications cited in Snyder (2003) suggest that while spherical electrodes are theoretically superior to cables, no significant difference in catch rate or the incidence of brands was observed between the two; that spherical anodes and cable cathodes appear to be the best combination; and that anodes should be kept high in the water to draw fish to the surface, where they can be easily netted.

- Species such as lamprey are more effectively captured using non-circular anode rings that direct the current into the substrates. A qualified biologist would know what is appropriate for the conditions.

Minimize channel dewatering impacts on HCP species by taking the following precautions:

- If pumps are used to temporarily divert a stream to facilitate construction, an acceptable fish screen must be used to prevent entrainment or impingement of small fish (NMFS 2001).

- Adhere to performance criteria for fish screens on pumped diversions presented by NMFS (1996a) and WDFW (1998). Compliance will minimize the risk of incidental take due to entrainment.

- Pump sediment-laden water (from the work area that has been isolated from surrounding water) to an infiltration treatment site.

- Dispose of debris or sediment outside of the floodplain.

- Stabilize disturbed areas at the work site with sediment corresponding to the ambient bed to prevent an influx of fine sediment once water is reintroduced to the site. Replace disturbed streambed materials with clean gravel of the appropriate size prior to rewatering to minimize an influx of fine sediment.

- Fish should be kept in the water as much as possible. Minimize exposure to the field and specimen handling by rapidly netting fish before they get too close to the anode and quickly, but gently, placing them in oxygenated holding water. After capture, place fish into a temporary holding bucket to allow recovery prior to transferring the fish to a safe release site. The time in the bucket should be as brief as possible. Process the fish frequently to reduce crowding, and change the water frequently to maintain cool, well-oxygenated water (Snyder 2003).

- The release location should near the capture site, but appropriately located either upstream of the construction activities, or a sufficient distance downstream to avoid increased turbidity from construction activities.

11.1.5.4 Dredging and Fill

Dredging and fill are necessary components of project construction and maintenance for many HPA-permitted activities. The permitted in-water work window for these structures should consider the full range of HCP species likely to occur in the vicinity and should be timed to avoid the presence of sensitive species and/or life-history stages where practicable. In cases where adverse impacts on HCP species cannot be avoided effectively (e.g., a nursery site for buried lamprey amocoetes), alternative designs that avoid dredging and fill impacts should be considered.

Where practicable, dredging and fill activities should be conducted within an exclusion area (dewatered or watered as appropriate) following fish removal. This will help to limit elevated turbidity and sediment impacts. Creation of exclusion areas and fish removal and relocation should be conducted using standardized protocols for these procedures.

A number of techniques have been developed that may be used to avoid or mitigate the effects of dredging (Smits 1998) and placement of fill materials on sensitive ecosystems such as wetlands (Sheldon et al. 2005). Dredging associated with fish screens is typically coupled with the installation and/or maintenance of a water diversion system. Placement of fill material is typically associated with the installation of water diversion system or may be incidental during construction.

General recommendations to avoid and minimize the impacts of dredging are provided in the 2001 Dredging: Marine Issues white paper (Nightingale and Simenstad 2001a) and include:

- Use multiseason pre- and postdredge project biological surveys to assess animal community impacts more extensively;
- Incorporate cumulative effects analysis into all dredging project plans;
- Increase use of landscape-scale planning concepts to plan for beneficial use projects most suitable to the area's landscape ecology and biotic community and food web relationships;
- Further identify turbidity and noise thresholds to assess fish injury risks; and
- Further analyse and synthesize knowledge about the spatial and temporal distribution of fish and shellfish spawning, migration behavior, and juvenile rearing to evaluate environmental windows for dredging on a site-specific basis.

The following recommendations are intended to reduce the effects of dredging on HCP species:

- For new marine, riverine, and lacustrine projects and significant expansions beyond general maintenance dredging, thoroughly assess the large-scale, cumulative impacts of the resulting changes in bathymetry, habitat loss, and change to estuarine/nearshore marine ecosystem dynamics (e.g., salinity intrusion).
- Require hopper dredges, scows, and barges, trucks or any other equipment used to transport dredged materials to the disposal or transfer sites to completely contain the dredged material.
- For long-term projects where continuous dredging and onloading to barges occur, require periodic movement of the barge to reduce shading.
- Modify in-water work windows to take into consideration what is known about site-specific spatial and temporal distribution of fish and shellfish eggs, larvae, and juveniles.
- Evaluate the application of in-water work windows on a site-specific basis based on the location and features of the site, such as sediment composition, plant and animal assemblages, and timing of seasonal and migration patterns.
- Use presampling bathymetric surveys, records from previous dredging events, and best professional judgment to estimate the volume of sediments likely to be dredged; base sampling and testing requirements on this estimated volume.
- Avoid projects and expansions that convert intertidal to subtidal habitat. If such conversion is unavoidable, employ comprehensive, large-scale risk assessment to identify the cumulative effects of site-specific changes to ecosystem dynamics.
- Select dredging equipment types according to project-specific conditions, such as sediment characteristics.

- Base turbidity threshold testing for dredging operations on background site turbidity.
- In areas where dredging is proximal to sensitive habitats (or in projects where sediments both suitable and unsuitable for unconfined open water disposal will be dredged adjacent to each other), use the "Silent Inspector" (a computerized electronic sensor system) to monitor dredging operations. This tool can assist in operational documentation and regulatory compliance by providing record accessibility and clarity. It also offers advantages for planning, estimating, and managing dredging activities.
- Increase the use of multiseason, preproject surveys of benthos to compare with postproject surveys to understand dredging impacts.
- Where applicable and involving uncontaminated sediments, consider beneficial use of dredged materials that can contribute to habitat restoration, rehabilitation, and enhancement, particularly for projects that incorporate a landscape ecology approach.
- Avoid beneficial use projects that impose unnatural habitats and features on estuarine, marine, and riverine landscapes.
- Use hydrodynamic models to predict system-wide changes in salinity, turbidity, and other physicochemical regimes for project assessment planning that avoids or minimizes impacts on aquatic habitat.
- Dredging should be conducted to a depth not greater than a navigation channel depth at the seaward end. If necessary, authorize dredging to depths greater than the navigation channel at the seaward end only in berthing areas and turning basins for commercial shipping purposes.

11.1.5.5 Vessel Activities

Issues related to vessel activities (including barges) during construction include vessel grounding in sensitive habitats (such as eelgrass), the effects of propeller wash, the risk of accidental spills of fuel or other contaminants, the risk of introducing noxious weeds, and noise.

- WDFW's standard HPA provisions already prohibit vessel grounding in areas of eelgrass, macroalgae, or forage fish spawning (e.g., "Eelgrass and kelp shall not be adversely impacted due to project activities [e.g., vessels shall not ground, anchors and spuds shall not be deployed, equipment shall not operate, and other project activities shall not occur in eelgrass and kelp," from Marine Boat Ramp Maintenance and Repair provisions in the Hydraulic Pemit Management System]).
- It may be appropriate to require construction vessel operation plans for larger projects or projects located in particularly sensitive habitat to ensure that the potential for vessel and construction activity impacts to sensitive habitats and species is minimized.
- To reduce vessel impacts to the nearshore environment at the Clinton ferry terminal, Thom et al. (1995, in Haas et al. 2002) recommended constructing a longer deck that keeps vessels in deeper water.

- Elevated ambient noise levels are produced when construction vessels are operated continuously around a project site. To protect HCP species from the resulting stressors, operation of vessel engines and motorized equipment should be limited to the extent necessary to support construction work and the working environment. Where available and practicable, vessels with noise-deadening technology should be employed to reduce underwater noise levels produced.

- HPA standard provisions should include:
  - Clean propellers before putting boats into the water to reduce the spread of noxious weeds.
  - File a spill prevention plan.
  - Maintain vessels on a routine basis as well as prior to its use on the construction site.

- Floats should be sited in deeper water to reduce the potential impacts associated with propeller wash.

11.1.6 Aquatic Vegetation Recommendations

HPA-permitted activities can impact aquatic vegetation through altered autochthonous production, habitat complexity, and nutrient cycling. Mitigation of impacts to aquatic vegetation is best achieved through avoidance. To protect and restore aquatic habitat functions, management strategies and development of shoreline regulations should:

- Avoid or minimize the removal or disturbance of aquatic vegetation. Locate facilities in areas that are currently devoid of native aquatic vegetation or in areas that will minimize the potential impacts, such as in deeper water or further offshore.

- Minimize impacts from vessels associated with HPA-permitted structures. The typical effects of vessels on aquatic vegetation vary with both distance and propeller speed, both of which may be important factors in loosening sediment particles and eroding the vegetation.
  - Manage equipment and vessel operations and establish no-construction or no-vessel activity buffers around existing aquatic vegetation to protect this habitat and its contribution to ecological functions.
  - Require the control of turbidity during construction and operation of the facility to minimize prop wash and bubbles, and prevent suffocation or excessive shading of plants.

- Site structures in deeper water to minimize shading and physical impacts on aquatic vegetation.

- Do not allow floats to ground out on low tides.

- Encourage the use of upland boat storage areas and the use of slings to minimize shading of aquatic vegetation.

- Place the potential shade-casting structures perpendicular to the arc of the sun (i.e., north–south placement) to maximize transmission of light under the structure.
- Any walkways should be 100 percent grated; floats and docks should be at least 60 percent grating.
- Orient grating to maximize transmission of light under the structure.
- Minimize the amount of pier area that directly contacts the shoreline, to allow light penetration to the nearshore intertidal and shallow subtidal areas.

11.1.6.1 Eelgrass

If HPA-permitted structures (including "water crossing structures" such as bridges, "overwater structures," and larger complexes such as marinas) are designed and located so that they do not reduce available light below approximately 325 $\mu M/m^2/sec$, then eelgrass impacts may be avoidable (Thom et al. 1996, in Simenstad et al. 1999).

Where projects result in a direct loss of eelgrass during in-water construction, revegetation can be achieved through natural regrowth or transplanting (Thom et al. 2001); however, transplanting eelgrass is not always successful and the science is still developing. For one project in the San Juan Islands, post-disturbance monitoring of eelgrass beds indicates that where substrate, depth, light availability, and currents are suitable and adjacent eelgrass remains intact, natural revegetation can recolonize disturbed areas at a rate of greater than 1 foot per year (Jones and Stokes 2005).

In Washington, transplanting has been used with some success to revegetate eelgrass beds, although a review of eelgrass restoration projects concluded that eelgrass restoration is "possible, with difficulty" (Thom et al. 2001). New eelgrass beds can be established where conditions that prevent eelgrass from growing (e.g., shade, depth, substrate, or current velocity) are remedied (Thom et al. 2001).

11.1.6.2 Freshwater Aquatic Vegetation

Mitigation of impacts to aquatic vegetation should focus on ecosystem functions (Hruby et al. 1999). Although all non-noxious aquatic plants are considered beneficial, replacement of vegetation lost or disturbed during project installation may be less beneficial than other ecosystem renovation methods, depending on the plant coverage, density, species, and setting involved. For example, guidance on assessing the functions and values of riverine flow through wetlands in Western Washington (Hruby et al. 1999) does not include aquatic vegetation as a variable in evaluating the functions and values to anadromous or resident fish. Likewise, the matrices of ecosystem functions and pathways for making ESA determinations of effect at the watershed scale (NMFS 1996; USFWS 1998) do not include aquatic vegetation as an indicator of ecosystem function. However, this is partly because both of these evaluation systems are largely designed to address salmonid habitat requirements; re-evaluation is warranted for many potentially covered species having a stronger dependence on freshwater aquatic vegetation (e.g., Olympic mudminnow or California floater). In many settings, aquatic vegetation can recolonize through natural seeding and vegetative growth if conditions are suitable. Depth, substrate, shade, and competition among plant species are all factors that determine which species of plants colonize and survive (Chambers et al. 1999).

Using the functional approach to assessing potential impacts to aquatic vegetation (Hruby et al. 1999), which is an important habitat component for many of the potentially covered species (e.g., Olympic mudminnow and California floater), and determining appropriate mitigation for the loss of freshwater aquatic vegetation are likely to result in minimal potential for incidental take related to aquatic vegetation loss.

11.1.7 Riparian and Shoreline Vegetation Recommendations

The following measures could help avoid and minimize incidental take arising from impacts to riparian and shoreline vegetation:

- Avoid and minimize any impacts on riparian, aquatic, and shoreline vegetation by protecting the vegetation.

- Consider whether projects that require extensive in-water work, which may require extensive access and which have high-quality riparian habitat, should have work performed entirely within the wetted channel to avoid impacts to riparian vegetation. The short-term impact to a stream channel may be of less consequence than the long-term impact that may be incurred to riparian vegetation, due to the respective rate of recovery.

- To the extent practicable, do not permit removal or disturbance of riparian vegetation in areas with high erosion hazard (Knutson and Naef 1997).

- Where riparian vegetation has been removed, isolate disturbed areas from aquatic resources using erosion control features until disturbed areas are stabilized.

- Consider all ecological functions when developing a riparian management strategy.

- If it is not possible to leave vegetation, prepare and carry out revegetation plans to restore the riparian vegetation. The revegetation plans should identify areas to be replanted, when construction is complete, with native riparian vegetation endemic to the area. The proximity of the vegetation to the aquatic habitat and the size of the vegetation should be such that it can restore the ecological benefits, such as temperature regulation and allochthonous inputs.

- Replanted vegetation should be monitored [5]. The project proponent should be required to ensure 100 percent of all plantings are viable and healthy at the end of one year and 80

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[5] Some of the original white papers recommended a three-year monitoring period, with two monitoring reports; one at the end of the first year, another after three or five years. Other white papers recommended monitoring every other year or every third year for an unspecified period of time.

percent of all plantings are viable and healthy by the end of the three-year monitoring period. These recommendations are based on provisions in WAC 220-110 and on general conditions provided by the Corps, NMFS, and USFWS for Corps ESA Section 7 programmatic consultations.

- Submit monitoring reports to WDFW. Similar to the requirement of the Corps for ESA Section 7 individual and programmatic consultations, the first monitoring report should be submitted one year after project completion. After 3 years, monitoring and reporting should be completed every other year or every third year[6]. The monitoring reports must include information on the percentage of plants replaced, by species. Monitoring reports should also state the cause of any plant failure, a provision generally required by the Corps, NMFS, and USFWS for Corps ESA Section 7 programmatic consultations. In addition, any specific conditions provided by the U.S. Army Corps of Engineers (for project permits) or NOAA Fisheries and USFWS (for ESA Section 7 compliance) must be implemented.

- Save vegetation (specifically large trees and root wads) removed for the project for later use in restoration efforts. This condition has often been required in recent individual and programmatic Section 7 consultations. Even if the material is not specifically useful for the permitted action, a WDFW area habitat biologist will generally know of ongoing or pending restoration projects in need of LWD and root wads.

- Require performance bonds for projects disturbing large areas (e.g., >500 square feet) of riparian vegetation.

- Enforce revegetation requirements.

- Consider establishing buffers and setbacks that protect the functions of the riparian system and its contribution to ecosystem. The term “buffer,”as applied in a specific management context, denotes an area set aside and managed to protect a natural environment from the effects of surrounding land-use or human activities (May 2003; Knutson and Naef 1997). Depending on the context, buffers may be designed to perform a specific function or set of functions, such as filtering pollutants or providing shade (May 2003).

- Establishing buffer areas is an important regulatory tool both to keep development activities in this habitat to a minimum, and (for developed or redeveloping sites) to trigger mitigation sequencing to deal with project impacts on riparian vegetation. May (2003) provides a review of riparian functions as a factor of buffer width. As indicated in May (2003), there is no consensus in the literature recommending a single buffer width for a particular function or to accommodate all functions. Knutson and Naef (1997) resolved the variability in the literature by averaging effective buffers widths reported for specific riparian functions. Knutson and Naef (1997) show that for streams, a buffer

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[6] See previous footnote.

width of 147 feet is effective in providing five of the seven riparian functions including: sediment filtration, erosion control, pollutant removal, LWD, and water temperature protection. Table 11-1 provides a summary from the scientific literature of how different riparian habitat widths protect function.

Table 11-1. Riparian buffer functions and widths (widths reported in feet)

Riparian Function	May (2003)			Knutson and Naef (1997)	
	Notes on Function	Range of Effective Buffer Widths	Minimum Recommended Widths	Range of Effective Buffer Widths	Average of Reported Widths
Sediment removal/erosion control	For 80% sediment removal	26 – 600	98		
Erosion control				100 – 125	112
Sediment filtration				26 – 300	138
Pollutant removal	For 80% nutrient removal	13 – 860	98	13 – 600	78
LWD recruitment	1 SPTH based on long-term natural levels	33 – 328	164	100 – 200	147
Water temperature protection	Based on adequate shade	36 – 141	98	35 – 151	90
Wildlife habitat	Coverage not inclusive	33 – 984	328	25 – 984	287
Microclimate	Optimum long-term support	148 – 656	328	200 – 525	412

SPTH = site potential tree height.

Brennan and Culverwell (2004) recommend the following for consideration as part of any coastal management strategy and development of shoreline regulations associated with marine riparian habitat:

- Preventing additional losses of riparian vegetation is both critical and cost-effective. Once riparian functions are lost, they are difficult and expensive to restore, if restoration is possible at all.

- Fill data gaps. The lack of empirical data for Northwest coastal ecosystems and limited recognition of riparian functions have led to poor management practices and protection standards for coastal resources. Research and documentation are critical to establish a scientific foundation for creating adequate policies and practices for protection and restoration.

- Establish appropriate buffers and setbacks. Buffers and setbacks are essential, functional, and cost-effective tools for preserving important processes and functions, preventing environmental degradation, and protecting valuable coastal resources.

- Maintain and/or restore riparian vegetation for human health and safety. Flooding, storm, and erosion hazards are common problems in coastal areas and become a greater threat when shoreline development does not consider the functions and values of maintaining riparian vegetation buffers.

- Identify, evaluate, and incorporate multiple functions into a management strategy. Any management strategy should be based on maintaining all natural processes and functions, determined by an evaluation of the specific requirements for maintaining individual and collective functions over space and time (e.g., LWD recruitment; life history requirements of multiple species of fishes and wildlife).

- Use a multidisciplinary approach in developing riparian management zones. Experts in a wide range of natural sciences should collaborate on an integrated and multidisciplinary assessment.

- Maintain and/or restore riparian vegetation for pollution abatement and soil stability. Vegetative buffers would likely be of benefit by reducing contaminants in runoff and reducing costly reactionary measures to clean up waterways.

- Maintain and/or restore riparian vegetation for fish and wildlife. It is clear that as vegetation is eliminated, the food supply, and thus the carrying capacity of the coastal ecosystem, is reduced.

- Protect marine riparian areas from loss and degradation. Riparian areas provide a wide range of functions that are beneficial to humans, fish, and wildlife. Every effort should be made to preserve remaining marine riparian areas from further degradation, fragmentation, and loss.

11.1.8 Water Quality Recommendations

Based on the findings of Bash et al. (2001) on turbidity effects on salmonids, the following mitigation measures are recommended to avoid direct and indirect effects on HCP species:

- Prior to project construction, determine background suspended sediment concentrations and collect information on particle size and shape, to understand the ambient turbidity to which animals have adapted.

- Review existing watershed assessments to consider pollution loads that may be from sources outside the project to evaluate the project's cumulative effects on turbidity levels.

- Once existing turbidity and sources have been determined, establish acceptable project increases to background turbidity that are similar to those set in the Implementing Agreement between WSDOT and Ecology (WSDOT and Ecology 1998), which states:

  > "All work in or near the water, and water discharged from the site shall meet the State's Water Quality Standards, WAC 173-201A. A mixing zone for turbidity is authorized within WAC 173.201A-030 during and immediately after necessary in-water or shoreline construction activities that result in the disturbance of in-place sediments. Use of a turbidity mixing zone is intended for brief periods of time (such as a few hours or days) and is not an authorization to exceed the turbidity standard for the entire duration of the construction. Use of the mixing zone is subject to the constraints of WAC 173-201A-100(4) and (6), requiring an applicant have supporting information that indicates the use of the mixing zone shall not result in the loss of sensitive or important habitat, substantially interfere with the existing or characteristic uses of the water body, result in damage to the ecosystem, or adversely affect public health. The mixing zone is authorized only after the activity has received all other necessary local and state permits and approvals, and after the implementation of appropriate best management practices to avoid or minimize disturbance of in-place sediments and exceedances of the turbidity criteria. Within the mixing zone, the turbidity standard is waived, and all other applicable water quality standards shall remain in effect. The mixing zone is defined as follows:
  >
  > 1. For waters up to 10 cfs [cubic feet per second] flow at time of construction, the point of compliance shall be 100-feet downstream of project activities.
  > 2. For waters above 10 cfs up to 100 cfs flow at time of construction, the point of compliance shall be 200-feet downstream of project activities.
  > 3. For waters above 100 cfs flow at the time of construction, the point of compliance shall be 300 feet downstream of project activities.
  > 4. For projects working within or along lakes, ponds, wetlands, estuaries, marine waters or other non-flowing waters, the point of

> compliance shall be at a radius of 150-feet from the activity causing the turbidity exceedance."

- As an indicator of pre-construction conditions, assess the PAH and metals contamination levels of the water body and sediment prior to construction. Consider the existing watershed condition and account for point and nonpoint source pollution loads from watershed sources other than the project and from legacy impacts of the system when evaluating cumulative impacts from PAHs, metals, and turbidity. Professional experience and information on urban stormwater pollutants presented by Menzie et al. (2002) and numerous others support this measure as reasonable.

- Set stockpile areas back from the bank and include erosion prevention BMPs, such as silt fencing and tarp covers.

- Locate the structure deep enough to avoid prop wash resuspension of sediments and contaminants.

- Given the large size of terminals and the large number of pilings required for marinas, use alternatives to treated wood (e.g., materials such as metal, concrete, plastics, and composites) to avoid potential impacts for both new and/or replacement structures.

- If treated wood is used,
    - it should be encased or sealed to prevent leaching of harmful chemicals.
    - Sawdust, drillings, and trimmings from treated wood should be contained with tarps or other impervious materials and prevented from contact with the bed or waters of the state.
    - Structures built of treated wood should incorporate features such as steel, plastic, or rubber collars, fendering, or other systems to prevent or minimize the abrasion of treated wood by floats, ramps, or vessels.

Many of the following mitigation measures regarding aquatic applications of treated wood are based on those suggested by Poston (2001).

- Use alternative materials such as metal, concrete, or composites, or for temporary projects use untreated wood.

- If possible, install immersed treated wood products when potentially covered species are not present near the site. This measure is based on information on rapidly diminishing leaching rates reported by Poston (2001).

- Pre-soak treated wood in confined water to reduce impacts by capturing the initial surge of most concentrated leachate, particularly in the case of ACZA- and CCA Type C-treated products, for which leaching rates appear to drop dramatically after a few days.

- Phase and stagger the installation of ACZA- and CCA Type C-treated structures by a few weeks or more, which may dramatically reduce the concentration of leached metals in surrounding water and the instantaneous extent of the area of impact. This measure is based on information on rapidly diminishing leaching rates reported by Poston (2001).

- Use semi-transparent, water-repellent stain, latex paint, or oil-based paint on above-water portions of treated wood structures, which may reduce leaching of arsenic, chromium, and copper into stormwater generated by that portion of the structure (Lebow et al. 2004).

Additional mitigation measures for water quality include:

- Require that stormwater runoff be 100 percent contained. Route stormwater from the structure and adjacent impervious surfaces to a treatment system.

- If possible, determine a spatial limit, beyond which no water quality effects will extend. Within this limit, monitoring will be required to ensure that established water quality standards are met. If at any point during construction/dredging/demolition these standards are exceeded, construction/dredging/demolition activities will cease until water quality standards are met.
- Existing Washington State Department of Ecology regulatory requirements for Clean Water Act Section 401 certification and the Hydraulic Code limit the in-water curing of concrete as necessary to avoid pH effects and the use of appropriate BMPs to avoid leakage of concrete leachate to surface waters.

11.1.9 Hydrologic and Geomorphic Recommendations

11.1.9.1 Channel Hydraulics

WDFW could consider requiring that HPAs for any structure that will place fill within the OHWL include a hydraulic model of probable structure effects on sediment transport and channel hydraulics to ensure that impacts such as scour, deposition, and embedding due to fine sediment deposition are avoided or minimized[7].

A modeling requirement would ensure that effects of the structure on the channel, and by extension on potentially covered species, are as well understood as practicable. The results of such studies can be summarized so as to allow monitoring of the quantitative impact of authorized projects on channel hydraulics. Such results would be useful in estimating cumulative impacts of the HPA program, incidental take, and identifying appropriate compensatory mitigation measures.

11.1.9.2 Littoral Drift

Impacts to littoral drift can be avoided or minimized through the following measures:

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[7] Some of the original white papers recommended that the hydraulic model provide a summary of effects “to a quantitatively ascertainable degree”, while others said that the hydraulic model need not be numerical; conceptual or qualitative models may suffice for some settings.

- Design pile-supported structures with maximum open space between pilings to allow waves, currents, and sediment to pass beneath (MOEE 1995).

- Minimize certain impacts from floating structures placed perpendicular to shorelines, which dampen wave action and prohibit natural shoreline erosional processes, by minimizing the dimensions of these types of structures.

- Utilize floating breakwaters or ramps in place of breakwater walls to reduce effects on littoral drift (Nightingale and Simenstad 2001b).

- Do not allow floats to ground at low tide.

The effects of these measures are site-specific, and thorough study of the littoral drift cell and potential habitat affected should be conducted on projects that could affect the system's littoral currents and wave action. Avoiding or minimizing alterations in littoral processes would allow shoreline sediment conditions to change at the scales and rates that match those that potentially covered species have evolved to adapt to, minimizing the potential for incidental take through alterations in shoreline substrate distribution and consistency.

11.1.10 Artificial Light

Kahler et al. (2000) recommends that to reduce impacts on salmonid predation, additional shoreline or pier lighting on lakes should not be permitted, and Tabor et al. (1998) suggests that reducing artificial light in the Cedar River would benefit emigrating sockeye salmon. Tabor et al. (1998) also observed that any reduction in artificial lighting must be balanced with safety and other public concerns.

11.1.11 Lost Opportunities

The hydraulic and geomorphic modifications induced by many HPA-permitted projects (e.g., dams, weirs, tide gates, beaver dam removal and large woody debris removal) can result in lost-opportunity impacts. Mitigation for lost opportunity requires mitigation for channel processes affected by a project. In some situations, off-site mitigation may be the only option (WDFW 2003). According to WDFW (2003), the concept of mitigation for lost opportunity should only be applied when consistent, acceptable assessment methods or site-specific information is available. More detailed information on mitigation for lost-opportunity is provided in WDFW (2003).

11.2 Activity-Specific Avoidance, Minimization, and Mitigation Actions

The measures for avoiding or minimizing take include conservation measures and best management practices. Many of the activity-specific measures presented below are generally applicable to several types of activities.

Conservation measures are design elements that are intended to avoid or minimize impacts to habitats and species.

Best Management Practices (BMPs) are those measures used during the construction phase to avoid or minimize impacts.

11.2.1 Bank Protection

11.2.1.1 Avoidance and Minimization Techniques

Impact reduction measures for bank protection include both conservation measures and BMPs. Many of these practices have been identified in the published literature as well as guidance documents, and they may be required by regulatory agencies as permit conditions. Table 11-2 summarizes these measures as currently known and practiced, organized by mechanism of impact.

Table 11-2. Bank Protection Conservation Measures and BMPs

Mechanism of Impact	Conservation Measures[ab]	Best Management Practices
Construction	Require construction set-back that will avoid the risks associated with slope retreat (high and low-no-bank sites) (Gerstel and Brown 2006). Manage all surface water to contain and direct it appropriately to the base of the bluff (high-bank sites) (Gerstel and Brown 2006). Develop guidelines for channel dewatering, including a protocol for WDFW review and approval of proposed dewatering plans. Adopt guidance/protocols for fish and invertebrate removal and exclusion. Specifically, this refers to guidance/protocols for fish capture (including seining and electrofishing), fish handling, and reporting on the number and types of fish captured, fish injured, injuries observed, and mortality. An example protocol is provided by the Washington State Department of Transportation (WSDOT 2006b). Define the qualifications of "qualified personnel" who can perform fish capture and handling activities or develop an appropriate training or qualification process for biologists. In addition, maintain a list of qualified fish biologists who can perform fish removal and exclusion activities. Initiate channel dewatering to allow for volitional movement out of area. Then conduct fish and invertebrate removal activities. Have qualified personnel present to survey the area during dewatering and remove any additional fish and invertebrates encountered.	Construction activities should be timed to occur when sensitive life stages of potentially covered species are less likely to be present. As appropriate, species surveys (including forage fish egg surveys) should be conducted at site prior to initiation of construction to ensure no species present or to allow for removal plan to be prepared and implemented. Use temporary erosion control measures, including application of mulch, hydroseeding, geotextiles, or soil stabilizers (Saldi-Caromile et al. 2004). Use temporary soil trapping measures, including silt barriers such as straw bales or silt fences (Saldi-Caromile et al. 2004). Use temporary bank protection techniques during construction (relevant to bank pull-back and revegetation; installation of deformable bank toes) (Saldi-Caromile et al. 2004). The following mitigation measures regarding suspended sediment are based on those proposed by Bash et al. (2001): • Prior to project construction, determine suspended sediment concentrations and collect information on particle size and shape as indicators of the nature of existing turbidity. • When evaluating cumulative impacts from turbidity, consider information from existing assessments of watershed condition to account for point and nonpoint source pollution loads from watershed sources other than the project, as well as legacy impacts of the system. • Set stockpile areas back from the bank and include erosion prevention BMPs, such as silt fencing and tarp covers. Use spill prevention plans and pollution and erosion control plans.

Mechanism of Impact	Conservation Measures[ab]	Best Management Practices
		To minimize noise generation: - Avoid use of impact hammer during any pile installation. - Use air bubble curtains and/or pile caps to attenuate sound pressure waves. - Fabric barriers or cofferdams can also serve to attenuate sound generation. Require that construction vessels and propellers are washed and free of noxious weeds or invasive animals prior to entering water. Avoid barge grounding. Avoid propeller scour. Require a spill prevention plan.
Channel Process Modifications	Adhere to guidelines in *Stream Habitat Restoration Guidelines* (Saldi-Caromile et al. 2004) and *Integrated Streambank Protection Guidelines* (Cramer et al. 2003) for project development and implementation. Minimize structure footprint. Site structure above OHWL and as far outside the active channel as possible. Evaluate fluvial geomorphic processes, and consider natural and locally modified processes in project design and construction. Develop and maintain upland infrastructure carefully and with consideration of potential effects on slope stability (high-bank sites) (Gerstel and Brown 2006). Discourage backshore filling to create new home or other construction sites (Gerstel and Brown 2006).	For activities requiring dewatering, plan for at least a one-year flow event to occur during construction and design dewatering systems accordingly (Saldi-Caromile et al. 2004).

Mechanism of Impact	Conservation Measures[ab]	Best Management Practices
Substrate Modifications	If traditional armoring techniques are used, consider applying measures that reduce substrate and wave impacts (e.g., floating energy attenuators, weir-like revetments, walls open near bottom) (Cox et al. 1994). Minimize area of large substrate placement. Use suitably sized materials to minimize potential for displacement and scatter during high-flow or storm events. Site structure above OHWL and as far outside the active channel as possible. Reduce slope and/or integrate vegetated or riprapped bench areas, supporting sediment retention (Zelo et al. 2000).	Schedule construction for times when project area is dry (or substrate is frozen) (Saldi-Caromile et al. 2004).
Habitat Accessibility Modifications	Locate bank protection structures as far outside of the floodplain as possible to minimize the potential for precluding access to off-channel areas.	No specific best management practices identified.
Aquatic Vegetation Modifications	Avoid impacts by locating structures away from aquatic vegetation, especially eelgrass, whenever possible. This will require a pre-construction survey of vegetation location, species assemblage, and density. Require post-construction monitoring of vegetation for up to 10 years to investigate potential project impacts.	Minimize the area of impact by using land-based construction operations that avoid trampling of aquatic vegetation. Avoid barge grounding. Avoid propeller scour.
Riparian Vegetation Modifications	Promote bank stability by leaving as many existing trees and vegetation in place as possible, early seeding in disturbed areas (Nunnally 1978). Use and/or maintain native plant revegetation as a means to stabilize banks, where possible (Gerstel and Brown 2006; Lund 1976; Knutson and Woodhouse Jr. 1983; Myers 1993; Manashe 1993; MacDonald et al. 1994; Downing 1983; Cox et al. 1994; Zelo et al. 2000).	To protect riparian habitat, construct any necessary access points and roads with the least impact possible, according to several activities listed by Saldi-Caromile et al. (2004) as lower impact:: • Access the site using an existing access point. • Access the site from the opposite bank and cross the stream (if necessary using a floating platform or driving equipment across the channel during low flows).

Mechanism of Impact	Conservation Measures[ab]	Best Management Practices
	Above high-water level, cover riprap with soil and revegetate (Lund 1976). To the extent practicable, do not permit removal or disturbance of riparian vegetation in areas with high erosion hazard (Knutson and Naef 1997). If such removal or disturbance is permitted, require replanting with native riparian vegetation or other appropriate erosion control measures. Prepare revegetation plans for projects that temporarily disturb vegetation during construction. The revegetation plans should identify areas to be replanted with native riparian vegetation when construction is complete. Replanted vegetation should be monitored over several years (up to a 10-year period), and performance standards for plant survival and non-native plant exclusion should be established and required. Submit monitoring reports to WDFW as part of the revegetation plan. Similar to the requirement of the U.S. Army Corps of Engineers (the Corps) for ESA Section 7 individual and programmatic consultations, two monitoring reports should be required, one to be submitted one year after project completion and the other to be submitted after the final required monitoring event. The monitoring reports must include information on the plant survival by species and maintenance activities (including plant replacement) needed during each monitoring cycle in order to meet performance standards. Monitoring reports should also state the cause of plant failure, a provision generally required by the Corps, NOAA Fisheries, and USFWS for Corps ESA Section 7 programmatic consultations. WDFW should prepare or locate a revegetation guidance document that describes appropriate native vegetation to use; water, shade, and soil requirements; time of year most appropriate for planting; and other pertinent information to promote successful revegetation efforts. Suggest that vegetation (specifically large trees and root wads) removed for the project be saved for later use in restoration efforts. This condition has often been required in recent individual and	• Construct any necessary access roads perpendicular to the streambank, implementing a rock work platform as needed and restoring following removal of platform. Other practices regarding access: • Clearly mark access through the riparian area to minimize impacts (Saldi-Caromile et al. 2004). • Use temporary mats to ”walk” equipment across sensitive areas, or fit applicable vehicles with extra wide tracks to reduce weight impacts and soil compaction (Saldi-Caromile et al. 2004). • In sensitive landscapes, use track-driven equipment when possible, as opposed to tire-driven, to distribute vehicle weight more evenly across surface (Saldi-Caromile et al. 2004).

Mechanism of Impact	Conservation Measures[ab]	Best Management Practices
	programmatic Section 7 consultations. Even if the material is not specifically useful for the permitted action, a WDFW area habitat biologist will generally know of ongoing or pending restoration projects in need of LWD and root wads.	
Water Quality Modifications	Manage all surface water to contain and direct it appropriately to the base of the bluff (high-bank sites) (Gerstel and Brown 2006). Evaluate and design for surface and groundwater flow issues (Gerstel and Brown 2006). Avoid placing structures in areas that may affect flow connection from cold-water groundwater sources to surface water.	No specific best management practices identified.

Notes: a) In addition to these measures and BMPs, all applicable conservation measures should be applied from the Washington State Department of Ecology's Stormwater Management Manuals for Eastern and Western Washington (Ecology 2002, 2005), and all actions should be in compliance with the Hydraulic Code and its implementing rules.

[b]) Many of the measures discussed in this table are also given in the *Integrated Streambank Protection Guidelines* (Cramer et al. 2003).

11.2.1.2 Mitigation Strategies

Mitigation for bank protection projects may be required by regulatory authorities when it is determined that the project will cause an adverse impact to species, habitats, or conservation values. General strategies may include acreage-based habitat restoration, enhancement, or creation at an on- or off-site location or the acquisition of additional high-quality habitat property for preservation purposes. Because of the long-term positive impact on habitat, many bioengineering and beach nourishment techniques are discussed and referred to in the literature as self-mitigating due to their support of additional habitat and vegetation to the project site (Cramer et al. 2003; Gerstel and Brown 2006). The *Integrated Streambank Protection Guidelines* (Cramer et al. 2003) provides a matrix that identifies the bank protection actions likely to be self-mitigating and to what extent (Chapter 5, Matrix 3 in Cramer et al. 2003). Several specific measures that may be used to mitigate for various impact mechanisms are summarized in Table 11-3.

Table 11-3. Bank Protection-Specific Mitigation Strategies

Mechanism of Impact Impact	Compensatory Mitigation Strategy	Function of Mitigation
Construction Activities	Several of the below strategies are typically used in or combined to mitigate for unavoidable construction impacts. BMPs are also used in conjunction with these measures.	Mitigate for unavoidable construction-related impacts.
Channel Processes and Morphology Modifications	Use energy dissipation structures for wave or flow (Gerstel and Brown 2006).	Reduce wave or flow energy at shoreline to prevent or stem further erosion.
Substrate Modifications	Use soft shore armoring or bioengineered solutions, some of which may be self-mitigating (Chapter 5, Matrix 3, Cramer et al. 2003). Spawning gravel supplementation or beach nourishment (may require periodic supplementation) (Zelo et al 2000; Parametrix 1985; Simenstad et al. 1991).	Reduce impact of armoring on shoreline habitat. Varied functions can be improved (e.g., long-line cabled logs can self-mitigate, contributing to ongoing capture of gravel, increase in local channel roughness and bank complexity, and protection or growth of riparian vegetation [Nichols and Sprague 2003]). Provide additional or higher-quality substrate for forage fish (nearshore marine habitats) and salmonid spawning (freshwater channel habitats).
Habitat Accessibility	Off-site construction of side channel(s) (reconnect side channel or oxbow) (Bonnell 1991; Cowan 1991).	Provide additional rearing and spawning habitat.
Aquatic and Riparian Vegetation	Replace lost aquatic vegetation and re-establish riparian buffer along bank shoreline (Saldi-Caromile et al. 2004). Retain removed vegetation for future restoration or mitigation effort (including LWD). Mitigation to eelgrass and macroalgae is best achieved through avoidance, but if this vegetation is unavoidably impacted, apply natural regrowth or transplant methods (Thom et al. 2001).	Provide additional vegetation for shoreline shading and detritus inputs. Provide additional macroalgae habitats for juvenile salmonid prey production and forage fish habitat.
Water Quality	Stormwater treatment or flow buffering for point sources (Osborne and Kovacic 1993) existing prior to bank protection project.	Improve water quality and quantity of delivery to habitat by buffering of flows and/or reduction of pollutants to the project site.

11.2.1.3 Management Strategies

Management strategies provide the best opportunity for WDFW to guide the construction and design of bank protection structures. These strategies are intended to lead to better information for design and review of projects, enhance the sharing of information, provide additional resources to contribute to lessening potential project impacts, and provide WDFW biologists and the entire department with the legal authority to prohibit activities that are not adequately protective of potentially covered species. Each of the recommendations requires additional WDFW staff availability because additional project oversight is recommended, and existing project oversight is already a significant challenge according to WDFW biologists around the State (Anchor Environmental et al. 2006).

11.2.1.3.1 Regulatory Recommendations

Regulatory recommendations are those changes to the WACs that are recommended in order to avoid, minimize, or mitigate impacts associated with bank protection structures. The WACs establish the rules that WDFW requires for bank protection projects. Many of the conservation measures, BMPs, and mitigation strategies could be incorporated into the WACs. In addition, the following regulatory recommendations have been identified:

- Require pre- and post-construction project monitoring to investigate conditions in the project area and adjacent areas.

- Require inspection during construction to ensure compliance with the HPA and a "sign off" by the inspector. WDFW could hire inspectors or license private engineering/environmental firms to inspect specific construction requirements related to fish habitat. Project components that would most benefit from inspection during construction are structural design, an instream habitat and/or instream mitigation, riparian vegetation, and revegetation progress.

- Prohibit bank protection structures that disconnect sediment sources unless life or property is at risk.

- Allow beach nourishment as a mitigation technique to address impacts of new and existing bank protection structures.

- Establish freshwater construction timing restrictions at the smallest geographic scale possible (ideally, basin-specific) based on species distributions and periodicity. Revisions to the WAC are recommended to address the lack of freshwater construction timing provisions, as well as saltwater timing provisions, based on consideration of the entire potentially covered species list to minimize the risk of take.

- Establish partnerships with other entities (e.g., the Corps and port authorities) to beneficially reuse clean dredged material to nourish beaches and have available as mitigation.

- Provide incentive mechanisms to promote "good" projects. Examples of potential incentives are simplified and accelerated permit review (i.e., "top of the stack") and conducting or funding the monitoring activities required for the project. Such monitoring is envisioned to be conducted by crews (similar to Washington Conservation Corps or Ecology Youth Corps crews) whose sole responsibility is monitoring, rather than by WDFW biologists.

- As incentive, identify grant funding opportunities for projects incorporating habitat restoration components.

- Limit programmatic coverage to certain size, types or locations of bank protection structures. For example, the USACE (2002) Nationwide Permit 13 limits the size of the proposed bank protection structure to 500 feet or less in order to be eligible under the programmatic coverage. Similarly, the USACE (2005) Regional General Permit for Pend Orielle River and Lake Chelan limits the size of a bank protection structure to 250 feet or less.

11.2.1.3.2 Education Recommendations

The recommendations focused on public education specific to bank-protection projects include:

- Educate the public on potential impacts of bank protection projects and alternative techniques available.

- Develop a paper or web-based resource that highlights representative “good” and “bad” bank protection projects to help citizens understand the differences. The resource could consist of concise case studies for a variety of marine, estuarine, and freshwater settings (e.g., Eastern and Western Washington; feeder bluffs and accretion shoreforms; large, moderate, and small systems; high gradient and low gradient).

- Educate the public on the limitations of bank protection projects at providing full protection from extremely high-flow events to discourage construction close to shorelines or bluffs.

- Have staff available to assist in project design and/or implementation of the *Integrated Streambank Protection Guidelines* (Cramer et al. 2003) and the *Stream Habitat Restoration Guidelines* (Saldi-Caromile et al. 2004).

11.2.2 Shoreline Modifications

This section provides recommendations of strategies for the protection, conservation, mitigation, and management of HCP species based on a review of the scientific literature of shoreline modifications. Because of the nature of the scientific literature (i.e., papers are written years after an action has been taken), some of these recommendations may already be commonplace. However, it is important to document support for those activities that have a basis in empirical science. Where citations are not provided, it should be assumed that no direct evidence for that recommendation exists, but the recommendation is based on a reasonable conclusion from the collective information surveyed in preparing this white paper.

11.2.2.1 Jetties

Jetties cause more damage to nearshore ecosystems than any other single shoreline modification measure. They intercept littoral transport, cut off groundwater supply, and disturb natural nearshore circulation. They also encourage vessel traffic. There are two primary ways to reduce major littoral and nearshore circulation impacts from jetties.

One of the easiest means for reducing the impact of littoral drift disruptions is to develop a sediment bypass strategy (see Figure 11-1). The strategy typically consists of collecting sediment on the updrift side of a set of jetties (in the deposition basin in the figure). This sediment is then dredged and piped, trucked, or barged to the downdrift side of the other jetty. Sediment bypass is a common practice along the Gulf Coast (Seabergh and Kraus 2003) but has seen limited application in sheltered settings (NRC 2007) like Puget Sound. Although large tidal fluctuations can complicate the design, a large number of installed systems have indicated that the mean tide level is a reasonable crest elevation (Seabergh and Kraus 2003). In Puget Sound, for example, this would allow much of the sediment to bypass the jetty in the active sediment transport corridor on the upper foreshore (Finlayson 2006).

The alterations of the water column (tidal prism, nearshore circulation, stratification, etc.) in the vicinity of the jetty are more difficult to mitigate. However, the possibility exists to use engineered logjams (ELJs) or other secured (untreated) woody debris to provide the same function as a riprap or walled structure. These types of structures have been used successfully as groins on riverine shorelines, even in locations where critical infrastructure is meant to be protected (Herrera 2004). To prevent the isolation of channel waters, these structures can be built to be semi-permeable to allow for water with differing salinity to pass through, thus minimizing the impacts on nearshore circulation. Such structures can also be built so that fish can pass through them.

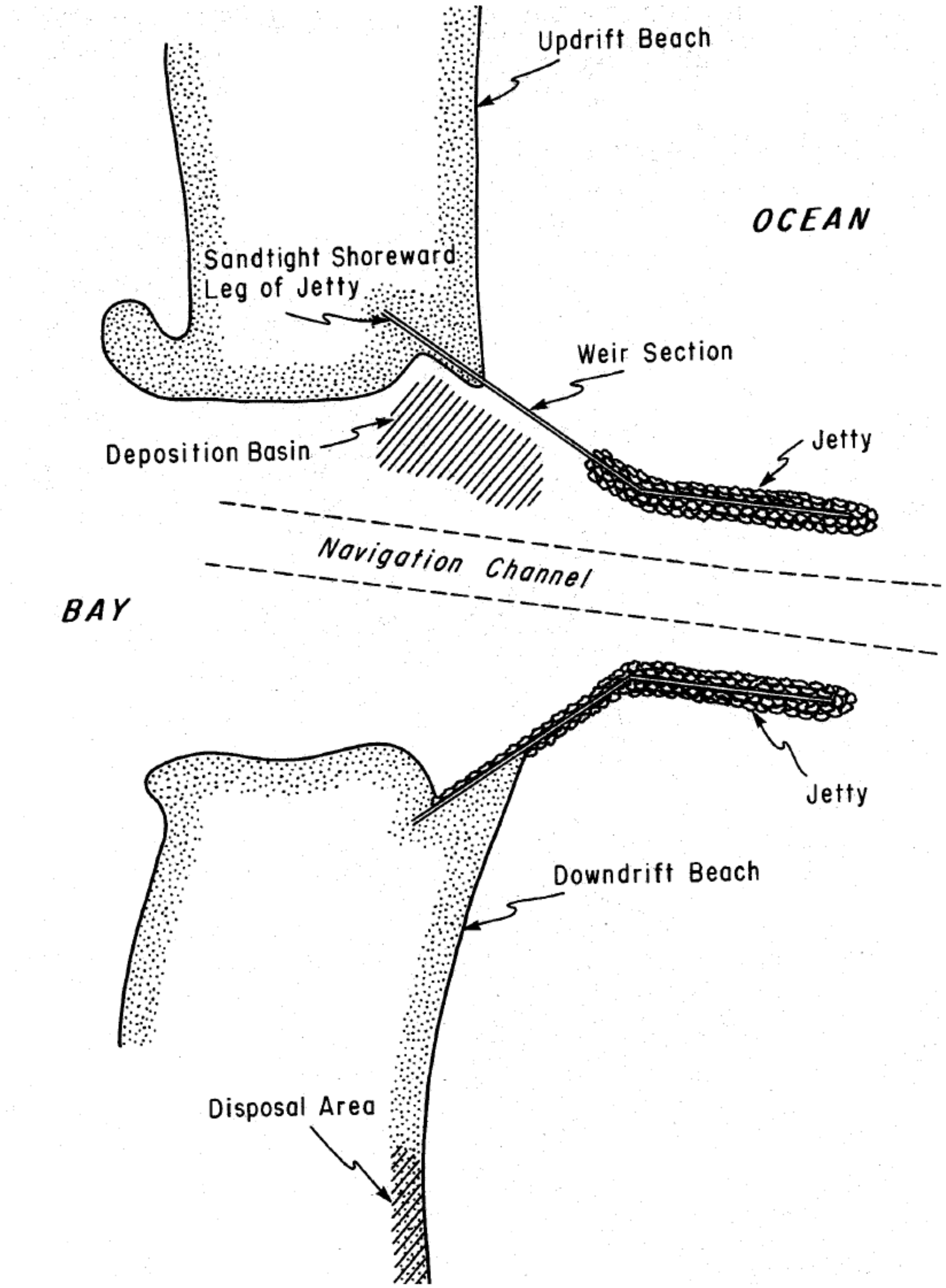

Figure 11-1. Schematic of sediment by-pass system (Seabergh and Kraus 2003).

11.2.2.2 Breakwaters

Thirty years of research on breakwaters and artificial reefs has led to a rich data set that has identified pitfalls in offshore, shore-parallel, hard-substrate deployment.

- Where possible, use temporary, removable shore-protection measures, such as moored floats (Thompson et al. 2002).
- If a permanent breakwater is necessary, locate the breakwater(s) to best connect the activity site to other areas of hard-rock habitat in order to reduce the probability of an invasive species infestation (Thompson et al. 2002).
- Use clean earthen materials where possible (i.e., no materials that would leech metals or other exotic organic compounds [e.g., creosote-treated wood]) (Thompson et al. 2002).
- Avoid the use of vertical walls (Bulleri and Chapman 2004). Where possible, mimic the slope of predevelopment shoreline, which in most cases in Puget Sound is between 6:1 and 10:1 (Finlayson 2006).
- Submerge the breakwaters where possible (i.e., in areas of small tides and large waves, the outer coast) (Thompson et al. 2002).
- Where possible, use removable, temporary floating breakwaters in place of permanent, continuous breakwater walls (Thompson et al. 2002).
- Avoid simple geometric designs. A complex landscape has been shown to be more productive for wide variety of fishes than simple geometries (Moschella et al. 2005; West et al. 1994).
- Provide a rough, complex surface on which a variety of organisms can colonize. Gullies and small caves can be especially fruitful (Moschella et al. 2005), particularly if they are large enough to allow sand to accumulate (Fabi et al. 2006).

11.2.2.3 Groins and Bank Barbs

On marine shorelines, the impacts of groins or groin-like structures can be minimized by following these guidelines:

- Minimize the structure’s cross-section in the shore-parallel direction.
- Minimize groin wetted length.
- Use earthen materials or untreated wood where possible.
- Avoid the use of structural members that would interrupt groundwater exchange between the sea and the shore.

- Avoid the use of vertical walls (Bulleri and Chapman 2004).

- Allow wrack to accumulate in and around the structure.

- Reduce the protrusion of the structure into the flow as much as possible.

On riverine shorelines, modifications to stabilize banks should mimic natural geomorphic and riparian conditions to the extent possible to limit incidental take. Along riverine shorelines, this would include the placement of engineered logjams (see Figure 11-2), the reconnection of floodplains, and the restoration of riparian forests (Collins et al. 2003). In general, groins and bank barbs provide greater habitat diversity than simple rock revetments (Hjort et al. 1983; Li et al. 1984), and thus are preferred over the construction of rock revetments. Because rivers are dynamic systems, localized bank stabilization efforts can shift the ongoing channel response to an adjacent river segment (Leopold et al. 1964). Bank hardening projects should therefore consider such impacts and take appropriate measures to mitigate these effects.

Figure 11-2. Example of bank protection before (left) and after (right) removal of the rock revetment and installation of engineered logjams in the Mashel River near Eatonville, Washington.

To repair an existing groin, a licensed engineering geologist with experience evaluating projects should determine if removing the structure will cause more damage to the shoreline than letting it remain, or if significant impacts will occur to life or property if the groin is removed. Erosion occurring along adjacent beaches as a result of pre-existing geomorphic conditions near the property should not be considered a significant impact. In addition:

- The replacement structure should be designed to allow uninhibited passage of alongshore sediment movement.

- The footprint along the shoreline should be minimized to the greatest extent possible.

11.2.3 Overwater Structures

11.2.3.1 Shading

Nightingale and Simenstad (2001b), Carrasquero (2001), and Thom et al. (1995, in Haas et al. 2002) provide impact minimization measures for the design, construction, and revetment of a variety of overwater structures. WDFW might want to consider following the guidance provided by these authors, such as:

- Increasing the height of overwater structures to allow light transmission under the structures.
- Decreasing structure width to decrease the shade footprint.
- Aligning the structure in a north-south orientation to allow the arc of the sun to cross perpendicular to the structure, which reduces the duration of light limitation each day.
- Using the smallest number of pilings possible, allowing more light beneath the structure.
- Using grated surfaces or including openings in the deck surface to pass light, as opposed to prisms. Gayaldo and Nelson (2006) found that grating (with 37 to 58 percent open space) transmits 10 times more light under piers than do acrylic prisms. In addition, light that passes through open grating penetrates the water evenly under the pier, whereas light transmitted though prisms concentrates beams of light that do not always reach the water surface. The U.S. Army Corps of Engineers Regional General Permit for residential overwater structures in inland marine waters within Washington State (USACE 2005) requires ramps to be grated, and floats are required to have grating account for a minimum of 30 percent of the surface area; the grating must have 60 percent open area and be oriented to maximize light penetration (USACE 2005). Additionally the Regional General Permit for residential overwater structures in inland marine waters prohibits pier widths greater than 6 feet, float widths greater than 8 feet and lengths greater than 20 feet, and the construction of new or the modification of existing fingers, “ells,” and T structures onto floats (USACE 2005).

Southard et al. (2006) provides additional recommendations on minimization measures specific to shading impacts on juvenile salmonids.

- To minimize the shade-related impacts to migrating juvenile salmonids created by ferry terminals, overwater structures should be designed and constructed to allow incidental light to penetrate as far under as possible, while still providing the necessary capacity and safety considerations necessary to support their intended function. The physical design (e.g., dock height and width, dock orientation, construction design materials, piling type and number) will influence whether the shadow cast on the nearshore covers a sufficient area and level of darkness to constitute an impediment. Construction of closely spaced terminal structures should be avoided to minimize the potential cumulative impacts of multiple overwater structures on juvenile salmonid migration (Nightingale and Simenstad 2001b).

- Experiment with technologies and designs that can soften the light-dark edge to minimize potential temporary inhibition of movement.

- The incorporation of light-enhancing technologies in the design of overwater structures is likely to maintain light levels under overwater structures greater than what is required by juvenile salmonids for feeding and schooling (i.e., estimated at between 0.0001 and 1 foot-candles, depending on age and species). To encourage daytime movement under terminals and other overwater structures, it would be beneficial to decrease the dark-edge effect as much as possible. Providing even a small amount of light in a regular pattern under a dock may encourage fish to swim underneath. Natural lighting for fish could also be enhanced if the underside of the dock were reflective.

- Continued research is needed to improve our understanding of the relationship between overwater structures and the behavior of migrating juvenile salmonids. Acoustic tagging-tracking technology should be further used to address the data gaps in our knowledge.

- Fish feeding behavior during temporary delays of movement should be investigated. If prey resources and refuge habitat are adequate, fish may benefit from holding in an area adjacent to a terminal.

Kahler et al. (2000) recommends the following measures to mitigate or avoid the undesirable impacts of overwater structures on salmonids in lakes:

- No net increase in overwater coverage should occur in the Lake Washington system — permits for new construction should be contingent on permits for replacement structures. Only replacement structures that demonstrate a reduction in overwater coverage should be permitted. The amount of overwater coverage eliminated from the replacement pier could be held in a “surface area mitigation bank,” which new piers would have to draw from. Gradually lower the total net coverage over local lakes.

- All piers, both new and replacement structures, should be restricted to a 3.5-foot-wide cantilever bridge that spans the nearshore area to a narrow moorage structure of the minimum size necessary to moor the applicant’s boat.

- Cantilever bridge structures should be grated and as high off the water as practicable, and moorage structures should be no less than 24 inches above OHWL. Floating structures should have maximum light penetration and be removed annually after boating season.

- Prisms and grating should be studied to determine their efficacy at providing sufficient ambient light for macrophyte production under piers. The best products should be utilized in all new or replacement overwater structures to minimize losses of primary productivity.

11.2.3.2 Aquatic Vegetation

Where conditions are suitable for eelgrass growth, impacts of overwater structures should be avoided or minimized by use of the following measures:

- Avoid impacts by locating structures away from eelgrass beds whenever possible.
- Minimize the area of impact by using the best available installation methods.
- Minimize shading by using the lowest possible number of pilings.
- Space pilings to minimize shade to areas suitable for eelgrass.
- Minimize dimensions of the structure to reduce shade.
- Incorporate design elements such as grated decks or deck openings to reduce shade.
- Whenever possible, orient structures to reduce the shade in habitat that is otherwise appropriate for eelgrass growth (e.g., structures oriented east-west cast a shadow on a single area for a longer period of the day than do structures oriented north-south).
- Locate the structure as high above the water as practical to reduce shade.
- Encourage shared-use docks to minimize cumulative impacts.
- Remove floats during off season and store at an upland location.
- Avoid vessel impacts to eelgrass by maximizing the vertical and horizontal distance between vessel propellers and eelgrass to the extent practicable, maintaining a minimum clearance of 1 foot below the propeller.

Adopting these measures would likely result in avoidance and minimization of eelgrass and macroalgae impacts to the greatest extent practicable. However, it is likely that some projects would still require compensatory mitigation to completely offset temporal loss of eelgrass function and site-specific and cumulative impacts on eelgrass.

11.2.3.3 Substrate Modifications

In the nearshore environment, where overwater structures alter the benthic environment via shellhash deposition and establishment of invertebrate communities on pilings, use of fewer and more widely spaced pilings will help to reduce sea star and crab bioturbation of the benthos (Thom et al. 1995, in Haas et al. 2002).

Prohibiting overwater structures from grounding out during low tide events will avoid potential impacts such as affecting aquatic organisms by directly crushing the organisms or changing the character of the substrate. The U.S. Army Corps of Engineers prohibits the grounding of floats on tidal substrates at any time in their Regional General Permit No. 6 (USACE 2005).

11.2.4 Marinas and Terminals

General methods to minimize and mitigate the impacts from construction apply to marinas and terminals. In addition, site selection , facility design strategies, and operations best management practices can be used to avoid, minimize, and mitigate negative impacts of marinas and terminals on habitats and HCP species.

11.2.4.1 Facility Planning

11.2.4.1.1 Site Selection Strategies

- Site marinas/terminals away from areas with littoral and aquatic freshwater vegetation, where practicable.
- Locate marinas/terminal in areas that are naturally deep enough to avoid resuspension of sediments associated with prop wash.
- Locate new shipping terminals and marinas: (1) in existing developed areas where nearshore areas have already been dredged, or (2) in areas where the natural bathymetry of the shoreline steeply drops off close to shore.
- Locate marinas/terminal in areas with low or impaired biological integrity.

11.2.4.1.2 Facility Design and Operation Strategies

All Environments (Marine, Riverine, and Lacustrine)

- If possible, all marina activities should include some component of on-site habitat enhancement. Types of enhancement include prototype (soft) shoreline stabilization techniques, planting, and beach nourishment. The mix of these activities should be consistent with the preactivity conditions.
- In the design of the marina itself, a number of alterations to traditional designs can minimize the impacts associated with hydraulic and geomorphic modifications. For instance, submerged breakwaters and weir jetties should be used in place of structures that are exposed. However, both submerged and emergent breakwaters have hydraulic and geomorphic impacts on adjacent areas. Weir jetties are particularly effective at reducing the littoral disruptions associated with marina activities, especially when accompanied by a strategy of beach nourishment and sediment routing (Seabergh and Kraus 2003).
- Construction of marina/terminal facilities often involves project activities such as channel modifications, bank hardening, groins and bank barbs, and other such projects. If hydraulic and geomorphic modification cannot be avoided, identify the area affected by the impacts. For example, the area of alteration includes areas affected by embedding, scour, or deposition. For projects of this size, hydraulic modeling of potential impacts using well-established sediment transport models should be required (Miller et al. 2001).
- Minimize width of the structure over the water.

- When applicable and feasible, require construction of longer marina/terminal decks that keep vessels in deeper water to avoid/minimize propeller wash effects on sensitive habitat areas such as eelgrass beds.

- Residential/recreational floats should be sited in deeper water to reduce the potential impacts associated with propeller wash.

- Use the smallest number of pilings necessary to carry the load.

- Design pile-supported structures with maximum open space between pilings to allow waves, currents, and sediment to pass beneath.

- Allow light transmission wherever possible along the shallowest areas of migratory corridors and over any areas near or adjacent to submerged aquatic vegetation.

- Locate the structure as high as practical to increase light transmission.

- Use light-reflecting materials on underside of docks, whenever feasible.

- Consider solar-powered artificial lighting under the dock, if light transmission is not possible. However, compared to full sunlight, grating transmits 10 times more light under a pier than, for example, acrylic prisms (Gayaldo and Nelson 2006); hence, the use of grating is always a better option than prisms.

- Construct marinas/terminals so that most of the overwater coverage is beyond the photic zone.

- Increase the distance between the dock and the water to allow greater light penetration.

- Place the potential shade-casting structures perpendicular to the arc of the sun (i.e., north–south placement) to maximize transmission of light under the structure.

- Install grating with maximum open spacing and ensure that the open space is kept uncovered or unshadowed by other pier features or gear.

- Orient grating to maximize transmission of light under the structure.

- Promote community-use docks to minimize the proliferation of single-family residential docks along shorelines.

- Site slips for smaller boats in shallow water, with slips for larger boats placed in deeper water.

- Facilities should be sited, if possible, so that dredging is not required.
- Require the use of rub strips on treated wood piles or timbers that are abraded by vessels (fender piles) or docks (guide piles) to reduce physical breakup of the piles.
- Use low-intensity artificial nighttime lighting and shield the lighting to prevent artificial light transmission to the ambient nighttime underwater light environment.
- Encourage the use of upland boat storage areas and the use of slings.
- Require that stormwater runoff be 100 percent contained.
- Encourage designs that create shallow-sloped pocket beach areas instead of continuous vertical bulkheads or riprap.
- Avoid the use of continuous sheet (impermeable) piles and encourage the use of permeable geomaterials (e.g., geotubes) (Oh and Shin 2006).
- Where possible, mantle the waterward side of the pile with natural materials (i.e., sediment consistent with the environment).
- Where possible, use removable, floating breakwaters in place of permanent, continuous breakwater walls.

Additional Considerations in Marine Environments

- Minimize the amount of pier area that directly contacts the shoreline to allow light penetration to the nearshore intertidal and shallow subtidal areas.
- Locate breakwaters to best connect to other areas of hard-rock habitat.
- Use submerged breakwaters in place of exposed breakwaters where appropriate (i.e., in areas of small tides and large waves, on the outer coast).

Additional Considerations in Lacustrine Environments

Considering that many reservoirs are located in a distinctly different climate than those in marine settings, there are some BMPs that are specific to these projects.

- Rock cribs, similar to jetties, were found to provide structural complexity for smaller fish in Lake Tahoe, California (Beauchamp et al. 1994), but this advantage may be outweighed by the interception of spawning materials from deposition in littoral zones and increased deposition of fine materials on rocky substrate. Coves, especially with inundated herbaceous vegetation, were found to yield the largest numbers of young fish in four Mississippi reservoirs (Meals and Miranda 1991).

11.2.4.1.3 Navigational Channel and Berthing/Maintenance Dredging

General recommendations to avoid and minimize the impacts of dredging are provided in the Dredging: Marine Issues white paper (Nightingale and Simenstad 2001a) and include:

- Use multiseason pre- and postdredge project biological surveys to assess animal community impacts;
- Incorporate cumulative effects analysis into all dredging project plans;
- Use landscape-scale planning concepts to plan for beneficial use projects most suitable to the area's landscape ecology and biotic community and food web relationships;
- Identify turbidity and noise thresholds to assess fish injury risks; and
- Analyse and synthesize what is known about the spatial and temporal distribution of fish and shellfish spawning, migration behaviors, and juvenile rearing to evaluate environmental windows for dredging on a site-specific basis.

The following recommendations are intended to reduce the effects of dredging on HCP species.

- For new marine, riverine, and lacustrine projects and significant expansions beyond general maintenance dredging, thoroughly assess the large-scale, cumulative impacts of the resulting changes in bathymetry, habitat loss, and change to estuarine/nearshore marine ecosystem dynamics (e.g., salinity intrusion).

- Require hopper dredges, scows, and barges or any other equipment used to transport dredged materials to the disposal or transfer sites to completely contain the dredged material.

- For long-term projects where continuous dredging and on-loading to barges occurs, require periodic movement of the barge to reduce unnecessary shading.

- Modify in-water work windows to take into consideration what is known about site-specific spatial and temporal distribution of fish and shellfish eggs, larvae, and juveniles.

- Evaluate the application of in-water work windows on a site-specific basis based upon the location and features of the site, such as sediment composition, plant and animal assemblages, and timing of seasonal and migration patterns.

- Use presampling bathymetric surveys, records from previous dredging events, and best professional judgment to estimate the volume of sediments likely to be dredged; base sampling and testing requirements on this estimated volume.

- Avoid projects and expansions that convert intertidal to subtidal habitat. If such conversion is unavoidable, employ comprehensive, large-scale risk assessment to identify the cumulative effects of site-specific changes on ecosystem dynamics.

- Select dredging equipment types according to project-specific conditions, such as sediment characteristics.

- Base turbidity threshold testing for dredging operations upon background site turbidity.

- In areas where dredging is proximal to sensitive habitats (or in projects where sediments both suitable and unsuitable for unconfined open water disposal will be dredged adjacent to each other), use a computerized electronic sensor system to monitor dredging operations. Such tools can assist in operational documentation and regulatory compliance by providing record accessibility and clarity. It also offers advantages for planning, estimating, and managing dredging activities.

- Increase the use of multiseason preproject surveys of benthos to compare with post project surveys to understand dredging impacts.

- Where applicable and involving uncontaminated sediments, consider beneficial use of dredged materials that can contribute to habitat restoration, rehabilitation, and enhancement, particularly for projects that incorporate a landscape ecology approach.

- Avoid beneficial use projects that impose unnatural habitats and features on estuarine, marine, and riverine landscapes.

- Dredging should be conducted to a depth not greater than a navigation channel depth at the seaward end. If necessary, authorize dredging to depths greater than the navigation channel at the seaward end only in berthing areas and turning basins for commercial shipping purposes.

- Use hydrodynamic models to predict system-wide changes in salinity, turbidity, and other physicochemical regimes for project assessment planning that avoids or minimizes impacts on aquatic habitat.

11.2.4.1.4 Vessel Activities

- Manage vessel operations to minimize the adverse effects of prop wash.

- Take precautions to avoid impacts from accidental spills of fuel and contaminants and post guidelines and protocols for handling spills for all personnel to view.

- Provide spill response training.

- Establish guidelines and protocols to avoid introduction of invasive species.

- If aquatic vegetation is present at or adjacent to a pier or wharf facility, establish guidelines and protocols outlining where vessel traffic should occur when entering or leaving the site.

11.2.5 Habitat Modifications

Habitat modification projects are designed to protect, create, and/or restore habitat. Unfortunately, not every project is successful and some projects result in habitat degradation due to poor design or site constraints. This section presents suggested measures which should be followed during the construction and design phase to increase the chance of success of the projects.

11.2.5.1 Beaver Dam Removal/Modifications

The habitat and species impacts of beaver dam removals can be decreased through a number of measures.

- Gradual drawdown of the beaver impoundment is important because it reduces the mobilization of sediments within the impoundment and provides motile organisms more time to evacuate the pond. This "notching" technique is frequently used in small dam removals (Doyle et al. 2003; Stanley et al. 2002).
- Other strategies to manage beaver include the application of flow devices which control the pond level so that flooding conditions are alleviated (Beaver Solutions 2007). This management strategy is ideal because the positive environmental benefit of the beaver pond is not lost while flooding issues are resolved simultaneously.
- Other strategies include the use of enlarged culverts. Beaver often use culverts as dam sites but research has shown that the application of enlarged culverts discourages dam building near the roadway (Jensen et al. 2001). It has been estimated that over the life of the enlarged culvert, the costs of installation will be less than those associated with beaver management activities (Jensen et al. 2001).

11.2.5.2 Large Woody Debris Placement/Movement/Removal

Historical forest clearing, river snagging, and splash damming has greatly reduced the quantity of woody debris in rivers and streams of the Pacific Northwest (Collins et al. 2002; Montgomery et al. 2003). Snagging records from the region suggest that wood loading in large Pacific Northwest rivers was 100 times greater than present-day wood loading and contained larger trees (Collins et al. 2002). Restoration efforts that increase wood loading must also consider the size and placement of wood pieces that will provide the stability necessary for habitat protection and function.

Logjams consisting of small pieces of wood are less stable than those jams anchored by large, key members (Braudrick and Grant 2000). Maclennan (2005) noted that overloading of loose wood in two Puget Sound estuaries resulted in reduced diversity and abundance of aquatic vegetation. Studies such as these suggest that stabilizing wood or adding wood that will not mobilize during flood events should be the goal of most LWD additions. If structural stability is a major goal of LWD additions, it is vital to either place large pieces of wood that will not move during the design flood event or provide stability using other means, such as piles. Observations from the undisturbed Queets watershed show that the size of key members capable of forming stable, natural logjams varies with channel depth and width (Abbe and Montgomery 2003).

Other factors increasing stability include the ability of logjams to accrete additional wood delivered by floods and the root cohesion and added roughness provided by vegetation growing on accumulations of woody debris.

The structural failure of wood placed in aquatic environments can impose construction impacts on HCP species. The adverse ecological impacts on HCP species caused by the structural instability and failure of instream woody debris can be minimized or avoided by ensuring that wood placed in rivers is properly engineered according to accepted engineering guidelines. Such guidelines are currently under development by the Washington chapter of the American Council of Engineering Companies (ACEC). Project success depends on a thorough understanding of the site-specific geomorphic constraints, quantifiable habitat goals, and the development of performance-based criteria that account for the anticipated hydrodynamic forces and the desired factor of safety for stability (Miller and Skidmore 2003; Slate et al. 2007). In addition, WDFW has published a series of guidelines through the department's Aquatic Habitat Guidelines Program. These guidelines are available from WDFW and include the Stream Habitat Restoration Guidelines document (Saldi-Caromile 2004), which provides guidance for habitat assessment, the development of restoration goals, and implementation of habitat restoration techniques.

Constructing stable logjams may involve harvesting large conifer trees from the few remaining patches of old-growth forest in the region. Habitat-protection measures can include the use of wood from blow-down or the wholesale purchase of trees from commercial harvest projects. Alternative sources of wood, such as salvaging trees from reservoirs, should also be considered to provide habitat benefits that will outweigh the impacts associated with project construction. As mentioned above, the use of piles in engineered log structures can eliminate the need for large, key-member logs that would otherwise be required for stability.

11.2.5.3 Spawning Substrate Augmentation

With the realization that early gravel augmentation projects were failing due to a lack of consideration regarding stream hydraulics and site geomorphology (Kondolf et al. 1996), practitioners began adopting new techniques and management strategies. It is recommended that every gravel augmentation project be based upon information gathered from detailed monitoring of site conditions and a geomorphic analysis of the reach, including estimates of sediment transport rates. Given the stochastic nature of riverine systems, even with detailed hydrologic and geomorphic information, the outcome of the project may be uncertain. Consequently, an adaptive management approach is recommended whereby the project is designed as an experiment. Bunte (2004) recommends the following adaptive management steps when conducting a gravel augmentation project:

1) *Pre-project Analysis*: In this step information is collected to formulate a conceptual model that explains how the stream should ideally function with an active gravel bed. For sustainable geomorphological and biological functionality, a channel shape must be attained in which:

- The 1.5-year recurrence interval flow fills the channel to its morphological bankfull stage

- Gravel is partially mobile every 1-2 years

- The flow regime is seasonally variable
- The timing of high and low flows corresponds with the needs of the salmon population.

2) *Measuring and modeling*: In this step data are collected and used to model the reach to predict gravel mobility and channel form under different restoration and stream discharge scenarios. The information derived here is used to inform the next step.

3) *Monitoring, evaluation, adjustment*: In this post-project step the site is monitored and evaluated to quantify channel response to the gravel augmentation. This step is vital and frequently not included in most restoration efforts due to a lack of funding. It is in this step that information regarding where the project may have gone wrong and how it might be remedied is collected. Without this step the project may fail and no lessons will have been learned. The ramifications of gravel augmentations are poorly understood (CALFED 2005), and it is only through projects which include monitoring programs that the science of gravel augmentation will progress.

11.2.5.4 In-Channel/Off-Channel Habitat Creation/Modifications

As with gravel augmentation, the prediction of the outcome of the majority of in-channel restoration work has some associated uncertainty. To reduce the risk associated with this uncertain outcome, a strategy must be in place to address the potential failure of the project. Project failure has been a common occurrence in past in-channel restoration efforts (Babcock 1986; Frissell and Nawa 1992) and every measure should be taken to prevent the failure of future projects. Suren and McMurtrie (2005) suggest that in-channel restoration efforts should focus on watersheds which have a natural hydrograph and minimal sediment loading. They argue that external drivers will dictate reach scale dynamics and that without a watershed based approach reach-scale restoration will be useless. In a separate study, Frissell and Nawa (1992) monitored 161 instream structures and found that 60 percent of the structures had the opposite of the intended effect on the stream. They attributed the high failure rate to the fact that structures were placed in both high velocity and sediment laden reaches. Other studies have found instream structures placed in Pacific Northwest streams to be more durable, with only a 20 percent failure rate after 5-year recurrence interval flood events (Roper et al. 1998). Regardless, most research indicates that instream structures are more likely to fail in large rivers (Roper et al. 1998), high energy environments (Frissell and Nawa 1992), and when sediment loading is elevated (Frissell and Nawa 1992; Suren and McMurtrie 2005).

These studies suggest a harsh reality which is that in-channel restoration is least likely to succeed in those reaches that need it the most. Streams with flashy hydrographs caused by watershed deforestation or urban development, streams with high sediment loads from anthropogenic disturbance in the watershed, these are the degraded systems that restoration practitioners focus on. These are also the systems where most restoration practitioners fail to achieve their goals. The recommendation which results from these studies is to focus in-channel rehabilitation efforts on those channels that have a natural hydrograph and average sediment loading. In more heavily impacted systems, a top-down approach whereby hydrology is addressed on the watershed scale may be more appropriate and effective (Roni et al. 2002).

Off-channel habitat modification has a higher success rate that in-channel work because the site does not receive the same flood induced shear stresses. The primary avenues of failure for off-channel habitat modification are infilling or isolation from the main channel, and improper hydrologic design. If off-channel habitat is too intimately connected with the channel, then the goal of increased habitat diversity will not be achieved. If the off-channel site is too disconnected from the channel, then entrapment may become an issue. This suggests that the most vital aspect of an off-channel habitat modification is the amount and duration of flows which flush the off-channel habitat. A recent study by Henning et al. (2006) indicated that floodplain habitat (i.e., enhanced wetlands) with flow control structures that provided an outlet for fish emigration and a longer hydroperiod for rearing, produced significantly higher age-1 coho abundance than unenhanced wetlands. Studies such as this suggest that off-channel restoration efforts should focus on the connectivity of the habitat with the main channel when designing the project.

11.2.5.5 Riparian Planting/Restoration/Enhancement

Some have argued that the maintenance of a healthy riparian system should be paramount in channel rehabilitation and should take precedence over in-channel work (Opperman and Merenlender 2004). Riparian restoration is indeed a powerful tool, but the project must be properly conducted in order for ecosystem benefits to be realized. As with most restoration efforts that do not attempt to remedy the processes driving ecosystem degradation, riparian planting will be best applied in watersheds that are either minimally or moderately impacted. If riparian planting is to be performed in highly degraded watersheds, the work needs to be conducted within the context of larger watershed restoration efforts. Riparian rehabilitation efforts which create a narrow corridor of improved habitat downstream of a degraded watershed may not improve stream conditions (Teels et al. 2006). In a study of forest fragments in agricultural areas of the South Island, New Zealand, Harding et al. (2006) found that forest fragments of 5-7 ha, located in the lower reaches of the study catchment did not mitigate the negative effects of upstream agriculture on stream functioning. They concluded that in order for a riparian buffer to be maximally effective the buffer should extend to all channels in the distributary network, even small first order tributaries.

A number of researchers have conducted literature reviews of the many riparian restoration research projects which have been performed since the practice became widespread in the 1970s (Hickey and Doran 2004). These synthesis papers have a number of recommendations regarding buffer width:

- Buffers should be between 33 and 165 ft (10 and 50 m) wide for effective nitrogen filtration (Mayer et al. 2005).

- Buffers should be greater than 98 ft (30 m) in width for effective nitrogen and phosphorus filtration (Hickey and Doran 2004).

- Forested buffer width should extend to the edge of the floodplain to reduce the impact of upslope silviculture practices on stream microclimate (Anderson et al. 2007).

- Buffer widths of 98 ft (30 m) or greater are required to protect the ecological integrity of the stream (Broadmeadow and Nisbet 2004).

The standard practice today is to create or maintain buffer widths that are approximately 3.2–33 ft (1–10 m) in width (Hickey and Doran 2004). The available research indicates that this range is too narrow to protect ecosystem functioning and that widths in excess of 98 ft (30 m) are preferable.

11.2.5.6 Wetland Creation/Restoration/Enhancement

Research has indicated that floodplain wetlands are most productive when hydraulic residence time on the floodplain is on the order of 2 to 10 days (Ahearn and Dahlgren 2005; Hein et al. 2004). Additionally, studies have indicated that when residence time on floodplains is below this threshold the floodplain becomes a net sink for algal biomass instead of a net source (Ahearn and Dahlgren 2005; Tockner et al. 1999). This suggests that small floodplain restorations may not increase food resources within the waterway and that restoration efforts should focus on large floodplains (or small floodplains which receive relatively low volumes of water). There is a delicate balance in the hydroperiod of restored wetlands; too much connectivity between the wetland and the main channel and the productivity of the wetland decreases; too little and the export of food resources to the channel is decreased while the probability of fish stranding on the floodplain increases. In a study located in the lower Chehalis River, Henning et al. (2006) collected juvenile Pacific salmon data in both natural wetlands and in wetlands that were enhanced with weirs designed to promote connectivity. They found that enhanced wetlands had significantly higher age-1 abundance than unenhanced wetlands that were a similar distance from the main-stem river. This study suggests that measures which promote connectivity between riparian wetlands and adjacent water bodies will benefit native fish species.

Several studies have examined the effectiveness of salt marsh restoration practices (French and Stoddart 1992; Williams and Orr 2002; Hood 2004; Konisky et al. 2006; Simenstad et al. 2006). These works lead to the following recommendations:

- Ensure that the marsh has not subsided below the elevation required for emergent marsh vegetation and, if so, provide sediment source such that this elevation will be reached shortly after the project has been constructed (Williams and Orr 2002).

- Consider the geomorphology of both the seaward and landward tide-channel network when designing the dimensions of tide channels (Hood 2004).

- Consider the project within the broader geomorphic context (Simenstad et al. 2006).

- Where possible, remove all dike structure so as not to compromise or constrict the tide channel network (Hood 2004).

11.2.5.7 Beach Nourishment/Contouring

Several decades of beach nourishment on the east coast and in Europe provide a track record of nourishment activities where nearshore organisms have been established (Speybroeck et al. 2006). Studies from these locales will be particularly germane to projects on the outer coast of

Washington, and in similar high-energy, sandy environments. Based upon an exhaustive survey of this work, (Speybroeck et al. 2006) makes the following recommendations:

- Choose nourishment grain size commensurate to the wave energy environment. Where possible, an estimate of storm wave height should be made to make this determination.
- Avoid short-term compaction by plowing immediately after construction (applicable only to sandy nourishment projects).
- Execute the nourishment in a period of low beach use by fish, birds, and other motile organisms.
- Break large nourishment projects into a number of smaller projects and stagger them such that nourishment in one reach feeds adjacent reaches (USFWS 2002).
- Select the nourishment technique consistent with the natural mode of sediment delivery (e.g., longshore transport on the outer coast and in the Strait of Juan de Fuca; bluff landslides in Puget Sound).

Other work, some of which has been performed in settings more typical of western Washington, similar to that of Puget Sound, has put forward other recommendations:

- Completely remove former bulkhead materials where possible (Gerstel and Brown 2006).
- Avoid using dredged materials from nearby marine elevations above wave base (Demir et al. 2004).
- When large projects are undertaken, reduce the size of individual renourishment placements by subdividing the site and alternately nourishing different portions of the project site (Munoz-Perez et al. 2001).
- Avoid nourishing areas immediately adjacent to eelgrass beds. If nourishment is carried out near eelgrass, ensure that sedimentation rates in the affected meadows do not exceed the rate found by Mills and Fonseca (2003) to cause significant mortality (i.e., >25% of the average stem length).

11.2.5.8 Reef Creation

Thirty years of artificial reef research have led to a rich data set that has identified pitfalls in offshore hard-substrate deployment (Thompson et al. 2002). The following suggestions are taken directly from this work:

- Locate the reef to best connect the activity site to other areas of hard-rock habitat to reduce the probability of an invasive species infestation (Thompson et al. 2002).

- Use clean earthen materials where possible (i.e., no materials that would leech metals or other exotic organic compounds, e.g., creosote-treated wood).
- Avoid the use of vertical walls (Bulleri and Chapman 2004).
- Place reefs completely below wave and tidal influence to minimize hydrogeomorphic disturbance to adjacent shorelines.
- Avoid simple geometric designs. A complex landscape has been shown to be more productive for a wide variety of fishes than simple geometries (Moschella et al. 2005; West et al. 1994).
- Provide a rough, complex surface on which a variety of organisms can colonize. Gullies and small caves can be especially fruitful (Moschella et al. 2005), particularly if they are large enough to allow sand to accumulate (Fabi et al. 2006).
- Use stable materials only. Materials that decay or that can become mobile during storms can endanger the communities that inhabit the reef and ultimately reduce fish numbers (USA-Today 2007).
- Protect reef areas from fishing (Guidetti et al. 2005).

11.2.5.9 Eelgrass and other Aquatic Vegetation Enhancement

Eelgrass planting has been traditionally considered a difficult enterprise (Thom 1990). However, recent work has demonstrated that it is possible to restore eelgrass populations if an adaptive management strategy is undertaken from the beginning of the restorative work (Thom et al. 2005). Further, Thom et al. (2005) described the elements necessary for a successful eelgrass restoration program:

- Clear goal statement—drives what is done
- Conceptual model—organizes understanding
- Monitoring—provides information for management decisions
- Evaluation framework—provides a mechanism to evaluate information openly and objectively
- Adjustment strategy—ensures clear plans and mechanisms to implement actions when adjustment is necessary
- Dissemination of information—lets others learn regionally and nationally.

In terms of the planting method, there are several eelgrass planting techniques (Pickerell et al. 2005). The most common approach is simply to manually plant adult shoots in the restoration area (Fonseca et al. 1998). Mechanized approaches have been attempted with mixed results (Pickerell et al. 2005). As any direct method of planting initiates some disturbance of the seabed (ultimately causing resuspension of sediment that is potentially harmful to nearby existing plants), several methods have sought to more closely simulate natural reproduction. In particular, Pickerell et al. (2005) put forth a technique to use buoys to broadcast seed across a particular area. This was demonstrated to be effective in encouraging the colonization of eelgrass without the impacts associated with intrusive planting, although it has not yet been proven effective in Washington State waters.

11.2.6 Channel Modifications

11.2.6.1 Dredging

A number of techniques have been developed that may be used to mitigate the effects of dredging on sensitive ecosystems (Smits 1998). However, many of these require a trade-off with regard to dredging efficiency and impacts on organisms. For example, in hydraulic dredging, the dredging rate can be adapted by increasing the amount of water pumped up relative to the amount of sediment that is dredged, which can help to reduce the extent of turbidity plumes, although the possibility of entrainment increases (Erftemeijer and Lewis 2006). Other examples of environmentally sensitive dredging equipment have been cited by Erftemeijer and Lewis (2006):

- Encapsulated bucket lines for bucket chain dredgers
- Closed clamshells for grab dredgers
- Auger dredgers
- Disc cutters
- Scoop dredgers and sweep dredgers.

In muddy environments that are underlain by sand, suction of material from below without exposing dredged material to the water column is also possible with new technology (RBW 2007). In this case, if implemented correctly, the effects to fish and invertebrates associated with entrainment and water-column turbidity could be virtually eliminated.

Measures to mitigate the destruction of aquatic resources found by Erftemeijer and Lewis (2006) include:

- confined land-disposal,
- turbidity modeling (plume prediction),
- turbidity thresholds,
- limits to allowable reduction in aquatic species productivity,
- minimizing the duration of dredging,
- seasonal restrictions to avoid fish use and aquatic flowering periods,
- limiting over-dredge quantities,
- use of silt screens,

- prohibiting dredging near eelgrass areas,
- stopping dredging when turbidity thresholds are exceeded, and
- adoption of legislation banning the use of certain (clamshell) dredging methods.

Contractual requirements have also been used to constrain the impacts on aquatic wildlife associated with dredging (Erftemeijer and Lewis 2006). In the bridge project to connect Denmark to Sweden, two major tools were introduced to ensure that dredging-induced turbidity was kept below the limits necessary to fulfill the environmental objectives and criteria of the project:

> (1) the contractor was held responsible through his contract for keeping the spill below specified limits varying in time and space, taking into consideration environmentally sensitive periods and areas;
>
> (2) a monitoring program was implemented to identify dispersal of significant turbidity occurrences, and documenting key variables related to the most sensitive benthic communities.

Dredging was stopped temporarily during peak tidal currents on twenty occasions to keep within these environmental restrictions (Thorkilsen and Dynesen 2001). These measures helped to ensure that there were no significant impacts from dredging and construction of this major infrastructure project.

Although a common practice in Washington State, the installation of physical barriers such as silt screens has not always proved as successful in practice (USACE 2005). Enclosure of dredging equipment with a silt screen is restricted mainly to use with stationary dredgers using pipeline discharge methods, and is always accompanied by some degree of leakage underneath. Protection of an environmentally sensitive area with silt screens may in some cases be viable, but only if the physical conditions of the site (especially waves and currents) allow their effective use (USACE 2005). As a result, a rigorous monitoring program is recommended to accompany any barrier method, such as silt screens.

Dredged spoils disposal presents another challenge to protect aquatic species. USACE (1983) classifies disposal into three categories: open water, confined (either in upland areas or at sea), and habitat development (usually beach nourishment in Washington State). Open ocean disposal of clean materials have been shown to have little effect on benthic invertebrate populations when strict procedures regarding release have been followed (Simonini et al. 2005). Typically, confined land disposal is more preferable when sediments are contaminated (USACE 1983). However, care should be used when disposing of sediments on land. Runoff from confined disposal sites have been shown to be a source of pollution (Peijnenburg et al. 2005). Beach nourishment of dredged materials presents its own challenges and is discussed at length in the Habitat Modifications white paper (Herrera 2007b).

11.2.6.2 Gravel Mining and Scalping

The ecological impacts and effects on HCP species of instream and pit mining can be significantly reduced or eliminated if future management of gravel mining emphasizes incentives to use alternative sources of construction aggregate such as glacial outwash deposits, reservoir deltas, quarries, and recycled concrete rubble.

If gravel mining is to occur in a riverine environment, several steps can be taken to minimize impacts on HCP species. To reduce the impacts of gravel mining on substrate conditions, Collins and Dunne (1989) recommended limiting instream gravel extraction rates to the ambient rate at which sediment is replenished by natural bedload transport processes. Additionally, quantitative site assessments should be performed to measure and document habitat changes and habitat use and preferences of salmonids before and after bar scalping activities, using both scalped and control sites.

Norman et al. (1998) offer several recommendations for planning and siting floodplain gravel mines. Wherever possible, large gravel mines should be located in uplands away from the river valley bottom. A poor second choice is to locate mining on terraces high above the active (100-year) floodplain. In Washington, upland glacial deposits offer ample rock supplies. Mining these deposits eliminates the potential for stream capture or river avulsion. Furthermore, pits in these locations have a good potential for successful long-term reclamation.

11.2.6.3 Sediment Capping

There are numerous ways in which to conduct a sediment capping project, and each technique is associated with different impacts. Some projects, like the Boston Harbor Capping Project (see Lyons et al. 2006), may require the construction of a confined aquatic disposal (CAD) cell to contain dredged material and a sediment cap. Construction of a CAD cell is associated with numerous impacts such as noise caused by pile driving, and contaminants leaching from treated wood products. Other projects may require the deposition of only a small in-situ cap. The impacts associated with these projects will be relatively small. However, independent of project size, practitioners should follow a number of common best management practices:

1. Practitioners should use clean capping material preferably dredged from areas where dredging was going to occur independent of the need for capping sediment. For instance, the sediment for the Eagle Harbor Sediment Cap was obtained from the Snohomish River Navigation Project (Palermo et al. 1998).

2. To ensure sufficient cap thickness, practitioners should account for bioturbation depth, erosion potential (USACE 1991c), and leaching potential. A minimum depth of 3 to 4 feet is recommended (USACE 1991a).

3. To avoid displacement of contaminated sediment, capping material should be of an equal or lesser density than the contaminated sediment that is to be covered (USACE 1991a).

4. Although such systems are expensive, practitioners should use an active barrier system (ABS), such as activated carbon (Murphy et al. 2006), zeolite (Jacobs and Forstner 1999), calcium carbonate (Hart et al. 2003), coke (McDonough et al. 2007), or a low hydraulic conductivity layer (Hull et al. 1999).

These recommendations can apply to any cap placement method; however, some methods have a greater impact than others. To reduce suspended solids concentrations and contaminated sediment displacement during construction, pump-down capping techniques should be used over point-dump techniques (USACE 1991b). Although more expensive, pump-down techniques allow for more control of where capping material is deposited while simultaneously reducing ambient suspended solids and contaminated material entrainment.

Capping is frequently associated with dredging, either to obtain the cap material (e.g., Eagle Harbor, Washington) or in projects where dredging spoils are capped (e.g., Boston Harbor, Massachusetts).

11.2.6.4 Channel Creation and Alignment

The adverse ecological impacts and effects on HCP species caused by channel creation and alignment activities can be diminished using techniques that are based on an understanding of site-specific geomorphic and ecological processes. For example, the engineered placement of wood, planting of riparian vegetation, avoidance of erosion-prone areas, and levee setback all illustrate techniques that can be incorporated into bank stabilization projects to promote desirable ecological outcomes. WDFW has published a series of guidelines through the department’s Aquatic Habitat Guidelines Program. These guidelines are available from WDFW and include the Integrated Streambank Protection Guidelines document (WDFW 2003), which provides guidance for assessing and selecting bank protection techniques, and the Stream Habitat Restoration Guidelines document (Saldi-Caromile 2004), which has an entire chapter devoted to of channel modification techniques.

The structural failure of wood placed in aquatic environments as mitigation for channel creation and alignment activities can impose construction impacts on HCP species. The adverse ecological impacts on HCP species caused by the structural instability and failure of instream woody debris can be minimized or avoided by ensuring that wood placed in rivers is properly engineered according to accepted engineering guidelines. Such guidelines are currently under development by the Washington chapter of the American Council of Engineering Companies (ACEC). Project success depends on a thorough understanding of the site-specific geomorphic constraints, quantifiable habitat goals, and the development of performance-based criteria that account for the anticipated hydrodynamic forces and the desired factor of safety for stability (Miller and Skidmore 2003; Slate et al. 2007).

USEPA (2007) recommends distributing small-scale practices throughout the landscape.

For activities that require dewatering, impacts can be minimized by performing work during low-flow or dry conditions and by pumping sediment-laden water from the work area to an infiltration treatment site. Disturbed areas within the channel should be stabilized with a layer of sediment corresponding to the ambient bed to prevent an influx of fine sediment once water is reintroduced to the site. Science-based protocols for fish removal and exclusion activities should be adopted to track and report the number and species of fish captured, injured, or killed. Projects should also require slow dewatering and passive fish removal from the dewatered area before initiating active fish-removal protocols. During passive fish removal, seining is recommended before using electrofishing, which carries a greater risk of mortality (NMFS 2006).

11.2.7 Water Crossings

11.2.7.1 Hydraulic and Geomorphic Modifications

11.2.7.1.1 Channel Hydraulics

HPAs typically require that structures such as bridges and culverts have capacity to convey flood flows and debris. Additional measures that can minimize impacts include finding an alternative to building the structure; siting the structure as far as possible outside of the active channel; minimizing the structure's footprint; and generally designing the structure to have the least possible effect on channel hydraulics (Bates 2003). Guidance for appropriate design of engineered channels is readily available; the Corps channel rehabilitation manual (Watson et al. 1999) provides a widely used example, and another useful source is the Corps manual *Hydraulic Design of Stream Restoration Projects* (Copeland et al. 2001). WDFW's culvert manual (Bates 2003) also provides excellent design guidance for culvert placement. Procedures for hydraulic design of culverts in steep (greater than 3 percent gradient) channels are detailed by Papanicolaou and Maxwell (2000).

Standard procedures for channel isolation and in-water work appear to be largely effective at minimizing channel hydraulic effects associated with work within the OHWL. However, some specialized additional measures may be appropriate for minimizing the risk of frac-outs from high pressure directional drilling (HPDD) water crossings. Examples of itemized measures intended to minimize the risk of frac-outs and expediently respond to their consequences are provided by Fisheries and Oceans Canada (2006) and California Coastal Commission (2000). WDFW may consider adopting these measures as rule (appropriately, within WAC 220-110-100(3)), periodically reviewing and revising them in consultation with the federal agencies and requiring them for all HPDD projects that need an HPA. In addition to minimizing adverse channel hydraulic impacts, the recommended measures also address substrate modifications and water quality impacts associated with HPDD operations. Compliance with such measures should ensure that incidental take due to frac-outs has been minimized to the greatest practicable extent, thereby meeting the ESA criterion.

Risk of damage from "catastrophic" events such as debris flows, dam-break floods, and rare conventional floods can be minimized by increasing the design standard (e.g., to 500-year flood capacity), using fords rather than culverts at sites where fish passage is not an issue, or siting piers/abutments so as to span the channel migration zone (see Bolton and Shelberg 2001 for discussion of channel migration zones).

11.2.7.1.2 Littoral Drift

Impacts to littoral drift can be avoided or minimized by avoiding or reducing those features that interfere with littoral transport processes through the following measures:

- Bury conduits so that they do not extend above the sediment surface (MOEE 1995) (currently required under WAC 220-110-100(2)).

- Design pile-supported structures with sufficient open space between pilings to allow waves, currents, and sediment to pass beneath (MOEE 1995).

- Avoid certain impacts from floating water crossings placed perpendicular to shorelines, which dampen wave action and prohibit natural shoreline erosional processes, by minimizing the dimensions of these types of structures.

The effects of these measures are site-specific, and thorough study of the littoral drift cell and potential habitat affected should be conducted on projects that could affect the system's littoral currents and wave action.

11.2.7.1.3 Substrate Modifications

The identified impacts of marine substrate modification (as distinct from substrate changes that occur in response to channel or shoreline hydraulic changes) are generally beneficial. The reviewed studies do not recommend specific habitat protection, conservation, mitigation, and management strategies. However, if the federal agencies express concern about the possible cumulative effects of marine substrate modifications on potentially covered species, it would be appropriate to track such effects in the course of overall HPA program monitoring. In this way, new data could be accumulated to help guide adaptive management of the program.

Substrate modification in freshwater environments generally consists of placing fill or culverts into aquatic habitat or adjacent riparian/floodplain habitat. Means of reducing the impact of such actions include:

- Minimizing fill placement by siting bridge abutments far enough apart to span the channel or using bottomless culverts that span the channel.
- Minimizing use of approach fills or including flood relief culverts in approach fills.
- Siting bridges or culverts, where possible, at locations where the channel is naturally confined.
- Oversizing culverts to ensure that they will pass LWD and large bedload particles.

11.2.7.2 Water Quality

No specific measures for water quality impacts from water crossing structures were identified.

11.2.7.3 Aquatic Vegetation

11.2.7.3.1 Eelgrass and Macroalgae

Where conditions are suitable for eelgrass growth, impacts of water crossing structures should be avoided or minimized by use of the following measures:

- Avoid impacts by locating structures away from eelgrass beds whenever possible.
- Minimize the area of impact by using the best available installation methods.
- Minimize shading of bridges over eelgrass and macroalgae by using the lowest possible number of pilings.

- Space pilings to minimize shade to areas suitable for eelgrass.
- Minimize dimensions of bridges to reduce shade.
- Incorporate design elements into bridges to reduce shade where feasible.
- Whenever possible, orient bridges to reduce the shade in habitat that is otherwise appropriate for eelgrass growth (e.g., structures oriented east-west cast a shadow on a single area for a longer period of the day than do structures oriented north-south).
- Locate the bridge deck as high above the water as practical to reduce shade.
- Avoid vessel impacts to eelgrass during water crossing construction by maximizing the vertical and horizontal distance between vessel propellers and eelgrass to the extent practicable.

Adopting these measures would likely result in avoidance and minimization of eelgrass and macroalgae impacts to the greatest extent practicable. However, it is likely that some projects would still require compensatory mitigation to completely offset temporal loss of eelgrass function and site-specific and cumulative impacts on eelgrass.

The reviewed literature did not identify minimization or mitigation techniques to address impacts to macroalgae.

11.2.7.4 Freshwater Aquatic Vegetation

No specific measures for freshwater aquatic vegetation impacts from water crossing structures were identified.

11.2.7.5 Riparian and Shoreline Vegetation

No specific measures for riparian vegetation impacts from water crossing structures were identified.

11.2.7.6 Artificial Lighting

For bridges, artificial lighting may not be avoidable; therefore, compensatory mitigation may be required to fully account for potential adverse impacts associated with artificial lighting.

11.2.7.7 Shading

Nightingale and Simenstad (2001b) and Carrasquero (2001) provide impact minimization measures for the design, construction, and revetment of a variety of overwater structures. Many of these measures appear to be applicable to water crossings, especially bridges. The guidance provided by these authors includes:

- Increasing the height of overwater structures (in this case, bridges) to allow light transmission under the structures.

- Decreasing structure width to decrease the shade footprint.
- Using the smallest number of pilings possible, allowing more light to penetrate beneath the structure.

It may also be helpful to construct bridges with a grated deck that allows some light transmission.

11.2.8 Fish Passage Structures

Best professional judgment of the design standards that are commonly used in the Pacific Northwest and elsewhere include the Draft Fish Passage Standards developed by NOAA Fisheries (NMFS 2001), draft revisions to these standards currently in development, WDFW culvert design guidelines (Bates et al. 2003), and WDFW fishway design guidelines (Bates 1997).

11.2.8.1 Design Criteria

Regardless of the structure type, it is apparent from available research that “one-size-fits-all” guidance for the design of fish passage structures will not yield adequate results where the passage of multiple HCP species at multiple life-history stages is a concern. Structure design and specific structural parameters should take into account these biological requirements to ensure long-term success. In this context, fish passage structures that attempt to mimic natural hydraulic and geomorphic complexity are likely to provide the most effective results. Current WDFW guidance emphasizes this approach.

Specific circumstances, such as the retrofitting of existing culverts or the development of fishways, may require engineered solutions based on the swimming abilities of target fish species. Where passage requirements for species of interest are uncertain, factors of safety should be incorporated to the extent practicable. Structure design must also accommodate the hydraulic and geomorphic context of the system in which it is being installed. This will increase the likelihood of successful operation over time, and ideally decrease the need for maintenance.

Consider the following parameters when developing design criteria for retrofitted culverts and fishways for juvenile salmonid passage:

- Design for the smallest size of fish anticipated to migrate through the structure.
- Create complex, interconnected low-flow velocity zones within the structure. Incorporation of roughness features (e.g., corrugation, gravel and cobble embedded within concrete, baffles) appears to aid in this objective by creating turbulence that induces low-velocity conditions in the boundary layer.

11.2.8.2 Culverts

In circumstances where culverts are required, structures that are designed appropriately for the hydraulic and geomorphic context of the project site can provide a high degree of fish passage and habitat protection. Accordingly, current design guidance directs project proponents in identifying the most appropriate type of structure for their specific circumstances.

Culvert design guidance has continually evolved in recent years as the result of ongoing research on fish passage requirements, as well as a growing understanding of the broader effects of culverts on the aquatic environment. WDFW guidance to date has emphasized the use of three design methods:

- the no-slope and stream-simulation options, which emphasize the placement and/or natural accumulation of bed material within the culvert to promote a hydraulically complex environment; and
- the hydraulic design option, which emphasizes the use of hydraulic calculations to design a structure based on the swimming performance of target species.

The stream-simulation option is currently the recommended approach to culvert design. The no-slope option is similar in concept, except that this method is limited to lower gradient environments with shorter culvert requirements. These geomorphically oriented designs attempt to accommodate natural fluvial processes to the greatest extent possible, thereby providing passage for a full range of aquatic species.

When properly designed for the hydraulic and geomorphic conditions present in the watershed, these geomorphic designs can provide a high degree of fish passage function with limited effects on ecosystem connectivity. However, any design that fails to incorporate the full range of current and future geomorphic conditions in the watershed may cause unintended effects on habitat conditions, or may ultimately fail to provide fish passage if channel conditions change. For example, a culvert design that fails to recognize the likelihood of migrating headcuts either reaching or being liberated by the structure may not allow the channel to adjust as required. Conversely, a culvert may be designed appropriately for current conditions, but the design may fail to recognize development trends in the watershed that could change local hydrologic and geomorphic conditions.

This speaks to the need for guidance for a predesign hydraulic and geomorphic assessment of current and likely future watershed conditions. This guidance should emphasize assessment of current conditions in the watershed (specifically with regards to channel evolution), and the hydraulic and geomorphic trajectory of the system. The latter should consider likely future land use patterns and their likely effect on hydrologic and geomorphic conditions. This guidance should also cover methods for addressing existing headcut conditions and channel incision, using grade control measures or other forms of habitat and/or channel modification as needed.

Culvert design guidance has evolved in recent years given acknowledgement of the complexities and uncertainties inherent when using the hydraulic design method to provide passage for a broad range of species. This method has become less favored over time because of its demonstrated failure to adequately provide juvenile fish passage, as well as other concerns. This weakness is due in part to limitations and uncertainties in the calculations used, failure to consider the design life of the project in the context of natural variability in channel conditions, and inappropriate criteria used to direct design guidance in the Hydraulic Code. With regard to the former, the hydraulic calculations employed in this method are limited from the standpoint that they may not fully capture the complexity of turbulence and boundary layer velocities within culverts that can aid or hinder fish passage.

Despite these limitations, the hydraulic design option is still employed in specific circumstances where retrofitting of a barrier culvert is required (e.g., when removal or replacement is impractical in the immediate future). In such cases, the use of rigorous hydraulic engineering methods is a desirable approach where fish passage must be considered. However, it must be stressed that the design approach be informed by the best available science on the swimming performance, behavior, and migratory requirements of all species and all life-history stages likely to be affected by the structure in question. As this information is developed, culvert design guidance should be updated accordingly. It is recommended that biological criteria not be included in the Hydraulic Code, however, because the code is updated too infrequently to reflect the most recent science.

Two examples illustrate the weaknesses inherent in the hydraulic design method. First, available data described throughout this white paper indicate that culverts and other fish passage structures need to accommodate the passage of fish species and life-history stages with a broad range of swimming abilities and behavioral requirements. Most research applicable to the retrofitting of culverts has focused on salmonids. However, protection of salmonids may not adequately protect the full range of HCP species. For many other HCPS species, data on swimming performance are too limited to be useful in guiding design, or do not exist at all.

Second, WAC 220-110-070 sets the design discharge criterion as the flow rate that is exceeded no more than 10 percent of the time during the months of active adult and juvenile migration (Bates et al. 2003; Powers and Saunders 2002). If the culvert velocities are less than or equal to the allowable velocity at the high passage design discharge, the WAC criterion is met. If not, the culvert is considered a barrier. However, barrier determinations made by the physical and hydraulic measurements described in the WAC, may not accurately represent the influence a culvert has on the movement of HCP species that are less well understood. Consequently, it is recommended that information be collected on the behavior of nonsalmonid fish species to document the actual effect of culverts on fish movement.

The Washington State Department of Transportation leads a cooperative program to study juvenile salmonid passage through culverts by systematically conducting statistically designed experiments in a full-scale culvert system at the Culvert Test Bed (CTB) at the WDFW Skookumchuck Hatchery near Tenino, Washington (Pearson et al. 2005, 2006). The CTB program is a unique opportunity to provide scientifically sound information that can be used to develop better designs for retrofitted structures (Pearson et al. 2005, 2006). However, WDFW staff have questioned the effectiveness of this program. If this program continues, research should focus on providing relevant understanding of the relationship between hydraulics and behavioral and physiological limitations necessary to develop sound design criteria. In this context, expansion of the program to evaluate the passage requirements of other HCP species may be valuable. As the need for retrofitted culverts declines over time (i.e., barrier culverts are removed or replaced, rather than retrofitted), the program can be retired.

Although not supported by direct citation from scientific literature, general recommendations regarding trash racks and livestock fences associated with culverts are provided in NMFS (2001). According to NMFS (2001), trash racks and livestock fences should not be used near the culvert inlets as accumulated debris may severely restrict fish passage and cause potential injuries to fish. Where fencing cannot be avoided, it should be removed during adult salmon upstream migration periods. Timely clearing of debris is also important, even if flow is getting around the fencing. Cattle fences that rise with increasing flow are highly recommended.

11.2.8.3 Fish Ladders/Fishways

Fishways are generally not recommended but may be useful in some applications, such as where excessive drops occur at a culvert outlet (NMFS 2001).

In general, given that fishways are commonly associated with dams, to the extent possible, owners should pursue notching or complete dam removal. The most biologically sound solution to fish passage related impacts from dams is to allow for free and unimpeded upstream and downstream migration at all times of the year.

Based on data and findings from the ongoing monitoring of constructed projects, FishXing (2007) offers the following recommendations with respect to fishway construction:

- Where applicable, design internal weirs with gradual side-slopes. Weirs with gradual side-slopes create a thin sheet of plunging water along the edges. The hydraulics of this thin sheet of water in the receiving pool creates good leaping conditions for smaller fish. Also, place a bevel on the downstream edge of V-notch weirs to create the best conditions for leaping by smaller fish. Placing the bevel on the upstream side may also improve debris passage.

- In a fishway, if the volume of each step-pool is relatively small, it may create excessive turbulence at relatively low flows. Assessing turbulence during the design process involves identifying the highest flow for passage through the step-pools and then sizing the pools to dissipate the energy associated with that flow. Turbulence in step-pools is assessed using the Energy Dissipation Factor (EDF). If the EDF is excessive at the high-passage design flow, either the pool volume should be increased, the drop height reduced, or the proportion of streamflow that bypasses the pools should be increased.

11.2.8.4 Roughened Channels

The effects of roughened channel construction are similar to those imposed by channel creation and realignment. Therefore, the habitat protection, conservation, mitigation, and management strategies discussed for channel creation apply also to roughened channel construction.

Based on constructed project monitoring data, FishXing (2007) provides the following recommendations for roughened channels associated with culverts:

- Construction of a roughened channel requires skilled equipment operators and on-site construction guidance from persons familiar with this type of design. Expert construction oversight is needed to avoid the construction of wider and shallower-than-designed roughened channels. These deviations from the design have the potential to create insufficient depth at lower fish migration time flows, possibly hindering fish passage.

- When rock must be used, the use of larger-than-specified rock to construct the bank of a roughened channel results in large voids within the bank rock. This will allow water flow behind the rocks, thus scouring the native bank material. The potential for this issue is greater when donated or “recycled” rock is used to construct the bank of a roughened channel, as it may not meet design specifications. If the problem occurs, it can be addressed with the use of smaller material added in the void areas to prevent water from flowing behind the rocks and scouring the native material.

- In roughened channel projects that extend through/past a culvert, using a continuous slope through the culvert rather than a short, oversteepened section would improve fish passage conditions.

- When designing roughened channel, consider the geomorphic and hydraulic impacts beyond the project area to avoid or minimize the potential for unintended impacts.

- Use an interdisciplinary team of engineers, hydrologists, fisheries biologists, and geomorphologists to identify and address potential problems beyond the project area during the preliminary design phase.

- Poor culvert alignment can increase the risk of debris plugging, scour adjacent banks, and reduce capacity. When extending or installing a culvert, consider the impacts on alignment between the culvert inlet and approaching channel.

- The natural streambed below a lined or hardened channel is typically susceptible to scour and downcutting. Therefore, it is advisable to include a transition area that dissipates energy and reduces velocity before flow enters the natural channel. Addressing this in the initial design phase may avoid the need for subsequent replacement or retrofits.

- Limiting the project length to the right-of-way can make it extremely difficult to satisfy fish passage objectives while maintaining a stable channel. To achieve the project's objectives, consider extending the project reach beyond the right-of-way. This will require coordination with adjacent property owners as well as stakeholders early in the project design.

11.2.8.5 Weirs

Using weirs to provide hydraulic controls in the channel upstream and/or downstream of a culvert can create a continuous low flow path through the culvert and stream reach intended to facilitate fish passage (NMFS 2001). These weirs should be designed to provide instream habitat complexity. To achieve this secondary objective, as well as to greatly improve their hydraulic performance, grade control weirs should be designed as complex structures, rather than simple or single-log structures. Simple or single-log structures are easily undermined and have often been observed in the field posing a barrier to fish after a few months of operation.

Where permanent weirs are desired to manage fish passage, these structures should be designed to limit hydraulic and geomorphic modifications to the greatest extent possible. Specifically, permeability to the downstream transport of water, LWD, sediment, organic material, and fish movement is desirable to limit broader ecological effects.

11.2.8.6 Trap and Haul

The principal biological benefits for a trap-and-haul system typically include connecting populations, increasing genetic exchange, and increasing access to habitat for multiple lifestages and species. These benefits can also be achieved with fishways. Given that trap-and-haul systems are typically less expensive than fishways, the former may be more appealing for some applications. However, although initially less expensive than a fishway, a trap-and-haul system has the disadvantage of higher annual maintenance to ensure that the mechanical equipment and systems work properly during the entire fish passage season (Ferguson et al. 2002). Due to these higher maintenance requirements, trap-and-haul systems are likely to cause more environmental disturbance than fishways, thus increasing their chance to affect HCP species (e.g., through water quality impacts).

Trap-and-haul programs present additional disadvantages that render them less desirable than volitional passage. Fish capture and handling are sources of potential injury and stress that can lead to immediate, delayed, or indirect mortality. In some cases logistical considerations may require release of transported fish at locations that significantly alter their migratory corridor. This in turn may lead to undesirable effects on survival, fitness, and/or spawning productivity. When imposed over several generations, these combined stressors have the potential to impose selection pressures that may result in undesirable evolutionary consequences.

Trap-and-haul programs are labor intensive, which translates to ongoing management costs. While failure to regularly maintain fish passage structures is likely to lead to a gradual degradation in function, trap-and-haul programs are entirely dependent on annual funding to function. In this light, structures that provide volitional passage are clearly preferable.

Given these inherent limitations, consideration should be given to the preferential construction of fishways over trap-and-haul systems where practicable to reduce the potential for undesirable effects on HCP species and their habitats.

11.2.9 Fish Screens

Several strategies exist for improving how fish screens are used in Washington State. These strategies fall into the following categories:

- Management strategies
- Strategies for improving fish screen design and structure

11.2.9.1 Management Strategies

11.2.9.1.1 Improved Training and Research

Designing an effective fish screen requires an integrated understanding of the engineering demands of the structure, site-specific performance requirements, and understanding of the biological needs of the species the screen is intended to protect. This combined knowledge is necessary to develop both an effective screen design, and to provide operational parameters for the water withdrawal or diversion when sensitive species are present that cannot be effectively protected.

WDFW currently provides training, design, and installation assistance for screening projects. WDFW-sponsored research conducted at the Yakima Screen Shop facility has produced many of the screen concepts and design criteria in current use in the region. There is some level of ongoing coordination among state and federal agencies in the Pacific Northwest on research and practical application of screening technologies. However, funding cuts in recent years have limited research and collaboration, leading to the abandonment of efforts targeted at developing and building effective screening technologies. The screen assistance and the screen research programs should be strengthened and coordinated with efforts at the federal level and in other states in the region.

Web-based case studies that evaluate the effectiveness of integrated design and operational parameters would be particularly useful.

11.2.9.1.2 Improved Rules and Guidance

The most current WDFW guidance on fish screen design is in incomplete draft form and has not been revised since 2001 (WDFW 2001a). This guidance document should be updated and improved based on the latest technical information and made available to managers and the public. A notable weakness in this and other fish screen guidance documents is the widespread use of inconsistent terminology, resulting in standards that are confusing and at times contradictory. The revised guidance document should be coordinated for consistency with NOAA guidelines, using a parallel format and consistent terminology to allow for easy cross-referencing among documents. Where state standards necessarily depart from federal guidance, the differences should be clearly highlighted and the rationale for the departure explained. The design guidance should also incorporate a set of typical design drawings for common screen designs and a range of flows, as well as provide contact information for manufacturers and vendors. The guidance should be supported by up-to-date web-based technical assistance, including current case studies that are regularly updated.

Currently, fish screens are typically designed conservatively around scenarios to provide protection of the smallest and weakest swimming salmonid life stages, the most extreme temperature conditions (which affect swimming performance), and the highest flow rates, conditions that are rarely observed in practice. Using the swimming performance and requirements of the smallest and weakest-swimming species and/or life-history stages likely to be exposed to the screen is presumed to provide broad protection for the full range of species and life-history stages likely to be exposed to the screen. This is a useful uniform recommendation that should be employed in all screen designs. However, screen facilities designed to such standards might impose a greater burden on the operator due to their operational limits and maintenance requirements, or engineering demands that are infeasible in certain cases. In such cases, operators have an incentive to contribute to research.

Even when properly engineered for site conditions, a fish screen may not be able to protect all HCP species/life stages. For example, planktonic larvae may be unavoidably entrained through even the most protective screen system. To provide additional protection where performance limitations cannot be overcome through design, WDFW may want to investigate expansion of authority under the Hydraulic Code to impose operational limits on water withdrawals, allowing water withdrawal restrictions to be included as part of the approval process under the HPA program for fish screens.

These restrictions would be enforced in circumstances where screens cannot provide adequate protection when sensitive life-history stages of various species are present. Water users in high-priority habitats (i.e., habitats where HCP species may be acutely vulnerable) should be required to develop an operational plan that is certified by state and federal agencies. Moreover, research should be dedicated to developing effective screen technologies for settings where flow restrictions are not practicable.

11.2.9.1.3 Improved Performance and Compliance Monitoring

More consistent monitoring and enforcement will greatly benefit the advancement of fish screen science, and help to ensure that existing screens are as protective of HCP species as possible.

Performance monitoring is a necessary tool to determine whether existing screens are functioning as intended and how effective they are at avoiding or limiting entrainment and impingement of sensitive species and/or life stages. For nearly two decades, the Bonneville Power Administration (BPA) has funded ongoing monitoring of fish screen systems on several of the larger irrigation diversions on Columbia River tributaries, including the Yakima, Walla Walla, Umatilla, and other river systems (Carter et al. 2003; Knapp 1992; McMichael and Chamness 2001; Vucelick and McMichael 2003; Vucelick et al. 2004). WDFW has received funding from BPA and NMFS through intergovernmental memoranda of agreement to conduct screen inspection and maintenance on screen systems throughout the Columbia River basin, and to a lesser extent in western Washington. WDFW currently operates a statewide screen maintenance and inspection program, partially subsidized by federal funds, that provides maintenance guidance and monitors maintenance compliance and screen performance.

Improved compliance monitoring is a necessary strategy to enhance protection of HCP species. Even when the best possible screen design and operational criteria are developed, some fish screens will not be operated or maintained as necessary to provide the level of protection desired. Noncompliance with permitting requirements is certain to be an issue of concern regardless of any advances in screen design and operational implementation. Full funding and expansion of the WDFW program would provide a useful and necessary means for training fish screen specialists, and provide case studies for demonstrating successful design and operational procedures. This type of program should consider the following objectives:

- Pre- and postconstruction review of fish screen designs and as-builts for all high-priority screen projects to confirm that the structure was built as intended.
- Incorporation of operational certification into the approval process under the HPA program, with a set recertification schedule based on inspection performance.
- Routine monitoring of fish screens (e.g., every other year, every 3 years) to evaluate compliance with maintenance and operational requirements for recertification purposes.
- Coordination with performance monitoring to provide a mechanism for addressing underperforming structures.

A comprehensive compliance program should include a mandatory but practical pathway for owners of noncompliant screens to address structural and operational issues as quickly as possible. Compliance incentives should first provide funding and technical assistance (building on existing state-level programs) to help owners meet recertification requirements, followed by enforcement and legal action as necessary.

11.2.9.1.4 Fish Screen Operations

In cases where the protection provided by fish screens is fundamentally limited, knowledge of when sensitive species and/or life-history stages are present can be used to manage the timing of water withdrawals. For example, intake systems that will unavoidably entrain fish larvae at high mortality rates could be shut down when larvae are most likely to be present. These management practices would require an expansion of WDFW's authority to regulate water withdrawals, which is currently limited.

Noise, Visual, and Physical Disturbance

Underwater noise, visual, and physical disturbance are, to a certain extent, unavoidable with screen systems that employ mechanical debris-clearing systems. Mechanical systems should be sound insulated and located above water to the extent practicable to limit continuous underwater noise that could contribute to auditory masking effects or avoidance behavior (except in circumstances where noise is being used as a behavioral deterrent). Air jet or hydraulic debris-clearing systems for in-channel screens should be calibrated to limit impulsive sound below established disturbance thresholds where practicable (e.g., 150 dB_{RMS} for salmonids). Proper siting of in-channel screens should limit behavioral avoidance of suitable habitats or other undesirable effects.

Entrainment and Impingement

Require design criteria to consider the full range of HCP species that are likely to encounter the fish screen. Screen mesh size, mesh material, and approach velocity are critical factors in determining entrainment and impingement risk. Current scientific understanding of the swimming performance and risk of entrainment or impingement-related effects is less than uniform across the range of HCP species. Current design criteria may not consider the full range of HCP species likely to occur and therefore may not be as protective as possible.

Introduction of Toxic Substances

Fish screens have the potential to introduce toxic substances to the aquatic environment through two primary pathways:

> (1) accidental spills of fuel, oil, lubricants, or other pollutants during construction and maintenance; and
>
> (2) screen equipment failure resulting in the release of toxic lubricants.

Construction, maintenance, and operational-related impacts can be avoided by requiring the project proponent or contractor to have an established spill prevention and spill containment plan in place, through proper equipment maintenance, and through the use of nontoxic, food grade hydraulic fluids and lubricants. Although these actions are commonly taken, making such actions mandatory could further reduce impacts.

11.2.9.2 Strategies for Improving Fish Screens

11.2.9.2.1 In-Channel Screens

In-channel screens vary widely in design configuration, ranging from simple screens on small, private water supply systems to large and complex structures on industrial water intake systems. Given this variety, the strategies identified lean toward recommendations specific to designs for certain applications or of a certain scale. Strategies identified include the following:

- **Infiltration galleries:** Develop guidance criteria for the siting, design, and operation of infiltration gallery screens. Alternatively, adopt NMFS criteria found in NMFS Anadromous Salmonid Passage Facility Design, Section 13 Infiltration Galleries (Experimental Technology (February 2008).

- **Fishway screening requirements:** Screen auxiliary intake systems for fishways and fish ladders, to prevent exposure of smaller sensitive fish to entrainment-related injury.

- **Siting of large intake systems:** Site intake systems at locations and depths where planktonic life-history stages of HCP species are less likely to occur. This highlights the value of incorporating biological expertise into the design of fish screens and related flow control structures.

11.2.9.2.2 Off-Channel Screens

Off-channels screens also encompass a range of designs appropriate for different conditions; therefore, design strategies are relatively specific to given design types.

- **General flow control for off-channel screens:** Screens should be designed to accommodate the hydrologic context of the system in question. Use automated headgate systems programmed to respond to changes in flow conditions. Overtopping by high flows or due to debris accumulation is the most common cause of screen failure and elevated entrainment risk. Other changes in flow conditions can change diverted flow rates, screen submergence, bypass flows, and other parameters in ways that adversely affect screen performance.

- **Flow control for inclined plate screens:** The screen design should provide for a minimum depth of water over the entire screen face. This depth should be based on expectations of the size and type of debris, size, and condition of fish (or other HCP species) requiring passage, and the potential variation in flow that could reduce the depth to below the desired minimum. To achieve these conditions, a substantial amount of bypass flow is typically required and flow conditions must be carefully monitored. Downward sloping screens require at least several feet of head loss to operate effectively. These constraints typically limit this type of screen to riverine applications. Because of the restrictive control of flow necessary for downward-sloping fixed plate screens to provide fish and debris clearance, this design is not recommended except where constant and precise flow control can be provided.

- **Hydraulic and geomorphic considerations:** Off-channel screen designs that incorporate bypass channels impose distinct hydraulic and geomorphic effects, because the bypass channel flow is removed from the main channel and is unavailable until it is discharged at some point downstream. These effects can be minimized by limiting the length of the bypass, discharging the return flow as short a distance downstream as practicable. This design criterion must be balanced against the need to provide sufficient head loss to maintain bypass flow velocities necessary to clear debris, and to discharge entrained fish at a safe location (e.g., areas unsuitable for loitering by lie-in-wait predators).

  These competing design requirements may lead to relatively long bypass channels. If the length of the affected reach is significant (e.g., greater than five times the average reach width) and the flow required to operate the bypass channel is a significant portion of the streamflow in the channel downstream of the diversion, then undesirable changes in channel morphology may occur due to factors such as vegetation encroachment.

- **Avoiding ecosystem fragmentation effects:**

  - Screens employing bypass channels must provide sufficient sweeping velocities to draw downstream migrant and dispersing fish into the bypass system, avoiding delay.

  - Site bypass systems to minimize the length of the bypassed reach. Site outlets to minimize predation on organisms exiting the system.

  - Do not locate bypass outlets in side channels or other channel features where the discharge could create attraction flows that delay upstream movement of migratory species.

  - Consider the potential cumulative effects of migration delays imposed by multiple screen systems when permitting the screen as well as the related flow control structure or channel modification.

11.2.10 Flow Control Structures

11.2.10.1 Construction and Maintenance Activities

In a recent document on procedures to minimize nonpoint source pollution from hydromodification projects, the USEPA (2007) proposed measures to minimize construction problems from sediment increases and chemical pollution. The management practices are specific to the location of the project, the local climate, and source of potential pollution.

Erosion and sediment control procedures are used to prevent sediment from entering surface waters during the construction or maintenance of flow control structures. Proper erosion and sediment control practices should be used to protect surface water quality because of the high potential for the loss of sediment directly to surface waters during these types of projects. Erosion control can be maximized by minimizing the area and time of land disturbance and by stabilizing disturbed soils to prevent erosion in a timely matter. USEPA (2007) has suggested using sediment and erosion control practices borrowed from other applications, such as urban development and construction activities. Potential erosion control activities include application of the following methods and practices:

- Bank shaping and planting
- Bulkheads and seawalls
- Check dams
- Coconut fiber roll
- Erosion control blankets
- Locate potential land disturbing activities away from critical areas
- Mulching
- Preserve on-site vegetation
- Retaining walls
- Revegetation
- Riparian improvements
- Sediment fences
- Sodding
- Vegetated filter strips
- Wind erosions controls.

Minimization of runoff will reduce potential impacts on water quality during construction activities. Practices for controlling chemicals and pollutants include the following (USEPA 2007):

- Check dams
- Constructing runoff intercepts
- Equipment runoff control
- Fuel and maintenance staging areas
- Locate potential land-disturbing activities away from critical areas
- Pesticide and fertilizer management
- Pollutant runoff control
- Preserve on-site vegetation
- Sediment traps
- Vegetated filter strips.

In the construction of new flow control structures, avoidance or minimization of impacts can be accomplished through site selection and facility design. For construction and maintenance activities, management strategies can be implemented to minimize underwater noise, dewatering and fish handling, and construction/maintenance dredging impacts.

11.2.10.2 Dams

Dams severely alter natural rivers systems in many ways including physically blocking the movement of migrating species, altering the natural flow regime, and reducing suitable habitats. Mitigation of these impacts can be divided into three general groups:

(1) actions to improve fish passage,

(2) actions to restore natural flow regime, and

(3) actions to reduce water quality impacts.

In addition, certain actions can be taken during the construction phase of dam projects. The special case of dam removal will often serve to reverse or greatly minimize impacts from dam projects in the long term.

11.2.10.2.1 Fish Passage

To minimize migratory impacts from dams, adequate fish passage structures are required that allow a majority of fish to reach upstream and downstream habitats. For example, Webber et al. (2007) concluded that the design of dams and fish barriers should have fast and slow portions to increase migration over these structures. In laboratory studies, the authors demonstrated that white sturgeon attempt to pass barriers with short bursts, followed by a resting period. Therefore, design of fish barriers (e.g., weirs, dams, step-pools) should have fast sections 2.76–8.27 ft/sec (0.84–2.52 m/sec), followed by slower sections 1.64–2.23 ft/sec (0.5–0.68 m/sec) for recovery (Webber et al. 2007). Information on optimal swimming velocities, height restrictions, diurnal migration patterns, and behavior at passage facilities for HCP species are necessary to optimize fish passage in the presence of dams.

11.2.10.2.2 Flow Regime

Numerous studies have concluded that in order to maintain the ecological integrity of riverine environments in the presence of dams, some return to a natural flow regime is needed (Bednarek 2001). A return to the natural flow regime maintains habitat complexity and connectivity, limits impacts from altered sediment transport and substrate composition, and improves species diversity. These are sometimes referred to as environmental flows (Chester and Norris 2006). In the Grand Canyon, attempts to remediate sediment movement by prescribed flooding or higher (elevation) releases of water through dams have taken place. Collier et al. (1997) documented that incised beaches and sand bars downstream of Glen Canyon dam were somewhat restored during these “flood” events. However, beaches and sandbars still suffered from a reduction in sediment supply.

Biodiversity is best protected where dam operation emulates a natural system. Food webs require variable flow regime and floodplain inundation (Power et al. 1996). Environmental flows used in Australia showed that macroinvertebrate communities were similar to those of unregulated flows in the region (Chester and Norris 2006). In addition, flow releases that simulate variable flows have been observed to improve the diversity of warmwater fish assemblages (Travnichek et al. 1995). On the Tallapoosa River (Alabama), the relative abundance of species classified as fluvial specialists increased from below 40 to more than 80 percent after initiating a more variable flow regime.

11.2.10.2.3 Water Quality

The primary impacts from dams on water quality include altered temperatures and altered dissolved oxygen concentrations. These modifications can be minimized if water releases from the reservoir can occur at multiple depths (Bednarek 2001). This mitigation practice will vary depending on the local conditions, as well as on what species are present; therefore, this practice should be analyzed on a case-by-case basis. In some cases, multiple-depth flow releases will solve these water quality problems; in other cases, they will not (Bednarek 2001).

11.2.10.2.4 Dam Removal

Dam removal is the best way to reestablish thermal regimes and natural sediment transport, restore habitat complexity, and minimize water quality changes. Dam removals are becoming more common as facilities are applying to renew licenses because, in some cases, dam removal is a more economical or safer option (Bednarek 2001). Dam removal, in general, restores natural sediment transport in the system by increasing habitat diversity in the former impoundments (Bednarek 2001) and replenishing coastal systems where beach erosion has proliferated (DOI 1995). Recently, eulachon have been observed in the Elwha River (Washington), and dam removal will likely increase the availability of sand and gravel sizes required for these fish to spawn (Shaffer et al. 2007). Dam removal allows organisms to migrate freely, reduces delays in migration, and reduces mortality caused by fish passage structures (Travnichek et al. 1993).

One significant environmental concern from dam removal projects is the release of stored sediment from the former impoundment. Stored sediments may cause increases in downstream sediment transport and turbidity; however, these increases will be a short-term impact while the river transitions back to a free-flowing system. Factors influencing the duration of impact from sediment releases from a dam removal include: (1) the length of time dam was present, (2) velocity and gradient of river, and (3) removal techniques (Bednarek 2001). The frequency of storms after removal is also important. The downstream effects from sediment releases can be on the order of days (Winter 1990) to many years. In some cases, sediment release will be equivalent to a periodic storm event (Winter 1990). Along with increases in turbidity, there is the potential for contamination arising from pollutants that are adsorbed onto sediment particulates. Pollutant contamination can be reduced by conducting a preremoval evaluation of sediments or dredging, and by conducting a slow drawdown of the reservoir prior to dam removal (Bednarek 2001).

11.2.10.3 Weirs

Weirs are similar to dams but are generally smaller in scale. As a result, mitigation activities associated with weirs are identical to those described for dams.

11.2.10.4 Dikes and Levees

Breaching of dikes and levees has been used to reconnect channel and floodplain habitats, with several documented benefits. After breaching levees on the Consumes River (California), floodplain geomorphology became more complex, with changes in topography, woody debris recruitment, and vegetation (Florsheim and Mount 2002). In addition, restored connectivity has been shown to enhance nutrient cycling by reducing nitrate loading downstream (Sheibley et al. 2006). Finally, levee breaches can influence algal dynamics and overall water quality of the restored floodplain (Ahearn et al. 2006).

Erosion and failure of levees may be reduced through planting vegetation. Conversely, vegetation removal is often encouraged on levees to provide access for inspection, fight flooding, reduce rodent burrowing, and to prevent root-induced water removal (Bolton and Shellberg 2001). However, this study also noted that grass and vegetation actually stabilize these structures, similar to vegetated stream banks. In addition, grass coverage on levees will cause a more even wetting and drying of the structures through transpiration, which will lessen cracking and failure from uneven drying after flood events. Taller vegetation may shade levees and reduce the cracking of earthen levees from extreme heat.

Where possible, dike and levee projects should be designed to retain as many natural hydraulic and geomorphic features as possible. This can be achieved by increasing the distance between the levees to allow channels to naturally meander, incorporating meanders into the channelization project, minimizing the reach length where levees are constructed, or creating artificial side channels (Bolton and Shellberg 2001). The creation of artificial side channels simulates a low-flow channel; when flooding occurs, water spills out into the "floodplain" and creates side channel and side pool habitats. Levee projects can be conducted so that in-channel (e.g., pools, riffles) features are preserved (Bolton and Shellberg 2001). This can easily be achieved by not dredging the channel after the levee or dike is constructed. As with all of these mitigation strategies, their feasibility depends on several site-specific factors, including the purpose of the project, the size of the project area, cost, and safety.

11.2.10.5 Outfalls

Hydraulic and geomorphic modifications associated with outfalls can be eliminated with a design that minimizes alterations to the physical environment surrounding the outlet. A few recommendations are:

- Locate all outfall infrastructure below-grade in areas where sediment transport is significant.
- Place submerged outfall outlets below the closure depth or light penetration depth, whichever is greater.
- Where possible, avoid discharges that are significantly different in density, temperature, salinity, and turbidity from the receiving water.
- Minimize the flow velocities of the discharged fluid. If the flow rates are expected to significantly alter the circulation or geomorphology in the vicinity of the outlet, perform hydrodynamic modeling to assess and limit the area of impact.
- To avoid scour associated with large discharge velocities, site the outfall outlet in an area of pre-existing immobile substrate, where possible.
- Screen the outlet to prevent fish entrainment into the outfall piping.
- Site exposed outfalls so they do not protrude or disrupt sediment transport. Where possible, placement of the outlet should be approved by a licensed geologist.

Where hydraulic and geomorphic modifications are unavoidable, mitigation of such effects is necessary. This could include routing the sediment around the geomorphic disruption.

Monitoring plans associated with submerged outfalls should include ongoing inspections of the outlet infrastructure for the presence of invasive species.

One of the most significant impacts from outfall projects is the alteration of water quality in receiving waters. These impacts can be minimized by:

- Ensuring that the contaminant load in the effluent has been reduced to the greatest extent possible (Williams and Thom 2001).
- Locating outfalls in marine areas where dilution and flushing are maximized (Williams and Thom 2001).
- In riverine environments, establishing a mixing zone will lower the effects downstream.
- Because sediments are associated with many types of pollutants (Murakami and Takeishi 1977), reducing the amount of sediment in the outfall discharge is desirable.

11.2.10.6 Intakes and Diversions

The primary hydraulic and geomorphic alterations associated with intakes and diversions are related to the piping infrastructure for these systems. The most common issue related to intakes and diversions is the entrainment of fish and invertebrates. This impact is mitigated by using fish screens.

Alteration of the amount of water removed and the timing of water removals can minimize impacts related to these structures. For example, a study of downstream drifting shrimp larvae showed that a large percentage of the larvae can be entrained in water intakes, with a mortality of 42 percent and almost 100 percent removed from water column during low flows (Benstead et al. 1999). However, the authors showed that most drift took place at night. When the intake was turned off for 5 hours at night, mortality was reduced to 11–20 percent (Benstead et al. 1999). This study demonstrates that knowing the migration and behavior patterns of HCP species will allow managers to minimize the impacts from flow control structures such as water intakes and diversions. In addition, Miller et al. (2007) discuss that to minimize impacts from diversions on macroinvertebrate communities, diversions should preserve environmental conditions as much as possible.

11.2.10.7 Tide Gates

Tide gates can significantly alter the migration of aquatic organisms and change the natural flow regime. The less time a tide gate is closed, the less likely the impacts on HCP species will be. The type of tide gate and the materials used for its construction can influence how long the gate remains open during the day. Tide gate design is summarized in Giannico and Souder (2005), and improvements for fish passage are described in Charland (1998).

- Tide boxes with side-hinged gates result in lower velocities required to open the gate compared to top-hinged gates because less force is needed to open them.
- Gates constructed of lighter aluminum need less water to open than heavier steel or cast iron gates of comparable size (Giannico and Souder 2005).
- Side-hinged gates open slower such that they also reduce bubbling, turbulence, and scour (Giannico and Souder 2005).

- If information is known about the local behavior and migration patterns of HCP fish or other species, tide boxes may be manually opened to maximize passage during migration and other high-use periods.

12 References[1]

Aarestrup, K., and A. Koed. 2003. Survival of Migrating Sea Trout (Salmo Trutta) and Atlantic Salmon (Salmo Salar) Smolts Negotiating Weirs in Small Danish Rivers. Ecology of Freshwater Fish 12(3): 169-176.

Abbe, T.B., and D.R. Montgomery. 1996. Large Woody Debris Jams, Channel Hydraulics and Habitat Formation in Large Rivers. Regulated Rivers-Research & Management 12(2-3): 201-221.

Abbe, T.B., and D.R. Montgomery. 2003. Patterns and Processes of Wood Debris Accumulation in the Queets River Basin, Washington. Geomorphology 51(1-3): 81-107.

Abbott, R. R., E. Bing-Sawyer, and R. Blizard. 2002. Administrative Draft Assessment of Pile Driving Impacts on the Sacramento Blackfish (Orthodon micrcrolepidotus). Caltrans, Oakland, California

Abdelrhman, M.A. 2003. Effect of Eelgrass Zostera Marina Canopies on Flow and Transport. Marine Ecology-Progress Series 248: 67-83.

Able, K.W., J.P. Manderson, and A.I. Studholme. 1998. The distribution of shallow water juvenile fishes in an urban estuary: the effects of man-made structures in the Lower Hudson River. Estuaries 21: 731-44. Cited in Nightingale and Simenstad 2001b.

Adams, P.B., and J.E. Hardwick. 1992. Lingcod. In California's Living Marine Resources and Their Utilization. Edited by W.S. Leet, C.M. Dewees and C.W. Haugen. Davis, California: California Sea Grant College Program.

Adams, P.B., C.B. Grimes, J.E. Hightower, S.T. Lindley, and M.L. Moser. 2002. Status Review for North American Green Sturgeon, Acipenser mediorstris. National Marine Fisheries Service, Southwest Fisheries Science Center. June 2002.

AFS and SER. 2000. Review of the 29 April 1999 Forests and Fish Report and of Associated Draft Emergency Forest Practice Rules. Northwest Chapter of the Society for Ecological Restoration.

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[1] These references were compiled from ten white papers prepared for the Washington Department of Fish and Wildlife in 2006 and 2007. Duplicate references were removed. Because the original white papers did not all use the same format for citations, some variation in format remains.

If one of the ten original white papers referenced different works by the same author(s) that were published in the same year, they are indicated by adding (a), (b), (etc.) after the year of publication.

However, if different works by the same author(s) in the same year were compiled from different white papers, such apparent duplications were not resolved in the compiled text, and each paper is listed here by year only.

Agostinho, C.S., A.A. Agostinho, F. Pelicice, D. de Almeida, and E.E. Marques. 2007. Selectivity of Fish Ladders: A Bottleneck in Neotropical Fish Movement. Neotropical Ichthyology 5(2): 205-213.

Ahearn, D.S., and R.A. Dahlgren. 2005. Sediment and Nutrient Dynamics Following a Low-Head Dam Removal at Murphy Creek, California. Limnology and Oceanography 50(6): 1752-1762.

Ahearn, D.S., J.H. Viers, J.F. Mount, and R.A. Dahlgren. 2006. Priming the Productivity Pump: Flood Pulse Driven Trends in Suspended Algal Biomass Distribution across a Restored Floodplain. Freshwater Biology 51: 1417-1433.

Ahearn, D.S., R.W. Sheibley, and R.A. Dahlgren. 2005. Effects of River Regulation on Water Quality in the Lower Mokelumne River, California. River Research and Applications 21(6): 651-670.

Ahn, I.Y. and Choi, J.W. 1998. Macrobenthic communities impacted by anthropogenic activities in an intertidal sand flat on the West Coast (Yellow Sea) of Korea. Marine Pollution Bulletin 36(10):808-817.

Ainslie, B.J., J.R. Post, and A.J. Paul. 1998. Effects of Pulsed and Continuous DC Electrofishing on Juvenile Rainbow Trout. North American Journal of Fisheries Management 18. (4): 905-918.

Airoldi, L., and S.J. Hawkins. 2007. Negative Effects of Sediment Deposition on Grazing Activity and Survival of the Limpet Patella Vulgata. Marine Ecology-Progress Series 332: 235-240.

Aitken, J.K. 1998. The importance of estuarine habitats to anadromous salmonids of the Pacific Northwest: a literature review. U.S. Fish and Wildlife Service, Lacey, WA. 25pp.

Aksnes, D.L., and A.C.W. Utne. 1997. A Revised Model of Visual Range in Fish. Sarsia 82(2): 137-147.

Alaska Natural Heritage Program. 2006. Longfin smelt. Compiled by Tracey Gotthardt, ANHP. Anchorage, AK.

Albers, W. D., and P.J. Anderson. 1985. Diet of Pacific cod, Gadus macrocephalus, and predation on the northern pink shrimp, Pandalus borealis, in Pavlof Bay, Alaska. Fishery Bulletin 83: 601-10. Cited in Nightingale and Simenstad 2001a.

Ali, M.A. 1959. The Ocular Structure, Retinomotor and Photobehavioral Responses of Juvenile Pacific Salmon. Ph.D. Thesis, University of British Columbia.

Ali, M.A. 1962. Influence of light intensity on retinal adaptation in Atlantic salmon (Salmo salar) yearlings. Canadian Journal of Zoology 40: 561-70. Cited in Simenstad et al. 1999, in Nightingale and Simenstad 2001b.

Ali, M.A. 1975. Retinomotor Responses. In Vision in Fishes, edited by M.A. Ali. New York, New York: Plenum Press.

Allen, H.H. and J.C. Fischenich. 1999. Coir geotextile roll and wetland plants for streambank erosion control. EMRRP Technical Notes Collection (ERDC TN-EMRRP-SR-04). U.S. Army Engineer Research and Development Center, Vicksburg, MS. Available at: www.wes.army.mil/el/emrrp.

Allen, H.H. and J.R. Leech. 1997. Bioengineering for streambank erosion control. Report 1, Guidelines Prepared for U.S. Army Corps of Engineers. 103 pp. Technical report EL-97-8 Report 1.

Altayaran, A.M., and I.M. Madany. 1992. Impact of Desalination Plant on the Physical and Chemical Properties of Seawater, Bahrain. Water Research 26(4): 435-441.

Amoser, S., and F. Ladich. 2005. Are Hearing Sensitivities of Freshwater Fish Adapted to the Ambient Noise in Their Habitats? The Journal of Experimental Biology 208: 3533-3542.

Anbutsu, K., T. Nakajima, Y. Takemon, K. Tanida, N. Goto, and O. Mitamura. 2006. Distribution of Biogeochemical Compounds in Interstitial and Surface Standing Water Bodies in the Gravel Bar of the Kizu River, Japan. Archiv Fur Hydrobiologie 166(2): 145-167.

Anchor Environmental, Coastal Geologic Services, KPG, and Shannon & Wilson. 2002. Seahurst Park Master Plan Technical Memorandum Summary of Background Information. Prepared for City of Burien.

Anchor Environmental, Jones & Stokes Associates, and R2 Resource Consultants. 2006. Scientific Needs Assessment of HPA Activities. Prepared for the Washington Department of Fish and Wildlife.

Anderson, J.J. 1990. Assessment of the Risk of Pile Driving to Juvenile Fish. Fisheries Research Institute. University of Washington, Seattle, Washington.

Anderson, P.D., D.J. Larson, and S.S. Chan. 2007. Riparian Buffer and Density Management Influences on Microclimate of Young Headwater Forests of Western Oregon. Forest Science 53(2): 254-269.

Angermeier, P.L., and J.R. Karr. 1984. Relationships between Woody Debris and Fish Habitat in a Small Warmwater Stream. Transaction of the American Fisheries Society 113: 716-726.

Angradi, T.R., E.W. Schweiger, D.W. Bolgrien, P. Ismert, and T. Selle. 2004. Bank Stabilization, Riparian Land Use and the Distribution of Large Woody Debris in a Regulated

Reach of the Upper Missouri River, North Dakota, USA. River Research and Applications 20(7): 829-846.

Anisfeld, S.C., and G. Benoit. 1997. Impacts of Flow Restrictions on Salt Marshes: An Instance of Acidification. Environmental Science and Technology 31: 1650-1657.

Anisfeld, S.C., M.J. Tobin, and B. Gaboury. 1999. Sedimentation Rates in Flow-Restricted and Restored Salt Marshes in Long Island Sound. Estuaries and Coasts 22(2A): 231-244.

Anthony, J.L. and J.A. Downing. 2003. Physical Impacts of Wind and Boat Traffic on Clear Lake, Iowa, USA. Lake and Reservoir Management. 19(1):1-14.

Appenzeller, A.R., and W.C. Legget. 1995. An Evaluation of Light-Mediated Vertical Migration of Fish Based on Hydroacoustic Analysis of the Diel Vertical Movements of Rainbow Smelt (Osmerus Mordax). Canadian Journal of Fisheries and Aquatic Science 52: 504-511.

Arkoosh, M.R., E. Casillas, E. Clemons, B.B. McCain, and U. Varanasi. 1991. Suppression of Immunological Memory in Juvenile Chinook Salmon (Oncorhynchus tshawystscha) from an Urban Estuary. Fish and Shellfish Immunology 1: 261-277.

Arkoosh, M.R., E. Casillas, E. Clemons, P. Huffman, A. Kagley, T.K. Collier, and J.E. Stine. 2001. Increased Susceptibility of Juvenile Chinook Salmon (Oncorhynus tshawytscha) to Vibriosis after Exposure to Chlorinated and Aromatic Compounds Found in Contaminated Urban Estuaries. Journal of Aquatic Animal Health 13: 257-268.

Arkoosh, M.R., E. Casillas, P. Huffman, E. Clemons, J. Evered, J.E. Stein, and U. Varanusi. 1998. Increased Susceptibility of Juvenile Chinook Salmon from a Contaminated Estuary to the Pathogen Vibro anguillarum. Transactions of the American Fisheries Society 127: 360-374.

Arkoosh, M.R., E. Clemons, A.N. Kagley, C. Stafford, A.C. Glass, K. Jacobson, P. Reno, M.S. Myers, E. Casillas, L.L. Johnson, and T.K. Collier. 2004. Survey of Pathogens in Juvenile Salmon (Oncorhynchus Spp.) Migrating through Pacific Estuaries. Journal of Animal Health 16: 186-196.

Arkoosh, M.R., E. Clemons, M. Myers, and E. Casillas. 1994. Suppression of B-Cell Mediated Immunity in Juvenile Chinook Salmon (Oncorhynchus Tshawytscha) after Exposure to Either a Polycyclic Aromatic Hydrocarbon or to Polychlorinated Biphenyls. Immunopharmacology and Immunotoxicology 16: 293-3214.

Arkoosh, M.R., E. Clemons, P. Huffman, and A.N. Kagley. 2001. Increased Susceptibility of Juvenile Chinook Salmon to Vibriosis after Exposure to Chlorinated and Aromatic Compounds Found in Contaminated Urban Estuaries. Journal of Aquatic Animal Health 13(3): 257-268.

Armstrong, D.A., B.G. Stevens, and J.E. Hoeman. 1982. Distribution and Abundance of Dungeness Crab and Crangon Shrimp and Dredging-related Mortality of Invertebrates and Fish in Grays Harbor, Washington. DACW67-80-C-0086. Seattle, Washington: School of Fisheries, University of Washington.

Assani, A.A., and F. Petit. 2004. Impact of Hydroelectric Power Releases on the Morphology and Sedimentology of the Bed of the Warche River (Belgium). Earth Surface Processes and Landforms 29(2): 133-143.

Astrup, J., and B. Mohl. 1998. Discrimination between high and low repetition rates of ultrasonic pulses by cod. Journal of Fish Biology 52, 205-208. Cited in Scholik and Yan 2002.

Atilla, N., M.A. Wetzel, and J.W. Fleeger. 2003. Abundance and Colonization Potential of Artificial Hard Substrate-Associated Meiofauna. Journal of Experimental Marine Biology and Ecology 287(2): 273-287.

Au, D.W.T., C.A. Pollino, R.S.S. Wu, P.K.S. Shin, S.T.F. Lau, and J.Y.M. Tang. 2004. Chronic Effects of Suspended Solids on Gill Structure, Osmoregulation, Growth, and Triiodothyronine in Juvenile Green Grouper Epinephelus Coioides. Marine Ecology-Progress Series 266: 255-264.

Auble, G., P. Shafroth, M. Scott, and J. Roelle. 2007. Early Vegetation Development on an Exposed Reservoir: Implications for Dam Removal. Environmental Management 39(6): 806-818.

Azous, A.L. 1991. An Analysis of Urbanization Effects on Wetland Biological Communities. Master’s Thesis, University of Washington, Seattle, Washington.

Azous, A.L., and R.R. Horner. 2001. Wetlands and Urbanization: Implications for the Future. Boca Raton: Lewis Publishers.

Babanin, A.V. 2006. On a Wave-Induced Turbulence and a Wave-Mixed Upper Ocean Layer. Geophysical Research Letters 33(20).

Babbitt, K.J., and G.W. Tanner. 1998. Effects of Cover and Predator Size on Survival and Development of Rana Utricularia Tadpoles. Oecologia 114(2): 258-262.

Babcock, W.H. 1986. Tenmile Creek - a Study of Stream Relocation. Water Resources Bulletin 22(3): 405-415.

Bacchiocchi, F., and L. Airoldi. 2003. Distribution and Dynamics of Epibiota on Hard Structures for Coastal Protection. Estuarine Coastal and Shelf Science 56(5-6): 1157-1166.

Backman, T.W., and D.C. Barilotti. 1976. Irradiance Reduction - Effects on Standing Crops of Eelgrass Zostera-Marina in a Coastal Lagoon. Marine Biology 34(1): 33-40. Cited in Nightingale and Simenstad 2001b.

Bailey, K.M. 1982. The Early Life History of the Pacific Hake, Merluccius Productus. Fishery Bulletin 80: 589-598. Cited in NRC 2001.

Bailey, K.M., T.J. Quinn, P. Bentzen, and W.S. Grant. 1999. Population Structure and Dynamics of Walleye Pollock, Theragra Chalcogramma. Advances in Marine Biology 37: 179-255. Cited in NRC 2001.

Baillie, B.R., and T.R. Davies. 2002. Influence of Large Woody Debris on Channel Morphology in Native Forest and Pine Plantation Streams in the Nelson Region, New Zealand. New Zealand Journal of Marine and Freshwater Research 36(4): 763-774.

Baine, M. 2001. Artificial Reefs: A Review of Their Design, Application, Management and Performance. Ocean & Coastal Management 44(3-4): 241-259.

Baker, C.F. 2003. Effect of Fall Height and Notch Shape on the Passage of Inanga (Galaxias Maculatus) and Common Bullies (Gobiomorphus Cotidianus) over an Experimental Weir. New Zealand Journal of Marine and Freshwater Research 37(2): 283-290.

Baker. P. 1995. Review of Ecology and Fishery of the Olympia Oyster, Ostrea lurida with Annotated Bibliography. Journal of Shellfish Research 14(2):501-518. Cited in WDNR 2006b.

Baldwin, D.H., J.F. Sandahl, J.S. Labenia, and N.L. Scholz. 2003. Sublethal Effects of Copper on Coho Salmon: Impacts on Nonoverlapping Receptor Pathways in the Peripheral Olfactory Nervous System. Environmental Toxicology and Chemistry 22(10): 2266-2274.

Baldwin, J.R., and J.R. Lovvorn. 1994. Expansion of Seagrass Habitat by the Exotic Zostera Japonica, and Its Use by Dabbling Ducks and Brant in Boundary Bay, British Columbia. Marine Ecology Progress Series 103: 119-127.

Ban, M. 2006. Rearing conditions to develop seawater tolerance in underyearling sockeye salmon smolt. Fisheries Science 72(1): 128-135.

Banner, A., and M. Hyatt. 1973. Effects of Noise on Eggs and Larvae of Two Estuarine Fishes. Transactions of the American Fisheries Society 102: 134-136.

Barber, L.B., S.F. Murphy, P.L. Verplanck, M.W. Sandstrom, H.E. Taylor, and E.T. Furlong. 2006. Chemical Loading into Surface Water Along a Hydrological, Biogeochemical, and Land Use Gradient: A Holistic Watershed Approach. Environmental Science & Technology 40(2): 475-486.

Barber, M.E., M.G. Brown, K.M. Lingenfelder, and D.R. Yonge. 2006. Phase I: Preliminary Environmental Investigation of Heavy Metals in Highway Runoff. Pullman, Washington: Washington State Transportation Center (TRAC), Washington State University.

Bargmann, G.C. 1980. Studies on Pacific cod in Agate Pass, Washington. Washington Department of Fisheries Progress Report No. 123.

Bargmann, G.C. 1998. Forage Fish Management Plan: A Plan for Managing the Forage Fish Resources of Washington. Olympia, Washington: Washington Department of Fish and Wildlife.

Barks, C.S. and J.E. Funkhouser. 2002. Effects of a simulated change in land cover on surface-water velocity distribution at a bridge in southeastern Arkansas: in Proceedings of the Second Federal Interagency Hydrologic Modeling Conference, Las Vegas, NV, July 29 - August 1, 2002: Subcommittee on Hydrology of the Interagency Advisory Committee on Water Information.

Barnard, B. 2002. Road Impounded Wetlands - Planning Guidance. Olympia, Washington: Washington Department of Fish and Wildlife - Habitat Program.

Barrett, K., W. Goldsmith, and M. Silva. 2006. Integrated Bioengineering and Geotechnical Treatments for Streambank Restoration and Stabilization Along a Landfill. Journal of Soil and Water Conservation 61(3): 144-153.

Barrett, M.E. 2005. BMP Performance Comparisons: Examples from the International Stormwater BMP Database. World Water Congress 2005, Anchorage, Alaska, USA, May 15, 2005. pp. 163.

Barrett, M.E., J.M. Malina, R.J. Charbeneau, and G.H. Ward. 1995. Characterization of Highway Runoff in the Austin, Texas. CRWR 263. Austin, Texas: Center for Research in Water Resources.

Bartholow, J.M. 2002. Estimating cumulative effects of clearcutting on stream temperatures. Available at: http://smig.usgs.gov/SMIG/features_0902/clearcut.html.

Barton, D.R., W.D. Taylor, and R.M. Biette. 1985. Dimensions of Riparian Buffer Strips Required to Maintain Trout Habitat in Southern Ontario Streams. North American Journal of Fisheries Management 5: 364-378.

Bartz, K.K., and R.J. Naiman. 2005. Effects of Salmon-Borne Nutrients on Riparian Soils and Vegetation in Southwest Alaska. Ecosystems 8(5): 529-545.

Bash, J., C.H. Berman, and S. Bolton. 2001. Effects of Turbidity and Suspended Solids on Salmonids. WA-RD 526.1. Olympia, Washington: Washington State Department of Transportation.

Bates, K. 1997. Fishway Design Guidelines for Pacific Salmon. Washington Department of Fish and Wildlife. Washington Department of Fish and Wildlife - Lands and Restoration Services Program. Olympia, Washington.

Bates, K. 2003. Design of road culverts for fish passage. Olympia: Washington Department of Fish and Wildlife. Available at http://wdfw.wa.gov/hab/engineer/cm/ (Accessed 2006.10.04).

Bates, K. 2007. Personal communication (telephone conversation), with E. Doyle, Herrera Environmental Consultants, December 14, 2007. Discussion of the potential for culvert removal or replacement to liberate arrested headcuts, effects on hydraulic and geomorphic conditions, and the prevalence of this issue in culvert replacement practice in Washington State. Consulting hydraulic engineer (formerly with WDFW engineering division). Seattle, Washington.

Bates, K. 2008. Personal communication with E. Doyle, Herrera Environmental Consultants, Seattle, Washington. January 8, 2008.

Bates, K., B. Barnard, B. Heiner, J.P. Klavas, and P.D. Powers. 2003. Design of Road Culverts for Fish Passage (Revised). Olympia, Washington: Washington Department of Fish and Wildlife, Aquatic Habitat Division.

Bates, K., D. Cenderelli, R.A. Gubernick, S.D. Jackson, and D.K. Johansen. 2008. Stream Simulation: An Ecological Approach to Providing Passage for Aquatic Organisms at Road-Stream Crossings. San Dimas, California: U.S. Forest Service. Partial draft, in progress.

Battelle Marine Sciences Laboratory. 2003. Immunocompetence of Juvenile Chinook Salmon Against Listonell Anguillarum Following Dietary Exposure to Polcyclic Aromatic Hydrocarbons, Profishent, Battelle Marine Science Lab: Palm, Powell, Skillman, Godtfredsen.

Battelle Marine Sciences Laboratory. 2004. Hydroacoustic Monitoring During Beach Pile Driving at Hood Canal Bridge on June 14th, 2004. Battelle Marine Sciences Laboratory, Sequim, Washington.

Bauer, B.O., M.S. Lorang, and D.J. Sherman. 2002. Estimating Boat-Wake-Induced Levee Erosion Using Sediment Suspension Measurements. Journal of Waterway, Port, Coastal, and Ocean Engineering 128(4): 152-162.

Baxter, C.V., and F.R. Hauer. 2000. Geomorphology, Hyporheic Exchange, and Selection of Spawning Habitat by Bull Trout (Salvelinus Confluentus). Canadian Journal of Fisheries and Aquatic Sciences 57(7): 1470-1481.

Baxter, J.S., and J.D. McPhail. 1999. The Influence of Redd Site Selection, Groundwater Upwelling, and over-Winter Incubation Temperature on Survival of Bull Trout (Salvelinus Confluentus) from Egg to Alevin. Canadian Journal of Zoology-Revue Canadienne De Zoologie 77(8): 1233-1239.

Baxter, R.M. 1977. Environmental Effects of Dams and Impoundments. Annual Review of Ecology and Systematics 8: 255-283.

Bay, S., B.H. Jones, and J. Schiff. 1999. Study of the Impact of Stormwater Discharge on Santa Monica Bay. SCU-T-99-001 C3. Alhambra, California: Los Angeles County Department of Public Works.

Baylar, A., and T. Bagatur. 2000. Aeration Performance of Weirs. Water SA 26(4): 521-526.

Bayley, P.B. 1991. The Flood Pulse Advantage and the Restoration of River-Floodplain Systems. Regulated Rivers Research and Management 6: 75-86.

Bayley, P.B. 1995. Understanding Large River Floodplain Ecosystems. Bioscience 45(3): 153-158.

Beacham, T.D., and C.B. Murray. 1990. Temperature, Egg Size, and Development of Embryos and Alevins of 5 Species of Pacific Salmon - a Comparative-Analysis. Transactions of the American Fisheries Society 119(6): 927-945.

Beamer, E., A. McBride, C. Greene, R. Henderson, G. Hood, K. Wolf, K. Larsen, C. Rice, and K. Fresh. 2005. Delta and Nearshore Restoration for the Recovery of Wild Skagit River Chinook Salmon: Linking Estuary Restoration to Wild Chinook Salmon Populations. LaConner, Washington: Skagit River System Cooperative.

Beamer, E.M., and R.A. Henderson. 1998. Juvenile Salmonid Use of Natural and Hydromodified Stream Bank Habitat in the Mainstem Skagit River, Northwest Washington. Prepared for United States Army Corps of Engineers, Seattle District, Environmental Resources Section, Seattle, Washington. September 1998.

Beamer, E.M., B. Hayman, and D. Smith. 2005. Linking Freshwater rearing habitat to Skagit chinook salmon recovery; Appendix C of the Skagit chinook recovery plan. Skagit System Cooperative, La Conner, WA.

Bearzi, G., D. Holcer, and G.N. Di Sciara. 2004. The Role of Historical Dolphin Takes and Habitat Degradation in Shaping the Present Status of Northern Adriatic Cetaceans. Aquatic Conservation: Marine and Freshwater Ecosystems 14(4): 363-379.

Beatty, D.D. 1965. A study of the succession of visual pigments in Pacific Salmon (Oncorhynchus). Canadian Journal of Zoology 44: 429-55.

Beauchamp, D.A., E.R. Byron, and W.A. Wurtsbaugh. 1994. Summer Habitat Use by Littoral-Zone Fishes in Lake Tahoe and the Effects of Shoreline Structures. North American Journal of Fisheries Management 14: 385-394.

Beaver Solutions. 2007. Beaver Solutions: Consulting and Beaver Management Services. Available at: http://www.beaversolutions.com/FAQ.asp (accessed July 21, 2007).

Beckett, D.C., T.P. Aartila, and A.C. Miller. 1992. Contrasts in Density of Benthic Invertebrates between Macrophyte Beds and Open Littoral Patches in Eau-Galle Lake, Wisconsin. American Midland Naturalist 127(1): 77-90.

Bednarek, A.T. 2001. Undamming Rivers: A Review of the Ecological Impacts of Dam Removal. Environmental Management 27(6): 803-814.

Beechie, T., E. Beamer, and L. Wasserman. 1994. Estimating salmon rearing habitat and smolt production losses in a large river basin, and implications for habitat restoration. North American Journal of Fisheries Management 14: 797-811.

Beechie, T.J., and T.H. Sibley. 1997. Relationships between Channel Characteristics, Woody Debris, and Fish Habitat in Northwestern Washington Streams. Transactions of the American Fisheries Society 126(2): 217-229.

Beechie, T.J., B.D. Collins, and G. Pess. 2001. Holocene and Recent Changes to Fish Habitats in Two Puget Sound Basins. In Geomorphic Processes and Riverine Habitat, edited by J.M. Dorava, B. Palcsak, F. Fitzpatrick, and D.R. Montgomery. Washington, DC: American Geophysical Union. pp. 37-54.

Beechie, T.J., E. Beamer, and L. Wasserman. 1994. Estimating Coho Salmon Rearing Habitat and Smolt Production Losses in a Large River Basin, and Implications for Habitat Restoration. North American Journal of Fisheries Management 14(4): 797-811.

Beeman, J.W., D.W. Rondorf, and M.E. Tilson. 1994. Assessing Smoltification of Juvenile Spring Chinook Salmon (Oncorhynchus-Tshawytscha) Using Changes in Body Morphology. Canadian Journal of Fisheries and Aquatic Sciences 51(4): 836-844.

Behlke, C.E. 1991. Fundamentals of Culvert Design for Passage of Weak-Swimming Fish: Final Report. Alaska Dept. of Transportation and Public Facilities.

Bell, M.C. 1991. Fisheries Handbook of Engineering Requirements and Biological Criteria. Prepared for the U.S. Army Corps of Engineers North Pacific Division Fish Passage Development and Evaluation Program. Portland, Oregon.

Bell, S.S., M.O. Hall, S. Soffian, and K. Madley. 2002. Assessing the impact of boat propeller scars on fish and shrimp utilizing seagrass beds. Ecological Applications 12:206-217.

Bellotti, G. 2004. A Simplified Model of Rip Currents Systems around Discontinuous Submerged Barriers. Coastal Engineering 51(4): 323-335.

Benda, L.E. and T.W. Cundy. 1990. Predicting deposition of debris flows in mountain channels. Canadian Geotechnical Journal 27(4) : 409-417.

Ben-David, M., T.A. Hanley, and D.M. Schell. 1998. Fertilization of Terrestrial Vegetation by Spawning Pacific Salmon: The Role of Flooding and Predator Activity. Oikos 83(1): 47-55.

Bennett, D.H., W.P. Connor, and C.A. Eaton. 2003. Substrate Composition and Emergence Success of Fall Chinook Salmon in the Snake River. Northwest Science 77(2): 93-99.

Bennett, M. 2007. Managing Himalayan Blackberry in Western Oregon Riparian Areas. EM 8894. Corvallis, Oregon: Oregon State University.

Bennett, S.J., T. Pirim, and B.D. Barkdoll. 2002. Using simulated emergent vegetation to alter stream flow direction within a straight experimental channel. Geomorphology 44: 115-126.

Benstead, J.P., J.G. March, C.M. Pringle, and F.N. Scatena. 1999. Effects of a Low-Head Dam and Water Abstraction on Migratory Tropical Stream Biota. Ecological Applications 9(2): 656-668.

Berg, L. 1982. The effect of exposure to short-term pulses of suspended sediment on the behavior of juvenile salmonids. P. 177-196 in G.F. Hartman et al. [eds.] Proceedings of the Carnation Creek workshop: a ten-year review. Department of Fisheries and Oceans, Pacific Biological Station, Nanaimo, Canada. In Bash et al. 2001.

Berg, L., and T.G. Northcote. 1985. Changes in Territorial, Gill-Flaring, and Feeding-Behavior in Juvenile Coho Salmon (Oncorhynchus-Kisutch) Following Short-Term Pulses of Suspended Sediment. Canadian Journal of Fisheries and Aquatic Sciences 42(8): 1410-1417.

Berman, C.H., and T.P. Quinn. 1991. Behavioral Thermoregulation and Homing by Spring Chinook Salmon, Oncorhynchus Tshawytscha (Walbaum), in the Yakima River. Journal of Fish Biology 39: 312.

Bernhardt, E.S., and M.A. Palmer. 2007. Restoring Streams in an Urbanizing World. Freshwater Biology 52: 738-751.

Bernhardt, E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G.M. Kondolf, P.S. Lake, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Ecology - Synthesizing US River Restoration Efforts. Science 308(5722): 636-637.

Berry, W., N. Rubenstein, B. Melzian, and B. Hill. 2003. The Biological Effects of Suspended and Bedded Sediments (SABS) in Aquatic Systems: A Review. Internal Report. U.S. Environmental Protection Agency, Office of Research and Development, Narragansett, RI. August 20, 2003.

Beschta, R.L. 1991. Stream Habitat Management for Fish in the Northwestern United States: The Role of Riparian Vegetation. American Fisheries Society Symposium 10: 53-58.

Beschta, R.L. 1997. Riparian Shade and Stream Temperature: An Alternative Perspective. Rangelands 19(2): 25-28.

Beschta, R.L., and R.L. Taylor. 1988. Stream Temperature Increases and Land-Use in a Forested Oregon Watershed. Water Resources Bulletin 24(1): 19-25.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1988. Stream temperature and aquatic habitat: Fishery and forestry interactions. In Streamside Management: Forestry and Fishery Interactions: Contribution No. 57. Seattle, Washington: University of Washington, Institute of Forest Resources. pp. 191-232.

Bestgen, K.R., J.M. Bundy, K.A. Zelasko, and T.L. Wahl. 2004. Effectiveness of High-Velocity Inclined Profile-Bar Fish Screens Measured by Exclusion and Survival of Early Life Stages of Fathead Minnow. North American Journal of Fisheries Management 24(4): 1228-1239.

Bhowmilk, N.G., T.W. Soong, W.F. Reichelt, and N.M.L. Seddick. 1991. Waves Generate by Recreational Traffic on the Upper Mississippi River System. Illinois State Water Survey.

Bilby, R. E. 1984. Post-logging removal of woody debris affects stream channel stability. Journal of Forestry 82(10): 609-613. Cited in Naiman et al. 2002.

Bilby, R.E., and J.W. Ward. 1991. Characteristics and Function of Large Woody Debris in Streams Draining Old-Growth, Clear-Cut, and 2nd-Growth Forests in Southwestern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48(12): 2499-2508.

Bilby, R.E., and P.A. Bisson. 1992. Allochthonous Versus Autochthonous Organic-Matter Contributions to the Trophic Support of Fish Populations in Clear-Cut and Old-Growth Forested Streams. Canadian Journal of Fisheries and Aquatic Sciences 49(3): 540-551.

Bilby, R.E., and P.A. Bisson. 1998. Function and Distribution of Large Woody Debris. In River Ecology and Management: Lessons from the Pacific Coastal Ecoregion, edited by R.J. Naiman and R.E. Bilby. Springer-Verlag, New York. pp. 324-347.

Bilby, R.E., B.R. Fransen, P.A. Bisson, and J.K. Walter. 1998. Response of Juvenile Coho Salmon (Oncorhynchus Kisutch) and Steelhead (Oncorhynchus Mykiss) to the Addition of Salmon Carcasses to Two Streams in Southwestern Washington, USA. Canadian Journal of Fisheries and Aquatic Sciences 55(8): 1909-1918.

Birtwell, I.K., G. Hartman, B. Anderson, D.J. McLeay, and J.K. Malick. 1984. A Brief Investigation Of Arctic Grayling (Thymallus Arcticus) And Aquatic Invertebrates In The Minto Creek Drainage, Mayo, Yukon Territory. Canadian Technical Report Of Fisheries And Aquatic Sciences 1287.

Bisson, P.A. and R.E. Bilby. 1982. Avoidance of Suspended Sediments by Juvenile Coho Salmon. North American Journal of Fish Management 2: 371-374.

Bisson, P.A., and R.E. Bilby. 1998. Organic Matter and Trophic Dynamics. In River Ecology and Management: Lessons Form the Pacific Coastal Ecoregion, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag.

Bisson, P.A., R.E. Bilby, M.D. Bryant, C.A. Colloff, G.B. Frette, and R.A. House. 1987. Large Woody Debris in Forested Streams in the Pacific Northwest: Past, Present, and Future. In Streamside Management: Forestry and Fishery Interactions, edited by E.O. Salo and T.W. Cundy. Seattle, Washington: University of Washington.

Bjornn, T.C., and D.W. Reiser. 1991. Habitat Requirements of Salmonids in Streams. In Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats, edited by W.R. Meehan. Bethesda, Maryland: American Fisheries Society Special Publication 19. pp. 83-138.

Bjornn, T.C., S.C. Kirking, and W.R. Meehan. 1991. Relation of Cover Alterations to the Summer Standing Crop of Young Salmonids in Small Southeast Alaska Streams. Transactions of the American Fisheries Society 120(5): 562-570.

Blaber, S.J.M., D.P. Cyrus, J.J. Albaret, C.V. Ching, J.W. Day, M. Elliott, M.S. Fonseca, D.E. Hoss, J. Orensanz, I.C. Potter, and W. Silvert. 2000. Effects of Fishing on the Structure and Functioning of Estuarine and Nearshore Ecosystems. ICES Journal of Marine Science 57(3): 590-602.

Black, J.A. and W.J. Birge. 1980. An Avoidance Response Bioassay for Aquatic Pollutants. Res. Report No. 123, U.S. NTIS PB80-180490. Water Resources Research Institute, University of Kentucky, Lexington. (As cited in Stratus 2005b).

Black, K.P., and G.D. Parry. 1999. Entrainment, Dispersal, and Settlement of Scallop Dredge Sediment Plumes: Field Measurements and Numerical Modeling. Canadian Journal of Fisheries and Aquatic Sciences 56(12): 2271-2281.

Black, M.C., D.S. Millsap, and J.F. Mccarthy. 1991. Effects of Acute Temperature-Change on Respiration and Toxicant Uptake by Rainbow-Trout, Salmo-Gairdneri (Richardson). Physiological Zoology 64(1): 145-168.

Blackley, T. 2004. Screening Irrigation Offtakes in the Murray-Darling Basin to Reduce Loss of Native Fish. Proceedings of the Workshop on Downstream Fish Movements. Murray-Darling Basin Commission, Canberra.

Blackmon, D., T. Wyllie-Echeverria, and D.J. Shafer. 2006. The Role of Seagrasses and Kelps in Marine Fish Support. U.S. Army Corps of Engineers. Wetlands Regulatory Assistance Program ERDC TN-WRAP-06-1. February.

Blanton, S.L., R.M. Thom, and J.A. Southard. 2001. Documentation of Ferry Terminal Shading, Substrate Composition, and Algal and Eelgrass Coverage. Seattle, Washington: Battelle Marine Sciences Laboratory.

Blaxter, J.H.S. 1975. Fish Vision and Applied Research. In Vision in Fishes: New Approaches in Research, edited by M.A. Ali. New York: Plenum Press.

Blaxter, J.H.S., J.A.B. Gray, and E.J. Denton. 1981. Sound And Startle Responses In Herring Shoals. J. Mar. Biol. Ass. U.K. 6: 851-69. Cited in Scholik and Yan 2001a.

Blinn, D.W., J.P. Shannon, L.E. Stevens, and J.P. Carder. 1995. Consequences of fluctuating discharge for lotic communities. Journal of the North American Benthological Society 14:233-248. Cited in Watters et al., 1999.

Block, D.G. 1955. Trout Migration and Spawning Studies on the North Fork Drainage of the Flathead River. University of Montana, Missoula, Montana.

Blomqvist, S., A. Gunnars, and R. Elmgren. 2004. Why the Limiting Nutrient Differs between Temperate Coastal Seas and Freshwater Lakes: A Matter of Salt. Limnology and Oceanography 49(6): 2236-2241.

Boehlert, G.W. 1980. Size composition, age composition, and growth of canary rockfish, Sebastes pinniger, and splitnose rockfish, S. diploproa, from the 1977 rockfish survey. Marine Fisheries Review 42:57-63. Cited in NRC 2001.

Boehlert, G.W., and J.B. Morgan. 1985. Turbidity enhances feeding abilities of larval Pacific herring, Clupea harengus pallasi. Hydrobiologia 123:161-170.

Boehlert, G.W., and M.M. Yoklavich. 1983. Effects of temperature, ration, and fish size on growth of juvenile black rockfish, Sebastes melanops. Environmental Biology of Fishes 8:17-28. Cited in NRC 2001.

Boehlert, G.W., and R.F. Kappenman. 1980. Variation of Growth with Latitude in Two Species of Rockfish (Sebastes Pinniger and S. Diploproa) from the Northeast Pacific Ocean. Marine Ecology-Progress Series 3: 1-10. Cited in NRC 2001.

Boehlert, G.W., M.M. Yoklavich, and D.B. Chelton. 1989. Time series of growth in the genus Sebastes from the northeast Pacific Ocean. Fishery Bulletin 87:791-806. Cited in NRC 2001.

Boese, B.L., B.D. Robbins, and G. Thursby. 2005. Desiccation Is a Limiting Factor for Eelgrass (Zostera Marina L.) Distribution in the Intertidal Zone of a Northeastern Pacific (USA) Estuary. Botanica Marina 48(4): 274-283.

Boggs, C.T., M.L. Keefer, C.A. Peery, T.C. Bjornn, and L.C. Stuehrenberg. 2004. Fallback, Reascension, and Adjusted Fishway Escapement Estimates for Adult Chinook Salmon and Steelhead at Columbia and Snake River Dams. Transactions of the American Fisheries Society 133(4): 932-949.

Bohn, C.C., and J.G. King. 2000. Stream Channel Responses to Streamflow Diversion on Small Streams of the Snake River Drainage, Idaho. Research Paper RMRS-RP-20. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, Colorado.

Bolam, S.G., and H.L. Rees. 2003. Minimizing Impacts of Maintenance Dredged Material Disposal in the Coastal Environment: A Habitat Approach. Environmental Management 32(2): 171-188.

Bolton, S. and J. Shellberg. 2001. Ecological issues in floodplains and riparian corridors. Submitted to Washington Department of Fish and Wildlife Washington Department of Ecology Washington Department of Transportation. Available at http://www.wsdot.wa.gov/research/reports/fullreports/524.1.pdf (Accessed 2006.10.04).

Bolton, S.M., J. Moss, J. Southard, G. Williams, C. Deblois, and N. Evans. 2002. Juvenile Coho Movement Study. Washington State Transportation Center (TRAC).

Bombace, G., G. Fabi, L. Fiorentini, and S. Speranza. 1994. Analysis of the Efficacy of Artificial Reefs Located in 5 Different Areas of the Adriatic Sea. Bulletin of Marine Science 55(2-3): 559-580.

Bonar, Dr. D.B. 1995. Juvenile Salmonid Outmigration at the Everett Homeport: Effects of Shoreline Shading and Offshore Pile Driving Activities. Aquatic Environmental Services. Final Report to Washington Department of Fish and Wildlife. July 26, 1995.

Bonnell, G. R. 1991. Construction, operation and evaluation of groundwater-fed side channels for chum salmon in British Columbia. Pages 109-124 in Colt, J. and R. J. White, editors. Fisheries bioengineering symposium. American Fisheries Society Symposium 10.

Bonner, T.H. and G.R. Wilde. 2002. Effects of turbidity on prey consumption by prairie stream fishes. Transactions of the American Fisheries Society 131:1203-1208.

Booth, D. B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38:835-845.

Booth, D.B. 1990. Stream Channel Incision Following Drainage Basin Urbanization. Water Resources Bulletin 26(3): 407-417.

Booth, D.B. 1991. Urbanization and the Natural Drainage System - Impacts, Solutions, and Prognoses. Northwest Environmental Journal 7: 93-118.

Booth, D.B., and P.C. Henshaw. 2001. Rates of Channel Erosion in Small Urban Streams. In Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas, edited by M. Wigmosta and S. Burges. AGU Monograph Series, Water Science and Application Volume 2. pp. 17-38.

Booth, D.B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38: 835-845.

Bordalo, A.A. 2003. Microbiological Water Quality in Urban Coastal Beaches: The Influence of Water Dynamics and Optimization of the Sampling Strategy. Water Research 37(13): 3233-3241.

Bostrom, C., and J. Mattila. 1999. The Relative Importance of Food and Shelter for Seagrass-Associated Invertebrates: A Latitudinal Comparison of Habitat Choice by Isopod Grazers. Oecologia 120(1): 162-170.

Bottom, D.L., K.K. Jones, T.J. Cornwell, A. Gray, and C.A. Simenstad. 2005. Patterns of Chinook Salmon Migration and Residency in the Salmon River Estuary (Oregon). Estuarine Coastal and Shelf Science 64(1): 79-93.

Botton, M.L., R.E. Loveland, and T.R. Jacobsen. 1994. Site Selection by Migratory Shorebirds in Delaware Bay, and Its Relationship to Beach Characteristics and Abundance of Horseshoe-Crab (Limulus-Polyphemus) Eggs. Auk 111(3): 605-616.

Boussard, A. 1981. The reactions of roach (Rutilus rutilus) and rudd (Scardinius erythrophthalmus) to noises produced by high speed boating. pp. 188-200. In: Proceedings of 2nd British Freshwater Fisheries Conference. Cited in Scholik and Yan 2001a.

Bowen, K.L., N.K. Kaushik, and A.M. Gordon. 1998. Macroinvertebrate Communities and Biofilm Chlorophyll on Woody Debris in Two Canadian Oligotrophic Lakes. Archiv Fur Hydrobiologie 141(3): 257-281.

Bowman, D., and E. Pranzini. 2003. Reversed Responses within a Segmented Detached Breakwater, the Tuscany Coast Italy - a Case Study. Coastal Engineering 49(4): 263-274.

Bowman, M.F., and R.C. Bailey. 1998. Upper pH Tolerance Limit of the Zebra Mussel (Dreissena Polymorpha). Canadian Journal of Zoology-Revue Canadienne De Zoologie 76(11): 2119-2123.

Box, J.B. and J. Mossa. 1999. Sediment, land use and freshwater mussels: prospects and problems. Journal of the North American Bentholological Society 18(1): 99-117.

Box, J.B., D. Wolf, J. Howard, C. O'Brien, D. Nez, and D. Close. 2003. The Distribution and Status of Freshwater Mussels in the Umatilla River System. Portland, Oregon: Bonneville Power Administration. Project No. 2002-037-00. Portland, OR. 74 pp.

Bragg, D.C., and J.L. Kershner. 2004. Sensitivity of a Riparian Large Woody Debris Recruitment Model to the Number of Contributing Banks and Tree Fall Pattern. Western Journal of Applied Forestry 19(2): 117-122.

Bramblett, R.G., M.D. Bryant, B.E. Wright, and R.G. White. 2002. Seasonal Use of Small Tributary and Main-Stem Habitats by Juvenile Steelhead, Coho Salmon, and Dolly Varden in a Southeastern Alaska Drainage Basin. Transactions of the American Fisheries Society 131(3): 498-506.

Brannon, E.L. 1984. Columbia River White Sturgeon (Acipenser Transmontanus) Enhancement, May 1-December 31, 1983 Final Report. DOE/BP-363; R&D Project: 1983-316-00; DE86001932. United States.

Brannon, E.L., M.S. Powell, T.P. Quinn, and A. Talbot. 2004. Population Structure of Columbia River Basin Chinook Salmon and Steelhead Trout. Reviews in Fisheries Science 12(2-3): 99-232.

Braudrick, C.A., and G.E. Grant. 2000. When Do Logs Move in Rivers? Water Resources Research 36(2): 571-583.

Bravard, J., G.M. Kondolf, and H. Piegay. 1999. Environmental and Societal Effects of Channel Incision and Remedial Strategies. In Incised River Channels: Processes, Forms, Engineering, and Management, edited by S.E. Darby and A. Simon. West Sussex, England: Wiley & Sons. pp. 452.

Breitburg, D.L. 1988. Effects of turbidity on prey consumption by striped bass larvae. Transactions of the American Fisheries Society 117:72-77.

Brennan, J. 2004. Riparian Functions and the Development of Management Actions in Marine Nearshore Ecosystems. In Proceedings of the DFO/PSAT Sponsored Marine Riparian Experts Workshop, Tsawwassen, BC, February 17-18, 2004, edited by J.P. Lemieux, J.S. Brennan, M. Farrell, C.D. Levings and D. Myers. Canadian Manuscript Report of the Fisheries and Aquatic Sciences No. 2680.

Brennan, J.S., and H. Culverwell. 2004. Marine Riparian: An Assessment of Riparian Functions in Marine Ecosystems. Published by Washington Sea Grant Program Copyright 2005, UW Board of Regents. Seattle, WA. 34 pp.

Brennan, J.S., K.F. Higgins, J.R. Cordell, and V.A. Stamatiou. 2004. Juvenile Salmon Composition, Timing Distribution, and Diet in Marine Nearshore Waters of Central Puget Sound in 2001-2002. Seattle, Washington: King County Department of Natural Resources and Parks.

Brett, J.R., and C. Groot. 1963. Some Aspects of Olfactory and Visual Responses in Pacific Salmon. Journal of the Fisheries Research Board of Canada 20: 548-559.

Brett, J.R., and M.A. Ali. 1958. Some Observations on the Structure and Photomechanical Responses of the Pacific Salmon Retina. Journal of the Fisheries Research Board of Canada 15: 815-829.

Bricelj, V.M., and R.E. Malouf. 1984. Influence of Algal and Suspended Sediment Concentrations on the Feeding Physiology of the Hard Clam Mercenaria mercenaria. Marine Biology 84(2): 155-165.

Bricelj, V.M., R.E. Malouf, and C. Dequillfeldt. 1984. Growth of Juvenile Mercenaria-Mercenaria and the Effect of Resuspended Bottom Sediments. Marine Biology 84(2): 167-173.

Brickhill, M.J., S.Y. Lee, and R.M. Connolly. 2005. Fishes Associated with Artificial Reefs: Attributing Changes to Attraction or Production Using Novel Approaches. Journal of Fish Biology 67(sb): 53-71.

Brim Box, J.B. and J. Mossa. 1999. Sediment, land use and freshwater mussels: prospects and problems. Journal of the North American Benthological Society 18(1): 99-117.

Brim-Box, J., D. Wolf, J. Howard, C. O'Brien, D. Nez, and D. Close. 2004. Distribution and Status of Freshwater Mussels in the Umatilla River System, 2002-2003 Annual Report, Project No. 200203700, 74 electronic pages (BPA Report DOE/BP-00011402-1). http://www.efw.bpa.gov/publications/I00011402-1.pdf (Accessed 2006.10.04).

Britt, L. 2001. Aspects of the Vision and Feeding Ecology of Larval Lingcod (Ophiodone longatus) and Kelp Greenling (Hexagrammos decagrammus). University of Washington.

Britt, L.L., W. McFarland, and B. Miller. 2001. Short Wavelength Vision in Larval Fishes: A Feeding Adaptation? Puget Sound Conference 2001.

Britt, Lyle L. 2001. Aspects of the vision and feeding ecology of larval lingcod (Ophiodone longatus) and kelp greenling (Hexagrammos decagrammus). M.S. Thesis, University of Washington.

Broadmeadow, S., and T.R. Nisbet. 2004. The Effects of Riparian Forest Management on the Freshwater Environment: A Literature Review of Best Management Practice. Hydrology and Earth System Sciences 8(3): 286-305.

Brock, C.S., P.R. Leavitt, D.E. Schindler, and P.D. Quay. 2007. Variable Effects of Marine-Derived Nutrients on Algal Production in Salmon Nursery Lakes of Alaska During the Past 300 Years. Limnology and Oceanography 52(4): 1588-1598.

Brookes, A. 1988. Channelized Rivers: Perspectives for Environmental Management. John Wiley and Sons. Chichester, U.K. Cited in Bolton and Shellberg 2001.

Brooks, A.P., P.C. Gehrke, J.D. Jansen, and T.B. Abbe. 2004. Experimental Reintroduction of Woody Debris on the Williams River, NSW: Geomorphic and Ecological Responses. River Research and Applications 20(5): 513-536.

Brooks, K.M. 1997. Literature Review, Computer Model and Assessment of the Potential Environmental Risks Associated with Creosote Treated Wood Products Used in Aquatic Environments. Prepared for Western Wood Preservers Institute. April 25, 1995; Revised June 1, 1997. Cited in Brooks 2004.

Brooks, K.M. 2004. Polycyclic Aromatic Hydrocarbon Migration from Creosote-Treated Railway Ties into Ballast and Adjacent Wetlands. Madison, Wisconsin: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory.

Brooks, K.M. 2000. Assessment of the Environmental Effects Associated with Wooden Bridges Preserved With Creosote, Pentachlorophenol, or Chromated Copper Arsenate. Prepared for Dr. Michael Ritter USDA Forest Service, Forest Products Laboratory, One Gifford Pinchot Drive, Madison, WI 53705-2398. Port Townsend, Washington: Aquatic Environmental Sciences.

Brosofske, K.D., J. Chen, R.J. Naiman, and J.F. Franklin. 1997. Harvesting Effects on Microclimatic Gradients from Small Streams to Uplands in Western Washington. Ecological Applications 7(4): 1188-1200.

Browman, H.I., I. Novales-Flamarique, and C.W. Hawryshyn. 1993. Ultraviolet Photoreception Contributes to Prey Research Behavior in Two Species of Zooplanktivorous Fishes. Journal of Experimental Biology 186: 187-98.

Brown, G.W. 1970. Predicting Effect of Clearcutting on Stream Temperature. Journal of Soil and Water Conservation 25(1): 11-13.

Brown, R.S., D.R. Geist, and the Yakama Nation. 2002. Determination of Swimming Speeds and Energetic Demands of Upriver Migrating Fall Chinook Salmon (Oncorhynchus Tshawytscha) in the Klickitat River, Washington. Pacific Northwest Laboratory; Bonneville Power Administration.

Brown, T.G. and G.F. Hartman. 1988. Contribution of seasonally flooded lands and minnow tributaries to the production of coho salmon in Carnation Creek, British Columbia. Transactions of the American Fisheries Society 117:546-551.

Brummer, C.J., T.B. Abbe, J.R. Sampson, and D.R. Montgomery. 2006. Influence of Vertical Channel Change Associated with Wood Accumulations on Delineating Channel Migration Zones, Washington, USA. Geomorphology 80(3-4): 295-309.

Brunet, R.C., G. Pinay, F. Gazelle, and L. Roques. 1994. Role of the Floodplain and Riparian Zone in Suspended Matter and Nitrogen-Retention in the Adour River, South-West France. Regulated Rivers-Research & Management 9(1): 55-63.

Bryan, M.D., and D.L. Scarnecchia. 1992. Species Richness, Composition, and Abundance of Fish Larvae and Juveniles Inhabiting Natural and Developed Shorelines of a Glacial Iowa Lake. Environmental Biology of Fishes 35(4): 329-341.

Bryant, M.D., and J.R. Sedell. 1995. Riparian Forests, Wood in the Water, and Fish Habitat Complexity. In Condition of the World's Aquatic Habitats. Proceedings of the World Fisheries Congress, edited by N.B. Armantrout. New Delhi, India: Oxford & IBH Publishing. pp. 202-224.

Bryant, M.D., T. Gomi, and J.J. Piccolo. 2007. Structures Linking Physical and Biological Processes in Headwater Streams of the Maybeso Watershed, Southeast Alaska. Forest Science 53(2): 371-383.

Buck, E.H. 1995. Acoustic Thermometry of Ocean Climate, Marine Mammal Issues. Report No. 95-603 ENR. Washington DC: Congressional Research Service, Library of Congress.

Buckley, R.M., and G.J. Hueckel. 1985. Biological Processes and Ecological Development on an Artificial Reef in Puget-Sound, Washington. Bulletin of Marine Science 37(1): 50-69.

Buell, J. 1992. Fish Entrainment Monitoring of the Western-Pacific Dredge R.W. Lofgren During Operations Outside the Preferred Work Period. Portland, Oregon.

Buer, K.Y., J.N. Eaves, R.G. Scott, and J.R. McMillan. 1984. Basin Changes Affecting Salmon Habitat in the Sacramento River. Pacific Northwest stream habitat management workshop, Arcata, California.

Buffington, J.M., and D.R. Montgomery. 1999. Effects of hydraulic roughness on surface textures of gravel-bed rivers. Water Resources Research 35: 3523-3530.

Bukaveckas, P. 2007. Effects of Channel Restoration on Water Velocity, Transient Storage, and Nutrient Uptake in a Channelized Stream. Environmental Science and Technology.

Bullard, S.G., G. Lambert, M.R. Carman, J. Byrnes, R.B. Whitlatch, G. Ruiz, R.J. Miller, L. Harris, P.C. Valentine, J.S. Collie, J. Pederson, D.C. McNaught, A.N. Cohen, R.G. Asch, J. Dijkstra, and K. Heinonen. 2007. The Colonial Ascidian Didemnum sp. A: Current Distribution, Basic Biology and Potential Threat to Marine Communities of the Northeast and West Coasts of North America. Journal of Experimental Marine Biology and Ecology 342(1): 99-108.

Bulleri, F., and M.G. Chapman. 2004. Intertidal Assemblages on Artificial and Natural Habitats in Marinas on the Northwest Coast of Italy. Marine Biology 145(2): 381-391.

Bulleri, F., M. Abbiati, and L. Airoldi. 2006. The Colonization of Human-Made Structures by the Invasive Alga Codium fragile ssp tomentosoides in the North Adriatic Sea (NE Mediterranean). Hydrobiologia 555: 263-269.

Bulthuis, D.A., and W.J. Woelkerling. 1983. Biomass Accumulation and Shading Effects of Epiphytes on Leaves of the Seagrass, Heterozostera Tasmanica in Victoria, Australia. Aquatic Botany 16: 137-148.

Bunn, S.E., and A.H. Arthington. 2002. Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. Environmental Management 30(4): 492-507.

Bunn, S.E., P.M. Davies, and M. Winning. 2003. Sources of Organic Carbon Supporting the Food Web of an Arid Zone Floodplain River. Freshwater Biology 48(4): 619-635.

Bunt, C.M., C. Katopodis, and R.S. McKinley. 1999. Attraction and Passage Efficiency of White Suckers and Smallmouth Bass by Two Denil Fishways. North American Journal of Fisheries Management 19(3): 793-803.

Bunte, K. 2004. State of the Science Review - Gravel Mitigation and Augmentation Below Hydroelectric Dams: A Geomorphological Perspective. Fort Collins, Colorado: U.S. Department of Agriculture Forest Service.

Burdick, D.M. and F.T. Short. 1999. The effects of boat docks on eelgrass beds in coastal waters of Massachusetts. Environmental Management 23(2): 231-240.

Burgess, W.C., and S.B. Blackwell. 2003. Acoustic Monitoring of Barrier Wall Installation at the Former Rhône-Poulenc Site, Tukwila, Washington. Tukwila, Washington: Greenridge Sciences, Inc.

Burgner, R.L. 1991. Life History of Sockeye Salmon (Oncorhynchus nerka). In: Groot, C. and L. Margolis [Eds.]. Pacific Salmon Life Histories. UBC Press, Vancouver, B.C.

Burner, L.C., and T.A. Rien. 2002. Incidence of White Sturgeon Deformities in Two Reaches of the Columbia River. California Fish and Game 88(2): 57-67.

Busby, M.S., and R.A. Barnhart. 1995. Potential Food Sources and Feeding Ecology of Juvenile Fall Chinook Salmon in California's Mattole River Lagoon. California Fish and Game 81(4): 133-146.

Busby, P.J, T.C. Wainwright, G.J. Bryant, L.J. Lierheimer, R.S. Waples, F.W. Waknitz, and I.V. Lagomarsino. 1996. Status Review of West Coast Steelhead from Washington, Idaho, Oregon, and California. US Department of Commerce, NOAA Technical Memo NMFS-NWFSC-27. Southwest Region, Protected Species Management Division. Long Beach, California. Available at: http://www.nwfsc.noaa.gov/publications/techmemos/tm27/tm27.htm. (Accessed 2006.10.06).

Bustard, D.R. and D.W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (Oncorhynchus kisutch) and steelhead (Salmo gairneri). Journal of Fisheries Research Board of Canada 32:667-680.

Butler, D.R., and G.P. Malanson. 1995. Sedimentation-Rates and Patterns in Beaver Ponds in a Mountain Environment. Geomorphology 13(1-4): 255-269.

Butler, D.R., and G.P. Malanson. 2005. The Geomorphic Influences of Beaver Dams and Failures of Beaver Dams. Geomorphology 71(1-2): 48-60.

Byerly, M., S.W. Johnson, M.L. Murphy, and P. Harris. [no date.] Importance of vegetated habitats to juvenile rockfish. [Poster] Available at http://www.afsc.noaa.gov/abl/habitat/images/juvrkpst.jpg. Accessed 2006.10.26.

Byrnes, M.R., and M.W. Hiland. 1995. Large-Scale Sediment Transport Patterns on the Continental-Shelf and Influence on Shoreline Response - St-Andrew-Sound, Georgia to Nassau-Sound, Florida, USA. Marine Geology 126(1-4): 19-43

Byrnes, M.R., R.M. Hammert, T.D. Thibaut, and D.B. Snyder. 2004. Effects of Sand Mining on Physical Processes and Biological Communities Offshore New Jersey, USA. Journal of Coastal Research 20(1): 25-43.

Caceres, I., A. Sanchez-Arcilla, B. Zanuttigh, A. Lamberti, and L. Franco. 2005. Wave overtopping and induced currents at emergent low crested structures. Coastal Engineering 52(10-11): 931-947.

Cada, G., T.J. Carlson, J. Ferguson, M. Richmond, and M. Sale. 1999. Exploring the Role of Shear Stress and Severe Turbulence in Downstream Fish Passage. U.S. Department of Energy, Oak Ridge National Laboratory.

Cada, G.F., and M.J. Sale. 1993. Status of Fish Passage Facilities at Nonfederal Hydropower Projects. Fisheries 18(7): 4-12.

Cada, G.F., M.G. Ryon, D.A. Wolf, and B.T. Smith. 2003. Development of a New Technique to Assess Susceptibility to Predation Resulting from Sublethal Stresses (Indirect Mortality). U.S. Department of Energy. ORNL/TM-2003/195. August 2003.

Cake, E.W., Jr. 1983. Habitat suitability index models: Gulf of Mexico American oyster. FWS/OBS/-82/10.57. US Department of Interior, Fish and Wildlife Service, Washington, D.C.

CALFED. 2005. Key Uncertainties in Gravel Augmentation: Geomorphological and Biological Research Needs for Effective River Restoration. Sacramento, California: CALFED Science Program and Ecosystem Restoration Program Gravel Augmentation Panel.

California Coastal Commission. 2000. W12b. Appeal No. A-1-MEN-00-043.

Camargo, J.A., and N.J. Voelz. 1998. Biotic and Abiotic Changes Along the Recovery Gradient of Two Impounded Rivers with Different Impoundment Use. Environmental Monitoring and Assessment 50: 143-158.

Camargo, J.A., K. Alonso, and M. de la Puente. 2005. Eutrophication Downstream from Small Reservoirs in Mountain Rivers of Central Spain. Water Research 39(14): 3376-3384.

Cameron, W.A., S.M. Knapp, and R.W. Carmichael. 1997. Evaluation of Juvenile Salmonid Bypass Facilities and Passage at Water Diversions on the Lower Umatilla River; 1991-1995 Final Report. Prepared for U.S. Department of Energy, Bonneville Power Administration by Oregon Department of Fish and Wildlife, Portland, Oregon.

Camp, D.K., S.P. Cobb, and J.F. Van Breedveld. 1973. Overgrazing of Seagrasses by a Regular Urchin, Lytechinus Variegatus. Bioscience 23: 37-38.

Canning, D.J., and H. Shipman. 1994. Coastal Erosion Management Studies in Puget Sound, Washington: Executive Summary. Coastal Erosion Management Studies Volume I. Water and Shorelands Resources Program. Report 94-74. Washington Department of Ecology. Olympia, WA.

Canning, D.J., and H. Shipman. 1994. The Cumulative Environmental Effects of Shoreline Erosion Control and Associated Land Clear Practices, Puget Sound, Washington. Coastal

Erosion Management Studies Volume 10. Water and Shorelands Resources Program. Report 94-74. Washington Department of Ecology. Olympia, WA.

Canning, D.J., and M. Stevens. 1989. Wetlands of Washington--a Resource Characterization. Olympia, Washington: Washington State Department of Ecology.

Cardoso, P.G., D. Raffaelli, and M.A. Pardal. 2007. Seagrass Beds and Intertidal Invertebrates: An Experimental Test of the Role of Habitat Structure. Hydrobiologia 575: 221-230.

Cardwell, R.D. and K.L. Fresh. 1979. Predation upon juvenile salmon. State of Washington Department of Fisheries Program Report Draft No. 8.

Cardwell, R.D., and R.R. Koons. 1981. Biological Considerations for the Siting and Design of Marinas and Affiliated Structures in Puget Sound. Olympia, Washington: Washington Department of Fisheries.

Cardwell, R.D., M.I. Carr, S.J. Olsen, and E.W. Sanborn. 1978. Water Quality and Biotic Characteristics of Birch Bay Village Marina in 1977 (October 1, 1976 to December 31, 1977). Olympia, Washington: Washington Department of Fisheries.

Cardwell, R.D., S.J. Olsen, M.I. Carr, and E.W. Sanborn. 1980. Biotic, Water Quality, and Hydrologic Characteristics of Skyline Marina in 1978. Olympia, Washington: Washington State Department of Fish and Wildlife.

Carling, P.A., and N.A. Reader. 1982. Structure, Composition and Bulk Properties of Upland Stream Gravels. Earth Surface Processes and Landforms 7: 349-365.

Carls, M.G., J.E. Hose, R.E. Thomas, and S.D. Rice. 2000. Exposure of Pacific Herring to Weathered Crude Oil: Assessing Effects on Ova. Environmental Toxicology and Chemistry 19(6): 1649-1659.

Carlson, T.J., D.A. Woodruff, G.E. Johnson, N.P. Kohn, G.R. Plosky, M.A. Weiland, J.A. Southard, and S.L. Southard. 2005. Hydroacoustic Measurements During Pile Driving at the Hood Canal Bridge, September through November. 2004. Prepared for Washington State Department of Transportation by Battelle Marine Sciences Laboratory, Sequim, Washington.

Carney, L.T., J.R. Waaland, T. Klinger, and K. Ewing. 2005. Restoration of the Bull Kelp Nereocystis Luetkeana in Nearshore Rocky Habitats. Marine Ecology-Progress Series 302: 49-61.

Carrasquero, J. 2001. Over-Water Structures: Freshwater Issues. Prepared for Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation by Herrera Environmental Consultants, Seattle, Washington. April 2001.

Carrasquero, J. Over-Water Structures: Freshwater Issues. White Paper. 2001. Herrera Environmental Consultants. Prepared for: Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation. April 12.

Carter, J.A., G.A. McMichael, and M.A. Chamness. 2003. Yakima River Basin Phase II Fish Screen Evaluations, 2002. PNNL-14205. United States.

Castro, J.C. 2003. Geomorphologic Impacts of Culvert Replacement and Removal: Avoiding Channel Incision. Portland, Oregon: U.S. Fish and Wildlife Service. pp. 19.

Caudill, C.C., W.R. Daigle, M.L. Keefer, C.T. Boggs, M.A. Jepson, B.J. Burke, R.W. Zabel, T.C. Bjornn, and C.A. Peery. 2007. Slow Dam Passage in Adult Columbia River Salmonids Associated with Unsuccessful Migration: Delayed Effects of Passage Obstacles or Condition-Dependent Mortality? Canadian Journal of Fisheries and Aquatic Sciences 64(7): 979-995.

Cech, J.J., and C.E. Crocker. 2002. Physiology of Sturgeon: Effects of Hypoxia and Hypercapnia. Journal of Applied Ichthyology 18(4-6): 320-324.

Cederholm, C., R.E. Bilby, P. Bisson, T. Bumstead, B. Fransen, W. Scarlett, and J. Ward. 1997. Response of Juvenile Coho Salmon and Steelhead to Placement of Large Woody Debris in a Coastal Washington Stream. North American Journal of Fisheries Management 17: 947-963.

Cederholm, C.J., D.B. Houston, D.L. Cole, and W.J. Scarlett. 1989. Fate of Coho Salmon (Oncorhynchus Kisutch) Carcasses in Spawning Streams. Canadian Journal of Fisheries and Aquatic Sciences 46(8): 1347-1355.

CEQ (Council on Environmental Quality). 1997. Considering Cumulative Effects Under the National Environmental Policy Act. Council on Environmental Quality, Executive Office of the President, Washington, DC.

Chaloner, D.T., and M.S. Wipfli. 2002. Influence of Decomposing Pacific Salmon Carcasses on Macroinvertebrate Growth and Standing Stock in Southeastern Alaska Streams. Journal of the North American Benthological Society 21(3): 430-442.

Chaloner, D.T., G.A. Lamberti, A.D. Cak, N.L. Blair, and R.T. Edwards. 2007. Inter-Annual Variation in Responses of Water Chemistry and Epilithon to Pacific Salmon Spawners in an Alaskan Stream. Freshwater Biology 52(3): 478-490.

Chaloner, D.T., K.M. Martin, M.S. Wipfli, P.H. Ostrom, and G.A. Lamberti. 2002. Marine Carbon and Nitrogen in Southeastern Alaska Stream Food Webs: Evidence from Artificial and Natural Streams. Canadian Journal of Fisheries and Aquatic Sciences 59(8): 1257-1265.

Chambers, P.A., R.E. DeWreede, E.A. Irlandi, and H. Vandermeulen. 1999. Management Issues in Aquatic Macrophyte Ecology: A Canadian Perspective. Canadian Journal of Botany-Revue Canadienne De Botanique 77(4): 471-487.

Chanseau, M., O. Croze, and M. Larinier. 1999. The Impact of Obstacles on the Pau River (France) on the Upstream Migration of Returning Adult Atlantic Salmon (Salmo salar L.). Bulletin Francais De La Peche Et De La Pisciculture (353-54): 211-237.

Chapman, D. 1996. Efficacy of Structural Manipulations on Instream Habitat in the Columbia River Basin. Northwest Science 5(4): 279-293.

Chapman, D.W. 1988. Critical Review of Variables Used to Define Effects of Fines in Redds of Large Salmonids. Transactions of the American Fisheries Society 117: 1-21.

Chapman, D.W., and E. Knudsen. 1980. Channelization and Livestock Impacts on Salmonid Habitat and Biomass in Small Streams of Western Washington. Transactions of the American Fisheries Society 109: 357-363.

Chapman, M.G., and B.G. Clynick. 2006. Experiments Testing the Use of Waste Material in Estuaries as Habitat for Subtidal Organisms. Journal of Experimental Marine Biology and Ecology 338(2): 164-178.

Charland, J. 1998. Tide Gate Modifications for Fish Passage and Water Quality Enhancement. Garibaldi, Oregon: Tillamook Bay National Estuary Project.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1992. Vegetation Responses to Edge Environments in Old-Growth Douglas-Fir Forests. Ecological Applications 2(4): 387-396.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1993. Contrasting Microclimates among Clear-Cut, Edge, and Interior of Old-Growth Douglas-Fir Forest. Agricultural and Forest Meteorology 63(3-4): 219-237.

Chen, J.Q., J.F. Franklin, and T.A. Spies. 1995. Growing-Season Microclimatic Gradients from Clear-Cut Edges into Old-Growth Douglas-Fir Forests. Ecological Applications 5(1): 74-86.

Chen, J.Q., S.C. Saunders, T.R. Crow, R.J. Naiman, K.D. Brosofske, G.D. Mroz, B.L. Brookshire, and J.F. Franklin. 1999. Microclimate in Forest Ecosystem and Landscape Ecology - Variations in Local Climate Can Be Used to Monitor and Compare the Effects of Different Management Regimes. Bioscience 49(4): 288-297.

Chen, L.Y., A.G. Heath, and R.J. Neves. 2001. Comparison of Oxygen Consumption in Freshwater Mussels (Unionidae) from Different Habitats During Declining Dissolved Oxygen Concentration. Hydrobiologia 450(1): 209-214.

Chen, X.Y., X.H. Wei, and R. Scherer. 2005. Influence of Wildfire and Harvest on Biomass, Carbon Pool, and Decomposition of Large Woody Debris in Forested Streams of Southern Interior British Columbia. Forest Ecology and Management 208(1-3): 101-114.

Cheney, D., R. Oestman, G. Volkhardt, and J. Getz. 1994. Creation of Rocky Intertidal and Shallow Subtidal Habitats to Mitigate for the Construction of a Large Marina in Puget-Sound, Washington. Bulletin of Marine Science 55(2-3): 772-782.

Cheng, W., and J.C. Chen. 2000. Effects of pH, Temperature and Salinity on Immune Parameters of the Freshwater Prawn Macrobrachium Rosenbergii. Fish & Shellfish Immunology 10(4): 387-391.

Cheong, T.S., M.L. Kavvas, and E.K. Anderson. 2006. Evaluation of Adult White Sturgeon Swimming Capabilities and Applications to Fishway Design. Environmental Biology of Fishes 77(2): 197-208.

Chester, H., and R. Norris. 2006. Dams and Flow in the Cotter River, Australia: Effects on Instream Trophic Structure and Benthic Metabolism. Hydrobiologia 572: 275-286.

Chisholm, I., M.E. Hensler, B. Hansen, and D. Skaar. 1989. Quantification of Libby Reservoir Levels Needed to Maintain or Enhance Reservoir Fisheries: Summary Report 1983-1985. Portland, Oregon: U.S. Department of Energy, Bonneville Power Administration, Oregon Division of Fish and Wildlife.

Chow, V.T. 1959. Open-Channel Hydraulics. San Francisco: McGraw-Hill.

Christie, M. C. and K.R. Dyner. 1998. Measurements of the turbid tidal edge over the Skeffling mudflats. Pg. 45-55 in K.S. Black, D.M. Patterson, and A. Cramp (eds.), Sedimentary Processes in the Intertidal Zone. London: Geological Society. Cited in Nightingale and Simenstad 2001b.

Chrzastowski, M.J., and T.A. Thompson. 1994. Late Wisconsin and Holocene Geologic History of the Illinois-Indiana Coast of Lake-Michigan. Journal of Great Lakes Research 20(1): 9-26.

Church, M., M.A. Hassan, and J.F. Wolcott. 1998. Stabilizing Self-Organized Structures in Gravel-Bed Stream Channels: Field and Experimental Observations. Water Resources Research 34(11): 3169-3179.

Clark, J. J.S. Banta, and J.A. Zinn. 1980. Coastal Environmental Management: guidelines for conservation of resources and protection against storm hazards. Washington, D.C.: Council on Environmental Quality. 161 p.

Clarke, D.G., and D.H. Wilber. 2000. Assessment of Potential Impacts of Dredging Operations Due to Sediment Resuspension. DOER-E9.

Clarke, D.G., C. Dickerson, and K. Reine. 2002. Characterization of Underwater Sounds Produced by Dredges. Proceedings of the Third Specialty Conference on Dredging and Dredged Material Disposal, Dredging '02, Orlando, Florida.

Clarke, S.J. 2002. Vegetation growth in rivers: influences upon sediment and nutrient dynamics. Progress in Physical Geography 26(2): 159-172.

Clary, W.P., C.I. Thornton, and S.R. Abt. 1996. Riparian Stubble Height and Recovery of Degraded Streambanks. Rangelands 18(4): 137-140.

Clemens, W.A. and G.V. Wilby. 1961. Fishes of the Pacific coast of Canada. Bulletin. Fisheries Research Board of Canada No. 68. 443pp.

Clifford, H.F., G.M. Wiley, and R.J. Casey. 1993. Macroinvertebrates of a Beaver-Altered Boreal Stream of Alberta, Canada, with Special Reference to the Fauna on the Dams. Canadian Journal of Zoology-Revue Canadienne De Zoologie 71(7): 1439-1447.

Close, D.A. 2000. Pacific Lamprey Research and Restoration Project: Annual Report 1998. Prepared for U.S. Department of Energy by Tribal Fisheries Program, Department of Natural Resources, Portland, Oregon. May 2000.

Close, D.A., M.S. Fitzpatrick, D.R. Hatch, A.D. Jackson, H. Li, and B.L. Parker. 1998. Pacific Lamprey Research and Restoration: Annual Report 1997. DOE/BP-39067-3. Portland, Oregon: U.S. Department of Energy, Bonneville Power Administration.

Cluett, L. 2005. The Role of Flooding in Morphological Changes in the Regulated Lower Ord River in Tropical Northwestern Australia. River Research and Applications 21(2-3): 215-227.

Clynick, B.G. 2006. Assemblages of Fish Associated with Coastal Marinas in North-Western Italy. Journal of the Marine Biological Association of the United Kingdom 86(4): 847-852.

Cohen, A., C. Mills, H. Berry, M. Wonham, B. Bingham, B. Bookheim, J. Carlton, J. Chapman, J. Cordell, L. Harris, T. Klinger, A. Kohn, C. Lambert, G. Lambert, K. Li, D. Secord, and J. Toft. 1998. A Rapid Assessment Survey of Non-Indigenous Species in the Shallow Waters of Puget Sound. The Puget Sound Expedition September 8-16, 1998.

Coho, C. and S.J. Burges. 1994. Dam-break floods in low order mountain channels of the Pacific Northwest. Water Resources Series Technical Report No. 138. Olympia, WA: Timber, Fish & Wildlife. http://www.ce.washington.edu/pub/WRS/WRS138, accessed 2006.10.04.

Collen, P., and R.J. Gibson. 2001. The General Ecology of Beavers (Castor Spp.), as Related to Their Influence on Stream Ecosystems and Riparian Habitats, and the Subsequent Effects on Fish - a Review. Reviews in Fish Biology and Fisheries 10(4): 439-461.

Colley, R.H., and J.E. Burch. 1961. A Small-Block Screening Test for Accelerated Evaluation of Wood Preservative for Marine Use. Proceedings of the American Wood Preservers Association. Vol. 57, pp. 1-11. Cited in Stratus 2005a.

Collier, M.P., R.H. Webb, and E.D. Andrews. 1997. Experimental Flooding in the Grand Canyon. Scientific American 276: 82-89.

Collins, B. 2000. Mid-19th Century Stream Channels and Wetlands Interpreted from Archival Sources for Three North Puget Sound Estuaries. LaConner, Washington: Skagit System Cooperative.

Collins, B., and T. Dunne. 1989. Gravel Transport, Gravel Harvesting, and Channel-Bed Degradation in Rivers Draining the Southern Olympic Mountains, Washington, USA. Environmental Geology and Water Science 13: 213-224.

Collins, B.D. 1997. Application of Geomorphology to Planning and Assessment of Riverine Gravel Removal in Washington. In Booth, D. B., Geology and Geomorphology of Stream Channels – Course Manual. University of Washington Center for Urban Water Resources Management, pp. IX-1-IX-46.

Collins, B.D., and D.R. Montgomery. 2002. Forest Development, Wood Jams, and Restoration of Floodplain Rivers in the Puget Lowland, Washington. Restoration Ecology 10: 237-247.

Collins, B.D., and T. Dunne. 1990. Fluvial Geomorphology and River-Gravel Mining- a Guide for Planners, Case Studies Included. California Division of Mine and Geology Special Publication 98.

Collins, B.D., D.R. Montgomery, and A.D. Haas. 2002. Historical Changes in the Distribution and Functions of Large Wood in Puget Lowland Rivers. Canadian Journal of Fisheries and Aquatic Sciences 59(1): 66-76.

Collins, B.D., D.R. Montgomery, and A.J. Sheikh. 2003. Reconstructing the Historical Riverine Landscape of the Puget Lowland. In Restoration of Puget Sound Rivers, edited by D.R. Montgomery, S. Bolton, D.B. Booth and L. Wall. Seattle: University of Washington Press. pp. 79-128.

Contor, D.R., and J.S. Griffith. 1995. Nocturnal Emergence of Juvenile Rainbow Trout from Winter Concealment Relative to Light Intensity. Hydrobiologia 299: 179-183.

Cook, M.A., K.M. Guthrie, M.B. Rust, and P.D. Plesha. 2005. Effects of Salinity and Temperature During Incubation on Hatching and Development of Lingcod Ophiodon Elongatus Girard, Embryos. Aquaculture Research 36(13): 1298-1303.

Cooper, A.B. 1990. Nitrate Depletion in the Riparian Zone and Stream Channel of a Small Headwater Catchment. Hydrobiologia 202(1-2): 13-26.

Cooper, A.C. 1965. The Effects of Transported Stream Sediments on the Survival of Sockeye and Pink Salmon Eggs and Alevins. International Pacific Salmon Fisheries Commission Bulletin 18. 71p.

Cooper, K., S. Boyd, J. Aldridge, and H. Rees. 2007. Cumulative Impacts of Aggregate Extraction on Seabed Macro-Invertebrate Communities in an Area Off the East Coast of the United Kingdom. Journal of Sea Research 57(4): 288-302.

Cooper, P.A. 1991. Leaching of Wood Preservatives from Treated Wood in Service. Prepared for Public Works Canada, Ottawa, Ontario. January. Cited in Stratus 2005a.

Coops, H., N. Geilen, H.J. Verheij, R. Boeters, and G. van der Velde. 1996. Interactions between waves, bank erosion and emergent vegetation: an experimental study in a wave tank. Aquatic Botany 53(3): 187-198.

Copeland, R. R., D. S. Biedenharn, and J. C. Fischenich. 2000. Channel-forming discharge. U.S. Army Corps of Engineers ERDC/CHL CHETN-VIII-5. 10pp.

Copeland, R.R., D.N. McComas, C.R. Thorne, P.J. Soar, M.M. Jonas, and J.B. Fripp. 2001. Hydraulic design of stream restoration projects. U.S. Army Corps of Engineers Engineer Research and Development Center: Coastal and Hydraulics Laboratory ERDC/CHL TR-01-28.

Copeland, R.R., D.S. Biedenharn, and J.C. Fischenich. 2000. Channel-forming discharge. U.S. Army Corps of Engineers ERDC/CHL CHETN-VIII-5. 10pp.

Copeland, Ronald R., Dinah N. McComas, Colin R. Thorne, Philip J. Soar, Meg M. Jonas, and Jon B. Fripp. 2001. Hydraulic Design of Stream Restoration Projects. U.S. Army Corps of Engineers Engineer Research and Development Center: Coastal and Hydraulics Laboratory ERDC/CHL TR-01-28.

Cordell, J.R. 1986. Structure and Dynamics of an Epibenthic Harpacticoid Assemblage and the Role of Predation by Juvenile Salmon. Seattle, Washington: University of Washington.

Cordone, A.J. and D.W. Kelly. 1961. The influence of inorganic sediment on the aquatic life of streams. California Fish and Game. 47:189-228.

Cordova, J.M., E.J. Rosi-Marshall, A.M. Yamamuro, and G.A. Lamberti. 2007. Quantity, Controls and Functions of Large Woody Debris in Midwestern USA Streams. River Research and Applications 23(1): 21-33.

COSEWIC (Committee on the Status of Endangered Wildlife in Canada). 2003. Rocky Mountain Ridge Mussel, Gonidea Angulata. Status Report. Ottawa, Ontario: Canadian Wildlife Service, Environment Canada.

Couch, D., and T.J. Hassler. 1990. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest) Olympia Oyster. PBS Record: 115470. U.S. Army Corps of Engineers and U.S. Fish and Wildlife Service.

Coughlin, D.J., and C.W. Hawryshyn. 1993. Ultraviolet Sensitivity in the Torus Semicurcularis of Juvenile Rainbow Trout (Oncorhynchus Mykiss). Vision Research 34: 1407-1413.

Counihan, T.D., A.I. Miller, M.G. Mesa, and M.J. Parsley. 1998. The Effects of Dissolved Gas Supersaturation on White Sturgeon Larvae. Transactions of the American Fisheries Society 127(2): 316-322.

Coutant, C.C. 2004. A Riparian Habitat Hypothesis for Successful Reproduction of White Sturgeon. Reviews in Fisheries Science 12(1): 23-73.

Cowan, L. 1991. Physical characteristics and intragravel survival of chum salmon in developed and natural groundwater channels in Washington. Pages 125-131 in Colt, J. and R. J. White, editors. Fisheries bioengineering symposium. American Fisheries Society Symposium 10.

Cowx, I.G. and R.L. Welcomme (eds), 1998. Rehabilitation of Rivers for Fish. Fishing News Books, Oxford. 160p. Cited in Petr 2000.

Cox, J., K. McDonald and T. Rigert. 1994. Engineering and Geotechnical Techniques for Shoreline Erosion Management in Puget Sound. Coastal Erosion Management Studies, Volume 4. Report 94-77. Shorelands and Coastal Zone Management Program, Washington Department of Ecology, Olympia, WA.

Cox, M., P.H. Rogers, A.N. Popper, and W.M. Saidel. 1987. Anatomical Effects of Intense Tone Stimulation in the Goldfish Ear: Dependence on Sound-Pressure Level and Frequency. Journal of the Acoustical Society of America 81(Suppl. 1): S7.

Crain, P.K., K. Whitener, and P.B. Moyle. 2004. Use of a Restored Central California Floodplain by Larvae of Native and Alien Fishes. American Fisheries Society Symposium 39: 125-140.

Cramer, M., K. Bates, D. Miller, K. Boyd, L. Fotherby, P. Skidmore, and T. Hoitsma. 2003. Integrated Streambank Protection Guidelines. Co-published by the Washington departments of Fish & Wildlife, Ecology, and Transportation. Olympia, Washington. 435 pp. http://www.wdfw.wa.gov/hab/ahg/strmbank.htm

Crecelius, E.A., D.L. Woodruff, and M.S. Meyers. 1989b. 1988 Reconnaissance Survey of Environmental Conditions in 13 Puget Sound Locations. Battelle Marine Science Laboratory.

Crecelius, E.A., T.J. Fortman, S.L. Kiesser, C.W. Apts, and O.A. Cotter. 1989a. Survey of Contaminants in Two Puget Sound Marinas. Battelle Marine Science Laboratory.

Creque, S.M., M.J. Raffenberg, W.A. Brofka, and J.M. Dettmers. 2006. If you build it, will they come? Fish and angler use at a freshwater artificial reef. North American Journal of Fisheries Management 26(3): 702-713.

Crocker, C.E., and J.J. Cech. 1997. Effects of Environmental Hypoxia on Oxygen Consumption Rate and Swimming Activity in Juvenile White Sturgeon, Acipenser transmontanus, in Relation to Temperature and Life Intervals. Environmental Biology of Fishes 50(4): 383-389.

Croft, A.R., and J.A. Adams. 1950. Landslides and sedimentation on the North Fork of Ogden River. USDA Forest Service Intermountain Forest & Range Experimental Station Research Paper INT-21, 4 p., illus.

Cuffey, K.M. 2002. Freshwater mussels in a California North Coast Range river: occurrence, distribution, and controls. University of California Water Resources Center Technical Completion Reports. http://repositories.cdlib.org/wrc/tcr/cuffey. (Accessed 2006.10.04).

Cummins, K.W. 1975. Macroinvertebrates. In River Ecology, edited by B.A. Whitton. Oxford, England: Blackwell Scientific Publications. pp. 170-198.

Curran, J.H., and E.E. Wohl. 2003. Large Woody Debris and Flow Resistance in Step-Pool Channels, Cascade Range, Washington. Geomorphology 51(1-3): 141-157.

Cushing, C.E., and J.D. Allan. 2001. Streams: Their Ecology and Life. San Diego, California: Academic Press.

Cyrus, D.P. and S.J.M. Blaber. 1987. The influence of turbidity on juvenile marine fishes in estuaries. Part 2. Laboratory studies, comparisons with field data and conclusions. Journal of Experimental Marine Biology and Ecology 109:71-91.

da Silva, J.F., and R.W. Duck. 2001. Historical Changes of Bottom Topography and Tidal Amplitude in the Ria De Aveiro, Portugal - Trends for Future Evolution. Climate Research 18(1-2): 17-24.

Dadswell, M.J. 1996. The Removal of Edwards Dam, Kennebec River, Maine: Its Effects on the Restoration of Anadromous Fishes. Draft environmental impact statement.

Dadswell, M.J., G.D. Melvin, and P.J. Williams. 1983. Effect of turbidity on the temporal and spatial utilization of the inner Bay of Fundy by American shad (Alosa sapidissima) (Pisces:Clupeidae) and its relationship to local fisheries. Canadian Journal of Fisheries and Aquatic Sciences 40(Supplement 1):330-332.

Dahl, T.E. 1990. Wetland Losses in the United States 1780's to 1980's. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service.

Dalbey, S.R., T.E. McMahon, and W. Fredenberg. 1996. Effect of Electrofishing Pulse Shape and Electrofishing-Induced Spinal Injury to Long-Term Growth and Survival of Wild Rainbow Trout. North American Journal of Fisheries Management 16: 560-569.

Dale, R.K., and D.C. Miller. 2007. Spatial and temporal patterns of salinity and temperature at an intertidal groundwater seep. Estuarine Coastal and Shelf Science 72(1-2): 283-298.

Danley, M.L., S.D. Mayr, P.S. Young, and J.J. Cech Jr. 2002. Swimming Performance and Physiological Stress Responses of Splittail Exposed to a Fish Screen. North American Journal of Fisheries Management 22(4): 1241-1249.

Dawes, C. J., R. C. Phillips, and G. Morrison. 2004. Seagrass communities of the gulf coast of Florida: status and ecology. Florida Fish and Wildlife Conservation Commission Fish and Wildlife Research Institute and the Tampa Bay Estuary Program. St. Petersburg, FL. Iv + 74 pp.

de Croux, P., M. Julieta, and A. Loteste. 2004. Lethal Effects of Elevated pH and Ammonia on Juveniles of Neotropical Fish Colosoma Macropomum (Pisces, Caracidae). Journal of Environmental Biology 25(1): 7-10.

Dean, R.G. 1986. Coastal Armoring: Effects, Principles, and Mitigation. Proceedings of Conference, American Society of Civil Engineers, pp. 1843-1857.

Dean, R.G., and R.A. Dalrymple. 2002. Coastal Processes with Engineering Applications. Cambridge, UK: Cambridge University Press.

deBruyn, A.M.H., D.J. Marcogliese, and J.B. Rasmussen. 2003. The Role of Sewage in a Large River Food Web. Canadian Journal of Fisheries and Aquatic Sciences 60(11): 1332-1344.

Demir, H., E.N. Otay, P.A. Work, and O.S. Borekci. 2004. Impacts of Dredging on Shoreline Change. Journal of Waterway Port Coastal and Ocean Engineering-ASCE 130(4): 170-178.

Dennison, W.C. 1987. Effects of Light on Seagrass Photosynthesis, Growth and Depth Distribution. Aquatic Botany 27: 15-26.

Dennison, W.C., R.J. Orth, K.A. Moore, J.C. Stevenson, V. Carter, S. Kollar, P.W. Bergstrom, and R.A. Batiuk. 1993. Assessing Water Quality with Submersed Aquatic Vegetation. Bioscience 43: 86-94.

Dernie, K.M., M.J. Kaiser, E.A. Richardson, and R.M. Warwick. 2002. Recovery of Soft Sediment Communities and Habitats Following Physical Disturbance. Journal of Experimental Marine Biology and Ecology 4069: 1-20.

Desbonnet, A., L.P. Pogue, D. Resiss, J. Boyd, J. Williams, and M. Imperial. 1995. Development of Coastal Vegetated Buffer Programs. Coastal Management 23: 91-109.

Desbonnet, A., P. Pogue, V. Lee, and N. Wolff. 1994. Vegetated Buffers in the Coastal Zone - a Summary Review and Bibliography. University of Rhode Island Graduate School of Oceanography.

Desjardin. 2003. Personal Communication as Cited in WSDOT 2006, Biological Assessment Preparation for Transportation Projects, Advanced Training Manual. Version 6. Olympia, Washington: Washington State Department of Transportation.

Dethier, M.N. 2006. Native Shellfish in Nearshore Ecosystems of Washington State. Technical Report 2006-04.

DeVries, P. 1997. Riverine salmonid egg burial depths: Review of published data and implications for scour studies. Canadian Journal of Fisheries and Aquatic Sciences 54:1685-1698.

Dewson, Z.S., A.B.W. James, and R.G. Death. 2007. Invertebrate Responses to Short-Term Water Abstraction in Small New Zealand Streams. Freshwater Biology 52(2): 357-369.

DFO (Department of Fisheries and Oceans Canada). 2007. Concrete Wash Water: Characteristics. Available at: http://www-heb.pac.dfo-mpo.gc.ca/water_quality/fish_and_pollution/conc_char_e.htm (accessed June 3, 2007).

Dias, J.M.A., and W.J. Neal. 1992. Sea Cliff Retreat in Southern Portugal - Profiles, Processes, and Problems. Journal of Coastal Research 8(3): 641-654.

Diaz, R.J., G.R. Cutter, and C.H. Hobbs. 2004. Potential Impacts of Sand Mining Offshore of Maryland and Delaware: Part 2 - Biological Considerations. Journal of Coastal Research 20(1): 61-69.

Dickerson, C., K.J. Reine, and D.G. Clarke. 2001. Characterization of Underwater Sounds Produced by Bucket Dredging Operations, DOER Technical Notes Collection (ERDC-TN-DOER-E14). Vicksburg, Mississippi: U.S. Army Engineer Research and Development Center.

Diederich, S. 2006. High Survival and Growth Rates of Introduced Pacific Oysters May Cause Restrictions on Habitat Use by Native Mussels in the Wadden Sea. Journal of Experimental Marine Biology and Ecology 328(2): 211-227.

Dietrich, W.E. and T. Dunne. 1993. The channel head. Pp. 175-219 in K. Beven and M.J. Kirkby (eds.), Channel Network Hydrology, J. Wiley and Sons.

Dietrich, W.E., J.W. Kirchner, H. Ikeda, and F. Iseya. 1989. Sediment Supply and Development of Coarse Surface Layer in Gravel Bedded Rivers. Nature 340(6230): 215-217.

Diez, J.R., S. Larranaga, A. Elosegi, and J. Pozo. 2000. Effect of Removal of Wood on Streambed Stability and Retention of Organic Matter. Journal of the North American Benthological Society 19(4): 621-632.

Dill, L.M., R.C. Ydenberg, and A.H.G. Fraser. 1981. Food Abundance and Territory Size in Juvenile Coho Salmon (Oncorhynchus kisutch). Canadian Journal of Zoology 59: 1801-1809.

Dobkowski, A.H. 1998. Dumptrucks Versus Dredges: An Economic Analysis of Sand Sources for Beach Nourishment. Coastal Management 26(4): 303-314.

Doeg, T.J., and J.D. Koehn. 1994. Effects of Draining and Desilting a Small Weir on Downstream Fish and Macroinvertebrates. Regulated Rivers-Research & Management 9(4): 263-277.

DOI (U.S. Department of the Interior). 1995. Final Environmental Impact Statement: Elwha River Ecosystem Restoration, Olympic National Park, Washington. Department of the Interior.

Dollar, E.S.J. 2000. Fluvial geomorphology. Progress in Physical Geography 24(3): 385-406.

Dolloff, C.A. 1986. Effects of Stream Cleaning on Juvenile Coho Salmon and Dolly Varden in Southeast Alaska. Transactions of the American Fisheries Society 115(5): 743-755.

Donahue, I and K. Irvine. 2003. Effects of sediment particle size composition on survivorship of benthic invertebrates from Lake Tanganyika, Africa. Archive fuer Hydrobiolgie. 157:131-144.

Dooley, K.M., C.F. Knopf, and R.P. Gambrell. 1999. Final Report: pH-Neutral Concrete for Attached Microalgae and Enhanced Carbon Dioxide Fixation - Phase I. Louisiana State University, Baton Rouge, Louisiana.

Dorn, P., L. Johnson, and C. Darby. 1979. The Swimming Performance of Nine Species of Common California Inshore Fishes. Transactions of the American Fisheries Society 108(4): 366-372.

Downing, J. 1983. The coast of Puget Sound: Its processes and development. Washington Sea Grant Program. University of Washington Press, Seattle, Washington. 126pp.

Downing, J.A. 1984. Assessment of Secondary Production: The First Step. In A Manual on Methods for the Assessment of Secondary Productivity in Fresh Waters, edited by J.A. Downing and F.H. Rigler. Oxford, England: Blackwell Scientific Publications.

Downing, J.A., Y. Rochon, M. Perusse, and H. Harvey. 1993. Spatial Aggregation, Body Size, and Reproductive Success in the Freshwater Mussel Elliptio Complanata. Journal of the North American Benthological Society 12: 148-156.

Dowty, P., B. Reeves, H. Berry, S. Wyllie-Echeverria, T. Mumford, A. Sewell, P. Milos, and R. Wright. 2005. Puget Sound Submerged Vegetation Monitoring Project 2003-2004 Monitoring Report. Olympia, Washington: Washington State Department of Natural Resources. Puget Sound Ambient Monitoring Program.

Doyle, M.W., E.H. Stanley, and J.M. Harbor. 2002. Geomorphic Analogies for Assessing Probable Channel Response to Dam Removal. Journal of the American Water Resources Association 38(6): 1567-1579.

Doyle, M.W., E.H. Stanley, and J.M. Harbor. 2003. Channel Adjustments Following Two Dam Removals in Wisconsin. Water Resources Research 39(1): 1-15.

Drinkwater, K.F., and K.T. Frank. 1994. Effects of River Regulation and Diversion on Marine Fish and Invertebrates. Aquatic Conservation-Marine and Freshwater Ecosystems 4(2): 135-151.

Duchrow, R.M., and W.H. Everhart. 1971. Turbidity measurement. Transactions of the American Fisheries Society 4: 682-690.

Dudley, S.J., J.C. Fischenich, and S.R. Abt. 1998. Effect of Woody Debris Entrapment on Flow Resistance. Journal of the American Water Resources Association 34(5): 1189-1197.

Duffy-Anderson, J. T., and K.W. Able. 1999. Effects of municipal piers on the growth of juvenile fishes in the Hudson River Estuary: a study across a pier edge. Marine Biology 133(3): 409-418. Cited in Nightingale and Simenstad 2001b.

Duke, J.R., J.D. White, P.M. Allen, and R.S. Muttiah. 2007. Riparian Influence on Hyporheic-Zone Formation Downstream of a Small Dam in the Blackland Prairie Region of Texas. Hydrological Processes 21(2): 141-150.

Dunn, J.R., and A.C. Matarese. 1987. A Review of Early Life History of Northeast Pacific Gadoid Fishes. Fisheries Research 5: 163-184. Cited in NRC 2001.

Dunne, T., and L.B. Leopold. 1978. Water in Environmental Planning. San Francisco: WH Freeman Co., pp. 590-594 and 693-695.

Dunne, T., W.E. Dietrich, N.F. Humphrey, and D.W. Tubbs. 1981. Geologic and Geomorphic Implications for Gravel Supply. Proceedings of the Conference on Salmon Spawning Gravel: A renewable resource in the Pacific Northwest? Pullman, Washington, pp. 75-100.

Durako, M.J., M.O. Hall, F. Sargent, and S. Peck. 1992. Propeller scars in seagrass beds: an assessment and experimental study of recolonization in Weedon Island State Preserve, Flordia. Pp. 42-53 in Webb, F.J., Jr. (ed), Proceedings of the Nineteenth Annual Conference of Wetlands Restoration and Creation, Hillsborough Community College, Tampa, FL. Cited in Dawes et al. 2004.

DWR (California Department of Water Resources). 2004. Merced River Gravel Augmentation Project: Monitoring Report. Sacramento, California: California Department of Water Resources, San Joaqauin District.

Dwyer, W.P. and R.G. White. 1997. Effect of Electroshock on Juvenile Arctic Grayling and Yellowstone Cutthroat Trout Growth 100 Days after Treatment. North American Journal of Fisheries Management 17:174-177

Dykaar, B.D., and P.J. Wigington. 2000. Floodplain Formation and Cottonwood Colonization Patterns on the Willamette River, Oregon, USA. Environmental Management 25: 87-104.

Ead, S.A., N. Rajaratnam, C. Katopodis, and F. Ade. 2000. Turbulent Open-Channel Flow in Circular Corrugated Culverts. Journal of Hydraulic Engineering 126(10): 750-757.

Ebersole, J.L., W.J. Liss, and C.A. Frissell. 2003. Cold Water Patches in Warm Streams: Physicochemical Characteristics and the Influence of Shading. Journal of the American Water Resources Association 39(2): 355-368.

Ebersole, J.L., W.J. Liss, and C.A. Frissell. 2003. Thermal Heterogeneity, Stream Channel Morphology, and Salmonid Abundance in Northeastern Oregon Streams. Canadian Journal of Fisheries and Aquatic Sciences 60(10): 1266-1280.

Eby, L.A., L.B. Crowder, C.M. McClellan, C.H. Peterson, and M.J. Powers. 2005. Habitat Degradation from Intermittent Hypoxia: Impacts on Demersal Fishes. Marine Ecology-Progress Series 291: 249-261.

Ecology (Washington State Department of Ecology). 1992. Focus--Wetlands in Washington State publication F-S-92-108. Olympia, Washington: Washington State Department of Ecology.

Ecology (Washington State Department of Ecology). 1992. Statewide Water Quality Assessment, 305(B) Report. Publication 92-04. Olympia, Washington: Washington State Department of Ecology.

Ecology (Washington State Department of Ecology). 1999. Working in the Water. Pub.#99-06. Available at http://www.ecy.wa.gov/biblio/9906.html (Accessed 2006.10.04).

Ecology (Washington State Department of Ecology). 2000. Alternative Mitigation Policy Guidance Interagency Implementation Agreement. Ecology Publication No. 02-06-007. February 2000.

Ecology (Washington State Department of Ecology). 2002. Evaluating Criteria for the Protection of Freshwater Aquatic Life in Washington's Surface Water Quality Standards, Dissolved Oxygen: Draft Discussion White Paper and Literature Summary. Publication Number 00-10-071. Olympia, Washington: Washington State Department of Ecology.

Ecology (Washington State Department of Ecology). 2004. Stormwater Management Manual for Eastern Washington. Publication No. 04-10-076.

Ecology (Washington State Department of Ecology). 2005. Stormwater Management Manual for Western Washington. Publication No. 05-10-30.

Ecology (Washington State Department of Ecology). 2005. Wetlands in Washington State - Volume 1: A Synthesis of the Science. Publication Number 05-06-006. Olympia, Washington: Washington State Department of Ecology.

Edinger, J.E., and V. S. Kolluru. 2000. Power Plant Intake Entrainment Analysis. Journal of Energy Engineering-ASCE 126 (1): 1-14.

Edwards, R.T. 1998. The Hyporheic Zone. In River Ecology and Management: Lessons from the Pacific Coastal Ecoregion, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag.

Eilers, H.P. III. 1975. Plants, plant communities, net production and tide levels: the ecological biogeography of the Nehalem salt marshes, Tillamook County, Oregon. Ph.D. thesis, Oregon State University, Corvallis. 368 pp. 7-47.

Eisler, R. 1998. Copper Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. Dept. of Interior, U.S. Geological Survey.

El-Asmar, H.M., and K. White. 2002. Changes in Coastal Sediment Transport Processes Due to Construction of New Damietta Harbour, Nile Delta, Egypt. Coastal Engineering 46(2): 127-138.

Elliott, S.T. 1986. Reduction of a Dolly Varden Population and Macrobenthos after Removal of Logging Debris. Transactions of the American Fisheries Society 115(3): 392-400.

Ellis, M. M. 1936. Erosion silt as a factor in aquatic environments. Ecology 17(1):29–42. Cited in Tucker and Theiling 1998.

Ellis, M.M. 1942. Fresh-Water Impoundments. Transactions of the American Fisheries Society, 71st Annual Meeting, pp. 80-93.

Embrey, S.S., and P.W. Moran. 2006. Quality of Streamwater in the Puget Sound Basin–a Decade of Study and Beyond [Poster]. Toxics in Puget Sound: Connecting Marine Environment to Human Health and the Economy, Puget Sound Action Team Forum, Seattle, Washington, April 5, 2006.

Emmett, R.L., S.L. Stone, S.A. Hinton, and M.E. Monaco. 1991. Distribution and abundance of fishes and invertebrates in west coast estuaries, Volume II: Species life history summaries. NOAA/NOS Strategic Environmental Assessments Division. Rockville, Maryland. ELMR Rept. No. 8. 329pp. Cited in NRC 2001.

Enger, P.S. 1981. Frequency Discrimination in Teleosts - Central or Peripheral? In Hearing and Sound Communication in Fishes, edited by W.N. Tavolga, A.N. Popper and R.R. Fay. New York: Springer-Verlag. pp. 243-255.

Ensign, S.H., and M.W. Doyle. 2005. In-Channel Transient Storage and Associated Nutrient Retention: Evidence from Experimental Manipulations. Limnology and Oceanography 50(6): 1740-1751.

Environment Canada. 1992. Interim Criteria for Quality Assessment of St. Lawrence River Sediment. St. Lawrence Action Plan. July. Cited in Stratus 2005b.

Erftemeijer, P.L.A., and R.R.R. Lewis. 2006. Environmental Impacts of Dredging on Seagrasses: A Review. Marine Pollution Bulletin 52(12): 1553-1572.

Erickson, D.L., J.A. North, J.E. Hightower, J. Weber, and L. Lauck. 2002. Movement and Habitat Use of Green Sturgeon Acipenser medirostris in the Rogue River, Oregon, USA. Journal of Applied Ichthyology 18(4-6): 565-569.

Eriksson, B.K., A. Sandstrom, M. Isaeus, H. Schreiber, and P. Karas. 2004. Effects of Boating Activities on Aquatic Vegetation in the Stockholm Archipelago, Baltic Sea. Estuarine Coastal and Shelf Science 61(2): 339-349.

Erskine, W.D., and A.A. Webb. 2003. Desnagging to Resnagging: New Directions in River Rehabilitation in Southeastern Australia. River Research and Applications 19(3): 233-249.

Eschmeyer, W.N., E.S. Herald, and H. Hammon. 1983. A field guide to Pacific Coast fishes of North America. Boston: Houghton Mifflin. 336pp.

Everest, F.H., R.L. Beschta, J.C. Schrivener, K.V. Koski, J.R. Sedell, and C.J. Cederholm. 1987. Chapter 4. Fine sediment and salmonid production: A paradox. P. 98-142 in E.O. Salo and T.W. Cundy (eds), Streamside Management: Forestry and Fishery Interactions. University of Washington, Institute of Forest Resources. Contribution No. 57.

Everett, P.O. 2006. Getting to Know Your Port. Available at: http://www.portofeverett.com/docs/52982_port_of_everett_facts_lo.pdf (accessed August 8, 2007).

Everett, R.A., and G.M. Ruiz. 1993. Coarse Woody Debris as a Refuge from Predation in Aquatic Communities. Oecologia 93(4): 475-486.

Everts, C.H. 1985. Effects of Small Protective Devises on Beach. California's Battered Coast: Proceedings of Conference on Coast Erosion, California Coastal Commission, pp. 127-137.

Fabi, G., S. Manoukian, and A. Spagnolo. 2006. Feeding Behavior of Three Common Fishes at an Artificial Reef in the Northern Adriatic Sea. Bulletin of Marine Science 78(1): 39-56.

Fausch, K.D., and T.G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences. 49:682-693.

Fausch, K.D., C. Gowan, A.D. Richmond, and S.C. Riley. 1995. The Role of Dispersal in Trout Population Response to Habitat Formed by Large Woody Debris in Colorado Mountain Streams. Bulletin Francais De La Peche Et De La Pisciculture (337-9): 179-190.

Faustini, J.M., and J.A. Jones. 2003. Influence of Large Woody Debris on Channel Morphology and Dynamics in Steep, Boulder-Rich Mountain Streams, Western Cascades, Oregon. Geomorphology 51(1-3): 187-205.

Fay, R.R. 1988. Peripheral Adaptations for Spatial Hearing in Fish. In Sensory Biology of Aquatic Animals, New York: Springer-Verlag. pp. 711-731.

Fay, R.R. and A.N. Popper (eds.). 1999. Comparative hearing: fish and amphibians. New York: Springer-Verlag.

Fayram, A.H. 1996. Impacts of largemouth bass (Micropterus salmoides) and smallmouth bass (Micropterus dolomieui) predation on populations of juvenile salmonids in Lake Washington. M.S. thesis,. University of Washington, School of Fisheries, Seattle.

Fayram, A.H. and T.H. Sibley. 2000. Impact of predation by smallmouth bass on sockeye salmon in Lake Washington. North American Journal of Fisheries Management 20: 81–89. Cited in Carrasquero 2001.

Feist, B.E. 1991. Potential impacts of pile driving on juvenile pink (Oncorhynchus gorbuscha) and chum (O. keta) salmon behavior and distribution. M.S. Thesis, University of Washington School of Fisheries.

Feist, B.E., J. Anderson, and R. Miyamoto. 1992. Potential impacts of pile driving on juvenile pink (Onchorhynchus gorbuscha) and chum (O. keta) salmon behavior and distribution. Pound Sounds Final Report. University of Washington, Seattle, WA.

Feist, G.W., M.A.H. Webb, D.T. Gundersen, E.P. Foster, C.B. Schreck, A.G. Maule, and M.S. Fitzpatrick. 2005. Evidence of Detrimental Effects of Environmental Contaminants on Growth and Reproductive Physiology of White Sturgeon in Impounded Areas of the Columbia River. Environmental Health Perspectives 113(12): 1675-1682.

Feller, M.C. 2005. Forest Harvesting and Streamwater Inorganic Chemistry in Western North America: A Review. Journal of the American Water Resources Association 41(4): 785-811.

FERC (Federal Energy Regulatory Commission). 2005. Final Environmental Impact Statement Volume I Capacity Replacement Project. Docket Nos. CP05-32-000, -001 FERC/EIS – 0178. Chapter 4 only. http://www.ecy.wa.gov/programs/sea/nw_capacity_replacement/, accessed 2006.10.04.

Ferguson, J., J. Williams, and Ed Meyer. 2002. Recommendations For Improving Fish Passage at the Stornorrfors Power Station on the Umealven, Umea Sweden, Report submitted to the Vindel River Fishery Advisory Board Umea, Sweden.

Fernald, A.G., D.H. Landers, and P.J. Wigington. 2006. Water Quality Changes in Hyporheic Flow Paths between a Large Gravel Bed River and Off-Channel Alcoves in Oregon, USA. River Research and Applications 22(10): 1111-1124.

Ferraro, S.P., and F.A. Cole. 2007. Benthic Macrofauna-Habitat Associations in Willapa Bay, Washington, USA. Estuarine Coastal and Shelf Science 71(3-4): 491-507.

Fetherston, K.L., R.J. Naiman, and R.E. Bilby. 1995. Large Woody Debris, Physical Process, and Riparian Forest Development in Montana River Networks of the Pacific-Northwest. Geomorphology 13(1-4): 133-144.

Feyrer, F., T. Sommer, and W. Harrell. 2006. Importance of Flood Dynamics Versus Intrinsic Physical Habitat in Structuring Fish Communities: Evidence from Two Adjacent Engineered Floodplains on the Sacramento River, California. North American Journal of Fisheries Management 26(2): 408-417.

Feyrer, F., T. Sommer, and W. Harrell. 2006. Managing Floodplain Inundation for Native Fish: Production Dynamics of Age-0 Splittail (Pogonichthys Macrolepidotus) in California's Yolo Bypass. Hydrobiologia 573: 213-226.

Fields, P.E. 1966. Final report on migrant salmon light guiding studies (Contract No. D.A.-45-108 CIVENG-63-29) at Columbia River Dams. University of Washington. College of Fisheries. Report for the Fisheries Engineering Research Program. U.S. Army Engineer Division, North Pacific Corps of Engineers, Portland, OR. Cited in Simenstad et al. 1999, and in Nightingale and Simenstad 2001b.

Fields, P.E., and G.L. Finger. 1954. The Reaction of Five Species of Young Pacific Salmon and Steelhead Trout to Light. Seattle, Washington: University of Washington, School of Fisheries.

Finlayson, D. 2006. The Geomorphology of Puget Sound Beaches. Ph.D. Thesis, University of Washington, Seattle, 216 pp.

Finstad, A.G., S. Einum, T. Forseth, and O. Ugedal. 2007. Shelter Availability Affects Behaviour, Size-Dependent and Mean Growth of Juvenile Atlantic Salmon. Freshwater Biology 52: 1710-1718.

Fischenich, J.C. 2001. Impacts of stabilization measures. EMRRP Technical notes Collection (ERDC TN-EMRRP-SR-32), U.S. Army Engineer Research and Development Center, Vicksburg, MS. www.wes.army.mil/el/emrrp

Fischenich, J.C. 2003. Effects of riprap on riverine and riparian ecosystems. ERDC/EL TR-03-4, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Fischenich, J.C. and H. Allen. 2000. Stream Management. Prepared for U.S. Army Engineer District, Ft. Worth, TX. Prepared by Water Operations Technical Support Program, Vicksburg, MS. (ERDC/EL SR-W-00-1). 295 pp.

Fischenich, J.C. and J. Morrow. 1999. Streambank Habitat Enhancement with Large Woody Debris. EMRRP Technical Notes Collection (ERDC TN-EMRRP-SR-13). U.S. Army Engineer Research and Development Center, Vicksburg, MS. Available at: www.wes.army.mil/el/emrrp.

Fischer, H.B., List, E. J., Koh, R. C. Y., Imberger, J., Brooks, N. H. 1979. Mixing in Inland and Coastal Waters. San Diego, California: Academic Press.

Fisheries and Oceans Canada. 2006. High-pressure directional drilling. http://www.dfo-mpo.gc.ca/regions/central/habitat/os-eo/prov-terr/sk/index_e.htm. Accessed 2006.12.06.

FishXing. 2007. Fishxing: Fish Passage Case Studies. USDA Forest Service. Available at: http://www.stream.fs.fed.us/fishxing.html (accessed October, 19, 2007).

Fitzsimons, J.D. 1996. The significance of man-made structures for lake trout spawning in the Great Lakes: Are they a viable alternative to natural reefs? Canadian Journal of Fisheries and Aquatic Sciences 53: 142-151.

Flebbe, P.A. 1999. Trout Use of Woody Debris and Habitat in Wine Spring Creek, North Carolina. Forest Ecology and Management 114: 367-376.

Flett, P.A., K.R. Munkittrick, G. VanDerKraak, and J.F. Leatherland. 1996. Overripening as the Cause of Low Survival to Hatch in Lake Erie Coho Salmon (Oncorhynchus Kisutch) Embryos. Canadian Journal of Zoology-Revue Canadienne De Zoologie 74(5): 851-857.

Flores-Cervantes, J.H., E. Istanbulluoglu, and R.L. Bras. 2006. Development of Gullies on the Landscape: A Model of Headcut Retreat Resulting from Plunge Pool Erosion. Journal of Geophysical Research-Earth Surface 111(F1).

Florsheim, J.L., and J.F. Mount. 2002. Restoration of Floodplain Topography by Sand-Splay Complex Formation in Response to Intentional Levee Breaches, Lower Cosumnes River, California. Geomorphology 44(1-2): 67-94.

Florsheim, J.L., and J.F. Mount. 2003. Changes in Lowland Floodplain Sedimentation Processes: Pre-Disturbance to Post-Rehabilitation, Cosumnes River, California. Geomorphology 56(3-4): 305-323.

Folmar, L. C., and W. W. Dickhoff. 1981. Evaluation of some physiological parameters as predictive indices of smoltification. Aquaculture 23: 309-24.

Fonseca, M.S., and J.A. Cahalan. 1992. A Preliminary Evaluation of Wave Attenuation by 4 Species of Seagrass. Estuarine Coastal and Shelf Science 35(6): 565-576.

Fonseca, M.S., and S.S. Bell. 1998. Influence of Physical Setting on Seagrass Landscapes near Beaufort, North Carolina, USA. Marine Ecology-Progress Series 171: 109-121.

Fonseca, M.S., J.C. Zieman, G.W. Thayer, and J.S. Fisher. 1983. The Role of Current Velocity in Structuring Eelgrass (Zostera-Marina L) Meadows. Estuarine Coastal and Shelf Science 17(4): 367-380.

Fonseca, M.S., W.J. Kenworthy, and G.W. Thayer. 1998. Guidelines for the Conservation and Restoration of Seagrasses in the United States and Adjacent Waters. Silver Spring, Maryland: National Oceanic and Atmospheric Administration.

Forrester, G.E., and M.A. Steele. 2004. Predators, Prey Refuges, and the Spatial Scaling of Density-Dependent Prey Mortality. Ecology 85(5): 1332-1342.

Francis, R.A. 2006. Allogenic and Autogenic Influences Upon Riparian Vegetation Dynamics. Area 38(4): 453-464.

Francis, T.B., and D.E. Schindler. 2006. Degradation of littoral habitats by residential development: Woody debris in lakes of the Pacific Northwest and Midwest, United States. Ambio 35(6): 274-280.

Francisco, M.D. 1995. Propeller Scour and Sediment Remediation at the Seattle Waterfront. Master’s Thesis, University of Washington, Seattle, Washington.

Franco, A., P. Franzoi, S. Malavasi, F. Riccato, P. Torricelli, and D. Mainardi. 2006. Use of Shallow Water Habitats by Fish Assemblages in a Mediterranean Coastal Lagoon. Estuarine Coastal and Shelf Science 66(1-2): 67-83.

Fredenberg, W.A. 1992. Evaluation of Electrofishing-Induced Spinal Injuries Resulting from Field Electrofishing Surveys in Montana. Montana Department of Fish, Wildlife and Parks, Helena. Helena, Montana: Montana Department of Fish, Wildlife and Parks.

Fredette, T.J., J.E. Clausner, M.R. Palermo, S.M. Bratos, and T.L. Prickett. 2002. Field Pilot Study of in Situ Capping of Palos Verdes Shelf Contaminated Sediments. ERDC TR-02-5. Vicksburg, Mississippi: U.S. Army Engineer Research and Development Center.

Freeman, G.E. and J.C. Fischenich. 2000. Gabions for streambank erosion control. EMRRP Technical Notes Collection (ERDC TN-EMRRP-SR-22), U.S. Army Engineer Research and Development Center, Vicksburg, MS. Available at: www.wes.army.mil/el/emrrp.

Freeman, P.L., and M.S. Schorr. 2004. Influence of Watershed Urbanization on Fine Sediment and Macroinvertebrate Assemblage Characteristics in Tennessee Ridge and Valley Streams. Journal of Freshwater Ecology 19(3): 353-362.

French, J.R., and D.R. Stoddart. 1992. Hydrodynamics of Salt-Marsh Creek Systems - Implications for Marsh Morphological Development and Material Exchange. Earth Surface Processes and Landforms 17: 235-252.

Fresh, K. 1998. Taking stock: anadromous salmonids and their habitats in the Puget Sound basin. In: Salmon in the City: Can Habitat in the Path of Development be Saved? Proceedings of a conference by American Public Works Association, Stormwater Managers Committee in Mount Vernon, Washington. Washington State University. Available at: http://depts.washington.edu/cwws/Research/Reports/salmoninthecity.pdf (Accessed 2006.10.04).

Fresh, K. 2000. Personal communication (review comments forwarded by the state salmon recovery program steering committee to José Carrasquero, Herrera Environmental Consultants, regarding the draft white paper on over-water structures [freshwaters]). Washington Department of Fish and Wildlife, Olympia. November 2000.

Fresh, K. 2006. Juvenile Pacific Salmon and the Nearshore Ecosystem of Puget Sound. Seattle: Puget Sound Nearshore Ecosystem Restoration Program.

Fresh, K. and D. Averill. 2005. Salmon in the Nearshore and Marine Waters of Puget Sound. Draft February 2005. Submitted as part of the Regional Nearshore and Marine Aspects of Salmon Recovery in Puget Sound that was delivered to Shared Strategy for inclusion in their regional salmon recovery plan. June 28, 2005.

Fresh, K. and R. Cardwell. 1978. Predation upon juvenile salmon. As cited in Nightingale and Simenstad 2001b.(Cardwell, R.D. and K.L. Fresh. 1979. Predation upon Juvenile Salmon. State of Washington Department of Fisheries Progress Report Draft No. 8.?)

Fresh, K., B. Williams, and D. Penttila. 1995. Overwater structures and impacts on eelgrass in Puget Sound, WA. Puget Sound Research '95 Proceedings. Seattle, WA: Puget Sound Water Quality Authority. Cited in Nightingale and Simenstad 2001b.

Fresh, K., B.W. Williams, S. Wyllie-Echeverria, and T. Wyllie-Echeverria. 2000. Mitigating impacts of overwater floats on eelgrass Zostera marina in Puget Sound, Washington using light permeable deck grating, Draft. Cited in Nightingale and Simenstad 2001b.

Frest, T.J., and E.J. Johannes. 1995. Freshwater Molluscs of the Upper Sacramento System, California with Particular Reference to the Cantara Spill. Sacramento, California: State of California, Department of Fish and Game.

Frest, T.J., and E.J. Johannes. 1995. Interior Columbia Basin Mollusk Species of Special Concern. Final report to the Interior Columbia Basin Ecosystem Management Project. Walla Walla, Washington. Cited in WDNR (2006b).

Fries, L.T., and D.E. Bowles. 2002. Water Quality and Macroinvertebrate Community Structure Associated with a Sportfish Hatchery Outfall. North American Journal of Aquaculture 64(4): 257-266.

Frihy, O.E., and P.D. Komar. 1993. Long-Term Shoreline Changes and the Concentration of Heavy Minerals in Beach Sands of the Nile-Delta, Egypt. Marine Geology 115(3-4): 253-261.

Frisch, A.J., and T.A. Anderson. 2000. The Response of Coral Trout (Plectropomus Leopardus) to Capture, Handling and Transport and Shallow Water Stress. Fish Physiology and Biochemistry 23(1): 23-24.

Frissell, C., and R. Nawa. 1992. Incidence and Causes of Physical Failure of Artificial Habitat Structures in Streams of Western Oregon and Washington. North American Journal of Fisheries Management 12(1): 182-197.

Fritioff, A., and M. Greger. 2003. Aquatic and Terrestrial Plant Species with Potential to Remove Heavy Metals from Stormwater. International Journal of Phytoremediation 5(3): 211-224.

Fuchs, S.A., S.G. Hinch, and E. Mellina. 2003. Effects of Streamside Logging on Stream Macroinvertebrate Communities and Habitat in the Sub-Boreal Forests of British Columbia, Canada. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 33(8): 1408-1415.

Fuchsman, P.C., T.R. Barber, J.C. Lawton, and K.B. Leigh. 2006. An Evaluation of Cause-Effect Relationships between Polychlorinated Biphenyl Concentrations and Sediment Toxicity to Benthic Invertebrates. Environmental Toxicology and Chemistry 25(10): 2601-2612.

Fuller, D.D. 1990. Seasonal Utilization of Instream Boulder Structures by Anadromous Salmonids in Hurdygurdy Creek, California. FHR Currents 3: 1-8.

Furniss, M.J., T.D. Roelofs, and C.S. Yee. 1991. Road Construction and Maintenance. Pp. 297-324 in W.R. Meehan (ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. Bethesda, Maryland: American Fisheries Society Special Publication 19.

Gabriel, A.O. and T.A. Terich. 2005. Cumulative Patterns and Controls of Seawall Construction, Thurston County, Washington. Journal of Coastal Research 21(3): 430-440.

Gadomski, D.M., and M.J. Parsley. 2005. Effects of Turbidity, Light Level, and Cover on Predation of White Sturgeon Larvae by Prickly Sculpins. Transactions of the American Fisheries Society 134(2): 369-374.

Gagnaire, B., H. Frouin, K. Moreau, H. Thomas-Guyon, and T. Renault. 2006. Effects of Temperature and Salinity on Haemocyte Activities of the Pacific Oyster, Crassostrea gigas (Thunberg). Fish & Shellfish Immunology 20(4): 536-547.

Galay, V.J. 1983. Causes of River Bed Degradation. Water Resources Research 19: 1057-1090.

Gale, S.B., and A.V. Zale. 2005. Evaluation of Entrainment Losses of Westslope Cutthroat Trout at Private Irrigation Diversions and the Efficiency of Fish Screens on Skalkaho Creek, Montana. Prepared for Wild Fish Habitat Initiative by Montana Cooperative Fishery Research Unit, U.S. Geological Survey, Department of Ecology, Montana State University, Bozeman, Montana.

Gallardo, A.H., and A. Marui. 2006. Submarine groundwater discharge: an outlook of recent advances and current knowledge. Geo-Marine Letters. Vol. 26. No. 2 June 2006: 102-113.

Galster, R.W. and M.L. Schwartz. 1990. Ediz Hook - A case history of coastal erosion and rehabilitation. Journal of Coastal Research Special Issue, 6: 103-113.

Gandy, C.J., J.W.N. Smith, and A.P. Jarvis. 2007. Attenuation of Mining-Derived Pollutants in the Hyporheic Zone: A Review. Science of the Total Environment 373(2-3): 435-446.

Garbaciak, S., P. Spadaro, T. Thornburg, and R. Fox. 1998. Sequential Risk Mitigation and the Role of Natural Recovery in Contaminated Sediment Projects. Water Science and Technology 37(6-7): 331-336.

Gardiner, W., G. Pascoe, L. Muench, B. Gregg, and D. Shreffler. 2006. An evaluation of the potential risks associated with creosoted pilings and their removal. Abstract. SETAC Annual Meeting, Baltimore, MD.

Gardner, F. (ed.). 1981. Washington Coastal Areas of Major Biological Significance. Washington State Department of Ecology, Baseline Studies Program. Olympia, Washington. Cited in WDNR 2006b.

Gardner, S.C., C.E. Grue, W.W. Major, and L.L. Conquest. 2001. Aquatic invertebrate communities associated with purple loosestrife (Lythrum salicaria), cattail (Typha latifolia), and bulrush (Scirpus acutus) in central Washington, USA. Wetlands 21(4): 593-601.

Garland, R.D., and K.F. Tiffan. 1999. Nearshore habitat use by subyearling fall chinook salmon in the Snake River. Pages 53-72 in K.F. Tiffan, D.W. Rondorf, W.P. Connor, and H.L. Burge, editors. Post-release attributes and survival of hatchery and natural fall chinook salmon in the Snake River. Annual Report to the Bonneville Power Administration, contract DE-AI79-91BP21708, Portland, Oregon.

Garland, R.D., K.F. Tiffan, D.W. Rondorf, and L.O. Clark. 2002. Comparison of Subyearling Fall Chinook Salmon's Use of Riprap, Revetments and Unaltered Habitats in Lake Wallula of the Columbia River. North American Journal of Fisheries Management 22(4): 1283-1289.

Garland, R.D., K.F. Tiffan, D.W. Rondorf, D. Anglin, and J. Skalicky. 2002. Assessment of chum and fall Chinook salmon spawning habitat near Ives and Pierce Islands in the Columbia River. Draft Research Report 1999-2001 to the Bonneville Power Administration , contract 04701, Portland, OR.

Garrad, P.N., and R.D. Hey. 1988. River management to reduce turbidity in navigable broadland rivers. Journal of Environmental Management 27(3): 273-288.

Garrison, K.J., and B.S. Miller. 1982. Review of the early life history of Puget Sound fishes. University of Washington Fisheries Research Institute, UW-8216. Seattle, Washington. 729pp.

Garrison, P.J., D.W. Marshall, L. Stremick-Thompson, P.L. Cicero, and P.D. Dearlove. 2005. Effects of Pier Shading on Littoral Zone Habitat and Communities in Lakes Ripley and Rock, Jefferson County, Wisconsin. Wisconsin Department of Natural Resources, Jefferson County Land and Water Conservation Department, and Lake Rioley Management District.

Garza, E.L., and R.L. Whitman. 2004. The Nearshore Benthic Invertebrate Community of Southern Lake Michigan and Its Response to Beach Nourishment. Journal of Great Lakes Research 30(1): 114-122.

Gaspin, J.B. 1975. Experimental investigations of the effects of underwater explosions on swimbladder fish, 1: 1973 Chesapeake Bay Tests. Naval Surface Weapons Center Silver Springs Maryland 20910. NSWC/WOL/TR 75-58.

Gatto, L.W., and W.W. Doe III. 1987. Bank Conditions and Erosion Along Selected Reservoirs. Environmental Geology and Water Sciences 9(3): 143-154.

Gay, P., G. Vellidis, and J.J. Delfino. 2006. The Attenuation of Atrazine and Its Major Degradation Products in a Restored Riparian Buffer. Transactions of the ASABE 49(5): 1323-1339.

Gayaldo, P.F. and K. Nelson. 2006. Preliminary Results of Light Transmission under Residential Piers in Lake Washington, King County, Washington: A Comparison between Prisms and Grating. Lake and Reservoir Management: Volume 22(3): 245 – 249.

Geist, D.R. 2000a. Hyporheic Discharge of River Water into Fall Chinook Salmon (Oncorhynchus Tshawytscha) Spawning Areas in the Hanford Reach, Columbia River. Canadian Journal of Fisheries and Aquatic Science 57: 1647-1656.

Geist, D.R. 2000b. The interaction of ground water and surface water within fall Chinook salmon spawning areas in the Hanford Reach of the Columbia River. Pp. 95-98 in Proceedings of the Ground-Water/Surface-Water Interactions Workshop, EPA/542/R-00/007.

Geist, D.R., and D.D. Dauble. 1998. Redd Site Selection and Spawning Habitat Use by Fall Chinook Salmon: The Importance of Geomorphic Features in Large Rivers. Environmental Management 22(5): 655-669.

Geist, D.R., R.S. Brown, V. Cullinan, S.R. Brink, K. Lepla, P. Bates, and J.A. Chandler. 2005. Movement, Swimming Speed, and Oxygen Consumption of Juvenile White Sturgeon in Response to Changing Flow, Water Temperature, and Light Level in the Snake River, Idaho. Transactions of the American Fisheries Society 134(4): 803-816.

Geist, D.R., T.P. Hanrahan, E.V. Arntzen, G.A. McMichael, C.J. Murray, and Y.J. Chien. 2002. Physicochemical Characteristics of the Hyporheic Zone Affect Redd Site Selection by Chum Salmon and Fall Chinook Salmon in the Columbia River. North American Journal of Fisheries Management 22(4): 1077-1085.

Gelfenbaum, G., T. Mumford, J. Brennan, H. Case, M. Dethier, K. Fresh, F. Goetz, M. van Heeswijk, T.M. Leschine, M. Logsdon, D. Myers, J. Newton, H. Shipman, C.A. Simenstad, C. Tanner, and D. Woodson. 2006. Coastal Habitats in Puget Sound: A Research Plan in Support of the Puget Sound Nearshore Partnership. Available at: http://www.pugetsoundnearshore.org/techical_papers/coastal_habitats.pdf.

Gendron, A. 2005. Water Table Dynamics at Lowell Point, Camano Island, Washington: A Designer Study for Puget Sound. University of Washington, Seattle, 30 pp.

Gerstel, W. J. and J. F. Brown. 2006. Alternative Shoreline Stabilization Evaluation Project; Final Report September 15, 2006. Prepared for: Puget Sound Action Team. Olympia, WA.

Giannico, G., and J.A. Souder. 2005. Tide Gates in the Pacific Northwest: Operation, Types and Environmental Effects. ORESU-T-05-001. Corvallis, Oregon: Oregon Sea Grant.

Giannico, G.R., and J.A. Souder, 2004. The Effects of Tide Gates on Estuarine Habitats and Migratory Fish. Oregon Sea Grant. National Oceanic and Atmospheric Administration grant number NA76RG0467, project number R/HBT-07-PD, pp. 10.

Gilbert, G.K. 1917. Hydraulic Mining Debris in the Sierra Nevada. Professional Paper 105. Washington, DC: U.S. Geological Survey.

Gillilan, D.M., and T.C. Brown. 1997. Instream Flow Protection: Seeking a Balance in Western Water Use. Washington, DC: Island Press.

Ginetz, R.M., and P.A. Larkin. 1976. Factors affecting rainbow trout (Salmo gairdneri) predation on migrant fry of sockeye salmon (Oncorhynchus nerka). Journal of the Fisheries Research Board of Canada 33: 19–24. Cited in Tabor et al., 1998.

Giorgi, A.E. 1981. The environmental biology of the embryos, egg masses and nesting sites of the lingcod, Ophiodon elongatus. NMFS, NWAFC Proceedings Report No. 81- 06. Seattle, Washington. 107pp. Cited in NRC 2001.

Gippel, C.J., B.L. Finlayson, and I.C. O'Neill. 1996. Distribution and hydraulic significance of large woody debris in a lowland Australian river. Hydrobiologia 318, 179-194.

Gipson, R.D., and W.A. Hubert. 1993. Spawning-site selection by kokanee along the shoreline of Flaming Gorge Reservoir, Wyoming-Utah. North American Journal of Fisheries Management 13(3): 475-482.

Gjovik, L. R. 1977. Pretreatment molding of southern pine: Its effect on the permanence and performance of preservatives exposed in sea water. In Proceedings of the American Wood Preservers' Association. 73:142-153. Cited in Cooper 1991.

Glasby, T.M. 1999. Effects of Shading on Subtidal Epibiotic Assemblages. Journal of Experimental Marine Biology and Ecology 234: 275-290.

Gliwicz, A.M. 1986. A Lunar Cycle in Zooplankton. Ecology 67(4): 883-897.

Godin, J.-G. J. 1982. Migrations of salmonid fishes during early life history phases: daily and annual timing. Pp. 22-50 in E.L. Brannon and E.O. Salo (eds.), Proceedings of the First International Salmon Trout Migratory Behavior Symposium University of Washington. Cited in Nightingale and Simenstad 2001b.

Goetz, F.A., E. Jeanes, E. Beamer, G. Hart, C. Morello, M. Camby, C. Ebel, E. Conner, and H. Berge. 2004. Bull Trout in the Nearshore (Preliminary Draft). U.S. Army Corps of Engineers, Seattle District.

Golder Associates, Inc. 2005. Upper Columbia River Juvenile White Sturgeon Monitoring: Phase 2 Investigations, Fall 2003–Spring 2004. Prepared for BC Hydro by Golder Associates, Inc., Castlegar, BC, Canada.

Gomi, T., R.C. Sidle, M.D. Bryant, and R.D. Woodsmith. 2001. The Characteristics of Woody Debris and Sediment Distribution in Headwater Streams, Southeastern Alaska. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 31(8): 1386-1399.

Gonor, J., J. Sedell, and P. Benner. 1988. What we know about large trees in estuaries, in the sea,and on coastal beaches. Pages 83-112 in: C. Maser, R. Tarrant, J. Trappe, and J. Franklin, editors. From the forest to the sea: a story of fallen trees. U.S.D.A, Forest Service, Pacific Northwest Research Station, General Technical Report PNW-GTR-229, Portland, OR.

Goodyear, C.P. 1977. Assessing the Impact of Power Plant Mortality on the Compensatory Reserve of Fish Populations with Demersal Eggs. In Assessing the Impact of Power Plant Induced Mortality on Fish Populations, edited by J.H.S. Blaxter. New York: Pergammon Press.

Gore, J.A. 1985. Mechanisms of colonozation and habitat enhancement for benthic macroinvertebrates in restored river channels. Pages 81-101 in The Restoration of Rivers and Streams, J.A. Gore (ed). Boston, Massachusetts: Butterworths.

Gosset, C., J. Rives, and J. Labonne. 2006. Effect of Habitat Fragmentation on Spawning Migration of Brown Trout (Salmo Trutta L.). Ecology of Freshwater Fish 15(3): 247-254.

Gottesfeld, A.S., M.A. Hassan, J.F. Tunnicliffe, and R.W. Poirier. 2004. Sediment Dispersion in Salmon Spawning Streams: The Influence of Floods and Salmon Redd Construction. Journal of the American Water Resources Association (JAWRA) 40(4): 1071-1086.

Gotthardt, T. 2006. Longfin Smelt. Anchorage, Alaska: Alaska Natural Heritage Program.

Goudreau, S.E., R.J. Neves, and R.J. Sheehan. 1993. Effects of Waste-Water Treatment-Plant Effluents on Fresh-Water Mollusks in the Upper Clinch River, Virginia, USA. Hydrobiologia 252(3): 211-230.

Govindjee, R. 1975. Introduction to Photosynthesis. In Bioenergetics of Photosynthesis, edited by R. Govindjee. New York, New York: Academic Press. pp. 1-50.

Gowan, C., M.K. Young, K.D. Fausch, and S.C. Riley. 1994. Restricted Movement in Resident Stream Salmonids: A Paradigm Lost. Canadian Journal of Fisheries & Aquatic Sciences 51: 2626-2637.

Gran, K., and C. Paola. 2001. Riparian Vegetation Controls on Braided Stream Dynamics. Water Resources Research 37(12): 3275-3283.

Grant, G.E., B. Burkholder, A. Jefferson, S. Lewis, and R. Haggerty. 2006. Hyporheic Flow, Temperature Anomalies, and Gravel Augmentation: Preliminary Findings of a Field Investigation on the Clackamas River, Oregon. Prepared for Portland General Electric, by USDA Forest Service and Department of Geosciences, Oregon State University, Corvallis, Oregon. December 6, 2006.

Grant, J., and B. Thorpe. 1991. Effects of Suspended Sediment on Growth, Respiration, and Excretion of the Soft-Shell Clam (Mya-Arenaria). Canadian Journal of Fisheries and Aquatic Sciences 48(7): 1285-1292.

Grant, J., C.T. Enright, and A. Griswold. 1990. Resuspension and Growth of Ostrea-Edulis - a Field Experiment. Marine Biology 104(1): 51-59.

Grapentine, L., Q. Rochfort, and J. Marsalek. 2004. Benthic Responses to Wet-Weather Discharges in Urban Streams in Southern Ontario. Water Quality Research Journal of Canada 39(4): 374-391.

Gray, A., C.A. Simenstad, D.L. Bottom, and T.J. Cornwell. 2002. Contrasting Functional Performance of Juvenile Salmon Habitat in Recovering Wetlands of the Salmon River Estuary, Oregon, USA. Restoration Ecology 10(3): 514-526.

Gray, G.A., and D.W. Rondorf. 1986. Predation on juvenile salmonids in Columbia basin reservoirs. Pp. 178–185 in G.E. Hall and M.J. Van Den Avyle (eds.), Reservoir Fisheries Management: Strategies for the 80's. Bethesda, Maryland: American Fisheries Society. Cited in Carrasquero 2001.

Gray, G.A., G.M. Sonnevil, H.C. Hansel, C.W. Huntington, and D.E. Palmer. 1984. Feeding activity, rate of consumption, daily ration and prey selection of major predators in the John Day pool. U.S. Fish and Wildlife Service, Annual report (Contract DI-AI79-82BP34796). Cook, Washington. Cited in Carrasquero 2001.

Gray, J.S., R.S.S. Wu, and Y.Y. Or. 2002. Effects of Hypoxia and Organic Enrichment on the Coastal Marine Environment. Marine Ecology-Progress Series 238: 249-279.

Greacen, E.L., and R. Sands. 1980. Compaction of Forest Soils - a Review. Australian Journal of Soil Research 18: 163-189.

Greathouse, E.A., C.M. Pringle, W.H. McDowell, and J.G. Holmquist. 2006. Indirect Upstream Effects of Dams: Consequences of Migratory Consumer Extirpation in Puerto Rico. Ecological Applications 16(1): 339-352.

Greene, C.M., and T.J. Beechie. 2004. Consequences of Potential Density-Dependent Mechanisms on Recovery of Ocean-Type Chinook Salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences 61(4): 590-602.

Gregory, R.S. 1993. Effect of Turbidity on the Predator Avoidance-Behavior of Juvenile Chinook Salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences 50(2): 241-246.

Gregory, R.S. and C.D. Levings. 1996. The effects of turbidity and vegetation on the risk of juvenile salmonids, Oncorhynchus spp., to predation by adult cutthroat trout, O. clarkii. Env. Biol. Fishes 47(3): 279-288.

Gregory, R.S., and C.D. Levings. 1998. Turbidity Reduces Predation on Migrating Juvenile Pacific Salmon. Transactions of the American Fisheries Society 127(2): 275-285.

Gregory, R.S., and T.G. Northcote. 1993. Surface, Planktonic, and Benthic Foraging by Juvenile Chinook Salmon (Oncorhynchus tshawytscha) in Turbid Laboratory Conditions. Canadian Journal of Fisheries and Aquatic Sciences 50: 233-240.

Greig, S.M., D.A. Sear, and P.A. Carling. 2005. The impact of fine sediment accumulation on the survival of incubating salmon progeny: Implications for sediment management. Science of The Total Environment 344(1-3): 241-258.

Greig, S.M., D.A. Sear, and P.A. Carling. 2007. A Review of Factors Influencing the Availability of Dissolved Oxygen to Incubating Salmonid Embryos. Hydrological Processes 21(3): 323-334.

Grette, G.B. 1985. Fish Monitoring During Pile Driving at Hiram H. Chittenden Lockes, August-September 1985. Report by Evans-Hamilton, Inc., to U.S. Army Corps of Engineers, Seattle, Washington.

Griffin, F.J., M.C. Pillai, C.A. Vines, J. Kaaria, T. Hibbard-Robbins, R. Yanagimachi, and G.N. Cherr. 1998. Effects of Salinity on Sperm Motility, Fertilization, and Development in the Pacific Herring, Clupea Pallasi. Biological Bulletin 194(1): 25-35.

Groberg, W.J., R.H. Mccoy, K.S. Pilcher, and J.L. Fryer. 1978. Relation of Water Temperature to Infections of Coho Salmon (Oncorhynchus-Kisutch), Chinook Salmon (O-Tshawytscha), and Steelhead Trout (Salmo-Gairdneri) with Aeromonas-Salmonicida and Aeromanas-Hydrophila. Journal of the Fisheries Research Board of Canada 35(1): 1-7.

Groot, C. and L. Margolis (eds.). 1991. Pacific Salmon Life Histories. Vancouver, British Columbia: University of British Columbia Press. Cited in WDNR 2006a.

Gross, H.P., W.A. Wurtsbaugh, and C. Luecke. 1998. The Role of Anadromous Sockeye Salmon in the Nutrient Loading and Productivity of Redfish Lake, Idaho. Transactions of the American Fisheries Society 127(1): 1-18.

Grubbs, S.A., and J.M. Taylor. 2004. The Influence of Flow Impoundment and River Regulation on the Distribution of Riverine Macroinvertebrates at Mammoth Cave National Park, Kentucky, U.S.A. Hydrobiologia 520(1-3): 19-28.

Guerra-García, J.M., J. Corzo, and J.C. García-Gómez. 2003. Short-Term Benthic Recolonization after Dredging in the Harbour of Ceuta, North Africa. Marine Ecology 24(3): 217-229.

Guidetti, P. 2004. Fish Assemblages Associated with Coastal Defense Structures in South-Western Italy (Mediterranean Sea). Journal of the Marine Biological Association of the United Kingdom 84(3): 669-670.

Guidetti, P., L. Verginella, C. Viva, R. Odorico, and F. Boero. 2005. Protection Effects on Fish Assemblages, and Comparison of Two Visual-Census Techniques in Shallow Artificial Rocky

Habitats in the Northern Adriatic Sea. Journal of the Marine Biological Association of the United Kingdom 85(2): 247-255.

Guidetti, P., S. Bussotti, and F. Boero. 2005. Evaluating the Effects of Protection on Fish Predators and Sea Urchins in Shallow Artificial Rocky Habitats: A Case Study in the Northern Adriatic Sea. Marine Environmental Research 59(4): 333-348.

Gurnell, A., and G. Petts. 2006. Trees as Riparian Engineers: The Tagliamento River, Italy. Earth Surface Processes and Landforms 31(12): 1558-1574.

Gurnell, A.M. 1998. The Hydrogeomorphological Effects of Beaver Dam-Building Activity. Progress in Physical Geography 22(2): 167-189.

Gustafson, R.G., T.C. Wainwright, G.A. Winans, F.W. Waknitz, L.T. Parker, and R.S. Waples. 1997. Status Review of Sockeye Salmon from Washington and Oregon, NOAA Technical Memorandum. Seattle, Washington: Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Gustafson, R.G., W.H. Lenarz, B.B. McCain, C.C. Schmitt, W.S. Grant, T.L. Builder, and R.D. Methot. 2000. Status Review of Pacific Hake, Pacific Cod, and Walleye Pollock from Puget Sound, Washington. NOAA Technical Memo NMFS-NWFSC-44. Seattle, Washington: Northwest Fisheries Science Center.

Guyette, R.P., W.G. Cole, D.C. Dey, and R.M. Muzika. 2002. Perspectives on the Age and Distribution of Large Wood in Riparian Carbon Pools. Canadian Journal of Fisheries and Aquatic Sciences 59(3): 578-585.

Haas, M.E., C.A. Simenstad, J.R. Cordell, D.A. Beauchamp, and B.S. Miller. 2002. Effects of large overwater structures on epibenthic juvenile salmon prey assemblages in Puget Sound, WA. Available: http://www.wsdot.wa.gov/Research/Reports/500/550.1.htm (accessed 2006.10.23).

Hadderingh, R.H. 1979. Fish Intake Mortality at Power Stations the Problem and Its Remedy. Aquatic Ecology 13(2): 83-93.

Hadderingh, R.H., and Z. Jager. 2002. Comparison of Fish Impingement by a Thermal Power Station with Fish Populations in the Ems Estuary. Journal of Fish Biology 61: 105-124.

Hagerthey, S.E., and W.C. Kerfoot. 2005. Spatial Variation in Groundwater-Related Resource Supply Influences Freshwater Benthic Algal Assemblage Composition. Journal of the North American Benthological Society 24(4): 807-819.

Haldorson, L. and L.J. Richards. 1986. Post-larval copper rockfish in the Strait of Georgia: Habitat use, feeding, and growth in the first year. Pages 129-141 in Proceedings International Rockfish Symposium. Anchorage, Alaska: Alaska Sea Grant College Program. Cited in NRC 2001.

Hall, J.L., and R.C. Wissmar. 2004. Habitat Factors Affecting Sockeye Salmon Redd Site Selection in Off-Channel Ponds of a River Floodplain. Transactions of the American Fisheries Society 133(6): 1480-1496.

Hall, L.W., and R.D. Anderson. 1999. A Deterministic Ecological Risk Assessment for Copper in European Saltwater Environments. Marine Pollution Bulletin 38(3): 207-218.

Hall, M.O., M.J. Durako, J.W. Fourqurean, and J.C. Zieman. 1999. Decadal Changes in Seagrass Distribution and Abundance in Florida Bay. Estuaries 22(2B): 445-459.

Hallermeier, R.J. 1981. A Profile Zonation for Seasonal Sand Beaches from Wave Climate. Coastal Engineering 4(3): 253-277.

Hallock, M. and P.E. Mongillo. 1998. Washington State Status Report for the Pygmy Whitefish. Washington Department of Fish and Wildlife. Olympia, Washington. Available at: http://wdfw.wa.gov/wlm/diversty/soc/status/whitfish/dftpwfsh.pdf (accessed 2006.10.23). Cited in WDNR 2006a.

Hamblin, P.F., D.Z. Zhu, F. Chiocchio, C. He, and M.N. Charlton. 2000. Monitoring Suspended Sediment Plumes by Optical and Acoustical Methods with Application to Sand Capping. Canadian Journal of Civil Engineering 27: 125-137.

Hammer, T.R. 1972. Stream Channel Enlargement Due to Urbanization. Water Resources Research 8: 1530-1541.

Hammerson, G.A. 1994. Beaver (Castor-Canadensis) - Ecosystem Alterations, Management, and Monitoring. Natural Areas Journal 14(1): 44-57.

Hansen, J.A., J.C.A. Marr, J. Lipton, D. Cacela, and H.L. Bergman. 1999a. Differences in neurobehavioral responses of chinook salmon (Oncorhynchus tshawytascha) and rainbow trout (Oncorhynchus mykiss) exposed to copper and cobalt: behavioral avoidance. Environ. Toxicol. Chem. 18:1972-1978. (As cited in Stratus 2005b).

Hansen, J.A., J.D. Rose, R.A. Jenkins, K.G. Gerow, and H.L. Bergman. 1999. Chinook Salmon (Oncorhynchus Tshawytscha) and Rainbow Trout (Oncorhynchus Mykiss) Exposed to Copper: Neurophysiological and Histological Effects on the Olfactory System. Environmental Toxicology and Chemistry 18: 1979-1991. Cited in Stratus 2005b.

Hansen, J.A., P.G. Welsh, J. Lipton, and D. Cacela. 2002. Effects of Copper Exposure on Growth and Survival of Juvenile Bull Trout. Transactions of the American Fisheries Society 131(4): 690-697.

Hansen, J.A., P.G. Welsh, J. Lipton, and M.J. Suedkamp. 2002. The Effects of Long-Term Cadmium Exposure on the Growth and Survival of Juvenile Bull Trout (Salvelinus confluentus). Aquatic Toxicology 58(3-4): 165-174.

Hansen, J.A., P.G. Welsh, J. Lipton, D. Cacela, and A.D. Dailey. 2002. Relative Sensitivity of Bull Trout (Salvelinus confluentus) and Rainbow Trout (Oncorhynchus mykiss) to Acute Exposures of Cadmium and Zinc. Environmental Toxicology and Chemistry 21(1): 67-75. Cited in Stratus 2005b.

Hard, J.J., R.G. Kope, W.S. Grant, F.W. Waknitz, L.T. Parker, and R.S. Waples. 1996. Status Review of Pink Salmon from Washington, Oregon, California. U.S. Department of Commerce., NOAA Technical Memorandum NMFS-NWFSC-25. Northwest Fisheries Science Center. Seattle, Washington. Available at http://www.nwfsc.noaa.gov/publications/techmemos/tm25/tm25.html (Accessed 2006.10.04).

Harding, J.S., K. Claassen, and N. Evers. 2006. Can Forest Fragments Reset Physical and Water Quality Conditions in Agricultural Catchments and Act as Refugia for Forest Stream Invertebrates? Hydrobiologia 568: 391-402.

Hardyniec, S., and S. Skeen. 2005. Pile Driving and Barotraumas Effects. No. 1941. Journal of Transportation Research Board 1941/2005: 184-190.

Harrahy, L.N.M., C.B. Schreck, and A.G. Maule. 2001. Antibody-Producing Cells Correlated to Body Weight in Juvenile Chinook Salmon (Oncorhynchus Tshawytscha) Acclimated to Optimal and Elevated Temperatures. Fish & Shellfish Immunology 11(8): 653-659.

Harris, C.T. 1974. The Geographical Distribution and Habitat of the Olympic Mudminnow (Novumbra Hubbsi). Master’s Thesis, University of Washington, Seattle, Washington.

Harrison, C.W. 1923. Planting Eyed Salmon and Trout Eggs. Trans. American Fisheries Society 53:191-200. Available http://afs.allenpress.com, accessed 2006.10.23.

Hart, B., S. Roberts, R. James, J. Taylor, D. Donnert, and R. Furrer. 2003. Use of Active Barriers to Reduce Eutrophication Problems in Urban Lakes. Water Science and Technology 47(7-8): 157-163.

Hart, C.W., and S.L.H. Fuller. 1974. Pollution Ecology of Freshwater Invertebrates. New York: Academic Press.

Hart, J.L. 1973. Pacific Fishes of Canada. Fisheries Research Board of Canada Bulletin 180: 730 pp.

Hartman, G.F., J.C. Scrivener, and M.J. Miles. 1996. Impacts of Logging in Carnation Creek, a High-Energy Coastal Stream in British Columbia, and Their Implication for Restoring Fish Habitat. Canadian Journal of Fisheries and Aquatic Sciences 53(S1): 237-251.

Hartwell, S.I., D.M. Jordahl, C.E.O. Dawson, and A.S. Ives. 1998. Toxicity of Scrap Tire Leachates in Estuarine Salinities: Are Tires Acceptable for Artificial Reefs? Transactions of the American Fisheries Society 127(5): 796-806.

Harvey, B.C., R.J. Nakamoto, and J.L. White. 1999. Influence of Large Woody Debris and a Bankfull Flood on Movement of Adult Resident Coastal Cutthroat Trout (Oncorhynchus Clarki) During Fall and Winter. Canadian Journal of Fisheries and Aquatic Sciences 56(11): 2161-2166.

Harvey, M.D., J. Pitlick, and J. Laird. 1987. Temporal and spatial variability of sediment storage and erosion in Ash Creek, Arizona. In: Beschta, R.L., Blinn, T., Grant, G.E., Ice, G.G., Swanson, F.J. (Eds.), Erosion and Sedimentation in the Pacific Rim. IAHS Publication, vol. 165. International Association of Hydrological Sciences Wallingford, Oxfordshire, UK, pp. 281–282.

Hasler, A.D., A.T. Scholz, and R.M. Horrall. 1978. Olfactory Imprinting and Homing in Salmon. American Scientist 66(3): 347-355.

Hasler, A.D., and A.T. Scholz. 1983. Olfactory Imprinting and Homing in Salmon. In Zoo-Physiol. 14, New York: Springer-Verlag. pp. 3-38.

Hastings, M.C. 1995. Physical effects of noise of fishes. Proceedings of INTER-NOISE 95, The 1995 International Congress on Noise Control Engineering, vol. II, pp. 979-984. Cited in Hastings and Popper 2005.

Hastings, M.C., A.N. Popper, J.J. Finneran, and P.J. Lanford. 1996. Effects of Low-Frequency Underwater Sound on Hair Cells of the Inner Ear and Lateral Line of the Teleost Fish Astronotus Ocellatus. Journal of the Acoustical Society of America 99(3): 1759-1766.

Hastings, M.C., and A.N. Popper. 2005. Effects of Sound on Fish. Prepared for California Department of Transportation by Jones and Stokes Sacramento, California. August 2005.

Hatch, A.C. and G.A. Burton, Jr. 1999. Sediment toxicity and stormwater runoff in a contaminated receiving system: Consideration of different bioassays in the laboratory and field. Chemosphere 39:1001-1017.

Hatch, A.C., and G.A. Burton. 1999. Phototoxicity of Fluoranthene to Two Freshwater Crustaceans, Hyalella Azteca and Daphnia Magna: Measures of Feeding Inhibition as a Toxicological Endpoint. Hydrobiologia 400: 243-248.

Hauer, F.R., G.C. Poole, J.T. Gangemi, and C.V. Baxter. 1999. Large Woody Debris in Bull Trout (Salvelinus Confluentus) Spawning Streams of Logged and Wilderness Watersheds in Northwest Montana. Canadian Journal of Fisheries and Aquatic Sciences 56(6): 915-924.

Hawkins, A.D. 1986. Underwater Sound and Fish Behavior. In The Behavior of Teleost Fishes, edited by T.J. Pitcher. Maryland: Johns Hopkins University Press. pp. 114-151.

Hawkins, A.D., and A.D.F. Johnstone. 1978. The Hearing of the Atlantic Salmon, Salmo Salar. Journal of Fish Biology 13: 655-673.

Hawkins, C.P., and J.R. Sedell. 1981. Longitudinal and Seasonal Changes in Functional Organization of Macroinvertebrate Communities in Four Oregon Streams. Ecology 62(2): 387-397.

Hawkins, C.P., M.L. Murphy, and N.H. Anderson. 1982. Effects of Canopy, Substrate Composition, and Gradient on the Structure of Macroinvertebrate Communities in Cascade Range Streams of Oregon. Ecology 63: 1840-1856.

Hawkins, C.P., M.L. Murphy, N.H. Anderson, and M.A. Wilzbach. 1983. Density of Fish and Salamanders in Relation to Riparian Canopy and Physical Habitat in Streams of the Northwestern United-States. Canadian Journal of Fisheries and Aquatic Sciences 40(8): 1173-1185.

Hawryshyn, C.W., and F.I. Harosi. 1993. Spectral Characteristics of Visual Pigments in Rainbow Trout (Oncorhynchus Mykiss). Vision Research 34: 1385-1392.

Hayman, R.A., E.M. Beamer, and R.E. McClure. 1996. Skagit System Cooperative Chinook Restoration Research Progress Report No. 1. FY 1995 Skagit River Chinook Restoration Research. Final Project Performance Report. Skagit System Cooperative, LaConner, WA.

Hazelwood, R.A., and J. Connelly. 2005. Estimation of Underwater Noise - a Simplified Method. International Journal of the Society of Underwater Technology 26(3): 51-57.

Healey, M.C. 1982. Juvenile Pacific Salmon in Estuaries: The Life Support System. In Estuarine Comparisons, edited by V.S. Kennedy. Academic Press. New York, New York. pp. 315-341.

Healey, M.C. 1991. Life History of Chinook Salmon (Oncorhynchus tshawytscha). Pp. 311-394230 in C. Groot and L. Margolis (eds.), Pacific Salmon Life Histories. Vancouver, British Columbia: University of British Columbia Press.

Heard, W.R. 1991. Life History of Pink Salmon (Oncorhynchus gorbuscha). Pp. 120-230 in C. Groot and L. Margolis (eds.), Pacific Salmon Life Histories. Vancouver, British Columbia: University of British Columbia Press. Cited in WDNR 2006a.

Heath, A.G., and G.M. Hughes. 1973. Cardiovascular and Respiratory Changes During Heat Stress in Rainbow-Trout (Salmo-Gairdneri). Journal of Experimental Biology 59(2): 323-338.

Heathershaw, A.D., P.D. Ward, and A.M. David. 2001. The Environmental Impact of Underwater Sound. Institute of Acoustics Proceedings 23 (4): 1-13. Cited in WSDOT 2006a.

Hecht, S.A., D.H. Baldwin, C.A. Mebane, T. Hawkes, and S.J. Gross. 2007. An Overview of Sensory Effects on Juvenile Salmonids Exposed to Dissolved Copper: Applying a Benchmark Concentration Approach to Evaluate Sublethal Neurobehavioral Toxicity. Technical White Paper. Lacey, Washington: National Marine Fisheries Service.

Heck, K. I. Jr. and R.J. Orth. 1980. Structural components of eelgrass (Zostera marina) meadows in the lower Chesapeake Bay--Decapod crustacea. Estuaries 3: 289-95. Cited in Nightingale and Simenstad 2001b.

Heck, K.I. Jr. and T.A. Thoman. 1984. The nursery role of seagrass meadows in the upper and lower reaches of the Chesapeake Bay. Estuaries 7(1): 70-92. Cited in Nightingale and Simenstad 2001b.

Heggenes, J., and R. Borgstrom. 1988. Effect of Mink, Mustela Vison Schreber, Predation on Cohorts of Juvenile Atlantic Salmon, Salmo salar L., and Brown Trout, S. trutta L., in Three Small Streams. Journal of Fish Biology 33(6): 885-894.

Hein, T., C. Baranyi, W. Reckendorfer, and F. Schiemer. 2004. The Impact of Surface Water Exchange on the Nutrient and Particle Dynamics in Side-Arms Along the River Danube, Austria. Science of the Total Environment 328(1-3): 207-218.

Heintz, R.A., B.D. Nelson, J. Hudson, M. Larsen, L. Holland, and M. Wipfli. 2004. Marine Subsidies in Freshwater: Effects of Salmon Carcasses on Lipid Class and Fatty Acid Composition of Juvenile Coho Salmon. Transactions of the American Fisheries Society 133(3): 559-567.

Heiser, D.W. and E.L. Finn. 1970. Observations of juvenile chum and pink salmon in marina and bulkheaded areas, Supplemental Progress Report. Washington Department of Fisheries, Olympia, WA. Cited in Nightingale and Simenstad 2001b.

Helfield, J.M., and R.J. Naiman. 2001. Effects of Salmon-Derived Nitrogen on Riparian Forest Growth and Implications for Stream Productivity. Ecology 82(9): 2403-2409.

Helfield, J.M., and R.J. Naiman. 2002. Salmon and Alder as Nitrogen Sources to Riparian Forests in a Boreal Alaskan Watershed. Oecologia 133(4): 573-582.

Helfman, Gene S. 1979. Fish attraction to floating objects in lakes. Pp. 49–57 in D.L. Johnson and R.A. Stein (eds.), Response of Fish to Habitat Structure in Standing Water. Bethesda, Maryland: American Fisheries Society. Cited in Carrasquero 2001.

Helfrich, L.A., C.R. Liston, B. Mefford, and R. Bark. 2001. Survival and Injury of Splittail and Chinook Salmon Passed through a Large Hidrostal Pump. North American Journal of Fisheries Management 21(3): 616-623.

Helvey, M. 2002. Are Southern California Oil and Gas Platforms Essential Fish Habitat? ICES Journal of Marine Science 59: S266-S271.

Henderson, P.A., and R.M.H. Seaby. 2000. Technical Evaluation of U.S. Environmental Protection Agency Proposed Cooling Water Intake Regulations for New Facilities. Lymington, England: Pices Conservation Ltd.

Henley, W.F., M.A. Patterson, R.J. Neves, and A.D. Lemly. 2000. Effects of Sedimentation and Turbidity on Lotic Food Webs: A Concise Review for Natural Resource Managers. Reviews in Fisheries Science 8(2): 125 - 139.

Henning, J. 2004. An evaluation of fish and amphibian use of restored and natural floodplain wetlands. Final Report EPA Grant number CD-97024901-1. Washington Department of Fish and Wildlife, Olympia, WA, USA. 81 p.

Henning, J.A., R.E. Gresswell, and I.A. Fleming. 2006. Juvenile Salmonid Use of Freshwater Emergent Wetlands in the Floodplain and Its Implications for Conservation Management. North American Journal of Fisheries Management 26(2): 367-376.

Hernandez, I., G. Peralta, J. Perez, and J. Vergava. 1997. Biomass and Dynamics of Growth of Ulva Species in Palmones River Estuary. Journal of Phycology 33: 764-772.

Hernandez-Miranda, E., A.T. Palma, and F.P. Ojeda. 2003. Larval Fish Assemblages in Nearshore Coastal Waters Off Central Chile: Temporal and Spatial Patterns. Estuarine Coastal and Shelf Science 56(5-6): 1075-1092.

Herrera Environmental Consultants, Inc. 2004. As-Built Considerations and Monitoring Recommendations: Hoh River Erosion Control Project U.S. Highway 101 at Milepost 174.4. Seattle: Washington State Department of Transportation.

Herrera Environmental Consultants, Inc. 2005. Marine shoreline sediment survey and assessment, Thurston County, Washington. Prepared for: Thurston Regional Planning Council. Olympia, WA.

Herrera Environmental Consultants, Inc. 2006. Landslide Disposal Site Identification: Sound Transit Everett-to-Seattle Commuter Rail. Seattle, Washington: Sound Transit.

Herrera Environmental Consultants, Inc. 2006. Simonson Place Boat Ramp and Stormwater Outfall Replacement Project: Coastal Processes Assessment. Seattle, Washington: Island County Public Works.

Herrera Environmental Consultants, Inc. 2007. Channel Modifications white paper. Prepared by Herrera Environmental Consultants, Inc., Seattle, Washington, for the Washington Department of Fish and Wildlife, Olympia, Washington. Draft, dated September 2007.

Herrera Environmental Consultants, Inc. 2007. Fish Passage White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, November 2007.

Herrera Environmental Consultants, Inc. 2007. Fish Screens White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, November 2007.

Herrera Environmental Consultants, Inc. 2007. Flow Control Structures White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, September 2007.

Herrera Environmental Consultants, Inc. 2007. Habitat Modifications White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, August 2007.

Herrera Environmental Consultants, Inc. 2007. Marinas and Shipping/Ferry Terminals White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, June 2007.

Herrera Environmental Consultants, Inc. 2007. Shoreline Modifications White Paper. Prepared for Washington Department of Fish and Wildlife by Herrera Environmental Consultants, Inc., Seattle, Washington. Draft, July 2007.

Herrera Environmental Consultants, Inc. 2007. Water Quality Statistical and Pollutant Loading Analysis: Green-Duwamish Watershed Water Quality Assessment. Prepared for King County Department of Natural Resources and Parks by Herrera Environmental Consultants, Inc. (Herrera), Seattle, Washington.

Hershey, A.E., and G.A. Lamberti. 1992. Stream Macroinvertebrate Communities. In Watershed Management – Balancing Sustainability and Environmental Change, edited by R.J. Naiman. New York: Springer-Verlag.

Hetrick, N.J., M.A. Brusven, T.C. Bjornn, R.M. Keith, and W.R. Meehan. 1998. Effects of Canopy Removal on Invertebrates and Diet of Juvenile Coho Salmon in a Small Stream in Southeast Alaska. Transactions of the American Fisheries Society 127(6): 876-888.

Hewitt, M., R. Schryer, A. Pryce, A. Belknap, B. Firth, and G. Van Der Kraak. 2005. Accumulation of Hormonally Active Substances by Wild White Sucker (Catostomus Commersoni) Exposed to Effluents Discharged to the Wabigoon River. Water Quality Research Journal of Canada 40(3): 315-327.

Heywood, M.J.T., and D.E. Walling. 2007. The Sedimentation of Salmonid Spawning Gravels in the Hampshire Avon Catchment, UK: Implications for the Dissolved Oxygen Content of Intragravel Water and Embryo Survival. Hydrological Processes 21(6): 770-788.

Hickey, M.B.C., and B. Doran. 2004. A Review of the Efficiency of Buffer Strips for the Maintenance and Enhancement of Riparian Ecosystems. Water Quality Research Journal of Canada 39(3): 311-317.

Hicks, B.J., J.D. Hall, P.A. Bisson and J.R. Sedell. 1991. Responses of Salmonids to Habitat Change. Pp. 483-518 in W.R. Meehan (ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. Bethesda, Maryland: American Fisheries Society Special Publication 19.

Hicks, B.J., M.S. Wipfli, D.W. Lang, and M.E. Lang. 2005. Marine-Derived Nitrogen and Carbon in Freshwater-Riparian Food Webs of the Copper River Delta, Southcentral Alaska. Oecologia 144(4): 558-569.

Higgins, K., P. Schlenger, J. Small, D. Hennessy, and J. Hall. 2005. Spatial relationships between beneficial and detrimental nearshore habitat parameters in WRIA 9 and the City of Seattle. Proceedings of the 2005 Puget Sound Georgia Basin Research Conference.

Hilderbrand, R.H., A.D. Lemly, C.A. Dolloff, and K.L. Harpster. 1997. Effects of Large Woody Debris Placement on Stream Channels and Benthic Macroinvertebrates. Canadian Journal of Fisheries and Aquatic Sciences 54(4): 931-939.

Hill, D.F., and M.M. Beachler. 2002. ADV Measurements of Planing Boat Prop Wash in the Extreme Near Field. Hydraulic Measurements and Experimental Methods. Proceedings of the Specialty Conference July 28-August 1, 2002, Estes Park, Colorado; Sponsored by environmental and Water Resources Institute of ASCE, and International Association of Hydraulic Engineering and Research.

Hill, M.J., E.A. Long, and S. Hardin. 1993. Effects of Dam Removal on Dead Lake, Chipola River, Florida. Apalachicola River Watershed Investigations F-39-R. Florida Game and Fresh Water Fish Commission.

Hilton, J., and G.L. Phillips. 1982. The Effect of Boat Activity on Turbidity in a Shallow Broadland River. Journal of Applied Ecology 19: 143-150.

Hinchey, E.K., L.C. Schaffner, C.C. Hoar, B.W. Vogt, and L.P. Batte. 2006. Responses of Estuarine Benthic Invertebrates to Sediment Burial: The Importance of Mobility and Adaptation. Hydrobiologia 556: 85-98.

Hinson, D., C.R. Steward, S.E. Wills, T.J. Kock, M.A. Kritter, T.L. Liedtke, and D.W. Rondorf. 2007. Adult Salmonid Migration Behavior in the North Fork Toutle River, Washington Following the 1980 Eruption of Mount St. Helens. American Fisheries Society/Sea Grant Symposium - Mitigating Impacts of Natural Hazards on Fishery Ecosystems. San Francisco, California.

Hinton, S.A., R.L. Emmett, and G.T. McCabe. 1992. Benthic Invertebrates, Demersal Fishes and Sediment Characteristics at and Adjacent to Ocean Dredge Material Disposal Site F, Offshore from the Columbia River, June 1989-1990.

Hiratsuka, J.I., M. Yamamuro, and Y. Ishitobi. 2007. Long-Term Change in Water Transparency before and after the Loss of Eelgrass Beds in an Estuarine Lagoon, Lake Nakaumi, Japan. Limnology 8(1): 53-58.

Hirose, T., and K. Kawaguchi. 1998. Sediment Size Composition as an Important Factor in the Selection of Spawning Site by the Japanese Surf Smelt Hypomesus Japonicus. Fisheries Science 64(6): 995-996.

Hjort, R.C., P.L. Hulett, L.D. LaBolle, and H.W. Li. 1983. Fish and Invertebrates of Revetments and Other Habitats in the Willamette River, Oregon. Vicksburg, Mississippi: Waterways Experiment Station, Army Corps of Engineers.

Hoar, W. S. 1951. The behavior of chum, pink, and coho salmon in relation to their seaward migration. Journal Fishery Research Board of Canada 8: 241-63. Cited in Nightingale and Simenstad 2001b.

Hoar, W.S., M.H.A. Keenleyside, and R.G. Goodall. 1957. Reactions of Juvenile Pacific Salmon to Light. Journal Fishery Research Board of Canada 14: 815-830.

Hoffmann, A. 2000. The Association of the Stream Caddisfly Lasiocephala Basalis (Kol.) (Trichoptera: Lepidostomatidae) with Wood. International Review of Hydrobiology 85(1): 79-93.

Holing, D. 1994. The Sound and The Fury: Debate Gets Louder Over Ocean Noise Pollution and Marine Mammals. The Amicus Journal 16(3):19-25. Cited in Radle 2005.

Holliman, F.M., J.B. Reynolds, and T.J. Kwak. 2003. Electroshock-Induced Injury and Mortality in the Spotfin Chub, a Threatened Minnow. North American Journal of Fish Management 23(3): 962-966.

Holm, T.E., and P. Clausen. 2006. Effects of Water Level Management on Autumn Staging Waterbird and Macrophyte Diversity in Three Danish Coastal Lagoons. Biodiversity and Conservation 15(14): 4399-4423.

Hood, W.G. 2004. Indirect Environmental Effects of Dikes on Estuarine Tidal Channels: Thinking Outside of the Dike for Habitat Restoration and Monitoring. Estuaries 27(2): 273-282.

Hood, W.G. 2006. A Conceptual Model of Depositional, Rather Than Erosional, Tidal Channel Development in the Rapidly Prograding Skagit River Delta (Washington, USA). Earth Surface Processes and Landforms 31(14): 1824-1838.

Horner, R.A. 1998. Harmful Algal Blooms in Puget Sound: General Perspective. Puget Sound Research: From Basic Science to Resource Management, Seattle, Washington, March 12 and 13, 1998. pp. 809-811.

Hoshikawa, H., Y. Sakai, and A. Kijima. 1998. Growth Characteristics of the Hybrid between Pinto Abalone, Haliotis kamtschatkana Jonas, and Exo Abalone, H. discus hannai Ino, under High and Low Temperature. Journal of Shellfish Research 17(3): 673-677.

Hossain, S., B.D. Eyre, and L.J. McKee. 2004. Impacts of Dredging on Dry Season Suspended Sediment Concentration in the Brisbane River Estuary, Queensland, Australia. Estuarine Coastal and Shelf Science 61(3): 539-545.

Howard, J.K., and K.M. Cuffey. 2003. Freshwater Mussels in a California North Coast Range River: Occurrence, Distribution, and Controls. Journal of the North American Benthological Society 22(1): 63-77.

Howell, M.D., M.D. Romano, and T.A. Rien. 2001. Draft outmigration timing and distribution of larval eucachon, Thaleichthys pacificus, in the lower Columbia River, Spring 2001. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife.

Hruby, T., T. Granger, K. Brunner, S. Cooke, K. Dublanica, R. Gersib, L. Reinelt, K. Richter, D. Sheldon, E. Teachout, A. Wald, and F. Weinmann. 1999. Methods for Assessing Wetland Functions Volume I: Riverine and Depressional Wetlands in the Lowlands of Western Washington. Publication #99-115. July. Washington Department Ecology, Olympia.

Hueckel, G.J. and R.L. Slayton. 1982. Fish foraging on an artificial reef in Puget Sound, Washington. Marine Fisheries Review 44:38-44. Cited in NRC 2001.

Hughes, G.W., and A.E. Peden. 1989. Status of the Umatilla Dace, Rhinichthys umatilla, in Canada. Canadian Field-Naturalist 103(2): 193-200.

Hull, J.H., J.M. Jersak, and C.A. Kasper. 1999. In Situ Capping of Contaminated Sediments: Comparing the Relative Effectiveness of Sand Vs. Clay Mineral-Based Sediment Caps. Proceedings of the 1999 Conference on Hazardous Waste Research, St. Louis, Missouri, May 24-27, 1999.

Hull, S.C. 1987. Macroalgal Mats and Species Abundance: A Field Experiment. Estuarine Coastal and Shelf Science 25: 519-532.

Hunt, R.J., M. Strand, and J.F. Walker. 2006. Measuring Groundwater-Surface Water Interaction and Its Effect on Wetland Stream Benthic Productivity, Trout Lake Watershed, Northern Wisconsin, USA. Journal of Hydrology 320(3-4): 370-384.

Hurst, C.K., and A. Brebner. 1969. Shore Erosion and Protection, St. Lawrence River – Canada. XXII International Navigation Congress, pp. 45-46.

Hutchings, J.A., and L. Gerber. 2002. Sex-Biased Dispersal in a Salmonid Fish. Proceedings of the Royal Society of London Series B-Biological Sciences 269(1508): 2487-2493.

Ikuta, K., Y. Suzuki, and S. Kitamura. 2003. Effects of low pH on the reproductive behavior of salmonid fishes. Fish Physiology and Biochemistry 28:407-410.

Illingworth and Rodkin, Inc. 2001. Noise and Vibration Measurements Associated with the Pile Installation Demonstration Project for the San Francisco-Oakland Bay Bridge East Span, Final Data Report.

Incardona, J. and N. Scholz. 2006. Cardiovascular defects in fish embryos exposed to polycyclic aromatic hydrocarbons. Available at:

http://www.nwfsc.noaa.gov/research/divisions/ec/ecotox/fishneurobiology/cardio.cfm (Accessed: 2006.10.23).

Incardona, J.P., T.K. Collier and N.L. Scholz. 2004. Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons. Toxicology and Applied Pharmacology 196(2):191-205.

Ingermann, R.L., M. Holcomb, M.L. Robinson, and J.G. Cloud. 2002. Carbon Dioxide and pH Affect Sperm Motility of White Sturgeon (Acipenser Transmontanus). Journal of Experimental Biology 205(18): 2885-2890.

Ingersoll, C.G., P.S. Haverland, E.L. Brunson, T.J. Canfield, F.J. Dwyuer, C.E. Henke, N.E. Kemble, D.R. Mount, and R.G. Fox. 1996. Calculation and Evaluation of Sediment Effect Concentrations for the Maphipod Hyalella Axteca and Midge Chrionomus Riparius. Journal of Great Lakes Research 22(3): 602-623. Cited in Stratus 2005b.

Ingram, J. 1982. Migration of Creosote and Its Components from Treated Piling Sections in a Marine Environment. Proceedings of the Annual Meeting of American Wood-Preservers' Association 78: 120-128.

Isaacson, M.A., Kennedy, and J. Baldwin. 1996. Wave Reflection Effects on Small Craft Motions. Canadian Journal of Engineering 23(2): 340-346.

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber Harvest Impacts on Small Headwater Stream Channels in the Coast Ranges of Washington. Journal of the American Water Resources Association 37(6): 1533-1549.

Jackson, G.A. 1984. Internal Wave Attenuation by Coastal Kelp Stands. Journal of Physical Oceanography 14: 1300-1306.

Jackson, N.L., K.F. Nordstrom, and D.R. Smith. 2005. Influence of Waves and Horseshoe Crab Spawning on Beach Morphology and Sediment Grain-Size Characteristics on a Sandy Estuarine Beach. Sedimentology 52(5): 1097-1108.

Jacobs, P.H., and U. Forstner. 1999. Concept of Subaqueous Capping of Contaminated Sediments with Active Barrier Systems (ABS) Using Natural and Modified Zeolites. Water Research 33(9): 2083-2087.

Jacobsen, E.E. and M.L. Schwartz. 1981. The use of geomorphic indicators to determine the direction of net shore-drift. Shore and Beach 49:38-43.

Jacobson, K.C., M.R. Arkoosh, A.N. Kagley, E.R. Clemons, T.K. Collier, and E. Casillas. 2003. Cumulative Effects of Natural and Anthropogenic Stress on Immune Function and Disease Resistance in Juvenile Chinook Salmon. Journal of Aquatic Animal Health 15(1): 1-12.

Jacobson, P.J., K.M. Jacobson, P.L. Angermeier, and D.S. Cherry. 1999. Transport, Retention, and Ecological Significance of Woody Debris within a Large Ephemeral River. Journal of the North American Benthological Society 18(4): 429-444.

Jacobson, R.B. 2006. Geomorphic Context and Effects of Instream Gravel Mining and Bank Stabilization Activities. Columbia, Missouri: United States Geological Survey Columbia Environmental Research Center Administrative Letter Report.

Jager, H.I. 2006. Chutes and Ladders and Other Games We Play with Rivers. I. Simulated Effects of Upstream Passage on White Sturgeon. Canadian Journal of Fisheries and Aquatic Sciences 63(1): 165-175.

Jamieson, G.S. 1999. Review of status of northern, or pinto, abalone, Haliotis kamtschatkana, in Canada. Department of Fisheries and Oceans Canada, Canadian Stock Assessment Secretariat Research Document 99/190.

JASCO. 2005. British Columbia Transmission Corporation, Vancouver Island Transmission Reienforcement Project, Atmospheric and Underwater Acoustics Assessment Report. Victoria, B.C., Canada. December 15.

Jennings, M.J., E.E. Emmons, G.R. Hatzenbeler, C. Edwards, and M.A. Bozek. 2003. Is Littoral Habitat Affected by Residential Development and Land Use in Watersheds of Wisconsin Lakes? Lake and Reservoir Management 19(3): 272-279.

Jennings, M.J., M.A. Bozek, G.R. Hatzenbeler, E.E. Emmons, and M.D. Staggs. 1999. Cumulative Effects of Incremental Shoreline Habitat Modifications on Fish Assemblages in North Temperate Lakes. North American Journal of Fisheries Management 19: 18-27.

Jensen, J.O.T. 2003. New mechanical shock sensitivity units in support of criteria for protection of salmonid eggs from blasting or seismic disturbance. Canadian Technical Report of Fisheries and Aquatic Sciences 2452. Fisheries and Oceans Canada Science Branch, Pacific Biological Station, Nanaimo, British Columbia.

Jensen, P.G., P.D. Curtis, M.E. Lehnert, and D.L. Hamelin. 2001. Habitat and Structural Factors Influencing Beaver Interference with Highway Culverts. Wildlife Society Bulletin 29(2): 654-664.

Jia, Y., T. Kitamura, and S.S.Y. Wang. 2001. Simulation of Scour Process in Plunging Pool of Loose Bed-Material. Journal of Hydraulic Engineering-ASCE 127(3): 219-229.

Johannes, R.E. 1980. The Ecological Significance of the Submarine Discharge of Groundwater. Marine Ecology-Progress Series 3(4): 365-373.

Johannessen, J.W., A. MacLennan, and A. McBride. 2005. Inventory and Assessment of Current and Historic Beach Feeding Sources/Erosion and Accretion Areas for the Marine

Shorelines of Water Resource Inventory Areas 8 & 9, Prepared by Coastal Geologic Services, Prepared for King County Department of Natural Resources and Parks, Seattle, WA.

Johnson, D.D. and D.J. Wildish. 1982. Effect of Suspended Sediment on Feeding by Larval Herring (Clupea harengus harengus L.), Bulletin of Environmental Contamination and Toxicology, 29, 261-267.

Johnson, G.E., B.D. Ebberts, D.D. Dauble, A.E. Giorgi, P.G. Heisey, R.P. Mueller, and D.A. Neitzel. 2003. Effects of Jet Entry at High-Flow Outfalls on Juvenile Pacific Salmon. North American Journal of Fisheries Management 23(2): 441-449.

Johnson, L.L., and J.T. Landahl. 1994. Chemical Contaminants, Liver Disease, and Mortality Rates in English Sole (Pleuronectes Vetulus). Ecological Applications 4: 59-68.

Johnson, L.L., E. Casilllas, T.K. Collier, J.E. Stein, and U. Varanasi. 1993. Contaminant Effects of Reproductive Success in Selected Benthic Fish Species. Marine Environmental Research 35: 165-170.

Johnson, L.L., G.M. Ylitalo, M.R. Arkoosh, A.N. Kagley, C. Stafford, J.L. Bolton, J. Buzitis, B.F. Anulacion, and T.K. Collier. 2007. Contaminant Exposure in Outmigrant Juvenile Salmon from Pacific Northwest Estuaries of the United States. Environmental Monitoring and Assessment 124(1-3): 167-194.

Johnson, L.L., J.T. Landahl, K. Kardong, and B. Horness. 1995. Chemical Contaminants, Fishing Pressure, and Population Growth of Puget Sound English Sole (Pleuronectes Vetulus). Puget Sound Research '95 Proceedings, Bellevue, Washington, pp. 686-698.

Johnson, L.L., T.K. Collier, and J.E. Stein. 2002. An Analysis in Support of Sediment Quality Thresholds for Polycyclic Aromatic Hydrocarbons (PAHs) to Protect Estuarine Fish. Aquatic Conservation: Marine and Freshwater Ecosystems 12: 517-538.

Johnson, O.W., M.H. Ruckelshaus, W.S. Grant, F. W. Waknitz, A.M. Garrett, G.J. Bryant, K. Neely, and J.J. Hard. 1999. Status Review of Coastal Cutthroat Trout from Washington, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC- 37. Northwest Fisheries Science Center. Seattle, Washington. Available at: http://www.nwfsc.noaa.gov/publications/techmemos/tm37/tm37.html (Accessed 2006.10.04).

Johnson, O.W., W.S. Grant, R.G. Kope, K. Neely, F.W. Waknitz, and R.S. Waples. 1997. Status Review of Chum Salmon from Washington, Oregon, and California, NOAA Technical Memorandum Seattle, Washington: U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Johnson, P.N., F.A. Goetz, and G.R. Ploskey. 1998. Unpublished Report on Salmon Light Study at Hiram M. Chittenden Locks. Stevenson, Washington: U.S. Army Corps of Engineers.

Johnson, P.N., K. Bouchard, and F.A. Goetz. 2005. Effectiveness of Strobe Lights for Reducing Juvenile Salmonid Entrainment into a Navigation Lock. North American Journal of Fisheries Management 25(2): 491-501.

Johnson, S.L., and J.A. Jones. 2000. Stream Temperature Responses to Forest Harvest and Debris Flows in Western Cascades, Oregon. Canadian Journal of Fisheries and Aquatic Sciences 57(S2): 30-39.

Johnson, S.W., and J.F. Thedinga. 2005. Fish Use and Size of Eelgrass Meadows in Southeastern Alaska: A Baseline for Long-Term Assessment of Biotic Change. Northwest Science 79(2-3): 141-155.

Johnston, C.A., and R.J. Naiman. 1990. Aquatic Patch Creation in Relation to Beaver Population Trends. Ecology 71(4): 1617-1621.

Johnston, S.G., P.G. Slavich, and P. Hirst. 2005a. The Impact of Controlled Tidal Exchange on Drainage Water Quality in Acid Sulphate Soil Backswamps. Agricultural Water Management 73(2): 87-111.

Johnston, S.G., P.G. Slavich, and P. Hirst. 2005b. Opening Floodgates in Coastal Floodplain Drains: Effects on Tidal Forcing and Lateral Transport of Solutes in Adjacent Groundwater. Agricultural Water Management 74(1): 23-46.

Jones and Stokes. 2002. San Juan Islands Cable Replacement Project. Nearshore Habitat Disturbance Year 1 Post-Construction Monitoring Report. November 21. (J&S 0P001.00.005) Bellevue, WA. Prepared for Bonneville Power Administration. Vancouver, WA.

Jones and Stokes. 2005. San Juan Islands Cable Replacement Project. Nearshore Habitat Recovery 2005 Post-Construction Monitoring Report. December. (J&S 0P011.03.002) Bellevue, WA. Prepared for Bonneville Power Administration. Vancouver, WA.

Jones and Stokes. 2006. Biological Assessment. Vancouver Island Transmission Reinforcement Project. May. (J&S 05197.05) Bellevue, WA. Prepared for British Columbia Transmission Corporation (BCTC).

Jones and Stokes. 2006. Overwater Structures and Non Structural Piling (White Paper). Prepared by Jones and Stokes Associates, in association with Anchor Environmental, L.L.C., and R2 Consultants for the Washington Department of Fish and Wildlife, Olympia, Washington.

Jones and Stokes. 2006b. Water Crossings White Paper. Prepared by Jones and Stokes Associates, in association with Anchor Environmental, L.L.C., and R2 Consultants for the Washington Department of Fish and Wildlife, Olympia, Washington.

Jones and Stokes. 2006. Biological Assessment. Vancouver Island Transmission Reinforcement Project. May. (J&S 05197.05) Bellevue, WA. Prepared for British Columbia Transmission Corporation (BCTC).

Jones, A.W. 1996. Concentration of Trace Metals in Two Species of Planktonic Copepods from the Duwamish River Estuary, Elliott Bay, and the Main Basin of Puget Sound (Abstract). Pacific Estuarine Research Society 19th Annual Meeting.

Jones, J.B., S.G. Fisher, and N.B. Grimm. 1995. Nitrification in the Hyporheic Zone of a Desert Stream Ecosystem. Journal of the North American Benthological Society 14: 249-258.

Joy, M.K., and R.G. Death. 2001. Control of Freshwater Fish and Crayfish Community Structure in Taranaki, New Zealand: Dams, Diadromy or Habitat Structure? Freshwater Biology 46(3): 417-429.

Jungwirth, M., O. Moog, and S. Muhar. 1993. Effects of River Bed Restructuring on Fish and Benthos of a 5th-Order Stream, Melk, Austria. Regulated Rivers-Research & Management 8(1-2): 195-204.

Junk, W.J., P.B. Bayley, and R.E. Sparks. 1989. The Flood Pulse Concept in River-Floodplain Systems. Proceedings of the International Large River Symposium, Ottawa, Canada, pp. 110-127.

Kadlec, R., and R. Knight. 1996. Treatment Wetlands. Boca Raton, Florida: CRC Press/Lewis Publishers.

Kahler, T.H. and T.P. Quinn. 1998. Juvenile and resident salmonid movement and passage through culverts. Prepared for Washington State Transportation Commission and U.S. Department of Transportation. Prepared by Fisheries Research Institute, University of Washington.

Kahler, T.H., M. Grassley, and D. Beauchamp. 2000. A Summary of the Effects of Bulkheads, Piers, and Other Artificial Structures and Shorezone Development on ESA-Listed Salmonids in Lakes. Prepared for The City of Bellevue by The Watershed Company, Kirkland, Washington. July 2000.

Kahler, T.H., P. Roni, and T.P. Quinn. 2001. Summer Movement and Growth of Juvenile Anadromous Salmonids in Small Western Washington Streams. Canadian Journal of Fisheries & Aquatic Sciences 58(10): 1947-1956.

Kail, J. 2003. Influence of Large Woody Debris on the Morphology of Six Central European Streams. Geomorphology 51(1-3): 207-223.

Kaller, M.D., and W.E. Kelso. 2007. Association of Macroinvertebrate Assemblages with Dissolved Oxygen Concentration and Wood Surface Area in Selected Subtropical Streams of the Southeastern USA. Aquatic Ecology V41(1): 95-110.

Kalmijn, A.J. 1988. Hydrodynamic and Acoustic Field Detection. In Sensory Biology of Aquatic Animals, edited by J. Atema, R.R. Fay, A.N. Popper and W.N. Tavolga. New York: Springer-Verlag. pp. 83-130.

Kang, S.-M., J.J. Morrell, J. Simonsen, and S.T. Lebow. 2003. Creosote Movement from Treated Wood Immersed in Fresh Water: Initial PAH Migration. IRG/WP/03-5. International Research Group on Wood Preservation. Prepared for the 34th Annual Meeting, Brisbane, Australia, 18-25 May. Cited in Stratus 2005a.

Kapoor, B.G., and B. Khanna, eds. 2004. Ichthyology Handbook. Berlin: Springer-Verlag. 1080 pp.

Karp, C.A., and G. Mueller. 2002. Razorback Sucker Movements and Habitat Use in the San Juan River Inflow, Lake Powell, Utah, 1995-1997. Western North American Naturalist 62(1): 106-111.

Karr, J.R. 1991. Biological Integrity - a Long-Neglected Aspect of Water-Resource Management. Ecological Applications 1(1): 66-84.

Karrow, N.A., H.J. Boermans, D.G. Dixon, A. Hontella, K.R. Solomon, J.J. Whyte, and N.C. Bols. 1999. Characterizing the immunotoxicity of creosote to rainbow trout (Oncorhynchus mykiss): A microcosm study. Aquatic Toxicology 45(4):223-239. Cited in Stratus 2005a.

Katano, O., T. Nakamura, S. Abe, S. Yamamoto, and Y. Baba. 2006. Comparison of Fish Communities between Above- and Below-Dam Sections of Small Streams; Barrier Effect to Diadromous Fishes. Journal of Fish Biology 68(3): 767-782.

Katopodis, C. 1992. Introduction to Fishway Design. Freshwater Institute, Central and Arctic Region, Department of Fisheries and Oceans, Canada.

Kauffman, J.B., R.L. Beschta, and W.S. Platts. 1993. Fish Habitat Improvement Projects in the Fifteenmile Creek and Trout Creek Basins of Central Oregon: Field Review and Management Recommendations. DOE/BP-18955-1; Other: ON: DE93019488. United States.

Kawamata, S. 2001. Adaptive Mechanical Tolerance and Dislodgement Velocity of the Kelp Laminaria Japonica in Wave-Induced Water Motion. Marine Ecology-Progress Series 211: 89-104.

Keller, E.A., and F.J. Swanson. 1979. Effects of Large Organic Material on Channel Form and Fluvial Processes. Earth Surface Processes and Landforms 4(4): 361-380.

Kelley, D.S., J.A. Baross, and J.R. Delaney. 2002. Volcanoes, Fluids, and Life at Mid-Ocean Ridge Spreading Centers. Annual Review of Earth and Planetary Sciences 30: 385-491.

Kelsey, K.A., and S.D. West. 1998. Riparian Wildlife. In River Ecology and Management: Lessons from the Pacific Coastal Ecoregion, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag.

Kemp, P.S., M.H. Gessel, B.P. Sandford, and J.G. Williams. 2006. The Behavior of Pacific Salmonid Smolts During Passage over Two Experimental Weirs under Light and Dark Conditions. River Research and Applications 22(4): 429-440.

Kendall, A.W. and W.H. Lenarz. 1986. Status of early life history studies of northeast Pacific rockfishes. Pages 99-128 in: Proceedings International Rockfish Symposium. Anchorage, Alaska: Alaska Sea Grant College Program. Cited in NRC 2001.

Kennish, M.J. 2002. Impacts of Motorized Watercraft on Shallow Estuarine and Coastal Marine Environments. Journal of Coastal Research Special Issue 37.

Kenworthy, W.J., and D.E. Haunert. 1991. The Light Requirements of Seagrasses: Proceedings of a Workshop to Examine the Capability of Water Quality Criteria, Standards and Monitoring Programs to Protect Seagrasses. National Oceanic and Atmospheric Administration.

Key, L.O., J.A. Jackson, C.R. Sprague, and E.E. Kerfoot. 1994. Nearshore habitat use by subyearling chinook salmon in the Columbia and Snake rivers. pp 120-150 in D.W. Rondorf and W.H. Miller. Identification of the spawning, rearing and migratory requirements of fall chinook salmon in the Columbia River Basin. Annual Report, 1992. U.S. Geological Survey, Columbia River Research Laboratory, Cook WA. Report to Bonneville Power Administration. Report No. DOE/BP-21708-3.

Kiefer, R.B., and J.N. Lockhart. 1995. Idaho Department of Fish and Game Intensive Evaluation and Monitoring of Chinook Salmon and Steelhead Production, Crooked River and Upper Salmon River Sites, Annual Progress Report January 1, 1993 - December 31, 1993.

Kiel, S. 2006. New Records and Species of Molluscs from Tertiary Cold-Seep Carbonates in Washington State, USA. Journal of Paleontology 80(1): 121-137.

King County. 2000. Selected ongoing and recent research on chinook salmon in the greater Lake Washington watershed. Conference held on November 8–9, 2000 in Seattle Washington by King County Department of Natural Resources, Wastewater Treatment Division. Cited in Carrasquero 2001.

King, D.M., N.J. Cooper, J.C. Morfett, and D.J. Pope. 2000. Application of offshore breakwaters to the UK: A case study at Elmer Beach. Journal of Coastal Research 16(1): 172-187.

Kingsford, R.T. 2000. Ecological Impacts of Dams, Water Diversions and River Management on Floodplain Wetlands in Australia. Austral Ecology 25(2): 109-127.

Kiorboe, T., F. Mohlenberg, and O. Nohr. 1981. Effect of Suspended Bottom Material on Growth and Energetics in Mytilus-Edulis. Marine Biology 61(4): 283-288.

Kishi, D., M. Murakami, S. Nakano, and Y. Taniguchi. 2004. Effects of Forestry on the Thermal Habitat of Dolly Varden (Salvelinus Malma). Ecological Research 19(3): 283-290.

Kitano, S., and K. Shimazaki. 1995. Spawning Habitat and Nest Depth of Female Dolly-Varden Salvelinus-Malma of Different Body-Size. Fisheries Science 61(5): 776-779.

Knapp, S.M. 1992. Evaluation of Juvenile Fish Bypass and Adult Fish Passage Facilities at Water Diversions in the Umatilla River, Progress Report 1990-1991. DOE/BP-01385-2, Oregon Dept. of Fish and Wildlife, Portland, Oregon.

Knighton, D. 1998. Fluvial Forms and Processes: A New Perspective. London: Arnold.

Knudsen, E.E., and S.J. Dilley. 1987. Effects of Riprap Bank Reinforcement on Juvenile Salmonids in Four Western Washington Streams. North American Journal of Fisheries Management 7: 351-356.

Knudsen, F.R., P.S. Enger, and O. Sand. 1992. Awareness Reactions and Avoidance Responses to Sound in Juvenile Atlantic Salmon, Salmo Salar. Journal of Fish Biology 40: 523-534.

Knutson, K. L., and V. L. Naef. 1997. Management recommendations for Washington’s priority habitats: riparian. Olympia, Washington: Washington Department of Fish and Wildlife. 181 pp.

Knutson, P.L. and W.W. Woodhouse Jr. 1983. Shore stabilization with salt marsh vegetation. Special Report No. 9. U.S. Army Corps of Engineers, Fort Belvior, Virginia.

Koch, E.M. 2001. Beyond Light: Physical, Geological, and Geochemical Parameters as Possible Submersed Aquatic Vegetation Habitat Requirements. Estuaries 24(1): 1-17.

Koch, E.W., and S. Beer. 2006. Tides, Light and the Distribution of Zostera Marina in Long Island Sound, USA. Aquatic Botany 53(1-2): 97-107.

Kock, T.J., J.L. Congleton, and P.J. Anders. 2006. Effects of Sediment Cover on Survival and Development of White Sturgeon Embryos. North American Journal of Fisheries Management 26(1): 134-141.

Kogan, I., C.K. Paull, L. Kuhnz, E.J. Burton, S. Von Thun, H.G. Greene, and J.P. Barry. 2003. Environmental impact of the ATOC/Pioneer Seamount submarine cable. www.montereybay.noaa.gov/research/techreports/cablesurveynov2003.pdf. (Accessed 2006.10.04).

Komar, P.D. 1998. Beach Processes and Sedimentation. Princeton, New Jersey: Prentice Hall.

Kondolf, G.M. 1997. Hungry Water: Effects of Dams and Gravel Mining on River Channels. Environmental Management 21(4): 533-551.

Kondolf, G.M., and M.G. Wolman. 1993. The Sizes of Salmonid Spawning Gravels. Water Resources Research 29(7): 2275-2285.

Kondolf, G.M., and R.R. Curry. 1986. Channel Erosion Along the Carmel River, Monterey County, California. Earth Surface Processes and Landforms 11: 307-319.

Kondolf, G.M., J.C. Vick, and T.M. Ramirez. 1996. Salmon Spawning Habitat Rehabilitation on the Merced River, California: An Evaluation of Project Planning and Performance. Transactions of the American Fisheries Society 125(6): 899-912.

Kondolf, G.M., M. Smeltzer, and L. Kimball. 2002. White Paper–Freshwater Gravel Mining and Dredging Issues. Berkeley, California: Prepared for Washington Department of Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Kondolf, M.G., M.J. Sale, and M.G. Wolman. 1993. Modification of Fluvial Gravel Size by Spawning Salmonids. Water Resources Research 29(7): 2265-2274.

Konisky, R.A., D.M. Burdick, M. Dionne, and H.A. Neckles. 2006. A Regional Assessment of Salt Marsh Restoration and Monitoring in the Gulf of Maine. Restoration Ecology 14: 516-525.

Konrad, C.P. 2000. The Frequency and Extent of Hydrologic Disturbances in Streams in the Puget Lowland, Washington. Ph.D. Dissertation Thesis, University of Washington, Seattle, Washington, 212 pp.

Korman, J., and P.S. Higgins. 1997. Utility of Escapement Time Series Data for Monitoring the Response of Salmon Populations to Habitat Alteration. Canadian Journal of Fisheries and Aquatic Sciences 54(9): 2058-2067.

Kosheleva, V. 1992. The impact of air guns used in marine seismic explorations on organisms, living in the Barents Sea. Fisheries and Offshore Petroleum Exploitation 2nd International Conference, Bergen, Norway, 6-8 April 1992. Cited in Turnpenny et al. 1994.

Kozloff, E. 1993. Seashore life of the northern Pacific coast : an illustrated guide to northern California, Oregon, Washington, and British Columbia. University of Washington Press. Seattle, WA.

Kozloff, E. 1983. Seashore life of the northern Pacific coast : an illustrated guide to northern California, Oregon, Washington, and British Columbia. University of Washington Press. Seattle, WA. Book.

Kozlowski, T.T. 2002. Physiological-Ecological Impacts of Flooding on Riparian Forest Ecosystems. Wetlands 22(3): 550-561.

Kramer, D. E. and O’Connell, V.M., 1995. Guide to Northeast Pacific rockfishes: Genera Sebastes and Sebastolobus. Alaska Sea Grant Marine Advisory Bulletin, 25. 78pp.

Kramer, D.E., and V.M. O’Connell. 1995. Guide to Northeast Pacific Rockfishes. Alaska Sea Grant Marine Advisory Bulletin No. 25.

Krone, C.A., D.W. Brown, D.G. Burrows, S. Chan, and U. Varanasi. 1989a. Butyltins in Sediment from Marinas and Waterways in Puget Sound, Washington State, USA. Marine Pollution Bulletin 20(10): 528-531.

Krone, C.A., D.W. Brown, D.G. Burrows, S.L. Chan, and U. Varanasi. 1989b. Tributyltin Contamination of Sediment and English Sole from Puget Sound. Ocean's 98, Seattle, Washington, pp. 545-549.

Krueger, K., P. Chapman, M. Hallock, and T. Quinn. 2007. Some Effects of Suction Dredge Placer Mining on the Short-Term Survival of Freshwater Mussels in Washington Final Report (Draft) USFSW HPA/HCP Grant E-29-HP. Olympia, Washington: Washington Department of Fish and Wildlife Habitat and Fisheries Programs.

Kruer, C.R. 1994. Mapping assessment of vessel damage to shallow seagrasses in the Florida Keys. Final Report for contract 47-10-123-L3. Florida Department of Natural Resources and Floarida Institute of Oceanography, St. Petersburg, Florida. Cited in Dawes et al. 2004.

Krumholz, L.A. 1943. A comparative study of the weberian ossicles in North American Ostariophysine fishes. Copeia 1943(1): 33-40.

Kynard, B., and E. Parker. 2005. Ontogenetic Behavior and Dispersal of Sacramento River White Sturgeon, Acipenser transmontanus, with a Note on Body Color. Environmental Biology of Fishes 74(1): 19-30.

Kynard, B., E. Parker, and T. Parker. 2005. Behavior of early life intervals of Klamath River green sturgeon, Acipenser medirostris with a note on body color. Environmental Biology of Fishes 72(1): 85-97. Available at http://www.springerlink.com/content/v55401j240453410/ (Accessed 2006.10.06).

Laasonen, P., T. Muotka, and I. Kivijarvi. 1998. Recovery of Macroinvertebrate Communities from Stream Habitat Restoration. Aquatic Conservation-Marine and Freshwater Ecosystems 8(1): 101-113.

Lagardère, J.P. 1982. Effects of Noise on Growth and Reproduction of Crangon crangon in Rearing Tanks. Marine Biology 71(2): 177-185.

Lagardère, J.P., and M.R. Régnault. 1980. Influence Du Niveau Sonore De Bruit Ambiant Sur Le Métabolisme De Crangon crangon (Decapoda: Natantia) En Élevage. Marine Biology 57(3): 157-164.

Lagasse, P.F., L.W. Zevenbergen, J.D. Schall, and P.E. Clopper. 2001. Bridge Scour and Stream Instability Countermeasures. Washington D.C. Federal Highway Administration, U.S. Department of Transportation.

Lagler, K.F., A.S. Hazzard, W.E. Hazen, and W.A. Tompkins. 1950. Outboard motors in relation to fish behavior, fish production and angling success. Pp. 280–303 in Transactions of the 15th Annual North American Wildlife Conference. Cited in Carrasquero 2001.

Lake, R.G., and S.G. Hinch. 1999. Acute Effects of Suspended Sediment Angularity on Juvenile Coho Salmon (Oncorhynchus Kisutch). Canadian Journal of Fisheries and Aquatic Sciences 56(5): 862-867.

Lamb, M.P., E. D'Asaro, and J.D. Parsons. 2004. Turbulent Structure of High-Density Suspensions Formed under Waves. Journal of Geophysical Research-Oceans 109(C12).

Lamberti, G.A., S.V. Gregory, L.R. Ashkenas, A.D. Steinman, and C.D. McIntire. 1989. Productive Capacity of Periphyton as a Determinant of Plant Herbivore Interactions in Streams. Ecology 70(6): 1840-1856.

Lampert, W. 1978. Release of Dissolved Organic-Carbon by Grazing Zooplankton. Limnology and Oceanography 23(4): 831-834.

Lancaster, S.T., S.K. Hayes, and G.E. Grant. 2001. Modeling Sediment and Wood Storage and Dynamics in Small Mountainous Watersheds. In Geomorphic Processes and Riverine Habitat, edited by J.B. Dorava, Montgomery, D.R., Palcsak, B., Fitzpatrick, F. Washington DC: American Geophysical Union. pp. 85-102.

Lane, E.W. 1955. The Importance of Fluvial Morphology in Hydraulic Engineering. Proceedings of the American Society of Civil Engineers 81: 745-761.

Lange, M. 1999. Abundance and diversity of fish in relation to littoral and shoreline features. M.S. Thesis. University of Guelph, Guelph, Ontario, Canada. 46 p. plus appendices. In Kahler et al. 2000.

Langer, O.E., B.G. Shepherd, and P.R. Vroom. 1977. Biology of the Nass River eulachon (Thaleichthys pacificus). Department of Fisheries and Environment Canada, Fisheries and Marine Service, Technical Report Series No. PAC, T-77-10. 56 p. Cited in Wilson et al 2006.

Larinier, M. 1998. Small-Scale Hydropower Schemes and Migratory Fish Passage. Houille Blanche-Revue Internationale De L Eau 53(8): 46-51.

Laroche, W.A. and S.L. Richardson. 1981. Development of larvae and juveniles of the rockfishes Sebastes entomelas and S. zacentrus (Family Scorpaenidae) and occurrence off Oregon, with notes on head spines of S. mystinus, S. flavidus, and S. melanops. Fishery Bulletin 79:231-256. Cited in NRC 2001.

Larsen, E.M., E. Rodrick, and R. Milner. 1995. Management recommendations for Washington’s priority species, Volume 1: Invertebrates. Olympia, WA: Washington Department of Fish and Wildlife.

Larson, K.W., and C.E. Moehl. 1990. Entrainment of Anadromous Fish by Hopper Dredge at the Mouth of the Columbia River. September 8-9, 1990. pp. 102-112.

Larson, M.G., D.B. Booth, and S.A. Morley. 2001. Effectiveness of Large Woody Debris in Stream Rehabilitation Projects in Urban Basins. Ecological Engineering 18(2): 211-226.

Latterell, J.J., J.S. Bechtold, T.C. O'Keefe, R. Van Pelt, and R.J. Naiman. 2006. Dynamic Patch Mosaics and Channel Movement in an Unconfined River Valley of the Olympic Mountains. Freshwater Biology 51(3): 523-544.

Lauck, B., R. Swain, and L. Barmuta. 2005. Impacts of Shading on Larval Traits of the Frog Litoria Ewingii in a Commercial Forest, Tasmania, Australia. Journal of Herpetology 39: 478-486.

Laughlin, J. 2004. Underwater Sound Levels Associated with the Construction of the SR 240 Bridge on the Yakima River at Richland. Seattle, Washington: Washington State Department of Transportation, Office of Air Quality and Noise.

Laughlin, J. 2005. Underwater Sound Levels Associated with the Restoration of the Friday Harbor Ferry Terminal. Seattle, Washington: Washington State Department of Transportation.

Laughlin, J. 2006. Underwater sound levels associated with pile driving at the Cape Disappointment boat launch facility, wave barrier project. Washington State Parks Cape Disappointment Wave Barrier Project. Seattle, Washington: Washington Department of Transportation.

Lawler, D.M. 2005. Turbidity and Nephelometry. In Encyclopedia of Analytical Science, edited by P.J. Worsfold, A. Townsend and C.F. Poole. Elsevier. pp. 343-351.

Leary, R.F., and F.W. Allendorf. 1997. Genetic Confirmation of Sympatric Bull Trout and Dolly Varden in Western Washington. Transactions of the American Fisheries Society 126: 715-20.

LeBlanc, R.T., and R.D. Brown. 2000. The Use of Riparian Vegetation in Stream-Temperature Modification. Journal of the Chartered Institution of Water and Environmental Management 14(4): 297-303.

Lebow, S, D. Foster and P. Lebow. 2004. Rate of CCA leaching from commercially treated decking. Forest Products Journal. 54(2): 81-88. Cited in Stratus 2005b.

Lebow, S.T., and M. Tippie. 2001. Guide for Minimizing the Effect of Preservative-Treated Wood on Sensitive Environments. Madison, Wisconsin: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory.

Lee, D.S., C.R. Gilbert, C.H. Hocutt, R.E. Jenkins, D.E. McAllister, and J.R. Stauffer, Jr. 1980. Atlas of North American Freshwater Fishes. North Carolina State Museum of Natural History. 867 pp. Cited in Alaska Natural Heritage Program 2006

Lee, R.F. 1985. Metabolism of Tributyltin Oxide by Crabs, Oysters and Fish. Marine Environmental Research 17: 145-148.

Lefebvre, S., P. Marmonier, G. Pinay, O. Bour, L. Aquilina, and J. Baudry. 2005. Nutrient Dynamics in Interstitial Habitats of Low-Order Rural Streams with Different Bedrock Geology. Archiv Fur Hydrobiologie 164(2): 169-191.

Leidholtbruner, K., D.E. Hibbs, and W.C. Mccomb. 1992. Beaver Dam Locations and Their Effects on Distribution and Abundance of Coho Salmon Fry in 2 Coastal Oregon Streams. Northwest Science 66(4): 218-223.

Lemieux, J.P., J.S. Brennan, M. Farrell, C.D. Levings, and D. Myers. 2004. Proceedings of the DFO/PSAT Sponsored Marine Riparian Experts Workshop. Department of Fisheries and Oceans, Puget Sound Action Team. Tsawwassen, British Columbia: Canadian Manuscript Report of Fisheries and Aquatic Sciences.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. Fluvial Processes in Geomorphology. San Francisco: W. H. Freeman and Company, pp. 156-160.

Leopold, L.B., R. Huppman, and A. Miller. 2005. Geomorphic Effects of Urbanization in Forty-One Years of Observation. Proceedings of the American Philosophical Society 149: 349-371.

Lepori, F., D. Palm, and B. Malmqvist. 2005. Effects of Stream Restoration on Ecosystem Functioning: Detritus Retentiveness and Decomposition. Journal of Applied Ecology 42(2): 228-238.

Lepori, F., D. Palm, E. Brannas, and B. Malmqvist. 2005. Does Restoration of Structural Heterogeneity in Streams Enhance Fish and Macroinvertebrate Diversity? Ecological Applications 15(6): 2060-2071.

Lessard, J.L., and R.W. Merritt. 2006. Influence of Marine-Derived Nutrients from Spawning Salmon on Aquatic Insect Communities in Southeast Alaskan Streams. Oikos 113(2): 334-343.

Levings, C. D. and G. Jamieson. 2001. Marine and estuarine riparian habitats and their role in coastal ecosystems, Pacific region. Canadian Science Advisory Secretariat, Research Document 2001/109. 41 p. Available at: www.dfo-mpo.gc.ca/csas/.

Levy, D.A. and T.G. Northcote. 1981. The distribution and abundance of juvenile salmon in marsh habitats of the Fraser River estuary. Tech. Rep. 25. University of British Columbia, Westwater Research Centre, Vancouver, B.C.

Levy, D.A. and T.G. Northcote. 1982. Juvenile salmon residency in a marsh area of the Fraser River estuary. Canadian Journal of Fisheries and Aquatic Sciences 39:270-276.

Lewis, A.F.J., M.D. McGurk, and M.G. Galesloot. 2002. Alcan's Kemano River eulachon (Thaleichthys pacificus) monitoring program 1988-1998. Consultant's report prepared by Ecofish Research Ltd. for Alcan Primary Metal Ltd., Kitimat, B.C. 136 p. Cited in Wilson et al. 2006.

Li, H.W., C.B. Schreck, and R.A. Tubb. 1984. Comparison of Habitat near Spur Dikes, Continuous Revetments, and Natural Banks for Larval, Juvenile, and Adult fishers of the Willamette River, USGS: Water Resources Research Institute Publication WRRI-95. Corvallis, OR. As cited in Bolton and Shellberg 2001.

Li, M.Z.L., and P.D. Komar. 1992. Longshore Grain Sorting and Beach Placer Formation Adjacent to the Columbia River. Journal of Sedimentary Petrology 62(3): 429-441.

Liang, H.Z., M.P. Lamb, and J.D. Parsons. 2007. Formation of a Sandy near-Bed Transport Layer from a Fine-Grained Bed under Oscillatory Flow. Journal of Geophysical Research-Oceans 112(C2).

Liknes, G.A. and P.J. Graham. 1988. Westslope cutthroat trout in Montana: life history, status and management. Pp. 53–60 in R.E. Gresswell (ed.), Status and management of cutthroat trout. Bethesda, Maryland: American Fisheries Society.

Linbo, T.L., C.M. Stehr, J.P. Incardona, and N.L. Scholz. 2006. Dissolved copper triggers cell death in the peripheral mechanosensory system of larval fish. Environmental Toxicology and Chemistry 25(2): 597-603.

Lisle, T.E., and J. Lewis. 1992. Effects of Sediment Transport on Survival of Salmonid Embryos in a Natural Stream - a Simulation Approach. Canadian Journal of Fisheries and Aquatic Sciences 49(11): 2337-2344.

Lister, D.B. and R.J. Finnigan. 1997. Rehabilitating off-channel habitat. In. P. A. Slaney and D. Zaldokas. Fish Habitat Rehabilitation Procedures. Watershed Restoration Technical Circular No. 9. Watershed Restoration Program, Vancouver, BC.

Lister, D.B., R.J. Beniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of streambank riprap. Pages 621-632 In Thorne, C. R., S. R. Abt, F. B. J. Barends, S. T. Maynord and K. W. Pilarczyk, editors. River, coastal and shorleline protection: erosion control using riprap and armourstone. John Wiley & Sons, Ltd.

Liu, H.Y., S.K. Zhang, Z.F. Li, X.G. Lu, and Q. Yang. 2004. Impacts on Wetlands of Large-Scale Land-Use Changes by Agricultural Development: The Small Sanjiang Plain, China. Ambio 33(6): 306-310.

Livingston, P.A. 1991. Food habitats and population level consumption of groundfish. Pages 9-88 in P.A. Livingston (ed.), Groundfish food habitats and predation on commercially important prey species in the Eastern Bering Sea from 1984 to 1986. U.S. Department of Commerce, NOAA Technical Memorandum NMFS F/NWC-207. Seattle, Washington. Cited in NRC 2001.

Lloyd, D.S. 1987. Turbidity as a Water Quality Standard for Salmonid Habitats in Alaska. North American Journal of Fisheries Management 7: 34-45.

Loflin, R.L. 1995. The Effects of Docks on Seagrass Beds in the Charlotte Harbor Estuary, Florida. Scientists 58: 198-205.

Loge, F.J., M.R. Arkoosh, T.R. Ginn, L.L. Johnson, and T.K. Collier. 2005. Impact of Environmental Stressors on Dynamics of Disease Transmission. Environmental Science and Technology 39: 7329-7336.

Long, E.R. and L.G. Morgan. 1991. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration, Seattle, WA. (Appendix L; as cited in MacDonald et al., 2000b.) Cited in Stratus 2005b.

Lorang, M.S., J.A. Stanford, F.R. Hauer, and J.H. Jourdonnais. 1993. Dissipative and Reflective Beaches in a Large Lake and the Physical Effects of Lake Level Regulation. Ocean & Coastal Management 19(3): 263-287.

Losada, I.J., J.L. Lara, E.D. Christensen, and N. Garcia. 2005. Modeling of velocity and turbulence fields around and within low-crested rubble-mound breakwaters. Coastal Engineering 52(10-11): 887-913.

Love, M.S. 1991. Probably more than you want to know about the fishes of the Pacific coast. Really Big Press, Santa Barbara, California. 215p.

Love, M.S., and A. York. 2005. A comparison of the fish assemblages associated with an oil/gas pipeline and adjacent seafloor in the Santa Barbara Channel, southern California bight. Bulletin of Marine Science 77(1): 101-117.

Love, M.S., and A. York. 2006. The Relationships between Fish Assemblages and the Amount of Bottom Horizontal Beam Exposed at California Oil Platforms: Fish Habitat Preferences at Man-Made Platforms and (by Inference) at Natural Reefs. Fishery Bulletin 104(4): 542-549.

Love, M.S., D.M. Schroeder, W. Lenarz, A. MacCall, A.S. Bull, and L. Thorsteinson. 2006. Potential Use of Offshore Marine Structures in Rebuilding an Overfished Rockfish Species, Bocaccio (Sebastes paucispinis). Fishery Bulletin 104(3): 383-390.

Love, M.S., P. Morris, M. McCrae, and R. Collins. 1990. Life history aspects of 19 rockfish species (Scorpaenidae: Sebastes) from the southern California bight. NOAA, NMFS Tech. Rept. No. 87. 38pp. Cited in NRC 2001.

Ludwig, M., D. Rusanowsky, and C. Johnson-Hughes. 1997. The impact of installation and use of a pier and dock assembly on eelgrass (Zostera marina) at Star Island, Montauk NY: Kalikow Dock Study., NMFS. USFWS. Cited in Nightingale and Simenstad 2001b.

Luecke, C., and W.A. Wurtsbauch. 1993. Effects of Moonlight and Daylight on Hydroacoustic Estimates of Pelagic Fish Abundance. Transactions of the American Fisheries Society 122: 112-120.

Lund, J.A. 1976. Evaluation of stream channelization and mitigation on fisheries resources of the St. Regis River, Montaana, U.S. Fish and Wildlife Service; FWS/OBS-76/06.

Luning, K. 1981. Light. Pp. 326-355 in C.S. Lobban and M.J. Wynne (eds.), The Biology of Seaweeds. Oxford: Blackwell Sci. Publ.

Lunz, J. 1985. An Analysis of Available Information Concerning the Entrainment of Oyster Larvae During Hydraulic Cutterhead Dredging Operations with Commentary on the Reasonableness of Seasonally Restricting Dredging Windows. Vicksburg, Mississippi: U.S. Army Corps of Engineers Waterways Experiment Station.

Lussier, S.M., J.H. Gentile, and J. Walker. 1985. Acute and chronic effects of heavy metals and cyanide on Mysidopsis bahia (Crustacea:Mysidacea). Aquatic Toxicology 7(1-2):25-35. Cited in Stratus 2005b.

Lyons, T., J.A. Ickes, V.S. Magar, C.S. Albro, L. Cumming, B. Bachman, T. Fredette, T. Myers, M. Keegan, K. Marcy, and O. Guza. 2006. Evaluation of Contaminant Resuspension Potential During Cap Placement at Two Dissimilar Sites. Journal of Environmental Engineering-ASCE 132(4): 505-514.

MacAvoy, S.E., S.A. Macko, S.P. McIninch, and G.C. Garman. 2000. Marine Nutrient Contributions to Freshwater Apex Predators. Oecologia 122(4): 568-573.

MacBroom, J.R. 1998. The River Book - The Nature of Streams in Glaciated Terrains. Hartford, Connecticut: Connecticut Department of Environmental Protection.

MacDonald, A. 1988. Predicting Channel Recovery from Sand and Gravel Extraction in the Naugatuck River and Adjacent Floodplain. Proceedings of the National Conference, Colorado Springs, Colorado, pp. p.702-707.

MacDonald, D.D. 1994. Approach to the Assessment of Sediment Quality in Florida Coastal Waters. Volume I: Development and Evaluation of Sediment Quality Assessment Guidelines. Prepared for Florida Department of Environmental Protection. November. Cited in Stratus 2005b.

MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000. Development and Evaluation of Consensus-Based Sediment Quality Guidelines for Freshwater Ecosystems. Archives of Environmental Contamination and Toxicology 39(1): 20-31. Cited in Stratus 2005b.

MacDonald, D.D., T. Berger, K. Wood, J. Brown, T. Johnsen, M.L. Haines, K. Brydges, M.J. MacDonald, S.L. Smith, and D.P. Shaw. 2000b. A Compendium of Environmental Quality Benchmarks. GBE/EC-99-001. Prepared for Environment Canada. Cited in Long and Morgan 1991 via Stratus 2005b.

MacDonald, J.S., E.A. MacIsaac, and H.E. Herunter. 2003. The Effect of Variable-Retention Riparian Buffer Zones on Water Temperatures in Small Headwater Streams in Sub-Boreal Forest Ecosystems of British Columbia. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 33(8): 1371-1382.

MacDonald, K.B., D. Simpson, B. Paulsen, J. Cox, and J. Gendron. 1994. Shoreline armoring effects on the physical coastal processes in Puget Sound, Washington. Coastal Erosion Management Studies, Volume 5. Shorelands and Coastal Zone Management Program, Washington Department of Ecology, Olympia, Washington.

Mace, P.M. 1983. Predatory-prey functional responses and predation by staghorn sculpins, Leptocottus armatus on chum salmon, Oncorhynchus keta. Doctoral dissertation, University of British Columbia, Vancouver, British Columbia. Cited in Tabor et al. 1998.

Maclennan, A. 2005. An Analysis of Large Woody Debris in Two Puget Sound Salt Marshes; Elger Bay, Camano Island, Sullivan Minor Marsh, Padilla Bay. Master's Thesis, Western Washington University, Bellingham, Washington.

Madej, M.A., and V. Ozaki. 1996. Channel Response to Sediment Wave Propagation and Movement, Redwood Creek, California, USA. Earth Surface Processes and Landforms 21(10): 911-927.

Magilligan, F.J., and K.H. Nislow. 2005. Changes in Hydrologic Regime by Dams. Geomorphology 71(1-2): 61-78.

Malins, D.C., B.B. McCain, D.W. Brown, S.L. Chan, M.S. Myers, J.T. Landahl, P.G. Prohaska, A.J. Friedman, L.D. Rhodes, D.G. Burrows, W.D. Gronlund, and H.O. Hodgins. 1984. Chemical Pollutants in Sediments and Diseases of Bottom-Dwelling Fish in Puget Sound, Washington. Environmental Science and Technology 18(9): 705-713.

Mamelona, J., and T. Pelletier. 2003. Butyltins Biomagnification from Macroalgae to Green Sea Urchin: A Field Assessment. Applied Organometallic Chemistry 17(10): 759-766.

Manashe, E. 1993. Vegetation management: A guide for Puget Sound bluff property owners. Shorelands and Coastal Zone Management Program. Washington Department of Ecology, Olympia, Washington.

Manga, M., and J.W. Kirchner. 2000. Stress partitioning in streams by large woody debris. Water Resources Research 36, 2373-2379.

Margolis, B.E., M.S. Castro, and R.L. Raesly. 2001. The Impact of Beaver Impoundments on the Water Chemistry of Two Appalachian Streams. Canadian Journal of Fisheries and Aquatic Sciences 58(11): 2271-2283.

Margolis, B.E., R.L. Raesly, and D.L. Shumway. 2001. The Effects of Beaver-Created Wetlands on the Benthic Macroinvertebrate Assemblages of Two Appalachian Streams. Wetlands 21(4): 554-563.

Marine, K.R., and J.J. Cech. 2004. Effects of High Water Temperature on Growth, Smoltification, and Predator Avoidance in Juvenile Sacramento River Chinook Salmon. North American Journal of Fisheries Management 24(1): 198-210.

Marsden, J.E., and M.A. Chotkowski. 2001. Lake Trout Spawning on Artificial Reefs and the Effect of Zebra Mussels: Fatal Attraction? Journal of Great Lakes Research 27(1): 33-43.

Marshall, D.W., M. Otto, J.C. Panuska, S.R. Jaeger, D. Sefton, and T.R. Baumberger. 2006. Effects of Hypolimnetic Releases on Two Impoundments and Their Receiving Streams in Southwest Wisconsin. Lake and Reservoir Management 22(3): 223-232.

Martens, D.W., and J.A. Servizi. 1993. Suspended Sediment Particles inside Gills and Spleens of Juvenile Pacific Salmon (Oncorhynchus Spp.). Canadian Journal of Fisheries and Aquatic Sciences 50: 586-590.

Martin, D.J. 2001. The Influence of Geomorphic Factors and Geographic Region on Large Woody Debris Loading and Fish Habitat in Alaska Coastal Streams. North American Journal of Fisheries Management 21(3): 429-440.

Maser, C., and J.R. Sedell. 1994. From the Forest to the Sea, the Ecology of Wood in Streams, Rivers, Estuaries, and Oceans. Delray Beach, Florida: St. Lucie Press.

Mason, C., W. Grogg, and S. Wheeler. 1993. Shoreline Erosion Study. Pleasure Island, Texas: U.S. Army Corps of Engineers.

Massong, T.M., and D.R. Montgomery. 2000. Influence of Sediment Supply, Lithology, and Wood Debris on the Distribution of Bedrock and Alluvial Channels. Geological Society of America Bulletin 112(4): 591-599.

Matishov, G.G. 1992. The reaction of bottom-fish larvae to airgun pulses in the context of the vulnerable Barents Sea ecosystem. Fisheries and Offshore Petroleum Exploitation 2nd International Conference, Bergen, Norway, 6-8 April 1992. Cited in Turnpenny et al. 1994.

Matthews, K.M. 1987. Habitat Utilization by Recreationally-Important Bottomfish in Puget Sound: An Assessment of Current Knowledge and Future Needs No. 264. Washington Department of Fisheries Progress Report.

Matthews, K.R. 1990a. A comparative study of habitat use by young-of-the-year, and adult rockfishes on four habitat types in central Puget Sound. Fishery Bulletin 88:223-239. Cited in NRC 2001.

Matthews, K.R. 1990b. An experimental study of the habitat preferences and movement patterns of copper, quillback, and brown rockfishes (Sebastes spp.). Environmental Biology of Fishes 29:161-178. Cited in NRC 2001.

Maudlin, M., T. Coe, N. Currence and J. Hansen. 2002. South Fork Nooksack River Acme-Saxon Reach Restoration Planning: Analysis of Existing Information and Preliminary Recommendations. Deming, WA: Lummi Natural Resources.

May, C.W. 1998. Assessment of the Cumulative Effects of Urbanization on Small Streams in the Puget Sound Lowland Ecoregion: Implications for Salmonid Resource Management. Seattle, Washington: University of Washington.

May, C.W. 2003. Stream-Riparian Ecosystems in the Puget Sound Lowland Eco-Region: A Review of Best Available Science. Watershed Ecology LLC.

Mayer, P., S. Reynolds, T.J. Canfield, and M.D. McCutchen. 2005. Riparian Buffer Width, Vegetative Cover, and Nitrogen Removal Effectiveness: A Review of Current Science and Regulations. EPA/600/R-05/118. Ada, Oklahoma: U.S. Environmental Protection Agency.

Mayfield, R.B., and J.J. Cech. 2004. Temperature Effects on Green Sturgeon Bioenergetics. Transactions of the American Fisheries Society 133(4): 961-970.

Mazur, M.M., and D.A. Beauchamp. 2003. A Comparison of Visual Prey Detection among Species of Piscivorous Salmonids: Effects of Light and Low Turbidities. Environmental Biology of Fishes 67(4): 397-405.

MBC Applied Environmental Sciences. 1987. Ecology of Important Fisheries Species Offshore California. Minerals Management Service, MMS 86-0093, Pacific Outer Continental Shelf Region. Washington, D.C. 252 pp. Cited in NRC 2001.

McAllister, T.L., M.F. Overton, and E.D. Brill. 1996. Cumulative Impact of Marinas on Estuarine Water Quality. Environmental Management 20(3): 385-396.

McAnally, W.H., J.F. Haydel, and G. Savant. 2004. Port sedimentation solutions for the Tennessee-Tombigbee Waterway in Mississippi. Ports and Waterways Division Mississippi Department of Transportation. http://www.gomdot.com/research/pdf/SS117.pdf, accessed 2006.12.06.

McAnally, W.H., J.F. Haydel, and G. Savant. 2004. Port Sedimentation Solutions for the Tennessee-Tombigbee Waterway in Mississippi. Department of Civil Engineering, James Worth Bhagley College of Engineering, Mississippi State University. 121 p.

Mccain, B.B., D.C. Malins, M.M. Krahn, D.W. Brown, W.D. Gronlund, L.K. Moore, and S.L. Chan. 1990. Uptake of Aromatic and Chlorinated Hydrocarbons by Juvenile Chinook Salmon (Oncorhynchus tshawytscha) in an Urban Estuary. Archives of Environmental Contamination and Toxicology 19(1): 10-16.

McClain, M.E., R.E. Bilby, and F.J. Triska. 1998. Nutrient Cycles and Responses to Disturbance. In River Ecology and Management. Lessons from the Pacific Coastal Ecoregion, edited by R.J. Naiman and B. R.E. New York: Springer-Verlag.

McCool, W.W., and J.D. Parsons. 2004. Sedimentation from Buoyant Fine-Grained Suspensions. Continental Shelf Research 24(10): 1129-1142.

McCormick, S.D., R.A. Cunjak, B. Dempson, M.F. O'Dea, and J.B. Carey. 1999. Temperature-Related Loss of Smolt Characteristics in Atlantic Salmon (Salmo Salar) in the Wild. Canadian Journal of Fisheries and Aquatic Sciences 56(9): 1649-1658.

McCullough, M.C., S. Spalding, and D. Sturdevant. 2001. Summary of Technical Literature Examining the Physiological Effects of Temperature on Salmonids. EPA-910-D-01-005. Washington, DC: U.S. Environmental Protection Agency.

McDonald, J. 1960. The behavior of Pacific salmon fry during their downstream migration to freshwater and saltwater nursery areas. Journal Fishery Research Board of Canada 17: 665-76. Cited in Tabor et al. 1998, and in Nightingale and Simenstad 2001b.

McDonough, K.M., P. Murphy, J. Olsta, Y.W. Zhu, D. Reible, and G.V. Lowry. 2007. Development and Placement of a Sorbent-Amended Thin Layer Sediment Cap in the Anacostia River. Soil & Sediment Contamination 16(3): 313-322.

McDowell, P.F. 2001. Spatial variations in channel morphology at segment and reach scales, Middle Fork John Day River, Northeastern Oregon. Geomorphic Processes and Riverine Habitat Water Science and Applications 4: 159-172. Cited in Brim Box et al. 2004.

McElhany, P., M.H. Ruckelshaus, M.J. Ford, T.C. Wainwright, and E.P. Bjorkstedt. 2000. Viable Salmonid Populations and the Recovery of Evolutionarily Significant Units. U.S. Department of Commerce, National Oceanographic and Atmospheric Administration, National Marine Fisheries Service.

McFarland, W.N., and F.W. Munz. 1975. Part II: The Photic Environment of Clear Tropical Seas During the Day and Part III: The Evolution of Photopic Visual Pigments in Fishes. Vision Resources 15: 1063-1080.

McFarlane, G.A., and R.J. Beamish. 1986. Biology and Fishery of Pacific Hake Merluccius Productus in the Strait of Georgia. International North Pacific Fisheries Commission Bulletin 50: 365-392. Cited in NRC 2001.

McGraw, K.A., and D.A. Armstrong. 1990. Fish Entrainment by Dredges in Grays Harbor, Washington. Effects of Dredging on Anadromous Pacific Coast Fishes, pp. 113-131.

McGurk, M.D. 1986. Natural Mortality of Marine Pelagic Fish Eggs and Larvae: Role of Spatial Patchiness. Marine Ecology Progress Series 34: 227-242.

McHenry, M.L., D.C. Morrill, and E. Currence. 1984. Spawning gravel quality, watershed characteristics and early life history survival of coho salmon and steelhead in five North Olympic Peninsula watersheds. April 1984.

McIntire, C.D. 1973. Periphyton Dynamics in Laboratory Streams: A Simulation Model and Its Implications. Ecological Monographs 43: 399-420.

McIntosh, M.D., M.E. Benbow, and A.J. Burky. 2002. Effects of Stream Diversion on Riffle Macroinvertebrate Communities in a Maui, Hawaii, Stream. River Research and Applications 18(6): 569-581.

McKinnell, S., J.J. Pella, and M.L. Dahlberg. 1997. Populations-specific aggregations of steelhead trout (Oncorhynchus mykiss) in the North Pacific Ocean. Canadian Journal of Fisheries and Aquatic Sciences 54: 2368-2376. Cited in WDNR 2006a.

McLaughlin, R. 2006. Research to Guide Use of Barriers, Traps, and Fishways to Control Sea Lamprey. Prepared for Sea Lamprey Research Program, Great Lakes Fishery Commission by Department of Zoology - University of Guelph.

McLeay, D.J., G.L. Ennis, I.K. Birtwell, and G.F. Hartman. 1984. Effects On Arctic Grayling (Thymallus arcticus) of Prolonged Exposure to Yukon Placer Mining Sediment: A Laboratory Study. Canadian Technical Report of Fisheries and Aquatic Sciences 1241.

McLeay, D.J., I.K. Birtwell, G.F. Hartman, and G.L. Ennis. 1987. Responses of Arctic Grayling (Thymallus Arcticus) to Acute and Prolonged Exposure to Yukon Placer Mining Sediment. Canadian Journal of Fisheries and Aquatic Sciences 44(3): 658-673.

McMahon, T.E., A.V. Zale, F.T. Barrows, J.H. Selong, and R.J. Danehy. 2007. Temperature and Competition between Bull Trout and Brook Trout: A Test of the Elevation Refuge Hypothesis. Transactions of the American Fisheries Society 136: 1313-1326.

McMichael, G., A.L. Fritts, T.N. Pearson. 1999. Lower Yakima River Predatory Fish Monitoring: Progress Report 1988. Oregon Department of Fish and Wildlife Workshop, Portland, Oregon, October 7-28, 1999.

McMichael, G.A., and M.A. Chamness. 2001. Walla Walla River Basin Fish Screen Evaluations, 2001: Burlingame and Little Walla Walla Sites. DOE/BP-00000652-7. Portland, Oregon: Bonneville Power Administration.

McMichael, G.A., J.A. Vucelick, C.S. Abernethy, and D.A. Neitzel. 2004. Comparing Fish Screen Performance to Physical Design Criteria. Fisheries 29(7): 10-16.

McMichael, G.A., L. Fritts, and T. N. Pearsons. 1998. Electrofishing Injury to Stream Salmonids; Injury Assessment at the Sample, Reach, and Stream Scales. North American Journal of Fisheries Management 18:894-904.

McNabb, C.D., C.R. Liston, and S.M. Borthwick. 2003. Passage of Juvenile Chinook Salmon and Other Fish Species through Archimedes Lifts and a Hidrostal Pump at Red Bluff, California. Transactions of the American Fisheries Society 132(2): 326-334.

Mcrae, G., and C.J. Edwards. 1994. Thermal-Characteristics of Wisconsin Headwater Streams Occupied by Beaver - Implications for Brook Trout Habitat. Transactions of the American Fisheries Society 123(4): 641-656.

Meador, J.P., J.E. Stein, W.L. Reichert, and U. Varansi. 1995. Bioaccumulation of Polycyclic Aromatic Hydrocarbons by Marine Organisms. Reviews of Environmental Contamination Toxicology 143: 79-165.

Meadows, G.A., S.D. Mackey, R.R. Goforth, D.M. Mickelson, T.B. Edil, J. Fuller, D.E. Guy, L.A. Meadows, and E. Brown. 2005. Cumulative Habitat Impacts of Nearshore Engineering. Journal of Great Lakes Research 31: 90-112.

Meals, K.O., and L.E. Miranda. 1991. Variability in Abundance of Age-0 Centrarchids among Littoral Habitats of Flood Control Reservoirs in Mississippi. North American Journal of Fisheries Management 11(3): 298-304.

Meehan, W.R. (ed.). 1991. Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats. Special Publication 19. Bethesda, Maryland: American Fisheries Society.

Meeuwig, M.H., J.M. Bayer, and J.G. Seelye. 2005. Effects of Temperature on Survival and Development of Early Life Stage Pacific and Western Brook Lampreys. Transactions of the American Fisheries Society 134(1): 19-27.

Meier, A.H., and N.D. Horseman. 1977. Stimulation and Depression of Growth, Fat Storage, and Gonad Weight by Daily Stimulus in the Teolost Fish, Tilapia Aurea. Eighth Annual Meeting World Mariculture Society, January 9-13, 1977. pp. 135-146.

Meldgaard, T., E.E. Nielsen, and V. Loeschcke. 2003. Fragmentation by Weirs in a Riverine System: A Study of Genetic Variation in Time and Space among Populations of European Grayling (Thymallus Thymallus) in a Danish River System. Conservation Genetics 4(6): 735-747.

Melo, E., and R.T. Guza. 1991. Wave-Propagation in Jettied Entrance Channels. II: Observations. Journal of Waterway Port Coastal and Ocean Engineering-ASCE 117(5): 493-510.

Menzie, C.A., S.S. Hoeppner, J.J. Cura, J.S. Freshman, and E.N. LaFrey. 2002. Urban and suburban storm water runoff as a source of polycyclic aromatic hydrocarbons (PAHs) to Massachusetts estuarine and coastal environments. Estuaries 25(2): 165-176.

Merz, J.E., and J.D. Setka. 2004. Evaluation of a Spawning Habitat Enhancement Site for Chinook Salmon in a Regulated California River. North American Journal of Fisheries Management 24(2): 397-407.

Merz, J.E., and L.K.O. Chan. 2005. Effects of Gravel Augmentation on Macroinvertebrate Assemblages in a Regulated California River. River Research and Applications 21(1): 61-74.

Merz, J.E., and P.B. Moyle. 2006. Salmon, Wildlife, and Wine: Marine-Derived Nutrients in Human-Dominated Ecosystems of Central California. Ecological Applications 16(3): 999-1009.

Merz, J.E., J.D. Setka, G.B. Pasternack, and J.M. Wheaton. 2004. Predicting Benefits of Spawning-Habitat Rehabilitation to Salmonid (Oncorhynchus Spp.) Fry Production in a Regulated California River. Canadian Journal of Fisheries and Aquatic Sciences 61(8): 1433-1446.

Mesa, M.G. 1994. Effects of Multiple Acute Stressors on the Predator Avoidance Ability and Physiology of Juvenile Chinook Salmon. Transactions of the American Fisheries Society 123(5): 786-793.

Mesa, M.G., and T.M. Olson. 1993. Prolonged Swimming Performance of Northern Squawfish. Transactions of the American Fisheries Society 122(6): 1104-1110.

Mesick, C. 2002. Knight's Ferry Gravel Replenishment Project. Prepared for CALFED Bay Delta Program by Carl Mesick Consultants, El Dorado, California.

Michael, H.A., A.E. Mulligan, and C.F. Harvey. 2005. Seasonal Oscillations in Water Exchange between Aquifers and the Coastal Ocean. Nature 436(7054): 1145-1148.

Michael, J.H. 2003. Nutrients in Salmon Hatchery Wastewater and Its Removal through the Use of a Wetland Constructed to Treat Off-Line Settling Pond Effluent. Aquaculture 226(1-4): 213-225.

Michelsen, T.C., C.D. Boatman, D. Norton, C.C. Ebbesweyer, T. Floyd, and M.C. Francisco. 1999. Resuspension and Transport of Contaminated Sediments Along the Seattle Waterfront. Part 1: Field Investigations and Conceptual Model Report 99-335. Olympia, Washington: Washington Department of Ecology.

Michny, F. and R. Deibel. 1986. Sacramento River Chico Landing to Red Bluff Project 1985 juvenile salmon study. USDI, FWS, Sacramento, CA. Prepared for USACOE, Sacramento District. 13 pp.

Mickett, J.B., M.C. Gregg, and H.E. Seim. 2004. Direct Measurements of Diapycnal Mixing in a Fjord Reach - Puget Sound's Main Basin. Estuarine Coastal and Shelf Science 59(4): 539-558.

Miles J., P. Russell, and D. Huntley. 2001. Field Measurement of Sediment Dynamics in Front of a Seawall. Journal of Coastal Research. 17: 195-206.

Millar, R.G., and M.C. Quick. 1998. Stable Width and Depth of Gravel-Bed Rivers with Cohesive Banks. Journal of Hydraulic Engineering-ASCE 124(10): 1005-1013.

Miller, B.S., C.A. Simenstad, and L.R. Moulton. 1976. Puget Sound Baseline Program: Nearshore Fish Survey, Annual Report July 1974-September 1975. Report No. 76-04. Seattle, Washington: Fisheries Research Institute, University of Washington.

Miller, D. E., P. B. Skidmore, and D. J. White. 2001. Channel design. Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Miller, D.E., and P.B. Skidmore. 2003. Establishing a standard of practice for natural channel design using design criteria. Restoration of Puget Sound Rivers, D. R. Montgomery, S. Bolton, D. B. Booth, and L. Wall, eds., University of Washington Press, Seattle, 340–360.

Miller, D.E., P.B. Skidmore, and D.J. White. 2001. Channel Design White Paper. Olympia, Washington: Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Miller, D.J. 1972. Changes in creosote content of immersed wood. Forest Products Journal 22(3):25-31. Cited in Cooper 1991.

Miller, D.J. 1977. Loss of creosote from Douglas-fir marine piles. Forest Products Journal 27(11):28-33. Cited in Cooper 1991.

Miller, D.J., and L.E. Benda. 2000. Effects of Punctuated Sediment Supply on Valley-Floor Landforms and Sediment Transport. Geological Society of America Bulletin 112(12): 1814-1824.

Miller, D.R., R.K.L. Emmett, and S.A. Hinton. 1990. A Preliminary Survey of Benthic Invertebrates in the Vicinity of the Coos Bay Oregon, Navigation Channel: Coastal Zone and Estuarine Studies. Prepared for U.S. Army Corps of Engineers by Northwest Fisheries Science Center, National Marine Fisheries Center, Seattle, Washington.

Miller, M.C., I.N. McCave, and P.D. Komar. 1977. Threshold of Sediment Motion under Unidirectional Currents. Sedimentology 24(4): 507-527.

Miller, M.C., R.M. Thom, G.D. Williams, J.A. Southard, S.L. Blanton, and L.K. O'Rourke. 2001. Effects of Shoreline Hardening and Shoreline Protection Features on Fish Utilization and Behavior, Washaway Beach, Washington. PNNL-13635-(2). Richland, Washington: Pacific Northwest National Lab.

Miller, S.W., D. Wooster, and J. Li. 2007. Resistance and Resilience of Macroinvertebrates to Irrigation Water Withdrawals. Freshwater Biology 52: 2494-2510.

Mills, K.E., and M.S. Fonseca. 2003. Mortality and Productivity of Eelgrass Zostera Marina under Conditions of Experimental Burial with Two Sediment Types. Marine Ecology-Progress Series 255: 127-134.

Minakawa, N., and G.F. Kraft. 2005. Homing Behavior of Juvenile Coho Salmon (Oncorhynchus Kisutch) within an Off-Channel Habitat. Ecology of Freshwater Fish 14(2): 197-201.

Minakawa, N., R.I. Gara, and J.M. Honea. 2002. Increased Individual Growth Rate and Community Biomass of Stream Insects Associated with Salmon Carcasses. Journal of the North American Benthological Society 21(4): 651-659.

Misitano, D.A., E. Casillas, and C.R. Haley. 1994. Effects of Contaminated Sediments on Viability, Length, DNA and Protein-Content of Larval Surf Smelt, Hypomesus pretiosus. Marine Environmental Research 37(1): 1-21.

Mitchell, N.L., and G.A. Lamberti. 2005. Responses in Dissolved Nutrients and Epilithon Abundance to Spawning Salmon in Southeast Alaska Streams. Limnology and Oceanography 50(1): 217-227.

Mitsch, W.J., and J.G. Gosselink. 2000. Wetlands. New York: John Wiley.

Mitson, R.B., and H.P. Knudsen. 2003. Causes and effects of underwater noise on fish abundance estimation. Aquatic Living Resources. 16 (3): 255-263. July 2003.

MOEE (Ontario Ministry of the Environment and Energy). 1995. Guidelines for Evaluating Construction Activities Impacting on Water Resources. Guideline B-6. Toronto, Ontario, Canada.

Moerke, A.H., and G.A. Lamberti. 2004. Restoring Stream Ecosystems: Lessons from a Midwestern State. Restoration Ecology 12(3): 327-334.

Mohlenberg, F., and T. Kiorboe. 1981. Growth and Energetics in Spisula subtruncata (Da Costa) and the Effect of Suspended Bottom Material. Ophelia 20(1): 79-90.

Molash, E. 2001. Washington State Department of Transportation. Personal communication. Cited in Bash et al. 2001.

Mongillo, P.E. and M. Hallock. 1998. Washington State Status Report for the Margined Sculpin. Washington Department of Fish and Wildlife. September 1998.

Mongillo, P.E. and M. Hallock. 1999. Washington State Status Report for the Olympic Mudminnow. Olympia, WA: Washington Department of Fish and Wildlife. 36 pp. http://wdfw.wa.gov/wlm/diversty/soc/status/mudmin/omudmin.pdf, accessed 2006.10.04.

Montgomery, D.R. 2000. Coevolution of the Pacific Salmon and Pacific Rim Topography. Geology 28(12): 1107-1110.

Montgomery, D.R. and J.M. Buffington. 1993. Channel classification, prediction of channel response, and assessment of channel condition. Timber, Fish & Wildlife TFW-SH10-93-002. http://www.stage.dnr.wa.gov/forestpractices/adaptivemanagement/cmer/publications/TFW_SH10_93_002.pdf, accessed 2006.10.04.

Montgomery, D.R. and J.M. Buffington. 1998. Channel processes, classification, and response, in River Ecology and Management, edited by R. Naiman and R. Bilby, Springer-Verlag, New York, NY, pp. 13-42.

Montgomery, D.R., and J.R. Buffington. 1997. Channel-Reach Morphology in Mountain Drainage Basins. Geological Society of America Bulletin 109: 596-611.

Montgomery, D.R., B.D. Collins, J.M. Buffington, and T.B. Abbe. 2003. Geomorphic Effects of Wood in Rivers. In The Ecology and Management of Wood in World Rivers, American Fisheries Society Symposium 37, edited by S.V. Gregory, K.L. Boyer and A.M. Gurnell. Bethesda, Maryland: American Fisheries Society. pp. 21-48.

Montgomery, D.R., E.M. Beamer, G.R. Pess, and T.P. Quinn. 1999. Channel Type and Salmonid Spawning Distribution and Abundance. Canadian Journal of Fisheries and Aquatic Science 56: 377-387.

Montgomery, D.R., J.M. Buffington, N.P. Peterson, D. Schuett-Hames, and T.P. Quinn. 1996. Stream-Bed Scour, Egg Burial Depths, and the Influence of Salmonid Spawning on Bed Surface Mobility and Embryo Survival. Canadian Journal of Fisheries and Aquatic Sciences 53: 1061-1070.

Montgomery, D.R., J.M. Buffington, R.D. Smith, K.M. Schmidt, and G. Pess. 1995. Pool Spacing in Forest Channels. Water Resources Research 31(4): 1097-1105.

Montgomery, D.R., S. Bolton, D.B. Booth, and L. Wall. 2003. Restoration of Puget Sound Rivers. Seattle, Washington: Center for Water and Watershed Studies in association with University of Washington Press.

Montgomery, D.R., T.B. Abbe, N.P. Peterson, J.M. Buffington, K. Schmidt, and J.D. Stock. 1996. Distribution of bedrock and alluvial channels in forested mountain drainage basins. Nature 381, 587-589.

Moore, H.L., and H.W. Newman. 1956. Effects of Sound Waves on Young Salmon. Special Science Report-Fisheries 172. U.S. Fish and Wildlife Service, 19p.

Moore, J.W., D.E. Schindler, J.L. Carter, J. Fox, J. Griffiths, and G.W. Holtgrieve. 2007. Biotic Control of Stream Fluxes: Spawning Salmon Drive Nutrient and Matter Export. Ecology 88(5): 1278-1291.

Moore, K.A. 2004. Influence of Seagrasses on Water Quality in Shallow Regions of the Lower Chesapeake Bay. Journal of Coastal Research 20: 162-178.

Moore, M.V., S.J. Kohler, and M.S. Cheers. 2006. Artificial Light at Night in Freshwater Habitats and Its Potential Ecological Effects. In Ecological Consequences of Artificial Night Lighting, edited by C. Rich and T. Longcore. Island Press. pp. 365-384.

Morioka, T., and H. Kuwada. 2002. The Upper Limit of Inhabiting Temperature and the Diet of Juvenile Pacific Cod Gadus Macrocephalus in the Northern Part of Nanao Bay and Its Vicinity, Japan. Nippon Suisan Gakkaishi 68(3): 345-350.

Mork, O. I. and J. Gulbrandsen. 1994. Vertical activity of four salmonid species in response to changes between darkness and two intensities of light. Aquacult. 127: 317-28. Cited in Simenstad et al. 1999, in Nightingale and Simenstad 2001b.

Morley, S.A., P.S. Garcia, T.R. Bennett, and P. Roni. 2005. Juvenile Salmonid (Oncorhynchus spp.) Use of Constructed and Natural Side Channels in Pacific Northwest Rivers. Canadian Journal of Fishery Aquatic Science 62(12): 2811-2821.

Morton, J.W. 1977. Ecological Effects of Dredging and Dredge Spoil Disposal: A Literature Review. Technical Paper 94. U.S. Fish and Wildlife Service, 33 p.

Moschella, P.S., M. Abbiati, P. Aberg, L. Airoldi, J.M. Anderson, F. Bacchiocchi, F. Bulleri, G.E. Dinesen, M. Frost, E. Gacia, L. Granhag, P.R. Jonsson, M.P. Satta, A. Sundelof, R.C. Thompson, and S.J. Hawkins. 2005. Low-Crested Coastal Defense Structures as Artificial Habitats for Marine Life: Using Ecological Criteria in Design. Coastal Engineering 52(10-11): 1053-1071.

Moser, H.G., N.C.H. Lo, and P.E. Smith. 1997. Vertical Distribution of Pacific Hake Eggs in Relation to Stage of Development and Temperature. California Cooperative Oceanic Fisheries Investigations Reports 38: 120-126.

Moser, M.L., A.M. Darazsdi, and J.R. Hall. 2000. Improving Passage Efficiency of Adult American Shad at Low-Elevation Dams with Navigational Locks. North American Journal of Fisheries Management 20(2): 376-385.

Moser, M.L., P.A. Ocker, L.C. Stuehrenberg, and T.C. Bjornn. 2002. Passage Efficiency of Adult Pacific Lampreys at Hydropower Dams on the Lower Columbia River, USA. Transactions of the American Fisheries Society 131(5): 956-965.

Mossop, B., and M.J. Bradford. 2004. Importance of Large Woody Debris for Juvenile Chinook Salmon Habitat in Small Boreal Forest Streams in the Upper Yukon River Basin, Canada. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 34(9): 1955-1966.

Moulton, L.L. 1977. Ecological analysis of fishes inhabiting the rocky nearshore regions of northern Puget Sound. Ph.D. Dissertation. University of Washington. Cited in NRC 2001.

Moyle, P., and J. Cech. 2004. Fishes: An Introduction to Ichthyology – fifth edition. Upper Saddle River, NJ: Prentice-Hall, Inc. Book.

Moyle, P.B., and D. White. 2002. Effects of Screening Diversions on Fish Populations in the Central Valley: What Do We Know? University of California, Davis. A report for the Science Board, CALFED Ecosystem Restoration Program. January 2002.

Moyle, P.B., and J.A. Israel. 2005. Untested Assumptions: Effectiveness of Screening Diversions for Conservation of Fish Populations. Fisheries 30(5): 20.

Moyle, P.B., and J.J. Cech Jr. 1988. Fishes: An Introduction to Ichthyology. Second Edition. New Jersey: Prentice Hall Publishing.

Moyle, P.G. 1976. Fish Introduction in California: History and Impact on Native Fishes. Biological Conservation 9: 101-118.

Mueller, G., P.C. Marsh, G. Knowles, and T. Wolters. 2000. Distribution, Movements, and Habitat Use of Razorback Sucker (Xyrauchen Texanus) in a Lower Colorado River Reservoir, Arizona-Nevada. Western North American Naturalist 60(2): 180-187.

Mulholland, P.J., E.R. Marzolf, J.R. Webster, D.R. Hart, and S.P. Hendricks. 1997. Evidence That Hyporheic Zones Increase Heterotrophic Metabolism and Phosphorus Uptake in Forest Streams. Limnology and Oceanography 42: 443-451.

Mulholland, P.J., J.D. Newbold, J.W. Elwood, L.A. Ferren, and J.R. Webster. 1985. Phosphorus Spiraling in a Woodland Stream - Seasonal-Variations. Ecology 66(3): 1012-1023.

Mulholland, R. 1984. Habitat Suitability Index Models: Hard Clam. National Coastal Ecosystems Team. Division of Biological Services Research and Development, Fish and Wildlife Service, U.S. Department of the Interior.

Mull, K.E. 2005. Selection of Spawning Sites by Coho Salmon (Oncorhynchus Kisutch) in Freshwater Creek, California. Master's Thesis, Humboldt State University, Humboldt, California, 56 pp.

Muller-Solger, A.B., A.D. Jassby, and D.C. Muller-Navarra. 2002. Nutritional Quality of Food Resources for Zooplankton (Daphnia) in a Tidal Freshwater System (Sacramento-San Joaquin River Delta). Limnology and Oceanography 47(5): 1468-1476.

Mullholland, R. 1984. Habitat suitability index models: Hard clam. FWS/OBS/-82/10.77. US Department of Interior, Fish and Wildlife.

Mulliss, R., D.M. Revitt, and R.B.E. Shutes. 1997. The Impacts of Discharges from Two Combined Sewer Overflows on the Water Quality of an Urban Watercourse. Water Science and Technology 36(8-9): 195-199.

Munoz-Perez, J.J., B.L.D. Roman-Blanco, J.M. Gutierrez-Mas, L. Moreno, and G.J. Cuena. 2001. Cost of Beach Maintenance in the Gulf of Cadiz (SW Spain). Coastal Engineering 42(2): 143-153.

Murakami, K., and K. Takeishi. 1977. Behavior of Heavy Metals and PCBs in Dredging and Treating of Bottom Deposits. In Management of Bottom Sediments Containing Toxic Substances, edited by S.A. Peterson and K.K. Randolph. Washington, DC: Proceedings of the 2nd US/Japan Experts Meeting USEPA-600/3-77-083. pp. 26-42.

Murphy, M. 1998. Primary Productivity. In River Ecology and Management: Lessons from the Pacific Coastal Ecoregion, edited by R.J. Naiman and R.E. Bilby. New York: Springer-Verlag. pp. 144-168.

Murphy, M.L. 1981. Varied Effects of Clearcut Logging on Predators and Their Habitat in Small Stream of the Cascade Mountains, Oregon. Canadian Journal of Fisheries and Aquatic Sciences 38: 137-145.

Murphy, M.L., and W.R. Meehan. 1991. Stream Ecosystems. In Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats, edited by W.R. Meehan. Bethesda, Maryland: American Fisheries Society Special Publication 19. pp. 17-46.

Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of Canopy Modification and Accumulated Sediment on Stream Communities. Transactions of the American Fisheries Society 110(4): 469-478.

Murphy, M.L., J. Heifetz, J.F. Thedinga, S.W. Johnson, and K.V. Koski. 1989. Habitat Utilization by Juvenile Pacific Salmon (Oncorhynchus) in the Glacial Taku River, Southeast Alaska. Canadian Journal of Fisheries and Aquatic Sciences 46(10): 1677-1685.

Murphy, M.L., S.W. Johnson, and D.J. Csepp. 2000. A comparison of fish assemblages in eelgrass and adjacent subtidal habitats near Craig, Alaska. Alaska Fishery Research Bulletin 7:11-21. Available http://www.adfg.state.ak.us/pubs/afrb/vol7/murphyv7.pdf (Accessed 2006.10.26).

Murphy, P., A. Marquette, D. Reible, and G.V. Lowry. 2006. Predicting the Performance of Activated Carbon-, Coke-, and Soil-Amended Thin Layer Sediment Caps. Journal of Environmental Engineering-ASCE 132(7): 787-794.

Mutz, M., and A. Rohde. 2003. Processes of Surface-Subsurface Water Exchange in a Low Energy Sand-Bed Stream. International Review of Hydrobiology 88(3-4): 290-303.

Myers, J.M., R.G. Kope, G.J. Bryant, D. Teel, L.J. Lierheimer, T.C. Wainwright, W.S. Grand, F.W. Waknitz, K. Neely, S.T. Lindley, and R.S. Waples. 1998. Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California. U.S. Department of Commerce, NOAA-NMFS Technical Memo NMFS-NWFSC-35. Northwest Fisheries Science Center. Seattle, Washington. Available at: http://www.nwfsc.noaa.gov/publications/techmemos/tm35/. Accessed 2006.10.06. Cited in WDNR 2006a.

Myers, M.M., L.L.K. Johnson, and T.K. Collier. 2003. Establishing the Causal Relationship between Polycyclic Aromatic Hydrocarbon (PAH) Exposure and Hepatic Neoplasms and Neplasia-Related Liver Lesions in English Sole (Pleuronectes Vetulus). Human and Ecological Risk Assessment 9: 67-94.

Myers, M.S., L.L. Johnson, T. Hom, T.K. Collier, J.E. Stein, and U. Varanasi. 1998. Toxicopathic hepatic lesions in subadult English sole (Pleuronectes vetuls) from Puget Sound, Washington, USA: Relationships with other biomarkers of contaminant exposure. Marine Environmental Research 45(1): 47-67.

Myers, R.D. 1993. Slope stabilization and erosion control using vegetation: A manual of practice for coastal property owners. Olympia, Washington: Shorelands and Coastal Zone Management program, Washington Department of Ecology.

Myrberg, A.A. 1972. Using Sound to Influence the Behavior of Free-Ranging Marine Animals. Plenum 2: 435-368.

Myrberg, A.A., and R.J. Riggio. 1985. Acoustically Mediated Individual Recognition by a Coral Reef Fish (Pomacentrus Partitus). Animal Behavior 33: 411-416.

Nagasaka, A., Y. Nagasaka, K. Ito, T. Mano, M. Yamanaka, A. Katayama, Y. Sato, A.L. Grankin, A.I. Zdorikov, and G.A. Boronov. 2006. Contributions of Salmon-Derived Nitrogen to Riparian Vegetation in the Northwest Pacific Region. Journal of Forest Research 11(5): 377-382.

Nagel, K.-O. 1987. Untersuchungen an einer Najadenpopulation (Bilvalvia: Unionidae) in einem Baggersee bei Kassel (Nordhessen). Philippia 5:383-395. Cited in Watters et al. 1999.

Naiman, R.J. 1992. Watershed Management: Balancing Sustainability and Environmental Change. New York: Springer-Verlag.

Naiman, R.J. and R.E. Bilby. 1998. River ecology and management in the Pacific coastal ecoregion. Pages 1-10 in R.J. Naiman and R.E. Bilby (Editors). River Ecology and Management: Lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York.

Naiman, R.J., C.A. Johnston, and J.C. Kelley. 1988. Alteration of North-American Streams by Beaver. Bioscience 38(11): 753-762.

Naiman, R.J., E.V. Balian, K.K. Bartz, R.E. Bilby, and J.J. Latterell. 2002. Dead wood dynamics in stream ecosystems. USDA Forest Service General Technical Report PSW-GTR-181.

Naiman, R.J., G. Pinay, C.A. Johnston, and J. Pastor. 1994. Beaver Influences on the Long-Term Biogeochemical Characteristics of Boreal Forest Drainage Networks. Ecology 75(4): 905-921.

Naiman, R.J., J.M. Melillo, and J.E. Hobbie. 1986. Ecosystem Alteration of Boreal Forest Streams by Beaver (Castor-Canadensis). Ecology 67(5): 1254-1269.

Naiman, R.J., R.E. Bilby, and P.A. Bisson. 2000. Riparian Ecology and Management in the Pacific Coastal Rain Forest. Bioscience 50(11): 996-1011.

Naiman, R.J., R.E. Bilby, D.E. Schindler, and J.M. Helfield. 2002. Pacific Salmon, Nutrients, and the Dynamics of Freshwater and Riparian Ecosystems. Ecosystems 5(4): 399-417.

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. O'Connor, P.L. Olson, and E.A. Steel. 1992. Fundamental Elements of Ecologically Healthy Watersheds in the Pacific Northwest Coastal Ecoregion. In Watershed Management – Balancing Sustainability and Environmental Change, edited by R.J. Naiman. New York: Springer Verlag.

Nakamoto, R.J., and T.T. Kisanuki. 1995. Age and Growth of Klamath River Green Sturgeon (Acipenser Medirostris). Project #93-FP-13. Arcata, California: U.S. Forest Service.

Nakano, D., and F. Nakamura. 2006. Responses of Macroinvertebrate Communities to River Restoration in a Channelized Segment of the Shibetsu River, Northern Japan. River Research and Applications 22(6): 681-689.

Nakayama, T., M. Watanabe, K. Tanji, and T. Morioka. 2007. Effect of Underground Urban Structures on Eutrophic Coastal Environment. Science of the Total Environment 373(1): 270-288.

Napolitano, M.B. 1998. Persistence of Historical Logging Impacts on Channel Form in Mainstem North Fork Caspar Creek. General Technical Report PSW-GTR-168-Web. Albany, California: United States Department of Agriculture Forest Service - Pacific Southwest Research Station.

National Conservation Training Center. 2004. The Analytical Approach to Consultation. Lacey, Washington: Advanced Interagency Consultation - Regional Training Curriculum.

Naughton, G.P., C.C. Caudill, C.A. Peery, T.S. Clabough, M.A. Jepson, T.C. Bjornn, and L.C. Stuehrenberg. 2007. Experimental Evaluation of Fishway Modifications on the Passage

Behavior of Adult Chinook Salmon and Steelhead at Lower Granite Dam, Snake River, USA. River Research and Applications 23(1): 99-111.

Nebeker, A.V. 1972. Effect of Low Oxygen Concentration on Survival and Emergence of Aquatic Insects. Transactions of the American Fisheries Society 101(4): 675-679.

Nebeker, A.V., S.T. Onjukka, D.G. Stevens, G.A. Chapman, and S.E. Dominguez. 1992. Effects of Low Dissolved-Oxygen on Survival, Growth and Reproduction of Daphnia, Hyalella and Gammarus. Environmental Toxicology and Chemistry 11(3): 373-379.

Neck, R.W. and R.G. Howells. 1994. Status survey of Texas heelsplitter, Potamilus amphichaenus (Frierson, 1898) Unpublished report, Texas Parks and Wildlife Department, Resource Protection Division and Inland Fisheries Division, Austin. Cited in Watters et al. 1999.

Nedeau, E., A.K. Smith, and J. Stone. 2005. Freshwater Mussels of the Pacific Northwest. Vancouver, Washington: U.S. Fish and Wildlife Service.

Nedwell, J., A. Martin, and N. Mansfield. 1993. Underwater tool noise: implications for hearing loss. In Advances in Underwater Technology, Ocean Science and Offshore Engineering, edited by Subtech '93. Dordrecht, the Netherlands: Kluwer Academic Publishers. pp. 267-275.

Nedwell, J., A. Turnpenny, J. Langworthy, and B. Edwards. 2003. Measurements of Underwater Noise During Piling at the Red Funnel Terminal, Southampton, and Observations of Its Effect on Caged Fish. Prepared by Subacoustics LTD, Hampshire, UK.

Nedwell, J., and B. Edwards. 2002. Measurements of Underwater Noise in the Arun River During Piling at County Wharf, Littlehampton. Prepared by Subacoustech, Hants, England.

Neff, J.M. 1985. Polycyclic aromatic hydrocarbons. in: Fundamentals of Aquatic Toxicology. G.M. Rand and S.R. Petrocelli(eds.), Hemisphere Publishing Corporation, New York, pp: 416-454.

Neitzel, D.A. and T.J. Frest. 1990. Survey of Columbia River Basin Streams for Columbia Pebblesnail and Shortface Lanx. Fisheries 15(2):2-3.

Neitzel, D.A. and T.J. Frest. 1989. Survey of Columbia River Basin Streams for Ashy Pebblesnail Fluminicola columbiana and Great Columbia River Limpet Fisherola nuttali. PNL-7103. Richland, Washington: Pacific National Laboratory. Cited in WDNR 2006b.

Neitzel, D.A., and T.J. Frest. 1989. Survey of Columbia River Basin Streams for Giant (sic) Columbia River Spire Snail, Fluminicola columbiana and Great (sic) Columbia River Limpet, Fisherola nuttalli. Richland, Washington: Pacific Northwest Laboratory.

Neitzel, D.A., and T.J. Frest. 1990. Survey of Columbia River basin streams for Columbia pebblesnail and shortface lanx. Fisheries 15(2) 2-3. Cited in WDNR 2006b.

Neitzel, D.A., C.S. Abernethy, and E.W. Lusty. 1990. A Fisheries Evaluation of the Wapato, Sunnyside, and Toppenish Creek Canal Fish Screening Facilities, Spring 1988. DOE/BP/01830-T1, Pacific Northwest Laboratory, Richland, Washington.

Nelson, D.R. 1965. Hearing and Acoustic Orientation in the Lemon Shark Negaprion Brevirostris (Poey), and Other Large Sharks. Bulletin of Southern Californian Academic Sciences 68(3): 131-137.

Nelson, D.R., R.H. Johnson, and L.G. Waldrop. 1969. Responses in Bahamian Sharks and Groupers to Low-Frequency, Pulsed Sounds. Bulletin of Southern Californian Academic Sciences 38: 131-137.

Nelson, D.R., R.H. Johnson, and L.G. Waldrop. 1969. Responses in Bahamian Sharks and Groupers to Low-Frequency, Pulsed Sounds. Bulletin of Southern Californian Academic Sciences 68:131-137.

Nelson, T.A., A.V. Nelson, and M. Tjoelker. 2003. Seasonal and Spatial Patterns of "Green Tides" (Ulvoid Algal Blooms) and Related Water Quality Parameters in the Coastal Waters of Washington State, USA. Botanica Marina 46(3): 263-275.

Nelson, T.A., and A. Lee. 2001. A Manipulative Experiment Demonstrates That Blooms of the Macroalga Ulvaria Obscura Can Reduce Eelgrass Shoot Density. Aquatic Botany 71(2): 149-154.

Nemeth, R. S. 1989. The photobehavioral responses of juvenile chinook and coho salmon to strobe and mercury lights. M.S. Thesis. University of Washington. Cited in Simenstad et al. 1999, and in Nightingale and Simenstad 2001b.

Neraas, L.P., and P. Spruell. 2001. Fragmentation of Riverine Systems: The Genetic Effects of Dams on Bull Trout (Salvelinus confluentus) in the Clark Fork River System. Molecular Ecology 10(5): 1153-64.

Newbold, S.C., and R. Iovanna. 2007. Population level impacts of cooling water withdrawals on harvested fish stocks. Environmental Science and Technology 41 (7): 2108-2114.

Newcomb, T.W., and T.A. Flagg. 1983. Some Effects of Mount St. Helens Volcanic Ash on Juvenile Salmon Smolts. Marine Fisheries Review 45(2): 8-12.

Newcombe, C.P. and J.O.T. Jensen. 1996. Channel suspended sediment and fisheries: A synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16: 693-727.

Newcombe, C.P., and D.D. MacDonald. 1991. Effects of Suspended Sediments on Aquatic Ecosystems. North American Journal of Fisheries Management 11(1): 72-82.

Newell, R.C., L.J. Seiderer, N.M. Simpson, and J.E. Robinson. 2004. Impacts of Marine Aggregate Dredging on Benthic Macrofauna Off the South Coast of the United Kingdom. Journal of Coastal Research 20(1): 115-125.

Nguyen, R.M., and C.E. Crocker. 2006. The Effects of Substrate Composition on Foraging Behavior and Growth Rate of Larval Green Sturgeon, Acipenser medirostris. Environmental Biology of Fishes 76(2): 129-138.

Nichols, R.A. and S.G. Sprague. 2003. Use of Long-Line Cabled Logs for Stream Bank Rehabilitation. In: Restoration of Puget Sound Rivers, D.R. Montgomery, S. Bolton, D.B. Booth, and L. Wall, eds. University of Washington Press.

Nightingale, B. and C. Simenstad. 2001a. Dredging Activities: Marine Issues. University of Washington. Prepared for the Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Nightingale, B. and C. Simenstad. 2001b. Marine Overwater Structures: Marine Issues. University of Washington . Seattle, WA. Prepared for Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Nittrouer, C.A. 1978. The Process of Detrital Sediment Accumulation in a Continental Shelf Environment: An Examination of the Washington Shelf. Dissertation Thesis, University of Washington, Seattle, Washington.

Nittrouer, C.A., and R.W. Sternberg. 1975. Fate of a Fine-Grained Dredge Spoils Deposit in a Tidal Channel of Puget-Sound, Washington. Journal of Sedimentary Petrology 45(1): 160-170.

NMFS (NOAA’s National Marine Fisheries Service). 1990. West coast of North America coastal and ocean zones strategic assessment: Data atlas. U.S. Dept. Commerce. NOAA. OMA/NOS, Ocean Assessments Division, Strategic Assessment Branch. Invertebrate and Fish Volume. Washington, D.C. Cited in NRC 2001.

NMFS (NOAA’s National Marine Fisheries Service). 1996. Making Endangered Species Act determinations of effect for individual or grouped actions at the watershed scale. Environmental and Technical Services Division, Habitat Conservation Branch. Lacey, WA.

NMFS (NOAA’s National Marine Fisheries Service). 1996a. Juvenile fish screen criteria. http://swr.nmfs.noaa.gov/hcd/pumpcrit.htm, accessed 2006.11.30.

NMFS (NOAA’s National Marine Fisheries Service). 1996b. Making Endangered Species Act determinations of effect for individual or grouped actions at the watershed scale. Environmental and Technical Services Division, Habitat Conservation Branch. Lacey, WA. E.

NMFS (NOAA’s National Marine Fisheries Service). 2001a. Biological Opinion: San Francisco-Oakland Bay Bridge East Span Seismic Safety Project. Consultation conducted by National Marine Fisheries Service, Southwest Region, Santa Rosa, CA.

NMFS (NOAA's National Marine Fisheries Service). 2001b. Biological Opinion for the Mill Creek Right Bank Levee Extension Project (NMFS No. WSB-00-177).

NMFS (NOAA's National Marine Fisheries Service). 2003. Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the United States Army Corps of Engineers – Seattle District Larson Bank Protection Project – Puyallup River. Consultation Number (NOAA Fisheries No. WSB-99-122).

NMFS (NOAA's National Marine Fisheries Service). 2003a. Biological Opinion for the Continued Maintenance of Weir Ponds and Proposed Fish Passage Improvement at Monitoring Facilities located on North Fork Caspar Creek and South Fork Caspar Creek, Jackson Demonstration Forest, Mendocino County, California. File No. 151422SWR02SR6251.

NMFS (NOAA's National Marine Fisheries Service). 2003b. Biological Opinion – Benicia-Martinez New Bridge Project 151422SWR02SR6292. NMFS, Southwest Region, Santa Rosa, CA.

NMFS (NOAA's National Marine Fisheries Service). 2004. Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Sucker Creek Bank Stabilization Project, Josephine County, Oregon (Corps No. 200300599).

NMFS (NOAA's National Marine Fisheries Service). 2004. Species of Concern and Candidate Species: Pinto Abalone. Office of Protected Covered Species Paper - Invertebrates 5-10 Resources. Available athttp://www.nmfs.noaa.gov/pr/species/concern/profiles/pinto_abalone.pdf (Updated 2005.05.31. Accessed 2006.10.06).

NMFS (NOAA's National Marine Fisheries Service). 2004a. Endangered Species Act - Section 7 Consultation Biological Opinion and Magnuson-Stevens Fisheries Conservation and Management Act Essential Fish Habitat Consultation SR 104 Edmonds Crossing Ferry Terminal Project, Snohomish County. HUC 17110019. NMFS Tracking No. 2003/00756.

NMFS (NOAA's National Marine Fisheries Service). 2004b. Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Sucker Creek Bank Stabilization Project, Josephine County, Oregon (Corps No. 200300599).

NMFS (NOAA's National Marine Fisheries Service). 2005. Endangered Species Act Interagency Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the US Highway 12 Naches River Bank Protection and Habitat Enhancement Project, Yakima County, Washington USGS HUC 170300020307 (COE Ref No.: 200401366). NMFS Tracking No: 2004/01714.

NMFS (NOAA's National Marine Fisheries Service). 2005a. Biological and Conference Opinion Pursuant to Section 7 of the Endangered Species Act (ESA) on the Effects of Northwest

Pipeline Corporation (Northwest) Capacity Replacement Project. NMFS Tracking No.: 2005-00271.

NMFS (NOAA's National Marine Fisheries Service). 2005a. Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Northwest Pipeline Corporation Capacity Replacement Project Docket Nos. P05-32-000, -001. NMFS Tracking No: 2005-00271.

NMFS (NOAA's National Marine Fisheries Service). 2005b. Biological Opinion - La Conner Wharf and Float Project, La Conner, WA (HUC 171100070202, COE Ref. No. 200401163). NMFS Tracking No. 2004/01826. Seattle, WA.

NMFS (NOAA's National Marine Fisheries Service). 2005b. Biological Opinion - La Conner Wharf and Float Project, La Conner, WA (HUC 171100070202, COE Ref. No. 200401163). NMFS Tracking No. 2004/01826. Seattle, WA.

NMFS (NOAA's National Marine Fisheries Service). 2006. Endangered Species Act-Section 7 Formal Consultation Biological and Conference Opinion and Magnuson-Stevens Fishery Management Conservation and Management Act Essential Fish Habitat Consultation Stream Crossing Structure Replacement and Removal Activities, Snake and Clearwater River Basins, 170601 & 170603, Idaho. NMFS Reference No.: 2005/06396, 2005/07365 and 2005/07366.

NMFS (NOAA's National Marine Fisheries Service). 2006a. Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Lafayette Highway Bridge Replacement, Yamhill River (HUC 170900080702), City of Lafayette, Yamhill County, Oregon (Corps No.: 200600248). NMFS Reference No.: 2006/02305.

NMFS (NOAA's National Marine Fisheries Service). 2006b. Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Installation of Puget Sound Energy Natural Gas Pipeline from Thorp to Cle Elum, Kittitas County, WA, Upper Middle Yakima River Subbasin, 170300010204 Lanigan Springs, 1703000010310 Dry Creek, 170300010309 Robinson Creek (COE No. 200401162). NMFS Reference No.: 2005/02668.

NMFS (NOAA's National Marine Fisheries Service). 2006c. Biological Opinion - Cascade Marina Expansion Project in the Columbia River, Franklin County, WA (Sixth Field Hydrologic Unit Code: Zintel Canyon 170200160603). NMFS Tracking No. 2005/06498. Seattle, WA.

NMFS (NOAA's National Marine Fisheries Service). 2006d. Biological Opinion - Bridge Creek Culvert Removal Project, Bridge Creek, a tributary to the Upper Middle Fork of the John Day River, Sixth field HUC 170702030105, Grant County, OR. ODOT Key No. 12661. NMFS Tracking No. 2005/03011. Seattle, WA.

NMFS (NOAA's National Marine Fisheries Service). 2006e. Endangered Species Act Section7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act

Essential Fish Habitat Consultation for the Tidewater Cove Marina, Columbia River, Sixth Field HUC 170800010901- Salmon Creek, Clark County, Washington. (COE No. 200401353) NMFS Tracking No: 2005/00228.

NMFS (NOAA’s National Marine Fisheries Service). 2006f. Biological Opinion - Biological Assessment for the National Marine Fisheries Service Entiat River Bridge-to-Bridge Restoration Project Fifth Field (HUC 1702001001), Chelan County, WA. NMFS Tracking No. 2006/01232. Seattle, WA. E.

NMFS (NOAA’s National Marine Fisheries Service). 2006g. Endangered Species Act-Section 7 Formal Consultation Biological and Conference Opinion and Magnuson-Stevens Fishery Management Conservation and Management Act Essential Fish Habitat Consultation Stream Crossing Structure Replacement and Removal Activities, Snake and Clearwater River Basins, 170601 & 170603, Idaho. NMFS Reference No.: 2005/06396, 2005/07365 and 2005/07366. E.

NMFS (NOAA’s National Marine Fisheries Service). 2006h. Biological Opinion - Moore Road Bridge Replacement, North Yamhill River (HUC 170900080604), Yamhill County, OR. Corps No.: 200500794. NMFS Tracking No.: 2006/01047. Seattle, WA.

NMFS (NOAA’s National Marine Fisheries Service). 2006i. Biological Opinion - Northwest Pipeline Corporation Capacity Replacement Project, Docket Nos. CP05-32-000, -001. NMFS Tracking No. 2006/02900. Seattle, WA. E.

NMFS (NOAA’s National Marine Fisheries Service). 2006j. Biological Opinion - Scholls Ferry Road Bridge Replacement, Tualatin River (HUC 170900100501), Washington County, OR. COE No. 200500709. NMFS Tracking No. 2006/00161. Seattle, WA.

NMFS (NOAA’s National Marine Fisheries Service). 2006k. Biological Opinion - Slate Creek Bridge Replacement located over the upper Salmon River within the Slate Creek Watershed (HUC 1706020108), Custer County, ID. NMFS Tracking No. 2005/006344. Seattle, WA. E.

NMFS (NOAA’s National Marine Fisheries Service). 2006m. Biological Opinion - Sucker Creek Bridge Replacement and Bank Stabilization Project, Sucker Creek, Lower Sucker Creek Sixth field (HUC 171003110304), Josephine County, OR. Corps No. 200300599. NMFS Tracking No. 2005/05756. Seattle, WA. E.

NMFS (NOAA’s National Marine Fisheries Service). 2006n. Biological Opinion - City of Sweet Home Water Line, Wiley Creek (HUC 170900060101), Linn County, OR. Corps No. 200500500. NMFS Tracking No. 2005/04541. Seattle, WA E.

NMFS (NOAA's National Marine Fisheries Service). 1996. Addendum Juvenile Fish Screen Criteria for Pump Intakes. May 9, 1996. Available at National Marine Fisheries website: http://swr.nmfs.noaa.gov/hcd/pumpcrit.htm (accessed October 10, 2006).

NMFS (NOAA's National Marine Fisheries Service). 1998. Draft Document - Non-Fishing Threats and Water Quality: A Reference for EFH Consultation. Seattle, Washington: National Marine Fisheries Service.

NMFS (NOAA's National Marine Fisheries Service). 2001. Guidelines for Fish Passage at Stream Crossings. Prepared by Southwest Region of the National Marine Fisheries Service, Hydraulic Engineering Staff. Santa Rosa, California.

NMFS (NOAA's National Marine Fisheries Service). 2004. Draft - Anadromous Salmonid Passage Facility Guidelines and Criteria. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region, Portland, Oregon.

NMFS (NOAA's National Marine Fisheries Service). 2004. Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation: SR 104 Edmonds Crossing Ferry Terminal Project, Snohomish County Prepared for U.S. Department of Transportation, Federal Highway Administration by NOAA’s National Marine Fisheries Service March 25, 2004.

NMFS (NOAA's National Marine Fisheries Service). 2005. Endangered Species Act Interagency Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the US Highway 12 Naches River Bank Protection and Habitat Enhancement Project, Yakima County, Washington. NMFS Tracking No: 2004/01714. Seattle, Washington: National Marine Fisheries Service.

NMFS (NOAA's National Marine Fisheries Service). 2006. Endangered Species Act–Section 7 Formal Consultation Biological and Conference Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation Stream Crossing Structure Replacement and Removal Activities, Snake and Clearwater River Basins, 170601 & 170603, Idaho. NMFS No. 2005/06396, 2005/07365, and 2005/07366. Seattle, Washington: National Marine Fisheries Service.

NMFS (NOAA's National Marine Fisheries Service). 2007a. Species of Concern and Candidate Species: Pinto Abalone. Office of Protected Covered Species Paper - Invertebrates 5-10 Resources. Seattle, Washington: National Marine Fisheries Service.

NMFS (NOAA's National Marine Fisheries Service). 2007b. Rationale for the Use of 187 dB Sound Exposure Level for Pile Driving Impacts Threshold. Unpublished memorandum. Seattle, Washington: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS (NOAA's National Marine Fisheries Service). 2008. Anadromous Salmonid Passage Facility Design. NMFS, Northwest Region, Portland, Oregon.

NMFS and USFWS (NOAA’s National Marine Fisheries Service and U.S. Fish and Wildlife Service). 2005. Endangered Species Act – Section 7 Consultation Biological and Conference

Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the Verlot Riverfront Tracts Bank Stabilization Project, Middle South Fork Stillaguamish River, Snohomish County, Washington. NMFS Tracking No.: 2005/00188

NOAA Fisheries. 2003. Non-Fishing Impacts to Essential Fish Habitat and Recommended Conservation Measures. Version 1. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NOAA Fisheries. 2006. Endangered Species Act-Section 7 Formal Consultation Biological and Conference Opinion and Magnuson-Stevens Fishery Management Conservation and Management Act Essential Fish Habitat Consultation Stream Crossing Structure Replacement and Removal Activities, Snake and Clearwater River Basins, 170601 & 170603, Idaho. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Noggle, C.C. 1978. Behavioral, Physiological and Lethal Effects of Suspended Sediment on Juvenile Salmonids. Masters Thesis, University of Washington, Seattle, Washington.

Nordstrom, K.F. 1992. Estuarine Beaches: An Introduction of the Physical and Human Factors Affecting Use and Management of Beaches in Estuaries, Lagoons, Bays and Fjords. New York: Elsevier Applied Science

Nordstrom, K.F. 2005. Beach Nourishment and Coastal Habitats: Research Needs to Improve Compatibility. Restoration Ecology 13(1): 215-222.

Norman, D.K., J.C. Cederholm, and W.S. Lingley. 1998. Flood Plains, Salmon Habitat, and Sand and Gravel Mining. Washington Geology 26(2/3).

Norris, J.E. 1991. Habitat Associations of Juvenile Rockfishes from Inland Marine Waters of Washington State: An Annotated Bibliography and Review.

Northcote, T.G. 1998. Migratory Behavior of Fish and Its Significance to Movement through Riverine Fish Passage Facilities. Fish Migration and Fish Bypasses. Edited by M. Jungwirth, S. Schmutz and S. Weiss. Fishing News Books.

Northcote, T.G., and P.A. Larkin. 1989. The Fraser River: A Major Salmonine Production System. Canadian Special Publication of Fisheries and Aquatic Sciences 106: 172-204.

Northcote, T.G., N.T. Johnston, and K. Tsumura. 1979. Feeding relationships and food web structure of Lower Fraser River fishes. Westwater Research Centre, Technical Report No.16. University of British Columbia, Vancouver, B.C. 73 p.

Novak, S.J., and C.R. Goodell, 2006. Using HEC-RAS 3.1.3 to Model and Design Tide Gate Systems. Unpublished technical memorandum. Portland, Oregon: National Oceanic and Atmospheric Administration, National Marine Fisheries Service. pp. 11.

Novales-Flamarique, I., and C.W. Hawryshyn. 1996. Retinal Development and Visual Sensitivity of Young Pacific Sockeye Salmon (Oncorhynchus nerka). Journal of Experimental Biology 199: 869-882.

NRC (Natural Resources Consultants, Inc.). 2001. Final species memorandum and habitat assessment in the King County HCP planning area. Volume 2: Marine Fish. Prepared for King County Wastewater Treatment Division.

NRC. 1995. Beach Nourishment and Protection. Washington, DC: Committee for Beach Nourishment and Protection, National Research Council.

NRC. 1996. Upstream: Salmon and Society in the Pacific Northwest. Edited by N.R.C. Committee on Protection and Management of Pacific Northwest Anadromous Salmonids. The National Academies Press.

NRC. 2007. Mitigating Shore Erosion along Sheltered Coasts. National Research Council of the National Academies. Washington, DC: National Academies Press.

NRS Canada. 2004. National Recovery Strategy for the Northern Abalone (Haliotis Kamtschatkana) in Canada.

NSC. 2007. Edmonds Underwater Park and Brackett's Landing Shoreline Sanctuary Conservation Area. Available at: http://www.nwstraits.org/uploads/pdfs/MPAs/Edmonds.pdf (accessed July 9, 2007).

Nunnally, N.R. 1978. Stream Renovation: An Alternative to Channelization. Environmental Management 2: 403-411.

Nybakken, J.W., and M.D. Bertness. 2005. Marine Biology: An Ecological Approach. San Francisco: Pearson Benjamin Cummings.

O’Neill, S.M., J.E. West, and S. Quinnell. 1995. Contaminant Monitoring in Fish: Overview of the Puget Sound Monitoring Program Fish Task. Puget Sound Research ’95 Proceedings, Bellevue, Washington, pp. 35-50.

O'Connell, V.M. and D.W. Carlile. 1993. Habitat-specific density of adult yelloweye rockfish Sebastes ruberrimus in the eastern Gulf of Alaska. Fishery Bulletin 91:304-309. Cited in NRC 2001.

O'Connor, J.P., D.J. O'Mahony, J.M. O'Mahony, and T.J. Gienane. 2006. Some Impacts of Low and Medium Head Weirs on Downstream Fish Movement in the Murray-Darling Basin in Southeastern Australia. Ecology of Freshwater Fish 15(4): 419-427.

Oh, Y.I., and E.C. Shin. 2006. Using Submerged Geotextile Tubes in the Protection of the E. Korean Shore. Coastal Engineering 53(11): 879-895.

Olla, B.L., M.W. Davis, and C.B. Schreck. 1995. Stress-Induced Impairment of Predator Evasion and Non-Predator Mortality in Pacific Salmon. Aquaculture Research 26(6): 393-398.

Olson, A.M., E.G. Doyle, and S.D. Visconty. 1996. Light Requirements of Eelgrass: A Literature Survey.

Olson, A.M., S.D. Visconty, and C.M. Sweeney. 1997. Modeling the Shade Cast by Overwater Structures

Olyphant, G.A., and S.W. Bennett. 1994. Contemporary and Historical Rates of Eolian Sand Transport in the Indiana Dunes Area of Southern Lake-Michigan. Journal of Great Lakes Research 20(1): 153-162.

Ona E., O.R. Godo, N.O Handegard, V. Hjellvik, R. Patel, and G. Pedersen. 2007. Silent research vessels are not quiet. Journal of the Acoustical Society of America 121 (4): EL145-EL150. April 2007.

Opperman, J.J., and A.M. Merenlender. 2004. The Effectiveness of Riparian Restoration for Improving Instream Fish Habitat in Four Hardwood-Dominated California Streams. North American Journal of Fisheries Management 24(3): 822-834.

Oren, U., and Y. Benayahu. 1998. Didemnid Ascidians: Rapid Colonizers of Artificial Reefs in Eilat (Red Sea). Bulletin of Marine Science 63(1): 199-206.

Orr, C.H., K.L. Rogers, and E.H. Stanley. 2006. Channel Morphology and P Uptake Following Removal of a Small Dam. Journal of the North American Benthological Society 25(3): 556-568.

Orth, R. J. and K.A. Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. Science 22: 51-52. Cited in Nightingale and Simenstad 2001b.

Orth, R.J. 1975. Destruction of Eelgrass, Zostera Marina, by the Cownose Ray, Rhinoptera Bonasus, in the Chesapeake Bay. Chesapeake Science 16: 205-208.

Orth, R.J.J., K.I. Heck, and J.V. Montrans. 1984. Faunal Communities in Seagrass Beds: A Review of Influence of Plant Structure and Prey Characteristics on Predator-Prey Relations. Estuaries 7: 339-350.

Osborne, L.L. and K.A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243-258.

Oullet, P., and J.J. Dodson. 1985. Tidal Exchanges of Anadromous Rainbow Smelt (Osmerus Mordax) Larvae between a Shallow Spawning Tributary and the St. Lawrence Estuary. Canadian Journal of Fisheries and Aquatic Sciences 42: 1352-1358.

Pacific Biodiversity Institute. 2006. Database search for “Anodonta californiensis” (California floater).

http://www.pacificbio.org/ESIN/OtherInvertebrates/CaliforniaFloater/CaliforniaFloater_pg.html, accessed 2006.12.19.

Pacific Biodiversity Institute. 2006. Endangered Species Information Network for Washington, Oregon, and Idaho. February 2006. http://www.pacificbio.org/ESIN/ESIN.html

Pacific Biodiversity Institute. 2007. Homepage. Available at: http://www.pacificbio.org/ (accessed July 2007).

Padma, T.V., R.C. Hale, M.H. Roberts, and R.N. Lipsius. 1999. Toxicity of creosote water soluble fractions generated from contaminated sediments to the bay mysid. Ecotoxicology and Environmental Safety 42:171-176.

Palermo, M., S. Maynord, J. Miller, and D. Reible. 1998. Guidance for In Situ Subaqueous Capping of Contaminated Sediments. EPA 905-B96-004. Chicago, Illinois: U.S. Army Corps of Engineers, Great Lakes National Program Office.

Palmer, M.A., E.S. Bernhardt, J.D. Allan, P.S. Lake, G. Alexander, S. Brooks, J. Carr, S. Clayton, C.N. Dahm, J.F. Shah, D.L. Galat, S.G. Loss, P. Goodwin, D.D. Hart, B. Hassett, R. Jenkinson, G.M. Kondolf, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, and E. Sudduth. 2005. Standards for Ecologically Successful River Restoration. Journal of Applied Ecology 42: 208-217.

Palmer, R.W., and J.H. Okeeffe. 1990. Downstream Effects of Impoundments on the Water Chemistry of the Buffalo River (Eastern Cape), South-Africa. Hydrobiologia 202(1-2): 71-83.

Paola, C., E. Foufoula-Georgiou, W.E. Dietrich, M. Hondzo, D. Mohrig, G. Parker, M.E. Power, I. Rodriguez-Iturbe, V. Voller, and P. Wilcock. 2006. Toward a Unified Science of the Earth's Surface: Opportunities for Synthesis among Hydrology, Geomorphology, Geochemistry, and Ecology. Water Resources Research 42(3).

Papanicolaou, A. and A. Maxwell. 2000. Equilibrium geomorphologic conditions for high gradient bed streams. Prepared for Research Office, Washington State Department of Transportation.

Paragamian, V.L., G. Kruse, and V. Wakkinen. 2001. Spawning Habitat of Kootenai River White Sturgeon, Post-Libby Dam. North American Journal of Fisheries Management 21(1): 22-33.

Parametrix and Battelle Marine Sciences Laboratory. 1996. Anacortes Ferry Terminal eelgrass, macroalgae, and macrofauna habitat survey report., Report for Sverdrup Civil, Inc. and WSDOT. Cited in Nightingale and Simenstad 2001b.

Parametrix and Battelle. 1996. Anacortes Ferry Terminal eelgrass, macroalgae, and macrofauna habitat survey report. Report for Sverdrup Civil, Inc. and WSDOT. In Nightingale and Simenstad 2001b.

Parametrix, Inc. 1985. Sand/gravel/riprap colonization study. Report to Port of Seattle.

Parametrix. 1996. Anacortes Ferry Terminal eelgrass, macroalgae, and macrofauna habitat survey report. Prepared for Sverdrup Civil, Inc. and the Washington State Department of Transportation by Parametrix and Battelle, Seattle, Washington.

Parkhill, K.L., and J.S. Gulliver. 2002. Effect of Inorganic Sediment on Whole-Stream Productivity. Hydrobiologia 472(1-3): 5-17.

Parkyn, S.M., J.M. Quinn, T.J. Cox, and N. Broekhuizen. 2005. Pathways of N and C Uptake and Transfer in Stream Food Webs: An Isotope Enrichment Experiment. Journal of the North American Benthological Society 24(4): 955-975.

Parsley, M.J., L.G. Beckman and G.T. McCabe. 1993. Spawning and rearing habitat use by white sturgeons in the Columbia River downstream from McNary Dam. Transactions of the American Fisheries Society 122: 217-227.

Pasternack, G.B., C.L. Wang, and J.E. Merz. 2004. Application of a 2D Hydrodynamic Model to Design of Reach-Scale Spawning Gravel Replenishment on the Mokelumne River, California. River Research and Applications 20(2): 205-225.

Patrick, P.H., and R.S. McKinley. 1987. Field Evaluation of a Hidrostal Pump for Live Transfer of American Eels at a Hydroelectric Facility. North American Journal of Fisheries Management 7(2): 303-305.

Patten, B.G. 1971. Increased predation by the torrent sculpin, Cottus rhotheus, on coho salmon fry, Oncorhynchus kisutch, during moonlight nights. Journal of the Fisheries Research Board of Canada 28(9):1352–1354. Cited in Tabor et al. 1998.

Paul, A.J., and J.M. Paul. 1998. Respiration Rate and Thermal Tolerances of Pinto Abalone Haliotis Kamtschatkana. Journal of Shellfish Research 17(3): 743-745.

Pauley, G.B., DA Armstrong, R. Van Citter, and G.L. Thomas. 1989. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Southwest)—Dungeness crab. U.S. Fish Wildl. Serv. Biol. Rep. 82(11.121). U.S. Army Corps of Engineers, TR EL-82-4. 20 PP. Available at: http://www.nwrc.usgs.gov/wdb/pub/0172.pdf (Accessed 2006.10.02).

Pauley, G.B., G.L. Thomas, D.A. Marino, and D.C. Weigand. 1989. Evaluation of the Effects of Gravel Bar Scalping on Juvenile Salmonids in the Puyallup River Drainage: University of Washington Cooperative Fishery Research Unit Report. Seattle, Washington: University of Washington.

Pauley, G.B., K.L. Oshima, and G.L. Thomas. 1988. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest)-- Sea-run

Cutthroat trout. U.S. Fish and Wildlife Service Biological Report 82(11.86). Cited in WDNR 2006a.

Peake, S. 1999. Substrate Preferences of Juvenile Hatchery-Reared Lake Sturgeon, Acipenser Fulvescens. Environmental Biology of Fishes 56(4): 367-374.

Peake, S. 2004. Effect of Approach Velocity on Impingement of Juvenile Northern Pike at Water Intake Screens. North American Journal of Fisheries Management 24(2): 390-396.

Pearlstine, L.G., W.M. Kitchens, P.J. Latham, and R.D. Bartleson. 1993. Tide Gate Influences on a Tidal Marsh. Water Resources Bulletin 29(6): 1009-1019.

Pearson, W.H. 2005. Protocols for Evaluation of Upstream Passage of Juvenile Salmonids in an Experimental Culvert Test Bed. Prepared for Washington State Department of Transportation, Pacific Northwest Division and Planning and Capital Program by Battelle Memorial, Institute. Report No. PNWD-3525, Richland, Washington.

Pearson, W.H., J. Southard, C.L. May, J.R. Skalski, R.L. Townsend, A.R. Horner-Devine, D.R. Thurman, R.H. Hotchkiss, R.R. Morrison, M.C. Richmond, and D. Deng. 2006. Research on the Upstream Passage of Juvenile Salmon through Culverts: Retrofit Baffles. Prepared for Washington State Department of Transportation, by Battelle Memorial Institute - Pacific Northwest Division, Report No. PNWD-3672, Richland, Washington.

Pearson, W.H., J.R. Skalski, and C.I. Malme. 1992. Effects of sounds from a geophysical survey device on behavior of captive rockfish (Sebastes spp.). Canadian Journal of Fisheries and Aquatic Sciences 49:1343-1356.

Pearson, W.H., R.P. Mueller, S.L. Sargeant, and C.W. May. 2005. Evaluation of Juvenile Salmon Leaping Ability and Behavior at an Experimental Culvert Test Bed. Prepared for Washington State Department of Transportation by Battelle Pacific Northwest Division, Olympia, Washington.

Peddicord, R.K. 1980. Direct effects of suspended sediments on aquatic organisms. In: Contaminants and Sediments, Vol. 1 (R.A. Baker, ed.). Ann Arbor Science, Ann Arbor, MI.

Pedersen, T.C.M., A. Baattrup-Pedersen, and T.V. Madsen. 2006. Effects of Stream Restoration and Management on Plant Communities in Lowland Streams. Freshwater Biology 51(1): 161-179.

Peijnenburg, W., A. de Groot, T. Jager, and L. Posthuma. 2005. Short-Term Ecological Risks of Depositing Contaminated Sediment on Arable Soil. Ecotoxicology and Environmental Safety 60(1): 1-14.

Penczak, T., G. Zieba, H. Koszalinski, and A. Kruk. 2003. The Importance of Oxbow Lakes for Fish Recruitment in a River System. Archiv Fur Hydrobiologie 158(2): 267-281.

Penland, S., P.F. Connor, A. Beall, S. Fearnley, and S.J. Williams. 2005. Changes in Louisiana's Shoreline: 1855-2002. Journal of Coastal Research: 7-39.

Pentec Environmental Inc. 1991. Port of Everett, Dredging Monitoring Surveys, 1990. Prepared for Port of Everett, Edmonds, Washington.

Pentec Environmental Inc. 1997. Movement of Juvenile Salmon through Industrialized Areas of Everett Harbor. Prepared for Port of Everett, Edmonds, Washington. Cited in Nightingale and Simenstad 2001b.

Penttila, D. 1978. Studies of the Surf Smelt (Hypomesus Pretiosus) in Puget Sound. Technical Report 42. Olympia, Washington: Washington Department of Fisheries.

Penttila, D. 1995. Investigations of the Spawning Habitat of the Pacific Sand Lance (Ammodytes Hexapterus) in Puget Sound. Proceedings of Puget Sound Research 1995, Seattle, Washington, pp. 855-859.

Penttila, D. 1995. Known spawning beaches of the surf smelt , Hypomesus, and the sand lance, Ammodytes, in southern Puget Sound, WA (Pierce, Thurston, and Mason Counties), as of March, 1995. April 1995 with charts updated and revised February 1999.

Penttila, D. 2000a. Forage fishes of the Puget Sound region. NWSC/PSAMP Data Conference, LaConner, WA. Washington Department of Fish and Wildlife. http://www.wa.gov/wdfw/fish/forage/forage.htm. (Accessed 2006.10.06). Cited in Nightingale and Simenstad 2001b.

Penttila, D. 2000b. Impacts of Overhanging Shading Vegetation on Egg Survival for Summer-Spawning Surf Smelt on Upper Intertidal Beaches in Northern Puget Sound, Washington. Draft. Washington Department of Fish and Wildlife, Marine Resources Division.

Penttila, D. 2001. Effects of Overhanging Shading Vegetation on Egg Survival for Summer-Spawning Surf Smelt on Upper Intertidal Beaches in Northern Puget Sound, Washington. Olympia, Washington: Washington Department of Fish and Wildlife, Marine Resources Division.

Penttila, D. 2007. Washington Department of Fish and Wildlife, Seattle, Washington. Personal communication with José Carrasquero of Herrera Environmental Consultants, Inc. Seattle, Washington, regarding Use of Nourished Beaches by Forage Fishes.

Penttila, D. and D. Doty 1990. Progress Report. Results of 1989 Eelgrass Shading Studies in Puget Sound. Washington Department of Fisheries, Marine Fish Habitat Investigations Division. Summary available at: http://depts.washington.edu/newwsdot/pentdot.html (Accessed 2006.09.29).

Penttila, D., and M. Aquero. 1978. Fish Usage of Birch Bay Village Marina, Whatcom County Washington in 1976. Washington Department of Fisheries Progress Report.

Perez-Ruzafa, A., J.A. Grarcia-Charton, E. Barcala, and C. Marcos. 2006. Changes in benthic fish assemblages as a consequence of coastal works in a coastal lagoon: The Mar Menor (Spain, Western Mediterranean). Marine Pollution Bulletin 53(1-4): 107-120.

Perillo, G.M.E., D.E. Perez, M.C. Piccolo, E.D. Palma, and D.G. Cuadrado. 2005. Geomorphologic and Physical Characteristics of a Human Impacted Estuary: Quequen Grande River Estuary, Argentina. Estuarine Coastal and Shelf Science 62(1-2): 301-312.

Persaud, D., R. Jaagumagi, and A. Hayton. 1991. The Provincial Sediment Quality Guidelines (Draft). Water Resources Branch, Ontario Ministry of the Environment, Toronto, Canada. May. Cited in Stratus 2005b.

Peters, R.J., B.R. Missildine, and D.L. Low. 1998. Seasonal Fish Densities near River Banks Stabilized with Various Stabilization Methods; First Year Report of the Flood Technical Assistance Project. Lacey, Washington: U.S. Fish and Wildlife Service, North Pacific Coast Ecoregion.

Petersen, W., E. Willer, and C. Willamowski. 1997. Remobilization of Trace Elements from Polluted Anoxic Sediments after Resuspension in Toxic Water. Water Air and Soil Pollution 99(1-4): 515-522.

Peterson, C.H., D.H.M. Hickerson, and G.G. Johnson. 2000. Short-Term Consequences of Nourishment and Bulldozing on the Dominant Large Invertebrates of a Sandy Beach. Journal of Coastal Research 16(2): 368-378.

Peterson, C.H., M.J. Bishop, G.A. Johnson, L.M. D'Anna, and L.M. Manning. 2006. Exploiting Beach Filling as an Unaffordable Experiment: Benthic Intertidal Impacts Propagating Upwards to Shorebirds. Journal of Experimental Marine Biology and Ecology 338(2): 205-221.

Peterson, H.W.U., and L. Amiotte. 2006. Decline of Skokomish Nation Spot Shrimp Catch in Low Dissolved Oxygen Waters of the Hood Canal, Puget Sound, State of Washington. Ethnicity & Disease 16(4): 17-17.

Peterson, J.T., N.P. Banish, and R.F. Thurow. 2005. Are Block Nets Necessary? Movement of Stream-Dwelling Salmonids in Response to Three Common Survey Methods. North American Journal of Fish Management 25: 732-743.

Peterson, J.T., R.F. Thurow, and J.W. Guzevich. 2004. An Evaluation of Multipass Electrofishing for Estimating the Abundance of Stream-Dwelling Salmonids. Transactions of the American Fisheries Society 133(2): 462-475.

Petr, T. 2000. Interactions Between Fish and Aquatic Macrophytes in Inland Waters. A review. FAO Fisheries Technical Paper. No. 396. Rome, FAO. 2000. 185p. Available at: http://www.fao.org/docrep/006/X7580E/X7580E00.htm#TOC (Accessed 2006.10.03).

Peven, C.M. 1987. Downstream Migration Timing of Two Stocks of Sockeye Salmon on the Mid-Columbia River. Northwest Science 61(3): 186-190.

Pflug, D.E. and G.P. Pauley. 1984. Biology of smallmouth bass (Micropterus dolomieui) in Lake Sammamish, Washington. Northwest Science 58(2): 118–130.

Pflug, David E. and Gilbert P. Pauley. 1984. Biology of smallmouth bass (Micropterus dolomieui) in Lake Sammamish, Washington. Northwest Science 58(2):118–130. Cited in Carrasquero 2001.

Phillips, R.C. 1984. Ecology of Eelgrass Meadows in the Pacific Northwest: A Community Profile. Seattle Pacific University, Seattle, WA. Prepared for: U.S. Fish and Wildlife Service. NTIS Publication Number PB86-110376. FWS Publication Number FWS/OBS-84/24.

Phillips, R.W., R.L. Lantz, E.W. Claire, and J.R. Moring. 1975. Some effects of gravel mixtures on emergence of coho salmon and steelhead trout fry. Transactions of the American Fisheries Society 3:461-466.

Pickerell, C.H., S. Schott, and S. Wyllie-Echeverria. 2005. Buoy-Deployed Seeding: Demonstration of a New Eelgrass (Zostera Marina L.) Planting Method. Ecological Engineering 25(2): 127-136.

Pickering, H., and D. Whitmarsh. 1997. Artificial Reefs and Fisheries Exploitation: A Review of the `Attraction Versus Production' Debate, the Influence of Design and Its Significance for Policy. Fisheries Research 31(1-2): 39-59.

Pickett, P.J. 1997. Pollutant Loading Capacity for the Black River, Chehalis River System, Washington. Journal of the American Water Resources Association 33(2): 465-480.

Pihl, L., S. Baden, N. Kautsky, P. Ronnback, T. Soderqvist, M. Troell, and H. Wennhage. 2006. Shift in Fish Assemblage Structure Due to Loss of Seagrass Zostera Marina Habitats in Sweden. Estuarine Coastal and Shelf Science 67(1-2): 123-132.

Pillard, D.A. 1996. Assessment of Benthic Macroinvertebrate and Fish Communities in a Stream Receiving Storm Water Runoff from a Large Airport. Journal of Freshwater Ecology 11(1): 51-59.

Pitt, R., R. Field, M. Lalor, and M. Brown. 1995. Urban Stormwater Toxic Pollutants: Assessment, Sources, and Treatability. Water Environment Research 67: 260-275.

Poe, T.P., H.C. Hansel, S. Vigg, D.E. Palmer, and L.A. Prendergast. 1991. Feeding of predaceous fishes on out-migrating juvenile salmonids in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120(4): 405–420. Cited in Carrasquero 2001.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.L. Stromberg. 1997. The Natural Flow Regime. Bioscience 47(11): 769.

Pollen, N. 2007. Temporal and Spatial Variability in Root Reinforcement of Streambanks: Accounting for Soil Shear Strength and Moisture. Catena 69(3): 197-205.

Pollock, M., M. Heim, and D. Werner. 2003. Hydrologic and Geomorphic Effects of Beaver Dams and their Influence on Fishes. American Fisheries Society Symposium 37: 213-233.

Pollock, M.M., G.R. Pess, and T.J. Beechie. 2004. The Importance of Beaver Ponds to Coho Salmon Production in the Stillaguamish River Basin, Washington, USA. North American Journal of Fisheries Management 24(3): 749-760.

Pondella, D.J., and J.S. Stephens. 1994. Factors Affecting the Abundance of Juvenile Fish Species on a Temperate Artificial Reef. Bulletin of Marine Science 55(2-3): 1216-1223.

Pondella, D.J., J.S. Stephens, and M.T. Craig. 2002. Fish production of a temperate artificial reef based on the density of embiotocids (Teleostei : Perciformes). ICES Journal of Marine Science 59: S88-S93.

Ponti, M., M. Abbiati, and V.U. Ceccherelli. 2002. Drilling Platforms as Artificial Reefs: Distribution of Macrobenthic Assemblages of The "Paguro" Wreck (Northern Adriatic Sea). ICES Journal of Marine Science 59: S316-S323.

Poole, G., J. Dunham, M. Hicks, D. Keenan, J. Lockwood, E. Materna, D. McCullough, C. Mebane, J. Risley, S. Sauter, S. Spaulding, and D. Sturdevant. 2001. Technical Synthesis Scientific Issues Relating to Temperature Criteria for Salmon, Trout, and Char Native to the Pacific. EPA 910-R-01-007. Environmental Protection Agency, Region 10.

Poole, G.C., and C.H. Berman. 2001a. An Ecological Perspective on in-Stream Temperature: Natural Heat Dynamics and Mechanisms of Human-Caused Thermal Degradation. Environmental Management 27(6): 787-802.

Poole, G.C., and C.H. Berman. 2001b. Pathways of Human Influence on Water Temperature Dynamics in Stream Channels. Environmental Management 27: 787-802.

Poole, G.C., J. Dunham, and M. Hicks plus nine authors. 2001. Scientific Issues Relating to Temperature Criteria for Salmon, Trout, and Char Native to the Pacific Northwest.   A summary report submitted to the Policy Workgroup of the EPA Region 10 Water Temperature Criteria Guidance Project. EPA 910-R-01-007. August 2001.

Popper, A.N. 2005. A review of hearing by sturgeon and lamprey. Environmental BioAcoustics, LLC. Rockville, Maryland. Submitted to the U.S. Army Corps of Engineers, Portland District.

Popper, A.N. and T.J. Carlson. 1998. Application of sound and other stimuli to control fish behavior. Transactions of the American Fisheries Society 127(5): 673–707.

Popper, A.N., and N.L. Clarke. 1976. The Auditory System of the Goldfish (Carassius Auratus): Effects of Intense Acoustic Stimulation. Comparative Biochemistry and Physiology 53: 11-18.

Popper, A.N., and R.R. Fay. 1973. Sound Detection and Processing by Teleost Fishes - Critical Review. Journal of the Acoustical Society of America 53(6): 1515-1529.

Popper, A.N., and R.R. Fay. 1993. Sound Detection and Processing by Fish - Critical-Review and Major Research Questions. Brain Behavior and Evolution 41(1): 14-38.

Popper, A.N., and T.J. Carlson. 1998. Application of Sound and Other Stimuli to Control Fish Behavior. Transactions of the American Fisheries Society 127(5): 673-707.

Popper, A.N., M.E. Smith, P.A. Cott, B.W. Hanna, A.O. MacGillivray, M.E. Austin, and D.A. Mann. 2005. Effects of Exposure to Seismic Airgun Use on Hearing of Three Fish Species. Journal of the Acoustical Society of America 117(6): 3958-3971.

Popper, A.N., T.J. Carlson, A.D. Hawkins, B.L. Southall, and R.L. Gentry. 2006. Interim criteria for injury of fish exposed to pile driving operations: a white paper.

Popper, Arthur N. and Thomas J. Carlson. 1998. Application of sound and other stimuli to control fish behavior. Transactions of the American Fisheries Society 127(5):673–707.

Portnoy, J.W. 1991. Summer Oxygen Depletion in a Diked New-England Estuary. Estuaries 14(2): 122-129.

Poston, T. 2001. Treated Wood Issues Associated with Overwater Structures in Marine and Freshwater Environments White Paper. Olympia, Washington: Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Poston, T.M., K.M. Krupka, and M.C. Richmond. 1996. Estimation of Treated Piling Emplacement and Piling Leachate Concentrations in the Columbia River. Richland, Washington: Pacific Northwest National Laboratory.

Power, M.E., W.E. Dietrich, and J.C. Finlay. 1996. Dams and Downstream Aquatic Biodiversity: Potential Food Web Consequences of Hydrologic and Geomorphic Change. Environmental Management 20(6): 887-895.

Powers, P.D., and C.S. Saunders. 2002. Fish-Passage Design Flows for Ungauged Catchments in Washington. Washington Department of Fish and Wildlife - Lands and Restoration Services Program.

Powers, P.D., and K. Bates. 1997. Culvert Hydraulics Related to Upstream Juvenile Salmon Passage. Washington Department of Fish and Wildlife, Land and Restoration Services Program. Environmental Engineering Services.

Pratt, T.C., and K.E. Smokorowski. 2003. Fish habitat management implications of the summer habitat use by littoral fishes in a north temperate, mesotrophic lake. Canadian Journal of Fisheries and Aquatic Sciences 60(3): 286-300.

Prepared for Dr. Michael Ritter USDA Forest Service, Forest Products Laboratory, One Gifford Pinchot Drive, Madison, WI 53705-2398. Prepared by:, Dr. Kenneth M. Brooks, Aquatic Environmental Sciences, 644 Old Eaglemount Road, Port Townsend, WA 98368

Pretty, J.L., S.S.C. Harrison, D.J. Shepherd, C. Smith, A.G. Hildrew, and R.D. Hey. 2003. River Rehabilitation and Fish Populations: Assessing the Benefit of Instream Structures. Journal of Applied Ecology 40(2): 251-265.

Prince, E.D., and O.E. Maughan. 1978. Freshwater Artificial Reefs - Biology and Economics. Fisheries 3(1): 5-9.

Prinslow, T.E., E.O. Salo, and B.P. Snyder. 1979. Studies of behavioral effects of a lighted and an unlighted wharf on outmigrating salmonids-March-April 1978. Final Report. Fisheries Research Institute, University of Washington, Seattle WA. Cited in Nightingale and Simenstad 2001b.

Protasov, V.R. 1970. Chapter 1: Distribution of Light in Water. In Vision and Nearshore Orientation of Fish. Academy of Sciences of the USSR, edited by V.R. Protasov. Jerusalem: Israel Program for Scientific Translations. pp. 175.

PSAT (Puget Sound Action Team). 2007. State of the Sound 2007. Publication No. PSAT 07-01. Seattle, Washington: Puget Sound Action Team.

PSAT (Puget Sound Water Quality Action Team). 2001. Eelgrass (Zostera marina). Available at http://www.psat.wa.gov/Publications/Fact_sheets/eelgrass.pdf (Accessed 2006.10.26).

PSNERP (Puget Sound Nearshore Project). 2003. Guidance for protection and restoration of nearshore ecosystems of Puget Sound. Available at http://www.cev.washington.edu/lc/PSNERP/guidance.pdf (Accessed 2006.10.26).

Puckett, K.J., and J.J. Anderson. 1987. Behavioral Responses of Juvenile Salmonids to Strobe and Mercury Lights. Seattle, Washington: Fisheries Research Institute, University of Washington.

Puckett, K.J., and J.J. Anderson. 1988. Behavioral responses of juvenile salmonids to strobe and mercury lights. Final report to Stone and Webster Engineering Corporation. Fisheries Research Institute, University of Washington, Seattle, Washington. Cited in Simenstad et al. 1999.

Puget Sound Steelhead Biological Review Team. 2005. Status Review Update for Puget Sound Steelhead. National Marine Fisheries Service Northwest Fisheries Science Center. July 26, 2005.

Pusch, M., D. Fiebig, I. Brettar, H. Eisenmann, B.K. Ellis, L.A. Kaplan, M.A. Lock, M.W. Naegeli, and W. Traunspurger. 1998. The Role of Micro-Organisms in the Ecological Connectivity of Running Waters. Freshwater Biology 40(3): 453-495.

Qian, P.Y., J.W. Qiu, R. Kennish, and C.A. Reid. 2003. Recolonization of Benthic Infauna Subsequent to Capping of Contaminated Dredged Material in East Sha Chau, Hong Kong. Estuarine Coastal and Shelf Science 56(3-4): 819-831.

Qiao, F.L., J. Ma, C.S. Xia, Y.Z. Yang, and Y.L. Yuan. 2006. Influences of the Surface Wave-Induced Mixing and Tidal Mixing on the Vertical Temperature Structure of the Yellow and East China Seas in Summer. Progress in Natural Science 16(7): 739-746.

Quinn, T.P. 2005. The Behavior and Ecology of Pacific Salmon and Trout. Seattle, Washington: University of Washington Press.

Quinn, T.P., and N.P. Peterson. 1994. The Effect of Forest Practices on Fish Populations. Olympia, Washington: Washington Department of Natural Resources, Timber-Fish-Wildlife Program.

Quinones, R.M., and T.J. Mulligan. 2005. Habitat Use by Juvenile Salmonids in the Smith River Estuary, California. Transactions of the American Fisheries Society 134(5): 1147-1158.

Quintella, B.R., N.O. Andrade, A. Koed, and P.R. Almeida. 2004. Behavioral Patterns of Sea Lampreys' Spawning Migration through Difficult Passage Areas, Studied by Electromyogram Telemetry. Journal of Fish Biology 65(4): 961-972.

Quirollo, L.F. 1992. Pacific hake. Pages 109-112 in: W.S. Leet, C.M. Dewees, and C.W. Haugen (eds.), California's Living Marine Resources and Their Utilization. California Sea Grant College Program. Davis, California. UCSGEP-92-12. 129pp.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman. 2001. Hypoxia in the Gulf of Mexico. Journal of Environmental Quality 30: 320-329.

Rabalais, N.N., W.J. Wiseman, R.E. Turner, B.K. SenGupta, and Q. Dortch. 1996. Nutrient Changes in the Mississippi River and System Responses on the Adjacent Continental Shelf. Estuaries 19: 386-47.

Radle, A.L. (undated). The effect of noise on wildlife: A literature review. http://interact.uoregon.edu/medialit/wfae/readings/radle.html#11, accessed 2006.10.12.

Radle, A.L. 2005. The effect of noise on wildlife: A literature review. World Forum for Acoustic Ecology. Web Site – College of Education – University of Oregon – Eugene, OR. http://interact.uoregon.edu/medialit/wfae/readings/radle.html#11. Accessed 2006.10.12.

Radomski, P., and T.J. Goeman. 2001. Consequences of Human Lakeshore Development on Emergent and Floating-Leaf Vegetation Abundance. North American Journal of Fisheries Management 21:46-61.

Rakocinski, C.F., R.W. Heard, S.E. LeCroy, J.A. McLelland, and T. Simons. 1996. Responses by Macrobenthic Assemblages to Extensive Beach Restoration at Perdido Key, Florida, USA. Journal of Coastal Research 12(1): 326-353.

Raleigh, R.F., W.J. Miller and P.C. Nelson. 1986. Habitat suitability index models and instream flow suitability curves: chinook salmon. U.S. Fish and Wildlife Service Biological Report 82(10.122). Cited in Bjornn and Reiser 1991.

Ram, J.L., P. Fong, R.P. Croll, S.J. Nichols, and D. Wall. 1992. The Zebra Mussel (Dreissena-Polymorpha), a New Pest in North-America - Reproductive Mechanisms as Possible Targets of Control Strategies. Invertebrate Reproduction & Development 22(1-3): 77-86.

Ranasinghe, R., and I.L. Turner. 2006. Shoreline response to submerged structures: A review. Coastal Engineering 53(1): 65-79.

Ranasinghe, R., I.L. Turner, and G. Symonds. 2006. Shoreline response to multi-functional artificial surfing reefs: A numerical and physical modeling study. Coastal Engineering 53(7): 589-611.

Rao, M.V. and V. Kuppusamy. 1992. Leachability of creosote : Fuel oil (1:1) wood preservative in marine environment. Journal of the Timber Development Association of India 38(3):42-45. Cited in Stratus 2005a.

Rao, Y.R., and D.J. Schwab. 2007. Transport and Mixing between the Coastal and Offshore Waters in the Great Lakes: A Review. Journal of Great Lakes Research 33(1): 202-218.

Ratte, L., and E.O. Salo. 1985. Under-Pier Ecology of Juvenile Pacific Salmon (Oncorhynchus Spp.) in Commencement Bay, Washington. FRI-UW-8508. Seattle, Washington: University of Washington Fisheries Research Institute.

Ratte, L.D. 1985. Under-pier ecology of juvenile Pacific salmon (Oncorhynchus spp.) in Commencement Bay, Washington. University of Washington. Cited in Nightingale and Simenstad 2001b.

Ray, H.L., A.M. Ray, and A.J. Rebertus. 2004. Rapid Establishment of Fish in Isolated Peatland Beaver Ponds. Wetlands 24(2): 399-405.

RBW (Royal Boskalis Westminster NB). 2007. Innovative Approach to Ground Level Lowering. Available at: http://www.boskalis.com/vervolg_1kolom.php?pageID=454&siteLang=en_US (accessed August 15, 2007).

Redding, J.M., C.B. Schreck, and F.H. Everest. 1987. Physiological Effects on Coho Salmon and Steelhead of Exposure to Suspended Solids. Transactions of the American Fisheries Society 116(737-744).

Regnault, M., and J.P. Lagardere. 1983. Effects of Ambient Noise on the Metabolic Level of Crangon crangon (Decapoda, Natantia). Marine Ecology 11: 71-78.

Reible, D., D. Hayes, C. Lue-Hing, J. Patterson, N. Bhowmik, M. Johnson, and J. Teal. 2003. Comparison of the Long-Term Risks of Removal and in Situ Management of Contaminated Sediments in the Fox River. Soil & Sediment Contamination 12(3): 325-344.

Reid, S.M., and P.G. Anderson. 1998. Effects of sediment released during open-cut pipeline water crossings. http://aplwww.alliance-pipeline.com/contentfiles/45____EffectsofSediment.pdf, accessed 2006.12.06.

Reid, S.M., F. Ade, and S. Metikosh. 2004. Sediment entrainment during pipeline water crossing construction: predictive models and crossing method comparison. Journal of Environmental Engineering and Science 3: 81-88.

Reilly, C.A., T.W. Wyllie-Echeverria, and S. Ralston. 1992. Interannual Variation and Overlap in the Diets of Pelagic Juvenile Rockfish (Genus: Sebastes) Off Central California. Fishery Bulletin 90: 505-515.

Reiman, B.E., and J.D. McIntyre. 1993. Demographic and Habitat Requirements for the Conservation of Bull Trout. Ogden, Utah: USDA Forest Service Intermountain Research Station.

Reiman, B.E., J.T. Peterson, and D.L. Myers. 2006. Have Brook Trout (Salvelinus fontinalis) Displaced Bull Trout (Salvelinus confluentus) Along Longitudinal Gradients in Central Idaho Streams? Canadian Journal of Fisheries & Aquatic Sciences 63: 63-78.

Reine, K., and D. Clarke. 1998. Entrainment by Hydraulic Dredges - a Review of Potential Impacts. Technical Note DOER-EI. Vicksburg, Mississippi: U.S. Army Corps of Engineers, Engineer Research and Development Center.

Reinelt, L.E., and R.R. Horner. 1995. Pollutant Removal from Stormwater Runoff by Palustrine Wetland Based on Comprehensive Budgets. Ecological Engineering 4(2): 77-97.

Reish, D.J. 1961. A Study of Benthic Fauna in a Recently Constructed Boat Harbor in Southern California. Ecology 42(1): 84-91.

Rempel, L., and M. Church. 2003. The Harrison Bar Gravel Removal Experiment: Final Report. Vancouver, BC.

RETEC. 2002. Remedial Investigation and Feasibility Study, Lower Fox River and Green Bay, Wisconsin, Prepared for Wisconsin Department of Natural Resources. Seattle, Washington.

Reyff, J. A. 2006. Russian River Replacement Bridge at Geyserville: Underwater Sound Measurement Data for Driving Permanent 48-inch CISS Piles. Illingworth and Rodkin, Inc., Petaluma, CA.

Reyff, J., P. Donavan, and C.R. Greene Jr. 2003. Underwater Sound Levels Associated with Seismic Retrofit Construction of the Richmond-San Rafael Bridge. Prepared for California Department of Transportation by Illingworth & Rodkin, Inc. and Greeneridge Sciences Sacramento, California.

Reynolds, W. 1987. Coastal structures and long-term shore migration. Proceedings, Coastal Zone 87. American Society of Civil Engineers. As cited in MacDonald et al. 1994.

Ribeiro, F., P.K. Crain, and P.B. Moyle. 2004. Variation in Condition Factor and Growth in Young-of-Year Fishes in Floodplain and Riverine Habitats of the Cosumnes River, California. Hydrobiologia 527(1): 77-84.

Rice, C.A. 2006. Effects of Shoreline Modification on a Northern Puget Sound Beach: Microclimate and Embryo Mortality in Surf Smelt (Hypomesus pretiosus). Estuaries and Coasts 29(1): 63-71.

Rice, C.A. 2006. Effets of shoreline modification on a northern Puget Soun beach: Microclimate and embryo mortality in surf smelt (Hypomesus pretiosus). Estuaries and Coasts 29(1):63-71.

Rice, C.E., and K.C. Kadavy. 1994. Riprap Design Downstream of Submerged Pipe Outlets. Transactions of the ASAE 37(1): 85-94.

Richard, J.D. 1968. Fish Attraction with Pulsed Low Frequency Sound. Journal of the Fisheries Research Board of Canada 25(7): 1441-1452.

Richards, K., J. Brasington, and F. Hughes. 2002. Geomorphic Dynamics of Floodplains: Ecological Implications and a Potential Modeling Strategy. Freshwater Biology 47(4): 559-579.

Richardson, E.V., and S.R. Davies. 2001. Evaluating Scour at Bridges. Fourth Edition. FHWA NHI 01-001 HEC-18, Hydraulic Engineering Circular No. 19. Washington D.C: Federal Highway Administration, U.S. Department of Transportation.

Richardson, J.S., R.E. Bilby, and C.A. Bondar. 2005. Organic Matter Dynamics in Small Streams of the Pacific Northwest. Journal of the American Water Resources Association 41(4): 921-934.

Richardson, M.D., A.G. Carey, and W.A. Colgate. 1977. Aquatic Disposal Field Investigations - Columbia River Disposal Site, Oregon. Appendix C: The Effects of Dredged Material Disposal on Benthic Assemblages. Vicksburg, Mississippi: Report to U.S. Army Corps of Engineers Waterways Expt. Station.

Richardson,W.J., C.R. Green Jr., C.I. Malme, and D.H. Thomson. 1995. Marine Mammals and Noise. Academic Press. San Diego, CA. Cited in Jones and Stokes 2006.

Richter, A., and S.A. Kolmes. 2005. Maximum Temperature Limits for Chinook, Coho, and Chum Salmon, and Steelhead Trout in the Pacific Northwest. Reviews in Fisheries Science 13(1): 23-49.

Riggs, C.D. and G.R. Webb. 1956. The mussel population of an area of loamy-sand bottom of Lake Texoma. American Midland Naturalist. 56:197-203. Cited in Watters et al. 1999.

Ringelman, J. 1991. Managing Beaver to Benefit Waterfowl, Fish and Wildlife Leaflet 13.4.7. Washington, DC: U.S. Fish and Wildlife Service.

Rinne, J.N., W.L. Minckley, and P.O. Bersell. 1981. Factors Influencing Fish Distribution in Two Desert Reservoirs, Central Arizona. Hydrobiologia 80(1): 31-42.

Roberts, B., P.J. Mulholland, and J. Houser. 2007. Effects of Upland Disturbance and Instream Restoration on Hydrodynamics and Ammonium Uptake in Headwater Streams. Journal of the North American Benthological Society 26(1): 38-53.

Robertson, D.R., D.G. Green, and B.C. Victor. 1988. Temporal Coupling of Production and Recruitment of Larvae of a Caribbean Reef Fish. Ecology 69: 370-381.

Robinson, G.D., W.A. Dunson, J.E. Wright, and G.E. Mamolito. 1976. Differences in low pH tolerance among strains of brook trout (Salvelinus fontinalis). Journal of Experimental Zoology 204: 33-42.

Robinson, J.E., R.C. Newell, L.J. Seiderer, and N.M. Simpson. 2005. Impacts of Aggregate Dredging on Sediment Composition and Associated Benthic Fauna at an Offshore Dredge Site in the Southern North Sea. Marine Environmental Research 60(1): 51-68.

Robinson, T.C., and J.M. Bayer. 2005. Upstream Migration of Pacific Lampreys in the John Day River, Oregon: Behavior, Timing, and Habitat Use. Northwest Science 79(2-3): 106-119.

Roby, D.D., K. Collis, D.E. Lyons, D.P. Craig, J.Y. Adkins, A.M. Myers, and R.M. Suryan. 2002. Effects of colony relocation on diet and productivity of Caspian terns. Journal of Wildlife Management 66(3): 662-673.

Rodgers, D.W., and P.H. Patrick. 1985. Evaluation of a Hidrostal Pump Fish Return System. North American Journal of Fisheries Management 5(3a): 393-399.

Rodriguez, M.A. 2002. Restricted Movement in Stream Fish: The Paradigm Is Incomplete, Not Lost. Ecology 83(1): 1-13.

Rodway, M.S., H.M. Regehr, J. Ashley, P.V. Clarkson, R.I. Goudie, D.E. Hay, C.M. Smith, and K.G. Wright. 2003. Aggregative response of Harlequin Ducks to herring spawning in the Strait of Georgia, British Columbia. Canadian Journal of Zoology-Revue Canadienne De Zoologie 81(3): 504-514.

Roegner, G.C., B.M. Hickey, J.A. Newton, A.L. Shanks, and D.A. Armstrong. 2002. Wind-Induced Plume and Bloom Intrusions into Willapa Bay, Washington. Limnology and Oceanography 47(4): 1033-1042.

Rolauffs, P., D. Hering, and S. Lohse. 2001. Composition, Invertebrate Community and Productivity of a Beaver Dam in Comparison to Other Stream Habitat Types. Hydrobiologia 459: 201-212.

Rolletschek, H., and H. Kühl. 1997. Die Auswirkungen Von Röhrichtschutzbauwerken Auf Die Gewässerufer (the Impacts of Reed Protecting Structures on Lakesides). Limnologica 27(3-4): 365-380.

Roman, C.T., W.A. Niering, and R.S. Warren. 1984. Salt Marsh Vegetation Change in Response to Tidal Restriction. Environmental Management 8: 141-150.

Roni, P., and T.P. Quinn. 2001. Effects of Wood Placement on Movements of Trout and Juvenile Coho Salmon in Natural and Artificial Stream Channels. Transactions of the American Fisheries Society 130(4): 675-685.

Roni, P., T. Bennett, S. Morley, G.R. Pess, K. Hanson, D. Van Slyke, and P. Olmstead. 2006. Rehabilitation of Bedrock Stream Channels: The Effects of Boulder Weir Placement on Aquatic Habitat and Biota. River Research and Applications 22(9): 967-980.

Roni, P., T.J. Beechie, R.E. Bilby, F.E. Leonetti, M.M. Pollock, and G.R. Pess. 2002. A Review of Stream Restoration Techniques and a Hierarchical Strategy for Prioritizing Restoration in Pacific Northwest Watersheds. North American Journal of Fisheries Management 22(1): 1-20.

Rooper, C.N., D.R. Gunderson, and B.M. Hickey. 2006. An Examination of the Feasibility of Passive Transport from Coastal Spawning Grounds to Estuarine Nursery Areas for English Sole. Estuarine Coastal and Shelf Science 68(3-4): 609-618.

Roper, B.B., D. Konnoff, D. Heller, and K. Wieman. 1998. Durability of Pacific Northwest Instream Structures Following Floods. North American Journal of Fisheries Management 18(3): 686-693.

Rosenthal, R.J., V. Moran-O'Connell, and M.C. Murphy. 1988. Feeding ecology of ten species of rockfishes (Scorpaenidae) from the Gulf of Alaska. California Department of Fish and Game 74:16-36. Cited in NRC 2001.

Roth, B.M., I.C. Kaplan, G.G. Sass, P.T. Johnson, A.E. Marburg, A.C. Yannarell, T.D. Havlicek, T.V. Willis, M.G. Turner, and S.R. Carpenter. 2007. Linking terrestrial and aquatic ecosystems: The role of woody habitat in lake food webs. Ecological Modeling 203(3-4): 439-452.

Rowe, D., J. Smith, B. Baillie, and M. Meleason. 2004. Wood in Streams: How Much Is Good for Fish? Water and Atmosphere 12(1): 16-17.

Rowe, D.K. and T.L. Dean. 1998. Effects of turbidity on the feeding ability of the juvenile migrant stage of six New Zealand freshwater fish species. New Zealand Journal of Marine and Freshwater Research 32:21-29.

Rowntree, K.M., and E.S.J. Dollar. 1999. Vegetation Controls on Channel Stability in the Bell River, Eastern Cape, South Africa. Earth Surface Processes and Landforms 24(2): 127-134.

Rulifson, R.A., and B.L. Wall. 2006. Fish and Blue Crab Passage through Water Control Structures of a Coastal Bay Lake. North American Journal of Fisheries Management 26(2): 317-326.

Russel, I.C., A. Moore, S. Ives, L.T. Kell, M.J. Ives, and R.O. Stonehewer. 1998. The Migratory Behavior of Juvenile and Adult Salmonids in Relation to an Estuarine Barrage. Hydrobiologia 371/372: 321-333.

Rychetsky, S., and G. Card. 2000. Infiltration Galleries Of Oregon. Developed in cooperation with the Jefferson County Soil and Water Conservation District, June 2000.

Saiki, M.K., and B.A. Martin. 2001. Survey of fishes and environmental conditions in Abbotts Lagoon, Point Reyes National Seashore, California. California Fish and Game 87(4): 123-138.

Saldi-Caromile, K., K. Bates, P. Skidmore, J. Barenti, and D. Pineo. 2004. Stream habitat restoration guidelines: Final draft. Co-published by the Washington Departments of Fish and Wildlife and Ecology and the U.S. Fish and Wildlife Service. Olympia, Washington. http://wdfw.wa.gov/hab/ahg/shrg/index.htm, accessed 2006.11.28.

Salim, M., and J.S. Jones. 1999. Scour Around Exposed Pile Foundations. Reston, Virginia.

Salo, E. O., N.J. Bax, T.E. Prinslow, C.J. Whitmus, B.P. Snyder, and C.A. Simenstad. 1980. The effects of construction of naval facilities on the outmigration of juvenile salmonids from Hood Canal, Washington. Final Report FRI-UW-8006. University of Washington, Fisheries Research Institute. Cited in Nightingale and Simenstad 2001b.

Salo, E.O. 1991. Life History of Chum Salmon (Oncorhynchus keta). In Pacific Salmon Life Histories, edited by C. Groot and L. Margolis. Vancouver, British Columbia: University of British Columbia Press. pp. 231-310.

Salo, E.O., N.J. Bax, T.E. Prinslow, C.J. Whitmus, B.P. Snyder, and C.A. Simenstad. 1980. The Effects of Construction of Naval Facilities on the Outmigration of Juvenile Salmonids from Hood Canal, Washington. Final Report FRI-UW-8006. Seattle, Washington: University of Washington, Fisheries Research Institute.

Salo, E.O., T.E. Prinslow, R.A. Campbell, D.W. Smith, and B.P. Snyder. 1979. Trident dredging study: the effects of dredging at the U.S. Naval Submarine Base at Bangor on outmigrating juvenile chum salmon, Oncorhynchus keta, in Hood Canal, Washington: final report, February to July 1977. Seattle, WA: University of Washington Fisheries Research Institute.

Sammut, J., I. White, and M.D. Melville. 1996. Acidification of an Estuarine Tributary in Eastern Australia Due to Drainage of Acid Sulfate Soils. Marine and Freshwater Research 47(5): 669-684.

Sampson, D.B. 1996. Stock Status of Canary Rockfish off Oregon and Washington in 1996: Appendix C in Pacific Fishery Management Council. Status of the Pacific Coast Groundfish Fishery through 1996 and Recommended Acceptable Biological Catches for 1997: Stock Assessment and Fishery Evaluation. Portland, Oregon: Pacific Fishery Management Council.

Sandahl, J.F., D.H. Baldwin, J.J. Jenkins, and N.L. Scholz. 2004. Odor-Evoked Field Potentials as Indicators of Sublethal Neurotoxicity in Juvenile Coho Salmon (Oncorhynchus Kisutch) Exposed to Copper, Chlorpyrifos, or Esfenvalerate. Canadian Journal of Fisheries & Aquatic Sciences 61: 404-413. (As cited in Stratus 2005b)

Sandecki, M. 1989. Aggregate Mining in River Systems. California Geology 42: 88-94.

Sand-Jensen, K. 1998. Influence of Submerged Macrophytes on Sediment Composition and Near-Bed Flow in Lowland Streams. Freshwater Biology 39(4): 663-679.

Sandstrom, A., B.K. Eriksson, P. Karas, M. Isaeus, and H. Schreiber. 2005. Boating and Navigation Activities Influence the Recruitment of Fish in a Baltic Sea Archipelago Area. Ambio 34(2): 125-130.

Sane, M., H. Yamagishi, M. Tatelshi, and T. Yamagishi. 2007. Environmental Impacts of Shore-Parallel Breakwaters Along Nagahama and Ohgata, District of Joetsu, Japan. Journal of Environmental Management 82(4): 399-409.

Sargeant, S.L., M.C. Miller, C.W. May, and R.M. Thom. 2004. Shoreline armoring research program; Phase II-Conceptual model development for bank stabilization in freshwater systems. Prepared for Washington State Dept. of Transportation. Olympia, WA.

Sargent, F.J., T.J. Leary, D.W. Crewz, and C.R. Kruer. 1995. Scarring of Florida’s seagrasses: an assessment and management options, Florida Marine Research Institute Technical Report TR-1. Florida Marine Research Institute, St. Petersburg, Florida. 43 pp. plus appendix. Cited in Dawes et al. 2004.

Sass, G.G., J.F. Kitchell, S.R. Carpenter, T.R. Hrabik, A.E. Marburg, and M.G. Turner. 2006. Fish Community and Food Web Responses to a Whole-Lake Removal of Coarse Woody Habitat. Fisheries 31: 321-330.

Saulnier, I., and A. Mucci. 2000. Trace Metal Remobilization Following the Resuspension of Estuarine Sediments: Saguenay Fjord, Canada. Applied Geochemistry 15(2): 191-210.

Scavia, D., and S.B. Bricker. 2006. Coastal Eutrophication Assessment in the United States. Biogeochemistry 79(1-2): 187-208.

Schaffter, R.G., P.A. Jones, and J.G. Karlton. 1983. Sacramento River and Tributaries Bank Protection and Erosion Control Investigation-Evaluation of Impacts on Fisheries. Prepared for U.S. Army Corps of Engineers, Sacramento District by The Resources Agency, California Department of Fish and Game, Sacramento.

Schemel, L.E., T.R. Sommer, A.B. Muller-Solger, and W.C. Harrell. 2004. Hydrologic Variability, Water Chemistry, and Phytoplankton Biomass in a Large Floodplain of the Sacramento River, California, USA. Hydrobiologia 513(1-3): 129-139.

Scheuerell, M.D., and D.E. Schindler. 2004. Changes in the Spatial Distribution of Fishes in Lakes Along a Residential Development Gradient. Ecosystems 7(1): 98-106.

Scheuerell, M.D., P.S. Levin, R.W. Zabel, J.G. Williams, and B.L. Sanderson. 2005. A New Perspective on the Importance of Marine-Derived Nutrients to Threatened Stocks of Pacific Salmon (Oncorhynchus spp.). Canadian Journal of Fisheries and Aquatic Sciences 62(5): 961-964.

Schille, P.C. 2007. Personal communication (contact with E. Doyle, Herrera Environmental Consultants, Seattle, Washington, regarding the use of pile driving in fish screen installation). WDFW, Yakima Screen Shop, Washington.

Schille, P.C. 2008. Personal communication (contact with E. Doyle, Herrera Environmental Consultants, Seattle, Washington, providing text description of modular screen systems developed by the WDFW Yakima Screen Shop during the 1990s). WDFW, Yakima Screen Shop, Washington.

Schindler, D.E., P.R. Leavitt, C.S. Brock, S.P. Johnson, and P.D. Quay. 2005. Marine-Derived Nutrients, Commercial Fisheries, and Production of Salmon and Lake Algae in Alaska. Ecology 86(12): 3225-3231.

Schindler, D.E., S.I. Geib, and M.R. Williams. 2000. Patterns of Fish Growth Along a Residential Development Gradient in North Temperate Lakes. Ecosystems 3: 229-237.

Schlesinger, W.H. 1997. Biogeochemistry: An analysis of global change. San Diego, California: Academic Press.

Schlosser, I.J. 1995. Dispersal, Boundary Processes, and Trophic-Level Interactions in Streams Adjacent to Beaver Ponds. Ecology 76(3): 908-925.

Schmetterling, D.A. 2003. Reconnecting a Fragmented River: Movements of Westslope Cutthroat Trout and Bull Trout after Transport Upstream of Milltown Dam, Montana. North American Journal of Fisheries Management 23(3): 721-731.

Schmetterling, D.A., C.G. Clancy, and T.M. Brandt. 2001. Effects of riprap bank reinforcement on stream salmonids in the Western United States. Fisheries 26(7): 6-13. 7-23; 7-48

Schmidt, K.M., J.J. Roering, J.D. Stock, W.E. Dietrich, D.R. Montgomery, and T. Schaub. 2001. The Variability of Root Cohesion as an Influence on Shallow Landslide Susceptibility in the Oregon Coast Range. Canadian Geotechnical Journal 38(5): 995-1024.

Scholik, A.R., and H.Y. Yan. 2001a. Effects of underwater noise on auditory physiology of fishes. Proceedings of Institute of Acoustics, United Kingdom 23(2): 27-36.

Scholik, A.R., and H.Y. Yan. 2001b. Effects of underwater noise on auditory sensitivity of a cyprinid fish. Hearing Research 152: 17-24.

Scholik, A.R., and H.Y. Yan. 2002. Effects of Boat Engine Noise on Auditory Sensitivity of the Fathead Minnow, Pimephales Promelas. Environmental Biological Fish 63: 203-209.

Scholik, A.R., and H.Y. Yan. 2002. The effects of noise on the auditory sensitivity of the bluegill sunfish, Lepomis macrochirus. Comparative Biochemistry and Physiology A133: 43-52.

Schoning, R.W., and D.R. Johnson. 1956. A Measured Delay in the Migration of Adult Chinook Salmon at Bonneville Dam on the Columbia River. Contribution No. 23. Portland, Oregon: Fish Commission of Oregon.

Schoonover, J.E., and K.W.J. Williard. 2003. Ground Water Nitrate Reduction in Giant Cane and Forest Riparian Buffer Zones. Journal of the American Water Resources Association 39(2): 347-354.

Schuett-Hames, D., B. Conrad, A. Pleus, and K. Lautz. 1996. Literature Review and Monitoring Recommendations for Salmonid Spawning Gravel Scour. TFW-AM9-96-001. Northwest Indian Fisheries Commission and Washington Department of Fish and Wildlife.

Schumm, S.A. 1971. Fluvial Geomorphology: Channel Adjustment and River Metamorphosis. In River Mechanics. Volume 1, edited by H.W. Shen. Fort Collins, Colorado. pp. 5-1 to 5-11.

Schumm, S.A., M.D. Harvey, and C.C. Watson. 1984. Incised Channels: Morphology, Dynamics, and Control. Littleton, Colorado: Water Resources Publications.

Schwarz, A.L. and G.L.Greer. 1984. Responses of Pacific herring, Clupea harengus pallasi, to some underwater sounds. Canadian Journal of Fisheries and Aquatic Sciences. 41: 1183-1192. Cited in Scholik and Yan 2001a.

Scruggs, G. D. 1960. Status of fresh-water mussel stocks in the Tennessee River. U.S. Fish and Wildlife Service Special Science Report on Fish 370:1–41. Cited in Tucker and Theiling 1998.

Seabergh, W.C., and N.C. Kraus. 2003. Progress in Management of Sediment Bypassing at Coastal Inlets: Natural Bypassing, Weir Jetties, Jetty Spurs, and Engineering Aids in Design. Coastal Engineering Journal 45(4): 533-563.

Sear, D.A. 1995. Morphological and Sedimentological Changes in a Gravel-Bed River Following 12 Years of Flow Regulation for Hydropower. Regulated Rivers: Research and Management 10: 247-264.

Sear, D.A., S.E. Darby, C.R. Thorne, and A.B. Brookes. 1994. Geomorphological Approach to Stream Stabilization and Restoration - Case-Study of the Mimmshall Brook, Hertfordshire, UK. Regulated Rivers-Research & Management 9: 205-223.

Sedell J. R., and K. J. Luchessa. 1982. Using the historical record as an aid to salmonid habitat enhancement. Pages 210–223 in N. B. Armantrout (ed.), Acquisition and utilization of aquatic habitat inventory information. Bethesda, Maryland: American Fisheries Society.

Sedell, J.R. and J.L. Froggatt. 1984. Importance of streamside forests to large rivers: The isolation of the Willamette River, Oregon, U.S.A. from its floodplain by snagging and streamside forest removal. Verhandlungen der Internationale Vereinigung für Limnologie 22:1828-1834.

Sedell, J.R., and K.J. Luchessa. 1982. Using the historical record as an aid to salmonid habitat enhancement. Pages 210–223 in N. B. Armantrout (ed.), Acquisition and utilization of aquatic habitat inventory information. Bethesda, Maryland: American Fisheries Society.

Sedell, J.R., F. J. Swanson, and S.V. Gregory. 1985. Evaluating fish response to woody debris. Pp. 222-245 in T.J. Hassler (ed.), Proceedings of the Pacific Northwest stream habitat workshop. Arcata, CA: California Cooperative Fishery Research Unit, Humboldt State University. Cited in Naiman et al. 2002.

Sedell, J.R., J.E. Yuska and R.W. Speaker. 1986. Habitats and salmonid distribution in pristine, sediment-rich river valley systems: S. Fork Hoh and Queets River, Olympic National Park. Pages 33-46 in W.R. Meehan, T.R. Merrell, Jr., and T.A. Hanley (eds.), Fish and wildlife relationships in old-growth forests. American Institute of Fishery Research Biologists.

Seitz, R.D., R.N. Lipcius, and M.S. Seebo. 2005. Food Availability and Growth of the Blue Crab in Seagrass and Unvegetated Nurseries of Chesapeake Bay. Journal of Experimental Marine Biology and Ecology 319(1-2): 57-68.

Seliskar, D.M. and J.L. Gallagher. 1983. The ecology of tidal marshes of the Pacific Northwest Coast: a community profile. U.S. Fish and Wildlife Service, Division of Biological Services. Washington , D.C. FWS/OBS-82/32. 65 p.

Selong, J.H., T.E. McMahon, A.V. Zale, and F.T. Barrows. 2001. Effect of Temperature on Growth and Survival of Bull Trout, with Application of an Improved Method for Determining Thermal Tolerance in Fishes. Transactions of the American Fisheries Society 130(6): 1026-1037.

Sergeant, C.J., and D.A. Beauchamp. 2006. Effects of Physical Habitat and Ontogeny on Lentic Habitat Preferences of Juvenile Chinook Salmon. Transactions of the American Fisheries Society 135(5): 1191-1204.

Servizi, J.A. and D.W. Martens. 1987. Some effects of suspended Fraser River sediments on sockeye salmon (Oncorhynchus nerka). In H.D. Smith, L. Margolis, and C.C. Wood, eds. Sockeye salmon (Oncorhynchus nerka) population biology and future management. Canadian Special Publication of Fisheries and Aquatic Sciences 96: 254-264.

Servizi, J.A., and D.W. Martens. 1991. Effect of Temperature, Season, and Fish Size on Acute Lethality of Suspended Sediments to Coho Salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 48(3): 493-497.

Servizi, J.A., and D.W. Martens. 1992. Sublethal Responses of Coho Salmon (Oncorhynchus kisutch) to Suspended Sediments. Canadian Journal of Fisheries and Aquatic Sciences 49(7): 1389-1395.

Servizi, J.A., and R.W. Gordon. 1990. Acute Lethal Toxicity of Ammonia and Suspended Sediment Mixtures to Chinook Salmon (Oncorhynchus tshawytscha). Bulletin of Environmental Contamination and Toxicology 44(4): 650-656.

Seymour, R., R.T. Guza, W. O'Reilly, and S. Elgar. 2005. Rapid Erosion of a Small Southern California Beach Fill. Coastal Engineering 52(2): 151-158.

Shafer, D.J. 1999. The Effects of Dock Shading on the Seagrass Halodule Wrightii in Perdido Bay, Alabama. Estuaries 22(4): 936-943.

Shafer, D.J. 2002. Recommendations to Minimize Potential Impacts to Seagrasses from Single-Family Residential Dock Structures in the Pacific Northwest. Seattle, Washington: U.S. Army Corps of Engineers, Seattle District.

Shaffer, J.A., D. Penttila, M. McHenry, and D. Vilella. 2007. Observations of Eulachon, Thaleichthys pacificus, in the Elwha River, Olympic Peninsula, Washington. Northwest Science 81(1): 76-81.

Shafroth, P.B., J.M. Friedman, G.T. Auble, M.L. Scott, and J.H. Braatne. 2002. Potential Responses of Riparian Vegetation to Dam Removal. Bioscience 52(8): 703-712.

Sharber, N.G., and S.W. Carothers. 1988. Influence of Electrofishing Pulse Shape on Spinal Injuries in Adult Rainbow Trout. North American Journal of Fisheries Management 8: 117-122.

Shaw, D.G., J.W. Farrington, M.S. Connor, B.W. Tripp, and J.R. Schubel. 1999. Potential Environmental Consequences of Petroleum Exploration and Development on Georges Bank, Report 99-3. Boston, Massachusetts: New England Aquarium.

Shaw, T.C. 1996. Effectiveness of an Excluder Device in Preventing the Entrainment of Benthic Invertebrates and Demersal Fishes in Grays Harbor, Washington. Proceedings of the Western Dredging Association Seventeenth Technical Conference.

Sheibley, R.W., A.P. Jackman, J.H. Duff, and F.J. Triska. 2003. Numerical Modeling of Coupled Nitrification-Denitrification in Sediment Perfusion Cores from the Hyporheic Zone of the Shingobee River, Minnesota. Advances in Water Resources 26(9): 977-987.

Sheibley, R.W., D.S. Ahearn, and R.A. Dahlgren. 2006. Nitrate Loss from a Restored Floodplain in the Lower Cosumnes River, California. Hydrobiologia 571: 261-272.

Sheibley, R.W., J.H. Duff, A.P. Jackman, and F.J. Triska. 2003. Inorganic Nitrogen Transformations in the Bed of the Shingobee River, Minnesota: Integrating Hydrologic and Biological Processes Using Sediment Perfusion Cores. Limnology and Oceanography 48(3): 1129-1140.

Sheldon, D., T. Hruby, P. Johnson, K. Harper, A. McMillan, T. Granger, S. Stanley, and E. Stockdale. 2005. Wetlands in Washington State - Volume 1: A Synthesis of the Science. Olympia, Washington: Washington State Department of Ecology. Publication #05-06-006. March 2005.

Sheldon, R.B., and C.W. Boylen. 1977. Maximum Depth Inhabited by Aquatic Vascular Plants. American Midland Naturalist 97: 248-254.

Sheoran, A.S., and V. Sheoran. 2006. Heavy Metal Removal Mechanism of Acid Mine Drainage in Wetlands: A Critical Review. Minerals Engineering 19(2): 105-116.

Shepard, B.B., K.L. Pratt, and P.J. Graham. 1984. Life Histories of Westslope Cutthroat Trout and Bull Trout in the Upper Flathead River Basin, Montana. Helena, Montana: Montana Department of Fish, Wildlife and Parks.

Shephard, B., R. Spoon, and L. Nelson. 2002. A Native Westslope Cutthroat Trout Population Responds Positively after Brook Trout Removal and Habitat Restoration. Intermountain Journal of Sciences 8(3): 193-214.

Sherwood, C.R., D.A. Jay, R.B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical Changes in the Columbia River Estuary. Progress in Oceanography 25(1-4): 299-352.

Shields, F., C. Cooper, and S. Knight. 1995. Experiment in Stream Restoration. Journal of Hydraulic Engineering 121: 494-502.

Shields, F.D. 1991. Woody Vegetation and Riprap Stability Along the Sacramento River Mile 84.5-119. Water Resources Bulletin 27(3): 527-536

Shields, F.D., and D.H. Gray. 1992. Effects of Woody Vegetation on Levee Integrity. Water Resources Bulletin 28: 917-931.

Shields, F.D., Jr., and C.J. Gippel. 1995. Prediction of effects of woody debris removal on flow resistance. Journal of Hydraulic Engineering 121(4), 341-354.

Shields, F.D., S.S. Knight, and C.M. Cooper. 1998. Rehabilitation of Aquatic Habitats in Warmwater Streams Damaged by Channel Incision in Mississippi. Hydrobiologia 382: 63-86.

Shipman, H., 2001. Beach Nourishment on Puget Sound: A Review of Existing Projects. Puget Sound Research Conference, Bellevue, Washington, February 12 – 14, 2001. pp. 1-8.

Shively, R.S., R.A. Tabor, R.D. Nelle, D.B. Jepsen, J.H. Petersen, S.T. Sauter, and T.P. Poe. 1991. System-wide significance of predation on juvenile salmonids in the Columbia and Snake river systems. U.S. Fish and Wildlife Service, Annual Report. Project 90-078 (Contract DE-AI79-90BP07096). Cook, Washington. Cited in Carrasquero 2001.

Shpigel, M., and L. Fishelson. 1989. Habitat Partitioning between Species of the Genus Cephalopholis (Pisces, Serranidae) across the Fringing-Reef of the Gulf of Aqaba (Red-Sea). Marine Ecology-Progress Series 58(1-2): 17-22.

Shreffler, D. K. and R. Moursund. 1999. Impacts of ferry terminals on migrating juvenile salmon along Puget Sound shorelines: Phase II field studies at Port Townsend Ferry Terminal. Contract GCA-1723. Olympia, Washington: Washington State Department of Transportation.

Shreffler, D.K., and W.M. Gardiner. 1999. Preliminary Findings of Diving and Light Surveys. Washington State Transportation Center.

Shreffler, D.K., C.A. Simenstad, and R.M. Thom. 1992. Foraging by juvenile salmon in a restored estuarine wetland. Estuaries 15(2):204-213.

Shteinman, B. and Y. Kameni. 1999. Study of long shore sediment transport in the vicinity of hydrotechnical constructions. In: Coastal Engineering and Marina Developments. C.A. Brebbia and P. Anagnostopoulos, eds. Southampton, Boston: WIT Press. Cited in Nightingale and Simenstad 2001b.

Shuman, J.R. 1995. Environmental Considerations for Assessing Dam Removal Alternatives for River Restoration. Regulated Rivers-Research & Management 11(3-4): 249-261.

Sibley, P.K., M.L. Harris, K.T. Bestari, T.A. Steele, R.D. Robinson, R.W. Gensemer, K.E. Day, and K.R. Solomon. 2001. Response of Zooplankton and Phytoplankton Communities to Liquid Creosote in Freshwater Microcosms. Environmental Toxicology and Chemistry 20(12): 2785-2793.

Sibley, P.K., M.L. Harris, K.T. Bestari, T.A. Steele, R.D. Robinson, R.W. Gensemer, K.E. Day, and K.R. Solomon. 2001. Response of zooplankton communities to liquid creosote in freshwater microcosms. Environmental Toxicology and Chemistry 20(2):394-405. Cited in Stratus 2005a.

Sibley, P.K., M.L. Harris, K.T. Bestari, T.A. Steele, R.D. Robinson, R.W. Gensemer, K.E. Day, and K.R. Solomon. 2004. Response of zooplankton and phytoplankton communities to creosote-impregnated Douglas fir pilings in freshwater microcosms. Archives of Environmental Contamination and Toxicology 47:56-66. Cited in Stratus 2005a.

Sigler, J.W. 1988. Effects of Chronic Turbidity on Anadromous Salmonids: Recent Studies and Assessment Techniques Perspective. In Effects of Dredging on Anadromous Pacific Coast Fishes, edited by C.A. Simenstad. Seattle, Washington: Washington Sea Grant Program.

Sigler, J.W. 1990. Effects of Chronic Turbidity on Anadromous Salmonids: Recent Studies and Assessment Techniques Perspective. September 8-9. pp. 26-37.

Sigler, J.W., T.C. Bjornn, and F.H. Everest. 1984. Effects of Chronic Turbidity on Density and Growth of Steelheads and Coho Salmon. Transactions of the American Fisheries Society 113(2): 142-150.

Sigler, John W. 1988. Effects of chronic turbidity on anadromous salmonids: Recent studies and assessment techniques perspective. In: C.A. Simenstad (ed.). Effects of dredging on anadromous Pacific coast fishes. Seattle, WA: Washington Sea Grant Program, University of Washington. 1990.

Sigourney, D.B., B.H. Letcher, and R.A. Cunjak. 2006. Influence of Beaver Activity on Summer Growth and Condition of Age-2 Atlantic Salmon Parr. Transactions of the American Fisheries Society 135(4): 1068-1075.

Silvester, R. 1977. The role of wave reflection in coastal processes. Proceedings of Coastal Sediments 1977 (American Society of Civil Engineers), 639-654. In Gabriel and Terich 2005.

Simenstad, C. A., K.L. Fresh, and E.O. Salo. 1982. The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function. Pages 343-364 in: V. S. Kennedy, editor. Estuarine Comparisons. Academic Press, Toronto, Ontario, Canada.

Simenstad, C., D. Reed, and M. Ford. 2006. When Is Restoration Not? Incorporating Landscape-Scale Processes to Restore Self-Sustaining Ecosystems in Coastal Wetland Restoration. Ecological Engineering 26(1): 27-39.

Simenstad, C.A. 1983. The ecology of estuarine channels of the Pacific Northwest: A community profile. U.S. Fish and Wildlife Service, Biological Services Program FWS/OBS 83/05. 250 pp.

Simenstad, C.A. 1990. Effects of Dredging on Anadromous Pacific Coast Fishes Summary and Conclusions from Workshop and Working Group Discussions. September 8-9, 1990. pp. 144-152.

Simenstad, C.A., and E.O. Salo. 1980. Foraging Success as a Determinant of Estuarine and Nearshore Carrying Capacity of Juvenile Chum Salmon (Oncorhynchus keta) in Hood Canal, Washington. Proceedings of the North Pacific Aquaculture Symptom, Fairbanks, Alaska, August. 18-27.

Simenstad, C.A., and R.M. Thom. 1996. Functional Equivalency Trajectories of the Restored Gog-Le-Hi-Te Estuarine Wetland. Ecological Applications 6(1): 38-56.

Simenstad, C.A., B. Nightingale, R.A. Thom, and D.K. Shreffler. 1999. Impacts of Ferry Terminals on Juvenile Salmon Migrating Along Puget Sound Shorelines: Phase I Synthesis of State of Knowledge. Research Project T9903 Task A2. Seattle, Washington: Washington State Transportation Center.

Simenstad, C.A., B. Nightingale, R.M. Thom, and D. K. Shreffler. 1999. Impacts of ferry terminals on juvenile salmon migrating along Puget Sound shorelines: Phase I synthesis of state of knowledge. Research Project T9903 Task A2. Washington State Transportation Center.

Simenstad, C.A., B.S. Miller, C.F. Nyblade, K. Thornburgh, and L.J. Bledsoe. 1979. Food web relationship of northern Puget Sound and the Strait of Juan de Fuca, EPA Interagency Agreement No. D6-E693-EN. Office of Environmental Engineering and Technology, US Environmental Protection Agency. Cited in NRC 2001 and in Nightingale and Simenstad 2001

Simenstad, C.A., J.R. Cordell, and L.A. Weitkamp. 1991. Effects of substrate modification on littoral flat meiofauna: Assemblage structure changes associated with adding gravel. FRI-UW-9124. Wetland Ecosystem Team, Fisheries Research Institute, School of Fisheries, University of Washington, Seattle, Washington. 91 pp.

Simenstad, C.A., J.R. Cordell, R.C. Wissmar, K.L. Fresh, S.L. Schroeder, M. Carr, G. Sanborn, and M. Burg. 1988. Assemblage Structure, Microhabitat Distribution, and Food Web Linkages of Epibenthic Crustaceans in Padilla Bay National Estuarine Research Reserve, Washington. FRI-UW-8813. Seattle, Washington: University of Washington, Fisheries Research Institute.

Simenstad, C.A., W.J. Kinney, S.S. Parker, E.O. Salo, J.R. Cordell, and H. Buechner. 1980. Prey Community Structure and Trophic Ecology of Outmigating Juvenile Chum and Pink Salmon in Hood Canal, Washington: A Synthesis of Three Years' Studies, 1977-1979. Seattle, Washington: Fisheries Research Institute, University of Washington.

Simmons, G.M. 1992. Importance of Submarine Groundwater Discharge (SGWD) and Seawater Cycling to Material Flux across Sediment Water Interfaces in Marine Environments. Marine Ecology-Progress Series 84(2): 173-184.

Simon, A. 1994. Gradation Processes and Channel Evolution in Modified West Tennessee Streams Process, Response, and Form. Professional Paper 1470, Denver, Colorado: U.S. Geological Survey, 84p.

Simon, A., and C.R. Hupp. 1992. Geomorphic and Vegetative Recovery Processes Along Modified Tennessee Streams: An Interdisciplinary Approach to Disturbed Fluvial Systems. Washington, DC: International Association of Hydrological Sciences.

Simonini, R., I. Ansaloni, F. Cavallini, F. Graziosi, M. Iotti, G.M. N'Siala, M. Mauri, G. Montanari, M. Preti, and D. Prevedelli. 2005. Effects of Long-Term Dumping of Harbor-Dredged Material on Macrozoobenthos at Four Disposal Sites Along the Emilia-Romagna Coast (Northern Adriatic Sea, Italy). Marine Pollution Bulletin 50(12): 1595-1605.

Sinclair, M. 1992. Marine Populations - an Essay on Population Regulation and Speciation. Books in Recruitment Fishery Oceanography. Seattle and London: Washington Sea Grant Program, University of Washington Press.

Sinsabaugh, R.L., M.P. Osgood, and S. Findlay. 1994. Enzymatic Models for Estimating Decomposition Rates of Particulate Detritus. Journal of the North American Benthological Society 13(2): 160-169.

Skalski, J.R., W.H. Pearson, and C.I. Malme. 1992. Effects of sounds from a geophysical survey device on catch-per-unit-effort in a hook-and-line fishery for rockfish (Sebastes spp.). Canadian Journal of Fisheries and Aquatic Sciences 49: 1357-1365.

Slate, L.O., F.D. Shields, J.S. Schwartz, D.D. Carpenter, and G. Freeman. 2007. Engineering design standards and liability for stream channel restoration. American Society of Civil Engineers Journal of Hydraulic Engineering 133: 1099-1102.

Smith, R.D., R.C. Sidle, P.E. Porter, and J.R. Noel. 1993. Effects of Experimental Removal of Woody Debris on the Channel Morphology of a Forest, Gravel-Bed Stream. Journal of Hydrology 152(1-4): 153-178.

Smith, S.D.A., and M.J. Rule. 2001. The Effects of Dredge-Spoil Dumping on a Shallow Water Soft-Sediment Community in the Solitary Islands Marine Park, NSW, Australia. Marine Pollution Bulletin 42(11): 1040-1048.

Smith, S.L., D.D. MacDonald, K.A. Keenleyside, C.G. Ingersoll, and L.J. Field. 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. Journal of Great Lakes Research 22(3):624-638. Cited in Stratus 2005b.

Smith, W.L. 1999. Local Structure-Induced Sediment Scour at Pile Groups. Master's Thesis, University of Florida, Gainesville, Florida.

Smits, J. 1998. Machines, Methods and Mitigation. Environmental Aspects of Dredging. The Hague, Netherlands: International Association of Dredging Companies (IADC).

Smokorowski, K.E., T.C. Pratt, W.G. Cole, L.J. McEachern, and E.C. Mallory. 2006. Effects on Periphyton and Macroinvertebrates from Removal of Submerged Wood in Three Ontario Lakes. Canadian Journal of Fisheries and Aquatic Sciences 63(9): 2038-2049.

Snoeyink, V.L., and D. Jenkins. 1980. Water Chemistry. New York: Wiley.

Snyder, D.E. 2003. Electrofishing and its harmful effects on fish. Information and Technology Report USGS/BRD/ITR-2003-0002. U.S. Government Printing Office, Denver, CO. U.S. Geological Survey. 149 p.

Snyder, N.P., S.A. Wright, C.N. Alpers, L.E. Flint, C.W. Holmes, and D.M. Rubin. 2006. Reconstructing Depositional Processes and History from Reservoir Stratigraphy: Englebright Lake, Yuba River, Northern California. Journal of Geophysical Research-Earth Surface 111(F4).

Soar, P.J. and C.R. Thorne. 2001. Channel restoration design for meandering rivers. U.S. Army Corp of Engineers. Report ERDC/CHL. September.

Sobocinski, K. 2003. The Impact of Shoreline Armoring on Supratidal Beach Fauna of Central Puget Sound. Master’s Thesis, University of Washington, Seattle, Washington, 92 pp.

Soil Conservation Service. 1968. Vegetative Tidal Bank Stabilization. College Park, Maryland: U.S. Department of Agriculture Soil Conservation Service.

Sommer, T.R., M.L. Nobriga, W.C. Harrell, W. Batham, and W.J. Kimmerer. 2001. Floodplain Rearing of Juvenile Chinook Salmon: Evidence of Enhanced Growth and Survival. Canadian Journal of Fisheries and Aquatic Sciences 58(2): 325-333.

Sommer, T.R., W.C. Harrell, and M.L. Nobriga. 2005. Habitat Use and Stranding Risk of Juvenile Chinook Salmon on a Seasonal Floodplain. North American Journal of Fisheries Management 25(4): 1493-1504.

Sotir, R.B. and J.C. Fischenich. 2001. Live and Inert Fascine Streambank Erosion Control. USACE Ecosystem Management and Restoration Research Program.

Soulsby, C., A. F. Youngson, H. J. Moir and I. A. Malcolm. 2001. Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: a preliminary assessment. The Science of the Total Environment. Volume 265, Issues 1-3, 29 January 2001, Pages 295-307.

Southard, S.L., R.M. Thom, G.D. Williams, J.D. Toft, C.W. May, G.A.M. Michael, J.A. Vucelick, J.T. Newell, and J.A. Southard. 2006. Impacts of Ferry Terminals on Juvenile Salmon Movement Along Puget Sound Shorelines. Prepared for Washington State Department of Transportation by Battelle Memorial Institute, Pacific Northwest Division, Richland, Washington.

Spadaro, P.A., D.W. Templeton, G.L. Hartman, and T.S. Wang. 1993. Predicting Water-Quality During Dredging and Disposal of Contaminated Sediments from the Sitcum Waterway in Commencement Bay, Washington, USA. Water Science and Technology 28(8-9): 237-254.

Spanhoff, B., W. Riss, P. Jakel, N. Dakkak, and E.I. Meyer. 2006. Effects of an Experimental Enrichment of Instream Habitat Heterogeneity on the Stream Bed Morphology and Chironomid Community of a Straightened Section in a Sandy Lowland Stream. Environmental Management 37(2): 247-257.

Spence, B.C, G.A. Lomnicky, R.M. Hughes, and R.P. Novitzki. 1996. An Ecosystem Approach to Salmonid Conservation. TR-4501-96-6057. ManTech Environmental Research Services Corp., Corvallis, Oregon. (December 1996). http://www.nwr.noaa.gov/1habcon/habweb/habguide/ManTech/front.htm

Speybroeck, J., D. Bonte, W. Courtens, T. Gheskiere, P. Grootaert, J.P. Maelfait, M. Mathys, S. Provoost, K. Sabbe, E.W.M. Stienen, V. Van Lancker, M. Vincx, and S. Degraer. 2006. Beach Nourishment: An Ecologically Sound Coastal Defense Alternative? A Review. Aquatic Conservation-Marine and Freshwater Ecosystems 16(4): 419-435.

Sprague, J.B. 1964. Avoidance of Copper-Zinc Solutions by Young Salmon in the Laboratory. Journal of the Water Pollution Control Federation 36: 990-1004. (As cited in Stratus 2005b).

Sprague, J.B. 1968. Avoidance reactions of rainbow trout to zinc sulfate solutions. Water Research 2:367-372. (As cited in Stratus 2005b).

Sridhar, V., A.L. Sansone, J. LaMarche, T. Dubin, and D.P. Lettenmaier. 2004. Prediction of Stream Temperature in Forested Watersheds. Journal of the American Water Resources Association 40(1): 197-213.

Stadler, J., NOAA Fisheries, Seattle, Washington. 2007. Email regarding NOAA Fisheries use of the Practical Spreading Loss model to estimate underwater noise intensity for the purpose of ESA consultation; sent to Eric Doyle of Herrera Environmental Consultants, Inc. Seattle, Washington, August 29, 2007.

Stamp, M.L., and J.C. Schmidt. 2006. Predicting Channel Responses to Flow Diversions. Stream Notes: To Aid in Securing Favorable Conditions of Water Flows. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Stanford, J.A., and J.V. Ward. 1992. Management of Aquatic Resources in Large Catchments: Recognizing Interactions between Ecosystem Connectivity and Environmental Disturbance. In Watershed Management – Balancing Sustainability and Environmental Change, edited by R.J. Naiman. New York: Springer-Verlag.

Stanford, J.A., and J.V. Ward. 2001. Revisiting the Serial Discontinuity Concept. Regulated Rivers-Research & Management 17(4-5): 303-310.

Stanford, J.A., J.V. Ward, W.J. Liss, C.A. Frissell, R.N. Williams, J.A. Lichatowich, and C.C. Coutant. 1996. A General Protocol for Restoration of Regulated Rivers. Regulated Rivers: Research & Management 12(4-5): 391-413.

Stanley, D.R., and C.A. Wilson. 2004. Effect of Hypoxia on the Distribution of Fishes Associated with a Petroleum Platform Off Coastal Louisiana. North American Journal of Fisheries Management 24(2): 662-671.

Stanley, E.H., and M.W. Doyle. 2002. A Geomorphic Perspective on Nutrient Retention Following Dam Removal. Bioscience 52(8): 693-701.

Stanley, E.H., M.A. Luebke, M.W. Doyle, and D.W. Marshall. 2002. Short-Term Changes in Channel Form and Macroinvertebrate Communities Following Low-Head Dam Removal. Journal of the North American Benthological Society 21(1): 172-187.

Starr, R.M., D.S. Fox, M.A. Hixon, B.N. Tissot, G.E. Johnson, and W.H. Barss. 1996. Comparison of Submersible-Survey and Hydroacoustic Survey Estimates of Fish Density on a Rocky Bank. Fishery Bulletin 94: 113-123.

Stasse, H.L. and H.S. Rogers. 1965. 1958 cooperative creosote project. II Marine tests. Analysis of marine panels after exposure for one to four years. In Proceedings of the Annual Meeting of the American Wood-Preservers' Association 61:81-85. Cited in Stratus 2005a.

Staurnes, M., F. Kroglund, and B.O. Rosseland. 1995. Water quality requirement of Atlantic salmon (Salmo salar) in water undergoing acidification or liming in Norway. Water Air and Soil Pollution 85: 347-352.

Stehr, C.M., D.W. Brown, T. Hom, B.F. Anulacion, W.L. Reichert, and T.K. Collier. 2000. Exposure of Juvenile Chinook and Chum Salmon to Chemical Contaminants in the Hylebos Waterway of Commencement Bay, Tacoma, Washington. Journal of Aquatic Ecosystem Stress and Recovery 7: 215-227.

Stein, D. and T.J. Hassler. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific southwest): Brown rockfish, copper rockfish, and black rockfish. U.S. Fish Wildlife Service, Biological Report No. 82 (11.113): 15pp. Cited in NRC 2001.

Stein, J.E., T. Hom, T.K. Collier, D.W. Brown, and U. Varanasi. 1995. Contaminant Exposure and Biochemical Effects in Outmigrant Juvenile Chinook Salmon from Urban and Nonurban Estuaries of Puget-Sound, Washington. Environmental Toxicology and Chemistry 14(6): 1019-1029.

Stein, J.N. 1970. A study of the largemouth bass population in Lake Washington. Masters thesis. University of Washington, College of Fisheries, Seattle. Cited in Carrasquero 2001.

Stevens, D.E., D.W. Kohlhorst, L.W. Miller, and D.W. Kelley. 1985. The Decline of Striped Bass in the Sacramento-San Joaquin Estuary, California. Transactions of the American Fisheries Society 114(1): 12-30.

Stevens, P.W., C.L. Montague, and K.J. Sulak. 2006. Fate of Fish Production in a Seasonally Flooded Saltmarsh. Marine Ecology-Progress Series 327: 267-277.

Stewart, T.L., and J.F. Martin. 2005. Energy Model to Predict Suspended Load Deposition Induced by Woody Debris: Case Study. Journal of Hydraulic Engineering-American Society of Civil Engineers 131(11): 1011-1016.

Stock, J.D., D.R. Montgomery, B.D. Collins, W.E. Dietrich, and L. Sklar. 2005. Field Measurements of Incision Rates Following Bedrock Exposure: Implications for Process Controls on the Long Profiles of Valleys Cut by Rivers and Debris Flows. Geological Society of America Bulletin 117(1): 174-194.

Stocker, M. 2002. Fish Mollusks and other Sea Animals, and the Impact of Anthropogenic Noise in the Marine Acoustical Environment. Prepared for Earth Island Institute by Michael Stocker Associates. Cited in WSDOT 2006a.

Stone, J., and S. Barndt. 2005. Spatial Distribution and Habitat Use of Pacific Lamprey (Lampetra Tridentata) Ammocoetes in a Western Washington Stream. Journal of Freshwater Ecology 20(1): 171-185.

Storry, K.A., C.K. Weldrick, M. Mews, M. Zimmer, and D.E. Jelinski. 2006. Intertidal coarse woody debris: A spatial subsidy as shelter or feeding habitat for gastropods? Estuarine Coastal and Shelf Science 66(1-2): 197-203.

Stratus (Stratus Consulting Inc., Duke University). 2005a. Creosote-Treated Wood in Aquatic Environments: Technical Review and Use Recommendations. Prepared for the National Oceanic and Atmospheric Administration's National Marine Fisheries Service.

Stratus (Stratus Consulting Inc.). 2005b. Treated Wood in Aquatic Environments: Technical Review and Use Recommendations. Prepared for NOAA Fisheries Habitat Conservation Division Southwest Habitat Conservation Division, Joe Dillon, by Stratus Consulting, Santa Rosa, California.

Streamnet Database. 2007. Available at: www.streanet.org/online-data/query-intro.html.

Strickland, J.D. 1958. Solar Radiation Penetrating the Ocean. A Review of Requirements, Data, and Methods of Measurement with Particular Reference to Photosynthetic Productivity. Fisheries Research Board of Canada 15: 453-493.

Sturm, T.W. 2004. Enhanced Abutment Scour Studies for Compound Channels. Report No. FHWA-RD-99-156, Office of Infrastructure Research and Development, Federal Highway Administration, McLean, VA.

Sudduth, E.B., and J.L. Meyer. 2006. Effects of Bioengineered Streambank Stabilization on Bank Habitat and Macroinvertebrates in Urban Streams. Environmental Management 38(2): 218-226.

Sukhodolov, A., C. Engelhardt, A. Kruger, and H. Bungartz. 2004. Case study: Turbulent flow and sediment distributions in a groyne field. Journal of Hydraulic Engineering-American Society of Civil Engineers 130(1): 1-9.

Sullivan, K., D.J. Martin, R.D. Cardwell, J.E. Toll, and S. Duke. 2000. An Analysis of the Effects of Temperature on Salmonids of the Pacific Northwest with Implications for Selecting Temperature Criteria. Portland, Oregon: Sustainable Ecosystems Institute.

Sullivan, K., J. Tooley, K. Doughty, J.E. Caldwell, and P. Knudsen. 1990. Evaluation of Prediction Models and Characterization of Stream Temperature Regimes in Washington. TFW-WQ3-90-006. Olympia, Washington: Washington Department of Natural Resources.

Sumer, B.M., J. Fredsoe, A. Lamberti, B. Zanuttigh, M. Dixen, K. Gislason, and A.F. Di Penta. 2005. Local scour at roundhead and along the trunk of low crested structures. Coastal Engineering 52(10-11): 995-1025.

Sumida, B.Y. and H.G. Moser. 1984. Food and feeding of Bocaccio and comparison with Pacific hake larvae in the California current. California Cooperative Oceanic Fisheries Investigations Report 25:112-118. Cited in NRC 2001.

Suren, A.M., and S. McMurtrie. 2005. Assessing the Effectiveness of Enhancement Activities in Urban Streams: II. Responses of Invertebrate Communities. River Research and Applications 21(4): 439-453.

Sutherland, A., and D. Ogle. 1975. Effect of Jet Boats on Salmon Eggs. New Zealand Journal of Marine and Freshwater Research 9(3): 273-282.

Suttle, K.B., M.E. Power, J.M. Levine, and C. McNeely. 2004. How Fine Sediment in Riverbeds Impairs Growth and Survival of Juvenile Salmonids. Ecological Applications 14(4): 969-974.

Swales, S., and C.D. Levings. 1989. Role of Off-Channel Ponds in the Life-Cycle of Coho Salmon (Oncorhynchus-Kisutch) and Other Juvenile Salmonids in the Coldwater River, British-Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46(2): 232-242.

Swanson, C., P.S. Young, and J.J. Cech. 2005. Close Encounters with a Fish Screen: Integrating Physiological and Behavioral Results to Protect Endangered Species in Exploited Ecosystems. Transactions of the American Fisheries Society 134(5): 1111-1123.

Swanston, D.N. 1991. Natural processes. Pp. 83-138 in W.R. Meehan (ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. Bethesda, Maryland: American Fisheries Society Special Publication 19.

Swartz, R.C. 1989. Acute Toxicity of Sediment from Eagle Harbor Washington, to the Infaunal Amphipod Rhepolxynius Abronius. Environmental Toxicology and Chemistry 8: 215-222.

Swartz, R.C., P.F. Kemp, D.W. Schults, G.R. Ditsworth, and R.J. Ozretich. 1989. Acute Toxicity of Sediment from Eagle Harbor, Washington, to the Infaunal Amphipod Rhepoxynius abronius. Environmental Toxicology and Chemistry 8(3): 215-222.

Sweeney, B.W., T.L. Bott, J.K. Jackson, L.A. Kaplan, J.D. Newbold, L.J. Standley, W.C. Hession, and R.J. Horwitz. 2004. Riparian Deforestation, Stream Narrowing, and Loss of Stream Ecosystem Services. Proceedings of the National Academy of Sciences of the United States of America 101(39): 14132-14137.

Sweka, J., and K. Hartman. 2006. Effects of Large Woody Debris Addition on Stream Habitat and Brook Trout Populations in Appalachian Streams. Hydrobiologia 559(1): 363-378.

Tabor, R. U.S. Fish and Wildlife Service. Personal communication, telephone conversation with Tom Kahler (The Watershed Company), 9 June 2000, and 11 July 2000. Cited in Kahler et al. 2000.

Tabor, R., F. Mejia, and D. Low. 2000. Predation of juvenile salmon by littoral fishes in the Lake Washington–Lake Union ship canal, preliminary results. Prepared for presentation at the American Fisheries Society, North Pacific International Chapter. 2000 conference, April 10–12, 2000, Mount Vernon, Washington. Cited in Carrasquero 2001.

Tabor, R., G. Brown, A. Hird, and S. Hager. 2001. The Effect of Light Intensity on Predation of Sockeye Salmon Fry by Cottids in the Cedar River. Lacey, Washington: U.S. Fish and Wildlife Service, Western Washington Office, Fisheries and Watershed Assessment Division.

Tabor, R.A., G. Brown, and V.T. Luiting. 1998. The effect of light intensity on predation of sockeye salmon fry by prickly sculpin and torrent sculpin. Lacey, Washington: U.S. Fish and Wildlife Service Western Washington Office, Aquatic Resources Division.

Tabor, R.A., H.A. Gearns, C.M. McCoy III, and S. Camacho. 2006. Nearshore Habitat Use by Juvenile Chinook Salmon in Lentic Systems, 2003 and 2004 report. United States Fish and Wildlife Service, Lacey, WA.

Tabor, R.A., J.A. Scheurer, H.A Gearns, and E.P. Bixler. 2004. Nearshore habitat use by juvenile Chinook salmon in lentic systems of the Lake Washington Basin. Annual Report, 2002, to Seattle Public Utilities, Seattle, Washington. 58 p.

Tabor, R.A., M.T. Celedonia, F. Mejia, R.M. Piaskowski, D. Low, and L. Park. 2004. Predation of Juvenile Chinook Salmon by Predatory Fishes in Three Areas of the Lake Washington Basin. Lacey, Washington: U.S. Fish and Wildlife Service.

Tabor, R.A., R.S. Shively, and T.P. Poe. 1993. Predation on juvenile salmonids by smallmouth bass and northern squawfish in the Columbia River near Richland, Washington. North American Journal of Fisheries Management 13:831–838. 1993. Cited in Carrasquero 2001.

Taft, E.P., Ill, and Y.G. Mussalli. 1978. Fish Diversion and Transportation System for Power Plant Application. Fisheries 3(3): 2-5.

Takami, T., F. Kitano, and S. Nakano. 1997. High Water Temperature Influences on Foraging Responses and Thermal Deaths of Dolly Varden Salvelinus malma and White-Spotted Char S. leucomaenis in a Laboratory. Fisheries Science 63(1): 6-8.

Tanner, C.D., J.R. Cordell, J. Rubey, and L.M. Tear. 2002. Restoration of Freshwater Intertidal Habitat Functions at Spencer Island, Everett, Washington. Restoration Ecology 10(3): 564-576.

Tarela, P.A., and A.N. Menendez. 2002. Numerical Simulation of the Wave Pattern within a Harbor Due to Ship Waves. International Journal of Computational Fluid Dynamics 16(4): 315-325.

Taylor, W. S. and W.S. Wiley. 1997. Port of Seattle Fish Mitigation Study: Pier 64/65 Short-Stay Moorage Facility: Qualitative Fish and Avian Predator Observations., Seattle, WA. Cited in Nightingale and Simenstad 2001b.

Teachout, E., Fish and Wildlife Biologist with U.S. Fish and Wildlife Service, Lacey, Washington. 2007. Email regarding USFWS use of the Practical Spreading Loss model to estimate underwater noise intensity for the purpose of ESA consultation; sent to Julie Hampden of Herrera Environmental Consultants, Inc. Seattle, Washington, August 29, 2007.

Teels, B.M., C.A. Rewa, and J. Myers. 2006. Aquatic Condition Response to Riparian Buffer Establishment. Wildlife Society Bulletin 34(4): 927-935.

Tenera Environmental. 2005. Draft - Carlsbad Desalination Facility Intake Effects Assessment. Prepared for Poseidon Resources Corporation by Tenera Environmental, Lafayette, California.

Teodoru, C., and B. Wehrli. 2005. Retention of Sediments and Nutrients in the Iron Gate I Reservoir on the Danube River. Biogeochemistry 76(3): 539-565.

Terich, T.A. 1987. Living with the Shore of Puget Sound and the Georgia Strait. Living with the Shore. Edited by O.H. Pilkey and W.J. Neal. Durham, North Carolina: Duke University Press.

Terich, T.A., and M.L. Schwartz. 1990. The Effects of Seawalls and Other Hard Erosion Protection Structures Upon Beaches: An Annotated Bibliography and Summary. Washington State Shorelands and Coastal Zone Management Program.

Terrados, J., C.M. Duarte, M.D. Fortes, J. Borum, N.S.R. Agawin, S. Bach, U. Thampanya, L. Kamp-Nielsen, W.J. Kenworthy, O. Geertz-Hansen, and J. Vermaat. 1998. Changes in

Community Structure and Biomass of Seagrass Communities Along Gradients of Siltation in SE Asia. Estuarine Coastal and Shelf Science 46(5): 757-768.

Thayer, G. W., W.J. Kenworthy, and M.S. Fonseca. 1984. The Ecology of Eelgrass Meadows of the Atlantic Coast: A Community Profile, FWS/OBSO-84/02. U.S. Fish and Wildlife Service, Washington D.C. Cited in Nightingale and Simenstad 2001b.

The Watershed Company. 2006. Shoreline Analysis Report. Kirkland, Washington: City of Kirkland.

Theurer, F.D., K.A. Voos, and W.J. Miller. 1984. Instream Water Temperature Model. Instream Flow Informational Paper 16 FWS/OBS-84/15. U.S. Fish and Wildlife Service.

Thom, R. M., and D.K. Shreffler. 1996. Eelgrass Meadows Near Ferry Terminals in Puget Sound. Characterization of Assemblages and Mitigation Impacts. Battelle Pacific Northwest Laboratories, Sequim, WA. Cited in Nightingale and Simenstad 2001b.

Thom, R. M., D. K. Shreffler, and J. Schafer. 1995. Mitigation plan for impacts to subtidal vegetation associated with reconstruction and expansion of the ferry terminal at Clinton, Whidbey Island, Washington. PNL-10844. Prepared for the Washington State Department of Transportation. Battelle, Pacific Northwest Division of Battelle Memorial Institute, Sequim, WA. 25pp. Cited in Haas et al. 2002.

Thom, R., C.A. Simenstad, J.R. Cordell, and E.O. Salo. 1988. Fisheries Mitigation Plan for Expansion of Moorage at Blaine Marina. Fishery Resource Institute, University of Washington.

Thom, R.M. 1990. A Review of Eelgrass (Zostera marina L) Transplanting Projects in the Pacific-Northwest. Northwest Environmental Journal 6(1): 121-137.

Thom, R.M. and D.K. Shreffler. 1994. The Ecological Consequences of Accelerated Sedimentation on Estuarine Processes, Estuarine Organisms, the Gain and Loss of Certain Habitats. PNWD-SA-4202A. Presented at the Forum on Impacts of Erosion and Sedimentation in the Tillamook Bay and its Watershed, sponsored by the Tillamook Bay National Estuary Project, January 12, 1995, Garibaldi, Oregon.

Thom, R.M., A.B. Borde, P.J. Farley, M.C. Horn and A. Ogston. 1996. Passenger-Only Ferry Propeller Wash Study: Threshold Velocity Determinations and Field Study, Vashon Terminal. Battelle Marine Sciences Laboratory Report to WSDOT. PNWD-2376/UC- 000. 15+pp. Cited in Simenstad et al. 1999, in Nightingale and Simenstad 2001b.

Thom, R.M., Bourne, A.B. Borde, G.D. Williams, J.A. Southard, S.L. Blanto, D.L. Woodruff. 2001. Effects of Environmental Stressors on Eelgrass Restoration Projects. Puget Sound Research. ’01. Conference Proceedings. Available: http://www.psat.wa.gov/Publications/01_proceedings/sessions/oral/7b_thom.pdf (Accessed 2006.10 02).

Thom, R.M., C.A. Simenstad, J.R. Cordell, and E.O. Salo. 1989. Fish and Their Epibenthic Prey in a Marina and Adjacent Mudflats and Eelgrass Meadow in a Small Estuarine Bay. Seattle, Washington: University of Washington.

Thom, R.M., D.K. Shreffler, and K. Macdonald. 1994. Shoreline Armoring Effects on Coastal Ecology and Biological Resources in Puget Sound, Washington. Report 94-80. Shorelands and Environmental Assistance Program, Washington Department of Ecology. Cited in Nightingale and Simenstad 2001b.

Thom, R.M., G. Williams, A. Borde, J. Southard, S. Sargeant, D. Woodruff, J.C. Laufle, and S. Glasoe. 2005. Adaptively Addressing Uncertainty in Estuarine and near Coastal Restoration Projects. Journal of Coastal Research: 94-108.

Thom, R.M., L.D. Antrim, A.B. Borde, W.W. Gardiner, D.K. Shreffler, P.G. Farley, J.G. Norris, S. Wyllie-Echeverria, and T.P. McKenzie. 1998. 1997 Puget Sound's Eelgrass Meadows: Factors Contributing to Depth Distribution and Spatial Patchiness. Puget Sound Research '98, Conference Proceedings. Available at: http://www.psat.wa.gov/Publications/98_proceedings/pdfs/2c_thom.pdf (Accessed 2006.09.29). Cited in Nightingale and Simenstad 2001b.

Thom, R.M., R. Zeigler, and A.B. Borde. 2002. Floristic Development Patterns in a Restored Elk River Estuarine Marsh, Grays Harbor, Washington. Restoration Ecology 10(3): 487-496.

Thomalla, F., and C.E. Vincent. 2003. Beach Response to Shore-Parallel Breakwaters at Sea Palling, Norfolk, UK. Estuarine Coastal and Shelf Science 56(2): 203-212.

Thompson, D.M. 2002. Long-Term Effect of Instream Habitat-Improvement Structures on Channel Morphology Along the Blackledge and Salmon Rivers, Connecticut, USA. Environmental Management 29(2): 250-265.

Thompson, D.S. 1995. Substrate additive studies for the development of hardshell clam habitat in waters of Puget Sound in Washington State: an analysis of effects on recruitment, growth, and survival of the Manila clam, Tapes phillipinarum, and on the species diversity and abundance of existing benthic organisms. Estuaries 18(1A):91-107.

Thompson, G.G. 1991. Determining Minimum Viable Populations under the Endangered Species Act. National Oceanographic and Atmospheric Administration, National Marine Fisheries Service.

Thompson, K.G., E.P. Bergersen, R.B. Nehring, and D.C. Bowden. 1997. Long-Term Effects of Electrofishing on Growth and Body Condition of Brown and Rainbow Trout. North American Journal of Fisheries Management 17: 154-159.

Thoms, M.C. 2003. Floodplain-River Ecosystems: Lateral Connections and the Implications of Human Interference. Geomorphology 56(3-4): 335-349.

Thomson Scientific Web of Science. 2007. Available at: http://scientific.thomson.com/products/wos/ (accessed May–July 2007).

Thomson, J.R., D.D. Hart, D.F. Charles, T.L. Nightengale, and D.M. Winter. 2005. Effects of Removal of a Small Dam on Downstream Macroinvertebrate and Algal Assemblages in a Pennsylvania Stream. Journal of the North American Benthological Society 24(1): 192-207.

Thorkilsen, M., and C. Dynesen. 2001. An Owner's View of Hydroinformatics: Its Role in Realising the Bridge and Tunnel Connection between Denmark and Sweden. Journal of Hydroinformatics 3(2): 105-135.

Tiemann, J.S., D.P. Gillette, M.L. Wildhaber, and D.R. Edds. 2004. Effects of Lowhead Dams on Riffle-Dwelling Fishes and Macroinvertebrates in a Midwestern River. Transactions of the American Fisheries Society 133(3): 705-717.

Tiemann, J.S., H.R. Dodd, N. Owens, and D.H. Wahl. 2007. Effects of Lowhead Dams on Unionids in the Fox River, Illinois. Northeastern Naturalist 14(1): 125-138.

Tikkanen, P., P. Laasonen, T. Muotka, A. Huhta, and K. Kuusela. 1994. Short-Term Recovery of Benthos Following Disturbance from Stream Habitat Rehabilitation. Hydrobiologia 273(2): 121-130.

Tockner, K., D. Pennetzdorfer, N. Reiner, F. Schiemer, and J.V. Ward. 1999. Hydrological Connectivity, and the Exchange of Organic Matter and Nutrients in a Dynamic River-Floodplain System (Danube, Austria). Freshwater Biology 41(3): 521-535.

Toft, J., C. Simenstad, J. Cordell, and L. Stamatiou. 2004. Fish Distribution, Abundance, and Behavior at Nearshore Habitats along City of Seattle Marine Shorelines, with an Emphasis on Juvenile Salmonids. SAFS-UW-0401. Wetland Ecosystem Team, School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA. Prepared for Seattle Public Utilities, City of Seattle, WA.

Toft, J., C. Simenstad, J. Cordell, and L. Stamatiou. 2007. Fish Distribution, Abundance, and Behavior along City Shoreline Types in Puget Sound. North American Journal of Fisheries Management 27: 465-480.

Toft, J.D. and J. Cordell. 2006. Olympic Sculpture Park: results from pre-construction biological monitoring of shoreline habitats. Technical Report SAFS-UW-0601, School of Aquatic and Fishery Sciences, University of Washington. Prepared for Seattle Public Utilities, City of Seattle. 36 pp.

Tonina, D. and J.M. Buffington. 2003. Effects of Discharge on Hyporheic Flow in a Pool-Riffle Channel: Implications for Aquatic Habitat. Eos, Transactions, American Geophysical Union 84 (46), Fall Meeting Supplement, Abstract H52A-1154.

Tonina, D. and J.M. Buffington. 2005. Biogeomorphology: Effects of Salmon Redds on River Hydraulics and Hyporheic Flow in Gravel-Bed Rivers. Eos, Transactions, American Geophysical Union 86(52), Fall Meet Supplement, Abstract H53D-0493.

Tonina, D. and J.M. Buffington. 2007. Hyporheic Exchange in Gravel Bed Rivers with Pool-Riffle Morphology: Laboratory Experiments and Three-Dimensional Modeling. Water Resources Research 43(1): 1-16.

Tops, S., W. Lockwood, and B. Okamura. 2006. Temperature-Driven Proliferation of Tetracapsuloides Bryosalmonae in Bryozoan Hosts Portends Salmonid Declines. Diseases of Aquatic Organisms 70(3): 227-236.

Travnichek, V.H., A.V. Zale, and W.L. Fisher. 1993. Entrainment of Lchthyoplankton by a Warmwater Hydroelectric Facility. Transactions of the American Fisheries Society 122(5): 709-716.

Travnichek, V.H., M.B. Bain, and M.J. Maceina. 1995. Recovery of a Warmwater Fish Assemblage after the Initiation of a Minimum Flow-Release Downstream from a Hydroelectric Dam. Transactions of the American Fisheries Society 124: 836-844.

Tribble, S.C. 2000. Sensory and Feeding Ecology of Larval and Juvenile Pacific Sand Lance, Ammodytes hexapterus. Master’s Thesis, University of Washington.

Triska, F.J., V.C. Kennedy, R.J. Avanzino, G.W. Zellweger, and K.E. Bencala. 1989. Retention and Transport of Nutrients in a 3rd-Order Stream in Northwestern California - Hyporheic Processes. Ecology 70: 1893-1905.

Tucker, J. and C. Theiling. 1998. Freshwater mussels. Chapter 11 in USGS: Ecological Status and Trends of the Upper Mississippi River System. Available at http://www.umesc.usgs.gov/reports_publications/status_and_trends.html (Accessed 2006.10.04).

Turnpenny, A.W.H., K.P. Thatcher, and J.R. Nedwell. 1994. The Effects on Fish and Other Marine Animals of High-Level Underwater Sound. Report FRR 127/94. Prepared for UK Defense Research Agency by Fawley Aquatic Research.

Tyack, P.L. and C.W. Clark. Communication and acoustic behavior of dolphins and whales. In: Au, W.W.L., A.N. Popper, and R.R. Fay (eds.), Hearing by whales and dolphins. New York, Springer, 2000, pp. 156-224. Book.

Urban, E.R., and C.J. Langdon. 1984. Reduction in Costs of Diets for the American Oyster, Crassostrea-Virginica (Gmelin), by the Use of Non-Algal Supplements. Aquaculture 38(4): 277-291.

Urban, E.R., and D.L. Kirchman. 1992. Effect of Kaolinite Clay on the Feeding-Activity of the Eastern Oyster Crassostrea-Virginica (Gmelin). Journal of Experimental Marine Biology and Ecology 160(1): 47-60.

Urick, R.J. 1983. Principles of Underwater Sound. Ch. 7 In The Noise Background of the Sea, Los Altos, California: Peninsula Publishing.

USACE (U.S. Army Corps of Engineers). 1983. Engineering and Design - Dredging and Dredged Material Disposal. U.S. Army Corps of Engineers. Washington, DC: CECW-EH-D.

USACE (U.S. Army Corps of Engineers). 1991a. Dredging Research Technical Notes: Design Requirements for Capping. DRP-5-03. Vicksburg, Mississippi: U.S. Army Corps of Engineers Waterways Experiment Station.

USACE (U.S. Army Corps of Engineers). 1991b. Dredging Research Technical Notes: Equipment and Placement Techniques for Capping. DRP-5-05. Vicksburg, Mississippi: U.S. Army Corps of Engineers Waterways Experiment Station.

USACE (U.S. Army Corps of Engineers). 1991c. Dredging Research Technical Notes: Site Selection Considerations for Capping. DRP-5-04. Vicksburg, Mississippi: U.S. Army Corps of Engineers Waterways Experiment Station.

USACE (U.S. Army Corps of Engineers). 2002. Decision Document for Nationwide Permit 13: Bank Stabilization. Available at: http://www.usace.army.mil/cw/cecwo/reg/nw2002dd/NW_13_2002.pdf

USACE (U.S. Army Corps of Engineers). 2005. Department of the Army Regional General Permit 4: Piling, Floats, Ramps, Piers, Minor Fills, and Bank Stabilization in the Pend Oreille River and Leke Chelan, in the State of Washington. Permit Number: CENWS-OD-RG-RGP4.

USACE (U.S. Army Corps of Engineers). 2005. Regional General Permit No. 6. Maintenance, Modification and Construction of Residential Overwater Structures in Inland Marine Waters of Within the State of Washington. Regulatory, Seattle District. 48 p.

USACE (U.S. Army Corps of Engineers). 2005. Silt Curtains as a Dredging Project Management Practice. Washington, DC: U.S. Army Corps of Engineers, Engineer Research and Development Center.

USACE (U.S. Army Corps of Engineers). 2007. Dredging Operations and Environmental Research Program. March 2007. Available at U.S. Army Corps of Engineers website, URL = http://el.erdc.usace.army.mil/dots/doer/ (accessed September 3, 2007).

USA-Today. 2007. Tire Reef Off Florida Proves a Disaster. 2/17/2007. Available at: http://www.usatoday.com/news/nation/2007-02-17-florida-reef_x.htm (accessed July 16, 2007, 2007).

USEPA (U.S. Environmental Protection Agency). 1985. Freshwater Wetlands for Wastewater Management: Environmental Assessment Handbook. U. S. EPA 904/9-85-135. Atlanta, Georgia: U.S. Environmental Protection Agency, Region 4.

USEPA (U.S. Environmental Protection Agency). 1996. Calculation and Evaluation of Sediment Effect Concentrations for the Amphipod Hyalella azteca and the Midge Chironomus riparius. Assessment and Remediation of Contaminated Sediments (ARCS) Program. EPA 905-R96-008. U.S. Environmental Protection Agency, Washington, DC. Cited in Stratus 2005b.

USEPA (U.S. Environmental Protection Agency). 1999. EPA Guidance Manual Turbidity Provisions. Available at http://www.epa.gov/safewater/mdpd/pdf/turbidity/chap_o7.pdf (Accessed 2006.10.04).

USEPA (U.S. Environmental Protection Agency). 2001. National Management Measures Guidance to Control Nonpoint Source Pollution from Marinas and Recreational Boating. EPA 841-B-01-005. Washington, DC: U.S. Environmental Protection Agency, Office of Water.

USEPA (U.S. Environmental Protection Agency). 2002. National Recommended Water Quality Criteria: 2002. EPA 822-R-02-047. U.S. Environmental Protection Agency, Washington, DC. November. Cited in Stratus 2005b.

USEPA (U.S. Environmental Protection Agency). 2003. EPA Region 10 Guidance for Pacific Northwest State and Tribal Temperature Water Quality Standards. EPA 910-B-03-002. Region 10 Office of Water, Seattle, WA.

USEPA (U.S. Environmental Protection Agency). 2007. National Management Measures to Control Nonpoint Source Pollution from Hydromodification. EPA 841-B-07-002. Washington, DC: U.S. Environmental Protection Agency, Office of Water.

USFS (U.S. Forest Service). 2007. Fish Resources. U.S. Forest Service website. Available at: http://www.fs.fed.us/r6/fishing/forests/fishresources/win_coldwater.html#redband.

USFS, NMFS, USBLM, USFWS, USNPS, and USEPA. 1993. Forest Ecosystem Management: An Ecological, Economic, and Social Assessment. Report of the Forest Ecosystem Management Assessment Team: U.S. Forest Service, National Marine Fisheries Service, U.S. Bureau of Land Management, U.S. Fish and Wildlife Service, National Park Service, and U.S. Environmental Protection Agency. Portland, Oregon: U.S. Forest Service, Pacific Northwest Region.

USFWS (U.S. Fish and Wildlife Service). 1986. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest) Steelhead Trout. Biological Report 82 (11.82). TR EL-82-4. Lafayette, Louisiana. US Fish and Wildlife Service, Coastal Ecology Group.

USFWS (U.S. Fish and Wildlife Service). 2000. Impacts of riprapping to ecosystem functioning, Lower Sacramento River, California. Prepared for U.S. Army Corps of Engineers by U.S. Fish and Wildlife Service, Sacramento, California. June 2000.

USFWS (U.S. Fish and Wildlife Service). 2000. Impacts of riprapping to ecosystem functioning, Lower Sacramento River, California. Prepared for: U.S. Army Corps of Engineers, Sacramento District. Prepared by USFWS Sacramento Office. June 2000.

USFWS (U.S. Fish and Wildlife Service). 1998. A Framework to Assist in Making Endangered Species Act Determinations of Effect for Individual or Grouped Actions at the Bull Trout Subpopulation Watershed Scale. DRAFT. February.

USFWS (U.S.Fish and Wildlife Service). 2002. Draft Fish and Wildlife Coordination Act Report: Bogue Banks Shore Protection Project, Carteret County, NC. Raleigh, North Carolina: Raleigh Ecological Services Field Office, U.S. Fish and Wildlife Service.

USGS (US Geological Survey). 1997. Washington's Wetland Resources. October 14, 1997. Available at U.S. Geological Survey website: http://wa.water.usgs.gov/pubs/misc/wetlands/ (accessed July 5, 2007).

Utter, F. 2001. Patterns of Subspecific Anthropogenic Introgression in Two Salmonid Genera. Reviews in Fish Biology and Fisheries 10(3): 265-279.

Vagle, S. 2003. On the Impact of Underwater Pile Driving Noise on Marine Life. Government Report. Canada Department of Fisheries and Oceans, Institute of Ocean Sciences, Ocean Science and Productivity Division.

Valett, H.M., C.L. Crenshaw, and P.F. Wagner. 2002. Stream Nutrient Uptake, Forest Succession, and Biogeochemical Theory. Ecology 83(10): 2888-2901.

Valett, H.M., M.A. Baker, J.A. Morrice, C.S. Crawford, M.C. Molles, C.N. Dahm, D.L. Moyer, J.R. Thibault, and L.M. Ellis. 2005. Biogeochemical and Metabolic Responses to the Flood Pulse in a Semiarid Floodplain. Ecology 86(1): 220-234.

Vallania, A., and M.D. Corigliano. 2007. The Effect of Regulation Caused by a Dam on the Distribution of the Functional Feeding Groups of the Benthos in the Sub Basin of the Grande River (San Luis, Argentina). Environmental Monitoring and Assessment 124(1-3): 201-209.

Valovirta, I. 1990. Conservation of Margaritifera margaritifera in Finland. Council of Europe Environmental Encounters Series 10:59-63. Cited in Watters et al. 1999.

Van Eenennaam, J.P., J. Linares-Casenave, X. Deng, and S.I. Doroshov. 2005. Effect of Incubation Temperature on Green Sturgeon Embryos, Acipenser medirostris. Environmental Biology of Fishes 72(2): 145-154.

Vandenavyle, M.J., and M.A. Maynard. 1994. Effects of Saltwater Intrusion and Flow Diversion on Reproductive Success of Striped Bass in the Savanna River Estuary. Transactions of the American Fisheries Society 123(6): 886-903.

Vannote, R.L., and G.W. Minshall. 1982. Fluvial Processes and Local Lithology Controlling Abundance, Structure, and Composition of Mussel Beds. Proceedings of the National Academy of Science 79: 4103-4107.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The River Continuum Concept. Canadian Journal of Fisheries and Aquatic Science 37: 130-137.

Varanasi, U., E. Casillas, M.R. Arkoosh, T. Hom, D.A. Misitano, D.W. Brown, S.-L. Chan, T.K. Collier, B.B. McCain, and J.E. Stein. 1993. Contaminant Exposure and Associated Biological Effects in Juvenile Chinook Salmon (Oncorhynchus Tshawytscha) from Urban and Nonurban Estuaries of Puget Sound. NOAA Technical Memorandum NMFS-NWFSC-8. Seattle, Washington: National Oceanographic and Atmospheric Administration.

Varanasi, U., J.E. Stein, W.L. Reichert, K.L. Tilbuery, M.M. Krahn, and S.-L. Chan. 1992. Chlorinated and Aromatic Hydrocarbons in Bottom Sediments, Fish and Marine Mammals in U.S. Coastal Waters. Laboratory and Field Studies of Metabolism and Accumulation. In Persistent Pollutants in the Marine Environment, edited by C. Walter and D.R. Livingstone. New York, New York: Permagon Press.

Vaudo, J.J., and C.G. Lowe. 2006. Movement Patterns of the Round Stingray Urobatis Halleri (Cooper) near a Thermal Outfall. Journal of Fish Biology 68(6): 1756-1766.

Vaughan, D.M. 2002. Potential Impacts of Road-Stream Crossings (Culverts) on the Upstream Passage of Aquatic Macroinvertebrates. Portland, Oregon: The Xerces Society.

Vaughn, C.C., and C.M. Taylor. 1999. Impoundments and the Decline of Freshwater Mussels: A Case Study of an Extinction Gradient. Conservation Biology 13(4): 912-920.

Vellidis, G., R. Lowrance, P. Gay, and R.K. Hubbard. 2003. Nutrient Transport in a Restored Riparian Wetland. Journal of Environmental Quality 32(2): 711-726.

Vile, J.S., T.A. Friesen, and M.J. Reesman. 2004. Diets of juvenile salmonids and introduced fishes of the Lower Willamette River. In: T.S. Friesen (ed.), Biology, Behavior, and Resources of Resident and Anadromous Fish in the Lower Willamette River. Final Report of Research, 2000-2004. Oregon Department of Fish and Wildlife.

Vines, C.A., T. Robbins, F.J. Griffin, and G.N. Cherr. 2000. The Effects of Diffusible Creosote-Derived Compounds on Development in Pacific Herring (Clupea pallasi). Aquatic Toxicology 51: 225-239. Cited in Stratus 2005a.

Vinyard, G.L., and J. O’Brien. 1976. Effects of light and turbidity on the reactive distance of bluegill (Lepomis macrochirus). Journal of the Fisheries Research Board of Canada 33: 2845–2849.

Visconty, S.D. 1997. Modeling the Shade Cast by Overwater Structures: A Technical Approach to Eelgrass Preservation. Master’s Thesis, University of Washington, Seattle, Washington.

Vogel, J.L., and D.A. Beauchamp. 1999. Effects of Light, Prey Size, and Turbidity on Reaction Distances of Lake Trout (Salvelinus Namaycush) to Salmonid Prey. Canadian Journal of Fisheries and Aquatic Sciences 56(7): 1293-1297.

Vucelick, J.A., and G.A. McMichael. 2003. Walla Walla River Basin Fish Screen Evaluations, 2003 Nursery Bridge Fishway and Garden City/Lowden Il. Pacific Northwest National Laboratory.

Vucelick, J.A., G.A. McMichael, and M.A. Chamness. 2004. Yakima River Basin Phase II Fish Screen Evaluations, 2003. Prepared for Bonneville Power Administration by Pacific Northwest National Laboratory, PNNL-14627.

WAC 173-201A. November 20, 2006. Water Quality Standards for Surface Waters of the State of Washington. Washington Administrative Code.

Wagner, C.M., and H.M. Austin. 1999. Correspondence between Environmental Gradients and Summer Littoral Fish Assemblages in Low Salinity Reaches of the Chesapeake Bay, USA. Marine Ecology-Progress Series 177: 197-212.

Wagner, E.J., R.E. Arndt, and M. Brough. 2001. Comparative Tolerance of Four Stocks of Cutthroat Trout to Extremes in Temperature, Salinity, and Hypoxia. Western North American Naturalist 61(4): 434-444.

Wagner, E.J., T. Bosakowski, and S. Intelmann. 1997. Combined Effects of Temperature and High pH on Mortality and the Stress Response of Rainbow Trout after Stocking. Transactions of the American Fisheries Society 126(6): 985-998.

Wahl, T.L. 1995. Hydraulic Testing of Static Self-Cleaning Inclined Screens. The First International Conference on Water Resources Engineering, San Antonio, Texas, pp. 14-18.

Wahl, T.L. 2003. Design Guidance for Coanda-Effect Screens. Report. Water Resources Research Laboratory. U. S. Bureau of Reclamation. R-Series(3): 37.

Wahl, T.L., and R.F. Einhellig. 2000. Laboratory Testing and Numerical Modeling of Coanda-Effect Screens. Building Partnerships- 2000 Joint Conference on Water Resource Engineering and Water Resources Planning & Management.

Waite, M.E., M.J. Waldock, J.E. Thain, D.J. Smith, and S.M. Milton. 1991. Reductions in TBT Concentrations in UK Estuaries Following Legislation in 1986-1987. Marine Environmental Resource 32(1-3): 89-111.

Wald, G., P.K. Brown, and P.S. Brown. 1957. Visual Pigments and Depths of Habitat of Marine Fishes. Nature 180: 969-971.

Wallace, J.B., J.R. Webster, S.L. Eggert, and J.L. Meyer. 2000. Small Wood Dynamics in a Headwater Stream. Verhandlungen der Internationale Vereinigung für Limnologie 27: 1361-1365.

Wallace, J.B., J.R. Webster, S.L. Eggert, J.L. Meyer, and E.R. Siler. 2001. Large Woody Debris in a Headwater Stream: Long-Term Legacies of Forest Disturbance. International Review of Hydrobiology 86(4-5): 501-513.

Wang, X.L., K. Klinka, H.Y.H. Chen, and L. de Montigny. 2002. Root Structure of Western Hemlock and Western Redcedar in Single- and Mixed-Species Stands. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 32(6): 997-1004.

Wantzen, K.M. 2006. Physical Pollution: Effects of Gully Erosion on Benthic Invertebrates in a Tropical Clear-Water Stream. Aquatic Conservation-Marine and Freshwater Ecosystems 16(7): 733-749.

Ward, J.V. 1989. The 4-Dimensional Nature of Lotic Ecosystems. Journal of the North American Benthological Society 8(1): 2-8.

Warmke, S., and D. Hering. 2000. Composition, Microdistribution and Food of the Macroinvertebrate Fauna Inhabiting Wood in Low-Order Mountain Streams in Central Europe. International Review of Hydrobiology 85(1): 67-78.

Warner, E. July 7, 2000. Personal communication (written comments to Tom Kahler, The Watershed Company). Muckleshoot Indian Tribe, Fisheries Division (as cited by Kahler et al. 2000). Cited in Kahler et al. 2000.

Warner, K. 1972. Further studies of fish predation of salmon stocked in Maine lakes. Progressive Fish-Culturist 34:217–221. Cited in Carrasquero 2001.

Warner, M.J., Kawase, M. and Newton, J.A. 2001. Recent Studies of the Overturning Circulation in Hood Canal. Puget Sound Research Conference, Vancouver, BC.

Warren, D.R., and C.E. Kraft. 2003. Brook Trout (Salvelinus fontinalis) Response to Wood Removal from High-Gradient Streams of the Adirondack Mountains (NY, USA). Canadian Journal of Fisheries and Aquatic Sciences 60(4): 379-389.

Warren, M.L., and M.G. Pardew. 1998. Road Crossings as Barriers to Small-Stream Fish Movement. Transactions of the American Fisheries Society 127(4): 637-644.

Warrington, P. 1999. Impacts of Recreational Boating on the Aquatic Environment. Available at: American Lake Management Society, http://www.nalms.org/bclss/impactsrecreationboat.htm.

Washington Forest Practices Board. 1995. Board Manual: Standard Methodology For Conducting Watershed Analysis. Appendix E. Stream Channel. Olympia: Washington Forest Practices Board. Available at http://www.dnr.wa.gov/forestpractices/watershedanalysis/manual/index.html (Accessed 2006.10.04).

Wasson, K., K. Fenn, and J.S. Pearse. 2005. Habitat differences in marine invasions of central California. Biological Invasions 7(6): 935-948.

Waters, G.T. 1999. Freshwater mussels and water quality: A review of the effects of hydrologic and instream habitat alterations. Ohio Biological Survey and Aquatic Ecology Laboratory. Ohio State University. Proceedings of the first Freshwater Mollusk Conservation Symposium.

Waters, T.F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society, Bethesda, Maryland. 251 p.

Watson, C.C., D.S. Biedenharn, and S.H. Scott. 1999. Channel Rehabilitation: Processes, Design, and Implementation. Vicksburg, MS: U.S. Army Corps of Engineers Research and Development Center.

Watson, L.R., A. Milani, and R.P. Hedrick. 1998. Effects of Water Temperature on Experimentally-Induced Infections of Juvenile White Sturgeon (Acipenser transmontanus) with the White Sturgeon Iridovirus (WSIV). Aquaculture 166(3-4): 213-228.

Watters, G.T. 1996. Small Dams as Barriers to Freshwater Mussels (Bivalvia, Unionoida) and Their Hosts. Biological Conservation 75(1): 79-85.

Watters, G.T. 1999. Freshwater Mussels and Water Quality: A Review of the Effects of Hydrologic and Instream Habitat Alterations. Proceedings of the First Freshwater Mollusk Conservation Symposium, Columbus, Ohio, pp. 261-274.

WDFW (Washington Department of Fish and Wildlife). 1997a. Washington State Forage fish fact sheet: Puget Sound herring fact sheet. WDFW, Forage Fish Management Unit. Olympia, WA. Available at http://www.wdfw.wa.gov/fish/forage/herring.htm (Accessed 2006.11.03).

WDFW (Washington Department of Fish and Wildlife). 1997b. Washington State Forage fish fact sheet: Washington State sand lance fact sheet. WDFW, Forage Fish Management Unit. Olympia, WA. Available at: http://www.wdfw.wa.gov/fish/forage/lance.htm (Accessed 2006.11.03). Cited in NRC 2001, and in Nightingale and Simenstad 2001.

WDFW (Washington Department of Fish and Wildlife). 1997c. Washington State Forage fish fact sheet: Washington State surf smelt fact sheet. WDFW, Forage Fish Management Unit. Olympia, WA. Available at http://www.wdfw.wa.gov/fish/forage/smelt.htm (Accessed 2006.11.03).

WDFW (Washington Department of Fish and Wildlife). 1998. Habitat engineering fish streams information: screening requirements for water diversions. http://wdfw.wa.gov/hab/engineer/fishscrn.htm, accessed 2006.11.30.

WDFW (Washington Department of Fish and Wildlife). 2000. Fishway Guidelines for Washington State. Olympia, Washington: Washington Department of Fish and Wildlife.

WDFW (Washington Department of Fish and Wildlife). 2001a. Fish Protection Screen Guidelines for Washington State - Draft. Draft Report. Olympia: Washington Department of Fish and Wildlife.

WDFW (Washington Department of Fish and Wildlife). 2001b Washington State Aquatic Nuisance Species Management Plan. Coordinated by Pamala Meacham of the Washington State Department of Fish and Wildlife for the Washington Aquatic Nuisance Species Committee. October 2001.

WDFW (Washington Department of Fish and Wildlife). 2003. Integrated Streambank Protection Guidelines. Olympia, Washington: Washington Department of Fish and Wildlife.

WDFW (Washington Department of Fish and Wildlife). 2004. Zebra mussels discovered at Washington-Idaho border. Washington Department of Fish and Wildlife. Available at: http://wdfw.wa.gov/do/newreal/release.php?id=may2504a (accessed June 22, 2007).

WDFW (Washington Department of Fish and Wildlife). 2007. Data provided by Washington Department of Fish and Wildlife, Olympia, Washington. MS Excel spreadsheet, WhitePaperdatatypes11_01_06_1.xls Provided to Herrera Environmental Consultants. May 9, 2007.

WDFW and ODFW (Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife). 2001. Washington and Oregon eulachon management plan. Olympia, WA: Washington Department of Fish and Wildlife. 32 p. Cited in Willson et al. 2006.

WDNR (Washington Department of Natural Resources). 2001. Shoreline Inventory: Nearshore Habitat Program. Olympia, Washington: Washington State Department of Natural Resources.

WDNR (Washington Department of Natural Resources). 2004. Maury Island Aquatic Reserve. Final Management Plan. October 29, 2004. Aquatic Resources Division. Olympia, WA. Available at http://www.dnr.wa.gov/htdocs/aqr/reserves/pdf/mgmtplan10_29_04.pdf (Accessed 2006.10.04).

WDNR (Washington Department of Natural Resources). 2005a. Draft Aquatic Resources Program Endangered Species Act Compliance Endangered Draft Covered Species Technical Paper. Olympia, Washington: Washington State Department of Natural Resources.

WDNR (Washington Department of Natural Resources). 2005b. Aquatic Resources Program Endangered Species Act Compliance Potential Covered Activities Technical Paper. Olympia, Washington: Washington State Department of Natural Resources.

WDNR (Washington Department of Natural Resources). 2005c. Forest Practices - Habitat Conservation Plan. Olympia, Washington: Washington State Department of Natural Resources.

WDNR (Washington Department of Natural Resources). 2005d. Standard Practice for the Use and Removal of Treated Wood Pilings on and from State-Owned Aquatic Lands. Standard practice memorandum SPM 02-07. Olympia, Washington: Washington Department of Natural Resources.

WDNR (Washington Department of Natural Resources). 2006a. Draft Covered Species White paper - Fish. Olympia, Washington: Washington Department of Natural Resources.

WDNR (Washington Department of Natural Resources). 2006b. Draft Covered Species White Paper - Invertebrates. Olympia, Washington: Washington Department of Natural Resources.

WDNR (Washington Department of Natural Resources). 2007. Aquatic Resources Program Endangered Species Act Compliance Project – Potential Effects and Expected Outcomes Technical Paper. Olympia, Washington: Washington Department of Natural Resources.

Webb, A.A., and W.D. Erskine. 2003. Distribution, Recruitment, and Geomorphic Significance of Large Woody Debris in an Alluvial Forest Stream: Tonghi Creek, Southeastern Australia. Geomorphology 51(1-3): 109-126.

Webber, J.D., S.N. Chun, T.R. MacColl, L.T. Mirise, A. Kawabata, E.K. Anderson, T.S. Cheong, L. Kavvas, M.M. Rotondo, K.L. Hochgraf, R. Churchwell, and J.J. Cech. 2007. Upstream Swimming Performance of Adult White Sturgeon: Effects of Partial Baffles and a Ramp. Transactions of the American Fisheries Society 136(2): 402-408.

Weber, E.D., S.M. Borthwick, and L.A. Helfrich. 2002. Plasma Cortisol Stress Response of Juvenile Chinook Salmon to Passage through Archimedes Lifts and a Hidrostal Pump. North American Journal of Fisheries Management 22(2): 563-570.

Webster, M.T., and R.C. Loehr. 1996. Long-Term Leaching of Metals from Concrete Products. Journal of Environmental Engineering-American Society of Civil Engineers 122(8): 714-721.

Wedemeyer, G.A., R.L. Saunders, and W.C. Clarke. 1980. Environmental Factors Affecting Smoltification and Early Marine Survival of Anadromous Salmonids. Marine Fisheries Review 42(6): 1-14.

Weigand, D.C. 1991. Effects of Gravel Scalping on Juvenile Salmonid Habitat. Seattle, Washington: University of Washington.

Weis, J.S. and P. Weis. 1994. Effects of Contaminants from Chromated Copper Arsenate-Treated Lumber on Benthos. Archives of Environmental Contamination and Toxicology 26: 103-109. Cited in Stratus 2005b.

Weis, J.S. and P. Weis. 1996. Effects of Using Wood Treated with Chromated Copper Arsenate in Shallow-Water Environments: A Review. Estuaries 19(2A): 306-310. Cited in Stratus 2005b

Weis, J.S., P. Weis, and T. Proctor. 1998. The extent of benthic impacts of CCA-treated wood structures in Atlantic Coast estuaries. Archives of Environmental Contamination and Toxicology 34:313-322. Cited in Stratus 2005b.

Weis, P, J.S. Weis, A. Greenberg, and T.J. Nosker. 1992. Toxicity of construction materials in the marine environment: A comparison of chromated-copper-arsenate-treated wood and recycled plastic. Archives of Environmental Contamination and Toxicology 22:99-106. Cited in Stratus 2005b.

Weis, P., J.S. Weis, and J. Couch. 1993. Histopathology and Bioaccumulation in Crassostrea Virginica Living on Wood Preserved with Chromated Copper Arsenate. Diseases Aquatic Organization 17: 41-46. Cited in Stratus 2005b.

Weisberg, S.B., W.H. Burton, F. Jacobs, and E.A. Ross. 1987. Reductions in Ichthyoplankton Entrainment with Fine-Mesh, Wedge-Wire Screens. North American Journal of Fisheries Management 7(3): 386-393.

Weitkamp, D.E. 1982. Juvenile chum and chinook salmon behavior at Terminal 91, Report to Port of Seattle. Parametrix Inc. Cited in Nightingale and Simenstad 2001a.

Weitkamp, D.E. 1981. Shilshole Bay Fisheries Resources, No. 81-0712-018 F. Parametrix Inc., Seattle, WA. Cited in Nightingale and Simenstad 2001a.

Weitkamp, D.E., and R.F. Campbell. 1980. Port of Seattle Terminal 107 Fisheries Study. Bellevue, Washington.

Weitkamp, D.E., and T.H. Schadt. 1982. 1980 Juvenile Salmonid Study. Document No. 82-0415-012f. Prepared for the Port of Seattle by Parametrix, Inc., Seattle, Washington.

Welch, E.B., and T. Lindell. 1992. Ecological Effects of Wastewater: Applied Limnology and Pollution Effects. London, New York: E & FN Spon.

Welch, E.B., J.M. Jacoby, and C.W. May. 1998. Stream Quality. In River Ecology and Management, edited by R.J. Naiman and R.E. Bilby. New York: Springer. pp. 69-94.

Welker, T.L., S.T. McNulty, and P.H. Klesius. 2007. Effect of Sublethal Hypoxia on the Immune Response and Susceptibility of Channel Catfish, Ictalurus Punctatus, to Enteric Septicemia. Journal of the World Aquaculture Society 38(1): 12-23.

Welton, J.S., W.R.C. Beaumont, and R.T. Clarke. 2002. The Efficacy of Air, Sound and Acoustic Bubble Screens in Deflecting Atlantic Salmon, Salmo Salar L., Smolts in the River Frome, UK. Fisheries Management and Ecology 9(1): 11-18.

Wenning, R.J., D.B. Mathur, D.J. Paustenbach, M.J. Stephenson, S. Folwarkow, and W.J. Luksemburg. 1999. Polychlorinated Dibenzo-P-Dioxins and Dibenzofurans in Storm Water Outfalls Adjacent to Urban Areas and Petroleum Refineries in San Francisco Bay, California. Archives of Environmental Contamination and Toxicology 37(3): 290-302.

West, J. 1995. Accumulation of Mercury and Polychlorinated Biphenyls in Quillback Rockfish (Sebasters Maliger) from Puget Sound, Washington. Puget Sound Research '95 Proceedings, Bellevue, Washington.

West, J.E. 1997. Protection and Restoration of Marine Life in the Inland Waters of Washington State. Puget Sound/Georgia Basin Environmental Report Series: Number 6. Puget Sound/Georgia Basin International Task Force. Puget Sound Action Team. Olympia, WA. Cited in WDNR 2006b.

West, J.E., R.M. Buckley, and D.C. Doty. 1994. Ecology and Habitat Use of Juvenile Rockfishes (Sebastes Spp.) Associated with Artificial Reefs in Puget-Sound, Washington. Bulletin of Marine Science 55(2-3): 344-350.

Westbrook, C.J., D.J. Cooper, and B.W. Baker. 2006. Beaver Dams and Overbank Floods Influence Groundwater-Surface Water Interactions of a Rocky Mountain Riparian Area. Water Resources Research 42(6): 1-12.

Whalen, P.J., L.A. Toth, J.W. Koebel, and P.K. Strayer. 2002. Kissimmee River Restoration: A Case Study. Water Science and Technology 45(11): 55-62.

White, D.K., C. Swanson, P.S. Young, J.J. Cech Jr., Z.Q. Chen, and M.L. Kavvas. 2007. Close Encounters with a Fish Screen II: Delta Smelt Behavior before and During Screen Contact. Transactions of the American Fisheries Society 136(2): 528-538.

White, D.S. 1990. Biological Relationships to Convective Flow Patterns within Stream Beds. Hydrobiologia 196: 149-158.

White, D.S. 1993. Perspectives on Defining and Delineating Hyporheic Zones. Journal of the North American Benthological Society 12: 61-69.

White, S.T. 1975. The Influence of Piers and Bulkheads on the Aquatic Organisms in Lake Washington. Master's Thesis, University of Washington.

Whitledge, G.W., C.F. Rabeni, G. Annis, and S.P. Sowa. 2006. Riparian Shading and Groundwater Enhance Growth Potential for Smallmouth Bass in Ozark Streams. Ecological Applications 16(4): 1461-1473.

Whitman, R.P., T.P. Quinn, and E.L. Brannon. 1982. Influence of Suspended Volcanic Ash on Homing Behavior of Adult Chinook Salmon. Transactions of the American Fisheries Society.

Whyte, J.J., R.E. Jung, C.J. Schmitt, and D.E. Tillitt. 2000. Ethoxyresorufin-O-Deethylase (EROD) Activity in Fish as a Biomarker of Chemical Exposure. Critical Reviews in Toxicology 30: 347-570.

Widdows, J., P. Fieth, and C.M. Worrall. 1979. Relationships between Seston, Available Food and Feeding-Activity in the Common Mussel Mytilus-Edulis. Marine Biology 50(3): 195-207.

Wijnberg, K.M. 2002. Environmental Controls on Decadal Morphologic Behaviour of the Holland Coast. Marine Geology 189(3-4): 227-247.

Wik, S.J. 1995. Reservoir Drawdown - Case-Study in Flow Changes to Potentially Improve Fisheries. Journal of Energy Engineering-American Society of Civil Engineers 121(2): 89-96.

Wilber, D.H., and D.G. Clarke. 2001. Biological Effects of Suspended Sediments: A Review of Suspended Sediment Impacts on Fish and Shellfish with Relation to Dredging Activities in Estuaries. North American Journal of Fisheries Management 21(4): 855-875.

Wilber, D.H., D.G. Clarke, and M.H. Burlas. 2006. Suspended Sediment Concentrations Associated with a Beach Nourishment Project on the Northern Coast of New Jersey. Journal of Coastal Research 22(5): 1035-1042.

Wilber, D.H., D.G. Clarke, and S.I. Rees. 2007. Responses of Benthic Macroinvertebrates to Thin-Layer Disposal of Dredged Material in Mississippi Sound, USA. Marine Pollution Bulletin 54(1): 42-52.

Wilber, D.H., D.G. Clarke, G.L. Ray, and M. Burlas. 2003. Response of Surf Zone Fish to Beach Nourishment Operations on the Northern Coast of New Jersey, USA. Marine Ecology-Progress Series 250: 231-246.

Wilcox, D.A., J.E. Meeker, P.L. Hudson, B.J. Armitage, M.G. Black, and D.G. Uzarski. 2002. Hydrologic Variability and the Application of Index of Biotic Integrity Metrics to Wetlands: A Great Lakes Evaluation. Wetlands 22(3): 588-615.

Wildish, D.J., and J. Power. 1985. Avoidance of Suspended Sediments by Smelt as Determined by a New Single Fish Behavioral Bioassay. Bulletin of Environmental Contamination and Toxicology 34(5): 770-774.

Williams, D.E., J.J. Lech, and D.R. Buhler. 1998. Xenobiotics and Xenoestrogens in Fish: Modulation of Cytochrome P450 and Carcinogenesis. Mutation Research 399: 179-192.

Williams, G.D. 1994. Effects of Habitat Modification on Distribution and Diets of Intertidal Fishes in Grays Harbor Estuary, Washington. University of Washington, Seattle, Washington.

Williams, G.D. and R.M. Thom. 2001. Development of Guidelines for Aquatic Habitat Protection and Restoration: Marine and Estuarine Shoreline Modification Issues. Prepared for the Washington State Department of Transportation, Washington Department of Fish and Wildlife, and the Washington Department of Ecology, by Battelle Marine Sciences Laboratory, Sequim, Washington. PNWD-3087.

Williams, G.D., and R.M. Thom. 2001. Marine and Estuarine Shoreline Modification Issues. A White Paper. Submitted to Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Williams, G.D., R.M. Thom, and J.E. Starkes. 2001. Reconnaissance Assessment of the State of the Nearshore Ecosystem: Eastern Shore of Central Puget Sound, Including Vashon and Maury Islands (WRIAs 8 and 9). Prepared for King County Department of Natural Resources, Seattle, Washington.

Williams, G.D., R.M. Thom, J.E. Starkes, J.S. Brennan, J.P. Houghton, D. Woodruff, P.L. Striplin, M. Miller, M. Pedersen, A. Skillman, R. Cropp, A. Borde, C. Freeland, K. McArthur, V. Fagerness, S. Blanton, and L. Blackmore. 2001. Reconnaissance Assessment of the State of the Nearshore Ecosystem: Eastern Shore of Central Puget Sound, Including Vashon and Maury Islands (WRIAs 8 and 9). Seattle, Washington: King County Department of Natural Resources.

Williams, J.D., M.L. Warren, K.S. Cummings, J.L. Harris, and R.J. Neves. 1993. Conservation Status of Fresh-Water Mussels of the United-States and Canada. Fisheries 18(9): 6-22.

Williams, P.B., and M.K. Orr. 2002. Physical Evolution of Restored Breached Levee Salt Marshes in the San Francisco Bay Estuary. Restoration Ecology 10(3): 527-542.

Williams, S.L., and M.H. Ruckelshaus. 1993. Effects of Nitrogen Availability and Herbivory on Eelgrass (Zostera Marina) and Epiphytes. Ecology 74: 904-918.

Williamson, R.B. 1985. Urban Stormwater Quality. New Zealand Journal of Marine and Freshwater Research 19: 413-427.

Willson, M.F., R. H. Armstrong, M. C. Hermans, and K Koski. 2006. Eulachon: a review of biology and an annotated bibliography. Alaska Fisheries Science Center and NOAA Fisheries. 243 pp.

Wilson, U. 1993. Eelgrass Zostera Marina, in the Dungeness Bay Area, Washington, During 1993. Sequim, Washington: Progress Report, U.S. Fish and Wildlife Service, Coastal Refuges Office.

Wilson, U.W., and J.B. Atkinson. 1995. Black Brant Winter and Spring Staging Use at Two Washington Coastal Areas in Relation to Eelgrass Abundance. The Condor 97(1).

Wilzbach, M.A., B.C. Harvey, J.L. White, and R.J. Nakamoto. 2005. Effects of Riparian Canopy Opening and Salmon Carcass Addition on the Abundance and Growth of Resident Salmonids. Canadian Journal of Fisheries and Aquatic Sciences 62(1): 58-67.

Winfield, I.J. 2004. Fish in the Littoral Zone: Ecology, Threats and Management. Limnologica 34(1-2): 124-131.

Winger, P.V., and P.J. Lasier. 1994. Effects of Salinity on Striped Bass Eggs and Larvae from the Savanna River, Georgia. Transactions of the American Fisheries Society 123(6): 904-912.

Winkelmann, C., S. Worischka, J.H.E. Koop, and J. Benndorf. 2007. Predation Effects of Benthivorous Fish on Grazing and Shredding Macroinvertebrates in a Detritus-Based Stream Food Web. Limnologica – Ecology and Management of Inland Waters 37: 121-128.

Winn, R.N., and D.M. Knott. 1992. An Evaluation of the Survival of Experimental Populations Exposed to Hypoxia in the Savanna River Estuary. Marine Ecology-Progress Series 88(2-3): 161-179.

Winter, B.D. 1990. A Brief Review of Dam Removal Efforts in Washington, Oregon, Idaho, and California. Technical Memo NMFS F/NWR-28. U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Winter, H.V., and W.L.T. Van Densen. 2001. Assessing the Opportunities for Upstream Migration of Non-Salmonid Fishes in the Weir-Regulated River Vecht. Fisheries Management and Ecology 8(6): 513-532.

Wipfli, M.S. 1997. Terrestrial Invertebrates as Salmonid Prey and Nitrogen Sources in Streams: Contrasting Old-Growth and Young-Growth Riparian Forests in Southeastern Alaska, USA. Canadian Journal of Fisheries and Aquatic Sciences 54(6): 1259-1269.

Wipfli, M.S. 2005. Trophic Linkages between Headwater Forests and Downstream Fish Habitats: Implications for Forest and Fish Management. Landscape and Urban Planning 72(1-3): 205-213.

Wisby, W.J., J.D. Richard, D.R. Nelson, and S.H. Gruber. 1964. Sound Perception in Elasmobranches. In Marine Bio-Acoustics, edited by W.N. Tavolga. New York: Pergamon Press. pp. 255-268.

WNWCB (Washington Noxious Weed Control Board). 2005. Myriophyllum spicatum. Web page. Last updated 09/29/05. Available at: http://www.nwcb.wa.gov/weed_info/Written_findings/Myriophyllum_spicatum.html (Accessed 2006.10.10).

WNWCB (Washington Noxious Weed Control Board). 2006. Noxious Weed Control Board Home Page. Available at http://www.nwcb.wa.gov/index.htm (Accessed 2006.10.02).

Wohl, E.E. 2001. Virtual Rivers: Lessons from the Mountain Rivers of the Colorado Front Range. New Haven, Connecticut: Yale University Press.

Wohl, E.E. 2004. Disconnected Rivers: Linking Rivers to Landscapes. New Haven, Connecticut: Yale University Press.

Wohl, E.E., and D.A. Cenderelli. 2000. Sediment Deposition and Transport Patterns Following a Reservoir Sediment Release. Water Resources Research 36(1): 319-333.

Wones, A. and C. Cziesla. 2004. A comparison of three cable installation methods. Jones and Stokes. Bellevue, WA. Poster presented at: Restore Americas Estuaries Conference. Seattle, WA. E.

Wood, P.J., and P.D. Armitage. 1997. Biological Effects of Fine Sediment in the Lotic Environment. Environmental Management 21: 203-217.

Woodsmith, R. D. and J.M. Buffington. 1996. Multivariate geomorphic analysis of forest streams: implications for assessment of land use impacts on channel conditions. Earth Surface Processes and Landforms 21:377–393.

Work, P.A., F. Fehrenbacher, and G. Voulgaris. 2004. Nearshore Impacts of Dredging for Beach Nourishment. Journal of Waterway Port Coastal and Ocean Engineering-ASCE 130(6): 303-311.

Worman, A. 1998. Analytical Solution and Timescale for Transport of Reacting Solutes in Rivers and Streams. Water Resources Research 34(10): 2703-2716.

WSDOT (Washington State Department of Transportation) and Ecology (Washington State Department of Ecology). 1998. Implementing Agreement between the Washington State Department of Ecology and the Washington State Department of Transportation Regarding Compliance with the State of Washington Surface Water Quality Standards.

WSDOT (Washington State Department of Transportation). 2005. Underwater Sound Levels Associated with the Friday Harbor Ferry Terminal. Agency Report. Washington State Department of Transportation, Office of Air, Noise, and Energy.

WSDOT (Washington State Department of Transportation). 2006a. Biological Assessment Preparation for Transportation Projects. Advanced Training Manual. Olympia, Washington: Washington State Department of Transportation, Environmental Affairs Office.

WSDOT (Washington State Department of Transportation). 2006b. Fish exclusion protocols and standards. Olympia, WA. 8 pgs.

WSDOT (Washington State Department of Transportation). 2006b. Washington Transportation Plan 2007-2026. Olympia, Washington: The Washington State Transportation Commission and the Washington State Department of Transportation.

WSDOT (Washington State Department of Transportation). 2006c. Highway Runoff Manual. Publication M 31-16. Olympia, Washington: Washington State Department of Transportation.

WSDOT (Washington State Department of Transportation). 2006d. BA Writers Guidance for Preparing the Stormwater Section of Biological Assessments. Unpublished Addendum To: Biological Assessment Preparation for Transportation Projects. Advanced Training Manual. Olympia, Washington: Washington State Department of Transportation.

WSDOT (Washington State Department of Transportation). 2007. Washington State Department of Transportation Project Index List. Available at: http://www.wsdot.gov/projects (accessed June 19, 2007).

WSDOT (Washington State Department of Transportation). 2006b. Fish Exclusion Protocols and Standards. Olympia, WA. 8 pgs.

Wu, J., W. Mersie, A. Atalay, and C.A. Seybold. 2003. Copper Retention from Runoff by Switchgrass and Tall Fescue Filter Strips. Journal of Soil and Water Conservation 58(1): 67-73.

Wurjanto, A., and N. Kobayashi. 1993. Irregular Wave Reflection and Runup on Permeable Slopes. Journal of Waterway Port Coastal and Ocean Engineering-ASCE 119(5): 537-557.

Wydoski and Whitney. 1979. Inland Fishes of Washington. Seattle: University of Washington Press.

Wydoski, R.G. and R.S. Wydoski. 2002. Age, Growth, and Reproduction of Mountain Suckers in Lost Creek Reservoir, Utah. Transactions of the American Fisheries Society 131(2): 320-328.

Wydoski, R.S. and R.R. Whitney. 2003. Inland Fishes of Washington, Second Edition. American Fisheries Society and University of Washington Press. 322 pp.

Wyllie-Echeverria, S., and R.C. Phillips. 1994. Seagrasses of the Northwest Pacific. In: Seagrass Science and Policy in the Pacific Northwest: Proceedings of a Seminar Series. Edited by S. Wyllie-Echeverria, A.M. Olson, and M.J. Hershman. Prepared for: U.S. Environmental Protection Agency. (SMA 94-1). EPA 910/R-94-004.

Wyzga, B. 1996. Changes in the Magnitude and Transformation of Flood Waves Subsequent to the Channelization of the Raba River, Polish Carpathians. Earth Surface Processes and Landforms 21(8): 749-763.

Xiao, Y., J. Simonsen, and J.J. Morrell. 2002. Effect of Water Flow Rate and Temperature on Leaching from Creosote-Treated Wood. Research Note FPL-RN-0286. U.S. Department of Agriculture, Forest Service, Forest Products Laboratory, Madison, WI. Cited in Stratus 2005a.

Yager, E., M. Schmeeckle, W.E. Dietrich, and J.W. Kirchner. 2004. The Effect of Large Roughness Elements on Local Flow and Bedload Transport. Eos Transactions American Geophysical Union 85(47), Fall Meet Supplement, Abstract H41G-05.

Yanai, S., and K. Kochi. 2005. Effects of Salmon Carcasses on Experimental Stream Ecosystems in Hokkaido, Japan. Ecological Research 20(4): 471-480.

Yang, M.-S., K. Dodd, R. Hibshman, and A. Whitehouse. 2006. Food Habits of Groundfishes in the Gulf of Alaska in 1999 and 2001. Seattle, Washington: Alaska Fisheries Science Center, National Marine Fisheries Service.

Yates, S. 2001. Effects of Swinomish Channel jetty and causeway on outmigrating chinook salmon (Oncorhynchus tshawytscha) from the Skagit River Washington. Bellingham, Washington: Western Washington University. 65 pp.

Yelverton, J.T., D.R. Richmond, W.Hicks, K Saunders, and E.R. Fletcher. 1975. The Relationship between Fish Size and their Response to Underwater Blast. Report DNA 3677T, Director, Defense Nuclear Agency. Washington D.C.

Yonge, D., A. Hossain, M. Barber, S. Chen, and D. Griffin. 2002. Wet Detention Pond Design for Highway Runoff Pollutant Control. National Cooperative Highway Research Program.

Young, G.K., S. Stein, P. Cole, and T. Kammer. 1996. Evaluation and Management of Highway Runoff Water Quality. Washington, DC: Federal Highway Administration, U.S. Department of Transportation.

Young, W.J. 1991. Flume Study of the Hydraulic Effects of Large Woody Debris in Lowland Rivers. Regulated Rivers: Research & Management 6: 203-211.

Young, W.T., and D.L. Scarnecchia. 2005. Habitat Use of Juvenile White Sturgeon in the Kootenai River, Idaho and British Columbia. Hydrobiologia 537(1): 265-271.

Yousef, Y.A. 1974. Assessing Effects on Water Quality by Boating Activity. U.S. Environmental Protection Agency, Technical Service.

Yousef, Y.A., W.M. Mclellon, and H.H. Zebuth. 1980. Changes in Phosphorus Concentrations Due to Mixing by Motor-Boats in Shallow Lakes. Water Research 14(7): 841-852.

Yozzo, D.J., P. Wilber, and R.J. Will. 2004. Beneficial Use of Dredged Material for Habitat Creation, Enhancement, and Restoration in New York-New Jersey Harbor. Journal of Environmental Management 73(1): 39-52.

Zalewski, M., B. Bis, M. Lapinska, P. Frankiewicz, and W. Puchalski. 1998. The Importance of the Riparian Ecotone and River Hydraulics for Sustainable Basin-Scale Restoration Scenarios. Aquatic Conservation-Marine and Freshwater Ecosystems 8(2): 287-307.

Zelo, I., H. Shipman, and J. Brennan. 2000. Alternative Bank Protection Methods for Puget Sound Shorelines. Shorelands and Environmental Assistance Program, Washington Department of Ecology, Olympia, Washington. 130pp.

Zemlak, R.J., and J.D. McPhail. 2006. The Biology of Pygmy Whitefish, Prosopium Coulterii, in a Closed Sub-Boreal Lake: Spatial Distribution and Diel Movements. Environmental Biology of Fishes 76(2-4): 317-327.

Zhang, W., X.L. Chen, J.H. Xu, G.C. Li, and Z.Q. Wang. 2000. Experimental Study on Scours Downstream of Floodgates. China Ocean Engineering 14(2): 243-254.

Zhang, Y.X., J.N. Negishi, J.S. Richardson, and R. Kolodziejczyk. 2003. Impacts of Marine-Derived Nutrients on Stream Ecosystem Functioning. Proceedings of the Royal Society of London Series B-Biological Sciences 270(1529): 2117-2123.

Ziller, J. 2005. Middle Fork Willamette River Bull Trout Spawning Gravel Augmentation Project. January 14, 2005. Available at: http://www.dfw.state.or.us/ODFWhtml/springfield/R&EBoard.pdf (accessed June 21, 2007).

Zimmermann, A.E., and M. Lapointe. 2005. Intergranular Flow Velocity through Salmonid Redds: Sensitivity to Fines Infiltration from Low Intensity Sediment Transport Events. River Research and Applications 21(8): 865-881.

Zisette, R., Aquatic Science Director with Herrera Environmental Consultants, Inc., Seattle, Washington. 2007. Discussion regarding Eagle Harbor project construction and monitoring with Dylan Ahearn of Herrera Environmental Consultants, Inc. Seattle, Washington, August 14, 2007.

Zweig, L.D. and C.F. Rabeni. 2001. Biomonitoring for deposited sediment using benthic invertebrates: a test on 4 Missouri streams. Journal of the North American Benthological Society 20(4):643-65.

Zydlewski, G.B., and J.R. Johnson. 2002. Response of Bull Trout Fry to Four Types of Water Diversion Screens. North American Journal of Fisheries Management 22(4): 1276-1282.