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^{o}F (21–22^{o}C) (Poole and Berman 2001a).
- Temperatures that impair smolting are above a range of between 52 and 59^{o}F (11 and 15^{o}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^{o}F (14^{o}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^{o}F (7–14^{o}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°F (23 and 26°C) can cause complete mortality, with upper limits for survival around 62.6–64.4°F (17–18°C) (Van Eenennaam et al. 2005).

- Dolly Varden show decreased appetite above 60.8°F (16°C) and lethal temperatures are observed above 68°F (20°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°F (4.85°C) and 40.9°F (4.97°C), respectively, with survival greatest for both at 64°F (18°C) and lowest at 71.6°F (22°C). Ammocoete exposure to a treatment temperature of 71.6°F (22°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 1.02×10^{-5} lb/ft^2 [50 mg/m^2]) with mortality and inhibition of feeding at higher levels of fine sediment (4.09×10^{-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.

[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) As	Cr	Cu	Zn	Reference
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).

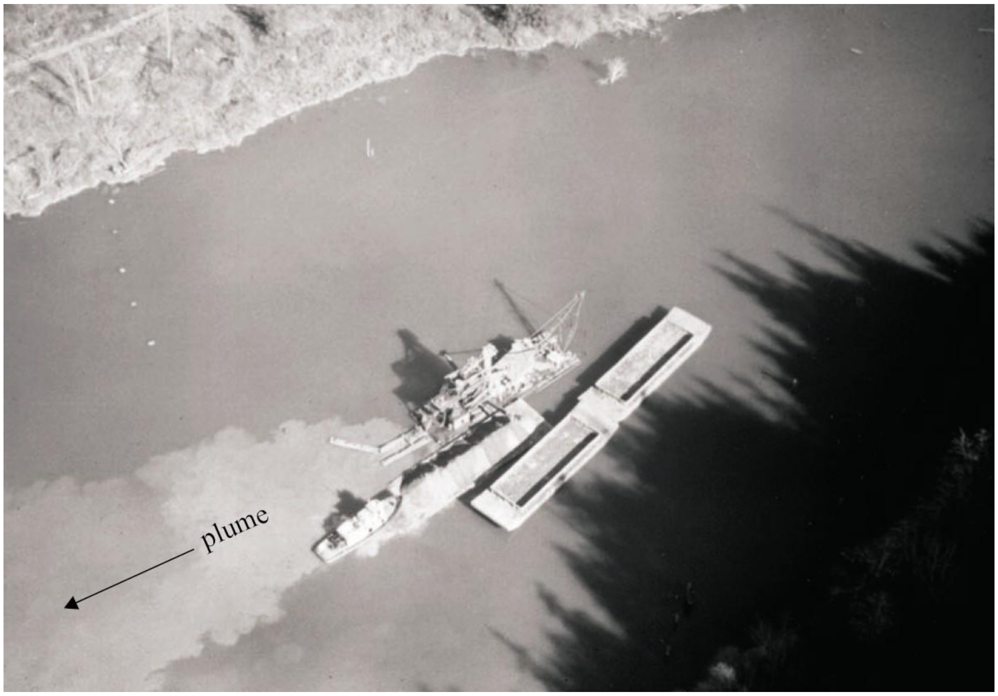

Figure 7-3. Turbidity plume associated with dredging. (Source Unattributed)

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

[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.