*Stock Assess*ment of Subtidal Geoduck Clams

(Panopea abrupta) in Washington

by

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January 2000

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Glossary

Annual mortality rate (A) - The number of animals which die during a year divided by the initial number.

Constant F **strategy -** A harvest strategy which sets the annual quota as a function of current population size and a recommended instantaneous fishing mortality rate.

CV - Coefficient of variation, a relative measure of statistical precision (the standard error of an estimator divided by the estimator, and expressed as a percentage).

DGPS - Differential Global Positioning System, a satellite navigational system which uses a shore-based slave station to provide extremely accurate position fixes on the surface of the earth.

DNR - Washington Department of Natural Resources.

Dig sample - A sample of geoducks (generally ten) dug with commercial water jet gear within a previously surveyed 900 ft' transect on a geoduck tract, used in estimating mean weight per geoduck.

Dimple - A visible depression or "show" caused by a geoduck or other clam siphon which is partially retracted in the substrate.

Equilibrium yield - The yield in weight taken from a stock when it is in equilibrium with fishing of a given intensity.

Exploitation rate (μ) - The fraction of the initial population removed fishing in one year; equivalent to the product of the annual mortality rate and the fishing mortality rate divided by the total mortality rate $(\mu = FA / Z)$.

Fishing mortality rate (F) - The ratio of number of animals harvested per unit of time to the population abundance at that time, if all harvested animals were to be immediately replaced so that the population does not change (an instantaneous rate). The portion of total instantaneous mortality due to fishing.

Geoduck Atlas - An annual WDFW publication listing all known geoduck tracts in Washington, along with maps of their location, their commercial status, estimates of geoduck biomass, and other summary information.

Grid line - The primary sampling unit in geoduck surveys, along which a series of 900 ft^2 trip transects is aligned; usually run perpendicular to shore.

Harvestable geoducks - Geoducks of a size in which the siphon or "show" is likely to be seen by a diver; generally, geoducks with a total weight > 300 grams and >5 yrs old.

Harvest rate - Same as exploitation rate, see above.

Harvest strategy - A quantitative plan which states how catch will be adjusted from year to year, usually depending on the size of the stock.

MLLW(Mean Lower Low Water) - The arithmetic mean of the lower low water heights of a mixed tide observed over a specific 19-year Metonic cycle at a specific tidal reference station. Used to correct ambient depths (from diver depth gauges) to a standard tidal datum.

Natural mortality rate (M) - The ratio of number of animals which die from non-fishing causes per unit of time to the population abundance at that time, if all dead animals were to be immediately replaced so that the population does not change (an instantaneous rate). The portion of total instantaneous mortality due to natural (i.e., non-fishing) causes.

Rafeedie decision - The popular term for United States v. Washington No. 9213, subproceeding 89-3, a federal district court decision regarding treaty tribal rights to shellfish, including geoducks.

Show - When applied to geoducks, either a geoduck siphon visible above the substrate surface, or a depression or mark left in the substrate which can be identified as having been made by a geoduck.

Show factor - The ratio of geoduck shows visible during a single observation of any defined area to the true abundance of harvestable geoducks in that area.

Show plot - Permanently-marked subtidal areas in which the absolute number of harvestable geoducks is known from repeated tagging; show plots are used to estimate geoduck show factors.

SPR (Spawning Biomass Per Recruit) - The biomass of sexually mature members of a stock, expressed in terms of weight per recruit. Mathematically, the product of numbers-at-age, weightat-age, and the proportion mature-at-age, summed over all ages in the population.

Stock-recruit (S-R) relationship - The functional relationship between the biomass (or number) of spawning stock and the resultant biomass (or number) of recruits.

Strip transect - See Transect below.

Subtidal geoduck - A geoduck living at a depth never uncovered by the tides (i.e., below the level of the extreme low spring tide at a given location).

TAC (Total Allowable Catch) - The number or weight of fish which may be harvested in a specific unit of time. As used in this report, the product of the estimated biomass of harvestable geoducks and the recommended annual harvest rate.

Total mortality rate (Z) - The ratio of number of animals which die from all causes per unit of time to the population abundance at that time, if all dead animals were to be immediately replaced so that the population does not change (an instantaneous rate).

Tract - A subtidal area with defined boundaries which contains geoducks. See *Definition of Key Terms* for a full discussion.

Transect - The secondary sampling unit for geoduck density. In this report, a standard strip transect 150 ft long by six ft wide (= 900 ft^2) within which divers count all geoducks which are "showing."

WDFW - Washington Department of Fish and Wildlife.

Abstract

WDFW is mandated to perform biological stock assessment of the commercial geoduck resource and to make annual recommendations on the Total Allowable Catch (TAC) for each geoduck management region. Systematically spaced strip transect surveys are used to estimate the density of harvestable geoducks within commercial tracts, and a sample of geoducks is taken from these transects to estimate average weight. Biomass estimates on commercial tracts are the product of mean biomass per unit area and the total area of the tract. Regional biomass estimates are the sum of all surveyed commercial tract estimates within the region. Regional TACs are the product of the regional biomass estimate and the recommended harvest rate. An age-based equilibrium yield model was used to predict the long-term consequences of various harvest rates, using geoduck life history parameters which were estimated from existing WDFW data and literature sources. The model predicts yield and spawning biomass per recruit over a range of fishing mortality rates. Five commonly-used constant harvest rate strategies were simulated with the model, including two based on yield-per-recruit analysis and three based on spawning biomass per recruit analysis. An $F_{40\%}$ strategy is recommended as a risk-averse policy for geoducks. Under an $F_{40\%}$ strategy, the recommended annual TAC is 2.7% of the current commercial biomass within a region.

Introduction

Geoduck clams *(Panopea abrupta)* dominate the biomass of benthic infaunal communities in many parts of Puget Sound. Goodwin and Pease (1989) summarized the biology and commercial dive fishery for geoducks, which began in 1970 in Washington state. Commercial fisheries also exist in British Columbia and Alaska (Campbell *et al.* 1998), and geoducks now provide the most valuable commercial clam harvest on the Pacific coast of North America. The average annual ex-vessel value of Washington's geoduck harvest from 1990-1998 was US$14 million. From 1971 through 1998, annual landings have averaged 3.3 million pounds.

The commercial geoduck fishery is jointly managed by two state agencies, Washington Department of Fish and Wildlife (WDFW) and the Department of Natural Resources (DNR), as well as the treaty Indian tribes with shellfishing rights affirmed by a 1994 federal district court judgement (the Rafeedie decision). WDFW's role is to perform biological stock assessment of the resource and to make recommendations on the Total Allowable Catch (TAC) which is expected to maintain a stable long-term commercial fishery.

WDFW began SCUBA diving surveys of geoducks in 1967, and surveys have continued on a yearly basis since that time, with a number of improvements and modifications. Treaty Tribes under co-management with the state began geoduck surveys in 1996. A modified Ricker yield-per-recruit model was adopted in 1981 for use in setting the statewide TAC. This initial research was updated and adopted by state and Tribal managers in 1997 with an equilibrium yield model.

This report describes the methods currently used by WDFW and treaty Tribes to assess subtidal geoduck populations and make annual recommendations on the TAC.

Part I of this paper describes the procedures used to estimate the biomass of harvestable geoducks on subtidal tracts of land. Part II describes the simulation of various harvest rate strategies via equilibrium yield modeling, and recommends a harvest rate based on this modeling to be used in the calculation of the TAC.

Goals and Objectives of Geoduck Stock Assessment

The long-term goal of geoduck stock assessment is to provide managers with the biological information needed to recommend a Total Allowable Catch (TAC). Currently, managers recommend separate TACs for each of six geoduck management regions, the boundaries of which are shown in Figure 1.

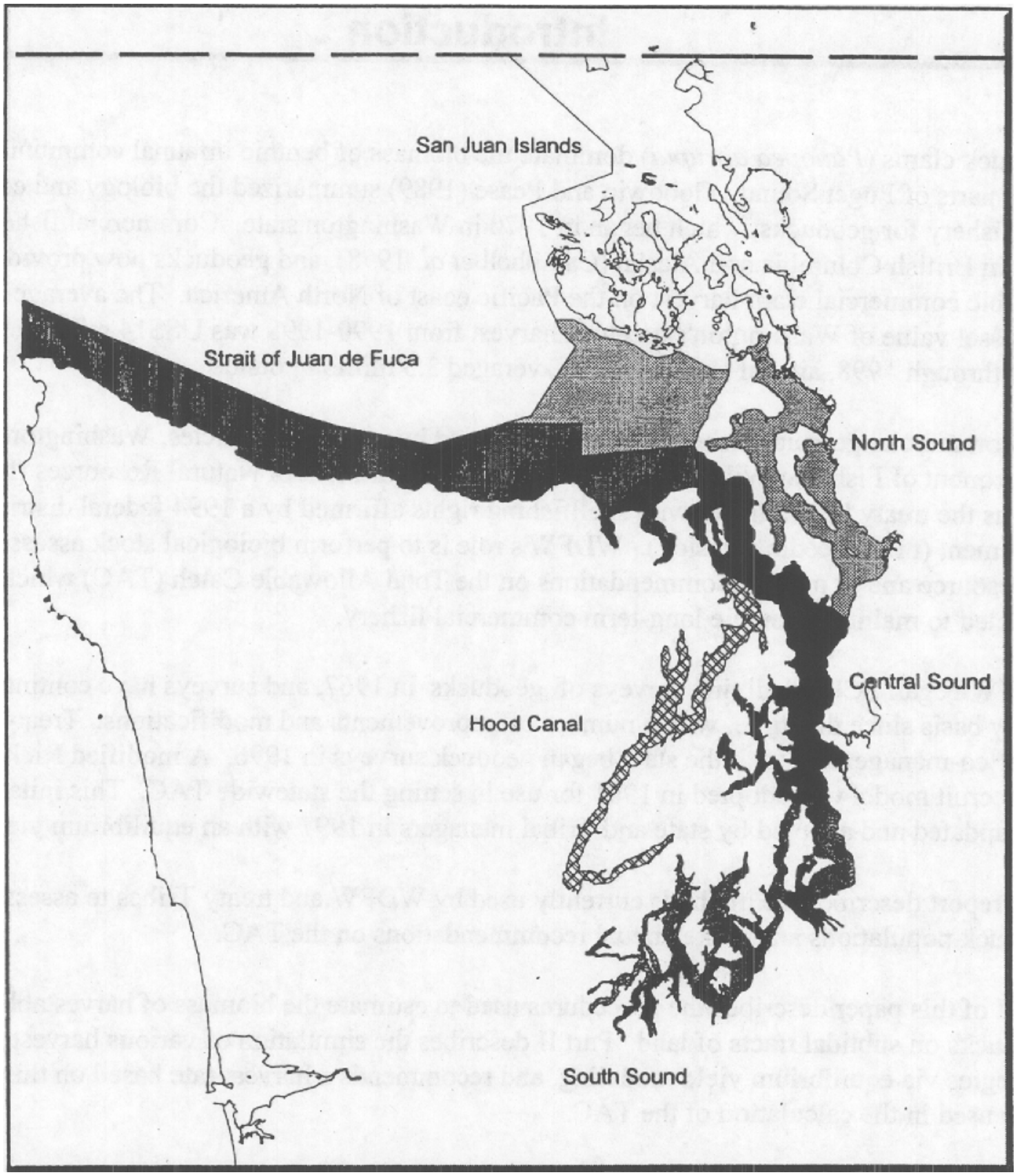


Figure 1. Six geoduck management regions.

The TAC for a given management region is the product of the current estimate of harvestable biomass of geoducks in the region and the recommended harvest rate for the region. Thus, the two short-term goals of geoduck stock assessment are: 1) To estimate harvestable geoduck biomass in each region, and: 2) To recommend a biologically sustainable harvest rate.

In order to reach the first of these short-term goals -- an estimation of harvestable biomass in each region -- dive surveys are carried out each year on relatively small subtidal areas known as "tracts." The objective of such surveys is to estimate the biomass of harvestable geoducks within the confines of the tract. The sum of biomass estimates on all commercial tracts surveyed within a region comprises the regional biomass estimate. Since only a few tracts can be surveyed each year, regional biomass estimates consist of the most recent estimate for each surveyed tract in the region, with known commercial catches subtracted from those tracts which are fished. Biomass estimates for all surveyed tracts are summarized yearly in the annual Geoduck Atlas. The Atlas is published by WDFW in collaboration with the treaty Tribes and is available from the Point Whitney Shellfish Laboratory. Part I of this paper describes the current procedures for making biomass estimates.

In order to reach the second short-term goal -- recommendation of an annual harvest rate -- estimates of important geoduck life history parameters were used to drive an age-based equilibrium yield model. The objective of this yield modeling was to predict the long-term effect of various harvest rates on equilibrium yield and spawning biomass per recruit, and to recommend one of these harvest rates for use in computing regional TACs. Part II of this paper describes the yield modeling and the rationale for recommending a particular harvest rate.

Figure 2 is a flow chart which shows these steps leading to regional TAC recommendations.

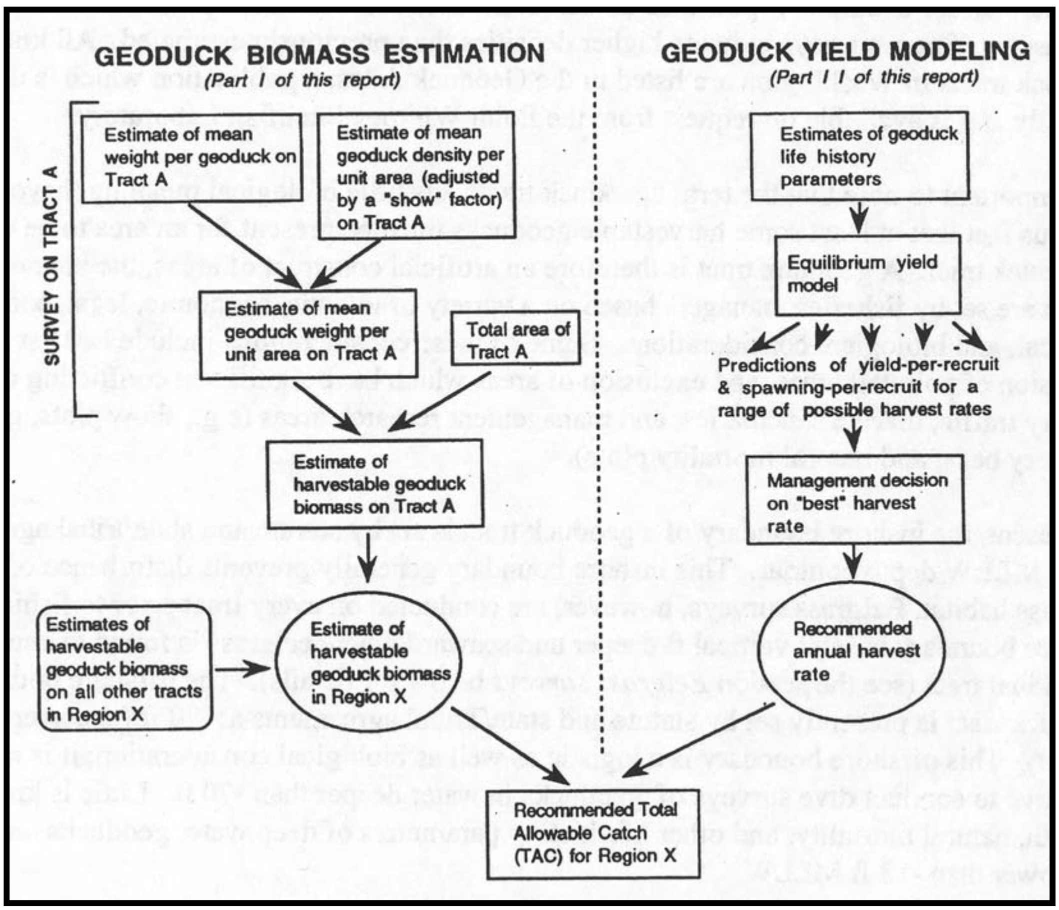


Figure 2. Steps leading to a geoduck biomass estimate on Tract "A" and an annual TAC recommendation for Management Region "X."

Definition of Key Terms

Three key terms -- geoduck tract, geoduck bed, and harvestable geoducks -- are used extensively throughout this paper. All three deserve a thorough definition because they are frequently a source of confusion for biologists, managers, and the public.

Geoduck Tract

A geoduck tract is any subtidal area with well-defined boundaries which contains geoducks. Boundary lines are typically referred to as inshore, offshore, and side boundaries. ***Commercial tracts*** are those tracts in which geoduck densities are considered high enough to support a fishery and which have no other drawbacks to fishing (e.g., pollution, narrow width, land-use conflicts, poor quality geoducks, difficult digging conditions, conflicts with threatened or endangered species or their habitats, etc.). ***Non-commercial tracts*** are those tracts which cannot be fished for one or more reasons, including those listed above. The status of a tract is always subject to change; for example, commercial tracts may become non-commercial if they become polluted, or if they are fished out following a commercial harvest; non-commercial tracts may become commercial, for example, if pollution abates, if the area recovers from past fishing, market prices increase, or if new surveys indicate higher densities than previously estimated. All known geoduck tracts in Washington are listed in the Geoduck Atlas, a publication which is updated annually and is available on request from the Point Whitney Shellfish Laboratory.

It is important to note that the term "geoduck tract" has little biological meaning, beyond the obvious fact that at least some harvestable geoducks must be present for an area to be considered a geoduck tract. A geoduck tract is therefore an artificial construct of areas, the boundaries of which are set by fisheries managers based on a variety of logistic, economic, legal, social, political, and biological considerations. Some of these considerations include harvest control, exclusion of polluted areas, and exclusion of areas which have significant conflicting uses -such as ferry traffic, marine sanctuaries, and management research areas (e.g., show plots, geoduck recovery beds, and natural mortality plots).

At present, the inshore boundary of a geoduck tract is set by statute and state/tribal agreements at -18 ft MLLW depth contour. This inshore boundary generally prevents disturbance of sensitive eelgrass habitat. Eelgrass surveys, however, are conducted on every tract prior to fishing, and the inshore boundary is set 2 vertical ft deeper and seaward where eelgrass is found to occur on an individual tract (see the section *Eelgrass surveys* below for details). The offshore boundary of a geoduck tract is presently set by statute and state/Tribal agreements at -70 ft (uncorrected for tide height). This offshore boundary is a logistic as well as biological consideration; it is not cost-effective to conduct dive surveys of geoducks in water deeper than -70 ft. Little is known about growth, natural mortality, and other life-history parameters of deep water geoducks and geoducks shallower than -18 ft MLLW.

Although the inshore and offshore boundaries of what is currently considered a geoduck tract are strictly defined by statute and state/tribal agreements as noted-above, the side boundaries of a tract are flexible. Side boundaries should enclose areas which have adequate survey information, and therefore may never lie more than 500 feet from the nearest grid line of transects (see *Variations on the standard grid line layout* below). Subtidal geoducks may be found along the entire shoreline of Puget Sound, albeit at very low densities in some areas. Thus, there is no specific point along the shoreline at which any tract can be said to "end" because geoducks are no longer present. In some cases, managers fix the "end" of a tract at the point along the shore where geoduck density falls below the current standard for commercial density. This is an arbitrary economic standard, however, which is subject to change with market prices. Virtually all commercial geoduck tracts contain areas within which density falls below the commercial level.

Besides the density of geoducks, other considerations in setting the side boundaries of tracts include: navigational channels, ferry lanes, steeply sloping bottom contours which "pinch" the tract to a width which makes commercial fishing impractical, and prohibited areas classified as such by the Washington Department of Health. Some tracts in the current Geoduck Atlas end simply at the point where biologists ran out of time during the survey season to continue surveys along the shoreline.

The side boundaries of a tract may be set before performing surveys, during the survey, or afterwards. In any case, the final side boundaries of a surveyed tract are usually modified based on survey findings. For example, survey transects may fall within areas which are later found to be polluted or near navigational hazards, and these transect data may later be eliminated from the tract. It is usually easier and more cost-effective to throw out survey data from small portions of a tract than to return to the field and perform additional surveys.

Because tract boundaries are set at the convenience of fisheries managers, there is, in theory at least, no limit on the size of a geoduck tract. There are, however, practical limits. The management and survey costs per acre increase dramatically as tracts become smaller, making very small tracts uneconomical to lease. The smallest tract listed in the 2000 Geoduck Atlas is four acres, and the smallest tract ever commercially fished was five'acres (Cooper Point). Extremely large tracts generally contain so much geoduck biomass that they may be divided into smaller tracts which can be fished in accordance with annual TACs or harvest shares. Large tracts also present compliance and enforcement problems. For example, the largest tract listed in the 1997 Geoduck Atlas was 2,452 acres (Jamestown 1, Tract #00450). Based on recent surveys this tract was subsequently reconfigured, and in the 2000 Geoduck Atlas appears as a 331 acre tract. The largest commercial tract in the 2000 Geoduck Atlas is 723 acres (Battle Point North, Tract #07000). The mean size of all tracts listed in the 2000 Geoduck Atlas is 106 acres (n = 267 tracts).

Existing tract boundaries may change annually to fit management needs. Large tracts are frequently divided into smaller ones, and small adjacent tracts are often joined to form a single, larger tract. New surveys may increase the side boundaries of certain tracts which had been previously surveyed.

Geoduck Bed

A geoduck bed is an aggregation of geoduck clams in the marine environment. Geoducks will recruit to areas with suitable substrate (sand or sand/mud mixtures), adequate current, sufficient food, and few predators. Geoduck beds occur from the intertidal zone to deep subtidal areas. A geoduck tract is typically a subset of a geoduck bed.

Harvestable Geoducks

Harvestable geoducks are those of a size in which the siphon or "show" is likely to be seen by a diver. Virtually all geoducks visible to experienced divers are of a marketable size. Washington samples indicate that geoducks first enter the fishery at 300 g, a weight which is usually attained between five and seven years (see the sections on *Growth* and *Fishery selectivity* in Part II of this paper). WDFW geoduck transect counts and weight samples made using the procedures described in this paper are assumed to closely mimic this commercial pattern of selectivity. In support of this assumption, we note that only 2% of the 11,181 geoducks sampled by WDFW divers during surveys from 1973-1985 weighed less than 300 g (Goodwin and Pease 1987).

Obviously, geoducks which are too small to be seen by divers are neither harvested by fishers nor counted by WDFW surveyers. Therefore, the procedures described in this paper for estimating the biomass of *harvestable* geoducks necessarily underestimate *total* geoduck biomass, because most geoducks <300 grams or <4 yr old are not counted. The only method which has been used to effectively sample geoducks smaller than this size in a quantitative way is excavation with a venturi suction dredge (Goodwin and Shaul 1984). Venturi samples, while useful for recruitment research on a very small spatial scale, are far too laborious and costly for estimating geoduck densities over large areas.

Part I. Estimation of Harvestable Geoduck Biomass

Sample Design for Estimating Geoduck Biomass

The objective of geoduck surveys is to estimate the biomass of harvestable geoducks within a specific tract. Biomass per unit area within the tract is estimated as the product of mean density and mean weight per geoduck; total biomass on the tract is estimated as the product of biomass per unit area and total area. Strip transect surveys are first carried out to estimate mean density within the tract, and a sample of geoducks is later taken from a subsample of these transects to estimate mean weight per geoduck.

The sample or target population is therefore all harvestable geoducks within the tract boundary. The experimental or sampling unit for geoduck density is a 900 ft^2 strip transect. The estimator is the mean density (in numbers per ft^2) of harvestable geoducks, i.e., the mean density from all transects taken within the tract. The experimental or sampling unit for geoduck weight is a cluster sample of ten geoducks haphazardly dug with commercial gear from a transect. The estimator is the mean weight (in grams) of all geoducks sampled within the tract.

The subsections below present the sampling and statistical methods used to estimate mean density mean weight per geoduck, total biomass, and the statistical precision of the total biomass estimate

Estimation of Mean Geoduck Density

The density of harvestable geoducks within a tract is estimated by a systematic sampling technique first developed in 1967 (Goodwin 1973; Goodwin and Pease 1991). A series of standard strip transects, each comprising an area six ft wide by 150 ft long (a total area of 900 ft^2) are taken along grid lines which run directly offshore from the -18 ft MLLW contour to the -70 ft contour **t uncorrected).** The grid lines (primary sampling units) begin at a randomly-selected starting point along the shoreline of the tract and are spaced systematically in both directions thereafter at 1000 ft intervals. Transects (secondary sampling units) are then taken back-to-back along each grid line. Figure 3 shows the arrangement of systematic samples on a typical tract. The section *Geoduck Survey Methods* below describes in detail the procedures used in the field.

The density of geoducks observed by divers within an individual transect is always an underestimate of the actual density present within that transect (Goodwin 1973; Goodwin 1977). Geoduck siphons may be retracted below the surface of the substrate, cryptic at the surface of the substrate, or obscured from view of the diver. The number of geoducks "showing" (i.e., observable to divers) compared to the number of geoducks actually present in the substrate is a function of various environmental factors such as food availability, water temperature, substrate type, algae cover, turbidity, and currents (Goodwin 1977).

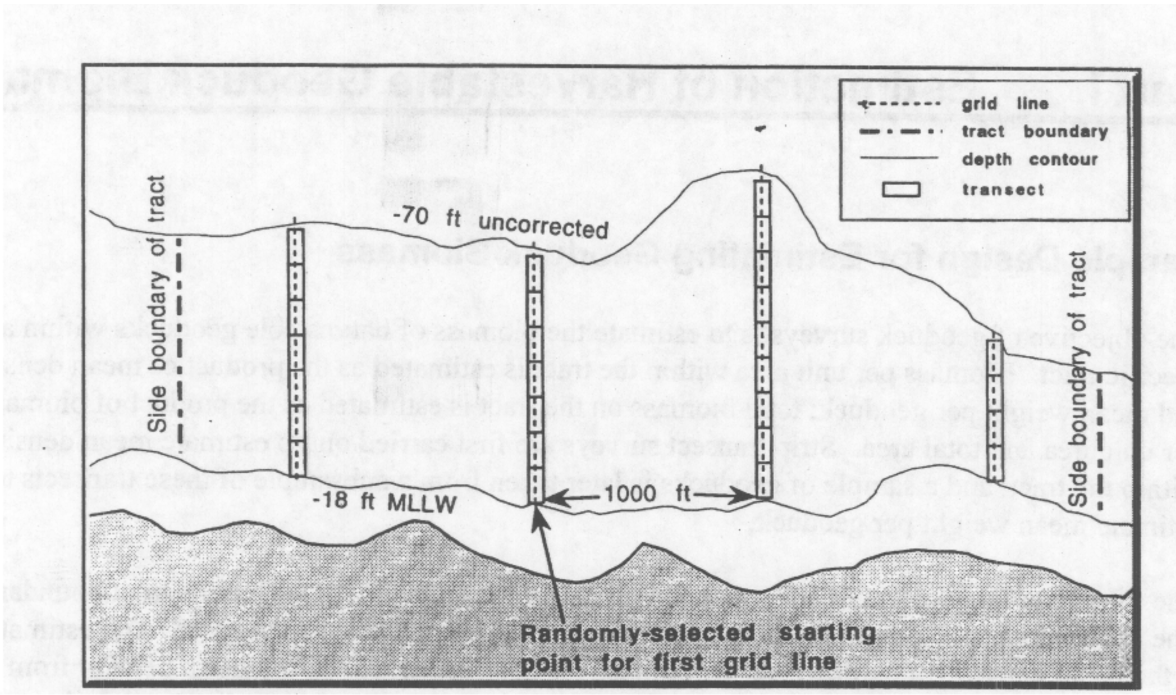


Figure 3. Typical layout of systematic grid lines and transects on a geoduck tract.

The estimate of geoduck density (number of geoducks per ft^2) for an individual transect is calculated by adjusting the observed density by a show factor as follows:

$$d_i = d_{obs}/S \quad (1)$$

where

d_i = density of geoducks (number per ft^2) on the ith transect

d_{obs} = density of geoducks (number per ft^2) observed by divers during a survey on the ith transect (for a 900 ft^2 transect, this is simply the total number of geoducks observed by both divers divided by 900). Note that the counts of both divers are summed to produce a single d_{obs} for each transect. Although it is tempting to consider each diver's count as a separate d_{obs} (therefore doubling the sample size), this would amount to "pseudoreplication" (Huribert 1984), because the two counts along the same 150-ft transect are obviously not independent.

S = "show factor" (within any defined area, the ratio of visible geoduck "shows" from a single observation and the true abundance of harvestable geoducks). A show factor of 0.75 is currently used to estimate density on all tracts for pre-fishing surveys unless there is a show plot established for a tract that will give site-specific data. Use of 0.75 as a constant show factor is a management decision that is assumed to give a conservative estimate of harvestable biomass. The section *Show plot surveys and show factors* below provides the basis for the 0.75 show factor, as well as detailed field procedures for establishing and counting a show plot.

The mean density of geoducks on a given tract is estimated as:

$$D = \Sigma d_i / n_D \qquad (2)$$

where

D = estimated mean density of geoducks

d_i= density of geoducks (number per ft^2) on the ith transect, adjusted by show factor as described above

n_D = sample n for density (number of transects surveyed)

The variance of the mean density (δ^2_D) is estimated as

$$\delta^2_D = \Sigma (d_i - D)^2 / n_D - 1 \qquad (3)$$

Estimation of Mean Weight per Geoduck

Following transect surveys, a series of cluster samples, each consisting of ten geoducks, are taken with commercial water jet harvest gear at systematically spaced intervals along each of the survey grid lines. Empirical studies suggest that reasonably precise and unbiased weight samples can usually be obtained by taking a cluster sample of ten geoducks systematically at one of every six to eight transects, beginning from a randomly selected transect (see the section *Sample size* below). This procedure ensures that all water depths are sampled, an important consideration because depth is a known biological gradient with respect to geoduck weight (Goodwin and Pease 1991). Because of the considerable set-up time involved in digging samples, cluster samples from a few systematically spaced transects are far more cost-effective than samples of individual geoducks from a large number of systematically or randomly chosen transects. The section *Geoduck dig (weight) samples* below provides detailed field and laboratory procedures for selecting dig stations, digging and processing the samples.

Mean weight per geoduck on a given tract is estimated as:

$$W = \Sigma w / n_W \qquad (4)$$

where

W = estimated mean weight per geoduck

w_i = weight of the ith geoduck from dig samples

n_W = sample n for weight

The variance of the mean weight per geoduck (δ^2_W) is estimated as:

$$\delta^2_W = \Sigma (w_i - W)^2/n_w - 1 \quad (5)$$

Estimation of Total Geoduck Biomass on the Tract

The estimate of total geoduck biomass on a tract is calculated as:

$$B_{tract} = (D)(W)(A) \quad (6)$$

where

B_{tract} = total geoduck biomass on a tract (in pounds)

D = estimated mean density of geoducks (number per ft^2, adjusted by a show factor as described above)

W = estimated mean weight per geoduck (in pounds)

A = total surface area of the tract (in ft^2) determined from GIS mapping software and tract maps prepared by DNR (see *Tract Mapping and Grid Line Placement Methods* below).

Precision of Geoduck Biomass Estimates

Statistical precision of the biomass estimate is reported in terms of the commonly-accepted 95% upper and lower confidence limits (i.e., $\alpha = 0.05$, two-tailed).

Confidence limits are calculated based on an estimate of the variance of the biomass (B), which is in turn the product of mean density and mean weight per geoduck. A standard variance-of products formula (Goodman 1960) is used to calculate an unbiased estimate of this product. If geoduck density and weight per geoduck (i.e., *D* and *W)* are independently subject to sampling error (i.e., there is no correlation between density and weight), then the variance of *B* is given by:

$$\delta^2_B = D^2[\delta^2_W/n_W] + W^2[\delta^2_D/n_D] - [\delta^2_D \delta^2_W/n_D n_W] \quad (7)$$

where

δ^2_B = variance of B, estimated geoduck biomass per ft^2

D = mean density of geoducks (number per ft^2 adjusted by a show factor as described above)

W = mean weight (pounds per geoduck)

δ^2_D = variance of mean density

δ^2_W = variance of mean weight

n_D = sample n for mean density (number of transects)

n_W = sample n for mean weight (number of geoducks weighed) The standard error (se) of *B* is calculated as the square root of δ^2_B.

The 95% confidence bound for a given geoduck tract of known size is given by:

$$B_{tract} \pm (t_{0.05,2,v})(se)(A) \qquad (8)$$

where

B_{tract} = estimated geoduck biomass on an entire tract (in pounds)

$t_{0.05,2,v}$ = tabled t-value, $\alpha = 0.05$, two-tailed, v = df (degrees of freedom)

se = standard error of *B*

A = total area of tract (in ft^2)

For pre-harvest surveys on commercial beds, state and Tribal managers have agreed on a required precision for total biomass estimates of ±30% at the $\alpha = 0.05$ confidence level. In other words, the 95% confidence limit as calculated above must lie within ± 30% of the estimate of *B*.

Sample Size

The goal of pre-fishing surveys on an individual tract is to survey a sufficient number of transects and dig a sufficient number of geoducks to allow an unbiased estimate of geoduck biomass with 95% statistical confidence bounds which lie within ± 30% of the biomass estimate itself (see the section above). On the majority of tracts, the sample size required to meet these goals can be achieved by running a series of transects along grid lines placed systematically every 1,000 feet along the -18 ft MLLW contour, and digging a cluster sample of ten geoducks at every sixth to eighth transect. However, it is not always possible to achieve the required precision with this sampling scheme, particularly on narrow tracts, small tracts, or tracts with highly variable substrates. Two methods are used to roughly estimate the sample size (i.e., the number of transects) needed to meet the statistical precision requirements:

1. Prior to performing the survey, an empirically derived "rule of thumb" may be used in conjunction with the known surface area of the tract to roughly estimate the required number of transects. Evidence from past surveys on a variety of beds indicates that the sampling intensity listed in Table 1 usually meets or exceeds the required degree of statistical

precision. Note, however, that these are rough guidelines for pre-survey planning only, and in no way guarantee that biomass estimates will meet the precision requirements.

Table 1. Empirically derived guidelines for roughly estimating the sample size (number of transects) needed to meet statistical precision requirements on geoduck tracts of different sizes.

Size of tract (acres)	Number of 900 ft^2 transects per acre
1-5	3
6-15	2
16-50	1
51-100	**0.66**
100+	0.33

2. Once transect surveys have been completed along grid lines spaced every 1,000 feet along the -18 ft MLLW contour, it is possible to determine whether additional transects must be run based on the variance of transect counts already performed. Table 2 shows the coefficients of variation (CVs) for both mean geoduck density *(D)* and mean weight per geoduck *(W)* from 13 recently-surveyed tracts, and suggests that precision of the biomass estimate is almost totally dependent on the variance of density. Doubling the CV of density almost always produces a result within one or two percentage points of the precision of the biomass estimate. For example, doubling the CV of density on the Eld Inlet East tract results in 28.2%; after geoduck samples were dug and weighed, confidence bounds on the estimate of biomass were ± 29.1 % of the estimate. By contrast, Table 2 shows that there is no such relationship between the CV of weight and the precision of biomass estimates. Thus, from the standpoint of statistical precision of the biomass estimate, the number of geoducks sampled for *weight* and the *variance of the mean weight* are irrelevant based on the tracts listed in Table 1.

The relationship shown in Table 2 between the CV of mean density and the 95% confidence interval makes it possible to predict with near certainty the precision of a tract's biomass estimate while still in the field, long before any geoducks are dug for weight samples.

A hand-held calculator with statistical function capabilities can be used to readily estimate the CV of density in the field, after transect surveys have begun. Actual geoduck counts from each completed transect may be used for this calculation, without applying either a show factor or converting the counts to a density estimate; CVs are unit-free relative measures of variance, and will therefore be identical in any case. The procedure is as follows:

1. Individually enter the geoduck counts from all transects (d_{obs}).

2. Have the calculator estimate the mean number of geoducks per transect and the sample variance (Ex: mean = 56.91 geoducks/transect and sample variance = 1,782.36).

Table 2. Sample size at 13 commercial tracts, coefficients of variation (CVs) for mean geoduck density and mean weight per geoduck, and the resulting 95% confidence intervals on the biomass estimates. Calculations are based on initial tract estimates.								
Tract	Size (acres)	n (number of transects)	Transects / acre	Dig samples	Transects / dig sample	CV of mean density *(%)*	CV of mean weight *(%)*	95% CI on biomass (as % of *B)*
Arcadia 2	26	27	1.04	6	4.5	9.6	4.5	20.8
Bridge	35	34	0.97	4	8.5	14.5	6.9	31.5
Eld Inlet East	54	31	0.57	5	6.2	14.1	4.6	29.1
Arcadia 3	55	43	0.78	7	6.1	12.2	4.8	25.7
Eld Inlet West	79	47	0.59	14	3.4	14.1	2.7	28.1
Arcadia 4	118	82	0.69	14	5.9	7.8	4.3	17.5
Skiff Point	126	41	0.33	11	3.7	8.6	3.9	18.5
Blake Is North	144	61	0.42	6	10.2	12.7	6.5	28.0
Port Gamble	217	161	0.74	45	3.6	7.7	1.8	15.6
Murden Cove	222	68	0.31	19	3.6	7.2	3.1	15.4
Olele Point	225	91	0.40	16	5.7	6.8	2.9	14.5
Jamestown	255	67	0.26	9	7.4	8.6	3.3	18.0
Warrenville	316	102	0.32	8	12.8	13.4	5.2	28.2

3. Divide the sample variance by the number of transects to produce the variance of the estimator (Ex: n = 117 transects, so 1,782.36/117 = 15.23).

4. Take the square root of this number to produce the standard error of the estimator *(Ex:* square root of 15.23 = 3.90).

5. Divide the standard error of the estimator by the mean number of geoducks per transect and multiply by 100 to produce the coefficient of variation *(Ex: CV* = (3.90/56.91)100 6.85 1

6. Double the CV to roughly estimate the width of the 95% confidence bound as a percentage of the biomass estimate (Ex: 2(6.85) = ± 13.7%. In this example, the precision lies well below the required limits of ± 30%, so that additional transects need not he taken). Doubling the CV roughly approximates the tabled t-value of 1.96 for an infinite number of samples (given a two-sided test with a = 0.05), and is usually sufficient for *rough* field calculations.

If using this method to determine when enough transects have been run, it is important to note that this estimate of sample size may only be carried out *after* transects have been taken in a representative fashion throughout the entire tract (e.g., along grid lines spaced systematically every 1.000 feet apart). It is entirely possible, for example, to reach the desired statistical precision after only a few transects have been taken in a tiny corner of the tract; such a sample would be precise, but would very likely be biased. The same is true for geoduck weight samples which, to avoid bias, should be taken at systematic, random, or stratified random intervals throughout the entire tract.

Additional transects may be needed for certain tracts to reach the desired level of precision. Placement of additional transects within a tract is discussed below in. the section *Variations on the standard grid line layout.*

Rationale for Systematic Sampling

A systematic grid sample was chosen to estimate geoduck biomass rather than a simple random sample for reasons of cost and convenience. To avoid decompression sickness, divers are limited in the amount of time they may spend sampling at depth, so that economizing bottom time becomes a paramount consideration in choosing underwater sampling designs. Systematic samples provide far greater information per unit time than simple random sampling, as illustrated in the example below.

Each transect typically takes experienced divers four minutes to complete, plus the time required to descend and ascend from the dive. Divers generally take about one minute to descend, become oriented, and record initial data. When surfacing following the dive, divers must ascend at a rate between 0.5 and 1.0 ft per second. Additionally, WDFW divers are required by safety regulations to perform a three-minute safety stop at -10 to -15 ft on every ascent to decrease the risk of decompression sickness (WDFW Diving Operations Manual, November 1991). Thus, a single transect at -60 ft would take between 9 and 10 minutes, and a random sample of 50 such transects would require as much as 500 minutes of diver time; only 200 minutes of this time is actually spent surveying geoducks, while the remaining time is used for descents, ascents, and safety stops.

A systematic grid sample, on the other hand, is considerably more economical in terms of diver time because there is only one descent and one ascent per grid line. Thus, the same 50 transects taken along systematically-spaced grid lines (assume, for this example, ten grid lines and five transects per line) would require only 260 minutes of diver time, and only 60 minutes of this time is used in descents, ascents, and safety stops. In practice, the time savings of systematic sampling versus random sampling are even greater, because the US Navy Dive Tables require that bottom times be rounded up to the nearest five minute increment, thus imposing an additional "penalty" for numerous single-transect dives. The time savings of systematic samples over random samples increases on tracts which are extremely wide (i.e., where each line consists of many transects) and decreases on tracts which are narrow due to steeply sloping bottom contours. Extremely narrow tracts, however, may be economically sampled using systematically spaced oblique or zig-zag lines (see *Variations on the standard grid line layout* below).

Besides the considerable savings in dive time, there are additional advantages to a systematic sample when surveying geoducks. Choosing a single random starting point along the shoreline and then spacing lines of transects every 1,000 feet is much simpler, with fewer start positions, and less prone to selection error than attempting to choose random 900 ft^2 samples throughout a tract. Systematic line sampling also permits the most precise mapping of boundaries and spatial patterns in geoduck density.

Despite the cost benefits of a systematic sample for geoducks, classic sampling theory cautions that there are two potential disadvantages to systematic sampling: 1) It is impossible to guarantee that the estimate of mean density derived from a systematic sample is unbiased, and; 2) It is not possible to obtain an unbiased estimator of the variance of mean density from a systematic sample.

While there is no guarantee, from a theoretical standpoint, that systematic samples of geoduck density will be unbiased, we believe that the sampling protocol outlined above is no more likely to produce biased estimates than a simple random sample of the same size. This is because there are no known biological gradients affecting geoduck distribution which occur systematically along the shoreline. Put another way, we know of no variations in geoduck density which occur periodically at 1,000-foot intervals along the shoreline. (Gradients in geoduck density do exist along shorelines, and some of these gradients are even predictable -- such as generally decreasing numbers of geoducks from the mouth of a bay to the stagnant head of the bay. But note that this is not *a systematic* gradient, and could be sampled in a representative way by both systematic or simple random schemes).

On the other hand, lines placed systematically along depth contours (or running parallel to the shoreline) invite biased results, because depth is a known biological gradient with respect to geoduck density (Goodwin and Pease 1991). This is particularly true of samples consisting of transects along only one or two such lines. Under the recommended sampling protocol above, each line of transects running from the shallow boundary of a tract to its deep water boundary cuts completely *across* the depth gradient, minimizing depth-related bias.

The second potential problem associated with systematic samples is that they do not produce an unbiased estimate of variance (in our case, variance surrounding the estimate of mean geoduck density). The sample design protocol used here calculates variance using the simple random sample formula. Thompson (1992) notes that this leads to unbiased variance estimates only if the population units are randomly distributed; in most natural populations, this procedure tends to overestimate the variance of the mean. Thus, estimates of variance surrounding mean geoduck density when using this sample design are likely to be higher than the true variance. This in turn will tend to inflate the variance estimate surrounding total biomass, and widen the 95% confidence bounds on biomass.

We believe that these are, on balance, minor concerns, and concur with Hilborn and Walters (1992) who recommend systematic samples over simple random samples for surveys of abundance.

Tract Mapping and Grid Line Placement Methods

Individual geoduck tracts are mapped prior to performing surveys. Precise mapping is required for the following reasons:

1. To provide an accurate estimate of the tract's total surface area, which is used in Equation 6 above to estimate harvestable geoduck biomass on the tract.

2. To provide surveyers with information on depth contours which may influence the alignment of the systematic grid lines (see *Variations on the standard grid line layout* below).

3. To provide surveyers with an estimate of the sample size (i.e., the number of transects) needed to meet the required level of statistical precision.

4. To provide surveyers and managers with an estimate of the labor and time costs required to survey the tract.

5. To provide a precise post-survey spatial mapping of both transect locations and geoduck densities within the tract.

6. To develop reproducible and verifiable survey results.

Tract mapping has evolved considerably since geoduck surveys began in Washington in the late 1960s. During these early years, survey locations were first estimated by eye and later came to rely on navigational fixes from LORAN equipment. In recent years, the availability of sophisticated electronic field equipment such as Differential Global Positioning Systems (DGPS) and laser range finders, as well as computerized Geographic Information Systems (GIS), has made it possible to plot survey locations and estimate the surface area of tracts far more precisely than in the past. The sections below describe the methods currently used to map tracts and lay out the sampling grid lines prior to a survey.

Tract Mapping and Surface Area Estimates

Tracts to be surveyed by WDFW are initially mapped by DNR at a scale of one inch = 1,000 ft using a survey-grade DGPS unit. For most current surveys, DGPS positions are justified and plotted on either the NAD 27 (North American Datum 1927) or the WGS 84 (World Geodetic Survey 1984) geographic survey datum. These maps show the shoreline, the inshore, offshore and side boundaries of the tract, and fixed aids to navigation which may be useful in laying out the systematic grid lines for the survey. Side boundaries for the initial tract map depend on information from previous surveys and other management considerations, and are likely to change once survey data are analyzed.

In the case of tracts which have never been surveyed before, exploratory dives are often made to determine the extent of commercial geoduck densities. These exploratory dives may involve underwater sledding, single "bounce" dives spaced haphazardly throughout the area, swims along the shoreline paralleling a depth contour, or haphazardly-placed transect surveys. Such exploratory dives, while useful in defining the geographic boundaries of a tract, do not constitute valid geoduck surveys and cannot be used to estimate either density or biomass for the following reasons: 1) The samples are not systematically or randomly placed, increasing the risk of bias; 2)

Sample size is usually too small to provide the required degree of statistical precision; 3) Variants on the transect method such as sledding or bounce dives cannot be reliably adjusted for either the area surveyed or by existing show factor data; and 4) Depth contour swims are likely to provide biased estimates of geoduck density because they parallel depth, a known biological gradient of geoduck density (Goodwin and Pease 1991).

Once the initial boundaries of a tract have been determined and mapped, estimates of a tract's surface area are estimated with a scaled overlay sheet. Overlay sheets available from the Washington Department of Natural Resources (DNR) Inventory Section are scaled to one inch = 1,000 ft, requiring that the map be scaled appropriately prior to estimating acreage. The overlay sheet is placed haphazardly over the map, and the number of dots on the overlay sheet lying within the tract boundaries is counted, each dot representing one acre. This procedure is repeated several times and the average is taken as the best estimate of surface area.

Tract area estimates are also made by digitizing tract boundary data in Maplnfo, a computerized geographic information system capable of calculating surface area. For most current surveys, DGPS fixes based on either the NAD 27 (North American Datum 1927) or the WGS 84 (World Geodetic Survey 1984) datum are used as input. It is absolutely essential that the same datum be used in creating maps and fixing positions in the field, or huge discrepancies in positions and area estimates will result. These computer-generated estimates of tract area are used to verify the estimate produced by the dot-overlay method.

The surface areas of tracts based on the initial mapping almost invariably change following the survey and prior to finalizing the biomass estimates on a tract. Side boundaries, for example, are likely to shrink if surveys or subsequent information indicate that low geoduck densities, polluted areas, difficult digging conditions, or narrow "pinched" depth contours merit a smaller tract. Inshore boundaries may be moved deeper, for example, if surveys find rooted eelgrass within two vertical feet of the -18 ft MLLW contour. In such cases, only survey data taken within the revised tract boundaries are used in the final estimation of biomass. The side boundaries of a tract may also be expanded -- as in cases where surveyers discover that commercial geoduck densities exist beyond the initial mapped tract -- but in such situations a new map of the expanded tract-is required.

Standard Layout of Systematic Grid Lines

Once the tract has been mapped, the beginning points for systematic grid lines of transects are determined and marked with buoys. Each line of transects begins at the -18 ft MLLW contour; this depth is determined in the field with a fathometer and a tidal correction factor from computer generated daily tide graphs for the area. A point along the tract's -18 ft MLLW contour is randomly selected and a heavily weighted buoy is dropped there. Buoys are subsequently placed at 1,000-ft intervals along the entire length of the tract's -18 ft MLLW contour. Distance between buoys is measured with a laser range finder; a band of reflective tape is wrapped around the top of each buoy to facilitate long distance laser fixes. If a laser range finder is not available, or if rough weather precludes its use, buoys may be placed using DGPS fixes. After this initial buoy placement, a diver descends and re-positions the line exactly along the -18 ft MLLW contour, if required, using a digital depth gauge and a correction factor from daily tide graphs for the area. The diver then anchors the buoy line in the substrate with two or three steel reinforcing bars.

Spacing grid lines every 1,000 ft ensures, in theory at least, that no point within the tract will lie more than 500 ft from the nearest surveyed point. There are cases, however, where systematic placement of grid lines results in larger unsurveyed areas. Because grid lines are spaced beginning from a random starting point, the final grid line on one side of a tract may end up, for example, 900 ft from the tract boundary. To make sure that no point on the tract lies more than 500 ft from the nearest grid line, there are three possible solutions in this example: 1) Extend the tract boundary anywhere between 100 and 600 ft, and run another grid line of transects 1,000 ft from the previous line; or 2) Move the tract boundary at least 400 ft closer to the grid line; or 3) Place another grid line of transects anywhere within 500 ft of the existing tract boundary. Since it is often impossible to extend tract boundaries (due to the presence of hazards, closed areas, other tracts, etc.), and because shrinking tract boundaries reduces fishing area, the third solution is used most frequently. For more details, see *Variations on the standard grid line layout* below.

Once buoys have been systematically placed along the -18 ft MLLW contour at 1,000 ft intervals, the buoy positions are mapped. Whenever possible, buoy positions are mapped with a combination of DGPS fixes and laser range.-finder fixes. Laser fixes rely on triangulation of laser ranges between the buoy and clearly identifiable landmarks appearing on the DNR-generated maps (e.g., fixed navigational aids, bridges, towers, jetties, docks). The laser range-finder currently used is capable of fixing positions marked with a reflective mirror from as far away as 4.8 km. Figure 4 shows and example of buoy mapping on a geoduck tract. On some tracts, it is impractical to shoot laser ranges to shore; in these cases DGPS fixes may be sufficient.

Once these steps have been taken to map the tract, dive surveys are initiated beginning at each of the anchored buoy lines. A series of 900 ft^2 transects are taken along a compass bearing headed directly offshore from each buoy. Figure 3 shows a typical survey layout with grid lines spaced at 1,000-ft intervals and running directly offshore.

Variations on the Standard Grid Line Layout

On some tracts, the survey layout described above -- systematic grid lines running directly offshore every 1,000 feet along the -18 ft MLLW contour -- requires modification. Variations on the standard layout are sometimes required for one or more of the following reasons:

1. To increase cost-effectiveness of the survey in terms of transects surveyed per unit of diver bottom time.

2. To reduce the likelihood of bias due to non-representative sampling of the tract area.

3. To meet the required standard for statistical precision described above.

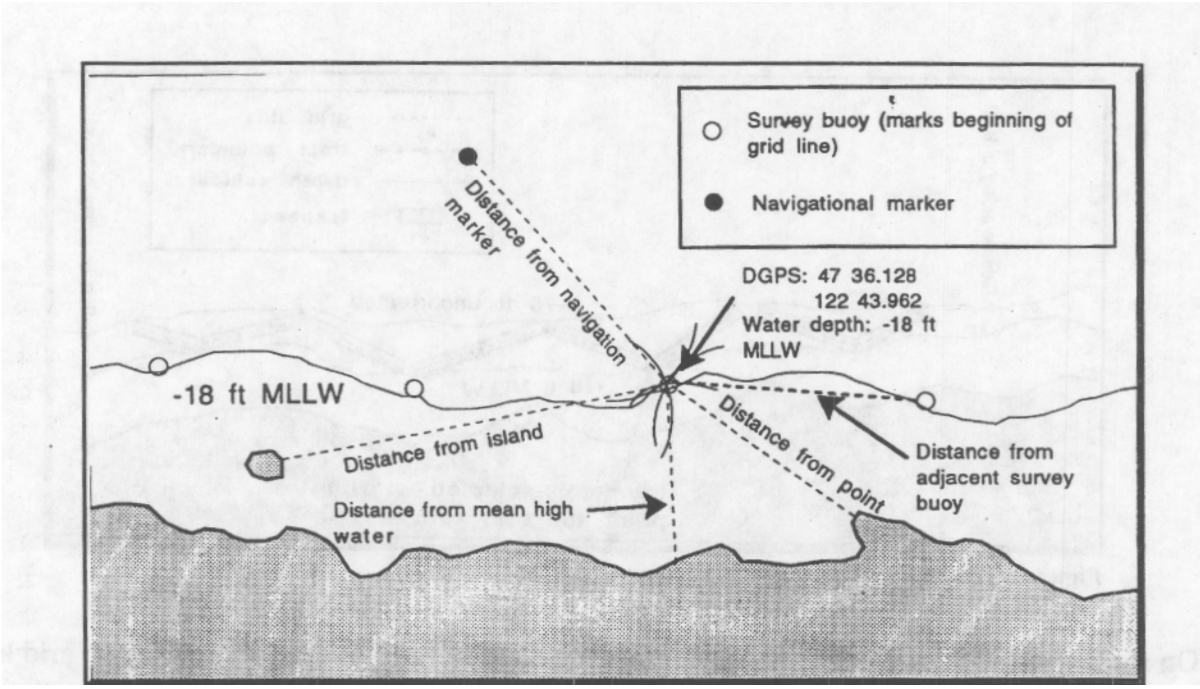


Figure 4. An example of survey buoy mapping on a geoduck tract.

Steep, narrow tracts frequently require a slightly different grid line layout to meet all three of the above goals. On such tracts, the steeply-sloping bottom often allows room for only one 150-ft long transect before divers reach the -70 ft contour. This single transect will require roughly ten minutes of bottom time, only four minutes of which are spent counting geoducks; the other six minutes are used on the descent, ascent, and three-minute safety stop.

In addition to being wasteful, a series of such single transects on a very narrow tract invites bias. This occurs when divers reach -70 ft prior to finishing the transect, and turn to finish the transect along the -70 ft contour. On long, narrow tracts, this may occur so often that a large proportion of the sampling effort takes place along the -70 ft contour. As noted earlier, depth is a known biological gradient with respect to geoduck density (Goodwin and Pease 1991), and transects running parallel to any depth contour are therefore a source of potential bias in the density estimate.

Finally, narrow tracts often fail to produce biomass estimates of the required statistical precision. This is because only one or two transects are possible every 1,000 ft, resulting in a low sample size (unless the tract is extremely long).

To remedy these problems on narrow tracts, grid lines are sometimes placed along oblique or zigzag angles rather than perpendicular to shore. Figure 5 shows examples of oblique and zigzag lines on narrow tracts. Obliques and zigzags allow more back-to-back transects to be surveyed before divers must surface, thus providing more information per unit time, as well as a larger sample size per length of tract shoreline. In the case of zigzag lines, the likelihood of bias is also reduced, because divers immediately turn inshore upon reaching the -70 ft contour, continuing to cut across depth gradients rather than surveying parallel to them.

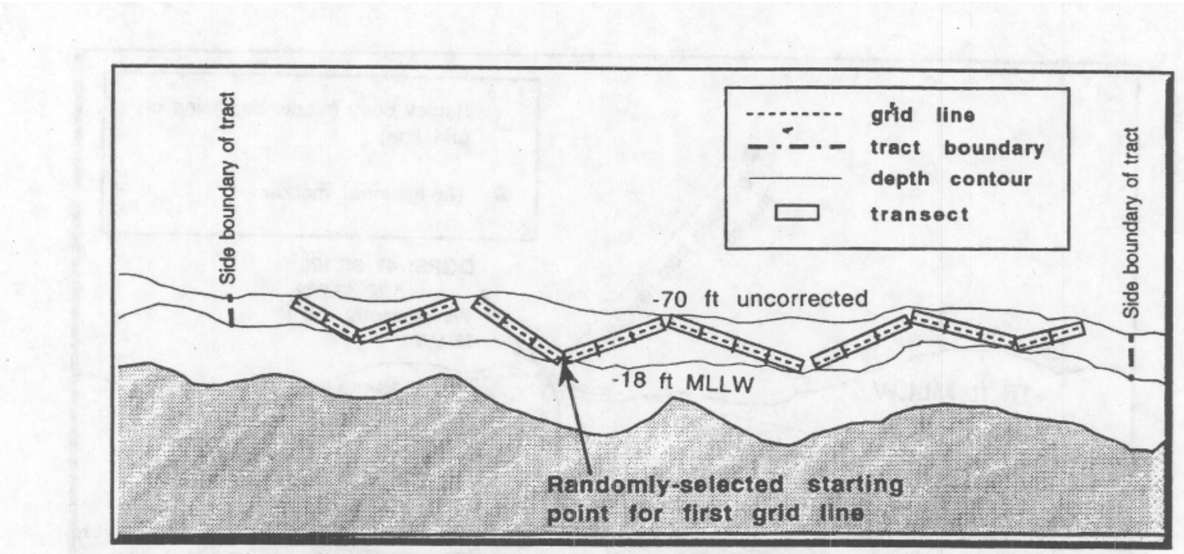


Figure 5. Zig zag layout of grid lines on a narrow geoduck tract.

On some tracts -usually small tracts, or those with highly variable geoduck density -- grid lines spaced systematically every 1,000 ft do not result in biomass estimates of the required statistical precision. Increasing the sample size (i.e., running more transects) is sometimes the answer. This is accomplished by splitting the existing grid lines -- in other words, running lines of transects every 500 ft rather than every 1,000 ft. Note, however, that to reduce the chance for sampling bias, new grid lines must be run between *all* existing lines rather than a select few. There is no limit on the number of times existing lines may be split in this manner to obtain more transects, but there are diminishing returns with respect to precision as sample size increases.

Strict adherence to systematic spacing of grid lines may sometimes result in samples that are not spatially representative of the tract, and are thus likely to be biased. As noted above, the goal of unbiased surveys is to ensure that no point on the tract lies more than 500 feet from the nearest grid line of transects. Because the grid lines are spaced beginning from a random starting point within the tract (rather than the tract boundary itself), situations may arise in which the final grid line of transects lies more than 500 ft from the tract boundary. As noted above, this is most easily remedied by simply adding another grid line of transects anywhere within 500 ft of the tract boundary. The shape of the shoreline may also require additional grid lines. Figure 6 shows an example in which, due to the shape of the shoreline, a large area of the tract would remain unsurveyed with systematically-spaced grid lines. In this example, the logical (but entirely *ad hoc)* remedy was adding another grid line of transects in the middle of the unsurveyed area, such that no point on the tract lies more than 500 ft from the nearest grid line. Similar *ad hoc* sampling schemes are sometimes called for on tracts with "dog-leg" shorelines, islands, or other unusual geographic contours.

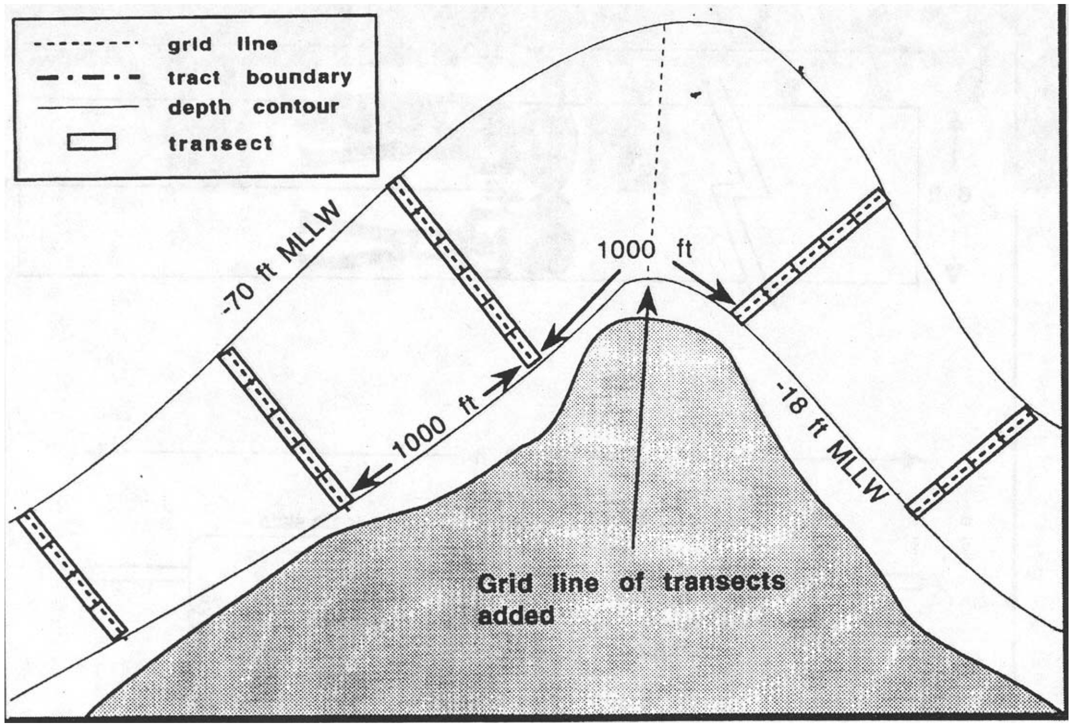


Figure 6. *Ad hoc* placement of an additional grid line of transects to achieve representative sampling of a large area of the tract.

Geoduck Survey Methods

Strip Transect Surveys

To estimate the number of harvestable geoducks within each 900 ft^2 transect, two divers swim side by side, each counting all geoduck siphons, or marks in the substrate which are judged to have been made by geoducks (also called "shows" or "dimples;" see the section *Identification of geoduck shows* below). An individual diver is responsible for counting the geoduck "shows" directly underneath his or her half of the six-ft wide transect rod and spool (Figure 7). Thus, each diver surveys a swath three-ft wide by 150-ft long. The sum of the two diver counts on an individual transect is the total observed number of harvestable geoducks on that transect (d_{obs} in Equation I above). In order to ensure consistent transect length and area, the transect line is periodically re-measured to detect and correct any stretch or shrinkage.

An individual diver attempting to survey geoducks in swaths wider than three ft will generally produce unreliable counts, due to the subtle character of geoduck shows and the poor underwater visibility in Puget Sound. Double counting of geoducks may occur when a diver must scan more than three feet in high-density geoduck areas. An additional problem with variants on the historically-used three-ft transect width is that geoduck show factors used in adjusting density have only been estimated on show plots of this width (see *Show plot surveys and show factors* below). Use of a different transect width may invalidate the use of the currently-accepted 0.75 show factor and require additional studies.

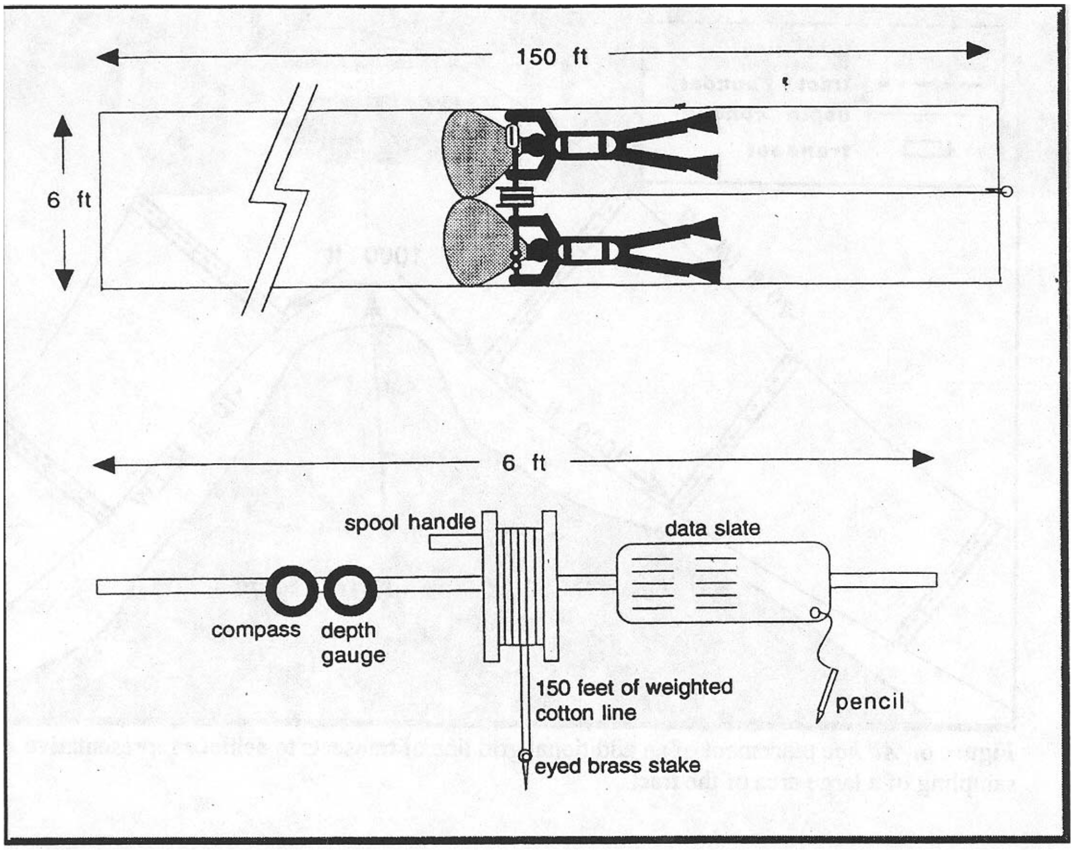


Figure 7. Two divers performing a strip transect survey of geoducks within a 900 square foot transect, and details of the transect pool.

The transect is initiated by planting a metal stake in the substrate which temporarily anchors the 150-ft long transect line. The first transect along any systematically-placed grid line begins at or near the anchored buoy marking that line along the -18 ft MLLW contour (see the section above). A compass course is determined prior to entering the water, generally directing the long axis of the transect perpendicular to the shoreline; oblique or zig-zag courses are sometimes used in surveying extremely narrow tracts as described above. Divers swim along the compass course and away from the shoreline, unspooling the 150-ft transect line as they swim. Each transect typically requires about four to five minutes.

If at the end of the 150-ft line, the -70 ft (uncorrected to MLLW) water depth has not been reached, another transect is initiated along the same compass course. The divers signal the start of each new transect for the boat tender by separating approximately 15-20 ft. One diver remains at the ending point of the transect, recording data for the transect on a dive slate, while the second diver swims back along the transect to respool the transect line. Meanwhile, the tender boat hovers near the divers' bubbles to record the starting position of each transect based on this separation of bubble streams (see *Recording data* below). When the -70 ft water depth is reached, the divers return to the surface and are moved to the next transect buoy to begin another line of transects. If divers reach -70 ft prior to reaching the end of the 150-ft long transect, they turn (generally upcurrent) and finish the transect obliquely toward shore. If a transect ends slightly shallower than the -70 ft contour, divers generally .return to the surface; this avoids the potential bias inherent in counting a transect which lies almost entirely along a depth contour.

Lines of such transects are completed at systematic intervals throughout the bed, generally spaced 1,000 ft apart, until the entire bed has been surveyed at a sampling intensity which produces biomass estimates of a specified statistical precision (see the sections *Precision of geoduck biomass estimates* and *Sample size* above).

Identification of Geoduck Shows

Geoduck siphons, when exposed above the surface of the substrate and pumping water, are easily recognized by their large size, elliptical or oblong shape, a flat (rather than rounded) siphon tip, the absence of tentacles along the inner portion of either siphon opening, and the fact that both siphon openings are the same size. When partially retracted, geoduck siphons may be identified by their elliptical or oblong shape, flat siphon tip, and sometimes by the presence of pellet-like particles of undigested particulate matter (pseudofeces) lying on the surface near the siphon tip. Such "dimples" may be probed with thin neoprene finger gloves for verification; geoducks have a characteristically soft, rubbery texture (as opposed to a slimy feel) with no horny plates on the siphon tip. When probed in this manner, geoducks typically retract their siphons slowly.

Subtidal geoduck tracts almost always contain other animals, however, whose siphons or shows may be confused with geoducks by inexperienced divers. These include other molluscs such as horse clams *(Tresus capax* and *T. nuttallii),* false geoducks *(Panomya spp.),* piddock clams *(Zirfaea pilsbryii),* cockles *(Clinocardium nuttallii),* horse mussels *(Modiolus rectus),* and truncated softshell clams *(Mya truncata), as* well as animals from other phyla (retracted sea pens, for example). Density and biomass estimates will obviously be biased if surveyers count these animals as geoducks, or if they fail to count geoducks under the assumption that they are something else.

Figure 8 shows the major differences between geoducks and those of other subtidal molluscs. Harbo (1997) provides an excellent chapter on siphon identification, including a key and color photographs of many north Pacific clam siphons. WDFW staff provide an annual class on geoduck survey methods which includes color slides of clam siphons and a touch tank containing various clam species buried up to their siphons. The class is open on a first-come basis to tribal shellfish biologists and biologists employed by ecological consulting firms.

The animals most easily confused with geoducks during subtidal surveys are horse or "gaper" clams of the genus *Tresus. Two* characteristics of the siphon tip serve to distinguish both species of horse clams from geoducks: 1) The presence of an inner ring of tentacles on the horse clam's siphon, and; 2) The presence of horny plates surrounding the siphon tip of horse clams. The tentacles are obvious when horse clam siphons are open and pumping. When the siphon is closed, or when the tip is not visible, divers with thin neoprene finger gloves can often probe the siphon and feel the horse clam's horny plates. Typically, horse clam siphons are oval or nearly round in cross-section, while geoduck siphons are elliptical. Horse clams generally retract their siphons faster than geoducks when disturbed, expelling a jet of water. Finally, horse clam pseudofeces are thin and stringy rather than pellet-like.

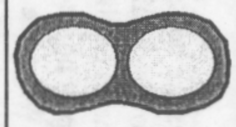
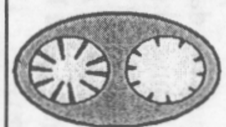
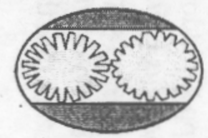
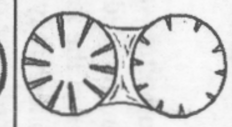
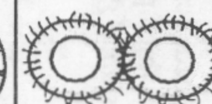
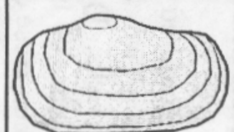
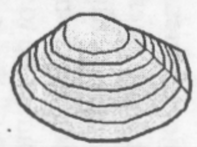
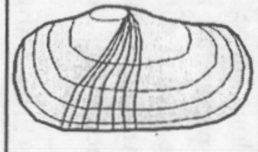
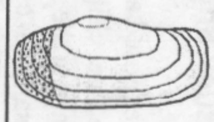
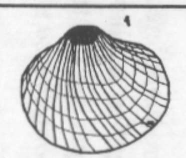
Scientific Name	*Panopea abrupta*	*Tresus sp.*	*Mya truncata*	*Panomya sp.*	*Zirfaea pilsbryii*	*Clinocardium nuttallii*
Common Name	**GEODUCK**	**HORSE CLAM**	**TRUNCATED MYA**	**FALSE GEODUCK**	**PIDDOCK**	**COCKLE**
siphon shape (topview)	 "Double Barrel Shotgun"	 Oval	 Oval	 "Double Barrel Shotgun"	 Bifurcated	 Two Circles
siphon shape (side view)						
tentacles	absent	present/distinct	present/fine	present/very fine	present/distinct	present/distinct
substrate depth	18 to 36 inches	8 to 15 inches	8 to 10 inches	8 to 15 inches	8 to 20 inches	at surface
substrate type	all (except clay)	grvl/cbble/sand	mostly mud/snd	all (except clay)	clay/rock/wood	snd/snd-mud
shell						
siphon color	brown to light brown w/ cream interior	grey/blue tentacles w/ brown exterior	dark brown w/ heavily wrinkled siphon	brown to light brown w/ red and cream circling siphon	mottled reddish and brown with cream white	creamy brown
other distinguishing features	large siphon, smooth/soft, obvious pseudofeces	horny plates on siphon, encrusted plates, hard tip when probed	leathery flaps, index finger shape to siphon	large, thin walled siphon, smooth/soft, different size siphon openings, cleft in shell, obvious pseudofeces	bifurcated siphon, slimy thin feel, toothed shell	"furry" look to siphon, very shallow, heavy round shell

Figure 8. Quick reference for subtidal clam identification.

False geoducks *(Panomya* spp.) are generally smaller than geoducks, and have a distinctive siphon tip with a thin pink or red ring encircling each siphon hole. Even when this color is not apparent, *Panomya* siphon tips appear rounded in side profile, as opposed to geoduck siphon tips, which are box-like when viewed from the side. *Panomya* can also be distinguished by their thinner siphon membranes and because the incurrent siphon, when open and pumping, is noticeably larger than the excurrent siphon. *Panomya* have a barely visible inner ring of very fine tentacles on the siphon.

Mya truncata are usually much smaller than geoducks, and have a thin, dark-brown, wrinkled siphon with leathery flaps at the tip. Piddocks *(Zirfaea pilsbryii)* are easily distinguished from geoducks by their bifurcated (forked) siphons, maroon or dark red siphon tips, and a distinctive white and reddish brown mottled pattern on the siphons. Piddock siphons are also very thin-walled and have a slimy, smooth feel unlike the rubbery siphon covering of geoducks. Piddocks are boring clams, and are therefore found only in substrates such as clay and wood, although this may not be readily apparent if there is a thin surface layer of sand or mud. Cockles *(Clinocardium nuttallii)* are readily distinguished by their white, "furry" siphon tips; they can also be easily dug by hand to verify their identity, since they do not burrow deeply into the substrate The siphon of the horse mussel *(Modiolus rectus)* appears as one or two narrow slits, usually in muddy substrates. Because the shell lies immediately below the substrate, they are easy to verify by hand-digging.

Non-molluscans such as sea pens *(Ptilosarcus gurneyii)* can sometimes produce a geoduck-like "dimple" H hen they are retracted into the sand. When probed, they feel soft to the touch like a geoduck siphon. But because sea pens have no siphons, they cannot retract further into the substrate when probed by hand. When visible, sea pens are a distinctive bright orange color.

The field experience of surveyers is crucial when distinguishing geoduck shows and siphons. Nev WDFW surveyers gain such experience in part by making practice "surveys" with experienced biologists, and by positively verifying their siphon identifications with dig samples. When making transect counts, WDFW surveyers include only shows which can be readily identified as belonging to a geoduck.

Recording Data

At the start and finish of each transect, the divers record water depth (i.e., ambient depth uncorrected to MILL W) to the nearest ft using a digital depth gauge. At the end of the transect an assessment of the surface substrate composition is recorded. The substrate is assigned. one or a combination of the following categories: mud (<63 microns), sand (63 microns-2 mm), pea gravel (2-20 mm). and gravel (>20 mm). Particle sizes and the dominant substrate throughout the length of the 150-ft transect are judged subjectively by the surveyers, and are not quantitatively measured with traditional screening techniques. Cobble, boulders, logs, wood debris, and other features associated with the substrate (e.g., sandy hummocks) are also recorded when present. The presence of readily visible macro flora and fauna is also recorded, including eelgrass, major algal groups, major epibenthic animals, and fish. The boat operator, hovering above the divers' bubbles at the start of each transect, records DGPS latitude and longitude to the nearest thousandth of a minute. Starting time for each transect is also recorded, so that the uncorrected transect depth reported by the divers may be later corrected to MLLW with the use of a tide graph for the area. DGPS latitude and longitude are also recorded at the end of the final transect in any line of continuous transects.

Appendix 1 contains sample data sheets. Appendix 2 lists the codes used for recording substrate composition and associated plant and animal data.

Geoduck Dig (Weight) Samples

As noted earlier, cluster samples of geoducks (called dig samples) are taken systematically at every sixth transect previously surveyed for density. Transects where the density of geoducks falls below currently accepted commercial levels (i.e., <0.04 geoducks per ft^2) are eliminated from this selection process. The dig samples provide an estimate of mean weight per geoduck (Equation 4), as well as information on market quality, difficulty of digging, and substrate composition below the surface layer.

Using DGPS fixes and corrected depth data from the transect surveys, the boat is anchored near the middle of each systematically-selected digging transect. A single line-tended diver descends immediately below the boat and haphazardly digs the first ten visible geoducks. The diver also records information on the surface substrate composition, the water depth at which geoducks were dug, and a subjective evaluation of the ease or difficulty of digging. The boat crew records DGPS latitude and longitude of the digging location, the number and condition of geoducks dug, and the time taken to dig the samples. The geoducks taken at each transect are kept separately in moist burlap sacks labeled with the transect number, and are periodically soaked with seawater to keep them alive. Appendix 1 contains an example of the data recorded for a typical dig transect.

The geoduck samples are kept cool and moist in burlap sacks, transported to the Point Whitney Shellfish Laboratory, and either processed the same day or placed in running sea water for later processing. Processing occurs as soon as possible to avoid moralities which may result from injuries sustained during digging. Whole wet weight (grams) is measured after a drainage time of a few minutes to two hours. All geoducks are weighed, but damaged clams -- those with broken valves or tissues blown apart by the water jet -- are noted and eliminated from the calculation of mean weight. The greatest anterior-posterior length of the right valve is measured with calipers to the nearest mm. The right valve is the valve on the observer's right side when the clam is held with the siphon down and the umbo facing the observer (Figure 9). The siphon is then cut from the body (Figure 9) and weighed separately. Siphon weight information is valuable for commercial marketers, since the siphon is the portion of the geoduck which currently determines the market price in many cases. Overall geoduck quality, which is a function of gross appearance, color, and size, is then judged as either commercial or non-commercial. Appendix 1 shows an example of typical weight and quality data as recorded on the data sheet.

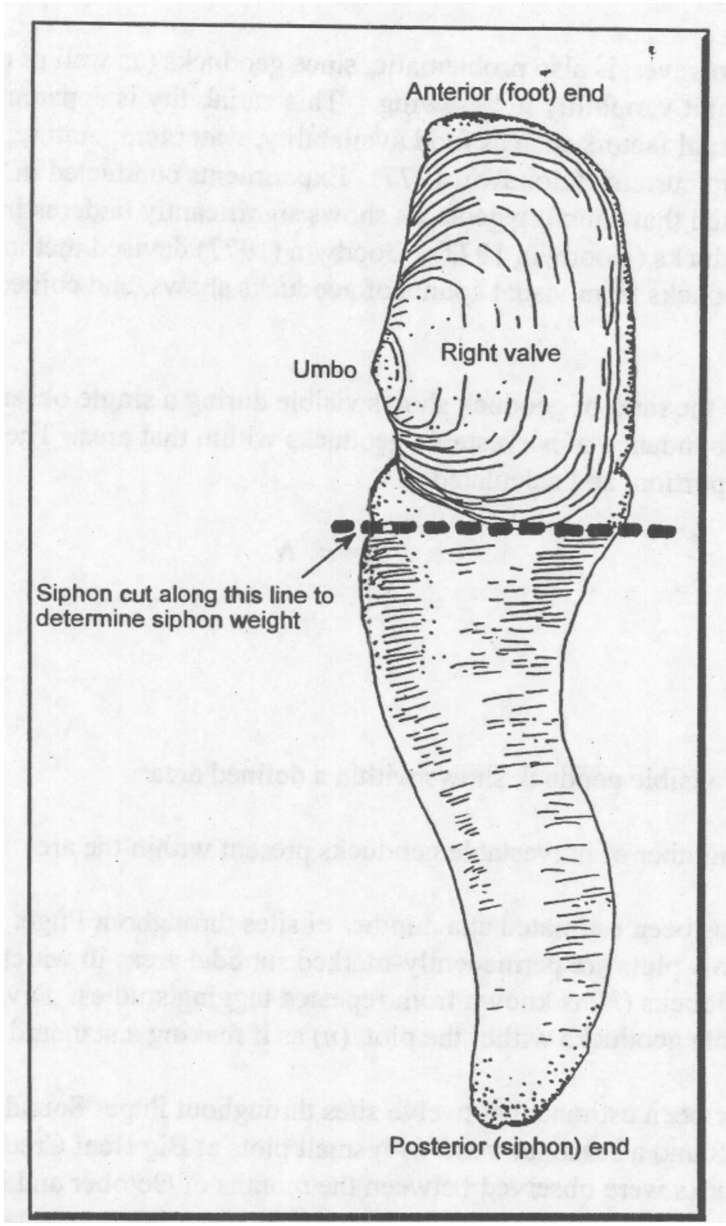


Figure 9. Geoduck clam.

Show Plot Surveys and Show Factors

A geoduck "show" is either a geoduck siphon visible above the substrate surface or a depression left in the substrate which can be identified as having been made by a geoduck siphon (Goodwin 1973). The only practical way to estimate geoduck density is to count such "shows" within measured transects. Digging numerous samples from the substrate, a method commonly employed to estimate the density of small intertidal clam populations, is not feasible for geoducks on a large spatial scale because they are buried too deeply in the substrate.

Counting shows, however, is also problematic, since geoducks (as well as other clam species, see Flowers 1973) exhibit variability in "showing." This variability is apparently a function of various environmental factors such as food availability, water temperature, substrate type, algae cover, turbidity, and currents (Goodwin 1977). Experiments conducted in Washington in the early 1970s indicated that counting geoduck shows significantly underestimated the true density of harvestable geoducks (Goodwin 1973). Goodwin (1977) devised methods of estimating the true density of geoducks from visual counts of geoducks shows, and coined the term "show factor."
The show factor is the ratio of geoduck shows visible during a single observation of any defined area and the true abundance of harvestable geoducks within that area. The show factor (S) is expressed as a proportion, and calculated as

$$S = n / N \tag{9}$$

where

S = show factor

n = the number of visible geoduck shows within a defined area

N = the absolute number of harvestable geoducks present within the area

This proportion has been estimated at a number of sites throughout Puget Sound with the use of "show plots." Show plots are permanently-marked subtidal areas in which the absolute number of harvestable geoducks (N) is known from repeated tagging studies. Divers then revisit the plot and count all visible geoducks within the plot *(n)* as if making a standard survey.

Show factors have been estimated at twelve sites throughout Puget Sound from 1984 to 1993. Goodwin (1977) found a seasonal trend with small plots at Big Beef Creek, Hood Canal, where zero or few geoducks were observed between the months of October and March. The average monthly geoduck show factor from the twelve sites in Puget Sound reached an average maximum for March of 0.7 3 (i.e., only 73% of the geoducks present would be expected to be observed during an instantaneous count by divers). There were small incremental declines each month to 0.54 in September and to 0.43 in October. Show factors also vary from year to year. For example. the average annual geoduck show factor for all show plots in 1986 was 0.51. In 1992, the average annual geoduck show factor for all show plots was 0.77. The Puget Sound average show factor for all show plots from 1984 to 1993 is 0.62.

Since establishing show plots for a tract is extremely time consuming, state and Tribal managers have agreed to use a show factor of 0.75 to estimate biomass for pre-fishing surveys on a given tract. In other words, we assume that 75% of the harvestable geoducks present are actually seen and counted during an instantaneous transect count. Using a standard show factor avoids the time and expense of establishing separate show plots for each tract being surveyed. A show factor of 0.75 is, for most tracts, conservative and will not lead to overestimation of geoduck biomass on a given tract. A show factor of 0.75 is used to estimate density on all tracts for pre-fishing surveys, unless there is a show plot established for a tract that will give site-specific data.

Some situations may arise in which surveyers may wish to establish a show plot despite the cost and time involved. Examples include: 1) Surveys carried out in habitats or depths for which no historical show plot data exist; 2) Surveys where risk-averse management policies dictate that a conservative (i.e., low) show factor be used (as, for instance, when geoduck densities below a certain threshold would permit developers to destroy a potentially commercial geoduck tract); 3) Surveys using non-standard methods (e.g., quadrat counts, transects more or less than three feet in width); and 4) Surveys carried out by inexperienced divers who wish to verify their counts. The following paragraphs describe the field methods used to establish and count a show plot.

Show plot surveys are carried out during the period March 1 - October 14, when geoducks are actively pumping water and the show factor is highest. Show plot counts made during the fall and winter are likely to underestimate the actual number of geoducks present within the plot because of the documented low show factor (Goodwin 1977), or else would require unreasonable effort and time to be certain that all geoducks within the plot had been detected.

Show plot sites are selected so that they are close to the tracts or areas being surveyed and to mimic, as closely as possible, the substrate and current conditions of the survey tract or area. Obviously, show plot sites must contain geoducks in roughly commercial densities. Show plots are usually situated along a depth contour which is midway between the depths being surveyed at the nearby tract or area. For example, most show plots for commercial geoduck surveys, which take place between the -18 ft MLLW and -70 ft contours, are situated at the - 40 ft MLLW depth. To avoid destruction of the show plot boundary markers, show plots are not sited in -areas where boats frequently anchor or where tidal currents sweep large amounts of algae along the bottom. Finally, show plots are not sited in areas with large populations of horse clams *(Tresus* spp.), which might confound the results.

Once a suitable site has been chosen, yellow polypropylene lines are staked on the bottom to delineate a standard 900 ft^2 geoduck transect, including a line down the center of the six-ft wide transect. In this way, two three-ft wide strips running for 150 ft are outlined with yellow line. Corners of the plot are staked with steel reinforcing bar, and the line is staked at intervals with smaller metal stakes to prevent the line from floating above the substrate.

Following placement of the plot boundary markers, divers begin "tagging" all geoducks which are showing within the plot. Two divers slowly swim the entire plot, each diver being responsible for his or her three-ft wide half of the plot. Geoduck shows are tagged by placing a sturdy wire stake -- usually 3/16 inch diameter and 12 inches in length -- next to the siphon. All such tags are oriented to either the left or right of the siphon to avoid confusion with other shows, and are carefully placed about 1.5 inches from the siphon to reduce the risk of injury to the animal. Tags are set roughly six inches into the substrate wherever possible. During tagging, divers situate themselves perpendicular to their half of the plot to prevent fins from dislodging tags that are already in place.

All geoduck shows are tagged in this manner throughout the show plot over a period of several days. Following each tagging session, divers record the total number of tags placed. Each successive tagging session requires fewer new tags, in the manner of classic "removal sampling" methods (Zippin 1956; Seber 1982). In this case, geoducks are "removed" by tagging, and we assume that the entire population within the plot has been censussed when, after several repeated tagging sessions, no new tags are required. This point is generally reached after about five days in most geoduck show plots, although tagging must continue as long as new shows are discovered during the previous tagging session. Several tagging sessions are sometimes done during a single day to speed up the "removal" process (i.e., to reach the point where repeated sessions encounter no new shows). However, repeated tagging sessions during the same day run the risk that at least some geoducks will not show because they have been disturbed by the divers, and therefore tagging should span a minimum of three days. To avoid bias of this sort, the final determination of complete "removal" is made on a day when no previous tagging sessions have occurred.

After repeated tagging sessions result in no new shows, divers carefully gather all tags from the plot and the total number of tags from both halves of the plot is assumed to represent N, the absolute number of harvestable geoducks within the plot.

Once N has been established, it is possible to estimate show factors by returning to the plot and counting the number of shows as if surveying a standard 900 ft^2 transect. Without disturbing geoducks, two divers locate the show plot and begin a routine transect survey, using the polypropylene line boundaries rather than the transect spool to delineate the transect. Each diver swims his or her half of the plot at a speed which is consistent with the swimming speed during normal transect surveys (roughly 4-5 minutes for a 150 X 6 ft transect), counting all shows. The total number of shows (n) is divided by N (known from the repeated tagging done previously) to produce the estimated instantaneous show factor (S) as in Equation 9.. Site specific show factors may be estimated in this way for successive days, weeks, or months; estimates after a year run the risk of bias due to changes in the geoduck population within the plot (N) due to recruitment or mortality. In estimating show factors on a daily basis, divers are rotated to reduce the chance of bias from an individual diver remembering the location of certain geoducks within the plot (Goodwin 1977).

Seasonal Considerations for Geoduck Surveys

State and Tribal managers have agreed that geoduck surveys will not made from October 15 through February 28, due to the low "show factor" of geoducks during the winter months (Goodwin 1977). Surveys made during this period of time would tend to produce highly unreliable density estimates; see the section *Show plot surveys and show factors* above.

Eelgrass Surveys

Eelgrass *(Zostera marina)* provides important :habitat for juvenile Dungeness crab, spawning herring, and other marine animals. The WDFW Habitat Division requires that geoduck harvest not occur within eelgrass beds. Prior to fishing, eelgrass associated with geoduck beds is surveyed and a two foot vertical buffer is established around occurrences of rooted eelgrass. On a tract where the slope is very slight, using this standard two-ft vertical buffer may needlessly exclude large portions of the commercial tract. Under these circumstances, a 180-ft horizontal buffer (seaward and deeper than the deepest eelgrass) may be used. Geoduck harvest is not allowed within these buffer zones. Thus, eelgrass surveys are an integral part of every prefishing geoduck survey, because eelgrass distribution determines the inshore or shallow boundary of the geoduck tract in many cases. This inshore boundary is required for a determination of total surface area, used in Equation 6 to estimate total geoduck biomass on the tract.

To determine whether the standard two foot buffer zone below eelgrass impinges on a commercial tract's inshore boundary (normally set at -18 ft MLLW), pre-fishing eelgrass surveys are conducted by divers swimming along the -16 ft MLLW contour. Occurrences and extent of eelgrass found deeper than -16 ft MLLW are noted using DGPS latitudes and longitudes. When eelgrass occurs deeper than -16 ft MLLW, divers characterize the occurrences, define the perimeter of eelgrass beds, and note the water depth at the deepest occurrence of eelgrass for that site. Normally a two foot vertical buffer along the entire length of the tract is set below the deepest occurrence of any rooted eelgrass found along the tract. Alternatively, a buffer zone of at least 180 ft around eelgrass beds deeper than -18 ft MLLW can' be used when the tract is marked to exclude eelgrass and the marking is visible to (livers within the tract.

Labor Costs of Geoduck Surveys

Table 3 shows the field time spent surveying geoducks at four recently-surveyed tracts, and provides a rough planning guide. Survey time includes not only running transects and digging geoduck samples, but also includes boat transit to and from the tract, boat maintenance, eelgrass surveys, and the placement and mapping of buoys which mark the sample grid lines. Laboratory time (weighing geoduck samples) and the time required for data entry and analysis, however, are not included here.

Table 3. Time budget (in person-hours) for geoduck field surveys at four commercial tracts.

Tract	**Size** (acres)	**Transects** I **Acre**	**Transect Survey Time** (hrs)	Dig Sample Time (hrs)	**Tract Mapping Time (hrs)**	**Eelgrass Survey Time (hrs)**	**Boat Maintenance Time (hrs)**	**Transit Time** (hrs)	Total **Time (hrs)**	**Hours** I Acre
Agate Pass	945	0.34	404	128	52	64	48	12	708	0.75
Jamestown	300	0.36	164	24	4	64	20	12	288	0.96
Olele Pt	160	0.59	174	56	28	48	4	12	322	2.01
Pt Robinson East	22	1.18	44	12	4	0**	0	12	72	3.27

* The data for this tract represent an initial survey area.
** Eelgrass surveys were not performed at this tract because the inshore tract boundary for non-Indian divers was roughly -35 ft MLLW throughout the tract, well below the deepest occurrence of eelgrass in Puget Sound.

As shown in Table 3, transect surveys consume most of the total geoduck survey time. Transect surveys required between 54 - 61 % of the total survey time at the four tracts. Note that the "transect survey time" in Table 3 includes not only actual diver bottom time, but also include all hours worked by the non-diving team aboard the boat, time spent during surface intervals, time spent suiting up, recording data, and other miscellaneous "diving" tasks which do not actually occur underwater.

Table 3 also suggests that as tract size decreases, the survey time required per unit of surface area increases. This occurs primarily because small tracts require more transects per acre to reach the statistical precision requirements (see *Sample size* above).

Surveys are usually conducted by four divers. A team of two divers begins the day by running transects until their no-decompression bottom time is expended. Meanwhile, the two remaining divers operate the boat, keep track of the divers, and record position and time data for the transects. The second team continues transect surveys while the first team completes a surface interval. Following the surface interval and the ascent of the second team, the first team typically re-enters the water to continue transects until their bottom time is expended. Digging typically requires one diver who actually digs the geoduck samples and at least two crewpersons who operate the boat, water pump, safety line, and record data.

Bottom times for WDFW divers must comply with the US Navy Tables, and each ascent must include a mandatory three to five-minute safety stop at -15 ft. Therefore, divers who utilize computers or who do not make recommended safety stops would obviously require less time to complete transect surveys and dig geoduck samples than WDFW divers.

Environmental Assessment

Geoduck beds which prove to have commercial concentrations of geoducks are then further studied. Inquiries are made to various agencies and groups to obtain additional ecological information, and to learn of possible interaction between geoduck fishing and other uses of the areas.

Washington Department of Ecology is contacted for water quality information. Divisions within WDFW and local Tribes are contacted to learn of sensitive habitats, important resources, or activities that may be affected by geoduck fishing. The county in which the proposed fishing will occur is contacted to learn of the shoreline designation of areas adjacent to geoduck beds. After receiving comments from all of the groups contacted, an environmental assessment is written by WDFW for each proposed geoduck fishing location.

The environmental assessment describes the size and location of the proposed tracts. Tract substrates and water quality are summarized, as well as the geoduck abundance, size, and quality. Other biota including fish, invertebrates, aquatic plants, marine mammals, and birds are discussed. The last part of the assessment covers activities including fishing, navigation (boat traffic), and other uses.

DNR then writes an adoption notice and notifies shoreline owners and other members of the public of the planned fishery.

Part II Equilibrium Yield Modeling

Methods

Data

Over 2,000 geoducks were sampled between 1979 and 1981 at 15 previously unharvested sites in Puget Sound and the Strait of Juan de Fuca to obtain information on age distribution and growth (Figure 10). The sites span four of the current six geoduck management regions, with six sites in the Hood Canal region, two sites in the Central Sound region, one site in the Strait region, and two sites in the South Sound region. Samples were taken randomly within each site at depths of -30 to -60 ft MLLW by washing geoducks from the substrate with a commercial water jet. Age was determined from annual growth increments in the hinge plate using the acetate-peel method (Shaul and Goodwin 1982). The von Bertalanffy growth parameters (L , k, t_0) were estimated for each of 234 sub-sampled geoducks with a nonlinear regression method. A two-factor ANOVA was used to test if growth parameters differed within or between management regions. Hoffmann *et al.*. (1999) provide a detailed description of the growth analysis.

Equilibrium Yield Model

Geoduck yield was modeled using a deterministic, age-structured equilibrium yield model. Given a set of parameter estimates for mortality, maturity, growth, and selectivity, the model collapses the number of geoducks at age for all cohorts in the population to a single cohort, assumed to represent the stable age distribution of the population. Population size was based on an initial unfished spawning population, a declining exponential function for survival at age, and by the Baranov catch equation. The model assumed continuous recruitment, the magnitude of which was based on a Beverton-Holt stock-recruitment relationship. Fishing mortality (F) was stepped from zero to a specified upper limit while computing yield per recruit (YPR) and spawning biomass per recruit (SPR) for each value of F. The model was constructed as a QuattroPro for Windows (Version 5.0) spreadsheet.

The model required the following user supplied inputs:

1. An instantaneous rate of natural mortality (M)

2. A shape parameter value for the Beverton-Holt S-R relationship (A)

3. The unfished ("virgin") spawning biomass ($B0_s$) in kg (only required to scale absolute biomass)

4. The fishery selectivity coefficient at age (v)

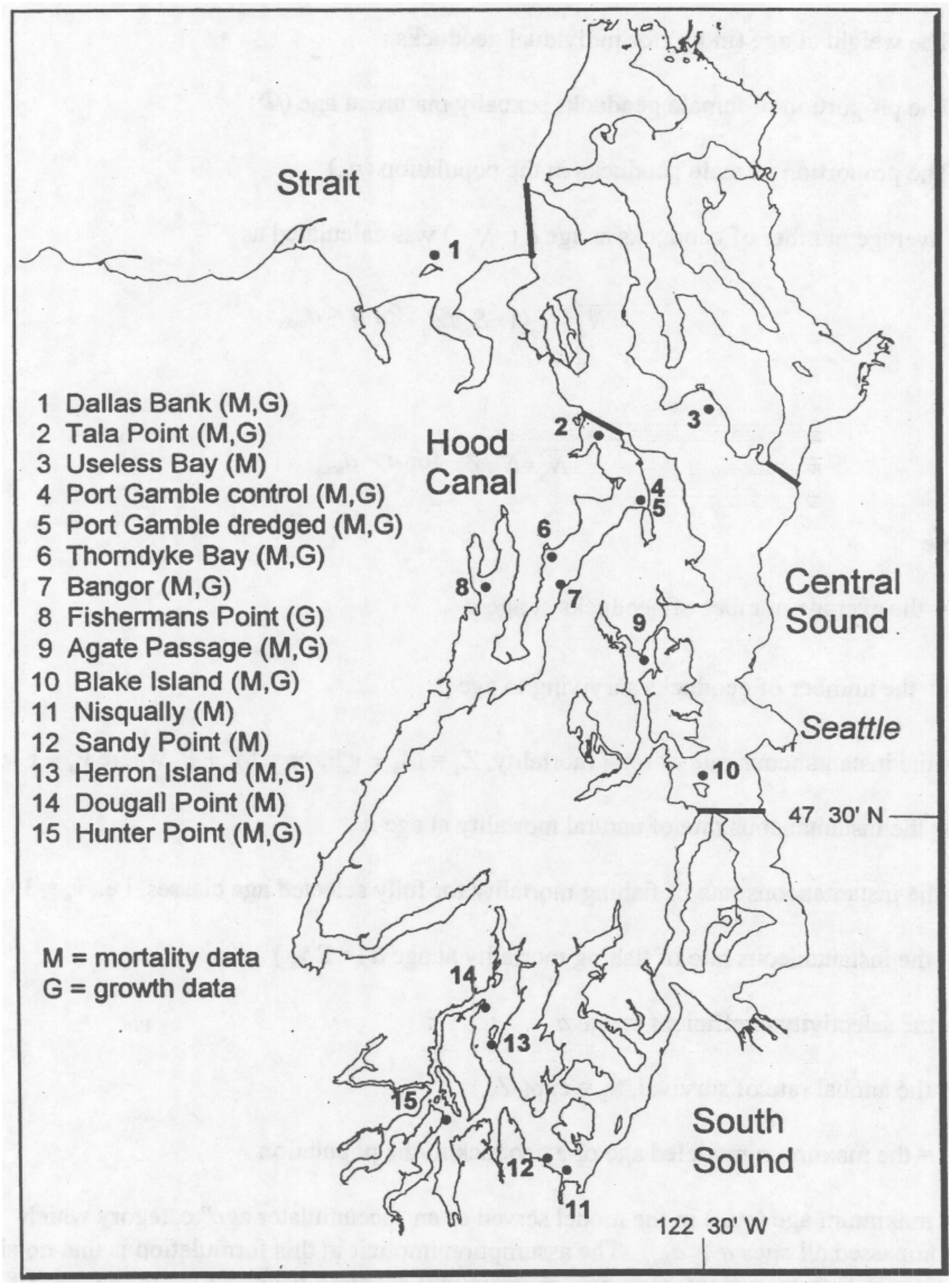


Figure 10. Sampling sites for geoduck natural mortality and growth.

5. The weight at age (in kg) for individual geoducks

6. The proportion of female geoducks sexually mature at age (Φ)

7. The proportion of male geoducks in the population (p_m)

The average number of geoducks at age *a* ($\bar{N}_a$) was calculated as

$$\bar{N}_a = N_a (1-S_a)/Z_a \text{ for } a < {}_{max} \qquad (1)$$

and

$$\bar{N}_a = N_a/Z_a \text{ for } a = {}_{max} \qquad (2)$$

where

$\bar{N}_a$ = the average number of geoducks at age *a*

N_a = the number of geoducks surviving to age *a*

Z_a = the instantaneous rate of total mortality, $Z_a = M_a + v_a F$, or $= M_a + F_a$ where $F_a = F\ v_a$

M_a = the instantaneous rate of natural mortality at age *a*

F = the instantaneous rate of fishing mortality for fully selected age classes, i.e., $v_a = 1$

F_a = the instantaneous rate of fishing mortality at age a ($= F\ v_a$)

v_a = the selectivity coefficient at age *a*

S_a = the annual rate of survival, $S_a = \exp(-Z_a)$

a_{max} = the maximum modeled age of a geoduck in the population

The maximum age (a_{max}) in the model served as an "accumulater age" category which encompassed all ages $a \geq a_{max}$. The assumption implicit in this formulation is that no significant changes in growth, weight, maturity, or selectivity occurred beyond a_{max}. In the case of geoducks, this assumption was reasonable and is addressed below. For other applications, the model spreadsheet could be simply extended to accommodate an unlimited number of older age classes.

For the first age class (a = 1), the number of geoducks surviving to age *a* (N_a) was calculated as

$$N_a = p_m \text{ for males} \tag{3}$$

and

$$N_a = 1\text{-}P_m \text{ for females,} \tag{4}$$

where p_m was the proportion of males in the population.

For $a > 1$, the number of geoducks surviving to age a (N_a) was calculated as

$$N_a = N_{a-1}\ S_{a-1} \tag{5}$$

The average biomass (in kg) of geoducks at age a ($\overline{B}_a$) was calculated as

$$\overline{B}_a = \overline{N}_a\ w_a \tag{6}$$

where

w_a = the weight (in kg) of an individual geoduck at age a

Weight at age a was calculated from an allometric length-weight relationship of the form $w_a = xL_a^y$, where L_a = shell length (in cm) at age a, and x and y were constants. Length at age was based on the von Bertalanffy growth equation:

$$L_a = L_\infty\ [1\text{- } \exp^{-k(a-t_o)}] \tag{7}$$

where L_a = shell length of a geoduck at age a , and L_∞, k, and t_o were estimated parameters.

Yield per recruit (in kg) at age a (YPR_a) was calculated as:

$$YPR_a = \overline{B}_a\ (F\ v_a) = F\ \overline{B}_a\ v_a \tag{8}$$

Total yield per recruit (in kg) for all ages (YPR) was calculated as:

$$YPR = \sum_a \overline{B}_a\ (F\ v_a) = F \sum_a B_a\ v_a \tag{9}$$

Spawning weight per recruit (in kg) at age a (SPR_a) was calculated for females only as:

$$\text{SPR}_a = \bar{B}_a \Phi_a \qquad (10)$$

Total spawning weight per recruit (in kg) for all ages (SPR) was calculated as:

$$\text{SPR} = \sum_a \bar{B}_a \Phi_a \qquad (11)$$

The fraction of the unfished spawning stock biomass remaining at a given level of fishing mortality (*P*) was a parameter of the Beverton-Holt spawner-recruit relationship, such that

$$P = 1 - (1/A)\,(1\text{-SPR / SPR0}) \qquad (12)$$

where

A = the shape or "steepness" parameter of the Beverton-Holt spawner-recruit function, a user-supplied input ($0 \leq A \leq 1$)

SPR = total spawning weight per recruit (in kg) from equation 11 above

SPR0 = total spawning weight per recruit (in kg) from equation 11 above when F = 0 (i.e., unfished spawning weight per recruit)

Spawning biomass (B_s) in kg when F > 0 was calculated as:

$$B_s = P\,B0_s \qquad (13)$$

where

P = the parameter in the Beverton-Holt S-R function which represents the fraction of the unfished spawning stock remaining at a given level of fishing mortality (see equation 12 above)

$B0_s$= unfished spawning biomass in kg, a user-supplied input

Recruitment to the fishery (R) in numbers was calculated using the re-parameterized form (Kimura 1988) of the Beverton-Holt spawner-recruit relationship, such that

$$R = (B_s / \text{SPR0}) / [1 - A\,(1 - P)] \qquad (14)$$

where

B_s = spawning stock biomass in kg when F > 0 (equation 13 above)

SPRO unfished spawning weight per recruit in kg (i.e., when F = 0)

and *A* and *P* were parameters of the Beverton-Holt spawner-recruit function as described above.

Yield (Y) in kg was calculated as the product of total yield per recruit (in kg) and the number of recruits:

$$Y = YPR\,(R) \tag{15}$$

The model is capable of returning a suite of fishing mortality benchmarks, such as F_{n}. , $F_{O.1}$, and $F_{xx\%}$. For example, the fishing mortality rate which produces, over the long run, the maximum yield per recruit corresponds to the F_{max} strategy, whereas $F_{0.1}$ represents a rate of harvest less than F_{max} (Deriso 1987, Gulland 1968).

The fraction of the unfished spawning weight per recruit remaining at a given level of fishing mortality was calculated as SPR/SPRO, and is achieved at a corresponding fishing mortality rate $F_{xx\%}$ where *xx* represents the ratio (SPR/SPRO)100 . Model predictions of this fraction formed the basis for SPR-based fishing strategies. For example, the fishing mortality rate which resulted in a value of SPR/SPRO = 0.35 corresponds to the $F_{35\%}$ strategy.

The harvest rate (μ) for fully selected age classes (i.e., when $v_a = 1$) when fishing and natural mortality operate concurrently (Ricker 1975) was calculated as:

$$\mu = F/Z\,[\,1-\exp(-Z)] \tag{16}$$

Parameter Estimates

Parameter estimates used in the equilibrium yield model are shown in Tables 4 and 5. The derivation of these parameter estimates is described below.

Table 4. Geoduck life history parameters held constant for all study sites.

Category	Parameter	Value
Spawning stock biomass when F = 0	B0	100 000 kg
Instantaneous natural mortality rate	M	0.0226
Length-weight relationship	x	0.349127
	y	2.972807
Maturity (simple logistic)	x	-1.9
	y	9.5
Fishery selectivity (simple logistic)	x	-1.5
	y	8.0
Beverton-Holt shape parameter (Eq. 14)	A	1
Proportion of males in population	P_m	0.5
Maximum (accumulator) age	a_{max}	25

Natural Mortality

The instantaneous rate of natural mortality (M) was estimated from the geoduck age-frequency distribution at 14 of the 15 sample sites (Figure 10) using two different catch curve models (Robson and Chapman 1961; Ricker 1975). Both models assume that mortality is constant for all ages used in the catch curve. The Robson and Chapman model is based on a geometric distribution and assumes that year class survival and recruitment are constant and all ages are equally selected. Geoducks are extremely long-lived, so that the number of animals observed in each one-year age class is typically low, even for sample sizes in which n > 1,000. Despite this problem, we chose to preserve the data in one-year age classes rather than aggregating ages, a procedure which potentially ignores real variability in the original data and may slightly inflate estimates of M (Noakes 1992). It was not possible to estimate site-by-site mortality rates using catch curves. because no individual site contained enough data to construct reliable catch curves. Age frequencies were therefore pooled from all 14 sites in order to create the catch curve.

To avoid arbitrary choices of the upper and lower ages used in the catch curve "right limb," we established a protocol for data inclusion: The initial upper age limit for the catch curve was the first age at which our sample contained no geoducks (i.e., the first gap in frequency). We then excluded younger age frequencies if they were identified as outliers by Weisberg's (1985) outlier test. Two methods were used to select the lower age limit for the catch curve: 1) The chi-square procedure described in Robson and Chapman (1961) was used to differentiate partially selected ages: and 2) Catch curve regressions were calculated for all possible lower age limits, and we used an *ad hoc* procedure to optimize the coefficient of determination (r^2) and the linearity of positive and negative residuals plotted against age. Once the lower and upper age limits for the catch curve were identified, a chi-square formula was then used to test goodness of fit of fully--selected ages to a geometric distribution (i.e., the Robson and Chapman model).

Sampled geoducks from the 14 previously-unfished sites ranged in age from 2 to 131 years (Figure II A) The mean age of geoducks was 46 years (SE = 0.56, n = 2,157). The initial upper age limit for the catch curve was 110 years, because no 111-year old geoducks were in our sample Examination of residuals showed a single large negative residual at the 99-year age class (only one geoduck of this age was in our sample), and this age class was eliminated from the analysis as an outlier. based on the test given in Weisberg (1985). Both the Robson and Chapman (1961) chi-square procedure and our *ad hoc* optimization procedure identified age 28 as the lower age limit for the catch curve. A chi-square was used to test goodness of fit of fully-selected ages (28-98) to a geometric distribution. The resulting chi-square was highly significant ($\chi 2$ = 326.56. df = 68), indicating that the age frequency was not geometric in distribution, and that data requirements for the Robson and Chapman model were not met. Ricker (1975) pointed out that in most stocks, difference in year class strength is the major source of variability, in which case the best estimate of survival would be obtained from a catch curve analysis with equal weighting. The Ricker catch curve based on ages 28 - 98 (Figure 11B) produced an estimate of M= 0.0226 y^{-1} (± 0.0018 SE, n = 71, r^2 = 0.70).

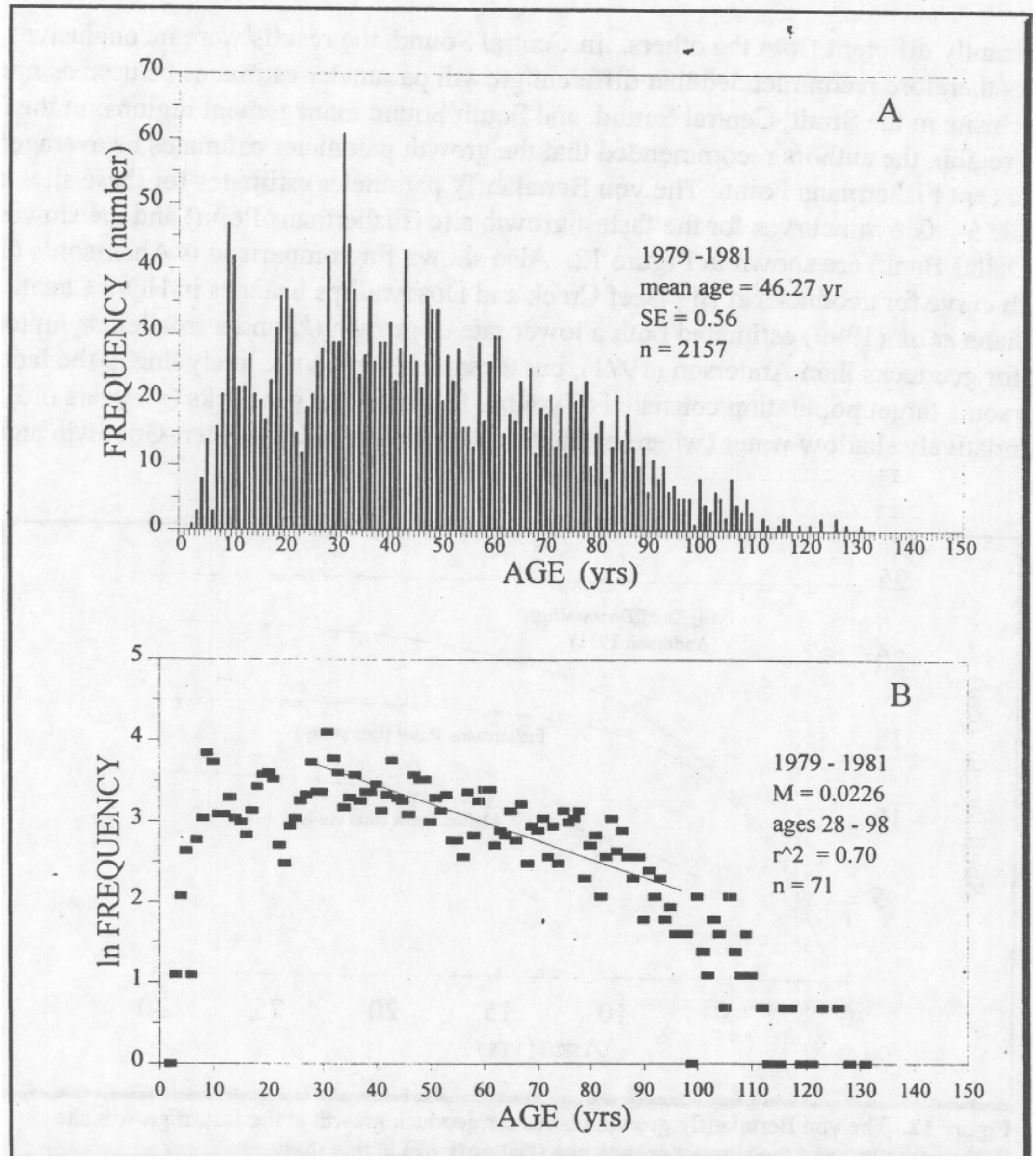


Figure 11. (A) Age frequency geoducks sampled at 14 sites in Washington. (B) Catch curve used to estimate the instantaneous natural mortality rate (M) for geoducks.

Growth

Of the three *von* Bertalanffy growth parameters, only one significantly influenced model-derived target fishing mortality rates: the growth constant *k* (Hoffmann *et al.* 1999). Statistically significant differences in *k* were detected among most of the sites within 3 management regions: Central Sound, Hood Canal, and South Sound. Further testing showed that in South Sound, the sites were also significantly different. In Hood Canal, only one site (Fishermans Point) was significantly different from the others. In Central Sound, the results were inconclusive. The authors therefore recommended that different growth parameter estimates be used as model input for each site in the Strait, Central Sound, and South Sound management regions; in the Hood Canal region, the authors recommended that the growth parameter estimates be averaged for all sites except Fishermans Point. The von Bertalanffy parameter estimates for these sites are shown in Table 5 . Growth curves for the fastest growth site (Fishermans Point) and the slowest growth site (Dallas Bank) are shown in Figure 12. Also shown for comparison is Anderson's (1971) growth curve for geoducks at Big Beef Creek and Dosewallips beaches in Hood Canal. Hoffmann *et al.* (1999) estimated both a lower rate of growth *(k)* and a smaller asymptotic size (L) for geoducks than Anderson (1971), but these differences are likely due to the fact that Anderson's target population consisted of young, fast-growing geoducks (<5 years old) sampled from relatively shallow water (where mean geoduck shell length is larger; Goodwin and Pease 1991).

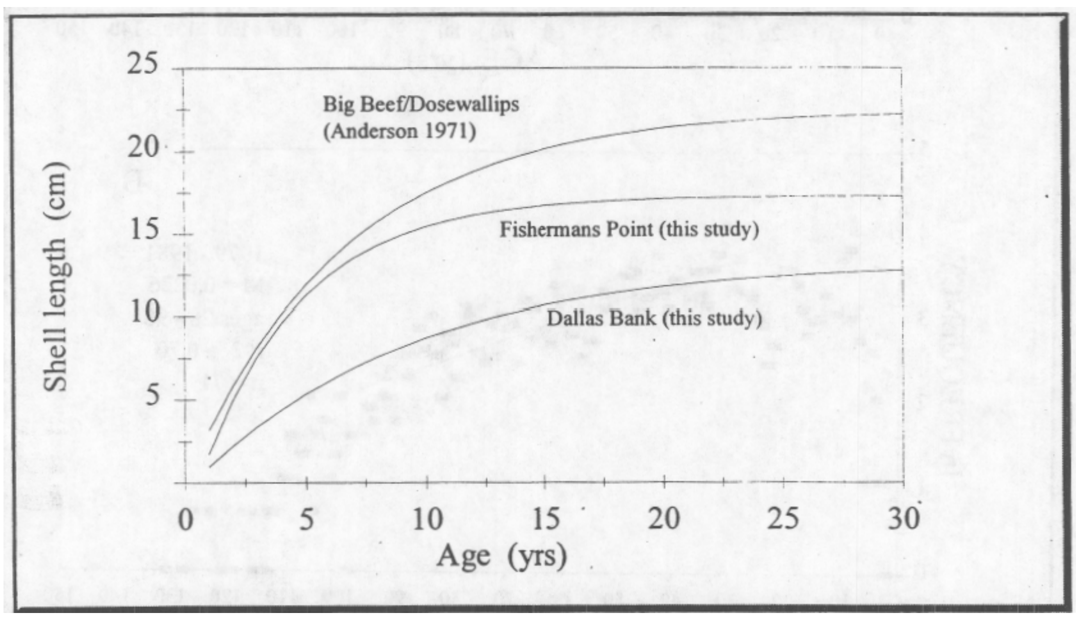


Figure 12. The von Bertalanffy growth curves for geoduck growth at the fastest growth site (Fishermans Pt.) and the slowest growth site (Dallas Bank) in this study.

Length-weight Relationship

Goodwin (1976) calculated an allometric length-weight relationship for Washington geoducks in log-log form. We converted this to the more familiar power curve form $w_a = xLQ$, where w_a = weight (in g) at age *a,* L_a= shell length (in cm) at age *a* (Table 4).

Sex Ratio

The proportion of males (p_m) in the geoduck population was set to $_{pm}$ = 0.5 based on a 50:50 sex ratio for geoducks older than 10 years (Goodwin and Pease 1989).

Maturity

The proportion of sexually mature geoducks at age was estimated by fitting a simple logistic curve to maturity data from published sources. Anderson (1971) found that 50% of his sample of geoducks was mature at 75 mm and an age that he estimated to be 3 years. The Washington growth curves described above suggest that this length would be attained in roughly 5 years, depending on the site. Sloan and Robinson (1984) reported that geoducks mature at 5 years and reproduce for at least a 100-year period with no "reproductive senility." They stated that "unequivocally mature geoducks" were 6 to 103 years old (late-active males) and 12 to 95 years old (late-active females). Based on these two sources, we fit a logistic curve with the least squares method and two data points, whereby 50% of the female geoducks would mature at 5 years and 100% by 12 years. The proportion of mature geoducks (Φ) at age a is described by

$$\Phi_a = l/(1 + \exp^{xa+y}) \qquad (17)$$

where a is age in years, x = -1.9, and y = 9.5.

Fishery Selectivity

Fishery selectivity at age was based loosely on Harbo et al. (1983), who reported that recruitment to the British Columbia geoduck fishery begins at 4 years and is complete by 12 years. We fit a simple logistic curve using the least squares method and two data points, assuming geoducks enter the fishery at roughly 4 years and, to more conservatively model fishery selectivity, assume that geoducks are fully selected by 8 years.

$$V_a = l/(1 + \exp^{xa+y}) \qquad (18)$$

where v_a is the proportion of geoducks of age a selected by the fishery, a is age in years, x = -1.5, and y = 8.

Stock-recruit Relationship

Nothing is known about the form or steepness of the stock-recruit (S-R) .relationship for geoducks. We therefore set the Beverton-Holt shape parameter (A) equal to 1.0 for all model runs. In other words, we assumed that recruitment was independent of spawning stock abundance. This assumption is reviewed below in *Discussion*.

Maximum Age

As a practical convenience, the equilibrium yield model uses an "accumulator age" category (a_{max}) as the final age category, encompassing all ages a ≥ a_{max} . For this study, we set a_{max}= 25, which implicitly assumes that there are no significant changes in growth, selectivity, or maturity beyond age 24. This assumption is reasonable for geoducks, which reach asymptotic size between the ages of 10-20 years (Hoffnann et al. 1999).

Results

Fishing Mortality Rates for Five Harvest Strategies

We ran the model for each site, varying only the growth parameters based on the analysis of growth presented in Hoffmann *et al.* (1999). The only sites where growth parameter estimates (specifically, the growth constant k) could be pooled were five of the six Hood Canal sites. In all other cases, site-specific growth parameters could not be pooled, and therefore separate model outputs were calculated for each site. All inputs except growth parameters were identical for each model run (Table 1). Growth parameters used as site-specific input are shown in Table 5.

Table 5. Bench mark instantaneous fishing mortality rates for fully-selected geoducks (v_a = 1.0) from seven sites in Washington. Model inputs except growth parameters are from Table 4. Growth parameter estimates are from Hoffmann *et al.* (1999).

Region	Site	n (sites)	L_∞ (cm)	k	t_o	F_{max}	$F_{o.1}$	$F_{35\%}$	$F_{40\%}$	$F_{50\%}$
South Sound	Hunter Point	1	16.4	0.23	0.72	0.090	0.036	0.036	0.029	0.020
	Herron Island	1	13.2	0.15	0.42	0.064	0.031	0.032	0.027	0.018
Central Sound	Agate Passage	1	15.8	0.20	0.18	0.085	0.035	0.035	0.029	0.020
	Blake Island	1	14.6	0.16	0.81	0.064	0.031	0.032	0.027	0.019
Hood Canal	Five sites pooled	5	12.8	0.16	0.47	0.067	0.032	0.033	0.027	0.019
	Fishermans Point	1	16.8	0.24	0.55	0.100	0.037	0.036	0.030	0.020
Strait	Dallas Bank	1	12.0	0.11	0.33	0.053	0.028	0.030	0.025	0.018

Values of the instantaneous fishing mortality rate (*F*) for five commonly-used constant harvest rate strategies are shown in Table 5. F_{max} is the fishing mortality rate that produces, over the long run, the maximum yield per recruit (YPR). $F_{0.1}$ is a common alternative to F_{max}, and is the rate of fishing mortality at which the marginal YPR is 10% of the marginal YPR for a lightly exploited fishery (Deriso 1987). $F_{35\%}$, $F_{40\%}$, and $F_{50\%}$ are spawning biomass per recruit (SPR) based harvest rates which reduce SPR to either 35%, 40% or 50% of the unfished level (Clark 1991).

F_{max} ranged from 0.053 to 0.100 depending on the site (Table 5). These rates correspond to annual harvest rates (μ) of 5.1 - 9.4% of the exploitable geoduck biomass. The Strait of Juan de FICA region, represented by the single sampling site at Dallas Bank, produced the lowest value, while Fishermans Point in Hood Canal produced the highest value. The F_{max} strategy reduced SPR to 15-21% of the unfished level, depending on the site. Values for $F_{0.1}$ ranged from 0.028 to 0.037, corresponding to annual harvest rates of 2.7 - 3.6%. This strategy reduced SPR to 3537% of the unfished level, depending on the site.

Values for $F_{3S\%}$ were, predictably, nearly identical to the $F_{0.1}$ rates, ranging from 0.30 - 0.36 (μ = 2.9 - 3.5%). F values for the $F_{40\%}$ strategy ranged from 0.025 - 0.030 (μ = 2.4 - 2.8%). The mean F value for the $F_{40\%}$ strategy was 0.028, corresponding to μ = 2.7%. F values for the $F_{50\%}$ strategy ranged from 0.018 - 0.020 (μ = 1.8 - 2.0%).

Model Sensitivity to Parameter Estimates

All the parameter estimates used to drive the model are subject to varying degrees of uncertainty. It is therefore reasonable to ask what might happen to our predictions if the true values of *M* or *k,* for example, were much lower or higher than our estimates. We tested the sensitivity of the model by running it with a range of values for each parameter in turn, while holding all other parameters constant. Values ranging from one-tenth the "best" parameter estimate (from Tables 4 and 5) to three times the estimated value were used in the analysis. Only the fishing mortality rates corresponding to the $F_{40\%}$ strategy were calculated, but the trend for other strategies would be similar.

The model was most sensitive to the estimate of *M*, with $F_{40\%}$ values ranging from 0.003 to 0.068 as *M* was increased from one tenth to three times our "best" estimate of M= 0.0226 (Figure 13). The model was far less sensitive to the other parameter estimates, as evidenced by the relatively flat $F_{40\%}$ trajectories for values of the growth coefficient *k,* the selectivity constant *y,* and the maturity constant *y.* For example, varying the value of *k* from one-tenth to three times our "best" estimate resulted in $F_{40\%}$ values which ranged only from 0.021 to 0.033.

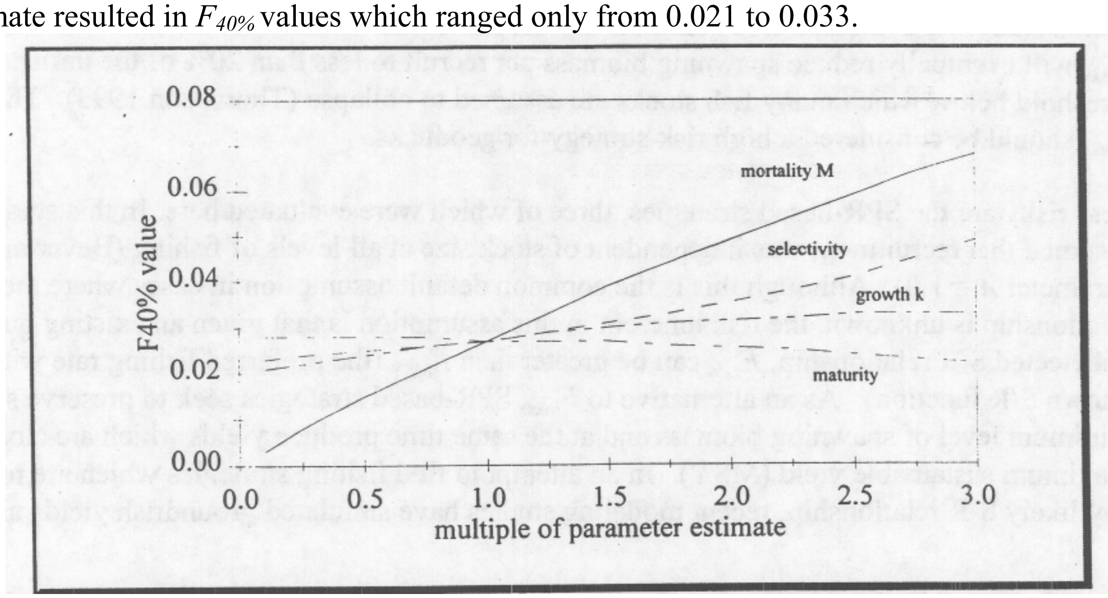


Figure 13. The effect of different parameter estimates on model-derived F $_{40\%}$ values

Discussion

Our primary objective in equilibrium modeling was to simulate the long-term results of various geoduck fishing strategies, both in terms of yield and spawning biomass per recruit. Before discussing our results, it is perhaps necessary to explain why we attach such importance to geoduck harvest rate strategies, particularly since the differences between many of the modeled options may appear trivial.

In many fisheries, especially those in which biomass is small or estimated with great uncertainty, debating a 1 % difference between annual harvest rate options would indeed be trivial. But in Washington's geoduck fishery, where the exploitable biomass is large (73,843 t in 1999; Sizemore and Ulrich 1999) and the price is high, even tiny incremental differences in the recommended harvest rate have tremendous economic significance. Moreover, because geoducks have a low M (and presumably a low intrinsic rate of increase), small differences in annual harvest rates can have profound cumulative effects on stock size, especially if the harvest rate is set too high. This is not to discount the importance of good biomass estimates, but we believe there are several reasons why Washington managers should place the greatest emphasis on improved harvest rate strategies rather than improved biomass estimates. First, biomass estimates for individual geoduck beds in Washington have coefficients of variation (CV) averaging about 20%. Simulation tests suggest that biomass estimation errors of this magnitude are unlikely to result in substantial degradation of long-term harvest performance (Frederick and Peterman 1995). Second, even greatly increased sampling is not likely to improve biomass estimate CVs very much. Third and most importantly, errors in biomass estimation are assumed to be reasonably unbiased. An error in setting the annual harvest rate, on the other hand, will have a persistent and cumulative effect on stocks in only one direction, either underharvest or overharvest. We therefore believe that, given reasonable estimates of stock size, choosing a harvest strategy remains the most critical aspect of geoduck management.

In this study we evaluated five common harvest strategies. Our model predicts that fishing at F_{max} will continually reduce spawning biomass per recruit to less than 20% of the unfished level, a threshold below which many fish stocks are assumed to collapse (Thompson 1993). Therefore, F, should be considered a high risk strategy for geoducks.

Less risky, arc the SPR-based strategies, three of which were evaluated here. In this study, we assumed that recruitment was independent of stock size at all levels of fishing (Beverton-Holt parameter A = 1 .0). Although this is the common default assumption in cases where the S-R relationship is unknown, the risk inherent in this assumption is that given an existing but undetected S/R relationship, $F_{xx\%}$ can be greater than F_{MSY} (the preferred fishing rate with a known S/R function). As an alternative to F_{max}, SPR-based strategies seek to preserve some minimum level of spawning biomass and at the same time produce yields which are close to the maximum sustainable yield (MSY). In an attempt to find fishing strategies which are robust for any likely S-R relationship, recent modeling studies have simulated groundfish yields using a range of typical life history parameters and realistic S-R models. Clark (1991) showed that fishing at $F_{35\%}$. would achieve at least 75% of MSY for a wide range of deterministic S-R relationships. On the basis of his results, $F_{35\%}$ has been adopted as a target rate for a number of fish stocks in Alaska and the U.S. Pacific coast. Clark (1993) later revised his recommendation to $F_{40\%}$ after considering variability in recruitment, but remarked that "it would be silly to argue very hard for or against any specific rate between $F_{35\%}$ and $F_{45\%}$." Mace (1994) also recommended $F_{40\%}$, which she claimed was a modest improvement over $F_{35\%}$. She states that $F_{40\%}$ represents a risk-averse fishing strategy in the common situation where there is adequate information to place bounds on all relevant life history parameters except the S-R relationship. Quinn and Szarzi (1993) modeled clam fisheries in Alaska and recommended SPR-based strategies equivalent to a range of $F_{30\%}$ to $F_{45\%}$.

On the basis of the results presented here, state and Tribal geoduck managers formally agreed on December 5, 1997 to an $F_{40\%}$, strategy for geoducks, applying an instantaneous fishing mortality rate of F= 0.028 ; the corresponding annual harvest rate for fully selected age classes ($\boldsymbol{\mu}$) is 0.027, or 2.7% of the exploitable biomass *(Appendix A* to state/Tribal geoduck agreements). Annual fishing quotas within each of the six management regions are calculated as the product of this harvest rate and the estimated exploitable biomass within the region (available from dive survey data). British Columbia managers calculate annual quotas using a fixed harvest rate of 1 (Campbell *et al.* 1998), but until recently this rate was applied to the estimated *virgin* biomass rather than current biomass estimates as is done in Washington.

Suggestions for Future Research

A secondary objective of our study was to determine which of the estimated geoduck life history parameters were most influential in predictions of yield and spawning biomass per recruit. The model was most sensitive to the estimate of natural mortality (M), while growth, selectivity, and maturity parameters had relatively little effect on SPR-based fishing mortality rates. This suggests that future research monies are best spent making more reliable estimates of M.

Our estimate of M= 0.0226 is similar to estimates from British Columbia. Sloan and Robinson (1984) estimated M= 0.035 at a single site, while Breen and Shields (1983) reported M= *0.01* to 0.04 in five populations. Noakes (1992) estimated M= 0.03 to 0.04 at three sites. Both our estimate and the British Columbia estimates relied on the catch curve method, which assumes that mortality rate is uniform with age and that recruitment has been constant over the range of age-groups analyzed. There is some suggestion in our age-frequency data that a shift in geoduck recruitment has occurred which could have biased the estimate of M. Age frequencies did not begin to decline until about age 25, a pattern in catch curves which is often due to inefficient sampling of younger age classes. But for geoducks, which grow quickly and are fully selected by the commercial fishery at half this age (Harbo *et al.* 1983), sampling inefficiency is not a plausible explanation for the low numbers of geoducks in the 10-25 year age group. Instead, low numbers of 10 - 25 year-old geoducks may indicate poor recruitment during the 15-year period prior to sampling. This suggests that recruitment declined during the period 1955-1970 (prior to the advent of a fishery), and perhaps more recently. Sloan and Robinson (1984) suggested the possibility of a similar decline in recruitment during the same time period in British Columbia.

Thus, catch curve estimates of *M* for geoducks based on older age classes may not accurately represent current trends in natural mortality. They likewise reveal nothing about *M* for younger geoducks. In either case, our results indicate that biases in the estimate of *M* will have a major influence on model-based predictions of yield and spawning biomass per recruit. Independent estimates of *M* should therefore be a high priority for research.

Given the fact that geoducks are entirely sedentary, direct estimates of *M* for adult geoducks are possible using non-invasive tags. In 1998 WDFW began testing a tagging method for estimating *M* at a previously unfished site in northern Hood Canal. Divers "tagged" 1,128 adult geoducks (>3-4 yrs) in May 1998 by placing thin plastic stakes next to geoduck siphons at a distance of 3 inches. Geoducks were tagged within 3 ft of three lines running offshore and anchored in depths of -18 m to -70 ft MLLW. One year later, we found 875 of the original 1,128 tags remaining in the substrate. Over a 6-day period, siphons were visible next to 856 of the tags. We used a venturi dredge to excavate the 19 tags with no visible siphons; 4 of these geoducks were alive, 14 were dead, and one tag had no sign of a living or dead geoduck. The annual survival rate (*S*) for all three lines was estimated as N_1/N_0 = 861/875 = 0.984 y^{-1}. (95% CI = 0.991 - 0.973) and the corresponding estimate of *M* was 0.016 y^{-1}. Estimates of *S* on individual lines ranged from 0.996 to 0.0.970, suggesting that survival and mortality rates vary widely even over small spatial scales. The direct estimate of *M* makes fewer assumptions than catch curve estimates and is less expensive. Now that the tagging method has proved feasible, experiments to estimate *M* at sites throughout Washington are recommended.

Although the model was not nearly as sensitive to growth parameter estimates as it was to *M*, Hoffmann *et al.* (1999) found evidence for site-specific differences in the growth parameter *k* which were of "managerial significance" (i.e., of a magnitude to influence model-derived target fishing mortality rates). However, since the growth sample sites were not selected at random, regional, estimates of *k* which are simply averages of the estimated site *k*'s will be biased. One solution, albeit a costly one, is to collect additional growth samples *from* a number of randomly-selected sites in all regions. Another possible solution is to analyze the empirical relationship between mean shell length at sites and the site-specific estimate of *k*; preliminary studies suggest that there is a positive linear relationship between the two. If this relationship proves significant, the huge volume of existing shell length data gathered every year since 1968 during pre-fishing surveys could be parsed by management region to obtain regional estimates of mean shell length. These could then be compared statistically and used to calculate empirical estimates of *k* for each region. This approach, if feasible, would not require any additional field work, but would instead rely on the large and already-existing morphological database for geoducks.

Finally, we plan to continue the empirical "recovery" study on at least 15 previously fished geoduck beds. This study tracks changes in geoduck density before fishing, immediately after fishing, and then at intervals following fishing. A recovery rate for each tract is estimated from the difference in density between the first post-harvest survey and the second post-harvest survey. The study is expected to provide empirical estimates of the time required for geoduck density to return to pre-fishing levels. Thus far, three surveys have been completed at all the sites: a survey prior to fishing, a survey immediately after harvest, and a second post-harvest survey. The decrease in geoduck density immediately after fishing averaged 72% and ranged from a low of 19% to a high of 95%. The elapsed time between the first and second post-harvest surveys ranged from 4 to 11 years, averaging 8 years. During this period following fishing, density increased on all the tracts. The average estimated time to recover to pre-fishing density (assuming 100% removal of all geoducks and linear recovery) was 39 years, ranging from a low of 11 years to a high of 73 years. Thus, the proportion of fished biomass replaced each year on average was 1/39 = 0.0256. A simple biomass dynamics model was used to compare the average recovery time estimated thus far (39 years) with the existing annual harvest rate of 2.7%. The model predicted that a recovery time of 39 years and fishing at 2.7% every year eventually reduced biomass to 49% of its unfished level. Since this is greater than the 40% target level for the $F_{40\%}$ strategy, the current harvest rate of 2.7% is considered conservative. However, the study must be continued at intervals to better define the shape of the recovery curve and the time required for recovery.

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Appendix 1
Geoduck Survey Data Sheets

✓ PREFISHING
__ POST FISHING
__ OTHER____

DIG ___

REFERENCE: 99207

GEODUCK TRANSECT (900 ft.²) DATA SHEET

LOCATION: PILOT POINT REGION CODE CS TRACT CODE 06200

STATION: 87 DATE: 4/28/99 Chart Datum NAD 27

START TIME: 1041 DEPTH CORRECTION: {+1.6}

DGPS POSITION: LATITUDE: 4752962 LONGITUDE: 12230611

UNCORRECTED BEG. DEPTH: 48 ft. CORRECTED BEG. DEPTH: 46 ft.

UNCORRECTED END DEPTH: 26 ft. CORRECTED END DEPTH: 24 ft.

GEODUCK COUNT 1: 7 GEODUCK COUNT 2: 20

SHOW FACTOR: 0.75

ADULT CUCS: 4 JUVENILE CUCS: Ø

SUBSTRATE RATING: Mud 1 Sand 2 Peagravel___ Gravel___ Shell___ Cobble___

Boulder___ Unstable___ Other wood debris

P1: 6 P2: 7 P3:___ P4:___ P5:___ P6:___ P7:___ P8:___ P9:___ P10:___ P11:___ P12:___

A1: 1 A2: 18 A3: 26 A4: 28 A5: 29 A6: 4 A7:___ A8:___ A9:___ A10:___ A11:___ A12:___

A13:___ A14:___ A15:___ OTHER:___

DIVER NAME(s) AB BS BOTTOM TIME 36 min.

FROM BUOY # 36 TRANSECT LINE 82 → 87

COMPASS COURSE 270° MAGNETIC

ADDITIONAL COMMENTS:

END @: 1117

DGPS 47°52971'

122°30641'

Appendix 2
Geoduck Data Sheet Codes

Taxonomer	Common Name	Group	Phylum
0 Elzippo nullus	NO ANIMALS	ENTROPY	KARMA
1 Butter, littleneck, venus'	HARDSHELL CLAM	BIVALVE	MOLLUSC
2 Tresus spp.	HORSE CLAM	BIVALVE	MOLLUSC
3 Ptilosarcus gumeyi	SEA PEN	MISC.	COELENTERATE
4 Parastichopus califomicus	SEA CUCUMBER	CUCUMBER	ECHINODERM
5 Unspecified	GHOST SHRIMP	SHRIMP	ARTHROPOD
6 Cancermagister	DUNGENESS CRAB	CRAB	ARTHROPOD
7 Cancerproductus	RED. ROCK CRAB	CRAB	ARTHROPOD
8 Cancergracilis	GRACEFUL CRAB	CRAB	ARTHROPOD
9 Strongylocentrotus	SEA URCHIN	URCHIN	ECHINODERM
10 Mya truncata	TRUNCATED MYA	BIVALVE	MOLLUSC
11 Unspecified Pecdnid	SCALLOP	BIVALVE	MOLLUSC
12 Chaetopterid polychaete tubes	ROOTS	MISC.	ANNELID
13 Unspecified Pholadid	PIDDOCK	BIVALVE	MOLLUSC
14 Panomya be tingiana	FALSE GEODUCK	BIVALVE	MOLLUSC
15 Unspecified	ANEMONE	ANEMONE	CNIDARIA
16 Polinices lewisi	MOON SNAIL	GASTROPOD	MOLLUSC
17 Stylatula elongata	SEA WHIP	MISC.	COELENTERATE
18 Pycnopodia helianthoides	SUNFLOWER STAR	SEA STAR	ECHINODERM
19 Unspecified	NUDIBRANCH	MISC.	MOLLUSC
20 Unspecified	HERMIT CRAB	CRAB	ARTHROPOD
21 Luidla foliiolata	SAND STAR	SEA STAR	ECHINODERM
22 Pisasterbrevispinus	SHORT-SPINED STAR	SEA STAR	ECHINODERM
23 Evastertas troschelli	FALSE OCHRE STAR	SEA STAR	ECHINODERM
24 Loligo opalescens	SQUID EGGS	CEPHALOPOD	MOLLUSC
25 Polinices lewisii	MOON SNAIL EGGS	GASTROPOD	MOLLUSC
26 Unspecified	FLATFISH	FISH	CHORDATE
27 Dendrasterexcentricus	SAND DOLLAR	SEA BISCUIT	ECHINODERM
28 Modiolusrectus	HORSE MUSSEL	BIVALVE	MOLLUSC
29 Henncia leviuscula	BLOOD STAR	SEA STAR	ECHINODERM
30 Unspecified Raja	SKATE	FISH	CHORDATE
31 Pachycenanthus fimbnatus	BURROWING ANEMONE	ANEMONE	CNIDARIA
32 Metrtdium senile	PLUMED ANEMONE	ANEMONE	CNIDARIA
33 Dermastenasimbncata	LEATHER STAR	SEA STAR	ECHINODERM
34 Hydrolagus collier,	RATFISH	FISH	CHORDATE
35 Unspecified cotdd	SCULPIN	FISH	CHORDATE
36 Unspecified	BURROWING CUCUMBER	CUCUMBER	ECHINODERM
37 Nassartus spp.	BASKET SNAIL	GASTROPOD	MOLLUSC
38 Ananhichthys ocellatus	WOLF EEL	FISH	CHORDATE
39 Unspecified	STARFISH	SEA STAR	ECHINODERM
40 Sebastes spp.	COLORED ROCKFISH	FISH	CHORDATE
41 Sebastes melanops	BLACK ROCKFISH	FISH	CHORDATE
42 Hexagrammos sp.	GREENLING	FISH	CHORDATE
43 Ophiodon elongatus	LINGCOD	FISH	CHORDATE
44 S. fransiscanus	RED URCHIN	URCHIN	ECHINODERM
45 S. purpuratus	PURPLE URCHIN	URCHIN	ECHINODERM
46 S. droebachiensis	GREEN URCHIN	URCHIN	ECHINODERM
47 Anthopleura xanthogrammica	LARGE GREEN ANEMONE	ANEMONE	CNIDARIA
48 Unspecified	MYSIDS	MISC.	ARTHROPOD
49 Pisasterochraceus	OCHRE STAR	SEA STAR	ECHINODERM
50 Scorpaenichthys marmoratus	CABEZON	FISH	CHORDATE
51 Crassadoma gigantea	ROCK SCALLOP	BIVALVE	MOLLUSC
52 Eschricdous robust us	GREY WHALE	MAMMAL	CHORDATE
53 Haliods kamtschatkana	ABALONE	GASTROPOD	MOLLUSC
54 Ammodytes hexapterus	SAND LANCE	FISH	CHORDATE
55 Unspecified embiotocid	PERCH	FISH	CHORDATE
56 Solasterspp.	SUN STAR	SEA STAR	ECHINODERM
57 Octopus spp.	*OCTOPUS*	MISC.	MOLLUSC

58 Balanus nubilis	GIANT BARNACLE	MISC.	ARTHROPOD
59 Cryptochiton stellen	GUMBOOT CHITON	MISC.	MOLLUSC
60 Chlamys rubida,C. hastata.	SINGING SCALLOPS	BIVALVE	MOLLUSC
61 Fusitriton oregonensis	OREGON TRITON	GASTROPOD	MOLLUSC
62 Unspecified	GOBIE	FISH	CHORDATE
63 Orcus orcinus	KILLER WHALE	MAMMAL	CHORDATE
64 Panopea abrupta	GEODUCK	BIVALVE	MOLLUSC
65 Telmessus cheiragonus	HELMET CRAB	CRAB	ARTHROPOD
66 Squalus acanthias	DOGFISH SHARK	FISH	CHORDATE
67 Mytilus califomianus	CALIFORNIA MUSSEL	BIVALVE	MOLLUSC
68 Stylastenas forreri	FISH-EATING STAR	SEA STAR	ECHINODERM
69 Clupea harengus pallasi	HERRING	FISH	CHORDATE
70 Syngnathus leptorhynchus	PIPEFISH	FISH	CHORDATE
71 Unspecified serpulid	TUBE WORM	MISC.	ANNELID
72 Raja spp.	SKATE EGGS	FISH EGGS	CHORDATE
73 Unspecified	ASSORTED SHRIMP	SHRIMP	ARTHROPOD
74 Clinocardium nuttalli	COCKLE	BIVALVE	MOLLUSC
75 Unspecified agonid	POACHER	FISH	CHORDATE
76 Poraniopsisinflata	SPINY STAR	SEA STAR	ECHINODERM
77 Crossaster papposus	ROSE STAR	SEA STAR	ECHINODERM
78 Mediasteraequalis	VERMILLION STAR	SEA STAR	ECHINODERM
79 Oncorhynchus spp.	SALMON	FISH	CHORDATE
80 Gadus macrocephalus	PACIFIC COD	FISH	CHORDATE
81 Cucumaria miniata	ORANGE CUCUMBER	CUCUMBER	ECHINODERM
82 Eupentacta quinquesemita	WHITE CUCUMBER	CUCUMBER	ECHINODERM
83 Urticina sp.	STRIPED ANEMONE	ANEMONE	CNIDARIA
84 Unspecified holothurian	BLACK CUCUMBER	CUCUMBER	ECHINODERM
85 Gorgonocephalus euchemis	BASKET STAR	SEA STAR	ECHINODERM
86 brthasterias koehieri	RAINBOW STAR	SEA STAR	ECHINODERM
87 Lopholithodes mandti	BOX CRAB	CRAB	ARTHROPOD
88 Unspecified Porifera	LARGE SPONGES	MISC.	PORIFERA
89 Diadora spera	KEYHOLE LIMPET	GASTROPOD	MOLLUSC
90 Patira miniata	BAT STAR	SEA STAR	ECHINODERM
91 Unspecified	CORAL	MISC.	COELENTERATE
92 Pterastertesselatus	ORANGE PEEL STAR	SEA STAR	ECHINODERM
93 Aulorhynchus flavidus	TUBESNOUT	FISH	CHORDATE
94 Pododesmus cepio	JINGLESHELL OYSTER	BIVALVE	MOLLUSC
95 Pterastertesselatus	SLIME STAR	SEA STAR	ECHINODERM
96 Hydrolagus colliei	RATFISH EGG CASE	FISH	CHORDATE
97 Ophiophobs aculeata	BRITTLE STAR	SEA STAR	ECHINODERM
98 Diopatra omata	DECORATING TUBEWORM	MISC	ANNELID
99 Pugettia spp.	DECORATOR CRAB	CRAB	ARTHROPOD
100 Unspecified arthropod	ARTHROPOD	MISC.	ARTHROPOD
101 Unspecified fish	FISH	FISH	CHORDATE
102 Unspecified cnidarian	CNIDARIA	MISC.	CNIDARIA
103 Unspecified echinoderm	ECHINODERM	MISC.	ECHINODERM
104 Unspecified mollusc	MOLLUSC	BIVALVE	MOLLUSC
105 Unspecified worm	WORM	MISC.	ANNELID
106 Unspecified marine mammal	MARINE MAMMAL	MAMMAL	CHORDATE
107 Unspecified fish eggs	FISH EGGS	FISH EGGS	CHORDATE
108 Composmyaxsubdiaphana	MILKY PACIFIC VENUS	BIVALVE	MOLLUSC
109 Glycymeris subobsoleta	BITTERSWEET ARKSHELL	BIVALVE	MOLLUSC
110 Humilaria kennerleyi	KENNERLYS VENUS	BIVALVE	MOLLUSC
111 Oregonia gracdis	DECORATOR CRAB	CRAB	ARTHROPOD
112 Terebelfid sp.	TEREBELLID TUBE WORM	MISC.	ANNELID
113 Solen sicarius	JACK KNIFE CLAM	BIVALVE	MOLLUSC
114 Semele rubropicta	ROSE SEMELE	BIVALVE	MOLLUSC
115 Opisthobranch sp.	OPISTHOBRANCH	MISC.	MOLLUSC
116 Sabellid sp.	SABELLID TUBE WORM	MISC.	ANNELID
117 Hippasteria spinosa	SPINY STAR	SEA STAR	ECHINODERM
118 Pentomera populifera	MUD CUCUMBER	SEA CUCUMBER	ECHINODERM
119 Chlamys rubida	PINK SCALLOP	BIVALVE	MOLLUSC
120 Chlamys hastata	SPINY SCALLOP	BIVALVE	MOLLUSC
121 Leptasterias hexactis	SIX-RAYED SEA STAR	SEA STAR	ECHINODERM

122 Patinopecten caunnus	WEATHERVANE SCALLOP	BIVALVE	MOLLUSC
123 Scyra acutifiuns	SHARP-NOSED CRAB	CRAB	MOLLUSC
124 Munida quadnspina	PINCH BUG	CRAB	MOLLUSC
125 Sebastes caurinus	COPPER ROCKFISH	FISH	CHORDATE
126 Sebastes maliger	QUILLBACK ROCKFISH	FISH	CHORDATE
127 Sebastes auriculatus	BROWN ROCKFISH	FISH	CHORDATE
128 *Platichthys stellafus*	STARRY FLOUNDER	FISH	CHORDATE
129 *Parophrys vefulus*	ENGLISH SOLE	FISH	CHORDATE
130 *Lepidopsetta bilineata*	ROCK SOLE	FISH	CHORDATE
131 *Pleuronichthys coenosus*	C-O SOLE	FISH	CHORDATE
132 *Psettichthys melonostictus*	SAND SOLE	FISH	CHORDATE
133 *Citharichthys sp.*	SANDDAB	FISH	CHORDATE
134 *Cnibnnopsis femaldi*	CRIMSON ANEMONE	ANEMONE	CNIDARIA
135 *Unspecified tunicate*	SESSILE TUNICATES	MISC.	ASCIDIAN
136 Unspecified bryozoan	MOSS ANIMAL	MISC.	BRYOZOAN
137 *Unspecified flatworm*	FLATWORM	MISC.	PLATYHELMINTHES
138 *Unspecified peanut worm*	PEANUT WORM	MISC.	SIPUNCULID

d:\forms\animal#.wd2

Geoduck Survey Plants List Last updated: 11/10/99

CODE	TAXONOMER	DESCRIPTION	GROUP	COLOR
0	*Elzippo nullus*	NO PLANTS		ENTROPY
1	*Laminaria and similar species*	LAMINARIA	Laminaria	BROWN ALGAE
2	*Nereocystis luetkeana*	BLADDER KELP	Laminaria	BROWN ALGAE
3	*Ulva spp.*	SEA LETTUCE		GREEN ALGAE
4	Zostera *marina*	EEL GRASS		ANGIOSPERM
5		SMALL MIXED ALGAE	red-brown-green	ALGAE
6	*Unspecified*	SMALL RED ALGAE		RED ALGAE
7	*Unspecified*	LARGE RED ALGAE		RED ALGAE
8	*Diatoms*	BROWN SLIME		YELLOW-BROWN ALGAE
9	*Unspecified*	SMALL GREEN ALGAE		GREEN ALGAE
10	*Unspecified*	SMALL BROWN ALGAE		BROWN ALGAE
11	*Pterygophora californica*	FEATHER PALM ALGAE	Laminaria	BROWN ALGAE
12	*Macrocystis integrifolia*	CALIFORNIA KELP	Laminaria	BROWN ALGAE
13	Unspecified	LARGE BROWN ALGAE		BROWN ALGAE
14	*Unspecified*	FILAMENTOUS BROWN ALGAE		BROWN ALGAE
15	*Unspecified*	FLUFFY BROWN ALGAE		BROWN ALGAE
16	*Unspecified*	FILAMENTOUS GREEN ALGAE		BROWN ALGAE
17	*Unspecified*	FILAMENTOUS GREEN ALGAE		GREEN ALGAE
18	*Corallina, Bosiella*	ARTICULATED CORALLINE ALGAE	Corrallinaceae	RED ALGAE
19	*Agarum spp.*	AGARUM	Laminaria	BROWN ALGAE
20	*Costaria costada*	COSTARIA	Laminaria	BROWN ALGAE
21	*AJaria nana*	ALARIA	Laminaria	BROWN ALGAE
22	*Pleurophycusgardneri*	PLEUROPHYCUS	Laminaria	BROWN ALGAE
23	*Desmarestia spp*	DESMARESTIA	Desmarestiales	BROWN ALGAE
24	*Gigartina papillata*	GIGARTINA	Gigartinales	RED ALGAE
25	*Porphyra spp.*	PORPHYRA	Bangiales	RED ALGAE
26	Lithothamnion, L!thophyllum	CRUSTOSE CORALLINE ALAGE	Corrallinaceae	RED ALGAE
27	*Opuntiella californica*	OPUNTIELLA	Gigartinales	RED ALGAE
28	*Gracilaria verrucosa*	GRACILARIA	Gigartinales	RED ALGAE
29	Sarcodiotheca gaudichaudi	SARCODIOTHECA	Gigartinales	RED ALGAE
30	*Polyneura spp.*	POLYNEURA	Ceramiales	RED ALGAE
31	Enteromorpha intestinalis	ENTEROMORPHA	Cladophorales	GREEN ALGAE
32	*Phyllospadix scouleri*	PHYLLOSPADIX	Surf Grass	ANGIOSPERM
33	*Egregia menziesi*	EGREGIA	Laminaria	BROWN ALGAE
34	*Fucus distichus* edentatus	FUCUS	Fucales	BROWN ALGAE
35	*Iridea cordata*	IRIDEA	Gigartinales	RED ALGAE
36	*Ceramium spp.*	CERAMIUM	Ceramiales	RED ALGAE