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Stock Assessment Framework for the British Columbia Geoduck Fishery, 2008

Cadre d'évaluation des stocks pour la pêche à la panope du Pacifique en Colombie-Britannique, 2008

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ABSTRACT

The stock assessment framework for the British Columbia (BC) geoduck fishery is described. Quota calculations rely on estimates of current biomass and use regional harvest rates. Biomass is calculated as the product of geoduck bed area, density and mean weight. The sources of data and of uncertainties are discussed for each parameter. Density categorization, based on qualitative information on geoduck bed densities from harvesters and On-Ground Monitors, is now used to extrapolate densities to un-surveyed beds. Sea otters have a progressively larger impact on the BC geoduck fishery as their range expands. The Limit Reference Point (LRP) currently in use in the BC geoduck fishery is to close a bed to harvest once biomass is reduced to 40% of pre-fishery biomass; which requires the calculation of pre-fishery biomass which is fraught with problems. An alternative to the current LRP is required and options are discussed. A system of reserves may be an effective conservation measure. Estimation of geoduck biomass outside of commercially harvested areas should be conducted to determine what proportion of the geoduck biomass in BC already exists in *de-facto* reserves.

RÉSUMÉ

Le cadre d’évaluation des stocks pour la pêche aux panopes en Colombie-Britanique (CB) est décrit. Les calculs de quotas sont basés sur les estimés de biomasse actuelle et utilisent des taux d’exploitations régionaux. La biomasse est estimée comme le produit de l’aire des lits de panopes, la densité et le poids moyen des panopes. Les sources de données et les incertitudes pour chaque paramètre sont discutées. La catégorisation de densité, basée sur des informations qualitatives à propos de la densité sur les lits de panopes, obtenues des pêcheurs et des surveillants en mer, est maintenant utilisée pour extrapoler les densités aux lits qui n’ont pas étés échantillonnés. Les loutres de mer ont un impact de plus en plus marqué sur la pêche aux panopes au fur et à mesure que leur distribution s’étend. Le Point de Référence Limite (PRL) utilisé pour la pêche aux panopes consiste à fermer la pêche sur un lit quand la biomasse est réduite à 40% de la biomasse initiale. Ceci requiert le calcul de la biomasse initiale, un procédé rempli de problèmes. Une alternative au PRL actuel est nécessaire et les options sont discutées. Un réseau de réserves pourrait être une mesure de conservation efficace. La biomasse de panopes à l’extérieur des lits qui sont pêchés commercialement devrait être estimée pour déterminer la proportion de la biomasse de panopes qui n’est pas sujette à la pêche commerciale.

1 INTRODUCTION

The geoduck clam (*Panopea generosa* Gould, 1850) (Vadopalas et al. 2010) is an infaunal bivalve with a geographic range from Alaska to southern California on the west coast of North America (Quayle 1970). Populations exist in almost all sedimentary substrates, but are generally only harvestable in soft sand, mud and small aggregate sediments. A commercial fishery began in British Columbia (BC) in 1976, and has since grown to be one of the highest valued fisheries in BC at $31 million in 2006 (Figure 1). The fishery developed prior to DFO's adoption of a national policy on new and emerging fisheries (Perry et al. 1999), and initially operated as an open-access, competitive fishery which was assessed with limited information. It has since evolved to a limited licence, individual quota fishery that, through time and the active involvement and financial contribution by the Underwater Harvesters Association (UHA), has become one of the more data-rich fisheries in BC.

Recently, the method and framework used to conduct stock assessments of BC geoduck populations has changed. The previous geoduck stock assessment framework (Hand and Bureau 2012) describes the methods used to calculate geoduck biomass and quota options for the geoduck fishery up to and including 2006 quotas. Until 2006, geoduck quotas were calculated based on estimates of virgin (pre-fishery) geoduck biomass (or Bzero, B_0). Quotas were calculated on a by-bed basis and a 1% exploitation rate was applied to the virgin biomass estimates to derive quotas. Due to inherent problems in the calculation of virgin biomass (Zhang and Hand 2007, Hand and Bureau 2012), and following recommendations from Zhang and Hand (2007), the approach for calculating quotas in the geoduck fishery changed to one based on estimates of current biomass (or Bcurrent, B_c), along with new exploitation rate estimates and a new Limit Reference Point (LRP). The geoduck biomass estimation process was further refined in 2006 by including density categorization for un-surveyed beds for which density information from fishery questionnaires and On-Grounds Monitors (OGM) were available.

The objectives of the current paper are to 1) document the geoduck stock assessment framework and quota calculation process currently in use (since calculation of 2007 quotas), 2) describe the density categorization process, 3) identify sources of uncertainties in parameter estimates, 4) discuss the LRP and make recommendations for further improvements in applying the Precautionary Approach (PA) in the geoduck fishery.

1.1 HISTORY OF ASSESSMENT AND MANAGEMENT

The early geoduck fishery in BC has been described in terms of regulations, landings and quota estimation by Cox (1979), Harbo and Peacock (1983), Farlinger and Bates (1985), Farlinger and Thomas (1988) and Harbo et al. (1986, 1992, 1993, 1994, 1995). The fishery was initiated at a time when much of the biology of geoducks was unknown. Estimates of virgin biomass came originally from surveys conducted by the Marine Resources Branch, Provincial Ministry of Environment, in 1977 and 1978, which were intended only to establish the range of commercial geoduck concentrations. The mean density estimated from these surveys was low, around 0.07/m^2, but the area over which it was applied was extensive.

Prior to 1979, there were no quotas (Figure 1). An initial quota of 3,600 t was set in 1979 (1,600 t in the North Coast and 2,000 t in South Coast), based on initial stock estimates from the Provincial surveys, evaluations of patterns of effort, historical landings, and expectations of additional undiscovered stocks. An arbitrary annual harvest rate of 2 %

to 5% was suggested (Harbo et al. 1992). In some areas, arbitrary exploratory quotas were set for new fishing grounds to promote development of the fishery. In 1980, a revised exploitation rate of 1.2% to 2.5% was suggested, based on early biological data from studies in Washington State (Harbo et al. 1986). Later that year, the exploitation rate was again revised to 0.75%-2% of estimated virgin biomass, based on the first estimates of growth, mortality and recruitment from analyses of BC data in south coast waters (Breen 1982). A value towards the lower end was chosen because Goodwin and Shaul (1984) suggested that fishing may have an adverse effect on recruitment.

The main problem facing managers was the uncertainty in stock biomass estimates. The coast-wide quota was reduced in 1981 to 6 million lbs (2,722 tons) and there were discussions about reducing the quota in a stepwise manner each year because of uncertainty in stock size. Surveys were conducted in select areas of the Strait of Georgia and southwest Vancouver Island (Breen and Shields 1983) and in the central coast of BC (Harbo and Adkins, unpublished data). These survey sites were selected for their high densities and the surveys were primarily designed to obtain estimates of biological parameters rather than estimates of abundance.

With poor knowledge of stocks hindering expansion of the fishery, managers requested that additional resource surveys be carried out. There were concerns that certain areas were being over-exploited, while the overall stock was potentially being underexploited. The feasibility of improving the geoduck stock assessment was investigated by Sloan (1985), where he concluded that accurate resource surveys could only be carried out by divers. It was acknowledged that such surveys would be costly due to the constraints of diving, and the patchy distribution and wide occurrence of geoduck populations.

Quotas were first based on commercial logbook data (which provided estimates of bed area) in 1988 (Harbo et al. 1992). In 1989, a pilot individual vessel quota (IVQ) program was initiated where the coastwide quota was divided equally among the 55 licences. In the same year, a three-year rotational fishery was initiated, primarily for logistical reason to reduce the number of delivery ports for validation of quotas. Also since 1989, in the north coast only, an OGM has been present with the commercial fleet during fishing operations. Because of the IVQ fishery and the requirement to validate all landings at dockside, catch and effort data since 1989 are accurate and collected in a timely fashion. Quotas decreased steadily between 1990 and 1997 as a result of the elimination of most exploratory fisheries, a reduction of density and bed area estimates, the introduction in 1994 of a 50% B_0 LRP (conservation closures in beds where the total landings were estimated to be greater than 50% of the original biomass (Harbo et al. 1994)), and the introduction in 1995 of a 50 year time-horizon ‘amortization’ where quotas were reduced to compensate for high quotas and landings that resulted from estimation errors.

The consistent downward trend in quotas during that time prompted stakeholders to request more quota stability in order to reduce growing concerns in the market. Quotas remained relatively stable after 1997, however by 2003 there were increasing concerns about the impact of sea otter (*Enhydra lutris*) predation and the status of stocks on the West Coast of Vancouver Island (WCVI). The WCVI returned to an annual fishery, with an industry-funded OGM to observe the fishery and to record observations on fishing success and evidence of otter predation. These concerns led to a decrease in quota in 2005.

1.2 GEODUCK BIOLOGY

The species name for geoducks was recently changed from *Panopea abrupta* Conrad, 1849 to *Panopea generosa* Gould, 1850 (Vadopalas et al. 2010). Geoducks will therefore be referred to as *Panopea generosa*, although most of the literature on geoduck for the last 25 years used the former name.

1.2.1 <u>Age, Growth, Longevity and Reproduction</u>

Geoducks are among the longest-lived animals in the world, often reaching ages over 100 years, and with a maximum recorded age of 168 years (Bureau et al. 2002). Geoducks grow rapidly in the initial 10 to 15 years, after which time the growth in shell length ceases while total weight increases at a slow rate through a thickening of the shell and an increase in meat weight (Harbo et al. 1983, Sloan and Robinson 1984, Goodwin and Shaul 1984, Bureau et al. 2002, 2003). Geoducks begin to recruit to the fishery at age 4 and are fully recruited by age 6 to 12 (Harbo et al. 1983, Campbell et al. 2004, Orensanz et al. 2004).

Spawning occurs annually, mostly from June to July in association with increases in seawater temperature (Sloan and Robinson 1984). Females release from 7 to 10-million eggs which are fertilized and develop in the water column until settlement on the bottom within 40 to 50 days (Goodwin et al. 1979, Goodwin and Shaul 1984). The settled post-larvae are active crawlers and can travel along the bottom, aided by a byssal thread parachute, for several weeks. At a shell length of approximately 2 mm, they begin to burrow into the substrate. For the first two years post-settlement, juvenile geoducks are vulnerable to a number of predators, including snails, sea stars, crabs, shrimp and fishes (Goodwin and Pease 1989). Fast growing clams can bury to a refuge of 60 cm or more in two years, and the end of the burrowing stage coincides with the beginning of annual reproductive activity. Sexual maturity is related more to size than age, and has been found to occur as early as two years on the WCVI near Tofino and three years from a sample of slower growing geoducks collected in the Strait of Georgia (Campbell and Ming 2003).

Growth rate and maximum size vary substantially between regions and even within the same bed (Bureau et al. 2002, 2003). As well, both slow and fast growing geoducks can attain small or large maximum sizes, depending on the location. Geoducks from southern BC were generally smaller, younger and had faster growth than geoducks from northern BC.

Estimates of natural mortality rate for mature geoducks in British Columbia populations range from 0.01 to <0.05 (Breen and Shields 1983, Harbo et al. 1983, Sloan and Robinson 1984, Noakes and Campbell 1992, Zhang and Campbell 2004). Studies have shown geoduck recruitment rates were relatively high from 1940 to 1960, decreased in the mid-1980s and have since rebounded (Orensanz et al. 2004, Zhang and Hand 2007). Age-frequency distributions from populations sampled during surveys show prominent modes, some of which appear coast-wide (Bureau et al. 2002, 2003) suggesting that these populations may be supported by widespread recruitment pulses, which may be correlated with climate conditions (Valero et al. 2004). The recent adoption of cross-dating methodology for ageing geoducks, borrowing from dendrochronology (tree ring analysis) techniques, suggests that recruitment can be even more episodic than previously thought (Black et al. 2008).

1.2.2 Population Distribution, Structure and Dynamics

Geoducks are found from the low intertidal to at least 110 m (Jamison et al. 1984). They occupy a wide range of un-consolidated substrate, from fine silt to pea gravel, and a range of habitats from low-exposure bays with little tidal flow to surf-swept outer coasts or tidal channels.

Geoducks have a patchy distribution, likely in response to varying small-scale conditions of substrate, exposure and tidal current. Within the patches, distribution is uniform (Breen and Shields 1983). The level of connectivity in geoduck metapopulations and the extent to which individual beds are self-sustaining is unknown. Anecdotes related by commercial fishermen and OGMs suggest a range of recruitment responses. For instance, some beds that once supported populations of fishable geoduck densities have not recovered from harvest pressure, whereas other beds have sustained high landings over long periods and continue to show good recruitment. Generally, the beds displaying steady recruitment are located where water movement is high (e.g. beds in the Tofino area). Similar observations have been made in Washington State geoduck populations (Orensanz et al. 2000).

2 QUOTA CALCULATIONS

Approximately 2,300 geoduck beds have been identified along the BC coast, many of which are comprised of several sub-beds, giving a grand total of 4,504. Quota options for the geoduck fishery are calculated on a by-bed basis and are later split to sub-bed quotas before being provided to fishery managers. The number of geoduck beds and sub-beds constantly change as new beds are discovered or multiple sub-beds merge as the fishing area expands.

Starting with the 2007 fishing season, geoduck quotas have been calculated using a fixed regional exploitation rate applied to estimates of current biomass, following recommendations of Zhang and Hand (2007):

$$Quota_b = A_b \times \overline{W_b} \times dc_b \times ER_r \qquad \text{Equation 1}$$

where the subscript *b* represents a geoduck bed, A_b is the area of bed *b* (m^2), $\overline{W_b}$ is the mean geoduck weight assigned to bed *b*, dc_b is the current density of geoducks (geoducks/m^2) in bed *b*, and ER_r is the exploitation rate set for region *r,* where *r* represents either Haida Gwaii, Prince Rupert, Central Coast (CC), WCVI, Area 24, Area 12 or Strait of Georgia. The following sections detail how each of the four parameters used in geoduck calculations are estimated. A list of all parameters used in the geoduck quota calculation process, with their description, is provided in Table 1.

Quota recommendations are provided to managers who then set the quotas based on management decision rules and from feedback from harvesters and OGMs. Bed quotas are then summed to calculate the Total Allowable Catch (TAC). Portions of the TAC are set aside for First Nations food, social and ceremonial (FSC) harvest and for biological sampling and the remaining TAC is divided equally amongst 55 geoduck (G tab) licences. The coast of BC is divided into three licensing regions for geoduck management: North Coast, West Coast Vancouver Island (WCVI) and Gulf (Figure 2). The number of licences fished in each region is based on the TAC for that region. The North Coast and Gulf regions are each further divided into three rotational areas fished once every three years while the WCVI is harvested annually. Rotation Areas in the North Coast are: Haida Gwaii (formerly Queen Charlotte Islands), Prince Rupert and

Central Coast. Rotation areas within the Gulf licence region are less well defined and have changed over the years (Figure 2). Each rotational area is further divided into a number of Geoduck Management Areas (GMAs). GMAs are generally portions of one or more Statistical Sub-Areas. GMA's were originally implemented to help spread fishing effort within large open areas. Since the move to by-bed management, GMAs have not been required to spread fishing effort. They are still used however in the opening and closing of areas to fishing.

Up to (and including) the 2006 quota calculations (done in 2005), quotas were calculated for only one or two rotations at a time. In 2006, the geoduck quota calculation system was re-designed from the ground up, allowing a seamless interface with a new database containing results from density surveys, incorporating density categorization (derived from the bed information database housing qualitative information on individual beds from harvester and OGM comments), and calculating quotas based on $B_{current}$ instead of B_0. The new system was designed so that biomass and quotas are estimated for every bed on the coast, every time quotas get calculated, allowing for a readily available estimate of the "big picture" of geoduck biomass for the entire BC coast.

The following sections describe how each parameter used in biomass estimation is derived. Uncertainties in parameter estimates are discussed.

2.1 GEODUCK BED AREA

Information used to estimate the area of geoduck beds comes from four sources: 1) harvest locations, 2) substrate mapping, 3) dive surveys of geoduck density and 4) comments from harvesters and OGMs.

2.1.1 Harvest Locations

Since the inception of the geoduck fishery, commercial licence holders have been required, as a condition of licence, to submit harvest logs accompanied by a map showing the location of harvest. Geoduck beds have been spatially defined from these records. Dive locations were transcribed onto hydrographic charts and used to construct harvest bed polygons, which were then digitized to calculated bed areas. The GIS software utilized for this purpose has evolved from GAP1, which provided only an estimate of area via planometric measurement, to Compugrid in which bed polygons were first digitized and displayed as computer-generated bed maps overlaid onto a coastal basemap, to the next generation GIS program, ArcView, where bed polygons can contain many attributes, including biological, environmental or harvest information.

Bed boundaries were initially defined by following the convention of upper and lower depth limits of 2 fathoms to 10 fathoms, respectively (later changed to 3 m and 20 m with the introduction of metric charts). As new harvest logs were submitted, fishing events were either coded to an existing bed polygon, or to a new bed code if fishing occurred on new ground. Geoduck beds, which are identified by unique bed codes, are often comprised of aggregations of neighbouring sub-beds.

Most of the existing bed polygons in the South Coast and many polygons in the North Coast were originally based on harvest charts submitted in early days of the fishery. These early reports were inaccurate, often taking the form of hand-drawn sketches, or photocopies of small-scale maps with an 'x' marking the spot. In addition, conventions in mapping and interpreting harvest location reports have become more conservative as more was learned about the spatial distribution and patchiness of beds with each passing year of the fishery. Since more of the recently mapped beds are in the North

Coast, the spatial area estimates are considered more accurate and more conservative than in the South Coast.

Structured observer-fishing, designed to verify the location and extent of geoduck beds and to address inconsistencies between the estimated biomass and feedback from industry, was conducted in a number of locations in the South Coast. A grid was placed over the assumed location of the bed in question, and the fishery conducted with an on-board observer who recorded catch and effort information and diver's observations of substrate, density, product quality and 'digability' in each 0.1 nm (nautical mile) by 0.1 nm grid square. A total of 25 observer fisheries were conducted between 1996 and 1999. Bed polygons on the reference charts were redrawn from the results of these fisheries and, generally, the bed areas were found to be smaller than initially estimated. Observer fisheries have been discontinued because the accuracy of geo-referencing and spatial detail was inferior to new remote-sensing technology now in use. Observer fisheries have the merit of providing information on geoduck quality, sub-bottom substrate characteristics (related to 'digability'), presence of juveniles, etc., and should still be considered, with some improvements in protocol, for some fishery areas in conjunction with remote substrate mapping.

In 1997, all of the bed polygons were transferred onto new paper charts because of the tattered state of many of them and also because new metric editions were available. At the same time, a crude revision of the bed polygons was undertaken by deleting areas of bed that were drawn over rocks and reefs, by downsizing beds where the density removed was extremely low (the ratio of geoducks harvested to the bed area would be low if the area was overestimated) and using information from observer grid-fishing, OGMs and surveys. Accordingly, the area estimate for every bed changed, even if no modifications were made other than merely transcribing the same polygon shape to a fresh chart. A small increase in estimated area occurred as a result of conversion from imperial to metric charts, which illustrates the sensitivity of estimates to the accuracy of hydrographic charts.

Historically, harvest locations were indicated on harvest charts without providing latitude and longitude. Imprecision in harvest-location reports often led to the over-estimation of bed area, especially for beds that were fished early in the history of the fishery (1980's mostly in the Gulf region). With the advent of accurate Global Positioning System (GPS) in 2000, reliable latitude and longitude data could be obtained. In 2004, recording of latitude and longitude on geoduck harvest logs was made mandatory and, for the North Coast and WCVI where OGMs were present, use of harvest charts was discontinued. For the Gulf Region, harvest charts are still in use along with latitude and longitude data.

Yearly geoduck harvest locations are imported into an ArcView GIS (Geographical Information System) environment. Harvest locations that do not fall on an existing bed are checked individually to determine if they constitute a new bed, an extension of an existing bed or a data error. Bed areas are then revised, also taking into account data from the other three sources of information available, i.e., substrate mapping, dive surveys and OGM/harvester comments.

Landings, originally assigned at the bed level in the logbook database, have been assigned at sub-bed level since 2006, thus increasing the spatial accuracy of landing information.

2.1.2 Substrate Mapping

Remote-sensing technology (QTC View) that uses acoustical back-scatter analysis and classification is now being used to determine the sediment composition of the top layer of the seabed in order to accurately define the perimeter and extent of geoduck beds (Murfitt and Hand 2004). This substrate surveying has become an invaluable tool for determining the spatial area of geoduck beds, by eliminating hard bottom and/or areas that are too deep for harvest (>20m chart datum depth), and for designing transect surveys. Bed area estimated from substrate mapping are generally lower than the bed area estimated from fishing locations on harvester's charts.

Initially, beds for which the area estimates were thought to be grossly over-estimated were selected and targeted for acoustic surveying and mapping. The focus then shifted to a systematic surveying of all the beds on the BC coast. Since 2001, bed mapping has usually been completed prior to areas being transect surveyed by dive crews for density estimation. The well-defined bed area allows the divers to target only areas of geoduck habitat, i.e., soft substrates, and avoid placing transects on bedrock or other inappropriate habitat. To date, acoustical surveys of 1,279 beds have been completed representing 9,759 hectares (ha) or 44.1% of the total coastal bed area (Table 2).

2.1.3 Dive Surveys

Every year, four to five geoduck density dive surveys are conducted in various locations along the BC coast. Dive survey methods have been described in Campbell et al. (1998a), Hand and Dovey (1999, 2000), Babuin et al. (2006) and are discussed in detail in section 2.3. Geo-referenced substrate and geoduck count data, recorded by divers along surveyed transects, are used to refine geoduck bed areas.

2.1.4 Comments from Harvesters and OGMs

In 2000, an intensive review of the bed maps in the Prince Rupert rotation was completed, using detailed information from the north coast OGM from the 1999 fishery, survey results and a limited examination of archival harvest charts. Objectives of this review were to revise the spatial extents of bed polygons using the best available information and to recode landings with finer spatial detail so as to rectify the loss of precision resulting from landings from multiple sub-beds being assigned the same bed code. The exercise was repeated in 2001 and 2002 for beds in the Haida Gwaii and Central Coast rotations, respectively. Since then, yearly Area Committee Meetings are held with harvesters and OGMs to review beds fished in the previous harvest season. Through this bed review process, harvesters and OGMs make recommendations on the extent or the location of beds.

Although an OGM has been present on the North Coast fishing grounds for every fishery since 1989, it was not until 1997 that GPS tools were used to compile fishing events in a convenient format for accurately mapping bed boundaries. Approximately one third of the beds in the North Coast were discovered in 1997 or later, and hence it is possible to map the location and extent of these beds with reasonable accuracy by referring to the maps provided by the OGM. The review of beds fished before 1997, but not since, involves a more time-consuming effort of retrieving original harvest charts from the logbook archives.

The management scale of the geoduck fishery has been getting progressively smaller. A switch from Geoduck Management Area (GMA) to bed-by-bed management was made in 2005, although North Coast OGMs were directing boats to harvest on a bed-by-bed basis earlier. The switch to management at the sub-bed level was made in 2006. The purpose of the reduction in the spatial scale of management was to encourage the fleet to visit as many of the known beds as possible to verify their existence and to get feedback, through bed questionnaires and area committee meetings, on parameters such as area, density and productivity. Through this on-going bed review process, some polygons are moved, some deleted altogether, some increased and some decreased in size.

Harvesters have been providing feedback on geoduck beds in the form of bed questionnaires since 2004. For the 2008 fishing season, the bed questionnaires became an integral part of the Geoduck Harvest Logbook. Information gathered from the bed questionnaires, as well as information from area committee meetings and OGMs, is input and coded to the Geoduck Bed Information database which is readily useable and query-able. Information recorded in the bed questionnaires includes comments on bed area and semi-quantitative data on geoduck density, quality, etc.

2.1.5 Sources of Uncertainty in Area Estimates

Beginning in 1997, when it was acknowledged that bed areas were estimated with some degree of imprecision, area estimates were assumed to be accurate within an arbitrary error of 10% of the mean estimate. The bed review process (previous section) produced useful information for calculating error estimates around mean area, from which to derive confidence intervals around quota options. Preliminary analyses indicated that the average decrease in bed area, for the subset of beds that were revised (i.e. no new ground added or bed aggregates split) was 13.8% and 5.2% for the Prince Rupert and Haida Gwaii rotations, respectively, with a combined average of 8.8%. The arbitrary 10% error imposed on the area estimate since 1997 is therefore not unreasonable, at least in the negative direction. Although approximately 10% and 16% of redrawn beds in the Haida Gwaii and Prince Rupert rotations, respectively, actually increased, the overall change in bed area was negative in both areas. Since the outcome of bed-area verification has predominantly been a decrease in area, it was deemed non-precautionary to continue to include an upper confidence bound for this estimate in quota calculations, and the application of a positive error on area estimates was therefore discontinued beginning in 2006, as recommended by Hand and Bureau (2012).

The accuracy of the spatial representation of geoduck beds is subject to the accuracy of the harvesters' geographic referencing, the accuracy of transposition and interpretation of fisher's information, and the accuracy of the charts themselves. The changes to area estimates that resulted from the use of new GIS software or from the conversion from imperial to metric charts illustrates the sensitivity of this estimate.

In the Gulf and WCVI, most bed polygons were drawn from information based on harvesters' charts. In the early days of the fishery, logbook charts were quite inaccurate and the protocol for transcribing the information onto the DFO reference charts was less conservative than it is currently. Therefore, the area of South Coast (Gulf and WCVI) beds, which were the target of early fishing effort, were likely overestimated. Since 2001, significant effort has been invested in substrate mapping of South Coast beds in order to refine the bed area estimates. North Coast beds are not as likely to be overestimated, since the fishery developed later in the North Coast, by which time a more conservative approach had been adopted in drawing new bed polygons. In addition, many of the new beds have likely not been fully explored.

Beds will continue to be systematically mapped using the QTC View system to help further refine estimates of geoduck bed areas.

2.2 MEAN GEODUCK WEIGHT

2.2.1 Historical Mean Weight Calculation Method

Quota calculations prior to and including 1995 utilized a mean geoduck weight of 2.348 lb (1.065 kg) coastwide, based on limited market sampling of geoducks collected from four sites on the WCVI, one site on the North Coast and one site from Inside Waters in 1981 and 1982 (Harbo et al. 1983). For 1996 quotas, mean weight estimates were calculated from a more extensive market-sample data set (Burger et al. 1995) and applied separately by Region (North Coast, WCVI, St. of Georgia, Johnstone Strait). Estimates varied by Region from 2.2 lb to 2.8 lb (Hand et al. 1998b). Mean weights for the 1997 and 1998 fisheries were calculated and applied on a finer geographic scale using additional market-sample data; estimates varied between 1.7 lb and 2.9 lb by Statistical Area (Hand et al. 1998c). For the 1999 and 2000 fisheries, additional market-sample data were available and estimates were applied at an even finer spatial resolution to geoduck bed, Statistical Subarea or Statistical Area, as available. The range in estimates by Statistical Area was similar, at 1.7 to 2.8 lb (Hand et al. 1998d).

2.2.2 Current Mean Weight Calculation Method

Methods for calculating geoduck mean weight have remained un-changed since 2001 when market sampling was abandoned as a source of data in favour of piece count (number of geoducks landed) information supplied by fishermen on harvest logs (Hand and Bureau 2012). Piece-count information has been recorded on harvest logs since 1997; it is available with greater spatial coverage than market-sample data and is more efficient to access, manage and utilize. Landed weight and the number of geoducks landed, by bed, for each validated landing, were extracted from the geoduck logbook database from records where it was noted that the recorded number landed were true counts rather than estimated. Data were checked for errors and mean weight and 95% confidence bounds were calculated on a by-bed, by-GMA, by-Subarea and by-Area basis following the flow chart in Appendix 2.2. Mean geoduck weight by Statistical Area, as calculated for the 2009 fishery quotas, are shown in Table 3. The overall average was 2.44 lb, with a range of 1.26 lb to 4.04 lb.

Mean weights from piece counts are not available for every geoduck bed. For beds missing a specific mean weight, the average weight over the GMA, Subarea or Area was used, as available. For the 2009 quota calculations, of the 2334 beds coast-wide, 23.9% have bed-specific mean weight estimates, 72.3% were assigned the mean weight value of the GMA, 1.9% beds were assigned the mean weight value of the Subarea and the remaining 1.9% of beds were assigned the mean weight value of the Area. Assignment of mean weights in an ever-increasing geographical scale assumes that geoducks in beds that are close to one another are more similar than those in more distant beds.

Regardless of the geographical scale of mean weight calculation, a minimum of 10 individual landing records is required for the data to be used. This is because the standard error to mean ratio for low sample sizes is usually high, leading to unrealistic confidence bounds in biomass estimates. If the number of records available is 10 or less, the GMA, Subarea or Area mean weight would be used. Bed-specific mean weight data are available for 1552 beds (66.5%) but only 557 beds had a sample size greater than 10 as of the 2009 quota calculations.

2.2.3 Sources of Uncertainty in Geoduck Mean Weights

In cases where more than one bed is harvested and recorded on a single harvest validation page (same vessel, same day), the weight landed is apportioned to the different beds according to the number of cages landed on each bed (as recorded on harvest logs). Cages are plastic containers (rectangular milk crates) that geoducks are packed in for transport. Cages are weighed several at a time during landing validation and cages from different harvest sites are not kept separate. Some uncertainty in mean weight estimates may arise from this splitting of weights. However, the logistics of landing at the dock, in addition to the transfer of cages from harvest vessels on to packer vessels would make it impractical to track the cages by site.

The calculation process utilizes all landings data since 1997 when piece-counts were recorded. Mean weight estimates used for the calculation of 2009 quotas were thus based on logbook data from 1997 to 2006: ten years of data, three rotational periods plus one year. As more data accumulate, the question arises as to whether the mean weight has changed over time such that an estimate based on all data may not be an accurate estimate of current mean weight of populations. The decision as to the number of years of harvest data to include involves a trade-off between spatial detail and accuracy. The potential increased accuracy from using the more recent years of harvest data may be outweighed by the loss of spatial detail by having fewer landing records (fewer beds will have enough data to calculate a bed-specific estimate). To maintain a balance between accuracy and spatial coverage, a minimum of two rotations (6 years) will be used, since not all beds are fished during each rotation.

Mean weights from logbook data could be biased if there is size selectivity occurring in the fishery. Size selectivity can occur either by the spatial allocation of harvester's effort to avoid areas where the geoducks size is undesirable by the market or by the ability of divers to select from a mixture of size classes and avoid small clams. Population biomass and quota estimates could be inflated if density data from dive surveys include counts of smaller (lighter) animals that are avoided in the fishery. So, whereas the density data might include small clams, the piece-count data would not, and there would be a data mis-match. Evidence of size selectivity has been demonstrated in depletion experiments conducted at Ritchie Bay on the WCVI (Campbell et al. 1998b). Mean weights remained relatively constant in the experimental catch until divers were forced to fish beyond the usual density threshold for commercial fishing, when mean weight of geoducks declined.

Biological samples that are collected during density surveys are another source of weight data which may be more representative of true distribution of geoduck weights in the population. Hand and Bureau (2012) recommended that estimates of mean weight from logbook data and from biological sample data be compared to determine whether there is a need to develop a correction factor in the quota calculation process and, if so, the best approach to incorporate it. Detailed analyses of data for beds where both sources of data are available are presented in Appendix 3. The results suggest that size selectivity may be occurring in the Haida Gwaii and Prince Rupert regions. However, sources of uncertainty exist that could impact the ability to compare the two sources, including effects of recruitment on mean weight, spatial coverage, sample size and the timing of harvest and sampling relative to recruitment events. Consequently, modest correction factors are recommended for the Haida Gwaii (WCR = 1.108) and Prince Rupert regions only (WCR = 1.082) (Appendix 3).

Estimates of mean geoduck weight are also used to calculate the density harvested from a given bed since the most recent survey, in order to adjust density estimates to current levels (see Section 2.4.2 for details). The density harvested since the last survey on a bed is calculated as bed-landings since the survey divided by the mean geoduck weight for the bed and the bed area (see Equation 10). In this case, uncorrected logbook mean weights best represent the mean weight of the catches.

2.3 GEODUCK DENSITY DIVE SURVEYS AND DENSITY ESTIMATION

2.3.1 Background

The first estimates of geoduck density came from large-scale dive surveys conducted by the Provincial Marine Resources Branch in 1977 (Queen Charlotte, Johnstone and Georgia Straits on the East Coast of Vancouver Island; Cox and Charman 1980) and in 1978 (WCVI and the north coast, unpublished data). These surveys were discussed and results tabulated in Harbo et al. (1992) and Sloan (1985). Sites were arbitrarily chosen by viewing nautical charts and roughly determining areas with suitable unconsolidated material within diving depths. Transect locations were determined beforehand. No information was provided on how the locations were determined, although from charts provided, they appeared to be randomly positioned within arbitrary areas. Transects (2 m wide by 50 m long) were laid perpendicular to the shoreline and counts were made of the number of visible geoduck siphons and probable shows, where the siphon tip could be felt beneath the sand. No correction was used to compensate for variability in geoduck shows. Estimated densities were very low (0.05 geoducks/m^2 in Pacific Fisheries Statistical Areas 12 to 18) over large expanses of area (362,800 ha). These surveys were not intended for stock assessment but to determine distribution and were considered of little use to determine quota options.

In the early 1980's, transect surveys were conducted over very small areas that were chosen for study because of their high densities of geoducks (Breen and Shields 1983, Harbo and Adkins unpublished data). Survey locations were based on commercial fishing experience and suggestions from industry. The main objective of these studies was to obtain estimates of mortality, recruitment and growth rate. Some of these surveys included a measure of 'show factor' to estimate the percentage of siphons visible to divers at the time of observation. Density estimates in these beds (some virgin and some harvested) ranged from 0.16 to 15 geoducks/m^2 over areas of 60 to 120 m^2. Other surveys were conducted by DFO staff in small areas of interest (100 to 250 m^2) and produced density estimates ranging from 0.9 to 12.3 geoducks/m^2 (Harbo et al. 1992).

Sloan (1985) recommended a program of shallow-water dive transect surveys to increase the reliability of stock estimates. It was not until 1992 that surveys of geoduck beds were conducted to specifically estimate density for biomass calculation purposes. Marina Island was surveyed (Campbell et al. 1996a) with the objective of determining the stock status in the closed bed, and also to determine the optimal sample size for efficiency and statistical power. Transects were placed systematically over the estimated location of harvested area at 100 m intervals. Results of the study suggest that optimal transect spacing should be between 200 and 300 m.

From 1993 onwards, surveys were conducted annually (Table 4), initially by DFO staff and commercial geoduck divers and then with the additional help from First Nations fisheries programs. Industry stakeholders took on a serious and active role in conducting surveys in 1995, when they hired a biologist who, working closely with DFO, designed survey protocols and supervised and participated in the collection of survey data by industry and First Nations divers. High standards of experience were set for the participation of industry divers.

Criteria for selecting areas to be surveyed for biomass estimation and quota setting have evolved over time. Initially, surveys were conducted to provide benchmark density estimates over a wide range of bed types and geographic locations. In any given year, priority was given to areas where little or no data existed and to beds that have supported significant commercial fisheries. Beds were chosen based on recommendations from fishermen or OGMs and reviews of catch history. Later, bed-groupings within a chosen region were selected at random. As more of the geographic data gaps were filled, priority shifted to re-surveying select beds in order to answer specific questions. These questions include monitoring the recovery in closed (e.g. Marina Island) or heavily harvested (e.g. Comox Bar) beds, verifying initial survey results where there was a conflict between the perception of commercial fishers and survey data (e.g. Houston Stewart Channel, Haida Gwaii) and, lately, to examine the impact of sea otter predation on geoduck populations (e.g. Winter Harbour, Mission Group). Between 1992 and 2008, at time of writing, 27.0% of the beds, representing 50.2% of the total estimated bed area in BC, have been surveyed. Surveys have, to some degree, concentrated on the larger beds that are more important to the fishery. Landings from surveyed beds account for 63.4% of total geoduck harvest to 2008. A total of 90 density dive surveys have been completed in BC to 2008 (Table 4). Only a few of these have been published (Campbell et al. 1996a, 1996b, 1998c; Farlinger and Thomas 1991; Hand et al. 1998a; Hand and Dovey 1999, 2000, Babuin et al. 2006).

2.3.2 Dive Survey Design

Surveys completed before 1996 were systematic in design. After 1996, a two-stage design of randomly-placed transects, with systematic sub-sampling of quadrats along each transect, was recommended (Campbell et al. 1998a). Optimal sampling intensity, in terms of the number of quadrats sampled within each transect, depends on the characteristics of the geoduck bed; wide bank-type beds could be surveyed with fewer quadrats per transect, allowing more transects to be completed. The current convention is to sample every fourth quadrat on transects measuring over 400 m, every third quadrat for transects lengths between 200 m and 400 m, every second quadrat for transects between 50m and 200 m and every quadrat for transects less than 50m.

Survey areas generally include many geoduck beds; transects are stratified by bed and transect locations are randomly placed within the bed. Over time, survey effort is increasingly focussed on defined geoduck habitat, as fishery-dependent (logbooks, OGMs) and fishery-independent (acoustic substrate surveys) data accumulates and improves.

Once a general survey area is chosen, the number of individual geoduck beds (strata) that can be surveyed in a 10-day period, with a target transect-spacing of 1 transect per 300 m of shoreline distance (Campbell et al. 1996a), is determined. Each bed is considered a stratum in the survey design. A reference line approximately parallel to the shoreline is drawn on a nautical chart for each geoduck bed (stratum) to be surveyed, and the number of transects to be surveyed on each bed is determined as the length (m) of the reference line divided by 300m. Transect positions are located randomly, by stratum, along this line using a random number generator. Lead-core transects, marked at 5-m intervals, are laid perpendicular to the reference line, extending from 3 m to 18 m (10 to 60 feet) chart-datum depth. Two SCUBA divers work together, one on either side of the transect, and count visible geoduck shows or dimples within 1 m (using a metre stick) of each side of the transect. At the end of each 5m segment, divers stop to record depth, the total number of geoducks and horse clams (*Tresus capax* and *T. nuttallii*), the dominant algal species and the three most dominant substrate types.

2.3.2.1 Show Factors

Individual geoduck siphons are sometimes withdrawn below the substrate surface due to physical and/or biological effects, and are thus not readily visible to divers (Goodwin 1977, Turner and Cox 1981). The proportion of the total abundance of geoduck siphons that are visible to divers during a single observation is called the 'show factor'. Show factors are estimated by monitoring 10m x 2m plots, the location of which are chosen to contain relatively high geoduck abundance and be representative of the substrate, depth and exposure encountered during the survey. On every day of the survey, the plot is visited and the position of every newly-visible geoduck is marked with a flag. The show factor for any given day is determined by dividing the number of shows observed on that day by the total plot population size. The plot population is taken as the total number of geoducks flagged in the plot during the survey.

Show factor is generally high for most surveys (~95%). A thorough review of all show factor data collected to date should be performed to investigate the variability in show factor between regions and years; and to determine the appropriateness of applying a constant show factor value to new surveys. Using a constant show factor value would eliminate the need to set-up and survey show factor plots during the dive surveys, thus enabling more time to be spent on surveying a greater number of transects. The number of show factor plots established per survey has decreased from three in early years to only one plot since the early 2000's. In some recent surveys, setting up a show factor plot has not been possible and the average show factor for previous surveys in the same region was used.

Show Factor on day *i* (SF_i) is calculated as:

$$SF_i = \frac{n_i}{N} \qquad \text{Equation 2}$$

where n_i = number of geoducks showing in plot on day *i* and *N* = total number of geoducks enumerated in plot during survey. SF_i is used in adjusting survey density estimates to the proportion of geoducks showing that day.

2.3.3 Analysis of Survey Data

Since the survey strata are defined by an imperfect knowledge of bed location, transects can sometimes be placed on bedrock or other unsuitable habitat or, conversely, can fall on high concentrations of geoducks. As well, some beds that are included in the survey are so small that they are assigned only one or two transects. The result is high between-transect variability. In an attempt to reduce this variability, analytical procedures include combining data from separate strata into "survey sites" to increase the sample size to a minimum of three transects, and omitting transects that are located outside of geoduck beds, as determined by independent mapping of the geoduck bed from the OGM or from acoustic substrate surveys. Survey sites can consist of a single bed, if the number of transects is sufficient (≥3), or a grouping of multiple beds, if beds are small and have fewer than three transects each. Density estimates are thus calculated on a by-survey-site basis. Beds are grouped by visual similarity of exposure and slope or by qualitative descriptions by the OGM. Data are also post-stratified on the basis of the substrate types recorded during the survey. Regardless of whether the entire transect was placed over suitable substrate, no individual quadrat data were omitted within a transect. Once the data set has been thus defined, analyses follow the procedures described in Campbell et al. (1998a) and Hand and Bureau (2000).

The number of geoducks on transect *t* (g_t) is calculated as:

$$g_t = \frac{\left(\frac{g_q}{a_q} \times a_t\right)}{SF} \qquad \text{Equation 3}$$

where g_q = number of geoducks counted in all quadrats

a_q = area of all quadrats = number of quadrats X 10m^2/quadrat

a_t = area of transect = transect length X 2m (transects are 2m wide)

SF = Show Factor value, fixed (e.g. 0.95) or daily show factor value from Equation 2

Equation 3 applies the show factor correction and also accounts for sub-sampling of quadrats on a transect.

The mean geoduck survey density (*ds*) for a given survey site (*s*) (ds_s in geoducks/m^2) is calculated as the ratio of sums for the number of geoducks over all transects in the site and the total transect area in the site, as:

$$ds_s = \frac{\sum_t g_t}{\sum_t a_t} \qquad \text{Equation 4}$$

Non-parametric bootstrapping methods (Efron and Tibshirani 1993) are used to calculate 95% confidence intervals on the mean density estimate, as described in Hand and Dovey (2000). The procedure randomly re-samples *n* transects with replacement from the *n* sampled transects. Each time a transect is re-sampled for the bootstrap, a show-factor is also sampled to correct the geoduck count. The corrected geoduck count for the re-sampled transect *j* (g^*_j) is calculated by:

$$g^*_j = \frac{b^*_j}{h^*_j} \qquad \text{Equation 5}$$

where b^*_j is the number of geoducks observed in the re-sampled transect (equivalent to the term between parentheses in Equation 3) and:

$$h^*_j = \frac{Binom(N, SF_i)}{N} \qquad \text{Equation 6}$$

where, $Binom(N,SF_i)$ is a random variate with a binomial distribution, *N* is the total number of geoducks observed in the show-factor plot and SF_i is the show-factor for the day (*i*) that the transect was surveyed.

The *n* g^*_j's are added, as are the corresponding areas for the re-sampled transects, and the mean density for the iteration (d_k^*) calculated as in equation 4. The process was repeated 1000 times to obtain 1000 estimated mean densities: d^*_1, d^*_2, ...d^*_{1000} . Bootstrap 95% confidence intervals were then constructed using the percentile method.

The 1000 bootstrap estimates of the mean were sorted and the 1000(0.025)th value and 1000(1-0.025)th value were used as the bounds of the 95% confidence interval.

Each geoduck survey is analysed multiple times at various levels of data post-stratification, as follows:

<u>Run 1</u>- Geoduck density (ds_s, geoducks/m^2) is calculated using equation 4, where the number of geoducks (g_t) is the sum of all individuals from all quadrats and transects within a survey site, irrespective of depth or substrate (whether the transects fall on a geoduck bed or not). Transect area (a_t) is calculated as transect length X 2 (transects are 2m wide), summed over all transects in the survey site. Biomass estimates for each survey site are then calculated by multiplying geoduck density by site area and mean geoduck weight for the site. Site area is calculated as the length of the survey site (shoreline along which transects are randomly placed) multiplied by the average transect length in the survey site.

<u>Run 2</u>- Geoduck density (ds_s, geoducks/m^2) is calculated using equation 4, where the number of geoducks (g_t) is the sum of all individuals from all transects within a survey site, irrespective of substrate but with quadrats at depths less than 3m excluded. Transect area (a_t) is calculated as transect length, at depth > 3m, X 2 (transects are 2m wide), summed over all transects in the survey site. Biomass estimates for each survey site are calculated by multiplying geoduck density by survey area and geoduck mean weight for the site. Where survey area is calculated as the length of the survey site (shoreline along which transects are randomly placed) x the average transect length in the survey site.

<u>Run 3</u>- Geoduck density (ds_s, geoducks/m^2) is calculated using equation 4, where the number of geoducks (g_t) is the sum of all individuals from only those transects that fell on a geoduck bed, with quadrats at depths less than 3m excluded. Transect area (a_t) is calculated as transect length, at depth > 3m, X 2 (transects are 2m wide), summed over the selected transects in the survey site. Biomass estimates for each survey site are then calculated by multiplying geoduck density by survey area and geoduck mean weight for the site. Where survey area is calculated as the length of the survey site (shoreline along which transects are randomly placed) x the average transect length in the survey site.

<u>Run 4</u>- Geoduck density (ds_s, geoducks/m^2) is calculated using the Run 3 methodology, however geoduck biomass is calculated using a different estimate of site area. Biomass estimates for each survey site are calculated by multiplying geoduck density and mean weight by the digitized bed area from the GIS shape file, which is considered the most accurate estimate of bed area.

The estimates of density are therefore the same between Runs 3 and 4, as only the area used for extrapolation is different. Density estimates from Run 2 are generally lower than those from Run 4. The mean site density for surveys where results from both Run 2 and 4 are available is 1.56 and 1.67 geoducks/m^2, respectively (1992-2008, n=402, Table 5). The Run 4 mean density was significantly higher than the Run 2 density (paired t-test, $t = -8.49$, $df = 401$, $p < 0.001$) by 7%. The corresponding lower 95% confidence bound on density is 0.86 and 0.96 geoducks/m^2 for Run 2 and 4 respectively, and the 12% difference is significant ($t = -6.78$, $df = 401$, $p < 0.001$). In quota calculations, results of Run 2 are used in density extrapolation to un-surveyed beds, since those bed areas have not been as thoroughly reviewed. The exception to this is systematic surveys conducted before 1996, where Run 4 results are used because too many transects fell off-bed in these early surveys to provide meaningful Run 2 results. Run 4 results are used for biomass estimation on the surveyed beds, as they are considered the most accurate. As an increasing number of beds have revised bed area estimates through substrate mapping, the extrapolation of the more conservative Run 2 density estimates to un-surveyed beds may be less warranted. To date, substrate mapping has been conducted on 21.5% (2,361/11,003 ha) of unsurveyed bed area. One advantage of using Run 4 results to extrapolate densities to un-surveyed beds is the lower coefficient of variation and therefore tighter precision compared to Run 2 results (Table 5).

A complete review and re-analysis of all geoduck dive surveys conducted since 1992 was initiated in 2005. After reviewing the data, each survey was re-analysed with a new analysis program which produces results in a standardized MS Access database format. The resulting density estimates and associated confidence bounds were then loaded into the master Geoduck Results database (a sub-set of tables in the Geoduck Biological Database, maintained by the Shellfish Data Unit). Density estimates stored in the Results database are linked to the Quota Calculation database. In 2005, Gulf, WCVI and Haida Gwaii surveys were reviewed. Surveys from Central Coast and Prince Rupert rotations were re-analysed in 2006 and 2007 respectively. From 2008 onwards, only the current year surveys require analysis.

The most efficient schedule for surveying the North Coast is to survey in a rotation-area the year after it has been fished. This allows one year to enter, check and analyse the data before results are required for quota calculations for the next rotation. For example; Central Coast was fished in 2004 and dive surveys were conducted in the summer of 2005, which provided enough time to analyse the survey data by summer 2006 when Central Coast 2007 quota options are required by fishery managers.

2.3.4 Precision, Accuracy and Uncertainty in Density Estimates

The precision of surveys has increased over the years. For Run 4, the coefficient of variation has decreased from 0.529 to 0.383 when comparing 1992-1996 surveys with 2003-2008 surveys (Table 5). As a measure of accuracy, the difference between Run 2 and Run 4 density estimates has decreased over time. The trend is largely attributable to the improved spatial estimate of bed area from substrate mapping conducted prior to surveying in recent surveys (2003 to present).

Campbell et al. (1996a) investigated transect spacing in relation to the precision of density estimates and suggested that the distance between transects be no more than 300m. The number of quadrats required to detect a given change in density was dependent on both the initial density and the change in density that needed to be detected. For example, a greater number of quadrats are required to detect a change in density of 1 geoduck/m^2 from 10 geoducks/m^2 than from 2 geoducks/m^2, because the relative change in density is smaller in the first case.

For red sea urchin surveys, sampling uncertainty within a transect is small relative to the transect-to-transect variation at a sampling threshold 20-30 quadrats in each transect (Skibo et al. 2008). Skibo et al. (2008) modelled the impact of various survey designs on estimation of red sea urchin densities and concluded that the simplest approach to improving precision of estimates may be to use a sampling method that provides significant time savings within a transect so that more time can be spent sampling a larger number of transects. Assuming the same applies to geoduck surveys, increasing between-quadrat spacing on geoduck transects could be a way to increase the number of transects that can be surveyed. However, a sufficient number of quadrats must be surveyed within each transect. The average number of quadrats per transect on geoduck surveys conducted to date is 17.3, near the threshold suggested by Skibo et al. (2008) for red sea urchins. If within transect variability is similar for geoducks and red sea urchins, decreasing the number of quadrats sampled per transect on geoduck dive surveys may not be warranted as it may increase the within-transect variability of the estimates.

The use of index sites has been suggested as an alternative to conducting surveys in different areas of the coast every year (Orensanz et al. 2004). However, the slow-paced dynamics of geoduck populations need to be considered when determining monitoring intervals. With low rates of mortality (natural and fishing) and recruitment, population abundance and age structure would change slowly. Furthermore, small changes in density over a short time period may be hard to detect against the wide confidence bounds of density estimates. Given these difficulties, monitoring of index sites at multi-year intervals may be appropriate. Several areas of the BC coast have been surveyed more than once and more areas get re-surveyed every year. Although not initially intended as index sites, areas that have been re-surveyed are *de-facto* index sites. If a system of index sites, surveyed at multi-year intervals, is established, areas of the BC coast that have been surveyed more than once already should be considered first to take advantage of the existing time-series of data.

Under current survey analysis procedures, all data from a transect are included, regardless of whether a quadrat falls on a geoduck bed. In some cases, portions of the transect may lie outside bed boundaries on unsuitable substrate where geoduck counts are low or zero. This results in a lower overall density estimate for the transect and possibly higher variability in the estimated site density. A possible improvement to analysis methods, in situations where the geoduck bed is well-defined and the spatial data are accurate, would be to post-stratify the data so that only those quadrats that fall on a bed are included in the analysis. The elimination of at least some zero counts would produce higher density and biomass estimates for that surveyed bed, which would be more accurate and possibly more precise. Survey data from two beds, the 1998 Comox Bar and 2008 Englefield Bay (west Haida Gwaii), were chosen for analysis to compare density estimates obtained from current methodology (Run 4) to a post-stratified dataset. Results showed that the mean estimate and 95% confidence bounds increased after off-bed data were excluded (Table 6). Mean density was from 2.8% to 48.3% higher than the estimates from untrimmed data. However, the precision, or coefficient of variation, was not always improved when using only on-bed quadrats. When comparing results on a by-transect basis, some estimates of density at Comox were actually lower in the trimmed dataset but in most cases, the density from on-bed quadrats was equal to or higher than density from all-quadrats. The decrease in the number of quadrats and transect length in the post-stratified dataset ranged from 0 to 67% on a per-transect basis and from 11 to 32% on a by survey site basis.

Post-stratification of quadrat data may be contemplated for surveyed beds with accurate spatial data, since the estimate of biomass would be more accurate for that bed and results may be more consistent with impressions of commercial harvesters (with whom DFO managers works closely to set bed quotas from within the range provided by Science). Density estimates from post-stratified quadrat data would not be extrapolated to other beds. Since data from only two surveys were used in this analysis, it is unknown whether the results are representative of surveys in general. If a change in methodology is considered, a more thorough review of the potential impacts should be conducted using all surveys from all regions on the BC coast. No change is considered at this time, as current methods are more precautionary.

Accuracy of geoduck counts is also affected by the show factor (variable proportion of the population visible at any one time), described in Section 2.3.2.1. Typically, show factors derived from these plots are in the order of 90-95%, and thus corrections applied to the observed data are not large. Conditions that affect the proportion of geoducks showing operate at both seasonal and diurnal time-scales, but these have not been specifically researched. Fishermen report that proportions showing can change drastically over a matter of hours due to changes in tidal currents. Disturbance due to storms also has an effect on siphon visibility, which usually lasts for days. What is not known is whether portions of geoduck populations may be dormant and buried for long periods of time.

Estimates of total population size in the show factor plots assume that all geoducks in the plot are flagged during the period of the survey, that no mortality occurs during the survey, that the plot boundaries do not change (from water current or other disturbance) or the geoduck neck positions do not change relative to the boundary of the plot. The first assumption has the largest impact on data accuracy. A typical survey is completed in 10 days, however usable show data are only available for a maximum of 9 days because geoducks in a plot take at least a day to recover from the disturbance of the plot being set up. Nine days is likely not long enough to obtain a full census of animals in the plot in all cases, resulting in a lower N in equation 2 and a more conservative show factor. In show factor plots that were established in long-term research plots, described in Campbell et al. (2004), field teams were able to find more geoducks after a month of repeated flagging (Alan Campbell, personal communication). Mortality experiments conducted in Washington have found geoducks that remain retracted for at least 5 days (Bob Sizemore, Washington Department of Fish and Wildlife, pers. comm.). Breen and Shields (1983) surveyed and intensively harvested five study plots in the Strait of Georgia and WCVI and found the ratio of the initial estimate (counts on the first day) to final estimate of density (total number of geoducks collected in plots over the course of sampling) was consistent around 0.53 (range 0.48 to 0.56). However, these studies took place in October and November, which is approaching winter storm season, so the proportions may be low compared to peak show period in the summer months. Researchers in Washington State have found an average show factor over 12 sites established throughout Puget Sound from 1984 to 1993 of 0.62 (Bradbury et al. 1999). Since 1994, they have used a standard show factor of 0.75 for all surveys between the months of March and October, which they consider to be conservative.

Establishing and monitoring show factor plots takes an estimated 25% of total field time, plus the additional effort required to process the data. An analysis of existing show factor data should be conducted to derive conservative standard show factors that can be used instead of relying on costly show factor plot data for every survey. Expensive field time could be better used to complete more transects. Alternatively, density data from surveys could go uncorrected, which would yield more conservative density estimates.

The accuracy of the visual counts may also be affected by the detection abilities of the survey divers and their ability to distinguish between geoducks and other similar species like horse clams (*Tresus capax* and *T. nuttallii*), piddocks (*Zirfaea pilsbryi*) and false geoducks (*Panomya ampla*). This source of error is considered to be minimal because the participants are all experienced commercial divers who are required to satisfy criteria regarding their fishing experience in the region being surveyed.

2.4 USE OF GEODUCK DENSITY ESTIMATES IN BIOMASS CALCULATIONS

Not all geoduck beds have been surveyed. Methods of biomass estimation therefore differ for surveyed beds and un-surveyed beds. Before biomass can be estimated, survey density estimates must be converted to current density estimates. The method for estimating current density does not take recruitment since the most recent survey into account, which means that the estimated current density will decrease over time with cumulative harvest. As current densities on surveyed beds decrease, the biomass estimates and consequently the quota options will decrease. In order to avoid this problem, periodic re-surveys of the beds will be required to update the density estimates and account for recruitment to the beds since the previous survey.

2.4.1 Surveyed Beds

Geoduck beds that have been surveyed for density are assigned a quota based on the biomass calculated using the density data for that specific bed. Calculations are done in the Current Biomass database (Appendix 2). Run 4 density estimates are used to calculate Survey Biomass (Bs_b), which are converted to estimates of Current Biomass (Bc_b) by subtracting the commercial landings since the survey.

$$Bc_b = (A_b \times \overline{W_b} \times ds_b) - LPS_b = Bs_b - LPS_b \qquad \text{Equation 7}$$

where the subscript *b* represents geoduck bed, A_b is the area of bed *b* (in m^2), $\overline{W_b}$ is the mean geoduck weight assigned to bed *b* calculated from logbook data (Section 2.2.2), ds_b is the density estimate for bed *b* (geoducks/m^2) from the most recent survey of that bed and LPS_b are the landings post-survey on bed *b,* calculated from logbook data.

Calculation of the uncertainty around biomass estimates on surveyed beds follows methods in Taylor (1982) for products, where the uncertainty of each parameter estimate is independent and random. The coefficient of variation (ratio of the 95% confidence interval to the mean estimate itself) of biomass (CV_B) is calculated by

$$CV_B = \sqrt{CV_A^2 + CV_D^2 + CV_w^2} \qquad \text{Equation 8}$$

where CV_A, CV_D and CV_w are the coefficients of variation for estimates of bed area, density and mean weight, respectively. CV_A was set to 10% for the low bound and 0% for the high bound (i.e., no positive error on area estimate). Upper and lower 95% confidence bounds on the mean Survey Biomass estimate, for each surveyed bed, are obtained by adding and subtracting the product of CV_B and Survey Biomass. Landings post-survey are then subtracted from the Survey Biomass estimates to convert the estimates to Current Biomass estimates.

$$\textit{95\% Confidence Bounds of } Bc_b = \left(Bs_b \pm \left(Bs_b \times CV_B\right)\right) - LPS_b \qquad \text{Equation 9}$$

2.4.2 Un-Surveyed Beds

The first step in preparing density data for extrapolation to un-surveyed beds is to convert survey density to current density by subtracting the density removed by the fishery since a survey was conducted from the survey density, as recommended by Zhang and Hand (2007). The current density in surveyed beds is calculated within the Quota Calculation database, which links to the Density Removed Database. Density removed is calculated as the landings since the most recent survey divided by the product of the mean geoduck weight (linked to the Mean Weight Calculation database) and the bed area:

$$dc_b = ds_b - \left(\frac{LPS_b}{\overline{W}_b \times A_b} \right) \quad \text{Equation 10}$$

where the subscript b represents a geoduck bed, dc_b is the current density, ds_b is the most recently estimated survey density, LPS_b is the landings post-survey, $\overline{W}_b$ is the mean weight estimate and A_b is the area of the bed (m^2).

For geoduck beds that have not been dive surveyed, biomass needs to be extrapolated from surveyed beds. The BC coast has been split into Quota Calculation Regions and density estimates from beds in each region are used to extrapolate to un-surveyed beds in the same region. For the North Coast, each rotational area is defined as a Quota Calculation Region: Haida Gwaii (coded as QCI in the databases), Prince Rupert and Central Coast. The WCVI was split into two Quota Calculation Regions: Area 24 and the remainder of WCVI because densities and growth rates in Area 24 are higher than the rest of WCVI (Bureau et al. 2002). The Inside Waters (Gulf) were split into Area 12 and Strait of Georgia since densities in Area 12 are generally higher than in the Strait of Georgia. The 2004 Lund survey results were excluded from the Strait of Georgia results used for extrapolations because density estimates from the Lund survey were extremely low and it was felt that they were not representative. Furthermore, some of the Lund beds have now been allocated to aquaculture or are closed to harvest and are therefore no longer available to the commercial fishery.

Biomass is calculated as the product of bed area, mean geoduck weight and mean current density. The uncertainty associated with all three of these estimates must be taken into account when calculating biomass. Discretization methods (Ismail and Ciesielski 2003) are used to estimate biomass on un-surveyed beds. Discretization is the process of dividing a continuous variable into a finite number of discrete elements. To facilitate arithmetic operations on probabilistic quantities, the corresponding probability density functions are discretized into equiprobable ranges.

Mean weight is represented by a normal distribution. Logbook data (Section 2.2) are used to calculate the estimated value $\overline{W_b}$, and associated variance, s_w^2. The actual mean weight, μ_W, is represented by a normal distribution:

$$\mu_W \sim N(\overline{W_b}, s_w^2) \quad \text{Equation 11}$$

The distribution is then represented by an array, W. W_i is the $\frac{i - \frac{1}{2}}{n}$ quantile of mean weight, as estimated from the normal distribution. $i = 1,2,...,n$ with n = 100. Each member of W represents a range of values with a probability of 1/n. Except for the first and last elements of W, each element is near the centre of the range it represents.

Bed area is represented by a truncated normal distribution (Figure 3). The area, $\hat{A}$, is estimated as described in Section 2.1 and the estimate of $\hat{A}$ is considered to be the maximum possible area for the bed. The actual bed area is assigned the probability density function:

$$f_{area}(x \mid A_b, s_{area}^2, x \le A_b) = 2 * \frac{1}{\sqrt{2 * \Pi} * s_{area}} * \exp\left(-\frac{(x - A_b)^2}{2 * s_{area}^2}\right)$$ Equation 12

$$f_{area}(x \mid, x > A_b) = 0$$

The standard error was arbitrarily assigned a value of $s_{area} = A_b / 10$.

f_{area} was represented by an array of values, A_j , $j = 1,2,...,n$ with n = 100. A_j was the $\frac{j - \frac{1}{2}}{n}$ quantile of bed area as determined from f_{area}.

Population density is represented by a set of m surveyed beds believed to be similar to the un-surveyed bed. Similarity is determined on the basis of geographical region and, if available, density categorization (Section 2.4.2.1). An estimate of population density is available for each of the similar surveyed beds. These estimates are put into a sorted array, P_k, $k = 1,2,...,m$. The kth member of P is used as the $\frac{k - \frac{1}{2}}{m}$ quantile for the population density of the un-surveyed bed. P is a discretized representation of the probability density function for the population density of the un-surveyed bed.

Arithmetic for probabilistic quantities is possible when probability density functions are represented by arrays such as W, A and P. The simplest representation of the product of mean weight and bed area is a matrix, $Q_{i,j} = W_i * A_j$, but $Q_{i,j}$ is a much larger representation of a probability density function than was used for the original quantities. Therefore to manage the size of the representation, the elements of Q are put into a sorted array, R, and only the indexed values $n * n * \frac{i-0.5}{n}, i = 1,2,3,...n$ (n = 100) are kept. Each element of R represents an equiprobable range of the product of W and A

Biomass is the product of R and P. Multiplication is performed as it was for W and A. The $n * m$ matrix is reduced to an n-element array, B. To approximate quantiles of biomass, the rank of B_i is interpolated against the value of B_i.

Two tables are created in the quota calculation database as inputs to the discretization process, which is a C++ routine performed outside of the Quota Calculation database. The first input table is a list of all beds with associated bed area and standard error, mean weight and standard error, quota calculation region and density category (if available). It provides the data necessary for calculation of the W and A arrays used for the calculation of the Q array and R array (subset of Q), for each bed.

The second input table is a list of current density along with quota calculation region and density category (if available) for all surveyed beds. Data from this second table are used to build the P arrays for each bed. Two sets of calculations are performed by the discretization routine for each bed, based on two different P arrays. All current density estimates from surveyed beds within the same quota calculation region are used in the creation of the first P array, to produce an estimate of biomass from regional density estimates. Secondly, a density-categorized P array is created with only the current density estimates from surveyed beds in the same region that have the same density category as the bed for which biomass is being calculated. For example, if a bed in the Central Coast and has a density category of 2, then only current density estimates from surveyed beds from the Central Coast with a density category 2 will be used in the creation of the second P array. The biomass arrays B are then calculated as the product of the R and P arrays. Quantiles are used in setting the confidence bound values of B.

The final product of the discretization routine is a list of all beds with estimated biomass and associated confidence bounds, on both a regional and density-categorized basis. The table is imported back into the Quota Calculation database where the remainder of the quota calculation process takes place. For reference, mean density for a bed is back-calculated from the mean biomass estimate from the associated estimates of bed area and mean weight. The back-calculated density is slightly lower than the density used by the model because the error estimate used for bed area is one-sided.

2.4.2.1 Density Categorization

Until 2006, extrapolation of density to unsurveyed beds was based on the assumption that beds in close proximity to one another are more similar than more distant beds, and unsurveyed beds were thus assigned the density from nearby surveyed beds. Results of surveys have shown that the assumption fails in many cases. Density categorization was therefore implemented to create an improved model for density extrapolation. The process relies on semi-quantitative comments on density of geoducks obtained at meetings with harvesters and OGMs, and from OGM reports and Bed Questionnaires filled out by harvesters. All geoduck beds with information on relative density are coded as low, medium or high density, and those groupings used to derive more appropriate density estimates for use in biomass calculations for unsurveyed beds. For example, an unsurveyed bed classified as ‘high density’ would be assigned a density estimate derived from only the surveyed beds within the same region that are also classified as high density.

Comments from bed questionnaires are provided by harvesters on a by-harvest-site basis. Likewise, an OGM may make density comments on several harvest sites within a bed. Bed questionnaires use a 4 level scale to describe densities, while OGMs use a 6 level scale. The density comments from OGMs and harvesters are therefore not readily comparable and are kept in separate fields in the Geoduck Bed Information database. A single density category per bed is required, and the biomass estimation procedure uses a 1 to 3 scale (low – mid – high) for density categories. The density comments must therefore be summarized and re-coded to derive these codes.

The first step of the process is to average the by-harvest-site values of the OGM comments (Density1) and bed questionnaires from harvesters (Density2) over the bed (within the Bed Information database). The bed-averaged values of Density 1 and Density 2 are re-coded into the DenCat1 and DenCat2 fields, respectively, within the quota calculation database. Re-coding of the OGM comments (Density 1) is straight forward, and is as follows: OGM codes 2 and 3 (low and below average) are re-coded as 1 (low) in the DenCat1 field in the quota calculation database, OGM codes 4 and 5 (average and above average) are re-coded as 2 (mid) and OGM codes 6 and 7 (high and very high) are re-coded as 3 (high).

The re-coding of the four categories from bed questionnaires (Density 2) into three codes is more complex. Density categories 1 and 4 from questionnaires are simply recoded to 1 (low) and 3 (high) respectively. The challenge is in deciding whether code 2 should be considered low or mid and whether code 3 should be considered mid or high. Survey densities, for beds that have Density2 values, are used to guide the decision. Average survey density is calculated for each of the 4 categories of Density2, on a regional basis, and compared (Table 7). The difference in mean survey density between

Density2 categories is calculated. If Density2 categories 1 and 2 are closest then Density2 category 2 is coded as low and 3 as mid. If Density2 categories 2 and 3 are closest then both get recoded as mid. Finally if Density2 categories 3 and 4 are closest then Density 2 category 2 gets coded as mid and 3 as high. For the WCVI, only Statistical Areas 23 and 24 were used for density categorization for the 2009 quotas since these are the only two areas that are fished on the WCVI, except a few surveyed beds in areas 25 and 26.

If only one of the two density categorization fields in the quota calculation process has data for a given bed, then the available data is used. If density categorization data are available from both sources (DenCat1 and DenCat2 fields in the quota calculation database), then their values are averaged and rounded up.

The number and proportion of surveyed beds with density category information increased from 74.4% in 2007 to 85.6% in 2009 (Table 8) and the proportion of all beds on the coast with density information increased from 49.9% to 65.3% over the same period. Part of the increase for surveyed beds was because Rupert region surveys were not reviewed until calculations of 2008 quotas were performed. Until 2007, density comments were provided on a voluntary basis on bed questionnaires filled out by harvesters, in addition to comments received at meetings with harvesters and OGMs. Since 2008, density comments are part of the daily harvest log which will likely increase the number of beds for which comments are available in the future.

Currently, the density categorization process utilizes all comments available for a given bed, i.e., all years of comments are used. In some cases, density may change over time with the change reflected in the bed comments. Trends in density over time may be masked if data over all years are averaged, which may lead to inaccurate density categorization for the bed. The number of years of data to use in density categorization may therefore need to be adjusted in the future and will be a trade-off between the number of beds for which density comments are available and how current the information is. The number of beds which receive density comments is expected to continue to increase, given that the questionnaire is now provided on logbooks, which could enable us to decrease the number of years of data used.

2.4.2.2 Density on Beds without Categorization

Biomass on un-surveyed beds for which no density categories are available is estimated by using the regional survey density estimates during the discretization process, as described in Section 2.4.2. Both the “density categorized” and the “non-categorized” sets of calculations are performed for all beds on the coast.

2.4.2.3 Effectiveness of Density Categorization

Analyses were conducted to evaluate whether density categorization improved the accuracy of extrapolated density estimates to unsurveyed beds. Survey densities are assumed to be the best approximation of the true geoduck population density. The density-categorized and regional estimates of geoduck density were compared to the survey estimates for the 552 surveyed beds with density categories. Two questions were asked: 1) Are density-categorized estimates of density closer to the survey estimates than the regional estimates? 2) Are density-categorized estimates different from survey estimates and, if so, are they lower (conservative) or higher than survey estimates?

To test the first question, the difference between the survey density estimate and density-categorized density estimate and between survey estimate and regional density estimate were calculated for each bed. The differences were then compared with a paired t-test. Density-categorized estimates were significantly closer to survey estimates (difference =0.473) than regional estimates (difference=0.605) ($t=-8.337$, $p<0.001$) indicating that density-categorized estimates are a more accurate than regional estimates.

Paired t-tests showed that survey densities were significantly higher than density-categorized estimates ($t=8.365$, $p<0.001$). Survey densities were also significantly higher than regional estimates ($t=10.238$, $p<0.001$), indicating that density-categorized and regional estimates are conservative. The use of conservative density estimates in extrapolation to un-surveyed beds is warranted, given the greater uncertainty. The lower densities observed for regional and density-categorized estimates (vs. survey estimates) are partially due to the fact that the regional and density-categorized estimates are back calculated from biomass estimates, from the discretization method described in 2.4.2. Since biomass extrapolation uses a one-sided error on bed area, the resulting density back-calculated from biomass is also on the low side.

2.5 EXPLOITATION RATE

2.5.1 Background

The first exploitation rates for the geoduck fishery were arbitrarily set in 1979 at 2% to 5% of virgin biomass estimates (Harbo et al. 1992). In 1980, a revised exploitation rate of 1.2% to 2.5% was suggested, based on early biological data from Washington studies. Later in 1980, this was again revised to 0.75%-2% of estimated virgin biomass, based on analyses and modelling of the first estimates of growth, mortality and recruitment from BC (Breen 1982). The negative recruitment effects of fishing hypothesized by Goodwin and Shaul (1984) suggested using the lower end of the estimate. Results from a study in British Columbia in 1989 (Noakes and Campbell 1992) confirmed the low productivity and also suggested that the range was reasonable. In 1992, two PSARC working papers were reviewed which simulated population dynamics using age-structured models to examine sustainable fishing patterns in BC: Breen (1992) suggested that the current 1% (of B_0) level was conservative while Campbell and Dorociez (1992) suggested that exploitation rates near 0.5% were more appropriate except where recruitment was shown to be higher, in which case 2% of the virgin biomass could be considered. Since both authors used the same estimates of natural mortality ($M=0.02$), the difference in their conclusions could be attributed to differences in assumed recruitment.

2.5.2 Current Exploitation Rates

Zhang and Hand (2006, 2007) modelled the impact of a range of exploitation rates on geoduck population levels. Recommended exploitation rates, projected to keep the stock at or above 50% virgin biomass over a 50 year time horizon, were modelled on a regional basis to be applied to estimates of current biomass. The recommended annual exploitation rates, by region, were: 1.6% for Haida Gwaii, 1.8% for Prince Rupert, Central Coast and Inside Waters and 1.2% for WCVI. For comparison, the harvest rate in the Washington State geoduck fishery is 2.7% of current biomass, based on the $F_{40\%}$ criterion (Bradbury and Taggart 2000).

For the 2008 fishing season on the WCVI, a harvest rate of 1.8% was used in GMAs where sea otters are present, while 1.2% was used in GMAs without sea otters. The 1.2% exploitation rate for the WCVI meets the target to maintain the geoduck stock at or above 50% virgin biomass over a 50 year time horizon. Zhang and Hand (2007) estimated that predation rates by sea otters on geoducks in the Mission Group, WCVI, were between 0.15 and 0.17, over 10 times higher than the recommended harvest rate. Therefore, if sea otters are present in an area and if predation rates are similar to those in the Mission Group, geoduck stocks are likely to decline below 50% of virgin biomass as a result of sea otter predation. Since the management goal of keeping biomass above 50% virgin biomass is unlikely to be met regardless of the exploitation rate used, a 1.8% exploitation rate was chosen by resource managers for areas where sea otters are present. Use of 1.8% harvest rate instead of 1.2% in sea otter areas translates into only a 3.7% ((1.8-1.2)/(15+1.2)) increase in total (human + sea otter) predation mortality, compared to a 50% increase, had it occurred in an otter-free area.

The geoduck fishery operates on a three-year rotation, except on the WCVI and a few beds in the Strait of Georgia that are fished annually. For areas under the three-year rotation, the annual quota estimates are tripled and the area is fished once every three years.

An improved method of geoduck age determination using cross-dating techniques (Black et al. 2008) is providing more accurate estimates of geoduck ages, which will translate into more accurate reconstructions of recruitment history. Once a sufficient amount of cross-dated age data has been collected, the model of Zhang and Hand (2006, 2007) will be re-run to determine what harvest rates would be predicted based on the improved recruitment history data.

2.6 SUB-BED STATUS CODING

Locations that are defined as geoduck sub-beds can lie in areas that prohibit harvest (e.g. parks, contaminated areas or research closures) or they can have attributes that impact successful harvest (e.g. competition with sea otters or areas that are logistically infeasible). A system of status coding is used identify whether a sub-bed should be allocated quota or not, and the reason why. The sub-bed status codes are defined in Table 9. A description of the loss and alienation of geoduck ground due to sea otter predation and other reasons is provided below.

2.6.1 Impact of Sea Otter Predation on the Geoduck Fishery

Sea otters (*Enhydra lutris*) have impacted the geoduck fishery in two regions of BC: WCVI and the Central Coast. Sea otters were re-introduced to the WCVI in Checleset Bay, South of Brooks Peninsula, between 1969 and 1972 (Nichol et al. 2005). Since then, their range on the WCVI has expanded northward around the northern tip of Vancouver Island to Hope Island in Queen Charlotte Strait and southward to Vargas Island near Tofino (Nichol et al. 2005). The Central Coast sea otter population was originally located in the Goose Islands and now ranges from the southern end of the Goose Group through Queens Sound to Cape Mark (Nichol et al. 2005).

There are two mechanisms by which sea otters can impact the geoduck fishery: 1) the otters may eat the majority of geoducks in a bed making the bed no longer commercially viable, or 2) otters may eat only a portion of the geoducks in a bed, so that the bed would still be commercially harvestable, but the resulting decreased quota may make it no longer logistically feasible to harvest the beds. For example, travelling to Quatsino from Tofino for a small quota is cost prohibitive due to the travel costs alone. Furthermore, any area open for harvest requires that bio-toxin (or PSP) monitoring stations be established and maintained, further adding to the costs. For these reasons, all locations north of Estevan Point on the WCVI (Statistical Areas 25 to 27) (except for a few beds) are no longer commercially harvested.

Harvesters also report that geoducks often have thin and long siphons, referred to as 'pencil necks', in areas where sea otters are present. Harvesters believe that the geoducks dig deeper into the substrate to avoid predation and therefore have to stretch their siphons farther to reach the sediment surface to breathe and feed. Another theory is that a geoduck can get buried under a mound of sediment when sea otters dig up adjacent geoducks and would need to similarly stretch its siphon. These geoducks are of lower market quality, further decreasing the interest of harvesters to harvest in areas where sea otters are present.

Identifying the geoduck beds that have been impacted by sea otters is relatively straightforward using feedback from harvesters and OGMs at annual meetings. However, determining the magnitude of predation on a given bed is more difficult since, once an area has become too difficult to harvest, it may not be re-visited for several years: timely information would no longer be available. Determining the impact of sea otter predation on the geoduck fishery (in terms of TAC) is therefore easier than determining the impact on population biomass. The two should not be confused. A measure of the impact of predation on geoduck biomass in a geoduck bed would require a dive survey, however there would likely be little interest and support from industry to conduct dive surveys in areas that the fleet is no longer interested in harvesting.

To date, 851 sub-beds, totalling 2423 ha in area, have been identified in the Central Coast and WCVI as being impacted to some extent by sea otters (Table 10). The impacted beds represent 28.9% and 33.0% of the Central Coast and WCVI total bed area, respectively. In terms of potential quota, 34.3% and 19.4% of the Central Coast and WCVI, respectively, are affected. The estimates of potential quota and biomass represent what was available in the affected areas before the appearance of sea otters, since few surveys have been conducted in areas inhabited by sea otters. The number of geoduck licences on the WCVI declined from 15 in 2001 to 9 in 2007 and the TAC decreased by 48.6% from 1,095,000 lbs to 562,500 lbs during the same time period. This decrease is partially due to the bed review process, however most is related directly or indirectly to sea otter predation. Sea otters have recently arrived in Tofino (Statistical Area 24), a highly-productive geoduck area, where they are likely to have an impact on geoduck biomass. However, the presence of sea otters in an area does not necessarily mean that a fishery becomes non viable. For example, sea otters have been established in the area of the Mission Group, Kyuquot, WCVI, since 1983-1988 (Watson and Smith 1996) and geoduck beds in the Mission Group are still harvested today.

Although impacts of sea otters have been limited in geographical scale to two regions of the BC coast, the impacts of otters are felt by all geoduck licence holders. Areas affected by sea otters that are no longer harvestable lead to a decrease in the Total Allowable Catch (TAC) for the geoduck fishery. Since the TAC is divided equally among all licences, the decrease in TAC due to sea otters is spread among all geoduck licences.

2.6.2 Geoduck Beds Closed for Reasons Other Than Sea Otters

A number of known geoduck beds are in parks, research areas and contaminated waters and are not open to fishing, as well as areas where it is logistically infeasible to harvest or where the geoducks are not easily fishable or marketable (Table 9). In total, 887 ha (4.0%) of documented geoduck bed area are closed to harvesting, not including sea otter-affected areas. If sea otter-affected beds were closed, then 3,310 ha (15.0%)

of documented geoduck bed area would be closed to harvesting. This is not a comprehensive inventory of closed or otherwise unavailable geoduck beds; for example, geoduck beds are known to exist in the Broken Group Islands within the Pacific Rim National Park, however since harvest is not allowed within the park, the beds are not mapped. Further examples include Ritchie Bay (near Tofino) and Bamfield Marine Science Centre research closures and Ladysmith Harbour sanitary closure where biological samples have been collected from known populations, although no commercial beds are defined because they have not been harvested. Geoduck populations also exist in areas of the coast that are logistically difficult to harvest. For example, geoducks have been observed on sea cucumber surveys in Tahsis Inlet yet no commercial geoduck harvesting has been documented. Only two commercial beds have been identified in all of Johnstone Strait due to the difficulty in harvesting in the high currents that prevail, and the full extent of geoduck beds in such areas is not known. The recording of geoduck abundance on a semi-quantitative basis (none, few: 1 to 10, many: 11 to 100 or abundant: >100), is now being done during sea urchin and sea cucumber surveys. This information will be valuable in identifying where geoducks are found outside of commercially harvested areas.

2.7 LIMIT REFERENCE POINT

DFO's Harvest Strategy Compliant with the Precautionary Approach (DFO 2006) defines Healthy, Cautious and Critical Zones. The Upper Stock Reference (USR) delineates the Healthy and Cautious zones, below which the removal rate is reduced to slow or stop the decrease towards the Limit Reference Point (LRP), the delineation between the Cautious and Critical Zones. LRP is defined as the stock level below which productivity is sufficiently impaired to cause serious harm but above the level where the risk of extinction becomes a concern (DFO 2006). In the Critical Zone, the stock is considered to be in a precarious state and management actions must promote stock growth by reducing fishing mortality. The responsibility for setting the USR lies with Fishery Managers, since economic and social factors need also to be considered, while the LRP is set by Science and is solely concerned with conservation. This section documents the current process, identifies problems with the use of a reference point based on estimates of current and virgin biomass and discusses possible alternative management rules that will satisfy objectives of the precautionary approach.

2.7.1 Background

Until the 2005 fishing season, a 50-year time horizon was used by resource managers whereby quotas were adjusted, down, to evenly distribute the remainder of estimated $0.5B_0$ over the remainder of the 50-year fishery, in any given bed. Quota adjustments were applied to beds that have been harvested at higher rates because of previous estimation inaccuracies. An Amortization Factor (AF) for each bed (*b*) was calculated as:

$$AF_b = \frac{50 - (TotalLandings_b \ / \ AnnualQuota_b)}{50 - (\#YearsElapsed_b)} \qquad \text{Equation 13}$$

The estimated bed quota was multiplied by the AF to derive the recommended quota for each bed. As biomass estimates decreased with revised estimates of area, density or mean weight, or with accumulated landings, the AF would increase to the point where beds were closed. Thus, a LRP was in place which prevented a population from falling below 50% of estimated virgin levels. The pace of the approach to that LRP was slowed by the application of AFs.

The application of AFs amplified the imprecision in quota estimates. Use of the AF was discontinued in the 2007 fishing season along with the adoption of a new LRP recommended by Zhang and Hand (2007).

2.7.2 Current Limit Reference Point

Zhang and Hand (2007) argued that the arbitrary LRP used in the geoduck fishery of 50% of virgin biomass was overly precautionary, and recommended that target and limit reference points be set to 50% and 40%, respectively, of pre-fishery biomass (B_0), based on provisional LRPs used in other jurisdictions and fish stocks. If the geoduck population in a bed reaches the LRP, harvest would cease and not resume until the population has increased above the target reference point of 50% B_0, determined by surveying.

The USR, below which the stock is in the Cautious Zone, has not been established for geoducks. DFO policy suggests a progressive decrease in harvest rate as the stock level approaches the LRP, the point at which productivity of a stock is impaired. This biological lower-limit is not known for geoducks but is likely to vary considerably between beds and is probably more dependent on the existence of spawning aggregations, environmental conditions and larval dispersal patterns than on overall abundance relative to some pre-harvested state.

The DFO Precautionary Approach is similar to the 40:10 Rule, accepted by the Pacific Fishery Management Council (PFMC, Portland, Oregon, Hilborn 2002), where the USR is 40% of virgin stock size and the LRP is 10% of virgin stock size. Stocks above 40% B_0 are harvested at a target reference exploitation rate u_{ref}, stocks below 10% B_0 are closed to harvest and the exploitation rate for stocks between 10% and 40% of virgin stock size increases from 0 to u_{ref} (Hilborn 2002). The approach currently in use in the BC geoduck fishery is therefore more conservative than that of the PFMC since harvest rate is dropped to zero when the stock reaches 40% B_0.

To implement the LRP in the geoduck fishery, stock biomass relative to the virgin biomass, termed the Stock Index (SI), is calculated each year for every bed. Virgin biomass B_0 is calculated by adding total recorded bed landings to the estimated current biomass. The current biomass is then divided by B_0 to obtain the SI.

$$SI_b = \frac{Bc_b}{(Bc_b + Landings_b)} \qquad \text{Equation 14}$$

Any bed where the stock index is <0.4 is then assigned a zero quota and closed to fishing. Currently, 2028 ha (9.2%) of the total commercial geoduck bed area falls in conservation closures (Table 11). Biomass on those beds represents only 2.6% of the estimated coastwide current biomass because the density on beds in conservation closures is generally low. Of the 138 beds under conservation closure, only 34 have been dive-surveyed. The majority of the closed area is located in the WCVI (1,194 ha) and Gulf (759 ha) regions, while only 76 ha is closed for conservation in the entire North Coast. This is not surprising since the fishery has operated for longer on the WCVI and Gulf than in the North Coast and some beds were harvested heavily in the early days of the fishery before quotas came into effect.

In 2008, the average *SI* value for geoduck beds above the 0.4 LRP and for all beds on the coast were 0.840 and 0.806, respectively, based on mean biomass estimates (using density-categorized values where available). On a regional basis, the average *SI* value for beds above the 0.4 LRP and for all beds (in brackets) was 0.872 (0.865) in the North, 0.741 (0.641) on WCVI and 0.783 (0.730) in the Gulf. Lower values for the WCVI and

Gulf were expected since these regions have a longer harvest history and most beds under the 0.4 cut-off fall in these regions. Nevertheless, overall geoduck biomass on the BC coast is well above the 0.4 LRP. Only two areas on the WCVI north of Estevan Point are still being harvested, even though many of the beds have *SI* values higher than 0.4. The beds that are being harvested (Mission Group and Rolling Roadstead) have *SI* values lower than the LRP but harvest is allowed on an experimental basis to gather data from beds subject to sea otter predation.

There are many concerns with the use of a LRP based on a proportion of pre-fishery biomass for geoduck populations. Problems associated with the calculation of B_0 were discussed by Zhang and Hand (2007) and is the main rationale for the switch to calculating geoduck quotas based on current biomass. Zhang and Hand (2007) highlight the need to move away from a threshold based on ratio of current to virgin biomass. Hilborn (2002) and Hilborn et al. (2002) describe some of the issues associated with the estimation of B_0. Following is a discussion of the problems of the current LRP as they relate to geoduck populations.

1) The LRP relies on accurate estimates of current and virgin biomass on all beds. Currently, the estimated biomass in 73% of geoduck beds is based on extrapolated density estimates. Estimates of virgin biomass rely on estimates of current biomass and on catch records for each bed on the coast. In early years of the fishery, catch records were not accurately geo-referenced. Furthermore, harvesters and OGMs report that, in some beds, conservation closures are a result of misreported landings, e.g., Stirling Is, Central Coast (Doug Stewart, North Coast OGM, pers. comm.).

2) In calculations of virgin biomass, recruitment and natural mortality are not taken into account in the back-calculations as they are assumed to be equal. This approach is considered necessary in the absence of data on bed-specific recruitment. Given the irregular nature of geoduck recruitment (Bureau et al. 2003, 2003), the assumption of balance between natural mortality and recruitment is unlikely to hold true in geoduck populations. If a bed has not been surveyed (or survey data is old) and recent strong recruitment has occurred, current biomass could be underestimated and the *SI* value calculated for the bed would be lower than it should be. Conversely, if a recently-surveyed bed with a long fishing history had high recruitment, those recruits would be included in the survey data and reflected in resulting biomass estimates. The addition of substantial fishery removals to the biomass estimates would produce inflated estimates of B_0, which again will lead to a low estimate of *S*I. Although both cases described above result in more conservative results, closing beds unnecessarily results in lost production in this fishery. Biological samples from commercial beds indicate that a high proportion of populations are comprised of animals that have recruited since the fishery began (Bureau et al. 2002, 2003).

3) The method of adding cumulative landings to the current biomass in order to estimate virgin biomass inevitably leads to all harvested geoduck beds reaching the LRP as the total catch ultimately equals 60% of the pre-fished biomass. Harvesters have often commented that some beds below the LRP are among the most productive beds on the coast, which partially explains the level of harvest. Again, the problem arises from the fact that recruitment and natural mortality rates are assumed to be equal in the calculation of the Stock Index. Expanding the geographical scale over which the Stock Index is calculated and applied, e.g., at the level of Geoduck Management Area (GMA), would not alleviate the problems associated with the calculation of B_0. Eventually all GMAs would reach a point where 60% or more of the initial biomass had been harvested and closure of the geoduck fishery would result.

4) The method of calculating B_0 does not account for recruitment to a bed since harvest began and disregards the possibility that harvest could have a positive effect on recruitment such as post dispersal compensation. A reduction in density of adults may increase survival of juveniles through increased resources (space and food). There is evidence to suggest that a reduction of spawning biomass through harvest has no negative effect on recruitment (Campbell et al. 2004, Zhang and Campbell 2004). Biological sampling from heavily-harvested beds has shown an abundance of recruits (Bureau et al. 2002, 2003). As well commercial harvesters report that many beds that have a low stock index are among the most productive. Unpublished data from Washington (discussed by Orensanz et al. 2004) suggests that higher-density plots recover faster than lower density plots, either through increased recruitment or decreased mortality and that faster-recovery plots rebound to higher density than pre-harvest levels. The cumulative landings in such high productivity beds would result in low stock indices, yet these populations would not, in reality, be at risk and a closure in such cases does not seem warranted.

5) Geoduck populations have been shown to have long-term changes in productivity, mainly due to fluctuating levels of recruitment (Zhang and Hand 2007), which makes the concept of virgin biomass tenuous. Based on back-calculated year-class strength from age frequency distributions, the recruitment rate of geoduck populations over the BC coast generally declined over many decades, reaching a minimum in 1975, when the geoduck fishery started in BC, followed by a rebound in all regions of the coast (Orensanz et al. 2004). Biomass on a bed is therefore likely to have fluctuated, albeit at a slow rate of change. Back-calculation of relative abundance of geoducks in Washington suggests that biomass in the 1970s was about 50% of the biomass during the 1920s (Orensanz et al. 2000). Stock indices relative to biomass immediately before fishing began in BC, considered to be the virgin state, would differ from indices derived from biomass estimates from a different era.

6) Geoducks are sedentary. Individuals cannot aggregate and are therefore susceptible to depensation effects when spawner density is low, leading to impaired fertilization rate. Even if biomass could be accurately estimated and if there were no long-term trends in productivity, closing a bed when 60% of the biomass has been removed does not address the need for broadcast spawners to have high density patches for successful fertilization. If a bed is geographically located so as to function as a recruitment sink rather than source of larvae, there is no conservation benefit to be derived from closure. Similarly, if a bed is naturally of low density, and harvest has reduced the density to 40% of the original abundance, the density of spawners may be too low to allow for successful fertilization. The bed may be closed because it is not commercially viable, but closure to allow recovery from the reproductive products of the resident population may not provide any benefit.

7) Geoducks are broadcast-spawners with larvae that remain in the plankton for about two months. There is no demonstrated stock-recruitment relationship for geoducks; spawning biomass in a bed is not related to subsequent recruitment in that bed. It is therefore not clear that preserving a percentage of an estimated virgin abundance confers any benefit to that bed.

The LRP currently in place for the BC geoduck fishery is not appropriate and a suitable alternative needs to be found. A number of Limit Reference Points have been reviewed in Caddy and Mahon (1995), however many require the existence and understanding of a stock-recruitment relationship, for instance F_{msy}, which is unknown for geoducks. Therefore, some of the commonly used LRPs cannot be applied to the geoduck fishery.

Size limits are not a viable management measure for a variety of reasons, not the least of which is the inability by divers to determine the size of a geoduck before pulling it from the sediment. Including recruitment in the calculations of B_0 may be an improvement to the current system but the variability in space and time that is known to occur, combined with the need to also include estimates of natural mortality, might render such an approach unworkable. Orensanz et al. (2004) advocate a transition from a Biological Reference Point-based management strategy for geoduck fisheries to one based on monitoring and feedback from select index sites.

Zhang and Hand (2007) suggest that a potentially useful LRP might be a minimum density threshold below which bed recovery is impaired. As geoducks are broadcast spawners where fertilization success is dependent on localized population density, they are thus susceptible to the Allee effect whereby fertilization success can drop dramatically if density decreases below a critical point. Depensatory effects on fertilization rate, although likely given geoduck's sedentary nature, remain undocumented (Orensanz et al. 2004). Convincing evidence for reproductive depensation is difficult to find; however, studies have suggested that reproductive depensation would generally be difficult to detect even if it did occur (DFO 2006). Modelling work on broadcast spawner fertilization rates by Lundquist and Botsford (2004) did not find a strict threshold of fertilization failure with decreasing density but did show densities at which fertilization rate declined and variability in fertilization success increased. They found that spawner aggregation can increase fertilization rate at low densities. However, since geoducks are sedentary and cannot aggregate, they may be more susceptible to the effects of decreased density. The effects of spawner density on fertilization rate have also been found to vary with population size (Levitan and Young 1995). Determining the effects of decreasing density on fertilization rates for geoducks would be extremely difficult since field spawning experiments would need to be conducted under a wide range of conditions and because geoducks cannot easily be induced to spawn in the field. Experiments of this kind have been performed on sea urchins, which can readily be induced to spawn in the field (Levitan 1991, Levitan et al. 1991, 1992, Levitan and Sewell 1998). A further complication to using a density threshold as a limit reference point is the reliance on survey data to both establish the bed status and determine when a bed has recovered and may be reopened. In the absence of convincing evidence for a depensatory response in recruitment, there is no completely non-arbitrary method of determining the point at which serious harm is occurring to the resource (DFO 2006).

2.7.3 No-take Reserves

Far easier to implement would be a network of closures serving as reproductive refugia throughout the BC coast. The fishery management goal of no-take reserves is to maintain sufficient reproductive capacity to provide recruits to adjacent fished areas (Morgan and Botsford 2001). Harvest refugia, with high densities of large fertile individuals, may provide a biologically feasible way to maintain high larval production in exploited populations of invertebrates (Claereboudt 1999).

The Gwaii Haanas National Marine Conservation Area (NMCA) has recently been established in Haida Gwaii by Parks Canada. Harvest closures are planned for portions of the Gwaii Haanas NMCA, ultimately up to a targeted 30% of the area within the NMCA boundaries. For geoducks, 48.5% of the bed area, 56.7% of the biomass and 56.5% of the quota of Haida Gwaii are located within the NMCA boundaries. Fishery closures in the Gwaii Haanas NMCA will begin to be implemented before the 2012 geoduck fishery rotation and the extent of quota reductions will depend on the exact location of the no-take zones. Although the closures will likely result in a decrease in available quota, there will be benefits to conservation through the protection of spawning biomass within the closures.

Much work has been published on the potential benefits of marine reserves for fisheries on marine invertebrates with sedentary adults and a larval dispersal phase. However because of the lack of information and difficulty of studying larval dispersal patterns, most of the research has concentrated on modelling work which relies on a number of assumptions and simplifications. Nonetheless, the published models provide insight into potential designs of marine reserve networks given particular life-history characteristics.

Species with a planktonic larval phase and sedentary adults, such as geoducks, are the most likely to benefit from marine reserves (Hastings and Botsford 2003, Kaplan and Botsford 2005). In such cases, the benefit would be solely from larval export (Botsford et al. 2003). Reserve networks can be optimized with knowledge of larval dispersal patterns; however, they are rarely known with certainty (Morgan and Botsford 2001). Transport of larvae through advection (currents) can play a dominant role in determining the effectiveness of different reserve configurations (Gaines et al. 2003). With strong current, multiple reserves can be markedly more effective than single reserves of equivalent total size (Gaines et al. 2003). The potentially 'open' nature of populations of marine organisms and the complexity of understanding larval dispersal is an argument for the establishment of networks of marine reserves, rather than isolated single reserves (Stobutzki 2001). Multiple reserves, spaced more closely than the average larval dispersal distance, are suggested to be an effective and conservative strategy for maintaining healthy populations and sustainable harvest levels (Quinn et al. 1993).

Increasing yield through use of marine reserves requires that larval export outside of reserves be maximized, which means that reserves should be as small as practically possible (Hastings and Botsford 2003). However, longer larval dispersal distance requires larger reserves for sustainability (Botsford et al. 2003). Species with short-lived larvae would be better protected by smaller, closely spaced reserves, while species with longer-lived larvae and high potential for extensive dispersal require larger reserves but can withstand greater distances between reserves (Morgan and Botsford 2001). Geoduck larvae are planktonic for 40-50 days before settlement (Goodwin et al. 1979, Goodwin and Shaul 1984), which suggests that reserves need not be closely spaced. Genetic studies of BC geoduck population structure showed that geoducks exhibit panmixia at small spatial scales of 50-300 km, and stepping stone gene flow at intermediate scales of 500-1,000 km (Miller et al. 2006). Genetic studies by Vadopalas et al. (2004) showed that geoducks from different localities within Puget Sound, Washington State, were generally genetically homogenous.

The pattern of larval dispersal is an important consideration in the design of a reserve network (Morgan and Botsford 2001). Research is necessary to investigate oceanographic features which accumulate larvae and learn more about geoduck larval dispersal patterns in relation to ocean current patterns in order to locate reserves and optimize the performance of a system of reserves. Errors in identifying either source or sink locations can have detrimental effects (Morgan and Botsford, 2001). Reserves would not replace controls on fishing mortality, but rather serve as a LRP on a broad spatial scale (Morgan and Botsford, 2001). New molecular techniques for detection of larval invertebrates in natural marine samples may be promising to enhance our understanding of larval presence in the ocean (Goffredi et al. 2006).

If reserves were to be considered as a conservation measure instead of the current LRP, then a first step should be to quantify geoduck populations outside of commercially harvested areas. A number of *de facto* geoduck reserves already exist, including official closures due to contamination and those for parks and research etc, and areas where geoducks are known to be present but are either not harvestable because of poor substrate, extremes of depth (deep or shallow) or exposure, or are undesirable due to poor market quality or logistical infeasibility. Perhaps the most significant component of the management system in the geoduck fishery is the existence of these reproductive refugia (Orensanz et al. 2004).

Geoducks are found from the intertidal zone to depths of over 110m (Jamison et al. 1984) but are only harvested between about 3m to 20m depth in BC. Depth distribution data from BC surveys indicate that geoduck density generally increases with depth to at least 12-18m (Campbell et al. 1996a, 1996b, 1998c, Hand and Dovey 1999, Babuin et al. 2006). No data on geoduck distribution are available from BC for depths greater than 20m, however there is anecdotal information of high geoduck densities at depths greater than 20m in some locations in BC. The upper depth range of geoduck populations is the intertidal zone, however geoduck harvest in BC is restricted to depths >3m below chart datum to protect eel grass beds. These deep- and shallow-water portions of geoduck populations constitute harvest refuge and potential sources of larvae, assuming the populations are reproductively viable. Some data from surveys exist to quantify the density and biomass of shallow populations, however the extent of deep-water populations is unknown. Conducting deep water (>20m depth) surveys of geoduck stocks using remotely operated vehicles (ROV) and/or high resolution drop-cameras may help quantify the geoduck populations residing at un-harvestable depths. ROV and drop-camera surveys would need to be calibrated in shallow waters against dive surveys, as ROV or drop-camera surveys are not likely to detect as many geoducks as divers can. The reproductive potential of deep water populations and their potential contribution to recruitment should also be investigated.

Quantifying geoduck populations found in areas where the substrate is too hard to harvest would also be worthwhile. This could be accomplished through post stratification of survey data so that only quadrats that fall outside of geoduck beds are analysed. Also, anecdotal and qualitative information exists on most of the geoduck beds on the coast, many of which are not harvested due to substrate. However, estimating the area over which to extrapolate such density estimates would be impossible since an inventory of such area is not available; geoduck surveys target commercial geoduck beds where the substrate is suitable for harvesting. These estimations would therefore be a minimum estimate of *de-facto* reserves.

Similar to reserves, geoduck aquaculture may help with geoduck conservation by providing a source of larvae to replenish wild populations. Geoducks can be sexually mature as young as 2 years old (Campbell and Ming 2003) whereas cultured geoducks need 8-10 years of growth to attain acceptable market quality. Cultured geoducks may therefore reproduce for up to 8 years before being harvested. The high density conditions under which cultured geoducks are planted would likely lead to high fertilization success during spawning events and thus to high larval production, although such enhancement would be difficult to quantify. To date, geoduck aquaculture in BC has been limited to the Strait of Georgia.

2.8 PROCESS OF CALCULATING QUOTA OPTIONS

The geoduck quota calculations are performed in a MS Access database which contains a number of linked tables to the raw data (on network drives) and a series of queries that perform the calculations. A new Geoduck Quota Calculation database is created every year to accommodate minor modifications, typically by copying the previous version and adjusting it to meet any new requirements. For example, quota options were calculated for individual beds in 2006 and for individual sub-beds in 2007. Creating a new Quota Calculation database each year provides a historical record of the quota calculation process for each fishing season.

For organizational purposes and to facilitate trouble-shooting, portions of the quota calculation process are done in different MS Access front-ends. The different database front-ends are linked to the tables containing the raw data and to the master quota calculation database. For example, the mean weight calculations are performed in the Mean Weight Calculation database. The quota calculation process involves approximately one hundred queries. Appendix 2 contains flow charts that describe the sequence of queries used in the quota calculation process, including flow charts for the other databases that support the quota calculations. Since the quota calculation process calls on several tables found in other database front-ends, the order in which each database is updated is critical. For example the Mean Weight Calculation database calls on the GMA database. Therefore, the GMA database has to be updated first, Mean Weights second, Density Removed third and Current Density fourth before Quota Calculations can be run.

The quota calculation process itself has several components. First, tables required to estimate biomass to un-surveyed beds, using both regional and categorized density estimates, are prepared (i.e., input tables to the discretization routine). Biomass is calculated for all beds using the discretization routine (Section 2.4.2). Results from discretization are linked to the Quota Calculation database. Estimates of current biomass on surveyed beds are then swapped in for those beds that have been surveyed. Finally, quota options are calculated from the estimates of biomass (at various confidence levels) by applying regional harvest rates. Confidence level options provided to managers include: Low 95%, Low 75%, Mean, and High 75% estimates. The stock index is calculated for each bed and beds that fall below the LRP are assigned a zero quota.

2.9 QUOTA RECOMMENDATIONS

As the final step in the Quota Calculation process, fields are added to the quota recommendations table to provide additional information such as sub-bed status, density category if available and whether QTC surveyed and/or dive surveyed. This information helps managers to choose among the quota options provided based on the Management Decision Rules (DFO 2008). Managers then review quota options with harvesters and OGMs before assigning quotas to beds. Quotas on a given bed are often set lower than quota recommendations, following harvesters' or OGMs' advice. Quotas are then summed to determine the regional and coastwide TACs. The number of geoduck licences assigned to each region is set by dividing the regional TAC by the Individual Quota value. The number of licences that are fished in each licence region may be adjusted from year to year.

3 GENERAL DISCUSSION

The stock assessment and management of the BC geoduck fishery have been conducted on a progressively finer spatial scale over time in recognition of the spatially-variable nature of the dynamics of geoduck populations. Such fine spatial scale is possible because geoducks are sedentary and exist within well defined habitats, combined with accurate catch and effort data captured with high geo-spatial detail and a large amount of fishery-independent data. Currently, biomass is calculated for over 2,300 beds, which are divided among over 4,500 sub-beds for management purposes. The process is complex, but undoubtedly more accurate than if biomass was calculated on a larger spatial scale.

Geoduck bed area estimates are constantly being refined through substrate mapping, dive surveys and feedback from OGMs and harvesters. Continued substrate mapping along with dive survey efforts and regular feedback from OGMs and the geoduck fleet will further improve our estimates of bed area.

Preliminary analyses suggest that the precision around quota estimates could be improved in two ways: 1- by using density estimates from Run 4s for extrapolation to un-surveyed beds that have been substrate mapped and 2- by estimating density for surveyed beds from only those quadrats that fall on-bed. The outcome of adopting either or both options would be a slight increase in the mean and lower 95% confidence bound estimates of density which would translate to slightly increased quota recommendations. Both options would require extensive changes to databases and analytical procedures. Option 1 could be more easily implemented than option 2, which would require plotting of all survey quadrats in GIS environment, a GIS routine to code which quadrats fall on-bed, and a re-analysis of all 90 geoduck surveys conducted to date. Option 2 would further be complicated by the lack of accurate transect position information for early surveys. Implementation of options 1 or 2 is not being considered at this time since methods currently in use are more precautionary.

Dive surveys have been getting more accurately-placed over time due to the practice of substrate mapping geoduck beds before they are surveyed so that fewer transects fall on inappropriate habitat. An additional way to improve precision of density estimates would be to increase the number of transects surveyed in a survey site. However, since the number of transects that can be surveyed in a season is limited, increasing the number of transects per survey site would mean surveying fewer sites. The sampling intensity that is currently used on geoduck surveys is based on a study of a single bed in the Strait of Georgia (Campbell et al. 1998a). During the 2009 season, some beds on the west coasts of Banks and Campania Islands, North Coast, were surveyed more intensively to evaluate the effects of transect spacing on the precision of density estimates. Results have not yet been analyzed.

The geographical range of sea otters on the BC coast is expanding and they are likely to have a progressively larger impact on geoduck and other marine invertebrate fisheries in the future. The exploitation rates and LRP chosen for the geoduck fishery were based on the management goal of keeping geoduck biomass above 40% of estimated virgin (pre-fishery) levels. In the presence of sea otters, the management goal is unlikely to be met irrespective of commercial harvest rate and LRP. According to the rules that were set, geoduck harvest should not occur in areas where current biomass is below 40% of estimated virgin biomass regardless of the cause. Exceptions to the rule were made in two areas on WCVI (Mission Group and Rolling Roadstead) in order to collect information on the impact of sea otter predation on geoduck populations and the extent to which the fishery can co-exist with sea otter populations.

Geoduck populations may have been at historically-high levels when the fishery developed. Before the mid-1800s and the extirpation of sea otters, these predators probably limited the size, abundance and distribution of many invertebrate populations (Watson 2000). With the elimination of the key predator, dense populations of large invertebrates resulted (Watson 2000). The B_0 estimates used in the calculation of the Stock Index for the geoduck fishery are likely to be greater than equilibrium biomass that existed with sea otters, and maintaining geoduck populations above 40% of B_0 estimates that are based on abundance levels resulting from prolonged absence of sea otters is unlikely to succeed. Further research is needed to determine the impacts of sea otters on geoduck populations and to determine if there is a level of harvest where the fishery and sea otters can co-exist without endangering geoduck stocks. With the expansion of the sea otter range in BC, finding ways in which the geoduck fishery can co-exists with sea otters is desired.

An alternative option to using the current LRP is needed for the BC geoduck fishery as the current LRP rules will inevitably lead to the closure of the geoduck fishery at some point in time. Protecting a portion of the geoduck population from harvest through closures or reserves could be a viable option. Only a small portion of known geoduck beds exist in closures of one type or another, however such is due to the fact that only through commercial harvest do geoduck beds get identified and mapped. We do not have an inventory of all geoduck populations on the BC coast. The fact that the geoduck fleet continues to discover new beds every year further attests to the fact that not all geoduck populations on the BC coast have been documented.

Much of the published research on marine reserve networks for benthic invertebrates suggests that a network of many smaller reserves may be preferable to a single or few larger reserves, and that the scale of the reserves is dependent on the scale of larval dispersal. The existing *de-facto* geoduck reserves are likely valuable as a conservation tool for the geoduck fishery. There may be considerable geoduck biomass outside commercially harvested areas, however efforts are needed to determine the extent of these populations and determine their potential benefit as a conservation tool in the BC geoduck fishery.

4 RECOMMENDATIONS

1. Regional correction factors should be applied to mean weight estimates from logbook data. A 10 % correction factor should be applied to the Haida Gwaii and 8% to Prince Rupert region until further work is conducted.
2. Use regional average show factor values until a thorough analysis of all show factor data collected to date is conducted.
3. Continue to use Run 4 density estimates for surveyed beds until further analysis have been conducted to determine the potential benefits of quadrat post-stratification in calculating geoduck density estimates.
4. Efforts should be made to quantify geoduck biomass outside of commercially harvested beds to help assess the amount of biomass that resides in *de-facto* reserves.
5. Alternative Limit Reference Points, or other conservation measures, need to be considered as alternatives to the current LRP for the BC geoduck fishery. The 40%

B_0 LRP should continue to be used until an alternative LRP or similar conservation measure has been developed, with the exception of experimental areas.

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Table 1: List and description of parameters used in the geoduck quota calculation process.

Parameter	Description	Units
Equation 1		
$Quota_b$	Quota estimate for bed *b*	lbs
A_b	Area of bed *b*	m^2
$\overline{W}_b$	Mean weight estimate for bed *b*	lbs
dc_b	Current density estimate for bed *b*	geoducks/m^2
ER_r	Exploitation rate for region *r*	
Equation 2		
SF_i	Show factor on day *i*	
n_i	Number of shows in plot on day *i*	# geoducks
N	Total number of geoduck shows observed in plot	# geoducks
Equation 3		
g_t	Estimated number of geoducks on transect *t*	# geoducks
g_q	Number of geoducks counted in quadrats *q*	# geoducks
a_q	Area of surveyed quadrats *q* (# quads X10)	m^2
a_t	Area of transect *t* (transect length X 2)	m^2
Equation 4		
ds_s	Estimated survey density for site *s*	geoducks/m^2
Equation 5		
g^*_j	Estimated # of geoducks on transect *j* corrected for show factor	# geoducks
b^*_j	Estimated # of geoducks on transect *j* not corrected for show factor	# geoducks
h^*_j	Show factor value use in bootstrapping	
Equation 6		
$Binom(N,SF_i)$	Random variate with a binomial distribution	
N	Total number of geoduck shows observed in plot	# geoducks
SF_i	Daily show factor value on day transect was surveyed	
Equation 7		
Bc_b	Current biomass on bed *b*	lbs
ds_b	Survey density on bed *b*	geoducks/m^2
LPS_b	Landings post-survey on bed *b*	lbs
Bs_b	Survey biomass on bed *b*	lbs
Equation 8		
CV_B	Coefficient of variation of biomass estimate	
CV_A	Coefficient of variation of bed area	
CV_D	Coefficient of variation of density estimate	
CV_W	Coefficient of variation of mean weight estimate	
Equation 9		
All parameters already defined		
Equation 10		
All parameters already defined		
Equation 11		
μ_w	Normal distribution representation of mean weight	lbs
s^2_w	Variance of mean weight estimate	
Equation 12		
s^2_{area}	Variance of bed area	
Equation 13		
AF_b	Amortization Factor on bed *b*	
$Landings_b$	Cumulative landings on bed *b*	lbs
$AnnualQuota_b$	Latest annual quota estimate for bed *b*	lbs
$\#YearsElapsed_b$	Number of years of harvesting elapsed on bed *b*	years
Equation 14		
SI_b	Biomass stock index for bed *b*	

Table 2: Geoduck sub-beds (number and area) that have been substrate mapped with QTC hydro-acoustic remote sensing, by management region, before 2009.

Licence	Management	QTC Surveyed			
Region	Region	# sub-beds	% sub-beds	Area (Ha)	% Area
North Coast	Haida Gwaii	264	34.9	979	38.7
North Coast	Prince Rupert	125	13.9	579	22.9
North Coast	Central Coast	325	29.1	726	33.5
West Coast	WCVI	222	23.0	2159	39.6
Gulf	Inside Waters	343	44.7	5316	56.3
Coastwide		**1279**	**28.4**	**9759**	**44.1**

Table 3: Mean geoduck weight estimates by Statistical Area, estimated from 1997-2006 commercial logbook data.

Statistical	Geoduck Weight		
Area	n	Mean (kg)	SE
1	126	1.1062	0.0157
2	5433	1.2446	0.0032
3	278	1.0336	0.0092
4	867	1.1557	0.0067
5	1400	1.0921	0.0047
6	2121	1.1145	0.0042
106	166	1.1341	0.0110
7	2925	1.0885	0.0042
8	449	1.2047	0.0102
9	183	1.1022	0.0157
10	198	0.9428	0.0125
12	382	1.1418	0.0085
13	370	1.1257	0.0126
14	1461	1.1422	0.0057
15	967	1.0301	0.0074
16	427	0.9931	0.0076
17	218	1.0486	0.0134
18	49	0.9593	0.0163
19	264	1.2704	0.0104
20	6	0.7771	0.0524
23	529	1.0226	0.0079
24	4764	1.1061	0.0030
124	13	1.3673	0.0944
25	928	1.0022	0.0070
26	917	0.9200	0.0063
27	575	0.9591	0.0085
29	17	0.9645	0.0301
Overall	**26033**	**1.1183**	**0.0014**

Table 4: Geoduck density dive surveys conducted in BC between 1992 and 2008.

Statistical Area	Year	Survey	Statistical Area	Year	Survey
1	2002	Parry Passage	7	1995	Goose/Wurtele/Seaforth
1	2002	Virago Sound	7	1996	S Bardswell/Prince Group
2	1994	Burnaby Island	7 and 8	1998	Hakai Passage
2	1995	Hotspring Island	7	2003	Spider Anchorage, 2003
2	1996	Houston Stewart Ch.	7	2005	Ivory to Stryker
2	1997	Cumshewa Inlet	7 and 8	2005	Tribal to Nalau
2	1998	Selwyn/Dana/Logan Inlets	12	1994	Goletas Channel
2	1999	Buck Ch.	12	1994	Malcolm Island
2	1999	Englefield Bay	12	1995	Duncan Island
2	1999	Gowgaia Bay	12	2003	Goletas Channel, 2003
2	1999	Hippa Island	12	2006	Port Hardy 2006
2	1999	Houston Stewart Ch. 1999	13	1992	Marina Island, 1992
2	1999	Port Chanal	13	2002	Marina Island, 2002
2	1999	W Skidegate Ch.	14	1993	Comox, 1993
2	2001	Kano Inlet	14	1996	Oyster River
2	2001	Tasu Sound	14	1998	Comox 1998
2	2004	Carpenter Bay SE	14	2005	Hornby Island
2	2006	Poole Inlet	14	2005	Mapleguard / Qualicum
2	2008	2008 Cumshewa	14	2007	Comox 2007
2	2008	2008 Englefield	15	2004	Lund 2004
2	2008	2008 Gowgaia	16	1999	Thormanby Island
2	2008	2008 Port Chanal	17	2000	Round Island
2	2008	2008 Tasu Sound	17	2001	Kulleet Bay
2	2008	Skedans	17	2008	Round Island
2	2008	Tanu	18	2001	Boatswain Bank
3	1998	Dundas Island	23	2000	Barkley Sound
4	2004	SE Prescott	23	2002	NE Barkley Sound
4	2004	Tree Nob	23	2005	Barkley 2005
4	2006	Melville/Baron	24	1994	Elbow Bank
5	1995	Griffith Harbour	24	1995	Yellow/Elbow Banks
5	1997	Principe Channel	24	1997	Millar Channel
5	2006	Griffith Harbour 2006	24	1997	Yellow Bank
5	2006	Larsen Harbour	24	2004	BlundenBartlett
5 and 6	2007	Deer Pt to Langley Pass	24	2004	Tofino 2004
6 and 7	1993	Price Island	24	2006	Tofino 2006
6	1994	Kitasu Bay	24	2007	Millar and Russell Channels
6	1994	West Higgins Pass	24	2008	Bartlett Is
6	1995	SW Aristazabal	25	2000	Nootka Sound
6 and 106	1995	Weeteeam Bay	25	2001	Rolling Roadstead
6	1996	Otter Pass	26	1998	Kyuquot
6	1996	West Aristazabal Island	26	2003	Mission Group, 2003
6	1997	Anderson/Laredo	27	1996	Quatsino 1996
6	2007	Kettle Inlet to Prior Passage	27	1996	Winter Harbour
106	1998	Moore Islands	27	2002	Winter Harbour, 2002
7	1994	McMullin Group	29	2000	Valdes Island

Table 5: Comparison of Run 2 (depth >3m, all transects) and Run 4 (depth >3m, on-bed transects only) estimates of geoduck mean density, lower 95% confidence bounds and coefficient of variation (CV). All surveys where both Runs 2 and 4 were available are included. n= number of survey sites. CV calculated as (Mean Density – Low 95% CB)/Mean Density.

Years	Run	Mean Density ($\#/m^2$)	Lower 95% CB	CV	n
1992-1996	2	1.627	0.670	0.588	59
	4	1.842	0.867	0.529	59
1997-2002	2	1.318	0.707	0.464	138
	4	1.469	0.818	0.443	138
2003-2008	2	1.709	1.023	0.402	205
	4	1.762	1.088	0.383	205
All Years	2	1.563	0.863	0.448	402
	4	1.673	0.963	0.425	402

Table 6: Comparison of density estimates, by survey site, using data from all quadrats on the transect vs. using only quadrats that fall on-bed, for the 1998 Comox and 2008 Englefield Bay surveys. Coefficient of variation (CV) calculated as (Mean Density – Low 95% CB)/Mean Density or (High95% CB – Mean Density)/Mean Density.

			Density (geoducks/m^2)			CV		On-Bed vs All Quadrats			
Survey	Site	Quadrats Used	Low 95 CB	Mean	High 95 CB	Low interval	High interval	Difference in Mean	% Diff. in Mean	% Diff. in Transect Length	% Diff. in No. of Quadrats
Comox	2	All	**0.240**	**0.333**	**0.410**	**0.281**	**0.230**				
		On-Bed	0.255	0.343	0.488	0.257	0.423	0.009	2.8	-32	-31
Englefield	1	All	**1.374**	**2.241**	**3.189**	**0.387**	**0.423**				
		On-Bed	2.004	3.323	4.053	0.397	0.219	1.082	48.3	-30	-37
	2	All	**1.207**	**2.768**	**4.358**	**0.564**	**0.574**				
		On-Bed	2.004	3.345	4.739	0.401	0.417	0.577	20.9	-20	-20
	3	All	**1.821**	**2.213**	**2.525**	**0.177**	**0.141**				
		On-Bed	1.853	2.498	3.298	0.258	0.321	0.284	12.8	-11	-12
	5	All	**0.414**	**1.328**	**2.948**	**0.688**	**1.219**				
		On-Bed	0.776	1.698	3.315	0.543	0.952	0.370	27.9	-23	-23
	6	All	**1.287**	**2.095**	**2.805**	**0.386**	**0.339**				
		On-Bed	1.772	2.529	3.207	0.299	0.268	0.434	20.7	-17	-17

Table 7: Re-coding scheme for "Density 2" information (density comments from bed questionnaires), based on survey density, by region. Density 1 are comments from OGMs.

Licence	Managment	Density 2		Survey Densities				Density 1	Quota Calculation
Region	Region	Dens2	n	n	Average	Min	Max	Average	Density Category
N	HG	1	6	3	0.67	0.47	0.99	4.20	1
N	HG	2	39	21	0.81	0.11	2.21	4.62	2
N	HG	3	51	34	1.64	0.48	4.59	4.95	3
N	HG	4	55	37	1.70	0.11	3.57	5.42	3
N	PR	1	7	3	1.26	0.13	1.85	4.57	1
N	PR	2	26	18	1.92	0.13	4.16	4.91	2
N	PR	3	34	17	2.23	0.56	5.08	5.00	2
N	PR	4	70	35	2.73	0.52	10.22	5.54	3
N	CC	1	12	1	0.65	0.65	0.65	3.80	1
N	CC	2	44	12	1.80	0.79	4.13	4.53	2
N	CC	3	109	34	2.10	0.43	6.72	5.13	2
N	CC	4	172	52	3.09	0.43	9.69	5.59	3
G	IW	1	36	8	0.16	0.09	0.33	3.58	1
G	IW	2	32	10	0.29	0.04	0.65	3.33	2
G	IW	3	6	2	0.73	0.30	1.15	5.50	3
G	IW	4	N/A	N/A	N/A	N/A	N/A	N/A	3
W	WCVI*	1	8	1	0.25	0.25	0.25	2.33	1
W	WCVI*	2	46	23	0.63	0.13	2.51	3.55	2
W	WCVI*	3	42	21	0.77	0.26	1.72	4.30	2
W	WCVI*	4	8	2	1.50	1.50	1.50	3.50	3

*: Only uses data from Statistical Areas 23 and 24

Table 8: Number and percentage of geoduck beds with density category data, by quota calculation region, for the 2007 to 2009 fishing seasons. 2008 data for surveyed beds were not available because the live-linked table in 2008 quotas automatically updated to 2009 values at the time of writing.

Quota Year	Quota Calc Region	Number of Surveyed Beds w Density Category	Number of Surveyed Beds Total	% Surveyed beds w Density Category	All Beds w Density Category	All Beds Total	% All Beds w Density Category
2007	Haida Gwaii	111	133	83.5	279	419	66.6
	Prince Rupert	N/A	N/A	N/A	N/A	N/A	N/A
	CentralCoast	142	161	88.2	520	686	75.8
	WCVI	16	51	31.4	52	279	18.6
	Area24	14	21	66.7	29	117	24.8
	GeorgiaStrait	8	25	32.0	49	262	18.7
	Area 12	0	9	0.0	0	98	0.0
	Overall	**291**	**391**	**74.4**	**929**	**1861**	**49.9**
2008	Haida Gwaii	N/A*	N/A*	N/A*	303	416	72.8
	Prince Rupert	N/A*	N/A*	N/A*	349	436	80.0
	CentralCoast	N/A*	N/A*	N/A*	555	687	80.8
	WCVI	N/A*	N/A*	N/A*	144	280	51.4
	Area24	N/A*	N/A*	N/A*	62	115	53.9
	GeorgiaStrait	N/A*	N/A*	N/A*	84	256	32.8
	Area 12	N/A*	N/A*	N/A*	5	98	5.1
	Overall	N/A*	N/A*	N/A*	**1502**	**2288**	**65.6**
2009	Haida Gwaii	146	161	90.7	308	437	70.5
	Prince Rupert	135	141	95.7	349	436	80.0
	Central Coast	145	162	89.5	556	687	80.9
	WCVI	43	55	78.2	145	280	51.8
	Area 24	20	22	90.9	61	116	52.6
	Georgia Strait	17	25	68.0	83	255	32.5
	Area 12	3	27	11.1	6	99	6.1
	Overall	**509**	**593**	**85.8**	**1508**	**2310**	**65.3**

*: Data un-available for 2008 quota calculations.

Table 9: Definition of Sub-Bed Status Codes, amount and percentage of bed area, potential quota and estimated geoduck biomass by sub-bed status code. Quota and Biomass are mean estimates derived from density-categorized values, when available.

Sub-Bed		Open or	Bed Area		% Potential	%
Status	Description	Closed	Area (Ha)	%	Quota	Biomass
0	Status Unknown or Unassigned					
1	Acceptable for Quota Calculations	Open	17,772	80.6	86.4	85.3
2	Closed seasonally	Open	116	0.5	0.8	0.8
3	Bed partially in closed area	Open	412	1.9	1.0	1.1
4	Test Area; not fishable or marketable.	Closed	17	0.1	0.1	0.1
5	Park / Reserve	Closed	185	0.8	0.1	0.4
6	First-Nations concern	Closed	38	0.2	0.2	0.2
7	Contaminated	Closed	181	0.8	0.7	0.7
8	Research, DFO or Bamfield	Closed	152	0.7	0.2	0.3
9	Not Commercially Feasible (logistically)	Closed	168	0.8	0.4	0.4
10	No Landings recorded in Harvest Logs.	Open	72	0.3	0.2	0.2
11	Offered up as Aquaculture sites, but not tenured yet	Open	353	1.6	0.4	0.4
12	Ex-wild bed, now tenured for aquaculture	Closed	147	0.7	0.1	0.1
13	New bed created via QTC survey (No Landings)	Open	2	0.0	0.0	0.0
20	Otters - impact unknown or minimal	Open	2,423	11.0	9.4	10.2
21	Otters - some impact	Open				
22	Otters - zero quota	Closed				
99	Obsolete Sub-Bed, no longer in existence	Closed				
	Total Open (with Status 20 Open)		**21,150**	**96.0**	**98.2**	**97.9**
	Total Closed (with Status 20 Open)		**887**	**4.0**	**1.8**	**2.1**
	Total Open (with Status 20 Closed)		**18,727**	**85.0**	**88.8**	**87.7**
	Total Closed (with Status 20 Closed)		**3,310**	**15.0**	**11.2**	**12.3**

Table 10: Number and percent of geoduck sub-beds, bed area, potential quota and biomass in areas where sea otters are present. Potential quota and biomass are based on mean estimates using the density categorized values where available.

Management Region		# of Sub-Beds		Bed Area		% Potential	%
		#	%	Ha	%	Quota	Biomass
CC	Otters	402	36.0	626	28.9	34.3	34.2
CC	All	1118		2,164			
WCVI	Otters	449	46.6	1,798	33.0	19.4	18.7
WCVI	All	964		5,449			
CC and	Otters	851	40.9	2,423	31.8	31.9	27.5
WCVI	All	2082		7,613			

Table 11: Number and percent of geoduck sub-beds, bed area, potential quota and biomass in Conservation Closures (below LRP). Potential quota and biomass are based on mean estimates using the density categorized values where available.

Licence Region	Management Region	Bed Area Closed		% Potential	%
		Ha	%	Quota	Biomass
N	HG	34.2	1.4	0.3	0.3
N	PR	15.2	0.6	0.2	0.2
N	CC	26.2	1.2	0.6	0.6
W	WCVI	1193.7	21.9	11.6	10.5
G	IW	758.6	8.0	4.2	4.2
Coastwide		**2027.9**	**9.2**	**1.6**	**2.6**

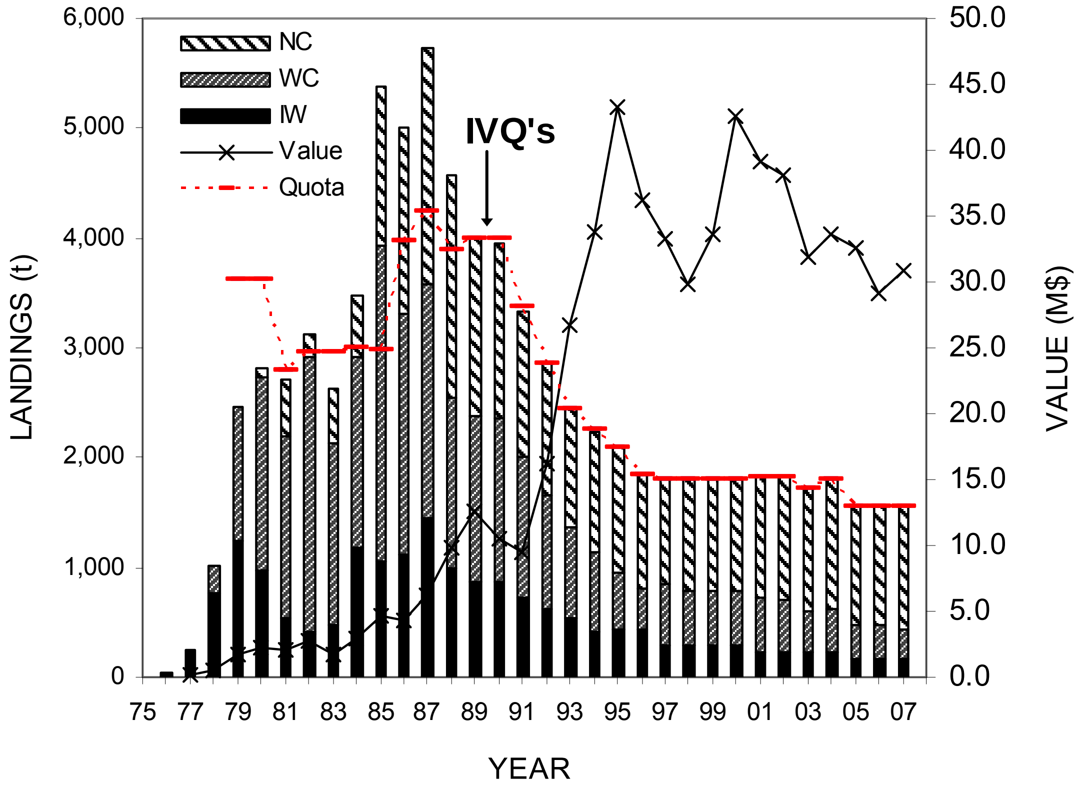


Figure 1: History of BC geoduck fishery landings (t) by Management Region (NC= North Coast, WC= West Coast of Vancouver Island, IW= Inside Waters), coast-wide quotas (t) and landed value ($million).

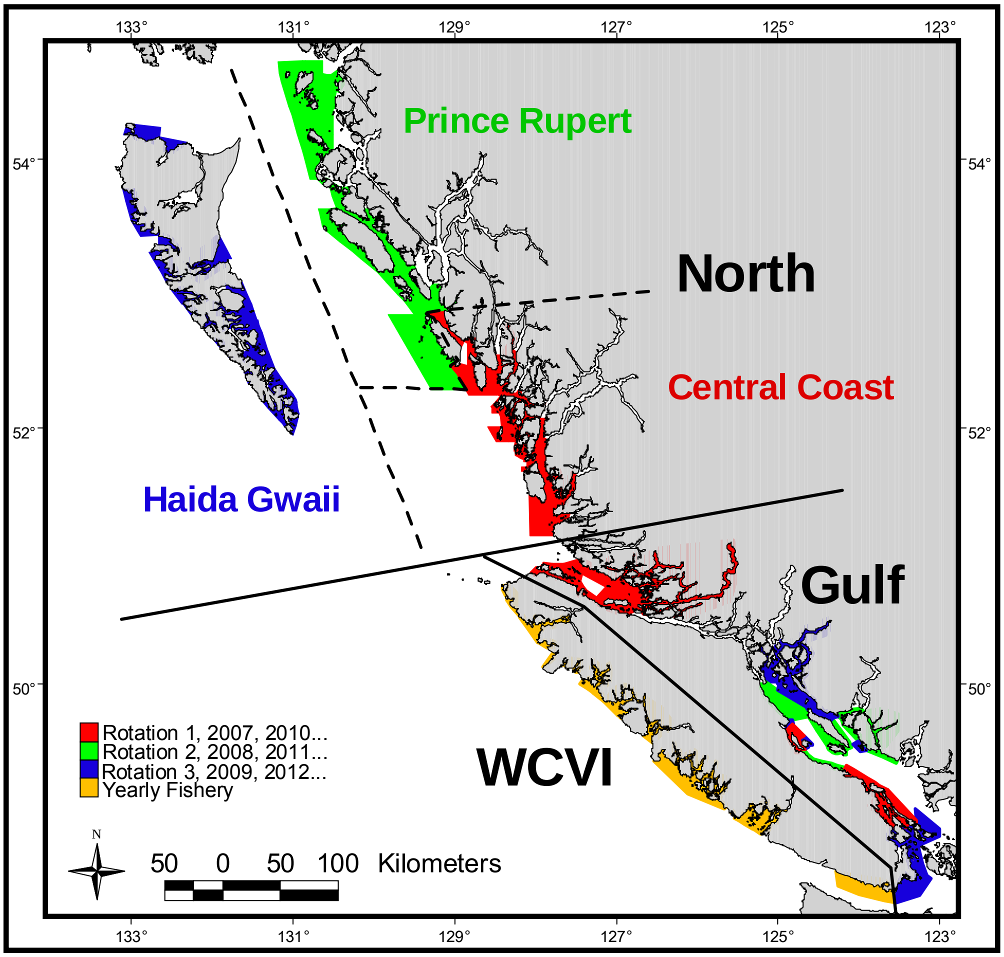


Figure 2: Map of BC coast showing geoduck "Licence Areas" (black letters, separated by solid lines) and "Rotational Areas" (North Coast rotational areas divided by dashed lines).

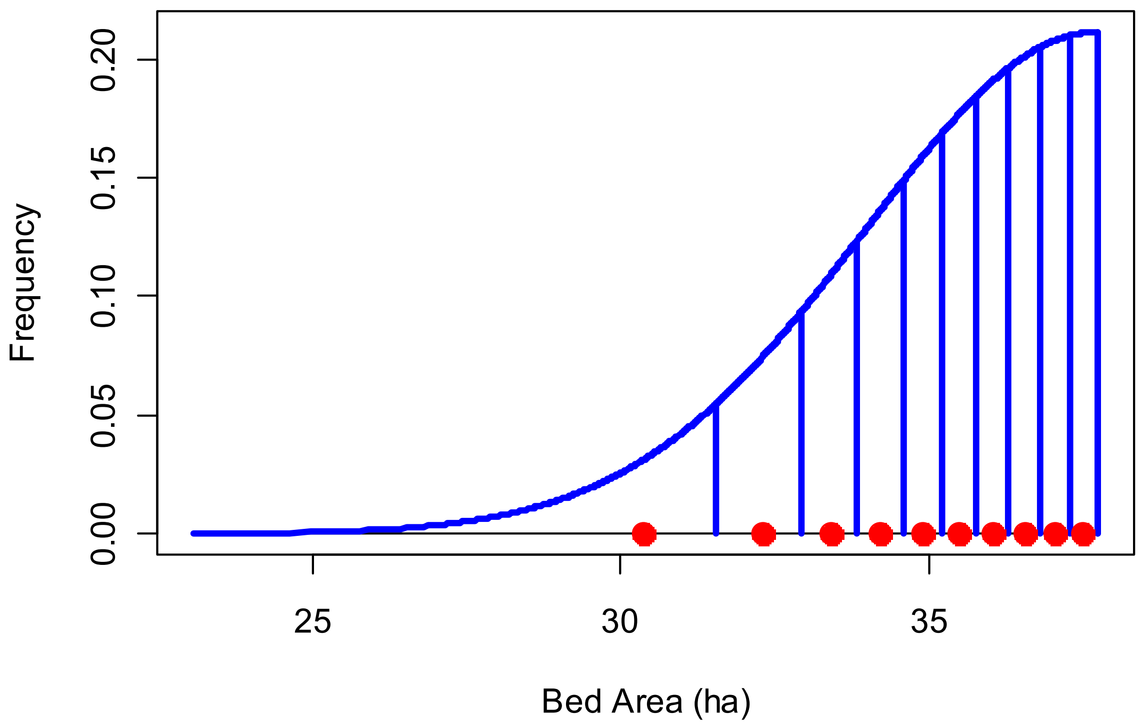


Figure 3: Distribution of values of bed area used in creating Array A, for an example bed of 37.8 ha, in the discretization process. The figure shows only 10 values for clarity, whereas the routine uses 100 values. Red dots represent the 10 values. The area under the curve for each segment is equal (equiprobable).

7 APPENDICES

7.1 APPENDIX 1 – REQUEST FOR WORKING PAPER

REQUEST FOR SCIENCE INFORMATION AND/OR ADVICE

PART 1: <u>DESCRIPTION OF THE REQUEST – TO BE FILLED BY THE CLIENT REQUESTING THE INFORMATION/ADVICE</u>

Date *(when initial client's submission is sent to Science) (dd/mm/yyyy)*:

Directorate, Branch or group initiating the request and category of request	
Directorate/Branch/Group	Category of Request
☒ Fisheries and Aquaculture Management	☒ Stock Assessment
☐ Oceans & Habitat Management and SARA	☐ Species at Risk
☐ Policy	☐ Human impacts on Fish Habitat/ Ecosystem components
☒ Science	☐ Aquaculture
☐ Other (please specify):	☐ Ocean issues
	☐ Invasive Species
	☐ Other (please specify):

Initiating Branch Contact:	
Name: Juanita Rogers/Rick Harbo	Telephone Number: (250) 756-7325
Email: juanita.rogers@dfo-mpo.gc.ca	Fax Number: (250) 756-7162

Issue Requiring Science Advice (i.e., "the question"): *Issue posed as a question for Science response.*
Document and evaluate the assessment framework currently in use, identify and prioritize future research, suggest changes to current framework to ensure a precautionary approach and sustainable fishing practices can be maintained.

Rationale for Advice Request:

What is the issue, what will it address, importance, scope and breadth of interest, etc.?

- In 1989, IVQs, a 3-year rotational fishery, and a North Coast On-Grounds Monitor were implemented in the geoduck fishery to provide greater stability, and improve the accuracy and timeliness of catch and effort data.
- Limit Reference Points (50% of original biomass, conservation closures) were implemented in 1994, and annual harvest rates were set at 1% of the estimated original (pre-fishery) biomass, identified by calculating the area of harvestable beds multiplied by estimated original density of geoducks on each bed.
- From 1997 to 2000 quotas remained unchanged.
- The quota calculation process was documented in the Stock Assessment Framework produced in 2002.
- Through 2005, the quota fluctuated following PSARC review of quota options, concerns about the West Coast of Vancouver Island stock, re-evaluation of bed status information, and a thorough bed-by-bed review in 2004.
- An improvement to biomass and quota calculations was presented to PSARC in 2005, using estimates of current biomass rather than original biomass, and implementing a lower limit reference point (40% of original biomass).
- In 2006, some areas were managed under the ‘original biomass’ model, while others were managed under the ‘current biomass’ model.
- In 2007 and 2008, all areas were managed under the current biomass model. Decision rules for allocating quota based on Science estimates were implemented.
- The new method of calculating and allocating quotas needs to be documented.

Questions to be Addressed:

Quota Calculation:

- Document current protocols for the collection and analysis of data on geoduck density, average geoduck weight and bed area and describe decision rules for applying all sources of data to geoduck biomass estimation and quota calculation.
- Describe and evaluate data precision, accuracy and sources of bias, and make recommendation, where necessary, for improved methods.

Survey Design:

- *Document current protocols for collecting geoduck density data through dive surveys.*
- *Evaluate current protocols for precision, accuracy and bias, and make recommendation, where necessary, for change.*

Reference Points:

- Evaluate utility of the 40% Bv Limit Reference Point with respect to the precautionary approach and sustainability principles and make recommendations where necessary.

Future Directions:

- Note any recommendations for ongoing work, future analyses and/or new request for advice.

Possibility of integrating this request with other requests in your sector or other sector's needs?

N/A

Intended Uses of the Advice, Potential Impacts of Advice within DFO, and on the Public:

Who will be the end user of the advice (e.g. DFO, another government agency or Industry?). What impact could the advice have on other sectors? Who from the Public will be impacted by the advice and to what extent?

Resource Managers, Research Scientists, Commercial Industry Association

No impact anticipated; documenting process.

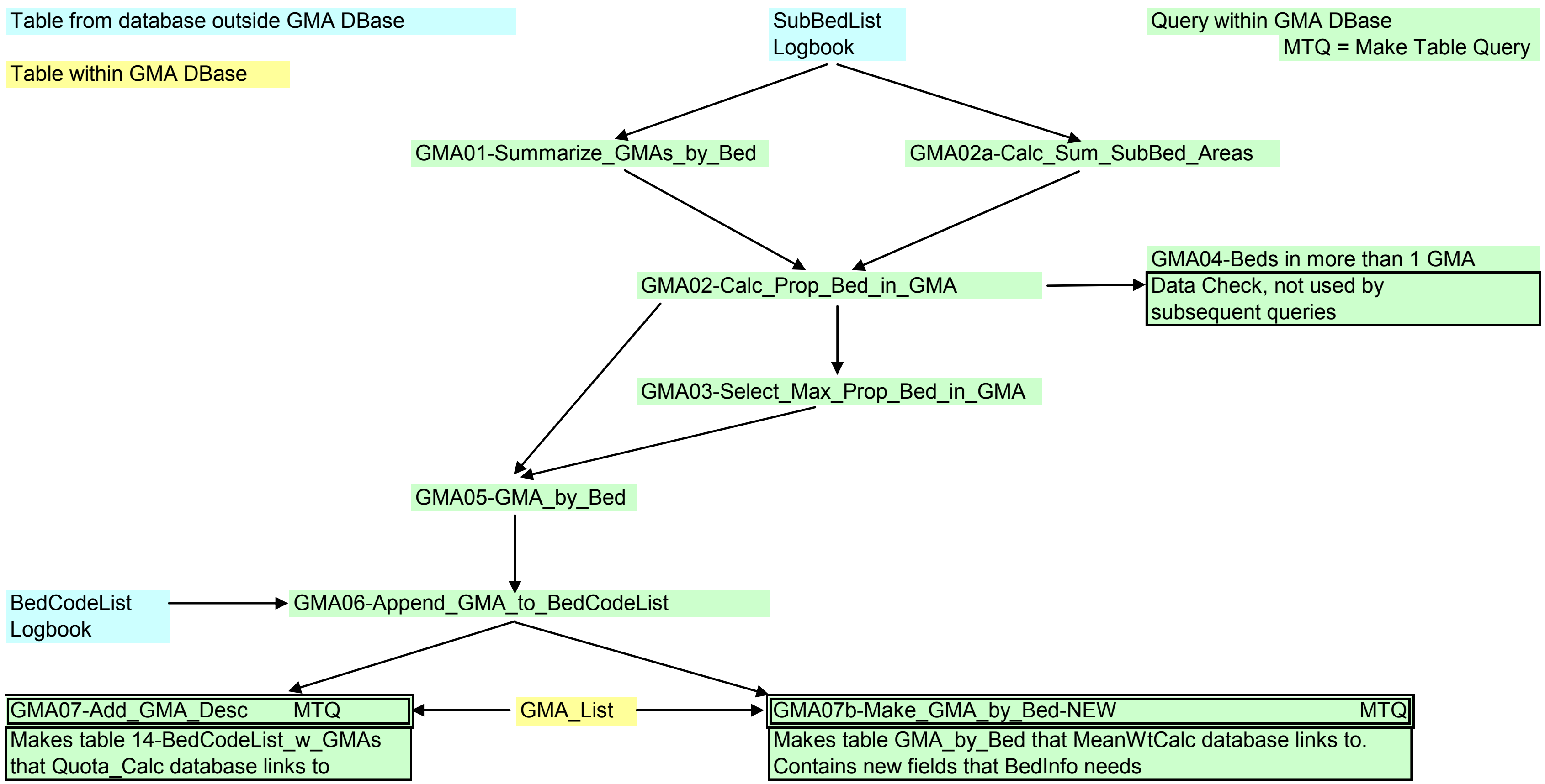
Date Advice Required:

Latest possible date to receive Science advice (dd/mm/yyyy): 26/11/2008

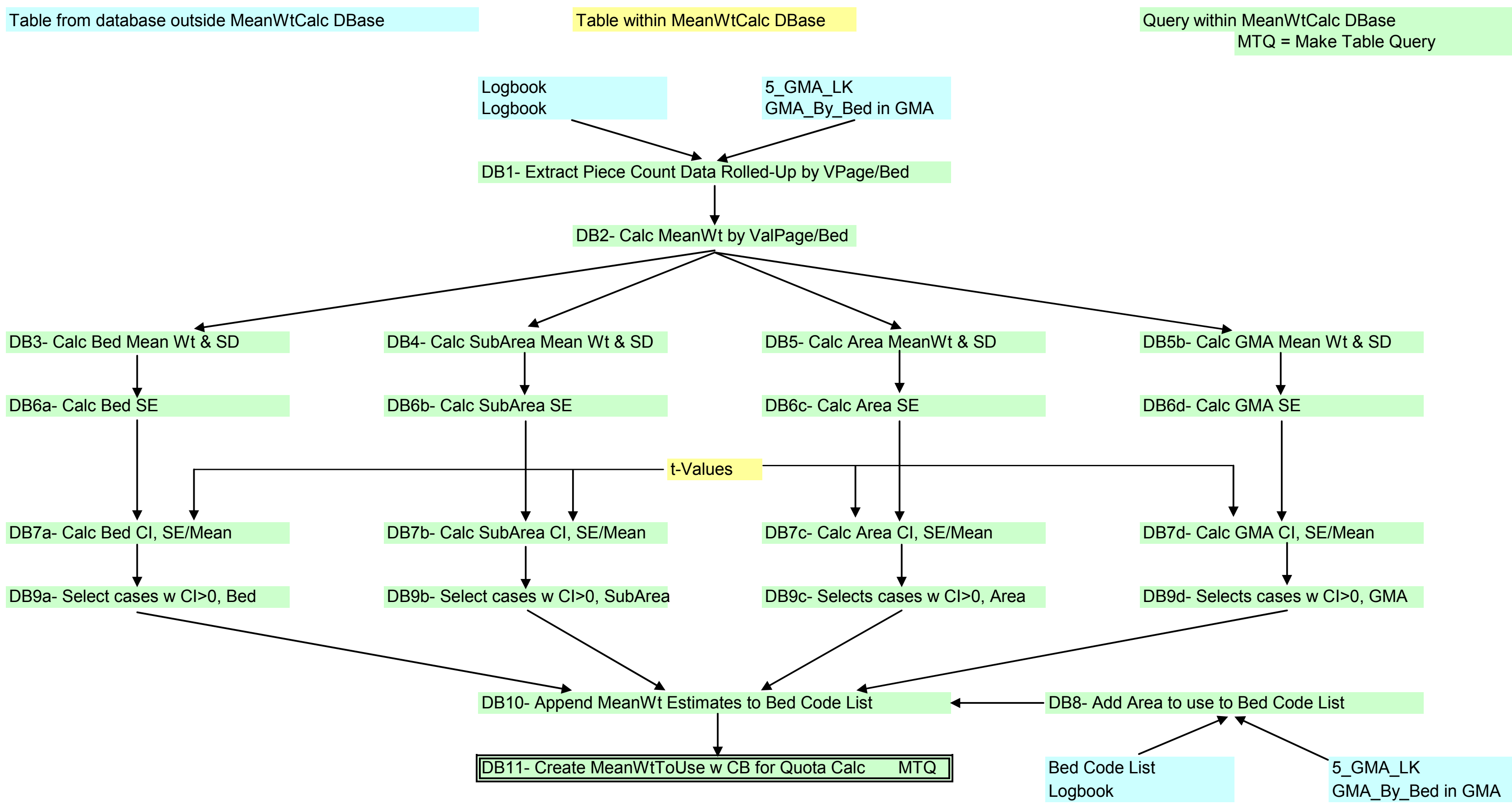
Rationale justifying this date: Scheduled Fall 2008 Invertebrate Subcommittee; paper then available to support 2009 decision-making.

7.2 APPENDIX 2 – QUOTA CALCULATION DATABASE FLOW-CHARTS

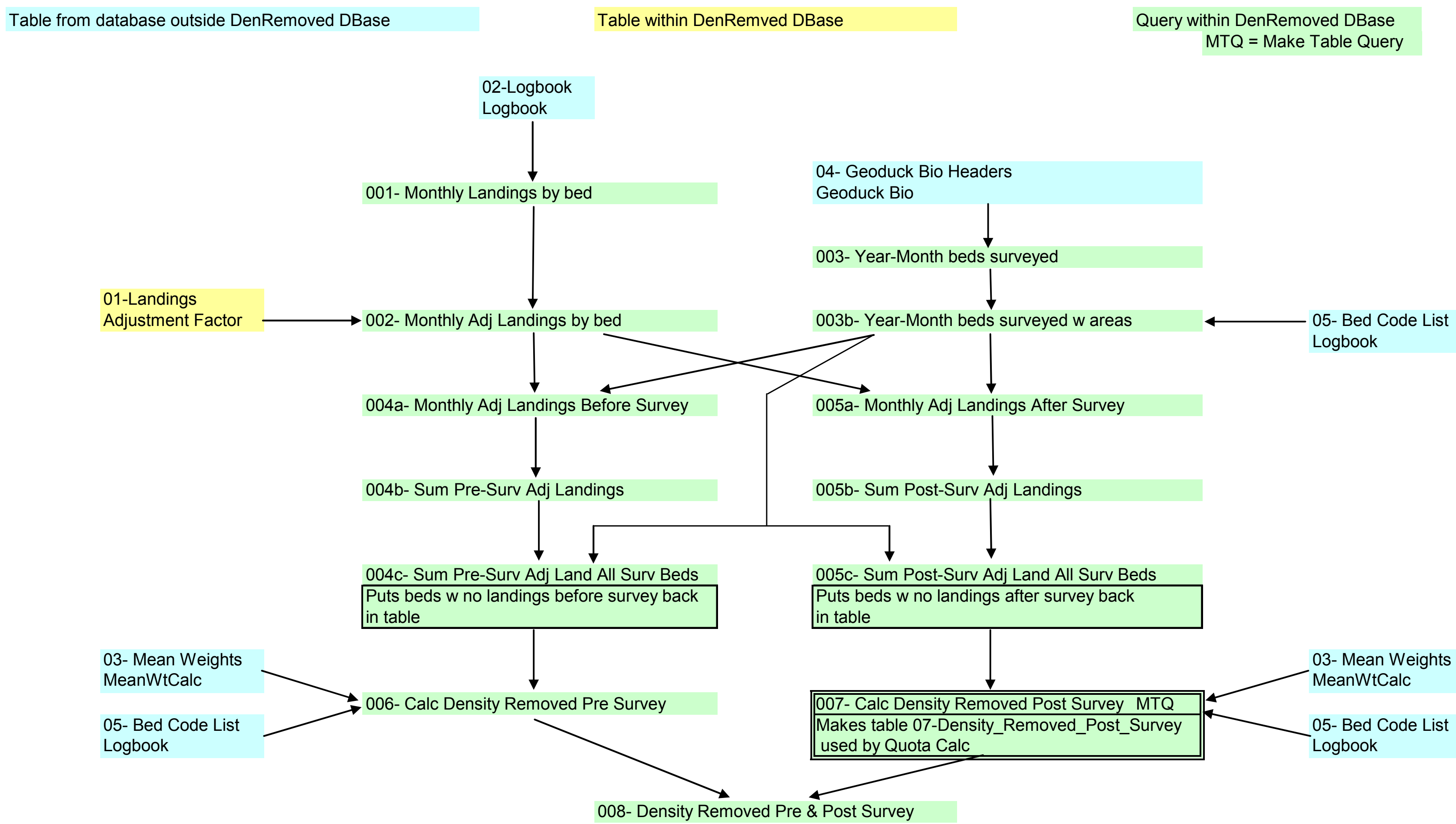
7.2.1 Appendix 2.1- GMA Database Flow Chart



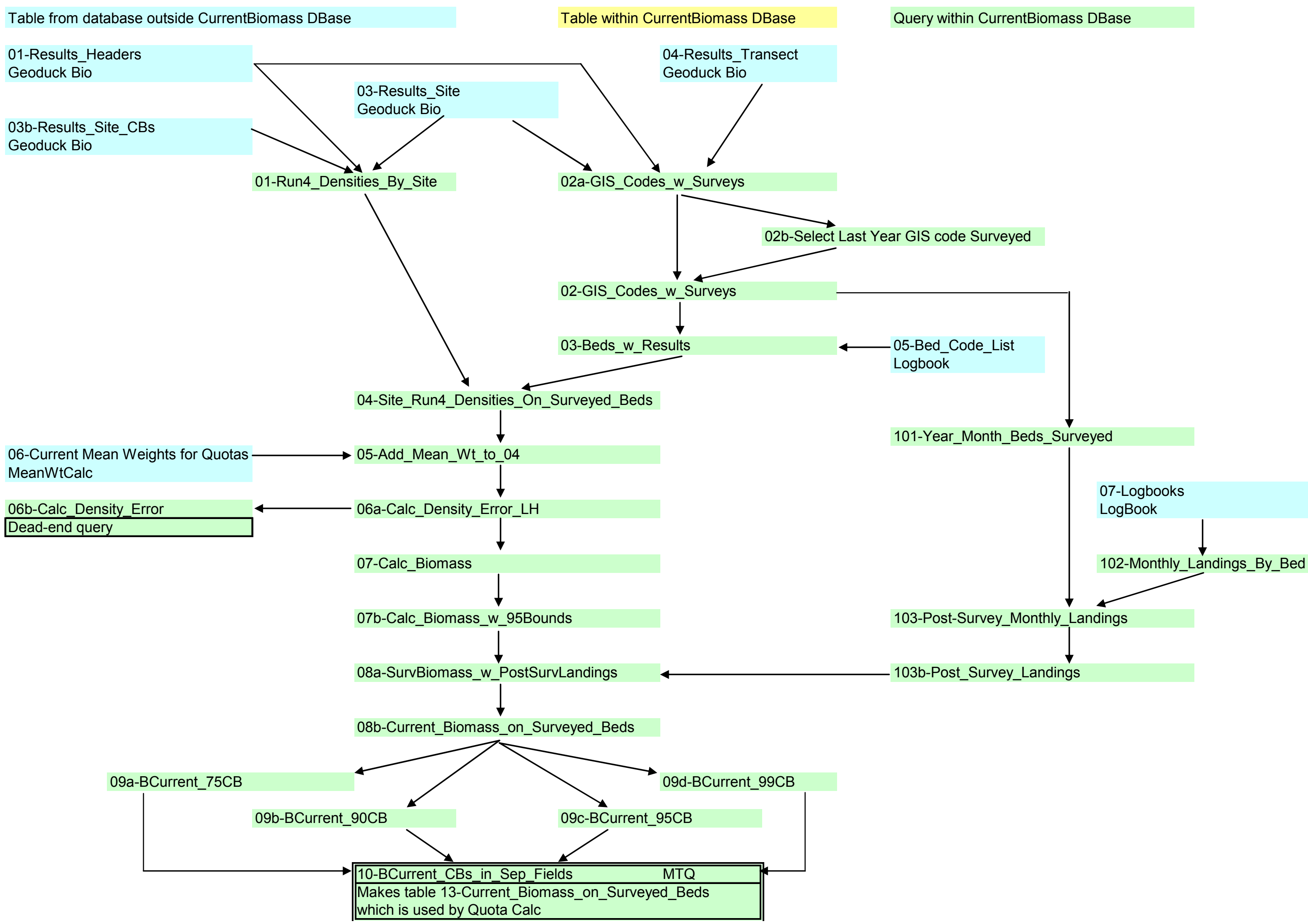
7.2.2 Appendix 2.2- Mean Weight Calculation Database Flow Chart



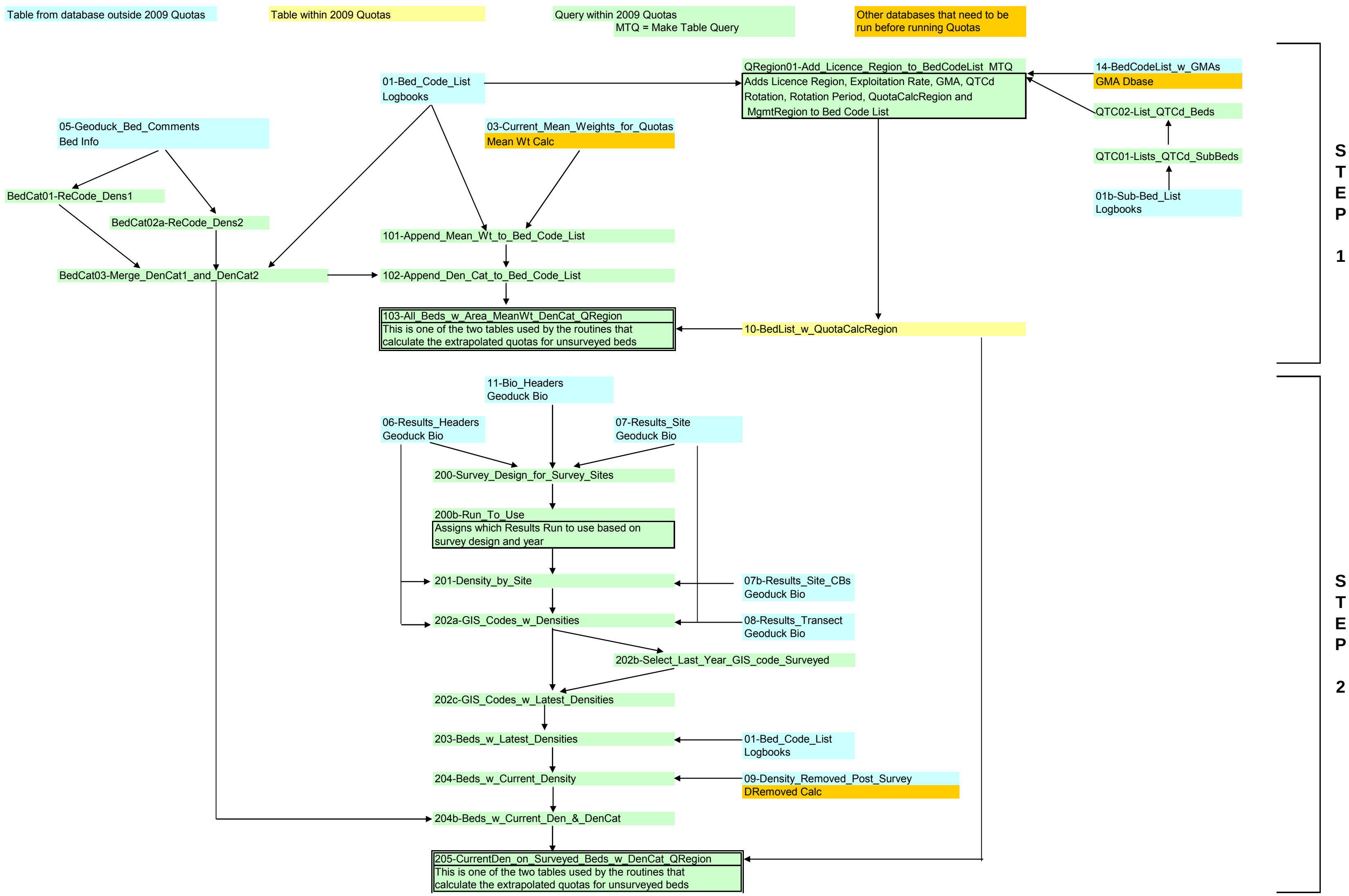
7.2.3 Appendix 2.3- Density Removed Database Flow Chart



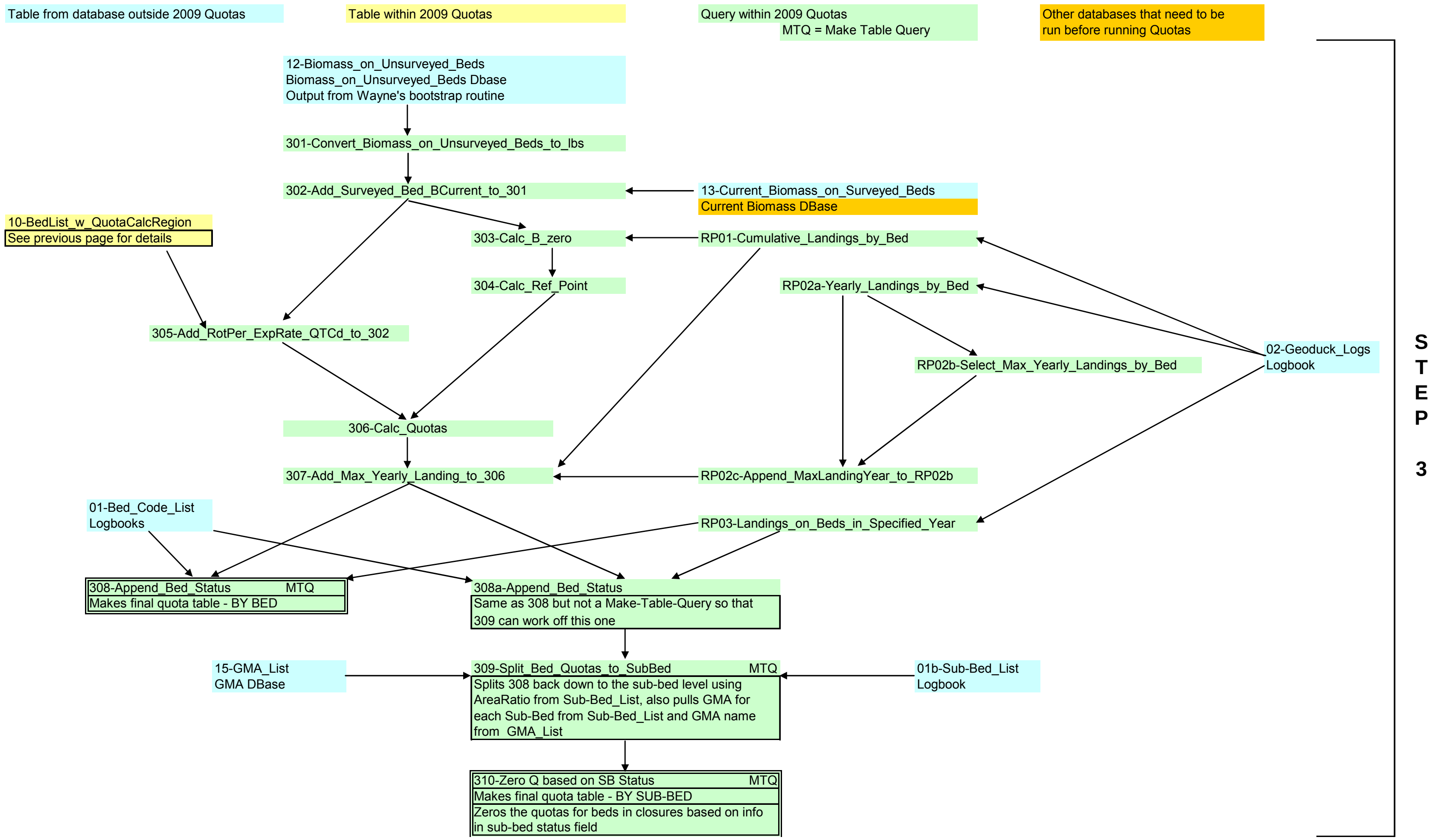
7.2.4 Appendix 2.4- Current Biomass Database Flow Chart



7.2.5 Appendix 2.5- Quota Calculation Database Flow Chart – Steps 1 & 2



7.2.6 Appendix 2.5- Quota Calculation Database Flow Chart – Step 3



7.3 APPENDIX 3 – COMPARISON OF GEODUCK MEAN WEIGHT ESTIMATES FROM COMMERCIAL LOGBOOK DATA AND BIOLOGICAL SAMPLE DATA

7.3.1 Introduction

Estimates of geoduck biomass are calculated as the product of estimated geoduck density, bed area and mean weight. From the 2001 fishing season to present, mean weights of geoducks have been estimated from logbook data. The use of logbook data to estimate mean geoduck weight has been questioned because, if size-selectivity is occurring in the fishery, biomass could be overestimated and quotas inflated. Size-selection while harvesting is considered to be low, since the size of a geoduck cannot be determined until it has been pulled out of the substrate. Size selection by avoiding harvest sites with small clams is more likely, but the extent to which it occurs is not known and would be difficult to determine.

The mean weight of a geoduck population may be better represented by biological sample data that are collected during surveys. The goal of biological sample collections is to obtain a complete population age-distribution for use in calculations of growth and mortality rates, in recruitment studies, etc. To achieve this goal, divers are instructed to collect all geoducks seen during the sampling process, irrespective of size. Small juvenile geoducks that incidentally pop out of the ground when a nearby adult is being harvested are also collected, but most of the smallest clams present in a population will not be detected and therefore will be under-sampled. True, un-biased, estimates of the geoduck mean weight in a given population are therefore not obtainable by using the size-selective traditional sampling methods. The only non size-selective method of collecting geoducks is the venturi dredge, however the method is time-consuming and not effective for broad-scale sampling.

Ideally, the estimates of mean weight used in geoduck biomass calculations should reflect the mean weight of geoducks that are counted on density surveys, since that is the detectable, and therefore fishable, population. The under-representation of juvenile geoducks in the biological sample is not a concern for harvestable biomass calculation because these small clams are usually too small to have visible siphon-shows and are unlikely to be seen by the survey divers as they count the animals. All we need concern ourselves with is that we have a reasonable estimate of the mean weight of counted animals that comprise the density estimate.

The objective of this analysis is to compare the two sources of geoduck weight data to determine whether the mean weight of harvested geoducks differs from the estimated mean weight of a given population from biological samples, whether there is evidence of size selectivity and whether there is a need to develop a correction factor for the more-extensive logbook dataset. Prior to comparing mean weight from both sources, biological sample data were corrected for the amount of water lost between the dock and the processing plants. The advantages and drawbacks of each mean weight calculation method are described and recommendations for further work required to determine representative and unbiased weight estimates are made.

7.3.2 **Methods**

7.3.2.1 Logbook Data

Multiple geoduck beds can be harvested on a single day by any given vessel. Harvest is recorded on geoduck harvest logs, with one day of activity per ‘validation page’ per vessel. Data recorded includes the number of geoducks (piece count) and cages landed, by bed, and the total weight of landings. The catch is packed into cages (rectangular milk crates) on the vessel and transported to designated ports where weights are recorded by a third-party validation company. After the dock-side validation, the weight of the cages and liners are subtracted from the gross weights to yield the net weight of geoducks, by validation page. The weight landed from each bed, by a given vessel on a given day, is apportioned according to number of cages landed from each bed. Since 1997, harvesters have indicated on their logs whether the number of geoducks counted was true or only an estimate. Logbook data where true piece counts were recorded were used to calculate geoduck mean weights.

Catch weight and piece count data were extracted from the geoduck logbook database on a by-validation page and bed basis. Mean weight for each page/bed combination was calculated by dividing the net weight of the catch by the number of geoducks landed. Generally, multiple values of mean weight were available for a given bed since more than one vessel can harvest the bed and/or the bed can be harvested on multiple days. The mean geoduck weight by bed was then calculated as the mean of the page values of mean weight for each bed.

7.3.2.2 Biological Sample Data

Biological samples, of approximately 300 to 500 geoducks each, have been collected during geoduck surveys from 1993 to present: data from samples collected between 1993 and 2005 were used in this analysis (Bureau et al. 2002, 2003). Samples were collected using standard geoduck commercial harvesting gear.

Sampled geoducks are packed into cages on the sampling vessels and weighed at the landing port on a by-cage or combined-cage basis before being shipped to processing plants in Vancouver. Net dock weight was obtained by subtracting the weight of the cages and liners from the gross dock weight. At the plant, geoducks were individually weighed by staff of Archipelago Marine Research and the mean net plant weight was calculated by averaging the individual geoduck weights measured at the plant.

7.3.2.3 Correction for Water Loss in Biological Samples

Mean weight from logbook data and biological samples are not directly comparable. Geoducks in biological samples experience more water loss due to the longer transport time in reaching processing plants in Vancouver compared to the closest port for the wild fishery. The first step towards making mean weight estimates from logbooks and biological samples comparable is to calculate the amount of water loss between the dock and the processing plant for each biological sample. The ratio of dock weight to plant weight was calculated for each cage and then averaged for each biological sample.

$$CF_s = Average\left(\frac{DW_c}{PW_c}\right) \qquad \text{Equation A1}$$

where CF_s is the correction factor for sample *s*, DW_c is the dock weight of cage *c* and PW_c is the plant weight of cage *c*, calculated as the sum of the individual geoduck weights measured at the plant in cage *c*. Sample-specific water loss correction factor values ranged from 0.991 to 1.140 (Table A1).

The sample-specific water loss correction factor was then used to convert the individual geoduck plant weights to dock-weight equivalents.

$$DWE_g = PW_g \times CF_s \qquad \text{Equation A2}$$

where DWE_g is the Dock Weight Equivalent of geoduck *g* and PW_g is the individual plant weight for geoduck *g*. If a sample-specific water loss correction factor (CF_s) was not available, the average conversion factor for the Region that the sample came from was used (Table A2). Regions were: Haida Gwaii, Prince Rupert, Central Coast, Inside Waters, WCVI and Area 24. Only five samples used a regional water loss correction factor. Regional water loss correction factors ranged from 1.025 to 1.066, with a coastwide average of 1.050. Dock weights were thus, on average, 5.0% heavier than the plant weights due to water loss occurring in transit between the dock and the plants.

The corrected (dock-weight equivalent) mean weight from biological samples was then calculated, by bed. For the purpose of the comparisons, analysis included only those beds where both a biological sample-estimate of mean weight and a bed-specific logbook-estimate of mean weight were available. Of the 112 beds with biological sample data, 89 had a bed-specific logbook estimate of mean weight (Table A4) and those beds formed the basis of analyses to compare the mean weight estimates.

7.3.2.4 Size selectivity

It is expected that biological sample mean weights, which include some pre-recruits (pop-ups), will be lower than mean weights estimated from commercial catch. However, the question of interest is to determine whether there is evidence of size selection of the visible population. For this purpose, biological sample data were truncated to exclude pre-recruited animals, those not vulnerable to the fishery, from the biological sample dataset and a second set of analyses was performed comparing logbook weights to truncated biological sample weights.

Two pre-recruit thresholds were investigated: one based on weight and the other on age. In two experimental plots, geoducks from the WCVI and the Strait of Georgia (Ritchie Bay and Marina Island) were determined to be 100% vulnerable to fishing at ages 6 and 7, respectively, based on neck weight data (Campbell et al. 2004). Geoduck growth rates vary between locations (Bureau et al. 2002, 2003) and vulnerability is more related to geoduck weight than age. Using total weight as a measure of vulnerability may be more appropriate than age. The mean weight of age 6 geoducks from Ritchie Bay and age 7 from Marina Island was 462.2 g, and this was used as the first pre-recruit threshold. The second threshold investigated was based on age. Bradbury and Taggart 2000 (cited in Orensanz et al. 2004) reported that age selectivity (of gear) is unlikely for geoducks older than 4-10 years. For practical purposes, it can be assumed geoducks are fully recruited by age 8-10 (Orensanz et al. 2004). Ten years was thus chosen as the second pre-recruit threshold (which corresponds to a mean weight of 719 g from all biological samples). Mean weights from biological samples were re-calculated after excluding weight data less than or equal to 462.2 g (before correction for water loss) and age data less than or equal to 10 yr.

7.3.2.5 Data Analysis

Mean weight from logbooks were compared to the various estimates of mean weight from biological samples (all sizes, > 462g and >10 yrs), by region and coast-wide, and tested for significance using paired t-tests. Due to the large number of tests conducted, we used 0.01 as the statistical significance level to minimize false positive p-values. Ratios of logbook mean weight to biological sample mean weight were calculated on a by-bed and by-region basis.

7.3.3 Results

7.3.3.1 Data Coverage

Logbook data provided mean weight estimates for 557 beds, or 23.9% of the beds (Table A3) while biological sample data provided mean weight estimates for 112 beds, or 4.8% of the beds. Logbook data thus yields five times more bed-specific estimates of mean weight than biological sample data. The logbook data included a measure of over 14 million geoducks from 26,033 validation events, while biological samples provided data on only 23,602 geoducks, a 600-fold difference.

7.3.3.2 Mean Weight Comparisons

Mean weight estimates from logbooks and from water loss-corrected biological samples are presented in Table A4. Using the full biological sample datasets, paired t-tests showed that mean weight from logbooks were significantly higher than mean weight from biological samples coast-wide and for all regions separately ($p<0.001$), except for the Central Coast ($p= 0.425$) and Area 24 ($p=0.618$) (Table A5). In comparisons using datasets that excluded geoducks 462g or less from the biological samples, paired t-tests showed that mean weight from logbooks was significantly higher than mean weight from biological samples coast-wide and for all regions ($p<0.005$) except the Central Coast ($p= 0.960$), Area 24 ($p=0.868$) and WCVI ($p=0.101$). Tests excluding geoducks 10 years or younger from the biological data showed that mean weight from logbooks was still significantly higher than mean weight from biological samples coast-wide ($p=0.004$), but on a Regional basis, only Haida Gwaii ($p<0.001$) and Prince Rupert were significantly different ($p=0.010$, Table A5).

The overall coastal ratio of the mean weight from logbooks to mean weight from biological samples (Log/Bio) with no pre-recruit cut-off was 1.131. Regional Log/Bio ratios ranged from 1.036 in the Central Coast to 1.265 in the Inside Waters, i.e., mean weights from logbooks were from 3.6% to 26.5% heavier than mean weights from biological samples (Table A4). The large difference between estimates in the Inside Waters was mostly due to three beds near Lund (Savary Island) where the Log/Bio ratio ranged from 1.629 to 1.827. When the Lund beds were excluded, the Inside Waters average Log/Bio ratio dropped to 1.203. The difference in mean weights at Lund can be explained by a strong recruitment event that occurred on sampled beds between the years when most of the harvest occurred and the year that the biological sample was collected.

Because strong recruitment events can influence population mean weight, the analyses were re-run using only logbook data from the same year that a biological sample was collected for any given bed. A total of 32 beds had commercial landings in the same year that a biological sample was collected (Table A6). Mean weight from logbooks was significantly greater than mean weight from biological samples on a coast-wide basis and for the Haida Gwaii region when no pre-recruit cut-off was applied (Table A7). When pre-recruits were excluded, using either the 462g or 10 year threshold, the Haida Gwaii Region continued to be the only Region where weight estimates were significantly different at the 0.01 level while the coastwide difference was no longer significant. Note that no data were available for the Prince Rupert Region in this analysis.

7.3.4 Discussion

Geoducks are challenging animals to survey and sample because both activities require that individual clams be visible. The estimate of mean geoduck weight that is used to calculate harvestable biomass and fishery quotas should be representative of population of geoducks counted by divers during dive surveys, since that is the basis of the density estimate. Concerns have been raised that mean weights estimated from logbook data may not be representative of geoducks counted during density surveys, due to alleged size selectivity of the fishery, and that biological samples may be a more representative source of mean weight data to use in biomass calculations. The notion that biological samples provide more accurate estimates of mean weight for biomass calculations relies on the assumption that biological samples are more representative of geoducks counted during dive surveys than the commercial catch.

Even if the smallest clams in a population are not detectable, geoduck surveys in areas of high recruitment (e.g. Bartlett Island 2008) have shown that survey divers can detect at least some geoducks that would either not be seen or be avoided by harvesters (Bureau, unpublished data). This observation lends support to the belief that the estimate of mean weight from biological samples is more representative of the population from which density is estimated than the mean weight estimate from logbook data. However, the extent of biological sampling is more limited in terms of the number of beds sampled, the number of sampling locations within a bed and the number of animals sampled. Since geoduck populations do not have a homogenous size-distribution, it is possible for a biological sample to be collected from a pocket of light-weight recently-settled clams or, conversely, from an area where recruitment has been low and the population is made up of older, heavier clams. Logbook data has higher spatial resolution than biological data. Logbook records describe many more fishing locations within a given bed (minimum 10) than do biological samples (maximum four) and thus may better capture the size-distribution over the whole bed. Also, logbook-derived mean weights are based, on average, on over 20,900 geoducks per bed, while only 150 to 500 geoducks per bed are collected in biological samples. Furthermore, logbook data provide a wider geographical coverage and include more beds than do biological samples. These are compelling reasons to utilize logbook data for mean weight estimation.

Choice of the most appropriate data source to use for estimating geoduck mean weight is a trade-off between representativeness of the data and the amount of data available. From the standpoint of data availability and spatial detail, using mean weights from logbooks is preferable, but the biological sample mean weight is likely a more accurate measure of the population.

Differences observed in mean weight estimates between the two sources are expected since biological samples do contain some pre-recruits which are not normally found in commercial catches. The purpose of the comparison, with pre-recruits excluded from the biological samples, was to determine whether a higher level of size selectivity was occurring in the fishery. If differences are found when pre-recruits are excluded from the biological samples, then evidence for size selectivity in the wild fishery is stronger.

Coastwide, mean weight from logbooks was significantly higher than mean weights from truncated biological samples, suggesting that some size selection may be occurring in the fishery, most likely through harvest site selection. At a regional level, however, the mean weights were not always significantly different, suggesting that size selection may not be occurring in all regions of the BC coast. For the Haida Gwaii and Prince Rupert regions, mean weights from logbooks were significantly greater than mean weights from biological samples, even when geoducks 10 years or younger were excluded, suggesting that size selectivity may be occurring in those two regions.

Applying a correction factor to mean weight estimates from logbooks for regions where size selectivity appears to occur may be the best option for providing wide geographical coverage and mean weight estimates that are representative of the geoducks that are counted on dive surveys. The choice of a correction factor should take into account the size-range of geoducks that are not harvested but likely to be counted on density surveys. Using the ratio of mean weight from logbooks to the mean weight from biological samples (all sizes) may initially seem to be the most appropriate, and the most conservative option with which to correct mean weights from logbooks. However, other factors besides size selectivity can explain differences between logbook and biological samples weights. The potential impact of recruitment on mean weight cannot be ignored, given the sporadic nature of geoduck recruitment and the increase in recruitment in recent decades.

Recruitment on a bed after the time when most commercial landings were harvested but before the collection of a biological sample can lead to large differences between the mean weight estimates for a given bed. The difference observed on Lund beds, for example, was largely due to recruitment. In the biological sample taken from Lund in 2004, 50% of geoducks were from the 1994 year-class or later which would not have recruited to the commercial fishery until 2001-2004, after most of the landings on the Lund beds had occurred. Similarly, for one of the Hornby Island beds, with a Log/Bio ratio of 1.551, 85% of geoducks found in the 2005 sample were of the 1992 year-class or later and were not fully vulnerable to the fishery in 1999 when most landings occurred.

The converse could also happen; the biological sample mean weight estimate from Elbow Bank in 1994 was higher than the estimate from logbooks which is based on harvest from 2000 to 2006. A recruitment pulse to the Elbow Bank bed after the biological sample was collected could explain the lower logbook mean weight observed. Mean weights from biological samples come from a single year and, while the mean weight from logbooks generally span several years of data, there are cases where most of the logbook data for a given bed came from only one or few years. The larger the time difference between the biological sample being collected and the years of harvest for a given bed, the more likely that recruitment events could influence the estimates.

Logbook mean weights get updated after every fishery season and reflect a potentially changing geoduck population structure, whereas biological sample mean weights represent a point estimate in time (except for a few beds sampled twice) and are thus more likely to miss changes in geoduck population structure. Updating the mean weight estimates from logbooks is a routine task performed each year whereby any bed harvested in the previous rotation will get a revised mean weight estimate. However, updating mean weight estimates from biological samples requires the collection of new samples. Historically no more than five biological samples have been collected in any given year due to sample processing time constraints. Biological sample data is therefore more difficult and costly to update than logbook data. Biological sample data is thus more likely to miss changes in geoduck population structure. Using logbook data therefore also seems preferable in terms of ease of updating estimates.

In the analysis of same-year data, only the Haida Gwaii region showed significant differences, with or without pre-recruit cut-offs. For one of the beds at Hippa Island, the ratio of logbook mean weight to biological sample mean weight (all sizes) was 1.499 (Table A6). In this case the discrepancy cannot be attributed to recruitment. However, the biological sample was collected from the bed on the North side of a point while commercial harvest was on the South side of the point, which is a good example of how mean weights can differ due to small-scale geographical differences in population size structure. In the same-year coast-wide analysis, there was a significant difference when pre-recruits were included in the analysis but when pre-recruits were excluded, using either cut-off, the estimates were not significantly different. These results therefore lend less support to the suggestion that size selection is occurring (beyond the exclusion of pre-recruits). Harbo et al. (1983) stated that there was no reason to suspect the fishery of age-selectivity among individuals fully recruited to the fishery.

Nevertheless, there seems to be enough evidence of some amount of size-selectivity that warrants the application of a regional correction factor to mean weight estimates from logbooks. However, taking into consideration the uncertainty in the data due to recruitment, small sample sizes and bias from limited spatial coverage in biological data, mean weight correction factors should be applied only for those regions where size selectivity by the fishery was demonstrated, i.e., Haida Gwaii and Prince Rupert. Furthermore, due to the uncertainties noted above, the ratio of mean weights from logbooks to biological sample mean weight using the 10yr pre-recruit cut-off is recommended to be used as a correction factor.

The correction factor should be applied as follows.

$$W_b = \frac{\overline{W}_b}{WCR_r} \qquad \text{Equation A3}$$

where W_b is the mean weight estimate to use for bed *b* in quota calculations, $\overline{W_b}$ is the mean weight for bed *b* calculated from logbook data and WCR_r , the Weight Correction Ratio for region *r,* is the ratio of logbook to biological sample mean weights using the 10yr pre-recruit cut-off (Table A5).

7.3.5 Recommendations

1- Continue to use mean weights estimated from logbooks to calculate geoduck biomass in order to take advantage of the wider spatial coverage (number of beds and number of locations within beds) and the greater amount of data (number of geoducks sampled), and the efficient computational process of updating data over time.

2- Regional correction factors should be applied (following Equation A3) to mean weight estimates from logbook data for regions where size selectivity was demonstrated, namely Haida Gwaii (WCR = 1.108) and Prince Rupert (WCR = 1.082).

7.3.6 References

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Table A1: Water loss correction factor, by survey.

Year	Survey Title	Water Loss Correction Factor	Water Loss Factor Source
Haida Gwaii			
1994	Burnaby Island	0.991	Survey
1995	Hotspring Island	1.044	Survey
1996	Houston Stewart Ch.	1.088	Survey
1997	Cumshewa Inlet	1.028	Survey
1998	Selwyn/Dana/Logan Inlets	1.037	Region
2000	Gowgaia Bay	1.050	Survey
2000	Hippa Island	1.059	Survey
2000	Tasu Sound	1.045	Survey
2002	Parry Passage	1.023	Survey
Prince Rupert			
1996	Otter Pass	1.027	Survey
1996	West Aristazabal Island	1.020	Survey
1997	Anderson/Laredo	1.044	Survey
1997	Principe Channel	1.044	Survey
1998	Dundas Island	1.031	Survey
1998	Moore Islands	1.068	Survey
2004	SE Prescott Island	1.071	Survey
2004	Tree Nob	1.029	Survey
Central Coast			
1993	Price Island	1.025	Region
1995	Goose/Wurtele/Seaforth	1.069	Survey
1995	Kitasu Bay	1.025	Region
1995	West Higgins Pass	1.025	Region
1996	S Bardswell/Prince Group	1.050	Survey
1998	Hakai Passage	1.032	Survey
2003	Spider Anchorage	1.030	Survey
2005	Ivory to Stryker	1.013	Survey
2005	Tribal to Nalau	1.023	Survey
Inside Waters			
1995	Duncan Island	1.130	Survey
1995	Goletas Channel	1.110	Survey
1996	Oyster River	1.071	Survey
1998	Comox 1998	1.055	Survey
1999	Thormanby Island	1.054	Survey
2000	Round Island	1.038	Survey
2001	Boatswain Bank	1.035	Survey
2002	Marina Island, 2002	1.140	Survey
2002	South Round Island	1.025	Survey
2003	Goletas Channel	1.072	Survey
2003	Ladysmith Harbour	1.066	Region
2004	Lund	1.094	Survey
2005	Hornby Island	1.031	Survey
2005	Maplegurad / Qualicum	1.057	Survey
West Coast Vancouver Island			
1996	Winter Harbour	1.065	Survey
1998	Kyuquot	1.043	Survey
2000	Barkley Sound	1.032	Survey
2000	Nootka Sound	1.035	Survey
2002	NE Barkley Sound	1.041	Survey
2002	Rolling Roadstead	1.072	Survey
2002	Winter Harbour, 2002	1.054	Survey
2003	Brady Beach, Bamfield	1.058	Survey
2003	Mission Group	1.065	Survey
2005	Barkley Sound	1.044	Survey
Area 24			
1994	Elbow Bank	1.034	Survey
1997	Millar Channel	1.063	Survey
1997	Yellow Bank	1.076	Survey
2003	Ritchie Bay	1.038	Survey
2004	Blunden / Bartlett Islands	1.053	Survey

Table A2: Mean regional water loss correction factor and number of pairs of dock weight to plant weight that were used to calculate the mean water loss correction factor (n). Coast Wide represents the pooling of all samples.

Region	n	Mean Water Loss Correction Ratio
Haida Gwaii	109	1.037
Prince Rupert	130	1.045
Central Coast	86	1.025
Inside Waters	210	1.066
WCVI	174	1.051
Area 24	70	1.054
Coast Wide	779	1.050

Table A3: Number (n) and percentage of beds for which a Bed, a GMA, a Sub-Area or a Statistical Area estimate of mean weight is used, as calculated from logbook data.

Mean Weight Estimate Used	n	% of Beds
Bed	557	23.9
GMA	1688	72.3
Sub Area	44	1.9
Statistical Area	45	1.9

Table A4: Mean weight estimates from logbook and biological sample data and ratios of logbook mean weight to biological sample mean weight.

		Mean Weight from Logbook Data	Mean Weight from Biological Samples Corrected for Water Loss			Ratio of Mean Weight from Logbooks to Mean Weight from Biological Samples		
Year Survey Title	Bed Code		All Sizes	>462.2g	>10 yrs	All Sizes	>462.2g	>10 yrs
Haida Gwaii		**1231.3**	**1077.9**	**1100.0**	**1111.3**	**1.142**	**1.119**	**1.108**
1994 Burnaby Island	02-13-02	1534.1	1408.9	1429.8	1422.4	1.089	1.073	1.078
1995 Hotspring Island	02-11-11	1160.2	946.6	1029.5	1043.3	1.226	1.127	1.112
1996 Houston Stewart Ch.	02-18-10	1224.7	1170.8	1190.2	1207.8	1.046	1.029	1.014
1996 Houston Stewart Ch.	02-31-02	1044.7	879.3	934.2	917.6	1.188	1.118	1.139
1996 Houston Stewart Ch.	02-31-03	1054.9	958.5	970.0	983.6	1.101	1.088	1.073
1997 Cumshewa Inlet	02-03-01	1354.3	1142.8	1157.7	1212.6	1.185	1.170	1.117
1997 Cumshewa Inlet	02-03-02	1436.7	1214.4	1230.1	1230.6	1.183	1.168	1.168
1997 Cumshewa Inlet	02-03-03	1387.1	1039.7	1052.7	1172.4	1.334	1.318	1.183
1998 Selwyn/Dana/Logan Inlets	02-06-04	1194.7	1020.6	1025.5	1020.6	1.171	1.165	1.171
1998 Selwyn/Dana/Logan Inlets	02-06-20	1234.1	1060.5	1074.2	1068.6	1.164	1.149	1.155
1998 Selwyn/Dana/Logan Inlets	02-08-06	1108.9	970.0	970.0	970.0	1.143	1.143	1.143
2000 Gowgaia Bay	02-38-01	1443.3	1308.1	1308.1	1322.6	1.103	1.103	1.091
2000 Gowgaia Bay	02-40-01	1612.8	1712.5	1725.2	1771.4	0.942	0.935	0.910
2000 Hippa Island	02-87-02	1052.6	741.2	773.7	741.2	1.420	1.361	1.420
2000 Hippa Island	02-87-04	1078.5	930.3	955.1	961.5	1.159	1.129	1.122
2000 Hippa Island	02-87-05	938.0	776.1	807.3	784.2	1.209	1.162	1.196
2002 Parry Passage	01-02-01	1072.0	1043.4	1066.9	1062.2	1.027	1.005	1.009
Prince Rupert		**1097.4**	**980.6**	**1009.5**	**1014.1**	**1.119**	**1.087**	**1.082**
1996 Otter Pass	06-09-01	1076.6	1096.7	1165.6	1234.5	0.982	0.924	0.872
1996 Otter Pass	06-09-14	1037.9	887.7	920.1	917.8	1.169	1.128	1.131
1996 West Aristazabal Island	06-13-15	981.8	871.8	882.3	879.5	1.126	1.113	1.116
1996 West Aristazabal Island	06-13-36	1191.8	1119.9	1119.9	1122.0	1.064	1.064	1.062
1996 West Aristazabal Island	06-13-38	1250.7	1130.7	1153.2	1189.6	1.106	1.085	1.051
1997 Anderson/Laredo	06-11-02	1085.3	604.6	750.5	737.6	1.795	1.446	1.471
1997 Anderson/Laredo	06-13-18	1117.3	1205.6	1221.3	1224.5	0.927	0.915	0.912
1997 Anderson/Laredo	06-14-01	1146.5	993.1	993.1	993.1	1.154	1.154	1.154
1997 Principe Channel	05-13-03	1042.1	862.3	894.2	862.3	1.209	1.165	1.209
1997 Principe Channel	05-13-04	1016.2	891.7	917.8	938.4	1.140	1.107	1.083
1998 Dundas Island	03-01-03	966.2	763.8	813.7	775.4	1.265	1.187	1.246
1998 Dundas Island	03-01-04	976.1	823.6	914.0	916.2	1.185	1.068	1.065
1998 Moore Islands	106-02-03	1166.8	1054.1	1059.9	1070.4	1.107	1.101	1.090
1998 Moore Islands	106-02-06	1149.1	1075.7	1083.6	1096.1	1.068	1.061	1.048
1998 Moore Islands	106-02-15	1201.4	1090.3	1090.3	1132.9	1.102	1.102	1.060
2004 SE Prescott	04-09-08	1119.4	1110.9	1115.9	1115.9	1.008	1.003	1.003
2004 Tree Nob	04-02-05	1141.9	1284.1	1290.9	1290.9	0.889	0.885	0.885
2004 Tree Nob	04-02-09	1098.2	940.9	955.9	940.9	1.167	1.149	1.167
2004 Tree Nob	04-13-03	1084.4	823.9	839.0	830.3	1.316	1.292	1.306
CentralCoast		**1088.2**	**1050.2**	**1090.6**	**1104.4**	**1.036**	**0.998**	**0.985**
1995 Kitasu Bay	06-18-02	1056.4	1229.9	1258.3	1282.0	0.859	0.840	0.824
1995 Kitasu Bay	06-18-03	1166.3	1185.2	1204.3	1199.8	0.984	0.968	0.972
1995 Kitasu Bay	06-18-06	1074.5	1270.3	1295.2	1292.7	0.846	0.830	0.831
1995 West Higgins Pass	06-16-01	1122.4	945.7	962.2	954.6	1.187	1.166	1.176
1996 S Bardswell/Prince Group	07-18-05	882.9	765.2	789.2	765.2	1.154	1.119	1.154
1996 S Bardswell/Prince Group	07-19-06	968.4	1194.2	1201.0	1211.9	0.811	0.806	0.799
1998 Hakai Passage	07-27-21	1224.3	1653.4	1713.8	1797.5	0.740	0.714	0.681
1998 Hakai Passage	08-02-01	1281.7	1157.6	1207.8	1260.8	1.107	1.061	1.017
1998 Hakai Passage	08-02-09	1085.0	904.2	995.2	964.9	1.200	1.090	1.124
2003 Spider Anchorage, 2003	07-27-07	1076.7	962.9	1033.9	1069.6	1.118	1.041	1.007
2003 Spider Anchorage, 2003	07-27-08	1265.4	1079.7	1114.0	1129.0	1.172	1.136	1.121
2005 Ivory to Stryker	07-18-05	882.9	636.1	688.6	639.5	1.388	1.282	1.381
2005 Ivory to Stryker	07-19-06	968.4	868.2	875.5	868.2	1.115	1.106	1.115
2005 Ivory to Stryker	07-32-01	1174.2	969.4	1054.5	1120.7	1.211	1.114	1.048
2005 Ivory to Stryker	07-32-11	1106.8	762.3	792.3	768.6	1.452	1.397	1.440
2005 Tribal to Nalau	07-27-21	1224.3	1279.2	1321.1	1390.9	0.957	0.927	0.880
2005 Tribal to Nalau	08-02-09	1085.0	1025.5	1056.4	1056.4	1.058	1.027	1.027
2005 Tribal to Nalau	08-04-01	942.2	1013.7	1066.7	1107.6	0.929	0.883	0.851

Table A4: (continued)

Year	Survey Title	Bed Code	Mean Weight from Logbook Data	Mean Weight from Biological Samples Corrected for Water Loss			Ratio of Mean Weight from Logbooks to Mean Weight from Biological Samples		
				All Sizes	>462.2g	>10 yrs	All Sizes	>462.2g	>10 yrs
Inside Waters			**1088.2**	**860.0**	**974.0**	**1038.0**	**1.265**	**1.117**	**1.048**
1995	Duncan Island	12-11-05	1157.3	1064.1	1117.7	1196.5	1.088	1.035	0.967
1996	Oyster River	14-13-01	1083.2	843.6	911.4	868.4	1.284	1.188	1.247
1996	Oyster River	14-13-08	1115.3	1082.9	1145.3	1124.2	1.030	0.974	0.992
1998	Comox 1998	14-10-01	1028.7	750.9	900.6	922.3	1.370	1.142	1.115
1998	Comox 1998	14-11-05	1234.8	892.8	1011.1	1048.7	1.383	1.221	1.178
1999	Thormanby Island	16-02-01	902.3	781.0	817.3	794.8	1.155	1.104	1.135
2001	Boatswain Bank	18-07-01	947.7	698.7	819.4	855.1	1.356	1.157	1.108
2002	Marina Island, 2002	13-15-01	1256.0	1105.0	1165.8	1249.1	1.137	1.077	1.006
2004	Lund 2004	15-02-01	925.2	568.0	720.8	648.7	1.629	1.284	1.426
2004	Lund 2004	15-02-02	1072.5	587.1	875.1	882.6	1.827	1.226	1.215
2004	Lund 2004	15-02-03	909.9	505.5	730.2	724.3	1.800	1.246	1.256
2005	Hornby Island	14-07-01	1159.1	964.4	1037.2	1142.3	1.202	1.118	1.015
2005	Hornby Island	14-07-03	1233.1	795.1	915.4	1368.5	1.551	1.347	0.901
2005	Hornby Island	14-09-01	1261.6	1022.9	1124.7	1200.0	1.233	1.122	1.051
2005	Hornby Island	14-10-03	989.7	1041.5	1079.1	1134.4	0.950	0.917	0.872
2005	Mapleguard / Qualicum	14-05-01	1082.8	975.9	1129.8	1312.6	1.110	0.958	0.825
2005	Mapleguard / Qualicum	14-05-02	1140.7	939.9	1056.7	1173.7	1.214	1.079	0.972
West Coast Vancouver Island			**920.3**	**797.3**	**868.2**	**877.6**	**1.154**	**1.060**	**1.049**
1996	Winter Harbour	27-03-02	1100.4	995.1	1031.0	1037.2	1.106	1.067	1.061
1996	Winter Harbour	27-07-02	920.9	692.0	764.4	711.2	1.331	1.205	1.295
1996	Winter Harbour	27-07-05	788.0	736.6	782.2	763.4	1.070	1.007	1.032
1998	Kyuquot	26-01-01	828.2	758.6	881.9	871.4	1.092	0.939	0.950
2000	Barkley Sound	23-10-01	1019.6	880.2	921.7	967.8	1.158	1.106	1.053
2000	Nootka Sound	25-06-04	969.0	1005.8	1070.4	1144.2	0.963	0.905	0.847
2000	Nootka Sound	25-06-09	948.7	676.2	746.5	696.4	1.403	1.271	1.362
2002	NE Barkley Sound	23-06-11	959.6	850.3	853.1	854.9	1.128	1.125	1.122
2002	Rolling Roadstead	25-13-13	1029.8	1018.7	1109.5	1320.1	1.011	0.928	0.780
2002	Winter Harbour, 2002	27-07-02	920.9	641.2	754.0	764.8	1.436	1.221	1.204
2002	Winter Harbour, 2002	27-07-05	788.0	725.4	852.6	831.2	1.086	0.924	0.948
2003	Mission Group, 2003	26-01-01	828.2	584.9	666.7	628.0	1.416	1.242	1.319
2005	Barkley 2005	23-05-04	862.2	800.1	852.9	818.9	1.078	1.011	1.053
Area 24			**1119.4**	**1061.0**	**1136.0**	**1172.5**	**1.055**	**0.985**	**0.955**
1994	Elbow Bank	24-06-32	1182.2	1540.3	1540.3	1553.0	0.768	0.768	0.761
1997	Millar Channel	24-06-02	944.3	785.1	916.4	951.4	1.203	1.030	0.993
1997	Yellow Bank	24-07-02	1105.7	1026.9	1124.7	1129.0	1.077	0.983	0.979
2004	BlundenBartlett	24-06-11	1170.9	908.0	967.4	1091.0	1.290	1.210	1.073
2004	BlundenBartlett	24-06-17	1194.1	1044.7	1131.1	1138.1	1.143	1.056	1.049
Coastwide			**1094.7**	**968.0**	**1022.9**	**1044.5**	**1.131**	**1.070**	**1.048**

Table A5: Paired t-test results comparing mean weight (g) from logbooks to mean weight from biological samples, by Region and coastwide. Tests were conducted using all geoducks from the biological samples, using geoducks >462g only and using geoducks > 10 years only.

Sizes from		Mean weight from		Ratio				
Bio-samples	Region	Logbook	Bio-samples	Log/Bio	Difference	t-value	df	P
All	Coastwide	1094.7	968.0	1.131	126.8	7.56	88	**0.000**
	Haida Gwaii	1231.3	1077.9	1.142	153.4	6.15	16	**0.000**
	Prince Rupert	1097.4	980.6	1.119	116.8	3.86	18	**0.001**
	Central Coast	1088.2	1050.2	1.036	38.0	0.82	17	0.425
	Inside Waters	1088.2	860.0	1.265	228.3	6.43	16	**0.000**
	WCVI	920.3	797.3	1.154	122.9	4.31	12	**0.001**
	Area 24	1119.4	1061.0	1.055	58.4	0.54	4	0.618
> 462g	Coastwide	1094.7	1022.9	1.070	71.8	4.79	88	**0.000**
	Haida Gwaii	1231.3	1100.0	1.119	131.3	5.36	16	**0.000**
	Prince Rupert	1097.4	1009.5	1.087	87.8	3.35	18	**0.004**
	Central Coast	1088.2	1090.6	0.998	-2.3	-0.05	17	0.960
	Inside Waters	1088.2	974.0	1.117	114.2	4.53	16	**0.000**
	WCVI	920.3	868.2	1.060	52.1	1.78	12	0.101
	Area 24	1119.4	1172.5	0.955	-53.0	-0.65	4	0.552
> 10 years	Coastwide	1094.7	1044.5	1.048	50.2	2.98	88	**0.004**
	Haida Gwaii	1231.3	1111.3	1.108	120.0	4.90	16	**0.000**
	Prince Rupert	1097.4	1014.1	1.082	83.2	2.86	18	**0.010**
	Central Coast	1088.2	1104.4	0.985	-16.2	-0.32	17	0.755
	Inside Waters	1088.2	1038.0	1.048	50.2	1.48	16	0.160
	WCVI	920.3	877.7	1.049	42.6	0.99	12	0.340
	Area 24	1119.4	1136.0	0.985	-16.5	-0.18	4	0.868

Table A6: Mean weight estimates (g) from logbook data, using only the data from the same year as the biological sample collection, and from biological samples and ratios of mean weight from logbook data to mean weight from biological samples.

Year Survey Title	Bed Code	Mean Weight from Logbook Data	Mean Weight from Biological Samples Corrected for Water Loss			Ratio of Mean Weight from Logbooks to Mean Weight from Biological Samples		
			All Sizes	>462.2g	>10 yrs	All Sizes	>462.2g	>10 yrs
Haida Gwaii		**1207.4**	**1044.3**	**1064.5**	**1077.1**	**1.156**	**1.134**	**1.121**
1997 Cumshewa Inlet	02-03-01	1402.7	1142.8	1157.7	1212.6	1.227	1.212	1.157
1997 Cumshewa Inlet	02-03-02	1346.1	1214.4	1230.1	1230.6	1.108	1.094	1.094
1997 Cumshewa Inlet	02-03-03	1286.2	1039.7	1052.7	1172.4	1.237	1.222	1.097
2000 Gowgaia Bay	02-38-01	1281.2	1308.1	1308.1	1322.6	0.979	0.979	0.969
2000 Tasu Sound	02-42-04	888.2	789.2	819.9	798.0	1.125	1.083	1.113
2000 Tasu Sound	02-42-08	1252.1	1147.3	1163.5	1168.9	1.091	1.076	1.071
2000 Tasu Sound	02-45-03	1457.5	1353.9	1376.6	1378.8	1.076	1.059	1.057
2000 Hippa Island	02-87-02	1111.3	741.2	773.7	741.2	1.499	1.436	1.499
2000 Hippa Island	02-87-04	1085.8	930.3	955.1	961.5	1.167	1.137	1.129
2000 Hippa Island	02-87-05	962.7	776.1	807.3	784.2	1.241	1.192	1.228
Prince Rupert								
Central Coast		**1248.6**	**1238.4**	**1305.6**	**1341.1**	**1.008**	**0.956**	**0.931**
1998 Hakai Passage	07-27-21	1195.8	1653.4	1713.8	1797.5	0.723	0.698	0.665
1998 Hakai Passage	08-02-01	1414.3	1157.6	1207.8	1260.8	1.222	1.171	1.122
1998 Hakai Passage	08-02-09	1135.8	904.3	995.2	964.9	1.256	1.141	1.177
Inside Waters		**1100.0**	**861.9**	**976.1**	**1050.9**	**1.276**	**1.127**	**1.047**
2005 Mapleguard / Qualicum	14-05-01	1160.0	975.9	1129.8	1312.6	1.189	1.027	0.884
2005 Mapleguard / Qualicum	14-05-02	1337.1	939.9	1056.7	1173.7	1.423	1.265	1.139
1998 Comox 1998	14-10-01	1025.4	750.9	900.6	922.3	1.366	1.139	1.112
1999 Thormanby Island	16-02-01	877.3	781.0	817.3	794.8	1.123	1.074	1.104
West Coast Vancouver Island		**830.5**	**754.8**	**831.5**	**842.8**	**1.100**	**0.999**	**0.985**
2005 Barkley 2005	23-05-04	858.2	800.2	852.9	818.9	1.072	1.006	1.048
2002 NE Barkley Sound	23-06-11	904.1	850.3	853.1	854.9	1.063	1.060	1.058
2005 Barkley 2005	23-06-13	687.1	771.8	829.6	816.1	0.890	0.828	0.842
2000 Nootka Sound	25-06-02	997.5	702.6	749.4	707.9	1.420	1.331	1.409
2000 Nootka Sound	25-06-04	1083.5	1005.8	1070.4	1144.2	1.077	1.012	0.947
2000 Nootka Sound	25-06-09	942.9	676.2	746.5	696.4	1.394	1.263	1.354
2002 Rolling Roadstead	25-13-13	919.0	1018.7	1109.5	1320.1	0.902	0.828	0.696
2003 Mission Group, 2003	26-01-01	798.9	584.9	666.7	628.0	1.366	1.198	1.272
2002 Winter Harbour, 2002	27-03-04	660.4	526.2	661.8	688.3	1.255	0.998	0.959
2002 Winter Harbour, 2002	27-07-02	557.5	641.2	754.0	764.8	0.869	0.739	0.729
2002 Winter Harbour, 2002	27-07-05	726.8	725.4	852.6	831.2	1.002	0.852	0.874
Area 24		**1135.5**	**941.2**	**1034.9**	**1077.4**	**1.206**	**1.097**	**1.054**
1997 Millar Channel	24-06-02	1085.7	785.1	916.4	951.4	1.383	1.185	1.141
2004 BlundenBartlett	24-06-11	1095.2	908.0	967.4	1091.0	1.206	1.132	1.004
2004 BlundenBartlett	24-06-17	1213.9	1044.7	1131.1	1138.1	1.162	1.073	1.067
1997 Yellow Bank	24-07-02	1147.1	1026.9	1124.7	1129.0	1.117	1.020	1.016
Coastwide		**1059.3**	**927.3**	**992.2**	**1018.1**	**1.142**	**1.068**	**1.041**

Table A7: Paired t-test results comparing mean weight from logbooks (only data from same year as biological sample collection used) to mean weight from biological samples, coastwide and by region. Paired t-tests performed using all geoducks from the biological samples, using geoducks >462g only and using geoducks > 10 years only.

Sizes from Bio-samples	Region	Mean weight from Logbook*	Mean weight from Bio-samples	Ratio Log/Bio	Mean Difference	t-value	df	P
All	Coastwide	1059.3	927.3	1.142	132.0	4.51	31	**0.000**
	Haida Gwaii	1207.4	1044.3	1.156	163.1	4.71	9	**0.001**
	Prince Rupert	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	Central Coast	1248.6	1238.4	1.008	10.2	0.04	2	0.969
	Inside Waters	1100.0	861.9	1.276	238.1	3.70	3	0.034
	WCVI	830.5	754.8	1.100	75.7	1.80	10	0.103
	Area 24	1135.5	941.2	1.206	194.3	5.09	3	0.015
> 462g	Coastwide	1059.3	992.3	1.068	67.0	2.26	31	0.031
	Haida Gwaii	1207.4	1064.5	1.134	142.9	4.31	9	**0.002**
	Prince Rupert	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	Central Coast	1248.6	1305.6	0.956	-57.0	-0.25	2	0.828
	Inside Waters	1100.0	976.1	1.127	123.9	2.22	3	0.113
	WCVI	830.5	831.5	0.999	-1.0	-0.02	10	0.984
	Area 24	1135.5	1034.9	1.097	100.6	3.20	3	0.049
> 10 years	Coastwide	1059.3	1018.1	1.040	41.2	1.22	31	0.232
	Haida Gwaii	1207.4	1077.1	1.121	130.3	3.91	9	**0.004**
	Prince Rupert	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	Central Coast	1248.6	1341.1	0.931	-92.5	-0.36	2	0.751
	Inside Waters	1100.0	1050.9	1.047	49.1	0.71	3	0.530
	WCVI	830.5	842.8	0.985	-12.3	-0.20	10	0.845
	Area 24	1135.5	1077.4	1.054	58.1	1.95	3	0.146

* only using data from same year as biological sample collected for each bed.