THE POTENTIAL IMPACT OF SEA OTTERS (*ENHYDRA LUTRIS*) ON THE BRITISH COLUMBIA GEODUCK CLAM (*PANOPEA ABRUPTA*) FISHERY

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ABSTRACT

I used fishery-independent survey data and catch curve analysis to assess the potential predation effects of sea otters (*Enhydra lutris*) on the British Columbia geoduck (*Panopea abrupta*) fishery on the west coast of Vancouver Island, by estimating geoduck total mortality rates across a gradient of sea otter abundance. Linear regression provided strong evidence of a fishing effort effect on geoduck total mortality while the main effect of otters was not significant. Harvesters, however, have increasingly reported seeing sea otters eating geoduck; thus a more balanced study design and greater sampling intensity are needed to increase the power to detect whether sea otters affect geoduck harvests. This paper concludes with an examination of the different legislative mandates of fisheries and wildlife management in Canada, and establishes that artificially limiting the sea otter's range in B.C. would be difficult under federal law and for socio-political reasons.

Keywords: Geoduck; sea otter; predation; catch curve analysis; total mortality rate; conflict.

DEDICATION

To my father Tom Reidy, who instilled in me the importance of education and whose intellectual strength and determination will inspire me always.

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Chapter 1
General Introduction

One of the main objectives of fisheries management is to provide for the sustainable yield from fish stocks over time (Hilborn and Walters 1992). A relatively new concern of management is emerging in regions where commercial fisheries co-occur with marine mammal populations that are recovering from a century of over-harvesting (Beverton 1985). As noted by Beverton (1985:4), “the world into which these mammal populations are recovering is not the same as it was when they were at their height decades ago.” Increases in natural predators may be influencing prey populations in ways that are important to conventional fisheries stock assessment and management (Jurado-Molina et al. 2005; Hollowed et al. 2000). Along the west coast of Vancouver Island (WCVI), British Columbia (B.C.), for example, sea otters (*Enhydra lutris*) were successfully reintroduced in the 1970s, and their expanding populations may now conflict with the management of high-value invertebrate fisheries that developed in their absence (Watson 2000; COSEWIC 2007).

Prior to commercial exploitation in the 18th and 19th centuries, sea otters almost certainly limited the quantity and distribution of their near-shore invertebrate prey (Estes and Palmisano 1974; Johnson 1982; Breen et al. 1982; Kvitek et al. 1992; Gerber et al. 1999; Watson 2000). Ecologists speculate that under reduced predation pressure by otters many invertebrate species increased in average size and abundance (Bodkin 2003), which may have led to the development of major new commercial shellfish fisheries (Gerber et al. 1999; Watson 2000). Following their reintroduction to the WCVI, sea otters were initially at very low levels near people and fishing grounds. This situation is changing, however, as fishermen are increasingly concerned about sea otters taking a large share of the shellfish resources (e.g., Green Sea Urchin IFMP 2010; IFMP 2010). As a result, scientists are being asked to broaden stock assessments to a multi-species perspective in regions where sea otters and shellfisheries overlap (IFMP 2010).

Despite considerable scientific research on the trophic ecology of sea otters (e.g., Estes and Palmisano 1974; Estes et al. 1978; Kvitek and Oliver 1992), we currently know little about how sea otters directly or indirectly affect fisheries. For instance, fisheries for abalone were once common along the Pacific coast of North America (Watson 2000). Abalones are a preferred prey of sea otters (Johnson 1982; Fanshawe et al. 2003), and expanding sea otter populations are known to limit the size, abundance, and distribution of abalone in California (Wendell 1994), in B.C. (Breen et al. 1982), and possibly in Washington State and southeast Alaska (Watson 2000). Intense human harvesting, however, has had the greatest impact on abalone population declines (DFO 2007a). Most commercial fisheries for abalone in North America were closed by the 1990s, because of the lack of knowledge about the life history characteristics of abalones and over-harvesting (DFO 2007a). The connection between increasing numbers of sea otters and the depleted abalone populations in B.C. is presently undetermined, but areas that are inaccessible to foraging sea otters (e.g., rock crevices) do support small populations of abalone, despite the occurrence of sea otters (Watson 2000; DFO 2007a). Reviews of the known (and expected) effects of sea otters on commercially important invertebrate species in B.C. can be found in Watson and Smith (1996).

The general lack of knowledge about the effects of sea otter predation on fisheries arises from a lack of quantitative studies aimed at partitioning shellfish mortality into different sources, including sea otters and fisheries (Watson 2000). By partitioning the total mortality into sea otter predation and fishery components, the trade-offs between sea otter recovery and fishery yield of the sea otter's invertebrate prey could be assessed. The purpose of my research is to evaluate the potential effects of sea otter predation on a WCVI fishery for the geoduck clam (*Panopea abrupta*). While previous studies have not detected an effect of sea otter predation on the deeply-burrowing geoducks in California (Kvitek et al. 1988) or in Alaska (Kvitek and Oliver 1992), Kvitek and Oliver suggested that predation on geoducks may possibly increase with a decline in the sea otter's preferred prey, such as sea urchins and other species of shallow-buried clams. In this paper, I review the biology and distribution of geoduck clams because the species is distinct from the typical commercially fished species of invertebrates. I also provide overviews of the geoduck fishery and the natural history of sea otters in B.C. In Chapter 2, I use spatial and temporal comparisons of geoduck populations along a gradient of sea otter occupancy, to explore the relations between geoduck total mortality rates and sea otter abundance. In Chapter 3, I examine a potential conflict between marine wildlife and fishery policy in British Columbia.

The geoduck clam

Geoduck habitat and biology

The geoduck clam has a patchy distribution in the northeast Pacific, occurring from southern Japan to Mexico (Coan et al. 2000; Harbo 1999; King 1986). Geoducks are the largest burrowing clams in the world and may weigh up to 4.5 kg (10 lbs) at 20 cm shell length (Harbo 1999). They are also among the longest-lived animals known, with a maximum-recorded age of 168 years (Bureau et al. 2002). They can bury to a depth of 1 m in a variety of unconsolidated substrates, such as mud, sand, and gravel, but tend to concentrate in large numbers in mud-sand or sand (Campbell et al. 1998) in beds of 20 to 50 ha (Orensanz et al. 2004) and at depths ranging from inter-tidal to 100 m (Bureau et al. 2002).

Geoducks are filter feeders, extending a long siphon to the surface of the substrate to take in food particles (e.g., phytoplankton) (King 1986). They are reproductively active for up to 100 years or more through annual broadcast spawning (Campbell et al. 1998). After drifting in the pelagic zone for about a month, the planktonic larvae stop swimming and begin crawling along the seafloor using byssal threads (Goodwin and Pease 1989). Soft substrates are thought to induce the post-larvae to insert a foot and begin burrowing (Goodwin and Pease 1989). Juveniles can attain depths of about 60 cm in the sediment within 2 years, and reach a maximum adult refuge depth of 1 m in 4 or 5 years (DFO 2000). Juvenile geoducks may be able to re-bury themselves following disturbance (King 1986), but adults lose their ability to dig after transitioning to the reproductive stage (King 1986; Campbell et al. 1998). Thus, adult geoducks are immobile and cannot re-bury if removed from their burrows (King 1986).

Geoduck recruitment to fisheries is dependent on growth and survival of larvae that settle in fishing areas. Studies indicate that recruitment declined for decades prior to the start of the fishery (Orensanz et al. 2004), though a recent analysis of age samples by Bureau et al. (2002; 2003), and subsequent recruitment simulations by Zhang and Hand (2006), suggest that frequent and strong geoduck recruitment events have occurred in

B.C. since the late-1980s. B.C. stock assessment scientists believe that geoduck populations are associated through the drifting and inter-mixing of larvae (Zhang and Hand 2006). In addition, some populations (e.g., deep-water stocks) of geoducks may be separate from the fishable populations that contribute to the recruitment processes (IFMP 2010; C. Hand, Fisheries and Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Road, Nanaimo B.C., V9R 5K6, Personal Communication, 2009). In the shallower beds, where fisheries occur, geoducks reach sexual maturity at around 3 years and marketable size of approximately 2 lbs between 6 and 12 years (Doherty 1990; Bureau et al. 2002).

Mortality from predation is high during the larval and juvenile stages (Goodwin and Pease 1989), but decreases considerably after four or five years when geoducks reach their maximum refuge depth (Sloan and Robinson 1984; Goodwin and Pease 1989). While the estimated average natural mortality rate (M) for geoduck populations varies between 0.014 and 0.054 yr^{-1} (Orensanz et al. 2004), Zhang and Campbell (2004) estimated M to be 0.036 yr^{-1} (SE 0.003 yr^{-1}) for the WCVI region. Zhang and Hand (2006) subsequently suspected that adult mortality was higher in regions in the WCVI where sea otters are established.

Geographic distribution of geoduck clams in B.C.

Geoduck densities are low in the intertidal zone to 1 m, increase with depth to 20 - 24 m then decrease to 110 m (Campbell et al. 1998). Commercial harvests and dive surveys target geoduck stocks from 3 m to the maximum commercial diving depth of 20 m (Campbell et al. 1998; Bureau et al. 2002). As a result, scientists and resource managers know little about geoduck densities beyond this range (Campbell et al. 1998).

Geoducks living at greater depths, or in harder substrates, are unsurveyed and are probably unexploited (Campbell et al. 1998; Bureau et al. 2002).

Information on the geographic distribution of geoduck beds in B.C. is based on fishing activities reported in commercial geoduck fisher logbooks, and GIS and acoustic mapping technology (Siddon 2007; IFMP 2010). The accuracy of bed area estimates mostly depends on the accuracy of information provided by fishermen, the correctness of logbook interpretation, and the accuracy of nautical charts (DFO 2000). Fisheries and Oceans Canada, the Underwater Harvesters Association (UHA), and First Nations began fishery-independent surveys of the exploited beds in 1992 to estimate geoduck density (IFMP 2010). These organizations now carry out joint annual surveys between May and September, primarily in Georgia Strait and along the WCVI region (Bureau et al. 2002). To date, 103 surveys have been conducted along the B.C. coast, which comprises about 57% of the total estimated area of commercial geoduck beds (G. Dovey, West Coast Geoduck Corp., 2009, unpublished data). Of these surveys, 22 (25%) were along the WCVI and provide specific fishery-independent biological information on geoduck abundance, distribution, and age composition.

Management of geoduck fisheries

The first recorded commercial exploitation of geoduck clams in B.C. occurred in 1976 (IFMP 2010). By 1979, the industry had grown to 101 vessels under an open access fishery model with no catch or effort limits (Hand and Bureau 2000; DFO 2000). Although fishing was initially concentrated in the sub-tidal waters along the east coast of Vancouver Island, market expansion resulted in a relatively fast spread of fishery exploitation to the west coast of Vancouver Island by 1979, and to the north coast by

1980 where new beds continue to be discovered (Hand and Bureau 2000; DFO 2000). In 1980, Fisheries and Oceans Canada (DFO) established geoduck management boundaries for Inside Waters (between the B.C. mainland and the east coast of Vancouver Island), the west coast of Vancouver Island (WCVI), and the North Coast (Hand and Bureau 2000).

DFO imposed further restrictions at the request of industry by instituting individual vessel quotas (IVQs), which reduced the fleet to 55 licences (Hand and Bureau 2000). Additionally, DFO implemented a three-year rotational harvest strategy to ensure that fishers were only harvesting along one-third of the B.C. coast in any given year, but at three times the effort (Hand and Bureau 2000). In 2002, however, DFO re-established annual harvesting in the WCVI region to obtain timely feedback about potential impacts from sea otters (IFMP 2010). Estimates of geoduck bed areas, densities, and recruitment, indicated that approximately 10% of the fishable geoduck beds were over-harvested and were closed for regeneration of the populations (DFO 2000).

The B.C. Underwater Harvesters Association, established in 1981 by commercial geoduck fishers and wholesalers, jointly manages the commercial geoduck fishery with DFO (UHA 2010). Funded by membership fees, the UHA's role is to provide biological survey data and commercial fishing information for DFO scientists and managers. DFO oversees the management of the fishery and makes recommendations on total allowable catches (TAC) that are expected to maintain a low interannual variation in yield (IFMP 2010). DFO sets each geoduck IVQ at 1/55th of the annual coast-wide quota and vessel owners must choose to fish in one of the three license areas (IFMP 2010). The geoduck fishery continues to operate under limited entry and individual vessel quotas (IFMP

2010), and the fishing effort has varied across fishing locations (Bureau et al. 2002). The primary goal of conducting regular geoduck surveys is to establish geoduck population parameters over time (i.e., natural recruitment and growth rates), as well as to study the long-term effects of harvesting on the fishable wild stocks (IFMP 2010).

Geoduck populations in B.C. cover approximately 26,400 ha of coast (DFO 2000) and support a profitable dive fishery (Zhang and Hand 2006). The average annual landed value between 2005 and 2009 was CDN$ 32 million, with approximately 11% harvested from the WCVI (Archipelago Marine Research Ltd., 2009, unpublished data). Although the fishery operates year-round, inclement weather and market fluctuations can influence harvest locations and the fishing effort, and paralytic shellfish poisoning (PSP) often causes closures in some areas (DFO 2000).

The long-term management goal is to maintain a bed's fishable biomass at above 50% of the estimated unfished level over a 50-year period (IFMP 2010; Zhang and Hand 2007). DFO managers set annual regional TACs by multiplying a target harvest rate (e.g., 1.2 to 1.8% for the WCVI) by the current biomass estimated for the beds in the region (IFMP 2010). Biomass estimates and harvest quotas change as a result of surveys and observer reports that provide up-to-date information about clam bed density (Hand and Bureau 2000; DFO 2000). Although the coast-wide TAC for 2010 is unchanged from recent years (1,574,454 kg), the TAC and number of licences for the WCVI declined in 2007 (IFMP 2010). The UHA and DFO suspect that declines in the stock estimates for this area are due to a history of intense commercial harvests and increased sea otter predation (IFMP 2010; Zhang and Hand 2007).

Geoduck harvesters are concerned that management based on biological reference-points is meaningless where sea otters are present. This is because the addition of predation mortality may cause a fished stock to fall below the limit reference point of 40% of virgin stock size, at which point fishing is not permitted (IFMP 2010). The management plan therefore recommends building on the existing research and monitoring programs to assess the potential impact of sea otter predation on geoduck populations (IFMP 2010).

The sea otter

Sea otter habitat and biology

Sea otters (*Enhydra lutris*) live in near-shore, marine habitats of the central and north Pacific Ocean, and in Canada are predominantly found along the WCVI and the central B.C. coast (DFO 2007b). Optimal habitats for sea otters are relatively shallow (<40 m) and near-shore (<1–2 km) where mixed rocky and soft-bottoms have an abundance of invertebrate resources (COSEWIC 2007). Sea otters forage by diving to the sea floor, capturing invertebrate prey with their forelimbs, and then consuming these prey items at the surface (DFO 2007b). Communities of clam species in soft-bottomed habitats are important resources that are currently abundant in B.C. (DFO 2007b).

Sea otters have high metabolic demands because they have no blubber for energy storage or insulation from the cold waters (Costa and Kooyman 1982). Instead, they rely on a high rate of food intake and air trapped in dense fur to keep warm. Adult male sea otters weighing up to 46 kg (101 lbs) (Bodkin 2003) can consume up to 22 to 33% of their body weight per day (DFO 2007b). Sea otters rest in large aggregations (rafts) of up to 100 or more animals that are comprised of either all males or all females and pups, which can remain in the same general area for months or years (Ralls et al. 1996). Radio-tracked sea otters in Alaska (Garshelis and Garshelis 1984) and in California (Ralls et al. 1996) regularly traveled beyond three kilometres from their resting areas while foraging, with male sea otters traveling much greater distances than females. Sea otters reach sexual maturity at around 5 years of age and generally remain reproductive throughout their 10 – 20 year lifespan (DFO 2007b). Although mating occurs year-round in B.C., peak pupping occurs in March and April, with females producing and feeding a single pup per year for up to eight months (DFO 2007b).

Geographic distribution of sea otters in B.C.

The sea otter was hunted to near extinction during the maritime fur trade that commenced in the 1700s, until sea otters were protected under the *International Fur Seal Treaty* in 1911 (COSEWIC 2007). There is no reliable estimate of the total number of otters in the north Pacific prior to exploitation (Kenyon 1969). However, data obtained from recent studies and historical documents from the fur trade suggest that sea otter populations once numbered over 200,000 animals in Alaska (Johnson 1982), and between 16,000 and 20,000 animals in California (Riedman and Estes 1990). Approximately 55,000 sea otter pelts were landed in British Columbia from unknown sources between 1789 and 1809, with at least 6,000 from the WCVI (COSEWIC 2007). In 1911, fewer than 2,000 animals remained in isolated populations, with many populations, including those in B.C., declining to extinction (COSEWIC 2007). The last records of sea otters in B.C. is of two single animals that were shot and killed in 1929 and 1930 on the WCVI (COSEWIC 2007).

During the 1960s, in response to a plan by the US Atomic Energy Commission to conduct underground nuclear tests in the Bering Sea near southwest Alaska, Canadian biologists received permission to transplant some Alaskan sea otters to the WCVI (Paul 2009). Between 1969 and 1972, 89 sea otters from Amchitka and Prince William Sound, Alaska were introduced to the WCVI (Bigg and MacAskie 1978). By 2008, the population of sea otters in B.C. had increased to more than 4,700 individuals (Nichol et al. 2009) spread over two distinct subpopulations that are currently increasing at annual rates of approximately 8% yr^{-1} (WCVI) and 12% yr^{-1} (central coast) (COSEWIC 2007).

DFO began to census B.C. sea otters in 1977 using fixed-wing aircraft and small boats (Bigg and MacAskie 1978). Since 1988, the sea otter range has been surveyed annually by helicopter or boat (COSEWIC 2007). The surveys are completed between April and September in areas known to be occupied by otters, and the direct counts provide measures of relative abundance for estimating trends in population abundance (Nichol et al. 2005). Nichol et al. (2005) suspect that some single otters that forage offshore, or are far up the inlets, are inevitably missed so that the survey counts likely underestimate the actual population size. Male aggregations gradually migrate into new areas and typically occupy the margin of the population range. Rafts of mothers and pups are established in areas abandoned by males (Nichol et al. 2009). The site fidelity and social structure of sea otters is comprised of over-lapping home ranges of aggregations that can extend up to a few kilometres along the coast (Nichol et al. 2009).

Status of sea otters in B.C.

Sea otters currently occupy 25-33% of their historical range within B.C. (Figure 1) (COSEWIC 2007). They are recognized as “keystone species” because they contribute to the structure and function of both rocky subtidal and soft-bottom habitats by reducing the size and abundance of invertebrate species (DFO 2007b). Sea otters are designated a special concern ("species at risk") under the federal *Species at Risk Act* (*SARA*) (Species at Risk Public Registry 2011), and as a threatened species under the *British Columbia Wildlife Act* (MNRO 2010a). Sea otters have partially recovered through legal protection under Schedule 1 of the *SARA* (Schedule 1 is the official List of Wildlife Species at Risk), the federal *Fisheries Act*, the *British Columbia Wildlife Act*, and the *Marine Mammal Protection Act* if they wander into adjacent U.S. waters (COSEWIC 2007). The management strategy for sea otters in B.C. is a "non-interventionist approach" (i.e., a hands-off approach) that allows natural range expansion and population growth for the species (DFO 2007b). The recovery goal is to ensure that the sea otter population is sufficiently large and widely spread to withstand known human-caused threats, such as oil spills, which could result in the extirpation of sea otters, or cause the population to become endangered (DFO 2007b).

Challenges to assessing conflict between sea otters and geoduck fisheries

Fishery and wildlife managers face important challenges in regions where otter foraging overlaps the fishery. Foraging sea otters and commercial geoduck fisheries harvest the same prey resource. Sea otters are voracious predators of invertebrates (Kenyon 1969), and along several beaches in California the expanding otter populations were at least partially responsible for the collapse of the recreational Pismo clam (*Tivela stultorum*) fisheries in a single season (Miller et al. 1975; Wendell et al. 1986). Although sea otters in Alaska also exert strong predation pressure on infaunal bivalve populations, especially reducing the shallow-burrowing butter clam (*Saxidomus giganteus*) populations (Kvitek and Oliver 1992; Kvitek et al. 1992), there is little evidence of sea otter predation on the deeply burrowing geoduck clams in Alaska when geoducks are abundant (Kvitek and Oliver 1992).

The intensity of competition between B.C. sea otters and geoduck harvesters is difficult to evaluate for ecological and statistical reasons. For instance, a group of sea otters that moved into a coastal lagoon in California had no detectable effect on the abundance and distribution of the deep-burrowing bivalve clams (*Tresu nuttallii* and *Saxidomus nuttalli*), despite these species comprising 61% of the otter's local diet (Kvitek et al. 1988). Nonetheless, harvesters in B.C. have increasingly reported seeing sea otters eating geoduck along the WCVI, so there is a need for a formal evaluation of the potential impact of sea otters on the fished geoduck populations.

Quantitative analysis of the effects of sea otter predation on geoduck populations presents several statistical challenges. A leading challenge is in the complexity of large-scale ecosystems where controlled experimentation is often impossible (Hilborn and Mangel 1997). In such situations, the assumptions of classical statistics that rely on strict experimental design and large sample sizes are difficult to meet (Clark 2007), and this ultimately leads to measurement error and high levels of uncertainty (Cressie et al. 2009). Samples that are collected may not be large enough to have sufficient statistical power, or the sample effect size (e.g., $\bar{x}_{\text{otter}} - \bar{x}_{\text{no otter}} / \sigma$) might be too small, relative to natural variability, to indicate a significant difference (Peterman 1990).

Despite the limitations associated with ecological data, resource managers responsible for managing the geoduck fisheries are obligated to the public and federal government to protect a marine mammal species at risk. From the large amount of biological survey data for geoduck populations in the WCVI, an initial step has been made for a fishery-impact analysis. Analysis is needed to quantitatively assess whether sea otter predation impacts are of sufficient magnitude to threaten the harvests of geoducks in B.C.

Study objectives and project overview

Potential interactions between sea otters and geoduck fisheries have not been investigated quantitatively. My objective in Chapter 2 is to improve our understanding of the potential impacts of sea otter predation on geoduck fisheries on the WCVI, B.C. by assessing the relation between sea otter presence and some fundamental life history parameters of geoduck clams. I use spatial and temporal comparisons of geoduck populations along a gradient of sea otter occupancy to address the following research objectives: 1) to estimate geoduck total mortality rates across a gradient of sea otter abundance and 2) to partition these total mortality rates into estimated fishery and sea otter components. Fishery-independent geoduck survey data from the WCVI and catch curve analysis are used to explore these relationships.

Measures designed to promote sea otter recovery have led to an increasing conflict in resource management, between the legislation for protecting sea otter recovery and conservation of shellfish resources. Geoduck harvesters are certain that sea otters are digging for geoducks and that substantial losses in economic yield should be expected in areas that are being re-colonized by sea otters (G. Dovey, West Coast Geoduck Corp., PO Box 781, Ladysmith, B.C., V9G 1A6, Personal Communication, 2008). Protection of sea otters via a harvesting ban, without a legal requirement to remove and relocate otters from fishery management zones (COSEWIC 2007), and uncertainty about the impact of sea otters on geoduck fisheries, is a concern of the geoduck industry, because over or under-estimating the effect of sea otters can have economic or logistical consequences. I evaluate these policy issues in Chapter 3. First, I address a potential conflict between marine wildlife and fishery policy in British Columbia that leads to the question: what might be done, if anything, to minimize any adverse effects of sea otter recovery on geoduck fisheries?

The federal government has a responsibility to encourage economic activity and development in Canada, such as that related to commercial fishing and development of new fishing opportunities (Parsons 1993). Second, because DFO has no official policy for compensating a voluntary restructuring of fisheries impacted by a species at risk, harvesters have taken up financing. The geoduck fishery is currently financing an assessment of management and potential ecosystem impacts of geoduck aquaculture in the Georgia Basin that is believed to be outside the sea otter's historical range (IFMP 2010). Because equitable allocation of fisheries resources among increasingly diverse stakeholders is a desirable goal (Hanna 1995), the policy question is: 3) How can DFO support the recovery of sea otters without unreasonably limiting opportunities for commercial geoduck fishing?

Chapter 2
Estimating Potential Sea Otter Predation Effects on the West Coast Vancouver Island Geoduck (*Panopea abrupta*) Fishery

Abstract

Fishery-independent survey data, from along a gradient of sea otter occupancy on the west coast of Vancouver Island, were used to calculate repeated, bed-specific estimates for three life history parameters of geoduck clams (*Panopea abrupta*): (*i*) density, (*ii*) mean age, and (*iii*) instantaneous total mortality rate. Retrospective power analysis revealed the geoduck transect data had insufficient power ($\beta < 80\%$) to detect large changes in mean density at the $\alpha = 0.05$ significance level. Linear regression provided strong evidence of a fishing effect on geoduck total mortality ($r^2 = 0.72$, $F_{2,17} = 25.09$, $P = 8.47\ 10^{-6}$), while the main effect of otters was not significant on geoduck total mortality. Mean geoduck age, however, was significantly different between study sites with and without otters ($F_{3,16} = 3.59$, $P = 0.037$), suggesting that a small but undetected predation effect could be operating on geoduck total mortality. A more balanced study design and greater sampling intensity is needed to increase the power to detect whether sea otters affect geoduck density and total mortality. High priority should be given to acquiring fishery catch-at-age data from the recently opened geoduck beds in the B.C. north coast in areas without sea otters. Samples from unfished geoduck populations will provide a better measure of how geoduck total mortality rates change when harvesting removes a known age-distribution and when sea otters eventually arrive.

Introduction

Increases in the number of natural predators may be influencing prey populations in ways that are important to conventional fisheries stock assessment and management (Jurado-Molina et al. 2005; Hollowed et al. 2000). Along the west coast of Vancouver Island (WCVI), for example, sea otters (*Enhydra lutris*) were successfully reintroduced in the 1970s, and their expanding populations may now conflict with the management of high-value invertebrate fisheries that developed in their absence (Watson 2000; COSEWIC 2007). Despite considerable scientific research on the trophic ecology of sea otters (e.g., Estes and Palmisano 1974; Estes et al. 1978; Kvitek and Oliver 1992), we currently know little about how sea otters directly or indirectly affect invertebrate fisheries, due to a lack of quantitative studies aimed at assigning the causes of shellfish mortality to different sources, including sea otter predation and fisheries (Watson 2000). By partitioning the estimated geoduck total mortality into sea otter predation and fishery components, the trade-offs between sea otter recovery and fishery yield of the sea otter's invertebrate prey can be assessed.

Geoduck (*Panopea abrupta*) populations in B.C. cover approximately 26,400 ha of coast (DFO 2000) and support a profitable commercial dive fishery (Zhang and Hand 2006). The geoduck is a large, deeply burrowing and extremely long-lived clam (> 100 years) (Harbo 1999; Bureau et al. 2002). Geoduck clams can burrow to a depth of 1 m in a variety of unconsolidated substrates, but tend to concentrate in large numbers in mud-sand or sand (Campbell et al. 1998) in beds of 20 to 50 ha (Orensanz et al. 2004) and at depths ranging from inter-tidal to 100 m (Bureau et al. 2002). Fisheries and Oceans

Canada (DFO), the Underwater Harvesters Association (UHA) and First Nations began annual fishery-independent surveys of the exploited beds in 1992 to estimate geoduck density (IFMP 2010). Commercial harvests and dive surveys target geoduck stocks from 3 m to the maximum commercial diving depth of 20 m (Campbell et al. 1998; Bureau et al. 2002). As a result, scientists and resource managers know little about geoduck densities beyond this range of depths (Campbell et al. 1998). Geoducks living at greater depths, or in harder substrates, are unsurveyed and are probably unexploited (Campbell et al. 1998; Bureau et al. 2002). The connection between increasing numbers of sea otters and the geoduck populations is presently undetermined, but many areas that are inaccessible to the fisheries may also be refugia from sea otter predation.

In shallower waters where the fisheries occur, geoduck mortality from benthic predators (e.g., fish, sea stars, crabs and snails) is high during the larval and juvenile stages (Goodwin and Pease 1989; Feldman et al. 2004), but decreases considerably after four or five years when geoduck reach their maximum refuge depth (Sloan and Robinson 1984; Goodwin and Pease 1989). While the estimated average natural mortality rate (M) for geoduck populations varies between 0.014 and 0.054 yr^{-1} (Orensanz et al. 2004), Zhang and Campbell (2004) estimated M to be 0.036 yr^{-1} (SE 0.003 yr^{-1}) for the WCVI region. Zhang and Hand (2006) subsequently surmised that adult mortality was higher in regions in the WCVI where sea otters are established.

DFO and the UHA co-manage the geoduck fishery in B.C. on a bed-by-bed basis (IFMP 2010). The long-term management goal is to maintain a bed’s fishable biomass at above 50% of the estimated unfished level over a 50-year period (IFMP 2010; Zhang and Hand 2007). Although the coast-wide total allowable catch (TAC) for 2010 is unchanged from recent years (1,574,454 kg), the TAC and number of licences for the WCVI declined in 2007 (IFMP 2010). The UHA and DFO suspect that declines in the stock estimates for this area are due to a history of intense commercial harvests and increased sea otter predation (IFMP 2010; Zhang and Hand 2007). Presently, geoduck harvesters are concerned that management based on biological reference-points is meaningless where sea otters are present. This is because the addition of predation mortality may cause a fished stock to fall below the limit reference point of 40% of virgin stock size, at which point fishing is not permitted (IFMP 2010).

Optimal habitats for sea otters are relatively shallow (< 40 m), are near shore (< 1-2 km) (COSEWIC 2007), and overlap the fishery. Sea otters are recognized as a "keystone species" because they contribute to the structure and function of both rocky and soft-bottomed habitats by reducing the size and abundance of invertebrate species (DFO 2007b). Communities of clam species in soft-bottomed habitats are currently abundant in B.C. and are important resources for sea otters (DFO 2007b). Despite being a potential competitor with commercial harvesters, however, the sea otter is protected by law and designated as being of special concern ("species at risk") under the federal *Species at Risk Act* (*SARA*) (Species at Risk Public Registry 2011), and as a threatened species under the *British Columbia Wildlife Act* (MNRO 2010a). The *Marine Mammal Regulations* of the federal *Fisheries Act* also directly protects sea otters by making it an offence to kill, harm, or harass a marine mammal.

Harvesters in B.C. have increasingly reported seeing sea otters eating geoduck along the WCVI, so there is a need for a formal evaluation of the potential impact of sea otters on the fished geoduck populations. Quantitative analysis of the effects of sea otter predation on these populations presents several statistical challenges. A leading challenge is in the complexity of large-scale ecosystems where controlled experimentation is often impossible (Hilborn and Mangel 1997). In such situations, the assumptions of classical statistics that rely on strict experimental design and large sample sizes are difficult to meet (Clark 2007), and this ultimately leads to high levels of measurement error and uncertainty (Cressie et al. 2009). Samples that are collected may not be large enough to have sufficient statistical power, or the sample effect size (e.g., $\bar{x}_{\text{otter}} - \bar{x}_{\text{no otter}} / \sigma$) might be too small, relative to natural variability, to indicate a significant difference (Peterman 1990).

Despite the limitations associated with ecological data, resource managers responsible for managing the geoduck fisheries are obligated to the public and federal government to protect a marine mammal species at risk. An initial step has been made for a fishery-impact analysis from the large accumulation of biological survey data for geoduck populations in the WCVI. Analysis is needed to quantitatively assess whether sea otter predation impacts are of sufficient magnitude to threaten the harvests of geoduck in B.C. This study had two primary objectives: (*i*) to estimate geoduck total mortality rates across a gradient of sea otter abundance and (*ii*) to partition these total mortality rates into estimated fishery and sea otter components. Fishery-independent geoduck survey data from the WCVI and catch curve analysis are used to explore these relationships.

Methods

Study area

This study examines the potential impacts of sea otters on geoduck fisheries on the Pacific coast of Vancouver Island, between Quatsino and Barkley Sounds (Figure 1). The region extends approximately 270 km from the northwest end of Vancouver Island southeast to Barkley Sound. The study region thus includes the area where sea otters were introduced near Kyuquot Sound in the early 1970s, and where otters have recently (i.e., 2004) expanded their range in Clayoquot Sound near Tofino. Some areas in Clayoquot Sound and all areas in Barkley Sound are presently outside the sea otter's foraging range. Most geoduck beds in the study region have comparable abiotic habitat components where the oceanographic conditions (e.g., water temperature, salinity, and density) and physical structure (e.g., depth and bathymetry) are similar (Thomson 1991). All geoduck beds are located near-shore in relatively high-energy islet and rocky reef environments and in less than 40 m depth.

Study design and site selection

In collaboration with the UHA and DFO, I selected study sites based on: (*i*) a relatively long time series of commercial geoduck fishing up to the dates in the analysis, (*ii*) similarity of bio-physical conditions, (*iii*) the presence and absence of sea otters, and (*iv*) availability of at least two years of survey data for the same geoduck bed. A long time-series of fishing was important for consistency, because survey data were available only for fished geoduck beds, and some beds have been fished longer than others have. A "study site" is a specific geoduck bed, or portion thereof, within a larger fishing area. Study sites classified "without otters" are those where: (*i*) no otters or very few sea otters have been observed by commercial harvesters; (*ii*) evidence is rare or non-existent of sea otter foraging (e.g., empty geoduck shells laying on the surface or otter holes in the substrate); and (*iii*) commercial fishing for geoduck clams has occurred over a long period. Sites classified "otters" are those where an abundance of sea otters have been reported in the area by commercial fishers or observed during otter surveys, and where evidence of otter foraging is generally common.

Although no bed-specific, repeated survey sites were available in Barkley Sound, I included these sites without otters for four reasons. First, they were the only other surveyed sites on the WCVI with a long time series of commercial geoduck fishing that are being fished today, and which are in a similar habitat to the other study sites with matching records in Quatsino, Kyuquot and Clayoquot Sounds. Second, they have both survey and age composition data available for the same geoduck bed in the same survey year. Third, they were relatively close in proximity to each other. Fourth, three of the sites were surveyed in 2002 and the others were surveyed in 2005. Therefore, these additional sites provide a useable time series in an area without otters for the comparison.

Although biological samples of geoducks have been collected throughout B.C. since 1992, I restricted my evaluation to be between 1995 and 2008, for two reasons. First, no matching records of bed-specific, repeated surveys were available for the WCVI prior to 1995. Being able to compare matching records was necessary for a temporal analysis. Second, 2008 was the most recent survey year with at least one site meeting the above criteria. This selection process resulted in 17 study sites within 8 commercially fished locations on the WCVI. Seven of these sites have geoduck age-composition data from two different survey years for comparison. The sites ultimately provided 16 survey years in otter areas in Quatsino, Kyuquot and Clayoquot Sounds, and 12 survey years in the areas without otters in Clayoquot and Barkley Sounds.

Geoduck survey data

Surveys took place over one or two weeks in each survey year. DFO followed a stratified random design, where geoduck beds were the strata and transect locations were drawn randomly within them on charts *a priori* (Bureau et al. 2002). Survey divers worked in pairs at one meter on either side of a transect, stopping every few meters to conduct a complete census of 10 m^2 secondary sampling units (quadrats). Eleven sites comprised repeat survey data, while the six supplementary sites in Barkley Sound each had a single survey year of data (Appendix Table B).

The average survey depth ranged from approximately 6 m to 13 m, and the dominant substrates were a mixture of sand and crushed shell (Table 1). I retained Yellow Bank (site *b*) even though its value as a “site” was not very high given one transect, because it also included an age sample from the same portion of the geoduck bed, from which an age sample was obtained nine years later. The number of study sites that had repeated survey data was already minimal, and those with repeated age samples were even less common.

The combined lengths of transects, and thus, the area surveyed in a study site was highly variable between years and among sites, with minimum and maximum areas surveyed of 250 m^2 (1 transect) and 7,040 m^2 (36 transects), respectively. The inconsistencies were largely a result of choosing only transects that overlaid the same portion of a bed in two different survey years. Divers collected a biological sample on the last day of a survey, using standard commercial fishing gear to harvest geoducks from the substrate (Bureau et al. 2002; 2003). In a single survey area, divers selected multiple sub-sampling locations (sample sites) by randomly choosing among the "eligible" transects within the surveyed beds, which collectively contained enough geoducks to comprise a single biological sample (Bureau et al. 2003). Divers attempted to sample all depths along the length of a transect, and to sample from among all possible geoduck sizes (Bureau et al. 2002; G. Dovey, West Coast Geoduck Corp., Personal Communication, 2009). Therefore, the samples are not representative of the commercial fishery, but rather a non-selective fishery-independent survey. All biological samples were aged at the Pacific Biological Station in Nanaimo using an acetate peel method described in Bureau et al. (2002).

Sea otter survey data

Information about sea otter abundance in B.C. is collected by small vessel surveys during summer months and is discussed in Nichol et al. (2005; 2009). Gaps in survey effort exist prior to 2000 and between 2005 and 2007 because of logistical problems (e.g., poor weather) (Nichol et al. 2009; COSEWIC 2007) and limited funding prior to the establishment of Canada's *Species at Risk Act* (J. Watson, Vancouver Island University College Professor, Nanaimo Campus, 900 5th Street, Nanaimo B.C., V9R 5S5, Personal Communication, 2010). DFO determines the location and extent of surveys on the WCVI *a priori* according to the known location of otters (Nichol et al. 2005). The otter range is subdivided into segments where, in good weather, each segment can be surveyed in one day by small boat (Nichol et al. 2009). New segments are added by DFO based on reliable reports (e.g., from fishers) of otter sightings in previously unsurveyed areas. A typical survey is conducted by small boat equipped with a GPS unit that is interfaced with Nobeltec software (Visual Navigation Suite, Nobeltec Corporation) (Nichol et al. 2009). An experienced observer of sea otters, with one or two additional observers on board, always leads a survey, while the assisting observers search for otters on either side of the boat. The lead observer follows a digital track line at a vessel speed of less than 10 knots (18.5 km/hour), and slows or stops the vessel to count animals (Nichol et al. 2009). The recorded data include the location and number of single and rafted otters, with female rafts discernible from male rafts by the presence of pups.

Number of sea otters in study sites

My geoduck study sites are very small (all < 8,000 m^2) relative to the foraging range of sea otters. I therefore used the nearest abundance estimates in Nichol et al. (2009; Table 2) to represent the number of otters in the geographic region of each study site around the time of the geoduck surveys (Appendix Table A). In my analysis, the "Kyuquot region" comprised three sea otter survey segments in Nichol et al. (2009). I summed the counts from all three segments in the year of a geoduck survey to obtain a single abundance estimate for the region (Table 1). In addition, no abundance estimates were available in Nichol et al. (2009) in the first geoduck survey year for the Kyuquot and Quatsino regions in 1998 and 1996, respectively. For those sites, I used the most recent sea otter abundance estimates prior to the geoduck survey year.

Data analysis

I used the Chapman and Robson (1960) method for estimating geoduck instantaneous total mortality rate (Z) from the age-frequency data. Repeated-measures analysis of variance (ANOVA) and ordinary linear regression analyses with at Type I error rate ($\alpha = 0.05$) were then used to assess relationships between estimates of Z and geoduck density, mean age and the number of sea otters.

Analysis of geoduck densities

For each study site, the mean density of geoducks in a survey year was the total number of geoduck siphons counted by survey divers along a transect divided by the total area sampled over all transects. I used a non-parametric bootstrap to obtain a 95% confidence interval for the mean geoduck density for each survey year. For 10 of the 11 study sites with repeated samples, I used a Welch Two-Sample *t*-test of the null hypothesis **H_0: the mean geoduck density in each site for the two survey years was equal.** The lack of adequate transect data for the first survey year at Yellow Bank (site *b*) resulted in its exclusion from the comparison.

Because geoduck recruitment oscillates over decades and is linked to environmental variability, a repeated survey of these particular study sites may be important for detecting statistically significant changes in bed densities due to sea otter predation. I used power analysis for a two-sample *t*-test to estimate the average minimum number of transects required to detect a 50% change in mean density in a future survey of the study sites. The power calculation was based on the sample size and standard deviation of the second survey year data (and the conventional $\alpha = 0.05$ and power $= 1 - \beta = 0.80$). This study design could lead to violations of important assumptions that underlie power analysis: equality of variances between samples, independent observations, and normally distributed data (Kupzyk 2011). Repeated measures of geoduck probably preclude these assumptions, especially given their patchy distributions along the B.C. coast (i.e., non-normality of metapopulations). Thus, the minimum number of transects are presented as under-estimates.

Analyses of age composition data

Age-composition sample size ranged from 141 to 562 geoducks per sample. In this analysis, a “sample” was comprised of one or more sub-samples taken from a single study site, within a larger surveyed area. A typical survey area contained either one very large geoduck bed, or many smaller beds. During dive surveys, the identification of transect locations varied from year to year, mainly due to changes in weather and sea conditions at the time of the survey. Consequently, divers surveyed within the same geographic area, but not necessarily the same area of a bed as in the preceding survey year (G. Dovey, West Coast Geoduck Corp., Personal Communication, 2009). Therefore, some study sites comprised only a portion of a geoduck bed that had been surveyed twice, with a single sub-sample of fewer than 200 clams collected each survey year. I assumed every sample was representative of the age composition of the geoduck population of the study site from which it was taken.

Geoduck mean age was statistically compared between sample years for the sites with paired samples. No paired age samples were available for the sites without otters in Barkley Sound. Most of the data did not meet the assumptions of equal variances and normality even after data transformation. Because non-normality and outliers could lead to erroneous conclusions using conventional parametric assessments (Crawley 2005), I used the Wilcoxon rank sum test as a non-parametric alternative to the two-sample *t*-test of the null hypothesis **H_0: the mean age of geoducks between the two survey years was the same.** I then used power analysis for a *t*-test to estimate the average minimum sample size required to detect at least a 30% change in mean age based on the mean and standard deviation of the second survey samples (and $\alpha = 0.05$ and $\beta = 0.80$). Minimum sample sizes for Barkley Sound were calculated based on the first age sample from those sites. Without making an explicit assumption about the distribution of the data (i.e., I assumed only that the data are non-normal), I report the sample sizes required for a *t*-test in Table 3, and recommend adding 15% for analyzing the data with a nonparametric test (i.e., Wilcoxon rank-sum test) (Lehmann 2006).

Catch curve analysis

In fisheries, the numbers of younger fish in commercial catches are well documented to often exceed the numbers of older fish (Hilborn and Walters 1992). Consequently, a graphical plot of the age-frequency distribution of a sample of catch (i.e., a catch curve) typically describes an ascending left limb, a domed middle segment, and a descending right limb. Hilborn and Walters (1992) ascribe this pattern to an increasing vulnerability to the fishing gear (ascending limb), which peaks as the younger individuals become “fully recruited” to the fishery, and to natural mortality and harvesting (descending limb), which effectively reduces the number of older individuals in the population over time. Therefore, the age composition of an exploited species is an indicator of the effect of harvesting that can be used to provide information about the total mortality rate on the stock (Hilborn and Walters 1992).

For commercially fished species, the natural mortality cannot be separated from the effects of harvesting unless there is a substantial change in the fishing mortality (Hilborn and Walters 1992). Catch curve analysis is a useful approach for estimating the average instantaneous total mortality rate (Z) of geoducks in a sample if the following assumptions are met (Hilborn and Walters 1992; Dunn and Doonan 2001): (*i*) annual recruitment is constant amid continuous natural and fishing mortality for all sites; (*ii*) no emigration or migration to the population; (*iii*) no stochastic variation in Z over time; (*iv*) geoducks in a sample are accurately assigned an age (aging errors would be an expected source of bias) (Dunn and Doonan 2002); and (*v*) the composition of a sample is proportional to abundance of the age classes (Hilborn and Walters 1992) for a specific geoduck bed. Chapman and Robson (1960) recognized that catch-at-age on the descending limb of a catch curve follows a geometric probability distribution from which the maximum likelihood estimator for the survival parameter (S) can be obtained in closed form. An estimate of the instantaneous total mortality rate (Z) can thus be derived by taking the natural logarithm of the survival rate $Z = -\log(S)$ (Chapman and Robson 1960).

I applied the Chapman and Robson (1960) estimator to estimate the total mortality rate, Z, for each bed-specific geoduck age sample and survey year. The Chapman-Robson method is a regression-based, minimum variance, unbiased estimator for the survival parameter $S = e^{-Z}$ based on age-frequency data (Dunn and Doonan 2002). This method assumes that some reference age exists above which vulnerability to the fishing gear is constant. The ages are recoded so that this reference age is equal to zero (Seber 1982). According to Chapman and Robson (1960) and Robson and Chapman (1961) this estimator of the survival parameter is

$$(1) \qquad S = \frac{\overline{X}}{1 + \overline{X} - 1/n}$$

where $\overline{X}$ is the mean re-coded age of the sample and n is the sample size. The Chapman and Robson (1960) estimator of Z is

(2) $Z = \log\left(\frac{1}{S}\right)$ = - log (S)

Chapman and Robson (1960) further showed that the variance of their estimator was approximately equal to the minimum bias (Dunn and Doonan 2002)

(3) bias (Z) = $\frac{\left(1-e^{-Z}\right)^2}{ne^{-Z}} \approx$ var (Z)

I compared the results from catch curve analysis between survey years and across sites to look for changes in the geoduck total mortality rates where otters have expanded their range.

Simulation model

The age-composition sample sizes appeared small relative to the number of age classes in the geoduck sample populations, so I investigated the performance of the estimator in a simulation model. I used a parametric bootstrap to simulate estimates of *Z* and calculate confidence intervals for comparison to the Chapman-Robson estimates of *Z*. The algorithm involved the following six steps: (1) each age-composition sample was re-coded so that age-x corresponded to re-coded age-0; (2) the age-vector was passed through an estimator function that calculated the age-frequency in each age class, from the youngest (set at > 10 years to cover the range of assumed recruitment ages of 8-10 years: Orensanz et al. 2004) to the maximum age in the sample, and counted the number of ages in the sample (*n*) and calculated the mean age ($\overline{X}$); (3) the estimator function then calculated the Chapman-Robson estimate of *Z* (using equation 2) and the variance (using equation 3), as well as the standard error that was easily calculated as

(4) se (Z) = $\sqrt{\mathrm{var}(Z)}$

To assess sampling variation, I included a calculation for the coefficient of variation of *Z* in the estimator function by

$$\text{(5)} \qquad \text{cv}\,(Z) = 100\,\frac{\sqrt{\text{var}(Z)}}{Z}$$

which is the standard deviation as a percent of the estimate of *Z* for a sample. The next step was to (4) simulate estimates of *Z* by sampling from a simulated population with a known total mortality rate and sample size (derived from the above estimator function). The simulation function generated 1,000 random age frequency data sets from an exponential distribution based on Dunn and Doonan (2002), passing each new data set to the estimator function. (5) A confidence interval was estimated for the average simulated *Z* by calculating the 2.5th and the 97.5th percentiles, and (6) the simulated *Z* (Z_{sim}) was compared to the Chapman-Robson estimate of *Z* for each of the samples by comparing the percent bias (%bias)

$$\text{(6)} \qquad \%\text{bias} = 100\,\frac{\left(Z - Z_{sim}\right)}{Z}$$

Analysis of the effects of sea otters

Using a repeated-measures analysis of variance (ANOVA) I tested the three null hypotheses that geoduck density, mean age and total mortality are the same in sites where sea otters were present and absent. I then used regression analyses to assess the associations between: (1) geoduck density and total mortality, (2) geoduck density and the number of sea otters, and (3) geoduck total mortality and the number of sea otters. There were no data on the catch per unit of effort from the fishery in the study sites (D. Bureau, Fisheries and Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Road, Nanaimo B.C., V9R 5K6, Personal Communication, 2010). I therefore summed the total site-specific landings from the beginning of fishing (1980s), up to the first and second survey years of the site. In sites where no fishing occurred in the year of a geoduck survey, I summed the total landings up to the most recent fishing year, prior to the year of a survey. Total diver fishing hours (thousands of hours) for the study sites almost perfectly predicted total fishery landings (millions of pounds) from the same sites ($r^2 = 0.99$, $F_{1, 11} = 966.9$, $P = 4.51\ 10^{-12}$) (Figure 2), indicating that the catch data was a good index of fishing effort. I thus used fishery "landings" in place of fishing effort in the analyses. I used the fishery-independent estimates of Z in a linear regression to test simultaneously the null hypotheses H_0: $\beta_L = \beta_O = 0$ where β_L and β_O are the coefficients for the variables "landings" and number of "otters" in the regression model respectively.

Results

Analysis of geoduck densities

Minimum and maximum mean geoduck density ranged from 0.17 to 2.22 m^{-2}, in the two otter areas near Kyuquot and Quatsino Inlet, respectively (Table 2a). Mean geoduck density in a site without otters in Yellow Bank (site *a*) decreased significantly from 2.15 to 0.81 m^{-2}, from the first survey in 1995 and the second survey in 2006 (t = 5.7, $P < 0.05$) (Table 2b). No statistically significant differences were observed in mean geoduck density between survey years for 9 out of 10 sites with repeated density surveys (all $P > 0.05$), although mean densities tended to decrease between survey years in the otter sites near Kyuquot (sites *a* and *b*), as well as in an otter site in Forward Inlet (site *b*). All other sites showed slight non-significant increases in density.

In order to interpret the non-significant results, I used retrospective power analysis for an unbalanced two-sample *t*-test (at the conventional $\alpha = 0.05$ and power = $1 - \beta = 0.80$) and calculated Cohen's standardised effect size (d) in each site using formulae in Nakagawa and Cuthill (2007; Equations 1, 2, 14 and 16). Cohen (1988) suggested that effect size values of d=0.2, 0.5, and 0.8 standard deviation units represent small, medium, and large effect sizes respectively. Thus, Cohen's d (Table 2a, 2b) is a measure of the size of the difference between two means, relative to the standard deviation in the data (Cohen 1988; Nakagawa and Foster 2004). For the majority of study sites, the statistical tests had insufficient power ($\beta < 80\%$) to detect "large" changes in mean density (i.e., when $d = 0.8$), and 95% confidence intervals around the d-value included zero, indicating no statistical difference between means. The result for Yellow Bank (site *a*) was statistically significant using both hypothesis testing and the effect size approach, although the effect size range is wide because of the small sample size. Only the otter site near Tofino and the site without otters in Millar Channel had sufficient power ($\beta \geq 80\%$) to detect a large change in mean density. At the extreme, a 14% chance existed of detecting a large change in mean density in Forward Inlet (site *a*), which had fewer than 5 transects in either survey year. Millar Channel (site *a*) had a 91% chance of detecting a change of 80% or more as a result of the large number of transects ($n > 30$) in both survey years, but the effect size was "tiny" ($d < 0.2$). The average minimum number of transects required to detect a 50% change in mean density in a future survey of these particular study sites ranged from 14 to 82 transects and are assumed to be under-estimated (Table 2a, 2b).

Analysis of geoduck age composition

Geoduck age samples spanned well over 100 years, ranging from 2 to 152 years and are presented in absolute terms rather than density (Table 3; Appendix Table C). Median and mean ages ranged from 9 to 63 years and 12 to 60 years, respectively. Of the seven sites with repeated age samples, only the otter site near Tofino did not show change in the age-distribution between survey years (2004 and 2008) (Figure 3a, 3b). In both survey years for this site, more than 80% of the sampled geoducks were under 20 years old, and less than 1% older than 60 years. Age samples from the two sites without otters (Millar Channel and Yellow Bank) also showed little change in the age distribution between survey years.

Across all sites, the median age for the otter sites near Kyuquot, Forward Inlet, and Tofino was lower than the median age for all sites without otters (Figure 4). Over 75% of the geoducks sampled in these particular otter sites were ≤ 20 years old in the second survey year, and fewer than 5% were older than 60 years. The 25%-75% quartile of the age distribution from the otter site at Forward Inlet shows a marked shift in the second survey toward a younger age distribution. The otter sites at Quatsino Inlet appeared anomalous, as they had older median ages and a greater spread than those of all other sites with repeated age samples. The age distribution of the biosamples from the two sites at Quatsino Inlet also shifted in the second survey year (toward a younger distribution), but the majority (> 50%) of geoducks sampled in both survey years were older than 20 years, with 30% older than 60 years.

The Wilcoxon test provided strong evidence against the null hypothesis of no difference in mean geoduck age between survey years for 4 of the 5 otter sites and for the site without otters in Millar Channel (all $P < 0.05$) (Table 3). No evidence was found to reject the null hypothesis of no difference in mean age between survey years for the otter site near Tofino (W = 38769, $P = 0.816$), or the site without otters in Yellow Bank (site *b*) (W = 40253, $P = 0.474$). In both cases, retrospective power analysis revealed that power was sufficient to detect a “small” difference in mean age when $d = 0.3$. For all sites, the effect size was zero because of large standard deviations and high variability in the age samples. The 95% confidence intervals around the *d*-value indicated the difference between samples was too small to have a biological (causal) effect, thus “no change” between survey years was just as well supported (Table 3). Finally, though the sites without otters in Barkley Sound do not have repeated samples for the statistical comparison, the observed median and mean age of geoducks was older in Barkley Sound than in all other sites (not including Quatsino Inlet). In a repeated sampling of these particular study sites, the average minimum sample size required to detect a 30% change in mean age ranged from 14 to 178 geoducks (Table 3). Ideally, however, sample size should be large (> 200 geoducks) to increase statistical power, and very large (possibly > 1,000 geoducks) for reliable catch curve analyses (Bradbury and Tagart 2000).

Estimates of geoduck instantaneous total mortality rate

Simulations confirmed that the Chapman-Robson method was nearly unbiased for the mortality estimates based on the sample sizes observed in this study (Table 4). Geoduck total mortality estimates ranged from 0.0158 yr^{-1} (SE ± 0.0014) in a site with otters in Quatsino Inlet (site *a*) in 1996 to 0.0478 yr^{-1} (SE ± 0.0044) in the otter site near Tofino in 2008. The otter site near Tofino had high estimates of Z in both survey years compared to the other sites with repeated age samples. In addition, the otter site near Kyuquot (site *a*) had the greatest increase in total mortality between the first survey in 1998 ($Z = 0.0280$ yr^{-1}, SE ± 0.0024) and the second survey in 2003 ($Z = 0.0445$ yr^{-1}, SE ± 0.0029). Estimated total mortality for the otter site in Forward Inlet (site *a*) and the site without otters in Yellow Bank (site *b*) both showed a similar magnitude of increase between surveys. Geoduck age-composition near Tofino, Kyuquot and Forward Inlet produced higher estimates of mortality because of the apparent scarcity of individuals older than 20 years in the second survey years.

The estimates of total mortality for the sites without otters in Barkley Sound and for the sites with otters in Quatsino Inlet were consistently lower (almost all $Z < 0.0200$ yr^{-1}) due to a strong presence of older age classes. Although the average Z was only slightly higher for sites with otters ($Z = 0.0291$ yr^{-1}, SE ± 0.0027) than without ($Z = 0.0234$ yr^{-1}, SE ± 0.0018), it was distinctly higher for the otter sites after removing Quatsino Inlet (average $Z = 0.0369$ yr^{-1}, SE ± 0.0034).

Analysis of the effects of sea otters

Mean geoduck density was not significantly different ($F_{3,24}$ = 1.61, P = 0.213) between sites with and without otters, although the main effect of otters was almost significant (P = 0.067). The interaction effect between the first and second survey years and the presence of otters was also not significant (P = 0.557). Similarly, estimated geoduck total mortality between sites with and without otters was not significantly different ($F_{3,16}$ = 1.02, P = 0.410). Neither the main effect of otters nor the interaction effect with year on Z was significant (P = 0.456 and P = 0.926, respectively). In contrast, mean geoduck age was significantly different between study sites with and without otters ($F_{3,16}$ = 3.59, P = 0.037). The main effect of otters on mean geoduck age was significant (P = 0.042), with an estimated effect size of -23.92 years (SE ± 10.84). However, the interaction effect of survey year and otters on mean geoduck age was not significant (P = 0.788).

Linear regression provided little evidence that mean geoduck density affects total mortality (r^2 = 0.01, $F_{1,18}$ = 0.776, P = 0.390), or that the number of otters affect mean geoduck density (r^2 = 0.01, $F_{1,18}$ = 0.727, P = 0.405). Most notable was the strong evidence of a fishing effort effect on geoduck total mortality (r^2 = 0.72, $F_{2,17}$ = 25.09, P = 8.47 10^{-6}). The minimal adequate model did not include an interaction effect between fishery landings and the number of otters. The main effect of otters was not significant (P = 0.356), but fishery landings was significant at P = 1.80 10^{-5}. The resulting regression equation is

$$Z = 1.86\ 10^{-2} + 5.06\ 10^{-3}\ (\text{landings}) - 5.94\ 10^{-6}\ (\text{otters})$$

which explains the observed variation in geoduck total mortality rate estimates fairly well (r^2 = 0.73, $F_{1,18}$ = 52.98, P = 9.16 10^{-7}) (Figure 5).

Discussion

In British Columbia, resolving the debate about whether sea otter predation is affecting the exploitable geoduck stocks is difficult because of the lack of quantitative information needed to disentangle otter predation mortality from natural and fishing mortality. Linear regression determined that fishing effort is strongly associated with higher geoduck total mortality, while the number of otters did not show a significant effect on geoduck total mortality. High abundances of sea otters occurred where commercial fishing effort is greatest; thus, a predation effect might be too small, relative to commercial fishing, to indicate a significant effect of sea otters on geoduck mortality. While the results of this study suggest that sea otters may not threaten the commercial harvest of geoduck, harvesters have increasingly reported seeing sea otters eating geoduck. A predation effect that is small and difficult to detect could be operating on geoduck total mortality given that geoduck age distribution was significantly younger in study sites where sea otters were present, versus absent. Further investigation is needed to improve our understanding of the predator-prey relationship between sea otters and geoduck fisheries.

Assessing whether mortality from predation and fishing are additive or non-additive factors in geoduck total mortality is relevant for determining sustainable geoduck harvests where sea otters are established. Under the additive-mortality hypothesis (Williams et al. 2002), geoduck total mortality is the sum of the independent predation and fishing mortality rates. Predation mortality could also be compensatory to fishing mortality where the mortality rates are negatively correlated (Servanty et al. 2010). In a compensatory relationship, the total mortality remains unchanged at a variety of low predation rates, due to compensating forces, but increases beyond a threshold mortality rate (Williams et al. 2002). Possible mechanisms of compensatory mortality include: recruitment to geoduck fisheries from spawning refuge populations (e.g., unexploited populations in deep water or harder packed substrates); otters eating clams pulled by divers and discarded on the surface (i.e., high grading), which is a concern in the geoduck fishery (Orensanz et al. 2004); or otters selectively foraging on lower quality geoduck (e.g., old or dying geoduck clams that are easily captured). Allen et al. (1998) hypothesized that strong recruitment events in a cyclical sportfish population produced peak abundances that exceeded their carrying capacity, leading in turn to density-dependent mortality of adults and compensation of moderate annual exploitation by anglers.

Recruitment pulses are reported for geoduck populations but on much greater time scales (Bradbury and Tagart 2000; Bureau et al. 2002; Orensanz et al. 2004). Compensatory post-dispersal density dependence for geoduck stocks in B.C. and Washington State may occur, but the supporting evidence is nonexistent (Orensanz et al. 2004). One difficulty in distinguishing additive from non-additive mortality in the geoduck survey data, however, is the strong effect of fishing, so data is needed about geoduck survival in the absence of fishing. Apart from regional estimates of geoduck unfished biomass that have been back-calculated under the assumptions of constant natural mortality and recruitment (e.g., Zhang and Hand 2006), no quantitative information is available about the geoduck populations in the WCVI prior to the fishery. In addition, conventional age-structured single species stock assessment methods (e.g., virtual population analysis (VPA) and statistical catch-at-age models) are useful tools for exploring past and future demographics of fished populations (Hilborn and Walters 1992), but they assume additive mortality that may over-estimate harvestable yields if mortality is, in fact, compensatory (Allen et al. 1998).

Catch curve analysis and age-structured models share several assumptions, such as closed populations, constant natural and fishing mortality, constant recruitment and vulnerability, and unbiased samples. Catch curve analysis further assumes no aging errors or stochastic variation in total mortality over time. Despite these strong assumptions, geoduck total mortality rate can be estimated quickly using catch curve analysis and is most relevant at the scale of a geoduck bed, rather than for a region or the entire coast. However, while it is useful for generating estimates of total mortality for comparing where sea otters are present and absent, catch curve analysis may not reveal general short-term trends in geoduck mortality for the coast, given the extremely slow-paced dynamics of geoduck populations (Orensanz et al. 2004).

Dunn and Doonan (2002) found the Chapman-Robson estimator is robust for determining total mortality from catch curves relative to other estimators, although it performs less well with increasing variation in recruitment and total mortality. In addition, however, geoduck populations comprise many year classes so that the age sample size might be too small to contain enough data to be truly geometric in distribution for the estimator (e.g., < 1,000 geoducks) (Bradbury and Tagart 2000). Bradbury and Tagart (2000) suggested that larger sample size, eliminating outliers (i.e., extremely old geoducks), and applying equal weighting could account for variability due to differences in year-class strength in catch curve analysis. Maceina and Bettoli (1998) demonstrated using a weighted regression with the catch curve method.

Estimates of geoduck total mortality

Estimates of total mortality were similar to the published estimates of mortality for geoduck populations in B.C. and in Washington State, varying between 0.010 yr^{-1} and 0.040 yr^{-1} (Breen and Shields 1983; Bradbury and Tagart 2000; Noakes and Campbell 1992; Orensanz et al. 2004; Zhang and Campbell 2004). Zhang and Campbell (2004) estimated geoduck mortality in B.C. to be 0.036 yr^{-1} or 0.039 yr^{-1} using Bayesian and Monte Carlo methods, respectively. The current estimated harvest rate used in the WCVI (1.8%) is within the range of estimates provided by Zhang and Hand (2007), which were based on natural mortality rates of up to 0.036 yr^{-1} using age-structured projection modelling. My analysis supports Zhang and Hand's concern that this rate was higher when sea otters co-occurred with the fishery, as the estimates of total mortality were noticeably high for two otter sites ($Z > 0.044$ yr^{-1}), although total mortality was similarly high for two sites without otters ($Z > 0.036$ yr^{-1}).

Sea otter foraging success and feeding rate on B.C. geoduck is presently unknown. Zhang and Hand (2007) did estimate a predation rate for the geoduck beds near Kyuquot, where sea otter predation on geoduck is reportedly "severe." By adding different predation rates to M in their projection model, they simulated geoduck abundance until it agreed with the survey-derived estimates. Zhang and Hand's estimated predation rate of between 0.150 yr^{-1} and 0.170 yr^{-1} (SE ± 0.019 – 0.013) implies that where sea otters are established, profitable geoduck fisheries are not possible under the current management regime of maintaining a geoduck bed's biomass at above 50% of the unfished level (Zhang and Hand 2007). In my analysis, a comparison of patterns between fishery landings from the study sites and the fishery-independent estimates of total mortality suggests that intensive harvesting, rather than sea otter predation, is probably mostly responsible for higher geoduck mortality rates in the WCVI (Figure 6). One caveat regarding my analysis method is that I retained all older age classes in the catch curves to compare *Z* between survey years in a site, which precluded information about the mortality of younger geoducks (Bradbury and Tagart 2000). My goal was not to construct a complex model but to show that catch curve analysis is a viable tool for measuring how geoduck total mortality rates change when sea otters are present, and to provide timely feedback to fishery managers.

The other variability of interest was the temporal variation in the estimated mean density and age of geoduck populations. However, the study is embedded within complex ecosystems of B.C., so it was not surprising that the geoduck density data did not have adequate power to detect large changes in a study site between survey years. Null hypothesis statistical testing allowed for simple dichotomous decisions (statistically significant or not), but revealed nothing about the magnitude of difference between the estimated means (the effect size) and measurement precision (Nakagawa and Foster 2004; Nakagawa and Cuthill 2007). Small to large effect sizes were indeed revealed using the Cohen's *d* measure of effect size, but the overall lack of a significant effect for the data implied that the temporal differences in mean density were not biologically significant. In contrast, the difference in geoduck mean age between survey years was most often statistically significant due to large sample sizes for the statistical test. Large standard deviations and highly variable ages, however, prevented detecting biologically important temporal differences between the age distributions in a study site.

Of importance to this study is that historical PSP levels within Quatsino Sound suggest a possible flaw in the classification of "otter sites." Sea otters in southeast Alaska can detect and avoid such toxins that accumulate in their prey and switch to alternate and less toxic prey (Kvitek and Bretz 2004). I assumed that sea otters foraged everywhere in Quatsino Sound by 2002. However, PSP levels were reportedly above the sea otter's toxic threshold in Quatsino Inlet in 2002 but not in Forward Inlet (CFIA 2010). If sea otters initially foraged on less toxic geoduck in Forward Inlet or on alternate prey in Quatsino Inlet, then my anomalous otter sites at Quatsino Inlet may actually be sites "without otters." Fishing activity was also historically low in the sites at Quatsino Inlet, relative to most other sites (Figure 6). The goal of this study was to improve our understanding of the potential impacts of sea otter predation on the geoduck fishery. In order to prevent the masking of a predation effect, one should ideally design a study that includes spatial and temporal PSP closures and differences in fishing activity.

Future study design

The majority of geoduck fisheries in B.C. now occur off the remote north coast where harvesters continue to discover new geoduck beds (IFMP 2010). A before-after-control-impact (BACI) study is a common approach for measuring "invasion" effects of species (Parker et al. 1999). A BACI design would involve repeated measures over time, made at a minimum of two geoduck control sites with no sea otters or fishing, and three "impact" sites comprising (*i*) sea otters only, (*ii*) fishing only, and (*iii*) sea otters and fishing. For instance, in just two years, Kvitek et al. (1992) identified differences at sites in Alaska where sea otters had recently become established, and where the abundance and size of infaunal prey decreased with increasing time of otter occupancy. Over time, this network of monitored sites could also provide insight about post-dispersal density dependence in geoduck stocks (Orensanz et al. 2004). Given the high costs of surveying geoducks, however, power analysis through simulation may provide more ecologically realistic estimates of required sample size when designing the study (Hilborn and Mangel 1997; Bolker 2008; Hirner and Cox 2007; Kupzyk 2011). For example, Hirner and Cox (2007) used simulation to determine the sample size required to attain a minimum 80% probability of detecting a 50% difference in amphibian abundance between lakes with and without rainbow trout (*Oncorhynchus mykiss*) in the southern interior of B.C. (at the standard $\alpha = 0.05$). Following their method, geoduck density estimates could be simulated where the true density of geoducks is set at 50% less in sites with otters, and then determine if a hypothesis test using the data is able to detect a significant difference in geoduck density between site types.

Field studies are also challenged by the spatial and temporal variability in species distribution patterns and spatial autocorrelation (MacKey and Lindenmayer 2001; Legendre et al. 2002). Geographic scale is, therefore, important for determining the overall impact of an invading species, where range size, average abundance, per-individual effect, and correlations among these factors are essential to the analysis (Parker et al. 1999). These factors are difficult to measure and vary by area (Parker et al. 1999), so it may not be possible to extrapolate the results from a BACI study to other regions in B.C. Alternatively, a modelling framework could be used to define a robust area-specific prediction of sea otter impacts despite uncertainty. Simulating the system is useful for evaluating future study designs (Bolker 2008), as well as different management options, given that biological reference points are questionable for geoduck (Orensanz et al. 2004). The impact of sea otters on geoduck harvests may be smaller than expected; thus, an operating model that allowed for a range of spatial and temporal variability covering the range of our understanding of sea otter and geoduck population dynamics may be useful for management decisions.

Limitations and extensions of the current study

Post-hoc analyses of ecological phenomena rarely provide complete data (Hilborn and Mangel 1997). I based the selection of study sites on the availability of survey data that, by chance, matched a repeated survey of the same portion of a geoduck bed in two different survey years. Because site selection excluded potentially valuable information from the geoduck database, statistical power would have been increased considerably if more transects and new biological samples were available for comparison of the study sites. In addition, sample size in the calculations of geoduck total mortality was given by the number of geoduck older than 10 years in the age-composition samples. "Trimming" the sample size in this way results in lower statistical power to detect significant differences in geoduck total mortality in sites where sea otters were present and absent.

Another problem was that I based my estimates of mortality on age data collected years ago and from areas where sea otters had barely established themselves at that time. Although sea otter occupation of "otter sites" was important for the analysis, a recent estimate of geoduck age was available only for the Tofino area (2008). Furthermore, all of my otter sites are in areas that have since become important feeding habitats for sea otters (DFO 2007b) and new data may provide higher power for defining a larger predation mortality effect, if one exists.

Geoduck-sampling methods also deserve particularly careful consideration when making inferences about the age structure of a population. For instance, because very small geoduck are hard to collect, an unintended sampling bias for the older age classes may have occurred, at least in the earlier surveys of the study sites. Survey divers initially reported losing many juvenile geoducks (≤ 2 years old) in strong water currents during sampling. The high proportions of younger clams in the second samples might be partly explained by a greater effort by survey divers to retain juvenile geoduck, which would bias comparisons of the mean age between sampling years. Therefore, my conclusions depend not only on sea otters causing increases in geoduck mortality, but also on constant selectivity by the survey divers.

Another useful extension of this analysis would involve a Bayesian regression approach. Classical statistical tests are only one way to describe a relationship and are misleading in some cases (Wade 2000). A Bayesian approach would allow for estimates of uncertainty in the mortality estimates. Specifically, it would use the data to estimate the range of underlying regression slope values and estimate a degree of belief (i.e., posterior probability) in the estimates. Hollowed et al. (2000) used a different Bayesian approach in a multispecies catch-age stock assessment model that accommodated predation mortality for Gulf of Alaska walleye pollock (*Theragra chalcogramma*). Their method relaxed the assumption of constant natural mortality by accounting for predation mortality from other fishes and a marine mammal species. Jurado-Molina et al. (2005) incorporated Bayesian methods into an age-structured multispecies statistical model for the Bering Sea. Nevertheless, fishery catch-at-age data and the eating habits of predators were some of the essential components for either model development, and are currently unavailable for the geoduck fishery and B.C. sea otters.

Potential effects of other recovering predators on geoduck

The impacts from sea otters may be small, but cumulative small effects of other large recovering predators might further increase geoduck mortality. For example, foraging gray whales may become a more significant local effect in the future as the population expands its feeding range in B.C. Small numbers of eastern Pacific gray whales migrate every spring to Clayoquot Sound to feed opportunistically on benthic invertebrates in the soft bottom habitats, at least when planktonic prey are less abundant in the area (Dunham and Duffus 2001; 2002). As the population of gray whales recovers to pre-whaling levels (Rugh et al. 1999), the population may be nearing the carrying capacity of the traditional Arctic feeding areas (Stelle et al. 2008), with more whales exploring southern habitats (i.e., the B.C. coast) along their migration route (Dunham and Duffus 2001; Stelle et al. 2008).

Gray whales are large-bodied predators that require high densities of prey to meet their metabolic requirements and they are well documented to excavate feeding pits in sandy substrates in central Clayoquot Sound (Dunham and Duffus 2001). Whales in Clayoquot Sound typically feed on benthic amphipods (Family *Ampeliscidae*) and benthic ghost shrimp (*Callianassa californiensis*) that form expansive beds in the top layer of soft sediments (Dunham and Duffus 2002) that may be associated with tracts of geoduck. Ghost shrimp inhabit sediments mostly in the intertidal zone, while amphipods live several centimetres below the sediment surface and at depths ranging from subtidal to 35 m (Dunham and Duffus 2002). Dunham and Duffus (2002) found that gray whales in Clayoquot Sound foraged mostly for amphipods in waters approximately 20 m deep, which overlap the depth range of commercial geoduck beds. During the second geoduck survey of the site near Tofino in 2008, divers reported seeing several large gray whale feeding pits in the geoduck bed, in addition to old otter holes. Geoduck buried at shallow depths are also vulnerable to other small benthic predators (e.g., sea stars, fish, crabs and snails) (Goodwin and Pease 1989; Feldman et al. 2004) that are quick to access buried clams after commercial harvests (and possibly whales) disturb sediments (Willner 2006).

Role of learned behaviour and an increasing trend in marine-mammal fishery interactions

Marine mammals are large and intelligent predators (Beverton 1985; Dudzinski et al. 2009). A worldwide total of 74 cetaceans, 31 pinnipeds and the sea otter interact with fisheries (Matthiopoulos et al. 2008). An increasing trend in some fisheries is depredation, in which marine mammals remove or damage fish caught in fishing gear, which can result in economic losses for fishermen and incidental mortality of some animals (Read 2008).

Marine mammals communicate through behavioural learning and are “creatures of habit” (Matthiopoulos et al. 2008; Dudzinski et al. 2009), with individual diets mostly determined through opportunistic feeding (Mate et al. 1987; Matthiopoulos et al. 2008; Dudzinski et al. 2009). Matrilineal dietary patterns in sea otters, for example, are transmitted to the dependent young that can be influenced by environmental phenomena at the population level (Estes et al. 2003). Kvitek and Oliver (1992) suggested that sea otters in Alaska had not yet learned to identify the deeply burrowing geoduck as a prey item. The situation might be different for B.C. sea otters, especially given the reports of commercial divers’ hand-feeding geoducks to a foraging mother sea otter with pup, while fishing geoduck near Kyuquot.

Another well-documented interaction is that of pinniped depredation on depressed salmonid (*Oncorhynchus spp.*) populations along the west coast of North America (NMFS 1997; Beeson and Hanan 1996). Salmon harvesters have long considered Pacific harbor seals (*Phoca vitulina*) competitors, where seal-caused damage to gear and catch is common, and bounty programs were used in the past to reduce their numbers (Mate et al. 1987). California sea lions (*Zalophus californianus*) depredate salmonids in river estuaries and the open ocean at rates that coincide with the animal's seasonal migration patterns (Beeson and Hanan 1996). Expanding pinniped populations are creating serious management problems (NMFS 1997; Scordino 2010). Reports of pinnipeds removing salmonids and other species from commercial and recreational fishing gear are increasing in the U.S. (NMFS 1997) and for B.C. salmon farms (Jamieson and Olesiuk 2001).

A different form of depredation is occurring in the southern and northern oceans between longline fisheries, sperm whales (*Physeter macrocephalus*) and killer whales (*Orcinus orca*) (Straley et al. 2005; Ashford et al. 1996; Purves et al. 2004; Kock et al. 2006; Sigler et al. 2008; Hill et al. 1999; Yano and Dahlheim 1995). In the northeast Pacific Ocean, sperm whales and killer whales remove longline catches of bottom fish from hooks, such as sablefish (*Anoplopoma fimbria*), Pacific halibut (*Hippoglossus stenolepis*), and lingcod (*Ophiodon elongatus*) (Straley et al. 2005; Sigler 2008). In the Bering Sea, pods of killer whales have followed vessels from one fishing area to another and depredated the largest fish on the lines (Yano and Dahlheim 1995). Increasing observations of this behaviour are reported for British Columbia longline fisheries (DFO 2010a). Foraging for bottom fish caught on longlines appears limited to individual whales and specific killer whale pods (Kock et al. 2006; Straley et al. 2005; Yano and Dahlheim

1995), and may be altering the natural diets and seasonal movement patterns of some whales (Kock et al. 2006; DFO 2010a). Revenue losses to fishermen may be important given the high market value of some fish species (e.g., B.C. and Alaska sablefish) and subsequent increases in vessel operating costs, such as lost gear, longer time spent fishing, and the associated higher fuel consumption (Sigler et al. 2008; DFO 2010a).

Conclusion

This analysis represents a pilot study to evaluate the use of fishery-independent survey data and catch curve analysis for quantifying the impact of sea otters on the B.C. geoduck fishery. It provides the foundation for future studies of sea otter predation on valuable geoduck resources, and introduces the gray whale as a potentially important predator that could affect geoduck total mortality. Success in managing a fishery largely depends on the mortality response of the fished populations (Allen et al. 1998), and the possibility for compensatory mortality should be investigated in the geoduck fishery. Expanding on the methods used in this study for geoduck beds in the B.C. north coast, such as using simulation to help design a before-after-control-impact study, may provide fishery scientists and managers with a useful real time monitoring tool. Unfortunately, there are no methods yet to control for departures from the unrealistic assumptions in catch curve analysis of constant recruitment and mortality rates (Dunn and Doonan 2001).

Data analysis demonstrated that commercial fishing effort is strongly associated with higher geoduck total mortality, while sea otters showed no statistically significant effect on geoduck total mortality, despite their association with younger geoduck age distributions. Greater sampling intensity at all study sites and the application of the latest sea otter counts would increase statistical power for determining whether sea otter predation impacts are of sufficient magnitude to threaten the harvests of geoduck in B.C. In light of mounting concerns from geoduck harvesters about the expanding sea otter population in the remote north coast, where the majority of geoduck fishing now occurs, further consideration should also be given to acquiring the consumption rate of sea otters on geoduck throughout the year. Trends in predator abundance could then be used to explicitly model predation mortality (Hollowed et al. 2000), and establish appropriate fisheries management decisions regarding geoduck harvests when sea otters are present.

Recommendations for scientists and managers

The intensity and importance of competition between sea otters and the geoduck fishery will depend on the predation rate on geoduck. I have four recommendations for future research and analyses:

1. Because sea otters are not migratory, meaningful management areas can be defined. The fishery is already recording the coordinates of dive sites within fishery management areas, as well as the occurrence of rafts of otters in newly established areas (IFMP 2010). An additional management tool would be to define otter zones that require divers to count the number of otters before each dive. This would require keeping track of "zeros" because records of no otters are just as important as the non-zero values (A. Salomon, Simon Fraser University Assistant Professor, 8888 University Drive, Burnaby B.C., V5A 1S6, Personal Communication, 2010).

2. My mortality estimates were generated using biological samples from areas where sea otters were beginning to expand their range. A repeated density survey of these particular study sites, with the minimum number of transects listed in Table 2, and new biosamples of possibly > 1,000 geoducks for reliable catch curve analyses, may provide a clearer distinction between sites with and without otters.

3. Design a before-after-control-impact study in the B.C. north coast to better assess the magnitude of impact on geoduck populations from sea otter predation. Power analysis through simulation would help to determine the most cost-effective sample size for detecting significant differences between areas with and without sea otters. A minimum of two geoduck control sites with no sea otters or fishing should be defined, and three "impact" sites comprising (*i*) sea otters only, (*ii*) fishing only, and (*iii*) sea otters and fishing. Quantifying this information would help managers determine which, if any, geoduck beds are altered beyond acceptable levels for commercial harvests.

4. Detailed reports about the weights of the catches from each vessel are collected throughout the year (IFMP 2010). While single species models (e.g., virtual population analysis (VPA) and statistical catch-at-age methods) were useful for estimating reference points and harvest quotas, scientists are being asked to include possible influences of predator interactions on their estimates of stock dynamics (IFMP 2010; Jurado-Molina et al. 2005). High priority should be given to collecting catch-at-age data from the landings from recently opened geoduck beds in the north coast in areas without sea otters. Samples from unfished geoduck populations will provide a better measure of how geoduck total mortality rates change when harvesting removes a known age-distribution and when sea otters eventually arrive. It would also facilitate accurate reconstructions of the historical numbers-at-age of the exploited geoduck populations, which is a necessary step for modeling in a multispecies context (Jurado-Molina et al. 2005; Hollowed et al. 2000).

TABLES

Table 1 Summary of the location and dates of geoduck surveys for 17 study sites in the WCVI. Sites are listed in order of sea otter range expansion. Millar Channel, Yellow Bank, and Barkley Sound are the sites without otters.

Survey Date	Survey Location	Study Site	Stat Area	Site Type	Average Survey Depth (m)	No. of Transects	Sum of Transect Lengths (m)	Total No. of Quadrats	Total Area Surveyed (m²)	Dominant Substrate	No. of Sea Otters
1998	Kyuquot	a	26	Otter	8.95	17	5915	307	3070	sand, crushed shell	770
2003	Kyuquot	a	26		9.76	30	7530	492	4920	mud, sand, crushed shell	1056
1998	Kyuquot	b	26	Otter	10.23	5	765	42	420	sand, crushed shell, crevices in bedrock	770
2003	Kyuquot	b	26		13.39	7	1215	109	1090	sand, crushed shell	1056
1996	Quatsino Inlet	a	27	Otter	8.92	7	890	89	890	sand, crushed shell	1
2002	Quatsino Inlet	a	27		11.52	6	720	66	660	sand, crushed shell	16
1996	Quatsino Inlet	b	27	Otter	7.86	6	725	73	730	sand, crushed shell	1
2002	Quatsino Inlet	b	27		11.51	5	620	60	600	sand, crushed shell	16
1996	Quatsino Inlet	c	27	Otter	8.37	11	1090	108	1080	crushed shell	1
2002	Quatsino Inlet	c	27		11.26	7	815	96	960	crushed shell	16
1996	Forward Inlet	a	27	Otter	8.25	4	340	34	340	sand	1
2002	Forward Inlet	a	27		11.17	3	270	32	320	sand	16
1996	Forward Inlet	b	27	Otter	9.27	10	3055	206	2060	sand, crushed shell	1
2002	Forward Inlet	b	27		11.06	9	2995	195	1950	sand, crushed shell	16
2004	Tofino	a	24	Otter	9.83	20	9610	490	4900	sand	181
2008	Tofino	a	24		10.25	17	6400	375	3750	sand	173
1997	Millar Channel	a	24	No Otter	9.37	36	14095	704	7040	sand	0
2007	Millar Channel	a	24		10.18	34	9050	574	5740	sand	0
1995	Yellow Bank	a	24	No Otter	8.24	9	5560	279	2790	sand, crushed shell	0
2006	Yellow Bank	a	24		12.4	14	3534	242	2420	mud, sand, crushed shell, gravel (< 3/4 in)	0
1997	Yellow Bank	b	24	No Otter	6.06	1	500	25	250	mud, sand	0
2006	Yellow Bank	b	24		7.45	21	6020	377	3770	sand	0
2002	Barkley Sound	a	23	No Otter	10.06	6	925	79	790	mud, sand	0
2002	Barkley Sound	b	23	No Otter	10.25	6	750	71	710	sand, crushed shell, crevices in bedrock	0
2002	Barkley Sound	c	23	No Otter	10.01	10	1260	148	1480	sand, bedrock and crevices	0
2005	Barkley Sound	d	23	No Otter	11.96	6	355	51	510	sand, crushed shell	0
2005	Barkley Sound	e	23	No Otter	13.38	9	600	77	770	mud, sand	0
2005	Barkley Sound	f	23	No Otter	12.91	9	720	80	800	sand, crushed shell	0

Table 2a Bed specific mean geoduck density for the 8 study sites with otters in the WCVI. Results are shown for the statistical comparison of mean densities between survey years and a retrospective power analysis (α = 0.05, β = 0.80, d = 0.8). Average minimum sample size (n) is based on the mean and standard deviation of the second survey samples (α = 0.05, β = 0.80, d = mean*0.5). Effect size (Cohen's *d*) is provided with 95% confidence intervals.

Survey Date	Survey Location	Study Site	Stat Area	Site Type	No. of Transects	Density (# geoducks/m^2) Mean	95% CI	Welch Two Sample *t*-test *t*	df	*P*-value	Power of *t*-test d=0.8 Power (%)	n	Cohen's *d*	95% CI
1998	Kyuquot	a	26	Otter	17	1.11	(0.74, 1.54)	1.9	21.59	0.0670	73	82	-0.17	(-0.75, 0.42)
2003	Kyuquot	a	26		30	0.55	(0.34, 0.77)							
1998	Kyuquot	b	26	Otter	5	0.46	(0.26, 0.70)	1	4.08	0.3856	24	14	-1.21	(-2.57, 0.15)
2003	Kyuquot	b	26		7	0.17	(0.15 0.20)							
1996	Quatsino Inlet	a	27	Otter	7	2.03	(1.60, 2.47)	-0.2	8.68	0.8432	26	46	0.08	(-1.10, 1.27)
2002	Quatsino Inlet	a	27		6	2.22	(1.61, 2.86)							
1996	Quatsino Inlet	b	27	Otter	6	0.75	(0.41, 1.13)	-1.38	8.77	0.2005	22	16	0.86	(-0.52, 2.25)
2002	Quatsino Inlet	b	27		5	1.52	(1.27, 1.77)							
1996	Quatsino Inlet	c	27	Otter	11	0.54	(0.38, 0.72)	-0.16	12.64	0.8731	34	49	0.06	(-0.92, 1.04)
2002	Quatsino Inlet	c	27		7	0.58	(0.41, 0.76)							
1996	Forward Inlet	a	27	Otter	4	0.28	(0.21, 0.34)	-0.88	3.26	0.4373	14	30	0.72	(-1.15, 2.59)
2002	Forward Inlet	a	27		3	0.46	(0.36, 0.54)							
1996	Forward Inlet	b	27	Otter	10	0.51	(0.42, 0.6)	1.7	16.49	0.1165	38	53	-0.49	(-1.47, 0.47)
2002	Forward Inlet	b	27		9	0.31	(0.22, 0.41)							
2004	Tofino	a	24	Otter	20	0.48	(0.34, 0.64)	-1.93	50.90	0.0592	80	43	0.08	(-0.46, 0.61)
2008	Tofino	a	24		17	0.76	(0.54, 0.99)							

Table 2b Bed specific mean geoduck density for the 9 study sites without otters in the WCVI. Results are shown for the statistical comparison of mean densities between survey years and retrospective power analysis ($\alpha = 0.05$, $\beta = 0.80$, d = 0.8), for two sites with repeated survey data. Average minimum sample size (n) is based on the mean and standard deviation of the second survey samples ($\alpha = 0.05$, $\beta = 0.80$, d = mean*0.5). Effect size (Cohen's *d*) is provided with 95% confidence intervals.

Survey Date	Survey Location	Study Site	Stat Area	Site Type	No. of Transects	Density (# geoducks/m^2) Mean	(L95, U95)	Welch Two Sample *t*-test *t*	df	*P*-value	Power of *t*-test d=0.8 Power (%)	n	Cohen's *d*	95% CI
1997	Millar Channel	a	24	No Otter	36	1.61	(1.13, 2.12)	-1.41	67.29	0.1629	91	22	0.06	(-0.41, 0.54)
2007	Millar Channel	a	24		34	2.04	(1.62, 2.48)							
1995	Yellow Bank	a	24	No Otter	9	2.15	(1.93, 2.36)	5.7	11.54	0.0001	43	14	-0.96	(-1.86, -0.05)
2006	Yellow Bank	a	24		14	0.81	(0.68, 0.94)							
1997	Yellow Bank	b	24	No Otter	1	1.59	NA							
2006	Yellow Bank	b	24		21	1.11	(1.00, 1.22)							
2002	Barkley Sound	a	23	No Otter	6	0.96	(0.74, 1.20)							
2002	Barkley Sound	b	23	No Otter	6	1.17	(0.82, 1.52)							
2002	Barkley Sound	c	23	No Otter	10	0.75	(0.52, 1.00)							
2005	Barkley Sound	d	23	No Otter	6	1.38	(0.97, 1.83)							
2005	Barkley Sound	e	23	No Otter	9	0.75	(0.44, 1.08)							
2005	Barkley Sound	f	23	No Otter	9	1.13	(0.81, 1.42)							

Table 3 **Geoduck age-composition statistics from 13 study sites in the WCVI. Results are shown for the statistical comparison of mean age between survey years and retrospective power analysis (α = 0.05, β = 0.80, d = 0.3), for the 7 sites with repeated age data. Average minimum sample size (n) is based on the mean and standard deviation of the second survey samples (α = 0.05, β = 0.80, d = mean*0.3). Effect size (Cohen's *d*) is provided with 95% confidence intervals.**

Survey Date	Survey Location	Study Site	Stat Area	Site Type	No. Geoducks Aged	Geoduck Age (years) Min	Max	Median	Mean	Wilcoxon Rank Sum Test W	*P*-value	Power of Wilcoxon Test (d=0.3) Power (%)	n	Cohen's *d*	95% CI
1998	Kyuquot	a	26	Otter	304	3	120	9	19.19	35709	3.68E-07	96	134	0.00	(-0.16, 0.16)
2003	Kyuquot	a	26		308	2	117	12	18.57						
1996	Quatsino Inlet	a	27	Otter	141	4	120	63	58.06	13976	1.67E-03	74	76	-0.03	(-0.25, 0.20)
2002	Quatsino Inlet	a	27		164	5	135	46	47.97						
1996	Quatsino Inlet	c	27	Otter	142	6	135	47.5	50.82	15855	3.12E-06	75	119	-0.04	(-0.26, 0.18)
2002	Quatsino Inlet	c	27		171	5	109	21	36.27						
1996	Forward Inlet	a	27	Otter	156	4	152	13	32.20	16337	1.76E-06	76	178	-0.10	(-0.31, 0.12)
2002	Forward Inlet	a	27		160	5	89	9	14.54						
2004	Tofino	a	24	Otter	296	2	73	12	13.71	38769	0.816	94	101	-0.01	(-0.18, 0.16)
2008	Tofino	a	24		259	3	62	9	12.31						
1997	Millar Channel	a	24	No Otter	277	2	96	15	24.56	70779	0.032	98	121	0.00	(-0.14, 0.14)
2007	Millar Channel	a	24		562	2	105	15	24.50						
1997	Yellow Bank	b	24	No Otter	186	2	95	15	24.80	40253	0.474	93	97	-0.01	(-0.16, 0.15)
2006	Yellow Bank	b	24		449	3	110	16	23.00						
2002	Barkley Sound	a	23	No Otter	183	3	108	53	47.65				41		
2002	Barkley Sound	b	23	No Otter	167	13	102	56	55.76				14		
2002	Barkley Sound	c	23	No Otter	151	7	120	60	59.83				38		
2005	Barkley Sound	d	23	No Otter	206	5	134	60.5	54.84				54		
2005	Barkley Sound	e	23	No Otter	103	4	126	24	36.19				103		
2005	Barkley Sound	f	23	No Otter	161	4	117	47	44.86				43		

Table 4 Estimated geoduck instantaneous total mortality (*Z*) for 13 study sites in the WCVI using the Chapman-Robson (1960) estimator. Performance of the estimator was tested using simulation (% bias).

Survey Date	Survey Location	Study Site	Stat Area	Site Type	No. Geoducks Aged*	CR Estimator: Z	CR Estimator: SE_Z	CR Estimator: CV_Z (%)	Simulated CR Estimator: Z_{sim}	Simulated CR Estimator: Bootstrapped CI L95	Simulated CR Estimator: Bootstrapped CI U95	% Bias
1998	Kyuquot	a	26	Otter	137	0.0280	0.0024	8.50	0.0273	0.023	0.032	2.50
2003	Kyuquot	a	26		229	0.0445	0.0029	6.60	0.0426	0.038	0.048	4.27
1996	Quatsino Inlet	a	27	Otter	130	0.0158	0.0014	8.80	0.0156	0.013	0.018	1.27
2002	Quatsino Inlet	a	27		129	0.0168	0.0015	8.80	0.0165	0.014	0.020	1.79
1996	Quatsino Inlet	c	27	Otter	135	0.0185	0.0016	8.60	0.0182	0.015	0.021	1.62
2002	Quatsino Inlet	c	27		105	0.0182	0.0018	9.80	0.0178	0.015	0.022	2.20
1996	Forward Inlet	a	27	Otter	84	0.0185	0.0020	10.90	0.0185	0.015	0.022	0.00
2002	Forward Inlet	a	27		52	0.0349	0.0048	13.90	0.0337	0.026	0.043	3.44
2004	Tofino	a	24	Otter	162	0.0477	0.0037	7.90	0.0456	0.039	0.052	4.40
2008	Tofino	a	24		118	0.0478	0.0044	9.20	0.0456	0.038	0.054	4.60
1997	Millar Channel	a	24	No Otter	152	0.0240	0.0019	8.10	0.0234	0.020	0.027	2.50
2007	Millar Channel	a	24		409	0.0312	0.0015	4.90	0.0303	0.028	0.033	2.88
1997	Yellow Bank	b	24	No Otter	387	0.0258	0.0025	9.50	0.0252	0.021	0.030	2.33
2006	Yellow Bank	b	24		111	0.0381	0.0019	5.10	0.0367	0.033	0.041	3.67
2002	Barkley Sound	a	23	No Otter	161	0.0186	0.0015	7.90	0.0183	0.016	0.021	1.61
2002	Barkley Sound	b	23	No Otter	167	0.0177	0.0014	7.70	0.0175	0.015	0.020	1.13
2002	Barkley Sound	c	23	No Otter	149	0.0162	0.0013	8.20	0.0160	0.014	0.019	1.23
2005	Barkley Sound	d	23	No Otter	186	0.0165	0.0012	7.30	0.0163	0.014	0.019	1.21
2005	Barkley Sound	e	23	No Otter	91	0.0244	0.0026	10.50	0.0237	0.019	0.029	2.87
2005	Barkley Sound	f	23	No Otter	155	0.0212	0.0017	8.00	0.0207	0.018	0.024	2.36

*Number of geoducks > 10 years old in the sample.

FIGURES

Figure 1 Distribution of sea otters in British Columbia. Dark areas represent the total occupied range as of 2009 along the WCVI and the Central Coast. Though close to the otter site near Tofino, Millar Channel and Yellow Bank are areas not yet “occupied” by otters. Barkley Sound is outside the sea otter’s current range. Modified from Nichol et al. (2009).

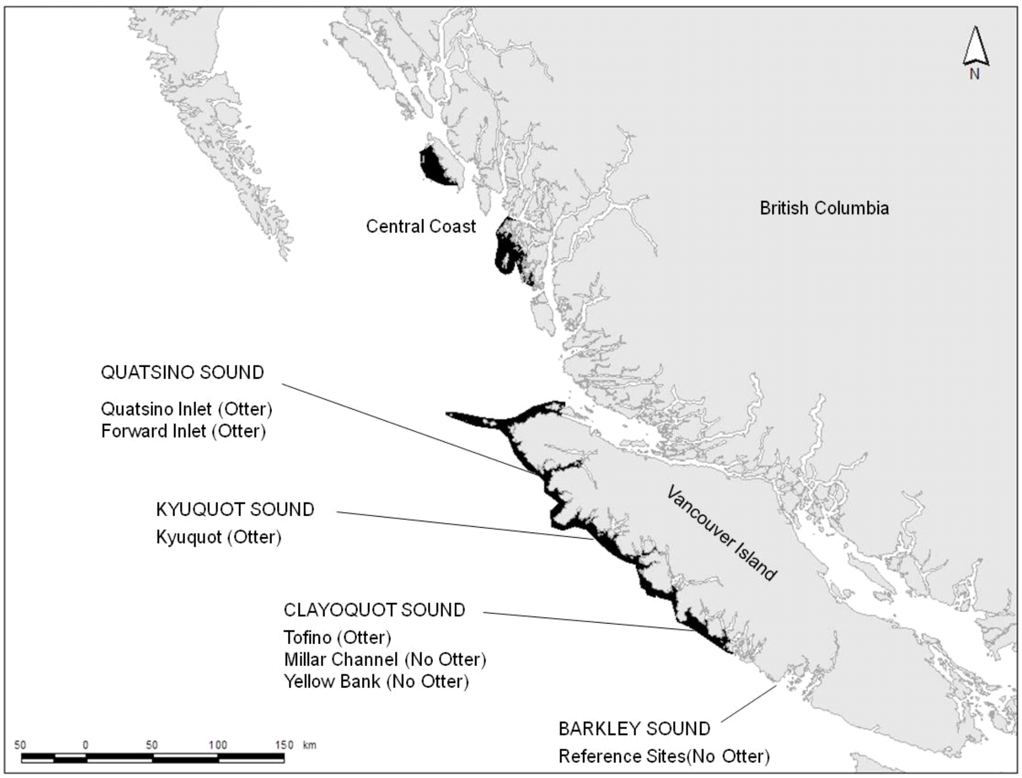


Figure 2 Total geoduck fishery landings and total diver fishing hours in the study sites. Data are from the beginning of the fishery in the WCVI in the 1980s to the second survey year in each study site. The line represents the regression fit, r^2 the coefficient of determination, and the P-value (*P*) represents the statistical significance.

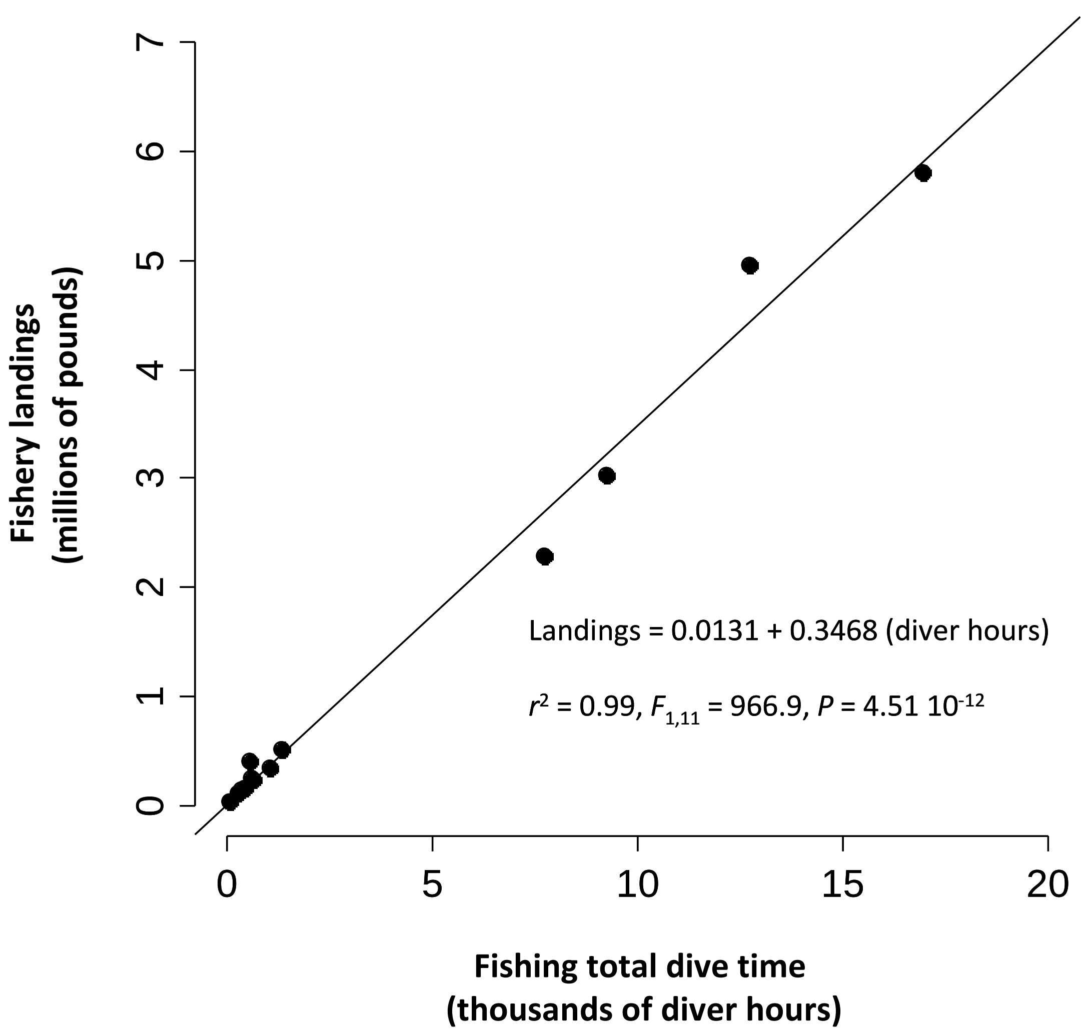


Figure 3a Age distribution of geoduck samples collected from 7 study sites with repeated age samples. The letters *a*, *b*, or *c* identify bed-specific geoduck samples from the WCVI, collected in two different survey years, with at least 3 years between surveys.

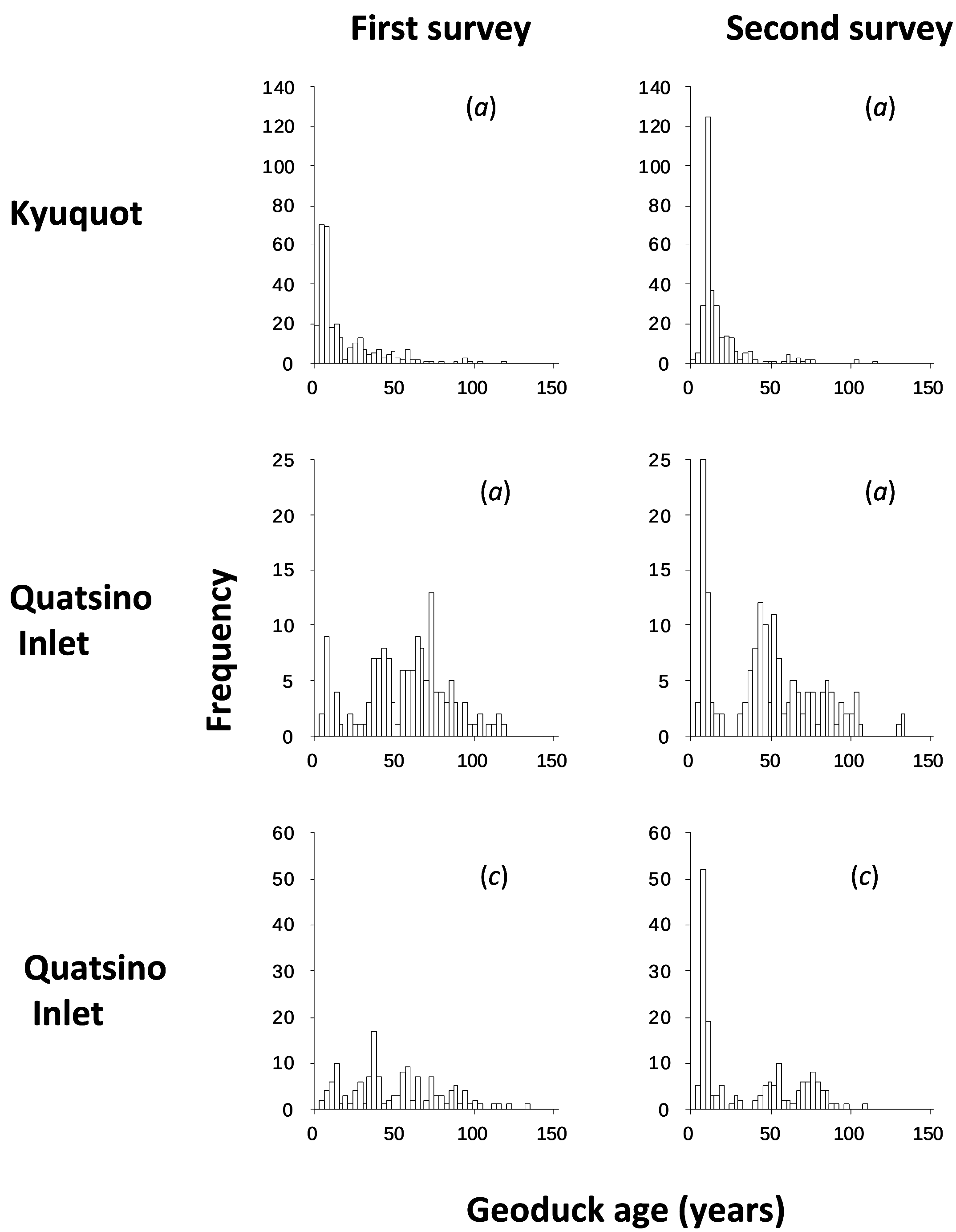


Figure 3a (Continued)

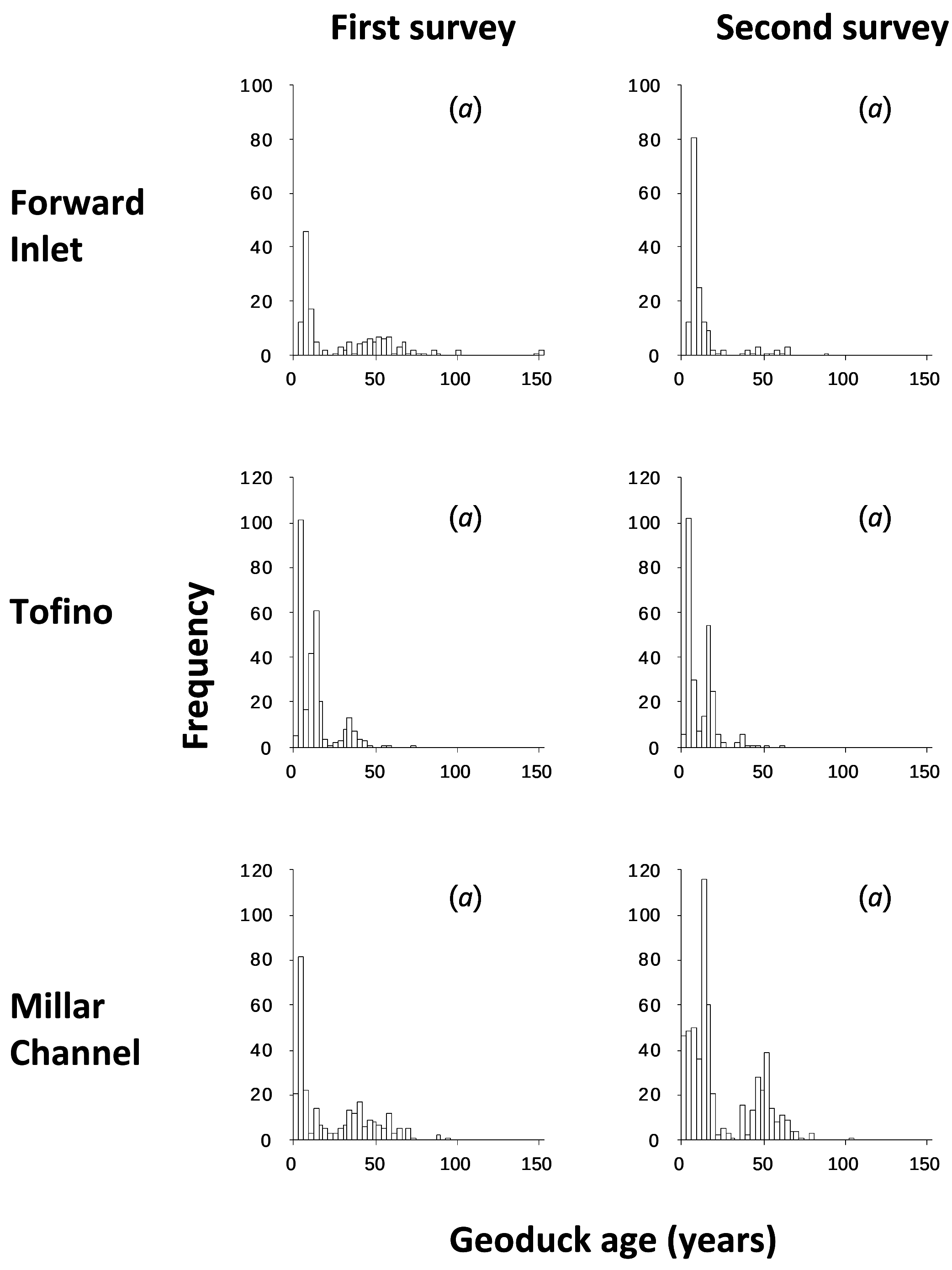


Figure 3a (Continued)

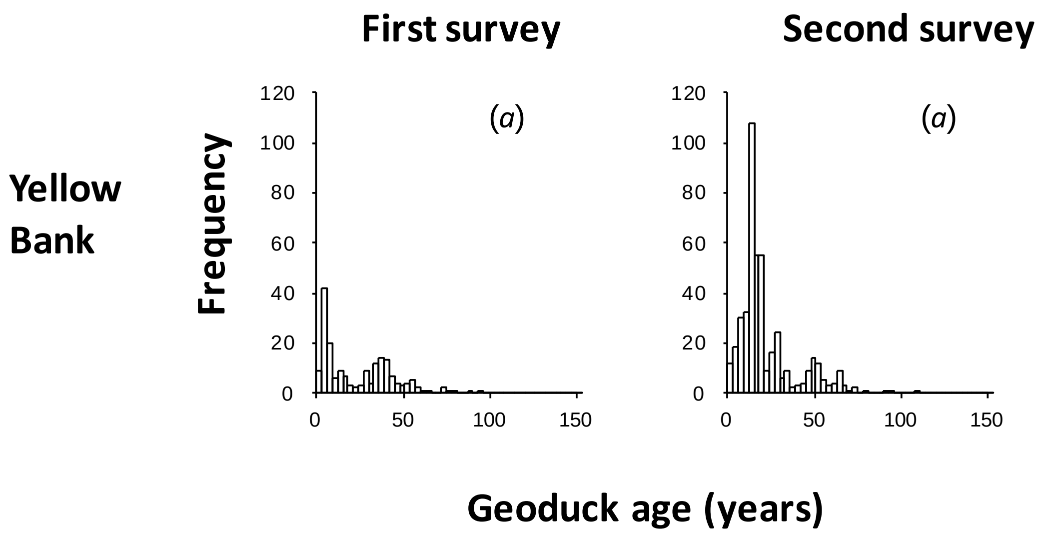


Figure 3b **Age distribution of geoduck samples collected from 6 supplementary sites without otters in Barkley Sound in the WCVI. No matching samples were available for the sites in Barkley Sound. Surveys were conducted in 2002 (First Survey) and in 2005 (Second Survey).**

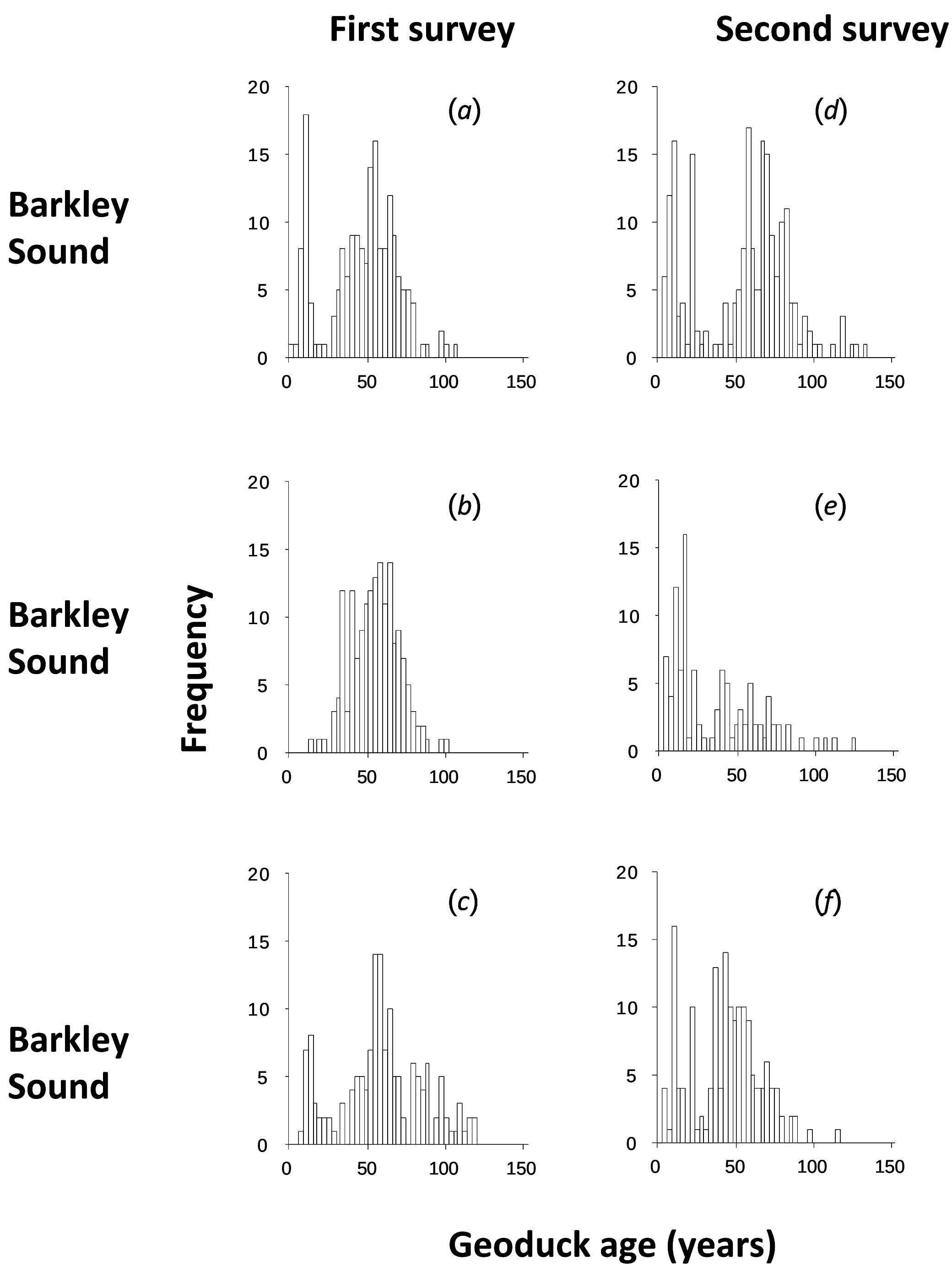


Figure 4 **Age distribution of geoduck samples collected from 7 study sites with repeated age samples and from 6 supplementary sites in Barkley Sound in the WCVI. Alternating colours show matching samples for comparison between survey years, with black bars showing median geoduck age, and letters *a-f* identifying bed-specific geoduck samples collected in two different survey years. Miller Channel, Yellow Bank and Barkley Sound are the sites without otters. No matching samples were available for the sites in Barkley Sound.**

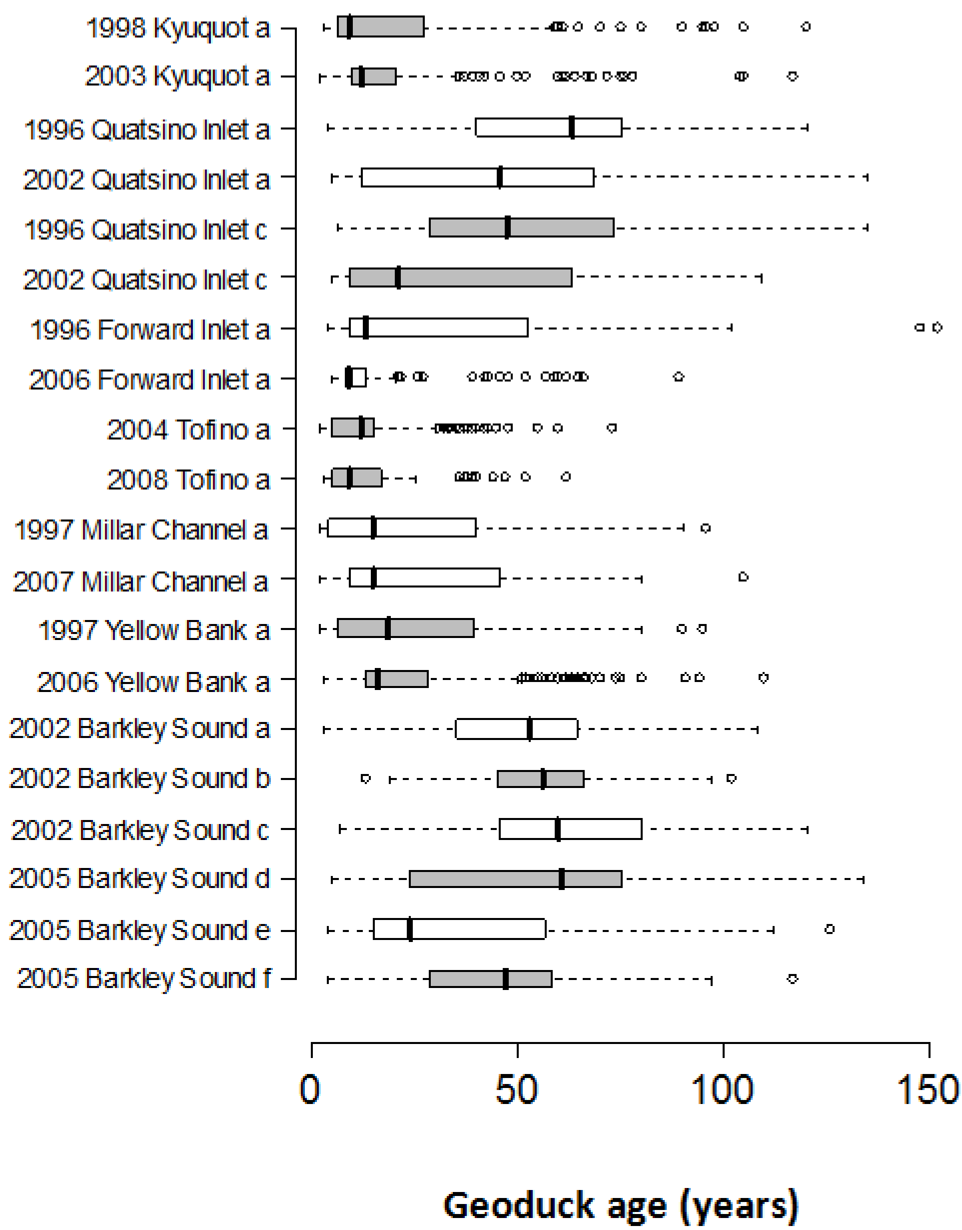


Figure 5 Chapman-Robson estimates of geoduck total mortality (*Z* year^{-1}) versus predicted geoduck total mortality (*Z* year^{-1}) obtained from the multiple regression equation. The line represents the regression fit, r^2 the coefficient of determination, and the P-value (*P*) represents the statistical significance.

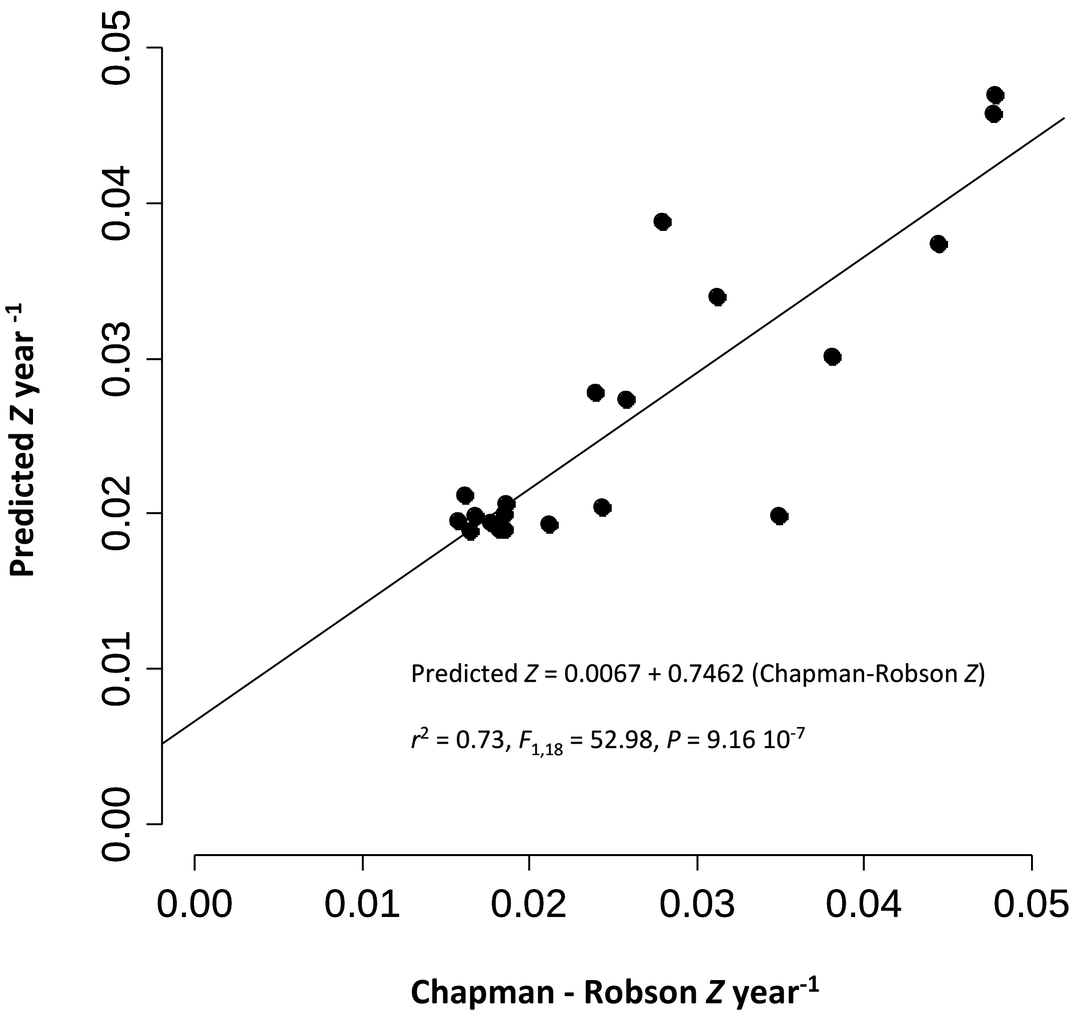
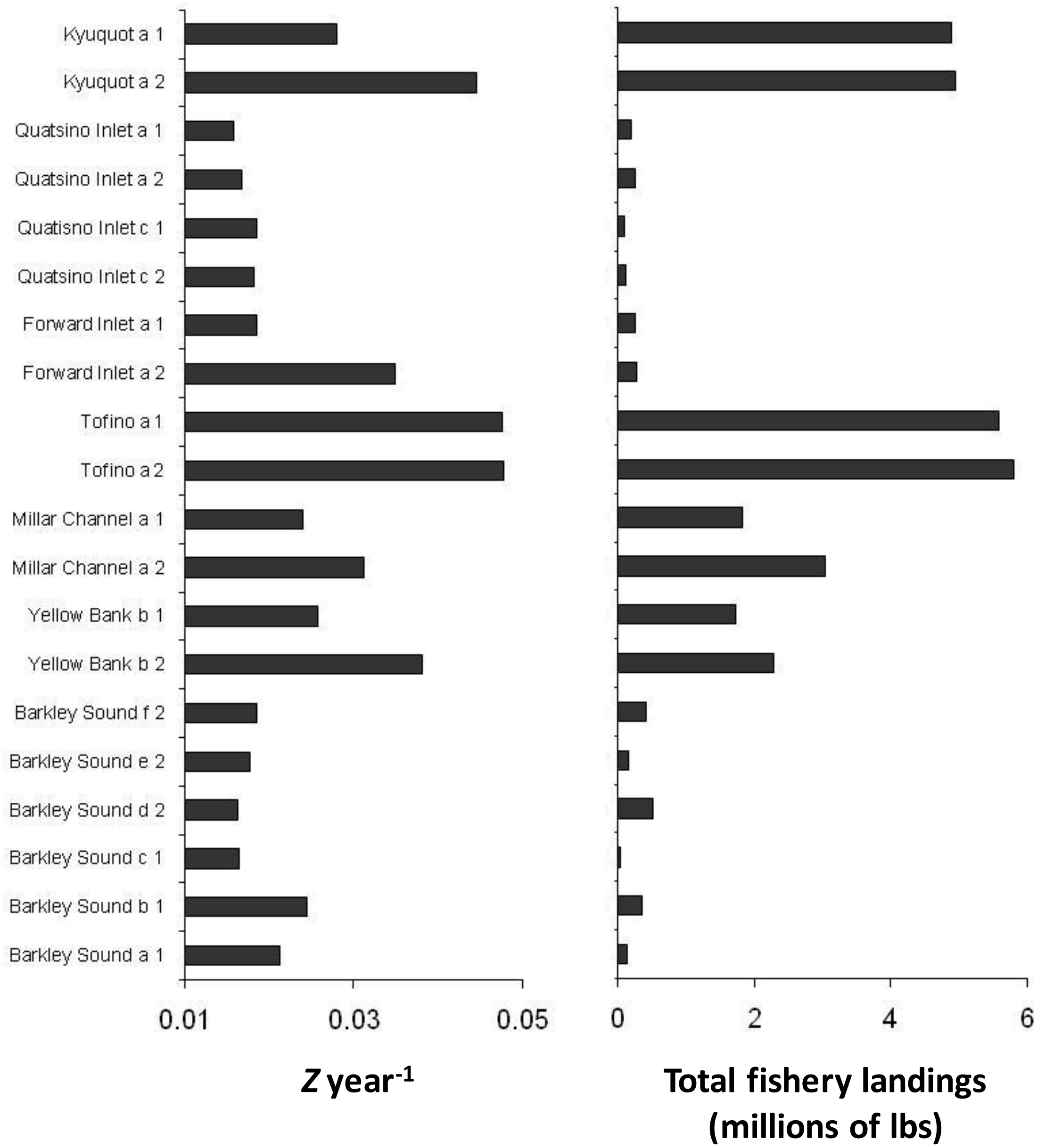


Figure 6 **Fishery-independent estimates of geoduck instantaneous total mortality (*Z* year^{-1}) (left) compared to total fishery landings (millions of pounds) from the study sites. Landings are from the beginning of the fishery in the WCVI in the 1980s to the first (1) and second (2) survey year in a study site. Millar Channel, Yellow Bank and Barkley Sound are the sites with no otters.**



CHAPTER 3
SEA OTTERS AND SHELLFISH FISHERIES IN BRITISH COLUMBIA: TOWARDS RESOLUTION OF CONFLICT AND UNCERTAINTY

Abstract

The reestablishment of sea otters in British Columbia has had repercussions that led to one of the more interesting resource management problems involving commercial fisheries in Canada. In this chapter, I examine: (1) the history of sea otter exploitation; (2) Canada's legislative framework for protecting marine fish and animal resources; and (3) the social context of the otter-fishery problem today. Fisheries and marine mammals in Canada are currently managed under overlapping legislative mandates and often with different goals. This analysis suggests that it is unlikely that the ban on killing and taking of sea otters will be lifted, and any attempt to artificially limit the sea otter's range in B.C. (i.e., conservation as active management) would be difficult under both the *Species at Risk Act* and the *Fisheries Act,* and also for socio-political reasons. Resolving concerns over sea otter predation on valuable geoduck resources may require government support of innovative research in geoduck aquaculture, outside the sea otter's historical range. Success in recovering sea otters in B.C. may depend on more than the ability to prevent potential losses from the otter populations, but also on the ability to engage formerly antagonistic stakeholders in the recovery of the species.

Introduction

Following the extensive hunt in the 18th and 19th centuries during the maritime fur trade, sea otters were absent along the B.C. coast for nearly 100 years. Between 1969 and 1972, the Alaska Department of Fish and Game, in cooperation with the British Columbia Fish and Wildlife Branch and Fisheries and Oceans Canada (Arctic Biological Station and Pacific Biological Station) reintroduced sea otters from Alaska to B.C. (Bigg and MacAskie 1978; Jameson et al. 1982). The objective of the transplants was to ensure the continued existence of the species following the planned nuclear detonations in the Bering Sea, close to where a population of sea otters was known to exist (Paul 2009). The reestablishment of sea otters had repercussions that led to one of the more interesting resource management problems involving commercial fisheries in Canada.

Conflict between sea otters and fisheries in B.C. has evolved through several stages based on perceptions, judgments and disagreement about the benefits and costs of recovering an extirpated marine carnivore. Decision-makers must now consider various ecological, social, political, and economic issues. Sea otters play a keystone role in coastal ecosystem processes, by limiting the number of herbivorous invertebrates like sea urchins and promoting the growth of productive kelp forests (Breen et al. 1982; Gerber et al. 1999; COSEWIC 2007). Sea otter recovery is associated with direct and indirect benefits for other species of invertebrates as well as fish from the subsequent increases in food and habitat resources (Bodkin 2003; COSEWIC 2007). In addition, public viewing of sea otters is becoming an important resource for many people through nature tourism and photography (Newsome et al. 2005), and the sea otter is now an iconic symbol of British Columbia's wilderness. On the other hand, evidence suggests there are potential economic costs associated with sea otter recovery. Some stakeholders consider predation by sea otters on shellfish resources to be unacceptably high where sea otters and shellfish fisheries co-occur. Understanding the evolutionary stages of the conflict is a critical first step towards developing an effective, equitable and acceptable solution.

This chapter examines: (1) the history of sea otter exploitation; (2) Canada's legislative framework for protecting marine fish and animal resources; and (3) the social context of the otter-fishery problem today. The final section is a discussion of two very different philosophies that are currently steering marine fisheries and wildlife management in Canada and the problems of reconciling these philosophies for effective otter-fishery management. As demonstrated in the previous chapter, fishery managers must also deal with the difficult problem of uncertainty. The challenge for geoduck managers is to support the recovery of sea otters without unreasonably limiting opportunities for commercial fishing. This chapter contains recommendations for addressing conflict and uncertainty in B.C. shellfish fisheries with special reference to geoduck fisheries and the recovery of sea otters.

Part One: The sea otter fishery

The sea otter fur trade grew rapidly in the 18th century on the west coast of North America under an open-access model with no prescribed limits on catch or effort (Lensink 1960). Hilborn and Walters (1992) describe the historical fishery for sea otters in the North Pacific as an aggregation-based fishery that collapsed because of hyperstability. That is, sea otters aggregate in large numbers at sites and times that are predictable, and at the height of the sea otter fishery, the catch per unit of effort remained elevated while the overall abundance of sea otters declined (Hilborn and Walters 1992).

The Pacific maritime fur trade began when the Bering Expedition discovered Alaska and the Aleutian Islands in the 1700s (Phillips 1961). That expedition encountered First Nations who used sea otters for food and clothing (Phillips 1961). Subsequently, following the arrival of Captain James Cook to the west coast of Vancouver Island in 1778, sea otter pelts became one of the main items of trade between the aboriginal peoples and the European merchants (Pethick 1980). By the mid-1800s, sea otters, once numbering between 150,000 and 300,000 animals in the North Pacific (COSEWIC 2007), became over-exploited to meet the foreign demand for otter pelts (Phillips 1961). The signing of the *North Pacific Fur Seal Treaty* in 1911, between the United States, Japan, Russia and Great Britain (on behalf of Canada) officially closed the fishery for sea otters (Lensink 1960; Phillips 1961; Gibson 1992), but not before the sea otter was extirpated in Canada (COSEWIC 2007).

The great hunt for fur-bearing animals was not limited to the sea otter. Commercial sealing has gone on for centuries, and in the two hundred years prior to the fur seal treaty, the hunt for fur-bearing seals devastated the North Pacific fur seal population (Kuokkanen 2002). As early as the 1700s, in waters outside any national jurisdiction, Russians, and later Americans, had harvested several million seals for their thick, waterproof underfur. Although the impetus for the fur seal treaty was to conserve the fur seal populations and to “maximize productivity” (Kuokkanen 2002; Alaska Department of Fish and Game 2008), the treaty ended the indiscriminant hunting of sea otters as well as seals. Pursuant to Articles I (Prohibition of pelagic sealing) and V (Protection of sea otters), the signing nations renounced sealing in open waters, and disallowed the killing, capturing, or pursuit of sea otters beyond the three-mile limit of their territorial seas. The *Fur Seal Treaty* of 1911, therefore, is significant in representing a shift to international regulation of a common resource (fur-bearing marine mammals) (NOAA 2007).

Recovery goal for sea otters in B.C.

In June 2003, the sea otter was listed under the Canadian *Species at Risk Act* (*SARA*) (S.C. 2002, ch. 29) as a threatened species (COSEWIC 2007). A Sea Otter Recovery Team appointed by the Canadian Minister of Fisheries and Oceans developed a Recovery Strategy for the Sea Otter in Canada (COSEWIC 2007). The recovery strategy recommends a "non-intrusive approach to recovery," or no interference with sea otters because the sea otter population in B.C. is rebounding on its own (DFO 2007b). Fisheries and Oceans Canada (DFO) is directed to allow the number of sea otters to continue to increase, based on the presumption that an increase in sea otter abundance and a corresponding increase in their distribution will reduce the threat from catastrophic events (DFO 2007b). The goal is to ensure that the sea otter population is large enough and dispersed widely enough to withstand known human-caused threats, such as oil spills, that could otherwise result in the extirpation of sea otters or reduce their population so that "recovery to pre-event numbers would be very slow" (DFO 2007b). In March 2009 the status of the sea otter under the *SARA* was changed from threatened to "special concern" (Species at Risk Public Registry 2011) through an *Order Amending Schedule 1 of the Species at Risk Act* (Canada Gazette, Part II 2009). However, the existing recovery strategy continues to apply until the Sea Otter Recovery Team issues a replacement management plan, which must be developed within three years after the change of the otter's status (L. Convey, Chairperson, Sea Otter Recovery Team, Fisheries and Oceans Canada, 3225 Stephenson Point Rd., B.C. V9T 1K3, Personal Communication, 2011).

Recovery objective

The recovery objective set by DFO in the recovery strategy for the sea otter is: "Identify and, where possible, mitigate threats to sea otters and their habitat to provide for recovery of the population" (DFO 2007b).

Recommended strategies to achieve recovery goal

The recovery strategy proposes the following four broad activities to identify and mitigate threats to sea otter recovery (DFO 2007b).

1. Research to clarify threats;
2. Population assessment surveys;
3. Protection from oil spills and other threats (e.g., disease, contaminants, entanglement in fishing gear, and illegal killing); and,
4. Communication to support recovery.

Part Two: Current laws protecting sea otters and regulating fishing

The allocation of jurisdiction over wildlife under the Canadian Constitution is complex, but generally the federal, provincial and territorial governments share authority for managing wildlife in Canada (Kerr 2011). The federal government is generally responsible for migratory birds, fish, marine mammals and wildlife on federal lands (MNRO 2010a). Provincial responsibility includes "most wildlife matters including the conservation of wildlife populations and habitat" that occur within provincial boundaries (MNRO 2010a).

The *British Columbia Wildlife Act*

The British Columbia *Wildlife Act* (R.S.B.C. 1996, Ch. 488), administered by the Ministry of Forestry, Lands and Natural Resource Operations (Fish and Wildlife Branch), is the main provincial law for protecting wildlife in B.C. (MNRO 2010b). Sea otters are designated as threatened under the *Wildlife Act* (MNRO 2010a). The *Act* provides clear prohibitions against killing or other direct harm to wildlife species in B.C., except when authorized under license or provided for by regulation. Currently, no license has been issued for the killing and taking of sea otters under the *Wildlife Act* (L. Convey, Chairperson, Sea Otter Recovery Team, Fisheries and Oceans Canada, Personal Communication, 2010).

The *Fisheries Act*

In addition to receiving protection under provincial legislation, sea otters are directly protected under the federal *Fisheries Act* (R.S. 1985, c. F-14) by the *Marine Mammal Regulations* (SOR/93-56) that make it an offence to kill, harm, or harass marine mammals. The only exceptions are when fishing for marine mammals under the authority of a license issued by the Minister under subsection 4(1), or under the *Aboriginal Communal Fishing Licenses Regulations* (SOR/93-332). No licenses have been issued yet under either regulation to fish for sea otters (L. Convey, Chairperson, Sea Otter Recovery Team, Fisheries and Oceans Canada, Personal Communication, 2010). The *Fisheries Act* is powerful in enabling the federal government to protect species and habitats "supporting those species that sustain fisheries, namely fish, shellfish, crustaceans, and marine mammals" (DFO 2007d). Under the *Fisheries Act*, Parliament designates Fisheries and Oceans Canada with the responsibility to conserve and protect fisheries and fish habitats in all Canadian waters. Both marine mammals and shellfish (and thus geoducks) are included in the *Act's* definition of "fish," and section 34(1) affirms, "fish habitat means spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes." Sea otters (and geoducks) are further protected under Section 35, the main provision dealing with habitat protection, as (1) "No person shall carry on any work or undertaking that results in the harmful alteration, disruption, or destruction of fish habitat."

The *Species at Risk Act*

Perhaps the most important legislation protecting the recovery of sea otters was created in 2002, when Parliament passed the federal *Species at Risk Act*, which became law in June 2003. In accordance with Canada's commitment to the *United Nations Convention on Biological Diversity* (1992), the *SARA* is designed to protect wildlife species from becoming extinct or extirpated. It expressly provides for the recovery of populations assessed as being endangered, threatened, or of special concern ("species at risk") (Species at Risk Public Registry 2010). The *SARA* protects critical habitats, and makes it illegal to kill, harm, harass, or capture an endangered or threatened species. Under the *SARA*, sea otters fall under the definition of "wildlife" and are currently designated as a species of "special concern" due to their small population size, limited distribution, and vulnerability to potentially harmful events like oil spills or disease that may trigger irreversible population declines (COSEWIC 2007). Their listing as a species of special concern under the *SARA* requires that a management plan be completed by

March 2012, with the intention of preventing them from becoming threatened or endangered.

The history of COSEWIC, the *SARA*, and the listing of sea otters

Environment Canada manages the implementation of the *SARA* for all listed terrestrial species. Fisheries and Oceans Canada is responsible for its implementation as applied to aquatic species, and Parks Canada is the agency responsible for aquatic species within park boundaries (Parks Canada 2009). The three departments share responsibility in collectively upholding the federal government's commitment to protect susceptible wildlife species, and to establish programs that encourage their recovery (Environment Canada 2009). The Committee on the Status of Endangered Wildlife in Canada (COSEWIC), created in 1977, is an independent committee of experts that reviews and determines the status of wildlife species in Canada that are suspected of being at risk. COSEWIC is the outcome of an agreement that was achieved at the 1976 Conference of Federal-Provincial-Territorial Wildlife Directors, to nationally standardize the process of classifying wildlife species at risk (Government of Canada 2009). Sea otters are the specific responsibility of the Marine Mammals Specialist Sub-committee of COSEWIC, as appointed by the federal Minister of the Environment (COSEWIC 2008). The subcommittee is comprised of marine mammal experts from Canada who volunteer to assist with the preparation of status reports, assessments, and status designations (COSEWIC 2008). By 1978, COSEWIC had produced its first assessments of Canadian wildlife species at risk, and immediately classified the sea otter as "endangered" in B.C. (COSEWIC 2007). COSEWIC's assessments, however, do not determine the legal status of a species under *SARA*. That status is determined by a political decision of the federal Governor in Council.

The *SARA* designates COSEWIC as an advisory body to the Minister of the Environment to evaluate and classify wildlife species, independent of the potential regulatory or socioeconomic effects that a listing might have on stakeholders (Irvine et al. 2005). COSEWIC makes its list public and forwards a rationale for its designations to the Minister of Environment and the Canadian Endangered Species Conservation Council, established in 1998 under the Accord for the Protection of Species at Risk in Canada (CESCC 2001). The CESCC evaluates COSEWIC's designations and then recommends appropriate government action.

The ultimate decision to list or change a species designation under the *SARA* rests with the federal Governor in Council, who may, after receiving a recommendation from the Minister of the Environment, add a species to (or remove a species from) the List of Wildlife Species at Risk in Schedule 1 of the *SARA*. The Minister of Fisheries and Oceans Canada advises the Minister of the Environment on issues pertaining to aquatic species. The Minister of the Environment is at liberty to consider the socio-economic costs of listing a wildlife species, and is required to publish a response to the Species at Risk Public Registry within 90 days of receiving the yearly assessments from COSEWIC. This response must indicate how the Minister initially intends to deal with COSEWIC's assessment. One option is to refer the matter for stakeholder consultation before making a recommendation to the Governor in Council.

Acting on the advice of the federal cabinet, the Governor in Council added the sea otter to Schedule 1 as "threatened," when *SARA* was created in 2002. Upon adding a species to the list, the government of Canada is obligated under the *SARA* to begin procedures for protecting and recovering that species. The designation of sea otters as threatened required the preparation of a recovery strategy detailing the "short-term objectives and long-term goals for protecting" sea otters (DFO 2010c). The recent change in legal status of sea otters from threatened to special concern in March 2009 changes the requirement, and instead requires the drafting of a management plan due three years after designation (in March 2012). The management plan will differ from the recovery strategy by setting broader goals and objectives to maintain "sustainable population levels" of sea otters (DFO 2010b).

Constitutional law at the helm of fisheries

Fishing has been a regulated activity in Canada since Confederation (Parsons 1993). For the Government of Canada, DFO is responsible for a stable fishery resource-base that "provides for an economically viable and diverse industry," while identifying the important stakeholder groups and capturing their views in Canadian fisheries policy (DFO 2010c). The constitutional authority for "seacoast and inland fisheries" was given to the Government of Canada under Subsection 91(12) of the *British North American Act, 1867*, later to be renamed the *Constitution Act, 1867*. In addition, the powerful *Fisheries Act* gives the Minister of Fisheries and Oceans the definitive authority to oversee conservation and enhancement of fish stocks in Canada (Parsons 1993).

Early fisheries legislation in Canada was primarily about salmon conservation in rivers and estuaries (Parsons 1993). The legislation was also intended to protect the marine fisheries resource from over harvesting and depletion while recognizing the "legitimate rights" of fishermen to the resource (Parsons 1993). In the last century, however, complex relationships among DFO, industry, researchers, academic institutions, the public, and First Nations have evolved, along with complex challenges in the fisheries. The existence of aboriginal rights to fisheries, for example, was confirmed and clarified in 1990, in the Sparrow case (*Sparrow v. The Queen* [1990] 1 S.C.R. 1075). In that case, the Supreme Court of Canada held that First Nations have "existing aboriginal and treaty rights" to fish in traditional territory waters under Section 35(1) of the *Constitution Act, 1982*. In contrast, although contemporary management tools like harvest "quotas" and "limited entry licensing" suggest private property rights of fishermen (Lane 1999), the guiding principle of the *Fisheries Act* is that Canada's fisheries resources remain common property, owned by all people of Canada (e.g., *Comeau's Sea Foods Ltd. v. Canada (Minister of Fisheries and Oceans)* [1997] 1 S.C.R. 12).

Part Three: Nature and context of the problem today

Sea otters on the west coast of Vancouver Island (WCVI) are a concern to commercial shellfish harvesters who have fished for at least 30 years in the region where otters are recovering (e.g., Green Sea Urchin IFMP 2010; IFMP 2010). The general structure of the problem can be broken down into three overarching dilemmas. First, the agencies involved in the original transplant of otters from Alaska to B.C. did not adequately consult the people inhabiting the WCVI (Dovetail Consulting 2004). For this and other reasons, some resource user groups that are currently affected by recovering sea otters are not altogether supportive of the recovery of otters. In particular, some First Nations and shellfish harvesters (e.g., urchin fishers) affected by the depletion of local shellfish levels, which they attribute to sea otters, do not view the legislation protecting sea otter recovery as legitimate (Dovetail Consulting 2004).

Second, while the commercial geoduck fishery was only beginning on the mainland coast when otters were reintroduced to the WCVI, the market expansion for geoduck resulted in a rapid spread of fishery exploitation to the WCVI by 1979 (Hand and Bureau 2000; DFO 2000). The distribution of fishing efforts that emerged along some sections of Vancouver Island was a direct response to the availability of shellfish and to the strengthening market for geoduck (Heizer 2000). With the complete ban on harvesting sea otters, the reintroduced population of otters has grown, and in some areas these animals may now be competing with harvesters for the commercially available geoduck (IFMP 2010). Under the *SARA*, however, DFO has been applying its legislative directive to protect sea otter recovery using a "non-intrusive approach" to otters (DFO 2007b), despite the possible adverse effects on the livelihoods of geoduck fishers (G. Dovey, West Coast Geoduck Corp., PO Box 781, Ladysmith B.C., V9G 1A6, Personal Communication, 2007).

Finally, developments in public attitudes about resource management may contribute to a conflict between fisheries management and the conservation of marine mammals (Beverton 1985). Increasing losses of wildlife species and their habitat have resulted in heightened public awareness and a call for integrated approaches to managing Canada's natural resources, especially wildlife (Gauthier 1995). Over time, through numerous non-governmental organizations like the Canadian Wildlife Federation, Greenpeace (Canada), and the World Wildlife Fund (Canada), many Canadians have petitioned for the protection of marine mammals, as well as for increased protective resource management activities such as designating protected areas, implementing international wildlife agreements, and making political pledges to protect biodiversity in Canada (Gauthier 1995; Quigley and Harper 2006).

Apart from aboriginal rights to harvest marine mammals (e.g., a communal right to harvest whales), there are no private rights to living marine mammals in Canada, as federal power extends to the management of these animals as a public resource (e.g., *Ward v. Canada (Attorney General* [2002] 1 S.C.R. 569). At present, many members of the public are asking DFO for increased protection of marine mammals through amendments to the existing *Marine Mammal Regulations* of the *Fisheries Act* (DFO 2003). DFO thus faces choices that are weighed down by political and value-based implications because the animals are common property and the legislation is discretionary. In addition, the scientific knowledge currently available to managers may not be able to resolve some of the marine mammal-fishery dilemmas (Beverton 1985). While the research discussed in the previous chapter did not detect an effect of sea otter predation on geoduck total mortality, greater sampling intensity is needed to increase the power to detect whether sea otters actually affect the number of geoduck available for harvest. At least for sea otters and commercial shellfish fisheries, the argument for and against management intervention is a live issue, not only because of the political and value-based considerations, but also because it is difficult to assess quantitatively the true effect of sea otters on shellfish resources in B.C. (COSEWIC 2007).

Discordant philosophies in fisheries and wildlife management

Vanderzwaag and Hutchings (2005) note that the “management of risk” to Canada’s living marine resources has revealed profound differences in values and interests, and divergent ideas about how marine resources should be managed. Although wildlife management in Canada dates back to the late 1800s (e.g., legislation protecting endangered wood bison) (Gauthier 1995), development of protective legislation during the environmental movement of the 20th century (Aronson 1977), and most recently through species at risk legislation, differs from the dominant historical approach to wildlife management. The mandates of modern marine fisheries and wildlife management are different from historical fisheries and wildlife management. The current mandates are sometimes incompatible with those of the past due to a philosophical and deep-seated "conflict of interests" (Beverton 1985).

A fundamental difference exists between the philosophies of "conservation" and "preservation" of natural resources. This difference has given rise to extensive public debate about ethics and economics in resource management (e.g. Rolston 1988; Taylor 1986; Nash 1990; Curtis 2002; Hampicke 1994). A frequently cited example of this philosophical divergence is the dispute about which resource and environmental management philosophy should apply in western forests (see Minteer and Corley 2007). As is routinely observed in fisheries, conservationists have strong interests in protecting natural resources but also see wildlife as a resource to be managed for efficient use by people (Manning 1989). This utilitarian or "pro-use philosophy" (Swanson and Barbier 1992) is to manage wildlife in the best interest for all, which might necessitate killing or relocating members of a population for a number of reasons, such as to protect fisheries by controlling "nuisance" animals that are conflicting with other users (Manning 1989). In contrast, protectionists assert a "strong environmental claim" (D'Amato and Chopra 1991) and support an "anti-use" approach to resource management (Swanson and Barbier

1992; Minteer and Corley 2007; Newsome et al. 2005). Emphasizing the right of existence, protectionists will defend against anthropogenic interference with an animal for ethical reasons (Regan and Singer 1989; Singer 1990). This argument has been advanced by some protectionists for rarely observed marine mammals (Manning 1989; D'Amato and Chopra 1991). Indeed, some protectionists go so far as to support management intervention (i.e., conservation) to support individuals of a wildlife population that are suffering as a result of natural events (e.g., from starvation) (Manning 1989). Norton (1986, 1991) proposed that a combination of the two approaches (conservation and preservation) is necessary for properly managing the competing uses of natural resources, while protecting an ecosystem within a larger framework of environmental management (Minteer and Corley 2007).

The protectionist philosophy gained momentum in North America in the last century when many people were moving from being impartial or indifferent toward marine mammals to showing fascination, while also "being presented with the hard facts of their decline" (Manning 1989). A civic interest in protecting Canada's marine wildlife species, including fish and fish habitats, became entrenched in programs designed to manage the ocean's resources (Parsons 1993). A major factor that pushed Canada to establish the *SARA*, for instance, was Canada's overdue obligation as a signatory to the 1992 Convention on Biological Diversity (Boyd 2003), and the 1996 Canadian Accord for the Protection of Species at Risk (Species at Risk Public Registry 2008). In 2003, Canada's environment minister David Anderson stated, "some of Canada's species have become extinct or extirpated, and a number of our species have become at risk," despite a long history of federal and provincial legislation protecting and managing wildlife (OAG

2003). In consequence of the far-reaching scientific, social, and political shift to address environmental concerns, a more “eco-centric” philosophy (Vanderzwaag and Hutchings 2005), especially in the protection of marine mammals, began challenging the long history of anthropocentric approaches for fisheries management (Beverton 1985; Manning 1989; Parsons 1993).

Legislation and policy reflect competing philosophies

Presently, these two very different philosophies are steering fisheries and wildlife management in Canada. On one hand, strong laws that are conservationist in nature (i.e., consideration is given to utilitarian social and economic factors) are regulating the fisheries. On the other hand, the new *SARA* and the *Marine Mammal Regulations* of the *Fisheries Act* are strongly protectionist where they apply. To a certain degree and on a socioeconomic level, the *SARA* occasionally conflicts with a frequently promoted objective of fishery management, which is to promote the efficient use of fishery resources (Hilborn and Walters 1992). For example, under the *SARA*, the Minister of Fisheries and Oceans is responsible for protecting aquatic species at risk and their habitats. In addition, under the same Minister, shellfish fishers are licensed and in essence are encouraged to harvest a shellfish resource that is managed for optimum sustainable yield (Parsons 1993; IFMP 2010). Recently, however, a species at risk (sea otters) has become established in areas that overlap the geoduck fishery. Under federal control, the reintroduced species at risk is in effect being given precedence over the fishery and allowed to take an unregulated portion of geoduck first, while industry quotas are calculated after taking into consideration the effects of predation on geoduck by sea otters. Although the *SARA* is promoted as placing science at the head of protected species legislation (Vanderzwaag and Hutchings 2005), many fishers see the current policy concerning otters as a victory for the preservationists, or in this case, an "otter-centric" management approach that is unnecessarily inflexible (Dovetail Consulting 2004).

The protection of sea otters on the WCVI demonstrates the magnitude of change in Canadian laws for protecting a vulnerable marine mammal species that was hunted indiscriminately in the past. Until the passing of the *Fur Seal Treaty* in 1911, historical fur trade records indicate that sea otter populations in Alaska were harvested annually since their discovery in the late 1700s, with thousands harvested from B.C. (Lensink 1960). In modern times, under federal rule, a complete ban is in effect in B.C. on the harvesting of sea otters. Even though the reintroduced population is growing, with some parts of its population assessed as being at equilibrium along Vancouver Island (DFO 2007b), the numbers are below the historical estimate of at least 6,000 animals for the WCVI (COSEWIC 2007).

Practical problems in managing sea otters

The conflict between sea otters and shellfish fisheries on the WCVI is partly due to the lack of negotiation about the tradeoffs involved in protecting a vulnerable species. Many shellfish harvesters feel that the sea otters have been awarded undue priority over economic considerations (Dovetail Consulting 2004). While Section 73 of the *SARA* leaves open the possibility of allowing "taking" of listed species by providing incidental harm permits under ministerial discretion (Vanderzwaag and Hutchings 2005), the Minister has never issued a license to kill sea otters. Some protectionist groups are exerting strong pressure at the political level to maintain the prohibition on harvesting sea otters (e.g., Lifeforce 2009), even though sea otters are no longer listed as threatened under the *SARA* (and are no longer subject to the Act's prohibitions against take). The protectionist groups argue that their position supports the "multiplicity of public values" for species at risk (Ecojustice 2008), and as some U.S. groups have claimed regarding the application of the *Marine Mammal Protection Act* (1972), they argue that an allowance for incidental harm is "a kind of conservationist exception to the law that contradicts the moratorium on take" (Manning 1989). If the Minister were to start issuing licenses under the *Fisheries Act* or the *SARA*, the hunting of sea otters in British Columbia might become the centre of a local and international debate on animal welfare comparable to the debate over the hunting of harp seals in Atlantic Canada (e.g., IFAW 2011; Daoust et al. 2002).

It also appears that any attempt to artificially limit the sea otter's range in B.C. (i.e., conservation as active management) would be difficult under both the *SARA* and the *Fisheries Act*, and also for socio-political reasons. Previously, Johnson (1982) recommended that Alaska manage its sea otter populations in accordance with the dominant resource use of each region. His proposed "active management program" was to allow the otter populations in some regions to increase for recreational viewing, but to restrict the movement of sea otters (through capture and relocation) where they competed with fisheries, or to authorize hunting for pelts where the animals were an "economic resource." Garshelis and Garshelis (1984) caution that sexual segregation in sea otters makes the populations especially sensitive to harvesting, where a harvest could impact an entire reproductive population. They argue that the most realistic management option is to prevent overall populations from increasing by focusing harvesting in areas where the numbers of sea otters are low (Garshelis and Garshelis 1984).

Competition for shellfish between people and sea otters is an increasing management problem in Alaska (J. Trent, U.S. Fish and Wildlife Service, Marine Mammals Management Office, 1011 East Tutor Road, Anchorage AK 99503, Personal Communication, 2011). However, because of prohibitions under the U.S. *Marine Mammal Protection Act* there have been no formal captures, harvests, or translocations of Alaskan sea otters in response to a resource competition issue (D. Burn, U.S. Fish and Wildlife Service, Marine Mammals Management Office, 1011 East Tutor Road, Anchorage AK 99503, Personal Communication, 2011). In California, the U.S. Fish and Wildlife Service (USFWS) did attempt a translocation program to protect sea otter recovery, while at the same time establishing a "no-otter" management zone in Santa Barbara County (Public Law 99-625) to restrict the movement of sea otters into valuable fishing areas (Loomis 2005). The translocations of sea otters from the no-otter management zone were largely unsuccessful due to the seasonal wanderings of otters in and out of the zone, and the potentially harmful effect that relocating large numbers of animals could have on otter populations (Watson 2000; Loomis 2005).

In British Columbia, even if the federal government were willing to apply its discretionary power and give greater weight to the B.C. geoduck industry's concern over economic losses, scientists do not actually know if culling a few sea otters would have any effect on the yield of the geoduck fishery (C. Hand, Fisheries and Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Road, Nanaimo B.C., V9R 5K6, Personal Communication, 2008). A large proportion of the WCVI sea otter population is probably what is relevant to the geoduck fishery, rather than individual otters (J. Watson, Personal Communication, 2008). Moreover, the complex ecology of the geoduck populations and environmental variability that happens over years or decades (Orensanz et al. 2004; Valero et al. 2004) further complicates the problem. For the moment, at least, DFO is straining to manage not only the different and sometimes opposing interests for the use of fishery and marine mammal resources (conservation vs. preservation), but fishery managers must also consider the often-unpredictable natural variability of the resources (Haward et al. 2003).

Conflicts that science may not be able to resolve

The life history attributes of marine mammals and the ethics associated with their conservation (or preservation) have no parallel in conventional fisheries (Beverton 1985). Sea otters present a particularly challenging situation not only because of strong federal protection, but because they also have “enthusiastic private supporters” (Armstrong 1979). In addition, scientific experts have indicated that killing some individual otters would probably not harm the B.C. population (DFO 2007c; COSEWIC 2007), which might add to existing tensions between DFO and those being regulated (Lane and Stephenson 2000). The Sea Otter Recovery Team, for example, calculated that a total human-caused mortality of 143 otters per year in B.C., spread out over the range of sea otters could be allowed, “so as not to constrain achievement of the recovery target” (DFO 2007c). Some stakeholders might believe that the government is now protecting individual sea otters in principle, not for ecological reasons, and giving preferred status to marine mammals over fisheries (Manning 1989; Dovetail Consulting 2004). Conversely, a decision that makes economic sense in favor of any fishing industry might lead to an expensive public review of the federal government’s “failure” to protect a wildlife species that scientific assessment confirms to be of concern (e.g., Ecojustice 2008).

Although both philosophies (conservation and preservation) support the protection of those species that are at risk of extinction, the philosophies deviate when the population being protected is not highly endangered. The dispute then becomes a "conflict of political and ethical values" between fisheries and wildlife management that is beyond the scope of science (Manning 1989).

The situation for sea otters is distinct from other high profile Canadian and U.S. policies for marine mammals, such as policies concerning whales. The latter have passed through "5 analytic stages" in regulating international whaling: "free resource, regulation, conservation, protection and preservation" (D'Amato and Chopra 1991). Sea otters disappeared from Canada when they were managed as a free resource, but then were reintroduced in B.C. in the 1970s as a protected species. Policies for sea otters have thus passed through only two analytic stages: free resource, followed by absolute protection. In addition, as for other marine mammals like the whale, a legal progression to a sixth stage of "entitlement to life" may be underway (D'Amato and Chopra 1991) that possibly will ascribe rights to sea otters to consume invertebrate resources. For now, the sea otter is listed as a species of special concern due to its limited distribution and a vulnerability to potentially catastrophic events like oil spills and disease, and because the total population is well below the estimated coast-wide carrying capacity of B.C. (COSEWIC 2007).

Uncertainty in integrated fisheries management

Management of uncertainty is a critical issue in fisheries (Hilborn and Walters 1992). The extent to which fisheries in general are competing with marine mammals is difficult to quantify, but food web competition is expected to increase (Trites et al. 1997).

Some scholars and managers claim that integrated, ecosystem-based management (EBM) can address the complex problems of ensuring sustainable fisheries harvests while meeting other goals (Hall and Mainprize 2004). According to DFO, the goal of marine EBM is to maintain healthy productive ecosystems primarily by transitioning from single species and sector management to collaborative, holistic planning (DFO 2009a). But some analysts argue that comprehensive examples of EBM in the ocean do not yet exist (Ruckelshaus et al. 2008; DFO 2009b). Ruckelshaus et al. (2008) review regional examples where incomplete forms of EBM are currently practiced, such as the highly productive waters of the Bering Sea in Alaska and the southern ocean around Antarctica. For the latter region, where krill is a major prey species of Antarctic predators and commercial fisheries, development and use of ecosystem models resulted in a 25% reduction in the harvest rate of krill to account for the importance of krill to non-human predators (Ruckelshaus et al. 2008). In North America, EBM for marine ecosystems is still in its formative years due in part to the political and economic challenge of reducing harvests to sustainable levels (Hall and Mainprize 2004). Thus, in B.C., although DFO is increasingly focused on multiple-use and sustainable management of marine systems (DFO 2009a), the re-colonization of sea otters may be outpacing development of appropriate tools to operationalize these concepts.

The 2005-2010 Strategic Plan for DFO, *Our Waters, Our Future*, indicates that regulatory reforms in Canada are largely in the planning stages and DFO is still discussing their practical implementation. Through the *Oceans Act*, a significant step toward ecosystem protection on the B.C. coast is the establishment of the Pacific North Coast Integrated Management Area (PNCIMA). PNCIMA is a marine spatial planning initiative in B.C., which spans from the Alaska/B.C. border in the north, to Campbell River and the Brooks Peninsula on Vancouver Island in the south, and extends offshore from the mainland to the base of the continental shelf (DFO 2009a). The overall objective of the PNCIMA initiative is to sustainably manage the north coast's ocean resources by balancing "ecological, economic, social and cultural interests," through the regular engagement of stakeholders and existing advisory processes (e.g., fishery advisory boards to DFO) (PNCIMA 2010b). The PNCIMA is not intended to be a replacement for the existing regulatory processes of governments, but is to provide a framework for equitable management among increasingly diverse stakeholders in the PNCIMA region. Resource users are to be internal to decision-making to reduce uncertainty for coastal communities and the public and private sectors, and to minimize conflicts between future uses (PNCIMA 2010a).

Even under integrated management, however, the sea otter is currently protected, and where it conflicts with a shellfish fishery, shellfish harvesters must heed federal law. Thus, geoduck harvesters are seeking to find a new niche that allows a geoduck fishery in spite of the recovery of otters (IFMP 2010). The industry is investigating geoduck enhancement in regions where sea otters are absent (IFMP 2010), and is working with a jointly-owned local hatchery to spawn geoducks and raise the planted seed in tenured areas outside the sea otter's historical range in the Strait of Georgia (Heizer 2000; IFMP 2010). The fishery and DFO presently do not assert ownership rights to the seeded geoducks but consider them a common property resource (Heizer 2000; B. Clapp, Underwater Harvesters Association, P.O. Box 39005, 3695 W. 10th Ave., Vancouver B.C., V6R 4P1, Personal Communication, 2010).

Towards a resolution

One important task for managers that has already been completed was to define the specific management objectives for recovering sea otters in British Columbia. Wintering habitats of sea otters in B.C. are possibly critical to the survival and recovery of the species (DFO 2007b). Yet species of special concern are not legally eligible for *SARA* “critical habitat” identification or protection and the management plan that will replace the recovery plan for sea otters may not include information about critical habitat. Although it is unlikely that DFO would restrict geoduck fishing in habitat that is important for sea otters, the animals are still protected by the Marine Mammal Regulations under the *Fisheries Act*. It is therefore in the best interest of harvesters to avoid fishing interactions with sea otters. Providing harvesters with additional information about how to avoid sea otter interactions could help them comply with the Marine Mammal Regulations.

To reduce tensions and uncertainty DFO should prioritize (1) identifying important wintering habitats for sea otters, and (2) limiting the regulatory burdens associated with the harvesting of seeded geoduck beds. I base these recommendations on two assumptions. First, that it is DFO’s interest to follow through with studying the animals it is mandated to protect and provide that scientific information to the public. Because sea otters are a popular species but a controversial one to shellfish fisheries, as much relevant information as possible should be made available for DFO to make sound management decisions, while accounting for the conflicting views on the value of sea otter recovery. This information would also help to address the fourth recommended strategy for achieving the recovery goal for sea otters: “Communication to support recovery.” Second, I assume that ecosystem-based fishery management will not only help to reduce conflict but is a management objective of the federal government. For example, accounting for predation mortality in B.C. geoduck stock assessments may eventually result in smaller fishing quotas where sea otters occur. Incentives for innovative research may be critical for encouraging developed fisheries to let go of traditional single-species management (Hall and Mainprize 2004). Incorporating the enhanced geoduck beds under the current management structure, with no additional regulatory conditions imposed could be an equitable means of compensating for potential losses of geoduck fishing areas where sea otters are recovering. It is one example of how stakeholders might collaborate for transitioning to integrated fisheries management.

Conclusion

Fisheries and marine wildlife in Canada are currently managed under overlapping legislative mandates and often with different goals. As demonstrated by my study and the literature, the current situation facing DFO and other managers of B.C. shellfish and sea otters is that of authority in a human-marine mammal conflict. While I could not demonstrate an impact of sea otter predation on the geoduck fishery, the weight of evidence in the literature generally indicates that the recovery of sea otters can be associated with losses of productive shellfish fisheries (e.g., Miller et al. 1975; Wendell et al. 1986; Watson and Smith 1996; Gerber et al. 1999; COSEWIC 2007). Specific fishery concerns mainly relate to the viability of economic opportunities. Success in recovering sea otters in British Columbia, therefore, may depend not only on the ability to prevent potential losses from otter populations, but also on the ability to develop alternative fishing opportunities and to engage previously antagonistic stakeholders in the recovery of the species.

APPENDIX

Table A **Sea otter counts in areas containing selected geoduck study sites in the WCVI. Sea otters were introduced in Checleset Bay near Kyuquot Sound and recently expanded their range to the outer coast of Clayoquot Sound near Tofino after 2000. Two geoduck study sites in Clayoquot Sound (Millar Channel and Yellow Bank) and the sites in Barkley Sound are without otters.** Adapted from Table 2 in Nichol et al. (2009).

	Kyuquot Region*					
Survey Year	**Checleset Bay**	**Mission Group**	**Kyuquot Sound**	**Quatsino Sound**	**Clayoquot Sound**	**Barkley Sound**
1977	55					
1978	51					
1980	60					
1982	97					
1984	196					
1987	234					
1988	201					
1989	329		25			
1990	288		173			
1991	230		50			
1992	257	4	74			
1993	272		91			
1994	413		397			
1995	530		240	1		
2001	663	83	372	52	229	
2002	667	72	417	16	234	
2003	683	80	293	39	183	
2004	740	111	296	86	181	
2008	882	179	461	197	173	

* Kyuquot region in my analysis comprised the sum total of three survey segments in Nichol et al. (2009): Checleset Bay, Mission Group, Kyuquot Sound.

Table B Geoduck survey records for the 17 study sites in the WCVI, summarized to the individual transect level. Letters (*a-f*) identify survey records for the same geoduck bed (or portion of a bed), completed in two different survey years. No repeated survey data were available for Barkley Sound.

Survey			Transect	Number	Number	Area	Mean Geoduck
Year	Location	Site	Length (m)	Geoduck	Quadrats	Sampled	Density (/m²)
1998	Kyuquot	a	265	229	14	140	1.64
1998	Kyuquot	a	285	130	15	150	0.87
1998	Kyuquot	a	600	615	30	300	2.05
1998	Kyuquot	a	525	1039	28	280	3.71
1998	Kyuquot	a	605	703	30	300	2.34
1998	Kyuquot	a	585	239	30	300	0.80
1998	Kyuquot	a	565	171	29	290	0.59
1998	Kyuquot	a	585	42	30	300	0.14
1998	Kyuquot	a	285	5	15	150	0.03
1998	Kyuquot	a	225	63	12	120	0.53
1998	Kyuquot	a	205	355	11	110	3.23
1998	Kyuquot	a	145	125	8	80	1.56
1998	Kyuquot	a	105	30	6	60	0.50
1998	Kyuquot	a	325	25	10	100	0.25
1998	Kyuquot	a	225	21	12	120	0.18
1998	Kyuquot	a	215	40	15	150	0.27
1998	Kyuquot	a	170	37	12	120	0.31
2003	Kyuquot	a	530	8	27	270	0.03
2003	Kyuquot	a	145	219	15	150	1.46
2003	Kyuquot	a	125	65	13	130	0.50
2003	Kyuquot	a	370	190	19	190	1.00
2003	Kyuquot	a	510	469	26	260	1.80
2003	Kyuquot	a	425	393	22	220	1.79
2003	Kyuquot	a	365	325	19	190	1.71
2003	Kyuquot	a	285	25	15	150	0.17
2003	Kyuquot	a	505	99	26	260	0.38
2003	Kyuquot	a	245	0	17	170	0.00
2003	Kyuquot	a	165	184	9	90	2.04
2003	Kyuquot	a	285	25	15	150	0.17
2003	Kyuquot	a	265	14	14	140	0.10
2003	Kyuquot	a	190	111	13	130	0.85
2003	Kyuquot	a	265	21	14	140	0.15
2003	Kyuquot	a	350	36	18	180	0.20
2003	Kyuquot	a	215	49	15	150	0.33
2003	Kyuquot	a	170	9	17	170	0.05
2003	Kyuquot	a	185	154	19	190	0.81
2003	Kyuquot	a	310	27	16	160	0.17
2003	Kyuquot	a	290	32	20	200	0.16
2003	Kyuquot	a	255	32	17	170	0.19
2003	Kyuquot	a	155	74	16	160	0.46
2003	Kyuquot	a	50	12	10	100	0.12
2003	Kyuquot	a	130	25	13	130	0.19
2003	Kyuquot	a	135	128	14	140	0.91
2003	Kyuquot	a	270	22	18	180	0.12
2003	Kyuquot	a	120	46	12	120	0.38
2003	Kyuquot	a	125	9	13	130	0.07
2003	Kyuquot	a	95	5	10	100	0.05
1998	Kyuquot	b	145	12	8	80	0.15
1998	Kyuquot	b	85	16	5	50	0.32

Table B (Continued)

Survey			Transect	Number	Number	Area	Mean Geoduck
Year	Location	Site	Length (m)	Geoduck	Quadrats	Sampled	Density (/m²)
1998	Kyuquot	b	65	66	4	40	1.65
1998	Kyuquot	b	305	31	16	160	0.19
1998	Kyuquot	b	165	1	9	90	0.01
2003	Kyuquot	b	145	18	15	150	0.12
2003	Kyuquot	b	80	26	16	160	0.16
2003	Kyuquot	b	330	27	17	170	0.16
2003	Kyuquot	b	135	14	14	140	0.10
2003	Kyuquot	b	125	22	13	130	0.17
2003	Kyuquot	b	195	30	20	200	0.15
2003	Kyuquot	b	205	48	14	140	0.34
1996	Quatsino Inlet	a	160	281	16	160	1.76
1996	Quatsino Inlet	a	150	13	15	150	0.09
1996	Quatsino Inlet	a	80	219	8	80	2.74
1996	Quatsino Inlet	a	180	212	18	180	1.18
1996	Quatsino Inlet	a	170	503	17	170	2.96
1996	Quatsino Inlet	a	55	200	5	50	4.00
1996	Quatsino Inlet	a	95	152	10	100	1.52
2002	Quatsino Inlet	a	220	96	15	150	0.64
2002	Quatsino Inlet	a	80	113	8	80	1.41
2002	Quatsino Inlet	a	150	165	15	150	1.10
2002	Quatsino Inlet	a	85	399	9	90	4.43
2002	Quatsino Inlet	a	110	100	11	110	0.91
2002	Quatsino Inlet	a	75	386	8	80	4.83
1996	Quatsino Inlet	b	75	20	7	70	0.29
1996	Quatsino Inlet	b	190	536	19	190	2.82
1996	Quatsino Inlet	b	65	0	7	70	0.00
1996	Quatsino Inlet	b	200	216	20	200	1.08
1996	Quatsino Inlet	b	140	36	14	140	0.26
1996	Quatsino Inlet	b	55	3	6	60	0.05
2002	Quatsino Inlet	b	185	194	13	130	1.49
2002	Quatsino Inlet	b	75	39	5	50	0.78
2002	Quatsino Inlet	b	55	86	11	110	0.78
2002	Quatsino Inlet	b	165	408	17	170	2.40
2002	Quatsino Inlet	b	140	300	14	140	2.14
1996	Quatsino Inlet	c	90	72	9	90	0.80
1996	Quatsino Inlet	c	50	11	5	50	0.22
1996	Quatsino Inlet	c	300	538	30	300	1.79
1996	Quatsino Inlet	c	140	39	14	140	0.28
1996	Quatsino Inlet	c	80	8	8	80	0.10
1996	Quatsino Inlet	c	160	135	16	160	0.84
1996	Quatsino Inlet	c	55	28	5	50	0.56
1996	Quatsino Inlet	c	45	5	4	40	0.13
1996	Quatsino Inlet	c	60	40	6	60	0.67
1996	Quatsino Inlet	c	50	12	5	50	0.24
1996	Quatsino Inlet	c	60	19	6	60	0.32
2002	Quatsino Inlet	c	295	50	20	200	0.25
2002	Quatsino Inlet	c	35	5	7	70	0.07

Table B (Continued)

Survey Year	Location	Site	Transect Length (m)	Number Geoduck	Number Quadrats	Area Sampled	Mean Geoduck Density (/m²)
2002	Quatsino Inlet	c	215	216	15	150	1.44
2002	Quatsino Inlet	c	75	148	15	150	0.99
2002	Quatsino Inlet	c	65	61	13	130	0.47
2002	Quatsino Inlet	c	70	14	14	140	0.10
2002	Quatsino Inlet	c	60	89	12	120	0.74
1996	Forward Inlet	a	70	10	7	70	0.14
1996	Forward Inlet	a	80	35	8	80	0.44
1996	Forward Inlet	a	90	6	9	90	0.07
1996	Forward Inlet	a	100	46	10	100	0.46
2002	Forward Inlet	a	150	115	15	150	0.77
2002	Forward Inlet	a	75	12	8	80	0.15
2002	Forward Inlet	a	45	41	9	90	0.46
1996	Forward Inlet	b	205	32	14	140	0.23
1996	Forward Inlet	b	445	237	30	300	0.79
1996	Forward Inlet	b	530	69	36	360	0.19
1996	Forward Inlet	b	670	183	45	450	0.41
1996	Forward Inlet	b	155	64	10	100	0.64
1996	Forward Inlet	b	125	59	8	80	0.74
1996	Forward Inlet	b	120	42	8	80	0.53
1996	Forward Inlet	b	155	103	11	110	0.94
1996	Forward Inlet	b	300	82	20	200	0.41
1996	Forward Inlet	b	350	56	24	240	0.23
2002	Forward Inlet	b	480	61	32	320	0.19
2002	Forward Inlet	b	555	33	28	280	0.12
2002	Forward Inlet	b	625	39	32	320	0.12
2002	Forward Inlet	b	270	98	18	180	0.54
2002	Forward Inlet	b	120	25	12	120	0.21
2002	Forward Inlet	b	135	65	14	140	0.46
2002	Forward Inlet	b	105	99	11	110	0.90
2002	Forward Inlet	b	350	30	24	240	0.13
2002	Forward Inlet	b	355	18	24	240	0.08
2004	Tofino	a	515	58	26	260	0.22
2004	Tofino	a	525	37	27	270	0.14
2004	Tofino	a	490	37	25	250	0.15
2004	Tofino	a	550	50	28	280	0.18
2004	Tofino	a	515	38	26	260	0.15
2004	Tofino	a	470	270	24	240	1.13
2004	Tofino	a	525	465	27	270	1.72
2004	Tofino	a	510	130	26	260	0.50
2004	Tofino	a	480	76	24	240	0.32
2004	Tofino	a	550	69	28	280	0.25
2004	Tofino	a	370	60	19	190	0.32
2004	Tofino	a	465	266	24	240	1.11
2004	Tofino	a	565	169	29	290	0.58
2004	Tofino	a	460	152	23	230	0.66
2004	Tofino	a	570	222	29	290	0.77
2004	Tofino	a	465	176	24	240	0.73

Table B (Continued)

Survey			Transect	Number	Number	Area	Mean Geoduck
Year	Location	Site	Length (m)	Geoduck	Quadrats	Sampled	Density (/m²)
2004	Tofino	a	470	60	24	240	0.25
2004	Tofino	a	530	42	27	270	0.16
2004	Tofino	a	300	16	15	150	0.11
2004	Tofino	a	285	36	15	150	0.24
2008	Tofino	a	160	300	17	170	1.76
2008	Tofino	a	295	228	20	200	1.14
2008	Tofino	a	350	341	24	240	1.42
2008	Tofino	a	500	288	25	250	1.15
2008	Tofino	a	355	163	24	240	0.68
2008	Tofino	a	330	139	22	220	0.63
2008	Tofino	a	250	235	17	170	1.38
2008	Tofino	a	375	384	19	190	2.02
2008	Tofino	a	500	232	22	220	1.05
2008	Tofino	a	335	220	23	230	0.96
2008	Tofino	a	400	309	20	200	1.55
2008	Tofino	a	370	466	25	250	1.86
2008	Tofino	a	395	404	27	270	1.50
2008	Tofino	a	440	203	22	220	0.92
2008	Tofino	a	440	159	22	220	0.72
2008	Tofino	a	415	54	21	210	0.26
2008	Tofino	a	490	25	25	250	0.10
2008	Tofino	a	350	205	18	180	1.14
2008	Tofino	a	440	204	22	220	0.93
2008	Tofino	a	470	280	24	240	1.17
2008	Tofino	a	455	280	23	230	1.22
2008	Tofino	a	450	173	23	230	0.75
2008	Tofino	a	450	233	23	230	1.01
2008	Tofino	a	215	5	15	150	0.03
2008	Tofino	a	340	3	23	230	0.01
2008	Tofino	a	240	0	16	160	0.00
2008	Tofino	a	200	2	14	140	0.01
2008	Tofino	a	150	2	15	150	0.01
2008	Tofino	a	430	33	22	220	0.15
2008	Tofino	a	480	8	24	240	0.03
2008	Tofino	a	230	4	16	160	0.03
2008	Tofino	a	135	3	9	90	0.03
2008	Tofino	a	95	2	10	100	0.02
2008	Tofino	a	185	89	19	190	0.47
2008	Tofino	a	200	77	20	200	0.39
1997	Millar Channel	a	100	227	5	50	4.54
1997	Millar Channel	a	300	574	15	150	3.83
1997	Millar Channel	a	300	547	15	150	3.65
1997	Millar Channel	a	300	266	15	150	1.77
1997	Millar Channel	a	300	311	15	150	2.07
1997	Millar Channel	a	300	183	15	150	1.22
1997	Millar Channel	a	300	315	15	150	2.10
1997	Millar Channel	a	300	303	15	150	2.02

Table B (Continued)

Survey			Transect	Number	Number	Area	Mean Geoduck
Year	Location	Site	Length (m)	Geoduck	Quadrats	Sampled	Density (/m²)
1997	Millar Channel	a	300	237	15	150	1.58
1997	Millar Channel	a	400	352	20	200	1.76
1997	Millar Channel	a	400	550	20	200	2.75
1997	Millar Channel	a	500	49	25	250	0.20
1997	Millar Channel	a	400	6	20	200	0.03
1997	Millar Channel	a	500	7	25	250	0.03
1997	Millar Channel	a	500	21	25	250	0.08
1997	Millar Channel	a	300	137	14	140	0.98
1997	Millar Channel	a	200	178	10	100	1.78
1997	Millar Channel	a	130	16	7	70	0.23
1997	Millar Channel	a	500	668	25	250	2.67
1997	Millar Channel	a	500	519	25	250	2.08
1997	Millar Channel	a	500	771	25	250	3.08
1997	Millar Channel	a	500	0	25	250	0.00
1997	Millar Channel	a	500	1	25	250	0.00
1997	Millar Channel	a	500	4	25	250	0.02
1997	Millar Channel	a	500	268	25	250	1.07
1997	Millar Channel	a	500	431	25	250	1.72
1997	Millar Channel	a	500	565	25	250	2.26
1997	Millar Channel	a	500	278	25	250	1.11
1997	Millar Channel	a	500	298	25	250	1.19
1997	Millar Channel	a	500	239	25	250	0.96
1997	Millar Channel	a	200	464	10	100	4.64
1997	Millar Channel	a	225	382	11	110	3.47
1997	Millar Channel	a	340	524	17	170	3.08
1997	Millar Channel	a	500	0	25	250	0.00
1997	Millar Channel	a	500	2	25	250	0.01
1997	Millar Channel	a	500	0	25	250	0.00
2007	Millar Channel	a	180	360	18	180	2.00
2007	Millar Channel	a	220	483	22	220	2.20
2007	Millar Channel	a	200	236	14	140	1.69
2007	Millar Channel	a	215	300	15	150	2.00
2007	Millar Channel	a	165	218	17	170	1.28
2007	Millar Channel	a	170	263	17	170	1.55
2007	Millar Channel	a	285	425	15	150	2.83
2007	Millar Channel	a	220	179	15	150	1.19
2007	Millar Channel	a	245	261	17	170	1.54
2007	Millar Channel	a	245	225	17	170	1.32
2007	Millar Channel	a	250	412	17	170	2.42
2007	Millar Channel	a	245	441	17	170	2.59
2007	Millar Channel	a	250	394	17	170	2.32
2007	Millar Channel	a	245	200	17	170	1.18
2007	Millar Channel	a	250	14	17	170	0.08
2007	Millar Channel	a	230	773	16	160	4.83
2007	Millar Channel	a	235	571	16	160	3.57
2007	Millar Channel	a	335	386	17	170	2.27
2007	Millar Channel	a	285	261	15	150	1.74

Table B (Continued)

Survey			Transect	Number	Number	Area	Mean Geoduck
Year	Location	Site	Length (m)	Geoduck	Quadrats	Sampled	Density (/m²)
2007	Millar Channel	a	315	236	16	160	1.48
2007	Millar Channel	a	170	506	12	120	4.22
2007	Millar Channel	a	265	513	14	140	3.66
2007	Millar Channel	a	325	431	17	170	2.54
2007	Millar Channel	a	350	646	18	180	3.59
2007	Millar Channel	a	360	523	18	180	2.91
2007	Millar Channel	a	500	845	25	250	3.38
2007	Millar Channel	a	260	184	18	180	1.02
2007	Millar Channel	a	440	444	22	220	2.02
2007	Millar Channel	a	300	77	15	150	0.51
2007	Millar Channel	a	405	409	21	210	1.95
2007	Millar Channel	a	230	62	16	160	0.39
2007	Millar Channel	a	275	36	19	190	0.19
2007	Millar Channel	a	150	10	15	150	0.07
2007	Millar Channel	a	235	352	12	120	2.93
1995	Yellow Bank	a	500	577	24	240	2.40
1995	Yellow Bank	a	735	1106	37	370	2.99
1995	Yellow Bank	a	755	1019	39	390	2.61
1995	Yellow Bank	a	1045	1382	52	520	2.66
1995	Yellow Bank	a	495	589	25	250	2.36
1995	Yellow Bank	a	550	569	28	280	2.03
1995	Yellow Bank	a	1080	931	53	530	1.76
1995	Yellow Bank	a	225	188	12	120	1.57
1995	Yellow Bank	a	175	85	9	90	0.94
2006	Yellow Bank	a	285	290	19	190	1.53
2006	Yellow Bank	a	200	177	14	140	1.26
2006	Yellow Bank	a	49	97	17	170	0.57
2006	Yellow Bank	a	645	317	33	330	0.96
2006	Yellow Bank	a	385	158	20	200	0.79
2006	Yellow Bank	a	525	323	27	270	1.20
2006	Yellow Bank	a	75	31	8	80	0.39
2006	Yellow Bank	a	100	61	10	100	0.61
2006	Yellow Bank	a	70	59	7	70	0.84
2006	Yellow Bank	a	160	187	16	160	1.17
2006	Yellow Bank	a	275	68	19	190	0.36
2006	Yellow Bank	a	180	95	12	120	0.79
2006	Yellow Bank	a	305	106	21	210	0.50
2006	Yellow Bank	a	280	70	19	190	0.37
1997	Yellow Bank	b	500	397	25	250	1.59
2006	Yellow Bank	b	265	285	18	180	1.58
2006	Yellow Bank	b	275	249	19	190	1.31
2006	Yellow Bank	b	175	83	11	110	0.75
2006	Yellow Bank	b	105	103	11	110	0.94
2006	Yellow Bank	b	150	110	15	150	0.73
2006	Yellow Bank	b	180	188	18	180	1.04
2006	Yellow Bank	b	200	129	14	140	0.92
2006	Yellow Bank	b	215	105	15	150	0.70

Table B (Continued)

Survey			Transect	Number	Number	Area	Mean Geoduck
Year	Location	Site	Length (m)	Geoduck	Quadrats	Sampled	Density (/m²)
2006	Yellow Bank	b	170	91	12	120	0.76
2006	Yellow Bank	b	290	300	20	200	1.50
2006	Yellow Bank	b	185	182	19	190	0.96
2006	Yellow Bank	b	155	166	16	160	1.04
2006	Yellow Bank	b	205	173	21	210	0.82
2006	Yellow Bank	b	250	209	17	170	1.23
2006	Yellow Bank	b	270	189	18	180	1.05
2006	Yellow Bank	b	320	259	22	220	1.18
2006	Yellow Bank	b	485	258	25	250	1.03
2006	Yellow Bank	b	555	238	23	230	1.03
2006	Yellow Bank	b	600	309	24	240	1.29
2006	Yellow Bank	b	525	299	21	210	1.42
2006	Yellow Bank	b	445	352	18	180	1.96
2002	Barkley Sound	a	215	140	15	150	0.93
2002	Barkley Sound	a	250	74	17	170	0.44
2002	Barkley Sound	a	175	29	18	180	0.16
2002	Barkley Sound	a	110	216	11	110	1.96
2002	Barkley Sound	a	70	112	7	70	1.60
2002	Barkley Sound	a	105	76	11	110	0.69
2002	Barkley Sound	b	110	124	11	110	1.13
2002	Barkley Sound	b	75	116	8	80	1.45
2002	Barkley Sound	b	115	330	12	120	2.75
2002	Barkley Sound	b	45	134	8	80	1.68
2002	Barkley Sound	b	245	0	17	170	0.00
2002	Barkley Sound	b	160	0	15	150	0.00
2002	Barkley Sound	c	210	60	21	210	0.29
2002	Barkley Sound	c	150	112	15	150	0.75
2002	Barkley Sound	c	85	11	17	170	0.06
2002	Barkley Sound	c	125	224	13	130	1.72
2002	Barkley Sound	c	145	166	15	150	1.11
2002	Barkley Sound	c	135	97	14	140	0.69
2002	Barkley Sound	c	175	358	18	180	1.99
2002	Barkley Sound	c	125	119	13	130	0.92
2002	Barkley Sound	c	40	0	8	80	0.00
2002	Barkley Sound	c	70	1	14	140	0.01
2005	Barkley Sound	d	75	97	8	80	1.21
2005	Barkley Sound	d	20	0	4	40	0.00
2005	Barkley Sound	d	50	358	10	100	3.58
2005	Barkley Sound	d	80	64	8	80	0.80
2005	Barkley Sound	d	80	86	16	160	0.54
2005	Barkley Sound	d	50	108	5	50	2.16
2005	Barkley Sound	e	45	136	5	50	2.72
2005	Barkley Sound	e	20	29	4	40	0.73

Table B (Continued)

Survey			Transect	Number	Number	Area	Mean Geoduck
Year	Location	Site	Length (m)	Geoduck	Quadrats	Sampled	Density (/m²)
2005	Barkley Sound	e	45	169	9	90	1.88
2005	Barkley Sound	e	30	22	6	60	0.37
2005	Barkley Sound	e	35	3	7	70	0.04
2005	Barkley Sound	e	25	8	5	50	0.16
2005	Barkley Sound	e	135	55	14	140	0.39
2005	Barkley Sound	e	140	33	14	140	0.24
2005	Barkley Sound	e	125	24	13	130	0.18
2005	Barkley Sound	f	35	0	7	70	0.00
2005	Barkley Sound	f	100	102	10	100	1.02
2005	Barkley Sound	f	140	307	14	140	2.19
2005	Barkley Sound	f	130	196	13	130	1.51
2005	Barkley Sound	f	80	7	8	80	0.09
2005	Barkley Sound	f	120	230	12	120	1.92
2005	Barkley Sound	f	70	111	7	70	1.59
2005	Barkley Sound	f	25	92	5	50	1.84
2005	Barkley Sound	f	20	0	4	40	0.00

Table C Geoduck age samples from sites with otters, disaggregated into 31 age-classes at intervals of 5 years. Letters *a* and *c* identify samples from the same geoduck bed, collected in two different survey years.

	Kyuquot		Quatsino Inlet		Quatsino Inlet		Forward Inlet		Tofino	
Geoduck	**1998**	**2003**	**1996**	**2002**	**1996**	**2002**	**1996**	**2002**	**2004**	**2008**
Age Class	**a**	**a**	**a**	**a**	**c**	**c**	**a**	**a**	**a**	**a**
5	70	5	1	2	0	4	9	8	95	90
10	97	74	10	33	7	62	63	100	39	51
15	29	119	4	9	15	13	8	22	92	18
20	15	36	1	3	4	4	2	10	25	77
25	13	23	2	1	2	5	1	2	1	10
30	18	16	2	0	9	3	3	2	5	0
35	10	6	2	4	5	2	6	0	17	0
40	11	7	14	12	24	2	4	1	12	9
45	5	2	10	15	4	3	7	3	6	1
50	10	2	10	12	5	8	11	3	1	1
55	4	1	5	15	6	12	10	1	1	1
60	8	1	8	6	14	8	10	3	1	0
65	4	5	15	8	7	3	3	2	0	1
70	1	3	11	5	3	7	6	2	0	0
75	1	3	15	5	8	9	3	0	1	0
80	1	2	8	4	6	14	2	0	0	0
85	0	0	6	6	3	5	1	0	0	0
90	1	0	5	8	7	4	2	1	0	0
95	2	0	1	4	4	1	0	0	0	0
100	2	0	4	2	4	1	1	0	0	0
105	1	2	2	6	1	0	1	0	0	0
110	0	0	1	1	0	1	0	0	0	0
115	0	0	3	0	2	0	0	0	0	0
120	1	1	1	0	0	0	0	0	0	0
125	0	0	0	0	1	0	0	0	0	0
130	0	0	0	1	0	0	0	0	0	0
135	0	0	0	2	1	0	0	0	0	0
140	0	0	0	0	0	0	0	0	0	0
145	0	0	0	0	0	0	0	0	0	0
150	0	0	0	0	0	0	1	0	0	0
155	0	0	0	0	0	0	2	0	0	0

Table D **Geoduck age samples from sites without otters, disaggregated into 31 age-classes at intervals of 5 years. Letters *a* and *b* for Millar Channel and Yellow Bank identify samples from the same geoduck bed, collected in two different survey years. No repeated samples were available for Barkley Sound.**

	Millar Channel		Yellow Bank		Barkley Sound			Barkley Sound		
Geoduck Age Class	**1997 a**	**2007 a**	**1997 b**	**2006 b**	**2002 a**	**2002 b**	**2002 c**	**2005 d**	**2005 e**	**2005 f**
5	101	87	45	25	2	0	0	3	4	2
10	24	66	30	37	20	0	3	17	8	4
15	16	143	11	138	10	1	13	17	17	19
20	11	74	9	104	1	1	3	4	17	4
25	4	10	5	22	2	1	5	17	7	10
30	8	7	10	33	3	3	2	2	2	3
35	15	1	13	13	10	14	2	2	1	2
40	29	16	27	6	10	6	2	1	7	19
45	11	14	10	5	17	18	8	5	7	15
50	16	45	7	15	14	14	8	4	3	15
55	9	52	7	22	20	22	14	9	3	18
60	16	14	4	6	19	23	22	22	7	15
65	6	20	2	11	16	20	15	11	3	7
70	6	6	0	6	16	18	10	25	4	10
75	2	3	2	2	8	11	4	17	4	6
80	0	3	2	1	6	7	4	14	2	6
85	0	0	0	0	4	4	10	13	2	1
90	2	0	1	0	1	2	7	8	0	3
95	0	0	1	2	0	0	2	3	1	0
100	1	0	0	0	3	1	6	3	0	1
105	0	1	0	0	0	1	2	2	1	0
110	0	0	0	1	1	0	3	0	1	0
115	0	0	0	0	0	0	4	1	1	0
120	0	0	0	0	0	0	2	3	0	1
125	0	0	0	0	0	0	0	1	0	0
130	0	0	0	0	0	0	0	1	1	0
135	0	0	0	0	0	0	0	1	0	0
140	0	0	0	0	0	0	0	0	0	0
145	0	0	0	0	0	0	0	0	0	0
150	0	0	0	0	0	0	0	0	0	0
155	0	0	0	0	0	0	0	0	0	0

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