# SIZE-SELECTIVITY OF BRITISH COLUMBIA'S SABLEFISH (*ANOPLOPOMA FIMBRIA*) FISHERIES AND IMPLICATIONS FOR THE ECONOMIC LOSSES ASSOCIATED WITH DISCARDING

by

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B.A. with Distinction, University of Victoria, 2004

RESEARCH PROJECT
SUBMITTED IN PARTIAL FULFILLMENT OF
THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF RESOURCE MANAGEMENT

In the
School of Resource and Environmental Management

Report No. 449



SIMON FRASER UNIVERSITY

Summer 2008

**APPROVAL**

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| **Degree:** | **Master of Resource Management** |
| **Title of Thesis:** | **Size-selectivity of British Columbia's sablefish (*Anoplopoma fimbria*) fisheries and implications for the economic losses associated with discarding** |
| **Report No.:** | **449** |

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## ABSTRACT

I used multiple mark-recapture experiments for British Columbia (B.C.) sablefish (*Anoplopoma fimbria*) to estimate size-selectivity functions for three commercial gear types employed in the B.C. sablefish fishery: (*i*) trap, (*ii*) trawl, and (*iii*) longline gear. Notable differences in selectivity were observed among gear types with the longline fishery selecting for large sablefish, the trap fishery selecting for intermediate-sized sablefish, and the trawl fishery selecting for small sablefish below the minimum size limit. Empirical estimates of gear selectivity were incorporated into yield-per-recruit (YPR) and spawner biomass-per-recruit models to evaluate the effects of at-sea discarding on long-term fishery yield. My results suggest that up to 49% of the total YPR is potentially lost because of at-sea discarding. Fishery regulations that minimize the capture of sub-legal sablefish in combination with economic incentives that encourage the retention of legal-sized sablefish can help to mitigate potential losses in fishery yield resulting from at-sea discarding.



**Keywords:** *Anoplopoma fimbria*; sablefish; size-selectivity; discard mortality; mark-recapture; yield-per-recruit model

**ACKNOWLEDGEMENTS**

I would like to thank my senior supervisor, Dr. Sean Cox, for his invaluable guidance, wisdom, and expertise as well as my second committee member, Robert Kronlund, for his insight, knowledge, and witty sense of humour. I would also like to thank the Canadian Sablefish Association who provided generous funding for this research, as well as SSHRC and NSERC, for additional financial support.

I am grateful to my colleagues in the Fisheries Science and Management Group at Simon Fraser University who helped me to understand, question, and explore the wonders of Fisheries Science. My cohort, in particular, provided relentless support and insightful discussions throughout this inspiring journey.

Finally, I would like to thank Brendan Connors who, as always, provided unconditional love and support along the way.

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# CHAPTER 1
# GENERAL INTRODUCTION

One of the primary goals of fisheries management is to provide for the sustainable yield from fish stocks over time (Gulland 1983; Hilborn and Walters 1992). Using a mix of regulatory controls (e.g., trip limits, area closures) and enhancement actions (e.g., habitat improvement, spawning channels), fishery managers seek to design policies that provide for the social and economic well-being of fishermen and the industries that depend on them while balancing conservation efforts (Hilborn and Walters 1992). Fishery agencies often engage in formal stock assessments in order to model the dynamics of an exploited stock (e.g., birth, death, and growth rates, as well as movement patterns) and make quantitative predictions about potential reactions of fish stocks to alternative management decisions (Hilborn and Walters 1992). Information on the total catch, relative abundance, and the life history of the species being assessed are important inputs into stock assessment models. This information is generally obtained from a mix of fishery-dependent and fishery-independent data sources (Cooper 2008). Fishery-independent sources refer to data derived from activities not involved in the commercial or recreational harvest of fish, such as trawl, acoustic, and video research surveys (Cooper 2008). Fishery-dependent data, on the other hand, refers to information derived from the fishing process itself and are generally collected by onboard observers, dockside monitors, and/or self-reporting (Cooper 2008).

The collection of unbiased fishery-dependent data is critical to the development of robust harvest strategies for commercially exploited fish stocks. Commercial catch statistics must reflect all fishery removals from the stock, including the mortality of the discarded bycatch, which are fish caught during the fishing process that were not specifically targeted for harvest (FAO 2008). Accounting for the mortality of discarded fish can be difficult, particularly when a species is harvested by a number of gear types and in a variety of fisheries. In the directed B.C. sablefish fishery, for example, both traps and hooks are used to harvest sablefish. Additional sablefish are harvested in the non-directed trawl fishery and as bycatch in other hook and line fisheries (DFO 2007). While some sablefish bycatch is landed and recorded in catch statistics, some is released at-sea because of legal requirements and/or market/economic considerations (FAO 2008).

A formal evaluation of the impacts of discarding on long-term fishery yield and revenue requires a basic understanding of the size-selectivity of commercial fishing gears (Chen and Gordon 1997). Selection ogives, describing how fishing mortality varies with age or size, are generally estimated for each gear type in a fishery using direct or indirect methods (Millar and Fryer 1999; Clark and Kaimmer 2006). When the selective properties of commercial fishing gears are known, in addition to information on growth and mortality, quantitative population models can be used to evaluate potential losses in yield and revenue that result from at-sea discarding (Chen and Gordon 1997).

A number of alternative models are available to describe the response of exploited fish stocks to at-sea discarding, including biomass dynamic models (BDMs) and age structured models (ASMs; Hilborn and Walters 1992). Biomass dynamic models are the simplest type of fisheries population model requiring only catch and effort data to model stock dynamics (Hilborn and Walters 1992). Biomass dynamic models make no reference to the age or length composition of the stock, but rather model the stock as a total aggregate unit (i.e., total biomass; Lleonart 1993; King 1995). Unfortunately, BDMs do not consider other factors affecting the dynamics of the stock, such as recruitment variability, time lags before recruitment to the spawning stock, or the differential vulnerability of fish to capture by a particular gear type (i.e., gear selectivity). A number of fisheries are, therefore, managed using age structured models that explicitly account for the age or size composition of the resource (Deriso 1987; Lleonart 1993; Sparre and Venema 1998; Malcolm 2001). ASMs are generally preferred to BDMs because they can incorporate auxiliary information on the stock, such as the number of recruits entering the fishery each year, the productivity of the population, growth and mortality, and the size-selectivity of the gear (Gulland 1983). In ASMs, total population size is broken down into age or length classes, where age or length is associated with differences in size, growth rates, fecundity, and vulnerability to capture (Hilborn and Walters 1992). In this paper, I use a length-based model of yield-per-recruit (YPR), which is a particular class of ASMs, to explore biological and economic losses in yield that result from at-sea discarding.

## Yield-per-recruit models

Yield-per-recruit (YPR) and spawner biomass-per-recruit (SBPR) models are the simplest form of age structured population models (Hilborn and Walters 1992). YPR analyses form the basis for the assessment of many fish stocks that lack the comprehensive catch-at-age and catch rate data required by more advanced age structured models (Gulland 1983; Griffiths 1997; Malcolm 2001). Incorporating the interplay between growth and the probability of death, YPR models predict the lifetime yield of a cohort under different combinations of fishing mortality and age-at-first-capture (Butterworth et al. 1989).

YPR and SBPR analyses provide a useful tool for fishery managers by allowing them to explore the theoretical effects of different fishing regimes on long-term fishery yield and revenue (Pikitch 1987; Chen and Gordon 1997). YPR models are frequently used to derive important biological reference points (BRPs), or values that indicate the state of a resource or stock status relative to an acceptable value or range (Caddy and Mahon 1995). At the policy and legal levels, BRPs are often involved in defining overfishing and rebuilding overfished thresholds for exploited fish stocks. A common BRP is $F^{max}$, or the fishing mortality rate that corresponds to the maximum on the YPR curve. Unfortunately, fishing mortality rates based on $F^{max}$ maximize YPR without regard to whether sufficient spawner biomass is conserved to ensure sufficient recruitment in the future (Deriso 1987; Clark 1991). Consequently, $F^{max}$ tends to cause overfishing and stock declines (Hilborn and Walters 1992). Most BRP recommendations are, therefore, based on the results of spawner biomass-per-recruit models (SBPR) that explicitly consider the effects of fishing mortality on the potential spawning stock (Butterworth et al. 1989; Griffiths 1997). Common YPR and SBPR models include the formulations by Beverton and Holt (1957), Ricker (1975), and Thompson and Bell (1934). Discrete YPR models analogous to Thompson and Bell (1934) are often favoured for their numerical simplicity (Chen and Gordon 1997) and their ability to incorporate age- or length-specific gear selectivity.

## Gear selectivity

Gear selectivity is a particularly important input into per-recruit models because it determines the probability of a fish dying before it reaches peak biomass-per-recruit and peak spawning output. When a sufficient understanding of gear selectivity is in place, fishery managers can better detect, and ideally avoid, growth and recruitment overfishing. Growth overfishing refers to a situation in which fish are removed from the population while they are still growing rapidly while recruitment overfishing refers to a scenario in which the spawning stock is reduced such that a sufficient number of recruits to the fishery is no longer produced (Walters and Martell 2004).

Gear selectivity is most commonly estimated using indirect methods, such as comparative catch studies, in which size distributions of catches among different gear variants are used to infer the relative selectivity of each gear type (e.g., Millar 1992; Suuronen and Millar 1992; Walsh et al. 1992; Millar and Fryer 1999). In comparative approaches, the selectivity of the gear and the size distribution of the population are estimated simultaneously and no prior knowledge of the size distribution of the stock is required (Millar and Fryer 1999). While such indirect experiments can provide valuable information on the relative selectivity of various gear types, unless the true size distribution of the population is known, comparative catch studies cannot be used to determine the functional form of the relationship between size and susceptibility to capture (Millar 1995; Myers and Hoenig 1997; Clark and Kaimmer 2006).

Direct estimates of selectivity are possible where the size structure of the population is known or can be reliably estimated through designed experiments, such as mark-recapture studies, in which animals are captured and tagged at random with unique identifiers before being released back into the natural environment (e.g., Hamley and Regier 1973). When animals are randomly recaptured, their tag information is recorded, and the animals are either retained or re-released. Comparisons between the size distribution of the initial marked population and the size distribution of the recovered individuals can be used to provide direct estimates of selectivity by length (Millar and Fryer 1999; Walters and Martell 2004). Mark-recapture studies can be particularly effective at providing direct estimates of selectivity (e.g., Hamley and Regier 1973; Myers and Hoenig 1997; Clark and Kaimmer 2006). Unfortunately, tagging studies are expensive to conduct because of the large sample sizes and high recapture rates needed to estimate gear selectivity (Ricker 1975; Hilborn and Walters 1992). The use of tagging data to estimate the size-selectivity of fishing gears has therefore only been applied in a few cases, including a gillnet study on tagged walleye (Hamley and Regier 1973), multiple tagging experiments on Atlantic cod (Myers and Hoenig 1977), and various mark-recapture studies for Pacific halibut (Clark and Kaimmer 2006).

In this study, I employ a direct method for estimating the size-selectivity of three commercial gear types employed in the B.C. sablefish (*Anoplopoma fimbria*) fishery: (*i*) longline trap, (*ii*) longline hook, and (*iii*) trawl gear. The resultant selectivity estimates are used as inputs into yield-per-recruit and spawner biomass-per-recruit models to explore the relationship between discarding and expected losses in yield and revenue for the B.C. sablefish fishery.

## B.C. sablefish fishery

Sablefish are one of the most economically important species fished in British Columbia with landings averaging 3,800 metric tonnes between 1995 and 2004 (DFO

2005a) producing an average annual landed value of CDN $26 million (MFCR 2001; MAFF 2001; MAFF 2002; MOE 2004). Sablefish, also referred to as black cod (AAC 2007), are endemic to the North Pacific Ocean (Allen and Smith 1988). Adult sablefish are generally found within 1 m of the ocean floor (Kreiger 1997) at depths greater than 200 m, although some sablefish have been captured at depths greater than 1,500 m (AAC 2007). Sablefish are distributed throughout the North Pacific from Baja California to the Gulf of Alaska, westward to the Aleutian Islands, and into the Bering Sea (DFO 2005). Spawning occurs in pelagic waters near the edge of the continental slope (300 m – 500 m) between January and March (DFO 2005). Larval sablefish drift on surface currents above the continental shelf and slope in April and May before migrating inshore to protected coastal habitats (Kreiger 1997). Juveniles remain in the nearshore environment for two to three years before migrating offshore where they reside as adults and recruit to commercial fisheries (Mason et al.1983; DFO 2005).

Sablefish have been harvested commercially for more than 100 years with the first commercial landings reported in 1913 (McFarlane and Beamish 1983). Between 1913 and 1981, foreign longline fleets from Japan, the former USSR, and the Republic of Korea (Haist et al. 2001) almost exclusively harvested sablefish. Foreign fishing was gradually phased out by 1977 with the development of Canada's 200-mile Exclusive Economic Zone (Haist et al. 2001). The management and assessment of B.C. sablefish are presently conducted cooperatively by Fisheries and Oceans Canada (DFO) and the Canadian Sablefish Association (CSA), which represents commercial sablefish license holders in the longline trap, longline hook, and trawl fisheries as well as non-governmental organizations (DFO 2007). Under the support of a joint project agreement, the CSA makes annual financial contributions towards various management and assessment activities for B.C. sablefish including biological studies, enforcement activities, tagging experiments, and stock assessments (DFO 2007). The fishery is managed using a total allowable catch (TAC) that is set annually based on assessment and yield recommendations identified by the Pacific Scientific Advice Review Committee (DFO 2007). An individual vessel quota (IVQ) system has been used to allocate catch among license holders in the directed sablefish fishery since 1990 and in the non-directed trawl fishery since 1997 (DFO 2007). Under an IVQ system, fixed shares of the TAC are allocated to individual licensed commercial sablefish vessels that were originally based on their historic average catch and overall vessel length (Haist et al. 2004).

The directed sablefish fishery currently operates year round (July 31$^{st}$ – Aug 1$^{st}$) under a category "K" license (Haist et al. 2004). Category "K" licensed vessels are permitted to harvest sablefish using longline trap and/or longline hook gear (Haist et al. 2004). Sablefish longline fishing is a technique that uses dozens of traps or hooks attached to a single groundline resting on the ocean floor (Haist et al. 2001). Heavy anchors temporarily secure both ends of the groundline to the ocean bottom (Wyeth and Kronlund 2003). In the commercial longline trap fishery (hereafter referred to as the trap fishery), between 50 and 80 conical Korean traps baited with a combination of frozen California squid (*Loligo* sp.) and frozen Pacific hake (*Merluccius productus*) are deployed along the groundline at 46 m intervals (Haist et al. 2004). Sablefish traps are generally deployed between 275 m and 1200 m; however, the majority of commercial trap fishing effort occurs between 460 m and 825 m (Haist et al. 2001). The longline hook fishery (hereafter referred to as the longline fishery) operates in much shallower waters, with over 80% of the fishing effort occurring in depths less than 460 m (Haist et al. 2001). In longline sets, between 500 to 1500 hooks baited with frozen California squid are attached to the groundline at 1.8 m to 3.6 m intervals (Haist et al. 2001).

The directed sablefish fishery receives the majority of the commercial sablefish TAC, of which 60% to 80% is harvested using trap gear (Haist et al. 2001). Another 8.75% of the sablefish TAC is allocated to the multispecies groundfish trawl fishery (DFO 2007). Operating under a category "T" license, the B.C. trawl fishery operates year round (April 1$^{st}$ – March 31$^{st}$). Only sablefish recovered in the Option A trawl fishery were considered in this analysis to restrict the study to sablefish residing in offshore areas.

## Sablefish tagging program

Sablefish have been tagged and released as part of the annual standardized research and assessment survey since 1991 (Wyeth and Kronlund 2003). During the assessment survey, sablefish collected in excess of the biological sampling requirements have been tagged and released at nine fixed indexing localities each year (Wyeth and Kronlund 2003). In 1994, a formalized tagging program was developed and six additional offshore localities were added to the annual research and assessment survey for the explicit purpose of tagging and releasing sablefish (Figure 1; Wyeth and Kronlund 2003). Under the formal tagging program, one thousand tagged sablefish are released at each of the six offshore tagging localities and three hundred sablefish are tagged and released from one additional set at each of the nine standardized survey localities (Table 1; Haist et al. 2001). Between fifty and seventy Korean traps, baited with a combination of

California squid (1 – 1.5 kg) and Pacific hake (3 – 4.5 kg), are deployed along a groundline between 457 m and 824 m (the depth zone known to produce the highest catch rates; Haist et al. 2001; Wyeth and Kronlund 2003). Between 1995 and 2004, approximately 131,000 sablefish were tagged and released during the formal sablefish tagging program (Figure 2).

Tagged sablefish are recovered throughout the year by the directed sablefish fishery and the non-directed trawl fishery (Haist et al. 2001). Additional tagged sablefish are recovered in the annual sablefish research and assessment survey and as bycatch in other longline fisheries (DFO 2007). Tagged fish recovered in the sablefish assessment survey are either retained by the survey vessel or re-released at-sea once fork length and tag identification numbers have been recorded (Wyeth and Kronlund 2003). Tagged sablefish recovered in the commercial groundfish fishery must be landed at designated offloading locations where they are validated by independent dockside observers (DFO 2007), who also collect important biological data including fork length, sex, and maturity data (Haist et al. 2004). While numerous incentives are in place to encourage the reporting of tagged sablefish by all commercial groundfish sectors (Haist et al. 2004), it is likely that not all tag recaptures are reported (Haist et al. 2001). During the ten-year period between 1995 and 2004, only 10% of the vessels operating in the directed sablefish fishery were required to have fishery observers onboard at any given time, generating some uncertainty in compliance with tag reporting requirements in the directed sablefish fishery (DFO 2003).

## Study objectives

The specific objectives of this study were two-fold: (*i*) to estimate the size-selectivity of the three gear types used in the B.C. sablefish fishery and (*ii*) to use empirical estimates of selectivity to evaluate the potential effects of discarding on projected fishery yield and revenue. Using a large number of mark-recapture experiments conducted on B.C. sablefish between 1995 and 2004, Chapter 2 describes a method of estimating the size-selectivity of the trap, trawl, and longline fisheries. The relationship between size and susceptibility to capture was estimated for each gear type, and differences in selectivity among and within gear types were identified over time. The resultant selectivity estimates were incorporated into length-based models of yield-per-recruit (YPR) and spawner biomass-per-recruit (SBPR), developed in Chapter 3, to explore the biological and economic impacts of at-sea discarding on the B.C. sablefish fishery. The traditional YPR and SBPR models were modified in Chapter 3 to reflect three different assumptions regarding the mortality of the discarded catch. Discard scenarios either assumed that:

1. all of the discarded catch survives,
2. all of the discarded catch dies, but discards are still included in estimates of YPR and SBPR, or
3. all of the discarded catch dies, but is ignored in estimates of YPR and SBPR.

A comparison among discard scenarios enabled me to examine the bias present in YPR and SBPR models that do not explicitly account for the mortality of the discarded catch when estimating fishery yield. Losses in yield were translated into economic losses per-recruit to provide an indication of the potential magnitude of losses in revenue that result from at-sea discarding.

This study is unique in that it uses direct estimates of selectivity-at-length to explore the potential impacts of discarding on biological and economic yields. Direct estimates of selectivity-at-length by gear type also improves the precision of total fishing mortality estimates in stock assessments and important management reference points (e.g., $F^{max}$). Although tagging data has been used extensively in previous sablefish assessments to estimate mortality rates and stock abundance (e.g., Haist and Hilborn 2000; Haist et al. 2001; Kronlund et al. 2002; Wyeth and Kronlund 2003; Haist et al. 2004; Haist et al. 2005), selectivity has not been previously estimated from sablefish tagging data.

# CHAPTER 2
# ESTIMATING THE SIZE-SELECTIVITY OF B.C.'S SABLEFISH (*ANOPLOPOMA FIMBRIA*) FISHERIES


## Abstract

Sablefish tagging data collected between 1995 and 2004 were used to estimate size-selectivity functions for three gear types employed in the B.C. sablefish (*Anoplopoma fimbria*) fishery: (*i*) trap, (*ii*) trawl, and (*iii*) longline gear. Differences in estimated selectivity were found among gear types, with the longline fishery selecting for the largest size-classes of sablefish, the trap fishery selecting for intermediate sizes, and the trawl fishery selecting for small sablefish below the minimum size limit. Size-selectivity functions were asymptotic for the longline fishery and exponential for the trawl fishery, whereas estimated selectivity functions for the trap fishery varied over time between asymptotic and dome-shaped. Monitoring and accounting for the size-selectivity of commercial fishing gears is critical to effective management of exploited fish stocks and the development of sustainable fishery harvest strategies. The temporal variations in size-selectivity observed in the commercial trap fishery suggest that further consideration should be given to the non-stationarity of size-selectivity functions in the ongoing management strategy evaluation for B.C. sablefish.

## Introduction

Gear selectivity describes the relationship between the size (or age) of a fish and its susceptibility to capture by a given type of fishing gear (Clark and Kaimmer 2006). Nearly every method of fishing removes a highly selective sample of fish from the population (Hilborn and Walters 1992). Identifying the size-selectivity of commercial fishing gears is therefore important for identifying appropriate harvest rates for a fishery (Rahikainen et al 2004), developing meaningful gear regulations for a fleet (Huse et al. 2000), and predicting the impacts of various gear types on stock abundance and fish community structure (Bianchi et al. 2000).

Fishing affects species diversity and the size composition of exploited fish stocks through the selective removal of target species, the bycatch of non-target species, and habitat modification (Pauly 1979; Haedrich and Barnes 1997; Sainsbury et al. 1997; Bianchi et al. 2000). Fisheries-induced evolution in life-history traits, especially in characters determining maturation, has been documented in a number of marine and freshwater fish (Ricker 1981; Conover and Munch 2002; Law 2007; Morita and Fukuwaka 2007) including stocks of all five species of Pacific salmon (*Oncorhynchus* sp.; Ricker 1981; Morita and Fukuwaka 2007), Northeast Atlantic cod (*Gadus morhua*; Hutchings 1996), and Pacific hake (*Merluccius productus*; Welch and McFarlane 1990). Earlier sexual maturation at smaller sizes and elevated reproductive effort has been linked to heavy fishing mortality with faster or slower growth resulting depending on the underlying relationship between growth, maturation, and fecundity (Law and Grey 1989; Walters and Martell 2004; Jørgensen et al. 2007). Research suggests that environmental factors alone are unlikely to account for such widespread changes in maturity schedules with fisheries-induced evolution consistently arising as the most parsimonious explanation after environmental factors have been considered (Conover and Munch 2002; Walters and Martell 2003; Jørgensen et al. 2007; Law 2007; Morita and Fukuwaka 2007).

With life history traits being among the foremost determinants of population dynamics, their evolution has important implications for stock biomass, stability, and demography, as well as for the recovery potential of exploited fish stocks (Law and Grey 1989; Conover and Munch 2002). Rapid evolutionary changes in life-history parameters, such as size- or age-at-maturity, can also affect the susceptibility of different age-classes to capture by different types of fishing gear (e.g., Parma 2002). In the case of the Pacific halibut fishery, a reduction in halibut size- and maturity-at-age between the 1970s and the mid-1990s delayed younger halibut susceptibility to commercial hook and line gear (Clark et al. 1999). Declining catch rates for smaller halibut in the longline survey were interpreted as a reduction in exploitable biomass, rather than the decreasing susceptibility of younger age classes to hook and line gear (Clark et al. 1999; Parma 2002). While it is not clear whether such changes in life-history traits were the result of fishing practices or a density-dependent response to high abundance, failure to account for changes in growth and their concomitant effects on gear selectivity within the assessment model nonetheless resulted in the persistent under-estimation of recruitment and stock abundance for Pacific halibut during the mid-1980s to the 1990s (Clark et al. 1999).

A similar scenario unfolded in the Atlantic cod fishery in Eastern Canada during the late 1980s, when Northeast cod stocks were managed under the assumption that the selectivity of bottom trawl fisheries decreased for older ages (Myers and Cadigan 1995; Myers et al. 1997). This assumption, however, was not supported by empirical research

(Myers and Cadigan 1995) and led to the overestimation of spawning stock biomass (Myers et al. 1997). Erroneous assumptions regarding the size-selectivity of bottom trawlers has since been identified as a key factor in the collapse of Atlantic cod stocks in Eastern Canada (Myers et al. 1997) and an important parameter to estimate for all commercially exploited fish stocks.

This study had two primary objectives: (*i*) to quantify the relationship between body length and the probability of capture for tagged sablefish harvested in the B.C. sablefish fishery and (*ii*) to identify differences in size-selectivity among the three gear types. To accomplish these objectives, I used mark-recapture data to generate direct estimates of selectivity by length for each gear type in the fishery. Three candidate models of selectivity (asymptotic, exponential, and dome-shaped) were considered and estimation and statistical tests were used to determine the best model fit to the data.

# Methods

## Model development

Size-selectivity functions were estimated from multiple mark-recapture experiments conducted on B.C. sablefish during the ten-year period between 1995 and 2004 (Table 1). A tagging experiment was defined as all sablefish released in a given length class in a given year. Releases of tagged sablefish were divided into 5 cm classes between 30 cm and 95 cm fork length (all length classes hereafter refer to fork length). Minimum recapture sample size requirements were calculated for each length category $l$, gear type $g$, and year $y$ based on pre-determined limits of error $\delta$ (Appendix). Recoveries were required in a minimum of three length classes in each year and only tags recovered within one year of release were considered in order to minimize the effects of growth and natural mortality during the time at liberty (Myers and Hoenig 1997).

Following the method of Myers and Hoenig (1997), the expected value of the reported catch of tagged fish, $E[C_{y,l}]$, is (notation for gear type $g$ is omitted),

**(1)** $$E[C_{y,l}] = N_{y,l}\ \pi_{y,l}\ ,$$

where $N_{y,l}$ is the total number of sablefish released in each length class and $\pi_{y,l}$ is the capture probability of a tagged fish in length class $l$.

If I assume that the probability of capture is the same for all fish of a given length and that the recoveries of tagged fish are independent of one another and occur at random during the course of the fishery, then the capture probability of a tagged fish can be separated into year- and length-based components, i.e.,

**(2)** $$\pi_{y,l} = S_{y,l} U_y R_y\ ,$$

where $S_{y,l}$ is the selectivity of the gear for length class $l$ in a given year $y$ and $U_y$ is the exploitation rate on the length class most vulnerable to the gear in a given year. $R_y$ is a year-specific random effect representing the combined effects of various factors that occur during the mark-recapture process including tag misreporting, tag loss, tagging mortality, and natural mortality (Myers and Hoenig 1997; Clark and Kaimmer 2006). Variables $U_y$ and $R_y$ are confounded in the above equation and thus a new term, $U'$, is defined to represent the combined effect of these two parameters. Therefore, the model for $\pi_{y,l}$ becomes,

**(3)** $$\pi_{y,l} = S_{y,l} U_y' \ .$$

Gear selectivity can be described by a number of statistical models including monotone selection curves (e.g., logistic), unimodal curves, lognormal curves, and inverse Gaussian curves (Millar and Fryer 1999). Logistic functions are ideal for modelling capture probabilities as a function of length in mark-recapture experiments for a number of reasons: (*i*) they are easy to comprehend and therefore good for parameter interpretation, (*ii*) they possess asymptotes at zero and one such that negative probabilities and probabilities greater than one can never be predicted, and (*ii*) they are eminently suited for the analysis of data collected retrospectively, such as mark-recapture data (McCullagh and Nelder 1989). In this study, I explored asymptotic and dome-shaped functional relationships between selectivity and sablefish length for the trap and longline fisheries. The asymptotic model, $S_{y,l}^{asy}$, is described by a two-parameter logistic function, which is the easiest mathematical expression to describe the size-selectivity of fishing gear (Millar and Fryer 1999),

**(4)** $$S_{y,l}^{asy} = \frac{1}{1+e^{-\beta(l-L_{50})}} \ ,$$

where $L_{50}$ is the length at which capture probability is 50% of the maximum $U_y'$ and $\beta$ is the steepness of the function at $L_{50}$. At sizes $l \gg L_{50}$, relative vulnerability approaches a constant maximum value of one indicating that large fish are equally vulnerable to harvest regardless of size.

In some cases, selectivity may decline as fish approach very large sizes due to either behavioural avoidance of the gear, natural factors such as spawning or migration that cause fish to leave the exploited areas (Özbilgin and Wardle 2002), or economic factors that discourage vessels from reporting large tagged fish (Haist et al. 2001). For instance, visual inspection of tag recoveries by length class indicated potential dome-shaped selectivity for trap gear because the ratio of recaptures to releases declined with increasing size-at-release. Therefore, I tested whether a more complex model improved the fit to the observed data for the longline and trap fisheries. Specifically, I developed the following four-parameter "dome-shaped" function, $S_{y,l}^{dom}$, that allowed for a reduction in selectivity for very large size classes,

**(5)** $$S_{y,l}^{dom} = \left[\frac{1}{1+e^{-\beta_1(l-L_{50})}}\right]\left[1-\frac{1}{1+e^{-\beta_2(l-(L_{50}+\beta_3))}}\right] ,$$

where $L_{50}$ represents the length at 50% selection as before, $\beta_1$ is the slope at $L_{50}$, $\beta_2$ is the slope at the 50% de-selection length, $L_{50}+\beta_3$. I modelled de-selection using the offset parameter $\beta_3 > 0$ so that the length at 50% de-selection always exceeded $L_{50}$.

For the trawl fishery, exploratory analyses suggested that the trawl selectivity pattern would be decreasing for all sizes so an exponential model was fitted to the trawl tag recapture data instead. The exponential model, $S_{y,l}^{\exp}$, is described by the following exponential function,

**(6)** $$S_{y,l}^{\exp} = \exp[-\beta(l-50)] \ .$$

where $\beta$ is a shape parameter and 50 is a fixed exponential model parameter.

Because fish tagged and released in mark-recapture experiments independently experience (by assumption here) one of two fates upon their release (i.e., they are either captured or they evade capture), the likelihood function of the reported catch, $C_{y,l}$, is binomial (McCullagh and Nelder 1989; Crawley 2007):

**(7)** $$C_{y,l} \big| \theta \sim Bin(\pi_{y,l}(\theta), N_{y,l}) \ ,$$

where the vector $\theta_{asym} = \{\beta_1, L_{50}, U_1, U_2, \ldots U_Y\}$ represents the parameters for the asymptotic model, $\theta_{dome} = \{\beta_1, \beta_2, \beta_3, L_{50}, D_{50} = L_{50} + \beta_3, U_1, U_2, \ldots U_Y\}$ for the dome-shaped model, and $\theta_{exp} = \{\beta, U_1, U_2, \ldots U_Y\}$ for the exponential model.

Each candidate model was fitted to the observed tag recovery data using a maximum likelihood estimation procedure (MLE; Hilborn and Mangel 1997). I used a simulation approach to characterize uncertainty in selectivity function parameter estimates. In particular, I used Markov Chain Monte Carlo (MCMC) with a Metropolis-Hastings algorithm to approximate the marginal probability distribution for each selectivity function parameter. MCMC methods are commonly used in cases where it is not possible, or as in my case, where it is not convenient, to derive analytical expressions for the posterior distributions of parameters. For each selectivity model and tag release-recovery experiment, I generated 5000 sample points from the joint posterior distribution of model parameters and then "thinned" the resulting sample by choosing every 10 values for inclusion in the final sample. This thinning step was performed to reduce the effects of autocorrelation within the MCMC chain. I summarized the resulting marginal posterior distributions for management parameters of interest using the 2.5$^{th}$, 50$^{th}$, and 97.5$^{th}$ percentiles and posterior standard deviations.

### Model checking

Identifying systematic and isolated discrepancies of the data from the fitted values is an important part of assessing the adequacy of a model for a particular data set. Deviance residuals, $r_{y,l}$, used in model checking were calculated as,

**(8)** $$r_{y,l} = \text{sgn}(O_{y,l} - \pi_{y,l})\sqrt{d_{y,l}} \ ,$$

where

$$d_{y,l} = 2\sum_{l=1}^{L}(O_{y,l}\log(O_{y,l}/E[C_{y,l}]) + (N_{y,l} - O_{y,l})\log\left(\frac{N_{y,l} - O_{y,l}}{N_{y,l} - E[C_{y,l}]}\right) \ ,$$

$O_{y,l}$ is the observed recoveries of tagged fish in each length class $l$, $E[C_{y,l}]$ is the expected catch of a tagged fish, and $N_{y,l}$ is the number of fish tagged and released in each length class in a given year. To inspect model fits, deviance residuals were plotted against the length of tagged sablefish at release for each year.

### Model selection

Information theoretic criteria are typically used to choose between models of differing complexity (e.g., Burnham and Anderson 2002). In these approaches, models are assessed by a balance of their fit to the data and the complexity of the model, which is usually measured as the number of parameters needed to obtain the fit. The most well known example of this approach is Akaike's information criterion (AIC; Akaike 1969), which proposes that the model with the fewest number of parameters is the optimal model. AIC is considered more appropriate than likelihood ratio tests because AIC takes into account both the value of the likelihood as well as the number of parameters in the model (Burnham and Anderson 2002).

In light of the small number of tag recoveries, a small-sample AIC was calculated for each candidate model based on the maximum log-likelihood value, $\ell$, in each year $y$ for each gear type $g$, the number of parameters to be estimated $K$, and $n_{l,y}$ the number of tagged sablefish released in each length class $l$ in each year,

**(9)** $$AIC_y = -2\ell_y + 2K + \frac{2K(K+1)}{n_{l,y} - K - 1} ,$$

The preferred model, given the candidate models considered, was the one that produced the smallest AIC value. Since AIC values cannot be used to compare models of different data sets (i.e., across years; Burnham and Anderson 2002), an optimal model was identified for each gear type within a given year.

## Results

Sparse tag recoveries for certain year and gear combinations (Table 2) resulted in some years falling short of the minimum data requirements and thus their exclusion from the analysis. For example, in January 2002 a coast-wide closure of the B.C. sablefish fishery was imposed mid-way through the 2001/2002 fishing year (DFO 2003) and thus, all gear types recovered significantly fewer sablefish in 2002. Low tag recoveries by the longline fishery in 1995, 2001, 2003, and 2004 also resulted in these years being excluded from the analysis. In the trawl fishery, the relatively low number of tags reported in 1998, 2000, 2002 – 2004 precluded robust estimates of selectivity in these years.

The asymptotic model provided the best fit to the observed tag recovery data for the longline fishery in all years considered (Figure 3). Deviance residuals indicated no outliers or isolated departures from the model (Figure 4). Longline selectivity patterns increased rapidly between 50 cm and 65 cm and slowly at lengths larger than 70 cm. Estimates of the length at 50% selection were highest for the longline fishery out of all gear types considered with the average $L_{50}$ = 58 cm (Table 3). Marginal posterior distribution summaries for $L_{50}$ are comparable to maximum likelihood estimates. Table 3 shows percentiles for $L_{50}$, a parameter of management interest, and posterior standard deviations for the remaining parameters ($U_y^{'}$ and $\beta$).

In contrast, a dome-shaped model in which selectivity decreased at fork lengths greater than 65 cm provided the best fit to the commercial trap fishery in the majority of years considered (Figure 5; Figure 6). Under a dome-shaped model, selectivity increased with length up to between 60 cm and 65 cm before decreasing, with the size at 50% de-selection occurring, on average, at 72 cm (Table 4). Estimated values of $L_{50}$ were slightly smaller for the commercial trap fishery relative to the longline fishery with mean $L_{50}$ = 53 cm. The exceptions to this pattern were 1999 and 2004, in which an asymptotic model provided the best fit to the observed trap tag recovery data. The trap fishery was the only case in which the preferred selectivity model changed with time. MCMC parameter estimates did not converge for the dome-shaped model and are therefore not shown in Table 4.

An exponential model provided the best fit to tag recovery data for the trawl fishery across all years (Figure 7). Deviance residuals indicate a good fit to the exponential model (Figure 8). However, in contrast to the trap and longline fisheries, the highest vulnerabilities in the trawl fishery were observed in the smallest length classes with the average $L_{50}$ = 24 cm (Table 5). In all years, trawl selectivity declined after approximately 60 cm, indicating the decreased vulnerability of larger sablefish to capture by trawl gear.

# Discussion

## Gear selectivity

Gear selectivity is of fundamental importance to fisheries stock assessment and management. Identifying the size-selectivity of commercial fishing gears, and how selectivity parameters change over time, allows fishery managers to assess the impacts of commercial fishing on an exploited fish stock and develop meaningful gear regulations for a fleet. Using direct estimates of gear selectivity from tagging experiments, I demonstrated differences in size-selectivity for B.C. sablefish among gear types as evidenced by the shape of the selectivity function and the length at 50% selection.

Gear selectivity patterns predicted for the longline fishery were similar to those observed in Alaska and Washington State, where asymptotic selectivity is commonly used to represent longline selectivity (Hanselman et al. 2005). In the trawl fishery, the pattern of selectivity was hyperbolic over the range of size classes observed with an exponential selectivity model providing the best fit to the tag recovery data. In contrast to the trap and longline fisheries, trawl selectivity declined dramatically for larger sablefish (> 60 cm), demonstrating the reduced vulnerability of large sablefish to towed gears. While estimates of $L_{50}$ are slightly higher in the Alaskan trawl fishery ($L_{50}$ = 40 cm; Hanselman et al. 2005), a similar decline in the vulnerability of large sablefish to trawl capture is also apparent in the Alaskan fishery (Hanselman et al. 2005). In Alaska, the reduced vulnerability of large sablefish to trawl gear is attributed, in part, to the operation of the trawl fleet in shallower waters where young sablefish reside (Hanselman et al. 2005). In B.C., the spatial distribution of the fleet and the depth distribution of the gear are likely to exhibit a similar influence on the size distribution of the catch. The B.C.

trawl fishery tends to operate in shallow waters between 100 m and 200 m (DFO 2005), while the trap and longline fisheries generally operate in depths greater than 200 m. The spatial and depth distribution of the fleet is therefore likely to account for some of the variation in size-selectivity observed among gear types (Haist et al. 2001; Clark and Kaimmer 2006; Jacobson et al. 2001).

A similar decline in selectivity of large sablefish was also observed in the trap fishery in seven out of the nine years considered. A number of fishery-related mechanisms potentially explain the decline in selectivity for larger fish including spatial and depth factors (mentioned above), targeting behaviours, market considerations, avoidance behaviours, and fish migration. Fishing effort tends to be concentrated in areas where fish of commercial sizes are abundant and economic return to effort can be maximized (Paloheimo and Dickie 1965; Caddy 1975; Hilborn and Walters 1992; Chen et al. 1998). Through the selection of location and depth, fishermen can vary their fishing practices in order to target certain species and sizes of fish (Caddy 1975; Hilborn and Walters 1992). In some fisheries, fishermen have been shown to vary their fishing practices by location and depth in order to track the migration of larger fish to deeper waters (Paloheimo and Dickie 1965; Caddy 1975; Chen et al. 1998; Walters and Bonfil 1999; Jacobson et al. 2001). In the case of herring in the North Pacific, walleye in Lake Erie, and cod stocks in the Northeast Atlantic, the persistent depletion of the oldest and largest fish in the stock led to an increase in the targeting of smaller and smaller fish (Myers and Mertz 1998). Such changes in targeting behaviour in order to harvest a constant quota from a declining stock have since been identified as a key factor in the collapse of these exploited fish stocks (Myers and Mertz 1998; Myers and Quinn 2002).

In the B.C. sablefish fishery, it is possible that the trap fishery is targeting areas and depths inhabited by more abundant intermediate size-classes in order to maximize catch rates. While similar targeting behaviours may also be present in the trawl and longline fisheries, targeting is less likely in these fisheries because these gear types generally harvest sablefish in conjunction with other groundfish species, which reduces the chances that a single size class will be targeted within a given trip. A price premium for larger sablefish may also encourage fishermen to misreport catches of larger tagged sablefish resulting in an apparent 'decline' in selectivity for larger sablefish (Haist et al. 2001). Although tag reporting was assumed to be independent of the length of a fish, an evaluation of tag reporting compliance would greatly improve interpretation of model results.

A rapidly descending right hand limb in the dome-shaped selectivity function could also represent fish that actively avoid trap or trawl gear. Previously captured fish may become 'gear-shy' and actively avoid the gear. Furthermore, spawning events or migration may also place large sablefish in unfished areas (Özbilgin and Wardle 2002). Unfortunately, disentangling the reasons for the observed decline in selectivity at length in the trap and trawl fisheries trap fishery remains a difficult task.

### Consequences of observed selectivity patterns

When all size-classes are equally vulnerable to the fishery, annual harvesting can lead to growth overfishing, a process in which fish are removed from the population while they are still growing rapidly (Walters and Martell 2004). In such cases, heavy fishing mortality on pre-recruits can lead to a waste of potential biomass by taking fish that would otherwise generate greater yields if they were allowed to grow larger prior to becoming vulnerable to the fishery (Armstrong et al.1990; Walters and Martell 2004).

My analysis found that 50% of the population became vulnerable to the trawl fishery at 24 cm. Female sablefish generally reach sexual maturity at approximately 61 cm fork length (Love 1996) causing a high proportion of immature female sablefish to be recruited to the trawl fishery. Current management regulations prohibit the retention of sablefish less than 55 cm fork length (DFO 2007) resulting in a large number of undersize sablefish being discarded at-sea. The mortality of discarded sablefish is highly variable and dependent on a suite of physical, biological, and environmental factors (Davis 2002; Davis et al. 2002). Research suggests, however, that smaller fish experience more behavioural impairments and higher mortality rates than larger fish (Neilson et al. 1989; Richards et al. 1995; Milliken et al. 1999; Parker et al. 2003; Davis and Parker 2004). If the incidental mortality of pre-recruits is high, discarding can lead to significant biological and economic losses for valuable commercial fisheries (Chen and Gordon 1997). And, as I show in Chapter 3, ignoring the mortality of discarded sablefish in stock assessment models can bias estimates of total fishing mortality and lead to substantial losses in fishery yield and revenue (Chen and Gordon 1997).

### Evolutionary implications

In many commercially exploited fish stocks, the mortality caused by fishing exceeds the mortality caused by natural factors (Pitcher and Hart 1982; Law and Grey 1989). Fishing, therefore, can influence the genetic composition of exploited fish stocks and induce changes in life history traits, such as age- or size-at maturity (Law and Grey 1989; Stokes et al. 1993; Heino 1998; Palumbi 2001; Ernande et al. 2004). For example, escape rings introduced in the commercial trap fishery in 1999 has resulted in more female sablefish being retained than males (females are generally larger at a given age relative to male sablefish; Haist et al. 2001). Preferentially selecting for the largest, and hence the most fecund fish in the population, can affect the total reproductive potential of the stock (Ernande et al. 2004). In the trawl fishery, the tendency to select for small, immature sablefish can also induce adaptive changes in sablefish life history parameters (Law and Grey 1989; Botsford et al. 1997; Heino 1998; Law 2007). Selecting for smaller fish may cause sablefish to mature at smaller sizes and younger ages in order to reach sexual maturity before becoming vulnerable to the fishery. Such changes in age- and size-at-maturation can have strong repercussions for population dynamics and sustainable harvesting (Ernande et al. 2004).

### Limitations and extensions

One of the foremost limitations of this analysis was the sparseness of data in certain years for certain size classes. Large data gaps precluded robust estimates of size-selectivity functions for some gear types (e.g., the trawl fishery) resulting in the exclusion of certain years from the analysis and the elimination of potentially informative data. A Bayesian approach (Rivot and Prévost 2002) would likely improve estimates of selectivity function parameters by providing a natural method for combining estimates from multiple tagging experiments. Such Bayesian hierarchical approaches have been used in the past to improve estimates of stock-recruitment parameters (Liermann and Hilborn 1997; Michielsens and McAllister 2004), gillnet efficiency (Harley and Myers 2001), and in mark-recapture analysis (Rivot and Prévost 2002). By borrowing tag recovery information from other years (or even other studies), a Bayesian approach could help to prevent unrealistic selectivity functions in certain years where sparse tag recoveries occur (Gazey and Staley 1986).

Another useful extension of this analysis would involve an examination of tag recoveries by the annual sablefish research and assessment survey to determine whether the observed decline in selectivity in the commercial trap fishery also occurs for fishery-independent recoveries. The B.C. sablefish assessment survey uses trap gear consistent with those employed in the commercial trap fishery (except escape rings are sewn shut in the sablefish survey; Wyeth and Kronlund 2003). A comparison of estimated selectivity patterns between the commercial trap fishery and the sablefish assessment survey would therefore be a worthwhile and potentially informative undertaking. If a similar decline in selectivity at larger sizes is evident in the assessment survey, then sparse tag recoveries or reporting biases are likely not responsible for the observed decline in selectivity at larger lengths I observed in the commercial trap fishery. While an analysis of survey tag recovery data was attempted in this study, robust estimates of survey selectivity were not possible given the short duration of the assessment survey (October – November) and the low number of survey tag recaptures.

Independent estimates of tag reporting rates and the quantification of tag loss, tagging mortality, and natural mortality would also assist in the interpretation of model results and the inferences drawn from my post-hoc analysis of tagging data. Blind tagging programs, in which fishermen are unable to tell the difference between marked and unmarked fish, could be used to obtain independent estimates of tag reporting rates (Martell and Walters 2002). Additional methods to improve estimates of tag reporting rates include high-reward tags (Pollock et al. 2001; Taylor et al. 2006), fishery observers

(Pollock et al. 2002), and planted tags (Hearn et al. 2003). Recent technological innovations, such as Passive Integrated Transponder (PIT) tags (e.g., Pengilly and Watson 1994), coded-wire tags (Jefferts et al. 1963), and genetic tagging methods (Palsboll 1999) are also possible.

## Conclusion

Mark-recapture experiments provide an effective means of obtaining direct and reliable estimates of gear selectivity. Knowledge of gear selectivity is critical to the success of fishery regulations especially where gear regulations, such as minimum mesh sizes in trawl and seines, are designed to minimize the capture of small fish and reduce the losses associated with discard mortality. Analysis of tagging data can also yield other useful information on exploited fish stocks including estimates of total mortality (Brownie et al. 1985), population size (Haist and Hilborn 2000), and movement rates (Myers and Hoenig 1997). Unfortunately, post-hoc analyses of tagging databases are rarely conducted and valuable information on the fishery is often overlooked (Myers and Hoenig 1997).

This study demonstrated differences in size-selectivity among gear types with the longline fishery selecting for the largest size-classes of sablefish, the trap fishery selecting for intermediate sizes, and the trawl fishery selecting for small sablefish below the minimum size limit. Size-selectivity functions were asymptotic for the longline fishery and exponential for the trawl fishery, whereas estimated selectivity functions for the trap fishery varied over time between asymptotic and dome-shaped. In light of the variation in trap selectivity observed with time, assumptions that fishery selectivity parameters are constant may be inappropriate in stock assessments for B.C. sablefish. Further consideration should therefore be given to changes in gear selectivity for all gear types in the ongoing management strategy evaluation for B.C. sablefish.

## Tables

**Table 1 Summary of the British Columbia sablefish tagging program between 1995 and 2004.**

| Location | Depth | Traps | Bait | Sablefish Catch Processing |
|---|---|---|---|---|
| 6 tagging localities | 457-824 m | ~ 60 | 1 kg squid plus 4.5 kg hake | all tagged until 1000 fish released in each locality |
| 9 indexing localities | 457-824 m | ~ 60 | 1 kg squid plus 4.5 kg hake | all tagged until 300 fish released in each locality |

Source: Wyeth and Kronlund (2003); Wyeth et al. 2004b.

**Table 2 Sablefish tag releases and total recoveries by gear type between 1995 and 2004.**

| | | Recoveries | | |
|---|---|---|---|---|
| Year | No. released | Longline | Trap | Trawl |
| 1995 | 9302 | 68 | 653 | 9 |
| 1996 | 6432 | 154 | 973 | 70 |
| 1997 | 13904 | 132 | 1966 | 60 |
| 1998 | 16051 | 119 | 816 | 91 |
| 1999 | 17532 | 324 | 818 | 92 |
| 2000 | 13618 | 251 | 789 | 124 |
| 2001 | 5900 | 128 | 826 | 128 |
| 2002 | 8798 | 31 | 106 | 58 |
| 2003 | 8596 | 88 | 270 | 34 |
| 2004 | 7260 | 64 | 448 | 27 |

**Table 3 Parameter estimates and AIC values for asymptotic model fits to trap and longline tag release and recovery data. Values in parentheses are the posterior standard deviations for the maximum likelihood parameter estimates.**

| | | | $L_{50}$ | | | | | |
|---|---|---|---|---|---|---|---|---|
| Year | Gear | $U_y'$ | MLE | 0.025 | 0.50 | 0.975 | $\beta$ | AIC |
| 1996 | L | 0.014(0.002) | 63 | 57 | 67 | 93 | 0.15(0.04) | 2139 |
| 1997 | L | 0.012(0.002) | 56 | 50 | 55 | 62 | 0.30(0.15) | 1827 |
| 1998 | L | 0.017(0.004) | 60 | 56 | 62 | 81 | 0.18(0.06) | 1537 |
| 1999 | L | 0.030(0.003) | 54 | 51 | 56 | 66 | 0.20(0.07) | 4597 |
| 2000 | L | 0.024(0.002) | 55 | 53 | 55 | 58 | 0.33(0.08) | 3595 |
| 1995 | Tr | 0.086(0.09) | 51 | 50 | 51 | 52 | 0.40(0.09) | 8476 |
| 1996 | Tr | 0.050(0.003) | 50 | 28 | 47 | 50 | 2.4(2.25) | 14155 |
| 1997 | Tr | 0.140(0.002) | 49 | 42 | 47 | 50 | 2.3(2.91) | 27608 |
| 1998 | Tr | 0.082(0.003) | 49 | 51 | 53 | 54 | 0.22(0.03) | 11158 |
| 1999 | Tr | 0.110(0.005) | 53 | 51 | 54 | 55 | 0.23(0.04) | 11344 |
| 2000 | Tr | 0.073(0.007) | 60 | 58 | 60 | 62 | 0.29(0.03) | 11209 |
| 2001 | Tr | 0.092(0.004) | 55 | 54 | 55 | 57 | 0.29(0.04) | 11535 |
| 2003 | Tr | 0.062(0.005) | 53 | 52 | 53 | 55 | 0.28(0.04) | 3489 |
| 2004 | Tr | 0.110(0.005) | 58 | 56 | 58 | 62 | 0.28(0.05) | 5797 |

Note: L = Longline, Tr = Trap, MLE = maximum likelihood estimate

**Table 4 Parameter estimates and AIC values for dome-shaped model fits to trap and longline tag release and recovery data.**

| | | | $L_{50}$ | | | | | | $\beta_3$ | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Year | Gear | $U'_y$ | MLE | 0.025 | 0.50 | 0.975 | $\beta_1$ | $\beta_2$ | MLE | 0.025 | 0.50 | 0.975 | AIC |
| 1996 | L | 0.01 | 63 | n.c. | n.c. | n.c. | 0.15 | 0.003 | 9.32 | n.c. | n.c. | n.c. | 2142 |
| 1997 | L | 0.01 | 58 | n.c. | n.c. | n.c. | 0.24 | 0.06 | 18.2 | n.c. | n.c. | n.c. | 1829 |
| 1998 | L | 0.02 | 66 | n.c. | n.c. | n.c. | 0.14 | 0.40 | 9.64 | n.c. | n.c. | n.c. | 1538 |
| 1999 | L | 0.03 | 54 | n.c. | n.c. | n.c. | 0.20 | 0.0001 | 483 | n.c. | n.c. | n.c. | 4599 |
| 2000 | L | 0.02 | 58 | n.c. | n.c. | n.c. | 0.23 | 0.38 | 12.3 | n.c. | n.c. | n.c. | 3589 |
| 1995 | Tp | 0.09 | 53 | n.c. | n.c. | n.c. | 0.25 | 0.20 | 20.3 | n.c. | n.c. | n.c. | 8462 |
| 1996 | Tp | 0.05 | 49 | n.c. | n.c. | n.c. | 0.60 | 0.11 | 29.0 | n.c. | n.c. | n.c. | 14144 |
| 1997 | Tp | 0.15 | 48 | n.c. | n.c. | n.c. | 0.23 | 0.10 | 24.2 | n.c. | n.c. | n.c. | 27579 |
| 1998 | Tp | 0.09 | 66 | n.c. | n.c. | n.c. | 0.11 | 0.31 | 5.46 | n.c. | n.c. | n.c. | 11134 |
| 1999 | Tp | 0.11 | 60 | n.c. | n.c. | n.c. | 0.23 | 0.01 | 1748 | n.c. | n.c. | n.c. | 11346 |
| 2000 | Tp | 0.08 | 60 | n.c. | n.c. | n.c. | 0.20 | 0.33 | 12.6 | n.c. | n.c. | n.c. | 11188 |
| 2001 | Tp | 0.09 | 58 | n.c. | n.c. | n.c. | 0.21 | 0.14 | 11.0 | n.c. | n.c. | n.c. | 11536 |
| 2003 | Tp | 0.06 | 64 | n.c. | n.c. | n.c. | 0.22 | 0.23 | 6.01 | n.c. | n.c. | n.c. | 3489 |
| 2004 | Tp | 0.10 | 62 | n.c. | n.c. | n.c. | 0.23 | 0.28 | 10.7 | n.c. | n.c. | n.c. | 5798 |

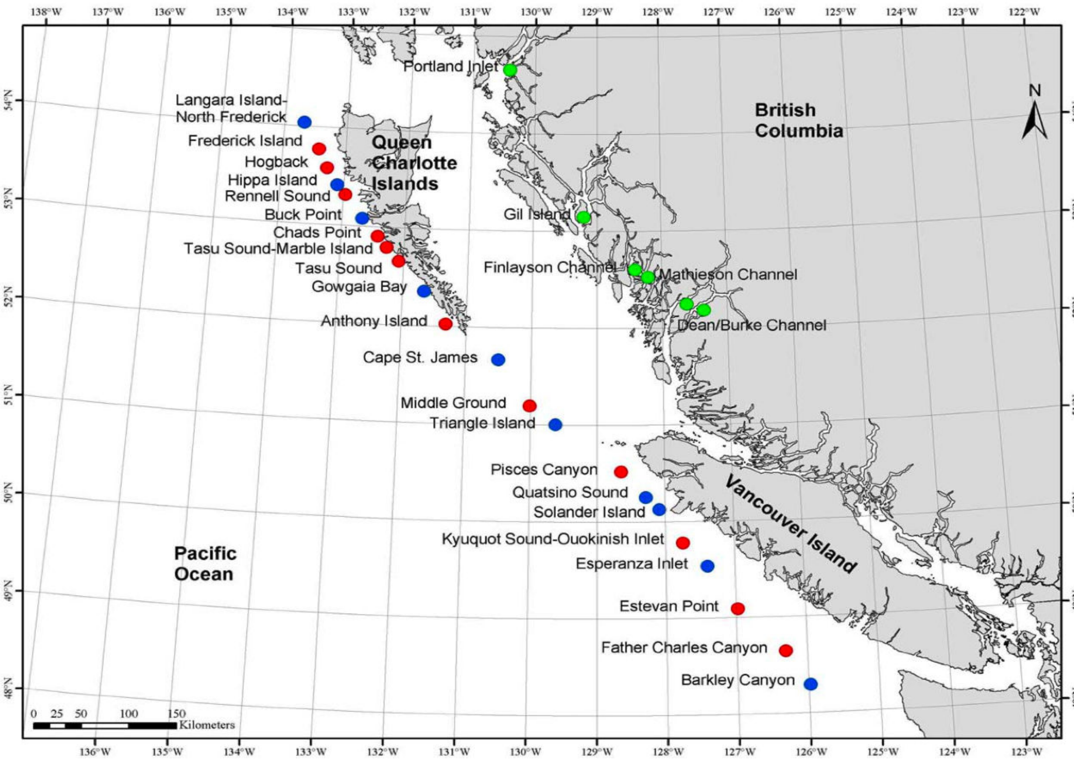
Note: L = longline, Tp = Trap, Tr = Trawl, n.c. = MCMC parameter estimates did not converge

**Table 5 Parameter estimates and AIC values for exponential model fits to trawl tag release and recovery data. Values in parentheses are the posterior standard deviations for the maximum likelihood parameter estimates.**

| Year | Gear | $U^{'}(sd)$ | $\beta\ (sd)$ |
|---|---|---|---|
| 1997 | Trawl | 0.107(0.03) | 0.010(0.002) |
| 1999 | Trawl | 0.07(0.03) | 0.010(0.002) |
| 2001 | Trawl | 0.085(0.02) | 0.020(0.002) |

## Figures

**Figure 1 Location of sablefish tag releases between 1995 and 2004.**




Source: Wyeth and Kronlund (2003)

**Figure 2 Length frequencies of sablefish tag releases between 1995 and 2004. Length classes denote the length at release and '*y*' and '*N*' correspond to the year of release and the total number of tagged sablefish released, respectively.**

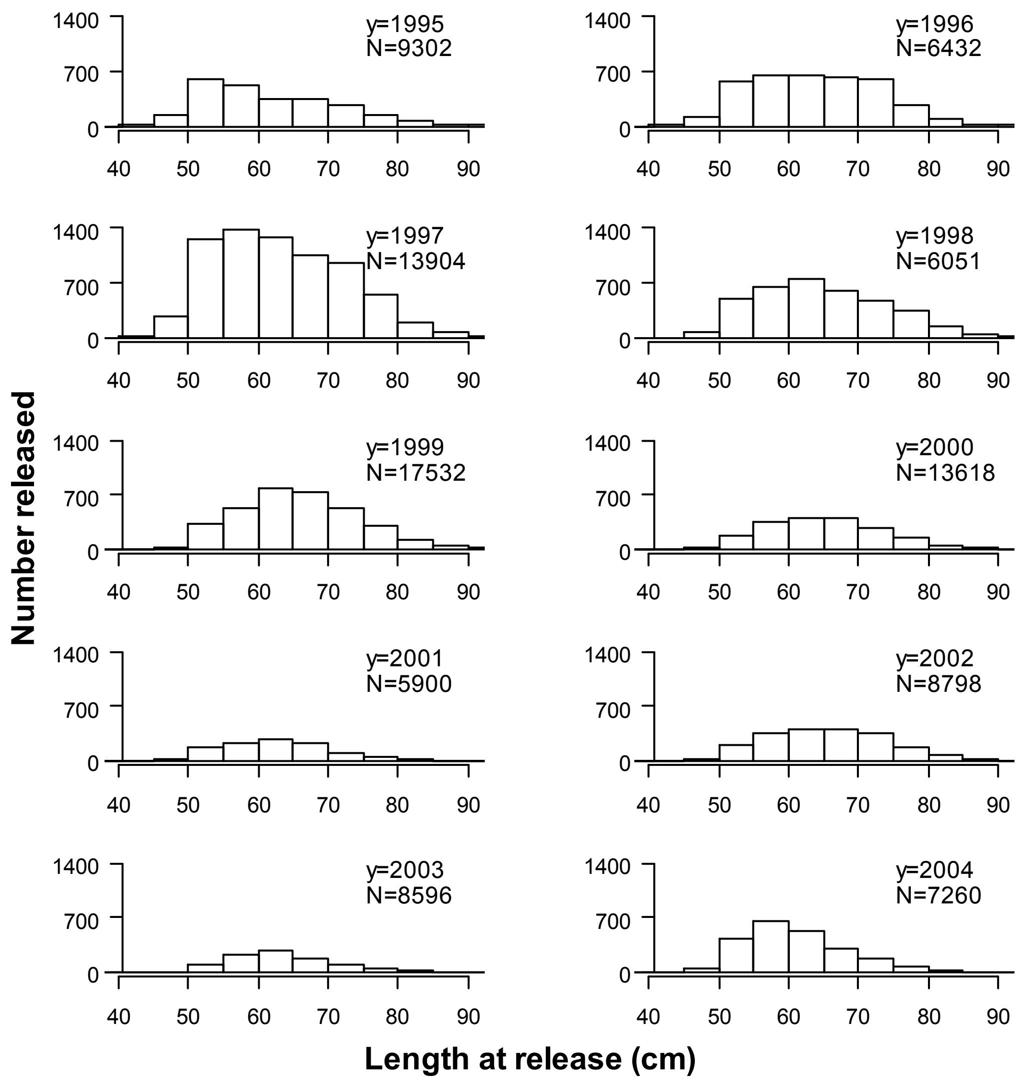


**Figure 3** **Relative selectivity-at-length for the commercial longline fishery. The solid lines represent asymptotic model fits and the solid circles represent the observed relative vulnerabilities for a given length class.**

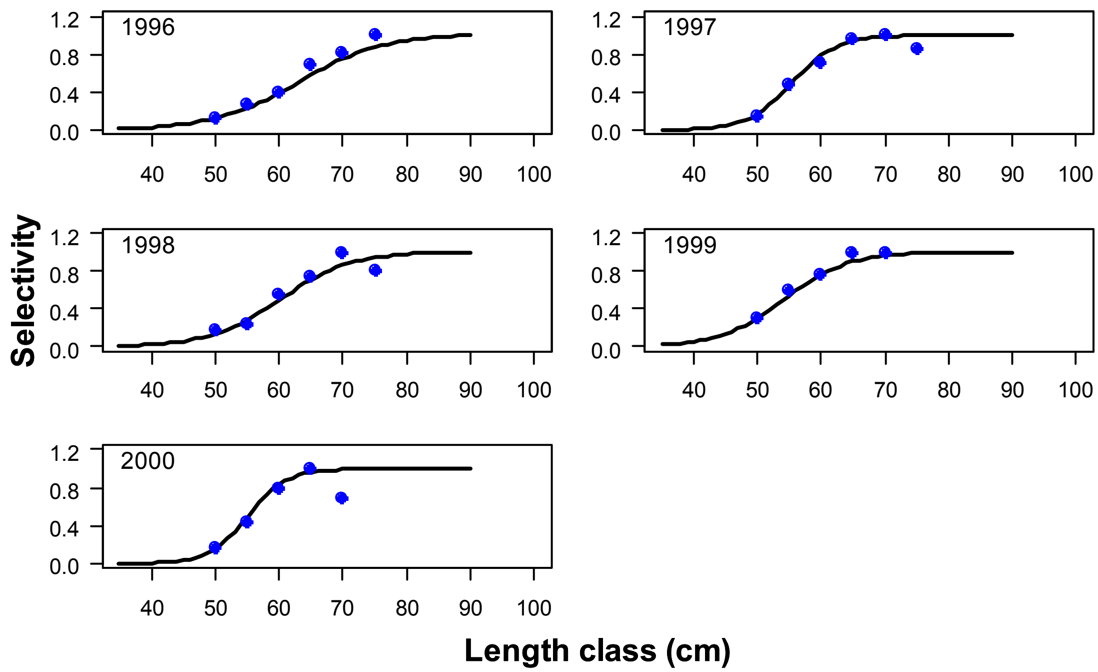



**Figure 4** **Deviance residuals by length for asymptotic model fits for the commercial longline fishery.**

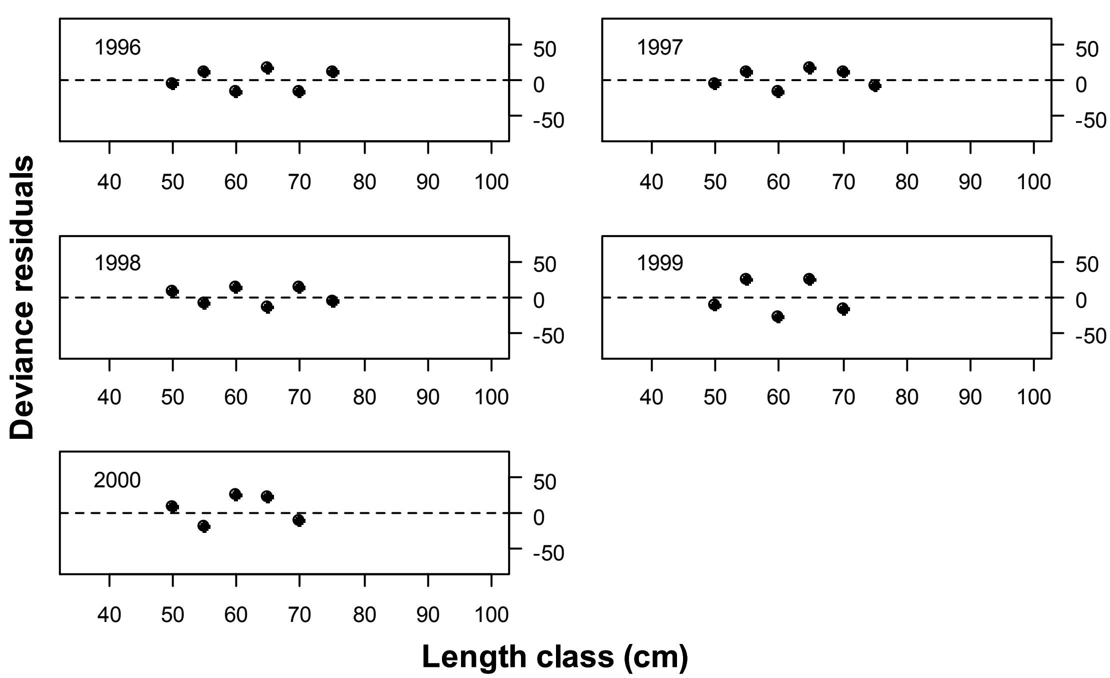


**Figure 5 Relative selectivity-at-length for the commercial trap fishery. The solid lines represent dome-shaped model fits and the solid circles represent the observed relative vulnerabilities for a given length class.**

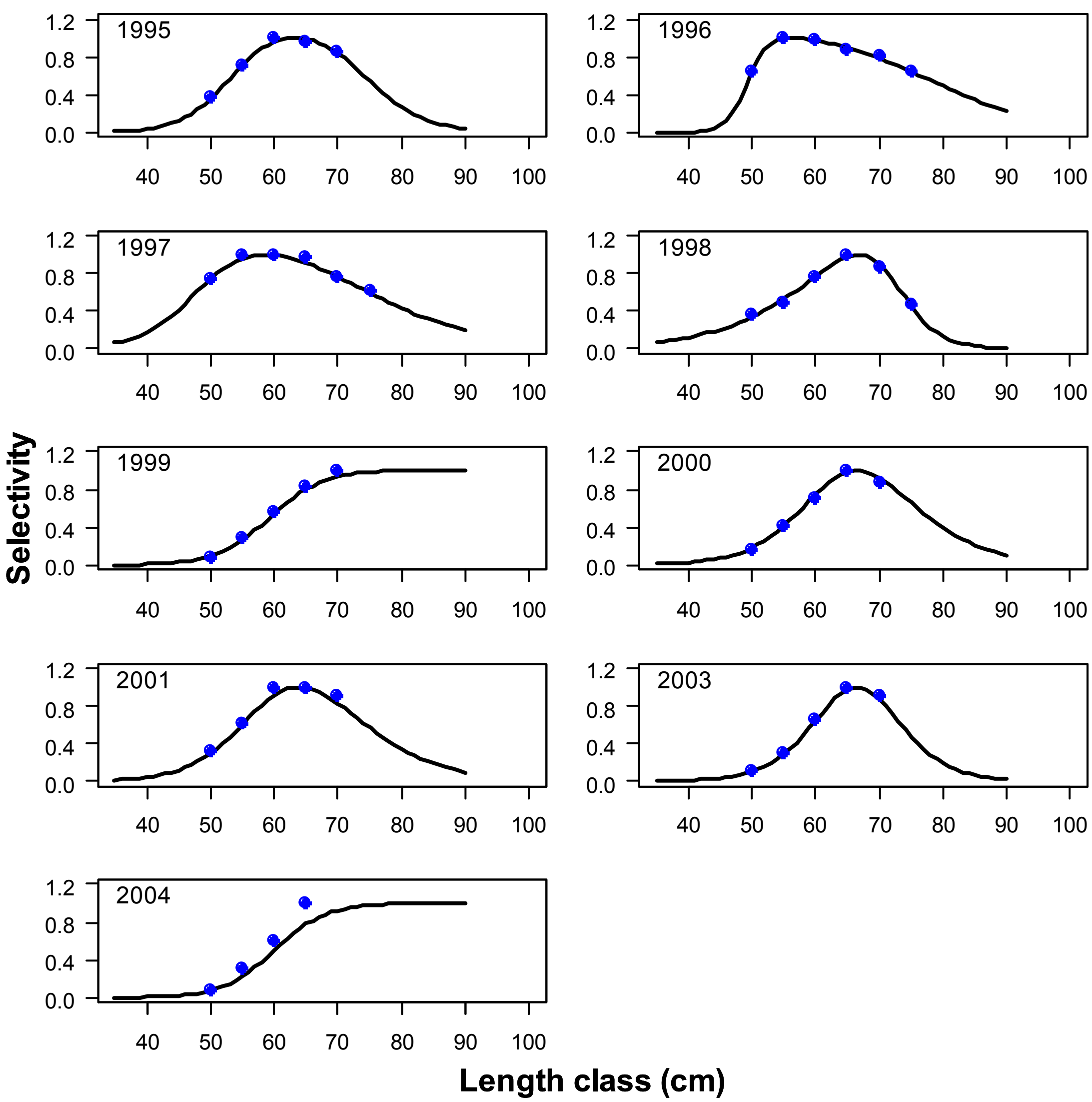


**Figure 6 Deviance residuals by length for dome-shaped model fits for the commercial trap fishery.**

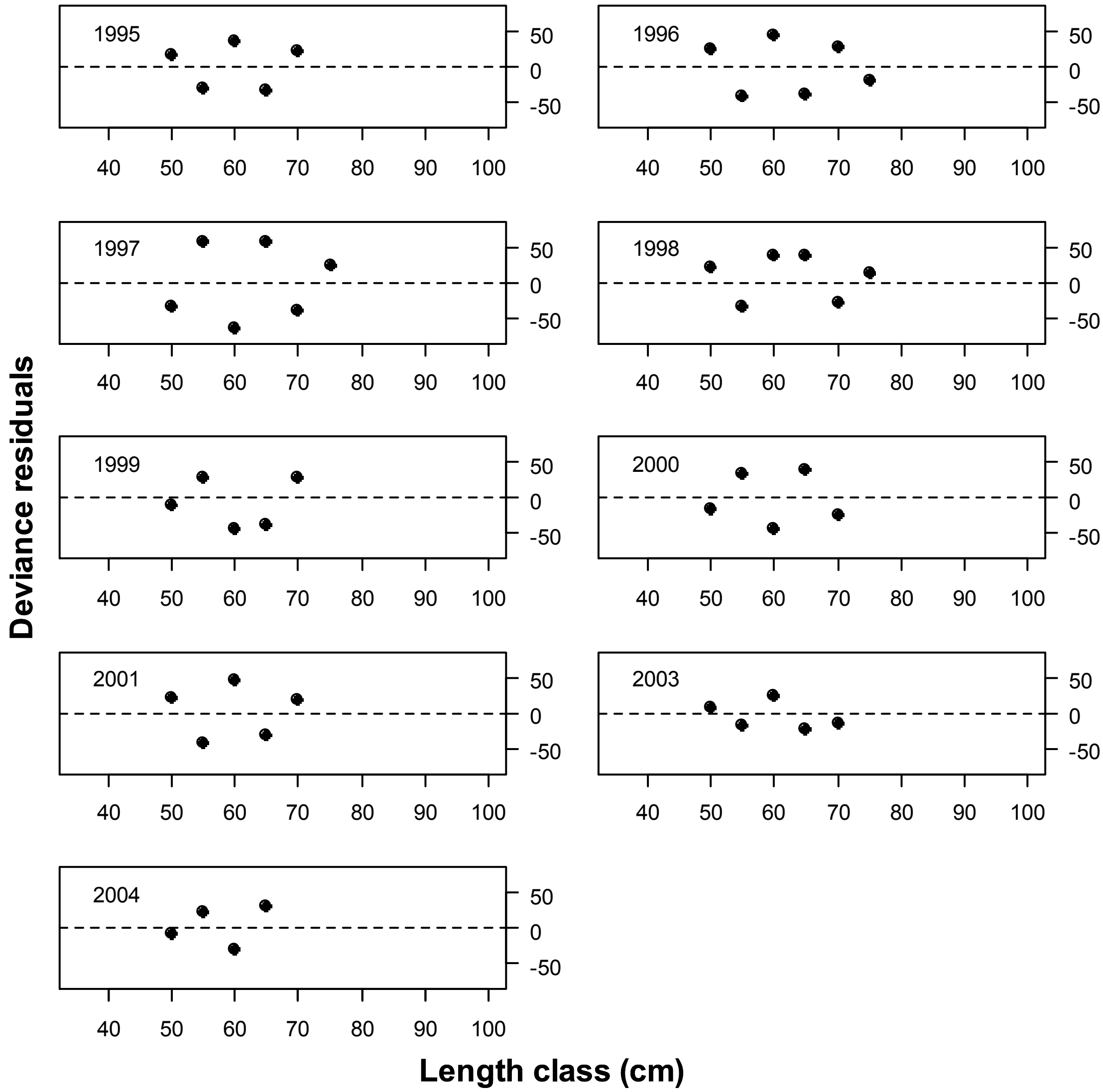


**Figure 7 Relative selectivity-at-length for the commercial trawl fishery. The solid lines represent exponential model fits and the solid circles represent the observed relative vulnerabilities for a given length class.**

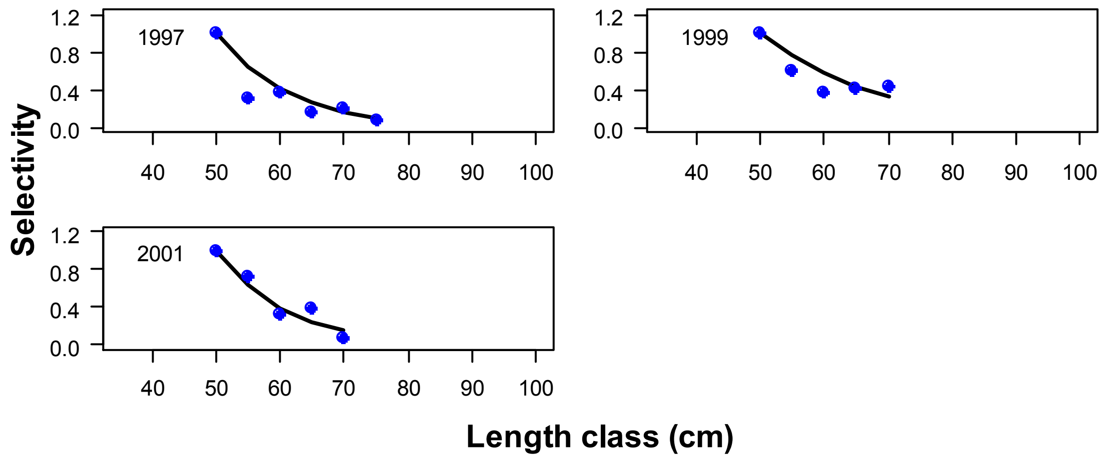
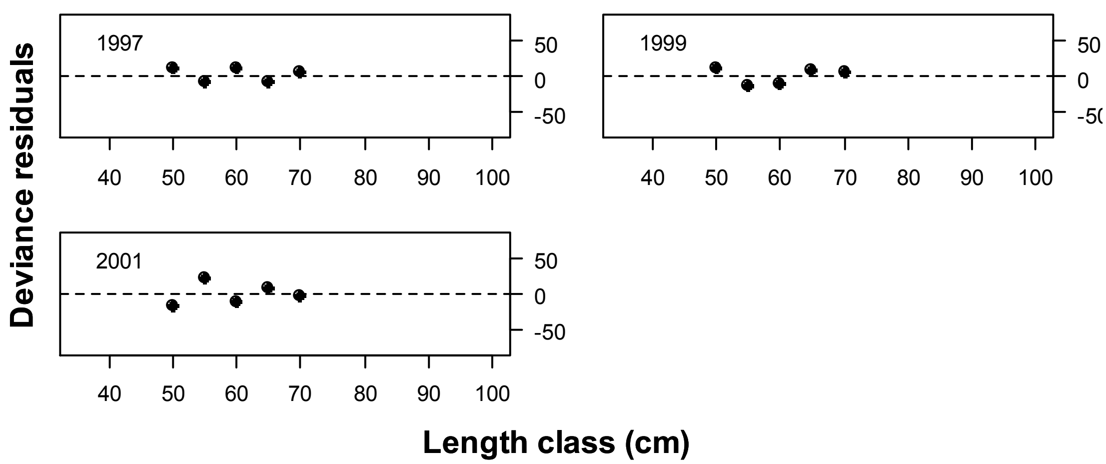



**Figure 8 Deviance residuals by length for exponential model fits for the commercial trawl fishery.**



# CHAPTER 3
# BIOLOGICAL AND ECONOMIC IMPACTS OF DISCARDING ON B.C.'S SABLEFISH FISHERIES


## Abstract

In many fisheries, a portion of the total catch is discarded at-sea because of legal requirements, market preferences, or economic considerations. High mortality of the discarded catch can lead to substantial losses in fishery yield and value for commercial fisheries. I examined the impacts of at-sea discarding for the B.C. sablefish (*Anoplopoma fimbria*) fishery using empirical estimates of gear selectivity. Yield-per-recruit (YPR) and spawner biomass-per-recruit (SBPR) calculations were used to evaluate economic and biological losses for three discard mortality scenarios in which (*i*) all of the discarded catch survives, (*ii*) all of the discarded catch dies, but discards are still included in estimates of YPR and SBPR, or (*iii*) all of the discarded catch dies and is ignored in estimates of YPR and SBPR. Expected losses of YPR and SBPR resulting from at-sea discarding were substantial for the B.C. sablefish fishery. Forty-nine percent of the total potential YPR was wasted because of at-sea discarding, which equates to economic losses of CDN \$5.44/recruit. This analysis suggests that gains in fishery efficiency and value could be realized if discarding were reduced or eliminated in the fishery either through the removal of perverse market incentives, such as price premiums for large sablefish, or regulatory approaches that restrict harvesting in areas where young sablefish reside.

## Introduction

The incidental capture, or bycatch, of non-target fish is an ongoing problem in the management of modern commercial fisheries (Alverson and Hughes 1996; Alverson 1999; Hall et al. 2000; Machias et al. 2001; Kennelly and Broadhurst 2002). Bycatch primarily arises because fishing gears and practices are not perfectly selective for the species and sizes of fish being targeted and because target species co-exist in habitats occupied by a wide range of species (FAO 2008).

In many fisheries, the decision to discard is driven by economic factors (Alverson 1999; FAO 2008). In an unregulated fishery, for example, fishers have an incentive to discard if the expected net price is negative and if the resultant costs incurred by landing are greater than those incurred by discarding (FAO 2008). There is an additional incentive to discard if the vessel has a limited holding capacity or, as in the case of sablefish, a better price per pound is paid for sablefish > 60cm than for those at the legal-size limit of 55 cm (Haist et al. 2001). In such cases, fishers tend to discard the low-value components of the catch in order to retain fish of higher value, a practice known as "highgrading" (Gillis et al. 1995a; Gillis et al. 1995b; FAO 2008).

Discarding of bycatch has long been recognized as wasteful (Harrington et al. 2005), although inevitable by virtue of the nature of fishing. It constitutes a loss of valuable food, has consequences for the environment and biodiversity, and can be aesthetically offensive (Alverson and Hughes 1996; Alverson 1999; Hall et al. 2000; Kennelly and Broadhurst 2002). In 1994, the Food and Agriculture Organization (FAO) estimated that global discards from the world's fisheries amounted to between 17.9 million and 39.5 million metric tonnes (FAO 1994). A re-evaluation of these estimates, together with adjustments allowing for subsequent reductions in discarding, indicates that current levels of discarding are approximately 20 million tonnes, or 25 percent of the reported annual production from marine capture fisheries (Kelleher 2004). If the mortality of discarded sablefish is high, the exclusion of discards from fishery assessments can lead to biased abundance estimates and large losses in fishery yield and revenue for the B.C. sablefish fishery (Pikitch 1987; Chen and Gordon 1997; Rahikainen et al 2004; Helfman 2007). Persistently failing to monitor and account for the discarded catch in estimates of total fishing mortality can also mask potential declines in stock abundance (Myers et al. 1997; Rahikainen et al. 2004) and lead to the over-exploitation of sablefish resources (Myers et al. 2000).

Considerable research has focused on estimating discard mortality rates for a wide range of species harvested by a variety of gear types (e.g., Olla et al. 1997; Davis et al. 2001; Davis 2002; Davis and Olla 2002; Davis and Parker 2004). Some of these studies have attempted to measure discard mortality rates directly in the field by either holding fish for short time periods after capture or using mark-recapture methods (Davis 2002). However, the logistical constraints of measuring discard mortality rates in the field (Kaimmer and Trumble 1998), combined with the limited application of these studies to a wider range of physical, biological, and environmental conditions, limit the utility of many field studies (Davis 2002). Laboratory experiments allow scientists to study the effects of capture stressors on discarded fish under a controlled range of physical, biological, and environmental factors (Davis 2002). Laboratory studies on discarded sablefish suggest that discard mortality is highly variable ranging from 0% to 100% depending on the life-history stage, gear type, handling time, gear deployment method (soak and trawl times), temperature, season, and a host of other factors (Olla et al. 1997; Davis 2002). The importance of exposure to elevated temperature following capture has been demonstrated in a number of studies, all of which indicate that rapid increases in temperature can magnify physiological changes and mortality, particularly for juvenile fish which generally experience more behavioural impairments and higher mortality rates relative to larger fish (Neilson et al. 1989; Richards et al. 1995; Milliken et al. 1999; Davis et al. 2001; Parker et al. 2003; Davis and Parker 2004). Rapid increases in temperature can induce mortality directly or indirectly by diminishing the capability to deal with basic ecological challenges such as food acquisition and predator avoidance (Olla et al. 1980; Schreck et al. 1997). When these behavioural impediments are added to the stress induced by capture, temperature can exert a potent influence on survival and induce acute levels of stress and mortality beyond that associated with capture processes alone (Olla et al. 1998). For demersal species such as sablefish, rapid changes in temperature are common during the gear retrieval process, particularly in the Pacific northwest Ocean where sharp thermoclines are present (Hunter et al. 1989; Tully 1964; Huyer 1977). The effects of temperature may be even more acute in years when an El Niño is present and warmer sea water temperatures follow climatic shifts (Huyer and Smith 1985).

In this chapter, I develop length-based models of yield-per-recruit (YPR) and spawner biomass-per-recruit (SBPR) to explore the impacts of at-sea discarding on biological and economic yields from the B.C. sablefish (*Anoplopoma fimbria*) fishery. The objective of this chapter was to use empirical estimates of selectivity, calculated in Chapter 2, to quantify the potential bias in YPR and SBPR estimates that ignore the mortality of the discarded catch in calculations of fishery yield. Potential biases in yield estimates were quantified by calculating YPR and SBPR under the following discard mortality assumptions:

1. all of the discarded catch survives,
2. all of the discarded catch dies, but discards are still included in estimates of YPR and SBPR, or
3. all of the discarded catch dies and is ignored in estimates of YPR and SBPR.

Analytical YPR and SBPR models are particularly useful for exploring the potential effects of alternative fishery scenarios, particularly where knowledge is limited to basic growth and mortality rates (Beverton and Holt 1957; Ricker 1975). YPR models are generally used to estimate fishery yield while SBPR models are often used to develop long-term management strategies that attempt to maintain or rebuild the reproductive capacity of the stock (Goodyear 1989). While YPR and SBPR models have been used in previous sablefish assessments to estimate target fishing mortality rates and guide management decisions (e.g., Klein 1986; Hilborn et al. 2001), to my knowledge, YPR and SBPR analyses have not been previously used to explore the biological and economic impacts of at-sea discarding for the B.C. sablefish fishery.

## Methods

YPR and SBPR models incorporate the interplay between growth and survival to predict the lifetime yield from a cohort under different combinations of fishing mortality and selectivity (Punt 1992). Tracking three state variables, numbers-at-age, length-at-age, and weight-at-length through time for a single cohort, YPR and SBPR models describe the dynamics of a cohort during its lifespan in a fishery (Chen and Gordon 1997). Most YPR analyses use an age-structured model to track the size and numbers of a cohort over its lifetime. In an age-structured model, time is divided into equal discrete steps so that in early stages of growth fish of several different sizes are lumped together (Chen and Gordon 1997). In contrast, length-based models of YPR divide time into intervals spent in each length class. Because fishery processes such as discarding are more correlated with length than with age (Hilborn and Walters 1992), a length-based model was considered appropriate for an analysis of the effects of at-sea discarding.

### Model development

Sablefish lengths were divided into 1 cm length classes, $j$, between $L_1$ = 35 cm and $L_n$ = 80 cm, which represent the length at first recruitment to the fishery and the maximum length of sablefish that still contribute to the fishery, respectively. Following the method of Chen and Gordon (1997), the time, $\Delta T_j$, it takes a fish to grow through each length interval $d_j = L_{j+1} - L_j$ was calculated from the von Bertalanffy growth function (VBGF; Ricker 1975), i.e.,

**(1)** $$\Delta T_j = \frac{1}{K} \ln \frac{L_\infty - L_j}{L_\infty - L_j - d_j} \qquad 1 \le j \le n-1 \quad ,$$

where $L_\infty$ represents the maximum theoretical length that fish can reach and $K$ is the Brody growth coefficient that determines the rate at which $L_\infty$ is attained (Beverton and Holt 1957).

With the exception of gear selectivity, for which empirical estimates were derived in Chapter 2, the input data required for parameterization of the model, including parameters in the VBGF, length-weight relationships, and the natural mortality rate were obtained from the 2006 Alaska sablefish assessment (Table 6; Hanselman et al. 2005).

As a cohort ages, it declines in total abundance as a result of natural and fishing mortality, while at the same time increasing in individual mass. Weight-at-length, $W_j$, was calculated using the allometric length-weight relationship,

**(2)** $$W_j = aL_j^{\beta} \quad ,$$

where $a$ and $\beta$ are the fixed weight-length parameters.

Baranov's catch equation was used to calculate the catch of fish, $C_{j,g}$, in length class $j$ for each gear type $g$, i.e.,

**(3)** $$C_{j,g} = N_j \frac{S_{j,g} F_g (1 - P_{j,g})}{Z_j} \left[1 - e^{-Z_j}\right] \quad ,$$

where

$$Z_j = (F_{j,\cdot} + M)\Delta T_j \quad ,$$

$$F_{j,\cdot} = \sum_{g=1}^{G} S_{j,g} F_g (1 - P_{k,g}) \quad .$$

$S_{j,g}$ is a gear-specific selectivity schedule, $F_g$ is the full-selectivity fishing mortality rate for each gear type $g$, and $Z_j$ represents the total mortality rate. The total number of recruits $R$ surviving to length $L_j$ follows the exponential survival function (Ricker 1975),

**(4)** $$N_j = R \cdot \exp[-\sum_{k=1}^{j-1}(F_{k,\cdot} + M)\Delta T_k] \quad .$$

The proportion discarded, $P_{j,g}$ ,was modeled using the following two-parameter logistic function, 1

**(5)** $$P_{j,g} = 1 - \frac{1}{1 + e^{-Q(L_j - R_{50})}} \quad ,$$

where $R_{50}$ represents the size at which 50% of the catch is retained by the vessel and $Q$ is the slope of the function at $R_{50}$. The proportion of sablefish retained at fork lengths less than 55 cm was set to one because management regulations prohibit the retention of sablefish smaller than the minimum size limit (DFO 2007). The proportion of the catch discarded at-sea is likely to differ among gear types and different values of $R_{50}$ were used for each gear type. For the trawl fishery, $R_{50}$ was set at the minimum size limit while in the trap and longline fisheries, $R_{50}$ was set at 60 cm because there may be an incentive to highgrade (Chen and Gordon 1997). However, with no empirically estimated values of $R_{50}$ available for the B.C. sablefish fishery, a range of values for $R_{50}$ were considered for each gear type during sensitivity analyses.

Yield-per-recruit for a given gear type was calculated using the YPR model analogous to that of Thompson and Bell (1934) such that:

**(6)** $$Y_g = \sum_{j=1}^{n} W_j C_{j,g} \quad ,$$

where $Y$ is the yield, $C_{j,g}$ is the catch of fish in each length class $j$ for each gear type $g$ as per Equation 3, and $W_j$ is the average weight of fish in the $j$th length class shown in Equation 2. Because the initial number of recruits is generally unknown, calculating Equation 6 can be difficult. Dividing both sides of the equation by $R$, however, allows Equation 6 to be expressed at the total yield-per-recruit.

Spawner biomass-per-recruit (SBPR) as a function of the full-selectivity fishing mortality was calculated as,

**(7)** $$SBPR = \sum_{j=1}^{n} W_j m_j e^{-Z_j} \quad ,$$

where the proportion mature in each length class, $m_j$, was calculated as,

**(8)** $$m_j = \frac{1}{1 + \exp^{-(h(L_j - m_{50})}} \quad ,$$

where $m_{50}$ is the length at which 50% of the catch are sexually mature and $h$ is the slope at $m_{50}$.

YPR and SBPR computations were made for a range of gear-specific fishing mortality rates. For the trap fishery, fishing mortality rates from 0.0 - 0.6 in increments of 0.03 were considered. Full-selectivity fishing mortality rates for the trawl and longline fisheries were calculated as 32% and 52% of the trap fishing mortality rate, respectively (Haist et al. 2001). Fishing mortality rates for the trap, trawl, and longline fisheries were based on the average share of the total sablefish landings by each gear type between 2002 and 2007 (Haist et al. 2001).

Because YPR and SBPR analyses require a single selectivity-at-length function as model inputs, the observed tag recovery data described in Chapter 2 was pooled across all years for each gear type. As in Chapter 2, two candidate models of selectivity (asymptotic and dome-shaped) were fitted to the pooled tag recoveries for the trap and longline fisheries and an exponential model was fitted to the pooled trawl tag recovery data. A small sample Akaike's Information Criterion was used to identify a single preferred model for each gear type (Table 7).

## Model scenarios

The YPR and SBPR models shown above were modified to provide an indication of how YPR and SBPR change under three different assumptions about discard mortality.

### Pseudo scenario

The first scenario, referred to as the "pseudo" discard scenario (Chen and Gordon 1997), optimistically assumes that all discarded catch survives. In the pseudo scenario, only the proportion of fish retained, 1- $P_{j,g}$ , are included in the catch and survival equations shown in Equations 3 and 4, respectively. The pseudo scenario is therefore equivalent to assuming that the discarded fish are not caught (Chen and Gordon 1997). The assumption of zero discard mortality, however, is highly unrealistic for B.C. sablefish because some level of mortality is generally associated with the discarded catch (Chen and Gordon 1997; Davis et al. 2002). Nonetheless, the pseudo discard scenario provides a useful basis for comparing estimates of YPR and SBPR calculated under different assumptions of discard mortality.

### Onboard scenario

The second scenario, referred to as the "onboard" discard scenario, models yield-per-recruit by including both the landed and discarded catches (Chen and Gordon 1997). Onboard YPR and SBPR were calculated using Equations 3 and 4, respectively; however, in contrast to the pseudo YPR and SBPR equations, $P_{j,g} = 0$ in the onboard YPR and SBPR models such that all fish (both landed and discarded) are subjected to the full-selectivity fishing mortality rate. This slight modification results in fewer sablefish surviving upon release and generates lower yields than those estimated under the pseudo YPR scenario. In the onboard YPR scenario, discards are still included in the total catch even though they are not landed by the vessel.

### Landed scenario

The third scenario, referred to as the "landed" discard scenario (Chen and Gordon 1997), calculates YPR based on the landed catch only. The landed YPR can be calculated by setting,

**(9)** $$F_{j,\cdot} = \sum_{g=1}^{G} S_{j,g} F_g \quad .$$

The catch $C_{j,g}$ and proportion surviving $N_j$ are calculating using Equations 3 and 4, respectively, however by using Equation 9 to calculate $F_j$, discards are treated as real removals from the stock (i.e., they are subjected to the full fishing mortality rate in the survival equation). Under this scenario, discards are excluded from estimates of the total catch resulting in the lowest fishery yield out of all three scenarios. SBPR for the landed discard scenario is calculated using the same equation used to estimate the onboard SBPR because in both scenarios the discarded catch is subjected to the full-selectivity fishing mortality rate.

The assumption of zero discard mortality made in the pseudo discard scenario is unrealistic for B.C. sablefish, particularly for the bottom trawl fishery, which is known to impose considerable physical damage on the harvested catch (Davis et al. 2001; Schirripa and Methot 2001; Davis 2002). The onboard and landed YPR scenarios, in which the discarded catch is subjected to the full-selectivity fishing mortality rate, therefore provide a more realistic treatment of discard mortality and more realistic YPR estimates. While I acknowledge that not all of the discarded catch will die upon their release, accounting for some level of discard mortality in fishery assessments is an obvious modelling choice and a more precautionary assumption in the formulation of advice to fisheries managers (Rahikainen et al. 2004). Although the onboard scenario acknowledges the mortality of the discarded catch, discards are still included in the catch equation thus artificially inflating estimates of YPR. The landed YPR scenario, which excludes discards from the catch equation but includes discard mortality in the survival equation, is therefore most likely to provide the most realistic estimates of YPR and SBPR for the B.C. sablefish fishery under the equilibrium assumptions and for the set of data inputs considered.

### Biological reference points

Biological reference points (BRPs) are commonly used to propose levels of fishing mortality required to achieve a given management objective (Caddy and Mahon 1995). I use one of the most common BRPs, $F^{max}$ or the fishing mortality rate that maximizes yield-per-recruit, to evaluate the performance of three discard scenarios. A one-sided derivative calculation was used to evaluate the derivative of yield-per-recruit versus fishing mortality over the range of fishing mortality values considered. A cubic spline function was fitted to this relationship and $F^{max}$ was identified by taking the root of the cubic spline versus fishing mortality. $Y^{max}$, the yield-per-recruit associated with $F^{max}$, was also calculated from this cubic spline function.

A second BRP, $F^{SBx}$, based on the results of the spawner biomass-per-recruit (SBPR) model, was also used to assess stock status. $F^{SBx}$ is defined as the fishing mortality rate at which spawner biomass-per-recruit is reduced to $x$% of its unfished level. Spawner biomass-per-recruit recommendations generally lie between 25% and 50% of unexploited levels (Deriso 1987; Goodyear 1989; Clark 1991; Punt 1992; Booth and Buxton 1997) and recruitment overfishing is said to occur when the relative SBPR is reduced to less than 20-30% of the unfished level (Clark 1991). These BRPs are based on temperate demersal marine species like sablefish with life-history characteristics that include high fecundity, pelagic spawning, and little or no parental care (Love 1995; Booth 2001). For this analysis, a reduction of spawner biomass per recruit to 35% of pristine levels, $F^{SB35}$, was considered sufficient to ensure high yields at low risk (Goodyear 1989). The fishing mortality rate corresponding to the quantity, $F^{SB35}$, was estimated graphically.

## Expected loss

One of the direct consequences of discarding is the loss of catch that is removed from the stock but not landed by the fishery. Such a direct loss in YPR is quantified by an index of direct loss (IDL), which compares YPR for the onboard and landed scenarios for a given level of fishing mortality (Chen and Gordon 1997),

**(10)** $$IDL = \frac{Y_{onboard} - Y_{landed}}{Y_{landed}} 100 \quad .$$

where $Y_{onboard}$ is the YPR for the onboard discard scenario and $Y_{landed}$ is the YPR associated with the landed discard scenario. The IDL calculates the percentage of total YPR that is wasted due to discard mortality (Chen and Gordon 1997). Higher IDL values indicate a less efficient fishery.

Economic losses per-recruit associated with the IDL, $E_{IDL}$, were evaluated using the following equation,

**(11)** $$E_{IDL} = \left(Y_{onboard}^{\max} - Y_{landed}^{\max}\right)p \quad ,$$

where $Y_{onboard}^{\max}$ is the value of $Y^{\max}$ obtained under an onboard discard scenario, $Y_{landed}^{\max}$ is the value of $Y^{\max}$ obtained under a landed discard scenario, and $p$ = \$8.50 is the ex-vessel price for sablefish (round weight; Hupert and Best 2004). Estimates of $p$ were based on the average ex-vessel value of the B.C. commercial trap and longline fisheries for the period 2000 – 2003 (Hupert and Best 2004).

In many fisheries, discards are not included in estimates of total fishing mortality and $F$ is calculated from the landed catch only. In this case, estimates of YPR will reflect that of a pseudo YPR scenario. The index of potential loss (IPL) quantifies the bias inherent in YPR under these circumstances. The IPL provides an indication of the maximum loss of fishery yield over the life of the cohort due to discarding (Chen and Gordon 1997) and is calculated as,

**(12)** $$IPL = \frac{Y_{pseudo} - Y_{landed}}{Y_{landed}} 100 \quad .$$

Potential economic loss per-recruit, $E_{IPL}$, was calculated by replacing $Y_{onboard}^{\max}$ with $Y_{pseudo}^{\max}$ in Equation 11. Given that the landed YPR scenario is most applicable to the B.C. sablefish fishery, potential biological and economic loses caused by at-sea discarding are best reflected by the IPL and the $E_{IPL}$.

### Sensitivity analysis

Sensitivity analyses were used to examine how deviations from key assumptions about (*i*) natural mortality, (*ii*) retention sizes, and (*iii*) size-selectivity functions affect estimates of yield-per-recruit and the associated BRPs. It is well established that YPR curves are most sensitive to the value of natural mortality, *M*, a parameter which is notoriously difficult to accurately and precisely estimate (Beverton and Holt 1957). Two additional values of *M* (0.08 $yr^{-1}$ and 0.12 $yr^{-1}$) were therefore used in additional YPR calculations.

In the second sensitivity analysis, I explored two alternative estimates of the length at 50% retention, $R_{50}$. Baseline values of the $R_{50}$ parameter were varied by ± 5 cm for each gear type. The final sensitivity analysis explored the effects of using an alternate size-selectivity function for the commercial trap fishery. To do so, asymptotic selectivity replaced dome-shaped selectivity in YPR and SBPR analyses and estimates of fishery yield and value were re-evaluated given this modification.

## Results

### Baseline yield-per-recruit

At low fishing mortality rates between 0.0 and 0.05, maximum differences in yield-per-recruit among the three scenarios were less than 25% (Figure 9). However, these differences became large as fishing mortality increased ($F > 0.05$), with the largest differences being observed between the pseudo and landed YPR scenarios at $F \geq 0.55$.

The landed YPR, which resulted in the smallest yields for a given level of fishing mortality, was dome-shaped with rapidly decreasing YPR at $F > 0.20$. The fishing mortality rates required to obtain $Y^{max}$ were also the lowest in the landed YPR scenario relative to the onboard and pseudo discard scenarios. In contrast to the landed discard scenario, YPR was asymptotic for the pseudo and onboard scenarios. Under these scenarios, yield-per-recruit increased rapidly with increasing fishing mortality before decreasing marginally at fishing mortalities greater than $F^{max}$. The pseudo discard scenario produced the largest estimate of YPR and SBPR for a given level of fishing mortality, a result that is not surprising given the assumption that all discards survived in pseudo YPR and SBPR models. Estimates of $Y^{max}$ associated with the pseudo YPR scenario were nearly double estimates obtained under the landed YPR scenario and 16% greater than estimates obtained under the onboard YPR scenario. The pseudo YPR also required the highest levels of fishing mortality ($F^{max} = 0.39$ and $F^{SB35} = 0.07$) to achieve $Y^{max}$ and reduce the pristine spawning biomass-per-recruit to 35% of its unfished level, respectively (Figure 10).

$F^{SB35}$ consistently provided the most conservative target fishing mortality rate. For example, under the pseudo YPR scenario, $F^{max}$ was nearly five times greater than $F^{SB35}$. While this difference was much smaller for the onboard and landed YPR scenarios (approximately two and three times greater, respectively), $F^{max}$ consistently exceeded $F^{SB35}$ for a given discard scenario.

The index of potential loss (IPL) increased quickly with increasing fishing mortality (Figure 11). While similar increases were observed in the IDL, these losses increased at a much slower rate relative to the IPL. At a fishing mortality rate of $F$ = 0.12, the IDL and the IPL were 43% and 49%, respectively, and economic losses per-recruit were CDN \$3.49 and CDN \$5.44 for the $E_{IDL}$ and the $E_{IPL}$, respectively.

## Sensitivity analysis

As expected, all types of yield-per-recruit varied with changes in natural mortality. A lower natural mortality rate caused the estimated yield-per-recruit curve to steepen and peak at a lower level of fishing mortality while a higher $M$ caused the curve to flatten and peak at higher natural mortality rates. For all three discard scenarios, the highest yields were found at the lowest level of natural mortality (0.08 $yr^{-1}$) since lower natural mortality rates allow for a greater accumulation of biomass (Table 8). Both target fishing mortality rates, $F^{max}$ and $F^{SB35}$, also increased with increasing natural mortality indicating that higher levels of fishing effort would be required to achieve $Y^{max}$ and reduce the spawning stock biomass to 35% of its unfished level, respectively. Decreasing $M$ also had a large affect on YPR and SBPR with decreases in natural mortality leading to large decreases in fishery yield and spawner abundance.

At a fishing mortality rate of $F = 0.12$, the IPL was unaffected by changes in the natural mortality rate with changes in $M$ causing proportional changes in the landed and pseudo YPR equations (Table 9). The IDL, however, was affected by changes in $M$ with decreases in $M$ from 0.10 $yr^{-1}$ to 0.08 $yr^{-1}$ causing a 9% reduction in the IDL and increases in the natural mortality rate from 0.10 $yr^{-1}$ to 0.12 $yr^{-1}$ resulting in a 14% increase. Changes in $M$ had the largest affect on the landed and onboard YPR scenarios since $M$ has a particularly strong influence on both the catch and survival equations in the landed and onboard scenarios.

I also explored the effects of changes in $R_{50}$ in sensitivity analyses. In the pseudo discard scenario, increasing $R_{50}$ by 5 cm resulted in an increase in $F^{max}$ and $Y^{max}$ by 60% and 3% respectively, but had no effect on $F^{SB35}$ (Table 10). In contrast, increases in $R_{50}$ resulted in a decline in $F_{max}$ and a lower estimate of $Y^{max}$ in the landed YPR scenario. The onboard scenario, which did not use $R_{50}$ in estimates of YPR, was unaffected. With increases in the length at 50% retention being intended to simulate highgrading in the fishery, it is not surprising that a reduction in fishery yield was observed in the landed discard scenario in which all of the discarded (or highgraded) catch was assumed to die.

Increasing $R_{50}$ resulted in decreases in both IPL and IDL compared to baseline values (Table 11). Decreasing the $R_{50}$ parameter by 5 cm resulted in a 123% increase in IDL and an 88% increase in the IPL. Economic losses per-recruit exhibited a similar pattern with increases in $R_{50}$ by 5 cm resulting in small losses in fishery value and a decrease in $R_{50}$ by 5 cm leading to greater economic losses relative to the baseline scenario.

Under an asymptotic selectivity model for the trap fishery, estimates of $Y^{max}$ increased slightly while $F^{max}$ decreased slightly for the onboard and landed scenarios (Table 12). Changes in the IDL and IPL under an asymptotic model were also negligible (7% and 2%, respectively), as were estimates of economic losses per-recruit (Table 13). This suggests that the choice of selectivity function for a particular gear type does have an affect on estimates of fishery yield and value. While this affect was rather small when considered on a per-recruit basis, if considered in the wider context of the fishery, differences in fishery yield and value may be considered substantial.

## Discussion

In this chapter, I evaluated the impacts of at-sea discarding for the B.C. sablefish fishery using empirical estimates of gear selectivity. My results suggest that expected losses in fishery yield and value resulting from at-sea discarding can be substantial. In the absence of discarding (i.e., the pseudo discard scenario), much larger estimates of YPR and SBPR were observed relative to scenarios in which discarding occurred. The pseudo scenario also required the highest level of fishing mortality to reduce the spawning biomass to 35% of its unfished value. Although the pseudo discard scenario represents the most desirable scenario from a biological or conservation perspective, it is highly unrealistic because it is likely that not all of the discarded catch survives upon release (Chen and Gordon 1997; Davis 2002; Rahikainen et al. 2004). In fact, in a number of fisheries, discard mortality is substantial and its exclusion from fishery assessments can bias estimates of fishery yield and value high (Chopin and Arimoto 1995; Chen and Gordon 1997; Davis et al. 2001; Davis 2002; Rahikainen et al. 2004). However, despite this limiting assumption, the pseudo discard scenario provides an important baseline value for comparing estimates of YPR and SBPR calculated under alternate assumptions of discard mortality. For example, comparisons between the pseudo and the landed YPR scenarios, quantified by the IPL, calculated the total potential loss of fishery yield over the life of a cohort due to discarding. In the B.C. sablefish fishery, up to 49% of the total YPR is potentially lost as a result of at-sea discarding, amounting to a potential economic loss of CDN $5.44/recruit. Given the economic importance of the sablefish fishery to the B.C. groundfish sector, such losses in fishery yield and value may be considered substantial and potentially threaten the long-term biological and economic viability of the fishery. While the onboard discard scenario presents a more realistic way of estimating fishery yield by acknowledging the mortality of discarded fish, estimates of YPR obtained under an onboard discard scenario are still likely to be biased high because the discarded catch is still included in the catch equation. The landed discard scenario, in which YPR is calculated from the landed catch only, therefore provides a more realistic way of calculating YPR and evaluating losses in fishery yield and value associated with at-sea discarding.

The difference between the onboard and landed discard scenarios, quantified by the IDL and referred to as the “discard-per-recruit” by Chen and Gordon (1997), calculates the loss of catch that is removed from the population, but not landed by the fishery (Chen and Gordon 1997). In the B.C. sablefish fishery, approximately 43% of the total potential YPR is wasted because of at-sea discarding suggesting that improvements in efficiency of the fishery could be achieved if discarding were eliminated or reduced either through regulatory measures or economic incentives. The current price structure for B.C. sablefish is such that sablefish greater than 65 cm are considerably less valuable than smaller legal-sized sablefish (Haist et al. 2001). This incentive structure is likely to lead to an increase in the discarding of small sablefish as vessels discard sablefish to stay within the quota or highgrade part of the catch in order to retain the larger and more valuable sablefish (FAO 2008). Removing such perverse economic incentives may help to reduce discarding and mitigate potential losses in yield and revenue for the B.C. sablefish fishery. Efforts to increase the survival of the discarded catch may also help to minimize the negative biological and economic impacts of discarding. For example, imposing strict handling guidelines for the discarded catch or restricting fishing effort when high sea surface temperatures are expected may help to reduce discard mortality.

As expected, changes in the natural mortality rate can have a large effect on expected losses in fishery yield and value. Because the mortality of the discarded catch can be incorporated into estimates of $M$ by simply inflating estimates of $M$ by an amount equal to the assumed discard mortality rate, exploring the effects of changes in $M$ on YPR provides an indication of how increases or decreases in discard mortality influence fishery yield and value. Increases in $M$ resulted in increases in $F_{max}$ under all discard scenarios because greater harvesting intensity early in the life of a cohort is required to ensure that enough fish are captured before dying of natural causes. When natural mortality rates decreased, larger yields were obtained at lower levels of fishing mortality. Lower natural mortality rates allowed fish to remain in the ocean longer thus providing fish with an opportunity to gain appreciable increases in mass before becoming susceptible to the fishery thus generating larger yields. This sensitivity analysis therefore suggests that discard mortality may have a large affect on fishery yield and value, with decreases in discard mortality leading to large increases in yield and economic revenue per-recruit. Management efforts should therefore focus on minimizing the catch of small sablefish either through improvements in gear selectivity or regulatory measures such as area or depth closures that restrict harvesting in areas in which small sablefish are known to reside. While commercial sablefish fishing is currently banned in nearshore areas (e.g., inlets and coastal areas; DFO 2006), small sablefish are still vulnerable to capture in shallow offshore areas $< 300$ m (Hanselman et al. 2005). Restricting fishing effort in shallow waters may therefore be an effective means of reducing the interception of small sablefish and mitigating losses in fishery yield and value as a result of at-sea discarding.

The effect of highgrading the catch at-sea, represented by changes in $R_{50}$, on losses in fishery yield and value were also explored during sensitivity analyses. Exploring the effects of changes in $R_{50}$ is important for developing sustainable harvest strategies for exploited fish stocks because the size at which 50% of the fish are retained is one of a handful of parameters under the control of fishery managers. Increases in $R_{50}$ allowed more legal-sized sablefish to be discarded at-sea. In the landed YPR and SBPR scenarios, which assumed 100% mortality of the discarded catch, increases in $R_{50}$ led to reduced fishery yields and spawning stock biomass. In the pseudo YPR and SBPR scenarios however, the opposite pattern was observed with increases in $R_{50}$ leading to increased fishery yield. This suggests that when the mortality of the discarded catch is low (i.e., zero), as in the pseudo discard scenario, increases in fishery yield could be obtained by increasing the size at 50% retention. In such cases, management measures that seek to reduce discarding using more selective fishing gear and practices, regulatory measures, or economic incentives may help to reduce losses in yield and fishery value associated with discarding.

In 2006, Fisheries and Oceans Canada developed a new integrated fisheries management plan (IFMP) for all fisheries participating in the B.C. groundfish sector (DFO 2007). The new IFMP requires all vessels to use new forms of information technology, such as video monitoring and geo-referenced positioning systems, to document catches of all marketable groundfish harvested in the B.C. groundfish fishery (DFO 2007). The new IFMP will greatly improve the collection of accurate catch data for all fisheries participating in the B.C. groundfish sector. Of particular importance will be estimates of sablefish discard quantities and catch rates for each gear type in the B.C. groundfish fishery. Yet, despite the best intentions of the new IFMP, further improvements to the IFMP are required to eliminate discarding in the fishery altogether. For example, longline vessels are currently required to demonstrate that all sub-legal discards are below the minimum size limit (DFO 2007). The same requirements, however, do not apply to the trap and trawl fisheries generating some uncertainty in the magnitude and size distribution of the discarded catch in these fisheries (DFO 2007). Furthermore, the new IFMP imposes no penalties for vessels for discarding sub-legal sablefish despite the fact that research suggests that smaller sablefish experience higher mortality rates relative to larger fish (Davis 2002). Until these issues are addressed within the context of the IFMP, discarding will continue to remain an ongoing problem in the management of the B.C. sablefish fishery.

The final sensitivity analysis quantified expected losses in fishery yield and revenue for alternate assumptions about the relationship between size and susceptibility to capture for the trap fishery. As we saw in the case of the Northeast Atlantic cod fishery, failure to accurately identify the shape of the selectivity function can cause management schemes based on maximum sustainable yield or alternative measures of ‘sustainability’ to fail (Hilborn and Walters 1992; Myers et al. 1997; Helfman 2007). When asymptotic selectivity was assumed for the trap fishery, increases in YPR and decreases in economic losses per-recruit (Table 14) were observed suggesting that the choice of selectivity function does have an effect on fishery yield and value. While, these effects were small (≥ 7%), these effects may become large if considered within the broader context of the fishery and not just on a per-recruit basis. This final sensitivity highlights the importance of monitoring and accounting for differences in selectivity in the ongoing management strategy evaluation for B.C. sablefish.

## Limitations

As with any model, YPR analyses are based on a number of assumptions that limit the conclusions that can be drawn from this analysis (Malcolm 2001). For example, assumptions that parameters for recruitment, growth, and mortality are constant over time and that the stock is in a steady state equilibrium are unrealistic for B.C. sablefish. In fact, sablefish recruitment is highly variable and a well-developed stock-recruitment relationship for B.C. sablefish has not been established (Schrippa and Colbert 2006). Furthermore, given the inherent variability of the biotic and abiotic environments, the parameters for recruitment, survival, and growth are likely to exhibit changes over time (Beverton and Holt 1957; Malcolm 2001). Management strategies that strictly adhere to $F^{max}$ or $F^{SB35}$ may therefore cause the population to decline rapidly to low levels during periods of poor recruitment, growth, or survival. However, sensitivity analyses including the use of two alternate levels of natural mortality, two different assumptions about length-specific retention rates, and an alternate model of selectivity for the trap fishery were used to test the robustness of model results.

An additional limitation of this analysis stems from the assumption of 100% discard mortality associated with the onboard and landed discard scenarios. While there is generally some level of mortality associated with the landed catch (Chen and Gordon 1997; Olla et al. 1997; Davis et al. 2001; Davis 2002), it is likely that not all discarded sablefish die upon their release. In fact, fixed gears such as trap and longline fisheries generally impose less physical damage on the discarded catch and lead to lower rates of discard mortality than mobile gears, such as bottom trawls (Davis 2002). However, the selectivity of fishing gear is just one of many factors influencing the survival of discarded fish. Environmental factors, such as air and sea surface temperature, also play a large role in survival and even the most benign gear types impose some level of mortality on the discarded catch. Therefore, including gear-specific discard mortality rates in subsequent YPR and SBR analyses would greatly improve the accuracy and interpretation of model results.

## Conclusion

The approach described in this chapter allows fishery managers to quantify expected losses in fishery yield and value for different discard mortality assumptions. By quantifying the bias present in fishery assessments that ignore the mortality of the discarded catch in estimates of total fishing mortality, fishery managers can formally evaluate biological and economic losses resulting from at-sea discarding.

In my analysis, I found that up to 49% of the total potential YPR was wasted because of at-sea discarding, which equates to economic losses of CDN $5.44/recruit. This analysis therefore suggests that gains in fishery efficiency and value could be realized if discarding were reduced or eliminated in the fishery either through the removal of perverse market incentives, regulatory approaches that restrict harvesting in areas where young sablefish reside, or the imposition of strict handling guidelines for the discarded catch. Given the economic importance of the B.C. sablefish fishery to the groundfish sector, evaluating the biological and economic impacts of at-sea discarding and taking measures to reduce discarding, is critical to ensuring the long-term sustainability of the B.C. sablefish fishery.

## Tables

**Table 6 Summary of parameter values used in baseline yield-per-recruit and spawner biomass-per-recruit calculations.**

| Symbol | Value |
|---|---|
| $L_1$ (cm) | 35 |
| $L_n$ (cm) | 80 |
| $L_\infty$ (cm) | 83 |
| $K$ (yr$^{-1}$) | 0.160 |
| $\alpha$ | 0.000010 |
| $\beta$ | 3.01 |
| $M$ (yr$^{-1}$) | 0.10 |
| $R$ | 1.0 |
| $R_{50}$ | 60 cm (trap/longline) |
| | 55 cm (trawl) |
| $Q$ | 0.50 |
| $m_{50}$ | 65 cm |
| $h$ | 0.40 |
| $p$ | $8.50/kg (CDN) |

**Table 7 Summary of length-based selectivity results for asymptotic, dome-shaped, and exponential model fits for all years combined.**

| Gear | Asymptotic Model | | | | Dome-shaped Model | | | | | | Exponential Model | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | $U'_y$ | $L_{50}$ | $\beta$ | AIC | $U'_y$ | $L_{50}$ | $\beta_1$ | $\beta_2$ | $\beta_3$ | AIC | $U'_y$ | $\beta$ |
| L | 0.02 | 54 | 0.26 | 18316 | 0.02 | 57 | 0.20 | 0.18 | 76 | 18322 | n.a. | n.a. |
| Tp | 0.08 | 52 | 0.26 | 104553 | 0.09 | 58 | 0.16 | 0.18 | 72 | 104502 | n.a. | n.a. |
| Tr | n.a. | n.a. | n.a. | n.a. | n.a. | n.a. | n.a. | n.a. | n.a. | n.a. | 0.09 | 0.01 |

Note: n.a. = not applicable, L = longline, Tp = Trap, Tr = Trawl

**Table 8 Biological Reference Points (BRPs) for each discard scenario for the baseline *M* = 0.10 and two additional *M* values (*M* = 0.08 $yr^{-1}$ and *M* = 0.12 $yr^{-1}$). Values shown in parentheses indicate the percent change from baseline BRP estimates.**

| | $F^{max}$ | $Y^{max}$ *(kg)* | $F^{SB35}$ |
|---|---|---|---|
| *M*=0.10 | | | |
| Pseudo | 0.39 | 1.47 | 0.07 |
| Onboard | 0.17 | 1.24 | 0.05 |
| Landed | 0.11 | 0.83 | 0.05 |
| *M*=0.08 | | | |
| Pseudo | 0.30 (-23%) | 1.68 (+14%) | 0.10 (+30%) |
| Onboard | 0.15 (-12%) | 1.40 (+11%) | 0.06 (+16%) |
| Landed | 0.10 (-9%) | 0.99 (+19%) | 0.06 (+16%) |
| *M*=0.12 | | | |
| Pseudo | 0.52 (+33%) | 1.30 (-12%) | 0.04 (-43%) |
| Onboard | 0.19 (+6%) | 1.10 (-11%) | 0.02 (-60%) |
| Landed | 0.11 (0%) | 0.70 (-17%) | 0.02 (-60%) |

**Table 9 Index of potential loss (IPL) and index of direct loss (IDL) and associated economic losses per-recruit ($E_{IPL}$ and $E_{IDL}$) for two alternate values of *M*.**

| *M* ($yr^{-1}$) | IPL (%) | $E_{IPL}$ ($/recruit) | IDL (%) | $E_{IDL}$ ($/recruit) |
|---|---|---|---|---|
| 0.08 | 49 | 5.87 | 39 | 3.39 |
| Baseline | 49 | 5.44 | 43 | 3.49 |
| 0.12 | 49 | 5.10 | 48 | 3.40 |

**Table 10 Biological Reference Points (BRPs) for each discard scenario and two alternate values for the length at 50% retention, $R_{50}$ (± 5 cm). Values in parentheses indicate the percent change from baseline BRP estimates.**

| | $F^{max}$ | $Y^{max}$ *(kg)* |
|---|---|---|
| $R_{50}$ = baseline | | |
| Pseudo | 0.39 | 1.47 |
| Onboard | 0.17 | 1.24 |
| Landed | 0.11 | 0.83 |
| $R_{50}$ = +5cm | | |
| Pseudo | 0.62 (+59%) | 1.51(+3%) |
| Onboard | 0.17(0%) | 1.24(0%) |
| Landed | 0.09(-18%) | 0.62(-24%) |
| $R_{50}$ = -5cm | | |
| Pseudo | 0.31(-21%) | 1.43(-3%) |
| Onboard | 0.17(0%) | 1.24(0%) |
| Landed | 0.12(+9%) | 0.98(+18%) |

**Table 11 Index of potential loss (IPL) and index of direct loss (IDL) and associated economic losses per-recruit ($E_{IPL}$ and $E_{IDL}$) for two alternate values of $R_{50}$ (± 5 cm).**

| $R_{50}$ | IPL (%) | $E_{IPL}$ ($/recruit) | IDL (%) | $E_{IDL}$ ($/recruit) |
|---|---|---|---|---|
| + 5 cm | 28 | 3.83 | 22 | 2.21 |
| Baseline | 49 | 5.44 | 43 | 3.49 |
| - 5 cm | 92 | 7.60 | 96 | 5.27 |

**Table 12 Biological Reference Points (BRPs) for each discard scenario using asymptotic model fits for the commercial trap fishery. Values in parentheses indicate the percent change from baseline BRP estimates calculated using dome-shaped model fits.**

| | $F^{max}$ | $Y^{max}$ *(kg)* | $F^{SB35}$ |
|---|---|---|---|
| Pseudo | 0.34(-13%) | 1.47(0%) | 0.07(0%) |
| Onboard | 0.14(-21%) | 1.27(+2%) | 0.04(-20%) |
| Landed | 0.09(-18%) | 0.89(+7%) | 0.04(-20%) |

**Table 13 Index of potential loss (IPL) and index of direct loss (IDL) and associated economic losses ($E_{IPL}$ and $E_{IDL}$) using asymptotic model fits for the trap fishery.**

| Selectivity Model for the Trap Fishery | IPL (%) | $E_{IPL}$($/recruit) | IDL (%) | $E_{IDL}$($/recruit) |
|---|---|---|---|---|
| Asymptotic | 48 | 4.93 | 40 | 3.23 |
| Dome-shaped | 49 | 5.44 | 43 | 3.49 |

## Figures

**Figure 9 Yield-per-recruit versus fishing mortality for each discard scenario calculated using baseline parameter values.**

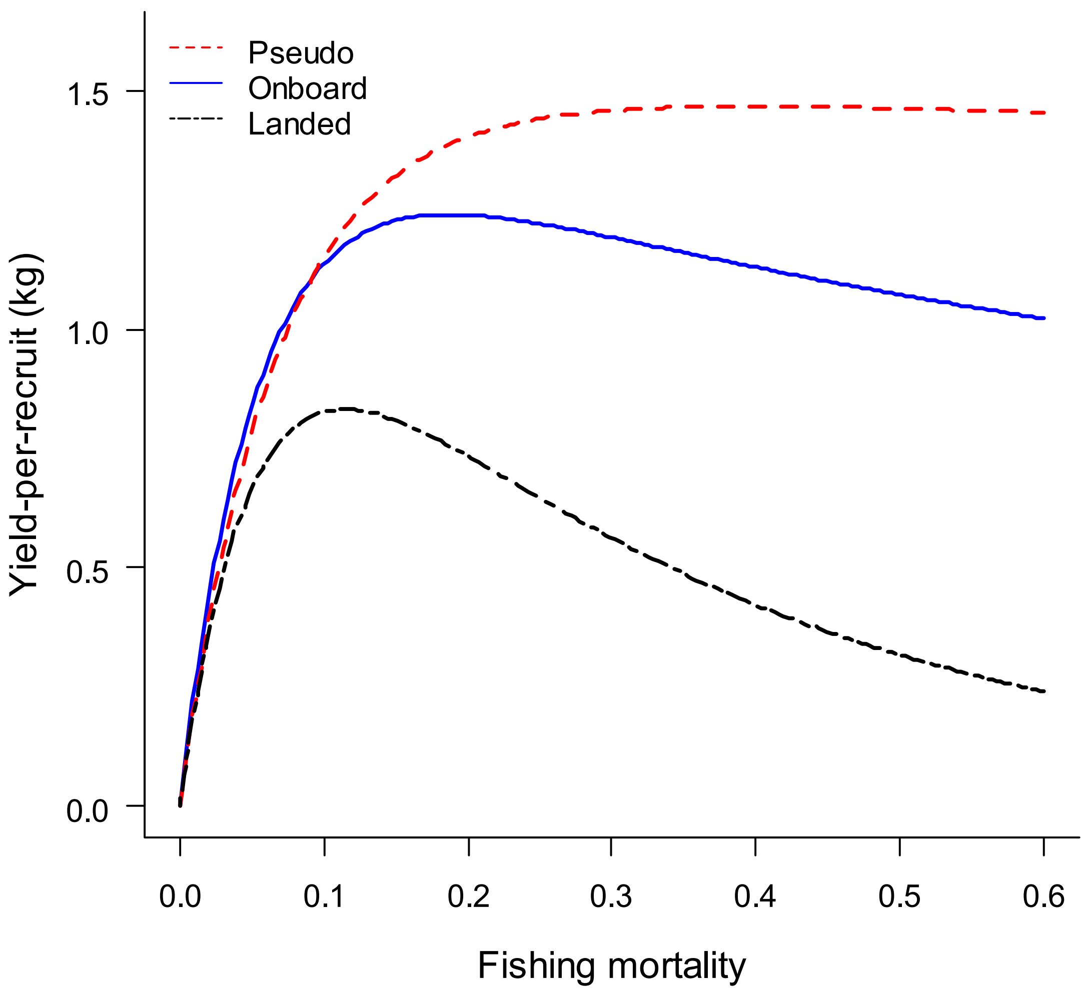


**Figure 10 Spawner biomass-per-recruit versus fishing mortality for each discard scenario calculated using baseline parameter values. The dotted lines indicate reference points for $F^{SB35}$ for each discard scenario.**

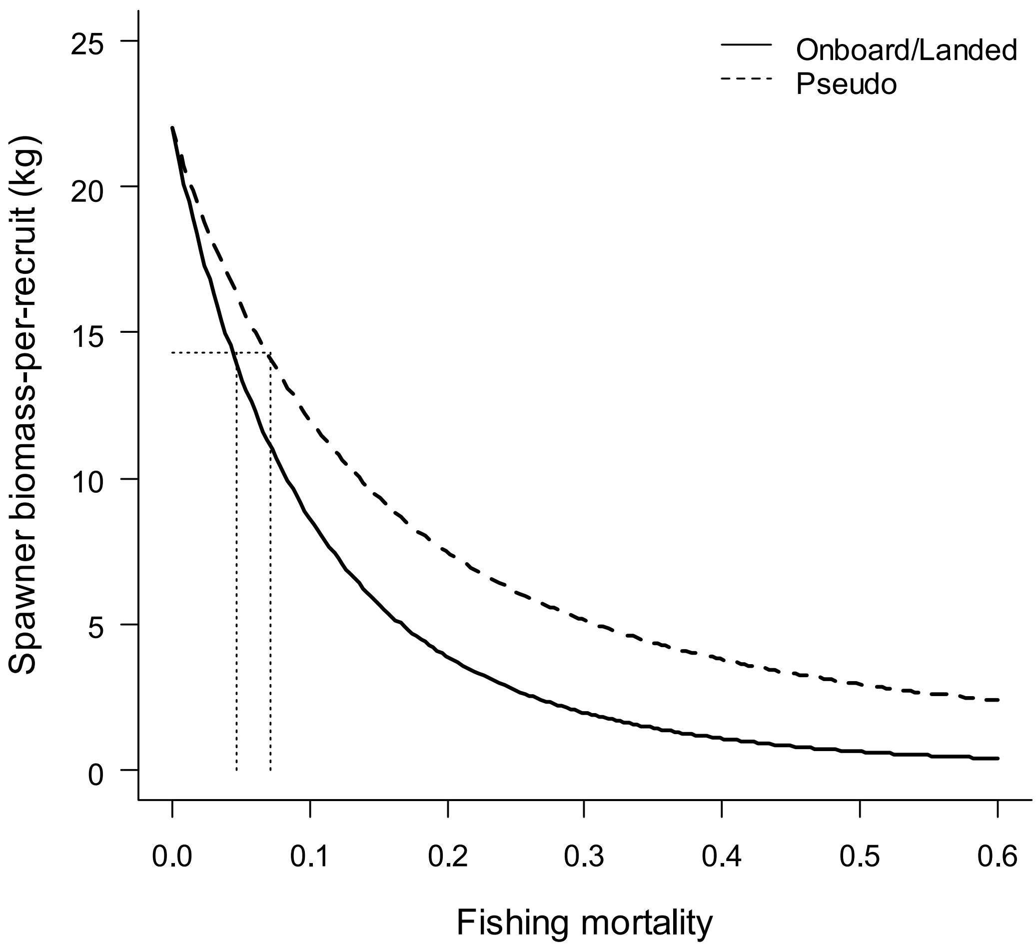
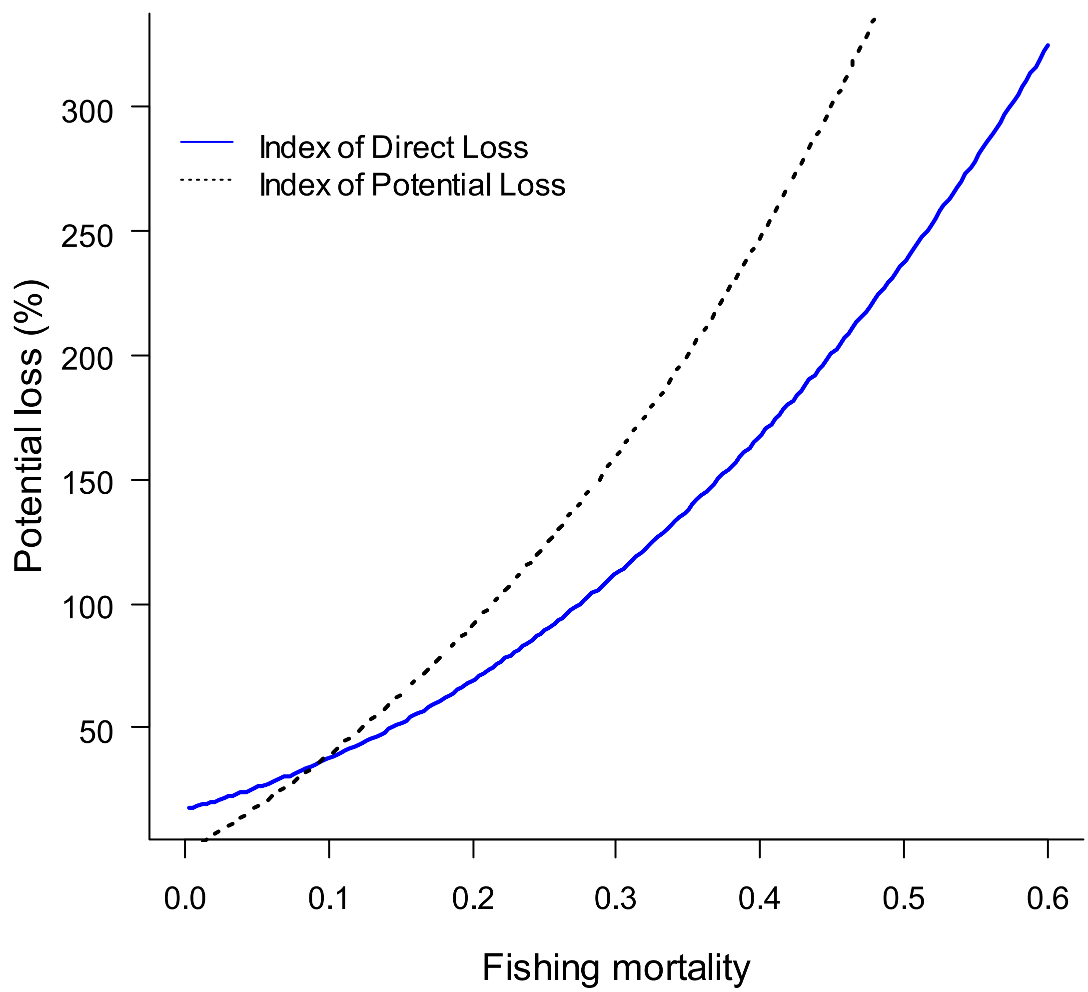


**Figure 11 Index of direct loss and index of potential loss calculated using baseline parameter values.**



# CHAPTER 4
# GENERAL CONCLUSIONS

This thesis employs a direct method for estimating size-selectivity functions for three commercial gear types employed in the B.C. sablefish fishery. Quantifying the relationship between size and susceptibility to capture and understanding how the shape of this relationship may change over time in response to physical, biological, or environmental factors, is important for evaluating the impacts of the fishery on stock abundance and composition. When empirical estimates of gear selectivity are available, in addition to knowledge on growth and natural mortality, YPR analyses can be used to identify target fishing mortality rates and evaluate different management actions such as size limits and minimum mesh sizes in trawls (e.g., Beverton and Holt 1957; Neilson and Bowering 1989).

This study revealed differences in selectivity among gear types. Differences in selectivity were apparent across all years with the longline fishery selecting for the largest size-classes of sablefish, the trap fishery selecting for intermediate size-classes, and the trawl fishery selecting for small sablefish generally below the minimum size limit. Size-selectivity functions were asymptotic for the longline fishery and exponential for the trawl fishery, whereas the estimated selectivity functions for the trap fishery varied over time between asymptotic and dome-shaped selectivity. Temporal variations in selectivity observed in the commercial trap fishery suggest that steady state assumptions regarding gear selectivity are inappropriate for the assessment of B.C.

sablefish. Instead, it is recommended that further consideration be given to estimating gear selectivity in the ongoing management strategy evaluation for B.C. sablefish.

Chapter 3 incorporates direct estimates of selectivity into length-based models of YPR and SBPR to evaluate potential losses in yield and revenue as a result of at-sea discarding. YPR analyses were largely affected by discarding with the exclusion of discard mortality resulting in 49% of the total YPR being wasted because of at-sea discarding, equivalent to a maximum economic loss of CDN $5.44/recruit. With Chapter 2 indicating that small sablefish are the most vulnerable to trawl gear, accounting for the mortality of small discarded fish is critical to producing non-biased abundance estimates and identifying ‘optimal’ harvest strategies for the fishery. As I show in Chapter 3, failure to account for the discarded catch in sablefish assessments can bias model outputs and lead to large losses in fishery yield and value for the B.C. sablefish fishery. Commercial catch statistics for B.C. sablefish fishery must therefore reflect all fishery removals from the stock, including the mortality of discarded sablefish. While the approach provided in this study describes an analytical method for evaluating the effects of discarding on sablefish yields and revenue, this approach can be applied to any fishery in which discarding is known to occur.

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# APPENDIX – ESTIMATION OF MINIMUM SAMPLE SIZES

The minimum number of tag recoveries, $n_{y,l,g}$ required to estimate capture probabilities with an error no greater than δ was calculated for each length class *l* in each year *y* for each gear type *g* using the following equation (Zar 1984 pg. 380; Cochran 1963 pg. 74),

**(1)** $$n_{y,l,g} = \frac{Z^2_{\alpha(2)} t_{y,l,g} q_{y,l}}{\delta^2},$$

where $Z^2_{\alpha(2)}$ is the upper critical value of the normal distribution (α = 0.05), $t_{y,l,g}$ is the number of tagged sablefish recovered in each year in each length class by gear type *g*, and $q_{y,l}$ is 1- $t_{y,l,g}$, or the number of tagged fish that were not recovered. A 95% confidence level is common (Cochran 1963) and, given that the allowable margin of error is arbitrary (Zar 1984), an error no greater than δ = 0.08 was deemed appropriate for this study.