

**RECONSTRUCTING LINGCOD BIOMASS IN GEORGIA STRAIT
AND THE EFFECT OF MARINE RESERVES ON LINGCOD
POPULATIONS IN HOWE SOUND**

By

STEVEN JAMES DEAN MARTELL

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Department of Zoology

The University of British Columbia
Vancouver, Canada

Date April 27/1999

Abstract

Commercial lingcod fishing in Georgia Strait was closed in 1990 due to severely depressed catches. Seasonal closures and size restrictions were added to sports fisheries in an attempt to rebuild the stock. There have been no fishery independent surveys to estimate lingcod abundance in Georgia Strait. The rate of recovery is monitored through changes in sports fishery catch rates. In this thesis, I use catch statistics and commercial catch and effort data to reconstruct the lingcod biomass time series for statistical area 4B (Georgia Strait and surrounding areas). This reconstruction was carried out using a technique called Stock Reduction Analysis (SRA), where the dynamics of an age-structured model is driven by annual removals. The SRA model was tested using a fake time series of relative abundance data and known parameter values. The model was able to estimate, within 10%, the correct parameter values used to generate the fake data. Catch and effort data (CPUE) are used as a relative abundance index to estimate unfished biomass and stock productivity. It is assumed that the relationship between CPUE and stock size is hyperstable in commercial fisheries. Results of the SRA indicate that Georgia Strait lingcod stock have been depleted by over 90%, and that their reproductive capacity is much lower than that of other temperate fish stocks.

Lingcod in Howe Sound are part of the Georgia Strait Stock. There are 3 small Marine Protected Areas (MPAs) in Howe Sound that are closed to fishing. An *in-situ* mark-recapture study was carried out to evaluate responses of lingcod populations in closed areas, and to determine whether or not lingcod move across reserve area boundaries. Changes in underwater encounter rates and length frequencies in each of 13

different survey sites provide evidence of small-scale seasonal movement patterns in lingcod. Densities and average lengths of lingcod have higher means in reserve areas compared to fished areas, but the differences are not significantly different. Lingcod in the oldest reserve area are significantly larger than lingcod in recently established reserve areas. However, this result may be due to artificial reefs that are present in the oldest reserve area.

The idea of using marine reserves as a fisheries management tool is relatively new in fisheries science. Current ecological questions focus on ideal reserve location and reserve area size. Until contentious issues surrounding the use of marine reserves have been resolved, and proper large-scale field experiments have been conducted to evaluate reserves, computer models can be used to speculate how reserve areas will perform. A spatially explicit computer model (FISHMOD) was used to evaluate the current network of "no-take" areas in Howe Sound. Specifically, the model was used to answer questions pertaining to movement rates in lingcod, spatial distribution of fishing effort, and to compare alternative management policies. Simulation results suggest that a disproportionate amount of fishing effort will be distributed along boundaries of reserve areas, especially at low stock sizes. As exchange rates between reserve area and fished area increase, more fishing effort is distributed along reserve boundaries. In comparison to increasing size limits or reducing fishing season length, a system of MPAs in Howe Sound is more effective.

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General Introduction

The lingcod, *Ophiodon elongatus*, is a top predator among fishes in the coastal ecosystems and is restricted to the Northeastern Pacific (Hart, 1973). The life history and movement of lingcod are not fully understood. For the last 110 years commercial fisheries have exploited this species using a variety of fishing gear types. Up until the 1980s, stock assessments for this species was restricted to analysis of catch and effort data, and more recently advanced age-structured techniques have been used to assess larger lingcod stocks. The commercial fishery for lingcod in Georgia Strait is now closed due to depressed stocks. Because of weak stock assessment techniques and apparent declines in ground fish stocks, governments and fishing industries are now interested in using permanent closed areas as refugia.

Lingcod range from Northern California to the Aleutian Islands in Alaska and are primarily found on rocky reefs (Cass *et al.*, 1990). Adults feed on herring (*Clupea pallasii*), Pacific hake (*Merrluccius productus*), and lesser amounts of other fish including salmon and rockfish. In coastal waters, female lingcod reside in deeper waters than males and during the spawning season they move inshore to spawn (Cass *et al.*, 1984; Gordon, 1994). The spawning season begins in mid January, peaks in mid February, and is complete near the end of March. After females have laid their eggs, they retreat back to deeper waters. Male lingcod aggressively defend one, and sometime two, egg mass(es) from predators for a 6 week incubation period. During this time period, anglers easily catch aggressive nest guarding lingcod. After the eggs hatch, the larvae spend 6-8 weeks feeding on pelagic zooplankton and ichthyoplankton (mostly copepods and herring larvae). Dispersal capabilities of the larvae are unknown. Larvae are most abundant in

areas where adults are also abundant (Cass *et al.*, 1990). Young of the year are not found in habitats occupied by older cohorts. The 50% age of maturity for female lingcod is 4 years and the 50% age of maturity for males is 3 years. Female lingcod spawn annually and live for a maximum of 20 years and male lingcod live to a maximum of 14 years (Cass *et al.*, 1990). In Georgia Strait, the asymptotic length for females is 107-cm total length (TL) and for males, 77-cm total length (Smith and McFarlane, 1995).

Commercial fisheries for lingcod were first documented in the 1860s, and the first catch statistics for this species began in 1889. During the early days of the fishery, lingcod were harvested using handlines. Prior to the development of trawl gear, the southern Georgia Strait (Comox-Powell River to Nanaimo-Vancouver) supported over 200 handline vessels (Cass *et al.*, 1990). Commercial lingcod fishing also occurred in Vancouver Harbor. In the 1940s the introduction of trawl gear in Georgia Strait resulted in a shift in gear types for harvesting lingcod and by the 1970s the majority of lingcod landed in Georgia Strait came from the trawl fishery. The 1940s was the “hey day” of the Georgia Strait lingcod fishery, with average landings of 4,000 t per year. At this time the lingcod fishery was the 4th largest fishery in BC, and was the main source of fresh fish in local markets.

The introduction of trawl gear allowed the fishery to expand to new grounds, namely the West Coast of Vancouver Island and Hecate Strait. The larger trawl vessels were suitable for working in more exposed areas, whereas the small handline vessels were restricted to working in protected waters. By the 1970s most of the smaller vessels discontinued fishing for lingcod in Georgia Strait, and in 1990 there was only a single handline vessel actively fishing lingcod. The Department of Fisheries and Oceans closed

the commercial fishery in 11 of the 13 statistical areas in 1990. In 1993, all 13 statistical areas were closed to commercial fishing.

Prior to 1980, there were no quantitative estimates of the impacts of the recreational fisheries on lingcod stocks in Georgia Strait. With catches and catch rates declining in Georgia Strait commercial fisheries, the Department of Fisheries and Oceans implemented a sports fishery creel survey program. At the time, the intention of the creel survey program was to collate information about the effects of sports fishing pressure on Georgia Strait salmon, lingcod, and rockfish stocks. From 1983 to 1989, there was no downward trend in the catch rates, and the sports fishery continued unregulated. When the commercial fisheries were closed in 1990, restrictions were put into place on the sports fishery as a precautionary measure. Sports fishery restrictions for lingcod include a seasonal closure from October 1 to May 31, a minimum size limit of 65-cm, and a bag limit of one fish per day and a maximum of 10 per year. The recommendation by stock assessment officials is a total closure for all lingcod fisheries in Georgia Strait. This was first recommended in 1987 (G. A. McFarlane, Pers. Comm.).

Like many other fisheries around the world, stock assessment techniques using commercial catch and effort data were also failing for the Georgia Strait lingcod stock. There was no apparent trend in the CPUE data that indicated the stock was about to collapse. The CPUE statistics generally reflected the appearance of a strong year class that recruits to the fishery (Beamish, *et al.*, 1994; McFarlane and Leaman, 1996). In the interest of maintaining commercially viable stocks, one approach to dealing with uncertainty is the use of closed areas (e.g. using closed areas to protect stocks when fishing mortality rates are unknown (Walters, 1998)). The use of closed areas or “no-

take" marine protected areas is the newest wave of fisheries management tools currently under investigation.

The idea of using marine reserves as a fisheries management tool in fisheries science was first explored in quantitative terms by Beverton and Holt (1957) (cited in Guenette *et al.*, 1998). Estimates of movement rates across the MPA boundary are required for evaluating the effectiveness of an MPA (Zellor and Russ, 1998; Roberts and Polunin, 1991). In many fish species movement rates across MPA boundaries have been quantified using tagging data (for examples see Roberts and Polunin, 1991) or ultrasonic telemetry data (see Matthews, 1992; Zellor and Russ, 1998). Previous tagging studies have been carried out to estimate lingcod movement rates in the Strait of Georgia (Matthews, 1992; Smith and McFarlane, 1990). Smith and McFarlane (1990) estimated movement rates for lingcod in an unbounded environment to be 500m/day and 1000m/day for males and females, respectively. In addition, several studies have shown tendencies for larger lingcod to reside in deeper waters (Smith and McFarlane, 1990; Gordon, 1994), indicating movement from shallow near-shore waters to deep offshore waters. There is some evidence that offshore movement is in part due to an ontogenetic shift in habitat preference, however, this was only observed for West Coast stocks (Jagiello, 1990; Jagiello, 1995). Lingcod stocks in Georgia Strait and Puget sound appear to mix with each other, however, there is little or no mixing between inside and outside stocks (Jagiello *et al.*, 1996).

The design of marine protected areas must incorporate the understanding of how dispersal influences population dynamics in a given area. Source-Sink theories are one method of considering dispersal influences on population dynamics (Roberts, 1998).

Areas of disproportionately high emigration rates (source areas) are more suitable for protection than areas of disproportionately high rates of immigration (sink areas).

Protecting a source population also benefits fisheries that operate adjacent to the reserve boundary. However, if movement rates across the reserve boundary are high, or the edge/area ratio is high, then reserve boundary fishing can be potentially damaging (Walters, 1999).

In this thesis, I reconstruct the biomass time series of lingcod in Georgia Strait, and examine the use of harvest refugia as a conservation tool for lingcod in Howe Sound. Computer simulations are used to compare alternative management policies to that of implementing marine protected areas. The following is a list of thesis objectives, where each objective is addressed in corresponding chapters.

Thesis Objectives:

Objective 1: To estimate the pre-fishery and present day biomass as well as the productivity of lingcod in Georgia Strait (statistical area 4B).

Objective 2: Estimate lingcod densities in several survey areas throughout Howe Sound using mark-recapture data, relative abundance data, and length frequency data.

Determine whether lingcod densities or age structure are significantly different in MPAs versus nearby fished areas.

Objective 3: To estimate movement rates of lingcod by using *in-situ* tagging data.

Objective 4: To evaluate the use of marine reserves as a conservation tool for lingcod under a range of hypotheses about fish movement and fishing effort dynamics. Compare the use of MPA's to alternative conservation measures, namely increase size limits and reduce fishing season.

Chapter I: Reconstructing the Georgia Strait Lingcod Stock

Introduction

This chapter deals with the estimation of unfished biomass and productivity of lingcod in statistical area 4B (Georgia Strait). Present-day biomass of lingcod stock in Georgia Strait is estimated using a stock assessment technique called Stock Reduction Analysis (SRA). This method, first proposed by Kimura and Tagart (1982), uses a catch time series to drive a model and estimate pre-fished biomass (B_o) and the initial slope of the stock-recruitment curve (k). The initial slope of the stock-recruitment curve describes stock productivity, and herein I will refer to k as the stock-productivity parameter. In this chapter, I will describe commercial lingcod fisheries in Georgia Strait, lingcod population parameters, an age-structured model, and the statistical method used to estimate two key population parameters (B_o and k). Part two of this chapter discusses results of SRA on lingcod in Georgia Strait. The stock productivity parameter estimated using this SRA model is used in a spatially explicit model presented in Chapter III.

Catch Statistics

Annual lingcod landings from the Georgia Strait lingcod fishery were used to estimate unfished biomass (B_o) and present-day biomass (B_{1998}). The lingcod fishery in Georgia Strait began in 1860, and catch statistics for this fishery were first recorded in 1889. Prior to 1927, lingcod was classified as a “cod-like fish” and grouped with rockfish species, however, at the time, lingcod was targeted and accounted for more than

90% of the landings (Cass et al., 1990). Three methods were used to harvest lingcod in Georgia Strait: a line fishery, a trawl fishery, and a longline fishery. The line fishery was responsible for the bulk of the catch (Figure 1.1). The trawl fishery started in the late 1940s and caught lingcod in trawlable grounds in area 4B. The longline fishery, which targeted rockfish (*Sebastes sp*), also harvested lingcod. In 1990, commercial lingcod fishing was closed in Georgia Strait (Statistical areas 13-19, 28, and 29; Beamish *et al.*, 1994). The closure was extended in 1991 to include Juan De Fuca Strait and Queen Charlotte Strait.

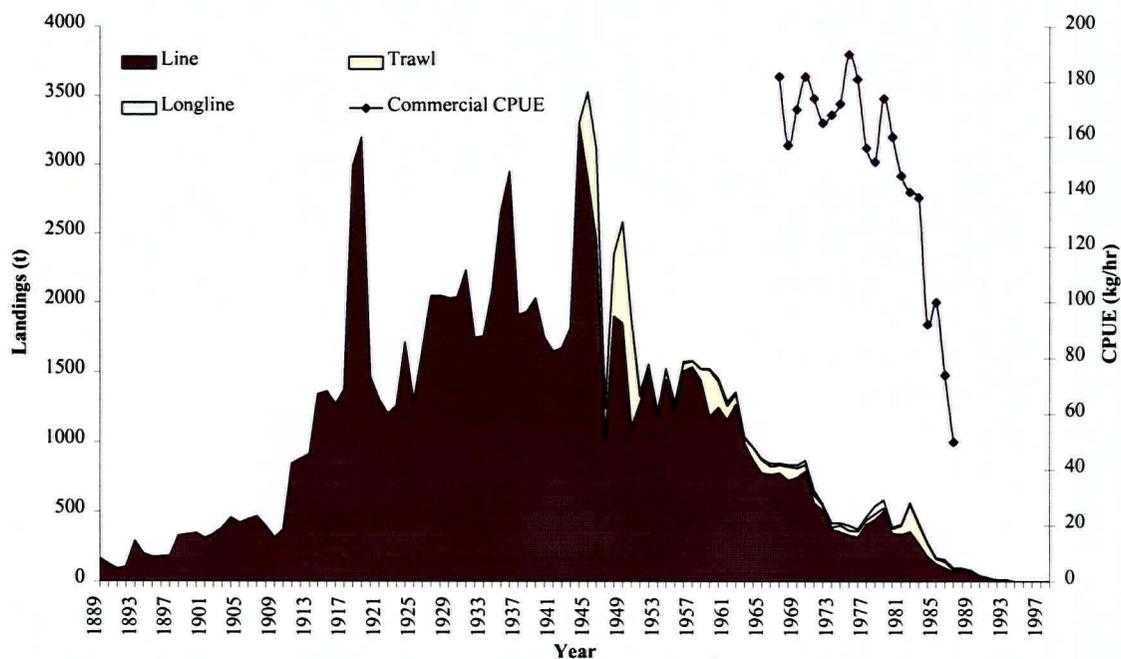


Figure 1.1. Estimated total lingcod landings removed by commercial fisheries from 1889 to 1998 in Georgia Strait. Catch per Unit Effort (CPUE) for line fisheries. Note that all areas represented in this graph were closed to commercial fishing by 1993. Sources: Beamish *et al.*, 1994; Cass *et al.*, 1990; Haist, 1994.

In 1993, all fishing grounds in area 4B (see Figure 1.2) were closed to commercial harvesting for lingcod.

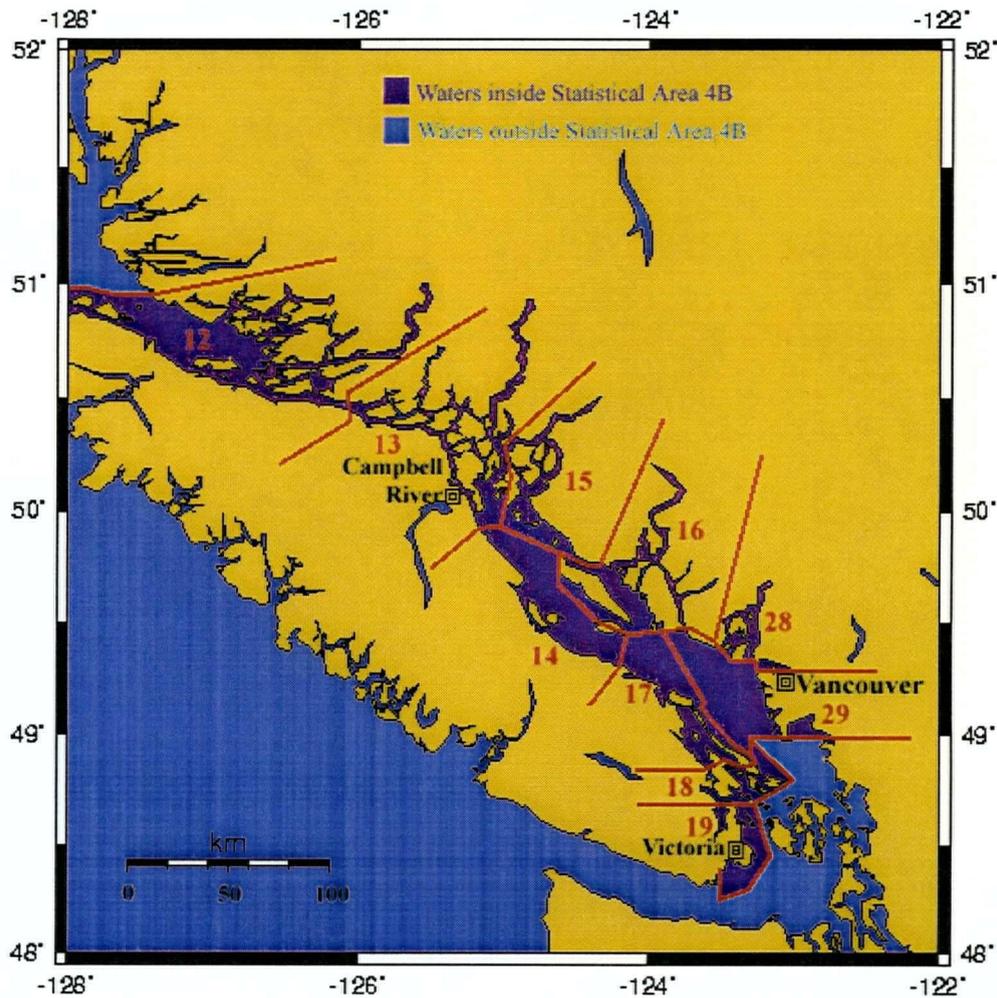


Figure 1.2. The dark blue region is statistical area 4B. The catch statistics from this region are used to drive SRA model. Population parameters estimated in the SRA model pertain to the lingcod stock in area 4B.

In the 1980s, sports fisheries have had a significant impact on lingcod stocks in Georgia Strait (Figure 1.3). Prior to 1980, no systematic data were collected on annual lingcod landings in sports fisheries; therefore, relative impacts on the stock from sports fisheries prior to 1980 are not known. Sports fishery landings in the 1990s have been less than 20% of the 1980s.

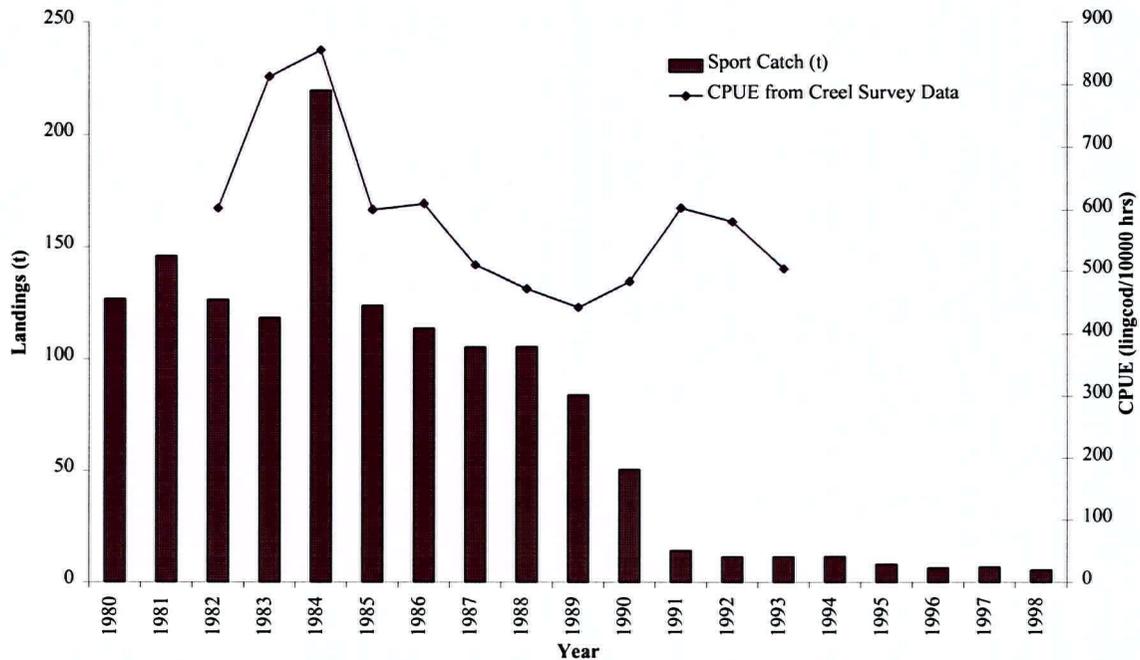


Figure 1.3. Georgia Strait sports-fishery landings for lingcod from 1980 to 1998 and relative abundance index calculated from creel survey data. Source: Beamish *et al.*, 1994; Haist, 1994; Nagy, 1999 (electronic communication).

Prior to 1990, sports fisheries were unregulated. After commercial fishing was closed, the sports fishery was restricted with a 58-cm volunteer size limit. In 1991 a seasonal closure was implemented. The fishery was restricted to the months of June to September, and the minimum size limit was increased to 65 cm. Daily bag limits of one fish/day and a voluntary limit of 10 fish per year were also implemented. After a recommending a total closure for lingcod in 1992 (Beamish *et al.*, 1994), annual bag limits of 10 fish per year became mandatory, and fishermen are now required to record their landings on the back of their fishing license. After these restrictions were implemented, the average size of lingcod landed increased from 58.3 cm in 1988 to 66.5 cm in 1993 (Beamish *et al.*, 1994).

These catch time series data are from the statistical area known as international statistical area 4B. It consists of 10 sub-areas, 3 of which are outside of Georgia Strait (Figure 1.2). Because the catch data cannot be separated into these areas, stock reconstruction analysis includes two sub-areas north and one sub-area south of Georgia Strait. The estimated unfished biomass and production parameters are for the lingcod stock within statistical area 4B, not strictly Georgia Strait alone.

Life History Information

Previous studies on lingcod growth have shown different growth rates for male and female lingcod (Cass *et al.*, 1990; Smith and McFarlane, 1994). Also, average weight-at-age for lingcod stocks inside Georgia Strait are small compared to off-shore stocks (Cass *et a.*, 1990). For stock reconstruction, I assumed a 50:50 sex ratio and use combined average weight-at-age in the SRA model (Figure 1.4). Under this assumption, I must also assume that mortality rates for both sexes are the same, and that sex ratios remains equal over time.

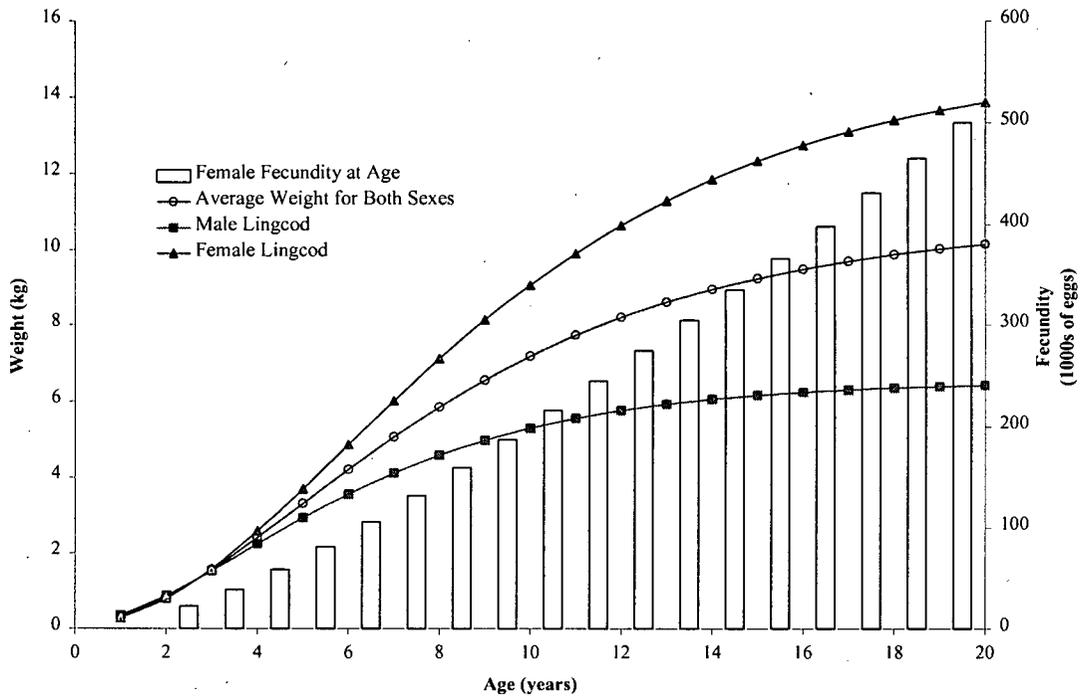


Figure 1.4. Estimated weight-at-age and fecundity-at-age for lingcod stocks in Georgia Strait. Source: weight-at-age from Smith and McFarlane, 1990; fecundity-at-age from Cass *et al.*, 1990.

The relationship between stock size and recruitment is unknown; however, there is a strong relationship between body size and fecundity (Cass *et al.*, 1990). Egg predators could have large effects on recruitment and populations even when exploitation rates are low. Female lingcod mature between 3-5 years of age (Cass *et al.*, 1990; Gordon, 1994). In the SRA model, a deterministic Beverton-Holt type stock-recruitment relationship was used to estimate annual recruitment. Recruitment is a function of egg numbers; therefore, the population age structure is important in determining annual egg production. Cass *et al.* (1990) estimated that female lingcod produce approximately 26 eggs per gram of body tissue. To simulate the effects of maturing between ages 3 and 5, I assumed that 3-year old fish produce 8 eggs per gram of body tissue. Four-year-old fish

produce 13 eggs per gram of body tissue, and 5+ year olds produce 26 eggs per gram of body tissue. The fecundity-at-age schedule shown in Figure 1.4 was used to predict annual egg production.

Effects of Fisheries Regulations on Lingcod Harvest

Management of the lingcod fishery over the course of the time series shown in Figure 1.1 included changes in minimum size limits. A minimum size limit of 58-cm was first implemented back in the 1940s for commercial fisheries only. In 1990, size limits were increased to 65-cm for the sports fishery. Because of changes in size limits, vulnerabilities-at-age schedules also changed over time. Prior to 1990, roughly 60% of the lingcod landed were below the minimum voluntary size of 58-cm (Figure 1.5).

- Average size of lingcod landed between 1985 and 1989 was $55.3 \text{ cm} \pm 12.3$ (one standard deviation). After 1990, average size of lingcod landed increased to $59.9 \text{ cm} \pm 13.3$ (one standard deviation). Again, roughly 60% of lingcod landed between 1990 and 1993 were below a minimum legal size of 65-cm (Figure 1.5)

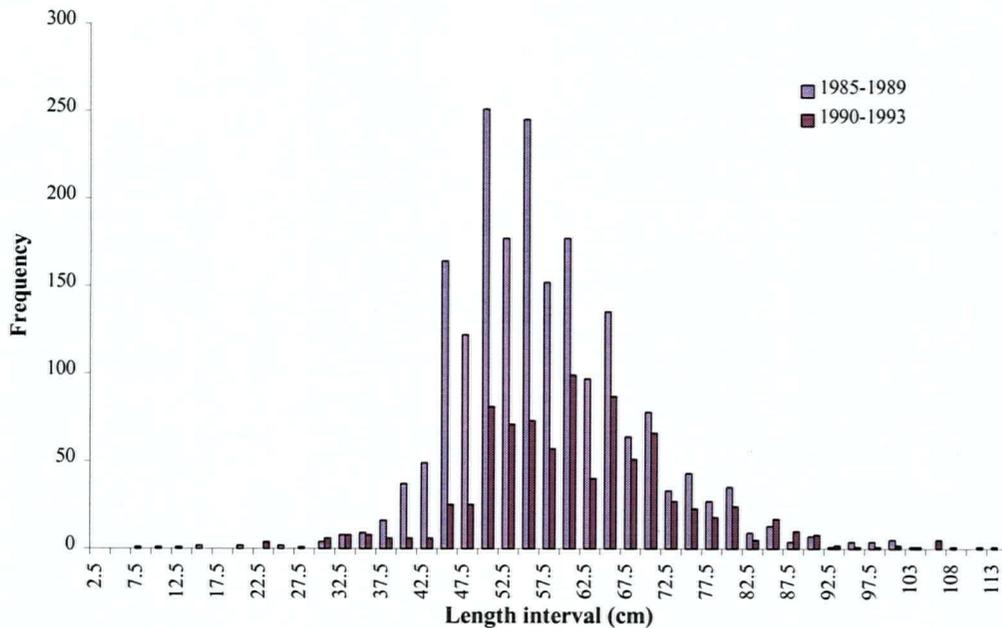


Figure 1.5. Length frequency distribution for lingcod caught in the Georgia Strait Sports fishery from 1985-89 and 1990-93. Data from Georgia Strait creel survey program (provided by Lynn Yamanaka, Department of Fisheries and Oceans).

Changes in length frequencies in the sport fishery catch are attributed to increases in minimum size limits. Beamish *et al.* (1994), reported higher discard rates by anglers after implementing a minimum size limit in 1990.

Length-frequency data from the sports fisheries catch were converted to catch-at-age data using the von Bertalanffy growth equation (Figure 1.6). Smith and McFarlane (1990) estimated parameters for the von Bertalanffy growth equation for Georgia Strait lingcod. Catch-at-age data were used to estimate the vulnerability-at-age schedules for the sports fishery. The addition of minimum size limits changes the vulnerability-at-age schedules.

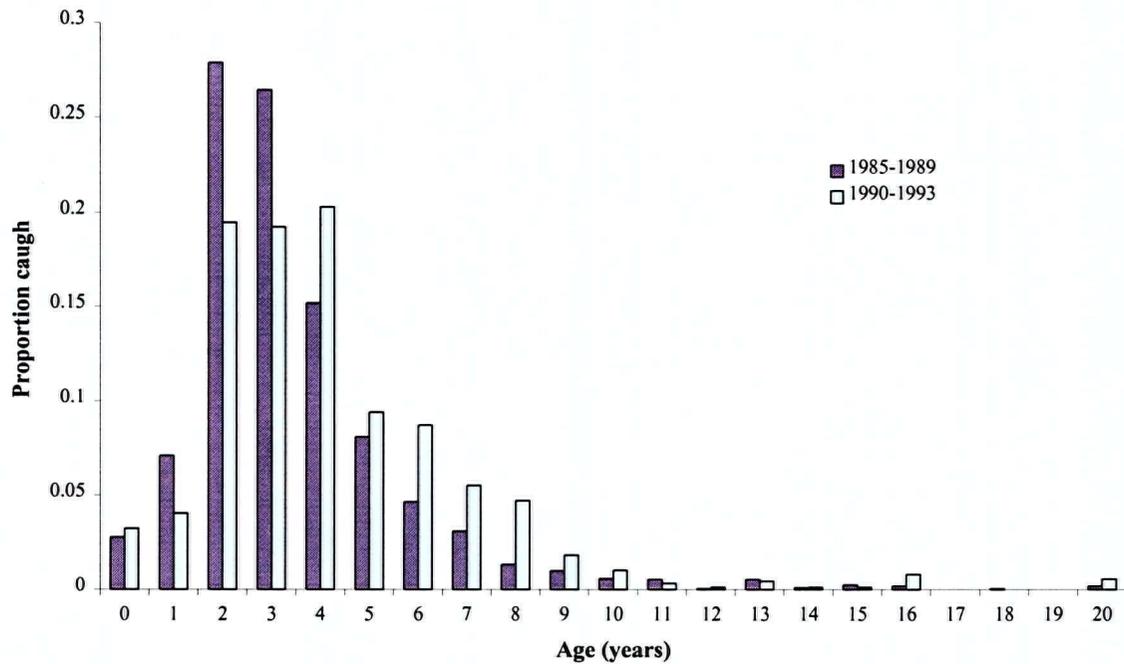


Figure 1.6. Catch-at-age data for lingcod caught in the Georgia Strait sports fishery between 1985-89 and 1990-93. Data converted from length frequency data in Figure 1.5 using the von Bertalanffy growth equation.

To represent historical management policies involving minimum size limits, vulnerability-at-age schedules were allowed to vary over time (Figure 1.7). In this stock reduction analysis I used a power function to represent gradual recruitment to the fishery over several age classes. The power function estimates a proportion of each age class that is vulnerable to the fishery. For the commercial fishery, Cass *et al.* (1990) estimated the age of recruitment to the fishery was between 5 and 6 years. The 50% age of vulnerability for the commercial fishery was 5.5 years (Figure 1.7). For the sports fishery, I used 3.4 years as the 50% age of recruitment before 1990 and 4.2 years after 1990 (Figure 1.7).

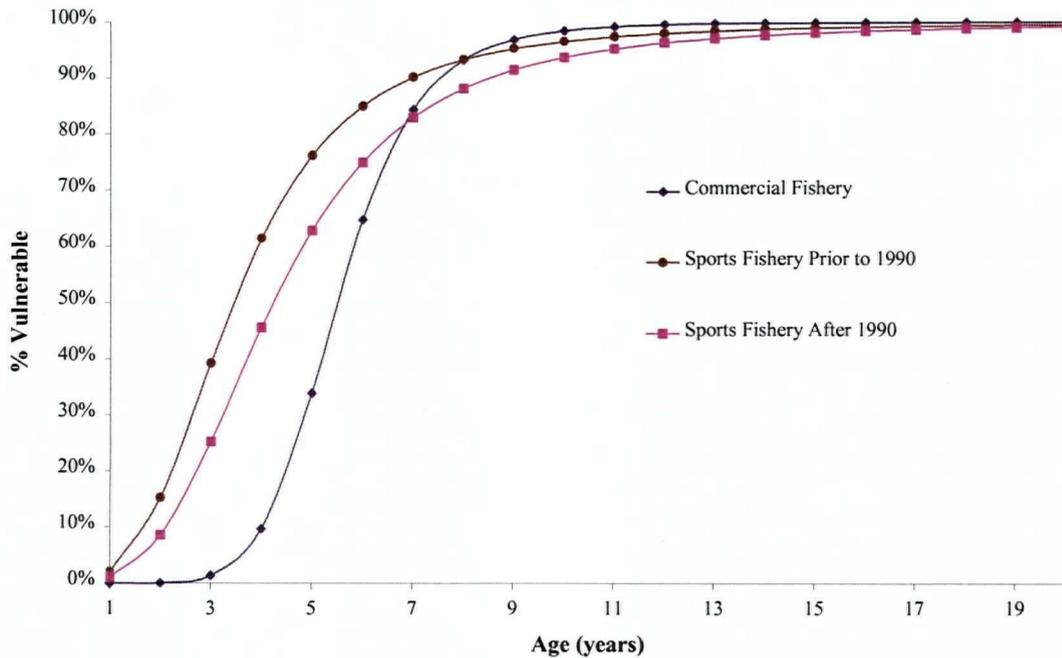


Figure 1.7. Vulnerability-at-age schedules used in the SRA model for commercial fisheries and sports fisheries before 1990 and after 1990 when a 65-cm size limit were imposed on the sports fishery.

Creel survey data set does not include information about the sex of each lingcod caught; therefore, I assumed an equal sex ratio in the sports fishery catch when estimating vulnerability-at-age schedules.

SRA Model Description

The SRA model was used to estimate two essential population parameters, the unfished biomass (B_0) and the initial slope of the stock-recruitment curve (α). The stock reconstruction was carried out using an age-synthesis model. I chose to use an age-structured model because of changes in vulnerability-at-age schedules over time.

A Beverton-Holt stock-recruitment relationship (equation 1.1) was used in the SRA model. Alpha corresponds to the initial slope of the stock-recruitment curve, and α/β corresponds to the asymptote of the stock-recruitment curve (Hilborn and Walters, 1992).

$$R = \frac{\alpha E}{1 + \frac{\alpha}{\beta} E} \quad (1.1)$$

The form of equation 1 expresses the number of recruits produced as a function of the total number of eggs (E) produced by the number of mature fish spawning. Beta is calculated, as a function of B_0 (*i.e.* is not a separate parameter to be estimated), by assuming equilibrium conditions prior to the start of the fishery. In the unfished state, I assume a steady state population; therefore, the equilibrium recruitment (R_0) is equal to:

$$R_0 = \frac{B_0}{\sum_a l_x \cdot W_a} \quad (1.2)$$

where B_0 is the unfished biomass, l_x is the survivorship vector, and W_a is the weight-at-age. The denominator of equation 1.2 is the unfished equilibrium biomass per recruit.

Under the equilibrium assumption, and using equation 1.2, I now have two points on the stock-recruitment curve (the origin $\{0,0\}$ and $\{B_0, R_0\}$). Given these two points and the natural survival rate (l_x schedule), equation 1.2 can be rearranged to calculate beta:

$$\beta = \frac{\alpha E_0}{\left[\frac{\alpha E_0}{R_0} - 1 \right]} \quad (1.3)$$

where E_o is the total number of eggs produced by population B_o and R_o is the equilibrium recruits for the given B_o population. The parameter alpha is required to estimate β , therefore, an estimate of α must be provided before using equation 1.3. Recall that α is the initial slope of the stock-recruitment curve. If $\alpha = R_o/E_o$ then β is undefined and recruitment is density independent. Therefore, alpha must be greater than R_o/E_o in order for the recruitment rate to have an asymptote at high stock sizes. It is better think of alpha as a multiple of R_o/E_o or $\alpha = k R_o/E_o$ (Walters and Bonfil, *in press*). In this formulation, the estimation procedure searches for an optimal k where $k \geq 1$ is the only constraint.

After using equations 1-3 to estimate the stock-recruitment parameters, it is now possible to predict annual recruitment in a dynamic model. In the SRA, the number of lingcod at age a at time t would propagate over time for a single fishery using equation 1.4 and 1.5:

$$N_{a+1,t+1} = S \cdot N_{a,t} \cdot (1 - U_t \cdot V_a) \quad (1.4)$$

and

$$N_{1,t+1} = \frac{\alpha E_t}{1 + \frac{\alpha}{\beta} E_t} \quad (1.5)$$

where S is the annual natural survival rate, U_t is the exploitation rate in year t , and V_a is the vulnerability at age a . The annual exploitation rate would be calculated as:

$$U_t = \frac{C_t}{\sum_a N_{a,t} \cdot W_a \cdot V_a} \quad (1.6)$$

where W_a is the weight-at-age, V_a is the vulnerability-at-age, and C_t is the total annual catch in the fishery.

There is a problem with using equation 1.5 to calculate the annual exploitation rate. The problem is that the catch from the sports and commercial fisheries come from different biomass pools. In the sports fishery, there is a larger vulnerable biomass than in the commercial fishery. Therefore, catches in the sports fishery will have an effect on the catches in the commercial fishery. Because of the differences in selectivity, an alternative approach is to use instantaneous mortality rates and simulate the dynamics using:

$$N_{a+1,t+1} = N_{a,t} \cdot e^{-Z} \quad (1.7)$$

where Z is the total instantaneous mortality rate, or the sum of the instantaneous natural mortality rate and the instantaneous fishing mortality rate. In this case, the instantaneous fishing mortality rate is the sum of the instantaneous sports fishery mortality and the instantaneous commercial fishery mortality. The instantaneous fishing mortality rates are calculated as follows:

$$F_i = -Ln\left(1 - \frac{C_i}{B_i}\right) \quad (1.8)$$

where F_i is the instantaneous fishing mortality rate for fishery i , C_i and B_i is the catch and vulnerable biomass ($\sum_a N_{a,t} V_{a,t} W_{a,t}$, for fishery i) in fishery i , respectively. By

calculating the fishing mortality rates separately, equation 1.7 is now written as:

$$N_{a+1,t+1} = N_{a,t} \cdot e^{-\left(M + \sum_i F_i\right)} \quad (1.9)$$

where M is the instantaneous natural mortality rate.

Estimating Key Population Parameters (B_0 and k)

The third part of the SRA is fitting an observation model to the relative abundance data shown in Figures 1.1. The first step is creating a set of predicted observations from the age-structured model. Next, a likelihood kernel is used to evaluate the parameter combinations given the observed and predicted observation data. This approach, assigning a credibility measure for an alternative parameter combinations, is a Bayesian approach (Walters and Ludwig, 1994). In this approach the parameter combinations are evaluated under Bayes theorem:

$$P(\text{parameter} | \text{data}) = \frac{P(\text{data} | \text{parameter}) \cdot (\text{Prior probability of the parameter value})}{\text{Total Probability of the data}} \quad (1.10)$$

which states that the probability of a parameter value given the data is equal to the probability of the data given the parameter value multiplied by a prior probability of the parameter value. The numerator in equation 1.9 is normalized by the total probability of the data, which is the sum over all parameter value likelihoods multiplied by prior probabilities. In this SRA I used a uniform prior for the unfished biomass and the initial slope of the stock recruitment curve.

The likelihood function used to assign a credible value for each parameter combination is:

$$L = e^{-\frac{1}{2}SS/\sigma^2} \quad (1.11)$$

where SS is the sum of squared differences between the relative abundance data (Y_i) and the scaled biomass ($q B_i$).

$$SS = (Y_i - q \cdot B_i)^2 \quad (1.12)$$

The catchability coefficient (q) is estimated from the relative abundance data and the predicted biomass by:

$$q = \frac{\sum Y_i B_i}{\sum B_i^2} \quad (1.13)$$

The variance (σ^2) in equation 1.11 is estimated as a sum of the variance in the relative abundance data and the variance in the biomass estimate:

$$\sigma^2 = (\sigma_{cpue}^2 + q^2 \sigma_{biomass,t}^2) \quad (1.14)$$

where the variance in the relative abundance data (σ_{cpue}^2) is approximated by using the residuals between the cpue and the scaled biomass. The variance in biomass at time t ($\sigma_{biomass,t}^2$) is estimated using Monte Carlo simulations. A process error term is added to the recruitment function in equation:

$$R = \frac{\alpha E}{1 + \frac{\alpha}{\beta} E} \cdot e^{Cv \cdot W_t} \quad (1.15)$$

where W_t is a log-normally distributed number between with a mean of 0 and a standard deviation of 1, and Cv is the coefficient of variation. In Monte Carlo simulations the Cv was set at 0.5.

Calculating likelihoods using equation 1.11 was a two step process. First, Monte Carlo simulations were carried out to estimate the variance in the biomass time series.

Secondly, using a range of hypotheses about unfished biomass and stock productivity, likelihoods were evaluated over all parameter combinations.

To search for the maximum likelihood estimate over the two key stock parameters (B_0 and α), a 2 dimensional grid was set up with a range of B_0 values as one axis and a range of k values as the second axis. The model was run for each combination of B_0 and k and the likelihood of the data given the parameter values was calculated. To ensure the model is working correctly I tested the model using a set of fake relative abundance data.

Testing the Model Using Fake Observation Data

It is good practice to test assessment models with fake data (Hilborn and Mangel, 1997), *i.e.*, to use the model with known parameter values and generate a fake relative abundance time series. After generating fake data, using a deterministic model (*i.e.* no observation error), the least squares criterion will equal zero and the estimated parameters equal the values used to generate the fake data. A second test is to generate fake data with observation error incorporated, and the following paragraphs describe this observation error test.

A set of fake relative abundance (Y_t) data was generated using a biomass time series and equation 1.21:

$$Y_t = q \cdot B_t \cdot e^{(Cv \cdot v)} \quad (1.21)$$

where Cv is the coefficient of variation, and v is a normally distributed number with a mean of zero and a standard deviation of one. The biomass time series was generated using the age-synthesis model with $B_0 = 40,000$ t, and $k = 3$. The coefficient of variation was set at 0.2. After generating the fake observation data, the correct parameter set was searched for over a 50 X 50 grid. The upper and lower bounds for B_0 were set at 50,000 t and 30,000 t, respectively. The upper and lower bounds for k were set at 5 and 1, respectively. The results of estimating B_0 and k from the fake data are shown in Figures 1.8 and 1.9, respectively.

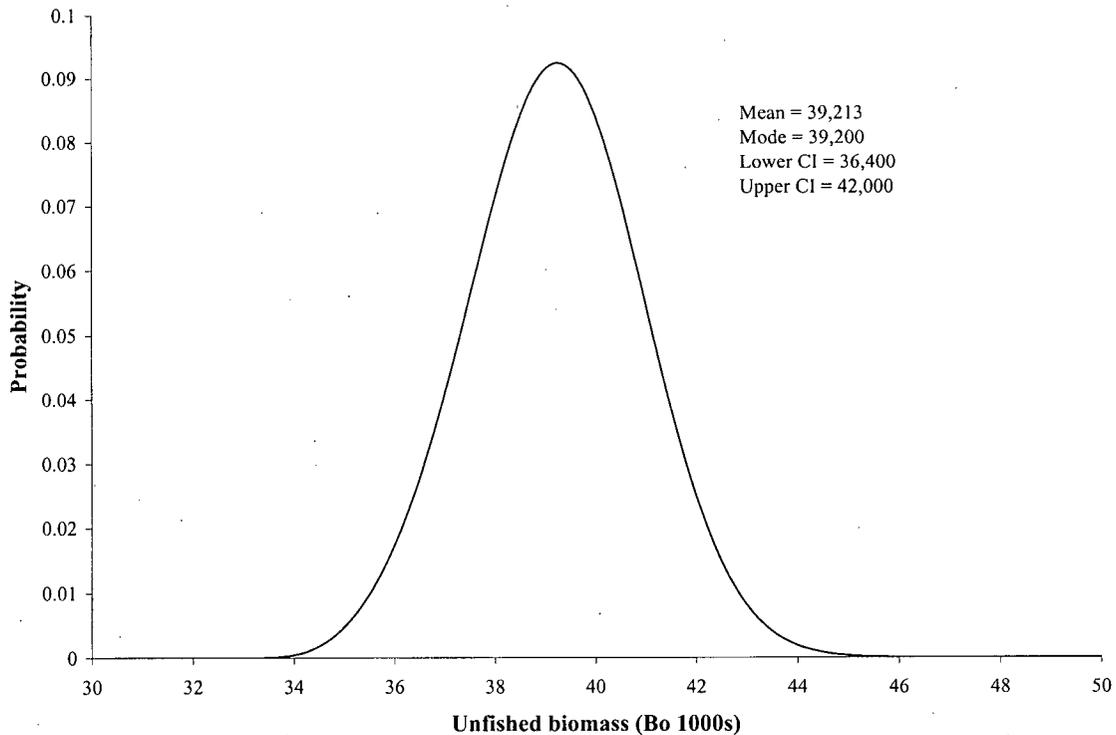


Figure 1.8. Marginal posterior probability distribution for unfished biomass when fitting the model to fake observation data.

The true unfished biomass in this simulation is 40,000 t, and the mean of the marginal posterior distribution for the unfished biomass (Figure 1.8) is 39,213 t. The difference between the true B_0 value and the predicted B_0 value is 787 t, an under-estimate of 1.97%. The true initial slope of the stock-recruitment curve is 3 and the mean of the posterior distribution in Figure 1.9 is 3.26, or an under-estimate of 7.98%.

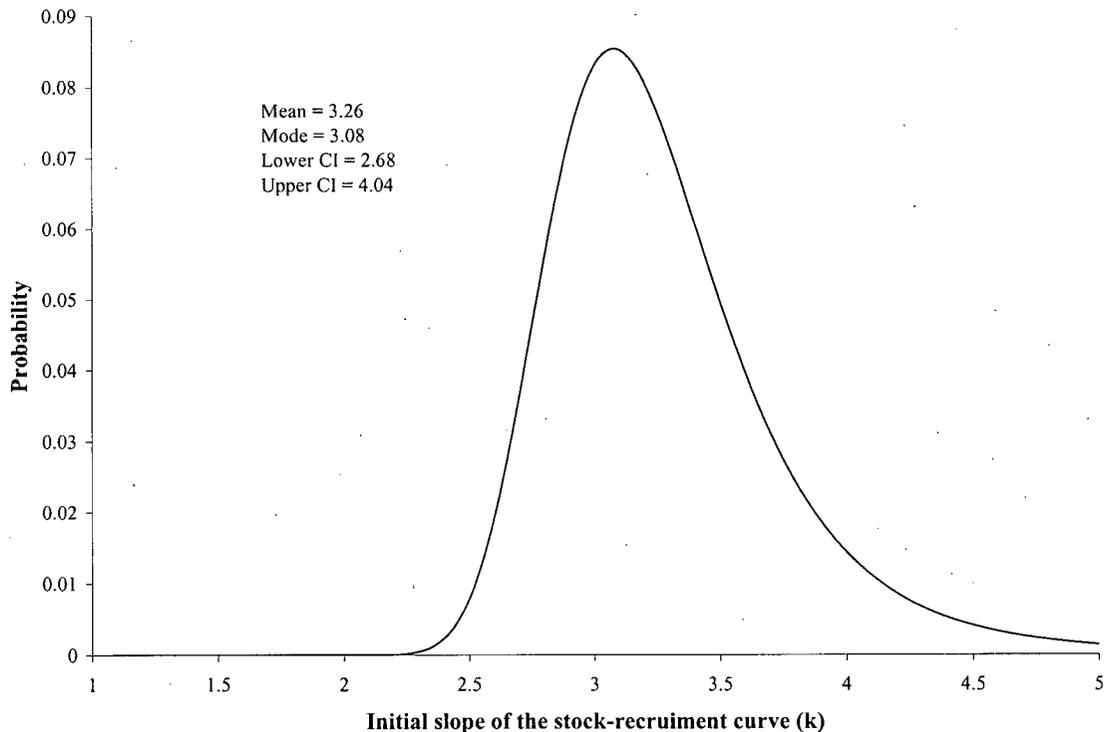


Figure 1.9. Estimating the stock productivity using marginal posterior probability distribution for the initial slope of the stock recruitment curve (α).

The results of the observation error test did not predict the exact parameter combination used to generate the fake data. This failure to predict the correct parameter combination is a result of the grid search method used to calculate the joint likelihood distribution, rather than bias in the estimation methods. When searching for the correct parameter combination in a two-dimensional matrix, it is easy to “miss” the optimal

parameter combination (Mangel and Adler, 1994; Hilborn and Mangel, 1997). The parameter values in the matrix may lie on either side of the more “correct” parameter. If there is a bias in the estimation methods, the model will continue to over-estimate or under-estimate a parameters value.

Reconstructed Lingcod Population in the Strait of Georgia

The joint likelihood surface for B_0 and α consists of a long ridge that is open-ended, *i.e.* neither parameter has an upper bound (Figure 1.10). The likelihood surface indicates a trade-off between the unfished biomass and productivity of the stock. The relative abundance data are insufficient to accurately describe the relationship between the two stock parameters. In other words, the data are equally likely to come from a small productive stock or a large unproductive stock when assuming catch and effort data are proportional to the stock size.

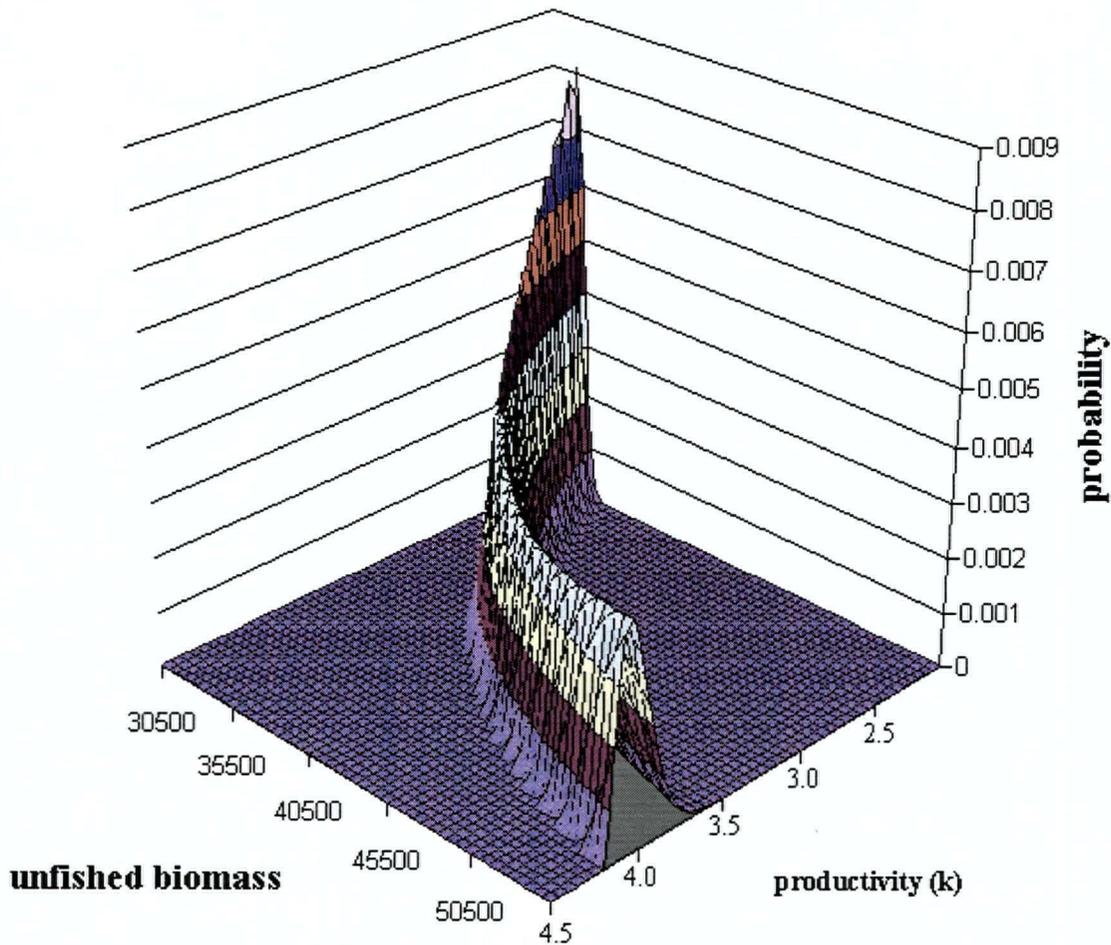


Figure 1.10. Joint likelihood profile for the two key population parameters being estimated by the SRA model (B_0 and the initial slope of the stock-recruitment curve).

The marginal posterior distribution for the unfished biomass was calculated by taking the sum of the joint probabilities for every hypothesis about stock productivity (Figure 1.11). Unfortunately, the commercial CPUE data are insufficient to estimate an upper bound for unfished biomass. The lower bound of the unfished biomass is roughly 34,000 t (Figure 1.11), and the mode of the marginal distribution is roughly 41,500 t.

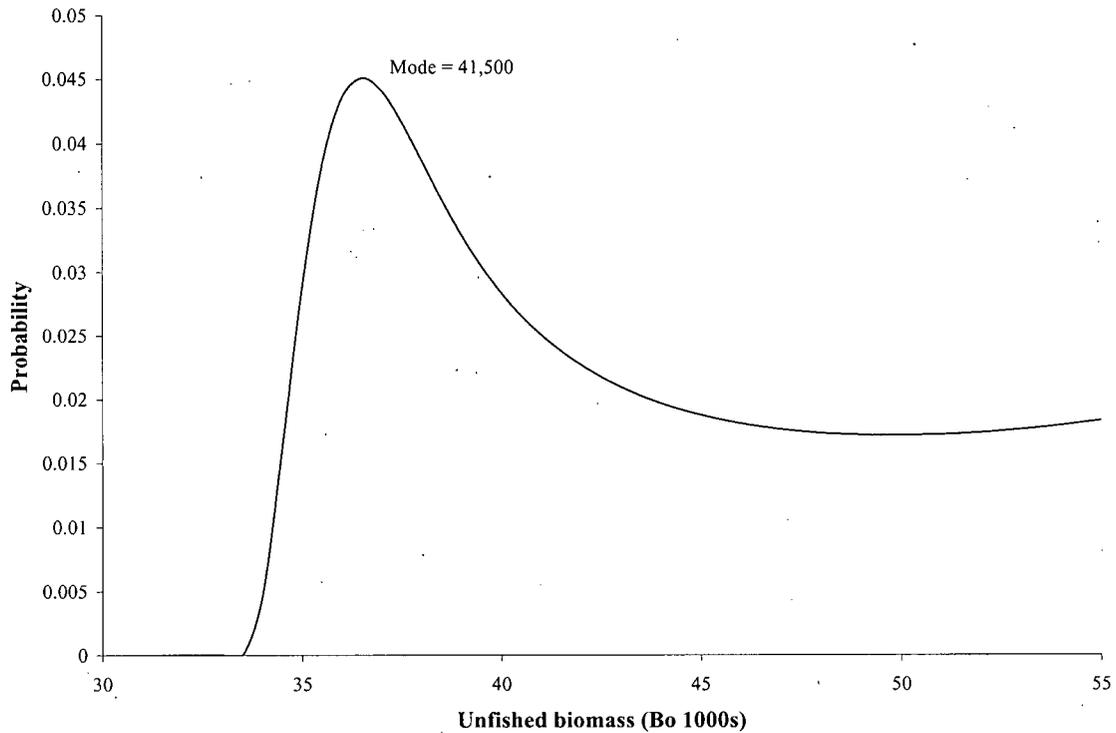


Figure 1.11. Marginal posterior probability distribution for unfished biomass (the biomass before the start of the fishery) in Georgia Strait. The peak of the distribution corresponds to the mode (41,500 t).

The marginal distribution for k is calculated just as the marginal for B_o , except in this case the distribution is integrated over all hypotheses about B_o . Again, the data are insufficient to estimate an upper bound for k , however, the lower bound, is roughly 1.76 (Figure 1.12).

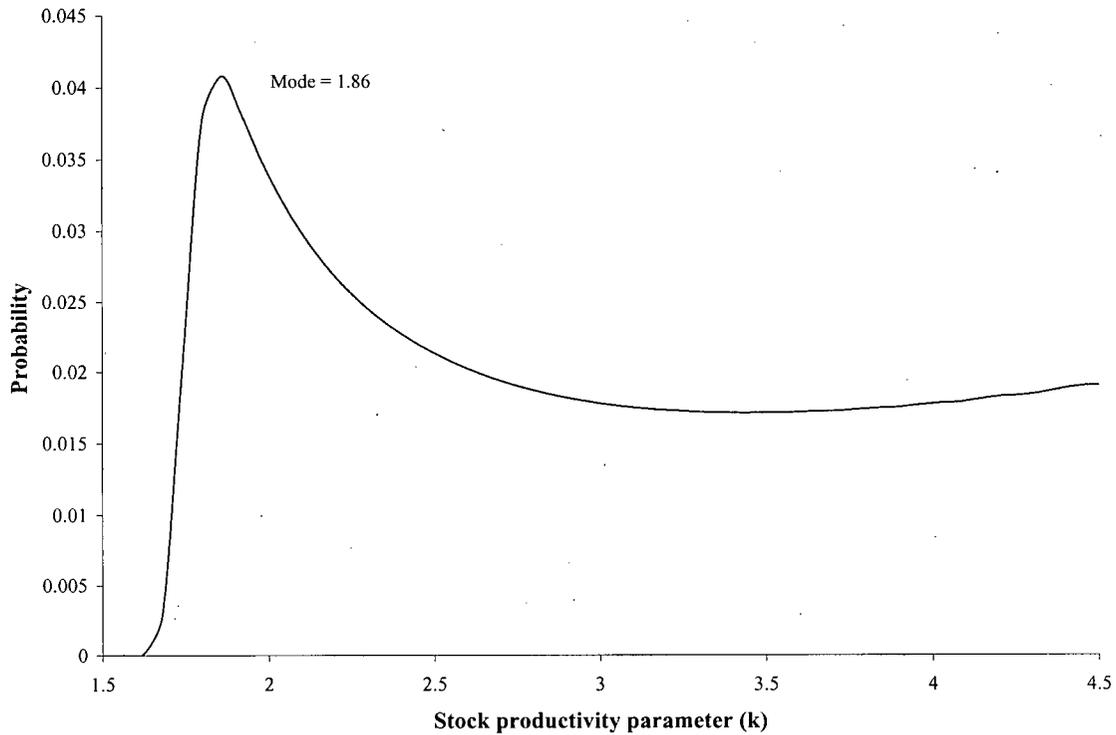


Figure 1.12. Marginal posterior probability distribution for the initial slope of the stock-recruitment curve for the lingcod stock in the Strait of Georgia. The mode of the distribution is 1.86.

Using the results of the parameter estimation procedure, the stock was reconstructed using the modes of the marginal posterior distributions. Two separate reconstructions were carried out, one using the maximum likelihood estimate for B_0 (Figure 1.13) and one using the maximum likelihood estimate for k (Figure 1.14). Both the best estimate of B_0 or k was used as a leading parameter, and the other parameter was searched for using a least squares criterion between biomass and CPUE/q. Model outputs consisted of ratios between the unfished biomass and present day biomass, maximum exploitation rates, minimum biomass observed and the catchability coefficient (Table I.I).

Table I.I. SRA model outputs and values obtained when using the best estimate for unfished biomass or the best estimate of stock productivity. Best estimates refer to the modes of the marginal distributions in Figures 1.11 and 1.12.

SRA Model Outputs	Use Mode for B_o Estimate	Use Mode for k estimate
Unfished Biomass B_o (t)	<u>41500</u>	47842
Stock Productivity k	2.435	<u>1.86</u>
Catchability q	0.0474	0.0383
B_{1998} (t)	5102	4626
B_{1998}/B_o	0.123	0.097
Minimum Biomass (t)	2175	2700
B_{min}/B_o	0.0524	0.0565
Year minimum biomass occurred	1984	1986
Max Commercial Exploitation Rate	0.54	0.37
Max Sport Exploitation Rate	0.14	0.11

The estimated biomass of lingcod in statistical area 4B (Georgia Strait area) in 1998 ranges between 4,600 t and 5100 t, a mere 9.7% or 12.2% of the estimated unexploited biomass. In the 1980s, the stock was estimated to be at its lowest biomass, between 2,175-2700 t. Using the low biomass estimate (Table I.I), the maximum exploitation rate for commercial fisheries was 0.54, and 0.14 for sports fisheries. However, maximum commercial exploitation rates are much lower using maximum likelihood estimate for k . Recall that prior to 1980 there was no estimate of the impact the sports fishery on the lingcod stocks in Georgia Strait. As it turns out, the sports fishery was having a significant affect on the lingcod stocks in Georgia Strait (Figure 1.13 and 1.14). Annual exploitation rates for the sports fishery peaked in 1984 at 14%. Since 1991, the estimated exploitation rates for the sports fishery are less than 0.5% per year. In general, if the unfished biomass was low, the stock productivity must be higher in order to sustain the observed catches. Alternatively, if the unfished biomass was very large, then the procedure for estimating parameters results in a low estimate of stock productivity in order to explain the observed CPUE data.

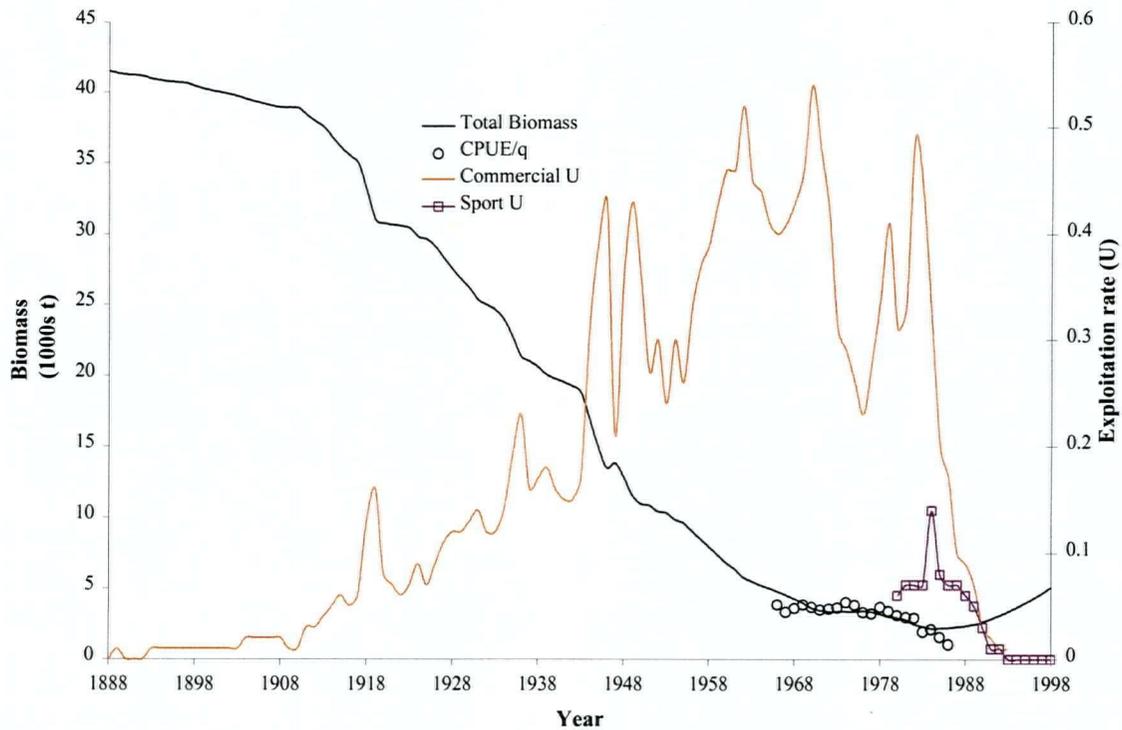


Figure 1.13. Reconstructed lingcod stock in Georgia Strait, annual exploitation rates for commercial and sports fishery, and the scaled commercial CPUE data using $B_o = 41,500$ t. This reconstruction used the mode of the marginal posterior distribution for the unfished biomass.

There are two important differences in Figure 1.13 and 1.14. First, the smaller unfished biomass in Figure 1.13 results in higher exploitation rates. As a result, stock productivity is higher. The second important difference is the difference in B_{1998}/B_o ratios. In Figure 1.13, the ratio is higher (12.3%) because the stock recovers faster due to higher productivity and in Figure 1.14 the ratio is lower (9.7%) do to larger initial biomass and a slower rate of recovery.

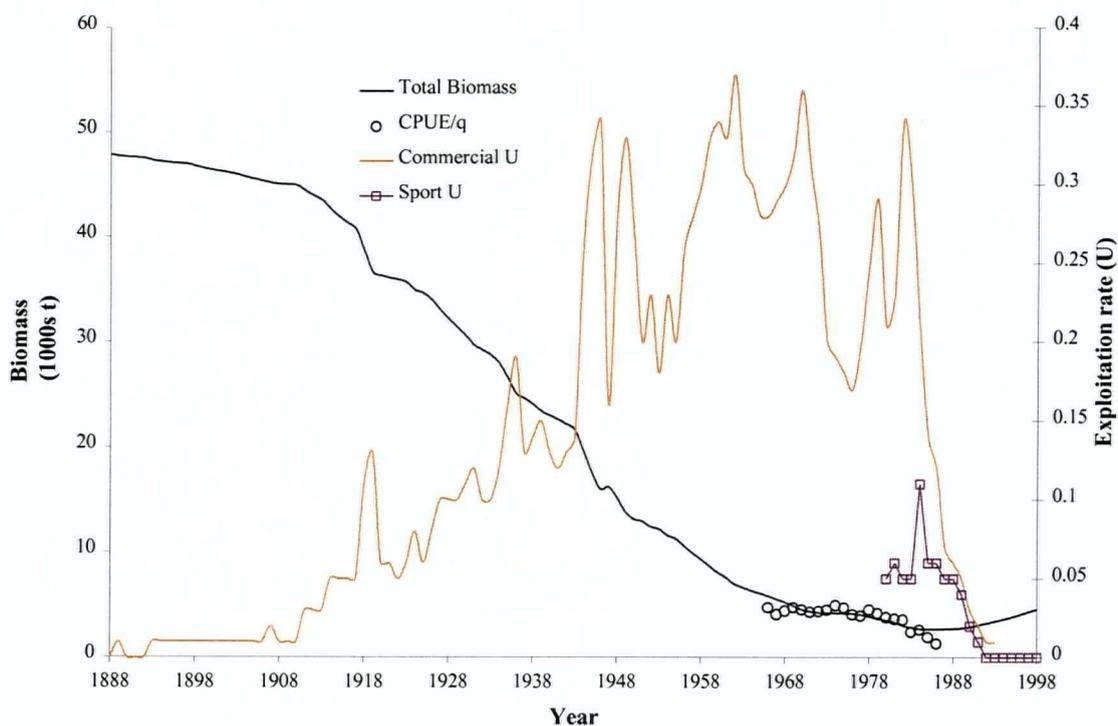


Figure 1.14. Reconstructed lingcod stock in Georgia Strait, annual exploitation rates for commercial and sports fisheries, and the scaled commercial CPUE data using $k = 1.86$. This reconstruction was carried out using the mode of the marginal distribution for the stock productivity parameter.

The estimated biomass time series shown in Figure 1.13 and 1.14 were generated assuming that natural survival rates, growth curves, and vulnerability-at-age schedules are correct, and more importantly have remained constant (or known in the case of vulnerability-at-age schedules) over the entire 110-year time series. The next section deals with how sensitive the model is to changes in these stock parameters.

Sensitivity Analysis

In the stock reduction analysis, the relative abundance data were used to estimate the unfished biomass and the initial slope of the stock-recruitment curve. So far, I have

assumed no uncertainty in other parameters such as the natural survival rates, growth parameters, and vulnerability-at-age schedules. To address uncertainty in these stock parameters, a sensitivity analysis was carried out. The optimal unfished biomass and recruitment slope parameters were calculated using a least-squares criterion while changing natural survival rates, weight-at-age and vulnerability-at-age schedules. The natural survival rate was changed by $\pm 10\%$ and the growth parameters (L_{inf} and k) were changed to alter the weight-at-age schedules. To increase the weight-at-age, L_{inf} was increased by 5%, k was reduced by 10% and vice versa for the decrease in weight-at-age scenario. The 50% vulnerability-at-age parameter was changed by ± 1 year. The results of the sensitivity analysis are shown in Tables I.II and I.III.

Table I.II. Estimated B_0 and α values for given changes in survival, vulnerability and growth parameters. Also included are the 1998 biomass predictions and the percent difference between the unfished biomass and present biomass.

Simulation	B_0 (t)	k	B_{98} (t)	B_{98}/B_0 (%)
Baseline	41,500	2.435	5,103	12.30
Survival (-)	42,070	1.924	4,723	11.23
Survival (+)	47,088	2.404	4,565	9.70
Vulnerability (-)	43,117	2.362	4,820	11.18
Vulnerability (+)	44,259	2.104	4,946	11.18
Growth (-)	52,527	1.992	5,566	10.60
Growth (+)	44,684	2.713	6,863	15.36

Results of the sensitivity analysis suggest that the growth parameters will have the largest impacts on estimation of the present day biomass. This is particularly true, if the growth rate of lingcod is under-estimated (Table I.III). A 10% increase in growth rate results in a 34% increase in the present day biomass. If lingcod grow faster then less biomass is required to sustain the observed catches. Vulnerability-at-age schedules have the least impact on estimation of biomass and stock productivity parameters. Changes in

the vulnerability schedule either increases or decrease the proportion of vulnerable stock. As the proportion of the vulnerable stock increases, the total biomass pool and or productivity of the stock must increase to sustain the observed catches. Conversely, as the vulnerable portion decreases, the stock can be less productive or smaller to sustain the observed catches. Decreases in the natural survival rate result in a smaller less productive stock and increases in natural survival rate result in a larger more productive stock. This may seem counter-intuitive, however, think of how longer-lived fish have more opportunities to reproduce over a lifetime and contribute more offspring.

Table I.III. Percentage changes in the estimated biomass (B_0 and B_{98}) and k values from the baseline prediction in Table I.II.

Simulation	B_0 (t)	K	B_{98} (t)
Survival (-)	1.37	-20.99	-7.45
Survival (+)	13.46	-1.27	-10.53
Vulnerability (-)	3.90	-3.00	-5.55
Vulnerability (+)	6.65	-13.59	-3.07
Growth (-)	26.57	-18.19	9.07
Growth (+)	7.67	11.42	34.49

Based on the results of the sensitivity analysis, the parameters estimated from the model output are most sensitive to growth parameters, then survival parameters and least sensitive to vulnerability schedules.

Discussion

The objective of the stock reduction analysis was to estimate the pre-fishery biomass, the present day biomass and productivity of lingcod in Georgia Strait. The best estimate of the unfished biomass ranges between 47,000 t and 58,000 t with a mean

estimate of 48,800 t. The best estimate of the B_{98}/B_0 ratio ranges between 0.097 and 0.123, or, in other words, the lingcod stock in Georgia Strait has apparently been reduced by 88% to 90% from its unexploited state 110 years ago. The productivity of the stock is a measure of how resilient the stock is to over-fishing, and the initial slope of the stock-recruitment curve quantifies this. The parameter α in equation 1.1 corresponds to the initial slope of the stock-recruitment curve, and its biological interpretation in this application is the maximum egg survival rate as spawning stock approaches zero, or a multiple (k) of egg survival rate in a steady state population. In this assessment of Georgia Strait lingcod, the estimated k ranged between 1.86 and 2.44.

The estimate of unfished biomass in this problem is very sensitive to the estimate of stock productivity. The only empirical way to obtain this parameter (k) is to over-fish the stock (Walters and Bonfil, 1999; Hilborn and Walters, 1992). Ideally, in assessing stock-recruitment relationships there should be a wide range of spawning stock biomass present in the stock-recruitment time series data to make direct inferences about stock productivity (Myers and Barrowman, 1996; Hilborn and Walters, 1992). Although no direct estimates of the relationship between spawning stock size (or egg production) and recruits are available for the Georgia Strait lingcod, the data suggest that the stock has been fished down sufficiently to indirectly estimate the initial slope of the stock-recruitment curve. Myers *et al.* (1997) estimated the same k parameter in stock-recruitment curves for 20 Atlantic Cod (*Gadus morhua*) stocks and found k to range between 1.9 and 23.1 recruits per spawner, and in 18 of the 20 stocks $k < 10$. In comparison, the estimated k parameter for Georgia Strait lingcod appears to be on the low side (an unproductive stock, $1.86 < k < 2.44$) of the range found by Myers *et al.* (1997).

If we examine the commercial catch series (Figure 1.1), the annual landings had fallen substantially in the early 1970s from the historical highs that were observed in the 1940s and 1950s. Small increases in k usually result in a rebuilding of the stock during the period of declining annual catches, as long as the initial unfished biomass remains constant. Based on the sensitivity of the k parameter, and the comparison of k to the work of Myers *et al.* (1997), I suspect that this SRA approach is under-estimating the productivity of the stock. As a consequence of under-estimating the productivity of the stock, the model compensates and over-estimates unfished biomass.

The biomass time series shown in Figure 1.12 and the parameters used to predict the biomass time series are a function of the relative abundance data. In the case of the Georgia Strait lingcod, the only relative abundance data comes from the commercial catch/effort statistics. Catch per unit effort data in general is a very poor index of relative abundance (Walters, 1986; Hilborn and Walters, 1992) and Walters and Ludwig (1994) “flatly recommend that catch/effort data never be used as a direct abundance index (assumed to be proportional to stock size)”. In the case of lingcod, the relationship is hyperstable (CPUE remains high at low stock sizes), because of the tendency for lingcod to aggregate in rocky reef areas (Cass *et al.*, 1990). A hyperstable relationship between CPUE would result in an over-estimation of stock productivity because the CPUE data would suggest a more abundant stock.

One other possibility that would have an effect on estimation of k is the relationship between lingcod and its predators or prey. As demonstrated in the sensitivity analysis, a decrease in the natural survival rate (through increases in predators or loss of prey) results in less productive stock (Table I.II and I.II). Smith *et al.* (1990) have

suggested that natural mortality rates on lingcod in Georgia Strait increased in the 1980s compared to earlier studies conducted in the 1950s. Present estimates of harbour seal, *Phoca vitulina*, consumption on lingcod are 294 t per year in Georgia Strait (Olesiuk *et al.*, 1990; cited in Smith, *et al.*, 1990). Also, roughly 70% of the lingcod diet consists of herring, *Clupea pallasii*, (Cass *et al.*, 1990). The Georgia Strait herring fishery collapsed in 1967 and was closed by the Federal government to allow recovery (Stocker, 1993). During the late 1960s and early 1970s the herring stocks in Georgia Strait were estimated to be at an all time low. At the same time, harbour seal populations in Georgia Strait were also at an all time low, estimated at 2000 seals in 1973 (cited in Smith *et al.*, 1990). The harbor seal population steadily increased to roughly 13,000 in 1988 (Olesiuk *et al.*, 1990). With the collapse of herring and increases in harbor seal abundance lingcod productivity should have declined.

In the age-synthesis approach to doing stock reconstruction, there is an underlying assumption that is apparent when using catch time series similar to Figure 1.1. This assumption is that exploitation rate is strictly a function of stock size (equation 1.6). In practicality, exploitation rate is a function of stock size, fishing effort, and catchability or area swept (*i.e.*, $F = e^{-qEB}$). In a retrospective analysis, however, fishing effort is not required to back calculate the stock size. This is similar to virtual population analysis (VPA) where catches-at-age are added back into the stock. In a forward synthesis, catches are subtracted from the population as a mortality rate (equation 1.8). Catch-at-age data are not required for a forward reconstruction; therefore, recruitment anomalies do not appear in the catch nor can they be estimated (in the absence of catch-at-age data). A catch time series with increasing catches followed by declining catches results in a

time period where the stock either continues to decline (assuming exploitation rates continue to increase) or recovers and exploitation rates decrease. In contrast, a catch series that only increases over time will always result in increasing exploitation rates.

Conclusion

The estimated biomass of lingcod in statistical area 4B in 1998 is roughly 5000 t. This is approximately 10% of the estimated biomass in 1888, the year before commercial catch statistics were first recorded for lingcod. The productivity of the lingcod stock is low in comparison to Atlantic cod stocks. An increase in predators or a decrease in prey (or both) would also have an impact on the productivity of the stock. In the absence of any fishery-independent data, I would be cautious about using results of this analysis for management decisions because the findings are based strictly on fisheries catch statistics. Finally, in the absence of age-structure data, recruitment anomalies are not incorporated into the model; therefore, it is assumed that exploitation rate is strictly a function of vulnerable biomass and the vulnerable proportion of the stock remains constant over time.

Chapter II: Lingcod Populations in Howe Sound

Introduction

This chapter deals specifically with the lingcod populations in Howe Sound. First I will describe the *in-situ* tagging methods used to mark lingcod in the field, the Roaming Diver Technique (RDT) used to re-sight tagged lingcod and measure all lingcod *in-situ*. The second part of the chapter is the results of a length-frequency analysis of lingcod inside and outside of the two Marine Protected Areas (MPAs). Following are the results of the mark-recapture data, including estimates of lingcod density in 11 different locations in Howe Sound, and two locations outside of Howe Sound. Finally, a brief discussion about the response of lingcod to the two MPAs, and proposed explanations for the observed length frequency and density differences observed.

Currently there are two MPAs in Howe Sound and one in nearby Lighthouse Park. The oldest is Porteau Cove, located in the northern end of Howe Sound. The most recent MPA is located at Point Atkinson, and Whytecliff Park is Canada's first official "no-take" MPA. Many studies have shown that densities and age-structure of fish or invertebrate populations within a reserve area are significantly different than nearby fished areas (Alcala, 1988; Ballantine, 1991; Russ and Alcala, 1996b; Horwood *et al.*, 1998). In this chapter I address the question: Are densities or age-structure of lingcod populations in Howe Sound MPA's greater or older, respectively, in comparison to nearby fished areas? To answer this question, I use a mark-recapture type study to estimate local abundance and density. Length frequency data are used to compare the age-structures between locations and within locations.

Another question about the use of MPA's as a conservation tool pertains to the movement rates across the boundary of the reserve area (Russ and Alcala, 1996; Zeller and Russ, 1998; Walters *et al.*, 1999). Small-scale seasonal movement patterns in lingcod have not been documented in the literature. Attempts have been made to quantify movement rates using simple tagging studies and complex ultrasonic tagging studies. Several studies have documented seasonal differences in depth distributions for female lingcod (Cass *et al.*, 1984; Gordon, 1994). To address the question of ontogenetic and seasonal shifts in habitat use, I look at the changes in length frequency over time within each of the study locations. During the spawning season, large lingcod (presumably mature females) are expected to move inshore to spawn, and subsequently return to deeper waters offshore. Recruitment of smaller (presumably juvenile or immature lingcod) to the near-shore has yet to be documented through field data, though it must occur for the lingcod life cycle to be closed.

Methods for *In-situ* Mark-Recapture Studies on Lingcod

In traditional mark-recapture studies for estimating abundance there are three components to the method. First, there is the marking event, where the study animal is given a permanent mark. Second is the recapture event, where the population is sampled and the proportion of tagged animals to untagged animals is recorded. The third component is analysis of the data, where the data are used to estimate population parameters such as natural mortality rate, growth rate, abundance, and movement rates (Seber, 1982; Krebs 1994). In this study, the mark event corresponds to tagging lingcod

in-situ. The second event is the underwater survey dive; lingcod are observed in their natural habitat and the presence or absence of a tag is noted. The mark-recapture data were then used to estimate local abundance using a sequential Bayes algorithm (Gazey and Staley, 1986).

Tagging Lingcod In-situ

Tagging dives took place over an 8-month period between January 13 and August 8, 1998, at 13 different locations (see Figure 2.1). All lingcod were tagged with an intramuscular tag (FIM-96: FLOY® Intramuscular Tag). Tags were labeled with a sequential tag number and a contact telephone number. The FIM-96 tag is designed for fish that are at least 50-cm in total length; therefore, lingcod less than 50-cm in total length were not tagged. The tags were inserted, *in-situ* using SCUBA, using a 1.2-m slingshot pole. Because we were not able to measure individual fish while tagging, the size of each individual was estimated visually and the diver determined whether the individual was large enough for tagging. The depth for tagging ranged from 1.8 to 32.6 meters.

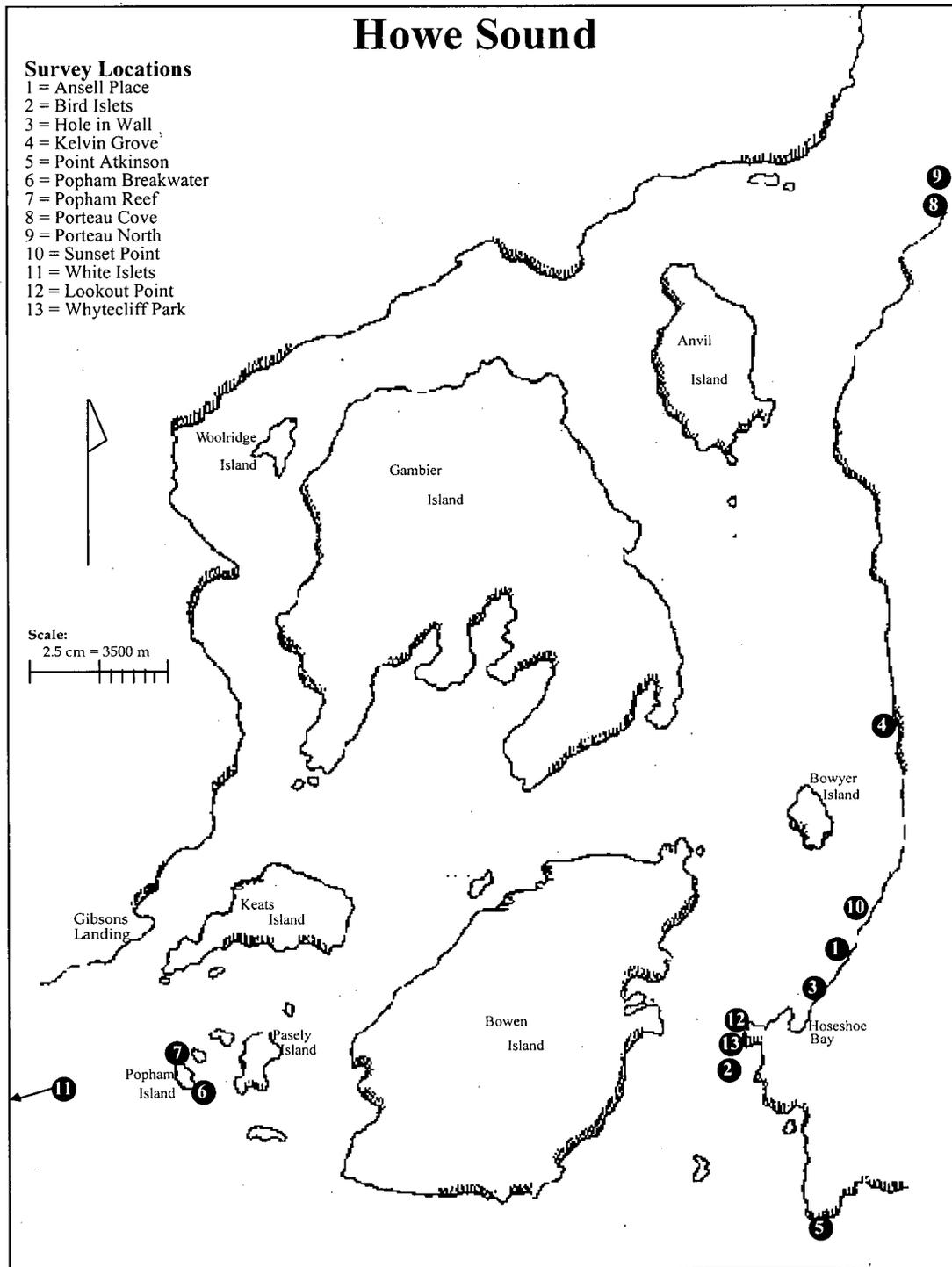


Figure 2.1. Location of the 13 different lingcod tagging sites in Howe Sound. Note that White Islets (location 11) does not appear on this map; it is located 16 km WSW of Popham Island.

The tag consists of a nylon dart head that is double barbed for strong retention in the dorsal muscular tissue (see Figure 2.2). Looped through the dart head is a 4.5 cm long 200 lb. test nylon monofilament fishing line that is attached to a 6.5 cm long spaghetti tag. The nylon monofilament fishing line serves two purposes; it allows the entry wound to close and heal, and it allows the spaghetti tag to trail next to the body thereby reducing drag. The dart head is hollow, and to apply the tag the dart head is inserted onto a 1.5-cm stainless steel applicator needle. The depth of insertion is controlled by a stop plate, located roughly 3 cm behind the tip of the applicator needle.

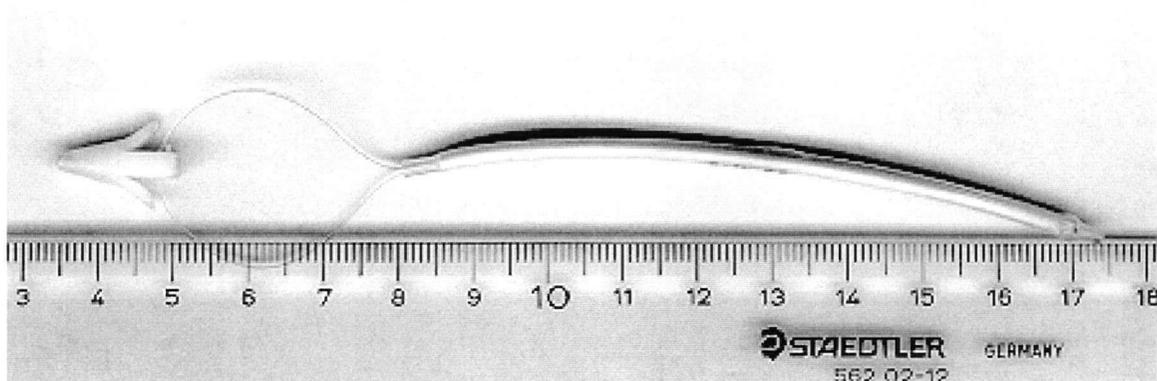


Figure 2.2. FIM-96 Intramuscular tag, manufactured by Floy Tag &Mfg., Inc.

To tag a lingcod, a diver slowly approaches a resting lingcod and aims the tip of the slingshot pole at the dorsal musculature area above the gut region. This region of the lingcod body contains the thickest muscle tissue. The tip of the slingshot pole is then positioned 10 to 40 cm above the target area and released. If the tagging attempt fails, the diver re-attempts to tag the fish, provided it remained in the area. Initially, lingcod were tagged only if they were resting on the bottom, however, as skill was acquired, slow swimming lingcod were also tagged.

Underwater Visual Census Dives

Survey dives were conducted on a weekly basis. The data collected included length frequency data, index of abundance data, and data on the proportion of tagged fish. Length frequency data were collected at the same 13 tagging locations from November 1997 to August 1998.

Survey dives were conducted using the roaming diver technique (RDT). The RDT method has pre-defined entry points and exit points, and once the diver is in the water, the depth contours surveyed depend on local water conditions. Due to small-scale local changes in underwater visibility, the roaming diver technique is advantageous over fixed line transect methods because lingcod, or other mobile prey items, can also respond to differences in local water conditions. One of the disadvantages of using this technique, is the variability of area swept among all surveys cannot be estimated because the linear distance searched is unknown (*i.e.*, the linear distance searched among all surveys is inconsistent). As a result, the area swept on each survey is inconsistent and unknown. The maximum and minimum depths surveyed were 36 and 1.8 meters, respectively.

The total length of the fish was estimated by measuring the distance between two points on the substrate (landmarks) below the resting lingcod. The anterior landmark was below the tip of the lower jaw, and the second landmark was below the posterior portion of the ventral caudal fin ray. Before a lingcod was disturbed from its resting position, the diver would locate the two landmarks, measure the distance between the two landmarks to the nearest centimeter, and record the measurement on an underwater slate. If the diver could not measure the lingcod (*i.e.*, only observed the fish swimming), then the

depth of the lingcod was recorded along with an estimated age. Unmeasured fish were not included in the length frequency analysis.

Analysis of Mark-Recapture Data

The number of lingcod sighted per hour of search time and the proportion of tagged fish versus untagged fish were used as indices of local abundance. The number of lingcod sighted per hour (z) on each dive was standardized.

$$z = \frac{(x - \bar{x})}{\sigma} \quad (2.1)$$

Where x is the number of lingcod encountered per hour, and σ is the standard deviation for each area.

The proportion of tagged fish in a population is represented by m/N_0 . The total number of tagged fish at large is equal to m , and the hypothesized population size corresponds to N_0 . If a total of n fish are observed on a dive, and y of those fish are tagged, then the likelihood of each N_0 is taken from the binomial distribution (Gelman *et al.*, 1995; Gazey and Staley, 1986):

$$P(y, n, m | N_0) = \left(\frac{m}{N_0}\right)^y \cdot \left(1 - \frac{m}{N_0}\right)^{n-y} \quad (2.2)$$

where N_0 is the hypothesis about lingcod abundance in the area. A uniform prior distribution was used to estimate the first posterior distribution (P_1) after the first underwater visual census survey (equation 2.3).

$$P_1(N_o | m, y, n) = \frac{P(y, n, m | N_o)}{\sum_{N_o=\min}^{\max} P(y, n, m | N_o)} \quad (2.3)$$

The minimum for N_o is constrained by the total number of tagged fish at large, and the maximum is the upper limit considered possible *a priori*. For additional survey data ($i+1, i+2 \dots i=k$) up to k surveys, the previous posterior distribution (P_{i-1}) was used as a prior to calculate the Bayes posterior probability distribution for N_o :

$$P_i(N_o | m_i, y_i, n_i) = \frac{L(y_i, n_i, m_i | N_o) \cdot P_{i-1}(N_o | m_{i-1}, y_{i-1}, n_{i-1})}{\sum_{N_o=\min}^{\max} L(y_i, n_i, m_i | N_o) \cdot P_{i-1}(N_o | m_{i-1}, y_{i-1}, n_{i-1})} \quad (2.4)$$

In equation 2.4, the posterior distribution is normalized by the sum of products between the likelihood function and the prior posterior distribution. The 95% credible interval is the population sizes corresponding to the 2.5% and 97.5% cumulative posterior distribution.

Estimates of lingcod density in each area were calculated by the mean population size, divided by the total suitable habitat in the survey location. Mean population sizes were calculated from equation 2.4, and were simply the means of the posterior distributions. Rocky reef area, or suitable lingcod habitat, in each location was estimated from dive surveys and bathymetry charts. Reef area estimates were purely subjective, as it is difficult to define what habitat is suitable and what is unsuitable. Also, dive surveys were restricted to depths less than 36 m, therefore, I was unable to qualify habitat types for depths below 36 m.

Results from Field Studies

Recall that the encounter rates for the entire year, in each location, are standardized using equation 2.1. As a result, the mean encounter rate over the entire survey period shown in Figure 2.3 corresponds to a z value of 0. Below average encounter rates and above average encounter rates correspond to a negative z value and a positive z value, respectively. Before the spawning season (October-December), only a single survey dive was conducted in the month of December, and the encounter rate was above average ($z > 0$). During the spawning season (January-March), the number of lingcod encountered was less than average ($z < 0$). As the summer months approach (April-June) the average number of lingcod encountered increased, and remained high during the summer (July-August). There were insufficient data to test if there is a statistical difference between encounter rates between months, or seasons, for each survey area.

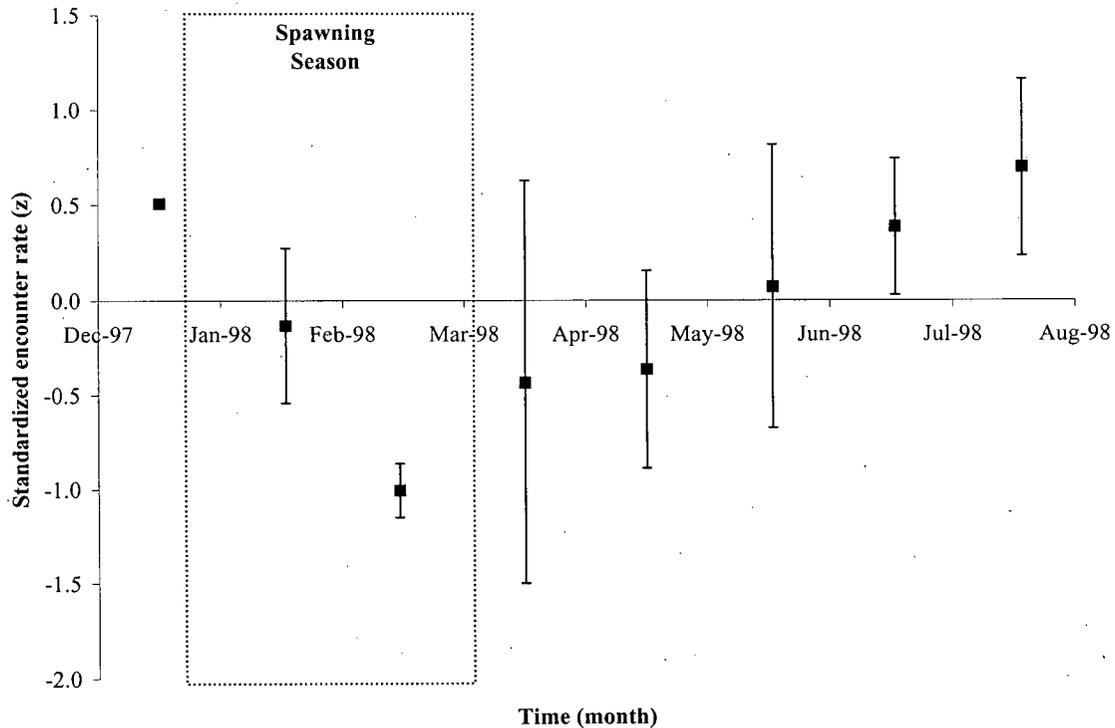


Figure 2.3. Lingcod encounter rates in Howe Sound from December 1997 to August 1998. Error bars represent 95% confidence intervals. Note only one survey dive in December-97, and greater than 4 survey dives per month after January-98.

Average encounter rates were higher in all reserve areas compared to fished areas in all months except June (Figure 2.4). In June, an exceptional number of lingcod was sighted at Popham Breakwater (ca. 45 lingcod). During the June survey dive at Popham Island, a new reef was discovered offshore in deeper waters. Lingcod that were tagged on the nearshore reef had moved to the offshore reef. As a result of this new discovery, I continued to survey both inshore and offshore reefs after June. The average encounter rate for fished areas was 23.5 lingcod per hour and the average encounter rate for lingcod in reserve areas was 25.6 lingcod per hour. There was no significant difference between encounter rates in fished areas versus protected areas ($F = 0.542 < F_{0.05(1),1,90} = 3.947$; $P = 0.465$).

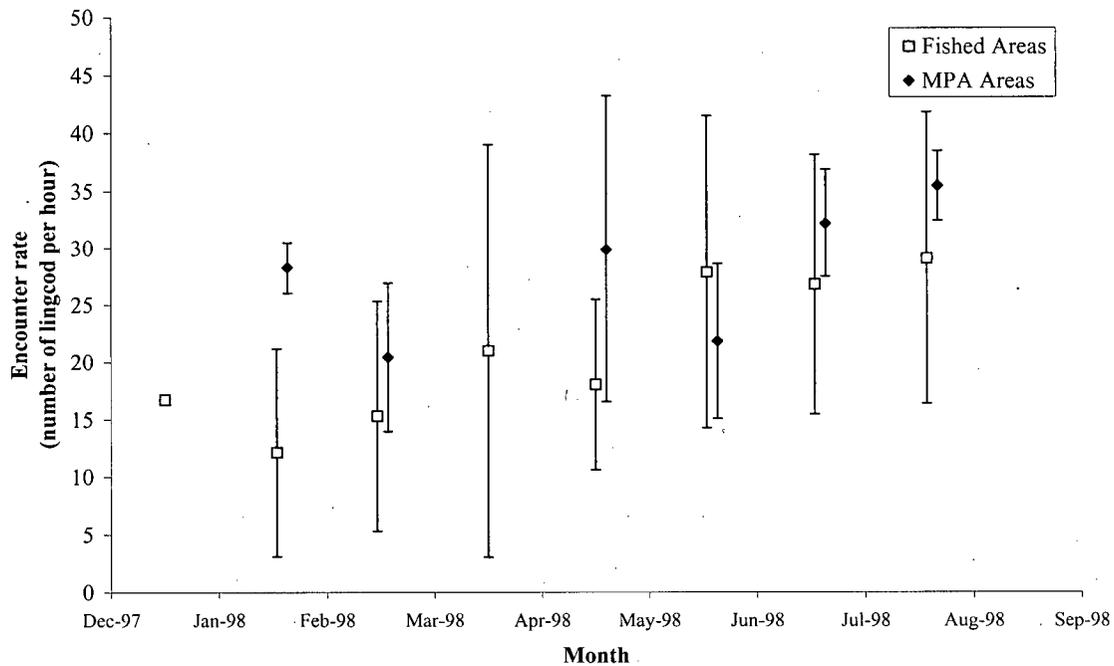


Figure 2.4. Average encounter rates for each month in three MPA sites and 10 fished sites. In all months, except June, the encounter rates are higher in the MPAs versus the fished sites.

There were differences in the encounter rates among the 13 different survey locations in Howe Sound and surrounding areas (Figure 2.5). A single factor Anova was used to compare the encounter rates, and the test yielded a significant difference ($F = 8.83 > F_{0.05(1),12,79} = 1.87$; $P < 0.001$). Encounter rates in the reserve areas were significantly higher than Hole in Wall, Kelvin Grove, Point Atkinson, and Porteau North. Three of these locations are easily accessible to shore fishing and Hole in Wall is close to Horseshoe Bay. Another interesting result is that encounter rates decreased rapidly immediately north of Porteau Cove (Porteau North location), whereas immediately south (and offshore) of Whytecliff Park (Bird Islets), the encounter rates remain relatively high.

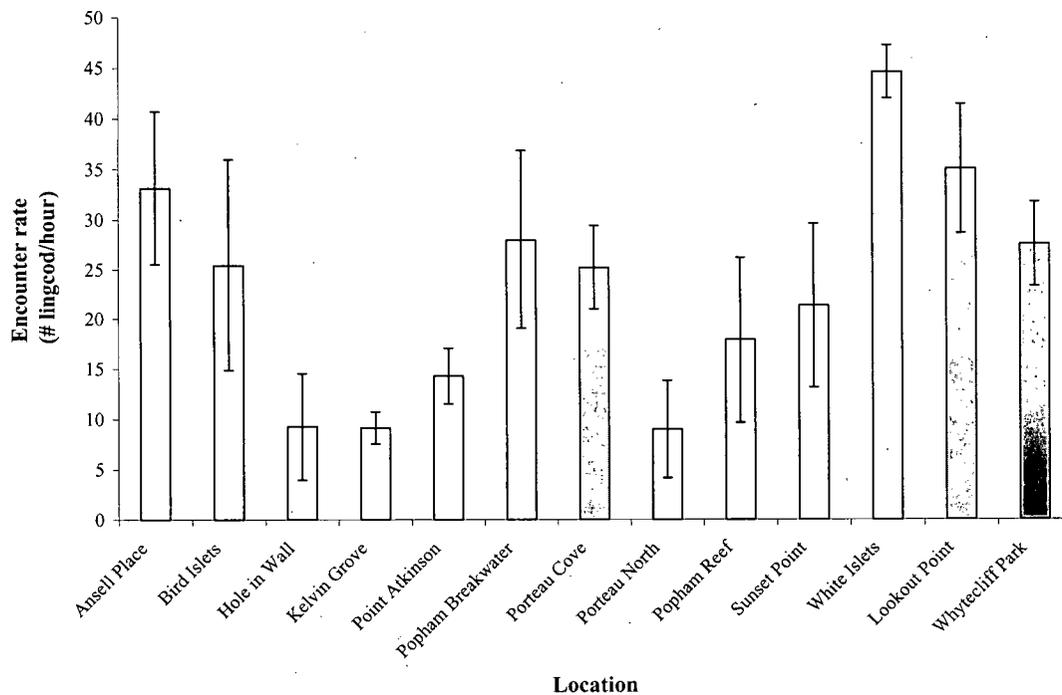


Figure 2.5. Average encounter rates and 95% confidence intervals for 13 survey locations in Howe Sound and adjacent areas. Note the shaded bars indicate that the survey is inside the boundaries of a marine reserve area.

In the fall, lingcod were observed primarily on rocky substratum, at depths ranging from 2 m to 33 m. The average encounter rate for all 13 areas during the fall season (October-December) was 23.1 ± 3.5 (SE) lingcod per hour. Early in the spawning season, large gravid females were first observed, and as the spawning season progressed, only male lingcod guarding egg masses were observed. The average encounter rate during the spawning season fell to 13.7 ± 2.9 lingcod per hour of bottom time. The average encounter rate in the summer jumped back up to 24.4 ± 2.1 lingcod per hour of bottom time. The first signs of new recruits (1-year-old fish) occurred in late summer. Most of the one-year-old lingcod were observed on sandy bottoms, adjacent to rocky reef

habitats. Changes in encounter rates were also related with slight changes in length frequency distribution over time (Figure 2.6).

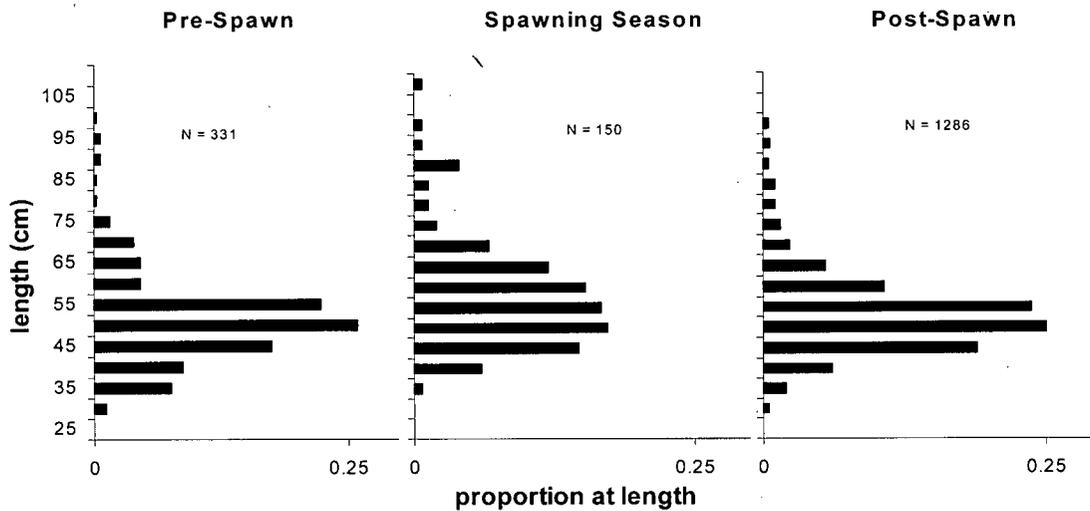


Figure 2.6. Length frequency distributions for lingcod in Howe Sound prior to the spawning season (November-December), during the spawning season (January-March), and after the spawning season and into summer (April to August).

In the fall, 331 lingcod were measured at all locations and the average length was 51.1 ± 0.6 cm. During the spawning season (January to March), the average length of lingcod measured increased to 58.1 ± 1.1 cm ($N = 150$), and in the following months the average length decreased to 53.4 ± 0.3 cm ($N = 1286$). The increase observed from the fall of 1997 to the summer of 1998 was less than expected due to growth, indicating that new (smaller) recruits moved into the survey areas during the summer months. The most striking feature in the length-frequency data was the sudden appearance of large fish during the spawning season. In January and February there was a temporary influx of mature fish into the transect sites. In late spring and into early summer, fewer large fish were observed in the transect areas and higher proportions of 2-year old lingcod (35-45

cm TL) were observed. During late summer, lingcod less than 35 cm (one year old fish) began to appear in transect areas.

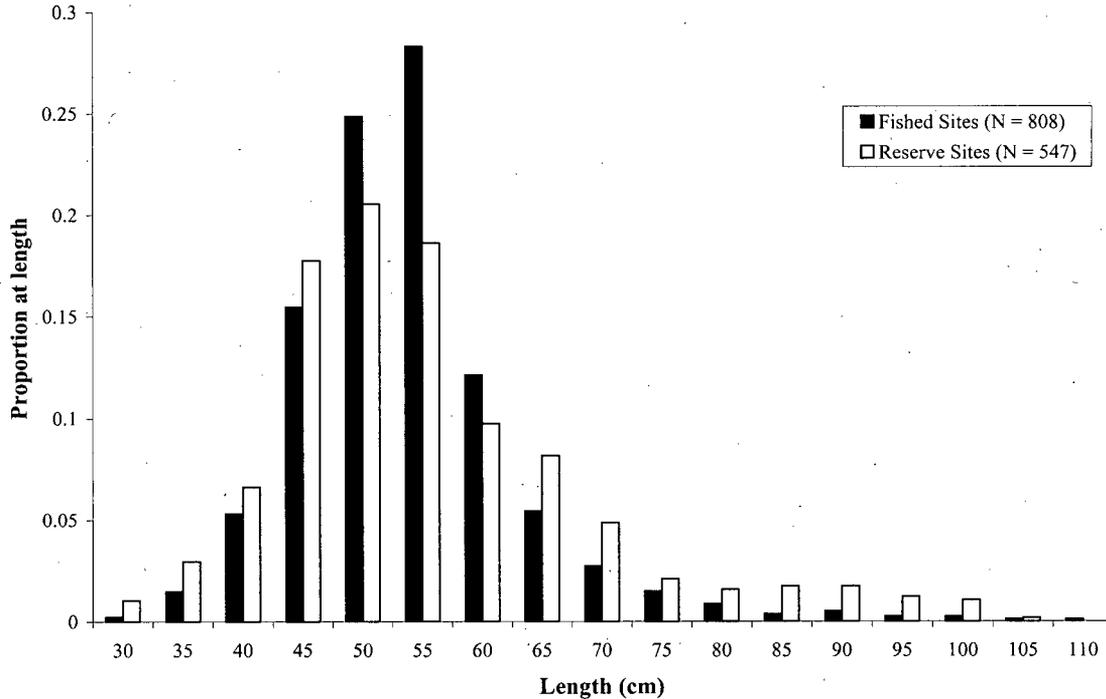


Figure 2.7. Proportion of lingcod in each length class in the fished areas and in the reserve areas. Note that the reserve areas consist of 3 study sites and the fished areas consist of 10 study sites.

Figure 2.7 compares length frequency data for reserve areas and fished areas.

Average length of lingcod in reserve areas was 53.7 cm TL and average length of lingcod in fished areas was 52.0 cm TL. There was no significant difference between the average lengths in reserve areas and fished areas. However, lingcod in the older reserve area (Porteau Cove) were significantly larger than those in Whytecliff Park and Lookout Point ($F = 93.6 > F_{0.05(1), 2, 571} = 3.01$; $P < 0.001$). The majority of large fish were observed in the Porteau Cove marine reserve (Figure 2.8). The average length of lingcod at Lookout Point was 46 cm, Whytecliff Park was 52 cm and, the average length at Porteau Cove

was 62 cm. Nearly 30% of the lingcod observed at Porteau Cove were greater than the minimum legal size of 65 cm TL. In contrast, less than 15% and 3% were of legal size in Whytecliff Park and Lookout Point, respectively. For fished areas, less than 7% of the lingcod observed were above the minimum legal size.

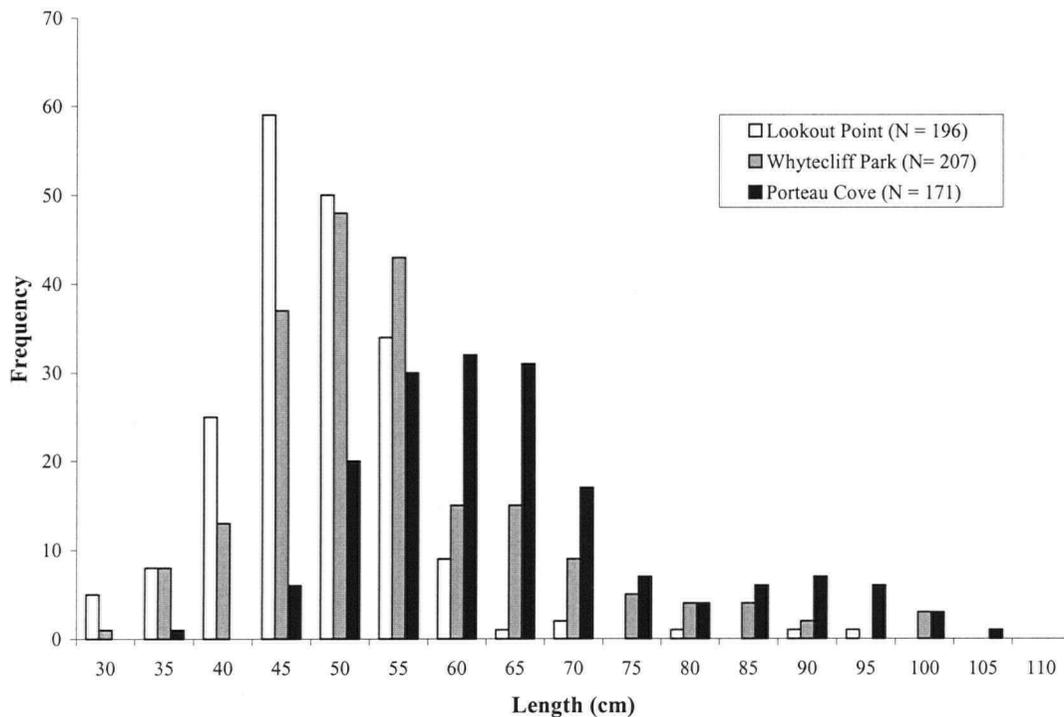


Figure 2.8. Length frequency distributions for lingcod at Lookout Point, Whytecliff Park, and Porteau Cove sampled over the 11-month study period. The average length of lingcod at Porteau Cove is significantly larger than populations within the other two reserve areas.

Results of population abundance and density estimates are presented in Table II.I, and Figures 2.9, 2.10 and 2.11. Figure 2.9 shows the outputs from the sequential Bayes algorithm. These outputs are probability distributions for the estimated number of lingcod present in the survey areas. Recall that the abundance estimates are based on the proportion of the tagged population that was observed in the field (equations 2.2-2.4). A

narrow posterior distribution corresponds to informative data about N_0 , and a wide distribution corresponds to uninformative data about N_0 . Also, the 95% credible intervals are calculated from these posterior distributions as a cumulative distribution. The data presented in Figure 2.9 are used to calculate the mean abundance and credible intervals (CI) presented in Table II.I.

Table II.I. The number of dive surveys at each of the tagging locations and the estimated abundance with the lower and upper 95% credible intervals (CI).

Location	Number of Surveys	Number of Fish Tagged	Average % Resighted	Mean estimate of abundance	Lower CI	Upper CI
Ansell Place	11	40	8.5	264	200	392
Bird Islets	4	16	4.9	312	184	908
Kelvin Grove	7	13	17.9	45	29	81
Point Atkinson	7	25	13.5	89	65	145
Popham Breakwater	9	43	10.9	151	119	203
Popham Reef	5	12	5.0	216	128	840
Sunset Point	8	31	15.5	135	103	195
White Islets	6	36	18.5	160	128	208
Lookout Point	7	28	8.9	316	224	576
Whytecliff Park	9	36	10.4	208	164	288
Porteau Cove	11	44	35.7	72	64	77

Its not surprising that the most accurate abundance estimate comes from Porteau Cove (N = 72 lingcod). This area is a well-defined region that consists of artificial habitat, placed over a sandy bottom. The entire area is easy to survey in a single dive, the lingcod present in this site are very used to divers, and on rare occasions they approach divers. Also, it is estimated that > 50% of the lingcod inhabiting this area were tagged.

The least accurate estimate of abundance was for Bird Islets (see Figure 2.9). There are two reasons the data at this location are less informative. First, only a very small proportion of the population was tagged (16 lingcod were tagged at this location). Secondly, in two of the four survey dives, no previously tagged fish were re-sighted, and

because I am using the binomial as a likelihood function, a zero re-sight count leads to an “open-ended” population estimate (*i.e.* the likelihood function is non-integrable at very large N ’s).

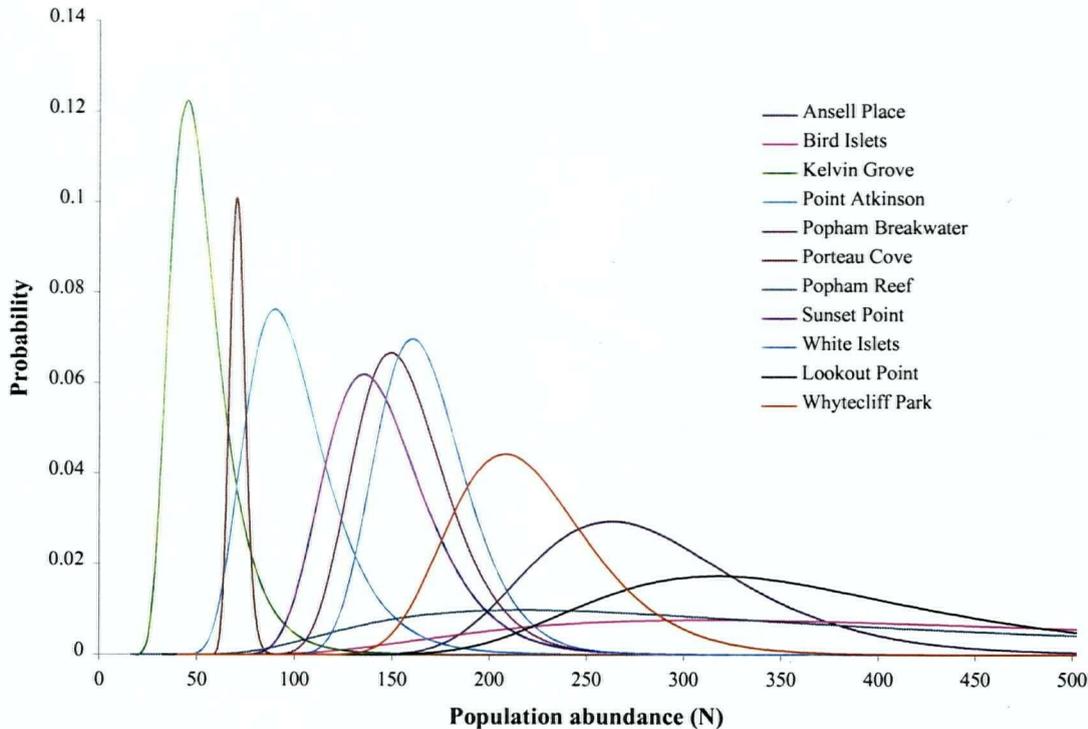


Figure 2.9. Posterior distribution for the estimated abundance of lingcod in 11 different tagging locations shown in Figure 2.1. These estimates are used to calculate density of lingcod shown in Figure 2.11.

The highest abundances of lingcod were concentrated near the Whytecliff Park MPA (Figure 2.9). Moving north of Hoseshoe Bay, the abundance of lingcod tends to decrease until Porteau Cove, where there is a slight increase in abundance. Also, survey sites that are located adjacent to islands (*i.e.*, which can only be accessed by boat) tend to have much higher numbers of lingcod present. During many survey dives along the east shore of Howe Sound, I frequently observed people fishing from shore. This is especially

apparent in the summer months. Also, much of the broken fishing line observed underwater was along the east shore of Howe Sound; however, no attempt was made to quantify the amount of fishing line present in each location.

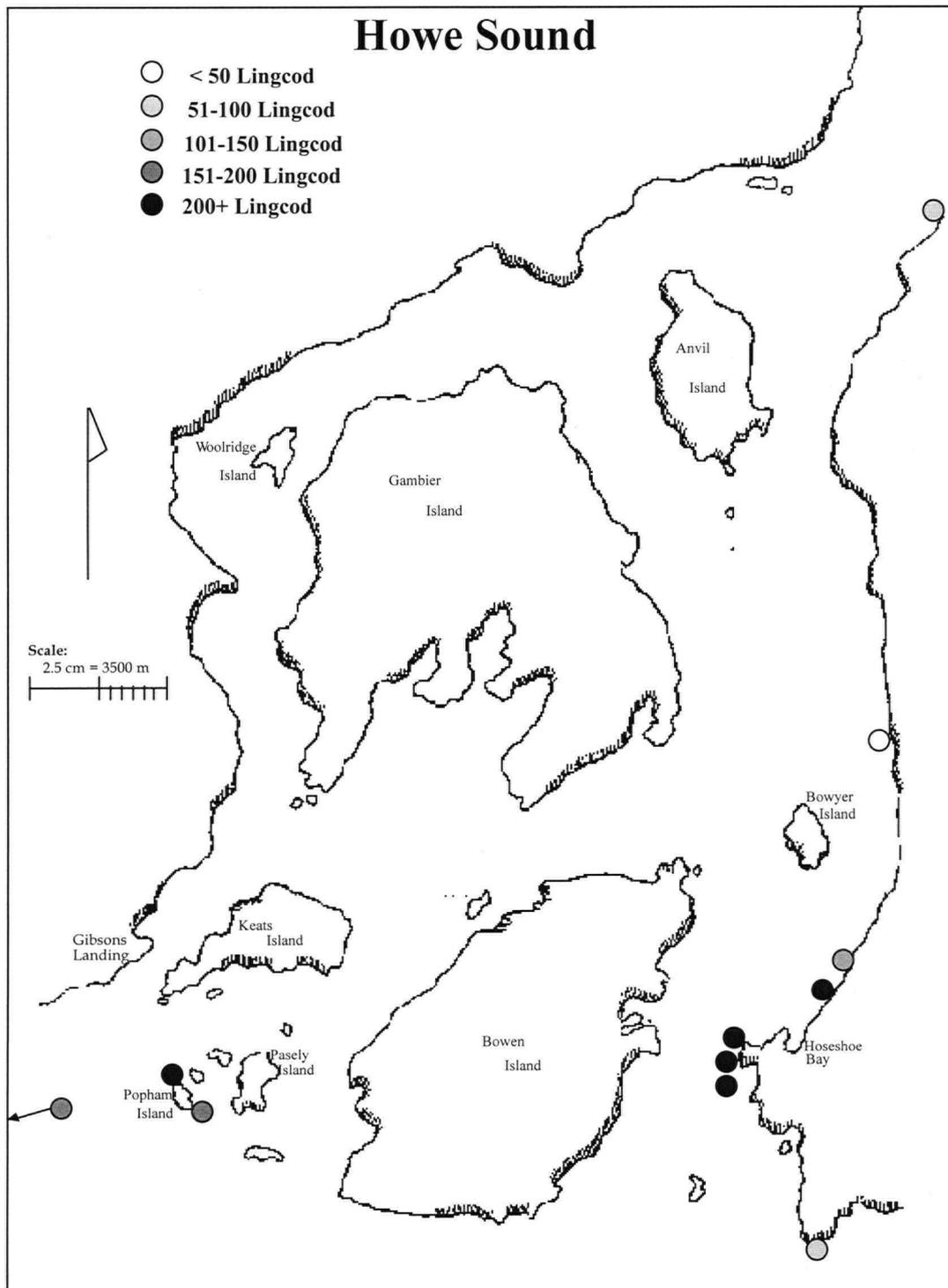


Figure 2.10. Abundance (estimated from mark-recapture data) of lingcod at 11 of the 13 different tagging locations in Howe Sound. Note that White Islets is not shown on this map.

Estimates of lingcod density at 11 of the 13 locations are presented in Figure 2.11. Unfortunately, there were insufficient data to estimate lingcod abundance at Hole in Wall and Porteau North. Among the 3 marine reserve areas, there was no significant difference in lingcod density. Among all 11 areas presented in Figure 2.9, Kelvin Grove had the lowest lingcod density (1.0 lingcod/1000 m²) and was obviously lower than Bird Islets, Popham Reef and Lookout Point. The highest density (7.7 lingcod/1000 m²) was observed in Lookout Point, one of the 3 marine reserve sites. At Bird Islets, immediately adjacent to Whytecliff Park, the density of lingcod was 6.2 lingcod/1000m².

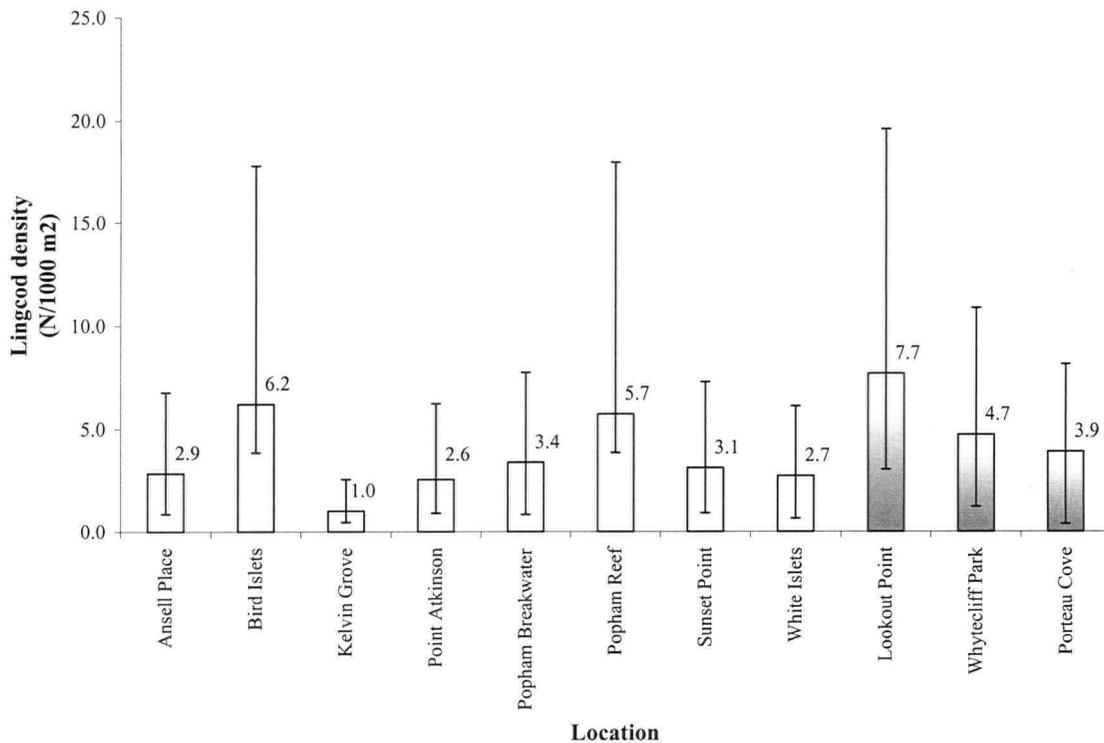


Figure 2.11. Estimated lingcod densities at 11 of 13 tagging locations in Howe Sound and at White Islets on the Sunshine Coast. The shaded bars indicate the survey area was within the boundaries of a marine reserve area.

Of the 332 tagged lingcod present in the field, two were recovered during the 1998-fishing season. Estimates of the total number of lingcod landed by anglers in Howe

Sound this year was 86 (L. Nagy, Department of Fisheries and Oceans, pers. comm.).

Therefore, a minimum estimate of the exploitation rate is $2/332 = 0.6\%$. One of the recaptured fish was caught roughly 5 km Northwest of the tagging location; the other was recaptured inside Whytecliff Park, where it had been tagged.

Discussion

The current network of marine reserves in Howe Sound consists of less than 1% of the total surface area of Howe Sound. The largest of the current “no-take” marine reserves is Whytecliff Park, and the smallest is Point Atkinson (100 m by 85 m). Using the movement rates estimated by Smith and McFarlane (1990), the maximum residence time for a moving lingcod in any of the MPAs in Howe Sound would be 3 days (or less if they travel the width of the reserve). In this study, I have repeatedly sighted tagged fish residing inside the boundaries of the reserves for a period of 14 months. However, changes in length frequency and encounter rates over time suggest that during the spawning season, small immature fish are displaced from the study sites and large mature fish move into the study sites. Immediately after the spawning season, the large mature fish then disappear from the study sites. At this time I am unable to provide conclusive evidence that demonstrates lingcod move across reserve boundaries in Howe Sound; however, based on the seasonal changes in length frequency and relative abundance, it is likely that a significant proportion of the fish are moving outside of the boundaries.

One of the many problems when conducting fish surveys using SCUBA is depth limitation. It is highly likely that the larger, “transient” type lingcod are just moving

offshore to deeper waters beyond the limitations of SCUBA. Moreover, many small juveniles and yearling lingcod may also reside in deeper offshore waters or in other habitat types not included in the surveys.

Of the 332 lingcod tagged in this study, anglers have reported only 2 tagged lingcod. One tag was recaptured at a significant distance from its tagging location (a 5-km displacement northwest from White Islets to Welcome Pass in 3 months). To date, this is the only direct estimate of movement rate for lingcod in near Howe Sound. There have been many tagging studies of lingcod in Georgia Strait over the last 5 decades that have tried to quantify rates of movement or look for migration in the stock (Hart, 1943; Chatwin 1956; Cass *et al.*, 1983, 1984, 1986; Smith *et al.*, 1990). In each of these studies, the author(s) report a small proportion (4-38%) of the tagged population being recovered at distances greater than 10 km from the release site. In another tagging study conducted in Neah Bay, Washington, Jagielo (1990) examined length frequency distributions of recovered tags and tested to see if "migratory type" lingcod are larger than "resident type" lingcod. "Migratory type" refers to lingcod that are recaptured at distances greater than 5 miles (8.6 km) from the release site. Jagielo concluded that lingcod recaptured further than 8.6 km away from the release site were larger than lingcod recaptured within 8.6 km of the release site. Also, a higher proportion of males were found to be migratory than females (Jagiello, 1990); however, I believe this to be an artifact because most of the lingcod were tagged near shore, and after the spawning season, thereby excluding most of the female population from the tagging study. In comparison to Jagielo's (1990) findings, the one lingcod that was recaptured 5 km away was smaller than the average size tagged at White Islets.

Smith *et al.* (1990) used mark-recapture data to estimate movement rates for male and female lingcod under the assumption of random movement. Their interpretation of the data suggested that female lingcod move at higher rates than male lingcod (1040 m/d for females and 500 m/d for males). In this study, the length frequency data indicate a large influx of large (probably mature females) into the survey areas during the spawning season. This should be of no surprise, as several authors have reported differential bathymetric distribution of the sexes and seasonal migrations of females to near-shore spawning areas (Miller and Geibel, 1973; Gordon, 1994; Cass *et al.* 1984 Cass *et al.* 1990). Therefore, one interpretation of changes in length frequency observed during the spawning season is the result of females moving inshore to spawn. However, not all lingcod stocks move inshore to spawn in shallow waters (O'Connell, 1993). Offshore stocks are capable of spawning in depths below 30 m on rocky pinnacles (O'Connell, 1993). Spawning site selection is suggested to be a function of water current velocity, not water depth. In the nearshore, water current velocities are typically faster than in deeper offshore waters due to tidal currents, and lingcod egg masses are less susceptible to hypoxic conditions (Giorgi and Congleton, 1984).

The other general observation about changes in lingcod density is recruitment of young-of-the-year lingcod. Prior to the summer months, young-of-the-year lingcod were not present in the rocky reef survey areas. To date there is conflicting information about where juvenile lingcod settle. Cass *et al.* (1990) reported newly settled juveniles rearing near kelp or eelgrass beds. During submersible observations of six-gill sharks, Dr. Jeff Marliave (pers. comm., Vancouver Aquarium Marine Science Centre) observed young-of-the-year lingcod at a depth of 123 m on mixed gravel, and mud slope at the base of a

cliff. In either case, small lingcod are probably avoiding older, cannibalistic, lingcod in much the same manner as Atlantic cod (*Gadus morhua*) cohorts segregate bathymetrically (Swain, 1993). During the late summer months (August-September), a growing number of lingcod less than 35 cm TL begin to appear in the survey areas, especially on sandy bottom. The appearance of yearling lingcod in the length frequency data accounts for the lack of apparent growth over the course of the time series shown in Figure 2.6. The sudden appearance of juvenile lingcod in the nearshore also corresponds to the appearance of small, newly born juvenile shiner perch (*Cymatogaster aggregata*). I never observed juvenile lingcod feeding on juvenile perch in the field, however, juvenile shiner perch were readily accepted by juvenile lingcod reared in captivity (Jeff Marliave, pers. comm.).

Fish movement across marine reserve boundaries is one of the central questions about the use of marine reserves in a fisheries management context (Roberts and Polunin, 1991; Watson and Walters, 1998). In this study I was not able to estimate movement rates across the reserve boundaries, but I was able to demonstrate that seasonal movement patterns exist, although not necessarily across reserve boundaries. An interesting question that was addressed by Zeller and Russ (1998) is the idea of home ranges and of movement within the home range. In their study on coral trout (*Plectropomus leopardus*), fish within the reserve moved greater distances than fish immediately outside the reserve areas. Also, coral trout densities in open versus closed areas were not significantly different; therefore, the apparent difference in dispersal distance did not appear to be density related. Zeller and Russ (1998) suggest a behavioral difference between the two populations as a result of fishing pressure. The results of this study do

not provide any direct estimates of movement rates inside or outside of the reserve areas. However, the proportion of observed tags to the total number of tags at large (Table II.I) can be used as an indirect measure of movement (e.g. the higher the movement rate the smaller the percent of re-sighted tags). Using this crude analysis, the results of this study indicate that the movement rates inside the reserve area are smaller than adjacent fished areas. This is especially true for Porteau Cove. During the course of this tagging study, fishing was never observed inside the Porteau Cove survey area, whereas shore fishing was frequently observed at the Whytecliff Park area.

There have been two previous telemetry studies on lingcod in Georgia Strait (Yamanaka and Richards, 1993; and Matthews, 1992). Both studies indicated that transplanted fish tend to wander, and sometimes leave the study area, whereas a control group (tagged fish that are not translocated) remains within 300 m of the capture-release site over the course of the study period. These results also conform to an earlier translocation experiment conducted by Buckley *et al.* (1984). Estimated movement rates from telemetry studies on lingcod range from 0.03 to 2.67 km per day (Yamanaka and Richards, 1993). Matthews (1992) was able to demonstrate that male lingcod do remain in small home range areas and return to the capture site when displaced shortly after the spawning season. At the lower end of this movement rate scale, and incorporating the idea of home ranges, lingcod could easily remain inside the small reserve areas in Howe Sound for the duration of their lives. In comparison to the offshore studies conducted by Jagielo (1990), little horizontal displacement is required to segregate sexes or cohorts by depth in Howe Sound due to rapid drop-offs.

It is difficult to judge how well the lingcod populations are doing in Howe Sound with one year of data, and I should emphasize that the results of this study should not be used as a historical baseline for future comparison. Such baseline shifts have been documented by Pauly (1995); therefore, I will add some anecdotal information provided by friends and colleagues who have more history with lingcod in Howe Sound.

Experienced divers have attested to the dramatic decline in lingcod abundance over the last 40 years at some of their favorite dive sites in Howe Sound (A. Lamb and B. Hanby, pers. comm.). Mr. Lamb and Mr. Hanby have been diving in Howe Sound since the 1950s. In their view, the abundance of lingcod at Whytecliff Park, and every other dive site they frequent in Howe Sound, has dramatically decreased. Also, they can recall large schools of yellowtail rockfish (*Sebastes flavidus*) and black rockfish (*Sebastes melanops*) in the Whytecliff Park area. Today, roughly a dozen small yellowtail rockfish have recruited to Whytecliff Park, and I have never seen a black rockfish in this area. In the 1930s, Popham Island was regarded as the "hot spot" in Howe Sound for lingcod by sports anglers (H. Bell Irving, pers. comm.). Today, very few people fish for lingcod at Popham Island. In general, my observations are that lingcod are more abundant near the entrance of Howe Sound, especially near Whytecliff Park. Further north along the east shore of Howe Sound, the abundance and density of lingcod decrease. I could find no published information about the historical abundance of lingcod in Howe Sound, other than that catch rates for the sports fishery have declined dramatically over the last 15 years (Haist, 1994).

Chapeter III: Use of Marine Reserves as a Fisheries Management Tool for Conserving Lingcod In Howe Sound.

Introduction

In this chapter, I will use a computer model (FISHMOD) to simulate the recreational lingcod fishery in those parts of Howe Sound included in my surveys and to examine the affect of MPAs in this fishery. There are three important aspects of a fishery that are of concern when using spatial closures as a fisheries management tool: 1) the movement of fish across reserve boundaries, 2) the spatial distribution of fishing effort, and 3) substrate structure as measured by the amount to which fish that spawn at a site and produce juveniles that eventually home to these sites for reproduction. Fish movement across the reserve boundary is one of the central questions about the use of marine reserves in a fisheries management context (Roberts and Polunin, 1991). The distribution of fishing effort along the reserve boundaries has several important theoretical implications about the efficacy of MPAs (Walters, 1999). Using the seasonal patterns of lingcod movement observed in Chapter II, and a range of hypotheses about rates of movement, I evaluate the efficiency of the 3 marine protected areas that presently exist in Howe Sound. I do this under fine-scale stock structure assumptions, namely that my studies sites represent 9 local populations that remain in small home range areas, and juveniles produced by each stock return to that area to spawn. Also, I compare the MPA policy to alternative management policies. The two alternative management policies are 1) increasing the minimum size limit, and 2) shortening the fishing season.

Methods: a general description of FISHMOD

FISHMOD is an age-structured, single-species population dynamics model that allows explicit representation of spatial distribution of fish and fishing effort over a map of grid cells. There are 3 major sub-models in FISHMOD. The three sub-modes pertain to population dynamics, movement of fish, and the distribution of fishing effort. The following three sections will briefly describe these sub-models.

Population Dynamics

If $N_{s,a,ij,t}$ is the number of fish originating from stock s , age a , in spatial cell ij (corresponding to map row and column) at time t , then dynamics in the absence of movement are represented by

$$N_{s,a+1,ij,t+1} = N_{a,a,ij,t+1} e^{-vaF_{ijt}-M} \quad (3.1)$$

where M = natural mortality rate and fishing rates F_{ijt} is predicted from fishing effort E_{ijt} allocated to cell ij during time t and age specific vulnerabilities v_a . Annual total egg production for each stock s (sum of numbers at age times fecundity at age, over all grid cells used by stock s) is used to predict $N_{s,1,t+1}$ using a Beverton-Holt stock-recruitment relationship. One year-old fish are distributed evenly over all cells designated as nursery cells for stock s .

The user must specify the maximum number of recruits per spawner as the spawning stock approaches zero (equivalent to k parameter in Chapter I). Assuming equilibrium during the initialization of the model (*i.e.* before the start of the fishery), the

model calculates the recruitment limitation parameter β . This method of predicting recruitment is the same as used in the Stock Reduction Analysis in Chapter I.

Equation 3.1 assumes that adults from stock s home to stock-specific nursery areas for spawning within the overall grid. Multiple stocks may be present in the same cell, however, during the spawning season, stock s distributes new recruits to rearing areas. Also, more than one stock can share the same nursery cell.

Movement of Fish

FISHMOD uses an Eulerian approach to move fish around a spatial grid and each stock follows an age-specific "life-history trajectory". For each simulated month, animals are redistributed among neighboring spatial cells one or more dispersal-migration computational steps. In each computational step, the animals in cell ij are divided into five proportions p : where p_0 = proportion of animals in cell ij that remain in cell ij , p_1 = proportion of animals that move to cell $i+1,j$ (if the adjacent cell has suitable habitat or is not defined as land), p_2 = proportion that move to cell $i,j-1$, if habitat is suitable, etc. Then $N_{s,a,ij,t}$ is reset to $p_0 N_{s,a,ij,t}$ plus the sum of proportions times numbers $p_k N_{s,a,i'j',t}$ moving into cell ij from (up to) four $i'j'$ cells surrounding it. The p 's are calculated by assigning a movement "weight" W_k to each of the five possible moves, where setting a high weight on W_0 represents a low dispersal rate. Distributing the weight evenly across W_k results in random movement around cell ij , and biased weight distributions on a particular k represents migration towards the cell represented by k . Then the p_k are calculated as just $p_k = W_k / \sum W_k$.

To represent seasonal migrations, the user specifies stock and age-specific monthly "target cells" $ij_{s,a,m}$; then at each movement step, the model places proportionally higher weight W_k on moving into those cells that are in spatial direction $ij_{s,a,m}$. The net results of this weighting procedure is to cause animals "near" the monthly target cell $ij_{s,a,m}$ to disperse randomly in all directions, but to cause a higher proportions of animals distant from $ij_{s,a,m}$ to move towards $ij_{s,a,m}$. The user specifies the random movement that occurs in animals near the target cell, where 0 corresponds to no random dispersal from the target cell, and higher number corresponds to increases dispersal abilities.

For investigations about the effects of fish movement rates across the reserve boundaries, I artificially increase the rate of movement across the reserve boundary by increasing the diffusion component of movement away from the optimum life history trajectory. For simulations in which there is no deviation from the life history trajectory, the random dispersal rate is zero. The following Figure shows an example of monthly trajectories that 3-year old lingcod were assumed to follow at Whytecliff Park. For the most part, the trajectories for the 9 mature populations used in these simulations followed the same southward progression from January-to June, and northward from July-December.

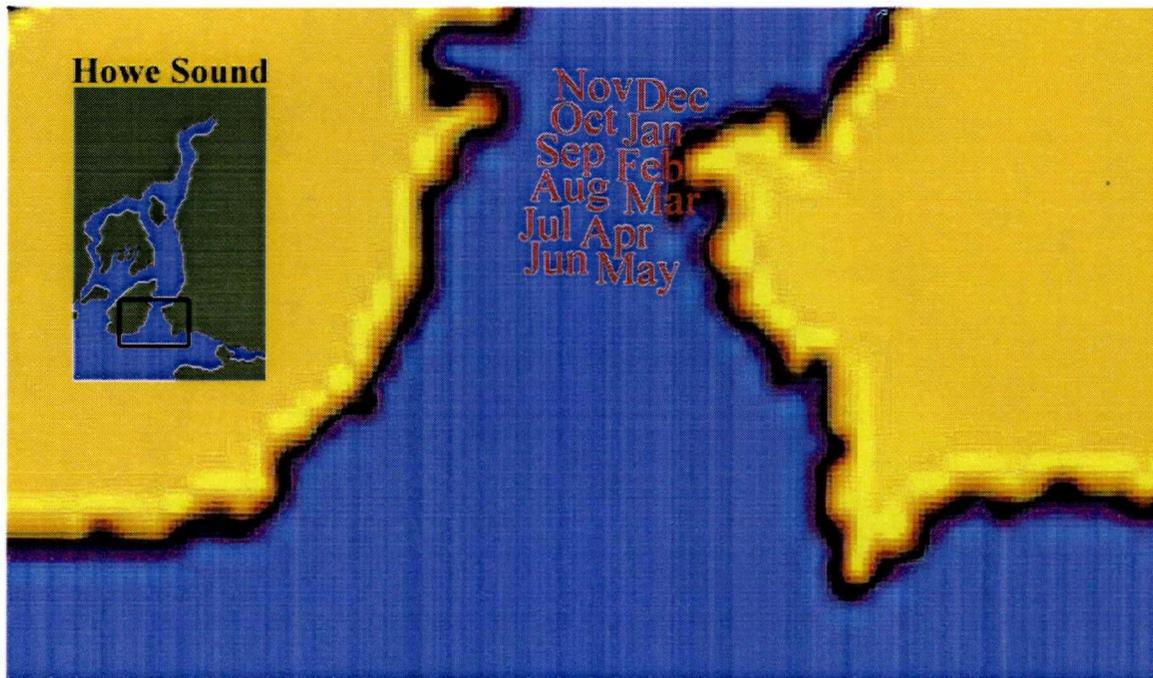


Figure 3.1. An example of seasonal movement patterns used in the FISHMOD model for the 3-year-old lingcod present at Whytecliff Park.

One other important aspect that is related to fish movement is the habitat type. For each cell in the spatial grid, the habitat must be defined as either suitable, marginal, or not used at all. These arbitrary definitions of habitat type limit the distribution of fish over the spatial grid, and as a consequence the model will not allow fish (or fishing effort) to be distributed in any cell that has unsuitable habitat (*i.e.*, cells that are not used at all).

Distribution of Fishing Effort

Fishing effort is distributed over the model spatial grid using a “gravity model” (Walters and Bonfil, 1999). The gravity model distributes a total monthly fishing effort to each cell based on 2 factors, the expected abundance in each cell, and the previous

catch rates in each cell. The idea behind the gravity model is simple. It is based on the assumption that anglers distribute themselves to maximize catch rate, and when an angler chooses a location to fish, the decision is based on two criteria: the previous experience and the expected abundance. Previous experience refers to the individual history of fishing in the area. For example, anglers who have experience fishing in a certain body of water are able to recall areas where catch rates were high, and are more likely to try these areas first before exploring new fishing grounds. Expected abundance can refer to a variety of things, the simplest being information sharing among anglers, or knowledge of the habitat preferences of lingcod.

This gravity model gives effects that are similar to, but more “smeared” in space than the Ideal Free Distribution methods proposed by Gillis *et al.* (1993). In essence the gravity model in FISHMOD allocates the total monthly effort to each cell in proportion to the “attractiveness” of that cell. In this context, attractiveness, referred to as θ , is a measure of the expected catch rate for a given ij cell:

$$\theta_{ij,t} = \left(\lambda \theta_{ij,t-1} + (1 - \lambda) \sum_{ages} V_{a,ij} B_{a,ij,t} \right)^m \quad (3.2)$$

where λ is a weighting parameter that must lie between 0-1. The products of the summation are the vulnerable biomass in cell ij at time t . The exponent (m) is the effort distribution power parameter. When m is greater than one, there is an exponential increase in the attractiveness weighting for each ij cell. The effort allocated to each cell ($E_{ij,t}$) is distributed according to the attractiveness, θ_{ij} and the total monthly effort E_{total} :

$$E_{ij,t} = \frac{\theta_{ij,t}}{\sum_i \sum_j \theta_{ij,t}} E_{total} \quad (3.3)$$

Equation 3.3 distributes the total monthly effort to each cell in proportion to the attractiveness of each cell. Walters and Bonfil (1999) show how equation 3.3 approximates the Ideal Free Distribution model. The total monthly effort used in the simulations is 1000 boat days, which roughly corresponds to the number of boat days observed in the creel survey data during the summer months in statistical area 28.

Simulations

All simulations were carried out with spatial reference to Howe Sound, and ran for 20 years. The starting biomass was 5000 t, and the stock parameters used are the same as those that were used in the stock reduction analysis. All parameters, with exception of the coefficient representing random diffusion in movement (cvm) and the effort-distribution-power parameter (m) were fixed for all simulations. Each simulation represented one hypothesis about the movement rate of lingcod and the angler response to the distribution of lingcod.

The model was set up to mimic the sports fishery in Howe Sound. At this present time the sports fishery restrictions for lingcod consist of a minimum size limit of 65 cm, a seasonal closure from October 1 to May 31, and a daily bag limit of one per day and ten per year. To simulate the minimum size limit in the model, the vulnerability-at-age was 50% for 4 year old fish (~ 65 cm TL), and to reach an asymptote of 100% vulnerable at 7 years of age. Seasonal closures were implemented by closing all cells to fishing from October to May. Unfortunately the model has no way to explicitly specify daily or

annual bag limits. The catchability coefficient estimated from the SRA model in Chapter I was used in these simulations ($q = 0.0474$).

A total of 20 separate parameter combinations were tested. The amount of diffusion in the movement trajectory ranged from 0 (no diffusion) to 100 (rapid diffusion rates). For each monthly time step, the movement sub-model was ran three times. That is the maximum dispersal distance in each time step is 3 cells. The effort distribution power parameter ranged from 1 to 3. The parameter combinations for which simulations were carried out are shown in Table III.I.

Finally, in these simulations I used the three existing “no-take” MPAs in Howe Sound. For each simulation, I kept track of the proportion of fishing effort that was distributed to the cells adjacent to the MPAs. Also, I assumed that poaching does not occur, *i.e.*, no fishing effort was distributed in cells that were specified as reserves.

For comparing different management policies, all simulations were compared to a baseline simulation. The baseline simulation used the aforementioned size limits, seasonal closures and 1000 boat days per month. There were no MPAs in the baseline simulation. The three policy options were: 1) implementation of 3 small (currently existing MPAs) in Howe Sound, 2) no MPAs and an increase in the minimum size limit to approximately 70cm, and 3) a two month fishery instead of a 4 month fishery. Both the effects on total biomass and catch rates under the different policy options were examined.

Results of FISHMOD Simulations

Recall that the aim of using FISHMOD was to explore the efficacy of marine protected areas on lingcod populations in Howe Sound. Also, it is important to note here that the model was not fitted to any field data, so all of the results shown below are strictly theoretical. First I will present the results that address issues of fish movement, then issues of fishing effort distribution. All of the parameter combinations will be evaluated in terms of the change in biomass over the 20 year time series, and the proportion of fishing effort that is distributed in the cells that border the 3 small "no-take" marine protected areas that currently exist in Howe Sound.

The second part of the analysis deals with the results of the 3 different policy options. The effects of the different policies are interpreted using the total biomass and catch rate time series outputs from FISHMOD.

Behaviour of the System in Response to Fish Movement

In general, as the simulated dispersal rate of lingcod was increased, the change in biomass over the 20-year time series also increased (Figure 3.2). When there is random dispersal in the seasonal trajectory (Movement = 0 in Figure 3.2), the biomass is reduced by just over 50% in 20 years. Approximately 30% of the remaining biomass (after 20 years) was found within the reserve cells when there is random dispersal. In contrast when diffusion rates are high, it takes less than 8 years to reduce the biomass by over 50%. Simulations with no reserve areas in the model result in a 50% biomass reduction in 3 years, and less than 10% of the biomass remains at the end of the 20-year simulation.

However, a small increase in simulated dispersal rate resulted in a huge biomass reduction due to straying across the boundaries of reserve cells. As dispersal rate increased, the exchange rate across the reserve boundary continued to increase and the biomass was reduced even further. The resulting biomass was directly related to the residence time inside the MPA, and the residence time is a function of dispersal rate. This relationship is probably best described by a "1/x" type function, where x is the rate of movement (e.g. if $x = 0$ then the animal spends its whole life inside the reserve). As diffusion rates increased further, however, biomass did not continue to decline within the reserves. Because the model was set up to allow fish to move a maximum of 3 cells for each monthly time step, a small portion of fish in cell *ij* left and returned to cell *ij* in the same month. This situation corresponds to all dispersal rates except when dispersal is 0, or movement = 0 in Figure 3.2. Approximately 10% of the biomass remained in reserve cells at the end of the time series when movement occurred across reserve boundaries.

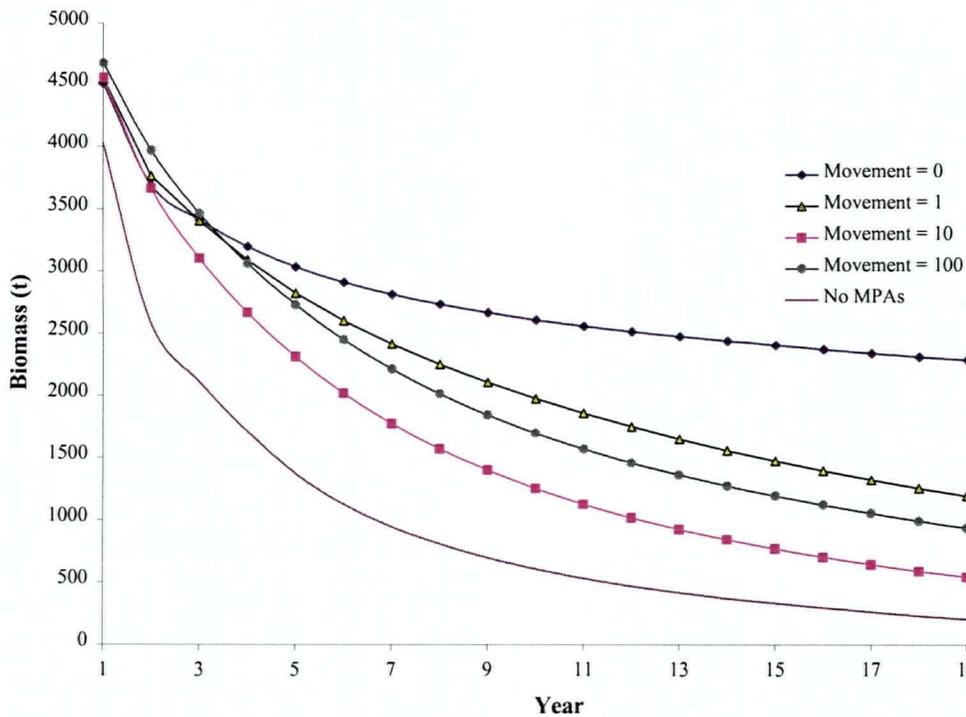


Figure 3.2. Mid season biomass estimates for a 20 year simulation in FISHMOD. In the first 4 simulations, the deviation in the dispersal rate is increased from 0 (corresponding to no movement) to 100 (corresponding to rapid movement rates). The fifth simulation is the biomass time series when all MPAs are removed from the model and the dispersal rate = 1.

As the biomass was depleted during the course of the fishery, simulated effort was redistributed to areas of higher fish concentrations. For low dispersal rates, the redistribution of effort to the boundary cells over the 20-year time series increased (Figure 3.3). However, at higher dispersal rates, the proportion of fishing effort initially started off high and remained relatively constant over the 20-year time series (Figure 3.3, Movement = 10 and 100). Only at extremely high rates of movement, did the proportion of effort along the boundary decrease.

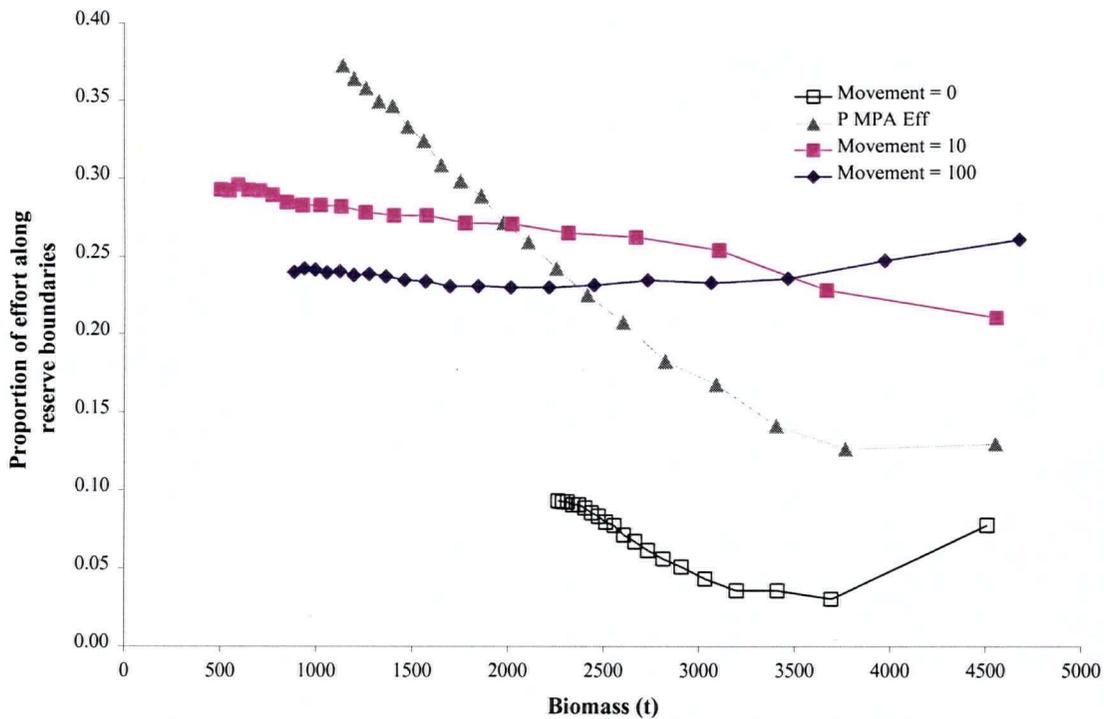


Figure 3.3. Proportion of fishing effort that was distributed in cells adjacent to reserve cells plotted against the remaining biomass. Each point along a line corresponds to the average proportion of fishing effort in boundary cells versus the mid season biomass.

Behavior of the System in Response to Changes in Effort Dynamics

Over the course of the 20-year time series, catch rates declined as simulated stock size was reduced (Figure 3.4). This result is expected because the relationship between stock-size and CPUE is proportional (*i.e.*, scaled by q , see equation 3.1). However, the catchability coefficient in all 7 scenarios presented in Figure 3.4 remained unchanged, yet the results indicate that spatial distribution of effort does influence catch rates.

As the effort-distribution power parameter was increased, annual catch rates decreased. The result of focusing fishing effort in cells with a higher expected abundance

was local depletion in such cells. Consequently, in the next time step the effort was redistributed into cells with the next highest abundance.

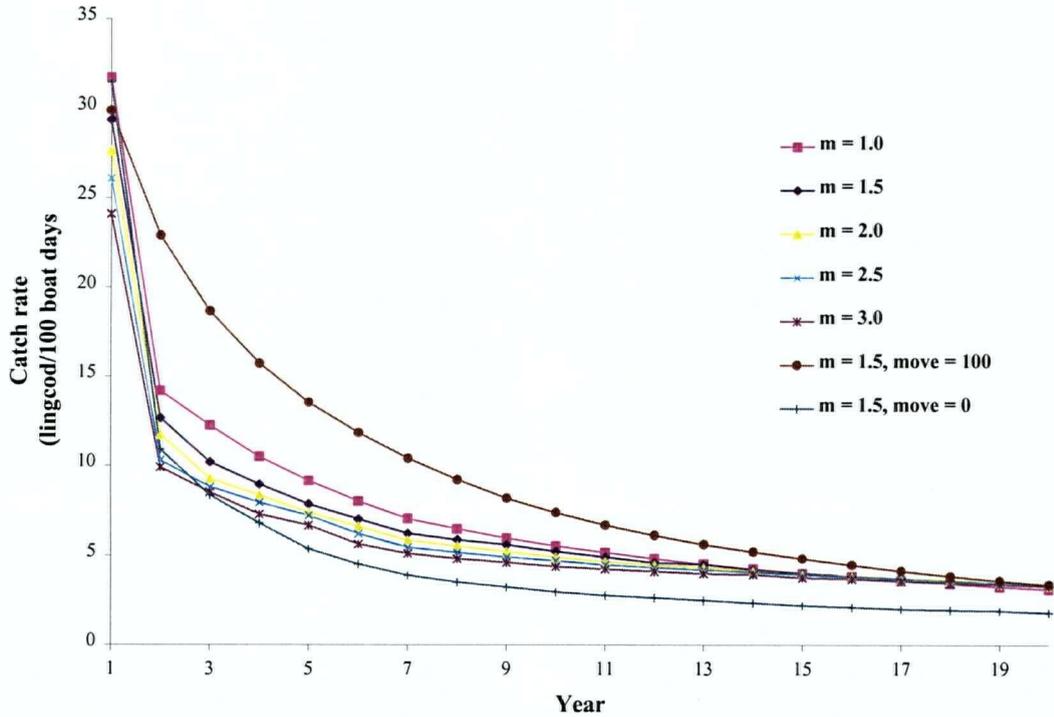


Figure 3.4. Annual average catch rates for 5 different hypotheses about effort distribution-power parameter m . All 5 scenarios were ran with the dispersal parameter = 1. Two scenarios with high and no dispersal rates are included for comparison.

The average proportion of fishing effort over the 20-year time series that was distributed in cells bordering MPA cells is presented in Table III.I. As dispersal rate increased, more effort was distributed adjacent to the MPAs. As the effort-distribution power parameter increases, less effort was distributed adjacent to the MPAs; however, this trend was only true for dispersal rates below 10. At extremely high rates of dispersal, there was less effort distributed adjacent to reserve areas as the effort-distribution power parameter increased (See Table III.1). This trend was also apparent in Figure 3.3, where

high rates of movement resulted in a decreased proportion of effort distributed in boundary cells as the stock size was reduced.

Table III.I. Average annual percentage of fishing effort distributed in cells bordering MPA cells under different combinations of movement rates and effort-distribution power parameters.

<i>m</i> \ move	0	1	10	100
1.0	7.02	26.49	27.42	23.78
1.5	4.52	19.62	28.81	27.54
2.0	3.73	15.68	26.28	30.21
2.5	3.75	13.59	24.70	31.89
3.0	4.18	11.91	23.68	32.44

Using the range of hypotheses presented for lingcod dispersal and the dynamics of effort distribution, there are 2 important results to report. First, dispersal of lingcod across reserve area boundaries has an effect on annual exploitation rates. In general, a higher exchange rate across boundary cells results in higher exploitation rates. Also, increases in exchange rates resulted in higher annual catch rates, because of immigration into previously depleted cells. Secondly, the proportion of fishing effort allocated in cells bordering reserve areas tended to increase as the stock was eroded over the course of the fishery. As dispersal rates increase, however, the change in the proportion of effort distributed in boundary cells decreased.

Finally, simulations in which there were no MPAs in place always resulted in the highest exploitation rates and the fastest reduction in biomass. This trend held true, regardless of dispersal rates and fishing effort dynamics.

Response to Alternative Fishing Policies

In comparing three different management policies to a baseline policy (No MPA, and current sports fishery restrictions), the safest policy involves the use of MPAs (Figure 3.5). Over the course of the 20-year time series shown in Figure 3.5, the biomass was reduced by over 90% for the baseline policy. As additional restrictions were implemented, there was an improvement in all policy options. Reducing fishing season length was the next best option to MPAs, and increasing the minimum size limit by 5-cm resulted in a small increase in overall biomass.

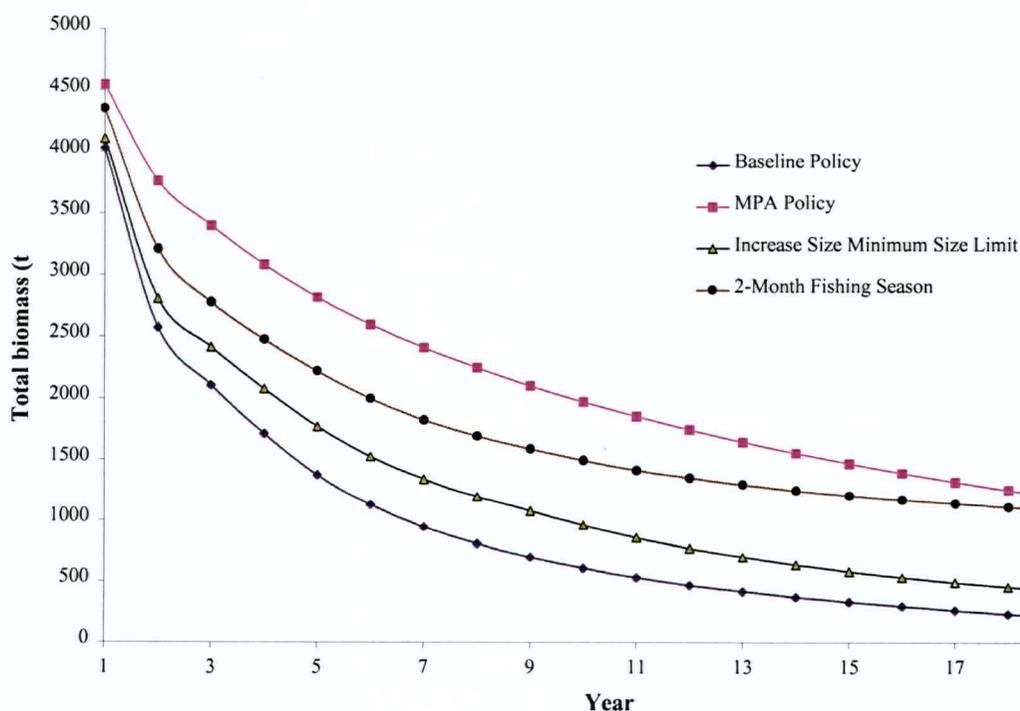


Figure 3.5. Biomass time series for a simulated 20-year fishery for lingcod in Howe Sound under 3 alternative fishing policies. The base line policy uses the current sports fishery regulations and no MPAs.

In the first year of the simulations, catch rates were highest for the baseline policy and lowest for the MPA policy (Figure 3.6). However, as time proceeded, catch rates for the baseline policy were lower than catch rates for the MPA policy. This switch occurred in year 6 (Figure 3.6). Catch rates for the MPA policy were roughly 3 times higher than the baseline policy during the last 10 years of simulation. An increase in minimum size limits also resulted in higher catch rates at the end of the time series, however, they were only twice as high as baseline simulations. The same result occurred when a 4-month fishing season was reduced to a 2-month fishing season.

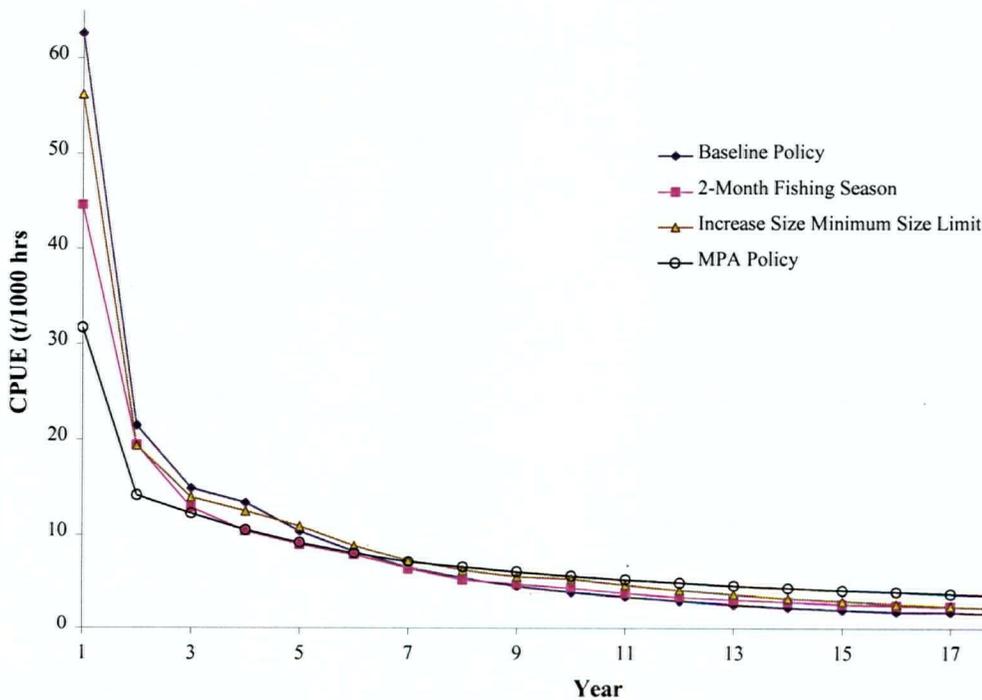


Figure 3.6. Catch rate time series for a simulated 20-year fishery for lingcod in Howe Sound under 3 alternative fishing policies. The base line policy uses the current sports fishery regulations and no MPAs.

In general, the use of MPAs as a management policy should reduce catch rates during the development of the fishery, but in the long run should result in high catch rates

because of a source population within the reserve exporting surplus biomass. The surplus biomass would be in the form of both new recruits to the fishery, and also fully recruited fish moving across the reserve area boundaries.

Discussion

The results of the FISHMOD simulations indicate that rates of fish movement and the distribution of fishing effort play an important role in the efficacy of Marine Protected Areas (MPAs). In this model, both the exchange rate across reserve area boundaries and the distribution of fishing effort affect stock dynamics. The system of MPAs present in Howe Sound is most effective if lingcod movement rates are essentially zero. Also, as the fishery responds more intensely to areas of high abundance (*i.e.*, the effort-distribution power parameter is > 1), the proportion of fishing effort distributed adjacent to reserve areas decreases at moderate to low movement rates. At high movement rates, the proportion of effort bordering reserves increases as the stock is eroded over the course of the 20-year fishery. Next I will relate the observations made from the simulations in FISHMOD to the actual sports fishery in Howe Sound.

To emulate the distribution of fishing effort in the recreational fishery, the gravity model weights the effort distribution in each cell based on two criteria. These two criteria are catch rates in the previous time step, and the vulnerable abundance of fish in the present time step. Weighting each location based on previous catch rates is equivalent to previous experience by anglers in a given location. For example, if anglers were lucky at one location a week ago, they're more likely to return to that location this

week, before exploring new fishing grounds. These types of choices are generally individual choices; however, the information could also be a result of information sharing among anglers. The second criterion pertains to the knowledge of the distribution of the stock. In the model, this is specified as the vulnerable biomass present in each cell. From an angler's perspective, the knowledge about the distribution of the stock is acquired through experience, and also information sharing among anglers. An omniscient angler would know the exact distribution of the stock; however, among all anglers there is a high degree of variability in the knowledge of stock distribution. As a result of this variability, some anglers catch more fish than others do (Hilborn, 1985). In general, the gravity model performs quite well at predicting the distribution of fishing effort, especially in commercial fisheries where the fishing ground attractiveness is a function of profit (Walters and Bonfil, 1999; Walters, 1999). However, in this application, the seasonal fishery has some interesting effects on the distribution of fishing effort.

Equation 3.2 behaves differently when the fishery operates continuously versus seasonally. This is particularly true when λ approaches 1, or the attractiveness is based entirely on the previous time step. In the case of a fishery that operates year-round, θ is updated every time step, and the fishery can "track" a moving stock. In the case of a seasonal fishery, θ is not updated during time steps when the fishery is closed, therefore, the fishery is not able to "track" the moving stock. Not being able to track a moving stock results in a learning process that has to occur in the first time step when the fishery reopens. Conversely, when λ approaches zero, fishing effort is distributed in exact proportion to the distribution of fish, provided the effort-distribution power parameter is

equal to 1. With perfect knowledge of the distribution of fish, there is no temporal effect and the fishery is very efficient at harvesting the resource. In both cases ($\lambda = 1$ or $\lambda = 0$), the catch rates decline over the 4-month fishing season; the difference is that when $\lambda = 0$, the catch rates decline in proportion to the decline of the stock.

The gravity model uses fish distribution and catch rates to distribute fishing effort. There are at least 2 other elements that are important for determining the distribution of fishing effort: the cost of fishing, and competition on the fishing grounds. In general, fishing grounds in close proximity to the dock are cheaper to fish (Gillis *et al.*, 1993). Travel to distant fishing grounds increases the cost of fishing, in both time spent travelling (*e.g.*, burning more fuel), and a reduction in time fishing due to increased travel time. My simulations assumed that the costs of fishing in all cells are equal. As a result, effort allocation at distant locations was just as likely as effort allocation at nearby locations, and in practicality costs of fishing in nearby areas was undoubtedly cheaper than fishing in distant areas.

Competition between anglers on the same fishing ground is both direct and indirect. Indirect competition occurs when 2 or more anglers visit the same fishing grounds on different days. In this case the first angler reduces the vulnerable stock on the ground and subsequent anglers are left with a smaller biomass pool to harvest. Direct competition occurs when 2 or more anglers visit the same fishing ground on the same day. This type of competition is most often referred to as interference competition (Gillis *et al.*, 1993). Gillis *et al.* (1993) used both of these competition arguments in an application of the ideal free distribution (IFD) theory for spatial allocation of fishing effort. The crux of their paper is that the catch rates in each fishing ground will

equilibrate via the spatial distribution of fishing effort due to competition among fishermen. The gravity model in FISHMOD behaves similarly to the IFD theory as applied by Gillis *et al.* (1993), however, it does not account for interference competition within each cell.

Modelling the proportion of effort that is distributed along the border of marine reserve areas is of great importance because fishing along the boundary of a reserve area has the potential to reduce the efficacy of the MPA (Walters *et al.*, 1998; Walters, 1999). There is some evidence that the distribution of fishing effort will shift disproportionately to the boundaries of the marine reserve in response to the perception of higher catch rates due to spill over from the reserve (Alcala, 1988; Alcala, and Russ 1990; Russ and Alcala, 1996 Guénette *et al.*, 1998). One of the best examples of boundary fishing is the “Plaice Box” in the North Sea (ICES, 1994). Also, the distribution of scallop fishing boats off the Coast of Maine have been monitored with new GPS technology, and roughly 90% of the fishing effort occurs along the boundary of closed conservation areas (Dr. S. Murawski, pers comm). In the FISHMOD simulations, > 50% of the total fishing effort is distributed along the boundary of the marine reserves occurs at low movement rates (move = 1) and the stock size has be reduced substantially (< 50% of initial stock size). Both the simulations and evidence from the literature demonstrate that a significant amount of fishing effort will occur along the boundaries of marine reserves when stocks are depressed and dispersal distances are small.

It is intuitively obvious that MPAs will work best as a fisheries management tool/conservation tool when the animals they are intended to protect do not move. For sessile fish and invertebrates that are anchored to the substrate, or have no means of

locomotion, movement out of the reserve is of no concern. The rate of movement in mobile animals is of primary concern (Polacheck, 1990); however, it is the rate of exchange across the reserve boundary that increases risk of mortality due to fishing. Therefore, the size of the reserve area is important in determining the rate of exchange across the reserve boundary. Larger reserve areas will have a lower overall exchange rate, and will be more effective for mobile animals than several small reserve areas of equal area.

MPAs appear to be much safer options than alternative management strategies, such as size restrictions or seasonal closures. In the simulations presented in Figure 3.5 and 3.6, the effect of MPAs is substantially different. In these simulations, roughly 30% of the stocks were partially protected from fishing (3 stocks were present in areas with marine reserves). Alternative approaches to reducing fishing mortality (reduce length of fishing season, or increase size limits) just seemed to prolong the misery. In practice, size limits are difficult to enforce. In Georgia Strait creel survey data, > 40% of lingcod landed by the sports fishery are less than the 65-cm size limit (Beamish, *et al.*, 1994). Also, seasonal closures are even more difficult to enforce. Currently, the lingcod sports fishery is closed from October to May, however, retention of rockfish and other sports fish are still permitted. Both rockfish and kelp greenlings (*Hexagrammos decagrammos*) live sympatrically with lingcod, and are caught using similar fishing gear. Discards of lingcod during the seasonal closures are reported, however, there have been no studies to my knowledge that quantifies the release mortality rate.

Conclusion

Movement across reserve area boundaries plays an important role in the spatial distribution of fishing effort. As exchange rates increase across reserve boundaries, a higher proportion fishing effort shifts to the borders of the reserve area, unless movement rates are so high that biomass does not accumulate in the reserve areas. Knowledge of stock distribution, and the amount of information sharing among anglers, also effects spatial distribution of fishing effort. Finally, even small MPAs are better policy options than changes in size limits and seasonal restrictions. Although it may be difficult to monitor and enforce closed areas, it is also difficult to monitor and enforce size limits and seasonal closures.

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