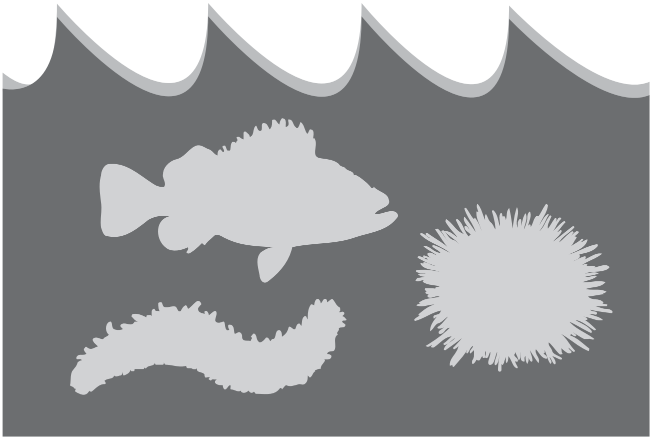
21ST LOWELL WAKEFIELD FISHERIES SYMPOSIUM

FISHERIES ASSESSMENT AND MANAGEMENT IN DATA-LIMITED SITUATIONS



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About the Symposium

The goal of the symposium “Assessment and Management of New and Developed Fisheries in Data-limited Situations,” held October 22-25, 2003, in Anchorage, Alaska, was to bring together scientists, fishery managers, and policy makers to synthesize historical track records, share recent advances, and discuss future avenues for the management of newly developing and small-scale fisheries under data-limited situations.

Expanding worldwide demand for seafood products is not limited to fishery resources with rich histories of stock assessment and fishery research. As most of the world’s large marine fisheries are fully exploited or overfished, new fisheries are being developed on marine species whose biology, productivity, and ecological relationships are little known.

Data-limited situations create challenges for fishery managers responding to societal demands to develop new fisheries while striving for precaution under the Code of Conduct for Responsible Fisheries. This has led to new applications of decades-old simple production models, spawned the development of new assessment techniques with meager data requirements, and led to creative fishery management schemes, including adaptive approaches, risk-averse methods such as establishment of no-fishing refugia, comanagement to share responsibilities among vested parties, and rights-based management systems.

The symposium was coordinated by Sherri Pristash, Alaska Sea Grant College Program, University of Alaska Fairbanks, with assistance by the organizing committee: Vince Gallucci, University of Washington; Doug Hay, Fisheries and Oceans Canada, Pacific Biological Station; Gordon Kruse, University of Alaska Fairbanks, Fisheries Division; Bill Wilson, North Pacific Fishery Management Council; Ian Perry, Fisheries and Oceans Canada, Pacific Biological Station; Randall Peterman, Simon Fraser University; Tom Shirley, University of Alaska Fairbanks, Fisheries Division; Paul Spencer, NOAA Fisheries, Alaska Fisheries Science Center; and Doug Woodby, Alaska Department of Fish and Game.

Symposium sponsors are Alaska Sea Grant College Program; Alaska Department of Fish and Game; North Pacific Fishery Management Council; Wakefield Endowment, University of Alaska Foundation; and NOAA Fisheries.

The Lowell Wakefield Symposium Series and Endowment

The Alaska Sea Grant College Program has been sponsoring and coordinating the Lowell Wakefield Fisheries Symposium series since 1982. These meetings are a forum for information exchange in biology, management, economics, and processing of various fish species and complexes, as well as an opportunity for scientists from high-latitude countries to meet informally and discuss their work.

Lowell Wakefield was the founder of the Alaska king crab industry. He recognized two major ingredients necessary for the king crab fishery to survive—ensuring that a quality product be made available to the consumer, and that a viable fishery can be maintained only through sound management practices based on the best scientific data available. Lowell Wakefield and Wakefield Seafoods played an important role in the development and implementation of quality control legislation, in the preparation of fishing regulations for Alaska waters, and in drafting international agreements for the high seas. In his later years, as an adjunct professor of fisheries at the University of Alaska, Lowell Wakefield influenced the early directions of Alaska Sea Grant. The Wakefield symposium series is named in honor of Lowell Wakefield and his many contributions to Alaska's fisheries. In 2000, Lowell's wife Frankie Wakefield made a gift to the University of Alaska Foundation to establish an endowment to continue this series.

Proceedings Acknowledgments

This publication presents forty-six symposium papers. Each paper was reviewed by two peers, and was revised according to recommendations by associate editors who generously donated their time and expertise: Gordon Kruse, Vince Gallucci, Doug Hay, Ian Perry, Randall Peterman, Tom Shirley, Paul Spencer, Bill Wilson, and Doug Woodby. The publisher thanks the editors and the authors of the papers.

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Fisheries Assessment and Management in Data-Limited Situations
Alaska Sea Grant College Program • AK-SG-05-02, 2005

Overview of World Status of Data-Limited Fisheries: Inferences from Landings Statistics

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Abstract

Data-limited fisheries are here considered to be fisheries lacking sufficient reliable biological information to infer the exploitation status of the targeted stocks. Considering species-specific catch data as the common minimum available data for assessing the status of a stock, in this paper we use the taxonomic breakdown of the reported landings statistics to FAO to make an approximate inference of data limitation of fisheries by region, country, and taxonomic groups. The paper also explores the possibility of extracting meaningful biological information from fisheries landings by applying a Bayesian approach to two selected fisheries. The contribution of data-poor fisheries to the world landings from marine capture fisheries is relatively low, but increasing (from 20 to 30% of world landings in the last 50 years). However, data limitation can be a substantial problem at the regional and country level, especially in areas with high species diversity, small stock sizes, and where fisheries play an important role for food security. Preliminary modeling results indicate that catch data, when combined with prior information about the dynamics of similar species/stocks and fisheries, could be useful for informing fisheries management in data-limited situations.

Introduction

The 25th Session of the Committee on Fisheries (COFI) recognized that, although information on the status and trends of fisheries is fundamental to the mandate of FAO, there are serious shortcomings in the reporting of information and many problems of data quality in fisheries. The availability and quality of information on fisheries is often so poor that it is very difficult not only to draw conclusions, but also to formulate sound policies for responsible fisheries management.

The problem of data limitation in fisheries can be examined in relation to the information required in a management plan. Ideally, fisheries managers should be engaged in a series of iterative activities (Cochrane 2002), including the definition of the geographical and institutional scope of the management plan; the definition of objectives, strategies, reference points, and performance measures; the formulation of rules (measures and controls) to achieve the desirable objectives; and the monitoring, assessment, and review of fisheries and resources status. Success in meeting management objectives depends largely on the availability and flow of information between these activities. There are many ways in which the management of a fishery can be limited by lack of data and information. Depending on the objectives of a management plan, different types of data are required to assess the performance of alternative management strategies. In practice, fisheries management involves balancing diverse (and often conflicting) interests or objectives, and the data/information requirements reflect this diversity (FAO 2003a). Nonetheless, in spite of the management objectives, information about the status of the resource base (fish stocks and supporting ecosystems) is a crucial indicator for the sustainable management of a fishery. In this paper, we concentrate our analysis and definition of data-limited fisheries on those fisheries considered to lack adequate reliable information to infer the exploitation status of the targeted resources.

Information about the status of a fishery resource can come from different sources: monitoring the fishery (landing, catch, and effort data); biological surveys; information from resource users; and from information on similar resources in other locations. Often the only data available to infer the status of a fishery in a particular location are the landings in weight or number of the targeted species. In such cases, catch data form the basis for fisheries assessment and management.

This paper is presented in two parts. First we use the taxonomic breakdown of landings statistics reported to the Food and Agriculture Organization (FAO) to evaluate the extent of data limitation of fisheries in a given country/region. In this evaluation it was considered that species-specific landings data are common minimum data for managing a fishery in a given location. This measure is only approximate because in many places fisheries exploit different stocks of the same species, and the stock-specific catches are not often reflected by the resolution of the landings statistics. We also recognize that landings and catches are not necessarily the same because of problems of data reporting, discards, and illegal fishing. Using landings data reported to FAO, data limitation in fisheries around the globe is described, and some factors behind the limitation are discussed. In the second part of the paper we explore the use of a Bayesian approach to infer meaningful biological information from landings data alone using prior information about the dynamics of fishing effort and of marine populations. The approach is tested using data from two previously assessed fisheries.

Methods

Data limitation in fisheries around the world

As part of its mandate FAO compiles information and data on various aspects of food and agriculture from all countries. Country-level fisheries data are collected through tailored questionnaires sent annually to member countries. Data reported by countries are supplemented as much as possible with information from other sources, including regional fishery bodies, field projects, independent surveys, and literature reviews (FAO 2001). Overall, FAO landings statistics are expected to include all available data by country and thus reflect the quality of their system of fisheries statistics. Reported data are categorized by species, genus, family, or higher taxonomic levels into 1,291 statistical categories called species items. Some countries tend to report their catch only in higher taxonomic categories and sometimes only as “miscellaneous fishes” or “marine fish not elsewhere included.” Landings not defined to species are termed here “nonspecific fisheries landings.”

In this study, marine fisheries landings statistics reported to FAO from 1950 to 2001 were disaggregated by taxonomic categories and spatially by ocean, FAO Statistical Area, and country. The percentage of the total landing volume composed of nonspecific items, i.e., items identified only to genus or higher taxonomic levels, was computed for each geographical region. The same percentage was also calculated for major taxonomic groups, i.e., fish, crustaceans, cephalopods, and other mollusks (excluding cephalopods). These percentages are used here as indices to describe data limitation in fisheries.

Example of an approach to extract meaningful biological information from landings data

The approach explored in this paper is based on the assumption that, in the absence of effective management, fisheries normally follow a pattern of development where the relative rate of increase in catch is zero at an undeveloped stage, increases rapidly during a development phase, drops to zero again when a mature stage has been reached, and becomes negative during a senescent phase (Caddy and Gulland 1983). Underlying the trend in catches is a pattern of change in fishing effort, which typically increases rapidly during the development phase. The effort and resulting fishing mortality can continue to increase, stabilize, or decline when the maturity phase is reached, depending on diverse factors such as changes in technology that make harvest more efficient, market conditions that affect the bioeconomic equilibrium, or regulatory regimes. While information and data exist about changes in fisheries landings over time, not much has been done to characterize the dynamics in effort and fishing mortality (Hilborn and Walters 1992, Seijo et al. 1998). Given the different possible trajectories of effort and in an attempt to keep the number of estimable parameters low, we propose two models that can be used to describe the observed pattern of change in fishing effort with time: a linear and logistic model. Effort and biomass equations are combined to develop a model that simulates catches over time. The fundamental assumption of the model is that time series of catch data contain information on both fishing effort and stock biomass dynamics. The model evaluates the change in catches to infer stock status, production, and exploitation rate.

In the linear model fishing effort increases by a constant amount with time:

$$E_{t+1} = E_t + xE_0 \tag{1}$$

where E_0 is the effort at the early phase of fishery development and x is a multiplier that defines the amount of increase in mortality with time. This model is particularly suited to describe fisheries in the developing stage or where effort continues to increase after the maximum catch is reached.

In the logistic model effort increases as a function of the stock abundance:

$$E_{t+1} = E_t[1 + x(I_t/I_e - 1)] \tag{2}$$

where x is the intrinsic rate of increase in effort, I_t is catch per unit effort (CPUE) in year t, and I_e is the CPUE at the bioeconomic equilibrium. According to this model fishing mortality will increase with time until the equilibrium abundance is reached, and will decrease if the stock continues to decline. As such, the model is useful to describe fisheries that went through the developing, mature, and senescent phases and may be more appropriate for commercial fisheries where economic performance will tend to drive effort.

The standard catch equation is:

$$C_t = qE_tB_t \tag{3}$$

where C_t is catch in year t, q is the catchability coefficient, E_t is effort in year t, and B_t the stock biomass in year t. Biomass dynamics is modeled according to the Schaeffer model:

$$B_t = B_{t-1} + rB_{t-1}(1 - \frac{B_{t-1}}{K}) - C_{t-1} \tag{4}$$

where r is the intrinsic rate of increase in biomass and K is the virgin biomass or stock at carrying capacity.

Expressing the temporal changes in fishing effort according to equations 1 and 2, and assuming that $I_t = B_t q$, then equation 2 can be expressed as:

$$E_{t+1} = E_t[1 + x(B_t/B_e - 1)]$$

where B_e is the biomass at a "bioeconomic" equilibrium, being $B_e = Ka$, where $0 < a < 1$.

Assuming that q is constant between years, then:

$$qE_{t+1} = qE_t + q(xE_0)$$

$$qE_{t+1} = qE_t[1 + x(B_t/B_e - 1)]$$

Let $qE_t = P_t$ = the proportion of the biomass caught in year t, then

$$P_{t+1} = P_t + xP_0 \tag{5}$$

$$P_{t+1} = P_t[1 + x(B_t/B_e - 1)] \tag{6}$$

Therefore, combining equations 6 (or 5) and 4 into equation 3 yields:

$$C_{t+1} = P_t[1 + x(\frac{B_t}{B_e} - 1)]\,[B_t + rB_t(1 - \frac{B_t}{K}) - C_t] \tag{7}$$

Assuming that the stock is lightly fished at the beginning of the time series (i.e., $B_0 = K$), and that the first catch (C_0) is measured without error (so $P_0 = C_0/B_0$), then four parameters must be estimated when effort model (2) is used (r, K, x, and a) or three parameters when effort model (1) is used (r, K, and x). We attempt to estimate parameters by fitting equation 7 (or the equivalent with the linear effort model) to time series of annual landings. We test how the performance of the model varies as informative priors for parameters r, K, x, and a are included in a Bayesian estimation procedure (see below). The model was tested using data from two fisheries previously assessed with biomass dynamic models and catch and effort data (Table 1). The performance of the model was evaluated comparing the estimated and "true" parameters and reference points listed in Table 1.

Table 1. Parameters and reference points of fisheries used to test the performance of the model. *K* and *msy* are measured in metric tons (thousands).

Fishery	r	K	x	msy	F/F_{msy}	B/B_{msy}
Yellowfin tuna	0.80	819	0.50	152	0.87	1.18
Namibia hake	0.39	2,709	0.40	266	1.89	0.77

Reference points F/F_{msy} and B/B_{msy} refer to ratio between the most recent fishing mortality and stock biomass compared to F_{msy} and B_{msy}, respectively. Only catch data from the period when the fisheries were unmanaged were used in the analysis: yellowfin tuna (1950-1993, ICCAT 2001), Namibia hake (1965-1976, Hilborn and Mangel 1997).

The likelihood function used for the estimation (Punt and Hilborn 2001) is:

$$L(D \backslash \underline{\theta}) = \prod_t \frac{1}{\sigma C_t \sqrt{2\pi}} \exp[-\frac{1}{2\sigma^2}(\ln C_t - \ln C_{t,est})]^2$$

θ is assumed to be known and equal to 0.4. $C_{t,est}$ is the estimated catch in year t, as calculated from equation 7. Informative prior probabilities for parameters r, K, x, and a are assumed to be normally distributed with means equal to the "true" parameter values in Table 1 and coefficient of variation of 40%. Alternatively, we used uniform probabilities for the same parameters as a source of non-informative priors. In this case parameters are assigned equal probabilities in the following ranges: yellowfin tuna ($0.1 < r < 1.5$; $100 < K < 5{,}000$; $0 < x < 10$; $0 < a < 1$); hake ($0.1 < r < 1.5$; $500 < K < 10{,}000$; $0 < x < 2$; $0 < a < 1$). The likelihoods were set to zero for parameter values below the minimum and above the maximum limits of the uniform prior distributions. The posterior probability distribution for each alternative parameter value was computed by combining prior information with the estimated likelihoods using Bayes rule. The computation of posterior probabilities was carried out using a Markov Chain Monte Carlo method implemented in an Excel spreadsheet (Punt and Hilborn 2001). The method generally converged to the posterior distributions after 50,000 runs. We disregarded the first 10,000 runs as a "burn in" period and selected the parameter vectors after every fortieth run as the basis to construct the posterior distributions.

Results

Data limitation in fisheries as indicated by FAO landings statistics

The global trend in nonspecific fisheries landings (i.e., landings not defined to species) shows an increase from approximately 20% to 30% during the 50 years of data available (Fig. 1). The relatively small percentage of nonspecific landings volume globally reflects the fact that a few fish species dominate world landings (the top 10 species in 2000 accounted for ca. 35% of world marine capture fisheries; FAO 2002a). However, the analysis of the same data by oceanic basins and FAO Statistical Areas reveals substantial geographical variation in data limitation (Fig. 1). The highest percentages of nonspecific landings are in the Indian Ocean (both Western and Eastern Indian Ocean show similar trends) and in the Western Central Pacific, where over 60% of the landings are reported in highly aggregated items. Overall, the Atlantic Ocean shows the lowest percentages of nonspecific landings among the oceans, but in the breakdown by fishing statistical areas the large regional differences in data quality are evident. Fisheries landings data are less limited in the Northeast and Northwest Atlantic while the worst areas in recent years are the Eastern Central, Western Central, and Southeast Atlantic. The same regional differences are evident in the Pacific Ocean. While the Western Central Pacific present up to 80% of the total landings volume reported as nonspecific items, in the Southeast Pacific this percentage has always been under 20% because of the dominance in landings of fisheries for very abundant small and medium pelagics (sardine, anchoveta, and jack mackerel) off the west coast of South America. The Mediterranean and Black Sea show a general improvement in fisheries statistics over time but still ca. 30% of the reported landings are nonspecific. The Antarctic fishing areas are overall very specific so the percentage of nonspecific records is normally low, but with some variability in the Antarctic areas of the Pacific Ocean (for instance, in 1989 100% of the reported 1,100 tons captured in the area were recorded as Myctophidae). In general, we found that the index of data limitation used here reflects the available information on the exploitation status of stocks in each FAO statistical area (FAO 2005).

Further analyses of these data were undertaken to describe the characteristics of fisheries in data-poor situations and to investigate some of the factors leading to poor quality of data. The quality of fisheries statistics correlates better with the number of fish species caught for human use than with the total fisheries production in the FAO Statistical Areas (Fig. 2). Areas of high biological diversity, expressed by the diversity of species commercially harvested, in general have a higher percentage of landings unidentified to the species level. This leads to two conclusions. First, the importance of an area in terms of its contribution to world supply of capture fisheries is not a factor influencing fisheries assessment

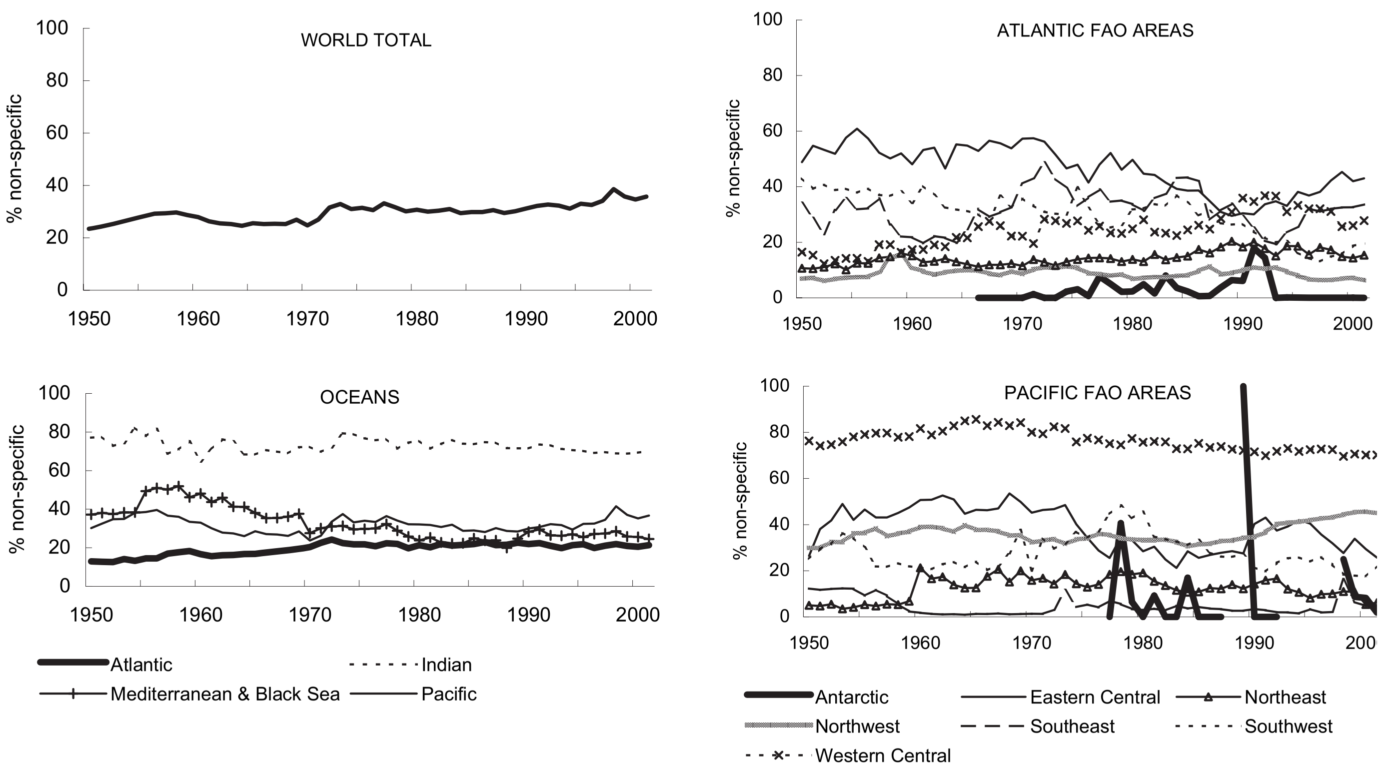

Figure 1. Trends in percentage of fisheries landings non-identified at the species level by oceans and FAO Statistical Areas.

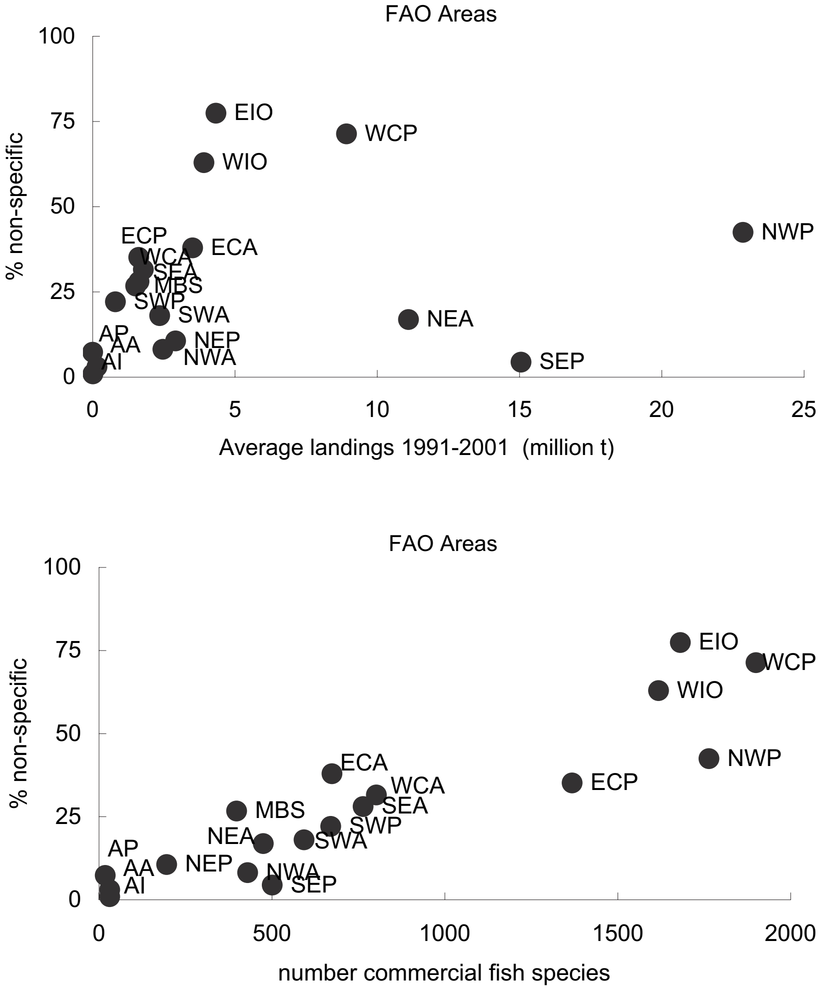

Figure 2. Relationship between percentage of fisheries landings nonspecific against average landings from 1991 to 2001 (upper panel) and the number of fish species used for diverse human uses (lower panel). The number of fish species by FAO Statistical Area was obtained in FishBase (Froese and Pauly 2000). FAO Statistical Areas are indicated by their initials: AA, Antarctic Atlantic; AI, Antarctic Indian; AP, Antarctic Pacific; ECA, Eastern Central Atlantic; EIO, Eastern Indian Ocean; ECP, Eastern Central Pacific; MBS, Mediterranean and Black Sea; NEA, Northeast Atlantic; NEP, Northeast Pacific; NWA, Northwest Atlantic; NWP, Northwest Pacific; SEA, Southeast Atlantic; SEP, Southeast Pacific; SWA, Southwest Atlantic; SWP, Southwest Pacific; WCA, Western Central Atlantic; WIO, Western Indian Ocean; WCP, Western Central Pacific.

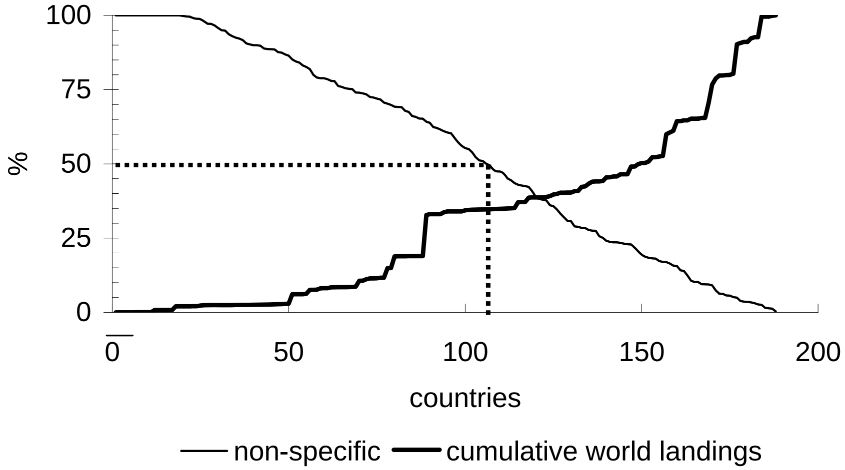

Figure 3. Percentage of nonspecific fisheries landings by country and contribution to global capture fisheries production of each country that reports fisheries data to FAO.

and monitoring efforts at a regional level. Second, there is a general limited capacity to monitor fisheries in areas of high biological diversity.

The geographical extent of data limitation in fisheries was examined further by breaking down the information by countries. One hundred and six (56%) of the 188 countries that submit landings statistics to FAO report more than 50% of their fisheries landings as nonspecific items (Fig. 3). Combined, these countries are responsible for ca. 40% of world fisheries capture production. They are mostly concentrated in Asia, Africa, Oceania (islands of Central and South Pacific), the Caribbean, and Central America. With the exception of Central America, where the average contribution of fish to the total supply of animal protein to the population (5%) is below the world average of about 15% (data for the year 2003 in the FAO Statistical Database on Food Supply; http://faostat.fao.org), all other regions are composed of countries that depend heavily on fisheries as a source of animal protein supply (> 15%). Paradoxically, the regions where fisheries play a major role for food security, i.e., where harvested fish provides most of the nutritious food to meet the dietary needs and food preferences of the population, are also the ones where generally the management of capture fisheries are most data limited. The paradox rests on the assumption that the limited capacity to monitor the status of fisheries resources in these areas also impairs the capacity of countries to plan the rational use of resources so as to secure the long-term availability and stability of food supply.

By defining a fishery as a unique combination of fished species and country in the FAO statistics database, it was also possible to evaluate the extent of data limitation by fishery size (Fig. 4). Despite the dominance in numbers of small to medium sized fisheries, a few large fisheries lead the global yield and present the best landings data. Results support the hypothesis proposed by Mahon (1997) that, regardless of the numerical dominance and greater socioeconomic dependence on small to medium stocks at a global level, most fisheries research and management effort has been directed at the largest stocks. The problem of data limitation is compounded by the fact that the preponderance of small stocks is in tropical areas of high biological diversity (Fig. 1), where fisheries, mostly small-scale, are an important source of subsistence and food for low income countries.

Finfish fisheries presented a steady increase in the percentage of landings not identified to the species level from 21% in 1950 to 32% in 2001 (Fig. 5). The trend in finfish landings mirrors that of world landings (Fig. 2) since fish dominates the world landings volume. The nonspecific records of fish landings are dominated by the generic category “marine fish not elsewhere included,” followed by demersal fish stocks and, with less importance, pelagic fish stocks. Landings statistics for crustacean and cephalopod fisheries have always been of poorer quality than those for finfish. Since the mid-1970s there has been a general improvement in the statistics for crustaceans and a worsening of the statistics for mollusk and cephalopod fisheries. However, approximately half of the reported landings of fisheries directed to the three invertebrate groups in the most recent years are not species specific.

What can be learned about the status and dynamics of stock from catch data alone?

The previous section highlighted serious limitations in data availability throughout the world. Nevertheless, catches are the minimum and often the only information available to inform management decisions in many fisheries. Therefore, in this section we begin to explore the question of whether catch data alone can be used to provide meaningful biological information about a fish stock, using as examples the fisheries for yellowfin tuna and Namibia hake.

Yellowfin tuna

The data used in the test were restricted to the period from 1950 to 1993, when the fishery for yellowfin tuna, *Thunnus albacares,* was unregulated. Measures to limit effort were first implemented in 1994 (ICCAT 2001). The fit of the linear effort model was better than the logistic model (Fig. 6;

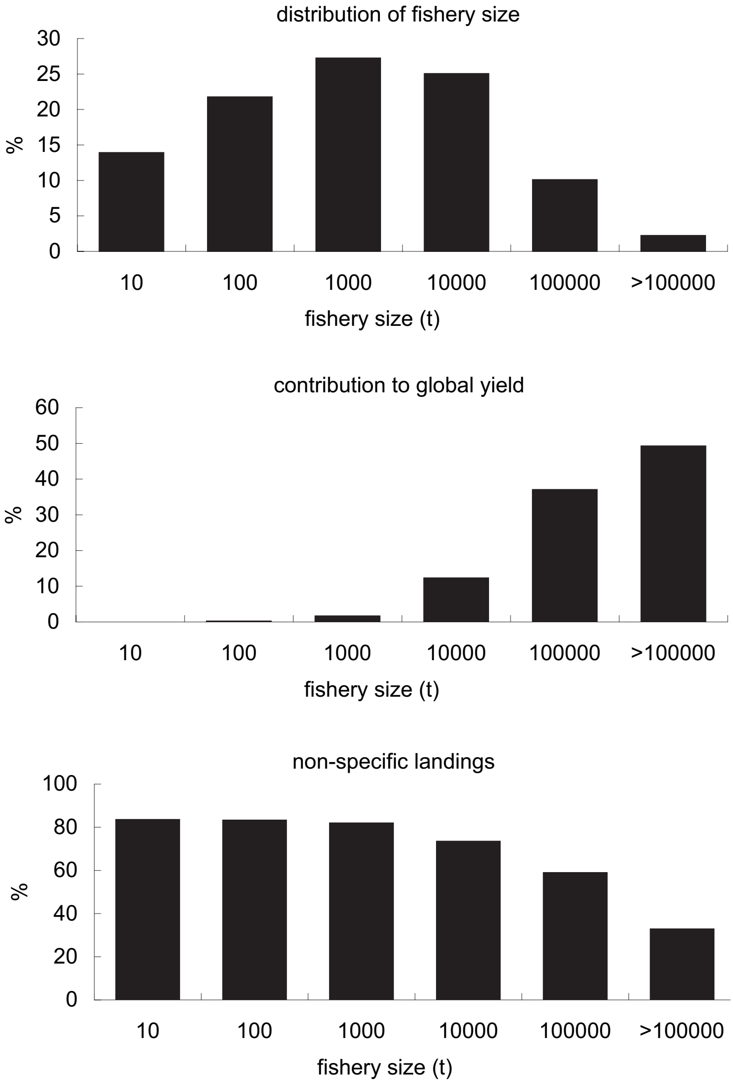

Figure 4. **Distribution of fishery size, percentage contribution to global capture fisheries production, and quality of information (measured as the percentage of fisheries with nonspecific landings) by fisheries size categories at the global level. Fisheries were grouped in size categories according to their most recently reported annual landings (2000-2001). In this analysis a fishery is defined by a unique combination of species item and country, excluding the highly aggregated species items (such as "miscellaneous marine fish").**

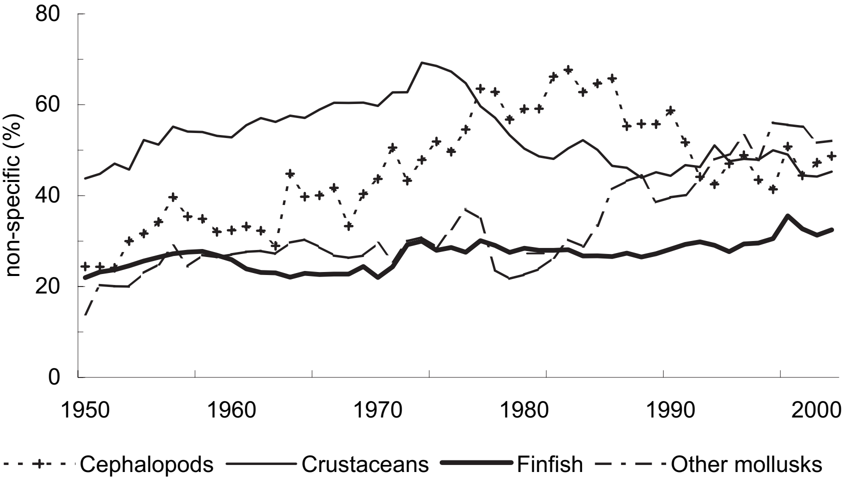

Figure 5. Percentage of fisheries landings nonspecific by main fisheries taxonomic groups.

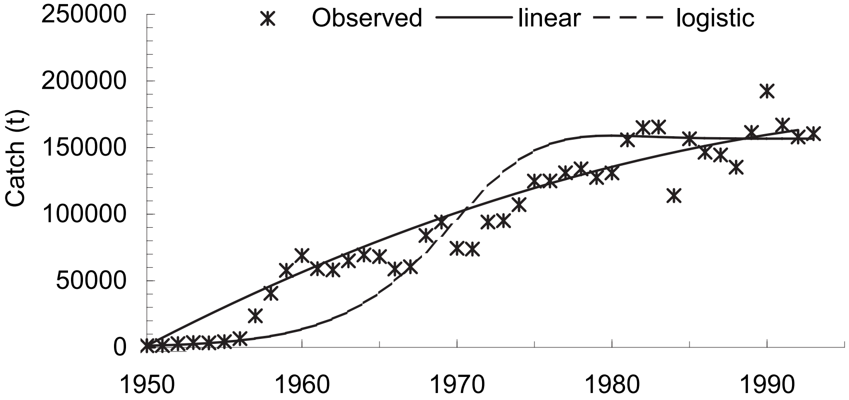

Figure 6. Best fit models to observed catch data of yellowfin tuna using the linear and logistic effort models. The goodness of fit was measured using the residuals of untransformed data because they were more homoscedastic than the residuals of log-transformed data.

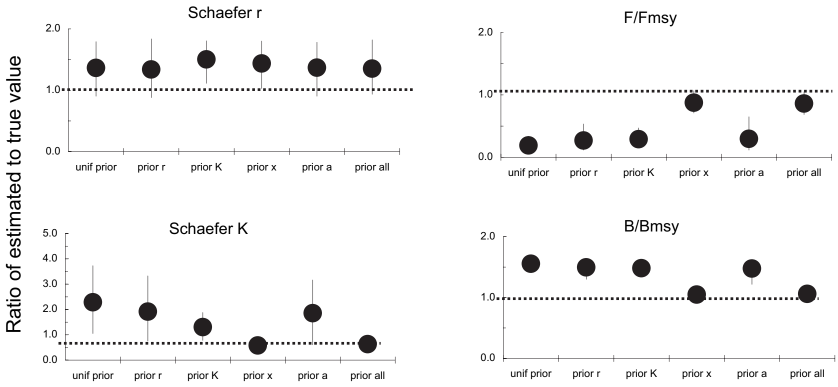

Figure 7. Performance measures of the logistic model applied to yellowfin tuna data. The y-axis is the ratio between predicted and the "true" value of parameters obtained in their respective stock assessments. The x-axis indicates the prior information used in the Bayesian analysis. Dots are average values and the lines indicate the 10% and 90% percentiles.

sum of squares linear: 9.09×10^{9}; logistic: 2.03×10^{10}), although both models are miss-specified to some degree because neither meet the criteria of randomness in the distribution of residuals at the 95% confidence interval using a runs test (randomness linear: –3.53; logistic: –3.73). Predictions of the trend in catch data differ according to the model used. While the linear model predicts a continued increase in catches over the time period of the analysis, the logistic predicts that catches would have stabilized since the early 1980s.

Despite providing a poorer fit to the data, the logistic effort model resulted in better estimates of parameters than using a linear model and those are shown here. Predicted biological and policy parameters for the logistic model are shown in Fig. 7. The model overestimated r (between 27 and 50% higher than ICCAT's estimates) irrespective of introducing informative prior information about the parameter. With prior information about the effort parameter x, the model underestimated K. In all other cases K was overestimated, even when priors on K were introduced. The critical parameter for the logistical model was in this case x, as good priors on x produced relatively good estimates of all policy parameters. In all other cases the model predicted that catches come from a much larger and productive stock with biomass close to the virgin stock size, a result that could indicate insufficient contrast in the data.

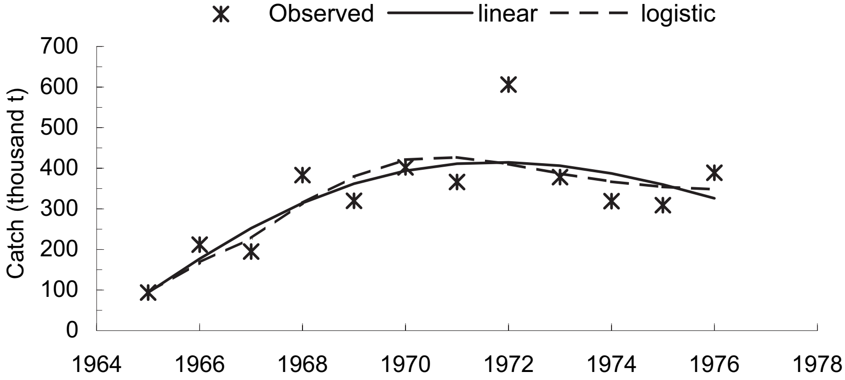

Figure 8. Best fit of models to observed catch data of Namibian hake using the linear and logistic effort models. The goodness of fit was measured using the residuals of untransformed data because they were more homoscedastic than the residuals of log-transformed data.

Namibian hake

To test the model on the Namibian hake (*Merluccius capensis* and *M. paradoxus*) fisheries we used data only from 1965 to 1976, when the fishery was unregulated. In this case, the fits were very similar for both effort models (Fig. 8; sum of squares linear: 6.07×10^4; logistic: 6.14×10^4) and the test of randomness indicated that both models were well specified (randomness linear: 0.62; logistic: 0.62). In fact, fishing mortality increased more or less linearly during this period (Hilborn and Mangel 1997). The models predicted a decline in catches after peaking in the early 1970s.

In this case, the linear effort model produced better parameter estimates than the logistic model. Results obtained with the linear model are shown in Fig. 9. Parameter r was overestimated (ca. 50% higher than Hilborn and Mangel's estimates) without informative priors about the parameter. The same was observed for parameter K, which was overestimated (between 10 and 35% higher) without informative priors. In general, the performance of the model in predicting policy parameters was substantially better than in the case of yellowfin tuna. Even without prior information the model was able to predict F/F_{msy} and B/B_{msy} close to the values assessed with catch and effort data.

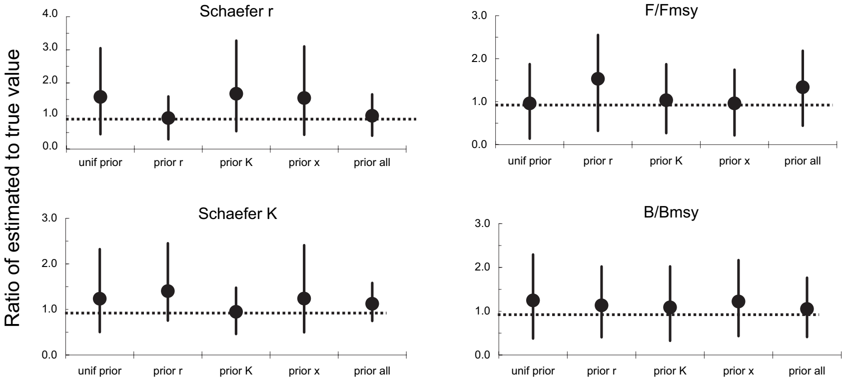

Figure 9. Performance measures of the linear model applied to the Namibian hake. The y-axis is the ratio between predicted and the "true" value of parameters obtained in their respective stock assessments. The x-axis indicates the prior information used in the Bayesian analysis. Dots are average values and the lines indicate the 10% and 90% percentiles.

Discussion

Results obtained by examining the fisheries landings statistics reported to FAO indicated some general characteristics of fisheries in data-limited situations. Although their importance to the world supply of marine capture fisheries is relatively low, but increasing (20-30%), data limitation can be a substantial problem at regional and country level. In terms of targeted resources, data limitation is more pronounced in invertebrate fisheries, despite their typical higher value. Among the finfish fisheries, the demersal fisheries have always been more deficient in specific landings data than the pelagic fisheries. Even so, a large proportion of finfish landings are reported as a generic category "marine fish not elsewhere included." Data limitation is more prominent in areas with high species diversity and small stocks where fisheries play an important role for food security, such as in many tropical, low-income countries of Africa, Asia, Oceania, and the Caribbean. The development of more cost-effective monitoring and assessment methods is therefore particularly urgent in these cases.

Recognizing the problem of data limitation in fisheries, the 25th Session of the Committee on Fisheries endorsed a draft "Strategy for Improving Information Status and Trends of Capture Fisheries" (FAO 2003b). The strategy calls the attention of states to the fact that many small-scale fisheries and multispecies fisheries, particularly in developing countries, are not well monitored. As a result, these fisheries are probably underestimated and underrepresented in current fisheries status and trends and consequently they are not adequately considered in the development of plans and policies for fisheries. One of the key actions of the strategy is to seek support for the development of cost-effective methods to acquire and validate data on small-scale and multispecies fisheries, including approaches for data-poor situations and participatory processes that closely associate resource users and their organizations to the data collection schemes (FAO 2003b).

As pointed out by FAO (2002b) the proportion of catch identified at the species level decreased during recent decades in response to the depletion of large stocks, the diversification of fisheries, and the decrease in financial support for the development and maintenance of national fishery statistical systems. The deterioration of data collection systems is occurring during a period when the statistical requirements have been increasing dramatically for bycatch and discards, fishing capacity, illegal fishing, among other socioeconomic data (FAO 2002b). Data limitation in fisheries can also be related to other inter-related factors, among them we highlight: (1) difficulty to monitor and assess fisheries in tropical areas of high biological diversity, dominated by multispecies and multi-fleet small-scale activities, where conventional fisheries assessment methods are not suited (Cochrane 1999, Berkes et al. 2001); (2) tendency of states to allocate human and financial resources preferentially to large and economically important fisheries (Mahon 1997, NMFS 2003); (3) management regimes that create incentives/disincentives for data gathering and reporting (OECD 1997); and (4) the weak institutional linkages between research and small-scale fisheries communities in many data-poor areas (SFLP 2003).

Catch information alone is a poor indicator to infer the exploitation status of a fishery and its potential productivity, but when combined with prior information about the dynamics of similar species/stocks and fisheries we suggest it could provide meaningful information, at least in some cases, to inform management decisions in data-limited situations. Other similar attempts have been made by Grainger and Garcia (1996) and Gaertner et al. (2001) to extract information on stock status and trend using catch data alone. Both those methods evaluate the change over time of the relative rate of catch increase to determine when a stock has reached an overfished status. The methods do not, however, provide information on the population dynamics of the resource or on biologically based reference points, in contrast to the Bayesian approach considered in this paper.

Some preliminary conclusions can be drawn regarding the model and information content of catch data. First, the performance of the model is sensitive to the assumptions about effort dynamics. Models using linear and logistic effort equations were not always well specified to describe the changes in catches over time (case of the yellowfin tuna), resulting in poor estimates of biological and policy parameters. The fact that the linear model provided a better fit to the data for yellowfin tuna but the logistic model provided better parameter estimates, suggests that an alternative, better-specified effort model could have been more appropriate. Only two types of effort models were applied in this study, and more appropriate effort dynamic models could substantially improve the performance of the combined effort-stock dynamic model. That could include making more realistic representations about how catchability changes in response to changes in technology, behavior of fish and fishers and environmental processes (Arreguin-Sanchez 1996). Other effort models have also been proposed (Seijo et al. 1998) and we suggest that further investigation of appropriate simple effort models for different circumstances could substantially improve the usefulness of the basic modeling approach we have proposed here.

The second conclusion is that the performance of the model was influenced by the information content of data, which appears to vary according to the phase of a fishery development. Data from the maturity or senescent phase of a fishery (such as the Namibian hake) provided more information about the status of the stock. The well-known principle that "you cannot determine the potential yield from a fish stock without overexploiting it" (Hilborn and Walters 1992) also applies here. However, more tests need to be done on the model presented here before any conclusions can be drawn with respect to the information content of catch data. In this respect, tests will be undertaken using simulated fisheries in an operating model with known dynamics and error structure. Lastly, informative priors about biological parameters r and K and the effort parameter x improved the performance of the model in estimating biological and policy parameters.

The definition of prior distributions is a very contentious issue in Bayesian analysis (Punt and Hilborn 1997, 2001). Prior distributions for parameter r, the intrinsic rate of increase in biomass, can be constructed combining demographic methods (McAllister et al. 2001) with meta-analysis data on maximum reproductive rate of fish stocks (Myers et al. 1999). Developing prior distributions for K, the virgin stock size, is more problematic because K is not only related to intrinsic biological characteristics of a species but is also a function of ecological characteristics such as size and productivity of the habitat. It is often recommended to assume a uniform prior for K over an appropriate interval based on the cumulative catch (Punt and Hilborn 2001). Prior distributions for the effort models and parameters could also benefit from expert opinion and empirical generalizations of fisheries bioeconomic equilibria based on a synthesis of available examples of fishing effort dynamics.

Notwithstanding the need for further work to test the potential value of the model thoroughly, we believe that these preliminary results are positive and that the Bayesian approach described here may provide a means of gaining important information for management in data-limited fisheries. We plan to explore some of these issues, including the suggested sources of prior information, in the near future.

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Fisheries Assessment and Management in Data-Limited Situations
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Risk-Averse Management of Eulachon in the Columbia River System

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Abstract

This paper reports on the development of a management strategy for eulachon (*Thaleichthys pacificus)* in the lower Columbia River where data are sparse. Eulachon are an anadromous species, of the family Osmeridae, which ascend the lower Columbia River and tributaries to spawn. Starting in 1994, the abundance of the spawning run declined sharply as evidenced in the performance of the commercial fishery. The decline prompted the need to develop a management plan for these fisheries in the absence of management or biological information. Oregon and Washington managers worked with the fishing industry to develop a management plan. The plan incorporated goals of maintaining healthy populations of eulachon, considering the role of eulachon in the Columbia River ecosystem, and developing a risk-averse management strategy. The plan adopted three levels of fishing effort. Fisheries are monitored in-season and fishing level changes are made depending on the results of the monitoring. The ability to adjust fishing levels in-season is important to the fishing industry, general public, and managers.

Introduction

Eulachon, or Columbia River smelt, (*Thaleichthys pacificus*) are a small, schooling, anadromous fish species found in the northeast Pacific Ocean. The largest run of eulachon south of Canada spawns in the Columbia

River and tributaries. The catches of eulachon in the Columbia River system declined through most of the 1990s (WDFW/ODFW 2001), and this prompted a series of fishery restrictions and the development of management policies and practices for both commercial and recreational fisheries.

Biology of eulachon

Eulachon are the largest member of the smelt family, Osmeridae, and the only member of its genus. Eulachon are small fish averaging about 200 mm in length (Scott and Crossman 1973) and reaching a maximum length of about 300 mm (McAllister 1963). Eulachon are an anadromous species. Demersal eggs are deposited in the winter or spring and the larvae drift rapidly to sea where the fish live until returning to freshwater to spawn. Most, if not all, of the fish die after spawning (WDFW/ODFW 2002).

Little is known of eulachon while they are in the ocean. There are indications that significant numbers of eulachon in the ocean off British Columbia originated in the Columbia River (Hay et al. 1997).

Eulachon spawn in the lower Columbia River below Bonneville Dam and in major tributaries of the main river (Fig. 1). Adult eulachon enter the Columbia River system during the winter months. Timing of the run can vary considerably between years. Peak adult abundance is usually in February, but may be as late as April. Eulachon movements are sensitive to water temperature (Smith and Saalfeld 1955), which may affect the timing of the upstream movement.

Fisheries for eulachon

Eulachon are caught in commercial and sport fisheries in the lower Columbia River and tributaries downstream from the Bonneville Dam. The catch is used for human consumption, food for zoo animals, and bait in recreational fisheries.

Commercial fishery

The commercial fishery in the mainstem Columbia River is largely done with gillnet, although a small otter trawl fishery existed in the recent past. Dip nets are authorized but rarely used. Since 1938, commercial catches of eulachon have averaged about 181 t annually in the mainstem Columbia River.

The Cowlitz River produced the bulk (approximately 75%) of the commercial catch in the tributaries of the Columbia River system. Between 1938 and 2001, the commercial catch of eulachon averaged 682 t annually in the Cowlitz River. Dip nets are the only commercial fishing gear authorized for use in the tributaries.

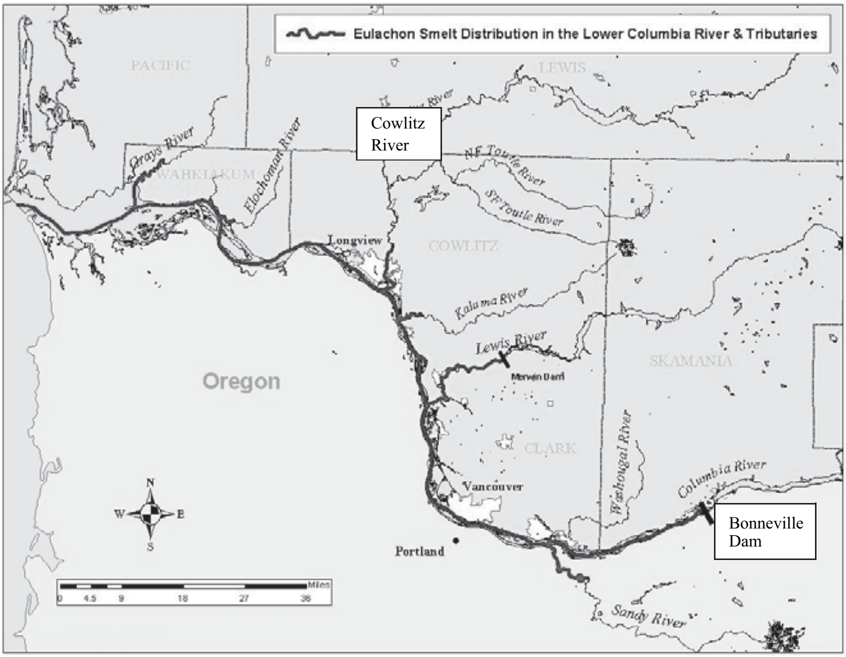

Figure 1. Eulachon spawn in the lower Columbia River below Bonneville Dam and in major tributaries of the main river. Source: Washington Department of Fish and Wildlife.

Recreational fishery

Recreational fisheries for eulachon are conducted primarily in the tributaries using dip nets. The recreational fishery is not regularly monitored, and no estimates of participation or catch are available. However, it is evident that recreational fishing for eulachon is an extremely popular activity with traffic jams common during the peak of the fishery. No quantitative estimates of the harvest by recreational fishing are available.

History of exploitation

Eulachon were harvested by Native groups for centuries. The Lewis and Clark expedition reports trading with local Natives for eulachon on February 25, 1806, and reported the fish were taken in great quantities (Moulton 1990).

Commercial harvest of eulachon can be traced back to the late 1800s. The first reported commercial landings of eulachon in the Cowlitz River were in 1872. By 1892, the commercial catch in the Cowlitz River was approximately 220 t (Hinrichsen 1998).

The commercial fishery is monitored by use of Fish Receiving Tickets. A Fish Receiving Ticket is required by regulation to be completed at the conclusion of each successful fishing trip. The ticket records fishing gear utilized, the general area fished, and the catch in pounds by species.

Consistent records of the commercial landings of eulachon are available back to 1936 (Fig. 2). It is unclear if the magnitude of the total annual landings reflects the size of the eulachon run. Reports from the fishing industry indicated that during years of strong abundance, the market for eulachon would become saturated and fishing effort would diminish (Smith and Saalfeld 1955). Catch per unit effort (CPUE) may not be useful to judge stock abundance, as CPUE may not have a direct relationship to run size (Beverton 1990, Hilborn and Walters 1992).

Between 1938 and 1989, commercial landings averaged 950 t annually and fluctuated without apparent trend. Beginning in 1993 landings dropped due to low abundance of spawning eulachon. In 1994 only 19.5 t were landed, the lowest level since modern record-keeping began in the 1930s. Market demand for eulachon was high, with prices over $3.00 per kg ($7.00 per pound) (WDFW/ODFW 2002).

Beginning in 1995, a series of fisheries restrictions were enacted and the first active management of this resource started. Restrictions consisted largely of reducing the length of the fishing season (Table 1).

Development of a management plan

Because of the fishery decline and the need for active management, Oregon and Washington managers developed a joint Eulachon Management Plan designed to guide eulachon management decisions (WDFW/ODFW 2001). This plan established four major policy directions for the management of the eulachon fishery:

- Maintain healthy populations of eulachon.
- Management actions will consider the role of eulachon in both marine and freshwater ecosystems and the need to maintain sufficient populations of eulachon for proper ecosystem functioning.
- A precautionary approach to resource management shall be utilized.
- Management will consider the best scientific information available and strive to improve the information base for eulachon.

In addition, the policy directed that fishing opportunity be maintained for both the commercial and recreational segments of the fishery.

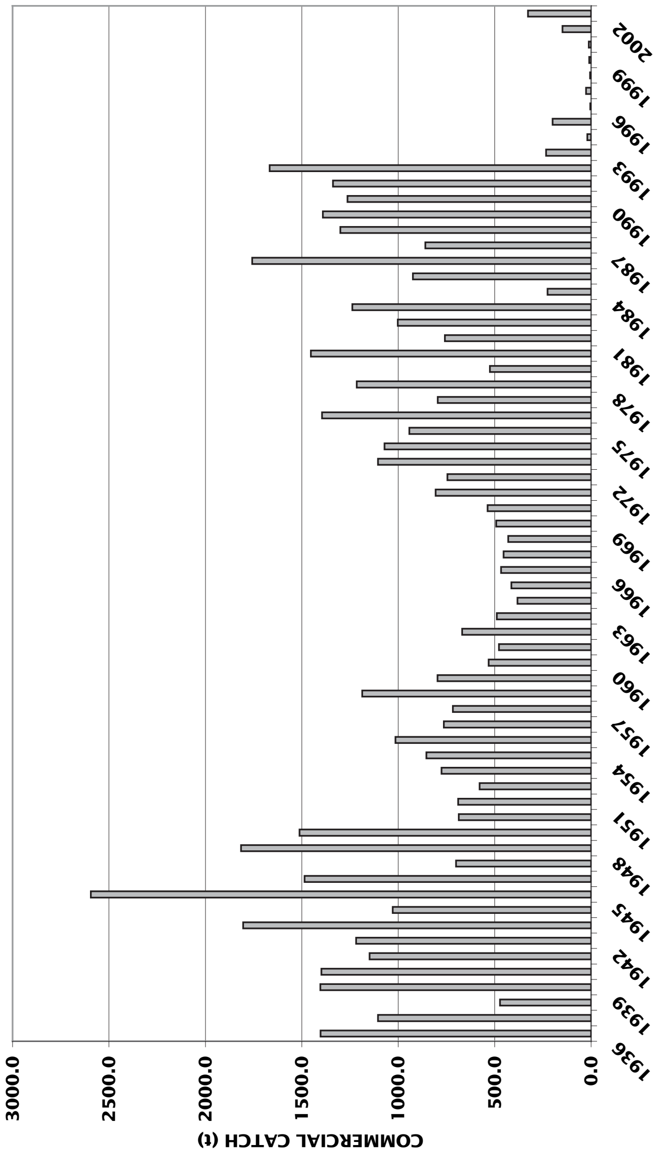

Figure 2. Commercial fishery landings of eulachon from the Columbia River system, 1936-2002.

Table 1. Days open during the Columbia River commercial eulachon fishing seasons, 1960-2002.

Seasonal year[a]	Days open to fishing
1960-1985	255 to 365
1986-1994	121
1995	86
1996	96
1997	63
1998	44
1999	30
2000	35
2001	47
2002	66

[a] A seasonal year is from December 1 to November 30.

To implement the formal policy directives, it became necessary to translate the directives into a management plan. To do so, the management agencies agreed on the following strategies:

- *Commit to work with the fishing industry.* By utilizing the industry's knowledge and interest in the resource, information could be gained at a low cost and an atmosphere of cooperation would be fostered.
- *Fishing levels would be based on abundance.* To meet policy directives there needed to be some knowledge, even qualitative, of the abundance of eulachon. Additionally, during years of low run sizes, fishing would be constrained to help assure healthy populations of eulachon and proper ecosystem function.
- *Acknowledge uncertainty.* For the foreseeable future, considerable gaps in needed knowledge are likely to continue. Management should openly identify these uncertainties and adopt fishery regulations that account for uncertainty.
- *Build flexibility into fishery management*. Managers should have the ability to change regulations in-season based on acquired information.

Table 2. Description of the three fishing levels for eulachon in the Columbia River.

Level 1—Level 1 fisheries are utilized when there is great uncertainty in run strength or indication of a poor return. Level 1 fisheries are the most conservative and scheduled to produce an annual harvest rate of 10% or less. The purpose of Level 1 fisheries is to gain insight on the spawning runs while minimizing the risk of overexploitation. Typical Level 1 fisheries might consist of one 12-24 hour fishing period in the mainstem Columbia River and one additional day in the Cowlitz River per week. Recreational fisheries would be limited to one 2-24 hour period per week in the Cowlitz River. Days and hours to be fished would be developed in conjunction with fishery participants.

Level 2—When fishery data indicate a promising abundance in the spawning run and other indices are favorable, fishing time would be increased to collect additional data. The trigger to move from a Level 1 to a Level 2 fishery is not specified, but should be carefully deliberated. Typical fishing opportunities for both recreational and commercial fisheries would be two or three days of fishing per week. The harvest rates expected under a Level 2 fishery are not quantified.

Level 3—Level 3 fisheries are the most liberal fishing seasons. Level 3 fisheries are adopted when there are indicators of strong stock abundance and productivity. Typical Level 3 commercial fisheries would be open four days per week and recreational fisheries four to seven days per week. In Level 3 fisheries the daily bag limit for recreational may be increased as well. The harvest rates expected under a Level 3 fishery are not quantified.

Following a series of formal and informal consultations with fishing groups and other interested parties, a fishing strategy was developed. Key aspects of the strategy were that fishing regulations would be conservative to start the season and fishing opportunities would be increased or decreased in-season depending on the results of the initial fishing.

To formalize the approach, three levels of fishing opportunity were developed (Table 2). Prior to the fishery season, managers would make a determination of the appropriate level of fishing to begin the season. During the fishing season, fishery results would be analyzed and ancillary information collected, such as density of eulachon larvae in the river. If warranted, fishing levels would be adjusted in-season.

Preseason indicators

Prior to each fishing season, a determination of anticipated run strength is made using a series of indicators including:

1. *Parental stock abundance as indicated by fishery data.* Although the age composition of spawning eulachon has not been verified,

managers assume that most spawning fish are three or four years of age. By examining the catches of fish two and three years prior to the upcoming season, a general idea of the abundance of the parental stock can be obtained.

2. *Juvenile production as indexed by larval abundance.* During the spawning season, the Columbia River and major tributaries are regularly sampled to measure the density of eulachon larvae. By comparing densities from year to year, an idea of the relative size of the spawning stock can be obtained (WDFW/ODFW 2002).

Risk management

The term "risk" is used in this paper in a nontechnical, unquantified sense to identify the probability of an undesirable outcome. In the case of the Columbia River eulachon fishery, the undesirable outcome is stock collapse (i.e., a severe decline in abundance and failure to recover [Francis and Shotton 1997]). Much of the risk associated with this fishery is related to uncertainty; the abundance of the stock of eulachon is unknown and can only be evaluated through the magnitude of landings by the commercial fleet. Only a portion of the fishery (the commercial fishery) is monitored and catches measured. A substantial segment of the harvest (the recreational fishery) is lightly monitored and catches are not estimated. Commonly used indicators of fishery performance (such as CPUE) may be uninformative or even misleading, due to factors such as patchy distribution of fish and market saturation.

Management of the risk of fishing on the eulachon resource is based on reduction of the amount of time fishing, either commercial or recreational, that is allowed in the Columbia River system. No harvest quotas have been established for two reasons (1) lack of a scientific basis to specify a quota amount and (2) lack of ability to estimate the harvest in the recreational fishery. It is unclear if the closed periods provide sufficient protection to the eulachon.

One facet of the fishery management plan allows some amount of fishing under all stock conditions. Our experience is that at lower run sizes, fishing effort is also low. This plan allows knowledge of run size to be gained at little expense or risk to the resource.

Conclusion

The implementation of this strategy was successful over the short term. The fishing industry has access to the resource, even at low abundance levels. This access provides continuity of markets and opportunity for fishers to gauge eulachon abundance. It provides managers with needed information at little financial cost. It appears to provide protection to eulachon during periods of low abundance. There is a risk of foregoing harvest during periods of high eulachon abundance, when fisheries are initially set at a low level. It remains unclear if the strategy is sufficiently risk-averse during periods of low abundance.

Keys to the apparent success of this approach have been (1) starting seasons at a conservative level that provides protection to the resource, (2) the ability to change fishing regulations in-season, and (3) the continuing communication between the fishing industry and managers. It is important that the managers have followed the plan and increased fishing opportunity during the season when indicators of abundance became favorable. This action has greatly improved industry trust in the plan.

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Managing Depleted Snapper Stocks in Inner Shark Bay, Western Australia

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Abstract

Snapper (*Pagrus auratus*) have been a major attraction for recreational boat fishers visiting the inner gulfs of Shark Bay, Western Australia, since the 1970s. In the mid-1990s, despite some long-held concerns over the high exploitation of spawning aggregations, data limitations hampered management efforts to take action to protect snapper stocks. In 1997, a fishery closure in the eastern gulf was overturned through public objection, lack of data being the underlying reason. From 1998 to 2002, fishery independent estimates of mature biomass, recreational catches, and age composition data were collected. Despite the biomass estimates being highly variable, and the age data being scant, models yielded results consistent with the available data for the three stocks of snapper in the inner gulfs of Shark Bay. Likely trajectories of mature biomass were explored for a range of future catch levels and these were compared to a biological reference point, 40% of the virgin mature biomass. In 2003, based on these assessments, a TAC (total allowable catch) was set for each snapper stock, a significant innovation in the management of marine recreational fisheries in Australia.

Introduction

Exploitation rates in marine recreational fisheries are an increasing concern for Australian fisheries managers. Although such fisheries were thought unlikely to suffer collapses akin to many of the world's commercial fisheries (Post et al. 2002), recent studies have shown that overfishing by recreational fishers can deplete spawning populations and collapse fisheries (Cockcroft et al. 1999, Cox et al. 2002). Management

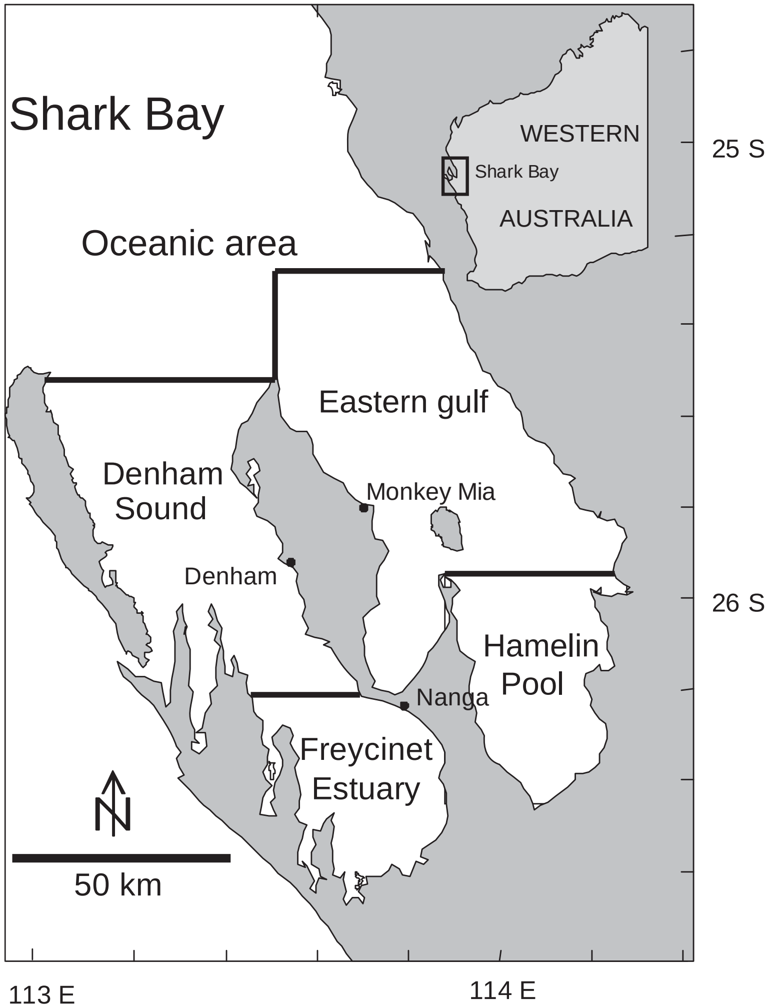

Figure 1. The four management areas of Shark Bay, Oceanic Area, Denham Sound, Freycinet Estuary, and eastern gulf. Hamelin Pool is permanently closed to all fishing.

of recreational fisheries is particularly challenging because most have unrestricted access (McPhee et al. 2002, Moore 2003), effort is generally increasing (McPhee et al. 2002, Sumner 2003), and data such as catch, effort, and biomass estimates are seldom available (Post et al. 2002). Consequently, managers must make decisions on anecdotal evidence or preliminary research data although such information is often insufficient to secure public and political support for the measures needed for stock rebuilding.

Shark Bay on the central coast of Western Australia (Fig. 1), particularly the sheltered inner gulfs, is a major recreational fishing destination with recreational fishing effort estimated at 35,000 to 49,000 fisher days per year between 1998 and 2001 (Sumner and Malseed 2002). Although recreational boat fishers catch a variety of species, the main attraction historically has been snapper (*Pagrus auratus,* Sparidae), which make up more than half the total recreational boat catch in the inner gulfs (Sumner and Malseed 2001). Snapper have been abundant in the past and are relatively easy to catch during winter (May to August) when they form spawning aggregations.

Stock structure of snapper in the Shark Bay region is unusually complex with little or no apparent mixing between local populations either during the pelagic egg and larval phases or later in life (Johnson et al. 1986; Edmonds et al. 1989, 1999; Moran et al. 1998, 2003; Bastow et al. 2002; Nahas et al. 2003). On this basis, snapper in the Shark Bay region are managed as four separate stocks: “oceanic” in waters outside of Shark Bay, “eastern” in the eastern gulf, “Denham Sound” in northern waters of the western gulf, and “Freycinet Estuary” in southern waters of the western gulf (Penn 2002, Fig. 1). The oceanic stock supports a major commercial fishery (managed independently under a limited-entry and individual transferable quota system) while the inner gulf stocks (eastern, Denham Sound, and Freycinet Estuary) are the basis of the recreational fishery, and are the focus of this paper.

Although some commercial snapper fishing occurred inside Shark Bay from the 1930s, commercial effort in the inner gulfs has steadily declined since the 1960s and 1970s, and has shifted to oceanic waters outside Shark Bay. In contrast, recreational fishing effort is thought to have increased from the 1970s through to the 1990s (Jackson et al. 2003). Over this period, improved vehicle access and increased use of GPS and color sounders resulted in the increasing efficiency of recreational fishers targeting local snapper.

By the mid-1990s, some in the local community had become concerned that recreational snapper catches, especially in the eastern gulf, had reached unsustainable levels. Proposals to introduce more stringent management measures, such as reduced daily bag limits during the spawning season, were unpopular, and not supported, as visiting recreational fishers make a significant contribution to the region's tourism-based economy.

High recreational catches in 1994-1996 led to heightened concerns by managers that the eastern gulf spawning stock was seriously depleted. In May 1997, the eastern gulf was completely closed to snapper harvest; however, the closure was overturned after only two months in response to public objection (Marshall and Moore 2000). The basis of the objection was the paucity of quantitative information, and it became apparent that estimates of stock size and recreational catch were needed to gain community support for any stock rebuilding strategy.

Since 1997, mature biomass (biomass of sexually mature male and female snapper) has been estimated using the daily egg production method (DEPM) (Jackson and Cheng 2001). This method involves simultaneously estimating the number of recently spawned eggs, the average weight of spawning females, the proportion of females that spawn, the number of eggs released by a female of average weight, and the ratio of females to males. These estimates are combined to calculate the mature biomass. In addition, recreational catches of snapper have been estimated between 1998 and 2002 (Sumner and Malseed 2001, 2002; Sumner et al. 2002).

Various measures have been employed to maintain or rebuild spawning stocks, including a complete prohibition on the take of snapper in the eastern gulf (June 1998 to March 2003), increases in the minimum legal size, protection of larger fish, reductions in the daily bag limit in Denham Sound and the Freycinet Estuary, and a six-week spawning season closure (August to September) in the Freycinet Estuary.

This paper describes age-structured models developed to improve the quality of research advice to managers responsible for this important fishery. The models explored likely trajectories of mature biomass for a range of past and future catches. Based on these assessments, TACs were set for the three inner gulf snapper stocks in 2003, a significant innovation in the management of marine recreational fisheries.

Methods

A separate age-structured model was developed for each of the three inner gulf snapper stocks, the details being shown in the appendix. The virgin recruitment and the recruitment at the beginning of the data set (1983) were estimated assuming a constant harvest rate H, set at 0.05, up to 1983. Subsequently numbers in each age class and each year were reduced by natural mortality and annual catches. Recruitment each year (number of 0-year-old fish) was calculated using a Beverton and Holt stock-recruitment relationship (SRR) with an annual recruitment deviation parameter ε_y. The mature biomass at the virgin level and also from 1983 was calculated from the age composition, sexual maturity at age, and the weight at age.

Table 1. Estimates of mature biomass (D_y, t) and standard deviation (σ_y) for inner gulf snapper stocks from daily egg production method surveys, 1997-2002.

	Denham Sound		Freycinet Estuary		Eastern gulf	
Year	D_y	σ_y	D_y	σ_y	D_y	σ_y
1997	na	na	842	619	12	4
1998	365	89	324	216	22	6
1999	62	24	50	13	97	22
2000	366	284	51	19	543	284
2001	142	27	19	6	95	27
2002	183	55	94	32	118	55

Estimates of mature biomass for each stock (Table 1) were obtained using the DEPM (Jackson and Cheng 2001) from surveys conducted between 1997 and 2002. There was considerable uncertainty in biomass estimates initially (1997, 1998) due to limited knowledge of the location of key spawning aggregations, precise timing of spawning, and some possible misidentification of snapper eggs.

Recreational snapper catches from the 1960s through the 1980s are unknown apart from an estimate in 1983 for Denham Sound, Freycinet Estuary, and the eastern gulf obtained from an aerial and boat ramp survey (Mike Moran, Department of Fisheries of Western Australia, Perth, Sept. 2003, pers. comm.). A catch estimate was made in 1995 in the eastern gulf following observations by fisheries officers. Subsequently recreational catch estimates were available from annual recreational fishing surveys in the period 1998-2002 (Sumner and Malseed 2001, 2002; Sumner et al. 2002) with total weight estimated from length frequency data and weight at length.

Anecdotal information obtained by interviews with commercial and recreational fishers indicated that recreational catches had increased gradually from the late 1970s to the early 1990s and then increased rapidly to the mid-1990s (Gary Jackson, Department of Fisheries of Western Australia, Perth, Oct. 2003, pers. comm.). In this assessment, recreational catches were ramped up from 1984 to 1997 to provide annual estimates of the catch from 1983 to 2002 (Table 2). Two other catch scenarios, low and high, were obtained by reducing and increasing this catch by 33% between 1984 and 1997.

The mortality of discarded fish below legal size was ignored in this assessment as caging experiments found that release mortality of snapper was less than 5% in water less than 30 m depth (Jill St John, Depart-

Table 2. **Recreational (R) and commercial (C) catches (in t) of snapper taken in Denham Sound (A), Freycinet Estuary (B) and the eastern gulf (C), between 1983 and 2002. Catches in bold are estimated from recreational fishing surveys and commercial catches reported in statutory production returns. Catches in other years were allocated according to anecdotal information from fishers.**

Area	Catch	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
A	R	**12**	13	14	15	16	17	18	18	19	19	20	20	20	18	15	**12**	11	**10**	**8**	**15**
	C	12	12	12	12	12	12	12	12	12	**12**	**9**	**5**	**13**	**16**	**12**	**5**	**1**	**2**	**3**	**2**
B	R	**17**	18	19	20	21	22	23	24	25	26	27	28	29	30	30	**30**	25	**20**	**23**	**19**
	C	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	0.5	0.3	0.3	0.3	0.3
C	R	12	13	14	15	16	17	18	18	19	19	20	20	20	18	15	12	11	10	8	15
	C	12	12	12	12	12	12	12	12	12	12	9	5	13	16	12	5	1	2	3	2

Table 3. **The selectivity of pink snapper v_a, at age *a*, for each of the inner gulf stocks, Denham Sound (A), Freycinet Estuary (B), and eastern gulf (C) for (1) 1983-2000, and (2) after 2000.**

Area	Period	*a*	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	15
A	1	v_a	0.00	0.00	0.01	0.05	0.15	0.36	0.64	0.85	0.95	0.98	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	2	v_a	0.00	0.00	0.00	0.01	0.05	0.15	0.36	0.64	0.85	0.95	1.00	1.00	0.5	0.02	0.01	0.00	0.00
B	1	v_a	0.00	0.00	0.02	0.13	0.93	0.99	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	2	v_a	0.00	0.00	0.01	0.42	0.67	0.92	1.00	1.00	1.00	0.85	0.72	0.60	0.55	0.41	0.26	0.10	0.00
C	1	v_a	0.00	0.00	0.00	0.07	0.39	0.76	0.83	0.99	0.94	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	2	v_a	0.00	0.00	0.00	0.00	0.11	0.36	0.69	0.83	0.89	0.85	0.93	0.75	0.50	0.00	0.00	0.00	0.00

ment of Fisheries of Western Australia, Perth, Aug. 2004, pers. comm.) as found in the inner gulf waters.

Age composition data were obtained from sectioned otoliths of legal sized fish collected from research surveys and recreational catches. Fish older than 12 years from Denham Sound and 14 years from eastern gulf were allocated to 13+ and 15+ groups respectively. Fish from the Freycinet Estuary older than 19 years were allocated to a 20+ group. The selectivity at age v_a, was determined from the proportion of fish in each age class with lengths greater than the minimum legal size of 41 cm from 1983 to 2000 and the proportion of fish in each age between the legal sizes of 50 cm to 70 cm after 2000 (Table 3).

For each stock, the parameters were estimated using the software AD Model Builder (Otter Research Ltd. 2000). The parameters estimated were initial recruitment (R), recruitment in 1983 (R_{1983}), natural mortality (M), and 20 parameters for annual recruitment deviations ε_y The virgin recruitment R, and R_{1993}, were assumed to have a uniform prior. The instantaneous rate of natural mortality M, was assumed to be constant for all ages and to have a lognormal prior. The parameters of the SRR were determined for the virgin stock with the steepness fixed at 0.75. The recruitment deviation ε_y was assumed to have a lognormal prior with CV = 0.5.

A Bayesian procedure was used to determine estimates of uncertainty of the mature biomass D. A joint posterior probability distribution was generated using the Markov Chain Monte Carlo (MCMC) procedure. A subset of 2,000 values from the joint posterior distribution was obtained by saving every 1,000th from 2,500,000 cycles of the MCMC and rejecting the first 500 generated during the "burn in" period.

The mature biomass after 2002 was determined using the subsample from the joint posterior distribution for each of the three catch scenarios and various levels of TAC in the future. The recruitment determined from the SRR was multiplied by the exponent of the annual recruitment deviations randomly sampled from a lognormal distribution with mean zero and variance taken as that from the historical annual recruitment deviations. From the marginal posterior distribution of mature biomass, the mode and 95% confidence intervals were determined. TACs in the future were evaluated by determining the proportion of values in the distribution of mature biomass that were above the biological reference point, i.e., 40% of the virgin mature biomass for each year, stock, and each of the three catch scenarios.

Results

Virgin recruitment was highest in Denham Sound and lowest in Freycinet Estuary (Table 4). For each stock, the low-catch scenario resulted in a low-

Table 4. Model estimates of virgin recruitment *R* (thousands of fish), with standard deviations σ_R, for the three catch scenarios (low, medium, and high) for inner gulf snapper stocks.

	Denham Sound			Freycinet Estuary			Eastern gulf		
	Low	Medium	High	Low	Medium	High	Low	Medium	High
R	33.83	34.71	36.39	9.92	11.17	12.71	19.78	42.55	26.58
σ_R	18.22	16.72	16.36	1.34	1.48	1.76	8.00	11.05	10.74

est virgin recruitment and the high-catch scenario resulted in a highest estimate of virgin recruitment.

The estimated natural mortality (M ± 1 sd) was lowest in Freycinet Estuary (0.08 ± 0.01) and higher in Denham Sound (0.18 ± 0.08) and the eastern gulf (0.19 ± 0.04). There was little variation between the three catch scenarios for each stock.

Modeled age composition (Fig. 2) generally fitted the observed data for the three stocks. Relative recruitment, with a mean of one, exp(ε_y) indicated a strong recruitment in Denham Sound in 1989-1990 corresponding the 7- and 8-year-old age classes in 1998 (Table 5). In Freycinet Estuary there appeared to be a strong recruitment in 1985, corresponding to strong 12-year-old and 13-year-old age classes in 1997 and 1998 and strong recruitment in 1989-1990 corresponding to the 8-year-old age class in 1998 and 1999 and the strong 10-year-old age class in 2001. In the eastern gulf the model indicated strong recruitment in 1989 and 1997 corresponding to the 11-year-old age class in 2001 and 5-year-old age class in 2002. The values of relative recruitment, exp(ε_y) are 1.0 after 1999 as there is no information in the age composition data until 2-year-old fish enter the fishery.

The agreement between estimated mature biomass and DEPM estimates varied greatly between years (Fig. 3). In Denham Sound, there was good agreement in 1999, 2001, and 2002 and the model indicated that the mature biomass in 1993 had been reduced to 34%, 33%, and 32% of the virgin level for the low, medium, and high catch scenarios, respectively. For the low, medium, and high catch scenarios, the model indicated that the spawning stock recovered to 62%, 55%, and 53% of the virgin level in 2002.

In Freycinet Estuary, there was agreement between DEPM and model estimates of mature biomass in 2002, but DEPM values were much higher in 1997 and 1998 and considerably lower in 1999-2001. In the Freycinet Estuary, mature biomass was estimated to have declined to 30%, 28%, and 27% of the virgin level in 2002 for the low, medium, and high-catch scenarios, respectively.

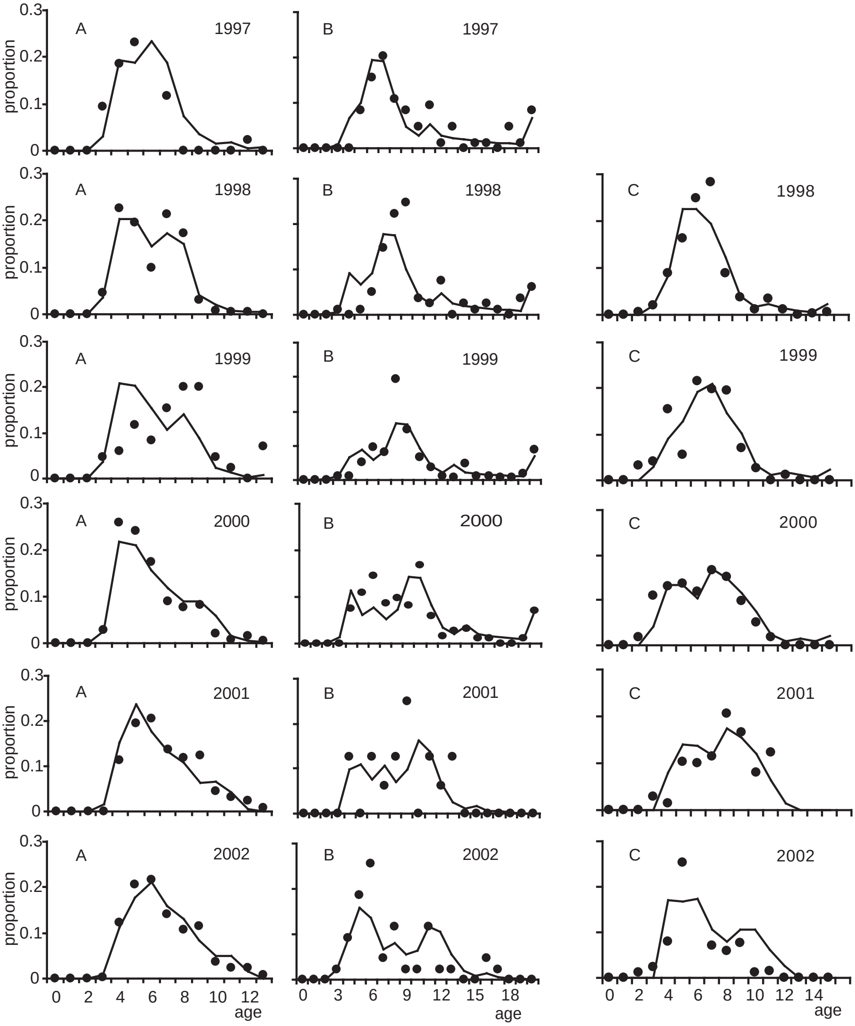

Figure 2. Observed (filled circles) and model predicted (lines) proportional age composition for Denham Sound (A), Freycinet Estuary (B), and eastern gulf (C) derived from samples collected between 1997 and 2002.

Table 5. Annual relative recruitment exp (ε_y), and standard deviation ($\sigma_{\exp(\varepsilon_y)}$) for the three inner gulf pink snapper stocks Denham Sound (A), Freycinet Estuary (B), and eastern gulf (C).

	y	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999
A	$\exp(\varepsilon_y)$	0.71	0.80	1.05	0.55	0.78	0.89	1.83	1.70	1.04	1.12	1.19	1.32	1.59	1.24	0.92	0.45	0.96
	$\sigma\exp(\varepsilon_y)$	0.30	0.32	0.41	0.19	0.22	0.21	0.30	0.24	0.13	0.13	0.15	0.20	0.28	0.25	0.22	0.17	0.46
B	$\exp(\varepsilon_y)$	1.08	1.10	1.69	0.84	1.08	1.96	2.77	2.25	0.95	0.57	0.63	0.41	0.59	0.70	0.60	1.21	1.01
	$\sigma\exp(\varepsilon_y)$	0.28	0.27	0.33	0.19	0.21	0.28	0.32	0.26	0.14	0.09	0.09	0.07	0.10	0.16	0.17	0.51	0.48
C	$\exp(\varepsilon_y)$	1.03	0.91	1.13	1.28	0.60	0.94	1.75	1.44	1.10	0.72	0.38	0.44	0.76	1.28	3.69	1.09	1.04
	$\sigma\exp(\varepsilon_y)$	0.48	0.42	0.50	0.44	0.22	0.26	0.34	0.24	0.17	0.11	0.07	0.08	0.16	0.33	1.38	0.53	0.50

In the eastern gulf the mature biomass estimates agreed with DEPM estimates in 1999, 2001, and 2002, especially for the low-catch scenario. The model estimate did not agree with DEPM values for 1997, 1998, and 2000 (Fig. 3). The estimated mature biomass appears to have declined rapidly from 1994, reaching a minimum in 1998 of 24%, 27%, and 28% of the virgin level for the low, medium, and high-catch scenarios. The stock has since recovered, after the snapper fishery closure in June 1998, to 47%, 53%, and 54% of the virgin level in 2002, for the three catch scenarios.

The mature biomass in each of the 2000 draws from the posterior distribution was used to calculate the probability that the mature biomass was above the biological reference point for each stock with different future TACs (Fig. 4). Level of future TACs was discussed with local stakeholders, including a closure to snapper fishing in the Freycinet Estuary. It was decided to set low TACs for each of the inner gulf stocks, 10 t in Denham Sound, 15 t in Freycinet Estuary, and 5 t in the eastern gulf. With the allocated TACs, the probability of achieving the target would be 0.90-0.94 in Denham Sound and 0.02-0.03 in Freycinet Estuary depending on the past catch scenarios. In the eastern gulf, the probability of achieving the target was 1.0 of all three past catch scenarios. With the low past catch scenario, the initial stock size was estimated to be smaller, the current stock status more optimistic, and the expected rebuilding slower. With higher historic catches, the initial stock size was estimated to be higher,

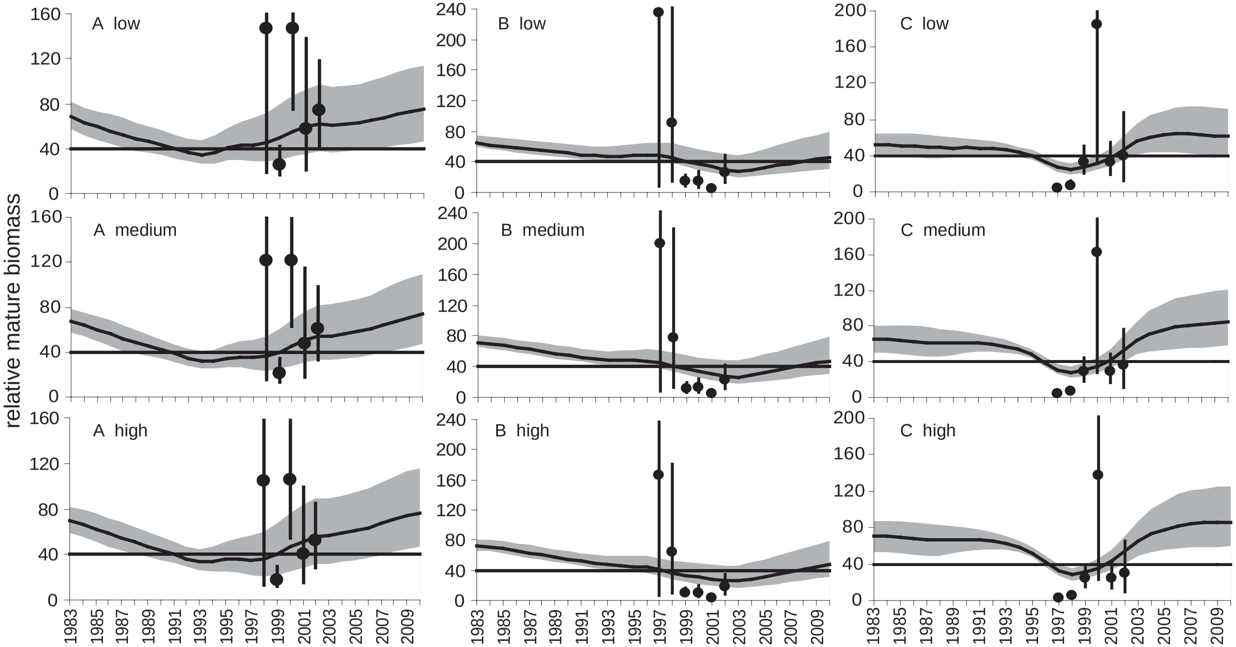

Figure 3. **Mature biomass as a proportion of 1983 level (black lines) with 95% confidence intervals (shaded area) for three catch scenarios (low, medium, high) in Denham Sound (A), Freycinet Estuary (B), and the eastern gulf (C). Horizontal lines represent the biological reference point, 40% of the virgin level. DEPM (daily egg production method) biomass estimates, as a percentage of the 1983 mature biomasses are shown as circles, with error bars representing 95% confidence intervals. The catch after 2002 was set at 10 t, 5 t, and 15 t for regions Denham Sound, Freycinet Estuary, and the eastern gulf respectively.**

the current status more pessimistic, and the rebuilding correspondingly more rapid.

Discussion

Managers of Australian fish stocks that are highly vulnerable to overexploitation by recreational fisheries are usually faced by acute data-limitations. Although strong anecdotal evidence of stock depletion may exist, attempts to take remedial management are often strongly challenged by stakeholders if quantitative information on stock size and catch levels are uncertain. However, fisheries scientists are typically faced with the reality of synthesizing the limited available data, however poor, and providing timely advice to fishery managers and stakeholders. The assessment of snapper stock status in the inner gulfs of Shark Bay presented here is the first formal assessment of a primarily recreational finfish fishery in Australia.

Concern in the mid-1990s that inner gulf snapper stocks may be overexploited led to a period of intensive data collection between 1997 and 2003. Although data for these years were available, catches are poorly known for most of the fishery history, except for a single boat ramp catch survey conducted in 1983. Despite this considerable limitation, an evaluation of the stock status was possible by assuming a plausible exploitation rate prior to the first catch estimate and then making “guess-estimates” of the annual catch, with considerable variation, for the period before survey data become available in 1998. Provided that the current catches and recent biomass estimates are realistic, the level of past catches were shown not to be critical in the medium term (5 years). If estimated historic catches were lower, the model indicated an optimistic current stock level, and a slow increase in mature biomass when the TAC was reduced. Conversely, if past catches were assumed high, the modeled current mature biomass was lower, and increased more rapidly when catches were reduced. This result indicates that lack of reliable past catch information need not be a reason for delaying analysis of stock status.

During the period when the take of snapper was totally prohibited in the eastern gulf (June 1998 to March 2003), recreational fishing for other species continued. This resulted in an estimated 11,000, 10,000, 7,000, and 11,000 snapper being caught and released in 1998, 2000, 2001, and 2002, respectively (Sumner and Malseed 2001, 2002; Sumner et al. 2002). Reduction in daily bag limit and an increase in the minimum legal size in both Denham Sound and Freycinet have increased the discarding of undersized fish. This could lead to the assessment being overly optimistic. The recent quantification of snapper handling mortality (Jill St John, Department of Fisheries of Western Australia, Perth, Aug. 2004, pers. comm.) will allow discard mortality to be incorporated in future assessments.

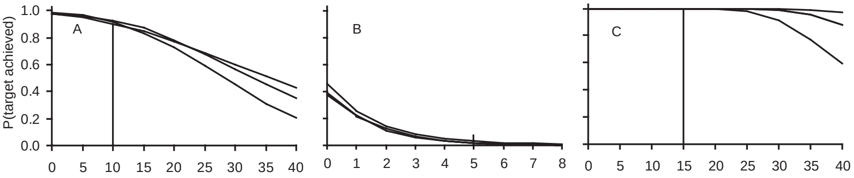

Figure 4. The probability of achieving the target, i.e., the mature biomass being above 40% of virgin level, in 2007, for various TACs (total allowable catch) in Denham Sound (A), Freycinet Estuary (B), and eastern gulf (C). The higher probability ogive is for the low-catch scenario, the middle ogive for the medium catch scenario, and the lower ogive for the high-catch scenario. Vertical lines illustrate the TAC set in 2003 for each stock.

Although there are estimates of mature biomass from DEPM surveys from 1997 to 2002, our knowledge of the current biomass is still limited. The DEPM estimates in 1997 and 1998 were obtained with limited knowledge of the extent of the spawning grounds, together with some difficulty in snapper egg identification. In Denham Sound non-snapper eggs may have been inadvertently included in the analysis, with egg counts overestimated by as much as 40% in some cases. Before any future assessment there is a need to repeat the egg counts and reanalyze the results for these years. In addition, the DEPM data in later years are inconsistent with large variation between years. Clearly, the mature biomass in the eastern gulf could not have varied from 97 t in 2000, to 540 t in 2001 and then 95 t in 2002. Although the early biomass estimates are considered to be poor, their high standard deviations reduce their weight in the log-likelihood in this analysis.

The TACs set in 2003 were designed to be sufficiently conservative to remain in place for a 3-year period. Mature biomass estimates will be determined from DEPM surveys in 2003 and 2004 and recreational catch surveys will be conducted in 2003 and 2004. These data will enable evaluation of the biomass trajectories of this assessment and assist in reviewing TACs for the period 2006-2009. In addition, the assessment model can be used to evaluate the extent to which age composition samples and future DEPM surveys are likely to reduce the uncertainty of future mature biomass estimates.

The fisheries of the inner gulf of Shark Bay are small and geographically isolated, and have been fished heavily for many years. They have special importance in Western Australia, as they are in a World Heritage Area and are a popular destination of recreational fishers. When anecdotal information from fishers indicated the stocks were overfished, managers had difficulty taking remedial action due to the paucity of information.

This scenario is typical of many small coastal recreational fisheries, which would generally not have a complete catch history. This study indicated that with little historic catch data and a research program lasting just 5 years, an assessment of the state of the stock is possible. The model was able to provide visual representations of possible stock levels that were useful in focusing attention on the key issues, and assisted managers and stakeholders in the decision-making process.

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Appendix 1.

The number of snapper in the fishery at the start of year *y* are determined by:

$$N^s_{a+1,y+1} = \begin{cases} R_y & \text{if } a = 0 \\ \left(N^s_{a,y}\, e^{-0.5M} - C_{a,y}\right) e^{-0.5M} & \text{if } 0 < a \le A \\ \left(N^s_{A,y}\, e^{-0.5M} - C_{A,y} + N^s_{A-1,y}\, e^{-0.5M} - C_{A-1,y}\right) e^{-0.5M} & \text{if } a = A. \end{cases}$$

The proportion of females mature at age *a* is

$$p_a = \left(1 + e^{-\ln(19)\left(\frac{a-a_{0.5}}{a_{0.95}-a_{0.5}}\right)}\right)^{-1}.$$

The biomass of spawning females (t) S_y, at the end of the year is

$$S_y = \sum_{a=0}^{A} N^f_{a,y}\, p_a W_a.$$

The number of snapper (thousands), at unfished equilibrium, the initial state, is

$$N^s_{a+1,init} = \begin{cases} (1-\rho) R_{init} e^{-aM} & \text{if } s = \text{male} \quad \text{and } 0 \le a < A \\ \rho R_{init} e^{-aM} & \text{if } s = \text{female} \quad \text{and } 0 \le a < A. \end{cases}$$

The number of 0-year-old recruits in each year is

$$R_{y+1} = \frac{S_y}{\alpha + \beta S_y}.$$

The stock-recruitment parameters α and β were calculated from the proportion of R_{init} that recruits when S_y is 20% of the virgin level (Hilborn et al. 1994),
where:

$$\alpha = \frac{S_1}{R_{init}}\left(1 - \frac{h-0.2}{0.8h}\right) \quad \text{and} \quad \beta = \left(\frac{h-0.2}{0.8hR_{init}}\right).$$

The number of snapper (thousands), in 1983, the start of the data set is

$$N^s_{a+1,1983} = \begin{cases} (1-\rho)R_{1983}(1-H)e^{-aM} & \text{if } s = \text{male} \quad \text{and } 0 < a < A \\ \rho R_{1983}(1-H)e^{-aM} & \text{if } s = \text{female} \quad \text{and } 0 < a < A. \end{cases}$$

The number of recruits at the beginning of subsequent years is given by

$$N_{0,y+1} = \begin{cases} R_y & y = 1983 \\ R_y e^{\varepsilon_y} & 1988 < y \le 2002 \\ R_y e^{N(0,\sigma_y{}^2)} & y > 2002. \end{cases}$$

The exploitation rate is

$$F_y = \frac{C_y}{\sum\limits_{s=m,f} \sum\limits_{a=0}^{A} N^s_{a,y} v_a W_a}.$$

The number of fish caught is

$$C_{a,y} = N^s_{a,y} e^{0.5M} F_y v_a W_a .$$

The estimated mature biomass at the end of year is

$$\hat{D}_y = \sum_{s=m,f} \sum_{a=0}^{A} N^s_{a,y} p_a W_a.$$

The logarithm of the likelihood function associated with the mature biomass observations is

$$\lambda_1 = -\frac{\sum\limits_{\substack{\text{y} \\ D_y \text{ exists}}} \left(D_y - \hat{D}_y\right)^2}{2\left(CV_y D_y\right)^2}.$$

The model estimated proportion at age is

$$\hat{p}_{a,y} = \frac{\sum_{s} N^{s}_{a,y}}{\sum_{s} \sum_{a=0}^{A} N^{s}_{a,y}}.$$

The logarithm of the likelihood function associated with age samples (ignoring constant terms) is

$$\lambda_2 = \sum_{\substack{y \\ \text{sample exists}}} K_y \sum_{a=0}^{A} p_{a,y} \log(\hat{p}_{a,y}).$$

The annual recruitment deviation in each area is assumed to be lognormally distributed with CV = 0.5. The contribution to the logarithm of the likelihood function is

$$\lambda_3 = -\frac{1}{2\sigma_r^2} \sum_{y} \varepsilon_y^2 .$$

The natural mortality M was assumed to have a lognormal prior with CV = 0.2 mean 0.12. The contribution to the logarithm of the likelihood function is

$$\lambda_4 = -\frac{(M - \tilde{M})^2}{2\sigma_M{}^2}.$$

The total log-likelihood, $\lambda = \lambda_1 - \lambda_2 - \lambda_3 - \lambda_4$ was minimized using AD Model Builder to estimate the parameters R_y, V_a, and ε_y.

Meaning of symbols

R_{init}	estimated number of 0-year-old fish at unexploited equilibrium.
$N^{s}_{a,init}$	number of fish of sex *s*, and age *a*, in the initial year.
H	exploitation prior to 1983.
$N^{s}_{a,y+1}$	number of fish of sex *s*, age *a*, in year *y*.
F_y	exploitation rate in year *y*.
V_a	proportion of vulnerable snapper of age *a*.
C_y	catch (t) in year *y*.
$C_{a,y}$	catch in numbers at age *a* in year *y*.
D_y	biomass of sexually mature snapper.

S_y	biomass of sexually mature female snapper.
A	maximum age.
α,β	parameters of the stock recruitment relationship.
h	steepness of the stock recruitment relationship.
p_a	proportion of snapper mature at age a.
$a_{0.5}$, $a_{0.95}$	ages when 50% and 95% of fish are mature.
ρ	proportion of females in the population.
R_y	recruitment in the fishery from the stock recruitment relationship.
ε_y	annual recruitment deviation for year y.
$p_{a,y}$	observed proportion at age in year y.
$\hat{p}_{a,y}$	model estimated proportion of fish at age a, in year y.
K_y	number of fish in the age composition sample.

Management of the Data-Limited Weathervane Scallop Fishery in Alaska

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Abstract

The weathervane scallop *(Patinopecten caurinus)* fishery in Alaska provides a case study of fishery resource management in a data-limited situation. The fishery progressed through several developmental phases since its inception in 1967. During the early years, the fishery was virtually unregulated. Harvests declined by the mid 1970s due to localized depletion of large scallops, establishment of closed areas to protect crabs and their habitats, and loss of product markets. Improved stock conditions and favorable seafood prices led to rapid growth of the fishery and concerns for overfishing in the late 1980s and early 1990s. Prior to 1993, the fishery was passively managed without a management plan. Since the mid 1990s, the fishery has been managed under state and federal fishery management plans (FMPs) that contain a suite of precautionary management measures including a limited entry program that prevents additional capitalization, conservative area-specific catch quotas to safeguard against recruitment overfishing, gear and crew size restrictions in part to prevent growth overfishing, and strict bycatch controls and area closures to minimize adverse fishing effects on large epifauna (e.g., crabs) and their habitats. A small fleet prosecutes the modern fishery; some vessels operate as a fishing cooperative to optimize harvest allocations among

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participants and minimize operational costs. A mandatory industry-funded onboard observer program collects data for fishery management and assures regulatory compliance. A combination of state, federal, and industry funding supports a small, ongoing research program to address extant data limitations.

Introduction

This paper describes the weathervane scallop *(Patinopecten caurinus)* fishery in Alaska as an evolving case study of fishery management with limited data. There were periods when landings statistics were the only data collected. Even today, there have been no full stock assessments of weathervane scallops in Alaska. Estimates of abundance, and therefore exploitation rates, remain unknown and biological reference points are still being developed. The evolution of this fishery reflects a rather tight coupling between development of data acquisition programs, fishery innovations, and management actions based on new data. Management is complicated by jurisdiction by the State of Alaska in state waters (<5.6 km from the coast) and federal jurisdiction in the exclusive economic zone (5.6-370.4 km offshore). However, even in this regard, state-federal management has evolved over time in attempts to improve efficiency and coordination of management actions. The fishing industry has been active to fund data collection and to help shape the current management regime. The current fishery is believed to be managed conservatively and is prosecuted by a fleet that is not overcapitalized. However, the colorful history of this scallop fishery includes the stages of development typical of most fisheries: discovery, bandwagon growth, fallback, and evolutionary development (Walters 1986). Kaiser (1986), Kruse (1994), Kruse and Shirley (1994a), Shirley and Kruse (1995), Kruse et al. (2000), Turk (2000), and Barnhart (2003) reviewed various aspects of scallop fishery history in Alaska. From these accounts we synthesize the interconnections among historical fishery development, chronology of management actions, and evolution of data collection methods to reduce levels of data limitation.

Weathervane scallops are distributed along the west coast of North America from central California to the eastern Bering Sea, and west to the Aleutian Islands (Foster 1991). Their depth distribution spans 0-300 m, but commercial densities generally occur at 46-128 m (Ronholt et al. 1977) on discrete beds, typically oriented with bottom currents that parallel bathymetry (Fig. 1). Beds may be composed of a wide range of substrates (Hennick 1973), but spatial analysis of sediment charts and fishing effort data indicates that scallop beds are typically associated with clayey silt, sand, and gravely sand sediments (Turk 2000). Most scallops mature at 76 mm shell height (SH), which is attained by age 3 or 4 years depending on area (Hennick 1970, Kaiser 1986, Ignell and Haynes 2000). The largest recorded Alaska specimen measured 240 mm SH with an

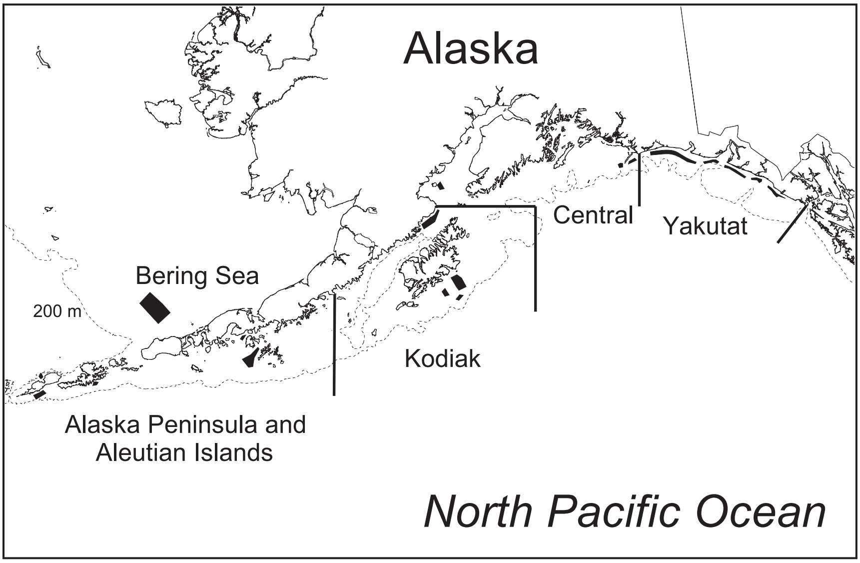

Figure 1. Map of the principal fishing grounds for weathervane scallops (solid black) in the Gulf of Alaska, Aleutian Islands, and Bering Sea, as indicated from onboard observer data. Small vessels without observers also harvest weathervane scallops from beds in Kamishak Bay, an embayment north of Kodiak Island on the western side of lower Cook Inlet in the central region. The dotted line shows the 200 m isobath.

adductor meat weight of 340 g, and the oldest specimen was 28 years old (Hennick 1973). Estimates of instantaneous natural mortality (M) range from 0.04 to 0.25, with median of 0.16, corresponding to 15% annual mortality (Kruse 1994).

In the current fishery, weathervane scallops are generally harvested by vessels towing two New Bedford style dredges. Dredges consist of a rectangular metal frame, typically 4.57 m wide weighing 1,180 kg (Barnhart 2003). An exception is a small scallop fishery in lower Cook Inlet where vessels are required to use a single 1.8 m dredge weighing 408 kg. Steel “shoes” on lower corners of the frame serve as runners on the seafloor. Attached to the frame is a bag made of 10.16 cm steel rings connected by chain links. A sweep chain footrope is affixed to the bottom of the bag. The top of the bag is composed of 15.24 cm polypropylene stretched mesh. A “club stick” (metal bar) is attached to the end of the bag to retain the shape of the ring bag and provide an attachment point for lifting and emptying its contents.

Weathervane scallops are shucked; only the single adductor muscle is retained. Scallop meats are 8-12% of total live weight, depending on area and season (Barnhart and Rosenkranz 2003). Meats are graded by size and sold to domestic seafood markets. Early attempts to market scallop mantles and gonads were unsuccessful, and even today a product with roe, which is highly desirable in European markets for some other scallop species, is not prepared from weathervane scallops partly due to concerns about paralytic shellfish poisoning in Alaska.

Fishery history

Initial fishery development (1967-1973)

The U.S. Bureau of Commercial Fisheries (later named National Marine Fisheries Service, NMFS) explored the distribution of weathervane scallops in Alaska during periodic surveys since 1953 (see Table 1 in Haynes and Powell 1968 and Appendix 3 in Turk 2000). Beds were located by scallop dredge surveys and analysis of scallop bycatch from groundfish trawl surveys. Despite these early efforts, it was not until 1967 that loss of fishing opportunities associated with declines of red king crab *(Paralithodes camtschaticus)* catches led to the first establishment of a small fishery (2 vessels) off Kodiak. During 1967 and part of 1968, scallops were delivered live to shoreside processors; however, this proved uneconomical and scallopers began delivering shucked scallop meats to processors for packaging and freezing (Kaiser 1986). In 1968, 19 vessels harvested 761 metric tons of scallop meats from the Yakutat and Kodiak management areas combined (Fig. 2). The fleet included numerous vessel types, including specialized vessels from New England and Alaskan fishing vessels converted from fisheries for crabs (pots), salmon seiners and gillnetters, halibut longliners, and shrimp trawlers (Kaiser 1986). The fishery peaked in 1969, when 19 vessels harvested 839 t of scallop meats. Thereafter, interest in the fishery began to wane perhaps due to a combination of weaker markets, high operating costs, and loss of crews who were attracted to higher paying jobs with the construction of the Alaska oil pipeline (Turk 2000). During 1970-1973, just 5-7 vessels landed an average of 527 t annually. Those remaining in the fleet were the most efficient vessels with capabilities to fish both in the day and night—vessels that measured 27-30.5 m in length and towed two 3.7-4.3 m wide dredges weighing 725-910 kg each (Hennick 1973). Shucked scallops were washed, placed in 18 kg cloth bags, and iced in the vessel's hold. During these early years, all scallop fishing was confined to the Kodiak and Yakutat areas, and catches were delivered to shoreside plants in Kodiak and Seward, Alaska.

Commercial landings statistics were collected since the inception of the fishery. Early data were limited to landings statistics, except during 1969-1972 when a small voluntary observer program involved a few cooperating vessels (Hennick 1973). Observer data included age, sex, maturity, size, fishing area, days fished, catch rate, tow length, depth, bottom type, catch and bycatch, and damage to caught species. These data showed that the percentage of scallops ≥7 years old in the catch declined from 74%-96% in 1968-1970 to 39%-71% in 1971-1972, reflecting the typical fishing-up effect (Walters 1986). Observer data showed that Dungeness crab *(Cancer magister)* appeared seldom as bycatch. One to four red king crab were caught per tow, and these rates varied by area (none off Yakutat) and season (highest in spring when king crabs migrate inshore to molt and mate). An average of 20-40 small (2.5-8 cm carapace width) Tanner crabs *(Chionoecetes bairdi)* were caught per tow (Hennick 1973).

From its inception, the Alaska weathervane scallop fishery was managed by the State of Alaska. During 1967-1968, no areas were closed to fishing, all gear types were allowed, and the only regulatory requirements were the purchase of a vessel license and commercial fishing license (Kaiser 1986). Although some concerns were expressed about harvest levels as early as 1970, most management concerns revolved around potential fishing effects on bycatch of fish, other shellfish, and their habitats. In 1969, the state regulatory body, Alaska Board of Fisheries (BOF), created time and area closures in selected waters off Kodiak Island and the Alaska Peninsula to protect molting and mating king and Tanner crabs. Regulations were based on crab bycatch data collected by observers. These regulations were renewed and modified annually until a June 1–March 31 scallop fishing season was established in 1971 for portions of the Kodiak fishing district. Few regulations were imposed in other areas of the state, except for portions of Cook Inlet where seasons and closed waters were also established. Gear was restricted to longlines, trawls, and dredges, and after June 1, 1969, dredges were required to have rings ≥102 mm diameter (Kaiser 1986).

Fallback phase (1974-1979)

The fishery steadily declined after 1973 (Fig. 2). Landings averaged just 139 t during 1974-1977 and no landings were reported in 1978. All harvests were taken near Kodiak and Yakutat. The geographic range of fishing trips was limited because fresh scallops were accepted only if they were caught less than 10 days prior to delivery; portions of catches were often discarded due to decomposition (Turk 2000). In the early to mid 1970s, vessels conducted numerous, generally unsuccessful, exploratory cruises in attempts to boost landings to prior levels (Hennick 1973). The decline in this period of the fishery was attributed to multiple causes: (1) area closures and season restrictions that reduced fishing opportunities; (2) limited distribution of scallop beds off Kodiak and Yakutat; (3) unreliable and generally declining prices paid by processors for landed

Table 1. Links among fishery managment, data acquisition, and weathervane scallop fishery developmental stages during 1967-2002.

Year	Management action	Data acquisition	Fishery stage
1953-1966	No fishery management plan (FMP).	Relative abundance and distribution from surveys.	None.
1967	Agencies promote fishery development.	Landings database (fish tickets) initiated.	First small commercial landings off Kodiak.
1968-1972	State management begins. Passive regulations include gear, seasons, and closed areas.	One survey and small onboard sampling program collect age, shell height, weight, size at maturity, meat recovery, and bycatch.	Full fishery develops off Kodiak and Yakutat, scallops depleted from fished beds, vessels decline from 19 to 5.
1973-1978	No new action.	Surveys and sampling programs discontinued.	Landings, CPUE, and vessels decline to zero in 1978.
1979-1989	No new action.	Fish ticket data (only) continues. First survey conducted in Kamishak Bay (1984).	Fishery re-develops and expands to SE Alaska, Cook Inlet, Aleut. Is. and Bering Sea. Mean vessel size increases.
1990-1993	High-impact emerging fishery declared. State adopts interim FMP (1993), including crew limits, crab bycatch limits. Shucking machines banned.	Mandatory industry-funded onboard observer program starts (1993). Data collected on age, shell height, weight, meat recovery, and bycatch. Early emphasis on crab injuries in bycatch.	Catches escalate, first catches from Prince William Sound, vessel and crew sizes increase, overfishing concerns for some areas.
1994	State adopts FMP, including registration areas, area-specific guideline harvest ranges (GHRs), new gear restrictions, and fishing seasons.	Fishery economic data studied for possible vessel moratorium.	Fishery considered as overcapitalized.

Table 1. (Continued.)

Year	Management action	Data acquisition	Fishery stage
1995	NMFS adopts emergency rule and federal FMP closes fishery.	Scallop survey conducted off Kayak Island (PWS).	1 vessel does not register, avoids state regulations, and catch is 200% of GHR in PWS.
1996	FMP Amendment 1 approved, federal regulations parallel state regulations.	Routine surveys resumed in Kamishak Bay and Kayak Island.	Fishery resumes. Bering Sea harvest is constrained by bycatch caps.
1997	FMP Amendment 2 (vessel moratorium for federal waters) established, and vessel moratorium established for state waters.	No change.	18 vessels qualify, harvest stabilizes at about half of highs.
1998	FMP Amendment 3 approved, management delegated to state. FMP Amendment 6 (definition of overfishing, MSY, and OY) approved.	No change.	Overall landings stable.
1999	FMP Amendment 5 (essential fish habitat) approved.	No change.	Overall landings stable.
2000	FMP Amendment 4 (license limitation program) approved.	Age-structured assessment model developed for Kamishak Bay.	7 vessels participate, fishing cooperative formed by industry.
2001	No change.	Establishment of a pilot video stock assessment program.	Participation falls to 6 total vessels, not all GHR harvested.

FMP = fishery management plan.
GHR = guideline harvest range.
MSY = maximum sustainable yield.
OY = optimum yield.
PWS = Prince William Sound.

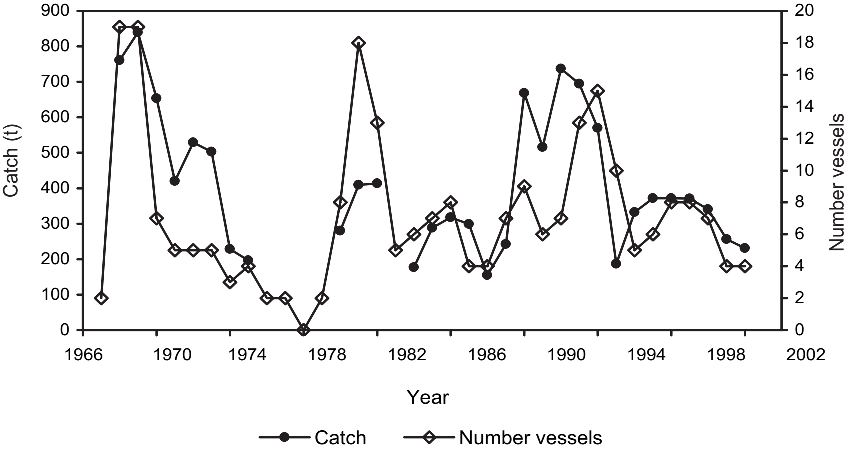

Figure 2. Numbers of participating vessels and reported landings (metric tons) of weathervane scallop meats in Alaska during 1967-2002. Scallop meats account for approximately 8-12% of the total (round) weight of harvested scallops, depending on area and season (Barnhart and Rosenkranz 2003). Landings are confidential and are not shown in years when fewer than three vessels participated in the fishery.

scallops in the early to mid 1970s (Fig. 3); and (4) increasing operational costs (Kaiser 1986).

Aside from landings statistics, virtually no biological and fishery data were collected during 1973-1978, a severely data-limited period. The observer program was discontinued owing to limited agency funding. Age composition data are unavailable, with the exception of one sample from the east side of Kodiak in 1975 which showed the continued predominance of young scallops (Kaiser 1986). In 1979, limited size composition data were collected, and catch rates were similar to previous historical averages (Kaiser 1986).

During 1973 and 1975, the BOF closed additional areas near Kodiak to scallop dredging, and fishing seasons were shortened to July 16–March 31 to afford greater protection to molting and mating crabs in late spring–early summer (Barnhart 2003). However, other portions of the Kodiak area and other areas of the state remained open to scallop dredging year-round.

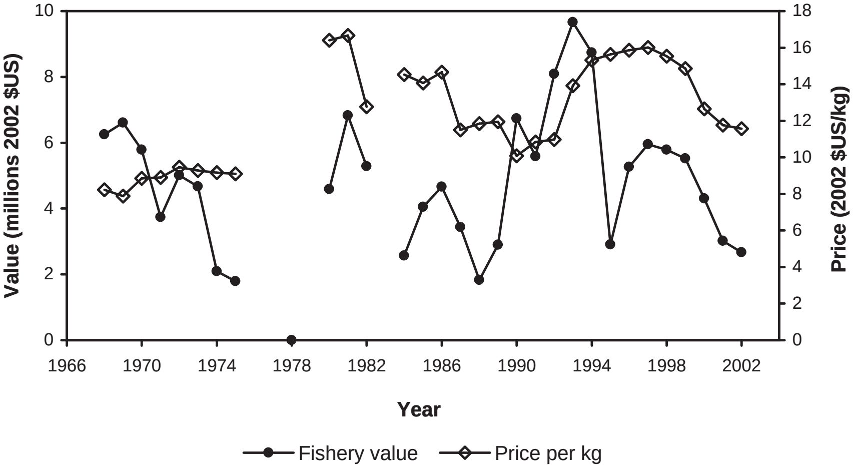

Figure 3. Estimated landed value and ex-vessel price (price paid to fishermen by processors) of weathervane scallops in Alaska during 1967-2002. Value and price have been adjusted to 2002 dollars based on the Consumer Price Index–Urban Research Series available from the U.S. Department of Labor, Bureau of Labor Statistics. Landings are confidential and are not shown in years when fewer than three vessels participated in the fishery.

Redevelopment phase (1980-1989)

Renewed interest in the Alaskan weathervane scallop fishery in the 1980s was partly attributed to both significantly improved prices (Fig. 3) and the overcapitalized Atlantic sea scallop *(Placopecten magellanicus)* fishery in New England, where numbers of U.S. vessels increased from 44 to 200 during 1975-1979 and landings declined 22% over 1978-1981 (NEFMC 1982). These same factors led to a boom-bust fishery off the Oregon coast in which 532 t of scallop meats were harvested in 1981 by 118 vessels; approximately 20 of these vessels were nonresident (Starr and McCrae 1983). Subsequent Oregon landings have been very small (Bourne 1991). During 1980-1983, Alaska landings increased from 280 t to 414 t in 1980-1982 (Fig. 3) despite declining participation (18 to 5 vessels). During 1984-1989, an average of 6 vessels delivered 247 t annually.

Vessels became increasingly specialized (Shirley and Kruse 1995). In 1983, vessels earned 60% of their incomes from scallops, the remainder coming from landings of crabs, shrimps, clams, herring, salmon, halibut, and other groundfish. By 1989 scallops accounted for 85% of total fishery earnings by vessels participating in the scallop fishery. Mean vessel size increased; during 1983-1987, the percentage of small (<18 m), medium (18-31 m), and large (>31 m) vessels averaged 49%, 34%, and 17%, respectively (Kruse and Shirley 1994b). By 1988-1989, there were 64% medium and 36% large vessels with no small vessel participation.

In the 1980s, harvests were first taken from areas outside of Yakutat and Kodiak including Southeast Alaska (1980), Alaska Peninsula and Dutch Harbor (1982), Cook Inlet (1983), and Bering Sea (1987). During the 1980s, Kodiak scallop beds still contributed most (44%) to statewide harvest, followed by the combined Alaska Peninsula–Aleutian Islands (mainly Dutch Harbor and Adak) areas (31%), Yakutat (22%), Bering Sea (2%), and the central region (1%), composed of Prince William Sound (near Kayak Island) and Kamishak Bay (west side of lower Cook Inlet). The fishery was lucrative. Mean landings and ex-vessel value (i.e., the price paid for fishery landings) of scallops per vessel increased from 35.0 t worth US$0.57 million in 1980 to 74.7 t worth US$0.86 million in 1987. Through the 1980s, scallop fishermen continued to ice scallop meats in 18 kg bags for delivery to shoreside processors for freezing and distribution (Turk 2000). Scallop product quality and prices remained variable owing to the length of fishing trips. Mean (inflation-adjusted to 2002) prices declined from $16.40 per kg in 1980 to $11.95 per kg in 1989.

The years 1980-1989 were severely data limited—only fish tickets (i.e., records of landings purchased by processors from fishermen) and commercial operators' annual reports (i.e., annual reports submitted by seafood processors which include processed product forms and seafood prices) were collected with one exception. In 1984, the first ADFG scallop survey was conducted in Kamishak Bay. The Alaska fishery remained passively managed with miscellaneous regulations concerning gear, fishing seasons, and closed areas. All Alaska waters were managed as a single registration area.

Bandwagon growth phase (1990-1993)

Harvests tripled from a mean of 224 t per year in 1983-1989 to 649 t per year in 1990-1993. The 1992 harvest of 737 t, worth US$8.1 million (2002 dollars), was the second largest in the history of the fishery. That year, mean ex-vessel value of scallop landings exceeded $1 million per vessel. The 1993 harvest of 694 t worth $9.7 million (2002 dollars) was the most valuable in the history of the fishery to date. During 1990-1993, participation increased from 9 to 15 vessels. Vessels were fully specialized, earning 100% of their fishery incomes from scallops (Kruse and Shirley 1994b). Mean vessel size increased 85% from 18.5 m in 1983 to 34.3 m in 1991 (Shirley and Kruse 1995). As vessels became larger and more specialized, crew sizes increased. In the early 1980s average crew sizes were 5-8 persons depending on area, but by 1993 all vessels except the smallest carried 12-person crews (Shirley and Kruse 1995). Automatic shucking machines were first used in 1991. Automatic shucking machines and larger crews facilitated economical processing of small scallops (Kruse and Shirley 1994b). During 1990-1993, most vessels installed onboard freezing operations to assure consistent, high-quality product. Partly as a result, ex-vessel prices increased $1 per kg during these three years.

As fishing effort shifted to new areas and harvests approached historical highs that were not sustained, conservation concerns developed. In 1992, ADFG released a draft interim FMP for public comment (Kruse et al. 1992). In 1993, ADFG declared the fishery to be a "high impact emerging fishery"—a State of Alaska designation applicable when at least one of four conditions are met: (1) harvesting effort recently increased beyond a low sporadic level; (2) the resource is harvested by more than one user group; (3) harvests approach levels that may not be sustainable; and (4) the BOF has not developed comprehensive regulations to address conservation and allocation issues. ADFG found that these conditions applied.

The interim FMP included a broad goal to maximize overall long-term benefits of scallop resources, while providing for conservation of scallop populations and their habitats. Five objectives were to: (1) ensure long-term viability of scallop populations; (2) minimize adverse effects on benthic species and habitats; (3) ensure conduct of manageable, steady-paced scallop fisheries that provide stable employment and supplies of high-quality products to seafood markets; (4) ensure harvest requirements of traditional users in coastal communities; and (5) gather new data relevant to attaining other objectives (Kruse et al. 1992). Key provisions of the interim FMP and subsequent BOF actions included: (1) establishment of eight (later became nine) separate registration areas; (2) reporting requirements; (3) gear specifications that limit dredge width ≤4.57 m and rings ≥10.16 cm inside diameter; (4) guideline harvest ranges (catch limits) with upper bounds based on long-term average catch excluding extreme highs and lows; (5) bycatch caps for red king crab and Tanner crab based on 1% of assessed crab abundance if the crab fishery was conducted or 0.5% if the crab fishery was closed owing to low crab abundance; (6) in-season adjustments, which allow managers flexibility to adjust to unforeseen circumstances; (7) closed waters to avoid crab bycatch and sensitive habitats; (8) fishing seasons; (9) observer requirements in which all scallop vessels (except small vessels fishing in lower Cook Inlet) are required to carry an observer at their expense; (10) a limit of 12 crew members per vessel; and (11) a ban on automatic shucking machines.

Some fishery participants were concerned about escalating fishing effort. At the time, the State of Alaska's Commercial Fisheries Entry Commission could limit participants, but not numbers of vessels. The North Pacific Fishery Management Council (NPFMC) can limit vessels in federal waters. Because vessel limitation was the preferred method, in 1993 the NPFMC prepared options for limited access considerations. At the same time, the NPFMC began to analyze the need for federal FMP for this fishery.

Fishery rationalization phase (1994-present)

Statewide landings of scallops were 570 t in 1994, fell to 186 t in 1995, averaged 358 t during 1996-2000, and averaged 244 t in 2001-2002 (Fig. 2). Prices were high, averaging $15.67 per kg (2002 dollars) during 1994-1998, but declined to $11.57 per kg in 2001-2002 (Fig. 3). Ex-vessel value was $8.7 million (2002 dollars) in 1994, dipped to $2.9 million in 1995, averaged $5.4 million over 1996-2000, and fell to $2.7 million in 2002 (Fig. 3). Statewide landings during 1994-2002 were taken primarily from Kodiak (35%), Yakutat (31%), and the Bering Sea (22%), with smaller amounts from the Alaska Peninsula–Aleutian Islands area (6%), and central region (6%).

In 1994, the state's BOF adopted a scallop FMP (Kruse 1994) largely patterned after the interim plan. The coordinated development of this state management plan and federal plan for license limitation was interrupted in 1995, when one vessel (F/V *Mister Big*) relinquished its state fishing permit and continued fishing outside state waters (>5.6 km) off Kayak Island (northeastern Gulf of Alaska) after the annual guideline harvest range was already taken for the area. Previously, all vessels fishing in state and federal waters were presumed to be subject to state regulations while fishing under a state fishing permit. As the F/V *Mister Big* no longer held a state permit, it continued to fish without regulation, thus challenging state authority over the fishery. Concerned about unregulated harvest in excess of the prescribed sustainable harvest for the area, the NPFMC met in emergency session and NMFS was requested to implement an emergency rule to close all federal waters to scallop fishing so as to temporarily close the loophole in state jurisdiction. Later that year, the NPFMC adopted a federal FMP (NPFMC 1995a), which formalized the closure of federal waters to scallop fishing. With this safeguard in place, some state waters were open in 1995, but most harvest was confined to Yakutat where catch rates were sufficiently high in state waters to warrant interest. Federal waters remained closed for 1.5 years during February 13, 1995–August 1, 1996. Thus, the F/V *Mister Big*, which relinquished its state license, was banned from fishing during this period.

Following the approval of the first federal FMP that closed federal waters to scallop fishing, six amendments to the federal FMP were approved during 1996-2000. In 1996, Amendment 1 adopted federal regulations parallel to those adopted by the state (NPFMC 1997). In 1997, Amendment 2 established a vessel moratorium in which 18 vessels qualified (NPFMC 1997). In that same year, by special action of the Alaska Legislature, a state-waters moratorium was established that qualified 10 vessels. In 1998, Amendment 3 to the federal FMP effectively delegated scallop fishery management to the State of Alaska (NPFMC 1998). The fishery was deemed to be overcapitalized in 1993 when 15 vessels participated (NPFMC 1995b). In 2000, Amendment 4 was approved, creating a license limitation program (NPFMC 2000). Seven vessels were permitted to fish statewide (except Cook Inlet) and two other vessels were permitted for Cook Inlet. These two small vessels are currently exempted from onboard observer requirements, but they are required to submit logbooks and they are limited to a single 1.83 m dredge. Notably, the F/V *Mister Big* was not among the nine licensed vessels. Finally, plan Amendments 5 and 6 satisfied recent federal requirements for designations of essential fish habitat and definition of overfishing (NPFMC 1999a,b).

Independently, most vessel owners formed a fishery cooperative in 2000. Within the cooperative, owners are allocated shares based on fishing history. Some owners arranged to have their shares harvested by other members and removed their vessels from the fishery. This further reduced fishing effort (just 4 of 7 cooperative vessels fished in 2001-2002), and extended the fishery for remaining participants. This action allows close coordination with fishery managers to precisely attain catch limits. Also, cooperative members are proactive in employing measures to reduce crab bycatch. Vessel operators provide confidential data to third-party consultants who review catch, bycatch, and location data and identify high crab bycatch areas. Near real-time analyses are provided back to the fleet, so they can adjust their operations to avoid triggering premature closures due to bycatch. An unforeseen side effect of effort reduction measures is that fishing capacity was too low in 2001 and 2002 to attain guideline harvests before regulatory fishing season closure dates that are set to protect molting and mating crabs. In part, this resulted from market-related decisions by the fleet. One vessel splits its fishing effort between scallops and the lucrative Bering Sea crab fishery. Also, in summer and early fall the small fleet tends to fish in the western Gulf of Alaska, where large, high-quality scallops command the best prices. Later, after catch quotas in the western gulf have been taken, the fleet moves to the central and eastern gulf. However, by then late fall and winter storms limit fishing opportunities and this region of the coast affords few places for vessels to hide from storms. As a result of these factors, an annual average of about 45 t of catch quota was not harvested from the eastern Gulf of Alaska in 2001 and 2002.

Since 1993, the imposition of an onboard observer program led to a marked increase in data collection from this fishery. Unlike the voluntary observer program on a few vessels in the early years of the fishery, this new program is mandatory and funded at the vessel owner's expense for all vessels fishing outside of Cook Inlet. Observers obtain data on scallop age, shell height, weight, meat recovery, and bycatch. As part of this observer program, vessel operators maintain logbooks with haul locations, tow duration, tow speed, bottom depth, and catch of retained scallops. These data have afforded analyses on bycatch, condition of discarded crabs and halibut, catch rates, and finer resolution of the location of the harvest (e.g., Barnhart 2003, Barnhart and Rosenkranz 2003). Detailed observer data allowed industry-funded analyses of scallop-habitat associations and preliminary biomass estimates using depletion estimators (Turk 2000).

Also, ADFG conducted fishery-independent assessment surveys in Kamishak Bay in 1996, 1998, and 1999, and near Kayak Island in 1996, 1998, and 2000 (e.g., Bechtol 2003, Bechtol and Gustafson 2002) using a research dredge donated by the Kodiak Fish Company. An age-structured stock assessment model was developed for the Kamishak Bay stock (Bechtol 2000), but time series of age data are too short to apply this model to other Alaskan scallop stocks at the present time.

Beyond routine stock assessment and fishery monitoring, federal funding supported recent applied research by ADFG into genetic stock structure, age validation, and development of video stock assessment technology. To date, analyses of allozyme variability and preliminary analyses of mitochondrial DNA (mtDNA) variability do not suggest much genetic differentiation among scallop beds within Alaska (James Seeb, Alaska Department of Fish and Game, Nov. 2004, pers. comm.). Recently, single nucleotide polymorphisms (SNP) in both mitochondrial and nuclear genes were identified for weathervane scallops (Elfstrom et al. 2005). These SNP developments, together with a full investigation of mtDNA variability, may be useful for delineating scallop population structure for refinement of stock boundaries. Regarding age validation, analyses of oxygen isotope ratios from samples off Kodiak confirmed that checks are formed on scallop valves annually in November-December (Barnhart and Carpenter 2003). Checks seem to be formed during the warmest months of the year associated with intense coastal downwelling. Two scallop fishing vessels have been equipped since 2003 with data loggers to verify seasonal temperature profiles on scallop beds relative to annual check formation. Pilot video stock assessments were conducted in the eastern Gulf of Alaska in 2002 using a towed sled equipped with a miniature video camcorder (Rosenkranz and Byersdorfer 2004). The sled was successfully deployed at 135 stations and over 12,000 scallops were enumerated from 124,000 m^2 of seafloor. These methods contrast with those of Stokesbury (2002), who used a video drop camera to survey *Placopecten magellanicus* on Georges Bank (U.S. East Coast). This video drop camera was attempted in Alaska, but scallop densities were too low and 90% of the drops contained no scallops. However, dredge video surveys were found to be a viable assessment approach for weathervane scallops, and routine surveys are planned for the three most commercially important fishing grounds in Alaska.

Conclusions

The Alaskan weathervane scallop fishery underwent a major transformation from a self-regulated, passively managed fishery in the 1970s to a very actively managed fishery highly constrained by regulations designed to maintain fishery sustainability and reduce adverse dredging effects on other benthic species and habitats. Limited access and formation of a scallop industry cooperative have been key to maintaining a lucrative fishery for the few remaining participants. Mean gross earnings per vessel during 2000-2002 were \$0.6-\$0.8 million (2002 dollars). Excellent cooperation among industry and fishery managers has been essential to attain harvest guidelines while minimizing bycatch.

Data availability also changed markedly since fishery inception. After an initial data collection program in the late 1960s to early 1970s, virtually no data were collected in the 1980s. Today, catch and bycatch of vessels are well documented by onboard observers. These data are used to set preseason harvest levels as well as for in-season fishery management. Two scallop stocks (Kamishak Bay and Kayak Island) are assessed by fishery-independent dredge surveys, and collection of specimens for age and growth studies is at an all-time high.

Several key data limitations remain, but the outlook for further improvements is promising. Lack of data has prevented a full stock assessment of major commercial scallop stocks. New video assessment technology should not only extend the current small assessment program, but should also allow estimation of dredge catchability, a critical parameter for stock assessments. Also, the continued accumulation of time series of age and size distribution data will allow broader application of age-structured assessment models (e.g., Bechtol 2000) to scallop stocks in Alaska. Moreover, these same time series will permit analyses of recruitment, growth, and mortality so that biological reference points, such as spawning stock biomass per recruit and minimum stock size thresholds, can be incorporated into the harvest strategies. An ultimate goal is to implement biomass-based exploitation rates.

The current level of data collection would not be possible without the cooperative efforts of government agencies and the fishing industry. The industry has funded graduate student projects (e.g., Turk 2000), purchased survey gear for ADFG, and funds the onboard observer program. In 2002, the annual cost of the observer program was about \$161,000, roughly 6% of the \$2.67 million in ex-vessel fishery value that year. State and federal funds cover the costs of ADFG's scallop research and management program with an annual cost of about \$400,000 in 2002. These combined costs, though relatively expensive compared to the value of this small fishery, are investments to attain fishery management goals and objectives.

Acknowledgments

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A Phased-Development Approach in the British Columbia Sea Cucumber Fishery

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Abstract

The sea cucumber (*Parastichopus californicus*) fishery has existed in British Columbia (BC) since 1980 under an arbitrary management regime. Although there were no indications of serious stock decline, the fishery was reviewed in 1995 due to a lack of biological data upon which to base management decisions. Following a review of available information on the biology, distribution, and productivity of sea cucumbers, new management restrictions were imposed in 1997. In the absence of abundance and life-history data from BC stocks, conservative estimates of density and productivity from studies conducted in Alaska and Washington state were used to calculate quotas for the BC fishery. The existing arbitrary quota in BC was maintained, but over only 25% of the total coastline. One half of the coastline was closed to harvest until sufficient knowledge had accumulated to permit informed management. Experimental fisheries, designed to investigate stock response to varying harvest rates, are being conducted in the remaining 25% of the coast. Abundance surveys are being conducted in open fishery areas as well. These projects are only possible through the strong cooperation with the commercial industry and First Nation stakeholders. Funds generated by the sale of experimentally fished product are used to fund the field activities and salary for a biologist with Fisheries and Oceans Canada. Four experimental fisheries are under way and are nearing the halfway mark of the planned 10-year time frame. Eight density surveys have been conducted over 30% of the area that is open to commercial fishing. Early results of these investigations are discussed.

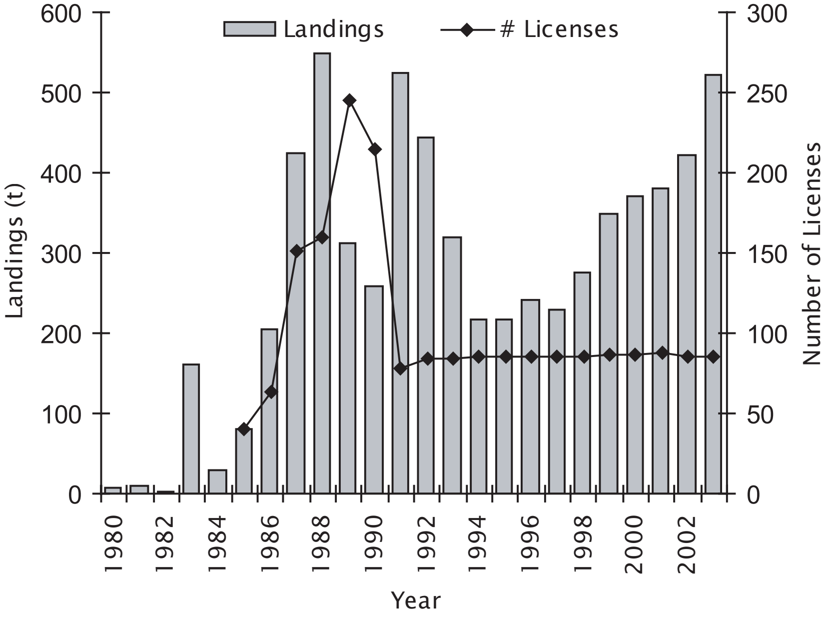

Figure 1. Historical landings (split weight, from logbook data) and number of commercial licenses in the British Columbia sea cucumber fishery. Landings from 1983 to 1990 were reported as round (unsplit) weights and were converted to split weights by dividing round weights by 2.73.

Introduction

Sea cucumbers (*Parastichopus californicus*) were first commercially harvested in British Columbia (BC) in 1971; however, the fishery was not regulated under commercial licensing until 1980 (Campagna and Hand 2004). Sea cucumbers are hand picked by divers and sold to Asian markets as trepang (brined and dehydrated skin) and frozen meat. The value of the fishery in 2003 was Can $2.0 million. A small food fishery is conducted by First Nation communities and a small sport fishery also exists.

The commercial sea cucumber fishery expanded rapidly after 1980 to exceed 1,500 t landed round weight, equivalent to 549 t split (sliced and gutted) weight in 1988 (Fig. 1). Management measures to limit the expansion of the fishery, namely area closures and arbitrary area quotas, were introduced in 1986. Landings and the number of licenses continued to increase, quotas were often exceeded, and CPUE declined in some areas, which prompted the implementation of quota reductions in 1989, 1993, and 1994 and license limitation in 1991 (Fig. 1). A three-year area rotation was introduced in 1993 in the south coast and a pilot individual quota (IQ) system, requiring validation of all landings, was introduced in 1995.

The sea cucumber fishery came under review in 1995 due to the lack of biological data upon which to base management decisions. Life history characteristics important to the understanding of productivity of sea cucumbers are largely unavailable in BC (Hand and Rogers 1999). Little is known about stock abundance and distribution, longevity, growth rates, rates of natural mortality, and recruitment (Phillips and Boutillier 1998). Growth studies are made difficult by the animals' ability to rapidly change shape and size, and by annual fluctuations in body mass due to resorption of visceral organs. Age determination studies have not been successful due to lack of hard body parts, the lack of length frequency modal patterns beyond three years, and the inability to tag and follow individuals for long periods (Phillips and Boutillier 1998). Therefore, in 1997, the sea cucumber fishery was classified as a "developing and data-limited fishery" and has since been managed under a precautionary regime known as the "phased approach" for new and developing fisheries (Perry et al. 1999). The phased approach consists of three phases. First, phase 0 involves a review of existing biological and fisheries information on the species of interest. Second, phase 1 consists of the collection of new information through surveys and experimental fisheries to fill information gaps identified in phase 0. Third, phase 2 consists of implementing the chosen management strategies in a commercial fishery and monitoring the fishery.

A phase 0 review of available biological and fisheries data was completed, critical information gaps identified, and recommendations for management and further research were made (Boutillier et al. 1998, Phillips and Boutillier 1998). Following some of these recommendations, the rotational fishery was discontinued in 1997 to provide an unbroken time-series of fisheries data, and fishery expansion was curtailed, pending the collection of abundance and productivity data from BC stocks. The fishery was restricted to 25% of the coast ("open areas"), while 25% of the coast was set aside for experimental fisheries and the remaining 50% of the coast is to remain closed to fishing until a biologically based and risk-averse management system can be implemented.

The sea cucumber fishery is currently in phase 1, i.e., data gathering through experimental fisheries and surveys (Hand and Rogers 1999). The arbitrary quota in open areas was replaced by a quota based on 4.2% exploitation rate (State of Washington, unpubl. data and analyses) applied to a presumed baseline estimate of 2.5 cucumbers per meter of shoreline (per m) from Alaska survey results (Larson et al. 1995, Boutillier et al. 1998). In 1997, a research program was developed with the Pacific Sea

Cucumber Harvesters Association (PSCHA). The research program has two main focuses aimed at filling the information gaps identified in the phase 0. First, density surveys in open areas ("open surveys") are conducted to gather data on abundance and distribution of sea cucumbers in BC for setting quotas. Second, experimental fishery areas (EFAs, see details below) were set up to help determine an appropriate harvest rate for sea cucumbers in BC. The research program was designed to be cost neutral; the PSCHA sells product harvested during experimental fisheries and uses the proceeds to charter vessels from the sea cucumber fishing fleet to complete the survey and harvest work. Additional profits go to the PSCHA research fund to pay for Open Surveys and for a biologist with Fisheries and Oceans Canada (DFO).

Open surveys are used to estimate sea cucumber densities for biomass and quota calculations, replacing the assumed baseline estimate of 2.5 per m. Thus, the "proven production potential" concept (Pearse and Walters 1992), where stakeholders are responsible for proving stock production before further expansion of the fishery, is being applied to the sea cucumber fishery (Perry et al. 1999). Areas covered by open surveys are re-surveyed every four years to monitor trends in density and mean weight (Campagna and Hand 2004). To date, eight open surveys have been conducted (two of which were re-surveys), covering over 30% of the coastline open to fishing. This large data set was used to calculate a new baseline density estimate and, as of 2003, unsurveyed areas are assigned a new baseline density of 5.1 per m, the lowest of the lower 90% confidence interval (CI) of all BC open surveys (Campagna and Hand 2004). Quotas for surveyed areas are based on the lower 90% CI of survey density. The original, more conservative density estimate of 2.5 per m was retained for exposed areas of low density or areas thought to have been overharvested. The revised density estimates from open surveys resulted in a more than doubling of the allowable catch from 230 t in 1997 to 520 t in 2003.

Four EFAs were set up throughout the BC coast (Fig. 2) to examine the effects of different exploitation rates on sea cucumber populations. Each EFA consists of five noncontiguous sites, measuring 10 km of shoreline each, which are harvested yearly at exploitation rates that bracket the 4.2% rate currently used in open areas: 0% (control), 2%, 4%, 8%, and 16%. The EFAs were established in Laredo Inlet and Tolmie Channel in the Central Coast of BC, Jervis Inlet in the Inside Waters of Vancouver Island, and Zeballos on the West Coast of Vancouver Island. The Kitasoo Fisheries Program, a local First Nations organization, participates in the surveying and harvesting of the two Central Coast EFAs. The EFA projects were envisioned to require approximately 10 years to yield useful results and have been ongoing for five to six years. Surveys and experimental harvests are conducted by sea cucumber harvesters and/or First Nations divers using their own vessels, but under the direction and direct super-

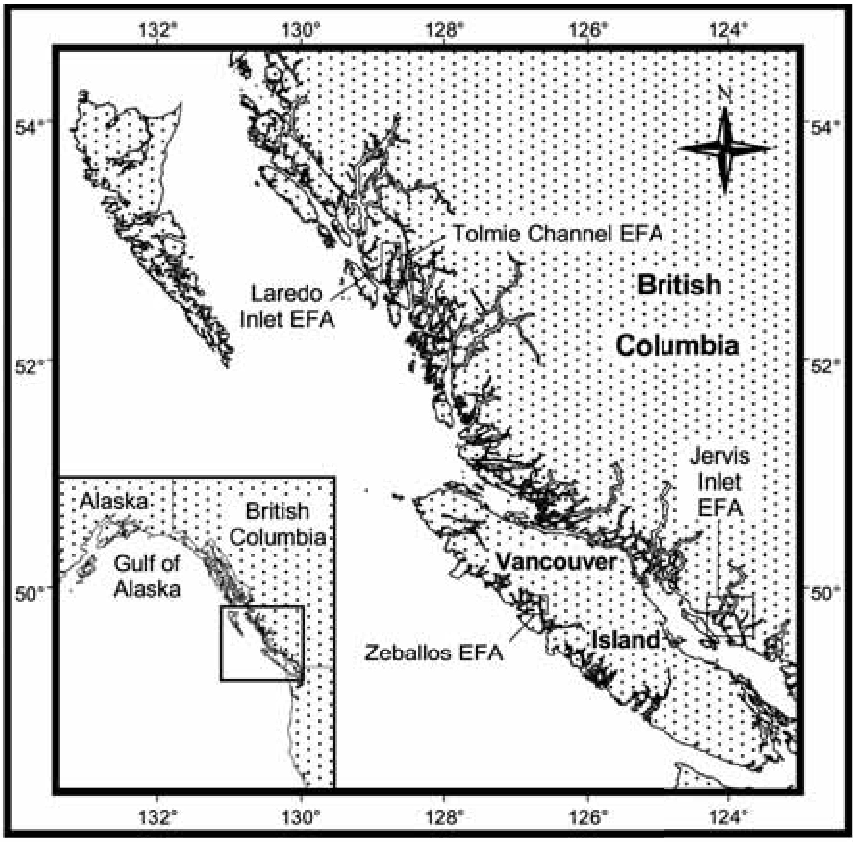

Figure 2. Map of the Northeast Pacific Ocean and the British Columbia coast showing the location of the four sea cucumber experimental fishery areas (EFAs).

vision of a DFO biologist. Here, we present preliminary results obtained from the EFA in Jervis Inlet.

Methods

Data collection

The Jervis Inlet EFA is located on the east side of the Straight of Georgia (Fig. 3). Two sites (8% and 16%) were on the edges of Malaspina Strait while the remaining three sites (0%, 2%, and 4%) were located in nearby inlets. Most sites are characterized by moderate to steep slopes with a

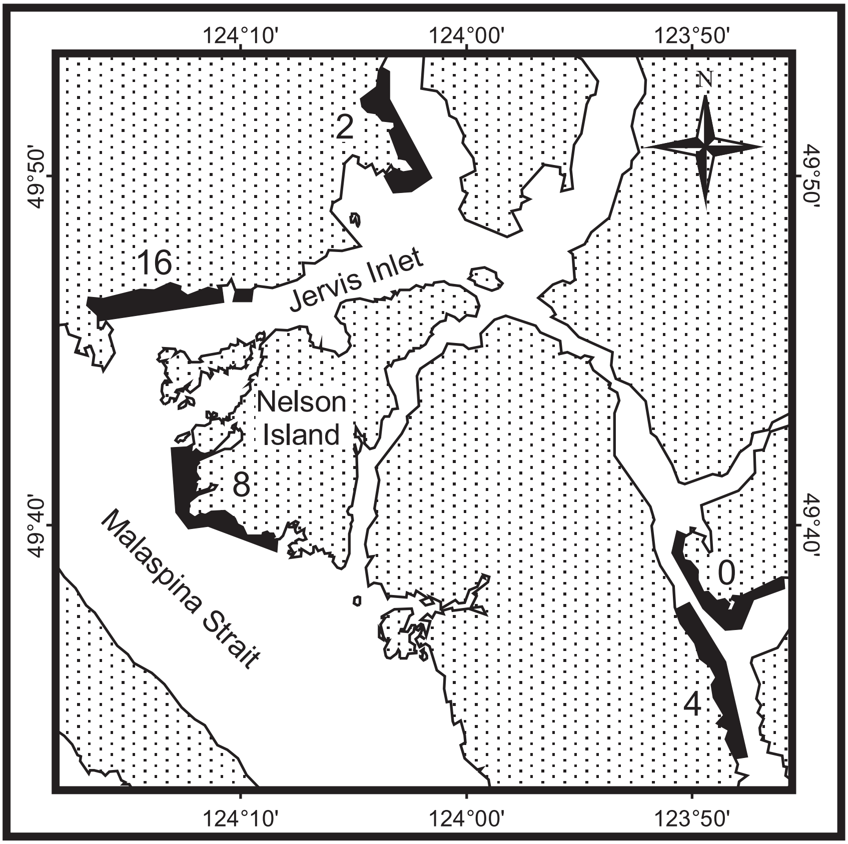

Figure 3. Map of the Jervis Inlet experimental fishery area (EFA) showing the location of the five study sites under exploitation rates of 0, 2, 4, 8, and 16%.

rocky substrate of mixed bedrock, boulders, cobble, and gravel (Campagna and Hand 1999).

A pre-survey of the Jervis Inlet sites was conducted in 1998 to get preliminary sea cucumber density estimates and to choose site locations. Results from the presurvey were used to determine the number of transects required at each site to yield density estimated with a 15% precision target (*D*) using the following formula (Elliott 1971):

$$D = \frac{SE}{Mean} = \frac{1}{\bar{x}}\sqrt{\frac{s^2}{n}}$$

Table 1. Density, experimental fishery quotas, and number of transects for each site in the Jervis Inlet sea cucumber EFA.

Site	1999 density (mean ± SE) (cucumbers per m shoreline)	Annual quota (no. cucumbers)	Number of transects
0% (control)	8.44 ± 1.28	0	25
2%	7.02 ± 1.58	1,440	25
4%	16.47 ± 3.35	6,588	15
8%	10.05 ± 2.70	8,040	25
16%	9.30 ± 2.00	14,880	16

where SE is the standard error of the mean density estimate, $\bar{x}$ is the mean density estimate, s^2 is the variance of the mean density estimate from the presurvey, and n is the desired number of transects. To achieve 15% precision on the density estimates, the required number of transects is given by:

$$n = \frac{s^2}{D^2\bar{x}^2} = \frac{s^2}{0.0225\bar{x}^2}$$

At Jervis Inlet, each site was assigned 15 to 25 randomly positioned transects which were surveyed prior to fishing (Table 1). The same transect positions were used every year that a site was surveyed. Because a noticeable fishery impact was not expected for some years, especially at the low exploitation rate sites, not all sites were surveyed every year (Table 2).

Transect lines, made of lead core rope marked at 5 m intervals, were laid perpendicular to the shoreline by survey vessels, from shore to a sounder depth of 18.3 m (60 feet). Survey depth was not adjusted for tidal height. A buoyed line was attached to the deep end of each transect. A pair of scuba divers descended on the float line to the deep end of each transect, and took positions on either side of the lead-line. Divers counted all sea cucumbers between the lead-line and the far edge of a 2 m aluminum bar on both sides along the transect in each 5 m section (quadrat). Divers looked under kelp and boulders within this strip transect, but rocks were not moved to look for cucumbers out of view. Each transect was therefore 4 m wide and left and right halves of each quadrat were each 10 m^2. At the end of each quadrat, divers recorded the number of sea cucumbers counted, substrate type, depth, the two most dominant algae types, and total percent cover of algae. Counts from the left and right side of each quadrat were summed.

Table 2. Detailed survey schedule for the Jervis Inlet sea cucumber EFA.

Year	Project year	Sites surveyed	Sites harvested
1999	1	0%, 2%, 4%, 8%, 16%	All
2000	2	None	All
2001	3	0%, 8%, 16%	All
2002	4	None	All
2003	5	0%, 2%, 4%, 8%, 16%	All
2004	6	None	All
2005	7	0%, 8%, 16%	All
2006	8	None	All
2007	9	0%, 2%, 4%, 8%, 16%	All
2008	10	None	All

Data analysis

Sea cucumber densities (*d*), by transect, were calculated as the sum of the number of cucumbers counted on all quadrats on that transect divided by 4 m, the transect width.

$$d = \frac{\sum_{i=1}^{n} C_i}{4}$$

where c_i is the number of sea cucumbers counted on quadrat *i*, and *n* is the number of quadrats surveyed on that transect. Thus, density was expressed in terms of the number of sea cucumbers per meter of shoreline. Mean sea cucumber density and 95% CIs for a site were then calculated from the transect densities. The Tukey multiple comparison test was used to compare mean density between years for each site.

Samples of 50 sea cucumbers were collected from each of two randomly chosen transects per site in order to estimate mean split weight. Samples were taken every year during the harvest from the same transects. Sampled cucumbers were split longitudinally and weighed individually (split weight) to the nearest gram. Data from both sample transects at each site were pooled and mean weight and 95% CIs were calculated. Significance tests for differences in mean weight were performed using Tukey's multiple comparison test between years for each site.

Weights of the experimentally harvested animals were also used to obtain mean split weight estimates by dividing the total landed weight for a site by the numbers of cucumbers harvested from that site. Confidence intervals are therefore not available for the mean weights estimated from harvested sea cucumbers. Linear regression coefficients of mean weight from harvest versus year were calculated for each site and tested for significance using ANOVA.

Densities for each transect were multiplied by the mean split weight of sea cucumbers estimated from experimental harvest data (sample transects for 0% site) to estimate the biomass (kg) per meter of shoreline. Mean sea cucumber biomass and 95% CIs for a site were then calculated from the transect biomass. The Tukey multiple comparison test was used to compare mean biomass between years for each site.

Experimental harvest

Experimental fishery quotas were calculated from the 1999 survey data (Table 1). The mean density at each site was multiplied by the shoreline length (10,000 m) to determine the total sea cucumber population at each site. The uncertainty in the mean density estimate was not considered when setting quotas for the EFAs. Each site was randomly assigned one of the five exploitation rates and the quota (in numbers of cucumbers) for each site was then calculated as a percentage of the total sea cucumber population on that site. The quotas were based on cucumber numbers, as opposed to weight, for ease of catch monitoring during the experimental fisheries. Quotas were kept constant across years.

Each harvest vessel had four crew: one biologist diver, one boat tender, and two sea cucumber divers. The divers hand picked sea cucumbers and placed them into mesh bags. The start and end times to fill for each bag were noted. The sea cucumbers were brought on board, split longitudinally, and enumerated by bag. Cucumbers were then packed in cages lined with plastic bags before delivery to the dock where the total weight of the catch was validated. The product was later transported to a plant for processing, and sold.

CPUE was calculated for both number of cucumbers and weight (kg) landed per dive-minute. The number of cucumbers landed (piece-counts) was available for each harvest bag during the experimental fisheries. For the piece-count CPUE, a per-bag CPUE was calculated from which the mean CPUE and 95% CIs were calculated. Significance tests were performed using Tukey's multiple comparison test between years for each site. For the weight-based CPUE, mean CPUE was calculated as the total landings for a site divided by total effort on that site. Therefore, confidence bounds could not be calculated for the weight-based CPUE. Linear regression coefficients of weight-based CPUE versus year were calculated for each site and tested for significance using ANOVA.

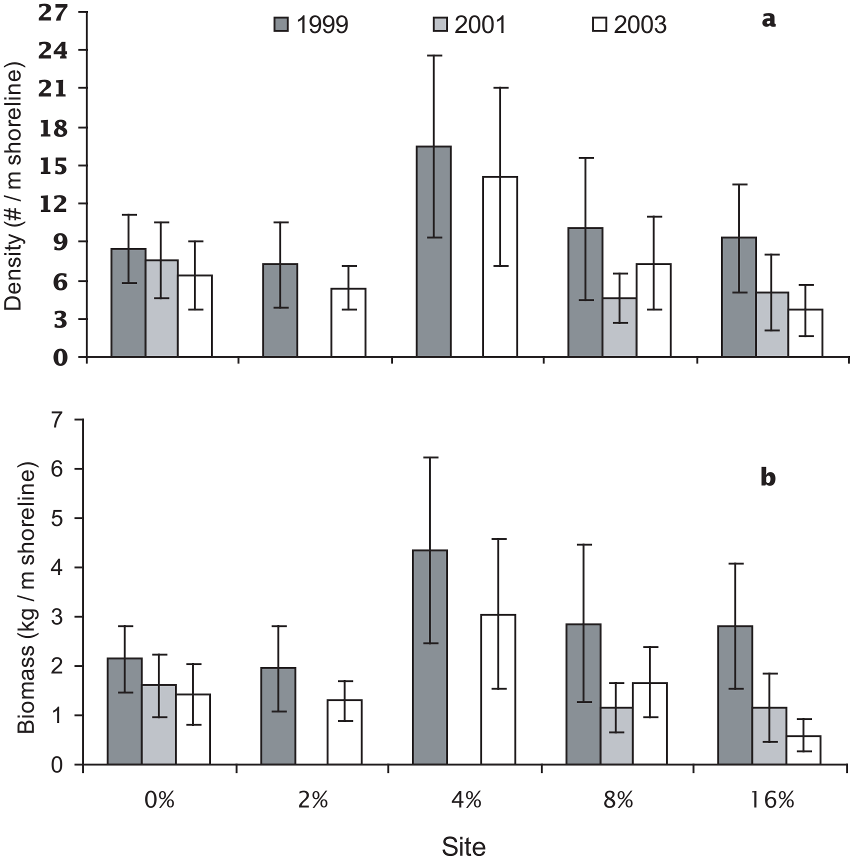

Figure 4. (a) Density of sea cucumbers (number per m of shoreline) and (b) biomass of sea cucumbers (kg per m of shoreline), by site and year, for the Jervis Inlet EFA. Error bars are 95% confidence intervals.

Results

The Jervis Inlet EFA project began in 1999, and the 2003 survey results reflect changes that have occurred after four years of harvesting. Density at the 16% site decreased significantly between 1999 and 2003 (df = 45, $P = 0.029$) from 9.3 to 3.6 per m (61% decrease). Other sites showed no significant changes in density between years, although a general decreasing trend was observed (Fig. 4a). Biomass at the 16% site decreased significantly between 1999 and 2003 (df = 45, $P = 0.001$) by 79%, from 2.8 kg per m to 0.6 kg per m. Biomass at the other sites showed no significant change, although a decreasing trend similar to that seen in the density data was observed (Fig. 4b).

Mean weight data from the sampled transects showed a significant decrease between 1999 and 2003 at the 2% and 16% sites (Fig. 5a; df = 526, $P = 0.008$ and df = 562, $P = 0.038$ respectively). Other sites showed no significant difference in mean weight between 1999 and 2003. The 4% and 16% sites showed a drop in mean weights after the first or second year of harvest followed by a relatively stable mean weight. The regression coefficient of mean weight from experimental harvest vs. year at the 16% site (–29.1) was significantly different from zero (ANOVA, $F = 19.8$, $P = 0.02$). Regression coefficients calculated for other sites were not significantly different from 0; however, a decreasing trend was observed (Fig. 5b). Because the experimental quotas were expressed in numbers of animals, the decrease in mean weight over time translated into a 48% decrease in total landings for the 16% site, from 4,530 kg in 1999 to 2,350 kg in 2003. Landings fluctuated less at the other sites.

Each site had been harvested four times before the 2003 survey. The total harvest from each site was therefore 8%, 16%, 32%, and 64% of the initial population for the 2%, 4%, 8%, and 16% exploitation rate sites, respectively. The amount of shoreline harvested to achieve the quota at each site was dependent on the exploitation rate, such that, at low exploitation rate sites (2% and 4%), only a small portion of the 10 km of shoreline needed to be harvested to reach the quotas. In the low exploitation rate sites, different sections of shoreline were fished in different years. At the 8% and 16% exploitation rate sites, however, approximately 60% and 90% of the 10 km of shoreline had to be harvested each year, respectively, to achieve the quota.

Piece-count CPUE was variable from year to year (Fig. 6a). There were no clear trends except for an apparent increase in 2003 at all sites which was significant at the 8% and 16% sites (df = 214, $P = 0.002$ and df = 360, $P < 0.001$ respectively).

The regression coefficient of weight-based CPUE vs. year at the 16% site (–0.278) was significantly different from zero (ANOVA, $F = 20.5$, $P = 0.02$). Regression coefficients for the other sites were not significantly different from zero, although a decreasing trend was present (Fig. 6b). Perhaps more important, the 2%, 4%, and 8% sites showed a drop in CPUE between 1999 and 2000 followed by a somewhat more stable CPUE.

Discussion

EFA

Although the EFA projects are still in early stages, results from Jervis Inlet to date reveal a significant decrease in densities and biomass at the highest exploitation rate site (16%). The implications of this decrease, however, must be considered in light of the apparent decrease in density

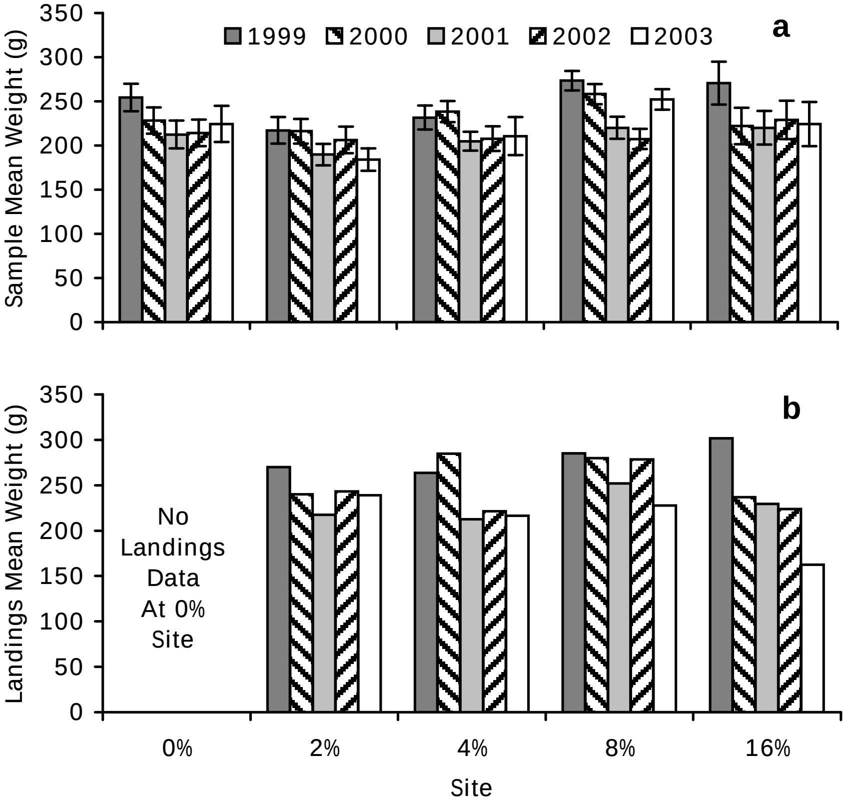

Figure 5. **(a) Mean split weight of sea cucumbers (g) from sample data (data from both sampled transects at each site are pooled), and (b) mean split weight of sea cucumbers (g) from landings of experimental harvest, by site and year, for the Jervis Inlet EFA. Error bars are 95% confidence intervals. Confidence intervals are not available for panel "b," as mean split weight from landings was calculated as total weight of experimental catch divided by total number of sea cucumbers harvested for each site and year.**

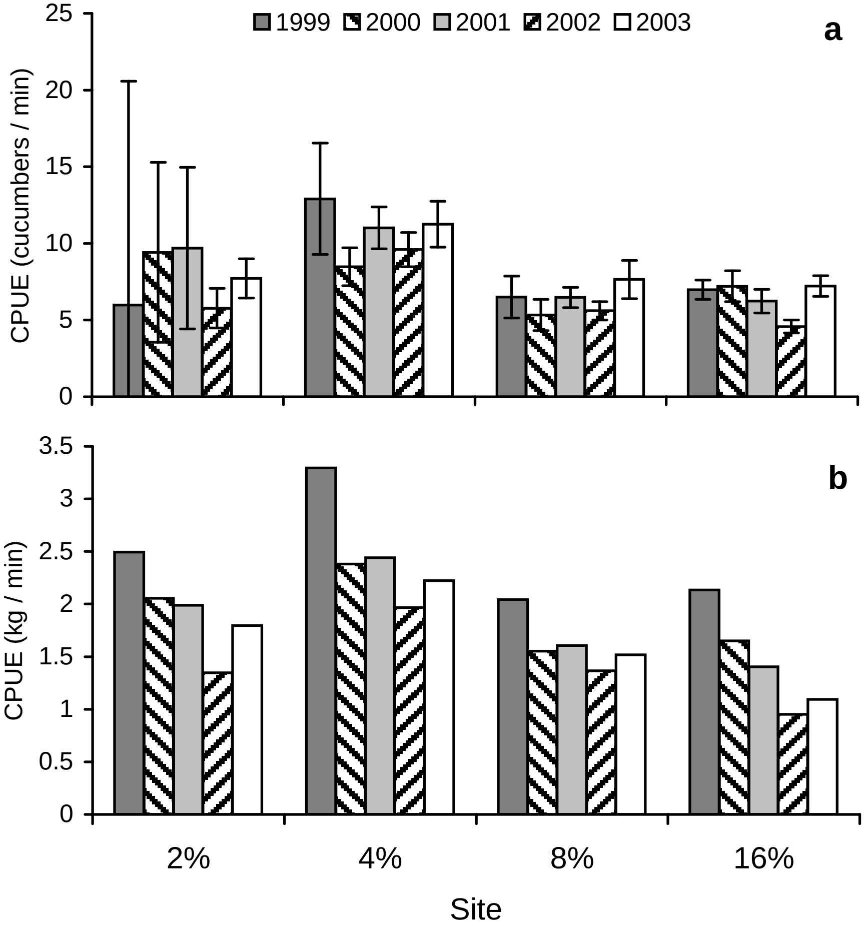

Figure 6. Catch per unit effort in: (a) numbers of sea cucumbers harvested per dive-minute, and (b) kg of sea cucumbers harvested per dive-minute, by site and year, for the Jervis Inlet EFA. Error bars are 95% confidence intervals. Confidence intervals are not available for panel "b," as CPUE was calculated as total weight of experimental catch divided by total effort for each site and year.

and biomass in the control and low exploitation rate sites. The data are quite variable and results are limited by small sample size. Completion of the study in another five years will improve the statistical power of the experiment.

Mean weight from samples decreased slightly over time (significantly at 2 of 5 sites). Mean weight from landings showed a larger decrease over time, although the decrease was significant at only one site. Because the mean weights from landings are based on larger sample sizes over a larger area than the sample transects, the mean weights calculated from landings may be better indicators of overall mean weight for a site. The pattern, observed at some sites, of decrease in mean weight after the first year of harvest, followed by a somewhat stable mean weight in later years, may be due to selective fishing for larger sea cucumbers in the first year of harvest (Ken Ridgway, Pacific Sea Cucumber Harvesters Association, pers. comm.). The apparent decrease in mean landed weight is likely a result of progressive selection and depletion of the largest sea cucumbers over time.

Anecdotal information from harvesters suggests that sea cucumber density has changed little in commercially exploited populations, but that the mean weight has decreased in some “open areas.” Preliminary results from re-surveys conducted in open areas in BC support this observation (unpubl. data). Woodby and Larson (1998) also made a similar observation for sea cucumber populations commercially exploited in Alaska.

Piece-count CPUE data were variable, but stayed relatively constant over time, despite a general decreasing trend in densities, suggesting that piece-count CPUE may not be a very good measure of sea cucumber density. At the lower exploitation rate sites, harvesters can move fishing locations from year to year (within each 10 km site), thus maintaining a high CPUE and potentially causing serial depletion. Also, catch rates may be limited by handling time and densities may still be high enough to not be limiting. Although harvest methods remained the same between years, other factors such as currents, underwater visibility, weather, or diver experience may have affected CPUE. However, CPUE based on weight landed showed a decreasing trend for all four harvest sites at Jervis Inlet, following the trend of decreasing biomass. The decrease in mean weight of sea cucumbers over time is most likely the cause of the decreasing weight-based CPUE. Selective fishing for large sea cucumbers in 1999 may explain the high weight-based CPUE for that year. Weight-based CPUE therefore seems to be more useful to monitor changes in biomass of sea cucumber populations than piece-count CPUE.

Because the quotas for EFAs were set in terms of numbers rather than weight of cucumbers, the effective exploitation rate in terms of percent of the original biomass harvested each year has decreased over time due to the decrease in mean sea cucumber weight. The commercial sea cucumber fishery quotas are expressed in terms of biomass. Therefore, if mean sea cucumber weight is decreasing over time in open areas, the effective exploitation rate will likely increase if quotas are based on outdated density and mean weight data. The frequent re-survey schedule (every four years) currently in place for open areas in BC should help ensure that effective exploitation rates do not increase. Mean weights from fishery data should be updated frequently, so that quotas can adjusted accordingly.

Phased approach

The phase 1 research program for sea cucumbers has answered some of the uncertainties associated with quota calculations in BC. Increased knowledge of local abundance and distribution has led to the use of BC baseline density estimates and to increased quotas. Results from the EFAs should eventually provide estimates of a biologically sustainable exploitation rate. Until such time, several conservation measures are in place to protect sea cucumber stocks in BC. First, only 25% of the coast is open to the commercial fishery. Second, the lower 90% CI of estimated densities are used in quota calculations for open areas, similar to what is done in Alaska (Woodby et al. 1993). Third, quotas are based on density surveys conducted to 18 m depth only; the portion of the sea cucumber population found at depths greater than 18 m is not accounted for in the assessments. Commercial divers generally harvest shallower than 18 m so the segment of the sea cucumber population found deeper than 18 m is generally protected from harvest. Woodby et al. (2000) found that about 30% of sea cucumbers were deeper than 20 m in a commercially viable area near Sitka, Alaska. Similar data are unavailable from BC.

The potential for the deeper portion of sea cucumber populations to migrate shallower and re-colonize harvested areas is unknown and should be investigated. If the deep portion of the population moves shallower to occupy habitat previously used by harvested cucumbers, declines in density may not be observed until the deep portion of the population becomes depleted. Further, CPUE may stay high until such a time that re-colonization from deep areas slows. The lag in response caused by the potential upward migration of the deep portion of the population may therefore have serious implications for the stock assessment and management of sea cucumber fisheries. If such a phenomenon occurs, an exploitation rate that appears sustainable in the short term may prove to be unsustainable in the long term. Boutillier et al. (1998) suggested using ROVs (remotely operated vehicles) to survey density and biomass of sea cucumbers at depths greater than 20 m. Establishing new EFAs that include both shallow water scuba surveys and deep-water ROV surveys to study potential post-harvest vertical migration of sea cucumbers would be valuable. An acceptable alternative could be to conduct deepwater ROV surveys in the current EFAs.

A strong cooperative relationship among scientists, managers, and stakeholders is essential to the success of a phased approach to providing scientific advice for the management of newly developing invertebrate fisheries (Perry et al. 1999). The BC sea cucumber research program relies heavily on a strong working partnership among industry, DFO (science and management), and First Nations groups, where research costs are financed by the sale of product harvested in EFAs and surveys are conducted on chartered industry and First Nations vessels.

The sea cucumber fishery in BC still faces data limitations. Little work has been published on growth and age of *Parastichopus californicus* (Fankboner and Cameron 1988, Cameron and Fankboner 1989). Maximum age of sea cucumbers has been used to determine the mortality rate of sea cucumbers, based on Hoenig's (1983) relationship between maximum age and mortality rate, which in turn has been used in determining exploitation rates (Woodby et al. 1993, Phillips and Boutillier 1998). Growth and longevity studies thus may be warranted to confirm or revise the estimates of natural mortality rates and reduce uncertainty in the assessment of the sea cucumber resource. Recruitment patterns of sea cucumbers are poorly understood and should be studied. Completion of the EFA projects, with an additional five years of data, will increase the statistical power of the experiments and help to estimate a sound harvest rate for sea cucumbers in BC. Future research plans in BC include the establishment of new experiments to evaluate the performance of rotational fisheries, compared to annual fisheries, as a way to mitigate the observed decreasing trend in mean weight data.

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Biological Considerations for Management of Data-Limited Surfperch (Family Embiotocidae) Populations in British Columbia

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Abstract

Eight species of Embiotocidae occur in British Columbia (BC): four are distributed coastwide (pile perch, shiner perch, striped seaperch, and kelp perch) and four (redtail surfperch, walleye surfperch, silver surfperch, and white seaperch) have distributions restricted to the extreme southern part of the province. Commercial fisheries target only the coastwide species but because the ranges of nontarget species overlap those of commercial ones, especially along the southern coast of Vancouver Island, there is potential for conservation concerns for nontarget species. Embiotocids are brooding, live-bearers that mature slowly and produce few young. As a result, populations may be vulnerable to overfishing even at moderate exploitation levels, with recovery limited by low fecundity and possibly small population sizes. The limits of the available data, however, make evaluation of these conservation concerns difficult. With the exception of kelp perch, which are restricted to kelp forest habitats, embiotocids are found around wharves and pilings where they are targeted by an unmonitored recreational fishery. Shiner and pile perch inhabit these and other habitats, including deeper waters where they are taken as bycatch in trawl fisheries. The current commercial fishery occurs nearshore using drag seine or hook and line gear and landings usually are small and intermittent. There is no documentation or monitoring of bycatch of nontarget species in the commercial embiotocid fishery. Although management of embiotocid fisheries in BC is hindered by data limitations, consideration of basic biological information on these species could be useful for improving management advice.

Introduction

The Canadian Department of Fisheries and Oceans (DFO) is implementing two new national policies that affect minor and nontarget species. The New and Emerging Fisheries Policy applies to new or developing fisheries (DFO 2001). This policy is based on the "phased approach" outlined by Perry et al. (1999) that provides a science-based framework for the assessment and potential development of fisheries for underutilized or data-limited species. The second is new legislation, the Species at Risk Act (SARA), implemented in 2003, which provides a framework for classifying species (or distinct populations) according to potential risk of extinction. In British Columbia (BC), there are eight species of embiotocids, all of which are affected by one or both policies. Researchers have been asked to provide information on the status of species within the family Embiotocidae (for SARA) and provide fisheries managers with information on the potential suitability of some populations for exploitation (new and emerging fisheries). As little work has been done on any of the eight species found in BC, providing defensible, science-based advice to resource managers is difficult in the existing data-limited situation.

Globally, the family Embiotocidae (surfperches) consists of 23 species, including one freshwater species. Most are restricted to the eastern Pacific Ocean with the number of species increasing from north to south. Of the eight surfperch species in BC, some have been fished both commercially and recreationally since the 1800s (Hart 1973, Eschmeyer et al. 1983). Four species are distributed coastwide: pile perch (*Rhacochilus vacca*), shiner perch (*Cymatogaster aggregata*), striped seaperch (*Embiotoca lateralis*), and kelp perch (*Brachyistius frenatus*) (Fig. 1). Four additional species are restricted to southern Vancouver Island (their northern range limit): redtail surfperch (*Amphistichus rhodoterus*), walleye surfperch (*Hyperprosopon argenteum*), silver surfperch (*Hyperprosopon ellipticum*), and white seaperch (*Phanerodon furcatus*) (Fig. 1). In general, embiotocids are widespread, inshore species occupying a variety of habitats including sandy beaches, rocky shorelines, reefs, kelp beds, and estuaries (e.g., Tarp 1952, Feder et al. 1974, Hobson and Chess 1986, Lamb and Edgell 1986). Embiotocids have specialized feeding niches so competition is limited within nearshore environments (e.g., DeMartini 1969) and

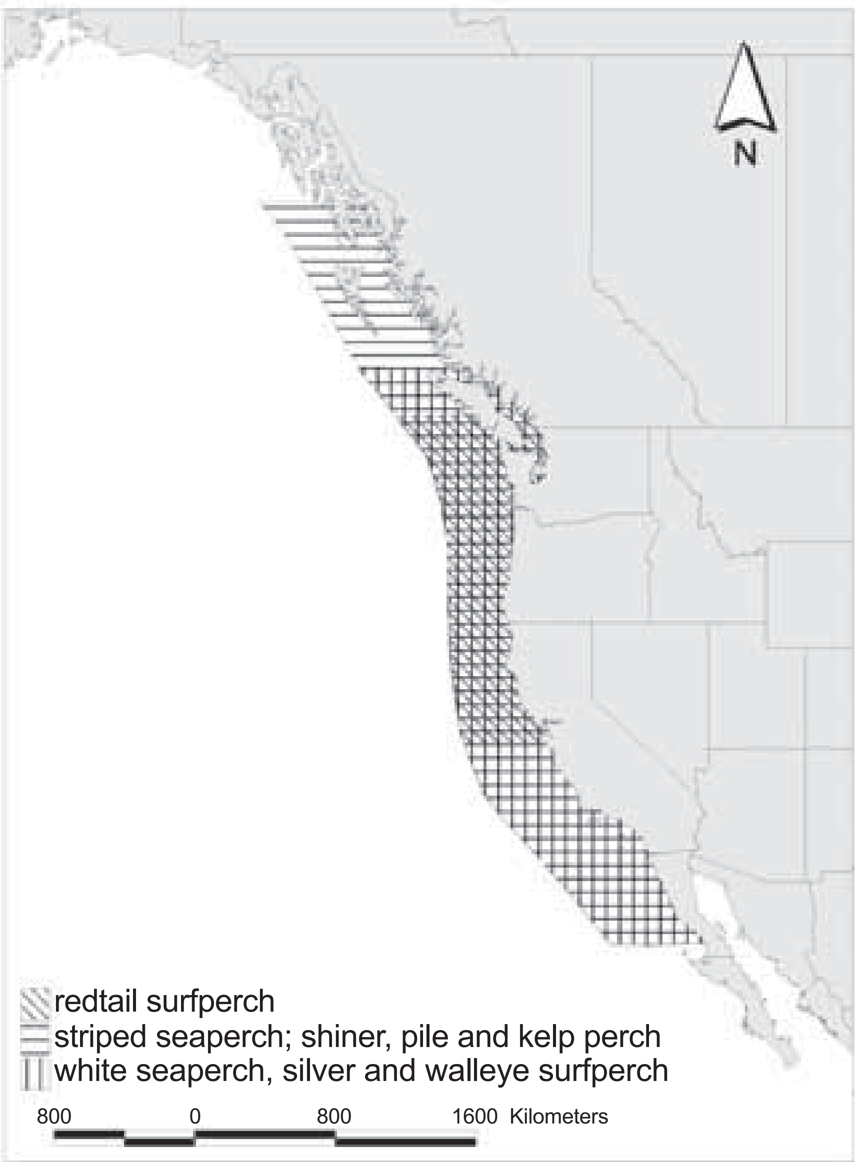

Figure 1. Reported distributions for the eight embiotocids encountered in British Columbia.

Table 1. Basic life history characteristics of British Columbia embiotocids.

Common name	Maximum age (years)	Age at maturity (years)	Average fecundity[a]	Average gestation (months)
Shiner perch	6	1	4-20	5
Pile perch	10	4	18-52	5
Striped seaperch	7	2-3	18-22	6
White seaperch	7	2	10-29	7
Redtail surfperch	8	4	7-45	5
Walleye surfperch	6	1	5-12	4
Silver surfperch	7	2	3-16	7
Kelp perch	N/A	1	N/A	N/A

[a] For all embiotocids fecundity increases with female size and age.

Data from Gordon 1965, Swedberg 1966, Gnose 1967, Wilson and Millemann 1969, Anderson and Bryan 1970, Miller and Lea 1972, Eckmayer 1975, Bennett and Wydoski 1977, Baltz 1984.

overlapping ranges among species is common. Water temperature is an important factor affecting distributions (Karpov et al. 1995).

Embiotocids are viviparous; females give birth to highly developed, free-swimming young (Turner 1947). Fecundity is low relative to nonviviparous fish species and varies by species and age, with older (larger) individuals producing more young (Cannon 1956; Table 1). Due to extremely low fecundity, maintenance of populations is likely dependent upon a high survival of the young (Quinnel 1986). Most species live six to ten years (Table 1) and growth rates are low. Females may attain greater sizes than males, but considerable variability can exist among populations (e.g., Hubbs and Hubbs 1954, Anderson and Bryan 1970, DeMartini et al. 1983). Recruitment to the parental population can occur at birth for some embiotocids (Schmitt and Holbrook 1990) and might result in increased vulnerability to fishing.

Directed embiotocid fisheries

In BC, embiotocids are landed both in recreational and commercial fisheries. Despite a long history of fishing activities, few quantitative data are available for either fishery but anecdotal information suggests the recreational catch is larger than the commercial catch. Recreational fishing for embiotocids is a common activity in BC, especially on wharves, docks, and piers. Recreational fishing is regulated under the Fisheries Act through the British Columbia Sport Fishing Regulations. The DFO Tidal Waters Sports Fishing Guide summarizes these regulations, which consider all embiotocids as "surfperch" with a combined daily harvest limit of eight fish using hook and line gear (DFO 2003). Harvesters are required to possess a valid tidal waters recreational fishing license when fishing for embiotocids. There are no recreational fishing closures or specified area closures specific to embiotocids. Creel surveys are conducted to estimate the recreational catch for some species, but embiotocids have not been included in survey efforts to date.

A specialized commercial fishery also exists for embiotocids in BC. Catches are taken mainly by beach seine, but hook-and-line and dip-net gear types are permitted. Licenses to commercially harvest embiotocids are available to fishermen who possess a vessel-based commercial harvesting license for at least one species. The use of beach seines has raised concerns about bycatch of nontarget species, especially juvenile salmonids, and negative impacts on nearshore habitats through destruction of macrophytes (i.e., eelgrass beds). Presently, only a small commercial fishery exists with few active fishers in any given year; therefore, potential habitat and bycatch impacts are considered negligible under current effort levels. However, this fishery has potential for considerable expansion under the current, nonrestrictive licensing system. A logbook requirement was established in 1984 as a condition of license but compliance has been poor (based on the number of logbooks received annually) and enforcement efforts have been minimal. Available logbook information indicates that the reported catch has fluctuated dramatically since the early 1980s with substantial landings in 1984 and 1992 and low landings in most other years, especially since 1997 (Fig. 2). In general, CPUE has tracked commercial landings so trends in the commercial catch and effort data are non-informative as a basis for interpreting trends in population abundance. Shifting market demand is most likely responsible for the observed trends in reported catch but a change in the number of resource users or incomplete catch reporting also can influence reported catch data. Sales are confined to specific, limited markets (mainly live or fresh fish markets in Vancouver) again suggesting market demand is the primary factor determining embiotocid catches.

Nondirected embiotocid fisheries: occurrence as bycatch

In addition to directed recreational and commercial fisheries, embiotocids are caught in several other fisheries. For example, shiner perch are routinely caught as bycatch in the shrimp trawl fishery (Hay et al. 1999). During the 1997-1998 season, shiner perch was the 16th most common species captured in shrimp trawls and accounted for 0.35% of the total catch. Unfortunately, little biological data exist because all bycatch in this fishery is discarded at sea and observer coverage is minimal. Also, catch rates varied considerably by area with shiner perch making up 0.20% of

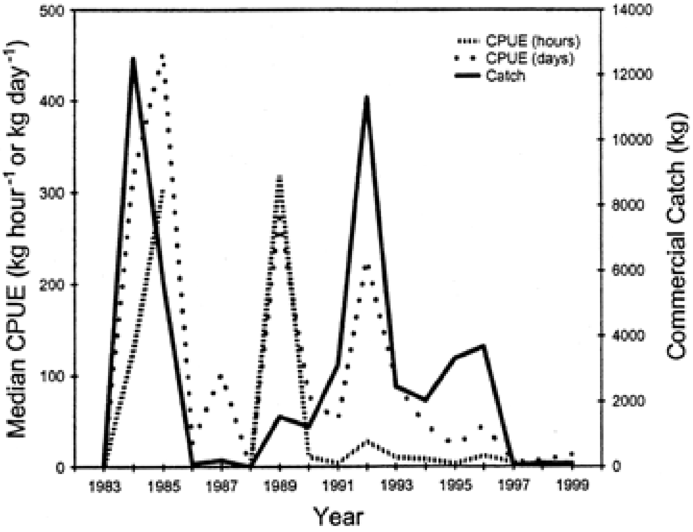

Figure 2. Reported commercial catch (kg) and CPUE (kg per hour or kg per day) for embiotocids landed in British Columbia since 1984.

the catch on the west coast of Vancouver Island, 0.58% in Prince Rupert district, and 2.65% of the catch in the Strait of Georgia (8th most common species). When the percent composition is applied to the total shrimp landed in each of these areas, shiner perch bycatch in the 1997-1998 shrimp trawl fishery was estimated to be 0.4 t on the west coast of Vancouver Island, 2.5 t in Prince Rupert, and 13.5 t in the Strait of Georgia. These estimated bycatch removals are considerably greater than the current commercial fishery or the expected recreational harvest. In contrast to the high levels of bycatch observed for shiner perch, pile perch only made up 0.0025% of the total shrimp trawl bycatch (Hay et al. 1999) with other embiotocids landed even less frequently. Even at this low level, the landed bycatch weight of pile perch (1.3 t) would exceed current commercial or recreational harvests. Thus, based on these calculations, it is probable that nondirected removals are greater than directed ones.

Discussion

Of the eight embiotocid species in BC, four species (shiner, pile, and kelp perch, and striped seaperch) occur throughout coastal BC waters, while four species (redtail, walleye, and silver surfperch, and white seaperch) have ranges restricted to southern BC. Because there is very little attention directed to embiotocids in BC, our understanding of their distributions is limited and ranges might be broader than currently recognized. For example, the redtail surfperch is known from the Tofino region of BC (west coast of Vancouver Island) where it is the focus of an intensive recreational fishery but it is unknown if their distribution is contiguous with populations in Washington state.

Four species have geographically restricted distributions in BC, limited to southern Vancouver Island. As this marks the northern extent of their ranges, it is possible that distinct (isolated) populations exist in BC. Under SARA, these populations could be considered for "listing." One key criterion used in the species listing process is the species' range within Canada. A species could be deemed threatened if its range is less than 20,000 km^2 or endangered if its range is less than 5,000 km^2 (COSEWIC 2003; Criterion B). Given that the depth distributions of these four species are restricted to 100 m of water (or less), the estimated maximum area of occurrence would be about 4,200 km^2 in the Canadian zone along the west coast of Vancouver Island, well below the minimum area of occupancy needed to be considered for listing. Further, as water temperature is a key factor determining the distribution of embiotocids (Karpov et al. 1995), the estimated area of occurrence could be reduced in years of cold water ocean regimes. Alternatively, under warm water regimes ranges could expand. We acknowledge, however, that the range area is only one of the criteria used to determine an actual listing but because we do not know population sizes or population trends, it could be an important factor. It is important to note that each of these four embiotocids has less restricted distributions in the United States. This raises the possibility that Canadian populations could be maintained from larger populations to the south but this hypothesis would need to be validated.

There have been no genetic studies to measure gene flow among embiotocid populations in BC but their viviparity and restricted geographical distributions suggest potential divergence from larger, ancestral populations. Species with restricted dispersal capabilities or limited larval times should have high speciation rates (e.g., Futuyma 1998). However, this is not the case within the family Embiotocidae and further study is needed to discern ecological or evolutionary factors responsible for speciation within this family (Bernardi and Bucciarelli 1999). A geographical barrier to dispersal of black surfperch (*Embiotoca jacksoni*) in California resulted in two major species clades within which gene flow was high but among which gene flow was low (Bernardi 2000). It is unknown what potential barriers are present in BC that might affect dispersal (and subsequent gene flow) of juveniles or adults or whether gene flow along the West Coast of North America is sufficient to prevent the establishment of distinct populations.

Embiotocids are actively pursued in both recreational and commercial fisheries in the United States, most notably California (Miller and Gotshall

1965, CDFG 2002), but also in Oregon and Washington (Quinnel 1986, ODFW 2002, WDFW 2002). In each of these states, recreational landings are substantially greater than commercial ones, even in California where the largest commercial embiotocid fisheries exist (Karpov et al. 1995). Although recreational catches in BC are unknown, the commercial fishery is so small that the recreational landings are probably larger. We anticipate that the New and Emerging Fisheries Policy will limit large-scale expansion of the commercial embiotocid fishery but this has yet to be tested. One of the goals of this policy is to only permit fisheries that are internally sustainable (i.e., the users are financially responsible for the stock assessment required to determine harvest levels). Recently, commercial landings have been small suggesting markets for this product also are small, but this fishery could grow if commercial markets expand.

Royce (1975) warned that species with low fecundity are vulnerable to overharvest; intense fishing pressure combined with low fecundity reduces the probability of strong year classes from developing. Therefore some embiotocids in BC could be prone to overfishing. Overfishing of embiotocids already may have occurred in California, where recreational and commercial catches have declined steadily since the 1960s (Fritzsche and Collier 2001). Because four embiotocid species in BC have very restricted distributions, local overfishing (or depletion) could result in loss of genetic diversity, raising concerns under SARA. Further, local depletions might act differently on different components of the population. For example, densities of sub-adults of both black and pile surfperch at two locations in California decreased while densities of adults remained constant over a 25 year period (Pondella et al. 2002). It is possible that increased recreational fishing activity has resulted in the decreased density of sub-adult fish and such a phenomenon could be universal.

In summary, the commercial harvesting of embiotocids in BC is subject to the New and Emerging Fisheries Policy as a data-limited fishery. Currently, adequate data to make sound assessments and catch recommendations are unavailable. Such data limitations confound attempts to adhere to a precautionary approach to fisheries management (e.g., FAO 1995) for these species in BC. The restricted distributions of some species in BC allow them to be considered under the new SARA legislation but with limited data it is not possible to infer trends in stock abundance or biomass, a fact vital both to SARA and fisheries managers who need to set harvest recommendations.

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Refining Management for Prince William Sound Sablefish

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Abstract

Sablefish is one of the most economically important resources in waters of the Gulf of Alaska. Prior to 1984, annual sablefish harvests from waters of state jurisdiction in Prince William Sound (PWS), Alaska, were less than 10 metric tons (t). As market demand increased, this fishery developed with a 12-month season but few management guidelines. More restrictive management measures were implemented as competition within the state-managed fishery increased due to improved product value and access limitation in adjacent federal waters. Initial measures included development of a harvest guideline based on a yield-per-recruit model using production data extrapolated from sablefish fisheries in Southeast Alaska. Fishery management continued to develop through state access limitation and into a quota share system wherein permit holders are allocated shares of the harvest guideline. Shares are equal within each of four vessel size classes, but differ among size classes. Due to the lack of stock assessment data, these management changes have been used as fishing effort controls to achieve a static annual harvest level. Recently, the Alaska Department of Fish and Game implemented a fishery-independent survey as an index to monitor changes in the PWS sablefish population. Future efforts will explore the utility of the survey to amend the harvest guideline. The development of the PWS sablefish fishery provides a case study of the evolution of data-limited management options for the harvest of a common property resource by a local, shore-based fleet.

Introduction

Commercial fisheries for sablefish *Anoplopoma fimbria* off Alaska began in the late 1800s (Bracken 1983, Rigby 1984, Sigler et al. 2002). Initially, landings occurred as incidental bycatch to the Pacific halibut *Hippoglos-*

sus stenolepis fishery. Directed sablefish fisheries developed around 1913. Annual harvests fluctuated widely in response to economic conditions, including market demand for alternative fish products such as halibut. Harvests also varied with the depression era of 1931-1935, cold storage capacity, and regulatory restrictions implemented during the 1940s-1960s in response to declines in catch rates and mean fish size. Until 1957, the fishery primarily involved U.S. and Canadian vessels. Japanese longline fleets entered the fishery in 1958, followed by fleets from Korea, Russia, and other countries (Low et al. 1976, McDevitt 1986). The longline fleets initially focused on the Bering Sea, but shifted to the Aleutian Islands and Gulf of Alaska in the mid-1960s as the Bering Sea fishing grounds were preempted by foreign trawl vessels. The Magnuson Fishery Conservation and Management Act in 1976 extended U.S. jurisdiction to 200 miles from shore and encouraged domestication of the groundfish fleet within those waters. Longline fishing by U.S. vessels expanded into the Gulf of Alaska in 1982, and full domestic utilization of sablefish resources in the Gulf of Alaska of the Exclusive Economic Zone (EEZ; formerly referred to as the Fishery Conservation Zone) occurred in 1984. The initial management strategy emphasized a “derby fishery” in which vessels competed to maximize harvest in as short a time as possible. As the “derby” fishery expanded, average fishing season length decreased from year-round in the early 1980s to less than one month in the early 1990s (Sigler and Lunsford 2001). Individual Fishing Quotas (IFQ) and an 8-month season were implemented in 1995 for the federal sablefish and halibut fisheries in EEZ waters off Alaska.

The Prince William Sound Management Area (PWS) is located along the northern Gulf of Alaska and adjacent to the EEZ and is managed under state regulations by the Alaska Department of Fish and Game (ADFG; Fig. 1). Prior to the late 1980s, groundfish fisheries in PWS occurred throughout the year at relatively low annual harvest levels (Bechtol and Morrison 1997). This fishery occurred in the absence of stock abundance data, and at a time that directed fishing was viewed as the primary means by which to evaluate resource availability. Biological and economic declines in salmon and crab fisheries, and subsequent diversification into groundfish by much of the fishing industry, increased effort in the groundfish fisheries and raised concerns of managers about sustainable yields from the groundfish resources. Improvements in both fishing technology and market conditions exacerbated these concerns. Fishery catch and effort increased from 50 metric tons (t) by 20 vessels in 1984 to 257 t by 126 vessels in 1995 (Table 1). The 1995 peak in PWS catch and effort was attributed to speculation about qualifying for a state limited entry permit. In 1996, a limited entry program was implemented for PWS sablefish to address the increasing effort that had begun to compromise management (Berceli et al. 2002). Following implementation of the limited entry program, fishery catch and effort through 2002 averaged 120 t and 47

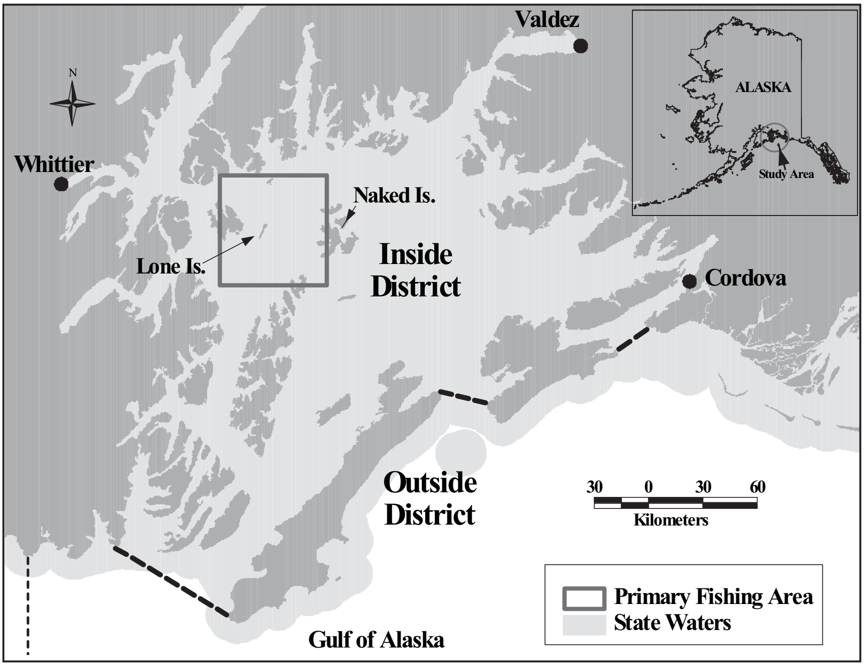

Figure 1. Inside District of the Prince William Sound Management Area.

vessels. In 2003, the PWS fishery changed to a quota share system in an effort to reduce hook loss and unretained bycatch.

Few biological data were collected during development of the PWS sablefish fishery. In addition, although PWS does not represent a closed population, mark-recapture efforts have been insufficient to draw definitive conclusions regarding movement of tagged fish between PWS and other management jurisdictions (unpubl. data). Thus, the ability to define stock structure has been limited, necessitating development of data limited management strategies. Based on historical knowledge of harvest practices, it has been appropriate to manage the PWS sablefish population independent of other harvest jurisdictions. This paper describes the evolution of management strategies, in the absence of stock assessment data, for PWS sablefish as the fishery changed from a year-round, open-access fishery to a limited entry, quota share system, with an extended fishing season featuring spring and fall fishing periods.

Table 1. Annual sablefish harvest from Prince William Sound, 1984-2003.

		Fishing effort			Harvest		Ex-vessel value	
Year	Season[a] (days)	Vessels	Permits	Landings	(t)[b]	%GHL	$US per lb	$US
1984	365.0	20	20	37	50	NA	0.46	$50,563
1985	323.0	28	34	108	174	NA	0.60	$229,974
1986	81.0	32	33	36	86	96	0.63	$119,606
1987	85.0	59	65	120	93	104	0.64	$131,424
1988	111.0	52	52	145	100	111	0.99	$217,222
1989	202.0	25	29	98	85	95	0.89	$167,357
1990	128.0	69	73	253	96	107	0.69	$145,925
1991	38.0	72	73	159	148	165	0.91	$296,874
1992	17.0	53	52	126	196	218	0.93	$401,920
1993	4.0	55	59	99	144	131	0.94	$297,606
1994	4.0	54	56	104	127	116	1.03	$289,121
1995	2.0	112	118	142	257	234	2.22	$1,255,514
1996	1.0	64	66	65	112	102	1.88	$465,385
1997	1.5	50	50	77	89	81	2.31	$451,307
1998	1.5	58	58	58	106	96	1.72	$400,703
1999	1.8	39	38	41	93	85	1.79	$368,846
2000	2.0	29	30	29	155	141	2.26	$773,666
2001	1.3	45	46	45	141	128	1.87	$579,737
2002	1.1	47	47	48	145	133	1.90	$609,288
2003	60.0	37	49	63	97	88	2.31	$494,447
Mean	71.5	50	52	93	125	124	1.35	$387,324

[a] Excludes closed periods during initial opening and final season closure.

[b] Round weight.

[c] Guideling harvest level.

Study area

Regulatory boundaries of PWS have been adjusted several times over the past three decades and are currently described as waters of Alaska from 144°00.00 W, near Cape Suckling, to the longitude of Cape Fairfield at 148°50.25 W. This area is divided into the Inside District and Outside District. Regulations restrict the PWS sablefish fishery to the Inside District, which is about 9,500 km^2 in surface area with a maximum depth of approximately 775 m (Fig. 1). Waters deeper than 275 m comprise more than 25% of the district and produce virtually all of the sablefish harvest (Bechtol and Morrison 1997).

Materials and methods

Waters managed under the jurisdiction of the State of Alaska extend from shore to three miles off shore. Regulations for state waters are adopted by the Alaska Board of Fisheries and implemented through in-season management by ADFG. The board considers proposals for regulatory changes on a three-year cycle. In addition to actions defined in codified regulations, “time-and-area” emergency order authority allows ADFG managers to open and close fisheries on an area-specific basis. Harvest data are derived from landing reports known as ADFG fish tickets, which are entered and archived in the ADFG Fish Ticket database. Among other information, fish ticket data include harvest amount by species, permit holder, vessel, port of landing, and processor.

The Commercial Fisheries Entry Commission (CFEC) implements limited entry programs in Alaska fisheries and may complement a program with additional fishing permit restrictions such as vessel size or gear limits. However, authority for resource allocation is reserved for the Alaska Board of Fisheries, which may allocate resources among user groups, but may not differentially allocate resources to individuals within a specific user group. Thus, allocations may differ among gear types or vessel sizes, but not among individual permit holders that have different harvest histories but identical user group criteria.

Results

Determination of the guideline harvest range

Prior to 1984, annual sablefish harvests from PWS totaled less than 10 t annually (Bechtol and Morrison 1997). Fishing effort increased on sablefish resources in state waters in response to improved markets and changes in EEZ sablefish fisheries such as increased competition and decreased season length. As a result, sablefish harvests in PWS expanded rapidly to 50 t in 1984 and 175 t in 1985. These increased harvests generated stock sustainability concerns because assessment data for the Inside District were lacking, particularly at the depths and areas typified by the fishery (Parks and Zenger 1979). In addition, fishery performance data were limited to commercial fish tickets and to voluntary logbooks that represented less than 10% of the 1985 harvest and were of uncertain accuracy.

Due to the lack of comprehensive stock data, Morrison (1987) adapted a yield-per-habitat model developed by S. Lowe (NMFS AFSC, Seattle, pers. comm.) to establish a guideline harvest range (GHR) for PWS sablefish. This process entailed estimation of suitable sablefish habitat using a grid overlay of nautical charts, and assumed depth as the primary selection criteria. The sablefish production rate was calculated as two standard deviations around mean areal sablefish harvest rates (annual catch per unit area) in the Clarence Strait fishery in Southeast Alaska from 1969 to 1979 (Bracken 1983). This provided an estimated annual yield ranging from 0.06 to 0.25 t per square nautical mile. Application of this productivity range to the area of suitable PWS habitat resulted in a potential production of 40-140 t (values are rounded), the GHR that was used for PWS sablefish management from 1986 to 1992. Due to improved bathymetric mapping, the area deemed suitable sablefish habitat was increased by 26% in 1993 and the GHR increased proportionately to 44-175 t (Bechtol and Morrison 1997). Management has typically targeted the midpoint of the GHR, or 93 t before 1993 and 110 t since 1993, as the guideline harvest level (GHL) or quota (Bechtol 1995, Berceli et al. 2002).

Although managers attempted to maintain a relatively fixed annual harvest for PWS sablefish, there has been little additional analysis of the suitability of this GHR in terms of stock productivity. Fishery performance data have been limited primarily to ADFG fish ticket harvest records. A refinement of harvest strategies for PWS sablefish requires better stock assessment information, particularly data on size and age composition, distribution, and productivity. In particular, fisheries-independent survey data were needed.

An annual survey was initiated in 1996 to determine the relative abundance and composition of groundfish caught on longline gear in PWS sablefish habitat (Bechtol and Vansant 1997, Berceli et al. 2002). The primary survey objective was to develop a fishery-independent index of sablefish abundance to monitor changes in the sablefish resource. Additional survey objectives were collection of biological data and determination of appropriate bycatch limits for non-target species in the PWS sablefish fishery. Preliminary survey analysis indicates sablefish catch rates increased from 1996 to 2001, declined in 2002, and then increased in 2003 (Table 2). Survey catch rates compared favorably to fishery catches in the subsequent spring for most years (Fig. 2). Although survey and fishery catches are not directly comparable, fishery duration is determined preseason based on anticipated effort and catch rates. Additional analysis is needed to learn if greater catches under a static annual harvest level are indicative of relative increases in stock abundance.

Table 2. Unstratified survey catch abundance (fish per set) in northwestern Prince William Sound, 1996-2003.

Year	Sets	Sablefish	Pacific cod	Pollock	Halibut
1996	31	53.3	7.7	4.2	27.1
1997	20	62.8	3.8	4.7	24.9
1998	26	56.7	6.9	5.4	18.7
1999	21	75.2	4.0	3.1	21.1
2000	24	85.8	0.8	1.4	8.5
2001	23	90.5	2.0	1.0	10.0
2002	27	45.5	9.7	1.3	12.1
2003	31	63.5	4.4	5.6	10.7

Year	Sets	Arrowtooth flounder	Rockfish	Skates	Sharks	Other
1996	31	2.3	3.6	14.5	2.0	0.3
1997	20	1.1	3.3	9.0	2.1	0.9
1998	26	2.2	2.9	13.8	28.8	0.2
1999	21	2.1	1.9	5.4	6.0	0.1
2000	24	0.7	2.7	8.4	3.2	0.0
2001	23	1.9	3.7	9.8	5.4	0.4
2002	27	2.4	2.4	6.7	3.3	0.4
2003	31	3.3	2.8	10.9	15.9	0.5

Evolving management measures

A requirement that participants obtain a sablefish interim use card from the CFEC was the primary management measure in place when the PWS sablefish fishery developed in the 1980s. Additional measures adopted by the Alaska Board of Fisheries in 1986 included

- Fishery participants must possess a Prince William Sound sablefish permit, also called a commissioner's permit, issued through local ADFG offices; and
- The fishery opened concurrently with the sablefish fishery in the adjacent EEZ and closed by emergency order when the PWS GHL was attained.

In addition to serving as an area registration, the PWS sablefish permit can be an important fishery management and data gathering tool because it allows ADFG to place additional management criteria, such as logbooks, department observers, or catch reporting requirements upon fishery

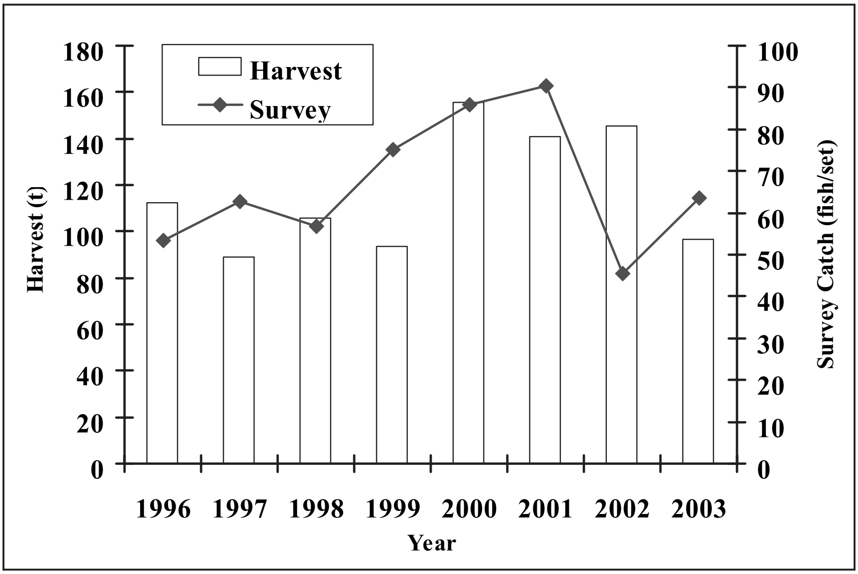

Figure 2. Fishery harvest and ADFG survey index for Prince William Sound sablefish, 1996-2003.

participants. The season opening measure was intended to limit potential fishing effort in PWS by having vessel operators elect to fish in either PWS or adjacent federal waters rather than serially fishing both areas. ADFG annually established the duration of the fishing period based on the GHL, the projected number of participants, and past fishery performance. Similar to the pattern in adjacent federal waters, PWS fishing seasons became shorter as effort and aggregate catch of the fleet increased (Table 1). Total fishery duration decreased from year-round in the early 1980s to seasons of 24 to 96 hours during the early 2000s. As season duration decreased, the PWS sablefish fishery became less manageable in an open access format and ADFG petitioned the CFEC to consider limiting entry. A limited entry program was initiated for the PWS sablefish fishery in 1996 using 1991 to 1994 as the qualifying years (Muse et al. 1995). The program established a target of 49 permanent permits, divided into four vessel size classes (≤11 m, 12-15 m, 16-18 m, and 19-27 m) and two gear classes, fixed (longline and pot) and trawl. The process of awarding permanent PWS sablefish permits is ongoing with 61 eligible permit holders in 2003 (Table 3).

Table 3. Number of eligible and registered permits, with permit quota share, for the Prince William Sound sablefish fishery, 2003.

	Eligible permits				Permit quota (t)	
Vessel size class (m)	Permanent	Interim-use	Total	2003 registered	Round	Dressed
A = 19-27	1	1	2	1	4.43	2.79
B = 16-18	2	1	3	2	4.43	2.79
C = 12-15	33	8	41	40	2.00	1.26
D ≤ 11	9	6	15	10	1.65	1.04
Total	45	16	61			

Although up to 20% of the PWS total harvest occurred in the Outside District in some years, the limited entry program considered landings only from the Inside District and the fishery has been restricted to this district since 1996 (Berceli et al. 2002). Most Inside District fishing effort has concentrated in a deepwater trench between Lone Island and the Naked Island group (Fig. 1).

Following adoption of the limited entry program, fishing effort intensified and fishery duration was reduced to 24 hours to avoid exceeding the GHL. Fishing periods were also adjusted to provide for daylight openings and closures, with longline fishing restricted immediately before and after the sablefish fishery to facilitate enforcement (Berceli et al. 2002). Alaska Department of Public Safety and ADFG monitored the fishery on the grounds via vessel boardings to verify regulatory compliance. Additional management measures included (1) sablefish must be landed within 24 hours of the season closure; and (2) sablefish may be retained only by PWS sablefish permit holders during the directed fishery opening.

Increased competition perpetuated the negative aspects of a derby fishery, including greater fishing effort, gear loss, and gear conflicts. Gear conflicts, typically related to tangled longlines and vessel crowding, resulted in lost gear when lines were parted. Another source of “lost gear” was vessels setting more gear than could be retrieved in the duration of the fishing period, often resulting in discarded gear. These problems were exacerbated when the average number of hooks fished per vessel-hour, where hour is defined as season duration, increased from 436 in 1998 to 668 in 2001, with some smaller vessels tendering additional gear to the grounds (Fig. 3).

The type and amount of gear set and lost typically varied with vessel size. For example, smaller vessels were more likely to use snap-on gear in which hooks are manually “snapped” onto the groundline when set-

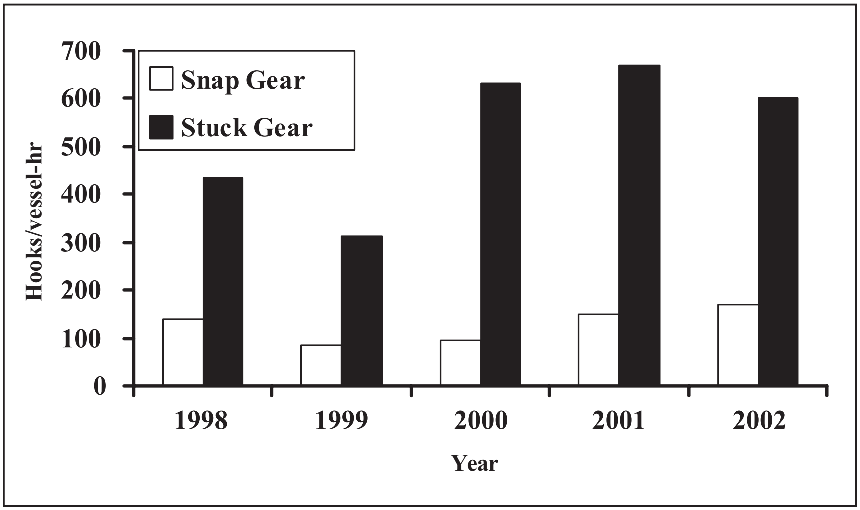

Figure 3. Estimated fishing effort by gear type, based on logbook data and vessel registrations, for the Prince William Sound sablefish fishery, 1998-2002.

ting gear. Alternatively, larger vessels were more likely to use stuck gear that has hooks on gangions, short lines that are permanently attached to the groundline at specified intervals. The greatest difference between these gear types was the amount of hooks that could be set and retrieved, particularly during a short duration fishery (Fig. 3). Minimum estimates of gear lost across the fleet ranged from 6,570 hooks in 1999 to 29,440 in 2001 (Fig. 4). Mortality attributable to lost gear is unknown, but was likely high.

A quota share program was implemented in 2003 to facilitate a more orderly fishery and address the conservation issues associated with the derby fishery. Under this program, half of the GHL is allocated equally among all registered participants, and the balance of the GHL is allocated according to the permit's vessel size class using the following percentages: Classes A and B (27 and 18 m maximum length) vessels = 18.53%; Class C (15 m maximum length) vessels = 70.33%; and Class D (11 m maximum length) vessels = 11.14%. These percentages were derived from the average harvest by vessel size class during 1999 to 2001. A preseason registration deadline allows ADFG staff to assess potential effort and establish quotas based on the number of participants registered for each vessel class.

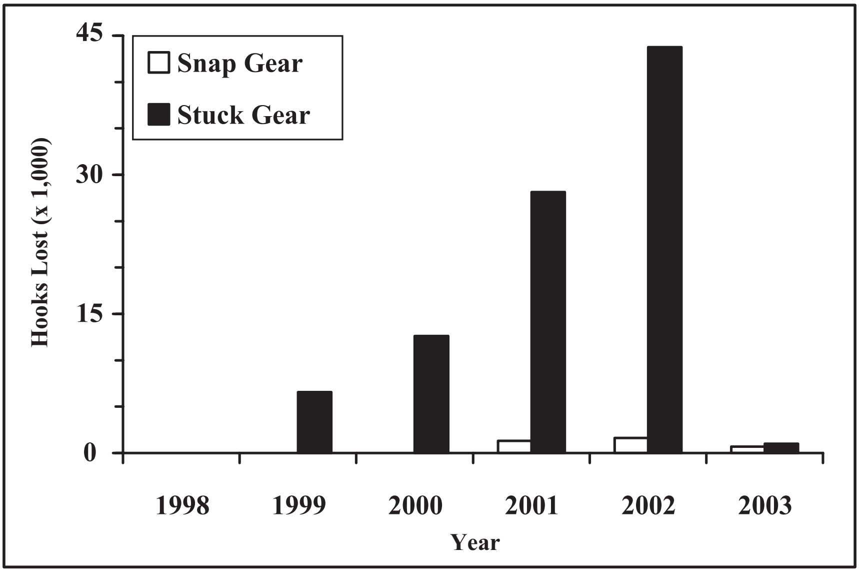

Figure 4. Logbook estimates of lost hooks by gear type for the Prince William Sound sablefish fishery, 1998-2003.

To encourage a less intensive fishery and improve economic efficiency and opportunity for fishery participants, new regulations in 2003 also established an extended, split fishing season of March 15 to May 15 and August 1 to August 21. Additional regulatory changes included

- Within 24 hours prior to the opening of the PWS sablefish season, sablefish may not be possessed in the Inside District on a vessel registered to participate in the PWS sablefish fishery.
- A vessel operator shall unload all sablefish taken in PWS before taking sablefish in another area.
- Sablefish may not be taken in PWS if sablefish from another area are aboard.
- Copies of all completed PWS sablefish fish tickets, issued to a sablefish permit holder for the current season, must be retained on board.
- PWS sablefish logbooks must be completed and returned to ADFG within five days of the closure of the sablefish season.

- Fishermen must provide at least six hours prior notice of landing and report the estimated pounds on board, estimated time of arrival, and the port of landing.

A total of 53 permits were registered for the 2003 fishery with quota shares ranging from 1.6 to 4.4 t (round weight) for individual permits based on vessel size class (Table 2). The 2003 harvest totaled 96 t (Table 1) with approximately 75% of the harvest occurring in the spring season. Compliance with the new management elements was generally good; quota overages and failures to provide prior notice of landing were the most common issues. Quota overages were sold with proceeds accruing to the State of Alaska.

Discussion

The shared quota program appears to have addressed the primary problems of gear conflicts that resulted in lost gear and unaccounted mortality, exceeding the target harvest limit, and retention of bycatch. An estimated total of 1,800 hooks was reported lost during the 2003 fishery, a dramatic reduction from the 2002 level of more than 45,000 hooks (Fig. 4). The harvest remained within the target GHL of 110 t. The prior notice of landing requirement facilitated opportunities to collect biological data and addressed most concerns that a longer season would reduce sampling opportunities. Increasing the prior notice of landing from six to twelve hours will improve the ability for biological sampling of landings in unstaffed ports.

One result of the change from the derby style to a less intensive, more extended fishery has been improved retention of non-sablefish species. In the traditional sablefish fishery, the vessel operator tried to maximize sablefish catch during the relatively short season. With the shift to an extended season, the number of vessels involved in the fishery has actually declined as permit holders, each with their own quota share, have pooled their fishing efforts aboard a single vessel. Increased retention of halibut bycatch from 20 t in 2002 to 57 t in 2003 indicates the slower paced fishery promoted the retention of halibut, thus improving full retention for vessel operators with halibut IFQ permit holders aboard (Table 4). This ultimately reduces competition and fishing effort because there is greater retention of the species that are caught, decreased discards, and less discard mortality.

Finally, the longline survey data will likely be incorporated into population modeling used to modify the GHR for PWS sablefish. The existing GHR, in the absence of additional data or analyses, has provided both sustained yield and relative stability for the industry since being implemented in the mid-1980s (Table 1). To some extent, observed variability in annual harvests results from changes in relative population abundance.

Table 4. Catch landings by Prince William Sound sablefish vessels, 1996-2003.

Year	Number of vessels	Number of landings	Sablefish (t)	Pacific cod (t)	Rockfish (t)	Halibut (t)
1996	64	65	112.3	0.5	9.2	N/A
1997	50	77	88.6	0.2	7.9	N/A
1998	58	58	105.7	0.4	8.7	N/A
1999	39	41	93.5	<0.1	3.1	N/A
2000	29	29	155.3	<0.1	6.6	0.3
2001	45	45	140.6	<0.1	6.9	11.3
2002	47	48	145.5	<0.1	7.1	20.2
2003	37	63	97.0	0.3	4.6	56.8

Management for a static harvest level has provided for sustained annual yield in this data-limited situation, but at a cost of foregone harvest opportunity in years of greater cohort abundance. PWS sablefish stocks may be able to sustain greater fishing mortality without a loss in population productivity during years of greater cohort abundance.

Any amendment to the GHR will need to consider migration of sablefish stocks both into and from PWS. Although some transboundary movement is indicated, current tag return data are insufficient to draw firm conclusions.

Although reports of killer whale *Orcinus orca* interactions increased substantially in 2003 (unpubl. data), the effect of different fishery management strategies on marine mammals is poorly understood. Killer whales have been documented to selectively forage on sablefish being retrieved on longline gear (Dalheim 1988). During the 1987 PWS fishery, killer whales were estimated to have consumed an additional 10% to 25% of the reported sablefish harvest off longline gear being retrieved (Matkin 1988, Schroeder and Morrison 1988). However, as PWS sablefish seasons became shorter, fewer whale interactions were reported. Additional research is needed to determine the effects of returning to a less intensive, longer-term fishery on killer whale interactions.

Rockfish (*Sebastes*) as a species group are susceptible to overfishing because they are relatively long-lived, slow growing, and late maturing. Captured fish, particularly those in the slope rockfish assemblage, tend to suffer 100% mortality due to embolism. Slope rockfish are commonly caught on longline gear in PWS waters deeper than 200 meters (Bechtol 2000). The effect on bycatch levels resulting from a shift to a less intensive, yet longer-term sablefish fishery cannot be determined at this time.

In particular, the effect of hook saturation on different fishing strategies needs to be examined.

Management strategies for the PWS sablefish fishery have been constrained by low-resolution harvest data, lack of biological data, and little information on migration or stock structure. These data limitations have resulted in a conservative approach using a static annual harvest level as the fishery has changed from year-round, open-access to a limited entry, quota system, with an extended fishing season featuring spring and fall fishing periods. Although we have identified some of the benefits and shortcomings of these developments, particularly managing for a static annual harvest level, this incremental management progression has provided sustained yield from the resource.

Acknowledgment

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Fisheries Assessment and Management in Data-Limited Situations
Alaska Sea Grant College Program • AK-SG-05-02, 2005

Indicators and "Response" Points for Management of Fraser River Eulachon: Protocols for Managing a Data-Limited Fishery

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Abstract

Future management of eulachon fisheries (*Thaleichthys pacificus*) in the Fraser River requires a management plan based on objective criteria. Ideally such criteria would be based on biological indicators and for each indicator there may be an explicit "reference" point that triggers a management decision. Usually reference points are determined from an understanding of population dynamics of a species. The main indicator for Fraser River eulachon is the spawning stock biomass (SSB) estimated annually since 1995 from egg and larval surveys. With such a short time series, we could not define explicit reference points based on population dynamic models. Instead, for the SSB and other indicators, we defined several "response" points. These response points are based on our judgment about the reliability of indicators that we think are precautionary and biologically realistic. We present these response points in the context of a "traffic light" scenario where, for example, an SSB of less than 150 t, for 2 consecutive years would represent a red light and fishing would not occur. A second indicator is an offshore eulachon biomass index estimated during annual shrimp trawl surveys in May. A biomass estimate of less than 1,000 t in offshore waters is a response point (yellow light) for concern about Fraser River eulachon fisheries. Another indicator is the assessment of the spawning run and catch data from the Columbia River fishery which occurs about 4 months prior to the fishery in the Fraser. A poor run or low catch in the Columbia may anticipate low Fraser River catches and could be a response point (yellow light) for the Fraser River. The last indicator is test-fishery data collected since 1995; but the reliability of these data remains uncertain—comparison of test-fishery catches with the SSB is promising but not convincing. However, these data provide a reassurance about run strength that may be useful for in-season management decisions. The paper concludes with a discussion of how to reconcile conflicting indicators in the context of the traffic light scenario.

Introduction

The terms “indicators” and “reference points” have specific meaning in the context of fisheries management. Within Fisheries and Oceans Canada, these terms refer to the implementation of Objective Based Fisheries Management Plans (OBFM) and Integrated Fishery Management Plans (IFMP). The requirement for the development of such plans for eulachons (*Thaleichthys pacificus*) led to the preparation of this report where we apply the term “indicator” to data or observations related to the state of population abundance of eulachons. Indicators may consist of a range of values, usually a time series of data. Within this range we identify specific points that provide a basis for a fisheries management decision, such as the opening of a fishery. Available data on Fraser River eulachon abundance are limited, however, so we cannot estimate specific points based on population dynamics or statistical models. Instead we provide different points, called “response points,” that are based on simple biological criteria. While such response points may lack statistical rigor, they are based both on our best understanding of the biology and history of eulachon fisheries in the Fraser River. Therefore the response points are precautionary and biologically realistic.

We compare several independent indicators for as many years as possible. These indicators include (1) over 60 years of annual commercial catch data for the Fraser and Columbia river eulachon fisheries from 1941 to 2002; (2) estimates of Fraser River spawning stock biomass (SSB) from egg and larval surveys, made annually since 1995; (3) indices of eulachon abundance in offshore waters based on incidental capture of eulachons in annual shrimp surveys conducted since 1973; and (4) data from an experimental test-fishery conducted since 1995 in the Fraser River.

The report begins with a brief description of eulachon life history and eulachon fisheries in the Fraser River and concludes with a review and commentary on the efficacy of the indicators. We present a simple “stoplight” scenario to distinguish and reconcile conflicting indicators and suggest recommendations for management of Fraser River eulachons.

Biology of eulachons and Fraser River fisheries

The biology of eulachons was reviewed by Hay and McCarter (2000); therefore we present only a brief overview here. Eulachons are smelts (Os-

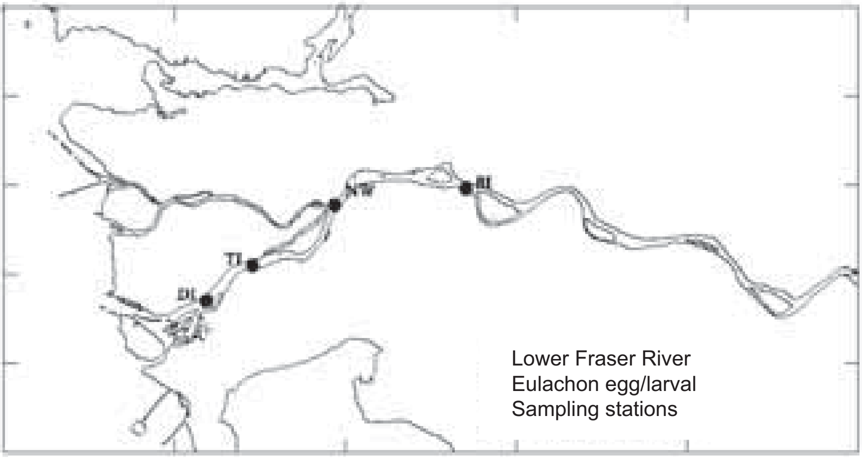


Figure 1. The key sampling locations examined throughout all years (1995-2002) for estimates of spawning stock biomass (SSB) from egg and larval surveys. The abbreviations refer to sampling stations as follows: Deas Island (DI), Barnston Island (BI), New Westminster (NW), Tilbury Island (TI). The distance between sampling stations is approximately 10 nautical miles. The test-fishery was conducted in the vicinity of New Westminster.

meridae) that occur from the southern Bering Sea to northern California. Eulachons spawn during the late winter and spring. The earliest spawning occurs in the Columbia River (the largest run in the world) in January and February, and the latest in the Fraser (perhaps the second largest) in April and May. Most populations in northern rivers in B.C. spawn in March and April. Fraser River eulachons are semelparous with most living for three years (Hay and McCarter 2000). Probably some spawn at age 2 and others at age 4 or 5.

A small commercial fishery for eulachons has occurred in the Fraser River since the early 1900s. The only other regularly occurring commercial fishery on the Pacific Coast is in the Columbia River. Eulachon stocks declined sharply in the Fraser River in 1994. Concerns about the apparently low spawning biomass led to research investigations to estimate spawning biomass and spawning locations and the introduction of regulations.

Figure 2. Temporal variation in spawning stock biomass (SSB) for the Fraser River, 1995-2002 (adapted from Hay et al. 2002).

Methods

Indicators and data

We examined all available data that were directly related or potentially related to eulachon abundance in the Fraser River (Fig. 1). To determine the efficacy of data sets as indicators, and to develop response points, we examined the temporal variation in each indicator, and when possible compared temporal trends among different indicators.

1. *Spawning stock biomass (SSB)* is estimated as the product of mean egg and larval (e + l) density (n per m^3) and river discharge (m^3 per s); methodological details are presented in Hay et al. (1997b, 2002). SSB is estimated as the biomass required to produce the observed e + l density. SSB estimates are made about one month after spawning, as eggs hatch, and continue for about an 8-week duration. The SSB estimate, as an after-the-fact estimation of abundance, has no direct application as an indicator to management of the fishery in the same year in which the SSB was made. The SSB estimates began in 1995. Spawning locations within the river change interannually, but in general SSB estimates are lowest in the most upstream locations, above most spawning locations, and greatest in the most downstream locations. In this paper we show the total SSB for the entire Fraser River as the sum of the production for the North Arm and South Arm (Fig. 2).

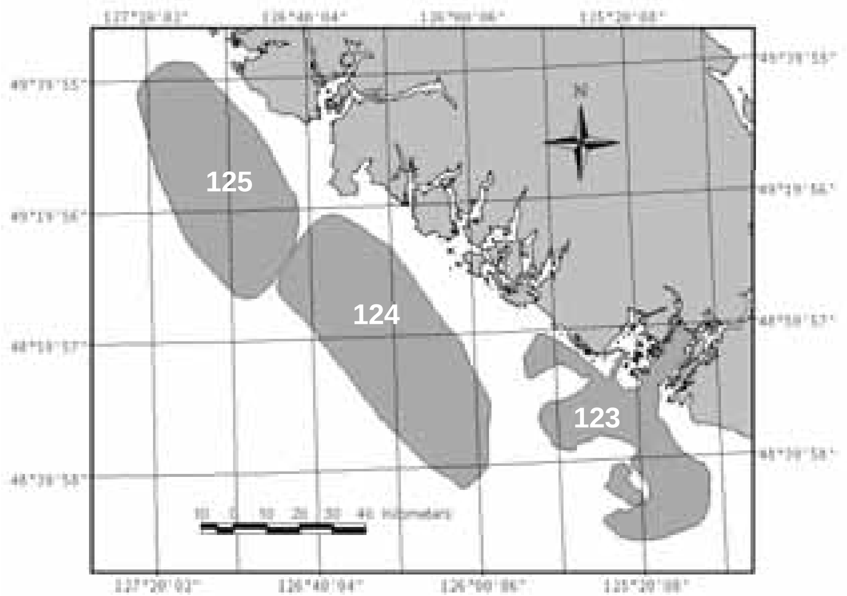


Figure 3. Offshore sample locations (gray polygons) examined during the annual shrimp trawl survey conducted in May of most years. The numbers within the polygons correspond to the statistical areas referred to in the text.

2. *An annual offshore biomass index* was estimated from analysis of bycatch in annual shrimp trawl research surveys conducted off the west coast of Vancouver Island since 1973 (Boutillier et al. 1997, Rutherford 2002). The shrimp trawl survey design is based on systematic sampling of the shrimp grounds with spatial analysis used to provide estimates of shrimp abundance (Hay et al. 1997a). Eulachon are caught as bycatch in this survey and the method used to estimate shrimp abundance also has been used to provide an index of eulachon abundance in the surveyed area. The detailed survey methodology for assessing shrimp stocks is documented in Boutillier et al. (1998) and Martell et al. (2000). Methods of estimating the eulachon biomass index from the survey are described in Hay et al. (1997a). The areas of concentration used for biomass indices are shown for several areas off the west coast of Vancouver Island: statistical areas 124 and 125 the combined areas of 121 and 123 (Fig. 3). The latter area is not used as part of the time series index, because it has only been surveyed for a few of the last 30 years.

The index is not completely represented for both areas 124 and 125 in all years, so these are treated as two separate indices (i.e., not summed or pooled). Prior to 1999 the total catch weight of eulachon taken during the research surveys was the only biological parameter recorded for eulachon. Beginning in 1999 eulachon were sampled for length and a subsample of the catch was counted and weighed to estimate an average number per kilogram. Then age composition was estimated from length frequency analysis of data (Fig. 4) from all the tows within a management area (Rutherford 2002).

3. *Fraser and Columbia River catch data* from commercial fisheries have been collected since the 1930s (Fig. 5). Columbia River catch data (ODFW/WDFW 1993) probably are roughly representative of trends in abundance, at least until the 1990s when catch restrictions were imposed (Hay and McCarter 2000). In the Fraser River, however, trends in catch data (Ricker et al. 1954, and DFO unpubl. catch data) may vary widely from trends in abundance, probably for most of the last four decades. Instead catch data probably reflects many factors other than abundance, including limited markets, incomplete reporting of catch, and changing spawning areas (Hay et al. 2002).

4. *Fraser River test-fishery data as a potential indicator*. Systematic catches in the New Westminster test-fishery have been conducted in the Fraser River since 1995, except 1999. All catches were made daily using identical gillnet gear (mesh size 3.18 cm or 1.25 inches, 50 fathoms or 92 m long and 380 meshes deep, fished at the same location [New Westminster], for the same duration [15 minutes], and at the same stage of tide [low slack at New Westminster]). All catch was sorted by sex, counted, and weighed, and a biological sample was collected for further analysis. Catch numbers were tallied each week.

Results

Temporal trends in indicators

SSB

A plot of the total Fraser River SSB shows substantial variation between 1995 and 2002 (Fig. 2), with lowest years between 1997, 1998, and 2000. The lowest year was 1997, with an SSB estimate of less than 80 t. As an indicator, a year with a low SSB (i.e., <150 t) should not necessarily be cause for fisheries management action, such as a closure or a catch restriction, in the next year. In most years, there are at least three eulachon cohorts present in the population. Eulachons are subject to considerable interannual fluctuations, so an SSB estimate <150 t would be cause for management caution, not alarm. Using the example of a traffic light, where

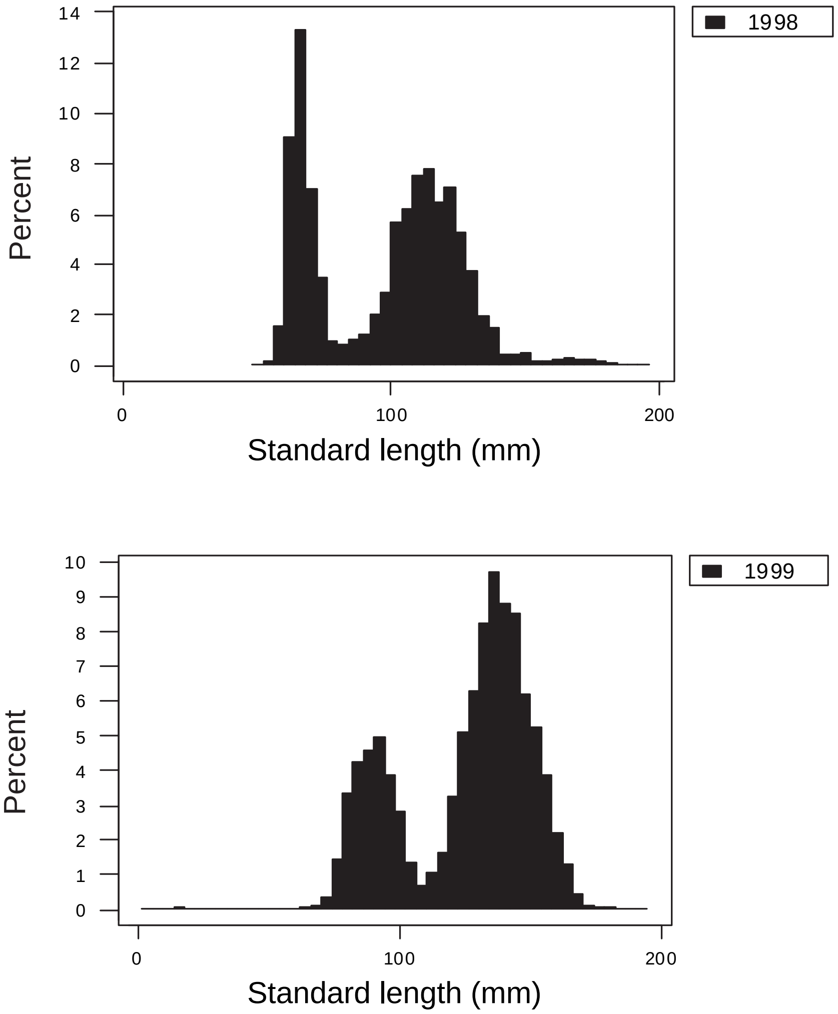
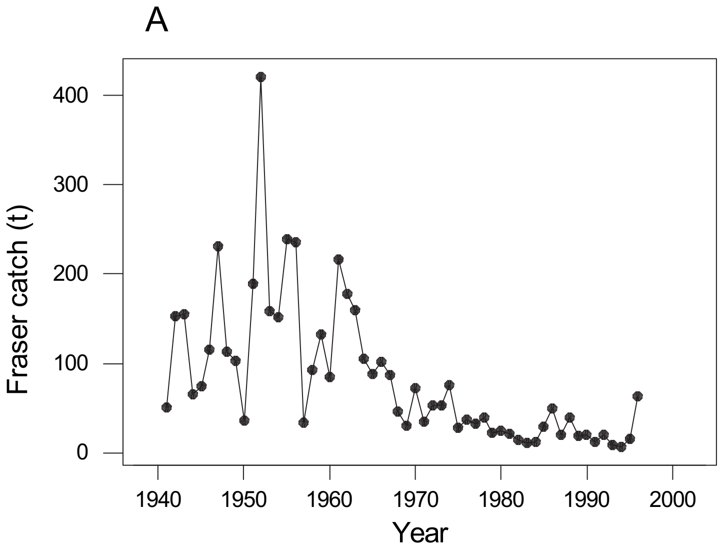


Figure 4. **Size modes of offshore eulachons, corresponding to age groups 1+ (about 15 months of age) and 2+ (about 27 months of age) from size composition data collected from eulachons in 1997 and 1998 during annual shrimp surveys in May, off the lower west coast of Vancouver Island (adapted from Hay and McCarter 2000).**



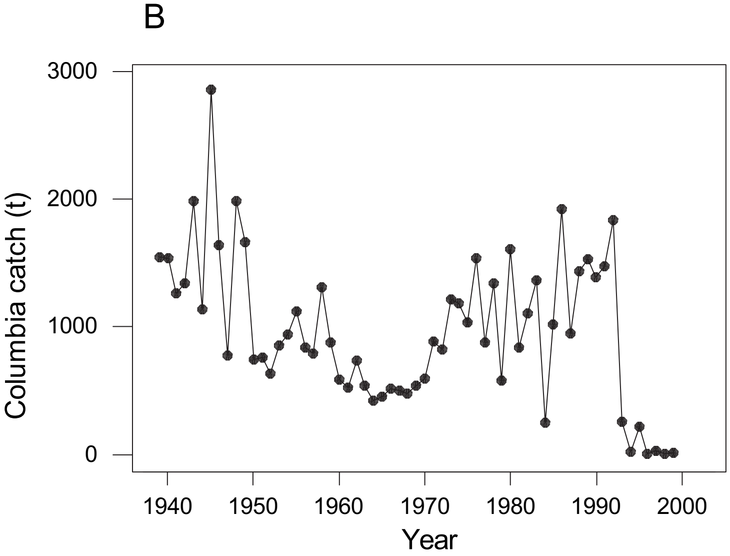


Figure 5. The commercial catches of the Fraser River (A) and Columbia River (B) by year. Low catch rates for recent years in both rivers reflect closures and catch restrictions based on conservation concerns.

a green light signals an advance, a yellow light signals an advance but slower and with caution, and a red light signals a full stop, then a single year with a low SSB could be seen as a yellow indicator. Two sequential years with a low SSB, however, would represent a red light indicator—a full stoppage of all removals.

Annual variation in Columbia and Fraser catches

In the mid-1990s the apparent abundance of eulachons in the Fraser River was lower than most previous years. Fraser River catches were relatively high in the 1940s and 1950s but declined in the 1960s and 1970s and remained low throughout the 1980s and 1990s (Fig. 5A). In contrast to the Fraser, Columbia River catches remained high throughout the 1970s and 1980s (Fig 5B). There were some years, such as 1983, when Columbia River catches were low and also some periods of fluctuations throughout the time series (see Hay et al. [1997a] and Hay and McCarter [2000] for a discussion of factors affecting catch sizes in rivers). Given the variation in Fraser River commercial catch data, it is unlikely that they are useful as an indicator of past trends in abundance. On the other hand, the relative magnitude of catches in the 1940s and 1950s may provide an approximate response point for determining present and future catch levels. Specifically, annual catch levels for much of the 1980s and 1990s was about 20 t and this appeared to be sustained over several decades.

Annual and spatial variation in offshore biomass indices

From 1973 to 1993 the eulachon index in waters offshore of Vancouver Island (statistical areas 124 and 125) was variable and without any apparent trend. From 1994 to 1999 eulachon abundance was low in all surveyed areas. Abundance increased sharply in most areas in 2000 and all areas were at record high levels in 2002. The biomass index for statistical area 124 fluctuated between low levels (<100 t) and nearly 2,000 t between 1973 and the early 1990s, when it declined sharply (Fig. 6A). It remained low through much of the 1990s but increased sharply in 2000, and the increase has continued into 2002, to unprecedented high levels. The offshore index in statistical area 125 follows a roughly similar trend, although the estimated biomass in most years between 1973 and 1993 is lower (<1,000 t). Like area 124, area 125 (Fig. 6B) has increased recently, but the sharp increase did not occur until 2002.

Variation in age composition and origin of offshore eulachons

Approximate age determination of offshore eulachons started in 1999, based on the distinct modes in length frequency (Fig. 3). The size distribution of eulachons in rivers corresponds to the largest size in the sea, or ages 2+ years. Because the offshore surveys occur in May, and because most eulachon probably hatched between March (Columbia River) and April or May (Fraser River), most eulachons observed in the May surveys

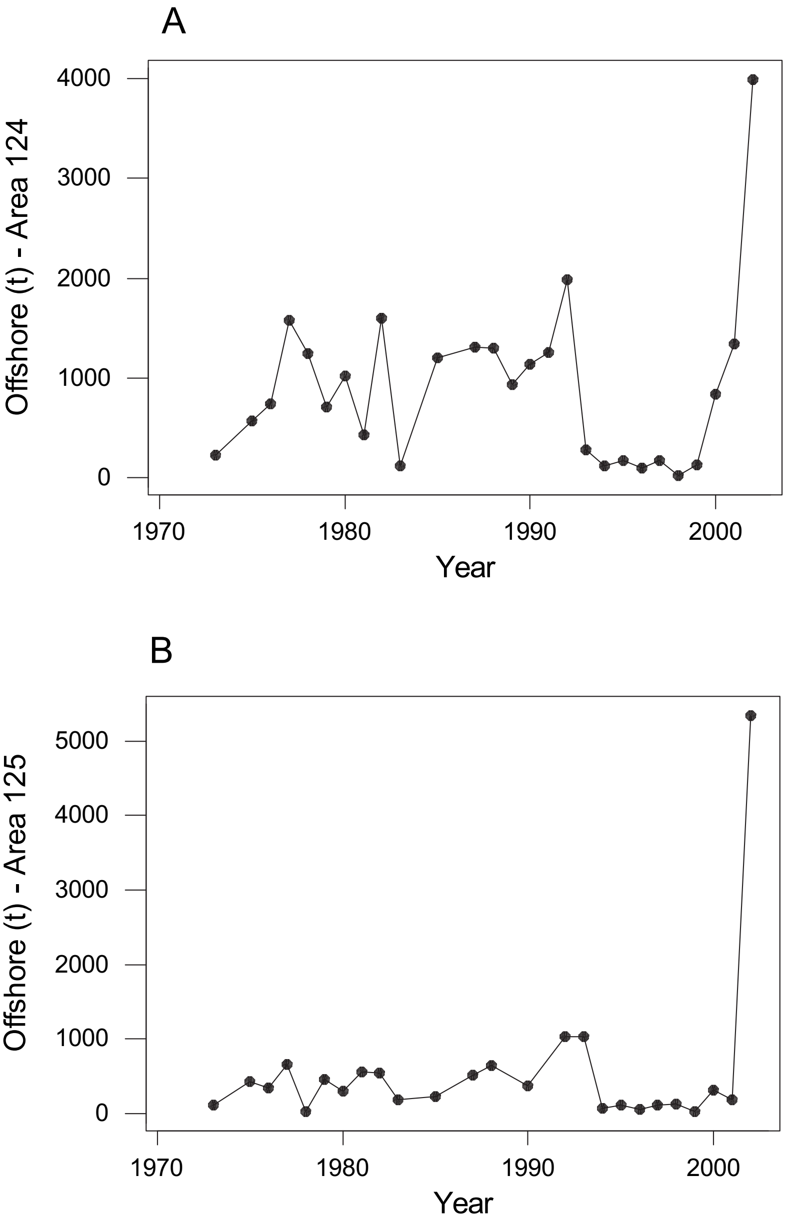


Figure 6. The offshore biomass index from 1973 to 2002 for statistical areas 124 (A) and 125 (B).

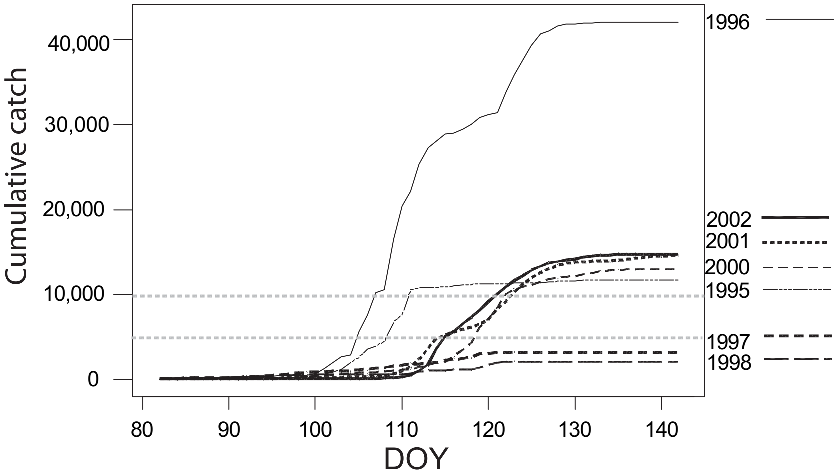


Figure 7. Patterns of cumulative catch from test fisheries for eulachons in the Fraser River, 1995-2002. In two of the years, the cumulative catch did not reach 5,000 pieces (lowest horizontal dotted line). In all other years, the cumulative catch exceeded 10,000 pieces (higher horizontal dashed line). See text for further explanation.

would be between 24 and 27 months of age. Most eulachons appear to spawn at age 3, so the larger size mode would correspond to the spawning fish in the next spring, following the survey. It follows that if reliable estimates of the relative abundance of age 2+ eulachons can be estimated in year *n*, in offshore waters, this would be a useful indicator of future spawning abundance in year $n + 1$. The difficulty with such an estimate, however, is that we are uncertain of the origin and destination of eulachons caught in offshore waters. Preliminary analysis of offshore mixed-stock samples, based on comparisons of genetic samples from most of the larger rivers in B.C., indicates that eulachons on the west coast of Vancouver Island consist of mixtures of Columbia and Fraser river fish, with most from the Columbia River (Beacham et al. 2005).

Temporal trends in test-fishery data

Although the start and finish dates of the test fishing vary slightly among years, the duration of the test-fishing activity has included the main spawning runs (Fig. 7). In general, the earliest fishing days begin approximately on Julian day 80 (March 21) and extend about 9 weeks until Julian day 143 (about May 23). Direct comparison of the catches among years indicates that (1) in some years catches occur earlier than others; (2)

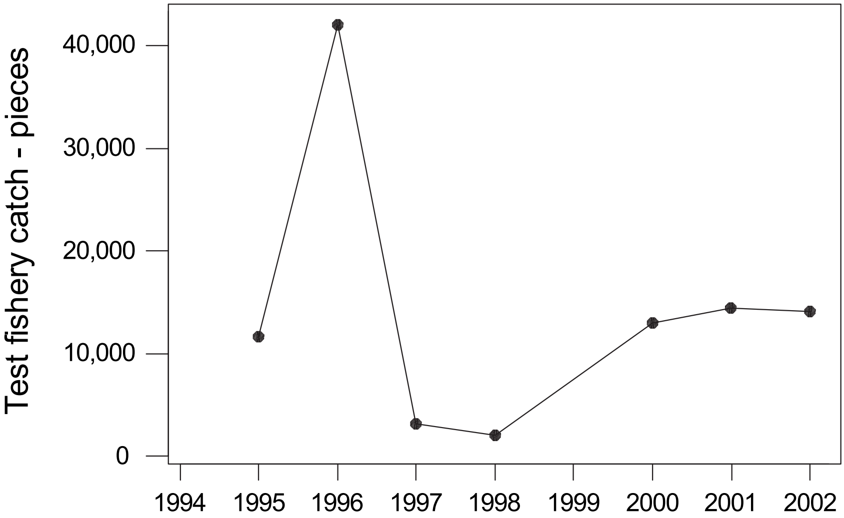


Figure 8. The cumulative catches of the test-fishery (in pieces, or numbers of individual fish) by year. The test-fishery was not conducted in 1999.

sometimes there were several periods (or waves) of high catches punctuated by periods with lower catches; (3) total (or cumulative) catches in some years were much greater than other years. The cumulative catch, compared among all years (Fig. 8) peaked in 1996, declined sharply in 1997 and 1998, and improved to moderate levels and is steady from 2000 to 2002. No test-fishery was conducted in 1999.

Comparison and contrast among indicator data

Comparison of Fraser and Columbia catches

There is no significant correlation between Fraser River and Columbia River catches. In some years both rivers had good runs (Fig. 9) but in many years, when Columbia River catches were high Fraser River catches were are low (see square symbols in Fig. 9). Therefore, high catches in the Columbia (i.e., high SSB) do not necessarily provide assurance of a good run (i.e., high catches or high SSB) in the Fraser River. More important, however, in all years when Columbia River catches were low (i.e., catch <500 t), indicated by the points to the left of the vertical dotted line in Fig. 9, the Fraser River catches also were low. Therefore, low catches in the Columbia River, where spawning and fishing occurs in January and February, may provide a rough but useful indicator of years with low SSB in the Fraser. Specifically, in previous years a catch of less than 500 t in

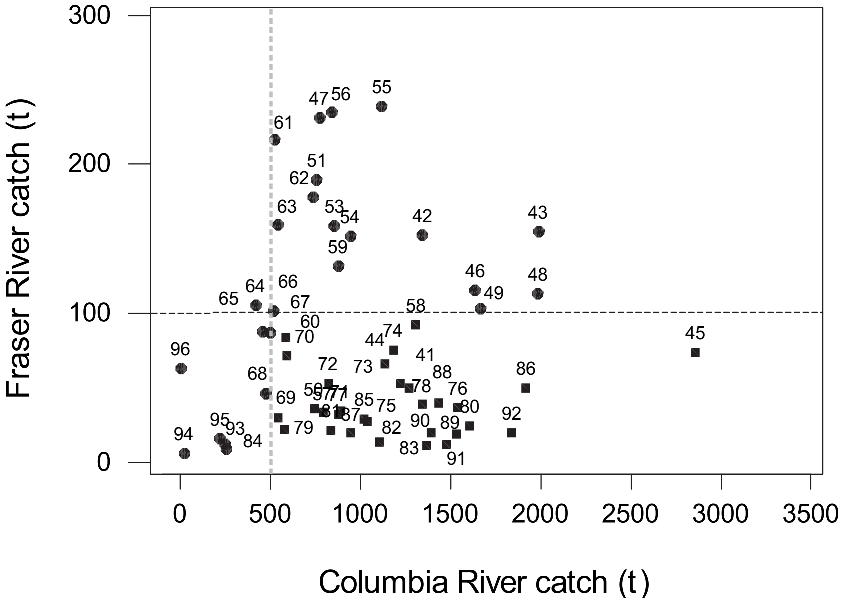


Figure 9. **Fraser River versus Columbia River catches, 1941-1996. Columbia River catches were high in some years (shown as the last two digits beside each point); but Fraser River catches were low, less than 100 t (all points below horizontal dashed line). In contrast, in all years when Columbia River catches are low (<500 t or all points to the left of vertical dashed line) the Fraser River also had low catches.**

the Columbia River could be a response point for pre-season Fraser River management. In more recent years, conservation concerns have led to catch restrictions in the Columbia River, so a simple estimate of total catch is not necessarily indicative of relative run size (i.e., SSB). In recent years, managers monitor daily catches and use other criteria, including eulachon larval surveys, to provide an appraisal of the spawning run size. If their post-season assessment of the relative size of the spawning run is judged to be "poor," this could be used as an indicator for caution (i.e., a yellow light indicator) in the Fraser River.

Comparison of offshore surveys and Fraser and Columbia catches

Hay et al. (1997a) noted that the offshore biomass index was positively and significantly correlated with Columbia River catches (Fig. 10A,B). The offshore biomass estimation techniques used in the 1997 report have been modified and the biomass estimates are reported for separate

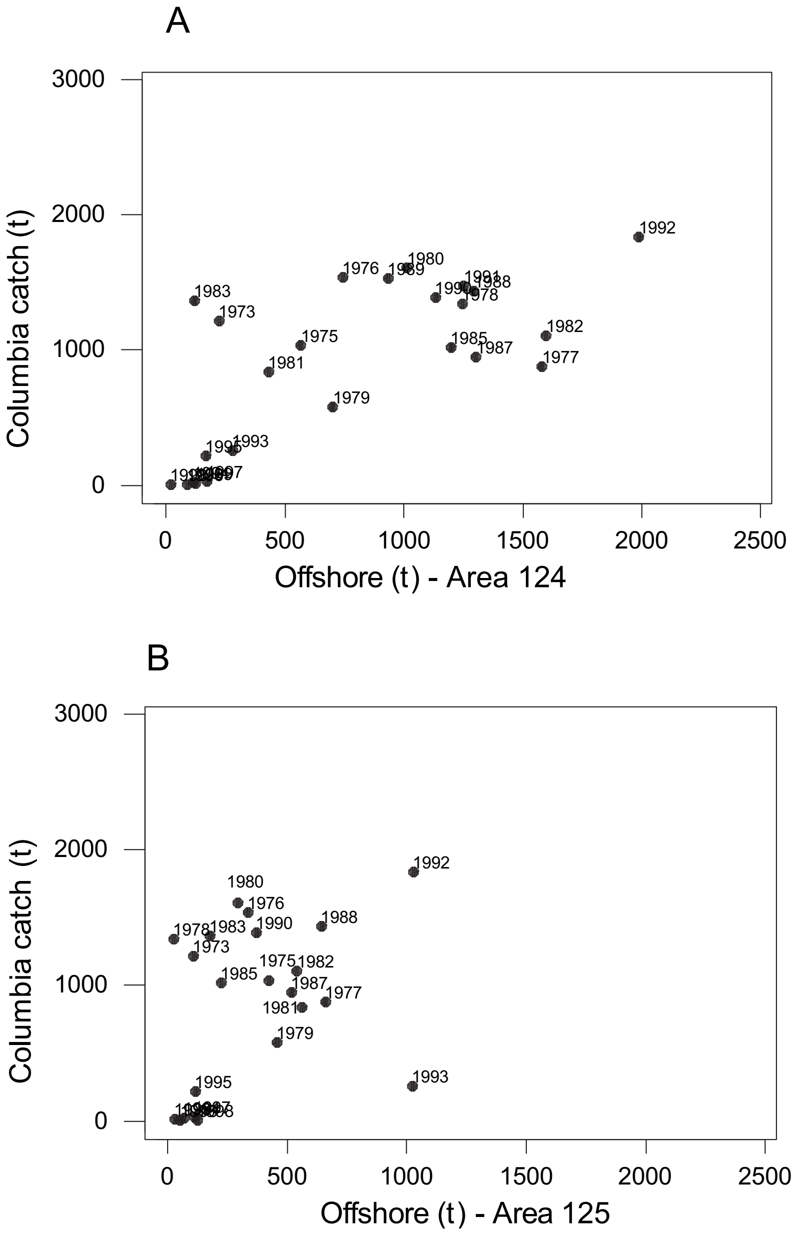


Figure 10. **The Columbia river eulachon catch (t) versus the offshore biomass index estimates for Fisheries and Oceans Statistical areas 124 and 125. The correlation coefficient ($r = 0.702$) for data shown in panel A is significant ($P << 0.01$). The correlation in panel B ($r = 0.363$) is not significant.**

statistical areas (Fig. 3) but the positive covariance between Columbia River and offshore indices remains, although only for the offshore biomass index from statistical area 124. There is, however, no apparent relationship between the offshore biomass indices and Fraser River catches (Fig. 11A,B). This precludes the use of the offshore biomass index as a direct predictor of Fraser River biomass. We caution that this lack of correspondence is based on a comparison of estimated offshore "biomass" that includes two separate age groups. Also it is made at a point in time that is one or two years prior to the time when these same fish would be able to spawn in the Fraser, if that were their ultimate spawning destination. Also the Fraser River catch data in the 1970s and 1980s are not necessarily accurate. For these reasons we cannot dismiss the possibility of a meaningful relationship between the observed offshore biomass and Fraser River SSB, but we can conclude the data are insufficient to allow us to use the offshore biomass as a direct pre-season indicator of Fraser River SSB. On the other hand, a correlation matrix between Columbia and Fraser river catches, and the offshore biomass index, indicates a significant positive relationship between offshore biomass and the Columbia River (Table 1.)

SSB and offshore biomass

The offshore biomass also can be compared with the much shorter data series of SSB, from egg and larval surveys in the Fraser River (Fig. 12). There is no apparent covariance in the comparison of eight years of data, but the year 1996 is of special interest. This year (1996) had an exceptionally large SSB estimate of about 1,900 t but the offshore biomass index was not exceptional. This indicates that the offshore biomass estimates may not apply directly to Fraser River fish. There are some years when the offshore biomass is low, and the Fraser appears to be relatively high, and vice versa. Nevertheless, the lack of a meaningful relationship may be more attributable to the inadequacies of the data, and when faced with insufficient information for management of the Fraser River, it is reassuring to know that eulachons are present in offshore waters. In the years when the offshore biomass index was very low (approximately between 1995 and 1999) the Fraser River catches also were low, with the exception of 1996. Therefore the very low estimates of offshore eulachon abundance observed between 1995 and 1999 (excluding 1996) would be cause for caution—but not alarm—about potential catches on the Fraser. An offshore index of less than 500 t (which occurs in 6 of the 27 years) would be sufficient to implement fishing restrictions—and this *could be a response point.* Such a point would be subject to revision, if the quantification methods for the offshore index were revised.

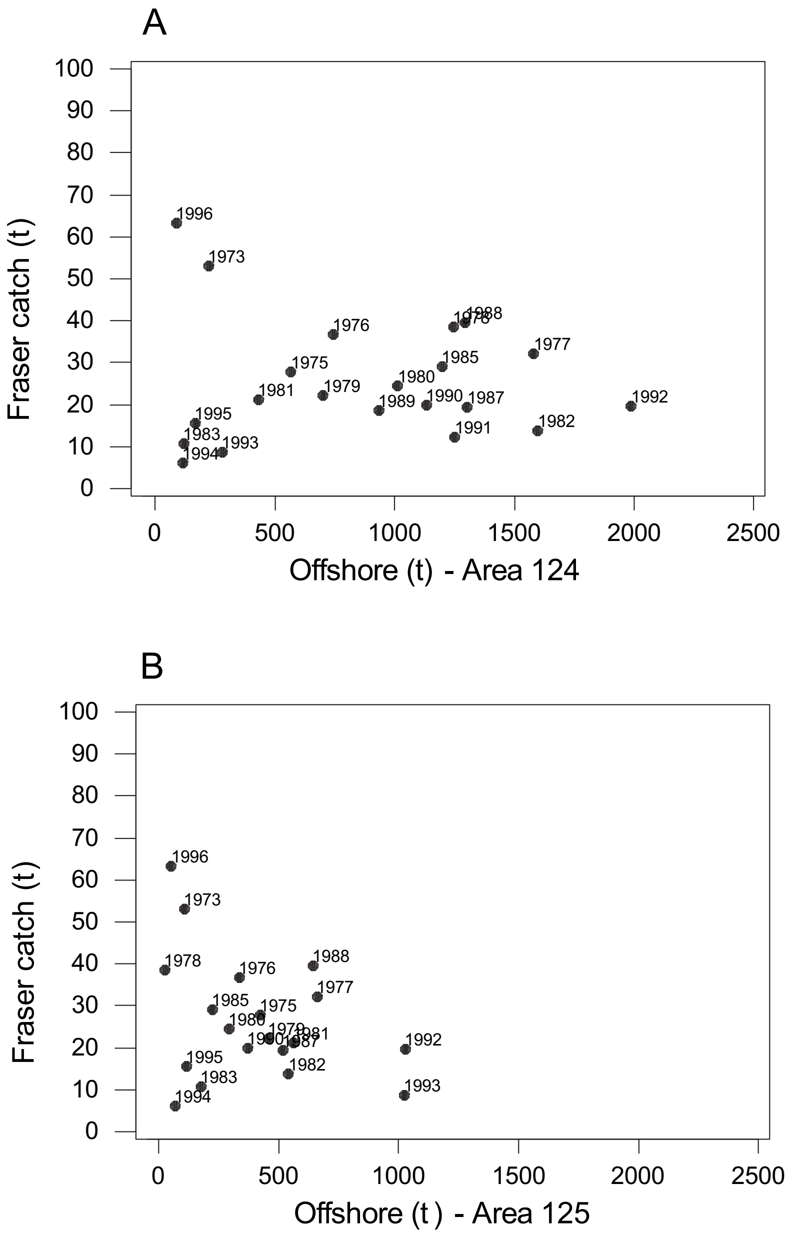


Figure 11. **The Fraser River eulachon catch (t) versus the offshore biomass index estimates for Fisheries and Oceans statistical areas 124 and 125. The correlation coefficient for the data shown in panels A and B are –0.075 and –0.337 respectively. Neither is significant.**

Table. 1. A correlation matrix between Columbia and Fraser river catches and the offshore area biomass indexes.

	Fraser catch	Columbia catch
124 Offshore index	–0.075	0.702
	0.746	0.000
125 Offshore index	–0.337	0.363
	0.158	0.097

The top number of each pair represents the Pearson correlation coefficient, and the lower number is the probability value. There is a significant positive relationship between offshore biomass and the Columbia River catches, but the Fraser River catches do not correspond to either the Columbia River or offshore areas.

Comparison of test-fishery data with catch, offshore, and SSB data

The test-fishery data are significantly correlated with the total Fraser River SSB ($r = 0.924$, $P = 0.003$) and with the SSB (Fig. 13). These relationships are based only on seven points, and the high significance is dependent only on the 1996 data point. The test-fishery occurs approximately at the New Westminster site (Fig. 1) but the year in which the high test-fishery catches were made, most of the spawning occurred below New Westminster (Hay et al. 2002). Therefore, we do not necessarily accept or dismiss the close relationship between the test-fishery and the SSB, because there are some uncertainties associated with test-fishery data. The main value of the test-fishery data would be as an in-season estimator whereby the results of the survey would be immediately evaluated and incorporated into predetermined management schedules. Utilization of test-fishery data should proceed cautiously because we cannot confirm that the test-fishery data vary with eulachon SSB. We acknowledge the significant correlation relationship between the test-fishery data and the SSB estimates (Fig. 13) but reiterate that the relationship consists only of seven points and the significance is dependent on a single data point.

From the test-fishery–SSB comparison (Fig. 13), the SSB in the two lowest years (1997 and 1998) was too low to support fisheries. In both years the cumulative test-fishery catches were below 5,000 pieces (Fig. 7). Therefore, if test-fishery data were to be used for in-season response points, two points can be identified. One is that there should be a minimum catch level before fisheries are considered, and this could be a cumulative catch of 5,000 fish. Such a catch failed to occur only in two years, 1997 and 1998 (Fig. 7). This could be regarded as an in-season response point for a “start” to a fishery, perhaps at a reduced scale. A second point, say 10,000 fish, could be used to establish a point where fishing would occur up to some predetermined point.

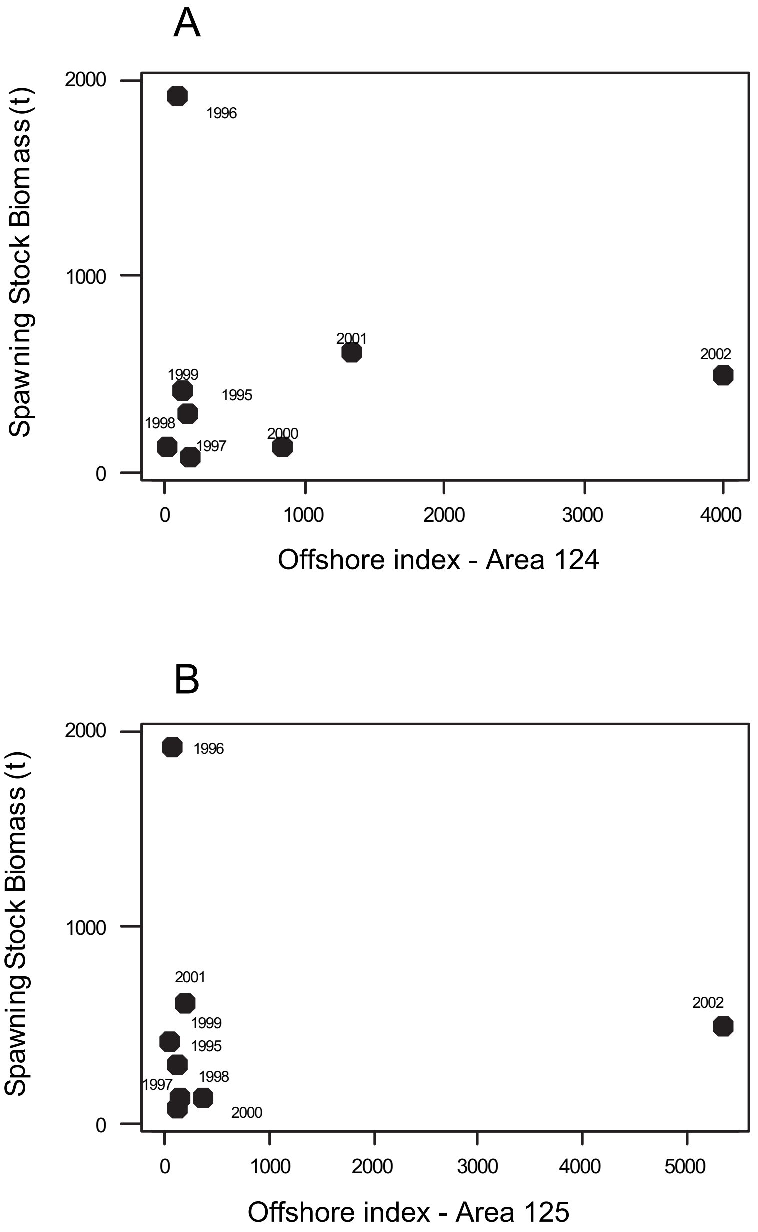


Figure 12. Fraser River SSB estimates versus the offshore biomass index for areas 124 (A) and 125 (B). There is no apparent relationship. The 1996 data point appears to be an outlier in the offshore index, but 1996 was a year of an exceptionally strong spawning run in the Fraser River.

Discussion

Management applications: integrated response points and sequential implementation

The sequential order of appearance of indicators in a year, response points, biological implications, and limitations are summarized in Table 2.

The earliest "pre-season" response point is the SSB estimate from the previous year. Given the short, three-year life cycle of eulachons (Hay et al. 2002) it would be unwise to consider a fishery opening in any year when the SSB in the previous two years was below a fixed response point, because in such a year two and perhaps three consecutive cohorts could have a low abundance. In this case, the capacity of the stock to increase could be compromised with a low SSB. For the Fraser River, the response point associated with the SSB be could be set at 150 t, which is slightly greater than the approximate biomass estimates observed during the lowest years of the 1990s (1997, 1998, and 2000). Therefore, if the SSB in two consecutive years were less than 150 t, no fishery would occur in the next year, regardless of other indicators. This estimate could be available by about October of each year.

The second sequential (pre-season) response point: offshore biomass. Although there is no apparent relationship between offshore biomass and Fraser River commercial catches, this may reflect incomplete data so it would be unwarranted to dismiss a possible connection with offshore eulachon abundance. Therefore, precautionary management should require a minimal level of eulachons in the offshore biomass index: we suggest 1,000 t. Based on the offshore time series, such a limit (i.e., response point) would have occurred for all years between 1994 and 1999.

The third sequential (pre-season) response point: the Columbia River catches. Although the Columbia River catch is not correlated to Fraser River catches, in most years when Columbia catches were low, the Fraser River catches also were low. A post-season appraisal of the Columbia river run, as a "poor" run and indicating a low SSB, is a useful qualitative response point. This appraisal of the Columbia run is available by late February or March of each year, several months prior to the Fraser spawning time.

The fourth sequential (in-season) response point: the test-fishery. The time series is short so the relationship of the test-fishery and the SSB is uncertain. Further, the test fishery catch data gathered to date do not appear to be sensitive to the wide changes in spawning abundance we observed through egg and larval surveys, especially at low levels of abundance. The appearance of a highly significant correlation between the test-fishery and the larval-based SSB is dependent only on the data from one extreme year (1996) when both the SSB and cumulative test-fishery catch was very high. Therefore until the test fishery data are shown to

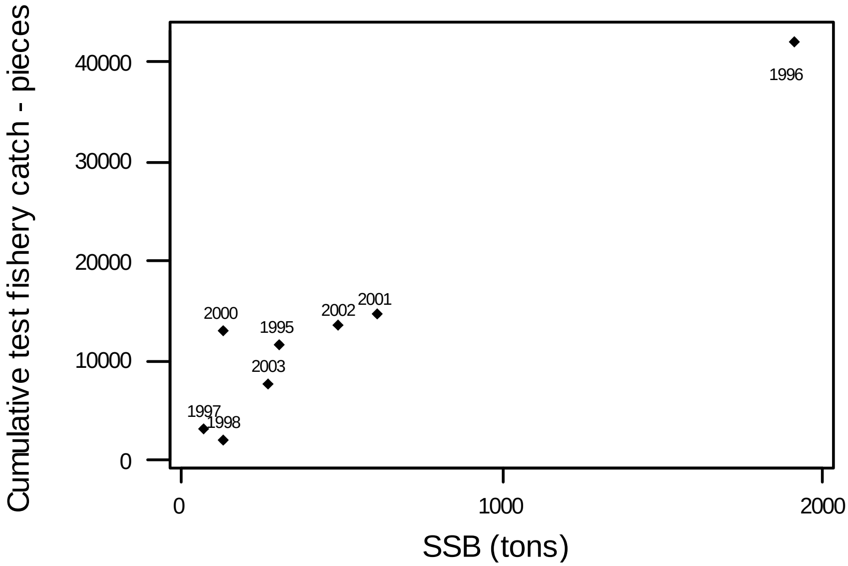


Figure 13. The Fraser River SSB versus the cumulative Fraser River test-fishery catch (in pieces). A highly significant correlation ($P < 0.01$) is dependent on the 1996 data point (see text for explanation).

be reliable, other indicators must be used to make decisions about fisheries. If other indicators are positive, then the test fishery has potential value as a within-season indicator. For instance, a cumulative catch of 5,000 pieces in the test fishery could be a response point requirement for a partial opening of other fisheries. A test-fishery cumulative catch of 10,000 pieces could be a response point required before a full opening of other fisheries.

Resolution of conflicting indicators: a "stoplight" approach

If all of the indicators—or response points—point in the same direction (i.e., all indicating downward or upward trends) then management decisions would be relatively simple. Management decisions would be more complex and difficult if the signals are mixed. Resolution of mixed signals can be facilitated by using decision rules that anticipate some of the combinations of possibilities. For instance, there are four distinct pre-season response points described in the preceding sections.

1. SSB < 150 t for one year.

2. SSB < 150 t for two consecutive years.

Table 2. Efficacy of assessment Indicators, response points, and application to fishery management.

Indicators of biomass	Response points		Purpose of indicator	Limitations
	Conservation concern	Fishable to various levels		
(1) Spawning stock biomass (SSB) (pre and post-season	< 150 t	> 150 t	Precautionary for SSB < 150 t for previous year. Conservation concern for SSB <150 t 2 consecutive years.	Limited time series, 8 years only.
(2) Offshore biomass (pre-season)	<1,000 t	>1,000 t	Indication of marine survival for multi-brood and multi-stock.	No apparent relationship to Fraser but positive to Columbia (Fig. 10). Index for Fraser may be improved by stock and age.
(3) Columbia River catch (pre-season)	Post-season appraisal of the runs as "poor."	Post-season appraisal of the runs as better than "poor."	Indication of overall survival for single return year and single stock (Columbia).	No convincing relationship to Fraser. However, when Columbia was low, Fraser was low (Fig. 9).
(4) Fraser test-fishery (in-season)	Less than 5,000 piece cumulative catch.	Greater than 5,000 piece cumulative catch.	Indicator of in-season returning abundance and overall survival of post-season for Fraser.	Limited time series, 7 years only. Correlation to SSB dependent on one year (Fig. 13). Temporal variations, cause of flow, tide, etc. unknown.

3. Offshore biomass < 1,000 t.

4. Columbia River run: post-season appraisal "poor."

There are at least two potential in-season indicators that might be used.

5. Test-fishery results < 5,000 pieces.

6. Test-fishery results < 10,000 pieces.

There are also at least three different potential management options: (1) a full fishery; (2) a partial fishery; and (3) no fishery. A partial fishery may be appropriate during years when the indicators are mixed. These options are illustrated, using a "stoplight" analogy, relative to nine different combinations of pre-season indicators (Table 3). In this table, the test-fishery results are shown to be most useful when there are one or more conflicting signals. In general, when two or more indicators indicate caution, the fishery should be stopped, or reduced in scale (i.e., a partial fishery). The decision scenarios in Table 3 are guides, subject to revision. Probably all the response points would require modification with future information. Also, there could be reason to consider new indicators, or reconsideration of some of the indicators identified here.

Precautionary catch levels for the Fraser River

Readers may notice that we presented our descriptions and analyses of indicators and response points without any mention of the recommended catch levels. This omission was deliberate, because there is no biological basis for recommending biologically sustainable catches except by referring to past catch levels. Ideally, catch rates or quotas for the management of any species should establish catch levels that consider the size of the spawning stock biomass, age structure, and the biological capacity of the population to replenish itself. Increasingly, ecosystem considerations also must be considered; eulachon in the Fraser River may have an important role as prey for other species, such as sturgeons, which may rely on eulachons as an important source of energy. For Fraser River eulachons, there is not enough information to identify catch levels based on any biological criteria, but past levels of catches may be useful. In the middle of the last century catches often exceeded 100 t (Fig. 9), but in the 1980s and 1990s catches were smaller yet relatively consistent among years (Fig. 11). Specifically, a commercial catch of about 20 t occurred for several decades in the 1970s and 1980s, but this estimate did not include catch from First Nations and recreational fisheries. The sizes of these additional catches is uncertain, but in the Columbia River, the total non-commercial catch may equal or exceed the commercial catch (G. Bargmann, Washington Department of Fish and Wildlife, Olympia, Washington, pers. comm.). If the same ratio applies in the Fraser River, then the combined catches from all "in-river" removals may have been as

Table 3. Array of sequential indicators showing similar and conflicting results for specific reference points, presented in the context of a traffic light regulation for nine different scenarios.

Scenario	SSB	Offshore biomass	Columbia catch	Management response
1.	**G** Y R	**G** Y	**G** Y	Proceed with full fishery.
2.	**G** Y R	**G** Y	G **Y**	Proceed with moderate caution, consult test-fishery results.
3.	**G** Y R	G **Y**	**G** Y	Proceed with moderate caution, consult test-fishery results.
4.	**G** Y R	G **Y**	G **Y**	Consider only partial fishery.
5.	G **Y** R	**G** Y	**G** Y	Proceed with caution, consult test fishery results.
6.	G **Y** R	**G** Y	G **Y**	Proceed with caution, consult test fishery results; consider only partial fishery.
7.	G **Y** R	G **Y**	**G** Y	Proceed with caution, consult test fishery results; consider only partial fishery.
8.	G **Y** R	G **Y**	G **Y**	Full closure justifiable.
9.	G Y **R**	G Y	G Y	Full closure necessary, conservation concern.

G (green) and Y (yellow) indicate an SSB (spawning stock biomass) of more than and less than 150 t, respectively. Similarly, G and Y indicate offshore biomass greater than or less than 1,000 t, respectively. Y and G indicate Columbia River post-season runs as "poor" or "better-than-poor," respectively. The R indicates a red signal. Each combination of the 9 different scenarios shows bold, underlined letters (**G** or **Y** or **R**), with alternates shown as non-bold. The suggested management response to each scenario is indicated on the right. In all instances of conflicting indicators, the suggested response is to (1) proceed with caution consulting the results of the test-fishery, or (2) proceed with extreme caution, and consider only partial fisheries. Three independent **Y** signals are sufficient rationale for a full closure, and a red (**R**) signal would require a full closure.

high as 40 t during this period. Based on the consistency of the catches in the last few decades, it is probable that a continuation of such modest catches ("modest" relative to the larger catches of the 1940s, 1950s, and 1960s) would be sustainable. Therefore, with appropriate combinations of indicators, response points, and decision rules, we suggest that a maximal catch of 40 t (from all removals) could be taken from the Fraser on an annual basis.

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Management of Lake Char in Great Bear Lake, Canada: Historical Perspectives and Future Directions

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Abstract

Although small scale, the majority of fisheries in the Canadian Arctic are culturally significant and provide an important source of protein to northerners. However, they are also data poor. Thus management has been conducted at a relatively unsophisticated level. In new and developing fisheries, such as the lake char (*Salvelinus namaycush*) harvest in the Keith Arm area of Great Bear Lake, there is often minimal or no prior detailed scientific knowledge of population sizes, safe harvesting levels, and food web relationships. In this paper we present a case history on lake char research and management in Great Bear Lake, Northwest Territories, to demonstrate how fisheries issues have traditionally been managed in the Canadian north. Historically, there has been an unstructured pattern of managing fisheries, and a lack of formal records of management actions making it difficult to retrace past decisions. In the current framework of land claims, however, there is a need to have transparency and accountability through a structured process that incorporates the opinions of multiple stakeholders in management decisions. There is also a growing interest in the use of traditional ecological knowledge (TEK) to complement scientific data. While the current DFO Regional Advisory Process does meet some of these needs, we suggest that decision analysis could provide a methodology to formally incorporate TEK, fishermen's opinions, and social and economic concerns into future management decisions for arctic fisheries. To demonstrate how this methodology could be applied to the lake char fishery in the Keith Arm area of Great Bear Lake, we used decision analysis techniques to incorporate the effects of TEK, scientific

analyses, and fishermen's opinions into a model to set total allowable harvest and regulations for the sport fishery.

Introduction

Fisheries in the Canadian Arctic and subarctic are mainly small scale and localized, but harvest a large number of stocks. Commercial and sport fishing activities in the north are relatively young, having only occurred since the shift to living in settlements by aboriginal peoples in the 1950s (Clarke 1993). Approximately 300 fisheries are managed under commercial quota with an equal or greater number prosecuted for subsistence purposes (Reist 1997). Noncommercial food fisheries are considered the most important as the right to subsistence harvest pre-empts all other concerns, save those regarding conservation. In addition to providing income, fisheries are culturally significant and provide an important source of protein to northerners. Resource users tend to have a vested interest in the sustainability of resources due to cultural ties and subsistence needs; thus there is often (but not always) a desire to self-regulate harvests.

Few resources have been allocated to research and management in the north. This problem is compounded by the fact that arctic research is logistically difficult and costly. In new and developing fisheries, such as the lake char (*Salvelinus namaycush*) harvest in the Keith Arm area of Great Bear Lake, there is often minimal or no prior knowledge of population sizes, safe harvesting levels, and food web relationships (Clarke 1993).

Fish species occurring in northern regions are generally characterized by slow rates of growth, late maturity, and extended longevity with a reduced spawning frequency and lower fecundity. Lake char represent an extreme case of this life history pattern. For example, those in Great Bear Lake are believed to reach maximum ages of 60+ years (Clarke et al. 1989), with age at first maturity from 15 to 26 years (Yaremchuk 1986). Most unexploited arctic fish populations have large standing stocks composed mainly of large, old fish with extremely low levels of recruitment (Johnson 1976). This is particularly evident in species at the top of the food chain such as lake char. The low productivity of these fisheries is not always apparent in the initial stages of exploitation as "fishing up" of the standing stock occurs (Ricker 1975). However, once stocks are reduced, replacement and recovery are usually slow since the existing stocks represent years of accumulated biomass. This problem is especially pronounced in lake char, which appear to have evolved an extremely low replacement rate as a natural population regulation mechanism (Falk et al. 1973).

To date, management of fish resources in the north has been conducted at a relatively unsophisticated level due to a lack of information regarding responses of such stocks to exploitation (McCart 1986). Various models have been used to determine safe harvest levels and manage fish stocks; however, most require estimates of stock size, harvest rates, and life history information such as age, length, maturity, etc. Unfortunately, stock size estimates are not available for most arctic fish populations. Although harvest levels are usually recorded for commercial fisheries, estimates of subsistence and recreational harvests are often incomplete or unreliable due to the fact that they are largely unregulated. Life history data are known for some stocks; however, few of these data have been collected as time series. As a result of the dearth of information, relatively simple methods have been used to set quotas and estimate safe harvest levels. Yield indices based on the size of a lake or river system or comparative information from other stocks or water bodies are typically used to set conservative quotas (McGowan 1989, Clarke 1993). This is usually followed by periodic monitoring of biological indicators of population status. Quotas and regulations are typically adjusted using an adaptive management approach through feedback from such monitoring programs.

An important feature of fisheries in the Arctic is the communication between management agencies and resource users. Research and management plans are typically discussed with users. Although this began to develop in the 1970s, land claims have led to formalized cooperative management through the formation of Renewable Resources Boards. Comanagement provides the opportunity for local input in planning and decision-making, integration of science with TEK, and education and training of resource users in research and management techniques. The main features of comanagement are that the government relinquishes control to resource users and the resource users take responsibility for their decisions. Renewable Resource Boards provide advice; however, the minister of the Department of Fisheries and Oceans (DFO) retains ultimate authority for decisions regarding conservation of the resource (Clarke 1993).

In this paper we present a case history on lake char research and management in Great Bear Lake to demonstrate how fisheries issues have traditionally been dealt with in the Canadian north. We then offer a possible approach for future management of arctic fisheries in the current framework of comanagement with land claims boards.

Great Bear Lake

Great Bear Lake (Fig. 1) is the fourth largest lake in North America with a surface area of 31,153 km^2 (Johnson 1975b). The lake has five arms (Keith, Smith, Dease, McTavish, and McVicar) radiating from a large central basin. Its physical characteristics are more similar to lakes of the arctic islands rather than the mainland of Canada (Johnson 1975b). Great Bear Lake is cold monomictic and essentially isothermal due to the short open

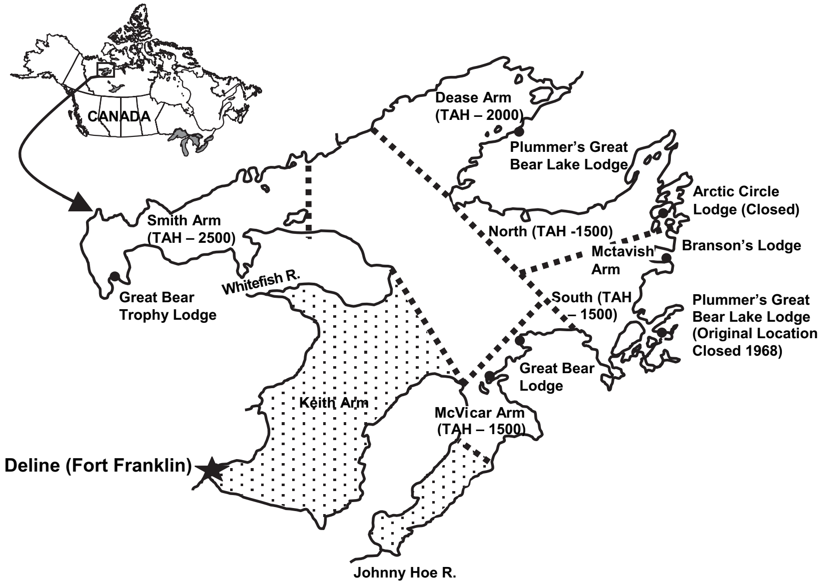

Figure 1. Map of Great Bear Lake, NWT, Canada, showing different management zones separated by dotted lines. Stippled areas are zones set aside for subsistence fishing only. Total allowable harvests (TAHs) of lake char are given as total numbers of fish.

water season and large volume of water. Surface water temperatures range from 4°C over the deepest areas of the lake to 15°C in sheltered areas during the time of maximum heat input in August (Johnson 1975b). The lake is deep (mean depth 90 m, maximum depth 446 m) and highly oligotrophic (Johnson 1975b). Productivity is low as indicated by high secchi depths (up to 30 m) and the low abundance of zooplankton. Great Bear Lake is characterized by an unusually low number of fish species (only 12 regularly occurring) considering its size (Johnson 1975a). The low productivity of this system is believed to result from the combination of cold temperatures and low nutrient input due to the relatively small catchment basin of insoluble rock (Johnson 1975b). Lake char, the primary species, reportedly utilize all areas of the lake, but are most common in waters of < 30 m (Howland et al. 2004, Johnson 1975a).

The lake and its fisheries have remained relatively isolated due to a lack of year-round road access. Deline (formerly Fort Franklin), a Dene community of approximately 800 people located at the south end of Keith Arm, is the only permanent settlement on the lake. Regularly scheduled flights between Norman Wells and Deline take place on a daily basis, and a winter road provides connections to more southerly communities for a short period of time each year. Fishing lodges located in other areas of the lake bring guests directly from southern Canada via chartered jet services.

The fishery

Subsistence

The Sahtudene ("Bear Lake People") traditionally utilized the entire area surrounding Great Bear Lake. They lived a nomadic lifestyle dictated by movements of caribou which were their main dietary staple (Hall 1978). Fish were typically captured as a supplementary food source at a variety of locations around the lakes (Hall 1978). In the early 1950s they began to take up permanent residence in the community of Deline at the southern end of Keith Arm. At this time large quantities of fish (mainly lake whitefish, *Coregonus clupeaformis,* and lake cisco, *Coregonus artedii*) were captured as food for sled dogs (Hall 1978).

With the increased use of snowmobiles and the move away from traditional ways of living there has been a decrease in the amount of fish required to feed dogs (Crawford 1989). Fishing, however, still remains an important part of the Dene culture and provides an essential source of protein. Deline is the main source of lake char in the Sahtu Region. Most of the harvest is consumed in Deline but fish are also shared and/or bartered with other communities (Bayha and Snortland 2002, 2003, 2004). Char are harvested using 5-5.5 inch mesh gillnets. Residents occasionally travel to other locations such as the mouths of the Johnny Hoe and Whitefish rivers (Fig. 1) to capture whitefish that congregate in these areas at particular times of the year (Miller 1947). Small mesh (3 inch) gillnets which were traditionally used to capture lake cisco are now rarely used.

There has never been a lake char quota for the Keith Arm area of the Great Bear Lake, and past harvest levels for this fishery are poorly recorded since there were no requirements for reporting of catch by subsistence fishers. With the settlement of the Sahtu Land Claim, surveys have recently been carried out to estimate the level of the subsistence harvest by Sahtu beneficiaries (Bayha and Snortland 2002, 2003, 2004).

Recreational

Great Bear Lake represents one of the two major sport fisheries for lake char in the Canadian north (the other is located in the east arm of Great Slave Lake) (McCart and den Beste 1979). Great Bear Lake and the east arm of Great Slave Lake are world renowned for containing large numbers of trophy size lake char (Low and Taylor 2004). Lodge development on Great Bear Lake began in the late 1950s, with the majority of lodges establishing themselves during the following decade (Johnson 1975b). Fishing activity by the lodges has been concentrated in the four arms of the lake excluding Keith Arm. There have been 4 to 6 lodges in operation at any given time (Fig. 1) with up to 1,300 mainly out-of-territory guests annually (Yaremchuk 1986). At the present time four lodges operate on various arms of the lake, and recently a fifth lodge was built in Deline (Fig. 1). Although this lodge has been running for several years, the outfitting operation was launched in August 2004.

Commercial

There has never been a commercial fishery on Great Bear Lake, with the exception of a limited fishery to supply the needs of other communities and itinerant anglers in the region (Clarke et al. 1989). Studies were conducted in the 1940s, to determine the commercial potential of a fishery (Miller and Kennedy 1948); however, based on the oligotrophic nature and limited productivity of the lake, recommendations were against any such development. Since this time there has been occasional renewed interest in the development of a commercial fishery by the community of Deline. Experimental fishery licenses were issued to the community in 1982, 1983, and 1986 and again in the early 1990s (DFO 1985, McGowan 1989, McGowan et al. 1993); however, catches were poorly reported and the fisheries were never converted to commercial status.

History of research and management in Great Bear Lake

Management of Great Bear Lake has largely been focused on the sport fishery. The history from the first biological surveys in 1945 to the present is presented in Table 1. The history of lodge development was characterized by an initial rapid expansion and then subsequent decline as fish resources became scarce. The fishery and lodges went through a recovery in the late 1980s and 1990s culminating with the development of the Grey Goose Lodge in Deline in the mid-1990s. Management actions have been primarily aimed at controlling the catch and possession limits of sport anglers. These limits have gradually declined from 5 and 10, to 1 and 2, respectively. Research into the harvest and state of the stock(s) has been discontinuous with the most intensive work occurring in the 1980s. Initial stock assessments (summarized in Yaremchuk 1986) were primarily based on standard creel census data collected between 1972 and 1980 from char stocks fished by the various lodges. These studies generated detailed time series of catch per unit effort, harvest and release rates, size and age structure (e.g., Fig. 2a), and growth of harvested char stocks as well as estimates of natural and fishing mortality and equilibrium yield. Later assessments (1984-1985) included experimental gillnetting and

Table 1. History of research and management in Great Bear Lake.

Time period	Lodge development	Management action	Assessment research	Reference
1945	Nil	Nil	First biological survey	Miller 1947, Miller and Kennedy 1948
Late 1950s	Sport fishing starts			
Early 1960s	Sport fishing expands to 4 lodges	CPL = 5 and 10	General limnological and fishery surveys, fish tagging program (1963-1965)	Johnson 1975a, 1975b; Falk et al. 1973
1970-1980	3 of 4 lodges closed or moved due to declining catches of trophy fish	CPL = 3 and 5 (1974). CPL = 2 and 3, only 1 char > 700 mm (1979).	Study of char sport fishery: • Standard creel census (1972-1980). • Experimental gillnetting and tagging (1978-1980).	Yaremchuk 1986 ; Moshenko and Gillman 1978; Falk et al 1973, 1974
1981-1990	No further lodge development	GBL management committee formed (1986). Lake divided into management areas. Moratorium on lodge development with exception for one lodge in Keith Arm.	Follow-up study of char sport fishery (1984-1986): • Standard and intensive (catch sampling of subsample of anglers) creel census. Experimental gillnetting. • TAHs per lake arm determined. Angler diary program initiated (1987-1990).	Roberge and Dunn 1988
1991-present	Sahtu Claim (1994). Keith Arm Lodge built	CPL = 1 and 2. Comanagement under SRRB (1994-present). GBL group formed (2002) to develop management plan for GBL watershed.	Char stock assessment studies (experimental gillnetting, genetic stock structure): • Keith Arm (2000-2004). • McVicar, McTavish, Dease, and Smith arms (2003-2006).	Anderson and Thompson 1991 This paper

GBL = Great Bear Lake; CPL = catch and possession limits; TAH = total allowable harvest; SRRB = Sahtu Renewable Resource Board.

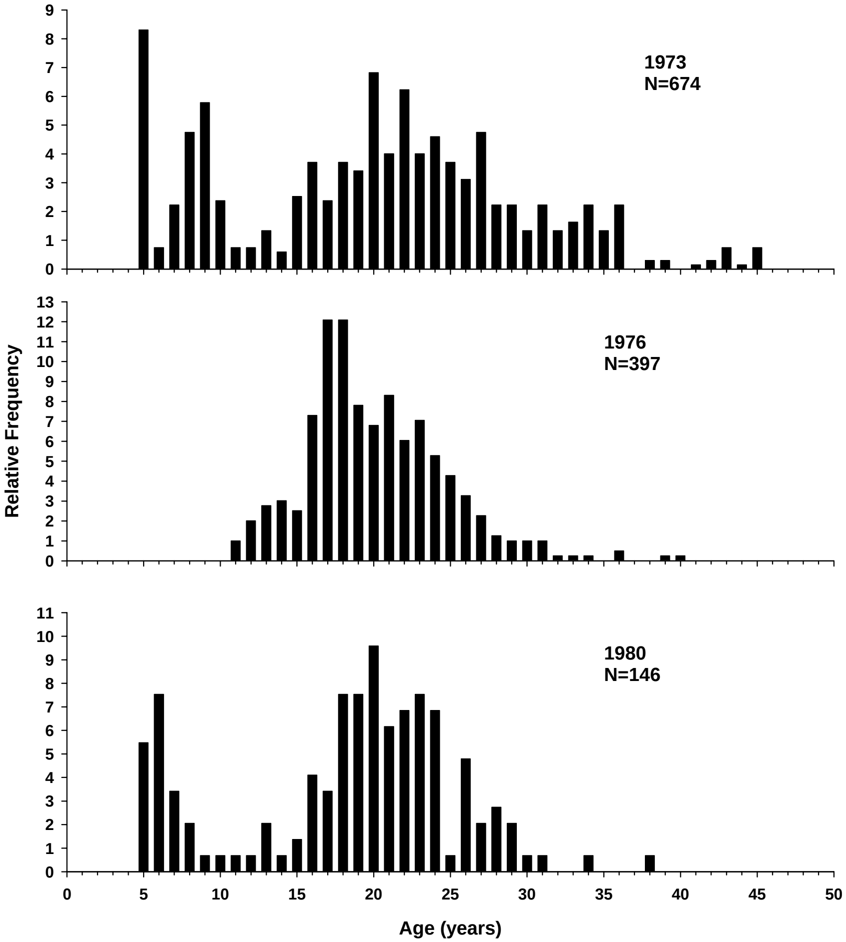

Figure 2a. Age frequency from angled catches at Great Bear Trophy Lodge, Smith Arm 1973, 1976, and 1980. Data from Yaremchuk (1986).

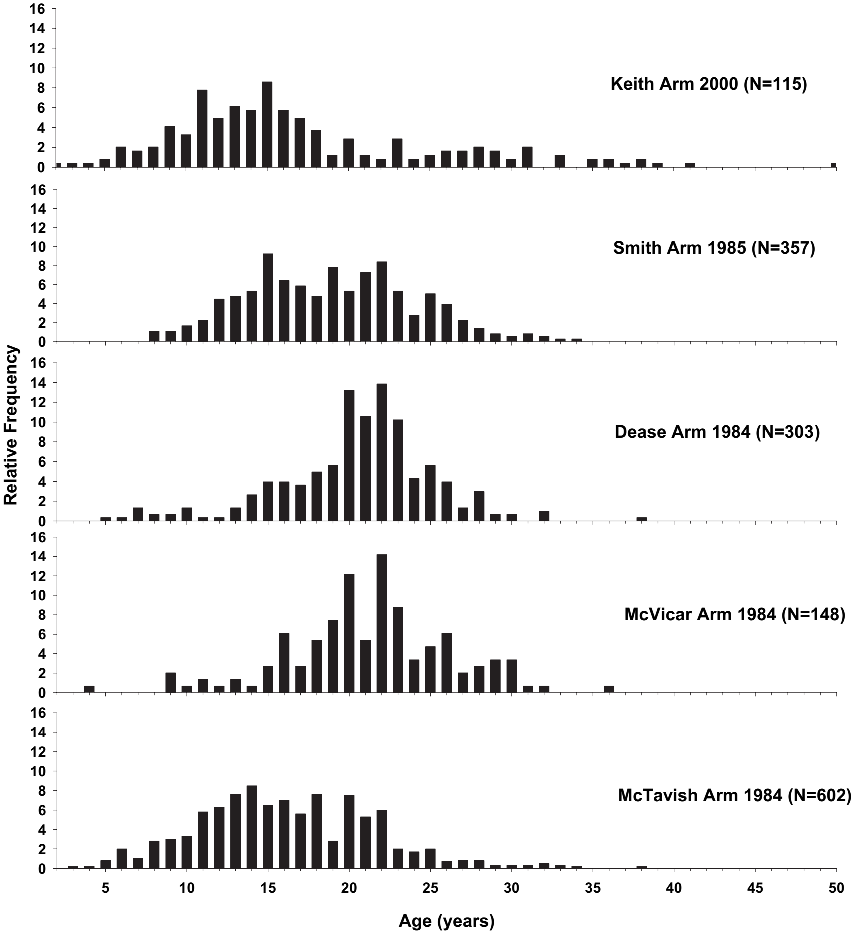

Figure 2b. Age frequency from Great Bear lake basins; Smith, Dease, McVicar, and McTavish arm data from Roberge and Dunn (1988); Keith Arm data from this study.

intensive creel census studies (Roberge and Dunn 1988). Lake char age structure variation based on these studies is shown in Fig. 2b.

The most significant changes in management and research occurred with the formation of the Great Bear Lake Management Committee in 1986 and the Sahtu Renewable Resource Board in 1994. A number of management decisions intended to preserve the trophy status of the fishery were implemented in the 1980s largely as a result of recommendations by the Great Bear Lake Management Committee, which was composed of representatives from government, the lodge industry, and the community of Deline. The most important recommendation was to divide the lake into management areas on the assumption that char had limited movements (Yaremchuk 1986). Total allowable harvests (TAHs) were then assigned to each area based on studies by Roberge and Dunn (1988). It was thought that this management system would encourage lodges to fish more conservatively. Keith Arm and the south end of McVicar Arm were reserved for subsistence use only. A moratorium was placed on further expansion of the recreational fishery with the exception of a provision for the community of Deline to build one additional lodge in Keith Arm. No commercial fishing was allowed in arms where lodges were situated. Lodge operators were urged to minimize harvest of lake char below trophy size; catch and possession limits were further decreased to one and two in 1991.

These management changes appear to have been effective as the lodges have gradually developed a more conservative approach to fishing. This was in part due to changes in regulations, but the lodges also appear to have learned through their own mistakes. It is in their own best interests to maintain trophy status of the fishery since this, along with the remoteness of Great Bear, is what makes it unique among sport fisheries and continues to bring in tourists. Allocating specific areas to individual lodges has probably encouraged lodge owners to manage their designated areas carefully and has allowed the fishery to recover with relatively little enforcement. Although regulations allow for a small number of fish to be retained, all lodges currently operating have a policy of releasing all char caught with the exception of some smaller char which are retained for shore lunches (G. Low, DFO Hay River, NWT, pers. comm.). Although the continued return of guests and the captures of world record trophies in the 1990s (Low and Taylor 2004) suggest the fishery is recovering, updated stock assessments are needed to confirm this.

Although recommendations were made by the Great Bear Lake Advisory Committee for continued monitoring of char stocks at less than 5 year intervals and the need to obtain a relative measure of stock abundance was identified (Clarke et al. 1989), no biological studies were conducted in the 1990s. The implementation of the Sahtu land claim (1994) and a desire to pursue development of recreational and commercial fisheries in Keith Arm has led to renewed interest in conducting stock assessment studies on Great Bear Lake. From this point the lake has been comanaged between the Sahtu Dene and Metis and DFO. Comanagement has led to a more complex but more realistic assessment of management issues due to the interplay between subsistence, sport, and potential commercial fisheries. We are currently conducting an intensive stock assessment of lake char in the Keith Arm area (never previously assessed) over a 5-year time frame (2000-2004). Additionally we have begun to reassess lake char in each of the other four arms of the lake.

Current management issues in Great Bear Lake

Management decisions to date have been somewhat ad hoc in nature. There is no clear record of management actions to date, nor the scientific bases upon which management actions were taken. Factors such as economics and opinions of resource users presumably had an influence on decisions; however, there are no records of how these factors were incorporated into the process.

In spite of the shortcomings, management appears to have worked relatively well over the long-term. This has, in part, been due to a desire by lodge owners to self-regulate harvests in the latter years of the sport fishery, and because the fishery was relatively simple with respect to overlap between resource users. To date, management decisions regarding fisheries in Great Bear Lake have mainly been concerned with regulating the sport fishery since there was no commercial fishery on the lake and subsistence fishing was largely isolated to the Keith Arm area.

With development of the lodge industry in Keith Arm, recurring interest in development of commercial fisheries by local residents of Deline, and a substantial subsistence fishery, there are now more complex management issues relating to allocation of fishery resources. Given that subsistence fishing is an integral part of the Dene culture, it will have to be balanced against the goals of any developing commercial or recreational fisheries. These issues may be further complicated by impending hydroelectric and hydrocarbon development, climate change, and demographic changes in the local population. We must try to determine what combination of subsistence, recreational, and commercial fishing will provide the greatest employment and economic benefits, while preserving the culture and lifestyle of the community and ensuring conservation of the fishery.

In the current framework of land claims we are faced with the need to incorporate opinions of multiple stakeholders in management decisions and to have transparency and accountability throughout the process. As well, there is a growing interest in the use of traditional ecological knowledge (TEK) to complement scientific data (Berkes 1994, 1999). With the increased availability of formalized TEK studies in northern regions, we are now in a position to utilize such information in fisheries management decisions. The regional advisory process (RAP) employed by DFO is designed to include all stakeholders in the fishery. This process brings together fisheries scientists, harvesters, aboriginal community representatives, and other groups to discuss the status of various stocks. The resulting stock status reports are accompanied by written proceedings of the meeting so that both the formal and informal aspects of the meeting are available for future reviewers to understand the outcome.

Although the RAP is a step forward in recording both scientific and non-scientific rationales for decision-making, it still may be difficult for future reviewers to see exactly how much weight was placed on one piece of information compared to another. There have also been difficulties in formally integrating TEK with scientific information since they represent different perspectives on the biological world (Berkes 1999). The former often focuses on unusual events because these are critical to the survival of the human population. The latter focuses more on the average or mean events with unusual events considered as outliers. Scientific data are numerical and can be analyzed using formal statistical procedures whereas TEK is generally not numerical and does not lend itself to conventional statistical analysis. Finally, TEK has alternately been treated as a quasi-religious entity that is unassailable or dismissed out of hand as being completely anecdotal or opinion (see Berkes 1999). This reverence and skepticism could be put aside if the information could be placed in a framework to test its reliability in decision-making.

Historically, fisheries science has relied upon formal statistical models. However, decision-making in fisheries management has not been undertaken using these models alone, but relies on guesses regarding areas of uncertainty and may be heavily influenced by political, economic, and cultural concerns. As noted by Wilimovsky (1985), when records are available, formal scientific predictions on fisheries have almost always been modified by judgment where there is uncertainty. There is no standardized approach to modifying fisheries management decisions in the face of uncertainty; thus two or more individuals may arrive at different management recommendations. For arctic fisheries the problem is to find a methodology that allows for formal consideration of uncertainty and the inclusion of non-scientific information and opinions of resource users without compromising the principles of scientific fishery management.

Future management approaches

Decision analysis is a tool that can be used when a decision must be made in the absence of complete information (Raiffa 1960). Decision analysis can formally incorporate both value-based and quantitative information into an analytical framework (Clemen 1996). Thus, decision analysis could provide a methodology to formally incorporate TEK, fishermen's opinions, and social and economic concerns into fishery management decisions.

The benefit of this approach lies not in removing the decision-making authority from decision-makers but in providing a focus for identifying and examining all components and factors (Clemen 1996).

Decision analysis framework for managing Keith Arm lake char in Great Bear Lake

To demonstrate how decision analysis could be applied to the fishery in the Keith Arm area of Great Bear Lake, we propose to use decision analysis techniques to incorporate TEK, scientific data, and fishermen's opinions into a model to set the TAH and associated regulations for the sport fishery. The objective is to maximize the lodges' fishing success while maintaining a stable fish protein supply for the community of Deline. We used decision programming language (DPL; Clemen 1996) to construct an influence diagram (Fig. 3) and thence a decision tree for the fishery problem.

The model

To model the goal of maximizing lodge success it was assumed that there was a simple relationship between costs and revenue (Fig. 3). Lodge success (profit) was influenced by three things: the number of guests (nights stayed) paying to attend the lodge, fixed costs of maintaining the lodge and boats (repairs, taxes), and operational costs (wages, fuel, meals). Thus:

Profit = (revenue from guest stays) – (fixed costs + operating costs),

where profit is lodge success. Operating costs were assumed to be related to the number of guest stays.

The number of guest stays was related to several factors: price per guest, the quality of the wilderness experience offered, and the frequency of trophy fish captured with time. The price per guest was fixed and therefore is without uncertainty for any year. The quality of the wilderness experience is influenced by the density of anglers and the presence of settlements. Lodge owners have stated that the presence of gillnets reduces the quality of the wilderness experience for the anglers. Thus, this factor was further influenced by the location of fisheries. The frequency of trophy fish taken was influenced by the number of large fish released during the subsistence harvest (which usually releases large fish) and the sport harvest as well as the mortality rates of released fish.

The likelihood that the community of Deline would have a stable food supply was considered to directly depend upon the stock size of lake char. The latter was modeled as:

Stock size after fishing = previous stock size + recruitment – removals from fishing (Fig. 3).

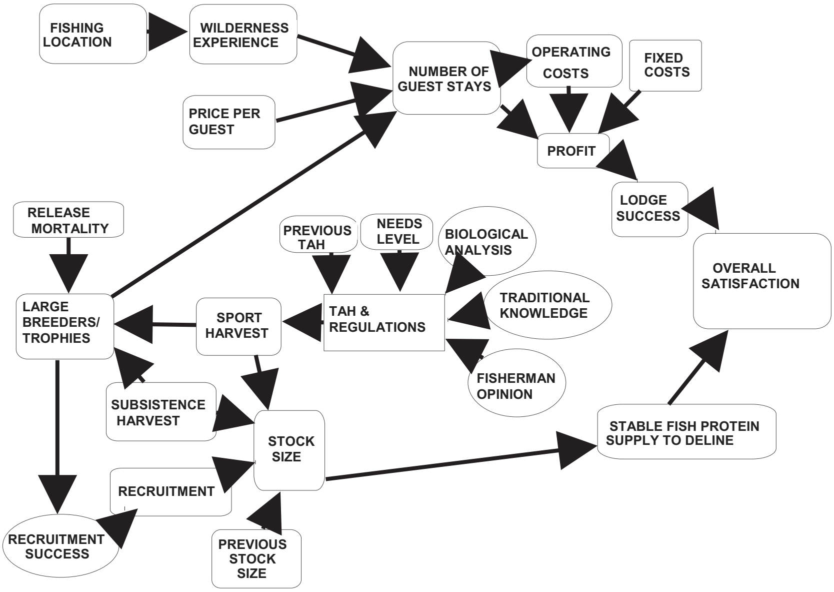

Figure 3. Generalized influence diagram for decisions on changing the total allowable harvest (TAH) in the Great Bear Lake lake char fishery. The rounded rectangles represent consequences or calculation nodes, the ovals represent chance or variable nodes, and the rectangle represents the decision node.

Recruitment success was considered to be uncertain and probabilistic and dependent upon the number of large breeders in the population. The latter was influenced by the success of the catch and release of large fish by both the subsistence and sport fisheries. Fishing was subdivided into sport and subsistence harvests.

The model to set the TAH and associated regulations was considered to be influenced by five factors: the previous quota; needs of the subsistence fishery; biological analyses; TEK; and fishermen's opinions. Subsistence needs were set at a fixed level that limited the number of fish available to the sport fishery. It was assumed that previous TAHs from other areas of Great Bear Lake and existing regulations would also influence the initial TAH for the Keith Arm sport fishery. Techniques for the scientific analysis of yield are well established and it is assumed that the stock assessments presented to the RAP represent the best possible estimates of sustainable yield given the available data. However, to allow for uncertainty this was analyzed as a chance node in the model. This allowed different risk options to be presented. Similarly, TEK was modeled as a chance node to reflect variation in opinion among elders in the community. Elders might be defined from the community perspective or by an age cut-off. Most elders are over the age of 55. The elders could be asked whether they favored a decrease in TAH to a certain level, status quo, or an increase to a certain level. Alternatively, elders could be asked if they thought it was a good idea to maintain the present regulations, or increase the daily allowable take per angler or decrease it, and the percentages for each choice used to develop a probability distribution.

The opinion of the subsistence fishers was modeled as a chance node to reflect variation in their opinion. The fishermen could be asked questions similar to questions asked of the elders, and the percentages for each response used to develop a probability distribution.

Posing a direct question about the decision or the state of the fish population is a method of collecting expert opinion known as the Delphi-Technique (Zuboy 1981). The assumption is that both elders and fishermen in a culture that passes on information by oral tradition would have considerable knowledge of the system, but little that is formally written down. They model the world by synthesizing a large number of factors in their minds and then giving an opinion on the state of the system.

An alternative approach to obtain TEK is to ask the local people a number of specific questions to fill knowledge gaps in the scientific information. For example, the elders may be asked "Are the fish larger or smaller than when you fished 30 years ago?" If the majority of the elders thought that fish were smaller after a few years of sportfishing, a researcher might conclude that the stock has been affected by this activity. The problem is that researchers must ask the right questions—perhaps the elders also know that the growing season has been much shorter for the last few decades or that during the war the military took out enormous amounts of fish (with a resulting increase in average size of the remaining stock). Secondly, with this method, researchers still rely upon a Western scientific interpretation of what the elders are saying. Because TEK and the scientific approach are intertwined, it is difficult to separate whether the traditional knowledge or the scientific data are flawed thereby producing erroneous conclusions. We are not suggesting that this standard approach is worse, but that there is a case for a simplified decision framework such as we present here. Alternatively a hybrid of these two approaches could be used within the framework we describe by using TEK to supplement biological data, while at the same time also directly asking opinions of elders and fishermen.

In our approach the output from fishermen's opinions, TEK, and scientific analyses were combined to determine if a change in TAH was desirable and to what level. The decision node was modeled as a "YES" or "NO" option. The "YES" option was modeled as:

$$\mathit{Change\ TAH} = [(A \times \mathit{Biological\ Analysis} + B \times \mathit{TEK} + C \times \mathit{Fishermen's\ Opinions}) \div 3] \div \mathit{Previous\ TAH}$$

in which *Biological Analysis* could be the expected value of the probabilistic node developed from RAP advice, TEK would be the expected value of the opinion of the community elders, and *Fishermen's Opinions* could be the averaged opinion of all the active subsistence fishermen. *A*, *B*, and *C* are coefficients for weighting the reliability or importance of the biological analysis, TEK, and subsistence fishers' opinions, respectively. In the general model *A*, *B*, and *C* are set to 1, but in many cases the weighting might be less than one for some of these inputs depending on the confidence in the information provided. The results from this equation would convert all information to a common set of values. The output would be a number multiplied against the previous TAH to determine the new TAH.

Conclusions

The case history of Great Bear Lake demonstrates some of the issues common to arctic fisheries, where research must be done with limited resources and decisions must be made with limited data. Historically, there has been an unstructured pattern of managing fisheries on this lake and a lack of formal records of management actions, making it difficult to retrace past decisions. The current DFO RAP process is a step forward in that both scientific and non-scientific rationales for decision-making are recorded. Still, there are difficulties in determining the relative influences of different pieces of information and in formally integrating TEK with scientific information.

We suggest that managers should consider the use of decision analysis as a flexible tool that can provide an approach to solving multiattribute problems presented by many fisheries. It permits integration of scientific information with TEK and other information sources (opinion, economics, cultural, and political). A decision analysis allows explicit description of both quantitative and qualitative components of the problem in a modeling framework and influence diagrams provide a framework for inputs by experts and informed laypersons. Furthermore, we believe that resource users are likely to identify with the method since it mimics their natural decision-making process. Thus, it may provide a reasonable approach for making future decisions in arctic fisheries.

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Density and Mortality in a Harvested Population of Quahog (*Mercenaria mercenaria*) in Nova Scotia, Canada

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Abstract

Innovative Fishery Products Inc. has managed a 1,682 ha quahog (*Mercenaria mercenaria*) lease in St. Mary's Bay, Nova Scotia, Canada, since 1997. A management strategy based on population modeling is desired to optimize production on a long-term basis. This requires a description of life history parameters, and data on the quahog population and its commercial exploitation. The objectives of this study were to describe the data collected on the commercial fishery, estimate quahog densities, and calculate preliminary mortality rates for the population. Mean densities ranged from 48.3 to 88.4 individuals per m^2 during surveys conducted in June 2001 and 2002, and May 2003. Densities were higher than those typically described for commercially harvested quahog beds. The mean age to market was 7 years. Spat recruitment was variable and age frequency graphs suggest immigration of juvenile quahogs between the ages of 3 and 6 years onto the intertidal portion of the lease area. Survival was estimated between 24 and 37% for 7-8 year old quahogs using catch curve and analysis of covariance techniques, where commercial exploitation only represented 5-10% of the loss. Causes of apparent high natural mortality are unclear, but winterkill due to ice abrasion or scouring, predation, and the movement of quahogs from the lease appear reasonable.

Introduction

Quahog population distribution in Atlantic Canada

The northern quahog (*Mercenaria mercenaria*) is a bivalve found in shallow coastal waters from the Gulf of Mexico to its northern limit in the southern Gulf of St. Lawrence. These bivalves are found in small patches or large beds in both intertidal and subtidal reaches of coastal embayments, from muddy sand to sand-based sediments (Grizzle et al. 2001). The geographical distribution of this species in Atlantic Canada is limited to areas where summer water temperature exceeds 20ºC (Landry and Sephton 1996), and therefore wild quahog populations typically occur in the southern portions of the Gulf of St. Lawrence (Fig. 1). Two populations have been documented in the Bay of Fundy region of Atlantic Canada, one of which in St. Mary's Bay, Nova Scotia, Canada. However, details on their origin and actual population structure have never been described (Whiteaves 1901, Dillon and Manzi 1992). Innovative Fishery Products Inc. (IFP) manages this population, which represents the only commercially viable quahog stock in the Bay of Fundy. This paper represents the first study of the St. Mary's Bay population.

Lease production and fishery management

Quahogs are harvested in St. Mary's Bay from May to November with the peak harvest period occurring from June to September. The annual harvest has ranged from 95 to 370 t since commercial harvesting began in 1997 (Fig. 2). Lease management is based on (1) routine visual inspections for quahogs of the intertidal portion of the lease prior to the harvest season; (2) harvest rotation whereby the lease area is harvested in plots and plots may not be harvested every year; (3) a minimum shell length of ≥50 mm, although the harvest may include a small percentage of individuals between 45 and 49 mm; (4) daily harvest monitoring; (5) a harvest season from May to November; and (6) active lease enforcement throughout the year where IFP reports illegal lease harvesting to Department of Fisheries and Oceans (DFO) enforcement officers. IFP and DFO entered into a four-year partnership to evaluate the use of population models to develop long-term management strategies to optimize quahog harvesting on the lease. St. Mary's Bay was considered to be ideal for population modeling. The St. Mary's Bay population appears to be an isolated population whereby immigration or emigration are currently considered negligible, the population can be readily surveyed, the lease area is managed by one user group, and good data on daily harvest and fishing effort are available. A precursor to population modeling is the requirement for a clear understanding of the life cycle of the population and basic population parameters. The objectives of this study were to describe the quahog population in relation to commercial harvesting in St. Mary's Bay and to estimate preliminary mortality rates using basic fisheries techniques.

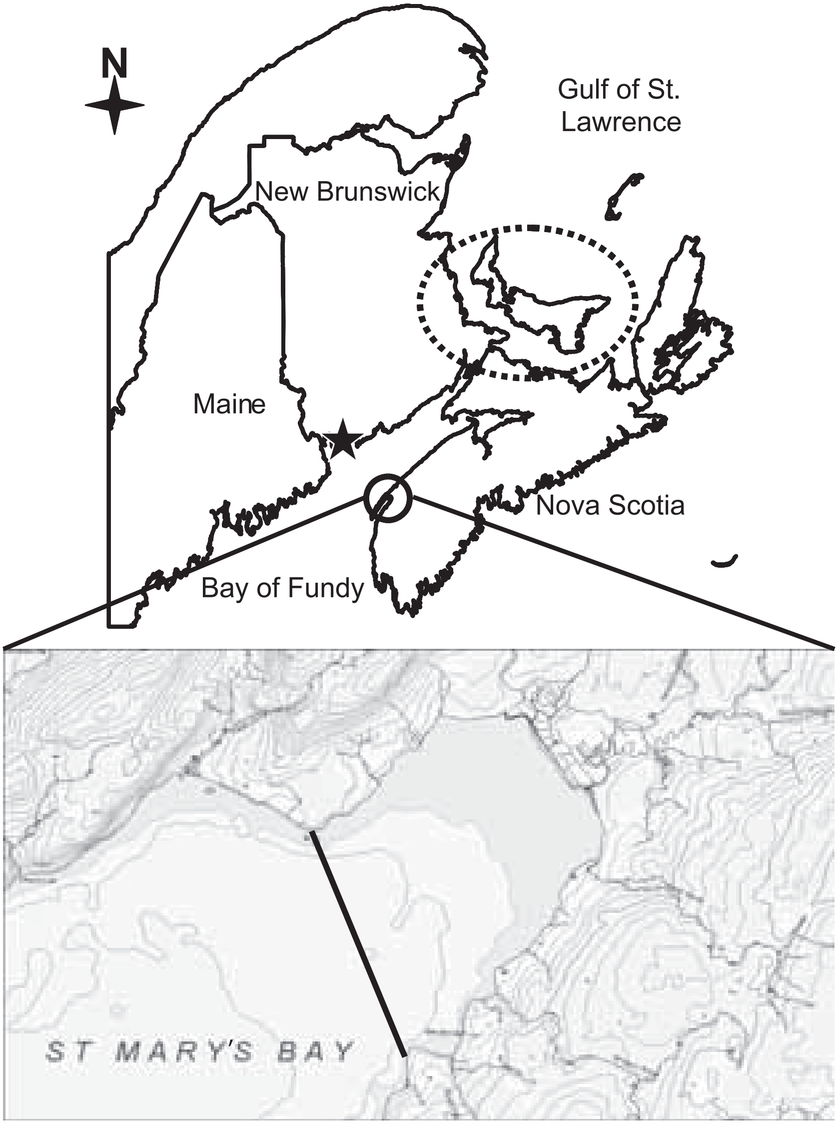

Figure 1. **Northern quahog lease located in St. Mary's Bay, Nova Scotia, Canada. Sam Orr's Pond, near St. Andrews, New Brunswick, is represented by the star. The northern quahog is typically found in the Gulf of St. Lawrence as described by dashed oval.**

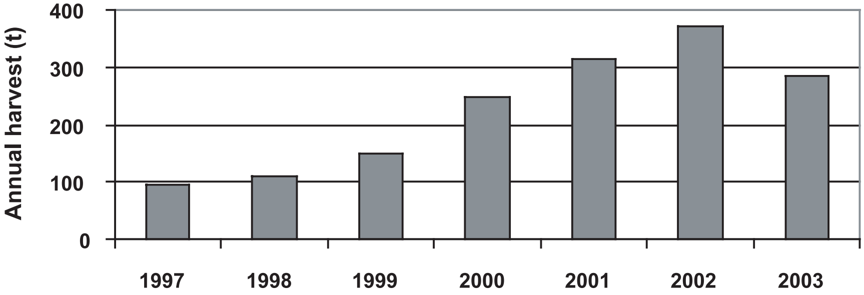

Figure 2. Annual harvest of quahogs from St. Mary's Bay, Nova Scotia, Canada.

Materials and methods

Population surveys

The study area includes the entire lease, which has a surface area of 1,682 ha with a maximum intertidal zone of 628 ha where the quahog is the dominant bivalve. The intertidal zone gradually slopes from the high to low tide mark and the substrate is largely mud and a mud-sand mixture. Pre-harvest population intertidal surveys were conducted in collaboration with IFP in June 2001 and 2002, and in May 2003. In June 2001 and 2002, surveys consisted of one sampling station per 500 × 500 m sampling unit for a total of 45 stations. A sampling grid 500 m east by 250 m south was used during the 2003 intertidal survey for a total of 95 stations. However, only the 45 traditional stations used in the June 2001 and 2002 surveys were used for survey comparisons with the May 2003 survey. During the May 2003 survey, 10% of randomly selected stations were also resampled.

At each sampling site, the upper sediment layer was collected to a depth of 25 mm from a 0.25 m^2 quadrat with small garden shovels and rinsed through a 2 mm mesh sieve, as spat and juvenile clams are typically found at this depth. Samples were bagged and frozen at –30°C until sample processing. All clams were then removed from the sediment by hand to a maximum depth of 15 cm. All clams were bagged and frozen at –30°C until sample processing. Shell length was measured to the nearest 1 mm with digital calipers. Whole frozen weight was measured to the nearest 0.1 g with a top loading digital balance.

Commercial harvest data

IFP measured the daily weight of quahogs harvested by each clam digger from 1997 to 2003. The harvested quahogs were sampled twice weekly during 2003. For each sample (n = 200), the length frequency, to the nearest millimeter, and the sample weight, to the nearest 0.01 kg, were recorded.

Age determination

Quahogs ($n = 362$) from the June 2002 survey were aged using techniques developed for surf clams, *Spisula solidissima* (Ropes and O'Brien 1979, Jones et al. 1990, Sephton and Bryan 1990). Thin sections were excised from the right-hand valve of specimens ranging from 25 to 110 mm shell length. The valve was secured to the manipulative support of an Isomet low speed geological saw and a 2 mm section was sliced between two diamond wafer cutting blades, one of the blades cutting just anterior of the umbo, yielding a highly polished thin section. The umbo side of the section was glued to a glass slide and viewed under a dissecting microscope at 25×. The number of annuli was counted within the outer and middle shell layers in the radial section from the umbo to the ventral margin.

Few quahogs older than 10 years were collected from the survey; thus the growth curve could not be properly estimated. Also, bivalves typically have highly variable growth rates whereby length frequency intervals of larger animals may encompass several age groups. Therefore, an age-length matrix coupled with the length frequency of the population was used to estimate the age composition rather than using a deterministic relationship between age and length (Hilborn and Walters 1992). Lengths for which age could not be determined were assigned to an unspecified group.

First the age-length matrix derived from the 2002 survey was used to calculate the proportion at age of quahogs for each 1 mm shell length interval. This age-length key was used in conjunction with the respective length frequencies for the June 2001 and 2002 and May 2003 population surveys. The numbers obtained for each age class were expanded to the survey area by multiplying the numbers at age by the ratio of the total survey area to the sampled area. The same age-length key was used to obtain an estimate of the age composition of the 2003 commercial harvest up to September 15, which made up most of the harvest.

Preliminary estimates of mortality rates and survival

Total instantaneous mortality rates (Z) were estimated from catch curve analyses on the yearly age compositions for the surveys and commercial harvest (Ricker 1975). This analysis assumes that recruitment and mortality rates are constant over the period determined by the number of age-groups used in the calculation. The slope of the descending limb of the natural logarithm of numbers at age is an estimate of Z. Only ages 7-10 years were used in the analysis because few individuals older than 10 years were collected.

Estimates of Z also can be obtained from catch curve analyses conducted along individual cohorts. This approach removes the assumption that cohorts are of similar abundance but requires data over several years. For the time series of pre-harvest surveys (2001-2003), a modified catch curve analysis was used. Sinclair (2001) used this approach to estimate total mortality rates of southern Gulf of St. Lawrence cod (*Gadus morhua*). The method is essentially an analysis of covariance and assumes that mortality rates in 2001-2002 and 2002-2003 were similar. We note that fishing effort over the lease area in 2001 and 2002 were relatively constant (3,145 and 3,198 harvester days respectively). This would imply that at least the fishing portion of the mortality rate may have been constant. The statistical model used was

$$\ln A_{ij} = \beta_0 + \beta_1 Y + \beta_2 I + \varepsilon$$

where A_{ij} is the number of quahogs of age i in year j; Y is a class variable indicating year class, and I is the covariate age. β_1 are year-class effects and β_2 is the estimate of total mortality in the time period.

For 2003, an exploitation rate for the fishery up to September 15 could be calculated because the harvest had been sampled. The exploitation rate was the ratio of the numbers of quahogs harvested to the numbers of quahogs estimated from the May 2003 pre-harvest survey for quahogs with shell length >45 mm. In addition, the fraction of the biomass removed by the fishery was estimated for all three years by dividing commercial landings by the estimated biomass of animals with shell-length >45 mm from spring surveys.

Finally, estimates of survival rate, S, were calculated using the standard equation $S = e^{-Z}$.

Results

Population surveys and age to market

Mean quahog densities ranged from 50 to 90 individuals per m^2 from 2001 to 2003 (Table 1). A comparison of mean densities with and without spat (quahogs with shell length ≤5 mm) suggested variable recruitment in 2002 and 2003 (Table 1). Few quahogs ≥10 years old were collected during the 2001-2003 surveys (Tables 2 and 3, Fig. 3).

The age composition of quahogs larger than 25 mm, sampled in the surveys of June 2001 and 2002 and May 2003, showed a similar age structure in the three years of the surveys (Fig. 3). Age 7 was the dominant age class. In 2003, age 7 was also the dominant age class of the commercial harvest, and the 258 t of harvested quahogs was equal to 4.2 million individuals.

Table 1. Mean quahog densities (individuals per m^2) for St. Mary's Bay, Nova Scotia, Canada.

Survey year	n	Density		Density[a] without spat		Density[b] shell length ≥30 mm	
		x	se	x	se	x	se
2001	45	54.8	13.1	n/a	n/a	46.8	12.2
2002	45	88.4	15.9	49.0	14.5	43.7	13.6
2003	45	48.3	10.5	42.0	10.4	57.7	13.4

[a] Spat were those individuals with a shell length of ≤5 mm.

[b] Data presented for comparison to Fegley (2001).

The symbols "n/a" indicate data are not available, "x" refers to the mean, and "se" refers to the standard error.

Estimates of mortality rates

Catch curve analyses of the 2001-2003 surveys (Fig. 4) as well as the fishery harvest (Fig. 5) suggested that total mortality of quahogs 7-10 years of age was high with estimates of Z ranging from 1.00 to 1.42, implying annual survival rates of only 24 to 37%. The modified catch curve analysis of survey numbers indicated no significant difference ($P < 0.05$) in year-class abundance for the 1992-1995 cohorts, quahogs aged 7-10 years in 2001-2003. The steepness of the common slope also suggested a high rate of mortality ($Z = 1.32$). As a result, a catch curve analysis was conducted using the pooled data which gave an estimate of $Z = 1.40$ equivalent to a survival rate of about 25% (Fig. 6).

While total mortality was estimated to be high, mortality attributed to commercial harvest of the lease appears to be low. The exploitation rate for 2003 was calculated to be 3.0% for quahogs ≥45 mm. For all three years, the estimated proportion of the fishable biomass taken in the fishery ranged from about 5 to 10% .

Discussion

Quahog densities in St. Mary's Bay were 3-10 times higher than commercially harvested populations in the Gulf of St. Lawrence (Landry et al. 1993). In North America, Fegley (2001) reported that 80% of density studies found relatively low population densities of 1-15 individuals per m^2 for quahogs ≥30 mm. The other studies documented densities of >500 individuals per m^2. Because the high densities described by Fegley (2001) were reflective of intensive shellfish aquaculture and rarely occur in nature, the densities observed in St. Mary's Bay (Table 1) were higher than other natural quahog populations in North America (Castagna 1984, Fegley 2001).

Table 2. Length frequency, shell length in mm, of northern quahogs from population surveys (n = 45 traditional stations) conducted in June 2001 and 2002 and in May 2003 in the lease area in St. Mary's Bay, Nova Scotia, Canada.

Shell length (mm)	2001	2002	2003
1-5	66	447	71
6-10	4	21	34
11-15	1	3	9
16-20	0	5	6
21-25	4	11	8
26-30	25	25	5
31-35	67	61	29
36-40	103	115	74
41-45	124	129	110
46-50	107	92	113
51-55	55	44	90
56-60	34	23	42
61-65	8	6	15
66-70	6	5	6
71-75	2	3	1
76-80	3	2	0
81-85	4	1	0
86-90	0	1	4
91-95	2	0	0
96-100	2	1	0
101-105	0	0	0
106-110	0	0	0
111-115	0	0	1
Total	617	995	618

Table 3. Age composition of quahogs of 25 mm and larger from population surveys (numbers expanded to the survey area) conducted in June 2001 and 2002 and May 2003 and from the 2003 commercial landings in the lease area in St. Mary's Bay, Nova Scotia, Canada.

	Population surveys (thousands)			Landings (thousands)
Age	2001	2002	2003	2003
3	2,425	2,039	1,996	0.1
4	4,369	3,949	1,955	0.2
5	22,119	24,061	12,392	19.2
6	59,755	56,786	52,854	692.8
7	107,617	101,312	104,149	1,965.2
8	46,912	44,341	46,625	807.9
9	4,523	3,256	5,442	206.5
10	2,061	3,178	1,861	110.5
11	0	0	0	0.0
12	472	1,417	630	43.4
13	472	472	157	8.0
14	0	0	0	0.0
15	0	472	0	10.1
Unspecified	6,138	472	3,305	344.8
Total	256,865	241,755	231,367	4,208.6

The unspecified group is composed of large individuals, >70 mm and equal to 11 years, for which no age assignment could be made.

The fishery appears to be exploiting quahogs of 6 years and older. Mean age to market for these quahogs was 7 years and ranged from 5 to 8 years. Populations in the Gulf of St. Lawrence reach commercial market size, shell length 50 mm, between 9 and 13 years (Landry et al. 1993) while southern populations reach market size in 2-5 years (Grizzle et al. 2001). In St. Mary's Bay, age to market appears to be faster than for populations of the Gulf of St. Lawrence. Differences in growth could be attributed to many factors including temperature, food quality and quantity, and salinity.

It is important to note that the age-length key for 2002 was used to calculate the age composition of the 2001-2003 surveys, as well as the commercial harvest in 2003 (Tables 2 and 3). This assumes that during this short period there have not been large changes in growth. The survey age-length key used for the calculations contained few large individuals,

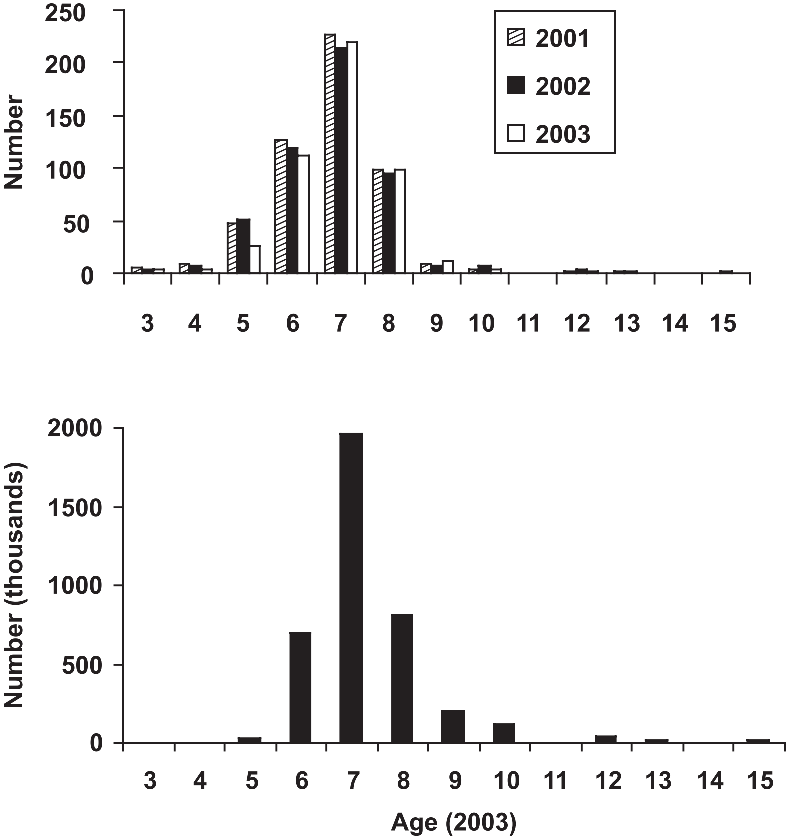

Figure 3. Age composition of quahogs from the 2001-2003 spring surveys (top), and commercial harvest of 2003 (bottom) from the lease in St. Mary's Bay, Nova Scotia, Canada.

which might have affected the estimation of the age composition of commercial harvest in 2003. However, for age groups used in the analyses (7-10 years) the age composition was based on 232 readings. There is usually little difficulty in identifying annuli for these age classes (Jones et al. 1990).

The increasing number of age 3-7 quahogs in the spring surveys for the three years of observations suggested that younger quahogs have a distribution that is larger than the survey area and that quahogs progressively "recruit" to the survey area. For St. Mary's Bay, passive transport may significantly affect the distribution of this population, particularly

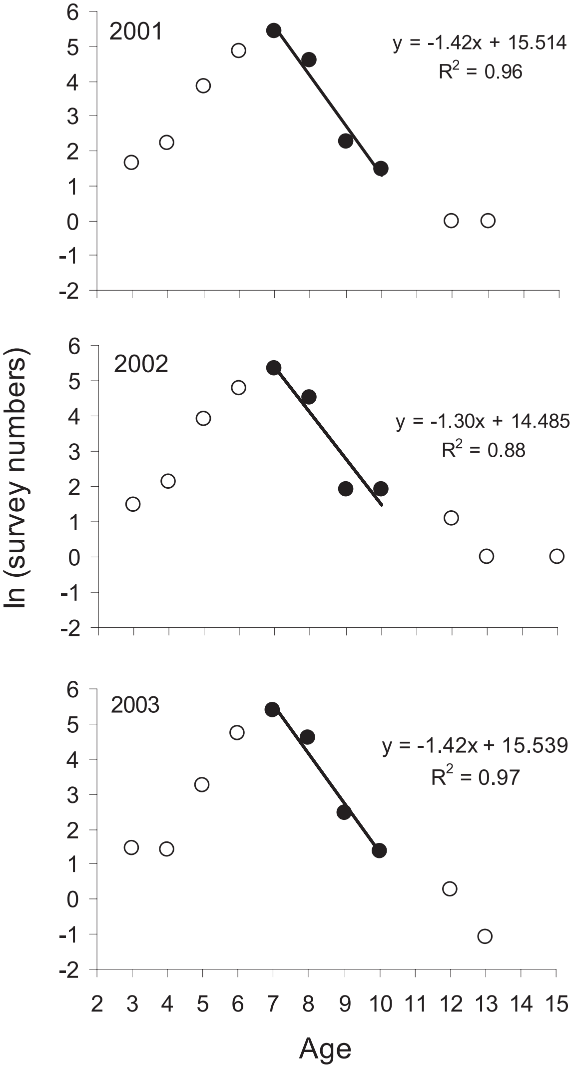

Figure 4. **Catch curves for quahogs in St. Mary's Bay from the 2001-2003 spring survey age compositions. Closed circles are data points used in the analysis; open circles were not.**

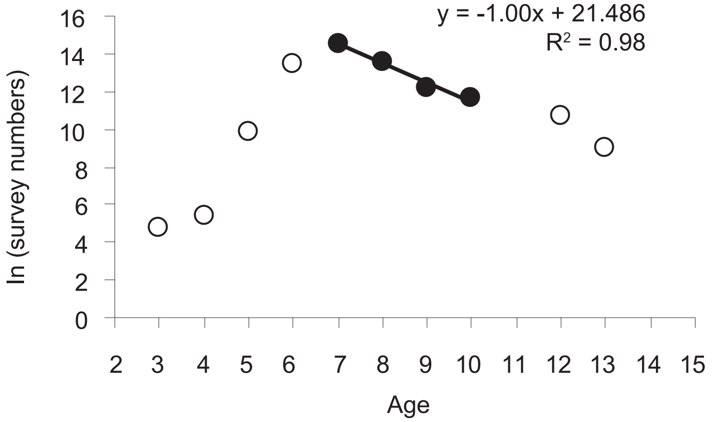

Figure 5. **Catch curves for quahogs in St. Mary's Bay from 2003 age composition of commercial harvest. Closed circles are data points used in the analysis; open circles were not.**

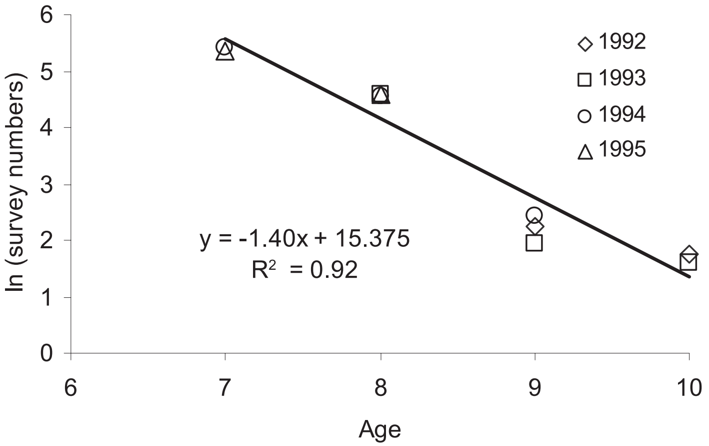

Figure 6. **Combined catch curve analysis of survey numbers along cohorts. Ages 7-10 were used in the analysis. Symbols indicate year classes.**

the younger and smaller animals. The movement of smaller quahogs may be caused by storm surges in the 7 m tidal range, which are characteristic to the area. Geospatial differences in distribution between spat, juveniles, and adults may have ecological implications to overall population fitness (Rice et al. 1989). Large concentrations of adult quahogs may also affect the recruitment of juvenile quahogs (Rice et al. 1989). Fegley (2001) has also documented that when there are large numbers of widely dispersed spawning quahogs, this can lead to lower fertilization rates.

All of the catch curve analyses of mortality rates suggested that total mortality for ages 7-10 was high. This would include both natural mortality, such as predation and disease, and the commercial harvest.

These estimates assume that the adult portion of the population is closed and thus not subject to emigration or immigration. Because there are no other known adult populations nearby, passive transport such as that hypothesized for small quahogs is considered less likely for the larger adult animals. The analyses conducted on the age-structured data from the surveys for individual years, or the 2003 harvest, also assumed constant recruitment. Survey data from 2001 to 2003 indicated that there was significant variation in recruitment in the area for year classes produced in the early 2000s. However, the analysis of covariance, which used all three years of survey data, showed that there was no significant difference in year-class strength for year classes produced in the early to mid-1990s (ages 7-10 in 2001-2003). This analysis, which took into account potential differences in year-class strength, also produced a similarly high estimate of total mortality. In summary, all estimates of Z, either from the survey or the harvest data and using various methods, were relatively high. We could not identify specific reasons to discount these estimates.

Given the apparent high total mortality, the low estimates of exploitation rate for 2003 and of the fraction of the fishable biomass taken by the fishery in 2001-2003 would imply that natural mortality on these age groups was unusually high.

Mortality rates for adult quahog populations from New England states are usually low and uniform throughout the year and rarely exceed 50% for age classes between 6 and 10 years (Kennish 1978). Predation may partly explain high mortality rates. Predation by seagulls is common over the lease area. Annual predation rates of seagulls on adult quahogs in the intertidal mudflat at Hamble Spit in Southampton, England, were estimated at 5-10 individuals per m^2 (Hibbert 1977). Large losses of adult quahogs may also be attributed to winterkill caused by ice scouring on the lease. Photographs taken of the lease area in December 2002 showed the presence of large ice cakes of 1.5 × 2 × 2 m (height × length × width). Ice cakes covered the intertidal region from January to April 2002-2004. Winterkill has been suggested for losses of large amounts of oysters (*Crassostrea virginica*) and quahogs throughout much of the

Gulf of St. Lawrence region in 2002-2003 (T. Landry, DFO, pers. comm. 2003). This phenomenon has not been quantified in Atlantic Canada but may be an important factor in the population dynamics of quahogs in St. Mary's Bay.

Low survival rates can have serious implications for the sustainability of a population. Size structure is important for the reproductive fitness of a population and in terms of fishery management. *M. mercenaria* is described as a protandrous consecutive hermaphrodite, meaning that males typically dominate the younger size classes, the sex ratio changes with age distribution, and the sex ratio of adults is not 1:1 with growth whereby males may still outnumber the females (Eversole 2001). Though separate sexes do exist, fully functional hermaphrodites are common to quahogs (Eversole 2001). In the Gulf of St. Lawrence, sexual maturity can be attained at 25 mm and 30 mm shell lengths for males and females respectively with one major spawning event usually occurring in mid-June (Landry et al. 1999). In this case, the fishery mainly harvests animals that are 45 mm and larger and the bulk of the mortality for quahogs 7 years and older appears to be largely due to causes other than exploitation. If further analyses confirm these results, this could be a natural characteristic of this population.

In conclusion, while the simple methods used are subject to a number of assumptions that need to be verified, these analyses provide a description of population structure and initial estimates of mortality for this understudied population. In our study, the limited data precluded the use of more complex models but the results underline the usefulness of basic methods to generate hypotheses about population dynamics, in this case high natural mortality. We hope that this information combined with continued sampling of the population and the fishery will lead to the use of age-structured population models, such as virtual population and statistical catch-at-age analyses, to gain a better understanding of population dynamics of quahogs in St. Mary's Bay.

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Fisheries Assessment and Management in Data-Limited Situations
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Timing of Parturition and Management of Spiny Dogfish in Washington

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Abstract

Management of the spiny dogfish (*Squalus acanthias*) fishery in the eastern North Pacific Ocean has historically been limited, and not focused on conservation of the species. Washington State Department of Fish and Wildlife (WDFW) recently adopted a new management strategy aimed specifically at protecting the spiny dogfish during the critical reproductive period. There is currently little information on the reproductive biology of the Puget Sound stocks of spiny dogfish, and much of it is anecdotal. The aim of this project is to improve the data poor nature of spiny dogfish fishery management. This paper reports some of the findings of an extensive investigation into the reproductive biology of Puget Sound spiny dogfish. The pupping season appears to be from May through November, longer than the anecdotal data indicate and much longer than current regulations were written to cover.

Introduction

The spiny dogfish (*Squalus acanthias*) fishery in the Puget Sound and Pacific Northwest waters has received little management and has been characterized by fluctuations in catch and effort. Management of this species in Puget Sound is based on poor data, managers have little local information about the species, and much of that information is anecdotal and from fishery stakeholders. Prior to 2003, spiny dogfish management goals were reduction of interactions with other fisheries, reduction of bycatch, or to conform to market practices (WDFW 2003).

In 2003, Washington Department of Fish and Wildlife (WDFW) amended the fishing regulations for setnet and set-line spiny dogfish fisheries. The 2003 amendments effectively closed the fishery in most of the Puget Sound areas during the summer months (June 16–September 15). This time frame was suggested by industry as the period when females were pupping and was adopted into the regulation. The impetus to put regulations in place was justified by the generally accepted need to practice precautionary management (FAO 1995) to conserve the species and maintain the fishery.

Recent stock assessments in the western North Atlantic have shown that the stocks are not stable, fishable biomass has greatly decreased, there is low recruitment of females, and the stocks may be fully exploited (Rago et al. 1998). Managers took dramatic steps to reduce the impact on the stocks as a whole and on large females in particular. While stocks have not had the same trend in Washington state waters, there is cause for concern. Catch rates have shown dramatic declines over the last two decades (Fig. 1) and recent stock assessments also suggest a decline (WDFW 2003). In neighboring British Columbia, stock assessments show their stocks to be stable. Given the geographic range of the stocks in the eastern North Pacific (WDFW 2003, McFarlane and King 2003), the populations are transboundary and require cooperative management between the two countries. This begins with creating accurate methods for assessing the status of these stocks, and managing accordingly.

Squalus acanthias is common, small, and easy to maintain in laboratory conditions. Literature on this species comes from many areas: North Atlantic (Rago et al 1998, Soldat 2002), North Sea (Stenberg 2002, Jones and Ugland 2001), Black Sea (Polat and Guemes 1995), and the North Pacific (Bonham 1954; Holland 1957; Ketchen 1972, 1975, 1986; Wood et al. 1979; McFarlane and Beamish 1987; Saunders and McFarlane 1993; McFarlane and King 2003). Laboratory studies with detailed examinations of the anatomy, physiology, and reproductive cycles, including endocrinology have been conducted (Tsang and Callard 1987, Koob and Callard 1999).

In 1948, Bigelow and Schroeder determined that the populations in the North Pacific were the same species as those in the North Atlantic. However, research indicates that the animals in these two areas do differ in some aspects. In the North Pacific, the spiny dogfish are longer lived, mature later and at a larger size, and they grow much larger than those

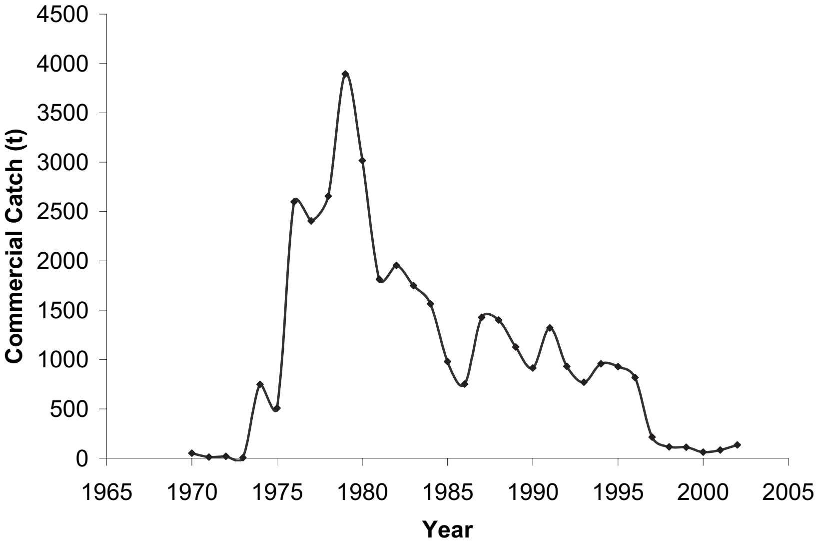

Figure 1. Commercial landings in metric tons for Puget Sound spiny dogfish.

in the North Atlantic (Ketchen 1972, 1975; McFarlane and Beamish 1987; Saunders and McFarlane 1993).

In the eastern Pacific Ocean, spiny dogfish research has focused on aging, migrations, and population dynamics (Bonham 1954, Holland 1957, Wood et al. 1979, McFarlane and Beamish 1987, Saunders and McFarlane 1993, McFarlane and King 2003) with less emphasis on life history. McFarlane and King (2003) show some animals move between Strait of Georgia and Puget Sound waters, and that the animals from the coastal tagging area (west coast Vancouver Island) are more prone to migration. Three separate stocks have been identified by McFarlane and King (2003) and WDFW (2003): coastal stocks (including Washington coast, and west coast Vancouver Island), northern (including the Strait of Georgia and the San Juan archipelago) and southern (waters from Port Townsend to the south).

Around the world, pupping and mating seasons vary by area. An in-depth analysis of reproductive biology for spiny dogfish found in British Columbia waters suggested that mating occurs from December through February and that parturition occurs October through November (Ketchen 1972). Soldat (2002) reported that pupping in the western North Atlantic occurs year-round with most activity between November and April. In waters near Sweden and Norway, pupping is reported to occur from November to December and mating from December to February (Stenberg 2002), while Jones and Ugland (2001) report pupping from September to December and fertilization (and onset of pregnancy) from October to February. It is important to note that the time of mating and the onset of pregnancy may not be closely linked if the female stores the sperm for a period of time prior to fertilization. The objective of this paper is to present results from an ongoing and in-depth study into the reproductive biology of spiny dogfish in Puget Sound. We propose that the timing of the critical reproductive events may differ from that previously reported, which has a direct impact on current management strategies.

Our investigation of the reproductive biology of the spiny dogfish is an effort to refine parameters for more accurate stock assessment and management strategies. Although this study focuses on the spiny dogfish in north Puget Sound (NPS), we also sampled spiny dogfish from the south Puget Sound (SPS), east Puget Sound (EPS), and coastal (C) areas, in an effort to compare reproductive timing and animal sizes (Fig. 2). The results presented in this paper will quantify the reproductive season for NPS spiny dogfish, contribute to a comparison of the timing of reproductive seasons with spiny dogfish from other areas, and suggest the role of this information in defining fishing seasons and harvest regulations. Information from this study will contribute to management of the spiny dogfish fishery in Puget Sound, and possibly to British Columbia management.

Materials and methods

This study has three parts: sample collection, lab analysis, and hormone analysis. Sample collections involved demographic information, size and sex distributions, and catch effort. The laboratory section included examination and measurement of reproductive tracts. The hormone component is not presented here but will appear in a future paper (M.S. thesis draft, Cindy A. Tribuzio).

Spiny dogfish samples were collected from November 2002 to October 2003. Fish were sampled from the catch of a commercial bottom trawl fisherman in the southern Strait of Georgia, Washington (48ºN, 123ºW) between 73 and 128 meters (40-70 fathoms) depth (Fig. 2). Up to 25 spiny dogfish were taken from the trawl catch on each sampling date and maintained in an onboard live tank with flowing seawater until brought to the dock. This was not considered a random subsample, as the vagaries of the collection effort did not allow for guarantee of randomization. Non-random samples may induce size bias, and while the captain was instructed to randomly sub-sample animals, it is the nature of the fishery to take the largest, and it is possible that our samples are upwardly biased. Given the nature of the project objectives, the non-random sampling and possible

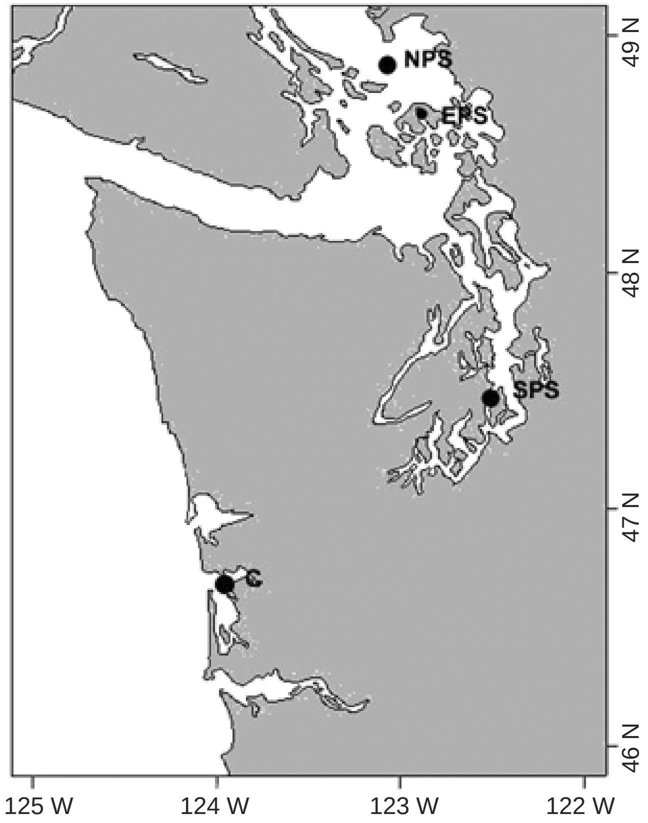

Figure 2. Map of fishing areas for this study. NPS = north Puget Sound, EPS = east Puget Sound, SPS = south Puget Sound, and C = coastal.

bias are inconsequential. Fish were processed at the Sea-K Warehouse in Blaine, Washington. Dockside processing allowed WDFW samplers to collect the data from fresh fish without requiring samplers to be onboard the vessel and without requiring samples to be frozen.

Each animal was weighed whole and data recorded for pre-caudal length (PCL), fork length (FL), and total length-natural (TL_{nat}) (Fig. 3). To compare length distributions to previous studies 146 spiny dogfish were randomly sampled from the same fishing ground and measured for TL_{nat} and TL_{ext} (Fig. 3). Male clasper inner length (CIL) was also measured. Blood

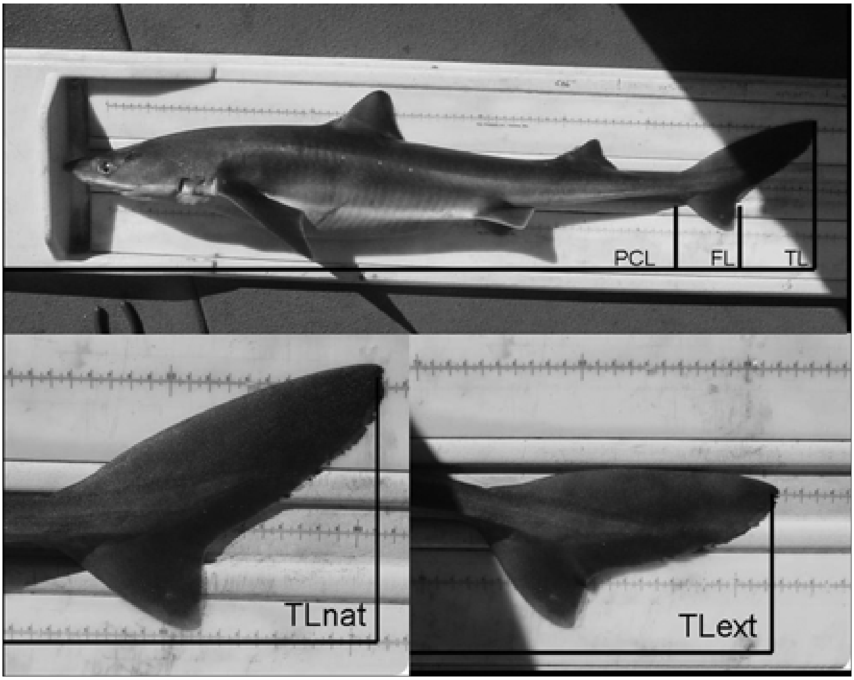

Figure 3. Length measurements for whole animal. Lower left, tail in natural position for measuring TL_{nat}. Lower right, tail extended to line up with body, for measuring TL_{ext}.

was collected (~5ml), in a 15 ml centrifuge tube, from the caudal vein by removing the caudal fin and catching the blood flowing from vein. These tubes were kept cold overnight to allow the blood to clot and the serum to separate out. The serum was drawn from each blood sample and divided into 3 separate 1.5 ml micro-centrifuge tubes, prior to shipping. Samples were kept on ice, not frozen, and shipped overnight to the University of Washington for further analysis.

The second dorsal spine was removed, from tip to vertebrae, for age determination. Maturity stages of the reproductive tracts (based on Stehmann 2002) were recorded, and the entire female reproductive tract or testes (males) were removed and kept on ice.

Female reproductive tracts were measured for length, width (both to the nearest millimeter) and weight (to the nearest tenth of a gram) of ovaries, oviducts, oviducal glands, and uteri. In adult females (Stehmann 2002, stages D-G), the diameters of the developing ova within the ovary were measured to the nearest millimeter. In pregnant females with

Table 1. Comparison of pregnancy stages from a previous study and this study.

Tsang and Callard 1987		This study	
Stage	Description	Stage	Description
A	Candles, embryos up to 3.5 cm, follicles 3-10 mm	A	Candle present, embryo not measurable
B	Embryos 3-10 cm, follicles 17-20 mm	B	Candle present, embryo measurable
C	Embryos 17-25 cm, follicles 28-34 mm	C	No candle, embryo TL < 10 cm
D	Embryos >25 cm, follicles 32-38 mm	D	Embryo TL 10.1-15 cm
		E	Embryo TL 15.1-17.5 cm
		F	Embryo TL 17.6-20.5 cm
		G1	Embryo TL 20.6-23 cm
		G2	Embryo TL 22.5-24.5 cm
		H	No external yolk sac embryo TL 10.1-15 cm
		I	Post-partum

TL = total length.

candles, the candles were weighed and the number of eggs counted. For those females with pups, each pup was weighed with and without the yolk sac, sexed, and measured for PCL, FL, and TL_{ext}. For male spiny dogfish, the testes were weighed and measured for length and width.

Maturity classifications in the field were based on Stehmann (2003). However, the reproductive analysis in this project required a fine scale. We initially used a scale proposed by Tsang and Callard (1987), with 4 reproductive stages and did not account for post-natal females (Table 1). Table 1 shows the stages used for this study. The stages of pregnancy were based on the presence of candles with embryos too small to measure with the naked eye, those with embryos in candles that were measurable, and those females with embryos free of the candle in the uterus based on size. Given the extended gestation in this species and the short period between pregnancies we classified all mature females with empty uteri as spent/post-natal or pre-fertilization (stage I).

Samples from outside the above sample area were also collected for comparison during the summer of 2003 (Fig. 2). For comparison, spiny dogfish from Willapa Bay, Washington (C) and south Puget Sound (SPS) were collected as bycatch in ongoing research by WDFW. Targeted hook and line fishing for spiny dogfish was conducted near Orcas Island, Washington (EPS). For most of these samples the only measurements taken

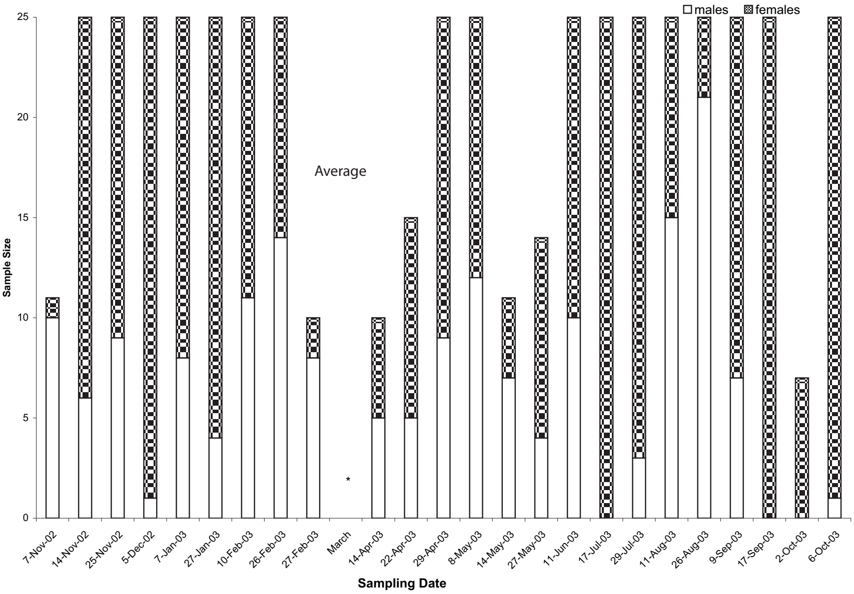

Figure 4. Total catch for each sampling date. Star (*) represents when no sampling occurred. N = 471.

were sex, lengths, weight, maturity, and blood, while a few were sent to the lab for further measurements.

Results

This paper reports on results from the first 11 months of a 12-month sampling program. Sampling catch rates were relatively constant throughout the sampling period, with the exception of early spring and early summer (Fig. 4). The sampling plan was for two trips each month, approximately every other week, with a targeted sample number of 25 fish for each trip. Due to weather (wind and tides) fishing did not always occur regularly. Inclement windy weather was the primary factor contributing to the low fishing effort during the period 26 February 2003 and 14 April 2003, resulting in no samples in March. Fishing was not affected by weather during the period from 14 May 2003 to 11 June 2003, but catches of spiny dogfish were lower during that period.

Numbers and maturities of males and females varied between sampling dates. Mature males were caught consistently from 7 November 2002 to 27 February 2003 (Fig. 5). Between 27 February 2003 and 29

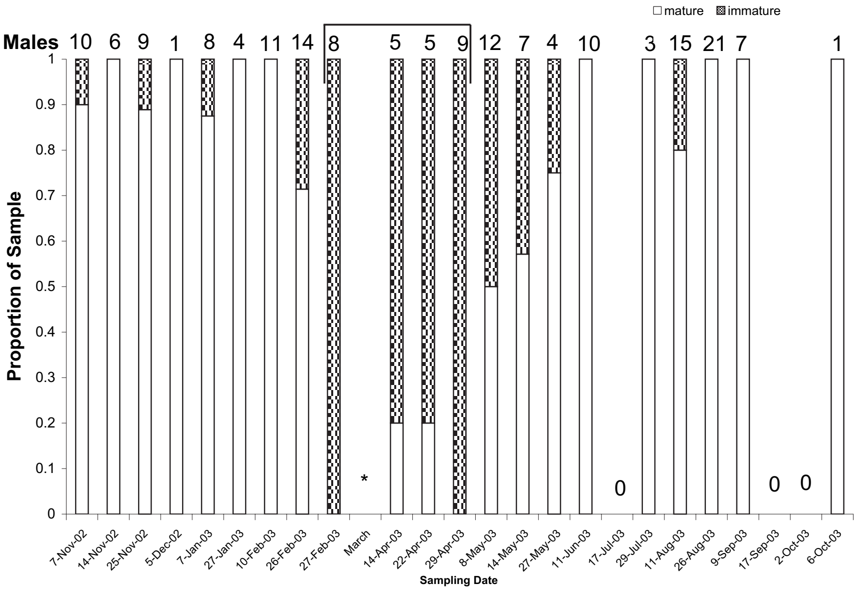

Figure 5. Male catches by month for mature and immature animals. Star (*) represents when no sampling occurred, numbers above the bars represent sample size. N = 170.

April 2003, the maturity of the males sampled changed significantly, with immature animals becoming prominent (*t*-test, alpha = 0.05, *P*-value = 0.002). In the early summer months (May and June) mature males become more prominent in the catch and peaked at 26 Aug 2003. The composition of the female catch varied more than the males throughout the sampling period (Fig. 6). Immature females made up most of the female catch 7 November 2002 to 5 December 2002, 26 February 2003 to 14 May 2003, 11 June 2003, and 29 July 2003 to 9 September 2003. At no point was the female catch composed entirely of mature animals, but between 26 February 2003 and 14 May 2003, and between 29 July 2003 and 9 September 2003, there was a significant increase in the proportion of immature females sampled (*t*-test, alpha = 0.05, *P*-value < 0.001).

Reproductive seasonality

The relative frequency of mature females in each stage for each month sampled shows that post-natal females (stage I, Fig. 7) were present much of the year, but were absent most of February (after 10 February 2003,

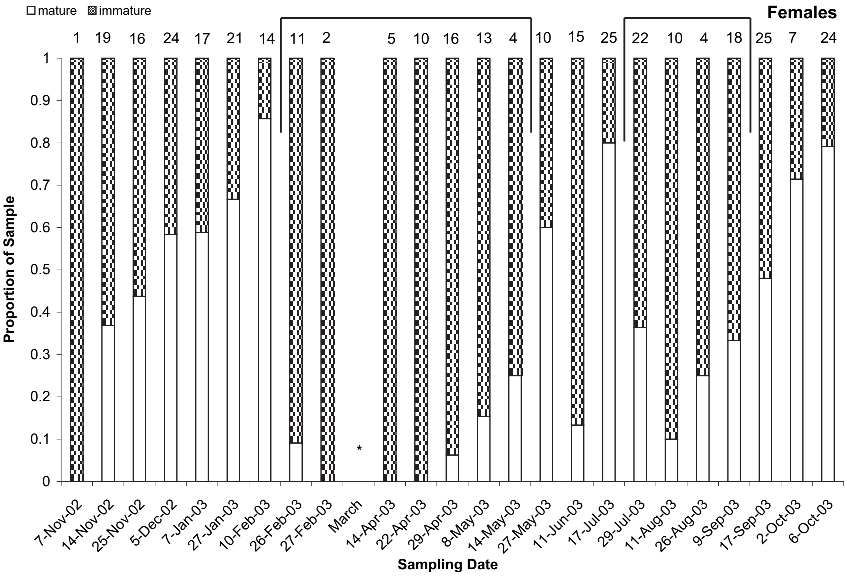

Figure 6. Female catches by month for mature and immature animals. Star (*) represents when no sampling occurred, numbers above the bars represent sample size. N = 333.

only two mature females were sampled) and April (one mature female was sampled during the last sampling event in April; no samples were collected in March). However, this was also the time of least intensive sampling. Stage I females were caught with the greatest frequency (the entire mature female catch) in June and September, and made up a large portion of the mature female catch in July and August. Late stage pregnancy females (stages G and H) were also present in July and August. The least frequent catches of stages, G-H, were in January through May, where the earlier stages of pregnancy were seen most often (stages A-B). Females may begin pupping in June and continue through November, while ovulation and fertilization occur November through February.

With the gestation occurring over about 22 months, in the population there are two groups of pregnant females: those in the first year of pregnancy and those in the second year. The graph of embryo size against sampling date shows this trend (Fig. 8). There were no females caught that were in the first year of pregnancy, in the months of August and September (in September, all females were stage I). In November and December 2002 and October 2003, embryos of three stages are seen

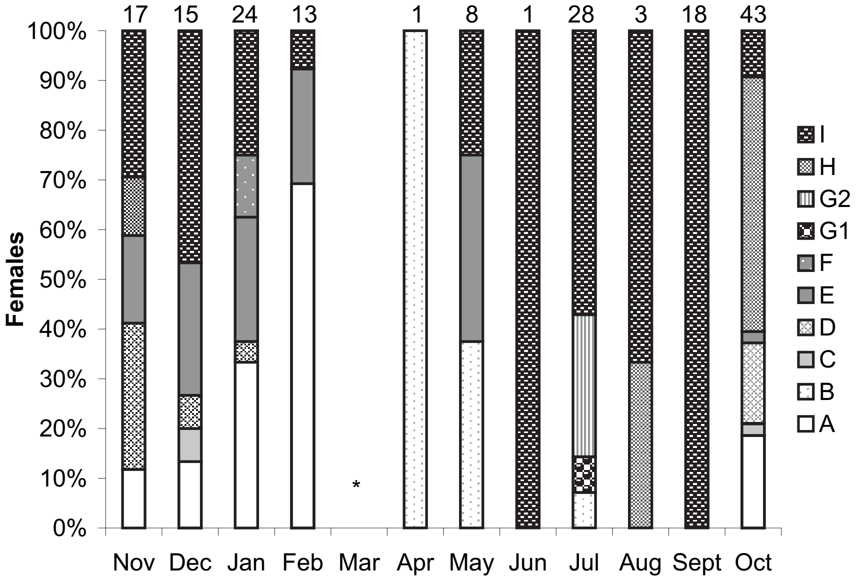

Figure 7. Frequency of pregnancy stages for mature females. Stage A represents the earliest stages of pregnancy through stage I which represents spent females. N = 171.

simultaneously, indicating some temporal overlap between pupping and fertilization.

Comparisons to other areas

Three areas sampled around Puget Sound and coastal Washington state waters were compared for differences in size and stage of maturity. For females all three areas were sampled in July and August, and just the north Puget Sound (NPS) and coastal (C) areas in September. In July, the sizes of the females caught were significantly different in the three areas (*t*-test, alpha = 0.05, all *P*-values < 0.004), but in August and September, the sizes were not significantly different from one another (*t*-test, alpha = 0.05, all *P*-values > 0.166). Males were encountered only in NPS and C and only in July and August. As with females, the sizes were significantly different between the two areas in July (*t*-test, alpha = 0.05, *P*-value = 0.01) but not in August (*t*-test, alpha = 0.05, *P*-value = 0.105). Also, tests between months within the same area showed that in NPS and C, different sized females were caught in July than in August (*t*-test, alpha = 0.05, *P*-values < 0.001), and between August and September in NPS (*t*-test, alpha

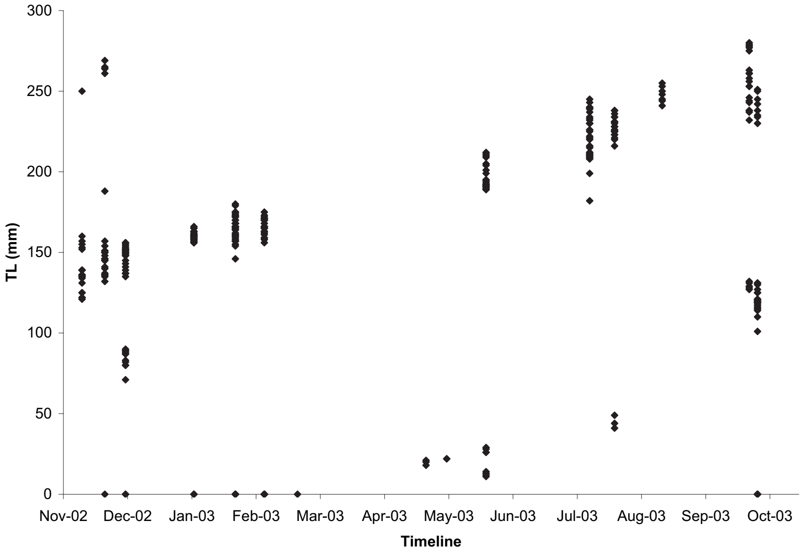

Figure 8. Embryo lengths by catch date. At least two distinct size classes are seen throughout the time frame of sampling. Zero values represent embryos that are still in the candle stage and are not measurable to the naked eye. N = 370.

= 0.05, *P*-value < 0.01). Coastal males caught in July and August were also significantly different in size (*t*-test, alpha = 0.05, *P*-value < 0.01).

The relative frequencies of females in each stage of the reproductive cycle were compared for the females in all three areas in the month of July (Fig. 9). A Mann-Whitney *U* test for ordinal data was used to test the null hypothesis that the regions have the same reproductive stage frequencies and thus the same timing of reproductive events. The null hypothesis was rejected for all three tests conducted (alpha = 0.05, all *P*-values < 0.001), and the three regions were all significantly different from each other in the frequencies of females in reproductive stages.

Discussion

The principal consideration in the WDFW 2003 management plan was timing of parturition. The NPS samples indicate that there may be a prolonged pupping season, which contrasts with earlier studies in nearby areas. Ketchen (1972) reported that pupping in British Columbia occurs in October and November and breeding December through February. At

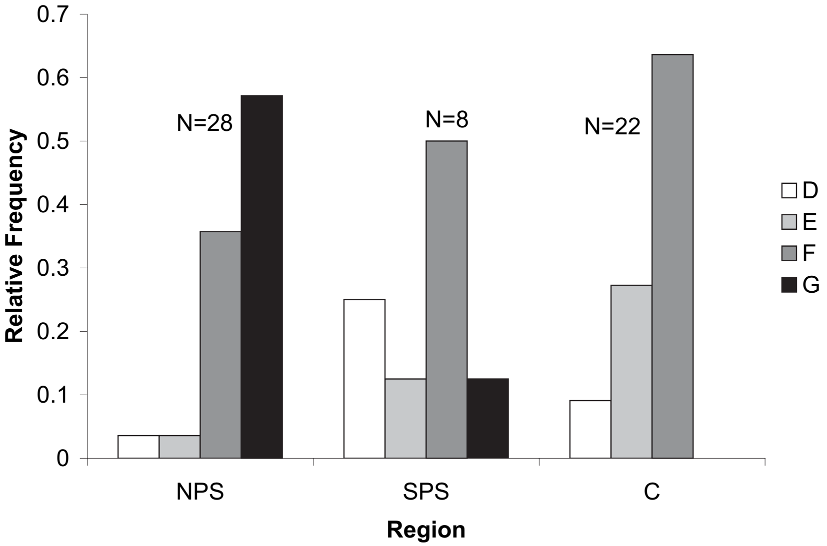

Figure 9. Comparison of uterine (pregnancy) stages in three different sampling areas. D = early pregnancy, E = mid pregnancy, F = late, and G = post partum (Stehmann 2002). NPS = north Puget Sound, SPS = south Puget Sound, C = coastal.

public aquariums in the Pacific Northwest, spiny dogfish show different reproductive timing. Pups have been found in December and January at the Seattle Aquarium, and records at the Point Defiance Zoo and Aquarium show pupping roughly March through June (Jeff Christiansen, Seattle Aquarium, pers. comm., September 2003; John Rupp, Point Defiance Zoo and Aquarium, Tacoma, pers. comm., September 2003). The animals on display at both aquariums were from stocks in Puget Sound. The Oregon Coast Aquarium, with spiny dogfish from coastal stocks near Newport, Oregon, reports pupping for all months except March, July, August, and November with peaks in May and October. Evidence of mating (damage to caudal and pectoral fins from biting males) has been recorded in March, June, July, and September (Colleen Green, Oregon Coast Aquarium, Newport, pers. comm., September 2003). Evidence of mating, however, is not necessarily indicative of fertilization and pregnancy; there may be a time lag due to the possibility of storage of sperm in the oviducal gland for a significant period of time. This phenomenon has not been studied in spiny dogfish.

Our study suggests that pupping occurs from May to November for the NPS stocks and ovulation and fertilization from October to January. It is also possible that the spiny dogfish exhibit a year-round reproductive seasonality with peaks in activity. Data on reproductive females were lacking between much of February through late April, and more intense sampling during that period may provide more insight into the seasonal pattern. Most reports for spiny dogfish are that they have either a specific seasonality or a prolonged seasonality to their reproductive cycle.

Females in the earliest stages of pregnancy (stage A: eggs are still contained within the candles in utero and the embryos are too small to be measured with the naked eye) were encountered from November through February. Jones and Ugland (2001) estimated the candle stage to last 13 months and Ketchen (1972) estimated length of embryos at the end of the first year to be about 14-15 cm. In this study, we consider the candle stage in two parts (Table 1): one where the embryo cannot be measured and the other where it can be measured by the naked eye. We estimate that the candle stage lasts less than one year and that stage A lasts about 1 month. The average monthly growth rate estimate for the first year is 11.67-12.50 mm (assuming that growth is constant during that year), based on Ketchen (1972). The smallest embryo we found not contained in a candle was 71 mm TL_{ext} and the largest within a candle was 49 mm TL_{ext}. Based on these observations and the proposed growth rate, the entire candle stage lasts somewhere between 3.9 months and 6.1 months. The smallest measurable embryos still in the candle were about 12.5 mm, which takes an estimated 1-1.07 months to achieve. We can assume that females in stage A of pregnancy have been pregnant for only about 1 month, thus those females encountered in stage A during November may have ovulated and fertilized in October, and those encountered in February became pregnant in January. Since samples were not taken in March, we were unable to determine if this period extends through February. No stage A females were encountered in April, indicating ovulation and fertilization were completed by March.

With the gestation being about 22 months long, the ovaries must continuously develop the next cohort of eggs instead of having a resting period between pregnancies. The next crop of eggs is then developing for the subsequent pregnancy while the current pregnancy is ongoing. After the female has pupped, the eggs finish development, ovulate, are fertilized, and move to the uteri. Maximum ova diameters are seen in September and November (prior to the final sampling event in October), and again in February. Given the broad temporal span of the pupping season, the measurements of ova diameter from spent females would be expected to vary over the time period. However, the variation in the size through time appears to fall into two groups of development, suggesting two broad and overlapping pupping seasons; the first being in the early summer and the next in the fall.

As with the developing ova, the developing embryos can be used to support the timing of reproductive events. Data for embryonic growth show two distinct groups of development, which are explained by the almost two-year gestation. Pregnant, mature females are either in their first or second year of gestation. Currently there is no evidence to suggest that females segregate by year of gestation. However, within these two-year classes of gestation, there does appear to be variation in the degree of embryo development, suggesting more broad timing of fertilization (Fig. 8).

Spiny dogfish harvest encompasses areas outside the NPS, so management strategies will need to be flexible enough to encompass the timing of reproductive events in the various areas. Comparisons between fishing areas in south Puget Sound (SPS) and coastal (C) waters were made. These comparisons did suggest differences in the timing of reproductive events in the three areas during the month of July. This relationship will be examined further in future reports.

This study was undertaken to investigate the timing of critical reproductive events to assist with development of management. The current management plan closes the fishery during the time period perceived to be when the females are most vulnerable. This paper is the first in a series that examines these events as well as reproductive physiology, endocrinology, and embryonic development. This paper specifically reports on the timing of pupping and fertilization, which is more broad than expected. The pupping season appears to extend well past the time frame over which WDFW enacted fishing closures, and is different from published data on nearby British Columbia populations. The data presented are for a one-year period. We have no evidence that year-to-year variability is minimal, so that the 2002-2003 data may not be representative. We suggest extending this study and refining the methods based on what was learned during this first year if more precise dates are needed.

Acknowledgments

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Developing Assessments and Performance Indicators for a Small-Scale Temperate Reef Fish Fishery

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Abstract

In Australia, the development of live fish markets in the early 1990s created strong demand for temperate reef fish species, particularly banded morwong (*Cheilodactylus spectabilis*). The fishery expanded rapidly over a very short period but has subsequently undergone a marked decline. Several management controls have been progressively introduced, including size limits, seasonal closures, and limited entry. Only simple performance indicators based on catch and catch rate trends have been utilized to monitor stocks.

Banded morwong are sedentary and appear to have a depth-structured sex and size distribution. They are long-lived (>80 years), and growth rates and maximum sizes are distinctly different for males and females. These life-history characteristics and the likely population structuring at small spatial scales have marked consequences for stock assessment.

A variety of simple assessment approaches, including catch rate standardization, yield-per-recruit and spawning biomass-per-recruit analyses, catch curve analysis, and biological indicators (median size and age, sex ratio) have been examined. These methods proved inconclusive in indicating whether current fishing levels are sustainable, or based on continued reduction of accumulated biomass and/or serial depletion of spatially structured populations. General uncertainty regarding data quality from the commercial fishery and spatial representation of biological data within the populations are of concern.

Because data intensive assessment techniques cannot be justified for this small-scale fishery, we propose the development of an operating model that can be used to evaluate whether simple biological and/or fishery indicators can assist performance monitoring and address issues of sustainability.

Introduction

The development of live-fish fisheries in Australia during the 1990s has placed increased pressure on tropical and temperate reef fish populations, with fisheries expanding rapidly during the mid to late 1990s (Rimmer and Franklin 1997). Recent annual production from the tropical fisheries has exceeded 1,500 metric tons (QDPI 2002), much of which is exported to Asian markets. By contrast, catches from the temperate fisheries, principally banded morwong (*Cheilodactylus spectabilis*) and wrasse (*Notolabrus* spp.), are low, in the order of 200 t, and service domestic live-fish markets.

Typically reef fish populations are spatially structured and reef fish fisheries tend to be small in size. This combination makes the application of data intensive assessment techniques difficult to justify both practically and economically. Using banded morwong as a case study, we examine the current stock assessment approaches applied in Tasmania, Australia. First, we evaluate catch and effort data and then investigate alternative assessments and reference points based on available biological information. Industry perceptions about resource status, fish behavior, and fishery developments have also been canvassed to supplement biological and fishery information. Our data, however, are patchy in space and time and present significant challenges when applied to assessing stock status.

Banded morwong fishery

The live-fish fishery for banded morwong is a coastal gillnet fishery off eastern Tasmania, with catches concentrated off the central and southeast coasts (Fig. 1). The fishery expanded rapidly in the early 1990s and reported catches peaked at almost 150 t in 1993-1994 (November-October fishing year, Fig. 2). Between 1994-1995 and 1999-2000, annual production declined steadily to below 40 t but has since stabilized at around 50 t, with catches generally tracking changes in effort.

Fishers generally operate out of dinghies or small vessels and target banded morwong using a fleet of large mesh gillnets (130-140 mm) over exposed rocky reefs. To minimize effects of barotrauma and thus maximize fish survival, fishing is largely restricted to maximum depths of about 25 m even though the species occurs to greater depths.

Despite low production and value, currently less than AU$ 0.7 million per year, the fishery is highly regulated. Minimum and maximum size lim-

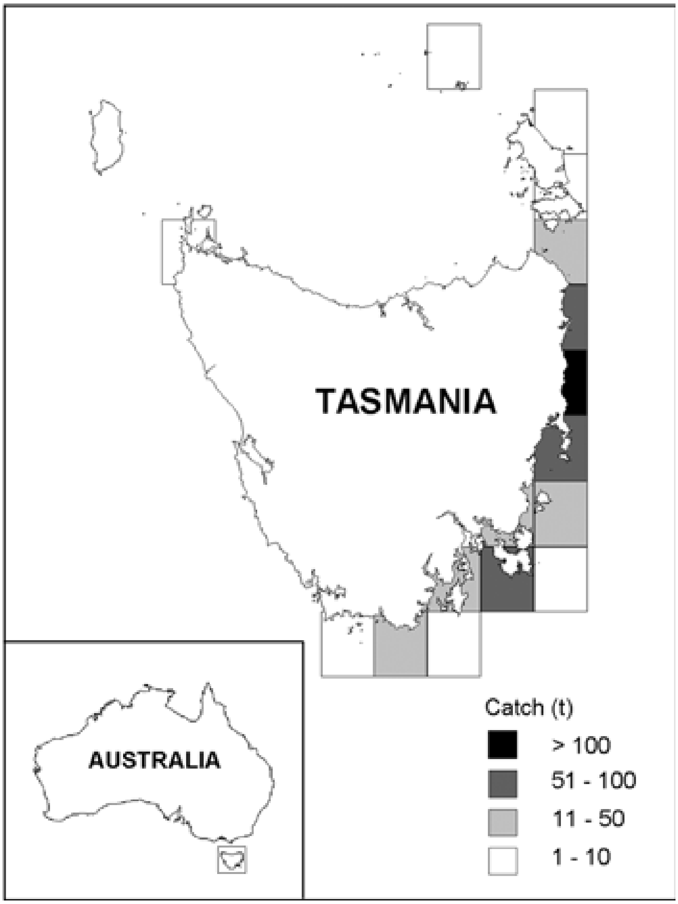

Figure 1. Map of Tasmania indicating total catches of banded morwong by 30 nautical mile fishing block summed over the period 1995-1996 to 2002-2003.

its (330 and 430 mm fork length, FL) were introduced in 1994 to protect large adults and permit spawning prior to recruitment to the fishery, in addition to matching market size requirements. In 1998 both limits were increased by 30 mm after it became apparent that they offered minimal protection to mature females and the lower limit was set close to the size at maturity (Murphy and Lyle 1999). A two-month closed season during the peak spawning period (March and April) was introduced in 1995 and has remained in place since that time. Interim licensing arrangements were implemented in 1996, with around 90 live-fish endorsements issued,

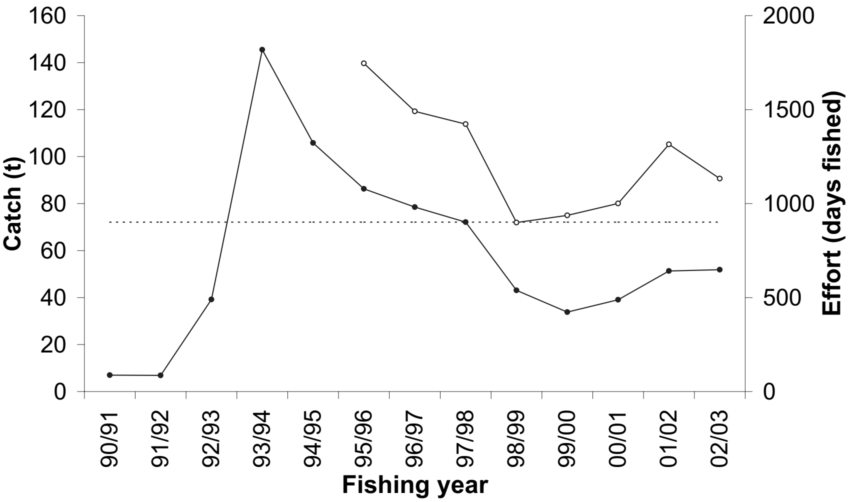

Figure 2. Annually reported catch (filled circles, in metric tons) and total effort (open circles, in days fished) for banded morwong in Tasmania from 1990-1991 to 2002-2003. Daily effort data have been reported only since 1995-1996. Dotted line represents minimum catch reference point level.

and replaced by a specific banded morwong license in 1998. There are currently 29 license holders.

Management of the fishery falls under the Tasmanian Scalefish Fishery Management Plan (DPIF 1998). The plan contains a series of performance indicators and reference points that are applied generically to all species. For routine assessments, annual summaries of catch, effort, and catch rates are evaluated against a set of reference points (Lyle et al. 2004). While target reference points are not part of this strategy, limit reference points or trigger points have been defined as levels or rates of change that are considered to be outside the normal variation of the stocks and the fishery. Limit reference points are reached when one or more of the following criteria are met:

- Total catch, fishing effort, or catch rates are outside levels of reference years;
- Total catch declines or increases in a year by more than 30% from the previous year;

- A significant change occurs in biological characteristics, e.g., size or age composition; and
- Any other indications of stock stress.

These reference points provide a framework against which the performance of the fishery is assessed and, if necessary, flag the need for management action. However, while the catch and effort reference points are clearly defined, the other definitions remain vague and leave much room for interpretation. In addition, apart from the implementation of a review process, management responses are not formally defined if reference points are reached.

Banded morwong life-history characteristics

Banded morwong is a large temperate reef fish that occurs around southeastern Australia and New Zealand to depths of at least 50 m (Gomon et al. 1994). Tagging studies have indicated that movement of juvenile and adult fish is very limited, implying that individuals remain largely site-attached (Murphy and Lyle 1999). Size- and sex-based population structuring by depth has also been observed in banded morwong, with females and juveniles more prevalent in the shallow sections of the reef and males tending to dominate deeper reef regions (McCormick 1989a). Adults display a complex spawning behavior, with males exhibiting territorial behavior and occupying the same area of reef over periods of several years (McCormick 1989b).

The species demonstrates an unusual combination of very fast initial growth, early age at maturity, and long life expectancy, with maximum ages for males and females of over 80 years (Murphy and Lyle 1999). In addition, growth rates and maximum sizes are distinctly different for the two sexes (Fig. 3). Growth in females is relatively rapid for the first 5-6 years to a size of about 350 mm, after which it slows dramatically. By contrast, males grow relatively rapidly for the first 10-12 years, up to about 450 mm, before growth slows. These growth characteristics in conjunction with spatial structuring have marked consequences for the assessment of this species.

Catch and effort assessment

Previous routine assessments have only been based on analysis of catch and effort information derived from compulsory logbook returns. Catch returns provide daily summaries of fishing operations, including method, location (based on 30×30 nautical mile fishing blocks), fishing depth, effort, catch weights, and whether seal interference had occurred. Catch returns are unverified and accuracy is uncertain.

Since 1995-1996 catches have, to a large degree, tracked changes in effort (Fig. 2). After apparent high fishing pressure early in the fishery,

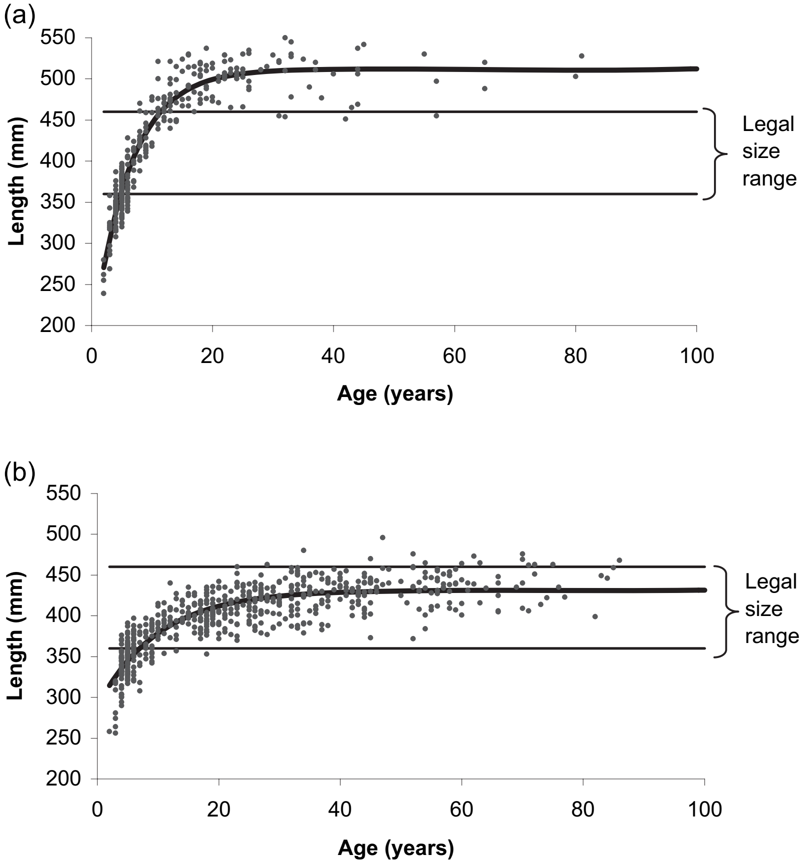

Figure 3. Size-at-age and von Bertalanffy growth functions for (a) male (*N* = 333), and (b) female banded morwong (*N* = 614) in 1996-1997 (after Murphy and Lyle 1999).

reported catches declined steadily to 1999-2000, but stabilized and even recovered slightly thereafter. Limit reference points based on catch have been breached every year since the management plan was implemented in 1998 (Fig. 2). Unstandardized catch rates based on geometric mean have generally declined since 1995-1996 and remained below reference levels each year since 1999-2000 (Fig. 4). Initial declines in catch and catch rates occurred at a time when there was ample capacity to take larger catches and markets were strong. This led resource managers and researchers to express concern about the status of the stocks, and biological and catch sampling was implemented. Recent stability is difficult to interpret in terms of stock condition, but may be linked to market demand which has become increasingly important in determining harvest levels, with fish buyers often placing fishers on catch limits.

Catch rate standardization using generalized linear models (GLM) has also been attempted to reduce the impact of obscuring effects such as region, depth, season, or skipper (Kimura 1981, 1988). The standardization was conducted on an annual base statewide and for all reports by fishers that had been in the fishery for at least two years and caught a median annual catch of at least one metric ton. The optimal model, fitted by assuming a lognormal distribution with an identity link, was based on Akaike's and the Bayesian information criterion (AIC and BIC; Burnham and Anderson 1998) and provided by:

$$\text{Ln CPUE} = \text{Constant} + \text{year} + \text{season} + \text{vessel} + \text{skipper} + \text{fishing block} \\ + \text{depth} + \text{seal interference} + \text{fishing block} \times \text{seal interference}$$

Standardization suggested a greater degree of stability by comparison with unstandardized catch rates, with catch rates remaining above reference levels (Fig. 4). The catch rate analyses generally support industry perceptions that fishing has had negative effects on the stocks, but that in recent years catch rates have stabilized or even improved.

Catch rates, however, offer limited insights into the sustainability of current harvest levels and stock status. There are several reasons for this. First, catch rates ignore the unique biological characteristics of the species. Second, we are dealing with spatially structured populations and the spatial scale for reporting may be too large to detect localized and/or serial depletions. In addition, fishers operate over relatively wide areas of coast to maintain catches and this behavior may mask localized depletion effects and result in apparent hyper-stability in catch rates. Third, data reliability is questionable.

In relation to data reliability, discussions with industry have identified a number of issues. For instance, catches are believed to have been substantially overstated before the introduction of live-fish endorsements in 1996, and data quality is considered variable at least up until approximately 2000. Furthermore, the "meaning" of catch and effort data has changed as a result of an increasing incidence of seal interference, resulting in loss of catch and damage to gear. Fishing practices have been altered as fishers have attempted to minimize these impacts, mainly by dispersing gear over large areas. In any case, fishing activity is not always reported for days when seal interactions are severe partly out of concern it could indicate poor catch rates and elicit a management response. Data quality and reporting consistency, along with the impacts of seal interactions, are therefore important factors affecting any interpretation of fishery-dependent information in this fishery.

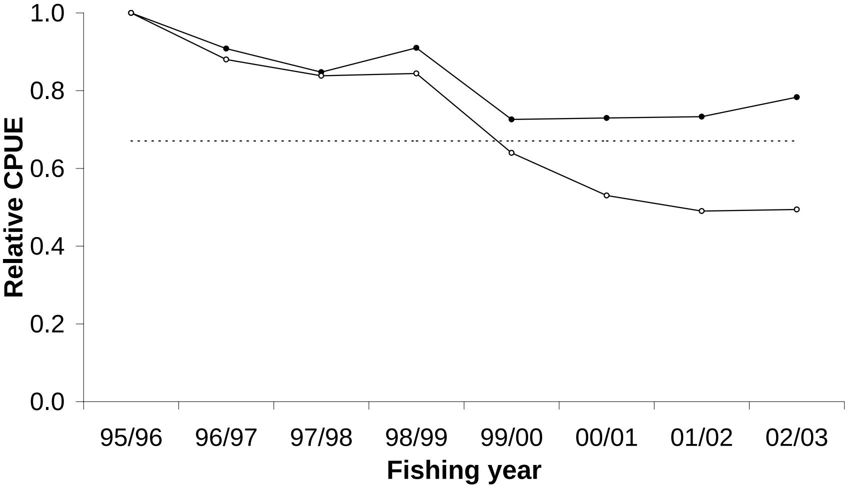

Figure 4. **Standardized (filled circles) and unstandardized catch rates (geometric mean, open circles) for banded morwong in Tasmania from 1995-1996 to 2002-2003. See text for standardization model. Given are values relative to 1995-1996. Dotted line represents minimum catch rates reference point level.**

Alternative assessments

Biological and catch sampling were undertaken between 1996 and 1997 by Murphy and Lyle (1999) and resumed in 2001 because of concerns about the stock status. These data provide the basis for assessments such as yield-per-recruit, spawning biomass-per-recruit, catch curve analysis, and examination of simple biological indicators.

Yield-per-recruit and spawning biomass-per-recruit analyses

The yield-per-recruit and spawning biomass-per-recruit have been calculated using the current size limits (360-460 mm) and a natural mortality of $M = 0.05$, derived by using the Sparre et al. (1989) approximation. These analyses highlight the dilemma faced when determining fishing reference levels for species with strong sex-based differences in growth parameters (Fig. 5). For instance, the reference level $F_{0.1}$ (Gulland and Boerema 1973) from the yield-per-recruit analysis was estimated as 0.25 for males but just 0.11 for females. Females recruit to the fishery at around six years of age and typically remain vulnerable for the remainder of their lives. On the other hand, males remain vulnerable to the fishery on average between four and ten years of age and can thus sustain much higher levels of fishing mortality. In practice, reference levels are probably conservative because the effect of lower selectivity by the fishing gear toward the upper end of the keyhole size limit has not been taken into account.

While $F_{0.1}$ often has been used as a target to minimize growth overfishing, it does account for recruitment (e.g., Clark 1991, 1993; Mace and Sissenwine 1993). To address recruitment overfishing for stocks where little is known about the stock-recruitment relationships and resilience, the fishing mortality rate $F_{30\%}$, which reduces spawning biomass-per-recruit to 30% of the unfished level, has been applied as a recruitment overfishing limit, and $F_{40\%}$ as a target limit (Clark 1993, 2002; Mace and Sissenwine 1993; Mace 1994). Using ovary weight-at-size as a proxy for fecundity, the spawning biomass-per-recruit curve drops fast with increasing fishing mortality, and hence fishing mortalities are low for both reference levels, with $F_{40\%} = 0.07$ and $F_{30\%} = 0.12$ (Fig. 5). This analysis suggests that only low fishing mortalities, close to $F_{0.1}$ for females, are sustainable. At mortality rates $F > 0.2$, relative spawning biomass falls below 20%.

A precautionary approach would favor reference mortality rates for recruitment overfishing of females as the basis for management advice. However, since it is impractical to manage the fishery differently based on sex, this level of fishing mortality would provide only about 50% of the theoretical maximum yield for males.

Increased minimum size limits would of course provide more effective protection to the female spawning biomass. The increase in the minimum size limit from 330 to 360 mm in 1998 represented a compromise between biological results from yield- and spawning biomass-per-recruit analyses and economic considerations (Murphy and Lyle 1999). Due to a market preference for small fish, any higher minimum size limits would have had a severe negative effect on access to the live-fish markets.

Yield-per-recruit and egg-per-recruit analyses are useful tools in setting management controls such as size limits and can provide target and limit reference levels for fishing mortality. But independent estimates or indicators of biomass and/or fishing mortality are still required to provide information about stock status.

Catch curve analyses

Research sampling conducted during the spawning closures in 1996 and 1997 and again in 2001, 2002, and 2003 has yielded a range of biological information for legal-sized as well as under- and over-sized individuals.

Age-frequency information is available from over 2,500 individuals, but because of longevity and spatial structuring, sample sizes at individual collection sites are low and thus may not be representative of the associated populations. Recognizing this uncertainty but attempting to

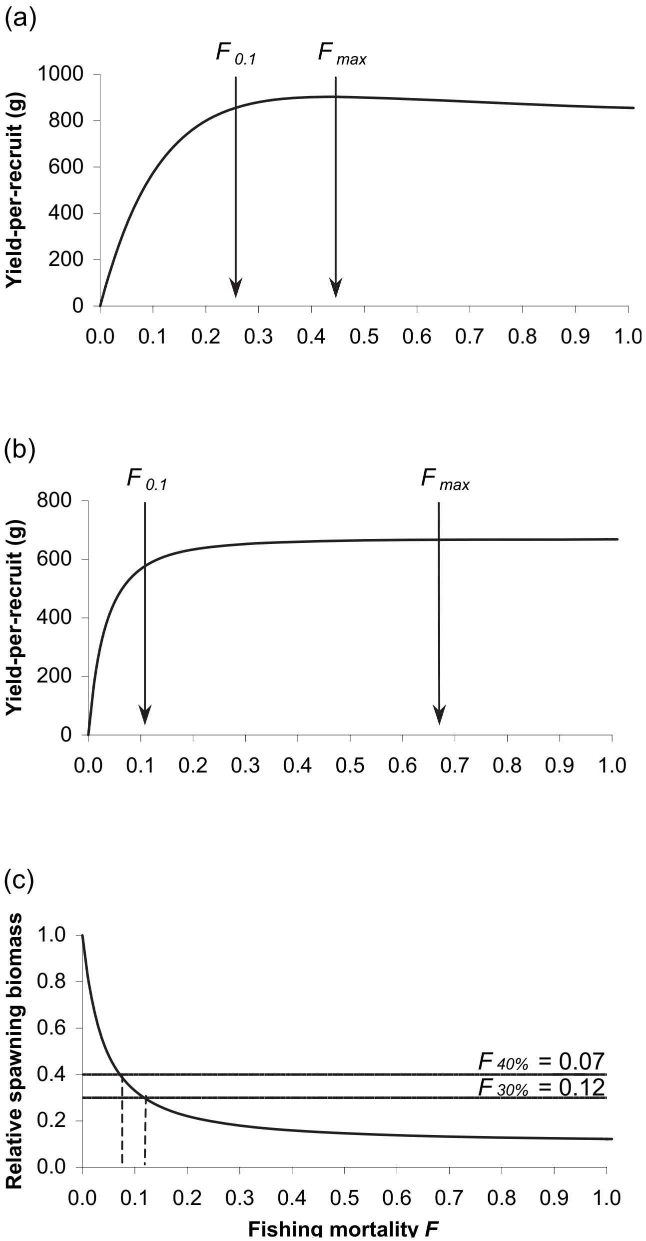

Figure 5. Yield-per-recruit curves for (a) male and (b) female banded morwong, and (c) spawning biomass-per-recruit curves. Results shown for size limits of 360-460 mm and natural mortality $M = 0.05$.

reduce noise in the data, we have pooled information across sites for each year. Age-frequency samples have been corrected for gillnet mesh selectivity (determined empirically and modeled using gamma distribution, after Millar and Holst 1997), and total mortality Z was estimated based on catch curve analysis (Beverton and Holt 1957, Ricker 1975).

This analysis revealed a considerable range of estimates of Z based on recruited ages up to 14 years, namely 0.16 for males and 0.22 and 0.12 for females in 1996 and 1997, respectively, and a range of 0.15-0.43 for males and 0.07-0.32 for females in the 2001-2003 samples (examples of 1996 and 2003 shown in Fig. 6). As over half of the reported catches had already been taken by 1996 (refer Fig. 3), even the earliest samples would have been impacted by the fishery. By comparison with many other fisheries, total mortality rates are low, but they do suggest that the reference points $F_{0.1}$, $F_{30\%}$, and $F_{40\%}$ for females derived from yield-per-recruit and spawning biomass-per-recruit analyses have been exceeded. Higher levels of noise in the data due to pooling of sites and low sample sizes, however, limits the utility of this analysis as an estimator for fishing mortality.

Age-frequency plots reveal an apparent dichotomy between trends for younger (to 14 years) and older age classes for both sexes. For males this is not unexpected, since most individuals over about 14 years are larger than the maximum size limit and are thus subject to much reduced fishing mortality. By contrast, females effectively remain vulnerable once recruited to the fishery, and thus the relatively high proportion of old females still present in the catches suggests that either catchability changes with age and/or fish down of accumulated biomass is still occurring.

If, as suggested by McCormick (1989a), large males concentrate at greater depths they may receive further protection from fishing through a “depth refuge,” since fishing rarely occurs at depths over 25 m. Any females found in deeper water will also be less vulnerable to capture. Anecdotal reports suggest that fish in the deeper waters are in fact large specimens and fishers contend that there is replenishment of fishing grounds throughout the year and associated with spawning, possibly due to movement of fish from deeper water. This gives rise to the suggestion that a portion of the population may be protected from fishing by depth. Unfortunately we have no information about the structure or relative abundance of populations in the deeper reef areas or potential mixing rates with the shallower areas. Fishing surveys of such areas and an understanding of the size and distribution of suitable deep reef habitat relative to the shallow fished reef areas may prove informative in evaluating the potential importance of depth refuges.

Biological parameters

Trends in the biological characteristics of exploited stocks have the potential to provide indications of more general changes in stock status (e.g., Caddy and Mahon 1995, Caddy 1998). Biological parameters, de-

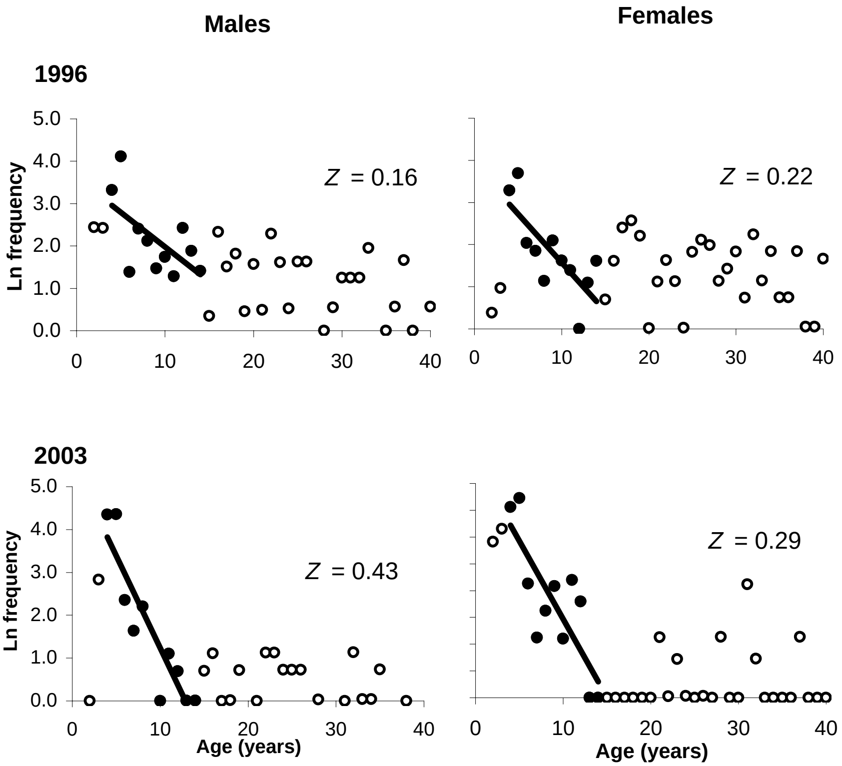

Figure 6. Catch curves for male and female banded morwong in Tasmania for 1996 and 2003. The log-transformed and selectivity-corrected age-frequencies are based on direct age estimates of *N* = 205 (1996) and *N* =216 males (2003), and *N* = 280 (1996) and *N* =164 females (2003). Data between 4 years (fish fully selected to fishing gear) and 14 years (first zero catches) have been used for mortality estimation (filled circles).

rived from spawning season surveys that include the full size range of fish available to the sampling gear, are summarized in Fig. 7.

Since 1996, the median size and age of the samples has declined for both sexes and, not surprisingly, these changes have been far more pronounced for females. For instance, median ages have fallen from around 20 to 7 years for females compared to 7 down to 5 years for males over the sampling period. Such variability between the sexes is clearly linked to growth rate differences and the effects of the legal size limits. Sex ratios also seem to have shifted from females dominating until 2001 to roughly equal numbers of males and females in the more recent samples. This biological response would at least be partly due to the greater selective fishing pressure on females.

Changes in the biological characteristics of populations are to be expected with exploitation (e.g., Buxton 1993, Harris and McGovern 1997, Helser and Almeida 1997). But without reliable and meaningful reference points, it is difficult to determine whether changes are indicative of stock stress and thus whether fishing pressure exceeds sustainable levels.

Discussion

Fished species, which are known or suspected to be spatially structured, often violate many of the assumptions typically used in stock assessment models (e.g., Parma et al. 2003). They can have heterogeneous stock structure, spatial gradients in life-history characteristics such as growth or size at maturity, and diverse stock-recruitment relationships. If the fishery is also small-scale and low in value, then extensive data collection and assessments at the appropriate spatial scales are difficult to justify.

Reliable but simple estimators of stock status together with management reference points that take into account the sedentary character and the specific life-history characteristics of the species are therefore needed. These can be based on simple biological indicators, singly or as a suite of indicators (Caddy and Mahon 1995; Caddy 1998, 1999). For example, Die and Caddy (1997) proposed a single reference point for total or fishing mortality which considers the size at first capture in relation to mean size at maturity to allow spawning before capture. Multiple reference points are generally considered "safer" than applying a single option especially when information sources are highly uncertain and insufficient. Caddy (1999) proposed a semi-quantitative approach where the fisheries management cycle incorporates a resource "traffic light" system indicating the state of the fishery on a multiple reference point board.

In our analyses for banded morwong, a range of basic assessment tools have been examined, but none have provided clear indications about stock status. Catch and effort analyses indicate recent stability but such stability could be based in part on the serial depletion of spatially structured populations and/or fish down of accumulated biomass of a large number of age classes. In effect, the catch and effort data suggest stability in the fishery but are generally uninformative about stock condition. Per-recruit and catch curve analyses suggest that sustainable fishing mortality levels are low and that current fishing pressure may be too high, though there is considerable variability in the data. Changes in biological indicators such as median age and size and sex ratio are consistent with the stocks having been impacted by fishing. Age structure information, particularly for females, indicates that the representation of old fish is still relatively high, supporting the notion that there is residual biomass

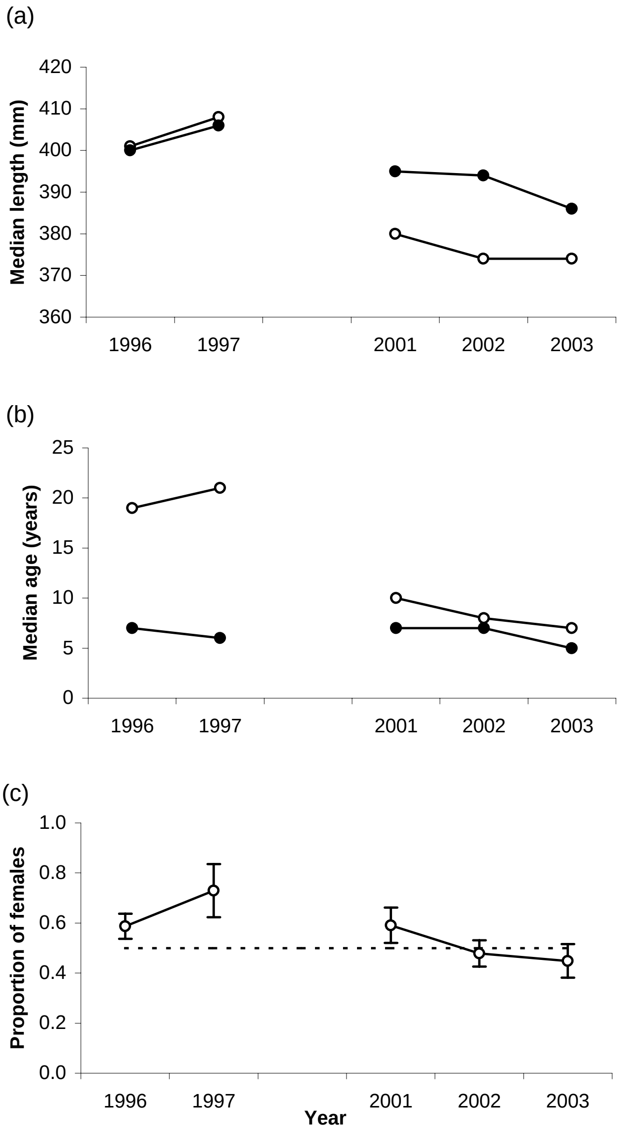

Figure 7. Biological indicators for banded morwong in Tasmania based on spawning season research surveys conducted between 1996 and 2003. (a) Median fork length and; (b) median ages (pooled samples) of males (filled circles) and females (open circles); and (c) proportion of females (mean of daily catches ± 95% confidence intervals).

still available to be fished. These observations are set against general uncertainties regarding data quality from the commercial fishery and representation of biological indicators in regard to the effects of spatial structuring within the populations.

Modeling offers a way forward to investigate the potential utility of various indicators and, in particular, to assist in determining which of these may or may not be informative about stock status. To progress, we propose to develop an operating model that incorporates spatial structuring and is tuned to the combined biological and commercial fishery data available for banded morwong (Haddon et al. 2005). Progressively, the model will be extended to simulate fishing activity, data sampling, stock assessment, and management responses. Model outputs will also be used to guide the development of cost-effective options for the monitoring and assessment of this fishery and may provide a template for the development of assessments for other spatially structured and data limited species.

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Use of Quality Control Methods to Monitor the Status of Fish Stocks

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Abstract

Many fisheries that are data-limited are also of low economic value. Therefore, not only are the fisheries data-limited, but there are limited human resources available for undertaking stock assessment. Qualitative methods such as "eyeballing" the data are then often used to assess such systems. Quantitative methods need to be developed that are objective, but less demanding than dynamic stock assessment models. In particular, simple methods that can signal trends in empirical stock-status indicators need to be explored. One such approach is the use of quality control methods such as Shewhart, moving-average, and CUSUM (cumulative sum) control charts. Originally designed for industrial quality control, these methods can be parameterized to detect transient or persistent causes with specific false-positive and false-negative error rates. These signals can be interpreted within a managerial context as trigger reference points.

Results of a simulated study of yellowfin bream (*Acanthopagrus australis*) stocks from New South Wales (Australia) are presented. Empirical stock-status indicators including catch, catch per unit effort, mean age, mean length, recruitment fraction, total mortality, and fishery-independent surveys were processed using quality control methods. Performance of these indicators and algorithms were measured with receiver-operator-characteristic curves, which captured both false-positive and false-negative error rates. Biomass surveys performed best, followed by mean age and length, and recruitment fractions. Commercial catch rates and catch had the worst performance but were still acceptable. Age-based total mortality performed poorly unless very large numbers of samples were taken. Potential applications of these methods include a rapid diagnostic tool in data-limited situations, development of empirical reference points,

and empirical rule-based management systems. These methods are easily applied even when there is a short time-series of low-contrast data but a range of caveats must always be considered.

Introduction

There is a wide range of quantitative methods to assess the status of a fish stock (Quinn and Deriso 1999) and predict the effect of alternative management choices (Hilborn and Walters 1992). Researchers working in data-limited fisheries are, however, frequently frustrated when even simple methods such as biomass dynamic or delay-difference models cannot be satisfactorily calibrated to observations. These models generally fail because of insufficient contrast, or information content, within the available data sets.

Another issue that can dominate data-limited stock assessment is lack of analytical expertise. Fisheries of low economic value are frequently both data-limited and expertise-limited because of small budgets for research, monitoring, and assessment. This constrains what can practically be achieved for many stocks. For example, issues associated with low-contrast data can be overcome using Bayesian methods that "borrow" information from other fisheries (Punt and Hilborn 1997). Expertise to apply such methods is expensive and unlikely to be used for low-value fisheries.

When data are limited and available expertise is constrained, fisheries are often assessed by "eyeballing" graphs of empirical indicators (such as commercial catch per unit effort [CPUE]). Anybody who has worked in data-limited fisheries must have seen this happen, and given the efficiency of the human eye for identifying patterns, this is neither surprising nor undesirable. Such methods are, however, somewhat subjective. Given the importance of developing objective and transparent rules for managing fisheries this approach could be criticized.

This paper attempts to identify a middle ground between "eyeballing" data, and calibrating and applying sophisticated dynamic models. Fisheries management systems have developed that utilize results from stock assessment via indicators (such as fishing mortality F) and biological reference points (such as $F_{0.1}$). Relationships between indicators and biological reference points are then used to signal particular management actions. Stock assessment therefore involves coupling data analysis and modeling with a signaling system for management. Signaling systems are a core feature of statistical quality control (QC), where indicators are monitored to flag uncharacteristic values or an undesirable trend in a sequence of values. A common application is to ensure that a manufacturing facility is producing goods with consistent characteristics. QC algorithms are simple and numerically stable but cannot, of course, provide the same insight into a fishery that a dynamic model might achieve. This paper examines the application of QC methods as a signaling system for fisheries management. Indicators derived directly from data would be analyzed (very simply) and signals from these indicators reported. Quality control methods may help fill the very large gap between “eyeballing” data and developing complex stock assessment systems.

This paper makes an important differentiation between empirical and estimated indicators of stock-status. An “empirical indicator” is one that is calculated directly from a specific set of raw data and, in the process of calculation, may use one or two parameters that can be easily defined. In contrast, an “estimated indicator” represents a fishery variable of interest that is derived from a range of data sets and is dependent upon additional parameters that may or may not be easily defined. For example, raw commercial CPUE and mean age are empirical indicators, while biomass and fishing mortality are estimated indicators. Empirical indicators would usually be calculated with a closed algorithm such as a database query, while estimated indicators frequently require iterative nonlinear algorithms. An exception to these generalities is the use of iterative general linear modeling algorithms to estimate an index-of-abundance.

Much fishery management science is oriented around the calculation and interpretation of estimated indicators. In data-limited situations where the calculation of estimated indicators is unreliable, the initial focus should be upon empirical indicators (also see Die and Caddy 1997, Caddy 1999). Several recent studies have measured the performance of empirical indicators with methods not dissimilar to those presented here. Punt et al. (2001) evaluated mean length, mean weight, and CPUE data collected in the east Australian broadbill swordfish fishery. Trenkel and Rochet (2003) completed a sophisticated power analysis on a range of stock-status indicators, including empirical indicators such as mean age and total mortality. Rochet and Trenkel (2003) extended these ideas to indicators of community structure. Scandol (2003) estimated the performance of landed catch as an indicator with CUSUM (cumulative sum, Page 1961) quality control methods. Previous research focused upon indicators, while this study examined the interpretation and performance of these indicators using methods from statistical quality control. Given the recent emphasis on the development of indicators for sustainable fisheries management (Garcia and Staples 2000), evaluation of statistical methods to interpret these indicators should be a research priority.

Materials and methods

Overview

This study used an operating model to generate observations that were transformed into nine stock-status indicators. Times-series of these indicators were then analyzed for transient and persistent causes (defined below) using three quality control methods: Shewhart control charts, CUSUM control charts, and moving-average Shewhart control charts. Signals from these quality control (QC) methods were then compared to the known state of the simulated biomass, and the sensitivity and specificity of these algorithms determined for each of the nine indicators. All of these QC methods required one or two parameters to specify an out-of-control observation.

The operating model

The operating model is detailed in the Appendix. The model was an age-structured population model for yellowfin bream (*Acanthopagrus australis*), a common species harvested off the coast of eastern Australia (Yearsley et al. 1999). The model was parameterized using data from Gray et al. (2000). A small fishery was modeled that included the likely variation in catchability, applied effort, and recorded effort among 50 commercial fishers. Simulated indicators for CPUE (U_t^{cpue}) and commercial catch (C_t) were generated by the model, and an unbiased indicator that represented the surveyable stock (U_t^{survey}) was also included.

Each year a random sample of fish from the simulated commercial fishery was taken and six other indicators were calculated: mean age (μ_t^{age}); mean length (μ_t^{len}); total mortality estimated with catch-curve analysis from the same of ages (Z_t^{age}); total mortality using the Beverton and Holt method from length statistics and growth characteristics (Z_t^{len}); and recruitment fractions (the fraction less than a threshold length or age) from age (θ_t^{age}) or length (θ_t^{len}) frequency distributions. Equations for these indicators are provided in the Appendix. More sophisticated analyses were obviously possible with these simulated data, but the aim was to capture the types of simple calculations that could be completed in a data-limited and expertise-limited environment.

The temporal sequencing of the operating model was important and there were four distinct phases. First, the model was run without any variation (but with the simulated fishery operating) until the exploitable biomass converged to an equilibrium value (B_{eq}). Second, the model was run for a further 30 years with variation to stabilize any transient processes. Third, the model entered a ten-year phase where historical data were collected and indicators calculated. Fourth, the model entered a ten-year future phase where additional data were collected and indicators calculated. Simulated observations from the future phase were used to measure the performance of the QC algorithms, and observations from the historical phase were used to provide a reference for the QC algorithms.

During the future phase, or the historical and future phases, various deleterious impacts were randomly imposed upon the fishery. These impacts affected recruitment (decreased); fishing effort (increased); catchability (increased); or natural mortality (increased). The maximal rate of an impact was a 20% change in a parameter value per year. The exact magnitude and type of an impact affecting a simulation were randomized, but these impacts were manipulated by two parameters: the probability of an impact occurring during the historical *and* future phases (P^{hist}); and the probability of an impact occurring during the future phase only (P^{futr}). This representation was required to generate scenarios where the fishery was historically unstable as well as stable.

The quality control algorithms

The quality control literature uses the term "transient cause" to describe a short-term influence that effects one or two consecutive observations in a time-series, and the term "persistent cause" to describe a long-term influence that distorts a process for a longer period and is responsible for a trend. Identification of an uncharacteristic value within a time-series is a relatively simple statistical problem and is the basis of the Shewhart control chart (Derman and Ross 1997). Processes are considered "in-control" if the observed mean of a consecutive set of observations is within a specified confidence interval. If the mean falls outside that interval, the process is considered "out-of-control." Shewhart control charts are very effective at detecting transient causes but less effective for persistent causes because they have no memory of past events. Observations at one time-step are evaluated independently of data from the previous time-step. This deficiency can be circumvented by a series of runs rules (e.g., three out of four signals in a row indicates a persistent cause) or the use of control charts with a memory of past events such as the cumulative sum or moving-average control chart.

Cumulative sum control methods are effective at detecting persistent changes in observed processes (Page 1961). The underlying algorithm is a simple cumulative sum of the deviation of observations from the mean (Hawkins and Olwell 1997). An alternative to CUSUM charts for detecting persistent causes is the moving-average control chart.

The three QC methods evaluated each required the indicators to be standardized (z transformed) using a control mean and standard deviation. This control mean and standard deviation could be specified as a managerial goal but could also be estimated from historical observations of that indicator. After all nine indicators were calculated for the historical and future phases of the simulation, they were standardized using data from the historical phase, and the QC algorithm applied to each of the ten future years. For each future year the algorithm generated a signal if an out-of-control process was detected. This signal was compared with the actual system state of exploitable biomass at time t (B_t^{ex}) defined by the biological reference point (ϕB_{eq}). If the exploitable biomass was less than this reference point and the QC algorithm generated an out-of-control signal then a true-positive result was registered. Other combinations of ($B_t^{ex} < \phi B_{eq}$) and QC signal results registered true-negative, false-positive, and false-negative results. These results were incremented for the ten years in the future phase of the simulation and the simulation replicated 1,000 times. This enabled an estimation of the probability of true-positive, true-negative, false-negative, and false-positive results of the QC algorithm for a particular set of parameter values. Due to dependencies among the future years these were not true probabilities. This could have been avoided by using different replicate simulations for each of the future years, but this would be inconsistent with the use of the scheme for a particular realization of a stochastic process (as would occur in any application of these methods).

Implementation of the QC algorithms

1. *Shewhart Control Chart.* This was the simplest to apply and required only a single control parameter, the decision interval (h). A signal is raised at time t if $|x_t|>h$, where x_t is value of the standardized indicator at time t.

2. *Moving-Average Control Chart.* An incremental development to the Shewhart control chart is the calculation of a moving-average of the last w observations of the indicator, and then application of a Shewhart control chart to these smoothed values. If the absolute value of the smoothed indicator is greater than the decision interval then a signal is raised.

3. *CUSUM Control Chart.* The final QC method examined was the cumulative sum or CUSUM control chart. This method calculated an upper or lower CUSUM ($\pm\phi$) with the following equations: $\phi_t^+ = \max(0, \phi_{t-1}^+ + x_t - k)$ and $\phi_t^- = \min(0, \phi_{t-1}^- + x_t + k)$, where x_t is value of the standardized indicator at time t and k is the chart tolerance (or variation that is ignored). These iterative equations were initialized using $\phi_0^{+/-} = 0$ and when $|\phi_t^{+/-}| > h$ an out-of-control signal was raised.

Both the moving-average chart and the CUSUM chart required additional parameters (w and k respectively), but these algorithms were more effective at signaling persistent causes because they captured the memory within a sequence of indicator values.

Measurement of performance and sensitivity analysis

Evaluation of the sensitivity and specificity of any type of test can be used to assess the performance of that test. The more sensitive a test, the more reliable it is at signaling an anomalous situation. The more specific a test, the more reliable it is at not signaling a situation that is not anomalous. These are well-established concepts in health research (Sackett et al. 2000, Gigerenzer 2002) but do not appear to have been used in the fisheries literature. Sensitivity and specificity are conditional probabilities that are defined:

$$sensitivity = P(T^+)/[P(T^+) + P(F^-)]$$

$$specificity = P(T^-)/[P(F^+) + P(T^-)]$$

where $P(T^+)$is the probability of a true-positive result and equivalent notation applies to the other terms. Different values of the decision interval h will yield different values of sensitivity and specificity for a test and illustrate the trade-off between these characteristics. A plot of the complement of the specificity against the sensitivity over a range of decision intervals (h) creates a receiver-operator-characteristic (ROC) curve.

The ROC curve is a valuable diagnostic tool because it enables an objective comparison of various indicators and QC algorithms. In this study the overall performance of an indicator and a QC algorithm is measured as the area under the ROC curve (Ω). This area was calculated using numerical integration. This performance measure had the additional advantage of integrating-out a nuisance parameter, the decision internal h, from the analysis. The ROC curve was also used to determine the value of the decision interval that yielded a threshold value of sensitivity, and estimate the specificity at that point. These calculations were completed using linear interpolation of available information from the ROC curve. Sensitivity analyses were also completed upon: the number of fish sampled for age- and length-based indicators; the coefficient of variation of recruitment; and the steepness of the stock-recruitment relationship. Consideration was also given to the CUSUM tolerance parameter k, the moving-average parameter w, the reference point ϕ, and the probability of an historical impact in the fishery P^{hist}. In all cases the sensitivity analyses are presented in terms of the overall performance (Ω).

Results

Illustrative results from a single replicate

Figure 1 includes correlation plots of the B_t/B_{eq} versus the indicator values from a single replicate simulation of the operating model. Figure 1a, 1b, and 1c illustrate the age-based, length-based, and abundance indicators respectively. The replicate was a scenario with no historical impacts ($P^{hist} = 0.0$) and surety of future impacts ($P^{futr} = 1.0$). The cluster of points at $B_t/B_{eq} \approx 1.0$ illustrates the behavior of the indicators during the historical equilibrium phase and the points when $B_t/B_{eq} < 1.0$ show the response of the indicators as the biomass collapses. As expected, there was a decrease in mean length and age as well as an increase in total mortality (Fig. 1a and 1b). The recruitment fraction indicators increased as the stock collapses. Within the abundance indicators there was strong correlation with the survey indicator (U_t^{survey}), but commercial catch (C_t) and U_t^{cpue} did not indicate biomass in a robust manner (Fig. 1c).

Results for the indicator mean age (μ_t^{age}) generated from this replicate were analyzed with the three QC algorithms examined in this paper. Figure 2a plots μ_t^{age} (unstandardized and standardized using data from 0

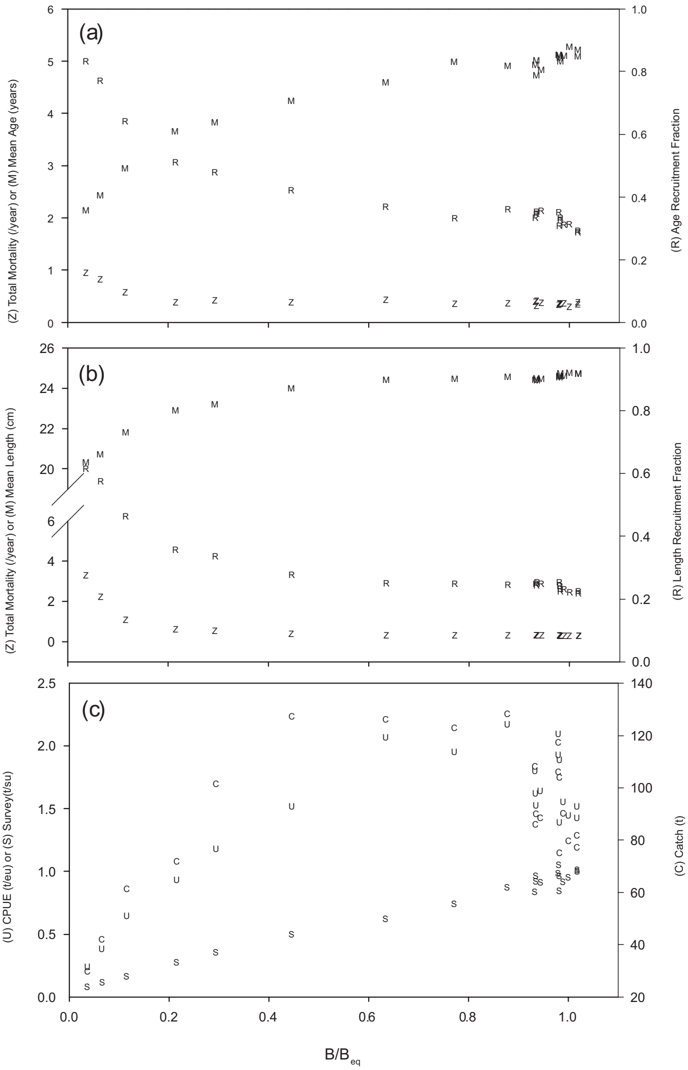

Figure 1. **Relationship between the empirical indicators and B_t/B_{eq}. (a) Age-based indicators Z_t^{age} (symbol Z) and μ_t^{age} (symbol M) are plotted on the left axis scale; θ_t^{age} (symbol R) is plotted on the right axis scale. (b) Length-based indicators Z_t^{len} (symbol Z), μ_t^{len} (symbol M), and θ_t^{len} (symbol R) are plotted in an analogous fashion to those on (a). (c) Abundance indicators U_t^{survey} (symbol S), U_t^{cpue} (symbol U), and C_t (symbol C). Unit notes: C_t (metric tons or t), U_t^{cpue} (tons per effort unit or t/eu), and U_t^{survey} (tons per survey unit or t/su).**

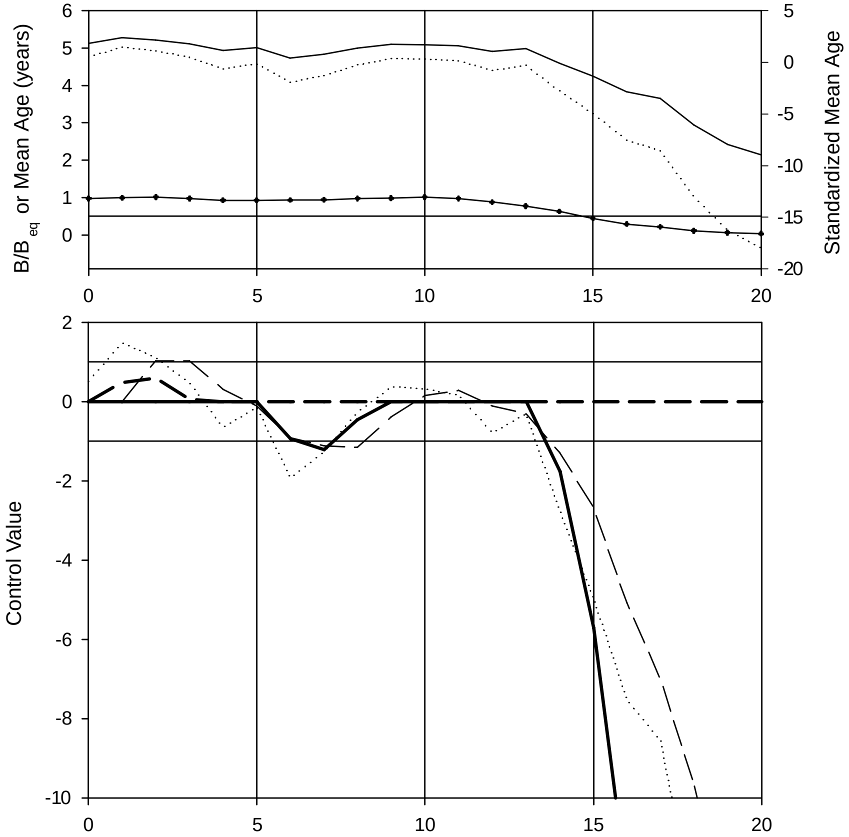

Figure 2. **Illustration of quality control charts. (a) Mean age (μ_t^{age}, solid line) and B/B_{eq} (solid line with symbols) plotted against time. Standardized mean age (dotted line) is also included. Note that $B/B_{eq} \approx 0.5$ at 15 years. (b) Control charts: Shewhart (dotted line); three year moving-average Shewhart (dashed line); CUSUM ($k = 1.0$, ϕ_t^+ heavy dashed line, ϕ_t^- heavy solid line). Signals are generated each time a control value exceeds $\pm h$, i.e., ±1 . The CUSUM method generated the strongest signal by year 15.**

to 10 years) along with B_t/B_{eq}. The reference point for this analysis was $B_t/B_{eq} = 0.5$, and the biomass fell below this value after 15 years. Before this time signals were false-positive and signals on or after 15 years were true-positive. Figure 2b provides the control charts for the standardized μ_t^{age} using a decision interval h of ±1. The Shewhart chart and moving-average (with a three year window) generated four false-positive signals before any impacts occurred (at ten years). The CUSUM chart (using $k = 1.0$) generated a false-positive signal at 7 years and 14 years but gave the strongest signal thereafter.

Comparison of QC algorithms

Replicate simulations of the above algorithm enabled the estimation of an ROC curve for a particular indicator and QC algorithm. Figure 3 illustrates these ROC curves for μ_t^{age}, C_t and U_t^{survey} using a CUSUM control chart ($k = 1.0$). When the decision interval h is 0 then all curves have a sensitivity of 1 and a specificity of 0. As h increased, the sensitivity decreases and specificity increases. Indicators and control schemes that are both sensitive and specific will have the greatest area under the curve (Ω). The ROC curve was also used to determine the specificity of an indicator at a certain value of sensitivity (or vice versa) as well as the value of the decision interval (h^*) that obtained these values. Table 1 summarizes the performance of indicators for Shewhart, CUSUM ($k = 0.5$, 1.0, 1.5) and moving-average control charts (with the average calculated over 2, 3, and 4 years). In general the choice of control chart did not have a large effect upon performance of an indicator, though the CUSUM charts always performed marginally better. This table also enables a comparison of indicators. As expected, U_t^{survey} had the best performance followed by μ_t^{age}, μ_t^{len}, Z_t^{len} (derived directly from μ_t^{len}),θ_t^{len} ,θ_t^{age} (all these age- and length-based indicators had very similar performance). Commercial catch rates (U_t^{cpue}) averaged lower performance across all QC methods than these previous indicators but were superior to $Z_t^{age}$and C_t. The number of age samples taken (1,000) was not sufficient for a precise estimate of total mortality. Ironically, Z_t^{len} was biased, but was a superior indicator to its age-based counterpart within this framework.

The table also presents the value of h^* and the test specificity when the sensitivity is 0.8. QC algorithms are commonly applied by defining a fixed sensitivity (true-positive rate). The decision interval required for such sensitivity is then estimated along with the test specificity. In time-series models, where assumptions of statistical independence between observations are invalid, simulation can be used to estimate these threshold values (otherwise analytical methods are available, Hawkins and Olwell 1997). The h^* values in Table 1 provide some insight in the decision intervals that could be used for the three QC algorithms and how specific the signals would be. These numerical values are, however, only valid for this simulated fishery. Note that for the better performing

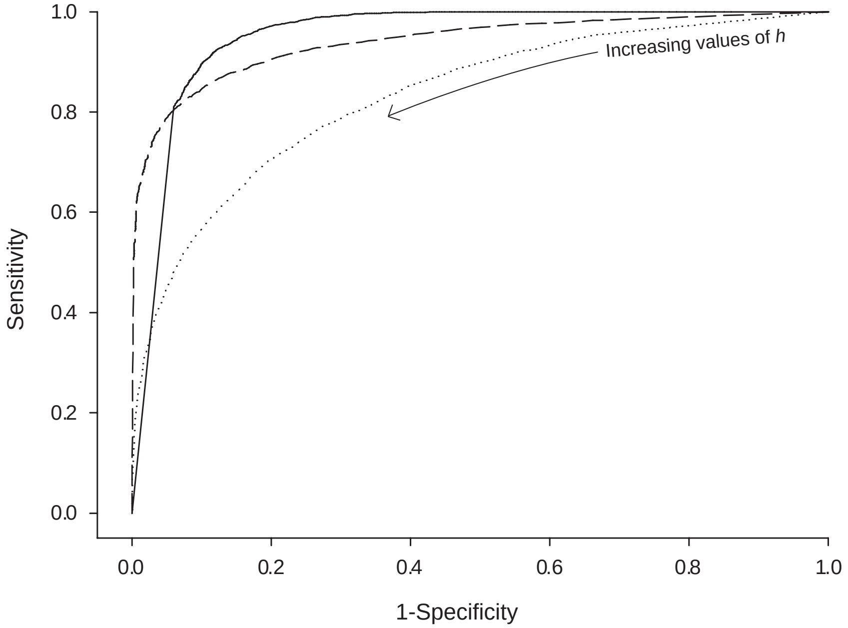

Figure 3. Receiver-operator-characteristic (ROC) curves for the CUSUM (k = 1.0) scheme for the empirical indicators: U_t^{survey} (solid line); μ_t^{age} (dashed line); and C_t (dotted line). The area under the curve for a particular indicator was used as the performance measure (Ω) in this study. Note that as the decision interval (h) increased, the sensitivity decreased and the specificity increased (following the ROC curve as annotated).

indicators such as μ_t^{age}, the h^* values are larger and are associated with greater specificity than with the inferior indicators such as C_t.

Reference points, effect size and effect timing

The reference point used for this analysis (by default 0.5 B_{eq} or $\phi = 0.5$) had a crucial role in this analysis and was analogous to an effect size in power analysis. Larger defined effects will be easier to accurately signal. Performance of indicators as a function of ϕ was estimated using the CUSUM scheme (k = 1.0) and the results plotted on Fig. 4a. As the effect size decreased (larger values of ϕ) the performance of all indicators decreased except surveys that continued to improve until $\phi = 0.8$. Large effects degraded the performance of survey indicator because of poor specificity (false-positive rate). Performance of the indicator total

Table 1 Performance (Ω) of empirical indicators for Shewhart, CUSUM, and moving-average control charts. Values of the specificity* and decision interval (h*) at 80% sensitivity are also tabulated.

			CUSUM			Moving-average		
Indicator	Metric	Shewhart	$k = 0.5$	$k = 1.0$	$k = 1.5$	$w = 2$	$w = 3$	$w = 4$
μ_t^{age}	Ω (%)	93	94	94	94	93	92	91
	_h_*	5.8	11.6	9.3	7.4	6.1	6.0	5.9
	_specificity_* (%)	94	94	94	94	93	92	90
θ_t^{age}	Ω (%)	92	93	93	92	91	91	90
	_h_*	4.1	7.8	5.7	4.2	4.2	4.1	4.1
	_specificity_* (%)	93	92	92	92	91	89	87
Z_t^{age}	Ω (%)	77	83	82	80	80	79	79
	_h_*	0.8	1.7	0.4	0.1	1.1	1.3	1.4
	_specificity_* (%)	49	68	61	55	55	57	56
C_t	Ω (%)	77	85	83	81	77	77	77
	_h_*	1.4	3.5	1.8	0.7	1.7	1.9	2.1
	_specificity_* (%)	54	71	68	65	54	55	55
U_t^{cpue}	Ω (%)	88	90	90	89	87	85	84
	_h_*	2.4	4.2	2.6	1.5	2.8	2.7	2.7
	_specificity_* (%)	84	83	83	82	79	75	72
μ_t^{len}	Ω (%)	93	94	94	93	92	91	90
	_h_*	6.2	11.9	9.7	8.0	6.1	5.7	5.5
	_specificity_* (%)	93	93	93	93	92	90	89
θ_t^{len}	Ω (%)	92	93	93	93	91	91	90
	_h_*	5.0	9.5	7.4	5.9	4.8	4.6	4.5
	specificity* (%)	92	92	92	93	91	89	87
Z_t^{len}	Ω (%)	93	94	94	94	93	92	91
	_h_*	7.4	13.7	11.4	9.8	7.1	6.5	6.3
	_specificity_* (%)	94	94	94	94	93	92	90
U_t^{survey}	Ω (%)	95	95	95	95	94	94	94
	_h_*	11.8	> 30.0	> 30.0	28.3	15.7	18.4	20.4
	_specificity_* (%)	91	< 93	< 94	94	91	91	92

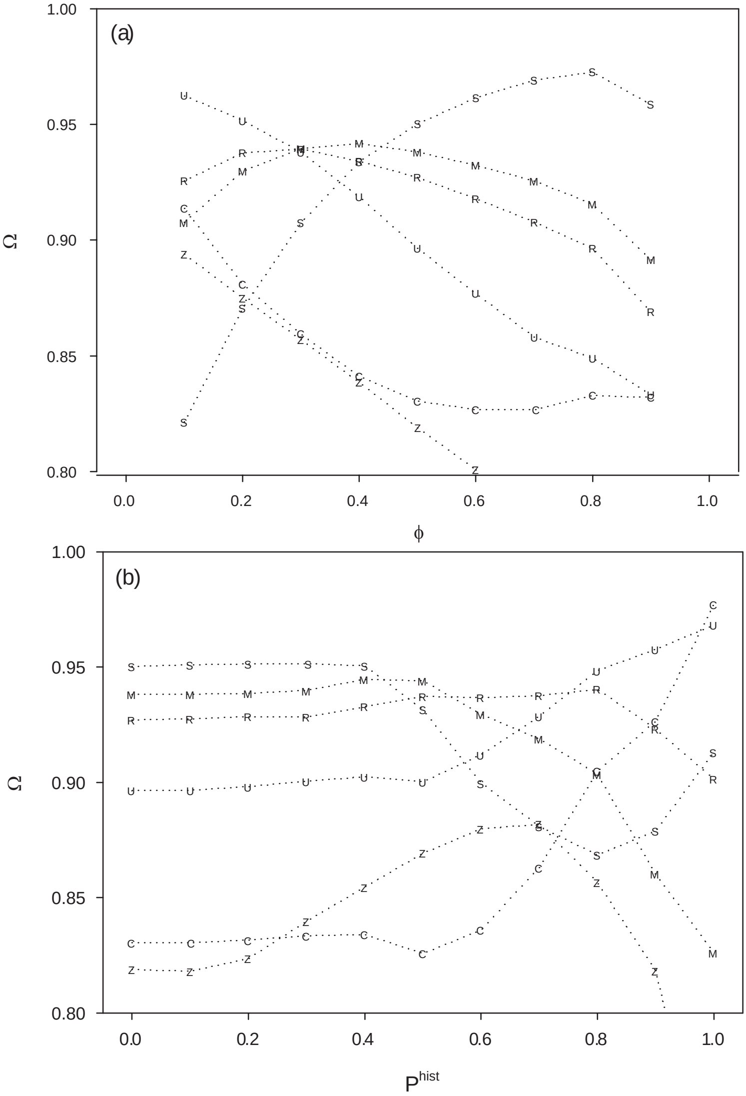

Figure 4. **(a) Performance of indicators as a function of the reference point ϕB_{eq}. Most indicators degrade as the effect size decreases. (b) Performance of indicators as a function of the probability of a historical impact P^{hist}. When $P^{hist} \geq 0.4$ the performance of all indicators became destabilized. Length-based indicators performed similarly to μ_t^{age} and θ_t^{age} in both analyses. Symbols used: U_t^{survey} (symbol S), U_t^{cpue} (symbol U), C_t (symbol C), Z_t^{age} (symbol Z), μ_t^{age} (symbol M), and θ_t^{age} (symbol R).**

mortality (Z_t^{age}) degraded badly when attempting to detect small changes to the underlying stock.

The controlling mean and standard deviation were calculated using observations from the historical years. Simulations to this point have assumed there were no impacts during this time ($P^{hist} = 0.0$). Figure 4b illustrates the effect on indicator performance when there have been changes to the stock during the historical years. Up to $P^{hist} \approx 0.4$ there was little effect on indicator performance, however; beyond this value performance changed in unpredictable ways. For example, the performance of U_t^{survey} decreased but that of U_t^{cpue} increased. Large amounts of historical variation will degrade the performance of all indicators using this scheme. These analyses on ϕ and P^{hist} were only completed for the CUSUM scheme, but the magnitude of the effects was far greater than the differences between the three QC algorithms considered. Similar patterns would be expected for the Shewhart and moving-average methods.

Sensitivity analysis

Using the CUSUM ($k = 1.0$) algorithm, results for four parameters are presented (Table 2): the number of fish aged; the number of fish measured; recruitment variability; and the steepness of the stock-recruitment parameter. Increasing the numbers of fish aged and measured increased the precision and accuracy of all of the age- and length-based indicators respectively with the exception of Z_t^{len} (which was always biased). There was, however, small marginal benefit from increasing samples from 1,000 to 10,000 fish. Most of increase in performance was obtained within 100~200 fish for all indicators except Z_t^{age}, which continued to increase in performance up to 10,000 fish. Actual fish populations would be expected to illustrate much greater variability in age and length structure than this model (which was not spatially structured and used an annual time-step). Nevertheless, these age- and length-based indicators appeared to perform very well with effective sample sizes in the hundreds rather than thousands of fish. The small (1%) performance variation in the indicators that were not age- or length-based reflected the random variation from different sequences of random numbers used in the simulations.

Table 2 also presents a sensitivity analysis on recruitment dynamics. Performance of indicators was extremely robust to changes in the steepness of the stock-recruitment parameter. Only very large values of the coefficient of variation of recruitment (50%) appeared to degrade the performance of these indicators. Other sensitivity results (not presented) were very predictable. For example, increasing the variability of surveys caused a marginal decrease in performance of U_t^{survey}. The results presented appeared to be robust to reasonable changes in the parameter values.

Table 2. Sensitivity analysis of the performance (Ω in %) of empirical indicators using the CUSUM scheme.

	Number of fish sampled for age- and length-based indicators				Coefficient of variation of recruitment				Steepness of stock-recruitment relationship		
	10	100	1,000	10,000	0.01	0.1	0.2	0.5	0.55	0.75	0.95
μ_t^{age}	78	92	94	95	94	94	92	86	93	94	94
θ_t^{age}	75	90	93	94	94	93	90	82	92	93	93
Z_t^{age}	57	77	82	90	82	82	81	75	82	82	82
C_t	83	83	84	83	83	83	82	77	83	83	83
U_t^{cpue}	89	90	90	90	90	90	89	83	90	90	90
μ_t^{len}	76	90	93	94	94	94	92	85	93	94	94
θ_t^{len}	71	88	92	94	94	93	91	83	92	93	93
Z_t^{len}	78	90	93	94	94	94	92	86	93	94	94
U_t^{survey}	95	96	96	95	95	95	95	94	95	95	95

Discussion

The lack of contrast in data is a major stumbling block for stock assessment in data-limited fisheries. In such situations, it may not be practicable to estimate fishing mortality rates or stock biomass from an index-of-abundance, landings, and age- or length-composition data sets. Alternative methods for identifying and signaling important trends in empirical stock-status indicators require examination. This study tested the application of quality control algorithms to detect signals in such empirical indicators and showed that QC-based systems using indicators of average age and length perform exceptionally well, with the integrated area under an ROC curve of between 90 and 95% under a wide range of situations. Indicators based upon recruitment fractions had similar performance. Averages can be calculated precisely with hundreds (rather than thousands) of samples, so detection of a persistent shift in these statistics is feasible. Mean lengths and ages also dampen transient signals resulting from variation in recruitment. Empirical indicators such as CPUE and catch performed worse than age- and length-based indicators, though a theoretical CPUE indicator (the ratio of actual catch and actual effort, rather than recorded effort) performed similarly to surveys. Total mortality estimated from catch-curve analysis was not a robust indicator unless the estimates used a very large sample size (many thousands of fish). Note that in real-world studies, there will be the usual challenge of obtaining representative samples of fish age and length. The above estimates of sample size have not considered spatial and within-year variation, both of which are likely to substantially increase the number of samples actually required.

There are three potential applications of the work presented here: (1) a simple signaling system for empirical indicators to prioritize research and management; (2) approximate calibration of observations to dynamic models and the use of these, along with a specification of required sensitivity/specificity to determine decision intervals and reference points; and (3) use of QC algorithms within decision-rules to ensure sustainable harvesting rates. These applications are discussed in turn.

Empirical indicators lie at the base of any type of stock assessment. In most cases (even within data-limited situations) there will be sufficient information on catch, CPUE, or length-composition to estimate a controlling mean and standard deviation. Estimation of these statistics, even from 2-3 years of data, will enable the standardization of new observations. QC algorithms could then be applied as an objective procedure to signal if the most recent observation is uncharacteristic and action should be taken. This approach is at the heart of any monitoring system. In most situations there will be ongoing uncertainty about the value of the decision interval *h* and the appropriateness of the controlling mean and standard deviation. If the recent historical state of a fishery is severely depleted, then there will be risks associated with using such information to estimate the controlling mean and standard deviation. In such cases, the decision interval could be asymmetrical, so that a signal is generated if there is *any* decrease in an indicator such as mean length or CPUE.

Results presented in Table 1 provide some guidelines on the decision interval values (for a sensitivity of 80%) but these vary greatly according to the indicator and the desired sensitivity. Rather than provide numerically specific recommendations about the application of these methods, the author recommends that interested readers compile relevant examples from their fisheries in a simple electronic spreadsheet to understand the trade-offs that occur when using these algorithms. Specification of the decision interval will depend upon the managerial context (what are the actions resulting from a signal), attitudes toward (and consequences of) false-positive and false-negative signals, and the historical context of the fishery (see above). In many data-limited cases, it is likely that the decision interval will need to be negotiated among the stakeholders of the fishery. Such an approach will certainly not be the basis of a rigorous stock assessment but will enable rapid identification of stocks where things are starting, or continuing, to go astray. Inferences will also be much stronger if there are two or more independent empirical indicators available (also see Caddy 1999, Scandol 2003). The simplicity of these schemes would enable them to be integrated into the reporting algorithms of database management systems that would (or should) be used to store the data required to compile empirical indicators.

The second, and much more technically sophisticated, application will require a meaningful calibration of a dynamic model to a particular fishery. This was the general approach taken by Punt et al. (2001). Once an estimated relationship between an empirical and estimated indicator is available, then the standard limit/target biological reference points (BRPs) such as $B/B_0 = 0.2$ or $F_{0.1}$ can be converted to values of an empirical indicator or indicators. The degree of precaution or risk-aversion to be applied with the management system then could be expressed via the sensitivity of the test and the corresponding value of the decision interval h. This signal system could then replace a BRP target/limit system. The main advantage that such a scheme could have over standard BRP management systems is improved transparency.

The third potential application of QC methods is within the managerial decision-rules that are now advocated by some fishery scientists (e.g., Starr et al. 1997, Hilborn 2002). Starr et al. (1997) used rules based upon moving-averages of CPUE to adjust the total allowable catch in a fishery to support sustainability. Similarly, the values from a CUSUM chart could be used to adjust fishing mortality using either input or output controls. Schemes that use control loops based upon indicators derived from age- or length-composition data should be particularly valuable as they will be more robust to changes in catchability. CUSUM charts are likely to be very effective because they deliver very strong signals if a persistent impact occurs (see Fig. 2b). Systems could be designed that are robust to uncertainties in most parameters.

Many fisheries management agencies that are dealing with data-limited fisheries have limited human resources to develop sophisticated stock assessments. There will, however, often be statutory requirements for performance reporting of fisheries. Simple graphical charts of empirical indicators will therefore be a mandatory and achievable goal. Charts are also a transparent and rapid method of communication. The research presented here introduces some simple ideas from the statistics of quality control to assist in the quantitative interpretation of these charts of empirical indicators. Such methods can be applied in a very simple manner but more sophisticated extensions are also feasible.

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Appendix: The operating model

Simulations were completed using the following age-structured model. Parameter values were, where possible, estimated from observations of yellowfin bream (*Acanthopagrus australis*) from Gray et al. (2000) or given assumed values. Default values of these parameters are given below. Average individual fish length (cm) at age *a* was represented with:

$$\bar{l}_a = 29.0\left\{1 - exp\left[-0.36(a+1)\right]\right\}$$

Lengths were converted to weights (in kg) with $w_a = 2.48\times10^{-5}\, l_a^{3.0}$. All selectivity and maturity schedules were based upon the logistic function:

$$\Lambda(x, x_{50}, x_{95}) = \left[1 + exp\left(-\ln(19)\frac{(x - x_{50})}{(x_{95} - x_{50})}\right)\right]^{-1}$$

Vulnerability at length was $v_l^{len} = \Lambda(l$, 24.4 cm, 29.4 cm$)$; vulnerability at age was $v_a^{age} = \Lambda(a$, 4.7 year, 8.8 year$)$; maturity at age was $m_a^{age} = \Lambda(a$, 3.7 year, 5.3 year$)$; and for surveys the vulnerability at age was $v_a^{surv} = \Lambda(a$, 0.5 year, 1.0 year$)$.

Exploitable and spawning biomass were respectively represented with:

$$B_t^{ex} = \sum_{a=0}^{A} N_{a,t}^{age} v_a^{age} w_a \quad \text{and} \quad B_t^{sp} = \sum_{a=0}^{A} N_{a,t}^{age} m_a^{age} w_a \ .$$

The maximum number of age classes (*A*) was 20 years and no “plus group” was included in this model. Standard deviation of length at age was $\sigma_a = 1.9 + 0.1a$ (cm) and the elements within the age-length key were calculated with

$$P_{i,j} = \int_{l-}^{l+} \frac{1}{\sqrt{2\pi}\,\sigma_j} \times exp\left[\frac{-(x - \bar{l}_j)^2}{2(\sigma_j)^2}\right] dx \ .$$

This integral was calculated over each of the 50 × 1 cm length classes. Distribution of fish at length *l* at time *t* was calculated using

$$N_{l,t}^{len} = \sum_{i=0}^{A} P_{l,i} N_{i,t}^{age} \ .$$

The model was initialized with $N_{a,0}^{age} = R_0\, exp(-M \times a)$ where $R_0 = 1{,}000$ $(\times10^3)$ fish and $M = 0.2$ year^{-1}. Initial spawning numbers were calculated

$$N_0^{sp} = \sum_{i=0}^{A} m_a^{age} N_{a,0}^{age}$$

which enabled parameterization of the stock-recruitment relationship using

$$A^{sr} = \frac{R_0}{N_0^{sp}}\left(1 - \frac{z - 0.2}{0.8z}\right) \quad \text{and} \quad B^{sr} = \frac{z - 0.2}{0.8zN_0^{sp}} \ .$$

The steepness parameter z had a default value of 0.75. Mean recruitment into the $t+1$ year was given by $\bar{R}_{t+1} = N_t^{sp} \big/ (A^{sr} + B^{sr} N_t^{sp})$.

The age-structured population was updated with

$$N_{a,t}^{age} = N_{a-1,t-1}^{age}\, exp[-(\mathrm{M} + \mathrm{F}_{a,t})]$$

(the fishing model for F is described below). Catch was calculated using

$$C_{a,t}^{age} = \frac{F_{a,t}}{M + F_{a,t}} \times N_{a,t}^{age} \times \{1 - exp[-(M + F_{a,t})]\}\ .$$

Numbers of fish in the zero age class were specified with

$$N_{0,t} = \bar{R}_t\, exp(R_{cv}\varepsilon - R_{cv}^2 \big/ 2)\ ,$$

where ε is a normally distributed random variable and R_{cv} has a default value of 10% (sensitivity analysis considered larger values of this parameter). Total fishing mortality F was the sum of individual activity from 50 fishers with individual effort ($e_{k,t}$) distributed as a random exponential variable (mean 0.4 units of effort). Individual catchability was lognormally distributed (mean 0.1, cv 50%). This gave an average fishing mortality (F) of 0.2 year^{-1}. Fishing mortality for an age class was assumed to be proportional to the vulnerability of that age class.

Impacts on a fishery were modeled by the generation of a random uniform variate (u) for each impact variable: natural mortality; catchability; effort; and recruitment (future impacts only). The geometrically imposed rate of impact r per year was

$$if\ (u \geq 1 - P)\ r = 0.2(u - P - 1)P\ else\ r = 0$$

where P was the probability of an impact (either P^{hist} or P^{futr}).

Indicators were calculated using the following equations:

$$C_t = \sum_{a=0}^{A} C_{a,t}^{age} w_a$$

$$U_t^{cpue} = C_t \Big/ \sum_k int(e_{k,t} + 1)$$ (tons per effort unit or t/eu; note the corruption of effort data)

$$U_t^{survey} = 0.001\left(\sum_{a=0}^{A} v_a^{survey} N_{a,t}^{age} w_a\right)[exp(0.05\varepsilon - 0.05^2 / 2)]$$

(tons per survey unit, or t/su).

Age- and length-based indicators were derived from random samples of fish (1,000 age samples, 3,000 length samples) drawn from the catch. Z_t^{age} was calculated using catch-curve analysis (with a threshold for recruitment of 4 years); μ_t^{age} and μ_t^{len} were calculated using the usual arithmetic equations; θ_t^{age} and θ_t^{len} were the fraction of the samples less than or equal to 3 years and 22 cm respectively. Finally $\mathrm{Z}_t^{len} = 0.36(29.0 - \mu_t^{len})/(\mu_t^{len} - 19.4)$.

Per-Recruit Simulation as a Rapid Assessment Tool for a Multispecies Small-Scale Fishery in Lake Malombe, Malawi, Africa

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Abstract

The 390 km^2 Lake Malombe supports a small-scale fishery that is dominated by the nkacha net, a locally developed purse seine. The nkacha fishery contributes in excess of 95% to the haplochromine cichlid catch from Lake Malombe. Since 1990 the annual haplochromine cichlid yield declined from 9,500 t to less than 4,000 t. In Lake Malombe, as in most African fisheries, the use of assessment methods that allow for dynamic simulation analysis incorporating the response of the stock to changes in management strategy was negated by the lack of directed age-based catch data and the lack of information pertaining to the biology of the target species. In this study, a rapid assessment framework for the assessment of biological and fishery input-parameters needed for the application of multispecies yield- (YPR) and spawner-biomass per recruit (SBR) analysis is presented. During a one-year assessment period, species selectivity by the nkacha fishery was determined. Length-frequency analysis allowed for a first estimate of growth rate for five major target species. These growth rate data were then used to estimate age-selectivity into the fishery, mortality rates, and age-specific maturity for each of the target species. Subsequently, YPR and SBR analyses were used to investigate commonly used "target reference points" as management targets for five target species in the fishery. The results are discussed with reference to the application

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[1] Deceased.

of per-recruit analysis for the rapid derivation of management advice for multispecies fisheries in data limited situations.

Introduction

Lake Malombe (14º30′-14º45′S to 35º12′-35º20′E) is situated in the Mangochi District of Malawi (Fig. 1). The lake is a shallow, 390 km^2 widening of the Upper Shire River about 15 km from its outflow from Lake Malawi, and provides a livelihood for about 2,000 small-scale fishers (Weyl et al. 2001b). While catch and effort has been monitored by the Malawi Department of Fisheries since 1976 (Tweddle et al. 1995), the monitoring system aggregates more than 60 species into 13 commercial categories (Tweddle et al. 1995). Despite this aggregation, major changes in the fishery have been detected. The fishery for tilapiine cichlids, which contributed more than 6,000 tons to the total catch in 1982, collapsed in 1991 and was replaced by a fishery for a variety of small haplochromine cichlid species (Tweddle et al. 1995). This haplochromine fishery yielded 9,500 tons in 1990, but by the mid-1990s had declined to less than 4,000 tons annually (Weyl et al. 2001a). Since a major objective of the Malawi fisheries policy is “the maximization of harvests within safe sustainable yield levels” (Government of Malawi 1999), the decline in this fishery necessitated a stock assessment.

Fishing gear used to harvest haplochromine cichlids on Lake Malombe includes gillnets, beach seines, purse seines, hooks, and traps (Weyl et al. 2004). Despite this diversity in gear use, 147 nkacha nets (purse-seine type gears that are operated offshore) contributed more than 95% to the total haplochromine catch (Weyl et al. 2001a). Nkacha nets utilize mesh sizes ≤ 19 mm and 57 species have been recorded in their catches (Weyl et al. 2004). While the species composition in the catch is highly diverse, five species, *Copadichromis chrysonotus*, *Copadichromis virginalis*, *Lethrinops turneri*, *Otopharynx argyrosoma* “ssp. red” (informal subspecies name after Turner [1996]), and *Otopharynx tetrastigma*, contributed more than 60% to the catch (Weyl et al. 2004).

The presence of an existing fishery on the lake requires the use of assessment methods that allow for dynamic simulation analysis incorporating the response of the multispecies stock to changes in management strategy (e.g., a change in selectivity or effort). However, the choice of suitable assessment methods was constrained by the lack of directed catch and effort data, length- or age-based catch data, and other biological data pertinent to the application of age-based models. Further complications arose from the high cost associated with the collection of these data and the immediate need for management recommendations. For this reason, a cost-effective stock-assessment technique that is not data intensive is required.

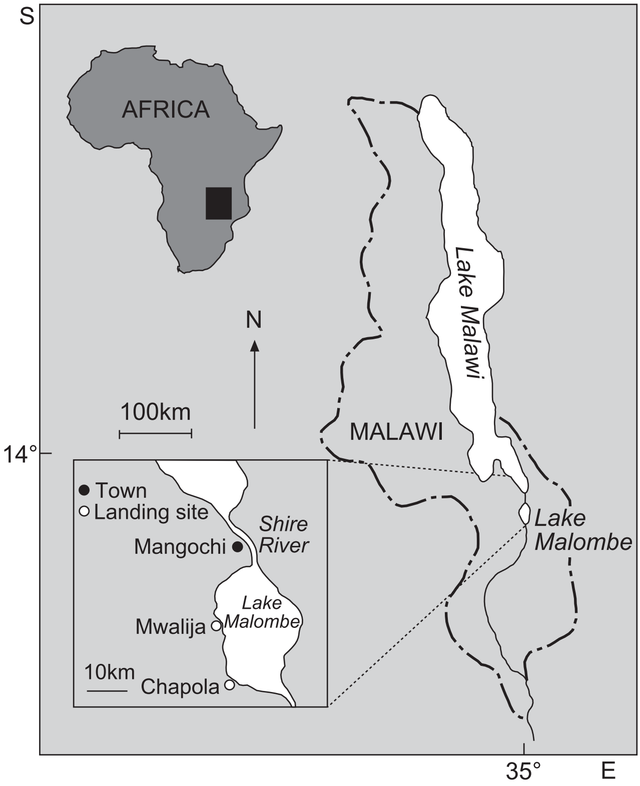

Figure 1. Map of Lake Malombe showing Mwalija and Chapola fish landing sites where nkacha net catches were sampled between April 2000 and March 2001.

As a result of similar constraints, fisheries managers in developing countries have focused on the application of the yield-per-recruit (YPR) model, which is an abbreviation of the full dynamic-pool model (Beverton and Holt 1956, 1957), in the management of lacustrine fisheries (Amarasinghe and De Silva 1992, Manyala et al. 1995, Thompson and Allison 1997). The application of these YPR models for the assessment of predefined fishing mortality targets (target reference points—TRPs) has become common practice in fisheries management (Clarke 1991, Punt 1993, Punt and Butterworth 1993, Caddy and Mahon 1995). The YPR approach allows for the determination of at least two commonly used TRPs: first, the fishing mortality which corresponds to the maximum of

Table 1. Input parameters used for the application of yield- and spawner-biomass per recruit models on *Copadichromis chrysonotus* (CC), *Copadichromis virginalis* (CV), *Lethrinops turneri* (LP), *Otopharynx argyrosoma* "ssp. red," (OA) and *Otopharynx tetrastigma* (OT) in Lake Malombe, Malawi.

	Species				
Parameter	CC[a]	CV	LT	OA	OT
L_∞	140[a]	118	144	149	139
K	1.0[a]	0.61	0.38	0.46	0.44
t_0	0[a]	0	0	0	0
α	0.000083[a]	0.00021	0.000043	0.000195	0.0000103
β	2.58[a]	2.33	2.70	2.33	3.03
ϕ	0.73[a]	1.02	2.54	1.09	1.65
σ_m	0.12[a]	0.20	1.23	0.25	0.27
t_r	0.57[a]	1.07	1.03	0.84	1.11
σ_r	0.07[a]	0.13	0.07	0.08	0.09
M	1.31[a]	0.7	0.3	0.39	0.38
F_{CUR}	1.36[a]	1.09	1.65	1.02	0.19
max	5[a]	5	5	5	5
q	0.009251701	0.007415	0.011224	0.006939	0.001293
R_i	1.52E+08	1.94E+08	3.93E+08	1.38E+08	2.14E+08

[a] Parameter estimates derived from Weyl et al. (2005).

L_∞ = asymptotic length (mm TL), K = Brody growth coefficient (yr^{-1}), t_0 = age at zero length, α and β = length weight parameters, ϕ = age-at-50%-maturity (years), σ_m = variance of logistic maturity ogive, t_r = age-at-50%-selection (years), σ_r = variance of logistic selection ogive (years), M = natural mortality rate (yr^{-1}), F_{CUR} = current fishing mortality rate (yr^{-1}), max = maximum age of fish (years), q = catchability coefficient), R_i = estimated recruitment.

the yield-per-recruit curve (F_{MAX}) and second, the marginal yield or $F_{0.1}$ strategy (Gulland and Boerema 1973, Deriso 1987), which is the rate of fishing mortality at which the slope of the yield-per-recruit curve falls to 10% of its value at the origin. However, the assumption that recruitment is constant and independent of spawner-biomass has led to some criticism of YPR-based TRPs and scientists concerned with the management of marine species have tended to base their TRP recommendations on the results of spawner biomass-per-recruit (SBR) models (Butterworth et al. 1989, Smale and Punt 1991).

In the absence of information on the surplus production function or the spawner biomass-recruitment relationship, SBR-based TRPs are currently considered the most robust, allowing for the determination of a fishing mortality rate that will provide relatively high yields at lower risks (Clarke 1991, Punt 1993). The definition of a SBR-based TRP ($F_{SB(x)}$)

involves setting the fishing mortality to a level at which spawner biomass-per-recruit is reduced to *x*% of its pristine level. Although there is no conventional $F_{SB(x)}$ TRP, the maintenance of SBR at between 25% and 50% of unexploited levels has been recommended (Deriso 1987, Sissenwine and Shepherd 1987, Butterworth et al. 1989, Punt 1993, Booth 2004). Because haplochromine cichlids are mouth-brooders (Turner 1996), strong density dependence between spawner-biomass and recruitment is implied and the maintenance of SBR at levels where the stock can replace itself is likely to be important for the sustainability of haplochromine-based fisheries.

This study utilizes data derived from a one-year survey of the nkacha net fishery to derive input parameters for, and apply YPR and SBR models to assess, four commonly used TRPs for the derivation of management advice for *C. chrysonotus*, *C. virginalis*, *L. turneri*, *O. argyrosoma* "ssp. red," and *O. tetrastigma* in the Lake Malombe nkacha net fishery.

Methods

The application of per-recruit analysis requires age-based estimates for mortality, maturity, selectivity, and the length-weight relationship. *Copadichromis chrysonotus* has been assessed using per-recruit analysis and all input parameters are available (Weyl et al. 2005). For the other four species, *C. virginalis*, *L. turneri*, *O. argyrosoma* "ssp. red," and *O. tetrastigma*, the length-weight relationship and length-based selectivity and maturity parameters were available in the literature (Banda 1995, Weyl et al. 2004).

Growth estimation

Due to the use of small mesh sizes (≤ 19 mm) by the nkacha net fishery resulting in negligible differences in size-selectivity (Weyl et al. 2004), the length-frequency of the target species in this gear showed discernable modal progression over time and was therefore considered suitable for the estimation of length-at-age (l_t) using length-frequency analysis (Pauly and David 1981, Wetherall 1986, Shepherd 1987).

Monthly length-frequency samples were collected from the nkacha net fishery at two major landing sites on the western shore of the lake, for three consecutive days a month from March 2000 to April 2001. The methods used during this survey are detailed in Weyl et al. (2004, 2005).

Length-frequencies of *C. virginalis*, *L. turneri*, *O. argyrosoma* "ssp. red," and *O. tetrastigma* in sampled catches were regrouped into 5 mm size classes and raised to represent the length-frequency of the estimated catch of the species during each sampling period using the methods outlined by Weyl et al. (2005) for *C. chrysonotus*.

To estimate length-at-age (l_t), length-frequency analysis (Pauly and David 1981, Wetherall 1986, Shepherd 1987) was used to derive estimates for asymptotic length (L_∞) and the growth coefficient (K) for the von Bertalanffy growth model: $l_t = \mathrm{L}_\infty(1 - \exp{-K^{(t-t_0)}})$ (Ricker 1975). A preliminary estimate of the growth parameter L_∞ was obtained from pooled monthly length-frequency samples using the Powell-Wetherall method (Wetherall 1986). Electronic length frequency analysis (ELEFAN) (Pauly and David 1981) was used to derive a first estimate of K. This analysis was performed using the FAO-ICLARM Fisheries Assessment Tools (FiSAT) software (Gayanilo et al. 1997). There was insufficient resolution in the data to calculate the age-at-zero length (t_0) and this parameter was, therefore, taken to be zero.

Mortality

A first approximation of the instantaneous rate of total mortality (Z) was estimated by catch-curve analysis (Ricker 1975). Annualized length-frequency data presented in Weyl et al. (2004) were analyzed by means of a linearized length-converted catch curve (Pauly 1983, 1984a,b). This method uses the von Bertalanffy growth parameters to plot $\ln(f/dt)$ against t, where f is the frequency of individuals in each length class and t is the relative age of the fish. The value dt is the time taken for the fish to grow through a particular length class and allows for decreased growth with increased age. The negative of the slope of the resultant linear regression line through the descending data points gives a first approximation of Z.

Natural mortality (M) was estimated using the Pauly (1980) empirical equation; $M = -0.0152 - 0.279 \ln L_\infty + 0.6543 \ln K + 0.463 \ln T$; where L_∞ (cm) and K are the von Bertalanffy growth parameters and T is the mean annual water surface temperature (26.2ºC, calculated from monthly temperature readings presented in Jambo [1997]). The estimates for M derived using the Pauly (1980) equation were similar to independent estimates for closely related species in Lake Malawi (Tweddle and Turner 1977) and were accepted as a first estimate for the five species.

The fishing mortality rate was derived by subtraction ($F = Z - M$). Because the fishery targets all five species, the coefficients of proportionality between fishing effort and fishing mortality (i.e., the catchability coefficients) will vary between species due to differences in their availability and vulnerability to the gear (Murawski 1984). Therefore, at any given level of effort, the F for each species in a multispecies fishery will be different. Catchability coefficients were estimated using the linear relationship, $F_i = q_i \times f$, where q_i is the catchability coefficient of species i, and f is the total effort by all gears in the fishery during the assessment year (i.e., 147 nkacha nets enumerated in 2001).

Maturity and selectivity

The proportion of mature fish-at-age ψ_t was estimated by age-converting the Banda (1995) length-based logistic maturity ogives as

$$t = t_0 - \frac{1}{K}\ln\left(1 - \frac{l_t}{L_\infty}\right).$$

The age-converted ogive width parameter σ_t was calculated from the length-based equivalents as

$$\sigma_t = t_0 - \frac{1}{K}\ln\left(1 - \frac{\sigma_l \ln 3}{L_\infty - \phi_l}\right) \text{ (Booth and Weyl 2004).}$$

Age conversion of the length-based selectivity for the four species provided in Weyl et al. (2004) was calculated using the maturity methods described above.

Per-recruit analysis

An assumption of per-recruit analysis is that the parameters for recruitment, growth, and natural mortality are constant from one year to the next and, therefore, the stock is in a steady state. Under these assumptions, yield-per-recruit [*YPR*(*f*)] and spawner biomass-per-recruit [*SBR(f)*] of any species *i* as functions of fishing effort *f*, were calculated as:

$$YPR(f)_i = \int_{t_r}^{\max} N_{t,i} W_{t,i} S_{t,i} q_i f dt$$

$$SBR(f)_i = \int_{t_r}^{\max} N_{t,i} W_{t,i} \psi_{t,i} dt$$

In all YPR and SBR models, for any species *i*, W_t is the mass-at-age *t* (derived from the relationship $W_t = \alpha(l_t)^\beta$ where l_t is length-at-age, α and β are parameters describing the length-weight relationship), S_t is the selectivity-at-age, t_r is the age of fish first recruiting into the fishery, ψ_t is the maturity-at-age, q_i is the catchability coefficient, *f* is the fishing effort (number of nkacha nets) and *max* the maximum recorded age. Because no direct estimate of age through hard part analysis was possible, the parameter *max* was set at 5 years for all species as this was the maximum number of modes seen in the length frequency distributions (Fig. 2).

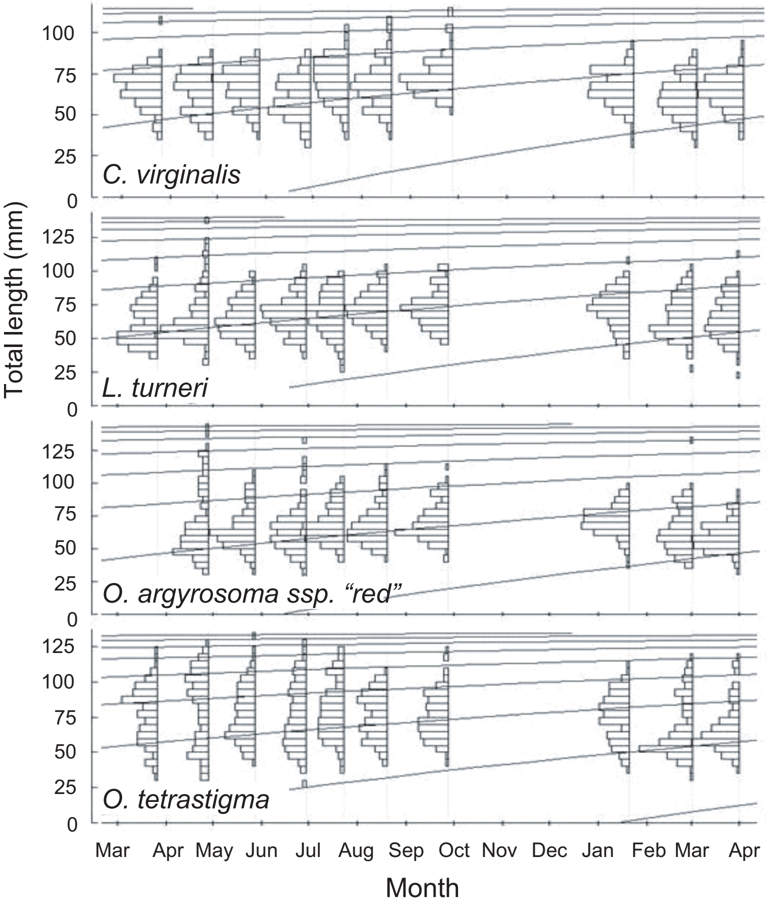

Figure 2. **Monthly length-frequency (bars) and fitted von Bertalanffy growth curves (lines) for *Copadichromis virginalis* (L_∞ = 118 mm TL, K = 0.61, t_0 = 0), *Lethrinops turneri* (L_∞ = 144 mm TL, K = 0.38, t_0 = 0), *Otopharynx argyrosoma* "ssp. red" (L_∞ = 149 mm TL, K = 0.46, t_0 = 0), and *O. tetrastigma* (L_∞ = 139 mm TL, K = 0.44, t_0 = 0) sampled from the Lake Malombe, Malawi, nkacha net fishery between April 2000 and April 2001.**

Numbers at age t (N_t), were calculated as $N_{t,i} = e^{-(t_{r,i}M_i)-(a-t_{r,i})(S_{t,i}q_i f+M_i)}$ where M is the age-independent rate of natural mortality. The YPR and SBR integrals were solved numerically using Simpson's rule with 50 steps.

To calculate total YPR for all five species at any given level of f, $TYPR_f$, the $YPR(f)_i$ of each species i was corrected according to the estimated recruitment R_i and $TYPR_f$ was then additive, such that:

$$TYPR_f = \sum_{i=1}^{5} YPR(f)_i \, .$$

Recruitment for species i, R_i was obtained by dividing the total catch of each species (Y_i) during the 2001 calendar year by the total yield-per-recruit (YPR_i) from the fishery determined at "base case" fishing and natural mortality scenarios.

Parameter values used in the analysis are summarized in Table 1.

Target reference points

Four target reference points (TRPs) were investigated for the YPR and SBR curves: F_{MAX} (that fishing mortality corresponding to the maximum of the YPR curve); $F_{0.1}$ (where the slope of the YPR curve is 10% of that at the origin); F_{SB50} and F_{SB35} (the fishing mortalities that correspond to a reduction in the SBR curve to 50% and 35% of its unexploited equilibrium level).

Due to the inherent difficulty in the estimation of the instantaneous rate of natural mortality (M), the sensitivity of the YPR and SBR models to changes in the rate M was assessed using a Monte Carlo estimation procedure (Manly 1998) described in Weyl et al.(2004). In this procedure 1,000 (U_{1000}) random mortality samples (M_U: U=1,2,…,U_{1000}) were generated, with a normally distributed error structure around the "base case" M of each species. A coefficient of variation (CV) of 25% around the M-estimate was utilized as this represented the CV of five M estimates derived for similar cichlid species in Lake Malawi (Tweddle and Turner 1977). Subsequently, a corresponding set of $\hat{F}^1$, $\hat{F}^2$,…, $\hat{F}^{U_{1000}}$ TRPs was computed for each $M \sim N[M,(0.25\times M)]^2$ and the mean, CV, and 95% confidence intervals derived. The percentile method was used to estimate 95% confidence intervals, where the 2.5% and 97.5% quartiles from the sorted $\hat{F}$ vector were chosen to represent the upper and lower 95% confidence intervals respectively (Buckland 1984).

Results

A total of 3,989 *C. virginalis*, 6,335 *L. turneri*, 3,512 *O. argyrosoma* "ssp. red," and 3,406 *O. tetrastigma* were measured during the assessment period, and the von Bertalanffy growth curves fitted to monthly length frequency distributions are shown in Fig. 2. The five species were relatively small with estimated L_∞ ranging between 118 and 140 mm TL (Table 1).

Table 2. Summary statistics for four target reference points (TRP) (F_{MAX}, $F_{0.1}$, F_{SB50}, and F_{SB35}) derived from a Monte Carlo estimation procedure using 1,000 iterations for five target species in the Lake Malombe nkacha net fishery.

		F (yr^{-1})		
Species	TRP	Mean	Lower-upper 95% CIs	CV (%)
C. chrysonotus	F_{MAX}	2.41	1.11-4.41	34.1
	$F_{0.1}$	1.14	0.64-1.75	24.5
	F_{SB50}	0.68	0.43-0.97	20.1
	F_{SB35}	1.13	0.70-1.64	20.6
C. virginalis	F_{MAX}	1.69	0.86-2.93	57.6
	$F_{0.1}$	0.76	0.53-1.03	17.2
	F_{SB50}	0.54	0.39-0.71	15.4
	F_{SB35}	0.94	0.65-1.29	17.9
L. turneri	F_{MAX}	0.62	0.53-0.71	7.4
	$F_{0.1}$	0.42	0.38-0.47	5.4
	F_{SB50}	0.28	0.27-0.30	3.5
	F_{SB35}	0.45	0.42-0.48	3.8
O. argyrosoma "ssp. red"	F_{MAX}	0.72	0.57-0.92	12.6
	$F_{0.1}$	0.46	0.39-0.55	8.8
	F_{SB50}	0.33	0.30-0.37	6.1
	F_{SB35}	0.53	0.47-0.61	6.7
O. tetrastigma	F_{MAX}	0.68	0.57-0.83	10.0
	$F_{0.1}$	0.46	0.40-0.53	7.1
	F_{SB50}	0.32	0.29-0.35	4.4
	F_{SB35}	0.50	0.46-0.55	4.6

C. chrysonotus data are from Weyl et al. 2005

Length-age converted catch curves and resultant Z estimates are shown in Fig. 3. First estimates of M, F, and q for *C. chrysonotus, C. virginalis*, *L. turneri*, *O. argyrosoma* "ssp. red," and *O. tetrastigma*, and age-converted selectivity and maturity parameters are summarized in Table 1.

For all species, the F required to attain the F_{MAX} TRP was considerably higher than that required to attain F_{SB50} and F_{SB35} (Fig. 4, Table 2). In all species, an F_{MAX} strategy resulted in severe SBR depletion and the $F_{0.1}$ TRP approximated the F_{SB35} TRP (Fig. 4, Table 2).

The response of YPR and SBR indicate differences in resilience to fishing effort by the different species. From a YPR perspective, both *C. chrysonotus* and *C. virginalis* are harvested at close to optimum levels with current fishing mortalities (F_{CUR}) between the $F_{0.1}$ and the F_{MAX} TRPs. From an SBR perspective, F_{CUR} reduces the SBR of the two species to about

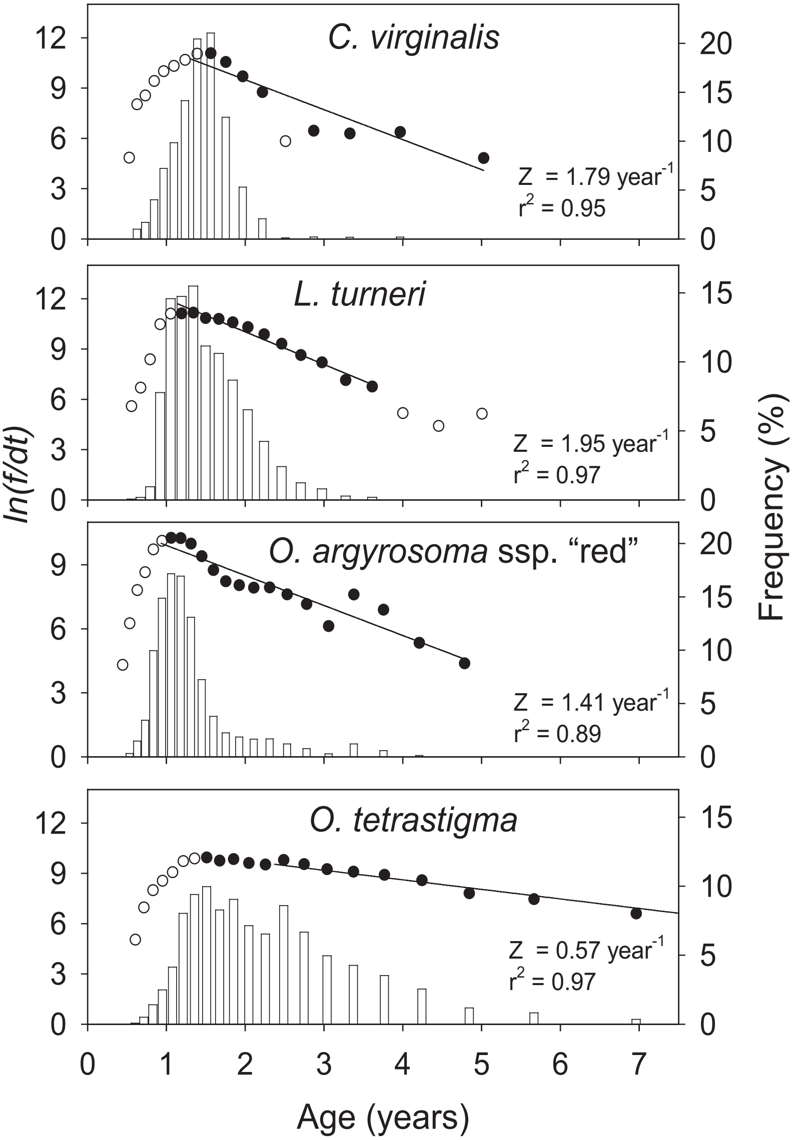

Figure 3. Length-age converted catch curves for *Copadichromis virginalis*, *Lethrinops turneri*, *Otopharynx argyrosoma* "ssp. red," and *O. tetrastigma* sampled from the nkacha net fishery in Lake Malombe, March 2000 to April 2001. Total mortality (Z) estimates were obtained through regression analysis of the descending limb (closed circles) of the linearized catch curve (closed circles). The ascending limb of the catch curve and outliers (open circles) were excluded from analysis.

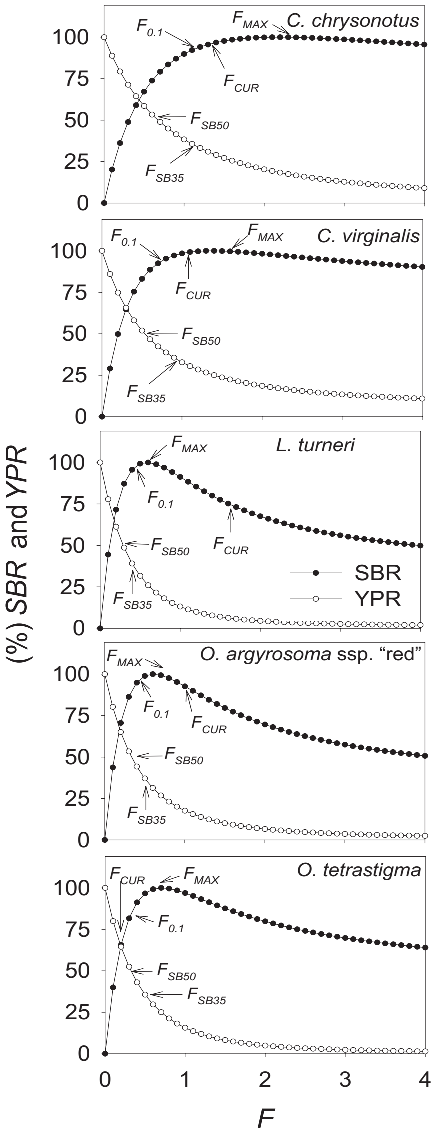

Figure 4. The response of *Copadichromis chrysonotus, C. virginalis, Lethrinops turneri, Otopharynx argyrosoma* "ssp. red," and *O. tetrastigma* yield-per-recruit (YPR) and spawner biomass-per-recruit (SBR) to fishing mortality (*F*) in Lake Malombe, Malawi. Point estimates for the current fishing mortality rate, F_{CUR}, and four target reference points, F_{MAX}, $F_{0.1}$, F_{SB50}, and F_{SB35} are illustrated. (*C. chrysonotus* data are from Weyl et al. 2005.)

30% which exceeds the F_{SB35} TRP. *L. turneri* is severely over-exploited from both an YPR and SBR perspective with F_{CUR} being more than twice as high as F_{MAX} and SBR being depleted to about 6% of pristine levels. In *O. argyrosoma* "ssp. red," F_{CUR} also exceeds F_{MAX} and depletes SBR to about 17% of unexploited levels. With F_{CUR} well below that required to attain $F_{0.1}$ and an SBR of about 66% of pristine levels, the *O. tetrastigma* stock appears healthy.

From a TYPR perspective, F_{MAX} was attained at *f* levels of about 130 gears and $F_{0.1}$ was attained at an *f* of 68 gears (Fig. 5). Effort levels of 130 gears, while lower than current effort, would not lead to a significant rebuilding of the SBR in the four over-exploited species. The $F_{0.1}$ strategy would maintain the SBR of *C. chrysonotus*, *C. virginalis,* and *O. tetrastigma* at levels above 50% of pristine, and reduce *L. turneri* and *O. argyrosoma* "ssp. red" SBR to 19% and 39% of pristine SBR, respectively.

Discussion

It has long been recognized that when a common gear harvests a number of species, it is impossible to manage each species at its optimum level (Beverton and Holt 1957, Anderson 1975, Pope 1979, Pikitch 1987). This is due to differences in the life histories of the different species which infers different population responses to exploitation rates as well as behavioral differences that affect the availability of each species to the gear (Murawski 1984). For instance, slow growing species with low natural mortality rates, long life-spans, late maturity, and precocial breeding habits are more likely to have relatively stable population sizes, and therefore relatively stable catch levels (Adams 1980). However, once they are overfished, it would require a relatively long period (depending on the extent of overfishing) for the stock to rebuild. Conversely, fast growing species with high rates of natural mortality, early maturity, and altricial breeding habits would support more productive fisheries, where fish can be harvested at younger ages from the population. However, these fisheries are likely to be of a "boom and bust" nature (Adams 1980), characterized by high initial stock sizes, and the potential for both growth and recruitment overfishing.

These effects are evident in the target species of the nkacha net fishery. While all five species are maternal mouth-brooders (Turner 1996), *L. turneri* had the latest maturity and lowest natural mortality rate (Table 1). Subsequently, this species appeared to be most vulnerable to the fishery with relatively low effort levels leading to asymptotic YPR and severe SBR depletion. At current effort levels, this species is over-exploited from both an YPR and SBR perspective. This over-exploitation is also evident from catch estimates for this species which show a decline in *L. turneri* catch from about 2,200 tons in 1991 to less than 900 tons in 2001 (Weyl et al. 2004).

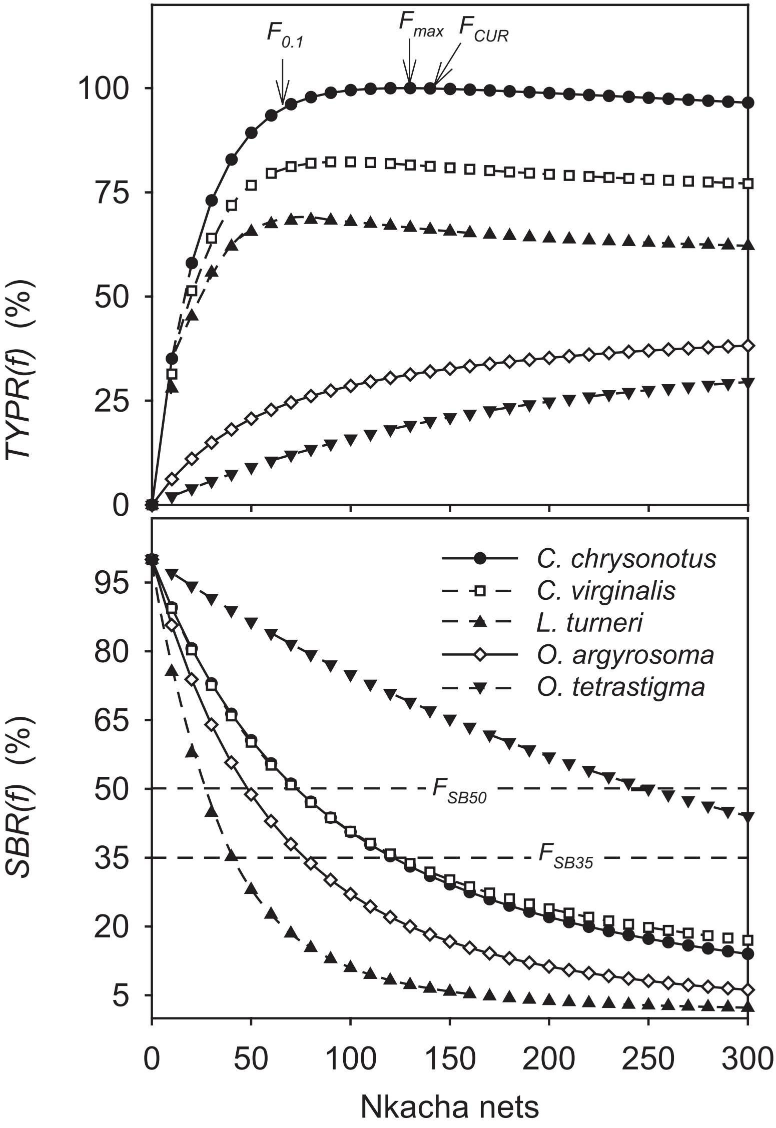

Figure 5. The response of total yield-per-recruit (TYPR) and spawner biomass-per-recruit of *Copadichromis chrysonotus, C. virginalis, Lethrinops turneri, Otopharynx argyrosoma* "ssp. red," and *O. tetrastigma* as a function of effort (expressed as the number of nkacha nets) in Lake Malombe, Malawi, Africa. Note that for TYPR the yield-per-recruit for each species considered is additive and expressed as a percentage of the maximum.

Conversely, the two *Copadichromis* species had the fastest growth rates and highest natural mortality rates. Consequently, the same effort levels that over-exploit *L. turneri* resulted in less severe YPR and SBR reduction in these two species. Despite similar catchability coefficients and SBR reductions to about 30% of pristine levels, *C. chrysonotus* catch increased from about 120 tons in 1991 to about 510 tons in 2000, while *C. virginalis* landings declined by more than 50% over the same period (Weyl et al. 2004). While this implies a higher vulnerability of *C. virginalis* to the fishery, further investigation of this effect was not possible.

The two *Otopharynx* species had similar growth and natural mortality rates, but responded differently to the fishery. Current effort levels exceed the F_{MAX} TRP for *O. argyrosoma* "ssp. red" and deplete its SBR to about 17% of unexploited levels (Fig. 4). *O. tetrastigma*, on the other hand, appears far less vulnerable to the gear and the species is underexploited on a YPR basis and its SBR is estimated to exceed 66% of unexploited levels (Fig. 4). Such effects imply different vulnerabilities of the five species to the fishing gear, which may be a consequence of differences in spatial distribution or behavior. Unfortunately, little is known about the distribution of the five species within the lake. What is known is that in the adjacent Lake Malawi, *O. tetrastigma* favors well-vegetated areas, while *O. argyrosoma* "ssp. red" inhabits open areas (Konings 1995). It is therefore possible that a part of the *O. tetrastigma* population is unavailable to nkacha nets, which operate in offshore areas. It must also be considered that, due to the shallow depth (maximum depth of 5 meters) and uniform sandy/muddy bottom, most of the lake is fishable with nkacha nets and spatial exclusion would be valid only for species favoring extremely shallow areas. The lack of knowledge on distribution and behavior of the five species therefore forms a major bottleneck in understanding the dynamics of the system. But all species have the same commercial value and it is unlikely that the fishery would target any one of the five species preferentially.

The Malawi fisheries policy (Government of Malawi 1999) stresses the maximization of harvests within safe sustainable yield levels. Peak yield-per-recruit is attained if an infinite fishing mortality is applied when the biomass of a cohort is at its maximum (Pereiro 1992). However, it is evident that, at current recruitment levels, higher yields are unlikely to be attained from this fishery (Fig. 5). In mouth-brooding cichlids recruitment may be dependent on spawner-biomass and the maintenance of the spawner stock at levels where it can replace itself is vital. Taking SBR recommendations for other species (Gabriel et al. 1989, Quinn et al. 1990, Clarke 1991, Punt 1993, Mace 1994, Caddy and Mahon 1995, Booth 2004) into account, the maintenance of SBR at levels between 35% and 50% of pristine was taken as an initial management target and, ideally, the SBR of all five major target species should be maintained at levels exceeding 35% of unexploited biomass.

A visual representation of the possible management choices, presented in Fig. 6, shows the variability inherent to the F_{SB35} management targets. Taking into account the variability of the F_{SB35} estimates, current effort levels are likely to maintain the SBR of *C. chrysonotus*, *C. virginalis*, and *O. tetrastigma* within acceptable levels. Effort would have to be reduced to at most 88 nkacha nets if the F_{SB35} TRP is to be attained for at least four species. If SBR is to be maintained at a F_{SB35} TRP for all five species, effort would have to be reduced to about 40 gears. However, as the nkacha net fishery cannot target any species in isolation and the market price for all five species is identical, a strategy maximizing total yield from the fishery, without regard for individual SBR, might be favored. But in the absence of information on interspecies effects and the SBR-recruitment relationship, such a strategy cannot be recommended.

While per-recruit models are relatively data-intensive and some data from previous studies (Banda 1995, Weyl et al. 2004, Weyl et al. 2005.) were used in the analysis, all input parameters for the models could have been estimated during a one-year assessment period. But it must be realized that the per-recruit models assume that age-specific maturity, age-specific selectivity, the growth equation, natural mortality and the current rate of fishing mortality are constant, that there is no recruitment variability, and that it is possible to impose a specified fishing mortality (Punt 1993). Despite these constraints, management is an immediate rather than a future concern in African fisheries and a per-recruit approach enables the rapid provision of a number of management options for a fishery given constraints in available historic data. It must also be noted that the age estimates derived from length-frequency analysis in this study only provide a first-estimate of growth for the five species until alternative aging methods can be applied. In this regard, sectioned otoliths are currently considered the most suitable hard tissue for age and growth determination in tropical and subtropical areas (Campana 2001). For this reason, it is vital that the per-recruit approach presented in this paper is seen as a first-assessment and that efforts be made to collect data for the future application of more quantitative methodologies.

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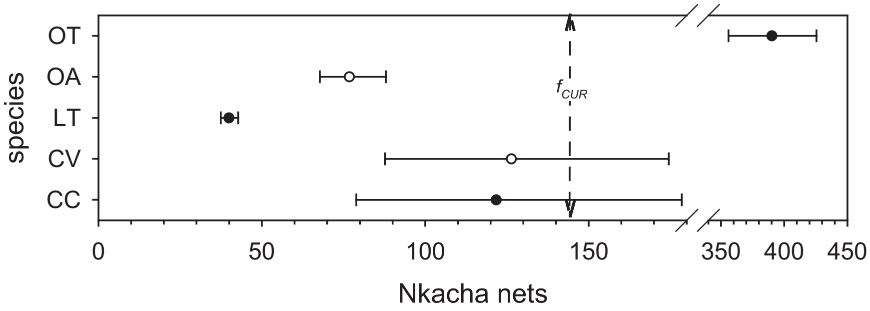

Figure 6. Mean and 95% confidence intervals (CIs) for the F_{SB35} target reference point (TRP) expressed as a function of fishing effort for *Copadichromis chrysonotus* (CC), *C. virginalis* (CV), *Lethrinops turneri (LT), Otopharynx argyrosoma* "ssp. red" (OA), and *O. tetrastigma* (OT) in the nkacha net fishery of Lake Malombe, Malawi, Africa. The 95% CIs were derived from a Monte Carlo estimation procedure using 1,000 iterations for five target species in the Lake Malombe nkacha net fishery. Current fishing effort f_{CUR} is illustrated.

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Do Commercial Fishery CPUE Data Reflect Stock Dynamics of the Baltic Herring?

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Abstract

Herring are important commercial fisheries of the Baltic Sea. The structure and dynamics of Baltic herring stocks has been monitored thoroughly since international management started in the mid-1970s but management has not always been successful, particularly during periods of rapid change. Shortcomings in certain assessment data are a concern and the difficulty of obtaining appropriate biological samples from the huge and extremely heterogeneous herring stocks is a problem. Another problem is the lack of good tuning data for age-structured analyses. Presently the results of an annual International Acoustic Survey are used for tuning in extended survivors analysis (XSA). A critical shortcoming is that the acoustic survey does not cover several key areas of herring distribution. Therefore, there is a need for alternative or supplementary data to minimize effects of biased acoustic results. CPUE (catch per unit effort) in commercial trawl fishery may be a potential source of additional data. In theory, the CPUE should be proportional to the stock size so changes in CPUE, adjusted for gear efficiency, could provide, at low cost, supplementary information on stock trends. The CPUE data collected from the Estonian pelagic trawl and pound-net fishery (a small part of the larger Baltic herring fishery) show good accordance with the results of VPA estimates of separate stock units in 1990-2000s. Further exploration of commercial CPUE as a stock status index would be reasonable.

Introduction

Pelagic stocks provide the vast majority of commercial fish production of the Baltic Sea. The total landings of Baltic herring (*Clupea harengus membras*) and the Baltic Sprat (*Sprattus sprattus balticus*) exceeded

600,000 t, sometimes reaching 700,000 t in the last decade. Like herring in other parts of the world, Baltic herring form local stocks or populations (Stephenson 1991). Probably there are nine or ten different Baltic herring populations, each with different biological characteristics and stock dynamics (Ojaveer 1981, Stephenson 1991). Three herring populations, the Gulf of Riga herring, the Gulf of Finland herring, and the open-sea herring of the northeastern Baltic, inhabit the northeastern part of the Baltic Sea. The gulf herring stocks spend all year within gulf waters, while the open-sea herring occur mostly in the Baltic proper, performing spawning migrations to the spawning grounds located in the gulfs or archipelago area.

The dynamics of Baltic herring populations (stocks) have been assessed and managed internationally since the mid-1970s. In early years, the separate assessments were performed for several local stocks, mostly defined on geographical basis (ICES 2001b). Since 1990, all local herring populations in the main basin and in the Gulf of Riga and the Gulf of Finland were combined and assessed as one stock (central Baltic herring in subdivisions 25-29 and 32). At present, the Baltic herring is assessed in five assessment units (Fig. 1):

- Herring in subdivisions 22-24.
- Herring in subdivisions 25-29 and 32 excluding the Gulf of Riga.
- The Gulf of Riga herring.
- Herring in subdivision 30.
- Herring in subdivision 31.

The international management of those stocks has not always been successful, particularly in case of the central Baltic herring where biomass has decreased continuously throughout the approximately 30 year period of observations (Ojaveer 2002). Therefore the complex stock structure of the Baltic herring could be one of the reasons for difficulties in understanding and managing these stocks.

There may be other reasons; for example difficulties in data collection also can have a substantial role (Raid 2002). In general, there are only few problems with collecting of general biological data from the commercial catches, discards, and landings. The respective data sets cover all Baltic herring fisheries, areas, and seasons for past 30 years reasonably well (ICES 2003). However, the tuning data for analytical assessments are fragmentary and this has been one of the shortcomings of herring assessment for years. At present, the results of the annual International Acoustic Survey (IAS) are used in order to tune the VPA in the XSA (extended survivors analysis) in the routine herring assessments by the ICES Baltic Fisheries Assessment Working Group. The IAS, initially directed on sprat

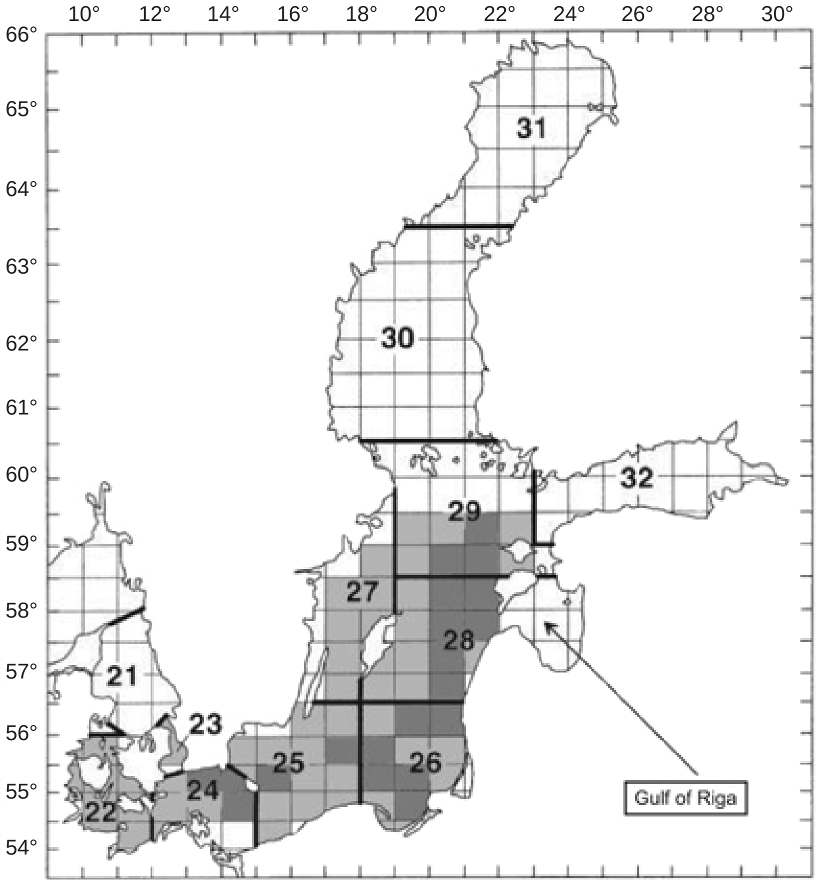

Figure 1. **ICES subdivisions in the Baltic Sea. The light-shaded area shows the coverage by the International Acoustic Survey. The overlapping areas between different national surveys are dark-shaded.**

stock, does not allow full coverage of the herring distribution area. Areas do not include a substantial part of the distribution area of the central Baltic herring, most of subdivision 29, the Gulf of Finland (subdivision 32), and the Gulf of Riga (part of subdivision 28) (Fig. 1). Consequently one herring population, the open-sea herring in the northeastern Baltic, has only partial coverage, while two populations (the Gulf of Finland herring and the Gulf of Riga herring), are not covered at all (Fig. 1). Therefore although there are considerable basic biological data on catch composition, there are essential gaps in the key data sets used for tuning of the assessments. In this sense, the Baltic herring fishery is data-limited. The limitation is not with the “amount” of data (which is voluminous) but rather with the limited geographic coverage. The main reason for poor coverage is high cost of acoustic surveys. In that respect, there is a clear need for alternative or supplementary data in order to fill the gaps in the existing tuning data and thus minimize the effect of inadequate acoustic estimates on assessment results.

CPUE data (catch per unit effort) from the commercial fishery may be a source of additional data for the herring stocks in the northeastern Baltic. The CPUE of the Finnish trawl fishery is routinely used in the assessments of herring stock in the Bothnian Sea (subdivision 30) and in the Bothnian Bay (subdivision 31; ICES 2001a, 2002, 2003). In theory, the CPUE of the commercial fishery should be proportional to the stock size (e.g., Gulland 1964), and consequent change in CPUE should track, at given gear efficiency, the dynamics in fish stocks. Such additional CPUE information could provide a low cost alternative or supplement to acoustic estimates used for tuning. The aim of the present paper was to explore if CPUE in commercial fishery could be a possible additional data source on stock status of the Baltic herring in the northeastern Baltic.

Materials and methods

Biological data for this study was collected from the Estonian commercial trawl and pound-net fishery during the routine data collection for assessment purposes. A sample of 100 fish was collected and analyzed every 10 days for all fishing seasons and fishing grounds in Estonian waters of subdivisions 28, 29, and 32 and in the Gulf of Riga (Fig. 1). Most of the data used in present work are from the period 1995-2002.

The pelagic trawl fishery takes about 95% of Estonian herring catches in most fishing areas, except the Gulf of Riga. In general, the fishing fleet and the gears have not been changed in the recent decade. Usually this fishery exploits the densest herring concentrations. Since no discarding or sorting occurs in the Estonian trawl fishery, catch data can be used to characterize the main trends in herring stocks. Approximately 5% of herring catches are taken by the pound-net fishery (but this can be up to 50% in the Gulf of Riga). The pound nets are big fixed trap-nets with open tops.

They are used in shallow coastal areas during the spawning season (second quarter of the year, April-June) to catch pre-spawning herring. Therefore only the spawning stock is represented in the pound-net fishery.

The CPUE data in trawl fishery (kilograms per hour) were compiled on a quarterly basis according to logbooks of fishing vessels. The respective data for herring pound-nets, in metric tons per check (the fishermen check the pound-net catches 3-5 times a week), were compiled on a monthly basis. The CPUE of the pound-net fishery was explored in the Gulf of Riga only (the main basin for that fishery). The VPA estimates of stock components were derived either from the reports of the ICES Baltic Fisheries Assessment Working Group (Gulf of Riga herring, central Baltic herring) or produced ad hoc, using input parameters (catch numbers, mean weights, and natural mortalities, etc.) available for the respective subdivisions in the same sources (ICES 2001a,b, 2002, 2003).

Results and discussion

The spawning stock biomass (SSB) of the complex central Baltic herring, including several local populations, has decreased by approximately 70% since 1974 (ICES 2003). Landings decreased from 300,000 t to below 200,000 t. Fishing mortality estimates have been in the range of 0.2-0.3; they were above the defined F_{PA} ($F = 0.19$, ICES 2003) until 1994, increasing thereafter up to almost 0.5 in 2000 (Fig. 2). The separate assessments of different stocks and results of hydroacoustic surveys of different subunits (herring in subdivisions 25-27, subdivisions 28, 29, and 32 and the Gulf of Riga herring) have revealed variation within pooled assessment units. For instance, fishing mortality of the Gulf of Riga herring has decreased but there was a sharp increase in mortality in the northeastern Baltic (subdivisions 28, 29, and 32). At the same time stock abundance and biomass of the Gulf of Riga herring increased to record high levels in early 2000s also allowing higher catches (ICES 2002, 2003; Fig. 3).

The average CPUE data from the Estonian herring pelagic trawl fishery, recorded since 1995, is presented in the Table 1. The comparison of the dynamics of mean CPUE and the SSB estimates in respective assessment units show similar trends (Figs. 2 and 3). However, the degree of coherence varies—the closest associations were between subdivision 32 (Gulf of Finland) and for the combined unit subdivisions 28, 29, and 32 (Figs. 4 and 5). The fit between the mean CPUE and the SSB estimate in subdivision 29 was the poorest. The results indicate that for the Gulf of Riga and subdivision 32, the trends in CPUE of the commercial trawl fishery are quite similar to the SSB trends. Deficiencies in the quality of VPA estimates due to the complexity of herring in the area might be the reason for the poor fit observed for subdivision 29. The area is an essential feeding area and a migratory area for several local herring stocks that mixing there, particularly during the late autumn and wintering period.

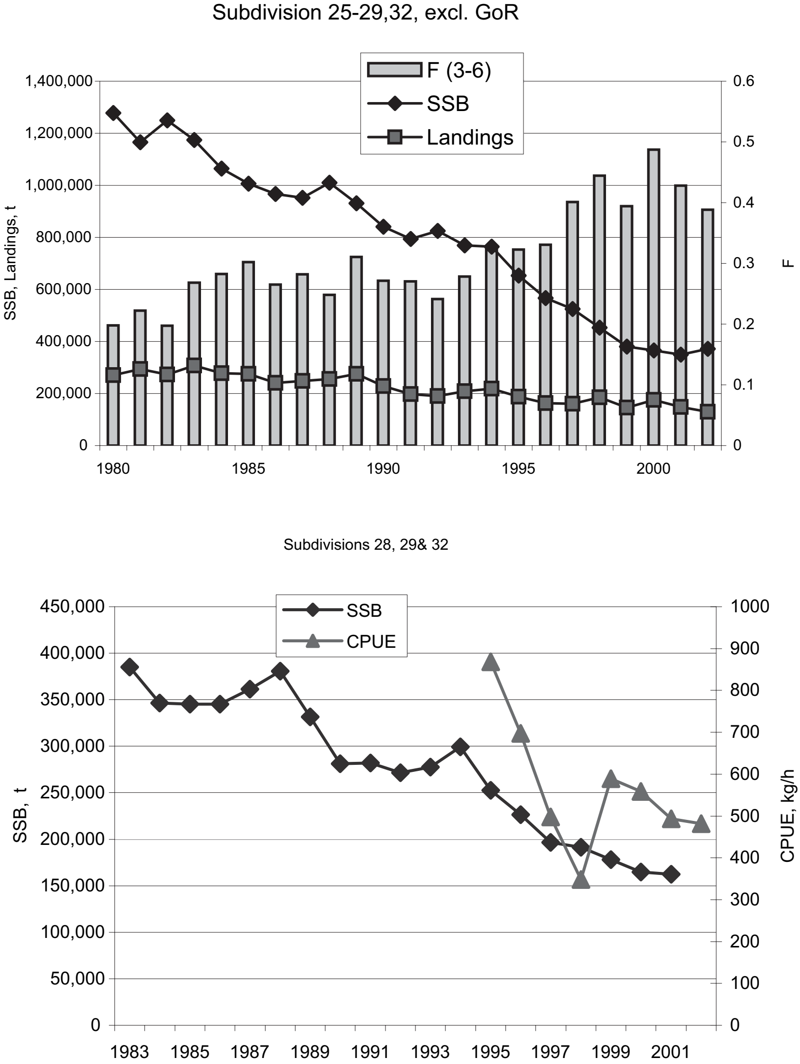

Figure 2. Upper panel shows dynamics of SSB, landings, and fishing mortality in the central Baltic herring stock in the 1980-2000s. Lower panel demonstrates SSB and CPUE trends in subdivisions 28, 29, and 32 (ICES 2002, 2003).

Table 1. Mean catch per hour in the herring pelagic trawl fishery (kg).

ICES sub-division, zone	1995	1996	1997	1998	1999	2000	2001	2002
Gulf of Riga	1,009	707	637	667	700	697	804	820
29	1,035	892	802	671	993	836	771	419
32-1	715	572	373	271	441	394	300	184
32-2	629	498	410	241	417	370	272	538
32	672	559	377	267	438	392	297	264
Mean	868	697	498	348	589	559	483	482

CPUE data may not be normally distributed so several authors advocate using median CPUE instead of the arithmetic mean (Hamley and Howley 1985, Reed 1986, Järvik 1989). The comparison of CPUE data from the herring pound-net fishery in the Gulf of Riga has a poor fit, regardless whether the simple arithmetic mean or median of CPUE was used (Table 2, Fig. 6). The absence of correlation could be explained by the pattern of the pound-net fishery, concentrated mostly on major spawning grounds. The actual number of fish in pre-spawning schools at each particular spawning ground depends on environmental factors, such as wind direction and temperature, and therefore can be rather variable regardless of spawning stock size. However, the total annual pound-net catches were found to be a quite good indicator of Gulf of Riga herring SSB (Järvik and Raid 2000).

The commercial CPUE data used in the present study cover only a small fraction of the conventional assessment units. To determine if CPUE data obtained in the limited part of the assessment unit is representative as an indicator of stock developments, the arithmetic mean of commercial CPUE of Gulf of Finland herring in 1995-2002 was compared with the available set of respective SSB estimates for different herring stock units and their combinations. The results (Fig. 7) indicate that the commercial CPUE data, obtained from the remote northern part of the distribution area of the central Baltic herring stock complex provides a satisfactory description of the main SSB trends in both combined stocks (subdivisions 25-29 and 32, and subdivisions 28, 29, 32).

A much looser relationship was found between the mean CPUE in the Gulf of Finland herring fishery and the SSB estimate for the neighboring subdivision 29, which probably indicates again at the uncertainties in the SSB estimates for that subdivision.

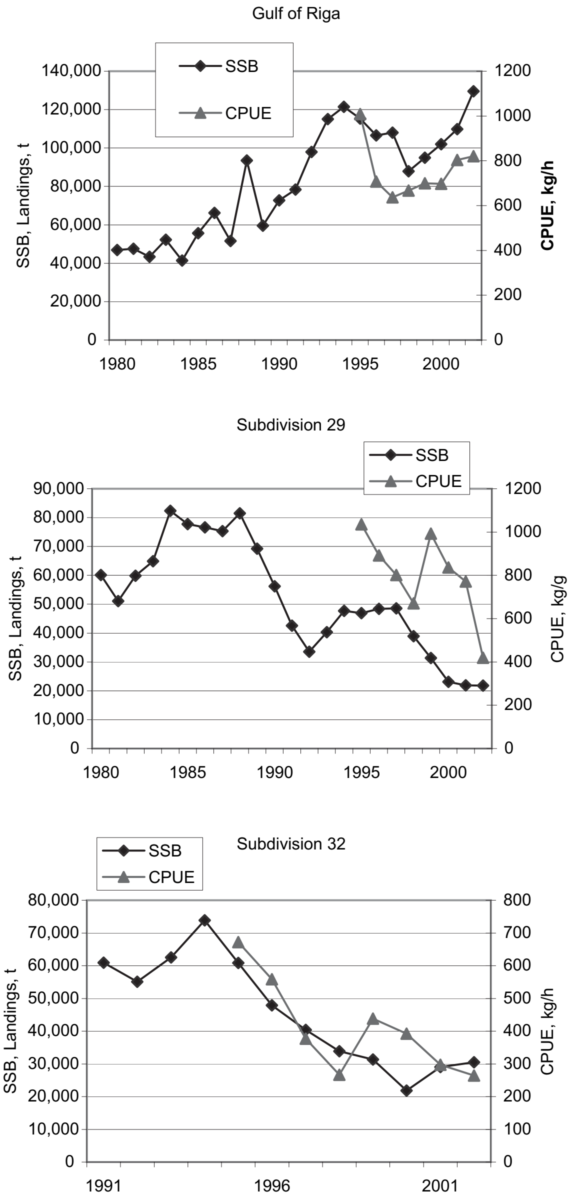

Figure 3. Trends in SSB and CPUE of herring in the Gulf of Riga (ICES 2003), and in subdivisions 29 and 32.

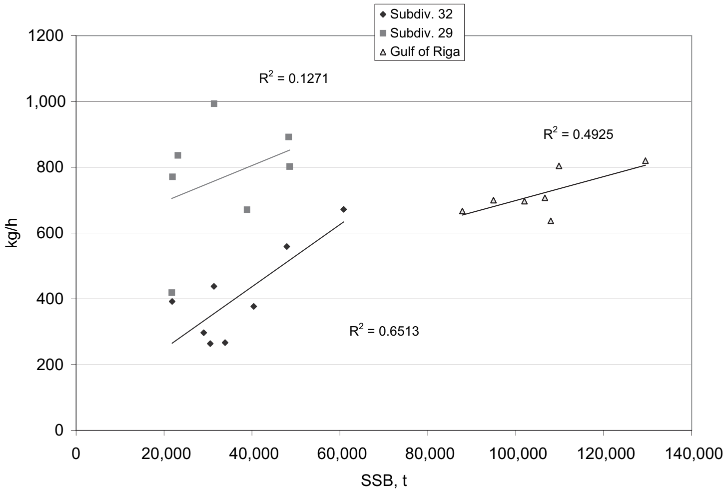

Figure 4. **Scatter plots of CPUE and SSB estimates in subdivision 32, subdivision 29, and Gulf of Riga assessment units (1995-2002).**

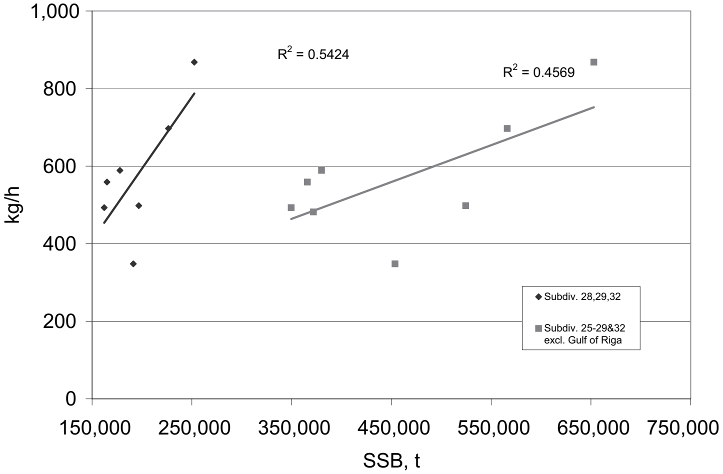

Figure 5. **Scatter plots of CPUE and SSB estimates in subdivisions 28, 29, and 32, and subdivisions 25-29 and 32 excluding the Gulf of Riga assessment units (1995-2002).**

Table 2. Mean and median catch per check in May-June in the pound-net fishery in the Gulf of Riga (t).

Category	1995	1996	1997	1998	1999	2000	2001	2002
Mean	2.62	2.77	1.92	1.70	2.40	1.27	1.62	2.29
Median	2.61	2.85	1.80	1.71	2.34	1.06	1.61	2.19

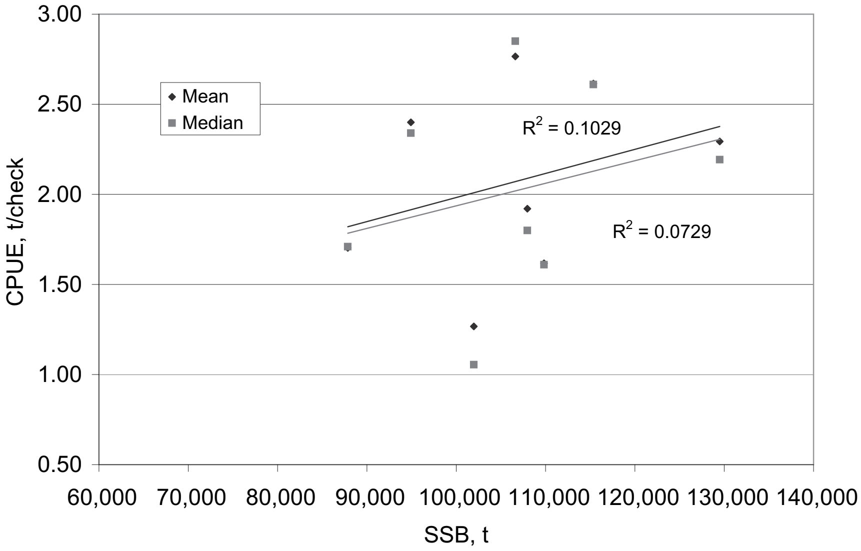

Figure 6. Scatter plots of pound-net CPUE and SSB estimates for Gulf of Riga herring (1995-2002).

No relationship was revealed between the Gulf of Finland herring CPUE and SSB estimates of the Gulf of Riga herring. The absence of any correlation in that case is not surprising, because of the fundamentally different pattern in stock developments in the Gulf of Finland and the Gulf of Riga (Fig. 3).

Although there is a relatively good temporal coherence between commercial CPUE and herring SSB estimates, the use of commercial CPUE as an indicator for stock trend should be taken with caution because the CPUE estimation depends on the distribution pattern of fish, and sometimes also on the behavior of the fishing vessel captain. For example,

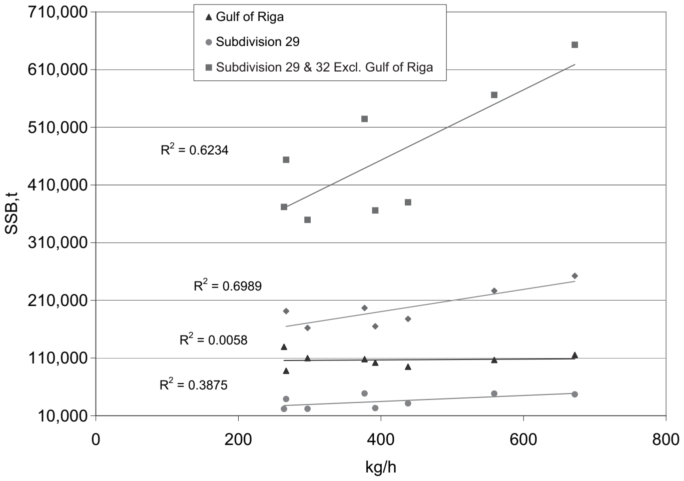

Figure 7. Mean trawl CPUE from subdivision 32 plotted against SSB estimates for different assessment units (1995-2002).

during daylight herring CPUE tends to be higher because herring are in dense shoals, whereas during the dark of night, herring are less dense and distributed more evenly. Hence, the source of CPUE data should be examined critically prior to including data in the assessment input.

Conclusions

Baltic herring fisheries are not limited by the *amount* of available data, but perhaps by the suitability of existing data. In particular, many of the Baltic herring fisheries do not have sufficient data to accurately describe relative trends in stock size. The tentative exercise of comparison of general trends in herring SSB estimates in different assessment units in the northeastern Baltic, to commercial CPUE data from the pelagic trawl fishery, show rather good fits in the case of units with less uncertain assessments. The CPUE data covering only a part of the remote section of the central Baltic herring stock follow the main trends in SSB estimates of the combined stock rather well, suggesting that further exploration of the possibilities of implementation of commercial CPUE data into assessment procedures of Baltic herring would be reasonable.

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Detecting Early Warnings of Recruitment Overfishing in Male-Only Crab Fisheries: An Example from the Snow Crab Fishery

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Abstract

Male-only crab fisheries often are assumed to be relatively well protected from recruitment overfishing by virtue of male polygyny and female sperm retention. These fisheries also offer the opportunity to directly assess the per capita female reproductive contribution. If the latter does not dwindle when a stock declines, then the decline cannot be attributed to overfishing. We explored expedient ways to assess per capita female reproductive contribution of snow crab (*Chionoecetes opilio*) and Tanner crab (*C. bairdi*) of the Eastern Bering Sea (EBS) by measuring clutch volume and estimating an objective "clutch fullness index" (CFI) based on modeling the "maximum boundary line" (MBL) of scatterplots of clutch volume vs. body size. We investigated three estimation methods, selected quantile regression for further analysis, and examined the distribution of the CFI in extensive samples collected during trawl surveys conducted in 1992 and 1993. An objective protocol for the visual assessment of clutch size (based on an estimated MBL) was introduced in 1994, with a dramatic effect on estimated CFI. CFI distributions proved very useful for

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the identification of patterns in the data. Average CFI declined between 1994 and 2000, then rebounded in 2001.

Introduction

The substantial sexual dimorphism of many commercially valuable crab species creates the opportunity for sex-selective (male-only) harvest tactics (Kruse 1993). Harvest rates imposed on males above the legal size in crab fisheries are often very high (Methot and Botsford 1982, Dufour and Coutu 1989, Zheng et al. 2002), yet it has been argued that brachyuran ("true") crab stocks are reasonably well protected from recruitment overfishing by virtue of male polygyny and female sperm storage. The stock of mature sublegal males and the escapement (albeit small) of legal-size males would suffice to fertilize the pool of receptive females. In addition, mated females would be able to retain sperm and use it in subsequent reproductive seasons, even if they did not mate again. Those assumptions, however, have been challenged by results indicating that there are limits to the long-term viability of stored sperm (Paul 1984, Paul and Paul 1992) and, maybe, to effective male polygyny (Sainte-Marie et al. 1995, 2002). Consequently, there is much current interest in assessing the possible occurrence of sperm limitation and recruitment overfishing in male-only crab fisheries (Kendall et al. 2001).

Interestingly, crab fisheries allow (at least in principle) for rather direct monitoring of the reproductive status of stocks. The reproductive contribution of an individual female in a given reproductive season can be estimated from the size of the clutch of fertilized eggs attached to its pleopods. If female survival is not affected by the fishery, then an absence of a trend in the per capita reproductive contribution of mature females implies no recruitment overfishing, even in the face of a stock decline (Orensanz et al. 1998). Making use of this opportunity requires the development of practical techniques for the assessment of female reproductive contribution, and the development of indicators that take into account the effect of significant factors, body size being the most obvious.

Scattergrams of female crab fecundity or clutch volume vs. body size (carapace width, CW) are typically "triangular": maximum clutch volume, conditioned on size, is constrained by the morphology of the incubating chamber underneath the abdominal flap (Figs. 1, 2). Data points are spread below the maximum boundary line (MBL), as the egg mass can be smaller than its possible maximum size due to senescence, egg mortality, incomplete fertilization, energetic limitation, etc. Estimation of the parameters of a boundary condition would allow the conversion of the observed clutch size (whether expressed as egg counts, clutch weight, or clutch volume) into an objective "clutch fullness index" (CFI), a reasonable indicator of the reproductive contribution of individual females relative to their maximum possible.

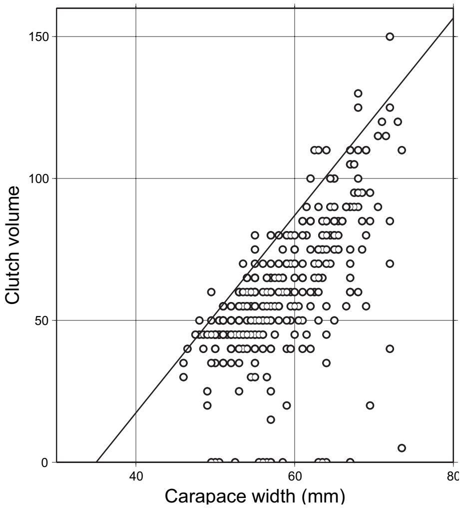


Figure 1. Clutch volume (in tenths of a ml) measured in a sample of snow crab collected during summer 1991 (NMFS survey).

The snow crab (*Chionoecetes opilio*) and Tanner crab (*C. bairdi*) fisheries of the Eastern Bering Sea offer good examples of the male-only harvest strategy. Information pertinent to female reproductive condition (size, maturity, shell condition, eggs color, relative clutch size) has been collected regularly during extensive surveys conducted by the U.S. National Marine Fisheries Service (NMFS) since the mid 1970s (Otto 1998, Stevens et al. 2002), but it has never been used to assess female reproductive potential. In this study we investigate practical indicators of female reproductive contribution in male-only crab fisheries using the EBS snow crab fishery as an example. We also present and discuss some preliminary results on Tanner crab. Specifically, we (1) explore different statistical methods for the estimation of parameters of boundary lines, (2) apply those to clutch volume data, and (3) examine the information retrieved from surveys before and after an objective CFI was introduced in 1994.

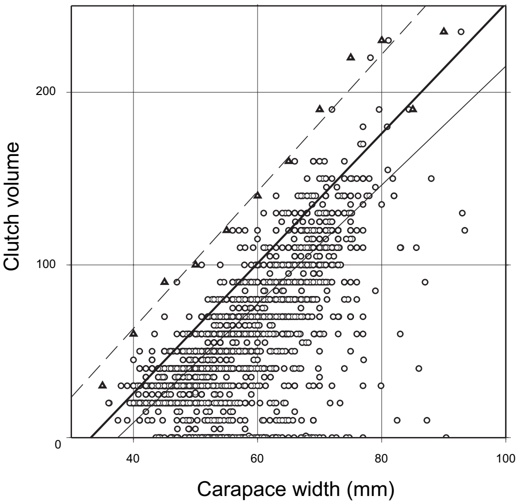


Figure 2. **MBLs (maximum boundary lines) fitted to snow crab clutch volume (in tenths of a ml) vs. body size (CW mm) data, 1992-1993 NMFS surveys combined, using three different statistical techniques. Thick line = quantile regression; dashed line = SSQ (sum of squares); thin line = trimming method. Triangles indicate the data retained for estimation of the MBL by the SSQ method.**

Material and methods

NMFS survey data

Data utilized in this study were collected during annual trawl surveys conducted by NMFS between 1975 and 2001 (Otto 1998, Stevens et al. 2002). Surveys follow a systematic sampling design, where stations are regularly spaced over a 20 nm × 20 nm grid. The core temporal window of the surveys is June-July. Several observations are made on each crab caught in a haul, or from a subsample when the catch was too large (for further information on the sampling protocol see the *Manual of crab sampling methods for the trawl survey in the Eastern Bering Sea*, available from NOAA Fisheries, Alaska Fisheries Science Center, PO Box 1638, Kodiak, AK 99615-1638). Results presented here for statistical populations (e.g., snow crab females from a given region) incorporate weighting factors that reflect the fraction subsampled. In the case of females, observations included carapace width (CW, in mm), sexual maturity based on shape of the abdominal flap (immature, mature), shell condition index (SCI), egg color, and clutch size. Measurement scales for the latter three follow.

Shell condition index

Female snow and Tanner crab undergo a terminal (puberty) molt. Their first mating follows immediately after that molt, while their shell is still soft (primiparous mating). During subsequent years females mate after the eggs hatch, and then the shell is hard (multiparous mating) (Somerton 1982, Conan et al. 1990, Stevens et al. 1994). The appearance of the shell changes gradually after the terminal molt due to wear and colonization by epibionts. Subjective SCIs based on shell appearance, ranging between 0 and 5, are roughly correlated with shell age; their interpretation has been discussed by Ernst et al. (2005). Females in SCI-0 and SCI-1 (ready to molt or molted within days before capture) are very uncommon in the surveys (Ernst et al., 2005), and were not included in our analyses. Females in SCI-2 and SCI-3 largely correspond to, respectively, primiparous and first-time multiparous females. Females in SCI-4 and SCI-5 (grouped here as category SCI-4^+) generally correspond to older multiparous females (two years or more past terminal molt).

Egg color

A scale of five colors was used, which in practice collapses to two clusters: yellow/orange, which are at least one year away from hatching, and brown/purple (eyed embryos), which are close to hatching. Female snow crab can have an annual or biennial reproductive cycle depending on the temperature regime experienced; in Tanner crab the cycle is always annual (unpubl. personal observations). Yellow/orange eggs can correspond to eggs that were spawned within recent months, or (in the case of a biennial cycle only) to eggs that have been carried by the female for about one year and will hatch about one year later. Only females with yellow/orange eggs were used in this study. The size of the clutch of females with brown/purple eggs (only 1.5% of the total, 1975-2001 surveys pooled) maybe comparatively small due to egg mortality during incubation. The rate of egg mortality cannot be estimated with the data available.

Relative clutch size

A visual scale of seven categories is used as a subjective index of fullness: over-full (“eggs bulging outside tail flap”), 100%, 75%, 50%, 25%, traces to 12.5%, empty.

Special data collected during the 1991-1993 NMFS surveys

During the 1991 survey Drs. Bob Otto (NOAA Fisheries, Kodiak) and Bob Elner (Canadian Wildlife Service, Delta) collected a sample of snow crab females ($n = 416$) that was frozen on board and later processed by author LO at the laboratory. The main purpose was to develop an objective index of clutch size to be used in a larger sampling program conducted during the 1992 and 1993 surveys. The abdomen of all specimens was photographed (open and closed) before processing. In addition to the information regularly recorded during the NMFS surveys (including the subjective visual index of clutch size), the protocol included the width of the pleon, presence and type of mating marks, volume of the egg mass (tenth of a ml), condition of the ovary (full, empty, partially evacuated, evacuated with residual eggs), color of the ovary (white, yellow, pink, orange, red), aspect of the sperm mass in the spermathecae, and reproductive status (primiparous vs. multiparous).

In the course of the 1992 and 1993 field seasons an extensive systematic sampling program of ovigerous females was conducted involving the entire Eastern Bering Sea NMFS summer survey. The target aimed at 10 specimens of each species per haul, but a smaller number was typically obtained. Given the geographical extension of the survey, the samples covered most of the distributional range of the female populations of both species (Stevens et al. 2002). Samples were frozen on board; due to constraints in holding space, legs were cut off before freezing. Specimens were processed in the laboratory by author LO. Total number of specimens processed was 2,083 in 1992, and 2,004 in 1993. Observations made in addition to those routinely recorded during the surveys included an objective scale of the color of eggs (Eiseman and Herbert 1990), volume of the egg clutch (in ml), color and dry weight of the ovaries (1993 only), and data on the spermathecae. SCI of specimens processed in the laboratory was considered unreliable; color of the carapace changed as a result of freezing-defrosting and leg removal impeded the assessment of the full complement of classification criteria; SCI was not used in the analysis of the data.

Changes in the protocol for recording clutch size after the 1992/1993 surveys

Clutch-size data obtained in 1991 were used to estimate an MBL to approximate maximum clutch size conditioned on body size (Fig. 1). Using the latter, individual clutch volume measurements were converted into objective indices of clutch fullness. Specimens that exactly matched the clutch size categories utilized in the NMFS surveys were selected, and their photographs utilized to prepare a chart to be used experimentally in the field, the ultimate purpose being to improve the objectivity of the assessment of clutch fullness on board. The chart was tried during the

Table 1. Statistical techniques used for fitting linear boundaries to "triangular" data sets.

Estimation technique	Point estimate	Uncertainty	Strong statistical basis	Reference
Quantile regression	Yes	Yes	Yes	Koenker and Bassett 1978
Parametric maximum likelihood	Yes	Yes	Yes	Kaiser et al. 1994
Nonparametric maximum likelihood	Yes	Yes	Yes	Maller et al. 1983
Trimming	Yes	No	No	Maller et al. 1983
Sum of squares	Yes	Yes	No	Blackburn et al. 1992

1992 and 1993 surveys, and fully incorporated to the field protocol starting with the 1994 survey.

Model of clutch size conditioned on body size

Clutch size (whether expressed as number of eggs, volume, or weight) should, in principle, be related to a linear measurement of body size (e.g., CW) by a power function. Observation of data for snow crab (e.g., Figs. 1-3), however, indicates that a linear approximation is reasonable for any practical purpose, at least for the size range of interest (almost all the records are in the 40-80 mm range of CW). In the two-parameter linear model,

$$Y_i = a + b\,X_i \ ,$$

a is the intercept, b is the slope, X_i is observed CW (in mm), and Y_i is predicted maximum clutch size (in ml).

Statistical models

Several statistical models proposed in the literature (Table 1) were considered, and three were selected as candidates to be evaluated for further analyses.

Quantile regression

Quantile regression is a statistical method developed to estimate and draw inferences about conditional quantile functions. The statistical foundation of this technique allows for fitting a variety of parametric functions, including linear and nonlinear models (Koenker and Hallock 2001). Parameter estimation is based on minimizing an asymmetric loss function of the absolute values of residuals, where positive residuals are given weights equal to τ and negative residuals are given weights equal to $1 - \tau$ (Cade et al. 1999):

$$\min \left[\sum_{i \in \{i | y_i > a+bx_i\}} \tau \left| y_i - (a + b x_i) \right| + \sum_{i \in \{i | y_i < a+bx_i\}} (1-\tau) \left| y_i - (a + b x_i) \right| \right]$$

We assumed that a τ of 0.95 represents the MBL. Scharf et al. (1998) did not find major differences between the parameter estimates associated to the 90 and 95 percentiles for a variety of data sets. To fit the model to the data we used the *R*-function (GNU, developed by Koenker 2003), which allows for standard error estimation and is based on a modified version of the Barrodale and Roberts linear programming algorithm for regressions (Koenker and d'Orey 1987,1994).

Least squares

Blackburn et al. (1992) estimated the slope of the MBL of animal abundance vs. body size scatter diagrams by least-squares optimization, using only the data near the upper bound; the slope of the MBL was adequately estimated in simulated "polygonal" data sets. Scharf et al (1998) examined the sensitivity of results with respect to alternatives for the identification and preprocessing of the points close to the upper bound, namely the grouping of the predictor variable (equal size increments, equal number of observations) and to how the values relate to the response variable (maximum value within the category). Following Scharf et al. (1998), we used 12 categories of the predictor and associated the maximum values of the response variable to the center of the respective CW intervals. The objective function to be minimized was

$$\min \left[\sum_{j=1}^{12} \left[y_j - (a + b x_j) \right]^2 \right]$$

where y_j is the maximum value of the response variable associated to the *j*th interval, and x_i the center of that interval.

Table 2. Point estimates and standard errors for the quantile regression fits ($\tau = 0.95$) to snow and Tanner crab clutch volume at body size data.

Species	Year	Slope	(se)	Intercept	(se)
C. opilio	1992	3.64	(0.15)	–113.64	(7.81)
	1993	3.97	(0.15)	–139.84	(8.01)
C. bairdi	1992	6.33	(0.42)	–290.67	(34.08)
	1993	6.90	(0.77)	–331.38	(61.48)

Trimming method

This method was introduced by Maller et al. (1983) to fit the regression model

$$y_i/p_i = a + b\,x_i + e_i,$$

where p_i represents the proportion with respect to the MBL, and comes from an unknown mixing distribution F on [0,1]. This model has two sources of variability: one around the MBL line and the other associated with unknown factors that determine realizations below the MBL. The method consists of a sequential "trimming" of the data over an iterative process that, in the end, identifies the data associated with the MBL.

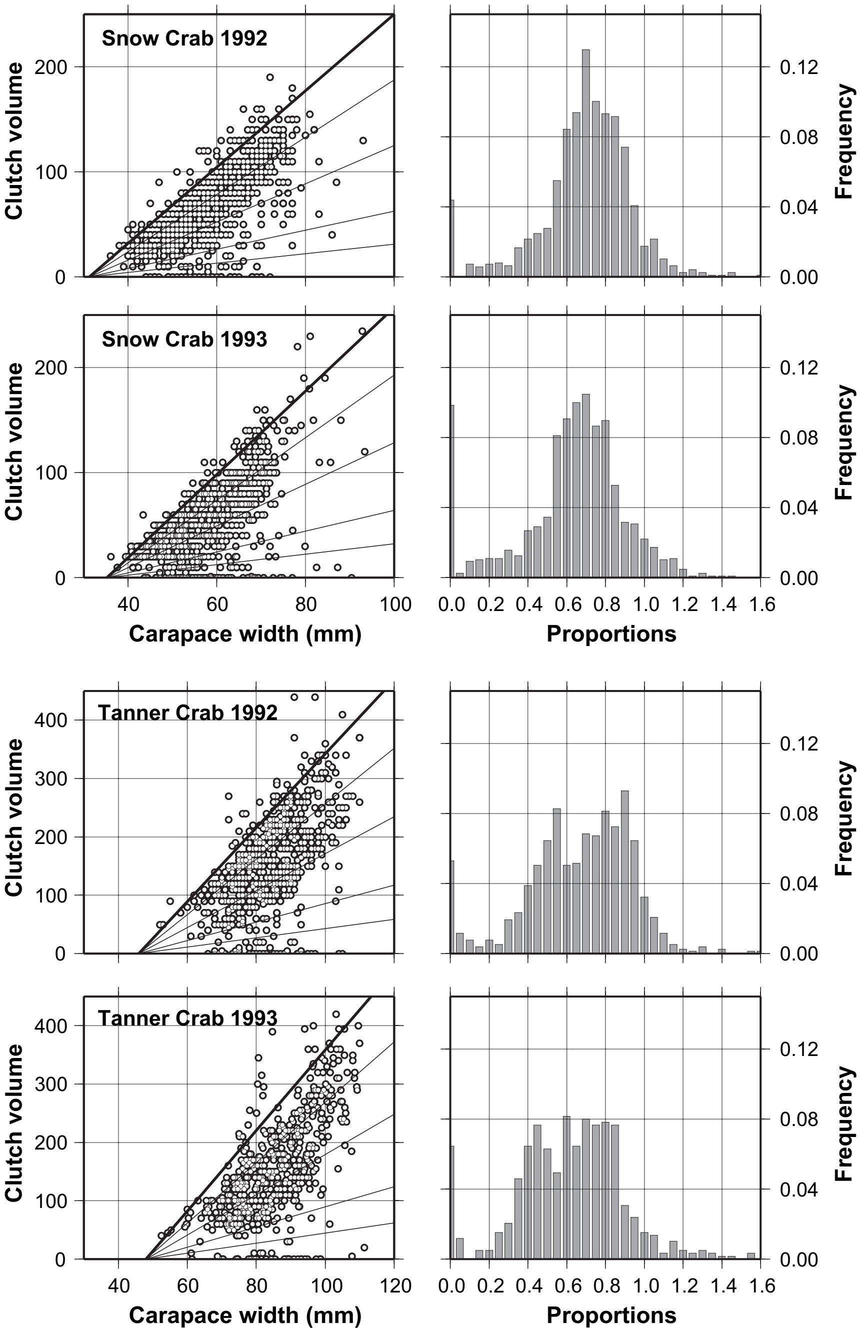
Clutch fullness index (CFI)

Once the parameters of an MBL are estimated, each clutch size observation can be expressed as a relative size or "fullness," i.e., as a fraction of the maximum clutch size predicted by the MBL given body size.

Results

While similar in body shape, mature females of the two species have very different sizes: most snow crab females are in the range of 40-80 mm while Tanner crab females are in the range of 60-110 mm (Fig. 3). Inspection and exploratory analysis of the data suggested that a linear approximation is reasonable for the MBL of each data set.

Differences in the MBL estimated for snow crab with the three methods relate largely to the intercept, and were rather small with regard to the slope (Fig. 2). The least squares approach of Blackburn et al. (1992) yielded an MBL that enclosed almost all the data. The trimming method yielded the lowest MBL, close to the 90% percentile in the linear quantile


Snow Crab 1992
Snow Crab 1993
Tanner Crab 1992
Tanner Crab 1993
Clutch volume
Carapace width (mm)
Frequency
Proportions
0
100
200
300
400
40
60
80
100
120
0.00
0.04
0.08
0.12
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6

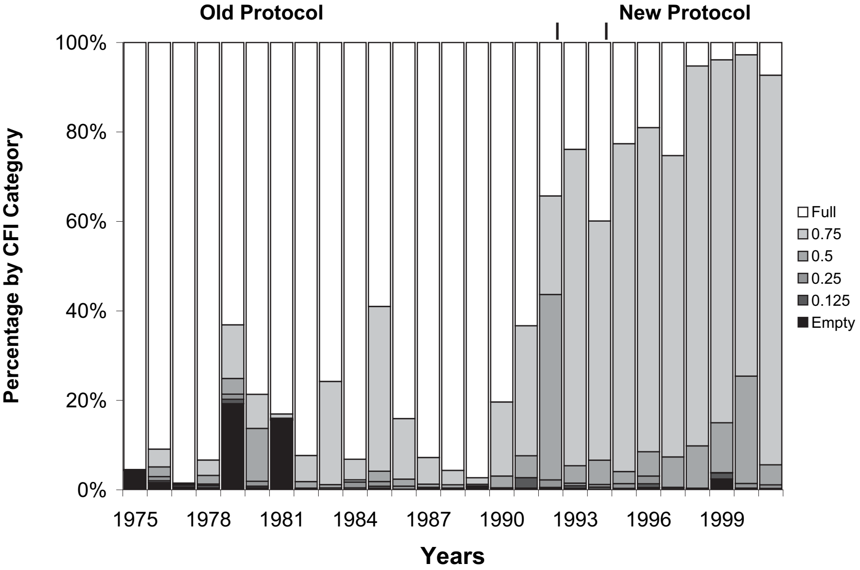


Figure 4. Time series of proportions of individual females in CFI (clutch fullness index) categories assessed visually for snow crab in SCI-2 (mostly primiparous), NMFS trawl surveys 1975-2001.

Figure 3. (Facing page.) Left: scattergrams of clutch volume vs. body size data for snow and Tanner crab, 1992 and 1993 survey samples pooled. Thick line = MBL (maximum boundary line) fitted with the quantile regression method ($\tau = 0.95$); thin lines = proportions of the MBL, corresponding to CFI (clutch fullness index) brackets used for visual assessment in the NMFS survey protocol (75%, 50%, 25%, 12.5%). Right: distribution of the CFIs.

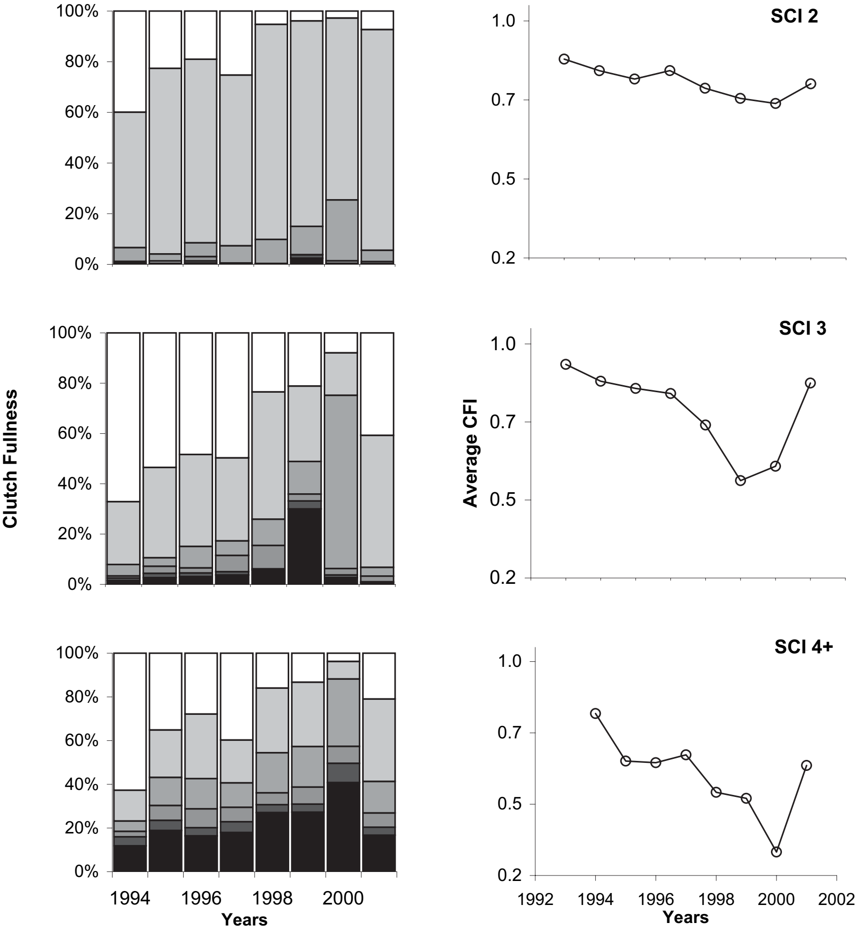


Figure 5. **Left: Time series of proportions of individual females in CFI (clutch fullness index) categories assessed visually for snow crab, 1994-2001, by shell condition index. Right: trends in average CFI. Shade scale as in Fig. 4. SCI-2 females are primiparous, SCI-3 are first-time multiparous, and SCI-4+ are older multiparous.**

regression model. A similar effect was observed by Maller et al. (1983) when they compared the trimming method to a nonparametric maximum likelihood approach. Quantile regression has a stronger theoretical foundation, is less reliant on ad hoc decisions, and in general is robust to outliers. Its main drawback is that there is no implicit association between a specific percentile and the boundaries that satisfy the purpose of the investigation. This association is left to the modeler, and therefore the need to explore other approaches. Quantile regression ($\tau = 0.95$) was selected for further analyses.

MBLs fitted to different data subsets (various combinations of species and years) using quantile regression yielded highly statistically significant slopes and intercepts ($P << 0.01$) and low coefficients of variation on those estimates (Table 2). This indicates that the data are very informative about the MBL, and that there was a high degree of consistency between the data and the model. Inter-annual variability in the estimated MBL was small for both species, indicating that it is a conservative property and that it is probably a good representation of the true relationship between clutch volume and body size, once other factors are removed (Fig. 3, left; Table 2). Distribution of the CFI differed between the two species, but for each species it was consistent between years (Fig. 3, right). In the case of snow crab the distribution was unimodal, with modal CFI at 70% of the estimated MBL. In the case of Tanner crab, instead, the distribution was bimodal in both years, with modes at 45-55% and 85-90% of the estimated MBL.

While a linear model is a reasonable and practical approximation for each species taken separately, it should not be used over a wider range of sizes, e.g., if the two species were to be compared. In that case a logarithmic transformation of the data would be required. This is evident in the parameters estimated using the linear model for the two species (Table 2). The slope and intercept estimated for Tanner crab are respectively higher and lower than those estimated for the smaller sized snow crab. Shape is very similar between the two species.

The graphical chart developed using the 1991 survey data (Fig. 1) was tried during the 1992 and 1993 surveys, and incorporated into the regular protocol starting in 1994. Introduction of an objective aid in the visual assessment of clutch fullness had a large effect (Fig. 4). It appears that before 1992 most specimens that were “75% full” were subjectively graded as “full” (B. Stevens, NOAA Fisheries, pers. comm.). The new protocol allowed the detection of patterns in the data that otherwise would have gone unnoticed. Mean CFI declined consistently between 1994 and 2000 for all SCI categories: 17% for SCI-2, 37% for SCI-3, and 63% for SCI-4^+ (Fig. 5).

Figure 6 shows the proportions of individuals by CFI, for different SCI categories (2, 3, and 4^+), in the 1995 survey. Most females in SCI-2 (primarily primiparous) were classified as “75% full,” followed by the

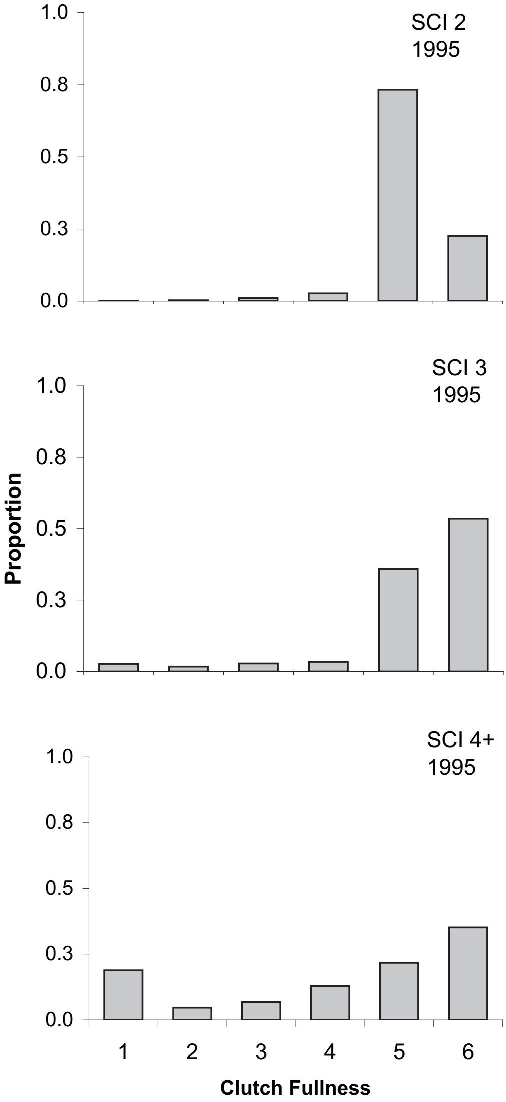


Figure 6. Snow crab, 1995 NMFS survey: relative frequency distribution of CFI (clutch fullness index) (visually assessed, new protocol) by shell condition category. SCI-2 females are primiparous, SCI-3 are first-time multiparous, and SCI-4$^+$ are older multiparous.

"full" category. Very few were classified as "50% full" or less. In the case of females of SCI-3 (primarily first-time multiparous) and SCI-4$^+$ (older multiparous) the relation is reversed, and the modal CFI category group corresponds to "full," followed by the "75% full" category. The proportion of females classified as having a clutch "50% full" or less increased among females with SC-4$^+$.

Discussion

The combined opportunity for male-selective harvests and direct assessment of the per capita reproductive contribution of females is unique to crab fisheries. Yet its potential for monitoring and management has not been fully appreciated until recently. Two main surrogates of per capita female reproductive contribution have been utilized in the past, both well exemplified by snow and Tanner crab:

- Absolute "fecundity" based on estimates of the number of eggs in a clutch (Haynes et al. 1976, Hilsinger 1976, Jewett 1981, Somerton and Meyers 1983, Elner and Beninger 1992, Watanabe 1992, Sainte-Marie 1993, Armstrong et al. 1995, Comeau et al. 1999); and
- Subjective indices of "clutch fullness," as recorded in the NMFS surveys, introduced earlier.

The two methods have significant limitations. Egg counts and other methods to estimate "fecundity" are costly and time consuming, and for those reasons impractical for monitoring purposes. Visual indices of clutch fullness not guided by a graphical or pictorial scale are unreliable. "Full" may mean different things for different observers. Besides, observers tend to internalize a subjective scale during a survey. This is problematic because the maximum observed level used as a reference for "full" (the minimum is always "empty") can vary between surveys, a typical case of "shifting baselines." As a result evidence of time trends in clutch fullness can be erased from the data. Here we propose the use of clutch volume as an indicator of the reproductive contribution of individual females. It is easily measured (even in the field) and it correlates well with other metrics (e.g., egg counts, clutch weight, etc.).

Measurements of all the surrogates of female effective reproductive contribution (egg counts, subjective appraisals, clutch weight, or volume) need to be standardized, as they are affected by several factors. First, and most obvious, they need to be related to body size (typically carapace width, CW). Another factor is the age of the clutch when measured, because of egg mortality and change in volume/weight of the eggs during ontogeny. Factors involved in egg mortality include abrasion, unfertilized eggs, developmental failure, and brood parasites (Hilsinger 1976, Wickham and Kuris 1990). Egg survival rate from oviposition to larval hatching in snow crab estimated in several regions (Kon 1974, Hilsinger 1976, Watanabe 1992, Comeau et al. 1999) ranges from 60% to 80%. The NMFS survey has been consistently conducted during the summer months, on average 4 and 2 months after oviposition of, respectively, primiparous and multiparous females, both of which carry yellow/orange eggs. In the analyses we excluded females with brown/purple eggs (close to hatching), which were always a small fraction of the samples (1.5%, all surveys pooled).

The most difficult problem for using clutch size (however measured) as a surrogate of a female's reproductive contribution in snow crab arises from the fact that females can have an annual or biennial reproductive cycle (i.e., carry the egg clutch for one or two years). This has been well documented in eastern Canada (Sainte-Marie 1993, Comeau et al. 1999), and there is good evidence that it also happens in the EBS (author J.M.O. unpubl. results). There is no evidence of the occurrence of a biennial cycle for Tanner crab. Patterns are largely related to water temperature, and can change over time (as documented in eastern Canada) or across space (apparently the case in the EBS). In the case of females that spawned earlier in the year when a survey is conducted it is not possible to distinguish between females that will carry their clutches for one or two years. Biennial females that have carried a clutch for one year have orange eggs, but they can be distinguished from recently spawned females (whether annual or biennial) by the development of the ovary. The issue of annual/biennial cycles is addressed in a forthcoming contribution.

The standard way to investigate the relationship between clutch size (usually expressed as number of eggs) body size, and other eventual covariates is by regression or correlation analyses (Haynes et al. 1976, Hines 1982, Comeau et al. 1999). Covariates may include ontogenetic stage of the eggs, insemination history of the female, condition of male partners, and multiparous vs. primiparous condition (Sainte-Marie 1993). The purpose in those cases is to determine, on average, the contribution of each factor to the variability in clutch size. Our objective is, instead, to estimate maximum clutch size conditioned on body size, and to express the residual variability in the form of a size-independent CFI. The relationship between clutch size and body size belongs to a general family of problems in which an observed covariate acts as a limiting factor on the dependent variable, but the latter is also affected by an unaccounted number of additional covariates. In the case of female crabs, clutch size is morphologically constrained by the room available for ovarian development, and egg protection. Manifestations of the "limiting factor principle" appear under a variety of names in different fields: edge-of-scatter diagrams, limiting relationships, upper bound plots, factor-ceiling distributions, production frontiers, boundary of a data set, upper bound plots, quantile regression, and triangular distributions. Ecological examples include the estimation of stomach capacity relative to body size (Maller et al. 1983, Maller 1990, Digiacomo et al. 1994), the relation between abundance and body size (Blackburn et al. 1992), and the relation between the size of predators and their prey (Scharf et al. 1998). Furota (1996) is the only author who, to the best of our knowledge, has considered this problem for the clutch size vs. body size relation in crabs. Statistical techniques for the estimation of boundary line parameters are still in development (Kaiser et al. 1994): approaches vary in model assumptions, data handling, and estimation methods (Table 1). With regard to the specific problem that we address here, future research should be aimed at the selection of an optimal statistical model to estimate the boundary line, and refinements such as nonlinear boundaries and boundaries specific to primiparous and multiparous females. We did not incorporate parametric and nonparametric maximum likelihood methods into our analysis, but do see potential in their implementation. Their statistical model structures and assumptions are based on a strong theoretical background. Simulation analysis should be conducted under a variety of scenarios to compare the performance of each statistical model.

Patterns in the distribution of the CFI observed in Tanner and snow crab proved very informative. The bimodal distribution consistently observed in Tanner crab reflects the difference in clutch size between primiparous and multiparous females reported by Somerton and Meyers (1983), because the gonad producing a primiparous brood is smaller, as it develops before the female reaches its final size at the puberty molt. This phenomenon was detected in the data collected during the NMFS survey, after the protocol was introduced in 1994 (Fig. 6). In 1995 modal CFI category was "75% full" for the primarily primiparous SCI-2 females, and "full" for the primarily first-time multiparous SCI-3 females. Increase in the fraction of females with a "50% full" or less among females in SCI-4^+ reflects reduced fecundity due to senescence. A difference in fecundity between primiparous and multiparous females was also documented for snow crab from eastern Canada by Sainte-Marie (1993). In the case of the EBS the observed unimodal distribution of the CFI may be due to the mixture of females on annual and biennial reproductive schedules, which could blur the pattern.

While a combination of an expedient method to measure clutch size and the application of MBL estimation methods in the calculation of CFIs may prove valuable to detect early signs of recruitment overfishing in male-only crab fisheries, the content of female spermathecae and male vas deferens are more proximate indicators of effective mating activity than CFIs. Although time consuming, monitoring of these indicators may be helpful in detecting sperm limitation. Field studies on exploited snow crab stocks from eastern Canada have shown evidence of sperm depletion in large morphologically mature males (Conan and Comeau 1986, Comeau and Conan 1992, Sainte-Marie et al. 1995). In the case of the NMFS surveys this would not be a good indicator because by the time the survey is conducted the mating season is over. Results from an extensive survey of female spermathecae collected in 1992, before the decline in CFI, will be presented in a forthcoming study. Comparison with samples collected recently (D. Pengilly and L. Rugolo, Alaska Department of Fish and Game, Kodiak, pers. comm.) may prove most informative.

The most important pattern revealed by the implementation of the revised protocol is a declining trend in clutch fullness during the period 1994-2000, followed by a rebound in 2001. Interestingly, results from the NMFS surveys (Stevens et al. 2002) show a 7-fold increase in the ratio of relative abundance of mostly mature "large females" (CW > 50 mm) to mostly mature "medium-to-large males" (CW > 78 mm) between 1996 and 2000, with a decline in 2001. This mirrors the trend in CFI described above. Interpretation of the pattern requires consideration of the complexities of the mating processes, including the diversity of mature females (primiparous, multiparous) and males (morphologically immature and mature) and their large scale patterns of spatial distribution.

Acknowledgments

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Using Local Ecological Knowledge in Ecosystem Models

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Abstract

We aim to construct marine ecosystem models of the past for northern British Columbia as a component of a Back to the Future (BTF) project. In order to satisfy the immense data requirements of an Ecopath with Ecosim (EwE) model, it has been necessary to recruit unconventional sources of information. Local ecological knowledge (LEK) has been called upon to supplement scientific, archeological, and naturalist records, particularly when dealing with data-limited species. The BTF team interviewed forty-eight northern British Columbia community members in 2001-2002: mainly commercial, aboriginal, and recreational fishers. Our LEK database now contains detailed anecdotal information on fisheries and some 130 marine species. The changes in abundance perceived by fishers are of special importance as we try to gauge anthropogenic effects over time. Validation showed that LEK comments agree with formal stock assessments in only 37% of instances, although agreement improves with reported fishing experience. LEK seems best suited to detect long time-scale changes: LEK abundance trends correlate poorly with stock assessment in species that exhibit high inter-annual variability. Trends offered by respondents are more likely to contradict stock assessment if they were reporting a decrease in abundance. This indicates that respondents are more likely to err on the side of pessimism and/or stock assessment is more likely to err on the side of optimism. The use of LEK information to supplement standard data sources may become an important tool. Besides consolidating and preserving community perception, we may establish criteria by which we can assess the quality of scientific data, challenging it with an independent authority and identifying where fishers' perceptions depart from the scientific understanding.

Introduction

The need for supplementary data

In modeling whole ecosystems, data deficiencies become apparent among species that hold no commercial appeal. Stock assessment records exist for only a small minority of species, so modelers must borrow parameters from other, often dissimilar systems, or rely on guesswork. Although Ecopath with Ecosim (EwE) (Christensen and Pauly 1992, Walters et al. 1997) grants modelers some reprieve by automatically estimating biomasses of data-poor groups based on the availability of food and abundance of predators, there is a clear need to reduce uncertainty in our estimates by incorporating supplemental information. Local ecological knowledge (LEK) held by community members is one such resource.

LEK can be used to fine-tune static Ecopath models, to confirm dynamic Ecosim function, or to inform us how the ecosystem might have been structured decades ago, before time-series data began for most species. LEK therefore holds obvious application to the Back to the Future (BTF) technique (Pitcher et al. 2004), which seeks to quantify past changes in the ecosystem over time. The key step in adapting LEK to our modeling needs comes in producing a quantitative data series from qualitative accounts. This article will discuss how that was accomplished, how abundance information from interviews compares to values from stock assessment, and how we use LEK in ecosystem simulations of northern British Columbia. This work supports the historic models of Ainsworth et al. (2002) for use in ecosystem restoration policy evaluations (Ainsworth and Pitcher 2005).

Methods

Interviews

A team from the University of British Columbia Fisheries Centre[1] interviewed forty-eight community members from the Prince Rupert region and Haida Gwaii, B.C. Interviewees represented a broad cross-section of commercial, recreational, and aboriginal fishers as well as processors and others who are familiar with the marine system of the study area: Hecate Strait, Dixon Entrance, and Queen Charlotte Sound (corresponding to Department of Fisheries and Oceans statistical areas 1-10 in northern British Columbia). As our aim was to improve the models, we did not select participants randomly, but sought the most knowledgeable contributors as recommended by partners and participants. Although we did not select interviewees based on years of fishing experience, or by the gear

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[1] Dr. Tony Pitcher (P.I.), Dr. Ussif Rashid Sumaila, Dr. Sheila Heymans, Dr. Melanie Powers, Nigel Haggan, Russ Jones (Haida Fisheries Council), Eny Buchary, Cameron Ainsworth, Pablo Trujillo, Louisa Wood, Richard Stanford, Erin Foulkes, and Aftab Erfan.

type they operated, we did record that information and so could look for apparent biases in the information provided.

One-hundred-eighteen flashcards of marine mammals, birds, fish, and invertebrates were shown to each of the interviewees, who were asked whether the abundance of these creatures had increased, remained the same, or decreased during their career (i.e., from their first year of fishing to their last); see Ainsworth and Pitcher (2004) for a more complete description of the interview process. This method assumes that respondents made implicit allowance in their answers for any changes in catchability arising from new methods or fishing technology. Other information recorded included known aggregations of animals, seasonal movements, fisheries interactions, and population changes. These data, along with demographic specifics, were processed to ensure anonymity and entered into the BTF Historical and Interview Database (contact C. Ainsworth).

Creating a time-series of relative abundance

For each organism, an interviewee's comment of increase, stable, or decrease was assigned as +1, 0, or −1 respectively. Every year that the respondent fished received one numerical "vote" for that organism. Summing votes from all respondents, the annual total was assumed to indicate the average perception for that year, where a positive value suggests the abundance had increased and a negative value suggests it had decreased. The resulting time-series indicated the perceived rate of change in abundance for that organism. The rate of change was converted into a running total—a time-series estimate of relative abundance.

Organisms were compiled into Ecopath functional groups (species aggregated according to trophic similarity) for comparison with the models (see Ainsworth et al. 2002 for a description of model groups). Some groups include several species, so we assumed that the abundance trend of the group most closely follows those species mentioned most often by respondents. For example, only eight comments concerning the functional group Odontocetae mentioned northern right whale dolphin, while 36 comments were made for orca. The abundance trend of Odontocetae therefore more closely reflects the trend for orca: it is a weighted average of the relative number of comments. Ideally, we would weigh the contribution of each species to the overall functional group abundance trend using some independent estimate of relative abundance. However, in the base Ecopath model, Ainsworth et al. (2002) has generally assigned important and commercial species (i.e., species for which abundance data are available), their own dedicated functional groups.

Weighting by expertise

The interviews captured a diverse sample of local knowledge. Many fisheries (and industries) were represented, and we expected each sector to carry its own special expertise in species of particular importance to their specialization. We therefore applied weighting to the votes offered by each participant according to their expertise. “Expert” opinions were taken to be worth twice as much toward calculating the average (i.e., +2 and –2 for increasing and decreasing votes); “novice” opinions were taken to be worth half as much (i.e., +0.5 and –0.5). What constituted an “expert” or “novice” vote was determined under the following rules:

1. Fishers are expert in their target functional groups.
2. Group interviews are novice in all functional groups.
3. First Nation group interviews remain unchanged in First Nation specialties.
4. Non-fishers are novice in all functional groups.
5. Recreational fishers are novice on all functional groups except their specialty, in which they are expert.
6. Interviewee 20 was judged expert in all functional groups.
7. Interviewee 21 was judged expert in all rockfish functional groups.

Group interviews operated on consensus; their vote was reduced in importance to limit the effect of influence between respondents in our analysis. As most of our First Nation respondents participated in group interviews rather than individual interviews, we did not wish to limit their contribution to the study, especially with regard to traditionally harvested species. We assume that non-fishers and recreational fishers spend less time at sea than commercial harvesters, so we weighted their contribution by only half as much. In addition to years of fishing experience, interviewees 20 and 21 had formal ecological training.

A weighting scheme based on years of experience could be used alternatively, although some degree of ranking by gear specialization should still be included. Information from experienced fishers does actually influence the LEK abundance trend more than information from less experienced fishers under the current methodology, since their comments apply to more years in the analysis. Also, we felt that retired fishers could be inclined to answer differently from those who still rely on the resource for income.

Comparing LEK to stock assessment information

To determine how often comments agreed with stock assessment records, we compared the qualitative change in abundance offered by each interviewee with time series abundance data assembled from DFO stock assessment data (i.e., various Canadian Science Advisory Secretariat and Pacific Scientific Advice Review Committee reports). For the period in which a respondent fished, a simple program consults DFO records to determine whether the recorded abundance of the subject functional group had increased, stayed the same, or decreased. It compares this result against the vote provided by the interviewee to determine agreement. This procedure was conducted for the eight functional groups that had continuous stock assessment information. Comments were used from only the fishers whose career spanned a period covered by stock assessment data.

For each comment made concerning a particular functional group, the span of the interviewee's career at sea was divided into two halves. The average abundance of that functional group in the first and second halves of the fisher's career was determined from stock assessment records. If the average abundance was greater in the latter half than in the former, the functional group was said to have increased. If the fisher had indicated an increase in abundance, then their comment was considered "true" (indicating only agreement between data sets). Similarly, if the stock had declined according to stock assessment, then comments that indicated a decrease in abundance were considered "true."

An arbitrary threshold was assigned so that if only slight increases or decreases in abundance had occurred during the fisher's career (according to stock assessment records), the functional group could be considered "stable," and comments providing that response would be considered "true." By decreasing the threshold required for a change in biomass to be considered significant, fewer comments indicating "stable" become true. The threshold was set as a fraction of the total amplitude of change in abundance witnessed for that group since time series stock assessment began (Table 1). It was found that when considering all functional groups together, a threshold of 15% change in absolute biomass yielded agreement equally often between increasing, stable, and decreasing votes ($\sigma^2 = 0.0018$). This assumes that fishers are equally likely to agree or not with stock assessment data, regardless of the direction of abundance change. That threshold was used for all calculations. For fishers whose careers were shorter than the 61 year maximum (the most experienced interviewee), the required threshold was decreased proportionately. For example, biomass would need to have only increased or decreased by approximately 7% of its maximum amplitude over the course of a 30-year career to be considered significant.

Analysis of trend

We compare the LEK relative abundance trend of commercial groups with stock assessment records. Time series are available for the following Ecopath functional groups: chinook, coho, transient salmon[2], flatfish, halibut, herring, lingcod, Pacific cod, sablefish, and seals/sea lions. In

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[2] Transient salmon includes (adult) pink, sockeye. and chum salmon, which pass through the marine ecosystem but do not feed.

Table 1. Percentage of interviewee comments that agree with stock assessment records ($N = 234$).

	Increase	Stable	Decrease	Total	Variance	Exact P
0	58.2%	0.0%	35.7%	33.8%	0.086	0.002
0.05	58.2%	21.8%	32.1%	37.2%	0.035	0.0004
0.10	49.3%	30.9%	30.4%	35.9%	0.012	0.005
0.15	37.3%	38.2%	30.4%	34.2%	0.002	0.082
0.20	37.3%	40.0%	28.6%	33.8%	0.004	0.063
0.25	28.4%	41.8%	25.9%	30.3%	0.007	0.129
0.30	19.4%	56.4%	23.2%	29.9%	0.041	0.126
0.35	14.9%	65.5%	17.0%	27.8%	0.082	0.158
0.40	14.9%	72.7%	14.3%	28.2%	0.113	0.171
0.45	14.9%	78.2%	12.5%	28.6%	0.139	0.159
0.50	11.9%	80.0%	8.9%	26.5%	0.162	0.161

Shown on left is the fraction of the maximum amplitude of abundance change required for the change over the fisher's career to be considered significant. In calculations, this threshold is proportionately reduced for fishers whose career is shorter than the maximum (61 years).

order to compare the LEK information with scientific data we converted the relative abundance time series suggested by the interviews into absolute abundance by assuming the same average and amplitude of change as in stock assessment data. The correlation of the LEK information to the assessment records was then measured using a non-parametric Spearman's rank sum test for the weighted and unweighted interview information. We then divided the time series into two periods, before and after 1965, and repeated the correlation analysis.

Challenging the model with LEK information

The models were challenged with the LEK data to verify their structure and dynamic functioning. LEK information is used here in two ways to challenge the models: first, as a test of the relative (static) structure of the 1950 and 2000 Ecopath models, and second, as a test of the dynamic function of a 50-year Ecosim simulation.

Static structure

LEK information can serve as a check in a comparison between Ecopath time periods. Here we attempt to verify that the relative abundance has increased, remained the same, or decreased from the static 1950 and 2000 models in accordance with the fishers' average perception. This process can validate the many data sources used. By weighing the credibility of our source against the magnitude of the disparity, a judgment can be reached whether to accept an alternate value (if one is available),

Table 2. Biomass estimates used in Ecopath models compared to LEK trend.

	1950	2000	Change	LEK	Agree?	Data pedigree[a]
Seals and sea lions	0.057	0.052	–0.005	+	n	6
Transient salmon	0.754	0.588	–0.166	+	n	5
Coho salmon	0.067	0.024	–0.043	+	n	6
Chinook salmon	0.026	0.018	–0.008	–	y	6
Dogfish	0.8	0.909	0.109	–	n	2
Forage fish	8.88	8.485	–0.395	–	y	1
Eulachon	1.758	1.661	–0.097	–	y	1
Herring	0.748	2.265	1.517	–	n	6
Piscivorous rockfish	0.416	0.654	0.238	–	n	1
Flatfish	0.221	0.392	0.171	–	n	5
Halibut	0.429	0.608	0.179	–	n	6
Pacific cod	0.086	0.163	0.077	–	n	4
Sablefish	0.602	0.301	–0.301	–	y	4
Lingcod	0.085	0.034	–0.051	–	y	2
Large crabs	0.31	0.46	0.15	–	n	2
Infaunal invertebrates	13.245	34.305	21.06	–	n	2
Epifaunal invertebrates	10.928	13.499	2.571	–	n	1

[a] Data pedigree column indicates the quality of data. This scale measures from 1, the lowest quality (data point estimated by Ecopath) to 6 (high precision, sampling based measures).

or allow Ecopath to estimate that parameter. Table 2 compares model biomass parameters with the LEK trend, and includes Ecopath's data ranking pedigree as a measure of data quality (Christensen et al. 2000). The pedigree describes the following ranking of data quality, where 1 indicates the lowest precision and 6 indicates the highest:

1. Estimated by Ecopath
2. From other model
3. "Guesstimate"
4. Approximate or indirect method
5. Sampling based, low precision
6. Sampling based, high precision

Ecopath's pedigree considers any user input to be more reliable than an internally generated value. However, the six criteria listed above are only established by convention—if the user has reason to believe that a value estimated by Ecopath is reliable, a higher data quality ranking can be entered manually. Similarly, if a "guesstimate" is made by expert opinion, it may warrant a better ranking than 3.

Dynamic function

LEK data is used to verify dynamic group interactions occurring in a 50-year harvest simulation from 1950 to present. Parameterizing the harvest simulation using real-world fishing effort, the dynamic model should recreate the patterns of abundance seen in stock assessment time series (the 1950 baseline static model is built on that information). For all functional groups, Ecosim's predicted abundance trend is compared with the suggested biomass trend from the LEK interviews, and stock assessment information (commercial groups only). A Spearman's rank correlation test determines whether abundance trends are in concordance with our two data sets.

Results

Comparing LEK comments to stock assessment information

Table 1 below records the fraction of instances where the interviewee comments agree with stock assessment records, varying the threshold of abundance change required to be considered significant. As that threshold is decreased, fewer "stable" comments become true.

Overall, agreement between data sets is poor, with a maximum of only 37% of comments concurring with stock assessment. This highest level of agreement occurs when the abundance change threshold is set to 5%. Votes that indicate increase, stable, and decrease are true most equally often when the threshold is set at 15% ($\sigma^2 = 0.002$). At most threshold levels, the "increase" votes show agreement with stock assessment more often than "decrease" votes. The discrepancy is significant ($\sigma = 0.05$) at all levels of threshold below 15%, according to Fisher's exact test (Table 1).

Figure 1 shows the fraction of comments that agree with stock assessment records per functional group at a threshold level of 15%. Not shown, flatfish comments ($n = 16$) were never in agreement with stock assessment.

A binomial test shows that comments provided for three functional groups disagree with stock assessment more often than could be expected by chance at $\alpha = 0.05$. By experts, the interview trend for chinook contradicts stock assessment ($P = 0.004$); by non-experts, the interview trend for transient salmon and Pacific cod contradicts stock assessment ($P = 0.002$ and 0.011 respectively). Non-expert comments agree with stock assessment more often than expert comments for all groups except transient

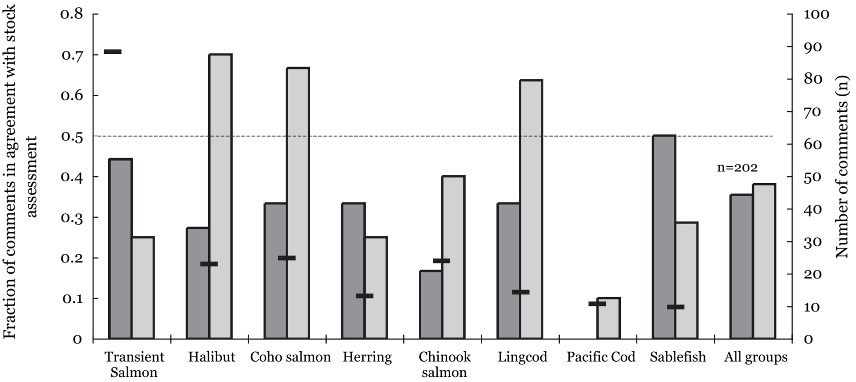

Figure 1. Fraction of comments that agree with Canada Department of Fisheries and Oceans records by functional group. Dark bars represent expert comments; light bars represent non-expert comments. Dashed line at 50% indicates proportion expected by random chance.

salmon, herring, and sablefish. Although non-experts are in agreement with stock assessment more often than experts, they are not as consistent across functional groups ($\sigma^2 = 0.010$ for experts, 0.046 for non-experts). Still, there is little evidence to support the division between expert and non-expert (non-expert including both unchanged and novice votes).

Figure 2 tests whether experienced interviewees are in agreement with stock assessment more often than less experienced ones. There is a trend suggesting that agreement between data sets improves as fishers' experience increases. Fisher's exact test reveals that interviewees with 40 or more years of experience provide a significantly ($P = 0.045$) higher fraction of comments that agree with stock assessment (41%, $n = 74$) than less experienced ones (31%, $n = 148$).

Analysis of trends

Figure 3 presents the absolute abundance for eight functional groups estimated from the interview materials, and from DFO stock assessment information. Abundance trends determined with and without the data quality weighting scheme (reflecting fisher's expertise by gear type) are shown. Figure 4 summarizes correlation between weighted and unweighted data sets with stock assessment information.

There is a significant positive correlation between the interview trends and stock assessment for four groups, using the unweighted LEK trend, and only three groups using the trend weighted for data quality. In fact, there are significant negative correlations for several groups, indicating

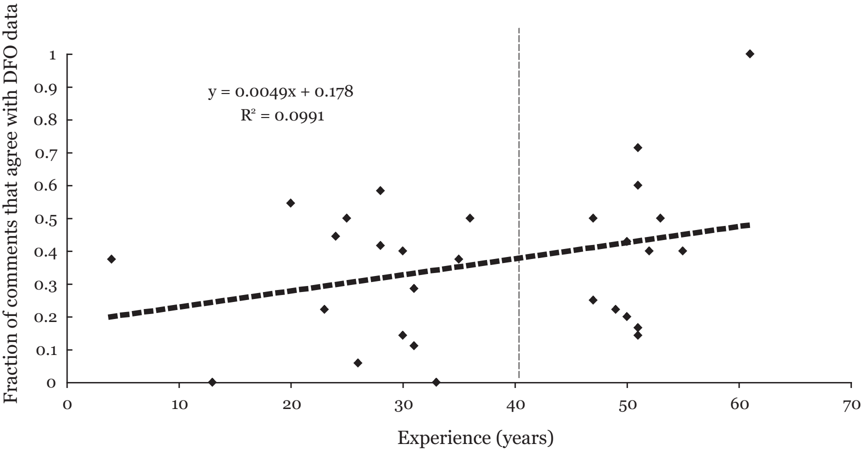

Figure 2. Interviewee agreement with stock assessment data by career length. Dots represent the fraction of comments that are in agreement with stock assessment; dotted line shows regression; dashed line shows 40-year division for Fisher's exact test.

that the average fisher perception is in contradiction to the scientific data set. The abundance trend weighted by fisher's expertise outperforms the unweighted trend only for chinook salmon.

With functional groups that display a large degree of inter-annual variability, we expect the correlation between interviewee and DFO data sets to suffer, since the abundance trend gleaned from the interview data is not suited to detect fine time scale changes in abundance. The LEK trend may be more suited to detect long-scale (e.g., decadal) changes. In fact, the best correlation occurs in lingcod and sablefish, two long-lived species whose abundances do not fluctuate greatly from year to year.

We tried dividing each data set into two series, before and after 1965. Since fewer interviewees could contribute to the early years of the analysis, we expected the LEK trend to match stock assessment information better, and more often (across functional groups) in the latter time series than in the former. However, not only did the LEK trend from the 1965-2001 data series achieve agreement with stock assessment less often than the 1933-1964 series, significant negative correlations were found for six functional groups, as opposed to three for the combined data set in Figure 4. This indicates that even when there was a maximum number of respondents contributing to the LEK trend, their perceptions of change in abundance still disagreed, and more frequently, with stock assessment information.

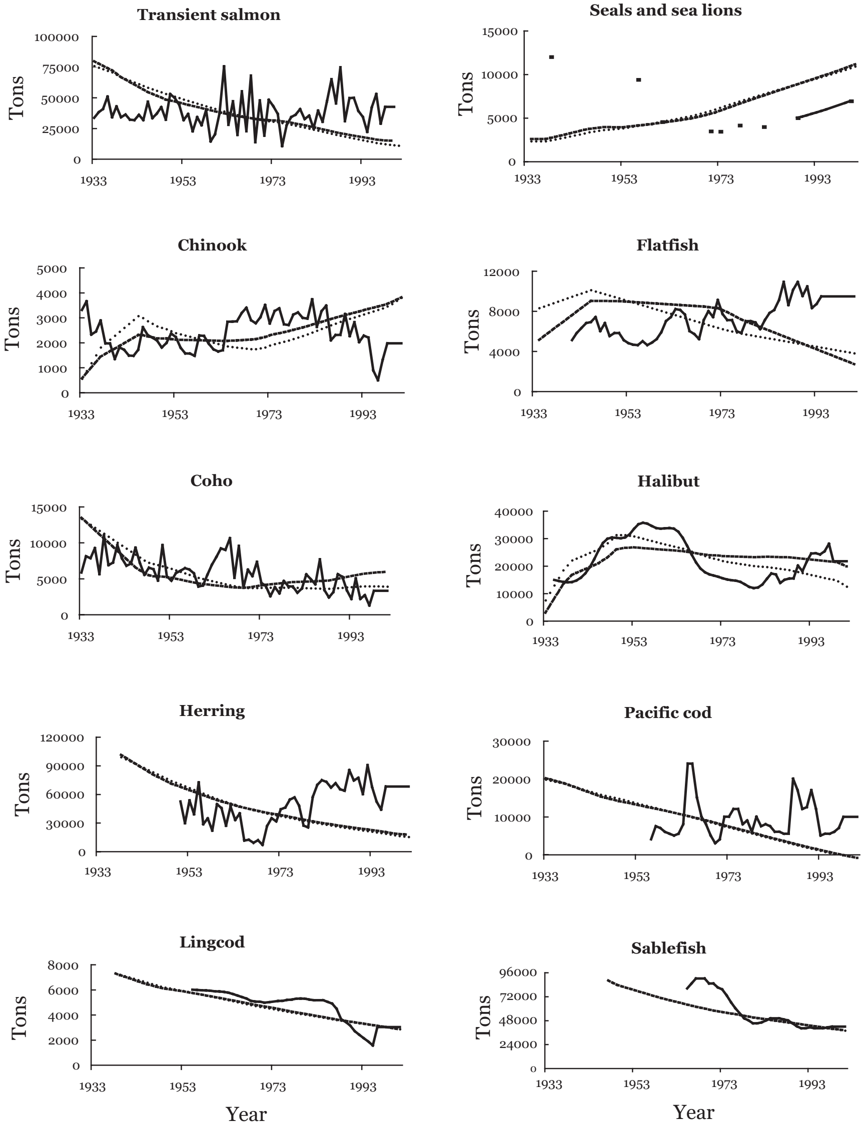

Figure 3. Weighted and unweighted LEK trends compared to stock assessment records. Solid line is stock assessment; large dots are unweighted LEK trend; small dots are weighted LEK trend. Absolute LEK trend is scaled using the same mean and amplitude as stock assessment.

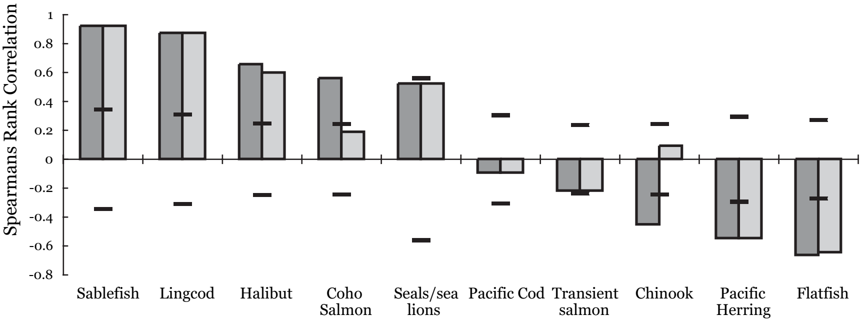

Figure 4. Rank correlation of weighted and unweighted LEK abundance trend with stock assessment information. Dark bars are unweighted LEK; light bars are weighted LEK; crossbars indicate significance level at $\alpha = 0.05$.

Challenging the model with LEK information

Static structure

Table 2 shows the biomass estimates used in the 1950 and 2000 Ecopath models. The net change is compared to LEK results. The LEK column indicates the net change in fishers' perception of abundance since 1950 according to the trends determined earlier. The data pedigree in the right column indicates the quality of data used in the 2000 model. Data quality in the 1950 model is generally poor for noncommercial groups as well. Not shown are the functional groups whose biomass remains constant between time periods.

For most functional groups, the LEK trend does not agree with the data used to construct the models. Fisher's exact test shows that agreement between LEK and the models' change in biomass is independent of the data pedigree ($P = 0.324$). Agreement between data sets is also independent from the direction of biomass change between periods, having increased or decreased from 1950 and 2000 ($P = 0.326$).

Dynamic function

A 50-year Ecosim simulation was run using the 1950 model as the starting point. Figure 5 shows how well LEK and stock assessment correlate with the Ecosim model's predicted biomass trend using a non-parametric Spearman's rank sums test. Dark bars indicate correlation with LEK information; light bars indicate correlation with stock assessment. Crossbars indicate the level of correlation needed for significance at $\alpha = 0.05$.

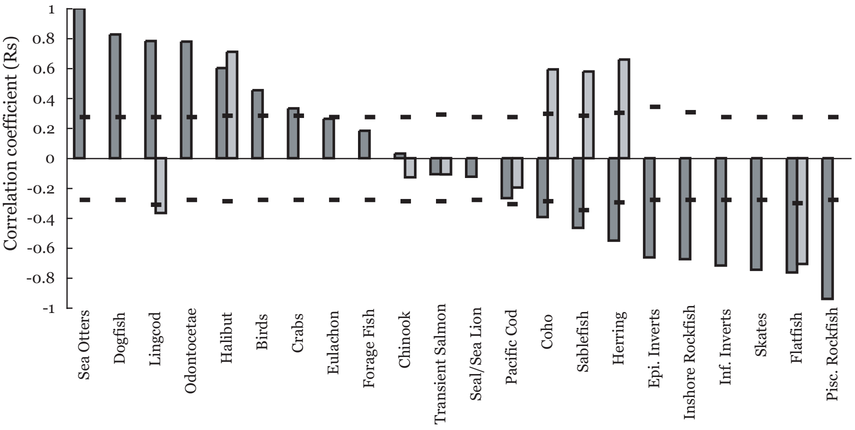

Figure 5. Correlation of LEK and stock assessment biomass trends to 50-year Ecosim model (beginning 1950). Dark bars show correlation with LEK; light bars show correlation with stock assessment; crossbars show significance at $\alpha = 0.05$.

Discussion

Validating LEK

In comparing the LEK trend with stock assessment, agreement is poor. The best correlation occurs in slow growing species, whose abundance does not fluctuate greatly from year to year. Agreement between data sets does improve with fishers' experience. However, considering that LEK comments generally indicate a decrease in abundance for the majority of functional groups, regardless of fishing experience, it is likely that fishers' perceptions come to resemble the scientific understanding only when considering the long-term trend. Presumably, a steady depletion (particularly among commercial species) becomes obvious over the course of several decades, where a short-term trend can be mired in fluctuations, and is open to some degree of interpretation. Interestingly, we found that interviewees were more likely to contradict stock assessment if they were reporting a decrease in abundance. This suggests that respondents are more likely to err on the side of pessimism and/or stock assessment is more likely to err on the side of optimism. We do not suggest that one data set is more accurate than the other, only that there is persistent bias in that direction. At any rate, discrepancies show where stock assessment records are in contradiction with fishers' perceptions.

Challenging the static Ecopath models

We tested the relative biomass values used in the static 1950 and 2000 Ecopath models against the LEK trend. The LEK trend verifies the change in biomass for only five out of seventeen functional groups. Of those that disagree with LEK, we can remain confident in our data sources concerning well-studied groups such as seals and sea lions, transient salmon, coho, herring, flatfish, and halibut. However, the LEK trend also disagrees with data-poor groups like dogfish, piscivorous rockfish, large crabs, and infaunal/epifaunal invertebrates. In the absence of empirical data, LEK offers our only guide to the abundance change of poorly studied species. But through the rigor offered by the mass-balance modeling process, it is possible to test whether the perceived abundance change of these noncommercial groups is congruent, or at least possible, within the trophic constraints imposed by the well-understood ecosystem components.

Testing dynamic Ecosim function

We compared the LEK and stock assessment data sets with the output of a 50-year simulation. Nine functional groups show a significant negative correlation with the LEK trend: coho, sablefish, herring, infaunal/epifaunal invertebrates, inshore rockfish, skates, flatfish, and piscivorous rockfish. Of these, the commercial groups coho, sablefish, and herring are vindicated by a strong positive correlation with stock assessment data. Suspect groups are therefore flatfish, whose Ecosim trend is contradicted by both stock assessment and LEK trends, as well as infaunal/epifaunal invertebrates, inshore rockfish, skates, and piscivorous rockfish. Lingcod shows a negative correlation with stock assessment records despite endorsement by LEK information. Future work to improve the dynamic function of the models will begin with these problematic groups.

We could attempt to improve the fit of Ecosim to LEK information by making fine adjustments to predator-prey vulnerabilities, which govern the simulated trophic interactions (Walters and Kitchell 2001). Or, if after tuning the model the time series biomass trajectories cannot be made to agree with the LEK trend, we could employ a second, more desperate option. That is, to force the biomass trend of problem groups to agree with our inputted trend from LEK. Although this robs Ecosim of its predictive power, the method could be used to treat a small number of highly influential groups, so that their dependent predator/prey groups would behave appropriately.

Future work

As one reviewer pointed out, abundance trends suggested by local ecological knowledge may partly reflect changes in the spatial distribution of species—particularly if LEK is more spatially restricted than broad-scale survey data. However, there are enough site-specific references in the interview materials that we could compare abundance trends with stock assessment, by area, to estimate this potential bias.

As this article goes to press, a new method is under development that would use this interview information, translated into a spatial context, to validate the behavior of a spatially explicit Ecospace model (Walters 1998).

Acknowledgments

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The Scotian Shelf Experience with Emerging Bivalve Fisheries

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Abstract

The development of commercial fisheries for offshore bivalves on Canada's East Coast is reviewed. The arctic surfclam fishery and the ocean quahog population, for which there is currently no fishery but increasing interest, are examined. For the arctic surfclam fishery a government survey took place prior to the fishery in 1981-1982, and an industry-funded survey was conducted in 1996-1997. This fishery has extensive industry involvement in funding sampling and research programs which have increased understanding of the life history of the species involved and aided management of the resource. The crew on the vessels regularly sample the catch for length frequencies, conversion factors, and bycatch, and collect samples of whole clams for morphometric analysis and aging of shells at a Department of Fisheries and Oceans (DFO) laboratory.

DFO has data for the ocean quahog population from the 1981-1982 survey, and can make comparisons to other populations that have ongoing fisheries. Different methods of determining the sustainable yield for these data limited populations are discussed, yield estimates for the arctic surfclam fishery examined, and a yield estimate for a possible ocean quahog fishery defined.

Current management is based on setting long term allowable catch levels, as there are no annual surveys and little contrast in the data. With the movement toward ecosystem based management more information will be required on the effects of the fishery on other species.

Introduction

The Scotian Shelf of Nova Scotia is home to a number of large bivalves. Arctic surfclams, *Mactromeris polynyma*, support a commercial fishery that started in 1986, and there is growing interest in harvesting ocean quahogs, *Arctica islandica*, and northern propellerclams, *Cyrtodaria*

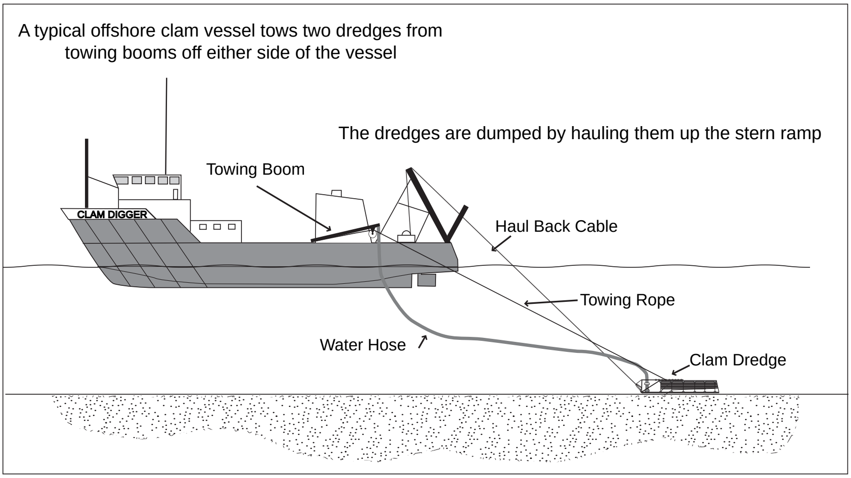

Figure 1. Schematic of the operation of an Atlantic Canadian offshore clam vessel with hydraulic dredges.

siliqua. On the adjacent Grand Bank the Greenland cockles, *Serripes groenlandicus*, are being harvested as a bycatch to the arctic surfclam fishery. These populations are on the offshore banks in waters between 50 and 100 m depth. For species such as Greenland cockles and northern propellerclams, little is known of their life history. For all of them the distance from shore and depths involved make sampling and surveys expensive and difficult. Arctic surfclams and ocean quahogs are long-lived species, 40 to 60 years for the arctic surfclams and over 100 for the ocean quahog. This means that sustainable exploitation rates have to be low, and there is likely to be little measurable population response to fishing.

The arctic surfclam fishery has evolved from the discovery of commercial concentrations on Banquereau Bank in the late 1980s, to a mature fishery with annual sales of $30-$50 million employing approximately 450 people. There are 3 large freezer-processor vessels in the fishery. These vessels use large hydraulic clam dredges that pump water down to the dredge to liquefy the sand at the front of the dredge so the clams can be separated out (Fig. 1). The fishing grounds are on Banquereau Bank on the Scotian Shelf, and Grand Bank off Newfoundland (Fig. 2). These are in different Department of Fisheries and Oceans (DFO) management regions, but the two banks are managed under a single management plan. In this paper I will concentrate on the fishery on the Scotian Shelf, but will include developments on Grand Bank as they influence management decisions.

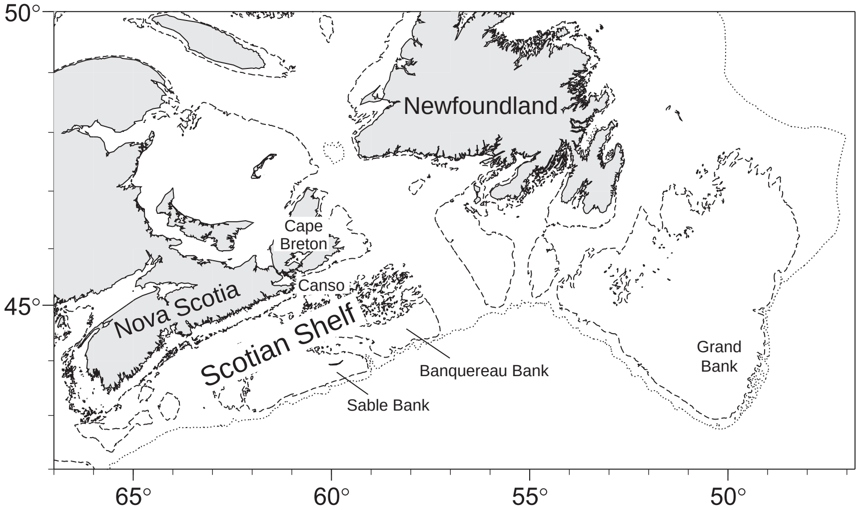

Figure 2. Map of the Scotian Shelf and Grand Bank on Canada's East Coast.

The offshore clam fishery started when there was a directed effort to get industry involved as a partner rather than an adversary in fisheries management. It was also a time when reduced government budgets restricted DFO's ability to fund research programs in support of emerging fisheries. This resulted in a high degree of industry involvement in research and management of this fishery, creating benefits and problems for government and industry.

There is no regular survey series for offshore clams, and there are no quantitative management targets for arctic surfclams. Targets have been constant harvest levels that have been based on empirical methods.

History of the fishery

The management of the fishery, developments occurring during each phase, and degree of industry involvement in research and management are discussed in relation to phases of the fishery (Table 1).

Development phase

In 1980 DFO initiated a fishery development plan to determine the resource potential of the ocean quahog (*Arctica islandica*) and other underutilized clam species on the Scotian Shelf (Fig. 2). During a series of offshore clam surveys from 1980 to 1982, commercial quantities of

Table 1. Chronology of the offshore clam fishery on the Scotian Shelf and Grand Bank.

Phase	Management program	Year	Management tools/changes	Events
Development phase		1980 1983		DFO developmental surveys on Scotian Shelf and Georges Bank
		1986	Offshore Clam Advisory Committee (OCAC) established	3-month test fishery with 3 participants using chartered U.S. vessels
Exploratory phase	3 Year Fishery Program	1987	TAC, limited entry, EAs, logbooks	Commercial fishing starts
		1988	OCAC requested to develop plan for Grand Bank	Requests for access to Grand Bank
		1989	Surfclams added to Canadian Atlantic fishery regulations as limited entry fishery	2 Exploratory licenses and 2 permits issued for Grand Bank
Expansion and consolidation phase	1990-1994 Arctic Surf Clam Management Plan	1990		Quota shared equally between 4 EAs
		1991		1 participant stops for financial reasons
Mature fishery phase	1995-1997 Arctic Surf Clam Management Plan	1995	Dockside monitoring	3-year industry-DFO survey of Banquereau Bank and Grand Bank starts
		1997	Offshore Clam Management Board established	
	1998-2002 Offshore Surf Clam Integrated Fishery Management Plan	2002	Banquereau Bank quota reduced to 24,000 t	
Efficiency and species expansion phase	1998-2002 plan extended for 2003	2003	Exploratory quahog fishing license issued	Industry-DFO quahog survey on Sable Bank

[a] EA = enterprise allocation.

arctic surfclams (*Mactromeris polynyma)* were found on Banquereau Bank (Rowell and Chaisson 1983, Chaisson and Rowell 1985). Areas with a large biomass of ocean quahogs were also found, but there was no commercial interest in this species.

In 1986 it was estimated that Banquereau Bank had a commercially exploitable arctic surfclam biomass of 561,000 t and maximum sustainable yield (MSY) of 16,821 t (Rowell and Amaratunga 1986). A three-month test fishery took place with three companies participating. Results from the test fishery increased the MSY estimate to 24,000 t (Amaratunga and Rowell 1986) based on the model $MSY = 0.5\ MB_0$, (B_0 = virgin biomass). They estimated natural mortality (M) with the equation $M = 3/T_{max}$ (T_{max} defined as the age at which 95% of a cohort would be removed by natural mortality, equation derived from $0.05 = N_t/N_0 = e^{-M}$ with $t = T_{max}$). It was recognized that this approach makes some assumptions that compromise use of the model, especially that of equilibrium conditions within the population, and that it was based on limited data. Another approach used by Amaratunga and Rowell (1986) was to consider the biomass as a finite resource, without assumptions about natural mortality, growth, or recruitment. In this way an annual level of exploitation is established that would remove existing biomass over a defined period of time. This approach was taken to examine the time period available to gather the data needed to estimate sustainable catch levels, and not to manage arctic surfclams as a non-renewable resource. Assuming an initial biomass of 600,000 t, annual level of removals to have the resource last 10, 20, or 25 years were 60,000 t, 30,000 t, and 24,000 t respectively.

Industry involvement in management was through the Offshore Clam Advisory Committee (OCAC), established in November 1986 as an open and public forum to provide advice to the minister on all issues affecting the fishery. OCAC is responsible for development of management plans and considers fishery development, access, and allocation issues.

Exploratory phase

In 1987 a three-year offshore fishery program was developed with industry consensus. The fishery was to be reviewed after the 1989 fishing season. Total allowable catch (TAC) was based on biological advice from the development phase, and an economic break-even analysis on the resource required to make a vessel and processor viable. TACs were set at 30,000 t for Banquereau Bank and 15,000 t for the rest of the Scotian Shelf. TAC was broken down into enterprise allocations (EAs), shares of the TAC allocated to individual enterprises, in this case license holders. EAs are “quasi property rights” used to prevent overcapitalization in a race for a common property. Other requirements were that shore-based processing be located in the Cape Breton-Canso area of Nova Scotia, an economically depressed region, and all vessels were to be Canadian-owned and crewed within a set period of time. Details on the development of the fishery up to 1989 can be found in Roddick and Kenchington (1990).

During this time exploratory fishing on the Grand Bank discovered commercial concentrations of arctic surfclams. DFO received requests for licenses for this area, and in 1989 two exploratory licenses and two exploratory permits were issued for Grand Bank, with a "precautionary" TAC of 20,000. These were issued to the three current participants plus a fourth, Newfoundland-based company, and TAC for the Scotian Shelf outside of Banquereau Bank was increased to 20,000 t to include this new company. There was no scientific basis behind these TACs, as there was no biomass estimate for Grand Bank, and the developmental surveys of the Scotian Shelf in the 1980s had not found commercial concentrations outside Banquereau Bank.

By the end of the program an arctic surfclam fishery was considered viable, and in February 1989, arctic surfclams were added to the Atlantic Fishery Regulations as a limited entry fishery. Industry involvement at this phase was through OCAC and regulatory requirements, including logbooks and occasional observer coverage.

Expansion and consolidation phase

The 1990-1994 Arctic Surf Clam Management Plan covered the fishery on both Scotian Shelf and Grand Bank. TACs and EAs were initially the same, but when one company went out of business in 1992, allocations were revised, giving remaining participants equal access and allocations on all banks. Any changes in TAC would be equally split between license holders. Additional consolidation occurred when one of the licenses was bought out by an existing participant, leaving just two companies controlling three active licenses. Since early 1993 there have been three factory processors fishing year-round. The catch and value of the fishery can be seen in Table 2.

When the fishery started, the expected market was the U.S. Atlantic surf clam (*Spisula solidissima*) market. Due mainly to color differences, there was little demand in the U.S. market for arctic surfclams. A market was found in northern Japan, but this market was limited. In the early 1990s participants and government invested in a successful generic marketing effort to expand this market, but demand continued to limit landings.

Industry involvement during this period was still through OCAC and regulatory requirements, but industry was lobbying DFO to conduct surveys of Banquereau Bank and Grand Bank. DFO responded that it did not have the resources available to conduct offshore clam surveys.

Mature fishery phase

The 1995-1997 Offshore Clam Fishery Multi-Year Harvesting Plan contained several key points. TACs and EAs did not change. It contained

Table 2. Landings and value for the arctic surfclam (*Mactromeris polynyma*) fishery.

	Surfclams					Others	
	Landings (t)			Value (×1,000)		Landings (t)	
Year	Grand Bank	Banquereau	Total	Landed value[a]	Export value[b]	Cockles	Propeller-clams
1987	0	717	718	$71			
1988	0	1,824	1,824	$2,724			
1989	402	7,666	8,068	$5,962			
1990	2,433	4,765	7,198	$8,160			
1991	6,753	746	7,500	$8,607			
1992	11,154	0	11,154	$15,277			
1993	18,905	60	18,965	$25,034	$35,182		
1994	15,881	4,590	20,471	$21,290	$39,448		
1995	14,108	10,427	24,535	$24,535	$51,374		
1996	6,458	18,745	25,203	$20,918	$34,991	96	153
1997	7,614	19,805	27,419	$24,202	$42,629	202	84
1998	963	24,695	25,658	$16,722	$33,514	36	63
1999	1,487	24,413	25,900	$23,783	$36,789	74	43
2000	3,775	19,989	23,764	$21,703	$31,699	57	33
2001	8,389	11,443	19,832	$17,538	$32,299	160	170
2002	6,901	12,492	19,403	$17,291	$46,665	757	437
2003[c]	10,265	16,883	27,148	$24,193	$36,850	1,209	544

[a] From 1997 to present landed value is estimated; statistics branch no longer uses actual sales slips.

[b] Statistics Canada, International Trade Division, Domestic Exports of Selected Commodities, total for all provinces, arctic surfclams were first separated in 1992.

[c] Preliminary.

commitments by participants to fund a dockside monitoring program to check landed weights; to fund an economic study of the fishery by a reputable third party; and to cost-share scientific studies with DFO. The plan noted an absence of precise estimates of key parameters on which to base TACs, especially current biomass of arctic surfclams.

In 1995, facing decisions on investment in the fishery, the clam industry proposed supplying vessel time, crew, and funding over three years for stock assessment surveys, if DFO would design and carry out the assessment. Banquereau Bank was surveyed in 1996, with additional stations conducted in 1997. In addition to survey results, the assessment had information available from other studies. There was an independent economic analysis of the viability of the offshore clam fishery, which

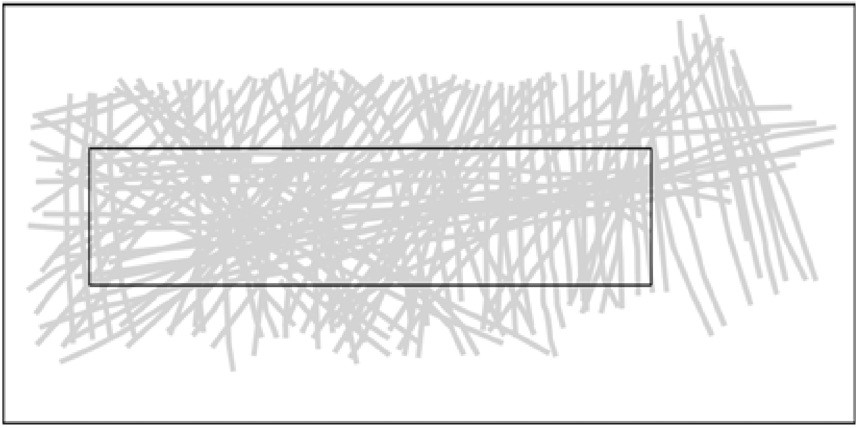

Figure 3. Tracing from sidescan image of clam dredge tracks from CFV Concordia on Banquereau Bank 1996. Dredge track width scaled to 12 feet wide (4 m). Box used for area analysis is 500 m × 125 m, percentage of ground covered within box is 67.42%.

gave the economic breakeven density for the fishery as 0.09-0.1 kg per m^2 of arctic surfclams. A study looking at the impacts of clam dredging on the bottom (Gilkinson et al. 2003) had taken high resolution sidescan images of the bottom in an area recently fished (Fig. 3). This provided estimates of the percentage the bottom dredges had covered when the vessel moved on to a new area (Roddick and Smith 1999). Discussions of results with vessel captains as to how much additional effort they would put into these areas gave an estimate of 75% coverage of the bottom in high biomass areas.

The survey gave an estimate of 500,000 t of biomass of which 470,000 t was above commercial size (85 mm shell length). There was 344,000 t within areas with a density above 0.09 kg per m^2. Of this it was estimated that 75% or 258,000 t was harvestable, as the vessels could not economically cover more than 75% of the bottom area. This indicates that almost 50% of the survey biomass would not be caught. This may be considered a reserve spawning biomass, although the majority of this biomass is in low density areas where spawning success may be limited. The current TAC of 30,000 t was judged not to be sustainable as it resulted in a fishing mortality on the exploited biomass (344,000 t) exceeding M (0.08). Setting $F = M$ is essentially fishing at MSY so it is usually recommended that F should be set below M (Quinn and Deriso 1999). The TAC was reduced to 24,000 t, which would result in an exploitation rate of $F = 0.9\ M$ on the exploited biomass and $F = 0.6\ M$ on the total biomass. For comparison, the Pacific Fishery Management Council has used $F = 0.75\ M$ for groundfish (PFMC 2001).

Industry began to look at other species they were catching as bycatch. The northern propellerclam (*Cyrtodaria siliqua*) was abundant on both banks, and the Greenland cockle (*Serripes groenlandicus*) on Grand Bank. Sales of these species in 1997-1998 were not followed by repeat orders and more marketing was undertaken to increase demand (Table 2). Ocean quahogs were not an abundant bycatch, but there was a large biomass on Sable Bank and southern Grand Bank. Industry made submissions to DFO for a directed ocean quahog fishery in these areas.

Following industry-funded surveys of Banquereau and Grand Bank, the clam industry entered into a series of joint project agreements (JPAs) with DFO to help fund research programs. Under these agreements the industry provides vessel time, manpower, and/or money depending on the requirements of the JPA. Part of the first JPAs signed for the arctic surfclam fishery covered an onboard sampling program. Industry provides personnel onboard the vessel to carry out sampling programs. Various types of data were collected, one of the most important being length frequencies of the catch. Samples were taken from raw catch, retained catch, and discards. Logbooks and dockside monitoring of catch are mandatory for any license holder. Logbook data lacks accurate discard information because most of the separation of clams from the dredge contents is done mechanically and not observed by the crew. Discard rates are therefore estimated from the sampling data. Catch composition samples are periodically taken from the raw catch and all material is sorted and weighed. Accurate bycatch information will become more important as ecosystem approaches to fisheries management are implemented.

Samples are also run through the processing line to calculate conversion factors. Since all vessels involved in the fishery are freezer/processors, catch data is recorded as processed weights, while TACs are round weights. Sampling personnel also collect samples of frozen whole clams which are sent to DFO for morphometric and age analysis. This data is used for growth studies and to estimate mortality rates. The length frequency data from these studies shown in Fig. 4 indicates a recruitment pulse, found on the eastern end of Banquereau Bank, but the rest of the population produces a single mode. The aging data, however, indicates that recruitment has not been constant. As well as the current pulse seen in the 5- to 10-year-old mode, there are at least two other modes in the age data, centered at approximately 19 and 33 years old. Since many assessment models assume constant recruitment, this would have to be taken into account.

Another JPA was signed in 1998 to help fund a multiyear program examining habitat impacts of clam dredges on Banquereau Bank (Gilkinson et al. 2003). It is providing valuable information on the impact of hydraulic clam dredging on the benthic community.

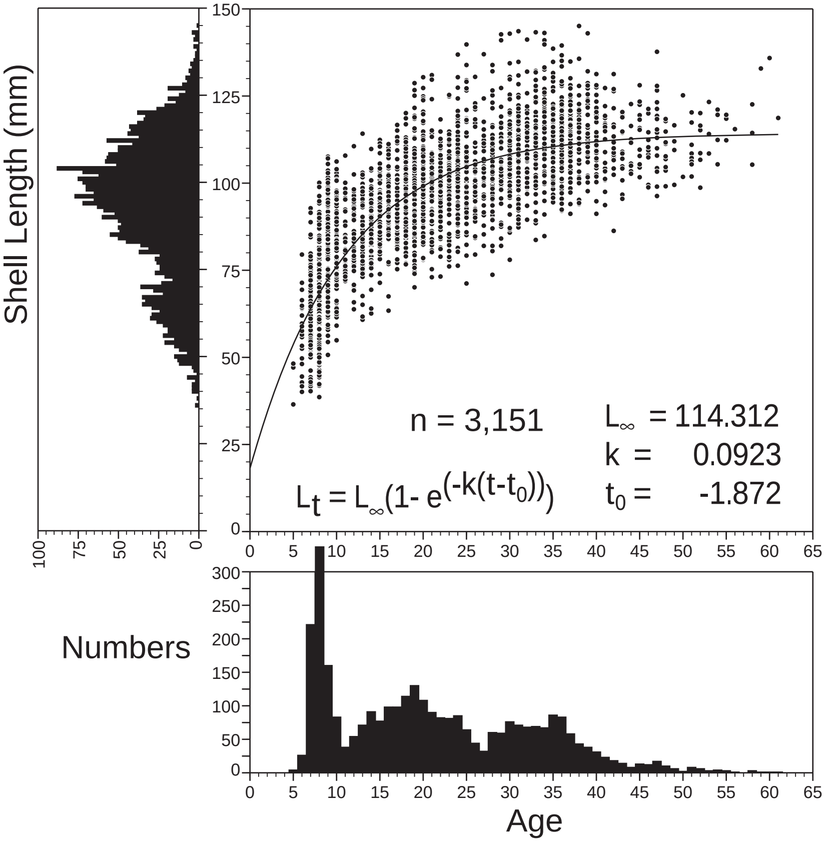

Figure 4. Aging and size frequency data for *Mactromeris polynyma* samples from Banquereau Bank. The growth model is a von Bertalanffy growth curve.

In addition to JPAs, industry involvement increased during this period with the establishment of the Offshore Clam Management Board (OCMB) in 1997. The OCMB consists of representatives from DFO and participants, and was established to oversee and direct the implementation of management plans. The management board is also responsible for recommending scientific research programs to be funded on a DFO-industry basis.

During this period there was further consolidation as the parent company of one participant bought out the clam operations of the other in 1998, so all existing participants are now controlled by one parent company, either directly or through subsidiary companies.

Efficiency and species expansion phase

The 1998-2002 Offshore Surf Clam Integrated Fishery Management Plan was extended for 2003. The fishery was still reporting landings below TAC due to market limitations. Participants invested in changes to vessels to improve efficiency of harvesting and processing operations to maximize return from their EAs. Tension compensating winches were installed on vessels to improve dredge efficiency during rough weather, and processing equipment was replaced and modified to increase product value and to produce different products from bycatch species. In 2003 an exploratory ocean quahog license for Sable Bank was issued, and a joint DFO-industry survey of the bank conducted. The newest integrated fishery management plan will run from 2004 to 2009 and incorporate the ocean quahog fishery.

Discussion

The offshore clam fishery has developed at a steady pace with few problems. The present exploitation rates appear to be sustainable, and the industry has shown its confidence in the future of the fishery with construction of a replacement vessel and a major refit of an existing vessel planned for 2004. It has stated that it will start fishing ocean quahogs in 2005. Part of this confidence may be based on knowledge that there is a large biomass of other species of bivalves on the banks. Information is collected on all clam species during surveys on the Scotian Shelf and Grand Bank, and samples of other species of clams are being collected from the bycatch in the arctic surfclam fishery for morphometrics and aging studies. For some species of interest such as northern propeller-clams, there is little information available in the literature. Preliminary age studies indicate that this species can live more than 70 years, putting their life span between arctic surfclams and ocean quahogs. Industry has encountered difficulties in marketing this species, due to its appearance when whole and the color of the flesh when processed. The Greenland cockle appears to be the fastest growing clam species of interest. It is another species of which there have been few studies. Greenland cockles have not been found in commercial quantities on the Scotian Shelf, but are an abundant bycatch on Grand Bank where their growth rate is faster than that of arctic surfclams. The data being collected will provide preliminary estimates of biomass, growth, and mortality rates. This will be available in the event directed fisheries are developed, and in preparation for the movement toward ecosystem approaches to management, which will require information on effects of the arctic surfclam fishery on other species.

There are problems in setting management targets when there are no regular surveys of the resource. MSY was used as the target reference point in the development phase of the fishery. When MSY is used as a reference point today, it is usually a limit that triggers actions aimed at reducing effort (Mace 2001). There is a variety of target reference points; a common one is a fishing mortality rate which takes a set fraction of the biomass. This can be based on empirical equations such as $F = xM$, as used for arctic surfclams after the 1996-1997 survey, or it can be set using a model such as yield-per-recruit (YPR). A YPR model can give two reference points, $F_{0.1}$ for the target mortality and F_{Max} for the upper limit. F_{Max} is the level of F that results in the maximum yield-per-recruit; $F_{0.1}$ is the fishing mortality rate corresponding to 10% of the slope of the yield-per-recruit curve at the origin (Gulland and Boerema 1973). It is a more conservative level that is below F_{Max} with minimal loss in yield. Another approach is to set a target biomass. The fishery is managed to keep current biomass above a specified fraction of the unfished biomass. These approaches reduce growth overfishing, but do not prevent recruitment overfishing. This can create problems, especially when size of recruitment to the fishery is less than size of maturity. Approaches that set a target in terms of spawning stock biomass are an attempt to address this problem. One such approach is to calculate the spawning stock biomass per recruit (SSBR), and set a target to keep this at some fraction of the unfished state: $\mathrm{SSBR}_{\mathrm{F}}/\mathrm{SSBR}_{\mathrm{F=0}}$. Targets of 0.3 to 0.5 have been used for New England groundfish. A common lower limit used in this approach is 0.2 (Mace and Sissenwine 1993, Mace 1994, Quinn and Deriso 1999). All of these approaches will adjust allowable catch to match changes in biomass, which may increase yields over the long term. They require estimates of current biomass and are usually used in conjunction with a full age-structured model. This implies regular survey and catch sampling programs to collect required data.

A different approach is to set a maximum constant yield (MCY) (Annala 1993, Caddy 1998). The MCY is a yield that can be taken at an acceptable level of risk at all probable levels of the stock. Catches taken under this approach will be lower over the long term than those taken by the previously mentioned approaches, as the MCY method does not change catch levels to match changes in stock biomass. An MCY approach can be used in data limited situations, but perhaps even more important, it does not need regular estimates of population numbers or biomass to see where the fishery is in relation to its target. An MCY is often set with such empirical formulas as $^2/_3\ MSY$, 0.25 MB_0, 0.5 $F_{0.1}B_{\mathrm{av}}$ or 0.5 MB_{av} where B_{av} is average historical biomass (Annala 1993, Zhang 1999). There are factors that will increase or decrease these levels as well. If age of recruitment is above age of maturity, the population is better able to withstand higher fishing pressure (Clark 1991), and yield can rise with increasing age of first capture. Although the management agency does not have to constantly check where the stock is in relation to the target, it must still monitor the stock, as any large changes in recruitment, growth rates, or other population parameters may produce biomass levels below that seen prior to setting the MCY and hence result in overfishing.

The MCY approach has been proposed for ocean quahogs on Canada's East Coast. The largest problem in managing an ocean quahog fishery will be that it is a very long-lived and slow growing species. In the U.S. fishery, much of the catch consists of ocean quahogs between 70 and 100 years old, and the oldest age to date was over 225 years old (Ropes and Murawski 1983). This means that sustainable allowable catch must be a small fraction of biomass. For fisheries on long-lived species such as this, low exploitation rates mean that effects of fishing will not be discernable from natural variations in the population under study. Since most methods used in fishery models require variation in the data in order to fit parameter estimates, model-based methods will probably not be capable of providing precise estimates of all parameters.

In 2002 there was a survey of an inshore ocean quahog bed funded by the inshore clam fishery. Since it is unlikely that a regular survey series or catch sampling program will be established in support of this small inshore fishery, it will have to be managed as a data-poor fishery. An MCY approach was proposed to set an allowable catch. Data available included a biomass estimate from the survey, mortality rates for ocean quahogs from the literature, and some age and size of maturity data for the area from Rowell et al. (1990). Estimating $MCY = 0.25\ MB_0$ provides a conservative catch level for this bed for start-up of the fishery. As the fishery develops, data for growth and mortality estimates specific to this area can be gathered and compared to those taken from the literature. Catch levels can be refined as more data becomes available, and a decision can then be made if it is worthwhile to move to more data intensive methods. A similar approach is envisioned for the start-up of an offshore ocean quahog fishery, but there is a higher probability that regular surveys and catch sampling will be established to allow for less conservative yield estimates.

A similar approach can be taken for the Greenland cockles and northern propellerclams. Their catches were recorded during clam surveys of Banquereau, Grand Bank, and Sable Bank, and aging of samples from commercial catches is supplying estimates of growth rates. There are no estimates of mortality rate in the literature, but preliminary estimates could be calculated using life history invariants, general relationships that exist between such life history traits as natural mortality, life span, and growth rate. Pauly (1980) has described an empirical relationship between M, the von Bertalanffy growth parameters, and mean annual water temperature. Froese and Binohlan (2000) have included such parameters as maximum length and length at maturity to derive their equations. These relationships are very general, but with the difficulties of deriving direct estimates of natural mortality they may be the best estimates available for a considerable period of time. The applicability of these relationships would have to be verified with bivalves, as most of the data used to derive them is from finfish.

Increasing interest in directed fisheries for bivalves on the Scotian Shelf at a time of decreasing DFO budgets means that initial management of these stocks will have to rely on methods that do not require many resources. As most developing fisheries take a few years for the landings to reach allowable catch levels, it is anticipated that by the time yields based on such methods as MCY have become restrictive, data required to move to models that are less conservative will be available. The risk is that if the fishery continues for this initial period without a problem, resources will not be put into gathering required data, and the assessment process will be no further ahead when landings reach the allowable catch.

Part of the decision on which approach to take will depend on the fishing industry. Does it feel that increased catches, made possible by methods that can react to changes in the population, are worth the additional costs they will have to bear to support the data requirements?

Managers also may have to deal with a mixed species fishery if the arctic surfclam fishery expands into other species. Of the species presently being examined, Greenland cockles are faster growing than arctic surfclams, and northern propellerclams are slower. If commercial sizes are above size of maturity it is probable that sustainable exploitation rates will be related to growth rates. For a mixed species fishery, does this mean that exploitation rates should be set at a level to keep the slowest growing species at a sustainable biomass? With the movement toward ecosystem and precautionary approaches to fisheries management, decisions on such issues will have to be made even in the absence of multispecies fisheries.

Managers will also have to address the question of independence of management advice when it is increasingly based on data provided by the industry being managed. The JPA approach to research questions is increasing throughout DFO, and there are both benefits and problems with this type of arrangement. The benefit to DFO is that costs are being absorbed by industry, although it must still provide people and resources for analysis and assessment activities. For industry, costs are lower than for an independent sampling program, such as the observer program on the East Coast. Sampling is also more tailored to the fishery than it would be under the observer program, which, because it was designed to cover a variety of fisheries, is a “generic” program. Another advantage is that industry and personnel onboard vessels are more directly involved in the assessment process, provided they are given some feedback on results. The largest disadvantage is that data are not collected independently. Sampling data can be compared with survey and monitoring data for verification, but it can always be argued that fishery participants could manipulate data to help pursue their own agenda. At present the arctic surfclam industry has little incentive to misreport. There is no reason to high grade or discard catch as the TAC is not limiting, and they have supported research on the habitat impacts of the fishing gear. This may not be true of all fisheries.

JPAs to date have covered regular sampling and research, a large research program on the habitat impacts of the clam dredges on Banquereau Bank (Gilkinson et al. 2003), a genetics study of the population structure of the arctic surfclam stocks on the East Coast, surveys of Banquereau Bank and Grand Bank, and a survey of the ocean quahog (*Arctica islandica*) stocks on Sable Bank. This is work that would not have been carried out without industry cooperation and funding; thus both DFO and industry are making decisions based on better information than they would have without these cooperative arrangements.

Conclusions

The offshore clam fisheries on the Scotian Shelf target long-lived, slow growing species. When this is combined with declining science budgets in DFO, management of this resource will depend on data limited methods or funding from participants to cover the cost of more data intensive methods. The level of industry cooperation and funding for research and stock assessments has been a positive influence on development and management of the resource. The research has provided information on the life history of both target and bycatch species, in some cases giving the first estimates of growth and mortality rates. Improved data and parameter estimates enhance the basis for advice, but raise questions on the independence of advice. Scientists and managers will have to be prepared to defend results based on data gathered by the industry that the data is being used to regulate.

At present the offshore clam fishery is healthy and appears to be sustainable. If it moves into new species and areas in the near future more research will be needed, and surveys should be conducted in advance of the fishery. As the fishery moves forward and management moves toward ecosystem-based approaches, there will be new questions on the optimal management strategy. More information will be required on the effect of the fishery on other components of the ecosystem, and how to balance the fishery components with broad system-wide ecosystem objectives.

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Estimating Economic Effects of Fishery Management Measures Using Geospatial Methods

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Abstract

One of the main limitations of fishery management is that many routine data collection efforts are outpaced by evolving analytical needs. Examples include area-based management measures such as fishing closures and marine protected areas that are emerging in the context of ecosystem-based management. Typically, data collection efforts were not designed to address spatial issues. In the absence of comprehensive observer coverage or vessel monitoring systems, this creates a motivation for reinterpreting old data in new, spatially explicit ways. In this paper, we present a pilot geospatial relational framework for mining and integrating existing data developed for the West Coast of the United States (Washington, Oregon, and California), and discuss its applications to fishery management in the context of area-based management alternatives such as the groundfish closures instituted in 2002. Mining a variety of ecological, fishery dependent, and fishery independent databases, we built an extensive relational database—the Ocean Communities "3 E" ANalysis (OCEAN) framework—which allows the user to jointly consider ecological, economic, and equity (hence "3 E") implications of marine management measures. We standardized the data, and conducted a meta-analysis on them, mapping trends over time and space in a geographic information system (GIS) that covers the length of the West Coast from Washington to California and covers the entire exclusive economic zone (EEZ). Using this framework, we analyzed various management scenarios

in terms of their effects on habitat areas and types, economic activity on shore, and likely implications for the fishing industry.

Introduction

Commercial fisheries off the coasts of Washington, Oregon, and California—as in other parts of the United States—are undergoing dramatic changes. Whether pelagic species, groundfish, or salmon, landings and revenues have been declining in most fisheries in the recent past. In the case of groundfish, the focus of this paper, commercial fishing for a complex of more than 80 species of soles, flounders, rockfish, sablefish, and whiting that are jointly managed by the Pacific Fishery Management Council (PFMC) dates back to the nineteenth century. The fishery has been prosecuted with gear types including hook and line, pots, traps, and trawl nets. After a period of expansion following the passage of the Magnuson-Stevens Fisheries Conservation and Management Act of 1976 (MSFCMA) and the subsequent "Americanization," landings of groundfish have declined over the past 20 years. Some species have declined to levels that trigger stringent rebuilding plans and bycatch avoidance measures, affecting the rest of the groundfish management complex.

The fishery is increasingly managed on the basis of time and area closures designed to minimize bycatch and allow for rebuilding of threatened species. While potentially better aligned with ecosystem mandates for marine resource management (EPAP 1999, NRC 2001), the comparatively new class of spatio-temporal management measures such as 2002 in-season closures poses considerable challenges to the management process. Specifically, fishery dependent and independent data that have been collected on the West Coast for over 20 years cannot be directly interpreted to analyze alternative spatiotemporal management measures or their impacts on the economic well-being of fishing communities.

Recognizing these limitations, a recent National Academy study, in reference to legally mandated essential fish habitat (EFH) assessments and management tools such as marine protected areas, notes that "NMFS [now NOAA Fisheries] and its partner agencies should integrate existing data . . . to provide geographic databases for major fishing grounds" (NRC 2002, p. 3). In the remainder of this paper we outline a spatial framework for linking ecological, fisheries, and socioeconomic information that utilizes existing, readily available data. We discuss the application of this approach to fishery management issues on the West Coast of the United States.

Materials and methods

The OCEAN framework

Conceptually, the Ocean Communities Economic/Ecological/Equity ANalysis (OCEAN) framework is a multilayered information system comprising geographic and other data in a set of linked, "smart" maps—maps with attached databases that relate many different data sources to each other and facilitate their manipulation and analysis at the landscape (or, in this case, oceanscape) scale. It is rooted in the growing literature of marine geographic information systems (GIS) that are being developed to address a host of oceanographic, coastal, and fisheries issues and problems (Kruse et al. 2001, Valavanis 2002, Breman 2002, Green and King 2003). OCEAN is essentially a meta-analytical tool for combining a range of data, using a relational database architecture and spatial analysis as the common currency.

Analytically, the OCEAN approach centers on the spatial association of a heterogeneous set of data. This kind of analysis has been used in other marine applications of GIS, e.g., to assess the location of fishing effort close to shore (Caddy and Carocci 1999) or to detect trends in global fishery statistics (Watson and Pauly 2001). The OCEAN approach operates at an intermediate, regional scale, with explicit consideration of the socioeconomic impacts in coastal communities. The system can be queried from within any one data layer, e.g., to find particular vessels or gear groups fishing in a habitat of interest, or to generate the ex vessel revenues associated with a particular species. Information can be manipulated both in database formats and map-based user interfaces, and results are summarized in map formats.

The centerpiece of this approach is the modeling of data that are already available in spatially explicit formats, and combining them with other, newly spatially interpreted, information. The challenge was to organize data from diverse sources, in diverse formats, and of varying quality, and to integrate them into a single framework. We began work on OCEAN by reviewing existing sources of data and compiling them into one relational database. Where necessary, we built new models to spatially interpret data, especially those pertaining to the distribution of fishing effort. Combining bathymetry and habitat information with fishing effort and species distributions then formed the basis for analyzing where vessels fish by gear type and target species. To this we added an economic model for assessing the relative socioeconomic impacts of different management scenarios. We present here results from an analysis of area-based management measures using "version 1.0" of OCEAN.

Data sources and their limitations

We obtained 14 years of data on landings, revenues, and vessels from the Pacific Fisheries Information Network (PacFIN), which stores and compiles fishery-dependent data that are routinely collected by the three states in our study area (Washington, Oregon, and California) (Sampson and Crone 1997). Data were summarized to individual vessels by port, gear, species, and year for all vessels reporting groundfish landings. The availability and quality of data for different fishing sectors vary considerably. The trawl fishery is documented the best, with at-sea logbooks augmenting the information on catch and landings that is reported port-side in the landing tickets. Trawl logbooks are spatially explicit, with trawl set points recorded for individual trawls (typically referenced by 10-minute blocks). Trawl duration is also recorded, thus providing a measure of effort.

The trawl logbook data had two major limitations. First, although skippers record trawl endpoints, these are not transcribed into the PacFIN database. Since there is as yet no comprehensive vessel monitoring system in place on the West Coast, estimating the precise extent of trawl activity remains rather difficult. To remedy this, we modeled tow tracks based on trawl set points and trawl duration. Essentially, we constrained vectors of possible trawl directions by using habitat and bathymetric consideration for each recorded tow. The result is a density map of probable tow tracks. Second, although the same vessel identifiers are used in both data collection efforts, there remain considerable gaps between the logbook and landing receipts record sets (Fox and Starr 1996, Sampson and Crone 1997). For our analysis, we used a record set provided by PacFIN in which the records were already matched up, thus subsuming any associated uncertainty.

No such logbooks exist for the fixed gear sectors of the fishery, making landing receipts the only source of information. These are less spatially explicit and typically contain no measure of fishing intensity or effort. With the exception of California, where all landings are recorded in 10 minute blocks, the spatial unit of PacFIN landing receipts are statistical areas defined by the now defunct International North Pacific Fisheries Commission (INPFC). There are only twelve INPFC areas for the entire West Coast from Cape Flattery in Washington to the Mexican border, each covering thousands of square miles. We developed an iterative algorithm (described below), drawing on all the data assembled in OCEAN to make the landing receipts more spatially explicit. This, in turn, formed the basis for considering the socioeconomic implications of management measures such as the in-season shelf closure, which affect vessels that used to fish in the now closed areas.

Bathymetry and other data on oceanographic characteristics were obtained from the National Oceanic and Atmospheric Administration (NOAA), the United States Geological Survey (USGS), and state agencies such as the California Department of Fish and Game. One key component for ecosystem management is habitat and the consideration of the impact of fishing activities on different parts of the seafloor. The continental shelf in our study area has been the subject of considerable habitat mapping efforts, such as the USGS habitat GIS for the Monterey Bay National Marine Sanctuary (Wong and Eittreim 2001). Using known habitat associations for various fish species, as well as the depth constraints on particular types of fishing gear, habitat data can be used to relate fishing effort to particular areas.

Trawl surveys conducted by NOAA Fisheries over the past 25 years are a major source of fishery-independent data. We obtained all available years of shelf and slope surveys from the NOAA Alaska Fisheries Science Center, 1977-2001. NOAA research vessels using trawl gear record the total number, size and age distribution, and weight of fish sampled along fixed transects, typically during the summer months (Lauth 2000, Weinberg et al. 2002). Because of the consistency of the sampling protocol the trawl surveys generate a comprehensive picture of species abundance, at least along the trawl transects and during the sampling period. Species and abundance (number of fish per species) are recorded for each trawl start point. We extracted individual records of species targeted in the commercial fishery from the surveys, normalized these by total effort, and generated species-specific density maps. Following an approach developed by NOAA's Biogeography Group (NOAA 2002), we summarized these to 9 km by 9 km analysis units, and derived single, cumulative species distribution maps for each target species.

The final component of OCEAN concerns the linkage between fishing activity and coastal communities. The obvious points of contact are the landing ports, where vessels sell their catch to fish buyers and processors. Together with other marine services and businesses, processing is a major contributor to income generated in coastal communities. This aspect of socioeconomic impacts is already captured in a regional input-output model used by the PFMC to assess the economic impacts of fishery policy (Jensen 1996, 1998). The fisheries economic assessment model (FEAM) belongs to a class of regional input-output models that treat the economic activity in a region as a set of interconnected sectors (Hewings 1985). Each dollar generated in one sector has a "multiplier effect" because it generates economic activity in other sectors. For example, fish are landed and the vessel is paid a price per pound for its catch. Out of this ex vessel revenue, crew shares, maintenance and moorage costs, and other expenses are paid, which in turn generate personal income, and revenues for the port district and other marine-related businesses. The FEAM estimates these effects for the two primary sectors affected by fishing activity, i.e., harvesters (fishermen and their families) and processors. We summarized these model outputs in a set of spreadsheets, which we integrated into OCEAN. This allowed us to consider the income impacts of changes in landings in a port resulting from particular management scenarios.

A key limitation of the FEAM analysis is that it is static in nature and provides only an incomplete snapshot in time. It is premised on the landings and revenues generated by the fishing fleet, but is silent on alternative sources of revenues in coastal communities such as tourism. Unlike other regional input-output models, FEAM is not designed to assess employment effects. Furthermore, a host of considerations beyond economic impacts are of importance to coastal communities and managers, but they are not yet routinely assessed. For example, the lifestyle aspects of fishing communities are important (The H. John Heinz III Center for Science Economics and the Environment, [Hanna et al, 2000]), as are concerns about the social and cultural resilience of ports and towns in response to the structural changes in the fishery (Langdon-Pollock 2002). By way of addressing these concerns, and to lay the groundwork for more in-depth analysis of coastal communities in future applications of OCEAN, we incorporated census statistics as well as qualitative information derived from port visits and interviews.

New tricks: methods for modeling fishing effort

To illustrate the spatial modeling conducted in OCEAN, consider the challenges inherent in determining the spatial extent of trawl and fixed gear fisheries, i.e., the distribution of fishing effort on the fishing grounds. This is key, for example, for assessing the socioeconomic implications of reducing certain sectors of the fleet, assessing the effects of area closures, or for determining the likely habitat interactions of particular gear types. In the absence of a comprehensive observer program (summarized data from the first year of west coast observer program are posted on the Northwest Fisheries Science Center's Web site, but do not provide any insights into the location of the fleet) or vessel monitoring systems, there is considerable uncertainty about where vessels using gear types other than trawl gear are fishing.

The OCEAN effort submodel essentially consists of a sequence of steps, programmed in ArcINFO, which successively constrain each landing record and subsequently apportion catch and revenue to equal area analysis units (9 km by 9 km blocks) based on probability of fishing activity in an area. In contrast to multivariate analysis used in terrestrial applications, which generally predicts what happens in a particular location (e.g., Hargrove and Hoffman 2003), we try to predict the location for known entities. The following steps characterize this process; Fig. 1 shows a flow chart of the model:

1. Each PacFIN record contains information on the gear used, species caught, landing port, vessel information, and one of twelve statistical management areas where the catch originated.

2. Impose a maximum range from the landing port that a vessel is likely to have fished, given its length and gear type used. This is

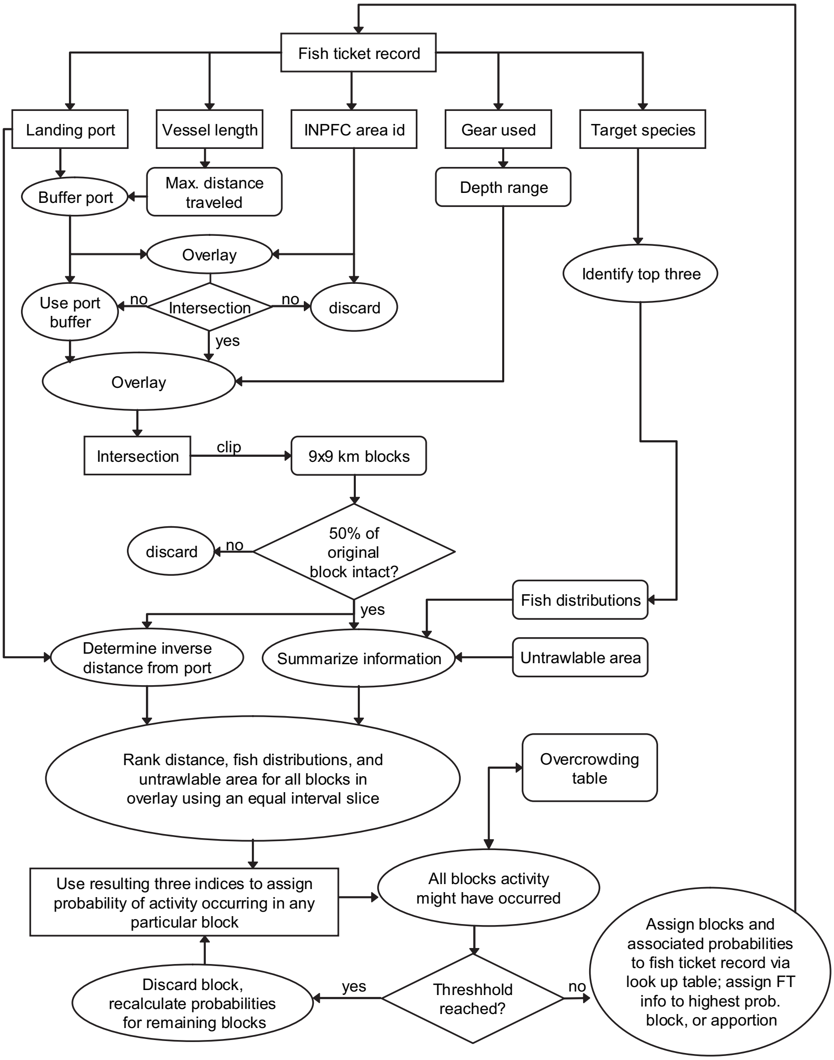

Figure 1. Commercial fishing effort model flow chart.

currently derived from expert witness testimonies, pending more formal studies of fishing behavior on the West Coast.

3. Impose depth restrictions on fishing gear used and target species. There are limits to the depth from which west coast trawlers can haul their nets, or in what depth various fixed gear types are used; similarly, different species of fish have known ranges of bathymetric associations.

4. Compare this to the species distribution densities derived from the fishery-independent surveys. Some areas are associated with higher frequencies of the target species in question, making it more likely that a fishing vessel would have gone there for its catch.

5. Within that maximum range, weight the species density clusters inversely by distance from port. This is a "friction of distance" idea: because travel is costly, vessels tend to fish closer to port even if they are slightly less likely to encounter the target species.

6. Impose habitat restrictions on fishing gear used. Trawlers do not operate in high relief areas, while these same areas tend to be frequented differentially by vessels using hook and line gear.

7. Apportion pounds caught and associated revenue from fish tickets. This can be done either deterministically, associating the entire catch and revenues with the block that has the highest likelihood of fishing having occurred there; or probabilistically, apportioning catch and revenues to fishing blocks within the maximum range based on probabilities derived from distance from port, targeted species densities, habitat restrictions, and previous activity.

8. Repeat for all records and map the resulting distribution of fishing activity. In principle, this can be normalized by number of records associated with an area, or—in the case of trawlers—number and duration of tows made there, to provide a measure of effort.

The maps resulting from this algorithm are probability surfaces of the distribution of fishing effort and the associated catches and revenues (see Fig. 2). The results discussed here are derived from an earlier, deterministic version of the model. We discuss the probabilistic model and its sensitivity to various assumptions in a forthcoming publication (Scholz et al. 2005). In general, however, it is most sensitive to assumptions about the maximum range of vessels from port and about the associations of gear types with particular habitats, as well as to the weight given to the overcrowding parameter. We are in the process of further refining this approach.

It is important to note that the OCEAN effort model constitutes a spatial re-interpretation of historic data. While it is conceivable to turn

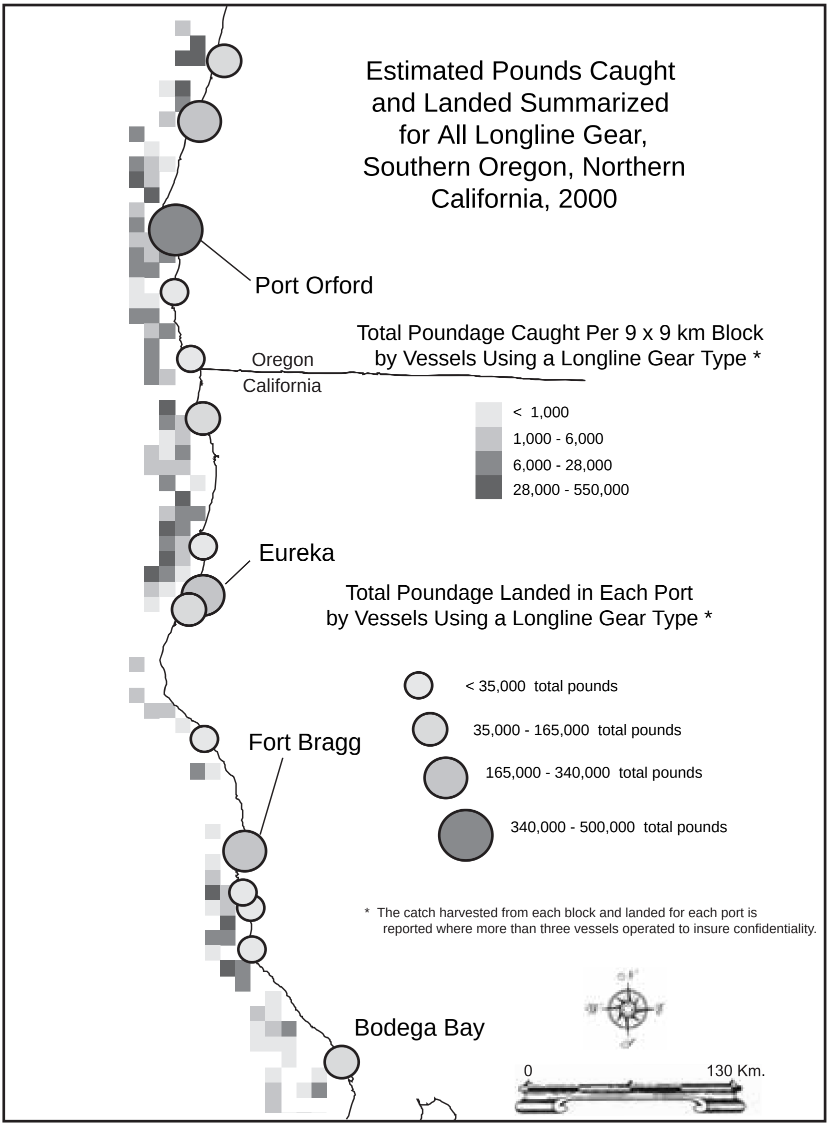

Figure 2. Sample output of commercial fishing effort model.

this into a predictive model, a major confounder lies in the absence of behavioral models of the fishery. In other words, there are few, if any, known rules that describe fleet behavior, and most economic models that attempt this are based on simplistic assumptions about rational actors and individual profit maximizing considerations.

Results: OCEAN applications to fishery management

A key characteristic of OCEAN is that it links fishing activities on the continental shelf to particular gear types, catches, and revenues, and thus to ports. To illustrate the utility of this feature for fishery management, consider the recent shelf closures put into effect on the West Coast of the United States. Implicit in area-based management measures is a displacement effect on fishing vessels. Depending on the size and depth covered by closure areas, some vessels may be induced to exit the fishery.

For example, the 2002 in-season shelf closures affected depths between 100 and 250 fathoms; fishing farther offshore, to the west of the closure area, is only feasible for a subset of the fishing fleet—vessels with sufficiently large engines and deepwater gear. Not all of the vessels that fished in the closure area will successfully relocate closer inshore. As an initial estimate of the potential displacement effect, in economic terms, we considered the shelf closures relative to the landings, revenues, and income generated inside it in the most recent year prior to the closure for which data were available to us, 2000. Figure 3 shows the extent of the shelf closure and the poundage caught there in 2000 by the west coast trawl fishery.

Using the 2000 fleet and effort distributions as a baseline, we identified the number and types (by gear, size, and species targeted) of vessels that likely fished in the closure area. Assuming that the same vessels would have fished there in 2002, we then computed an estimate of the coastwide income impacts associated with the landings initially lost due to the closure. Since this is a static analysis, we did not consider the adaptive effects, and consequently the estimates constitute the upper bound of the wealth effect. For the total coast, with landings worth (in 2000) $137 million, the income impacts generated by landings outside the closure area amount to around US$115 million, and thus the closure potentially results in lost income on the order of US$22 million if vessels were permanently displaced. More interesting, the estimated effects of the shelf closure vary along the coast, since fishing in the shelf areas is of varying importance for different ports, gear groups, and fishing vessels. The estimated income effect of the closure, therefore, varies accordingly, as shown in Fig. 4.

From San Diego in the south to northern Puget Sound on the U.S.–Canadian border, there are significant differences in the degree to which

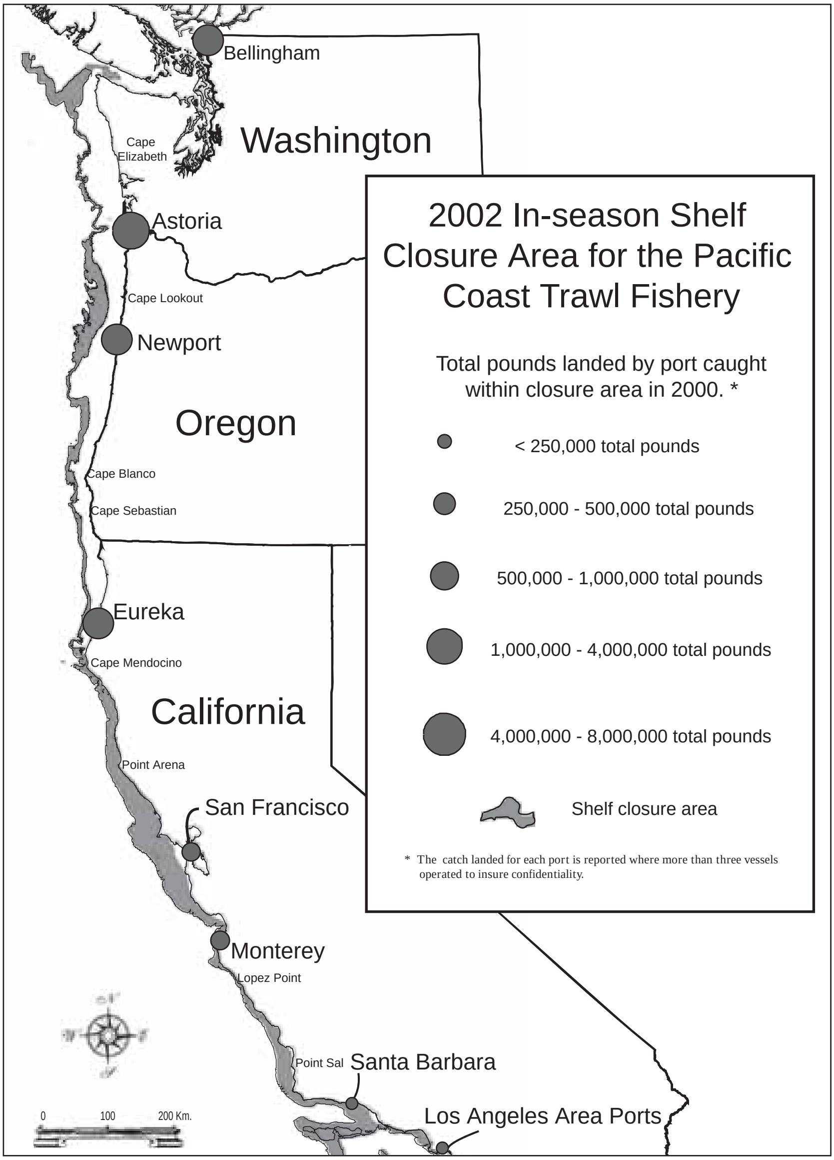

Figure 3. Extent of shelf closure and poundage of groundfish landed by the west coast trawl fishery in 2000 that originated there.

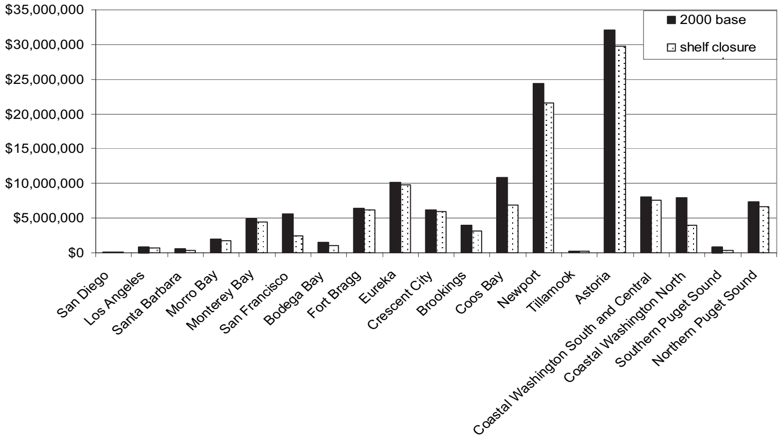

Figure 4. Estimated processor and harvester income from groundfish landings by port group before and after the 2002 shelf closure.

port groups were estimated to have been affected by the shelf closure. While ports farther north generally have higher total groundfish landings, the estimated income effect of the shelf closure area varies considerably relative to other port groups. For example, while Eureka and Crescent City appear relatively unaffected, Newport and Astoria were estimated to experience a larger impact, and a more pronounced income effect is estimated for Coos Bay and northern coastal Washington. These results are a reflection of the regional fishing patterns: the shelf is steeper near Crescent City, making for a narrower shelf closure area, which consequently is not as significant for the local fleet as the shelf off Newport and Astoria.

This is further illustrated by Fig. 5, which shows the percentage of landings and revenues derived from the shelf closure area. Although total income derived from groundfish in southern California ports between San Diego and Morro Bay is small (see base amounts in Fig. 4), the reliance on shelf closure areas can be quite large, e.g., accounting for over 50% of groundfish landings and revenues in Santa Barbara. In other words, what fishing there is for groundfish is highly dependent on the closed areas, which make up much of the area around the historically productive Channel Islands.

Another important aspect of the geographical difference of reliance on the shelf closure areas emerges from the difference between landings and revenues. For example, less than 5% of groundfish landings in Newport are estimated to come from the shelf, but these account for almost

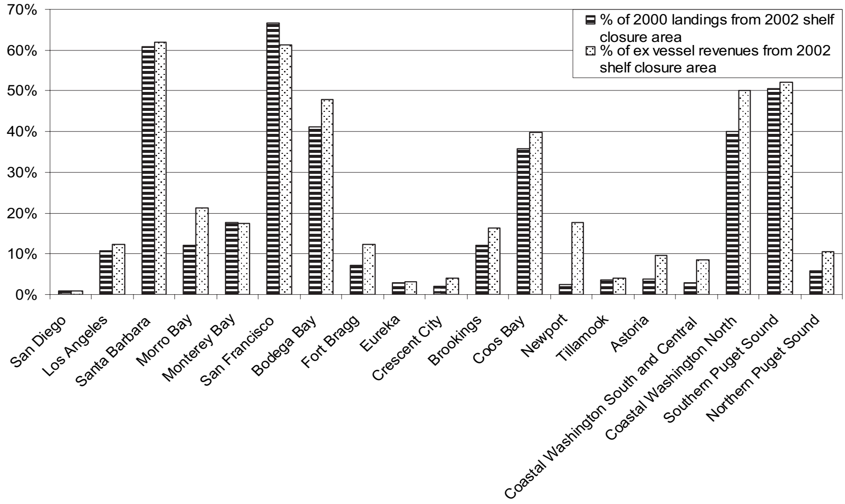

Figure 5. Commercial fishery landings and ex vessel revenues from inside the 2002 shelf closure area.

20% in ex vessel revenues in this port. This suggests that the shelf closure areas yield relatively more valuable species than other fishing grounds. Again, this illustrates the geographic differences of the fishery's reliance on the shelf closure area.

Conclusions

The analysis and results presented here are illustrative of the kind of assessments that can be conducted with a spatially integrated analytical framework such as OCEAN. The OCEAN approach makes use of currently available data and harnesses them in a framework that can be accessed by decision-makers and communities directly. By linking a GIS to fishery data (which, with the advent of electronic logbooks, could conceivably be accessed in real time) and economic impact models, we were able to generate estimates of the effects of fishery management measures, in this case area closures. Other measures that could be analyzed using this framework are fleet restructuring mechanisms such as vessel buybacks or individual quota systems, and measures designed to protect essential fish habitat or reduce bycatch.

Our analysis suggests that geography matters: scenarios have location-specific and differential effects in different parts of the coast, on differently composed fleets (by size and gear-types), and by the relative reliance on particular species or fishing grounds. We are cautiously optimistic that, as spatially explicit models such as OCEAN become more commonplace in fishery management, decision-makers and stakeholders will seize upon them for investigating synergistic effects, e.g., between habitat protection and increasing the economic feasibility of the fleet.

While there are many conceivable extensions of this approach to predictive modeling, an immediate benefit of OCEAN is that it makes visible and spatially interprets the existing data, and thus helps identify gaps and problems with current information sources. In particular, it remains to be seen if the spatial interpretation of historical landing receipts can be validated using the forthcoming observer data on the West Coast or a future vessel monitoring system.

Acknowledgments

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Sportfishing Catch and Harvest of Pacific Halibut (*Hippoglossus stenolepis*) in Glacier Bay National Park

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Abstract

Actions taken by a manager to reduce or increase allowable harvest are legitimized through the collection of accurate fishery harvest information. In many cases the fishery is large and/or remote, contains a large diversity of users, lacks historical data, or is subject to large reporting errors that hamper collection efforts and decrease the accuracy and precision of estimates. These data limited situations require a holistic approach when surveys are being designed. Glacier Bay National Park presents a data limited situation in that the existing survey data does not enable managers to estimate sportfish catch and harvest of Pacific halibut (*Hippoglossus stenolepis*) in park waters. This paper focuses on a combination of survey methodologies used to create a fishery survey program in Glacier Bay National Park to estimate halibut harvest and catch by sport anglers. Methods employed were a creel survey, aerial survey, mail survey, telephone survey, and boat observations. We found that the mail and telephone surveys provided the most precise estimates of halibut catch and harvest, while the creel and aerial surveys provided the least precise estimates. We also found that anglers included in the mail and telephone surveys accounted for the majority of the sportfish halibut catch and harvest in Glacier Bay proper and areas directly adjacent to Glacier Bay proper. Moreover, data from the aerial survey suggest that anglers misreported halibut catch and harvest levels in the creel survey. This paper is a contribution to a general methodology that could be employed in the management of the Glacier Bay National Park sportfishery. The framework created in this paper is flexible enough to be of value in the management of sport fisheries in other locations as well.

Introduction

Fishery managers are faced with the difficult task of modifying human behavior through regulations that control fishery harvest (Cicin-Sain and Knecht 2000). Actions taken by a manager to reduce or increase allowable harvest are legitimized through the collection of accurate fishery harvest information. In many cases the fishery is large and/or remote, contains a large diversity of users, lacks historical data, or is subject to large reporting errors that hamper collection efforts and decrease the precision of estimates (Crosby 1995, Crosby et al. 2000). These data limited situations require a holistic approach when surveys are being designed. Distinct attributes such as access points, level of use, political climate, and user group needs should be incorporated into a survey design (Pollock et al. 1994).

Mail and logbook survey programs implemented by Glacier Bay National Park (GBNP) and the Alaska Department of Fish and Game (ADFG) provide little to no information for sportfishing related catch, harvest, and effort for Pacific halibut (*Hippoglossus stenolopis*) occurring in park waters by user group (private and charter) and point of origin (Gustavus or Bartlett Cove). These data limited characteristics of GBNP have prompted resource managers to assess halibut sportfishing pressure within GBNP.

This study provides baseline data in GBNP for halibut catch, harvest, and effort by user group and port of origin. This study also outlines methodologies that can be applied to other recreational fisheries that have diverse user groups, are situated near marine protected areas, have

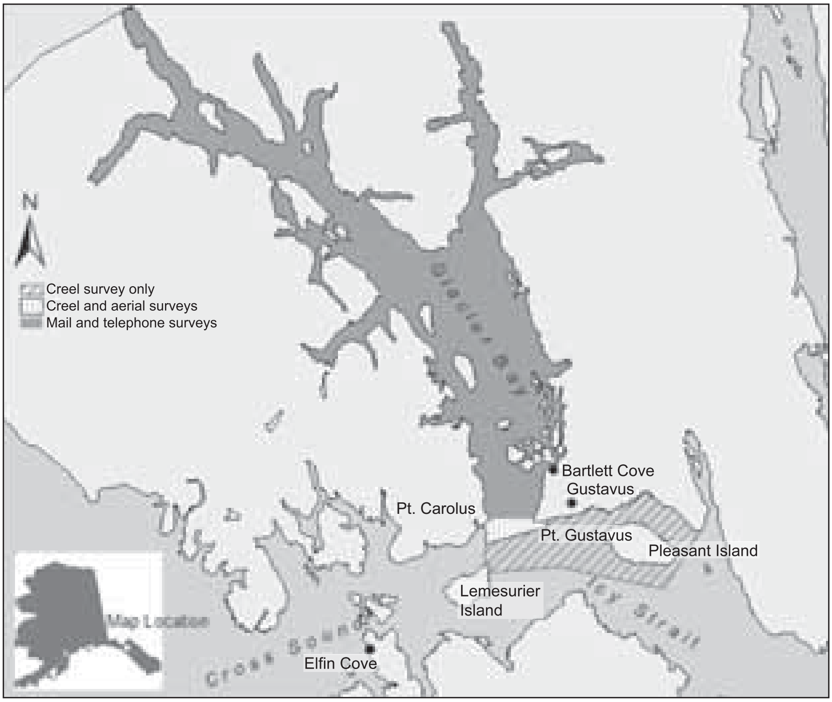


Figure 1. Map showing fishing areas covered by the creel, aerial, telephone, and mail surveys. Mail and telephone surveys covered the entire Glacier Bay.

complex licensing or permitting requirements, and/or have little or no baseline monitoring data.

Background

Glacier Bay National Park is located approximately 90 miles west of Juneau, Alaska, in remote northern Southeast Alaska. Marine waters in GBNP encompass 243,339 km^2 and include Glacier Bay proper, waters within the Cross Sound/Icy Straight region, and the Outer Coast from Cross Sound to Sea Otter Creek north of Cape Fairweather (Fig. 1). Approximately 45% of GBNP marine waters (110,000 km^2) exist outside Glacier Bay proper. The Glacier Bay area is a popular sportfishing destination with halibut, salmon (*Oncorhynchus* spp.), and rockfish (*Sebastes* spp.) being the primary target species. Sportfishing in the area is focused on the arrival of anadromous fish and the movement of halibut into the area during the summer months (May-September). The majority of sportfishing in the region occurs May 15-September 15. Fishing effort originates from three locations: Gustavus, Elfin Cove, and Bartlett Cove. There are four primary user groups in the sportfishery: marine anglers based out of Gustavus, Elfin Cove, and Bartlett Cove; and marine anglers entering Glacier Bay proper during a special GBNP permitting season (June 1-August 31). Sportfishing charters operate from the communities of Gustavus and Elfin Cove.

Vessel activity in the bay proper is further restricted by a limit on the total number of boats that are allowed to reside in the bay per day. The permitting system consists of a local permit and general permit. Local permits are issued to private marine boaters originating from the communities of Gustavus and Bartlett Cove. General permits are issued to private boaters (including locals from Gustavus and Bartlett Cove) entering Glacier Bay proper.

Charter businesses are allowed to operate within GBNP through a separate permitting system. A GBNP concessionaires permit or incidental business permit is required for sportfishing charters operating within park waters.

Private and charter recreational fishing activity (effort, catch, harvest) for anglers in the greater Glacier Bay region is documented by state and park sponsored programs. These programs are composed of a charter logbook program implemented by GBNP (since 1995) and a mail-out survey (Statewide Harvest Survey: SWHS) administered by ADFG.

The SWHS survey is not useful for understanding sportfishing activity relative to GBNP boundaries because it fails to delineate catch, harvest, and effort occurring within park boundaries. Natural Resource Management Guidelines for GBNP require that "fishing activities must be managed through . . . a harvest reporting system which addresses harvests within park boundaries." Similarly, the GBNP logbook program has a limited ability to assess fishing activity relative to park boundaries. The logbook program is for permitted charter anglers and thus excludes private anglers, non-permitted charter anglers, and charter anglers fishing illegally. The logbook program also may not accurately reflect charter fishing activity in GBNP, as it relies on self-reporting and has not been validated.

Project goals

The primary goal of this study is to estimate harvest and catch of sport-caught halibut in marine waters directly adjacent to and within Glacier Bay proper. Marine waters adjacent to Glacier Bay proper encompass an area between northeastern Lemesurier Island, Pt. Carolus, Pt. Gustavus, and approximately 5 km south of the GBNP boundary (Fig. 1). Three objectives are discussed in context with the primary goal: (1) provide halibut catch and harvest estimates for each survey method; (2) determine the precision of each survey method; (3) evaluate the accuracy of creel survey responses using an aerial survey. The final section of this paper provides a brief discussion on how this study improves the data limited situation in GBNP.

Survey methods described in this paper are also used to obtain estimates for rockfish, salmon, and lingcod (*Ophiodon elongatus*). For the purpose of this paper, only results for halibut are presented.

Methods

The complexity of the permitting system, remote nature of the fishery, and disparate use levels between charter and private anglers required a multifaceted survey program. Four survey methods were used to obtain an estimate of halibut catch and harvest in GBNP waters: (1) a creel survey at the Gustavus dock for private and charter anglers; (2) a mail-out survey for general permitted anglers; (3) an offsite telephone survey for local permitted boaters; and (4) an aerial survey of charter boats that were fishing in park waters directly adjacent to Glacier Bay proper. The following sections discuss each method in detail.

Gustavus creel survey

Anglers fishing in marine waters from a boat were intercepted at their landing site (dock) after their fishing trip was completed. Catch, harvest, and effort information relative to the location of fishing (ADFG statistical area and GBNP specific statistical areas), type of fishing being conducted (bottomfishing or salmon), and fishing trip type (private or charter) were collected at the Gustavus dock. Creel surveys were conducted from June 1 to September 15, 2003.

Although the Gustavus creel survey covered a wide geographic area and involved multiple species, this method is focused only on Glacier Bay proper, marine water adjacent to Glacier Bay proper, and fishing activity relevant to halibut.

Estimates of catch and harvest for halibut in the Gustavus fishery were made between June 1 and September 15, 2003, using a two-stage, stratified random design (Cochran 1977). The first stage consisted of the days sampled and the second stage was composed of boating parties. A single access point was used on a daily basis by private and charter boats in Gustavus. A pilot study showed that approximately 90% of the daily activity occurred between 1200 and 1900 hours (Gasper et al. 2004). Sampling effort was limited to that time period because it captured the majority of anglers completing their fishing activities.

Strata were divided into weekend and weekday strata due to potential differences in use that result from angler work schedules. All days within the weekend stratum were sampled and three to five days were randomly sampled during the weekday stratum. A census was attempted for all boating parties within a sampling period. To insure all parties who fished were counted, we asked parties who refused an interview if they had fished.

Analysis

The following equations were used to estimate catch and harvest by substituting the appropriate statistic. Equations 1.1-1.8 were obtained from Bernard et al. (1998) and Cochran (1977). Calculations of the fishery statistic $(\hat{Y}_{hi})$ on day i within stratum h were made for a user type and location as

$$\hat{Y}_{hi} = M_{hi}\bar{y}_{hi} \tag{1.1}$$

where:

M_{hi} = the count of anglers during day i in stratum h, and

$\bar{y}_{hi}$ = the mean statistic (catch or harvest) of interviewed anglers on the i^{th} day in the h^{th} stratum (weekend or weekday).

The estimate of the fishery statistic (catch or harvest) for stratum h is (Bernard et al. 1998)

$$\hat{Y}_h = D_h \frac{\sum_{i=1}^{d_h} \hat{Y}_{hi}}{d_h} \tag{1.2}$$

where:

D_h = the number of days in the sampling frame,

d_h = the number of days sampled, and

$\hat{Y}_h$ = the total statistic (catch or harvest) estimate for stratum h.

An estimate of the variance for catch or harvest is:

$$Var(\hat{Y}_h) = (1 - f_{1h})D_h^2 \frac{S_{1h}^2}{d_h} + f_{1h}^{-1} \sum_{i=1}^{d_h} M_{hi}^2 \left(1 - f_{2hi}\right) \frac{s_{2hi}^2}{m_{hi}} \tag{1.3}$$

where:

$$f_{1h} = \frac{d_h}{D_h}, \tag{1.4}$$

$$f_{2hi} = \frac{m_{hi}}{M_{hi}} \,, \tag{1.5}$$

m_{hi} = the total number of boating parties interviewed in stratum h on day i, and

M_{hi} = the total number of boating parties counted in stratum h on day i.

$$s^2_{2hi} = \frac{\sum_{j=1}^{m_{hij}} (y_{hij} - \bar{y}_{hi})^2}{m_{hi} - 1} \,, \tag{1.6}$$

y_{hij} = the fishery statistic (catch or harvest) for boating party j on day i in stratum h,

$$S^2_{1h} = \frac{\sum_{i=1}^{d_h} (\hat{Y}_{hi} - \hat{\bar{Y}}_h)^2}{d_h - 1} \text{ , and} \tag{1.7}$$

$\hat{Y}_h$ = the statistic (catch or harvest) for day i in stratum h. Note: $Y_{hi} = \sum y_{hij}$ when $M_{hi} = m_{hi}$.

The mean fishery statistic parameter estimate (catch or harvest) is

$$\hat{\bar{Y}}_h = \frac{\sum_{i-1}^{d_h} \hat{Y}_{hi}}{d_h} \,. \tag{1.8}$$

Aerial survey

Aerial surveys were used in conjunction with the Gustavus creel survey to estimate halibut catch and effort for GBNP waters between Pt. Gustavus and Pt. Corolus. Aerial surveys were used to estimate the number of boats fishing within park waters and the creel census was used to model catch or harvest based on effort (number of boats). Aerial surveys were conducted between June 16 and September 15, 2003. A total of 46 days were surveyed out of the 106 day sampling frame.

An estimate for the number of boats residing within GBNP using observations from the aerial survey was based on a two-stage randomly stratified design. The first stage, based on days sampled, was stratified into weekend and weekday sampling periods. The second stage, based on the time of day sampled, was stratified into equally sized morning and afternoon periods: 700-1200 for morning periods and 1201-1900 for afternoon periods. Two to four weekdays and one or two weekend days were randomly selected for each week surveyed. A census of all boats was assumed during each survey period because of the small size of the survey area, short flight duration across the sampling area (<5 min), and positive identification of boats using digital photography.

Charter vessels from Gustavus were visually identified using the five-digit Commercial Fishery Entry Commission (CFEC) vessel license number visible on a vessel's side, the vessel's name, or by comparison with photographs of vessels residing at the Gustavus dock. A unique characteristic such as gear configuration, boat color, motor configuration, or navigational equipment was also used to identify a digitally recorded vessel license number. It was generally easy to identify charter vessels from Gustavus because they are homogeneous in style and length, ranging from 24 to 28 feet and always having an enclosed cabin.

Several criteria were used to identify charter vessel activity: (1) visual conformation of fishing gear (i.e., rods and reels, rod holders, and downriggers, etc.) being used; (2) the presence of several (> 2) anglers fishing actively on deck; and (3) if the vessel was in transit (moving rapidly) or anchored. It was assumed an anchored vessel was fishing, while a vessel in transit (unless trolling gear deployed) was not fishing. To further document aerial survey observations, digital photographs of each observed vessel were archived and verified.

Analysis

Because the aerial survey was designed to monitor overall vessel use and activity without considering fishery estimates, sampling was not optimized to correspond with periods of peak charter use. Thus, the number of charter boats observed fishing within park waters was very low (10 boats). All strata were combined to facilitate statistical expansion by increasing sample size.

Calculation of halibut harvest and catch was possible because of a linear relationship observed during the creel survey between the number of boats fishing and number of halibut caught or harvested (equation 2.1). Separate linear regressions were observed for catch and harvest. This relationship is discussed further in the results section. Equations 2.1-2.4 were used to estimate the total harvest and catch of halibut $(\hat{h}_{reg})$ for charter vessels fishing in park waters.

$$\hat{h}_{reg} = b_1(K) - b_0 \tag{2.1}$$

where:

b_1 = slope of the regression line,

b_o = intercept of regression line, and

K = number of vessels.

The variance for each point estimate $S^2(\hat{h}_{reg})$ is described in (2.2) (Neter et al. 1996).

$$S^2(\hat{h}_{reg}) = MSE\left(\frac{1}{d_h} + \frac{(k_i - \bar{k})^2}{\sum (k_i - \bar{k})^2}\right) \tag{2.2}$$

where:

k_i = i^{th} observation of the independent variable, and

MSE= mean square error for the regression in equation 2.3.

A final harvest or catch estimate $(\hat{H}_{est})$ was calculated by expanding the mean harvest for all point estimates $(\hat{h}_{reg})$ calculated in 2.1 to the sampling frame (2.3).

$$\hat{H}_{est} = D_h(\hat{\bar{h}}_{reg}) \tag{2.3}$$

The variance for the final estimate of halibut harvest and catch is described in 2.4.

$$V\hat{a}r(\hat{H}_{final}) = D_h^2(1 - f_{1h})\frac{S^2}{d_h} + \sum MSE\left(\frac{1}{d_h} + \frac{(k_i - \bar{k})^2}{\sum (k_i - \bar{k})^2}\right) \tag{2.4}$$

Telephone and mail survey methods

Offsite methods were used to estimate halibut harvest and catch that occurred within Glacier Bay proper. The offsite methods were a telephone survey for private boaters using GBNP permits to enter Glacier Bay proper. In both surveys, vessel permit requirements within the bay proper presented a known respondent list for all boats legally fishing within the bay proper.

Telephone survey

The sampling frame for the telephone survey was June 12-August 31, 2003, and included all anglers using a local permit to enter Glacier Bay proper two or more times during the sampling period. Local permit holders were sampled upon their initial entry into Glacier Bay proper by mail survey, whereas the telephone survey estimated catch and harvest on second and later entries only.

To limit recall and telescoping bias, boaters were contacted via telephone within two weeks of their completed fishing trip. ("Recall bias" occurs when anglers cannot accurately recall events due to the passage of time [Pollock et al. 1994]; "telescoping bias" occurs when anglers confuse trips.) Boaters were called up to five times in an effort to contact as many as possible. The same information as described in the creel survey was elicited from respondents.

Halibut harvest and catch $(\hat{H}_{tel})$ were estimated using a direct expansion method as described in 3.1.

$$\hat{H}_{tel} = N \times \bar{Y} \times P \tag{3.1}$$

where:

N = population of respondents who fished,

Y = Mean harvest or catch per boating party, and

P = the proportion of parties who fished for halibut. Also, $P = n/N$, where n is the total number of respondents indicating they had fished for halibut.

Because a census was attempted, the sample variance was assumed to be an unbiased estimate of the population variance.

Mail survey

A census was attempted on all boaters over the age of 17 who had an Alaska fishing license and entered Glacier Bay proper aboard a private boat for the first time during the general permitting season: June 1-September 15, 2003. Boaters using the general or local permitting system are required by GBNP to complete an orientation at the GBNP visitor information station (VIS) upon entering the park for the first time each season. Following the required orientation, VIS staff urged respondents to complete a questionnaire and survey.

Boaters agreeing to complete the survey were given a packet containing the following: an introductory letter that explained management reasons for conducting the survey, a contact questionnaire that was completed before their trip, a survey form to be completed while on their trip, a park map showing predefined statistical areas, and a prepaid return envelope. The contact questionnaire was used to obtain general demographic information, whether respondents possessed a valid Alaska fishing license, and provided names and addresses used for follow-up mailings. The mail questionnaire asked respondents about the number of rods and hours spent fishing at each location for a type of fishing (halibut, salmon, or bottomfish). Catch and harvest information for each type of fishing at each location was also requested. Information was recorded for each day fished within Glacier Bay proper.

In theory, all visitors who entered the park for the first time during the 2003 season would have submitted a completed contact sheet, thus providing a record of the entire first-visit-of-the-season boating population. In reality a census was not obtained because survey technicians missed potential respondents during orientation. Instead, the total number of possible respondents was obtained from park records. The contact questionnaire was used to obtain contact information and evaluate response biases between groups of differing age and gender.

In an effort to increase response rates, three follow-up reminder letters were distributed to all participants who completed a contact sheet: the first reminder letter was sent to all participants within two weeks of initial contact; the second reminder, containing another survey packet, was mailed within two weeks of the initial reminder to those who had yet to return their mail survey; and a final third reminder was mailed six weeks after the distribution of the first letter to anyone who had not returned their survey.

Results

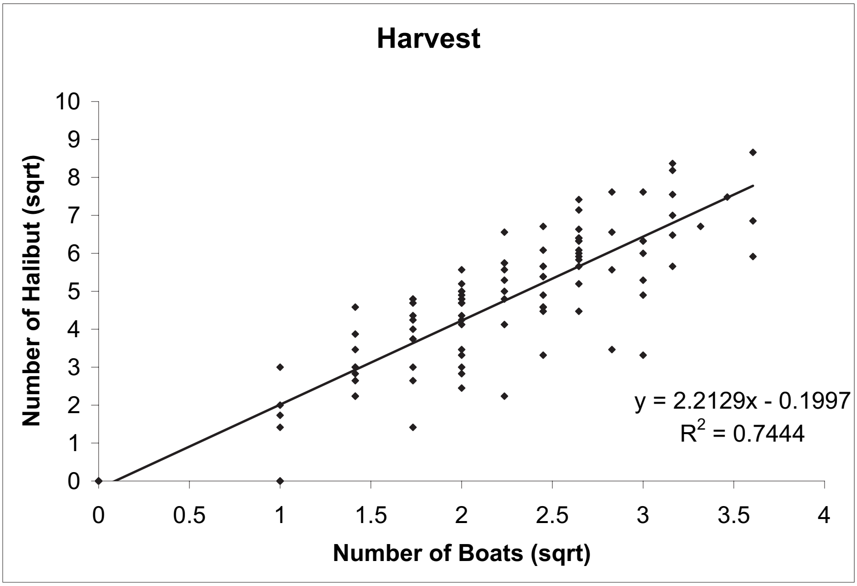
Gustavus creel survey

Halibut harvest and catch levels based on reports by charter anglers in Gustavus were low, as indicated by only 59 (SE = 12.5) halibut harvested and 80 (SE = 13.3) halibut caught directly adjacent to Glacier Bay proper. High variability in catch and harvest across days resulted in low precision for catch (32.9%) and harvest (41.9%). Moreover, it is likely estimates are biased low due to large misreporting error discussed later in this paper.

Aerial survey

To quantify misreporting that occurred during the Gustavus creel survey using aerial methods outlined in equations 2.1-2.4, linear regressions comparing catch and harvest (dependent variable) with effort (independent variable) were modeled (Fig. 2). Normalizing both the dependent and independent variables with a square root transformation reduced heteroscedasticity in the residuals. The resulting regressions had strong relationships between the dependent and independent variables as demonstrated by an R^2 of 0.74 for harvest and 0.70 for catch.

Estimates of halibut harvest and catch using aerial survey methods were 123% and 333% higher than creel survey estimates, respectively. The estimated halibut harvest and catch using aerial survey methods was 99 (SE = 30.7) and 197 (SE = 64.5) fish, respectively. Based on these results, charter anglers operating from the Gustavus dock accounted for 12% of the total catch and harvest in Glacier Bay (Table 1). Due to the small sample size in the aerial survey, the relative precision for estimates was low: 65% for catch and 61% for harvest estimates.



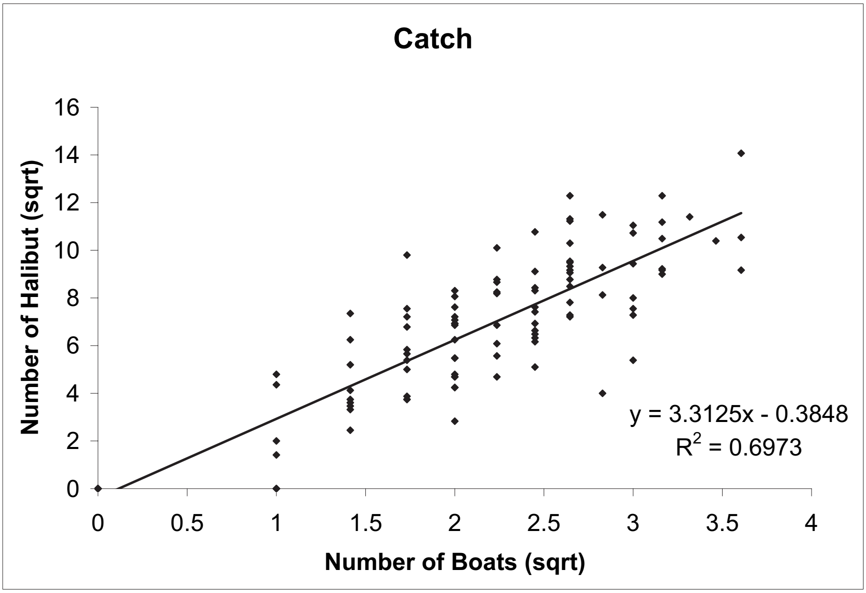


Figure 2. Linear relationships between halibut catch/harvest (individuals) and bottomfishing effort (rod hours) for charter sport boats originating from the Gustavus sampling site. (sqrt = square root.)

Telephone survey

During the survey period, a total of 115 respondent phone numbers were called, of which 5 were no longer in service and 86 were contacted (78.1% response rate observed). Of the 86 respondents contacted, 79.1% indicated they fished. Assuming this percentage reflects the entire population, then 79.1% of the local permit holders on their second or later visit had someone in their party fish. Thus, the total number of parties that could have fished during the survey period was 91 (115 × 0.791).

Local permitted anglers who entered Glacier Bay proper on their second or later trip accounted for 31% of both the total halibut catch and harvest (Table 1). An estimated 496 (SE = 6.54) halibut were caught and 261 (SE = 3.4) harvested in park waters. Most of the halibut catch and harvest occurred within 1 mile of Bartlett Cove. The near census resulted in very high precision for catch (2.6%) and harvest estimates (2.5%). The near census also insured the estimates were representative of the population and reduced the potential for a biased estimate. It is unlikely that recall bias was an issue for this survey because call-back periods were short (<2 weeks).

Mail survey

A total of 156 useable addresses were obtained from a population of 729 boaters who entered Glacier Bay on their first visit of the 2003 season. A response rate of 62.8% was observed, of which 46.2% indicated they used fishing gear that targeted halibut. Since only 21% (156/729) of potential respondents were contacted, the chances for biased estimates may be high.

Data from the questionnaire completed during orientation provided an opportunity to evaluate possible differences between respondents and non-respondents using chi square and *t*-tests. Such tests assessing possible non-response bias were focused on two categories of characteristics: (1) visitor characteristics, included gender, age, residence, and whether the visitor was the permit holder for the group; (2) group characteristics, included the number of visitors in the party, the type of visitor group (i.e., family, friends, etc.), whether the group included members under age 18, and how many people in the group had Alaska fishing licenses. Statistically significant differences in response rates were found for visitor age (characteristic 1). Respondents who returned the mail-back questionnaire averaged 54 years, whereas those who returned only the contact sheet averaged 50 years, $t\,(158) = -2.40$, $P = 0.02$. Response differences by age are very common in this type of survey (Osterhoudt et al. 2004) and additional analysis suggested the effects of respondents' age on the variable measured in this mail survey were not large enough to alter representativeness of the sample in important ways; however, it is impossible to completely rule out undetected examples of non-response bias.

Table 1. Estimated halibut harvest and catch by survey type.

	Catch			Harvest		
	Estimate	SE	Rel. Prec.[a]	Estimate	SE	Rel. Prec.
Mail	904	6.1	1.3%	485	2.3	0.9%
Telephone	496	6.5	2.6%	261	3.4	2.5%
Creel	80	12.5	41.5%	59	13.3	32.6%
Aerial	197	64.5	64.1%	99	30.7	60.8%
Total[b]	1,597	65.1[c]	8.0%	845	31.0	7.2%

[a] Rel. Prec. refers to the relative precision statistic: SE × 1.96/estimate × 100.

[b] Sum of mail, telephone, and aerial survey estimates.

[c] Standard error (SE) is calculated as the sum of the $\sqrt{V\hat{a}r}$ for each survey. Survey populations are independent from each other; thus the total SE is the square root of the sum of each survey variance ($SE_{total} = \sqrt{V\hat{a}r(telephone) + V\hat{a}r(mail) + V\hat{a}r(aerial)}$). The creel survey variance is excluded from the total SE.

Local and general permitted anglers who entered Glacier Bay proper for their first time accounted for 57% of both the total halibut catch and harvest (Table 1). An estimated 904 (SE = 6.1) halibut were caught, of which 485 (SE = 2.3) were harvested in park waters.

Discussion

The results of the four survey methods provided estimates of fishing harvest and catch for halibut within GBNP waters. The creel survey accounted for the smallest catch and harvest and was biased low when compared with aerial survey estimates. The mail and telephone surveys accounted for the majority of the catch and harvest. The aerial and creel surveys had the lowest precision, while telephone and mail surveys had the highest relative precision.

The large reporting error in the Gustavus creel survey, as demonstrated by the aerial survey, resulted in a gross underestimate of halibut harvest and catch. These high misreporting levels may have been due to some charter anglers not knowing (or not paying attention) to the location of park boundaries outside Glacier Bay proper, non-permitted charter vessels operating within the park, guides avoiding the need to file use reports or logbooks at the end of the season as required by GBNP, or apprehension about providing creel census data to the park due to privacy concerns. The reasons behind misreporting were not studied in this report. Therefore, the above-mentioned reasons are conjecture, but are provided to demonstrate the impact external survey factors can have on a survey program. Assessment of user attitudes toward a survey program is especially important in a data limited situation. Without the aerial survey, park managers would not have been able to quantify reporting bias in the creel survey and adjust estimates accordingly.

Aerial surveys generally are not used to estimate catch (Pollock et al. 1994) because it is difficult to determine a boat's origin and if the boaters were actively fishing. This problem was circumvented with the use of high-resolution digital photography. The identity of individual sportfishing boats could be accurately determined and matched with creel survey data. The remote nature of the GBNP sportfishery facilitated the meshing of creel surveys with the aerial survey because few high-use access points exist. Survey situations that have many access points and/or transient boaters from communities outside the sampling universe may have difficulties with this method because the catch or harvest statistic cannot be assessed using creel survey methods. Furthermore, the high correlation observed between catch and effort allowed development of a regression model for halibut fishing that could be used to estimate catch based upon observed effort. More variable fisheries may not have a strong enough relationship between catch (or harvest) and effort to establish a reasonably accurate model.

Despite the economic concerns associated with large complex surveys, it is particularly important in data limited situations to assess sources of bias. Brown (1991) best summed up the economics of survey design with the statement: "The problem is that the cheap biased data are not cheap. They are barely affordable." This was particularly relevant to this survey, where high misreporting was observed in the creel survey, a small portion of the population was captured in the mail survey (which may result in a biased estimate), and the telephone survey was a relative success. Despite the higher costs potentially accrued from using four different survey methods, all surveys were needed to assess bias and provide baseline data to inform future survey designs. In particular, large costs associated with aerial and creel surveys were offset by the information gained about creel survey bias.

Conclusion

Managers should view offsite survey results with caution due to recall and non-response bias. In this study the offsite telephone survey was the most accurate out of the three survey methods (creel, mail, and telephone surveys). For larger scale fisheries with multiple fishing sites, telephone surveys would be very difficult to implement. However, in specific areas (i.e., a bay or special use area), telephone surveys are useful for estimating catch, harvest, and effort in a sportfishery. Moreover, telephone surveys could be used to validate other survey methodology. For instance, a sample of respondents fishing in a known area, that was surveyed using mail-out methods, could be surveyed using a telephone survey (i.e., mail complemented with a telephone survey). This may expose problems in survey design such as fishing occurring on statistical or political boundaries, misreporting, and recall bias.

The data limited situation encountered in this study was characterized by a lack of baseline data on use levels, the distribution of use across ports and on the fishery grounds, and misreporting problems. Without the benefit of using an aerial survey, it would have been very difficult to determine reporting accuracy of the creel survey at the Gustavus dock. This survey did not evaluate the accuracy of mail and survey responses. Future studies need to assess the accuracy of the mail and telephone surveys and use fishery independent methods complemented with a creel survey at Gustavus to estimate the amount of halibut catch and harvest occurring in park waters.

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Use of Observer Data in Management of the Grooved Tanner Crab (*Chionoecetes tanneri*) Fishery off Alaska

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Abstract

The commercial fishery for grooved Tanner crab, *Chionoecetes tanneri*, in deep (typically more than 600 m) waters off Alaska developed rapidly from the late 1980s to its peak in 1995, when 1,300 t valued at $4.4 million were harvested from the Bering Sea, Aleutian Islands, and Gulf of Alaska. Development of management measures for this fishery has been hampered by lack of stock assessment surveys, resulting in a lack of annual stock abundance estimates, and scant information on life history and productivity parameters. Coupled with this poor information, there were conservation concerns related to the potential for localized depletion and high bycatch mortality. Due to the infeasibility of stock assessment surveys, observer coverage on vessels was used to gain information on the distribution of fishery catch and effort and on the life history of the species. We summarize the history of the *C. tanneri* fishery in Alaska, the uses of observer data to address information needs, and the management measures developed to address conservation concerns for a commercial fishery on these unsurveyed stocks in the Bering Sea, Aleutian Islands, and Gulf of Alaska.

Introduction

Directed commercial fisheries for the grooved Tanner crab, *Chionoecetes tanneri*, in Alaska have been prosecuted since only the late 1980s, with most of the effort and catch having occurred in the Alaska's Westward Region and only limited effort and catch in Alaska's Southeast Region. The data and analyses presented here are from the Westward Region, which

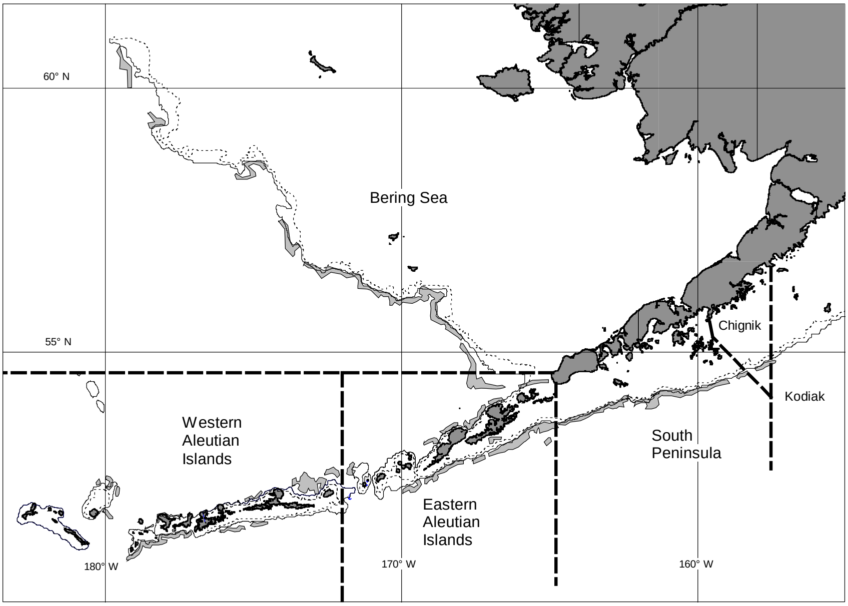

Figure 1. Commercial Tanner crab fishery management districts in the Westward Region, Alaska, with distribution of commercial fishery effort for *Chionoecetes tanneri* during 1994-1996 (shaded regions, offshore) as estimated from locations of all pot lifts randomly sampled by fishery observers. The 350 m depth contour is represented by the dotted line and the solid line represents the 700 m depth contour.

is divided into six districts for Tanner crab fishery management: Kodiak, Chignik, South Peninsula, Eastern Aleutians, Western Aleutians, and the Bering Sea (Fig. 1).

The *C. tanneri* fishery is a males-only fishery prosecuted using pots (traps) fished either singly or in a longline array. Prior to 1988, *C. tanneri* were landed only occasionally in the Westward Region as incidental catch in the golden king crab *Lithodes aequispina* fisheries of the Aleutians (ADFG 2003). A special-permit fishery for *C. tanneri* was established in 1988 and in that year two vessels obtained permits to fish for *C. tanneri* in the Bering Sea. No commercial landings for *C. tanneri* were reported from the Westward Region from 1989 through 1991 and in 1992 only two vessels landed *C. tanneri* as incidental catch in the golden king crab fishery. In 1993, however, seven vessels directed effort on *C. tanneri* in the Westward Region and reported landings from the Bering Sea and

Eastern Aleutian districts. Fishery effort, participation, harvest, and value increased steadily from 1993 through 1995, when 15 vessels harvested 1,300 t worth $4.4 million from the Bering Sea, Eastern Aleutians, Western Aleutians, and South Peninsula districts. Landings declined in 1996 to 360 t worth $0.7 million, harvested by nine vessels. The decrease in harvest between 1995 and 1996 was most notable in the Bering Sea and Eastern Aleutian districts, where 1996 harvests were roughly 10% of the 1995 level. Fishery performance in the Bering Sea and Eastern Aleutian districts had declined steadily from 1994 through 1996, with catch per unit effort (CPUE; measured as number of legal crabs per pot lift) annually falling to one-half of the previous year's value.

Throughout the period 1988-1996 the *C. tanneri* fishery in the Westward Region was managed as a special-permit fishery that allowed for exploration and development of the fishery without any target or predetermined upper limit to the annual harvest and without the benefit of a stock assessment survey. The special-permit fishery regulations allowed ADFG to restrict depth fished, establish season dates, establish areas of operation by statistical areas or district, establish minimum size for retained crabs, require presence of an onboard observer, and require logbook reporting of operations as conditions of the permit. However, given the low effort during 1988-1992, no restrictions other than area or district registration were placed on the permits until 1993, when a minimum size limit of 5 inch (127 mm) carapace width (including spines), corresponding with the size preferences for marketing, was established in issued permits.

In 1994, ADFG began requiring 100% onboard observer presence on vessels fishing for *C. tanneri*. Mandatory observer coverage was intended not only to better monitor fishing activities and practices (e.g., fishing locations, depths fished, and catch rates), but also to provide a means to establish a baseline of biological data on *C. tanneri* in the Westward Region. With the exception of a report on data collected from the Bering Sea in summer 1982 (Somerton and Donaldson 1996), most information on the biology of *C. tanneri* available in the early 1990s was collected from British Columbia (Jamieson 1990, Jamieson et al. 1990) or Oregon (Pereyra 1966, Tester and Carey 1986). Those sources provided some information about distribution by depth and season, size at maturity, growth per molt, fecundity, and the season of reproduction and molt. Existing information was often inconsistent among regions or uncertain, however, indicating the need to obtain information from each area fished in the Westward Region.

Chionoecetes tanneri typically occur at depths greater than 300 m and to depths in excess of 1,000 m (Pereyra 1966, Jamieson et al. 1990, Somerton and Donaldson 1996). That depth range in waters off Alaska represents only a narrow band of ocean floor along the steep continental slope. Hence there were concerns that serial localized depletion could occur before the effects became evident. In addition, there were concerns about the mortality of any females and undersized males captured at depth and discarded on the surface during the fishery.

By 1995 the *C. tanneri* fishery had developed to a level that its management as a developing or exploratory fishery needed to be reassessed. Although the need for fishery-independent data collection projects (e.g., stock assessment surveys) was clearly desirable, such projects were not practical or feasible. Instead, fishery data, particularly data collected by onboard observers during 1994-1996, were examined for information that could be used to identify conservation and management concerns and to establish practices or regulations that addressed such concerns. The data were investigated to determine trends in distribution of catch and effort, characteristics and levels of bycatch, biological seasons, and characteristics of male size distribution. Also, special data collection projects for onboard observers and cooperative data gathering with vessel operators, including deployments of ADFG biologists on fishing vessels, were initiated.

We reviewed results from the analyses of fishery data from the *C. tanneri* fishery through 1996 and their application toward management of the Westward Region *C. tanneri* fisheries after 1996.

Methods

As with all Westward Region crab fisheries, total landed catch, weight of dead-loss, average weight of delivered crabs, number of delivered crabs, and catch and effort by statistical reporting area were reported on a “fish ticket” (a bill of sale) from each commercial *C. tanneri* landing, with a copy delivered to ADFG.

During the 1994-1996 fishery seasons, onboard observers randomly sampled a predetermined number of pot lifts daily, with the number of pots sampled depending on season and district (Tracy 1995; Boyle et al. 1996, 1997). The date, location, depth, gear type (e.g., conical or rectangular pot, single or longlined), and soak time were recorded from each sampled pot. Composition of catch and bycatch within sampled pots was recorded as counts by species and, for commercial crab species, by sex and legal status. Measurements of carapace width (CW, excluding spines; Jadamec et al. 1999) in mm, sex, and shell condition of all *C. tanneri* and reproductive condition of all female *C. tanneri* in sampled pots were recorded when possible. Note that, due to the exclusion of spines in the measurement, 120 mm CW as measured by observers corresponds with the minimum size limit of 5 inch (127 mm) carapace width including spines (Tracy 1998). During 1994-1996, observers in Westward Region *C. tanneri* fisheries sampled nearly 16,000 individual pot lifts, counted 400,000 *C. tanneri*, and recorded data from 300,000 individual *C. tanneri*.

In addition, observers sampled 100 crabs from each landing by vessels at a shoreside or floating processor for size and shell-age frequency distribution and interviewed skippers to obtain information on daily effort and catch of legal-sized males by statistical area and gear type. Sampling of deliveries during 1994-1996 in the Westward Region resulted in size and shell-age data on nearly 30,000 delivered crabs.

Special data collection by observers included collection of morphometric data from males when possible. Chela height (Jadamec et al. 1999) was measured to the nearest 0.1 mm along with CW in subsamples of male *C. tanneri*, which allowed for classifying those males as either morphometrically mature or immature on the basis of the size of the chela relative to CW (Somerton 1980). Measurements of length (L; measured from the anterior of the rostrum to the ventral-posterior margin of the second abdominal somite), CW, and carapace width including spines in mm were also collected.

Results and discussion

Effort distribution and depth

Although there were no seasonal restrictions on fishing for *C. tanneri*, there was no effort during the months of January or February during 1994-1996. Initially, the fishery was prosecuted during late spring to late summer, periods when larger, more established Bering Sea crab fisheries were closed. In 1995, however, the fishery was prosecuted from March through December.

Data on pot lifts sampled during 1994-1996 by observers showed that the fishers typically set pots for *C. tanneri* at depths of 500 to 950 m (Fig. 2). Fishing effort was concentrated at depths of 700-800 m; median depth of sampled pots was 720 m in the Western Aleutians, 760 m in the Eastern Aleutians, 730 m in the Bering Sea, and 720 m in the South Peninsula. The concentration of fishing effort by depth corresponded well with the average depth of mature males reported from the Bering Sea (752 m; Somerton and Donaldson 1996), but was deeper than reported for commercial concentrations in British Columbia (580-670 m; Jamieson et al. 1990) and the typical depth for adult males in British Columbia (550-650 m; Workman et al. 2002) and in Oregon (500-685 m; Pereyra 1966). Plots of locations of sampled pots (Fig. 1) showed that fishing effort during 1994-1996 extended through a broad range of the available depth zones for commercial fishing, particularly in the South Peninsula district and in the Bering Sea district south of 57ºN.

Fishing effort by depth generally coincided with the highest catch per pot lift (CPUE) of legal-sized male *C. tanneri*. In those fisheries in which regulations allowed for fishing with longlined pots, which allows for fishing at greater depths, observer data on effort and catch relative to depth suggests that depth zones with commercial concentrations

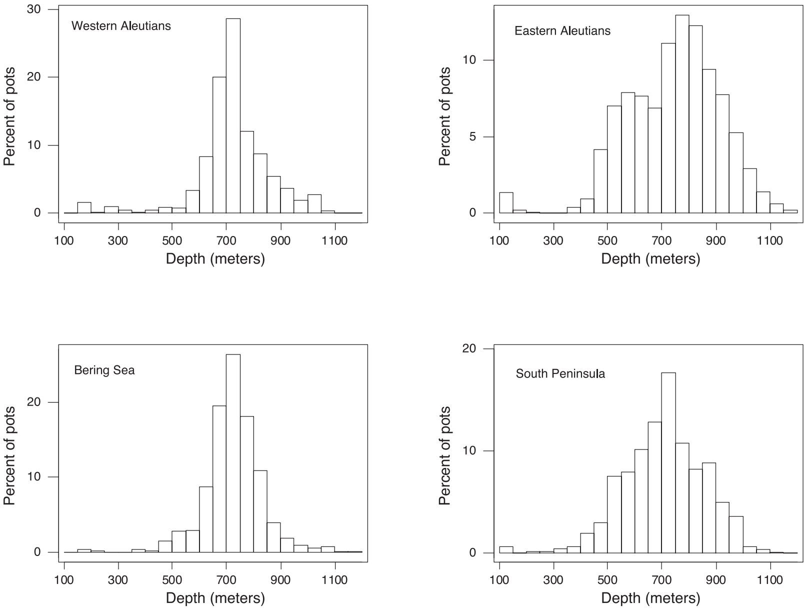

Figure 2. Depth fished by pots randomly sampled by fishery observers from the commercial fishery for *Chionoecetes tanneri* in four Westward Region, Alaska, management districts during 1994-1996.

were fully explored by the fishery. For example, in the Bering Sea 1995 fishery season, highest CPUE in sampled pots tended to be those fished at depths of 500-900 m, where effort was highest, and CPUE decreased markedly at depths greater than 900 m, where effort was low. On the other hand, longlining of pots was not permitted in the South Peninsula district until 1996 and observer data on effort and CPUE by depth in the South Peninsula fishery prior to 1996 suggested that depth zones with commercial concentrations of legal males were not fully exploited. For example, there was little fishing effort at depths greater than 800 m during the 1995 South Peninsula season although CPUE of legal-sized males remained high at depths of 800-900 m.

Changes in distribution of effort within districts during 1994-1996 cannot be detailed here because of the confidential nature of the fishery data. Certain core areas within districts were fished in each successive year. However, in each successive year, previously unfished areas within the preferred depth zone were exploited. Within districts the area fished expanded from 1994 to 1995 and, except for the South Peninsula district, contracted from 1995 to 1996. Depths fished in the Bering Sea district and

Western Aleutians district remained relatively stable over 1994-1996, with the median depth of fished pots for each year varying by less than 50 m, whereas in the Eastern Aleutians and South Peninsula districts there was a tendency toward fishing deeper after 1994.

Hence by 1996 the depth zone for commercial fishing of *C. tanneri* was well established and it was clear that the commercial fishery had explored the majority of available fishing grounds. Trends in changes in distribution of fishery effort during 1994-1996 raised concerns for managers. The contraction of effort seen from 1995 to 1996 in some districts, coupled with a marked decrease in landings, could be interpreted as indicating that areas explored in 1995 did not support sufficient commercial concentrations relative to the core areas that continued to be fished. It was unclear whether such core areas, which represented only a small portion of the identified fishing grounds, could sustain a fishery into the future. Those concerns were particularly noted for the Bering Sea district. In the South Peninsula district, on the other hand, the geographic range of fishing effort had increased from 1995 to 1996 and the reduction in harvest from 1995 to 1996 was not as marked as in other districts. However, it was troubling to managers that fishery performance in the South Peninsula was maintained by the increase in fishing depth afforded by a 1996 change in regulations that allowed for longlining of pots.

Bycatch

Chionoecetes tanneri composed nearly 90% of the commercially exploited crab species occurring in sampled pot lifts from the directed fishery for *C. tanneri* during 1994-1996 (Tracy 1995; Boyle et al. 1996, 1997). Of all the 409,530 *C. tanneri* observed in sampled pot lifts during 1994-1996, only one-half were retained legal-sized males; females accounted for 28% of captured *C. tanneri* and undersized males accounted for 22%. Hence the greatest concern relative to bycatch in the *C. tanneri* fishery was the possible effects to the female and sublegal male *C. tanneri* that were captured and discarded. The depths and steep slope at which the bycatch was captured made it likely that discards would be significantly displaced from the area of capture after settling to the bottom. Anecdotal observations by fishers and observers that discarded *C. tanneri* can float for prolonged periods at or just below surface water unless “burped” before discarding (Byrne and Cross 1997) increased displacement concerns and raised the additional concern of potentially high mortality in the discards. Other observations indicated that these deep-water Tanner crabs may be more fragile than the more shallow-water Tanner crab, *C. bairdi*, and snow crab, *C. opilio*. Dead-loss in the Bering Sea *C. tanneri* fishery, for example, accounted for 7-11% of the annual delivered harvest as compared to accounting for roughly 2% of the annual harvest in the Bering Sea Tanner and snow crab fisheries (ADFG 2003). Jamieson et al. (1990) noted problems in survival of retained crabs in the British Columbia fishery during the summer months, when surface water temperatures exceeded 10°C.

Data collected by at-sea observers showed that female *C. tanneri* occurring as bycatch were almost exclusively mature as evidenced by the presence of egg clutches or egg-case remnants. During the 1994-1996 fishery seasons 99% of sampled females were mature in the Eastern Aleutians, Western Aleutians, and South Peninsula fisheries and 94% were mature in the Bering Sea fishery (Boyle et al. 1997). Egg-bearing females constituted 95% or more of the mature females in the bycatch by district and year.

Prior to initiation of observer coverage, data on morphometric maturity status of males was first collected from the 1993 Bering Sea fishery through cooperation with a fisher. Eighty-one percent of 100 legal-sized males (117-176 mm CW) were morphometrically mature, whereas 40% of 126 incidentally captured sublegal males (82-127 mm CW) were morphometrically mature. The largest morphologically immature male was 146 mm CW and the smallest morphologically mature male was 83 mm CW.

During the 1994 through 1996 fishery seasons, observers were instructed to obtain measurements of chela height in addition to carapace width of males in sampled pots when possible. A total of 3,045 males were measured for morphometric maturity from the Eastern Aleutians, Bering Sea, South Peninsula, and Western Aleutians during 1994-1996. The results on male morphometric maturity from that sampling were too inconsistent among years and areas to characterize the proportion of captured sublegal or legal-sized males that were morphologically mature or immature. Coupled with the data collected in 1993, it could be concluded that morphologically mature males were captured as sublegal-sized bycatch at sizes down to less than 90 mm CW and that morphologically immature males were captured as legal-sized retained catch up to sizes in excess of 140 mm CW.

Jamieson et al. (1990) reported a trend of depth stratification by sex and size in the British Columbia stock of *C. tanneri* with an apparent vertical migration that resulted in a higher association of males and females in March and April. However, depth zones, geographic areas, or seasons of high bycatch could not be identified in the observer data from the Alaska fishery during 1994-1996. During the 1995 Bering Sea fishery, for example, females occurred most commonly at the depths most frequently fished for legal-sized males. Eighty-seven percent of the pot lifts sampled by observers during the 1995 Bering Sea fishery were fished at depths of 600-900 m and 31% of those contained at least one female (Fig. 3). That compares to only 8% of pots sampled during that fishery from depths <500 m or >900 m, of which only 5% contained at least one female. Of the pot lifts sampled during the 1995 Bering Sea fishery that contained at least one female, the median female catch per pot lift was 3. That value was constant over the 100 m depth zones between 600 m

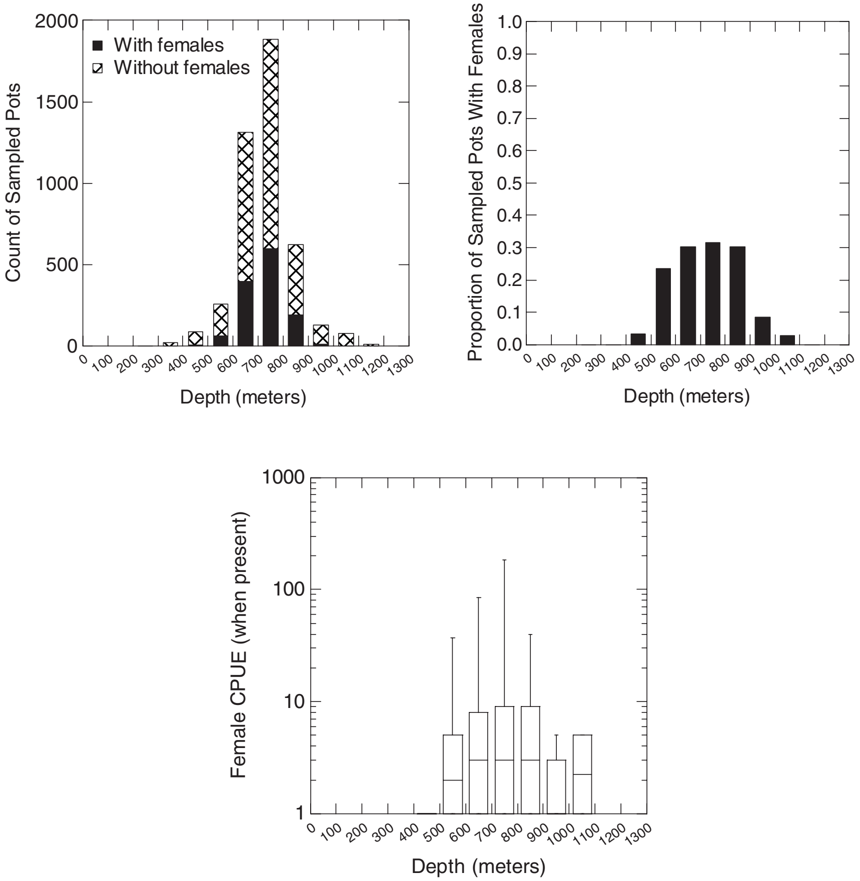

Figure 3. Bycatch of female grooved Tanner crabs relative to depth fished during the 1995 commercial Bering Sea grooved Tanner crab fishery as estimated from 4,409 pot lifts randomly sampled by observers. Top panel: distribution of depth for pot lifts sampled during the fishery with females present and without females present. Middle panel: proportion of pot lifts with females present by depth zone. Bottom panel: box plots of catch per pot lift (CPUE, log scale) of females by depth zone for those pot lifts with at least one female present (1,264 pot lifts).

and 900 m and was not exceeded in any shallower or deeper depth zone (Fig. 3). High catches of females occurred unpredictably throughout the fished depth zone. For example, in the observer data from the 1995 Bering Sea fishery, pots containing from 15 to 183 females occurred in only 4% of the sampled pots, but those catches accounted for 50% of the total females counted as bycatch. Likewise, monthly summaries of incidence and bycatch rates of females in the observer-sampled pots showed no consistent trend among fishery seasons or districts.

Biological seasons

Incidence of soft-shelled (recently molted) *C. tanneri* was well below 1% of the males and females sampled by observers for each district and fishery season (Boyle et al. 1997) and was too low to determine any seasonal pattern. In regard to determination of a molting season, one of the most experienced of the commercial *C. tanneri* fishers reported that he saw no evidence of any definable molting season. In his experience, recently molted crabs could occur occasionally at any time of the year at unpredictable locations; when encountered, recently molted crabs could be avoided by moving the fishing gear. Overall, the low incidence of soft-shell crab indicated no need to control the fishery's capture of fragile, soft-shelled crab.

Stage (eyed or un-eyed) of eggs carried by mature females was examined for indication of a predominant hatching season. Highest incidence of eyed eggs in the observer data tended to occur in March and April, similar to the findings reported for *C. tanneri* off British Columbia (Jamieson et al. 1990), indicating that hatching occurs predominately in the spring. Those results are consistent with a strongly seasonal hatch timing reported for *C. tanneri* in Oregon (Pereyra 1966) and suggested for *C. tanneri* in British Columbia (Jamieson et al. 1990) and the Bering Sea (Somerton and Donaldson 1996). Nonetheless, incidence of females with eyed eggs in the observer samples was not confined to the spring and, depending on district and year, could occur in the observer samples during any month that sampling occurred. Given the occasional contradictory data and a lack of any data from January and February, we could not draw any strong conclusions on hatching season from the observer data.

Size distribution of males and estimates of growth per molt

Observers obtained carapace width measurements from over 200,000 male *C. tanneri* that were encountered in sampled pots in the Eastern Aleutians, Bering Sea, and South Peninsula districts during 1994-1996. The presence of large males in the catch was maintained in each district from 1994 through 1996, although there was some reduction in the size of males captured in the Eastern Aleutians district during that period

Table 1. Mean and percentiles of distribution of carapace width (mm; excluding spines) of male *Chionoecetes tanneri* in randomly sampled pot lifts during the Eastern Aleutians, Bering Sea, and South Peninsula fisheries in Alaska, during 1994-1996.

Sample	*N*	Mean	5%	25%	50%	75%	95%
E. Aleutians, 1994	16,056	133.1	112	123	135	142	151
E. Aleutians, 1995	41,423	129.4	109	118	130	140	152
E. Aleutians, 1996	1,950	127.4	108	116	125	137	154
Bering Sea, 1994	5,747	135.9	107	127	136	146	162
Bering Sea, 1995	39,528	142.0	113	132	143	154	165
Bering Sea, 1996	3,600	141.1	114	130	140	154	165
S. Peninsula, 1994	9,366	124.4	111	117	122	131	142
S. Peninsula, 1995	66,732	125.6	110	118	124	133	143
S. Peninsula, 1996	40,730	126.9	110	118	126	136	146

(Table 1). Over 30,000 retained legal males were also sampled from deliveries during 1994-1996 and those samples indicated a decrease in size of delivered males from 1994 through 1996 only within the Eastern Aleutians district (Boyle et al. 1997). On the other hand, average weights of delivered crabs within all districts were maintained or increased slightly from 1994 to 1996 (ADFG 2002). Hence, although conclusions could not be drawn without fishery-independent data, there was no evidence from the fishery that the fishery had substantially impacted the male size distribution by the end of 1996.

Annual size-frequency distributions for males in sampled pots by district and year indicated the presence of two modes that we assumed represented molt instars. Those two modes were most evident in the size frequencies for samples obtained from the Eastern Aleutians and South Peninsula districts, both of which appeared to be composed largely of only two components. The size distribution for males captured in the Bering Sea fishery was broader and with a larger average size (Table 1), indicating the presence of at least one additional, larger molt instar. The annual size frequency distributions for the males sampled from the Eastern Aleutians and South Peninsula were fit as a mixture of two normally distributed components using the expectation-maximization (EM) algorithm (Titterington et al. 1985). The two estimated components, with means of approximately 119 mm CW and 138 mm CW (Table 2) provided good fits to the observed size frequencies (Fig. 4). Assuming that these two components represent molt instars, the difference between the means, 19 mm CW, provides an estimate of the mean growth-per-molt between instars. The mean of the smaller presumed molt instar (119

Table 2. Estimated normal distribution parameters (mean, SD, and percent contribution, P) for a two-component mixture distribution fit to the carapace width (mm, excluding spines) frequency distribution of male *Chionoecetes tanneri* from randomly sampled pot lifts during the Eastern Aleutians, Bering Sea, and South Peninsula fisheries in Alaska, 1994-1996.

		Component 1			Component 2			
Sample	*N*	P_1	$Mean_1$	SD_1	P_2	$Mean_2$	SD_2	$Mean_2 - Mean_1$
E. Aleutians, 1994	16,056	28%	117.8	6.02	72%	139.1	8.48	21.3
E. Aleutians, 1995	41,423	35%	115.8	6.02	65%	136.8	10.68	21.1
E. Aleutians, average	–	–	116.8	–	–	138.0	–	21.2
S. Peninsula, 1994	9,366	71%	119.5	6.22	29%	136.4	6.48	16.9
S. Peninsula, 1995	66,732	76%	121.5	7.68	24%	139.0	6.19	17.5
S. Peninsula, 1996	40,730	57%	119.3	7.05	43%	137.2	7.59	17.9
S. Peninsula, average	–	–	120.1	–	–	137.5	–	17.4
Average, all			118.8			137.7		18.9

mm CW) corresponds closely with the minimum size limit established in 1993. Additionally, the mean of 119 mm CW corresponds with the size at 50% morphometric maturity estimated for male *C. tanneri* in the Bering Sea (119 mm CW; Somerton and Donaldson 1996), Oregon (118 mm CW; Tester and Carey 1986), and British Columbia (112 mm CW; Workman et al. 2002).

Establishment of guideline harvest levels for districts and bycatch controls

By the end of 1996 ADFG fishery managers concluded that management measures for the *C. tanneri* fishery in the Westward Region were needed to control removals and to minimize bycatch mortality. The distribution of the stock was limited and had been well-explored geographically and by depth. Concerns were particularly acute for the Eastern Aleutians and Bering Sea districts, where district-wide CPUE had decreased by roughly one-half from 1994 to 1995 and again from 1995 to 1996 (ADFG 2003). Smaller scale investigation of fishery performance using observer pot sampling data indicated that stock depletion had likely occurred in some of the localized areas that had first supported the fishery. Although there were no estimates of bycatch mortality, management biologists believed it was prudent to assume that survival of discarded bycatch *C. tanneri* was very low.

Lack of fishery-independent stock assessment data posed problems for establishing harvest controls. Various measures to control harvests

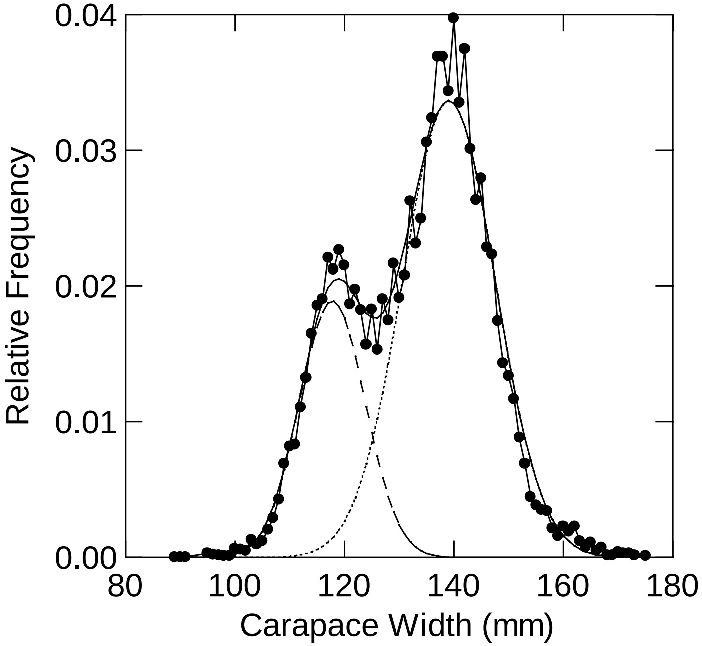

Figure 4. Observed (solid line with solid circles) carapace width (mm) frequency distribution for 16,056 male *Chionoecetes tanneri* in pot lifts randomly sampled by observers during the 1994 Eastern Aleutian, Alaska, commercial fishery compared with a mixture distribution fit to the data (solid line). The mixture distribution is the weighted sum of two normally distributed components: one with a weighting of 28%, a mean of 118 mm, and standard deviation of 6.0 mm (dashed line); the other with a weighting of 72%, a mean of 139 mm, and standard deviation of 8.5 mm (dotted line).

without defining a harvest limit were explored. Among the approaches considered were limiting fishing to only designated areas and setting the rest aside as refuges; subdividing districts and opening subareas rotationally; and limiting the harvest with a “slot size limit” by use of gear requirements to allow the escape of small crabs from pots and prohibit the entry of large crabs into pots. Harvest controls based on closing subareas, either permanently or in rotation, were rejected by management biologists due to uncertainties related to effectively subdividing districts. Likewise, management of the fishery using a slot size limit was rejected because bycatch-reducing gear modifications had not yet been tested. At the suggestion from participants at an annual interagency meeting on

Alaska crab research held late in 1996, it was determined that the most effective way to control harvest was to establish harvest limits based on historic harvest levels.

Determination of guideline harvest levels (GHLs) for districts was based on a "Tier 6" methodology, which was developed by the National Marine Fisheries Service (NMFS) to estimate the acceptable biological catch (ABC) for groundfish stocks where the only reliable information on stock size is catch history (NPFMC 1996). The ABC for a Tier 6 fishery is calculated by multiplying the historic average annual catch for a given groundfish species by a percentage less than or equal to 75%. To determine GHLs for the *C. tanneri* fishery, only the years when the fishery was fully developed were used to compute historic average harvests: 1995-1996 for the South Peninsula Area, 1994-1996 for the Eastern Aleutians Area, and 1993-1996 for the Bering Sea Area. Also it was determined that GHL would be based on the historic average annual harvest of males greater than or equal to one molt above the size of morphometric maturity. The intention here was to better maintain the size and age distribution of mature males by controlling the fishery removals of the larger males within the class of mature-sized males. That determination was also consistent with the rationale for the size limits established in other Alaska commercial Tanner crab fisheries, which typically have been set at one average molt increment above the estimated size of maturity (Donaldson and Donaldson 1992).

Given the estimated 50% size at morphometric maturity estimated for Bering Sea male *C. tanneri* (119 mm CW; Somerton and Donaldson 1996) and the estimated average growth per molt of 19 mm CW that was estimated from the size frequency data collected by observers, males ≥138 mm CW were considered to be one molt above the size at 50% morphometric maturity. The harvest in weight of males ≥138 mm CW was estimated from the size distribution of retained crabs sampled by observers coupled with the allometric CW-to-weight equation estimated for *C. tanneri* by Somerton and Donaldson (1996). Depending on year, males ≥138 mm CW were estimated to account for 73%-84% of the annual harvest from the Bering Sea during 1993-1996, 50%-72% of the annual harvest from the Eastern Aleutians during 1994-1996, and 35%-47% of the annual harvest from the South Peninsula during 1995-1996. Average annual harvests of males ≥138 mm CW were estimated at 185 t in the Bering Sea for 1993-1996, 176 t for the Eastern Aleutians for 1994-1996, and 121 t for the South Peninsula for 1995-1996.

Due to the decrease in harvests and CPUE that occurred through 1996, the GHLs for the Bering Sea and Eastern Aleutians districts were computed as 50% of the average annual harvest of males ≥138 mm CW. The GHL for the South Peninsula district was computed as 75% of the average annual harvest of males ≥138 mm CW. The computed GHLs for each of the Bering Sea, Eastern Aleutians, and South Peninsula districts were rounded up to 200,000 pounds (91 t) and announced for the 1997 season in February 1997. For each of the remaining districts that had little history of a *C. tanneri* fishery in the Westward Region, GHLs of 100,000 pounds (45 t) were established. The GHLs for each district were intended to serve as not-to-exceed catch limits.

Although the GHLs were established to limit the harvest of males ≥138 mm CW, the 5 inch CW (including spines) minimum size limit was retained. Males 120-137 mm CW constituted a large portion of the presorted and retained catch in the fishery and were marketable. Given the concerns for survival of discards and the possible handling effects due to sorting of the catch, it was determined that allowing retention of males 120-137 mm CW and applying their retention toward the GHL would reduce unnecessary bycatch mortality.

Requirements for the use of 4.5 inch (114 mm) escape rings were also established for the 1997 fishery season. The escape rings were intended to allow for the escape of captured females and sublegal males before fished pots were lifted. Because crabs will pass through the ring laterally, the inside diameter of the escape ring was determined from the relationship for male crabs between L and CW. Observers obtained measurements of L and CW during the 1996 fishery season from 320 males ranging in size from 91 mm CW to 172 mm CW. The two dimensions had a strong linear correlation ($r = 0.99$), allowing for a reliable prediction of L from CW by linear regression ($L = 6.64 + 0.93 \times CW$). The predicted L from CW = 119 mm is 117 mm (4.6 inches); the minimum size of required escape rings was established at 4.5 inches due to the pre-existing availability of 4.5 inch rings.

The requirement for 100% observer coverage on vessels fishing for *C. tanneri* was continued with the goal of gathering information on effectiveness of the required escape mechanisms, as well as to continue tracking distribution of effort, catch, and bycatch and gathering basic biological information.

The post-1996 fishery

There were no landings of *C. tanneri* in the three years following the establishment of GHLs, 1997-1999 (ADFG 2002). In 1999, the GHLs established in 1997 were used to define the upper limits of harvest ranges to clarify their intent as not-to-exceed harvest limits. In the subsequent three years, 2000-2002, participation in the Westward Region was low and sporadic, with only 1 to 3 vessels landing *C. tanneri* annually. Due to the reliance on fishery observers to obtain information for fishery management, the dearth of fishery effort and landings during 1997-2002 resulted in the addition of little new information.

The abrupt drop in fishery participation following the establishment of GHLs in 1997 coincided with a declining market value of *C. tanneri*. Market value and participation in the Alaska *C. tanneri* fishery both began decreasing between 1995 and 1996 and the cessation of effort during 1997-1999 reflects, at least partially, a continuation of that trend. Nonetheless it is possible that the established GHLs themselves, which were less than one-quarter of the 1995 harvests from each of the Eastern Aleutians, Bering Sea, and South Peninsula districts, also dampened the enthusiasm of potential fishery participants.

Conclusions

In 1999, two models for approaching the development of new, data-limited fisheries were published. One (Miller 1999) advocated an approach of first allowing commercial fishing with minimal interference, while deferring the collection of biological data to later stages of fishery development. The other (Perry et al. 1999) stressed the importance of identifying and collecting needed biological, distribution, or other data before fishing for commerce commences. Development of the *C. tanneri* fishery off British Columbia has largely followed the second, phased approach (Workman et al. 2002). Development of the *C. tanneri* fishery off Alaska followed an approach intermediate to the two; the effort was made to identify and collect needed information by deploying observers in the fishery during the largely unfettered developmental stage of that fishery. That approach was seen as a best-possible approach, given the charge to develop new fisheries, the managers' lack of authority to limit participation in the fishery, and the lack of resources needed to gather preliminary fishery-independent data.

The success of that approach was mixed. From 1994 through 1996 observers collected a large body of data on *C. tanneri* and fishery activities from the Alaska fishery. These data provided fishery managers with detailed information on the effort and performance of the fishery and on the characteristics of the catch and bycatch, as well as greatly expanding the data on biological characteristics of *C. tanneri* in Alaska. The data allowed managers to address concerns of stock depletion and bycatch. However, lacking fishery-independent information on pre- and post-fishing population distribution and conditions, management measures were still established in an atmosphere of great uncertainty.

The commercial fishery was not the ideal source for gathering basic biological information on Alaska stocks of *C. tanneri*. The data were limited by being drawn from only those locations and periods that fishers chose to fish and from only those crabs that recruited to the variety of fishing gears that were used in the fishery. Moreover, lacking a predetermined spatial and temporal design for collecting data from the stock, collecting data from the fishery is inherently inefficient relative to a designed survey. Collecting data from the fishery necessitated an approach where observers sampled from the catch as much as possible on a daily basis throughout the course of the fishery in an attempt to provide the greatest possible spatial and temporal distribution of sampling.

Despite the attempt to gain as much information from the commercial fishery as possible, the sampling effort still may not have been sufficient to estimate certain biological characteristics of interest. The difficulty in determining depth and seasonal trends in the catch of females, for example, may have been due as much to the use of the commercial fishery as a platform for data collection as it was to the actual distribution of females relative to the targeted legal-sized males. As well as limitations on depth zones and months of the year from which data was obtained, there were undoubtedly effects on catch rates due to individual vessels that could not be estimated or accounted for. Likely "vessel effects" on catch and bycatch rates include the gear and fishing practices that allowed some vessels to more effectively avoid bycatch than others. With different vessels targeting different areas or fishing during different periods, such effects due to individual vessels in the observer data may have obscured the true associations of female density with season, depth, or male density.

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Recreational Fisheries in Northeastern Brazil: Inferences from Data Provided by Anglers

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Abstract

Brazilian recreational fisheries became increasingly important after the 1990s. A plan to develop this segment of the fishery sector was launched in 1997, in spite of the lack of basic information, e.g., number of anglers, fishing effort, or total catch. This paper aims to fill this information gap by presenting a historical overview of Brazilian recreational fisheries. It also describes the profile of anglers from northeastern Brazil, a region dominated by onshore angling; estimates annual catch for this region; and analyzes changes in fish size through time. Anglers from northeastern Brazil are mainly non-licensed, consumption-oriented men who fish in both salt and freshwater bodies and participate in fishing tournaments, but normally do not travel great distances to fish. The total catch estimated from recreational fisheries off northeastern Brazil in 2001 was about 1,147 t, representing only 0.8% of commercial catches. This proportion may be between 0.2 to 4%, if uncertainty in the input parameters is considered. The main species or group of species caught were Ariidae, *Polydactylus* spp., rays (unidentified), *Caranx* spp., *Trachinotus* spp., *Menticirrhus americanus*, puffers (unidentified), *Centropomus* spp., *Scomberomorus cavalla*, *Scomberomorus brasiliensis*, sharks (unidentified), *Cynoscion* spp., *Paralonchurus brasiliensis*, *Megalops atlanticus*, *Eucinostomus gula*, *Coryphaena hippurus*, *Lutjanus analis*, *Epinephelus* spp., *Ocyurus chrysurus*, *Istiophorus albicans*, *Thunnus atlanticus*, *Thunnus obesus, Thunnus alalunga*, and *Sphyraena barracuda*. The mean size of fish caught in tournaments has been decreasing since the 1970s, but it has not been possible to separate the effect of recreational fisheries from that of commercial fisheries. The information presented in this paper will

assist in the integration of recreational fisheries into management of fish resources in this region.

Introduction

Recreational fisheries are well-recognized worldwide as an industry, dominated by the United States, some European countries, and Canada, and associated with expenditures of US$5-38 billion (Cowx 2002) . Although no precise global catch data for these fisheries are available, Coates (1995) gave an initial estimate of 2 million tons. Compiling such information is a difficult task, as some countries, including Brazil, do not have the mandate to gather this basic information for management of recreational fisheries. Other necessary information is the understanding of fishers' behavior, here referred to as "anglers." Some previous studies have shown the importance of understanding the heterogeneity within this sector in terms of consumptive habits of anglers and other motivation factors, and how these are translated into behavioral choices (Fedler and Ditton 1986, 1994; Matlock et al. 1988).

Though recreational fishing has been practiced in Brazil for a long time, its potential importance as an industry has become evident only since the 1990s (Mélega 1998). However, very little is known about this segment of the fishery sector along the Brazilian coastline (Dias Neto and Dornelles 1996). There is a possibility of obtaining higher economic benefits from this activity when compared to commercial fisheries, but there is still a lack of local studies dealing with valuation of recreational fisheries, except for Shrestha et al. (2002).

The main Brazilian recreational fisheries are found in the Amazonian and Pantanal regions (freshwater) and in southeastern Brazil (oceanic) (Fig. 1). Oceanic recreational fishing began in Rio de Janeiro (Barroso 2002), and has gradually extended toward the north, reaching the states of Espírito Santo, Bahia, Pernambuco, and Rio Grande do Norte. The latter three states are part of the region known as "northeast," which has comparatively lower economic power than the "south" (Goldsmith and Wilson 1991). This may be one reason that such oceanic fisheries, which demand higher costs to catch sailfish (*Istiophorus albicans*), blue marlin (*Makaira nigricans*), white marlin (*Tetrapturus albidus*), dolphinfish (*Coryphaena hippurus*), wahoo (*Acanthocybium solandri*), and yellowfin tuna (*Thunnus albacares*), are not well established in northeastern Brazil.

Onshore fisheries are much more developed in this region and are mainly structured around fishing clubs. However, much less is known about these fisheries (Schultz 2000), predominantly in relation to the possible conflicts between recreational and commercial fisheries. The objective of this paper is to present a historical overview of Brazilian recreational fisheries, to describe the profile of anglers from northeastern Brazil, to provide an initial estimate of annual catch from recreational

Figure 1. Bar graphs showing the number of fishing licenses issued in 2000, 2001, and 2002 for the four Brazilian states with the largest number of licenses (SP = São Paulo, MS = Mato Grosso do Sul, MG = Minas Gerais, and GO = Goiás). The tallest bar corresponds to 28,277 licenses sold in São Paulo in 2002. The number of licenses issued in northeastern Brazil is shown inside the circle (2,648 in 2002). MA = Maranhão, PI = Piaui, CE = Ceará, RN = Rio Grande do Norte, PB = Paraíba, PE = Pernambuco, AL = Alagoas, SE = Sergipe, BA = Bahia, ES = Espírito Santo, RJ = Rio de Janeiro, and RS = Rio Grande do Sul.

fisheries of that region and compare it with commercial catch, and to analyze changes in fish size through time. This information is essential to successfully integrate recreational fisheries into management of fish resources, as advocated by Sutinen and Johnston (2003).

Historical background of Brazilian recreational fisheries

Evolution of fishing clubs and tournaments

The first recreational fishing club in Brazil, "Sociedade Esportiva Aérea do Salto de Avanhadava," was established in 1923 in the southeastern region, and its members were mainly freshwater anglers (Revista Nacional da Pesca 1971). The first Brazilian tournament for which documentation is available occurred in 1948, also in southeastern Brazilian freshwater (Anuário da Pesca 1970). The southern region established a federation (FRAP) in 1955. This federation promoted the fifth South American fishing tournament and the first international fishing tournament of Rio Grande do Sul, both in freshwater bodies (Azevedo 1963, Rocha 1999). While the first sailfish was caught in Brazilian waters in 1950 (Barroso 2002), the first oceanic tournament took place twelve years later (1962-1963) in Rio de Janeiro state. Since then, tournaments have been promoted annually in this state (Paiva and Pires-Júnior 1983, Arfelli et al. 1994).

In northeastern Brazil, the Pâmpano Esporte Clube was the first fishing club established (1954). This club was based in Rio Grande do Norte state and its members were mainly fishing in salt water (Bezerra 1994). Its establishment was probably influenced by the presence of U.S. forces in that state. Since then, 28 other fishing clubs were established in that region. Some of these already have been dissolved, while others are quite active with more than 200 members and promoting monthly fishing tournaments.

Brazilian recreational fisheries have traditionally been organized around fishing clubs and the tournaments organized by those clubs. The Brazilian Fishing and Aquatic Sports Confederation is the national agency responsible for tournaments promoted by "official" clubs (CBPDS 2003). This does not prevent several "non-official" clubs from organizing their own fishing tournaments, creating conflicts that contribute to the weakening of the recreational fishing sector in the country.

Evolution of the fisheries management system

No institution was responsible for managing Brazilian recreational fisheries before 1933; this was probably unnecessary at that time, since this sector had only begun to develop. Since then, the following institutions sequentially took charge of recreational fisheries management:

- 1933-1961: Division of Fish and Game.

- 1962-1988: SUDEPE (National Institute for Fishery Development).
- 1989-present: IBAMA (Brazilian Institute for the Environment and Renewable Resources).
- 1997: PNDPA initiated (National Program for the Development of Recreational Fisheries.

The oldest legal instrument related to Brazilian recreational fisheries is the decree no. 221 of February 1967 (federal regulation). According to this decree, state branches of the now extinct SUDEPE should be responsible for issuing recreational fishing licenses. Data reported for 1977-1981 indicated the existence of 10 to 30 fishing clubs in Brazil during that period, and about 119,000 to 244,000 anglers (SUDEPE 1978, 1979, 1980, 1981, 1984), corresponding to 0.1-0.2% of the Brazilian population in that period (IBGE 2000).

IBAMA assumed the responsibilities for managing recreational fisheries when SUDEPE was dissolved in 1989. Decree no. 1.583 of December 1989 (federal regulation) established the rules for recreational fisheries, including those related to tournaments and registration of fishing clubs or associations. IBAMA regulates this fishing sector through the control of licenses, minimum catch size, and bag limits, although enforcement is rather poor. Fishing licenses cost is about US$9 (R$20) for onshore anglers and US$26 (R$60) for those who fish both onshore and in boats. In 2002, a total of 88,000 licenses were sold country-wide (PNDPA, unpubl. data), but this number is probably far below the actual number of anglers as there is a large number of unlicensed anglers in Brazil (pers. obs.); some of them are not even aware that they are supposed to have a fishing license.

In 1997, the Ministry of Sport and Tourism, through the Brazilian Tourism Enterprise (EMBRATUR), and the Ministry of Environment, through IBAMA, created a National Program for the Development of Recreational Fisheries (PNDPA 2003). The objective of this program was to promote recreational fisheries as a socioeconomic activity, with environmental conservation as one of its most important concerns. More specifically, the program intends to (i) identify new areas for recreational fishing; (ii) establish an industrial infrastructure, which would include hotels, restaurants, and guides; (iii) search for national and international investors interested in developing this sector; (iv) improve regulations, including those pertaining to fishing licenses, quotas, and minimum size; (v) establish and regulate areas reserved only for recreational fishing; and (vi) promote training courses for fishing guides and environmental education (PNDPA 2003). As part of these objectives, the PNDPA issued in 2001 a new license for anglers, including a mini-questionnaire related to fishing methods and areas, frequency of fishing trips, and practice of catch-release. However, the information gathered from this effort is not yet available. The most recent regulation regarding this fishery sector is Federal Regulation no. 30 of May 23, 2003, which defines a creel limit of 15 kg for marine waters plus one fish (any size). This regulation also establishes data types to be gathered through an observer program for fishing tournaments targeting oceanic species.

Methods

A tour to various fishing clubs (official and non-official) and fishing shops in seven out of the nine states of northeastern Brazil was conducted by the author in June-August 2001. The other two states were not visited owing to time constraints. The president or vice-president of each fishing club was interviewed in a semi-structured fashion, i.e., data were gathered on the establishment of the club (year), number of members, fees, and degree of activities (promotion of fishing tournaments), along with additional information.

All available data on tournaments organized by each fishing club were collected:

1. Points obtained by each angler, i.e., a combination of total number and weight of fish caught.
2. Total number and total weight of fish caught by all anglers combined (not categorized by species).
3. Total number and total weight of fish caught by each angler (not categorized by species).
4. Number of fish caught by each angler broken down by species and total weight of all species combined.
5. Weight of the largest fish caught by each angler (not categorized by species).
6. Weight of the largest fish caught identified to species level (i.e., by common name) by each angler.
7. Weight of the largest fish caught during the tournament identified to species level (by its common name).

Because of the heterogeneity in the data, it was difficult to perform analyses across states. However, the type (3) data available for Rio Grande do Norte state from 1975 to 2000 could be used to demonstrate changes in mean weight of the fish caught through time. Mean weight was calculated first dividing the total weight by the total number of fish caught by each angler in each tournament, and then averaging out all individual mean values obtained for all anglers in all tournaments in a specific year.

The analysis of the largest fish was performed when type (5) data was available. The results were analyzed separately for light and heavy tackle. There were only enough data to perform such a separated analysis for the tournaments that took place in Rio Grande do Norte state during 1975-2000.

A total of 1,500 questionnaires were distributed to all members of fishing clubs and to clients of fishing shops in the states visited. These contained 17 questions related to demography (sex, age, and occupation), fishing habits (type of fishery, fishing area, and catch-release practice), anglers' consumption habits, license holding, and participation in tournaments. This basic information was used to quantitatively describe the anglers' profile.

For each kind of fishery (onshore, onshore boat, offshore boat, and spearfishing), each respondent provided information on daily catch, number of fishing days, size of the fishing season, species caught, and bait used. This information, together with the ratio of license holders/non-holders obtained from the questionnaire and the number of licenses sold (PNDPA, unpubl. data), was used to estimate the annual catch from recreational fisheries off northeastern Brazil in 2001. A Monte-Carlo simulation was performed to consider uncertainty in the input data (daily catch, fishing days per week, and size of the fishing season). Ten thousand runs were carried out and random values of each input variable were selected from a triangular distribution. The simulation was repeated for three different ratios of license non-holders/holders: 1.25, 2.5, and 5. No attempt was made to split these catches among species.

Definitions

Recreational fisheries = "fisheries conducted by individuals primarily for sport but with a possible secondary objective of capturing fish for domestic consumption but not for onward sale" (FAO 1997).

Light tackle = tackle with lines thinner than 0.6 mm.

Heavy tackle = anglers can use any line specification, including thicker than 0.6 mm.

Onshore fisheries = take place by the beach, in the surfing zone, and off piers.

Onshore boat fisheries = anglers use boats, but stay in waters close to the coast.

Offshore boat fisheries = anglers use boats and go offshore to catch billfishes, tunas, and mackerels.

Spearfishing = "taking of fish with a handheld spear or spear gun by a snorkeler or diver while swimming in the water" (Schultz 2000).

Official fishing club = club officially recognized by the Brazilian Fishing and Aquatic Sports Confederation (CBPDS).

Table 1. Age distribution of anglers from northeastern Brazil obtained through questionnaires in June-August 2001.

Age (years)	Number	Percent
10-19	5	3
20-29	19	13
30-39	34	23
40-49	45	30
50-59	32	21
60 or older	15	10
Total	150	100

Commercial fisheries = include both artisanal and industrial fisheries.

Results

Anglers' profile

These results are based on the answers from 150 respondents, corresponding to a response rate of 10%. Ninety-five percent of the respondents were men, mainly 30-49 years old and self-employed or retired (Tables 1 and 2). Questionnaires were distributed only in coastal cities, where all fishing clubs are located. Therefore, none of the respondents were restricted to freshwater angling and 38% were restricted to marine waters; most of them actually fish in both.

Fifty-three percent of the respondents fish only in their residential state and only 18% go on fishing trips outside the limits of northeastern Brazil. Seventy-nine percent of them take part in tournaments. When fishing on their own, i.e., not during a tournament, 86% of the respondents said they sometimes release the fish they have caught, 71% consume their own catch, and 43% give away some fish to family, friends, or charitable foundations. Seven of the respondents sold part of their catch. Eighty-eight respondents participate in only one type of recreational fishery: 75 in onshore fisheries, 12 in onshore boats, and 1 in offshore boats (Table 3). Forty-four of them share any two types of fisheries, nine would share three types and only three participate in all four types (including spearfishing).

The respondents are quite aware of the problems involved in recreational fisheries: "predatory" fisheries (mainly trawlers); forest clear-

Table 2. Occupation of anglers from northeastern Brazil obtained through questionnaires in June-August 2001.

Occupation	Number	Percent
Self-employed	39	26
Retired	31	21
Private sector	23	15
Public servant	18	12
Student	14	9
Tourism sector	5	3
Unemployed	3	2
Fishermen	1	1
Others	15	10
No response	1	1
Total	150	100

cutting and its effect on coastal waters; lack of serious programs for fishery management; importance of minimum size; and limitation of the catch-release practice, to name a few. They are also unsatisfied with the performance of the institution responsible for managing recreational and commercial fisheries (IBAMA).

Initial estimate of annual catch from recreational fisheries off northeastern Brazil

The annual catch for this region was estimated at about 1,147 t in 2001. This estimate was obtained by multiplying the values presented in columns 2-5 of Table 4 and by correcting the result using a 2.5 multiplier. This value was calculated based on the proportion of license holders and non-holders obtained from the questionnaire. The results from the simulation, using a 2.5 multiplier, showed a high probability of catches between 1,000 and 2,000 t (Fig. 2). A 1.25 multiplier produced lower catches (300 to 1,500 t) and a 5 multiplier resulted in higher catches (1,200 to 5,500 t). Independent of the multiplier used, the catches from recreational fisheries are probably between 0.2 and 4% of the total annual catches from commercial fisheries.

Considering that mainly members of fishing clubs were targeted in this study (for logistic reasons), who are probably more avid anglers than nonmembers, this first estimate should be viewed as an upper limit of

Table 3. Number of anglers taking part in any combination of two recreational fisheries in northeastern Brazil, obtained through questionnaires during June-August 2001.

Fishery	Onshore	Onshore boat	Offshore boat	Spearfishing	Total
Onshore	75	39	1	0	115
Onshore boat	–	12	4	0	54
Offshore boat	–	–	1	0	5
Spearfishing	–	–	–	0	0

The sum of the values on the diagonal represents the number of anglers participating in only one type of recreational fishery. The total column does not add up to 150 (total number of respondents), as some anglers participate in more than two fishing types.

Table 4. Mean values obtained from questionnaires in June-August 2001, used to calculate a point estimate for catches from recreational fisheries off northeastern Brazil in 2001.

State	Days[a]	Months[b]	Catch (kg d^{-1})	Licenses[c]	Catch (t $year^{-1}$)
Bahia	1.3	9.9	4.5	71	39.6
Ceará	0.1	9.3	1.3	45	0.3
Maranhão	1.0	7.7	25.1	8	15.9
Paraíba	1.0	12.0	1.0	25	3.0
Pernambuco	2.4	7.2	40.5	78	535.1
Rio Grande do Norte	1.6	9.2	22.5	30	98.8
Sergipe	1.4	9.9	12.5	46	78.9
Piauí	1.2[d]	9.3[d]	15.3[d]	92	162.8
Alagoas	1.2[d]	9.3[d]	15.3[d]	120	212.3
Total				515	1,146.7

[a] Fishing days per week.

[b] Length of the fishing season.

[c] Number of licenses sold in each state (unpubl. data, Plano Nacional de Desenvolvimento da Pesca Amadora).

[d] Note that the mean values for Piauí and Alagoas correspond to the average for all other states.

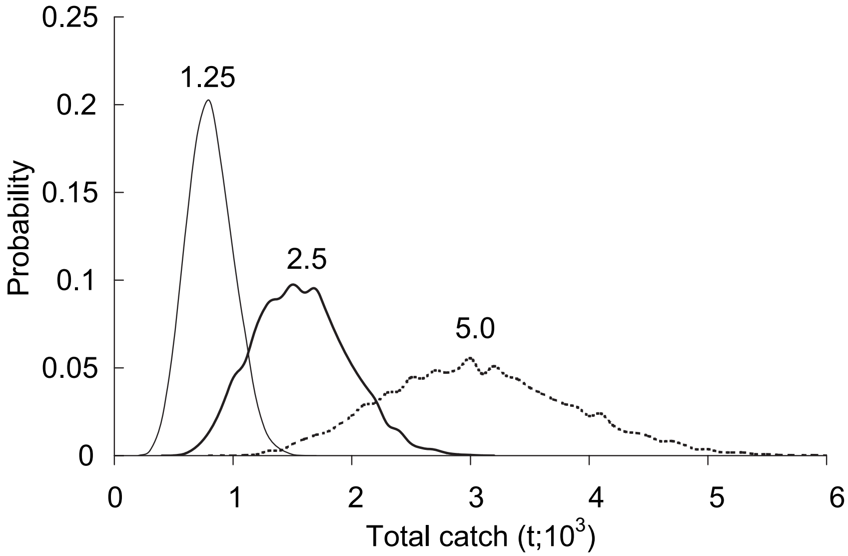

Figure 2. Annual catch from recreational fisheries in northeastern Brazil as obtained by a Monte Carlo simulation (no. of runs = 10,000). The numbers above the curves represent the ratio license non-holders/holders.

recreational catches in this region. However, there is no local evidence to confirm this source of bias, an issue to be addressed further. Avidness may also be leading to an overestimation of catches, as pointed out by Pollock et al. (1994), if we consider that more avid anglers are more prone to answer questionnaires. This effect is particularly important in this study where the response rate was low.

Eighty-six species or groups of species were recorded in the questionnaires as caught by anglers, with the main ones being catfish (Ariidae), threadfins (*Polydactylus* spp.), rays (unidentified), jacks (*Caranx* spp.), pompanos (*Trachinotus* spp.), southern kingcroaker (*Menticirrhus americanus*), puffers (unidentified), snooks (*Centropomus* spp.), king mackerel (*Scomberomorus cavalla*), serra Spanish mackerel (*Scomberomorus brasiliensis*), sharks (unidentified), weakfish (*Cynoscion* spp.), banded croaker (*Paralonchurus brasiliensis*), tarpon (*Megalops atlanticus*), jenny mojarra (*Eucinostomus gula*), dolphinfish (*Coryphaena hippurus*), mutton snapper (*Lutjanus analis*), groupers (*Epinephelus* spp.), yellowtail snapper (*Ocyurus chrysurus*), sailfish (*Istiophorus albicans*), tuna (*Thunnus atlanticus*, *T. obesus,* and *T. alalunga*), and barracuda (*Sphyraena barracuda*).

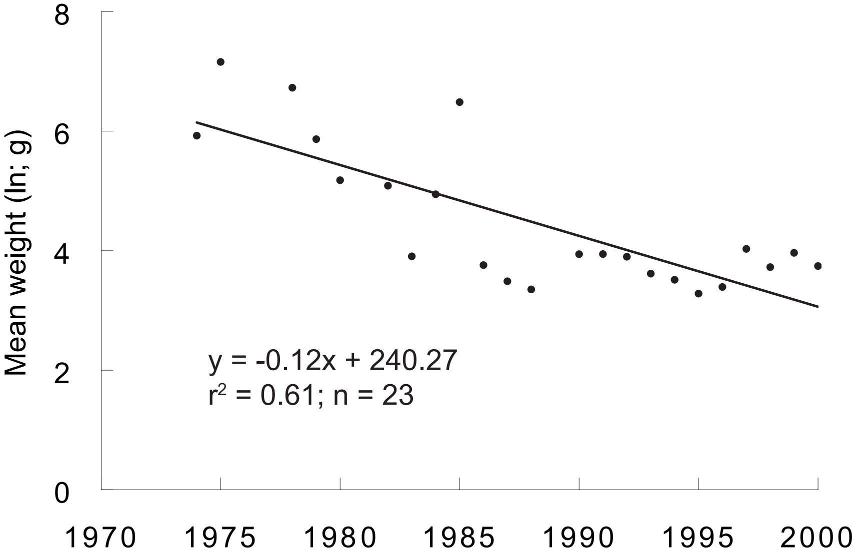

Figure 3. Mean weight of fish caught during onshore fishing tournaments in Rio Grande do Norte state (unpubl. data from tournament results). Each data point is based on the ratio between the total number and the total weight of fish caught in all tournaments promoted in a given year.

Changes in fish size through time

The analysis of the mean weight obtained from total data (total weight divided by the number of fish caught by each angler in each tournament) showed a decreasing trend from 1975 in Rio Grande do Norte state (Fig. 3). Figures 4 and 5 show decreasing trends for weights of the largest fish caught in tournaments using light tackle and heavy tackle, respectively. This trend is less evident for the heavy tackle, as anglers are allowed to use a wide range of line thickness during these tournaments. The five largest fish most often listed in the records of fishing tournaments in Rio Grande do Norte state are jacks ("aracanguira"), threadfins (*Polydactylus* sp.; "barbudo"), catfish ("bagre"), croakers (*Menticirrhus* sp.; "judeu"), and puffers.

Discussion

The lack of national, regional, or local initiatives to compile recreational catch statistics has led to the loss of a large amount of data potentially

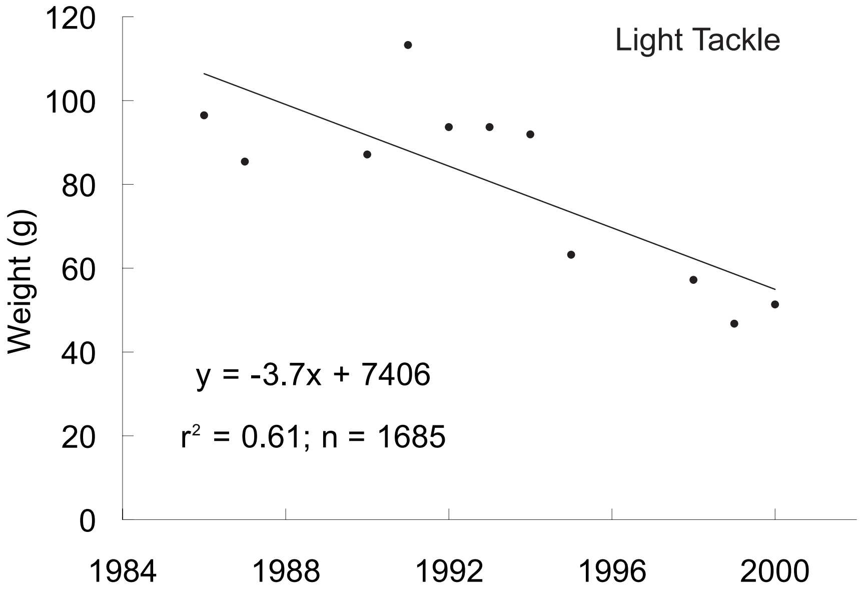

Figure 4. Mean weight for the largest fish recorded per angler in each onshore fishing tournament using light tackle (Rio Grande do Norte state; unpubl. data from tournament results).

useful for fisheries management. This is associated with the lack of interest by club presidents in keeping data archives owing to storage space limitations (most clubs are hosted in private homes). Thus, it is clear that a permanent infrastructure is needed to collect, compile, and store such data. Ideally, IBAMA/PNDPA should work in partnership with CBPDS to collect and compile recreational catch statistics by species and by state. In the past, the CBPDS made tournament data available online, though it was restricted to the most recent "official" events.

In addition, there seems to be insufficient dialogue among local institutions involved with Brazilian recreational fisheries and scientific advisors leading to the lack of understanding of these fisheries. Committees organizing fishing tournaments normally record results using the point system, information precious for anglers but of limited use for scientific analyses, i.e., number and weight of fish caught are the minimum required quantitative data for additional scientific analyses. Convincing anglers to incorporate their "local" knowledge in future data gathering projects can help to improve the compilation of necessary data. This should, however, involve buy-in strategies by tournament organizers highlighting the gain anglers would benefit from in spite of the "lost" time

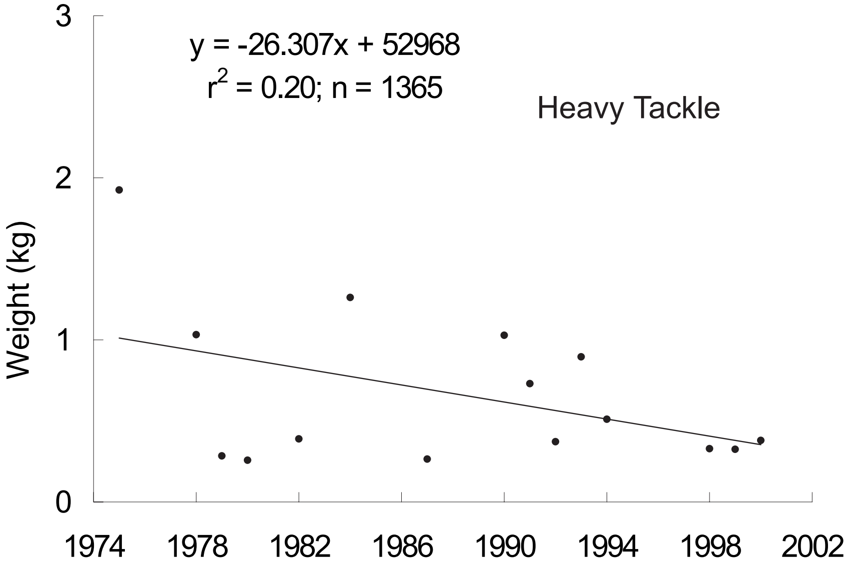

Figure 5. Mean weight for the largest fish recorded per angler in each onshore fishing tournament using heavy tackle (Rio Grande do Norte state; unpubl. data from tournament results).

due to, e.g., counting the number of fish caught by species and recording their individual weights. Data can also be gathered outside of tournament periods if, e.g., club presidents are able to convince their members to keep individual logbooks recording their daily fishing activities. PNDPA created a Fishing Logbook (Caderneta de Pesca) in 2003 and made forms available online to record daily fishing activities (http://www.ibama.gov.br/pescaamadora/caderneta/ [see fm_pescaria.php]). This initiative, however, will leave non-licensed anglers outside of the logbook program.

In data-limited situations, the number of licenses sold may be used to refer to the number of anglers. However, this method underestimates the total number of anglers, as not all of them have fishing licenses, e.g., 59% of the respondents in this study did not have fishing licenses. The number of licenses issued in Brazil has decreased from 244,000 in the 1970s to 88,000 in 2002, with a wide information gap between 1982 and 1999. This probably does not reflect a decrease in the number of anglers, but rather on their degree of satisfaction with IBAMA, the institution responsible for issuing and controlling angling licenses. Results of interviews with club presidents indicate there are 1,790 fishing club members in northeastern Brazil, while only 515 angling licenses were issued in that region in 2000.

Another cause for underestimation of the overall number of anglers is that mainly men (95%) are included in the counts, while in fact women do participate in this activity. Brazilian women have competed in fishing tournaments, including spearfishing events, at least since 1974 (Pâmpano Esporte Clube 1994). Their participation has increased since the 1970s. In 2002, about 22% of records available from fishing tournaments in Rio Grande do Norte state referred to women. Sixty-five-year-old men and sixty-year-old women (or older) and retired anglers are exempted from buying fishing licenses, if they are not affiliated with fishing clubs, as well as those who are younger than 18 years or fish only with handline or pole and line. This is another factor leading to the underestimation of the total number of anglers.

Anglers fish in both fresh and salt water in northeastern Brazil, mainly in their residential state. Initiatives to control and/or monitor fishing practices in freshwater may result in an increase in fishing effort in marine areas because anglers fish in both environments. Another concern is defending the practice of catch-release as a management tool. Catch-release is a well-established concept, introduced in Brazil in the 1990s by Rubens de Almeida Prado (Pescaventura, São Paulo, Brazil, Feb. 2003, pers. comm.). Some anglers were and still are skeptical about the effectiveness of this practice because of the occurrence of other more damaging activities such as trawling in areas less than 3 nautical miles from the coastline. This is a legitimate concern notably if hook mortalities are reviewed; e.g., mortalities vary from 3 to 88%, with the highest values associated with tropical marine areas (see references in Policansky 2002). These values could be even higher if post-release mortality is fully considered (Wilde and Pope 2003). Catch-release programs would also be more effective in highly productive temperate waters (Barnhart 1989). Thus, catch-release may not be the best management tool for areas such as northeastern Brazil, where temperature is high and productivity is low. In addition, enforcement would become a problem since most anglers are consumption-oriented and also give away fish caught as social contributions to charitable foundations. One fishing club in Paraíba state has even trained street children as anglers who have participated in fishing tournaments (and turned out as prize winners!).

Respondents showed a preference for onshore fisheries, probably associated with accessibility and low cost, though some of them combine one to three fishing types. The estimated total catch for recreational fisheries off northeastern Brazil, combined for all four fishing types (onshore, onshore boat, offshore boat, and spearfishing), of about 1,147 t represents only 0.8% of the total commercial catch of this region. However, the actual ratio may be between 0.2 and 4%, if uncertainty in all input parameters is considered. Due to the low response rate, this estimation could be biased toward a higher limit because of the avidness of respondents and also because most of them were members of fishing clubs.

All species reported in the questionnaires and in tournament results are also caught by artisanal fisheries. As recreational fisheries take place in areas much closer to the coast than artisanal fisheries, the former might be targeting juveniles of the species caught by the latter, a theme to be followed up elsewhere. Any attempt to analyze the impact of one fishery on another has to consider the correct identification of species in both fisheries, as catches are recorded by common names and the number of common names per species is very high (Freire and Pauly 2003). Some species can have up to 37 different common names, and names used by anglers can be quite different from those used by commercial fishers.

Although this study shows that recreational fisheries in northeastern Brazil do not result in high catches, its possible impact on commercial fisheries (through the capture of juveniles) and its ascending importance in this region demand a close monitoring program of this activity.

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Fisheries Assessment and Management in Data-Limited Situations
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Assessing the Effects of Fishing in Cape Verde and Guinea Bissau, Northwest Africa

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Abstract

The assessment of fishery resources in countries such as Cape Verde and Guinea Bissau is particularly difficult due to data limitations and a lack of consistent time series. Considering these limitations, several techniques were applied, depending on the type of data available. These techniques included nonparametric multidimensional scaling (MDS), abundance-biomass curves (ABC curves), correlation analysis, and general linear models (GLM), which were applied to trawl survey data. Furthermore, mean size of species groups was used as an indicator in the case of Guinea Bissau and time series analysis was applied to the Cape Verde catch statistics.

In Cape Verde, there was no apparent change over time in the structure of demersal fish communities. This is consistent with a gradual increase in fishing pressure over a period of 20 years, which was also observed as a trend for lower mean abundance of demersal fish species in recent years. In contrast, time series analysis of catch data indicates that a shift has occurred with decreasing catches of important pelagic species such as yellowfin and skipjack tuna and increasing catches of small pelagics and neritic tuna as well as some demersal species.

In Guinea Bissau, the effects of area and depth were found to be important in defining the structure of demersal fish communities. In contrast, the effect of time was not apparent. On the other hand, demersal fish abundance and mean size of groups decreased for recent years, from 1989 to 1992 in particular, which was a period of intensive fishing. Furthermore, an effect on the abundance/biomass relationship (ABC curves) was observed for the same period.

Introduction

Quantitative ecosystem indicators have been proposed in recent years to determine the effects of fishing on marine ecosystems from a global or broad perspective (e.g., Jennings and Kaiser 1998, Pauly et al. 1998, Bianchi et al. 2000, Gislason et al. 2000, Rice 2000, Jennings et al. 2002, Link et al. 2002). Many of these indicators provide a comprehensive bridge between different scientific disciplines, including the ecological, environmental, and fisheries perspectives, and constitute a more efficient way to communicate results for management purposes (Cury and Christensen 2001). These new complementary management tools may prove useful for improving on the current state of fisheries resources and their application would be particularly convenient in data-limited situations such as in many tropical fisheries.

The Cape Verde Archipelago and Guinea Bissau are located in Northwest Africa (Fig. 1). In spite of their relative proximity, these two countries are different in terms of ecosystem productivity and functioning, thus classified in different marine ecological provinces (Longhurst 1998). The Cape Verde ecosystem has strong links to the open oceanic ecosystem of the central Atlantic and is characterized by relatively low productivity and very limited shelves around the islands. The Guinea Bissau continental shelf is relatively extensive and considered highly productive and an important nursery ground in the Northwest African region with strong seasonal influence from upwelling and river runoff (Berrit 1977).

Trawl surveys have been undertaken sporadically in both countries, and therefore consistent time series of survey data were not available. The quality of survey data in Guinea Bissau is generally better in that sampling methodology and coverage has been more homogenous over time. This is related to the fact that trawl surveys are more difficult in Cape Verde due to the very narrow shelves and difficult bottom conditions (Fig. 1).

Cape Verde has had a reasonably well-functioning statistical collection system since 1986, but a consistent time series of fisheries statistics is lacking in Guinea Bissau. Total estimated catches in Cape Verde are around 10,000 t (INDP 2001) (Fig. 2). In Guinea Bissau, artisanal catch estimates are available for 1993 only, amounting to around 40,000 t (Thiam 1993).

Another difference is that industrial trawlers operate on the continental shelf off Guinea Bissau, while industrial trawling is prohibited in Cape

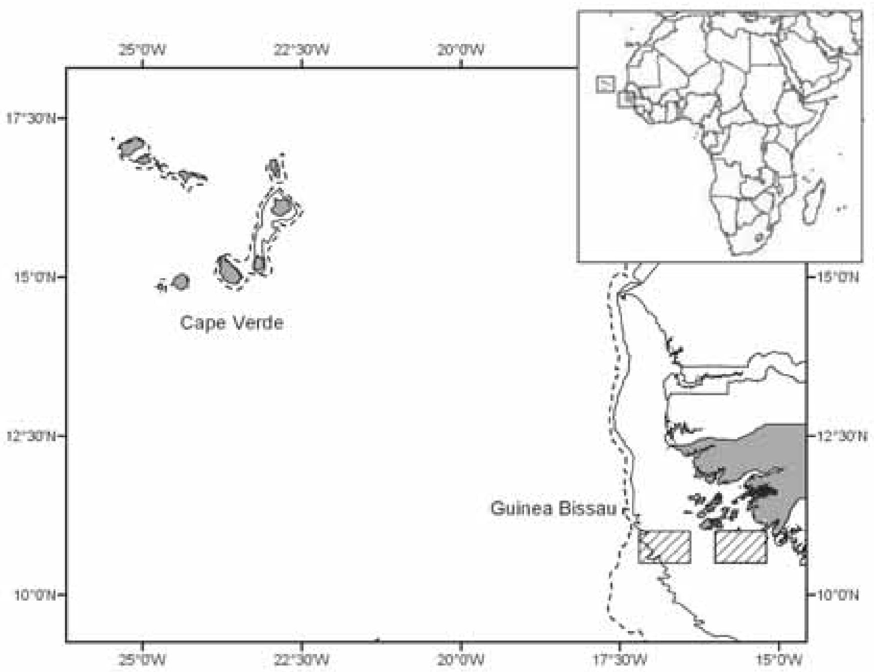


Figure 1. Geographic location of the study area including bathymetric lines corresponding to depths of 200 (dashed) and 1,000 (solid) meters. Also shown are the two "windows" defined off Guinea Bissau, representing areas characterized by strong seasonal upwelling (western window) and warmer/less saline waters (eastern window). Western window: 16.4°-17.2°W, 10.5°-11°N; eastern window: 15.2°-16°W, 10.5°-11°N (decimal degrees).

Verde. Estimates of these industrial trawl catches in Guinea Bissau have ranged between 20,000 and 45,000 t during the last two decades with a doubling in the period from 1990 to 1992 (Fig. 3) (Watson et al. 2000).

These two countries are thus quite different in terms of fishery exploitation and ecosystem characteristics, but the fisheries in both countries can be considered as data-limited situations. A pragmatic approach was adopted using methods that apply for the data available in order to determine the possible effects of fishing over time.

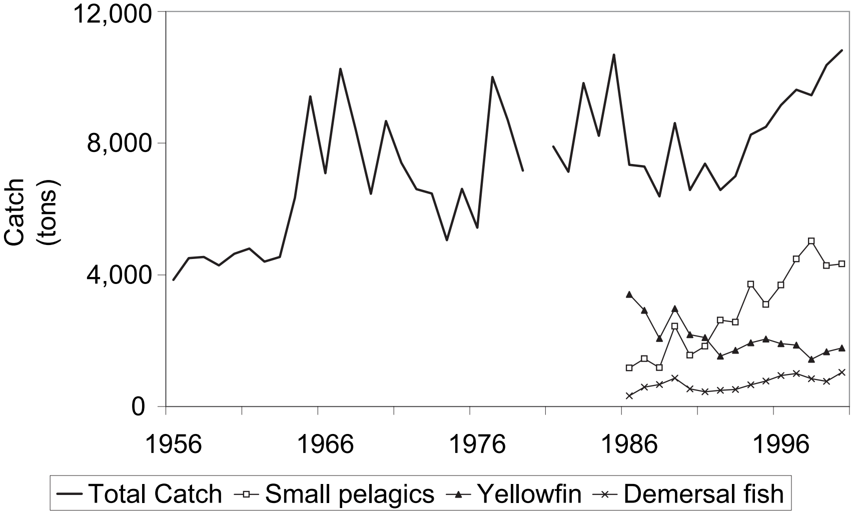


Figure 2. Estimated catches in Cape Verde, including the catches of selected important species or species groups (INDP 2001). The source of estimates up until 1979 was Watanabe 1981.

Materials and methods

One of the objectives of a recent project, the Fisheries Information and Analysis System project (SIAP), was to identify, recover, and compile data from historical trawl research surveys undertaken in the northwest African region (see also http://www.seaaroundus.org/Dakar/index.htm). As a result of these efforts, data from several surveys were made available concerning Cape Verde and Guinea Bissau (Table 1). Trawl survey catches were standardized (kg per nm^2) using the swept-area method involving haul duration, average trawling speed, and a fixed value for the horizontal net opening. This standardization was not possible on a haul-by-haul basis, as data on speed were often lacking and the net opening was not measured.

A revision of the identified fish species was undertaken using FishBase 2000 (Froese and Pauly 2000), which attempted to detect problems such as species misidentifications, inconsistencies, or contradictions in the observations. Habitat was designated for all fish, cephalopod, and crustacean species. For the subsequent data analysis, pelagic and bathypelagic species were omitted, because trawl gears are not considered suitable for sampling pelagic species. In Cape Verde, the catches of cephalopod and crustacean species were not considered as these data

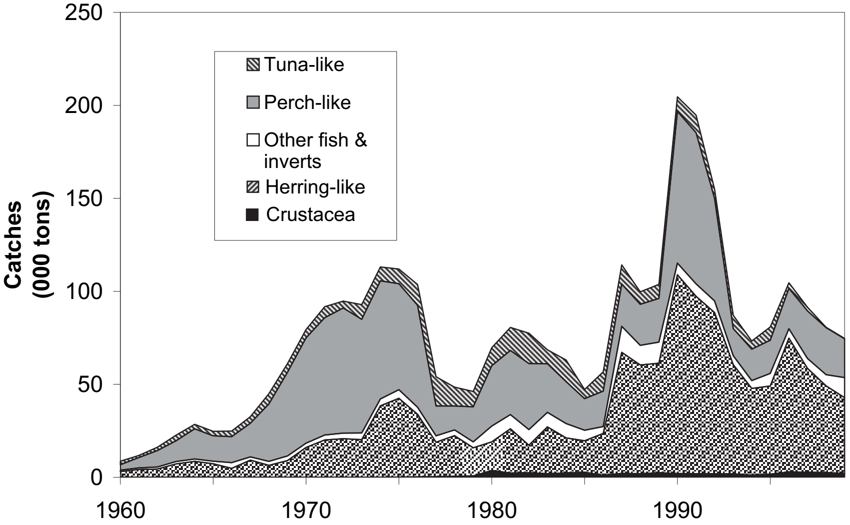


Figure 3. Estimated industrial catches (cumulative) in Guinea Bissau from Watson et al. 2000. Industrial trawl catches are composed of perch-like fish, other fish, crustaceans, and other invertebrates.

were rare occurrences. Furthermore, all trawl stations with depths over 200 m were omitted as these depths were sporadically sampled.

The standardized catches by species were aggregated by genera in order to avoid problems due to species identification. A total of 447 trawl stations (observations) and 172 genera were considered in the case of Guinea Bissau, while in Cape Verde there were 217 trawl stations and 99 genera.

Trawl stations were grouped according to year, depth (≤ 30 m; 30-80m; ≥ 80 m) and area. Two areas were defined in Guinea Bissau, based on the limit of 11º30′N as an approximate division between areas according to two broad sediment types: silt or silty sand sediments in the north and sand or sandy silt in the south (e.g., Amorim et al. 2004). In contrast, the area was divided according to the continental shelf in Cape Verde, including three consistently sampled areas: the shelf around the north-eastern island Sal, the northwest shelf system, and the more extensive shelf around Boavista and Maio islands (eastern islands) (Fig. 1).

The multivariate ordination method nonparametrical multidimensional scaling (MDS) was used to study changes over time and space in the species composition and abundance of demersal communities in both countries. This method (MDS) is generally applicable as fewer

Table 1. Trawl research surveys considered in the study, including the number of haul stations, depth coverage, year, and month for each survey.

	Month	Survey context	Depth (m)	No. of stations
Cape Verde				
1964	March	Spanish cooperation	42-75	6
1976	October	German (DDR) cooperation	45-195	16
1981	November	Fridtjof Nansen Programme	23-112	18
1985	June/July	Icelandic cooperation	20-338	37
1988	August	Icelandic cooperation	4-188	80
1994	August	Cape Verde/Iceland	20-137	60
Guinea Bissau				
1963	December	Guinean trawl surveys	10-200	25
1964	May	Guinean trawl surveys	10-500	24
1981	May/June and September	Fridtjof Nansen Programme	10-95	17
1986	August-September	Fridtjof Nansen Programme	12-200	30
1988	April-May	Portuguese-Guinean cooperation	12-374	35
1989	March-April	Portuguese-Guinean cooperation	10-304	88
1990	April-June	Portuguese-Guinean cooperation	10-250	105
1991	May-June	Portuguese-Guinean cooperation	10-480	33
1992	March	Fridtjof Nansen Programme	11-295	47
1995	May-June	Portuguese-Guinean cooperation	10-103	77

assumptions are made about the nature and quality of the data (Clarke and Warwick 2001). Thus, when considering biological survey data, it is not necessary to deal with problems related to the skewed distributions of the data and all species can be included, which is important in studies concerning species assemblages.

MDS is simple in concept, which is to create a sample map or ordination plot where the distances between points have the same rank order as the corresponding similarities (or dissimilarities) between samples (or haul stations in this case) (Clarke and Warwick 2001). MDS plots can be arbitrarily scaled or rotated, while maintaining the relative distance between points on two or three dimensions. This is in contrast to a classical principal components analysis (PCA) plot, where the PC axes explain a certain amount of variance based on linear combinations of species data (Euclidean distances). A measure of goodness-of-fit is termed the stress value in MDS:

$$Stress = \sqrt{\sum_j \sum_k (d_{jk} - \hat{d}_{jk})^2 \Big/ \sum_j \sum_k d_{jk}^2}$$

where $\hat{d}_{jk}$ is the distance predicted between the j^{th} and k^{th} sample points from the fitted nonparametric regression line corresponding to dissimilarity δ_{jk} . Stress values larger than 0.3 indicate an almost random position of points; values lower than 0.1 correspond to a good ordination with clear interpretation; and values between 0.1 and 0.2 indicate a potentially useful ordination but interpretation should be handled with care (Clarke and Warwick 2001).

MDS was applied to a similarity matrix calculated with the Bray-Curtis coefficient on square-root transformed data. The Bray-Curtis similarity coefficient has the desirable property of not being sensitive to joint absences or double zeros (Clarke and Warwick 2001), which is common in species occurrence or abundance data.

Another approach was the study of mean abundance of demersal fish resources over time in each country using general linear models (GLM) (Dalgaard 2002). This is a common procedure to obtain standardized abundance indices in stock assessment, based on fisheries catch and effort data including fleet characteristics (e.g., Hilborn and Waters 1992). The approach used here differs in that we are not standardizing according to fleets, but according to other factors such as depth and area strata. Survey catches were standardized using the swept-area method in an attempt to remove vessel effects. Standardized catches (kg per nm^2) were aggregated according to year, area, and depth as well as commercial value (using codes 0 and 1 for noncommercial and commercial, respectively).

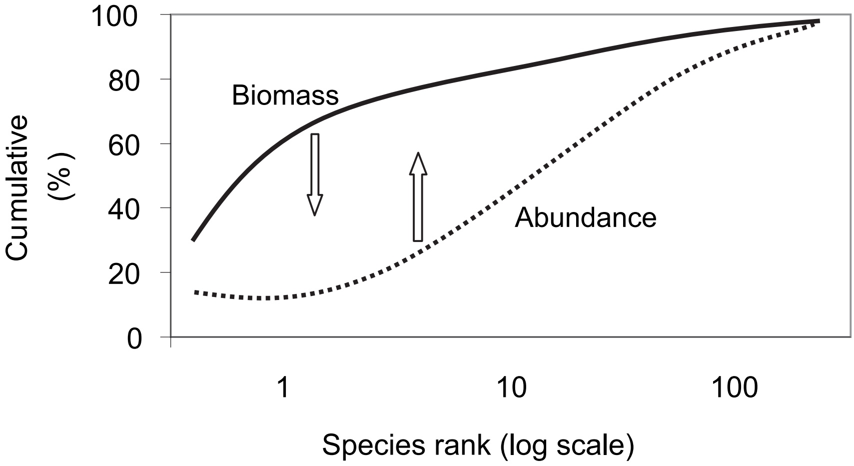


Figure 4. Schematic representation of an abundance and biomass curve (ABC). In a stable community without perturbations, the biomass (based on cumulative weight per species) curve is expected to be above the abundance curve (based on cumulative numbers of individuals per species). The arrows show the expected shift as a result of perturbation such as the effects of fishing.

Aggregated catches were log-transformed, which was straightforward as all catches were larger than zero. Due attention was given to diagnostics such as residual plots and distribution.

Guinea Bissau

Trawl surveys undertaken in Guinea Bissau provided data on both numbers and weights per species, which are suitable for constructing abundance and biomass (ABC) plots. These ABC curves are constructed by ranking species in terms of abundance (numbers) and biomass (weights) and plotting the respective curves as cumulative percentages (Fig. 4). Biomass curves tend to be above the abundance curve in undisturbed communities, characterized by fewer and larger individuals (*K*-selected species) and the inverse, a dominance of smaller individuals, is expected in highly perturbed or polluted communities (Clarke and Warwick 2001). Following the same line of reasoning, fishing or environmental effects are considered perturbations to a stable situation and the expected results would be a relative displacement of the biomass curve downward in relation to the abundance curve. In order to compare these various ABC curves, the summary statistic W was calculated:

$$W = \sum_{i=1}^{S} (B_i - A_i) / \left[50(S-1) \right]$$

W is based on the sum of differences between biomass (B) and abundance (A) rank percentages standardized to a common scale, so that comparisons can be made between samples of differing number of species (S) (Clarke and Warwick 2001).

This approach was not considered interesting for Cape Verde as there were only two surveys with data on both numbers and weight of individuals.

Correlation analysis was applied to study the possible relationships between fishing effort, precipitation, upwelling, fish abundance, and the mean size of fish, cephalopod, and crustacean species in Guinea Bissau. However, this was possible only from 1986 onward due to limitations in the environmental data available. Mean size was calculated as a simple weighted average over time. Fishing effort was estimated for the period as the number of industrial trawlers operating, using the estimated catches of demersal species and a mean yearly catch per industrial trawler, assuming that catches were proportional to effort.

The upwelling index was constructed based on the annual differences in sea surface temperature (SST) between two “windows” representing areas characterized by strong seasonal upwelling, and on the other hand warmer, less saline waters from the Gulf of Guinea Current (Fig. 1). SST monthly data were available from the NASA Physical Oceanography Distributed Active Archive Center at the Jet Propulsion Laboratory, California Institute of Technology (NASA, no date). Monthly precipitation data was available through the Web, courtesy of Dr. Mike Hulme at the Climate Research Unit, University of East Anglia, Norwich, UK (Climate Research Unit, no date). For both parameters, annual averages were calculated and used in correlation analysis as mentioned previously.

Cape Verde

A consistent time series of fisheries catch statistics was available from 1986 to 2000 in Cape Verde, thus making it possible to apply min/max autocorrelation factor analysis (MAFA). MAFA is a recently developed technique for analyzing short (15 to 25 years), non-stationary, multivariate data sets (Solow 1994, Zuur 2003).

MAFA is a type of principal components analysis (PCA) where the axes represent autocorrelations and give an indication of the amount of association between variable Y_t and Y_{t+k} where k is the time lag with values 1, 2, 3, etc. Unlike PCA where the first axis explains most of the variance, the first MAFA axis has the highest autocorrelation and since trends are associated with high autocorrelation it therefore represents the main trend in the data (Erzini et al. 2005). MAFA can be used to extract trends from multiple time series, to estimate index functions from time series, and for smoothing (Zuur 2003).

As in PCA loadings can be estimated and used to determine the relationship of individual response variables to particular MAFA axes. Cross-correlations between MAFA axes and the response variables, also known as canonical correlations, are a measure of the relationship between Y_t and X_{t-k} and can also be calculated for the same purpose. This allows significant relationships between trends and explanatory variables to be identified. We used the Brodgar software package to carry out the MAFA on catches in Cape Verde.

Catch statistics were available at the species level (58 species and species groups), but these were grouped into main ecological functional groups for MAFA analysis (e.g., Stobberup et al. 2004): small pelagics, yellowfin tuna, skipjack tuna, small tuna, pelagic predators, demersal predators, demersal fish, jacks, moray eels, and sparids. Variables such as fishing effort, yearly precipitation, an SST index (yearly differences between October and March mean values), and the North Atlantic Oscillation index (Hurrell 1995; NCAR, no date) were included for canonical correlation analysis. The sources of SST and precipitation data were the same as referred above in the section for Guinea Bissau.

Results

The results for Cape Verde and Guinea Bissau will be treated separately in the following, because of differences in the types of data available and techniques used.

Cape Verde

The MDS plot for Cape Verde shows that change over time in the structure of demersal fish communities were not apparent as expressed in the relatively high stress value (0.23) (Fig. 5). The effects of depth and area also were not clear. Such a stress value, between 0.2 and 0.3, indicates that the plot is difficult to interpret and the results are inconclusive. Other attempts without data transformation and aggregating species at the genus level did not alter the overall result, an uninformative MDS plot.

The GLM model was highly significant ($P < 0.0001$; R^2 adjusted = 0.23; Table 2) and showed that the factors year and depth had significant effects on abundance. The effect of depth was observed as a significantly lower abundance in depths between 30 and 80 m (depth2: $P < 0.001$; Table 2). Abundance was significantly lower in the period from 1985 to 1994 ($P < 0.01$) relative to 1964, which can also be seen as a downward trend in abundance (Fig. 6). However, the result for 1976 stands out with a significantly higher abundance. Although the GLM model was highly significant, it explained only 23% of the variance. The effect of factors such as Area and Commercial value were not significant and therefore omit-

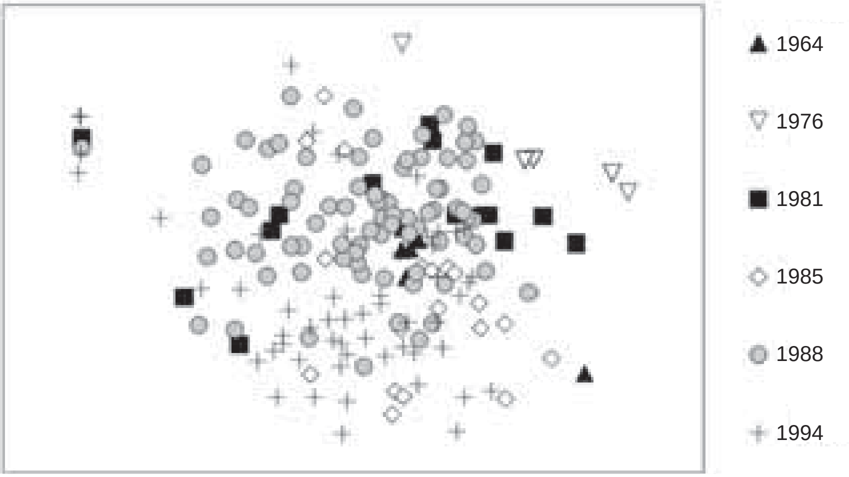


Figure 5. Two-dimensional MDS plot based on the similarity of haul stations in terms species of composition and abundance (square-root transformed). The symbols refer to survey year in Cape Verde. Note that MDS plots can be rotated without affecting the result (the relative distance between points) and the axes represent two arbitrarily chosen dimensions.

ted from the model. Also, an interaction term between year and depth was significant, but the improvement in terms of R^2 was considered low (about 3%) and thus not included in the final model.

Regarding MAFA, Fig. 7 shows the canonical correlations between species group catches and the first MAFA axis (Fig. 8). The significant negative correlation for yellowfin and skipjack tuna shows a decreasing trend in catches. A significant positive correlation was obtained for the other groups, except for demersal fish and jacks, indicating increasing catches over time. For demersal fish and jacks, catches have followed a different pattern with two peaks during the study period.

The first MAFA axis is shown in Fig. 8, which has the highest autocorrelation with time lag 1 (autocorrelation with time lag 1 = 0.97; Pr = 0.048). The underlying idea is that a trend is associated with high autocorrelation. Therefore, the first MAFA axis represents the significant trend or the main underlying pattern in the data—a trend of steadily increasing catches. Only the first axis was significant. As expected, fishing effort was correlated significantly with this increasing trend in catches (industrial effort = 0.79 and artisanal effort = 0.93; (critical level for a significant correlation was ± 0.52 (Pr = 0.05). The correlations to environmental

Table 2. GLM results for Cape Verde, including model specification and results as well as regression coefficients for each factor level. Model = log(std. catch) ~ year + d.strata.

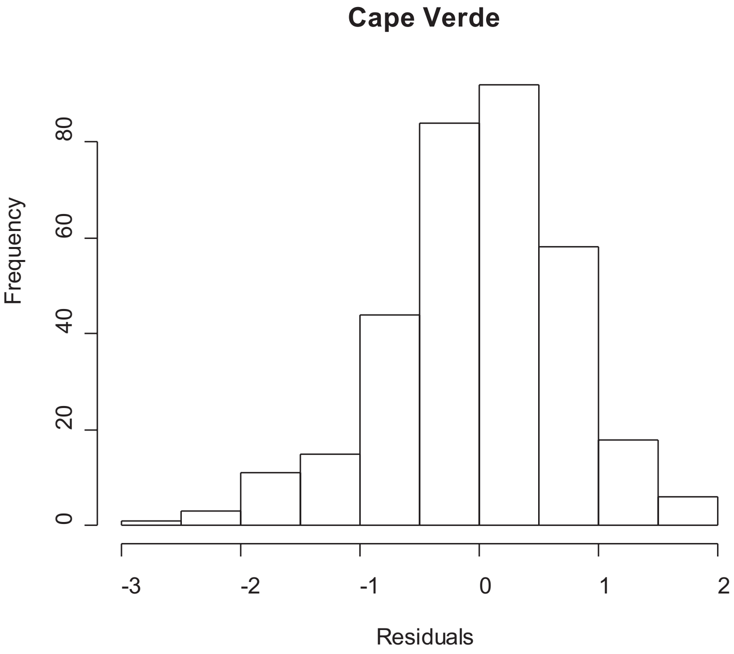
Coefficients	Estimate	Std. error	*t* value	Pr(>|*t*|)
(Intercept)	4.345	0.252	17.27	0.000
year_76	0.915	0.304	3.01	0.003
year_81	–0.252	0.266	–0.95	0.344
year_85	–0.891	0.265	–3.36	0.001
year_88	–0.672	0.232	–2.90	0.004
year_94	–0.840	0.235	–3.58	0.000
d.strata_depth2	–0.441	0.121	–3.64	0.000
d.strata_depth3	–0.229	0.143	–1.61	0.109

Residual std. error: 0.7634 on 324 d.f.

Multiple *R*-squared: 0.2453; adjusted *R*-squared: 0.229.

F-statistic: 15.05 on 7 and 324 d.f.; *P*-value: < 2.2e–16.

Only the two significant factors, year (Table 1) and d.strata (depth1: ≤ 30 m; depth2: 30-80 m; depth3: ≥ 80 m), were included in the model. Coefficients express the difference between each level of the factors and the first level. Mean abundance of demersal fish expressed as log kg per nm^2.



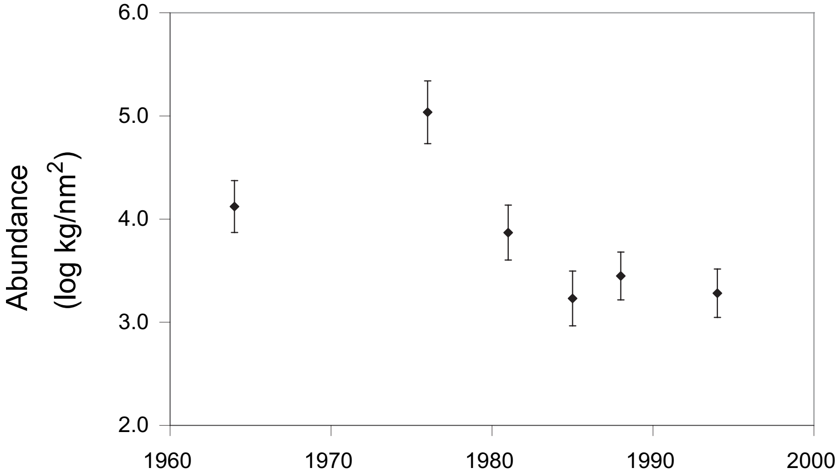


Figure 6. Mean abundance of demersal fish over time in Cape Verde. Estimated by GLM as the intercept plus the appropriate year coefficient plus the mean of depth coefficients (Table 2). Error bars represent ±SE (standard error).

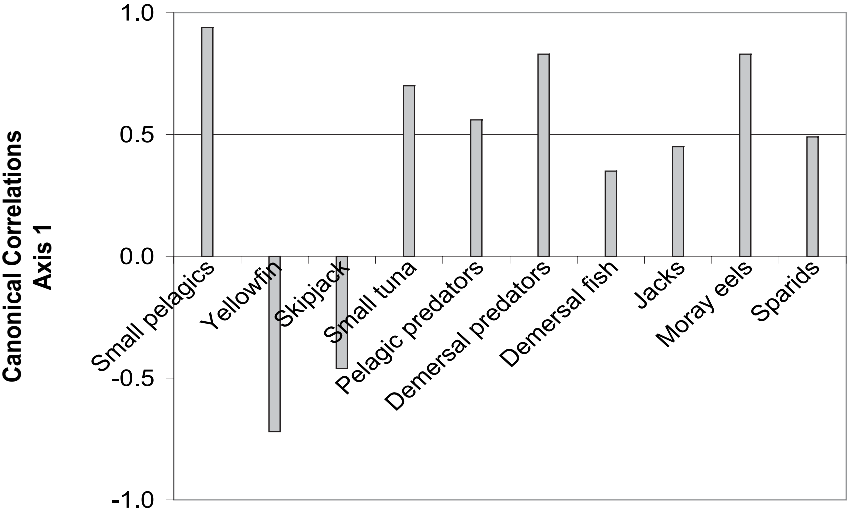


Figure 7. Plot of the canonical correlations between species groups and the first MAFA axis. Critical level for a significant correlation was ±0.52 (Pr = 0.05).

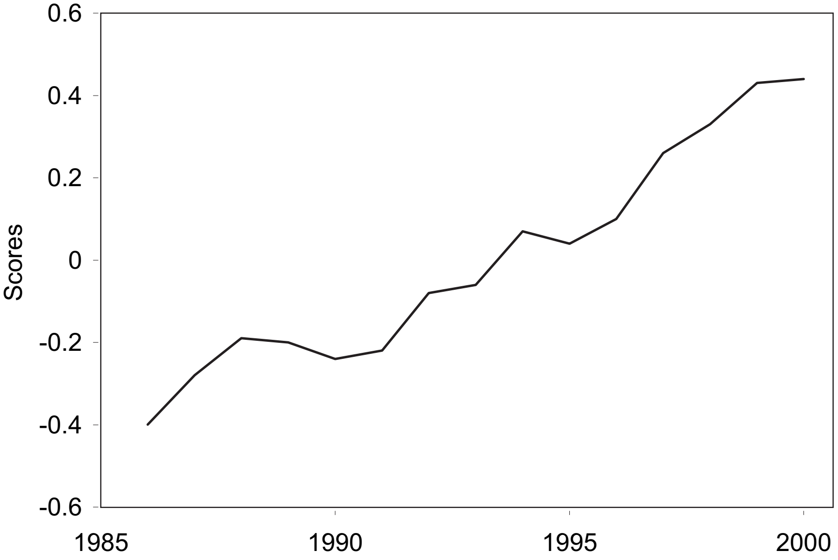


Figure 8. **Plot of the first MAFA axis showing the trend over time or the main underlying pattern in the data. MAFA scores on the y-axis are analogous to scores in PCA analysis, a measure of distance of points from the axis. The first axis is chosen so as to minimize the sum of squared distances of the points from the line and maximizing the variance of points projected on the axis. In MAFA however, the points are projected on an axis that represents a trend.**

variables such as yearly precipitation, North Atlantic Oscillation, and the SST index were non-significant.

Guinea Bissau

The MDS results for Guinea Bissau were similar to Cape Verde in that change over time in the structure of demersal fish communities was not apparent. In contrast, depth and area appeared to play an important role in defining demersal fish community structure in terms of species composition and abundance (Fig. 9).

On the other hand, the *W* statistic shows that change occurred in the period 1988 to 1992 concerning the relationship between abundance and biomass (Fig. 10). The increase in *W*, particularly in 1988, was the result of higher dominance by *Balistes* sp. triggerfish, which are relatively large individuals. However, the decrease in *W* from 1989 to 1992 was maintained even after the removal of *Balistes* sp. from the data, which appears to show that perturbation took place during this period.

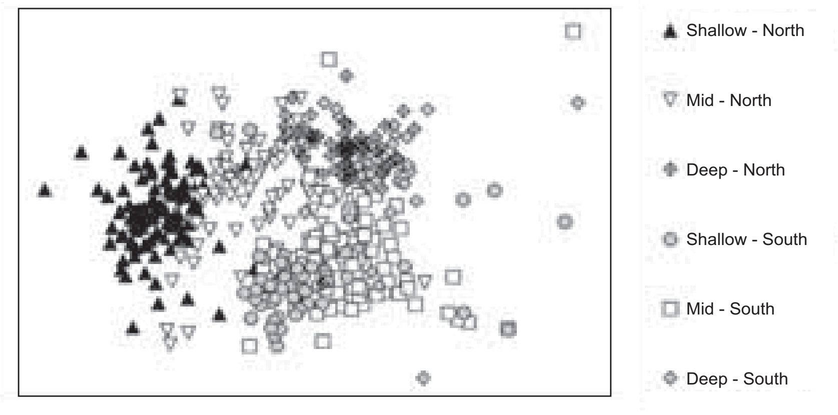


Figure 9. Two-dimensional MDS plot based on the similarity of haul stations in terms species of composition and abundance (square-root transformed). The symbols refer to area (north and south) and depth (shallow: ≤30 m; mid: 30-80 m; deep: ≥80 m). Note that MDS plots can be rotated without affecting the result (the relative distance between points) and the axes represent two arbitrarily chosen dimensions.

The GLM model was highly significant, explaining 38% of the variance in demersal fish abundance in Guinea Bissau ($P < 0.0001$; R^2 adjusted = 0.38; Table 3). In contrast to Cape Verde, the results for Guinea Bissau showed that the effects of all factors such as year, area (zone), depth (d.strata), and commercial value (c.value) were significant (Table 3). Abundance was significantly lower in the period 1988-1995 ($P < 0.01$), except for 1990, in relation to the beginning of the study period. There appears to be a stronger downward trend in abundance of commercial fish in relation to noncommercial fish, but the estimates for 1995 do not support this (Fig. 11). Furthermore, abundance was significantly higher in deeper waters, including both depth strata 2 and 3 as well as in the southern area ($P < 0.001$; Table 3).

The quality of trawl survey data from Guinea Bissau made it possible to calculate mean size of commercial species by group for several years (Fig. 12). It is interesting to note that there is a decrease in size of fish and cephalopods, which was particularly clear from 1988 onward and then an increase in 1995 for all groups. For crustaceans, the trend appears to be the opposite. GLM was used to test the Year effect on mean size of fish (the longer time series), including the effects of area, depth, and commercial value. The GLM model was highly significant ($P < 0.0001$; R^2 adjusted

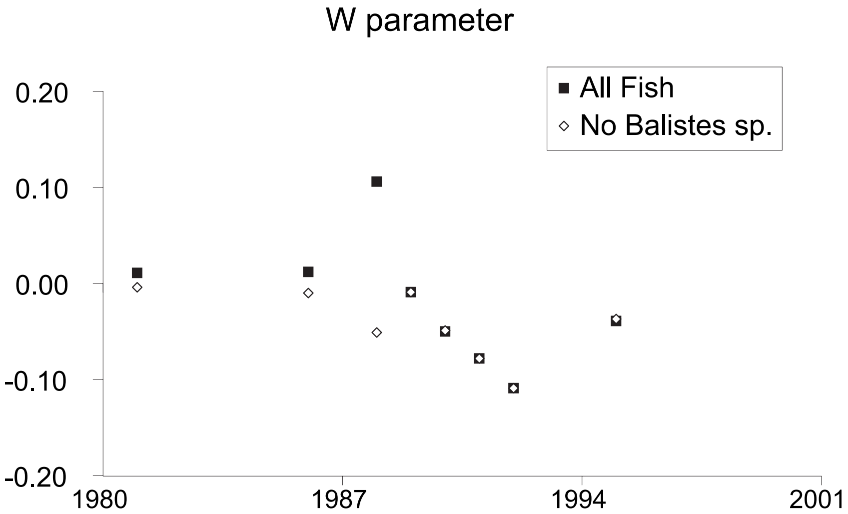


Figure 10. Plot of the *W* statistic over time, which shows the differences between biomass and abundance rank percentages over time. *W* was calculated for demersal fish, including and excluding *Balistes* sp.

= 0.35; d.f. = 88), showing that mean size of fish was significantly smaller between 1989 and 1992 ($P < 0.05$) in relation to the beginning of the study period (regression coefficients not shown).

Correlation analysis was used to explore the possible effects of fishing and the environment on abundance and mean size. This involved abundance of commercial and noncommercial demersal fish as well as mean size of fish, cephalopod, and crustacean groups. These were correlated with fishing effort as well as precipitation and upwelling with various time lags: no time lag, time lag 1 year, and time lag 2 years.

There was a highly significant negative correlation between fishing effort (industrial) and the mean size of commercial demersal fish ($r = –0.956$; $P < 0.001$). Also moderately significant correlations ($P < 0.05$) were the positive correlations between effort and the abundance of commercial cephalopods as well as rainfall (time lag 2 years) and the abundance of commercial cephalopods. All other correlations were non-significant ($P > 0.05$).

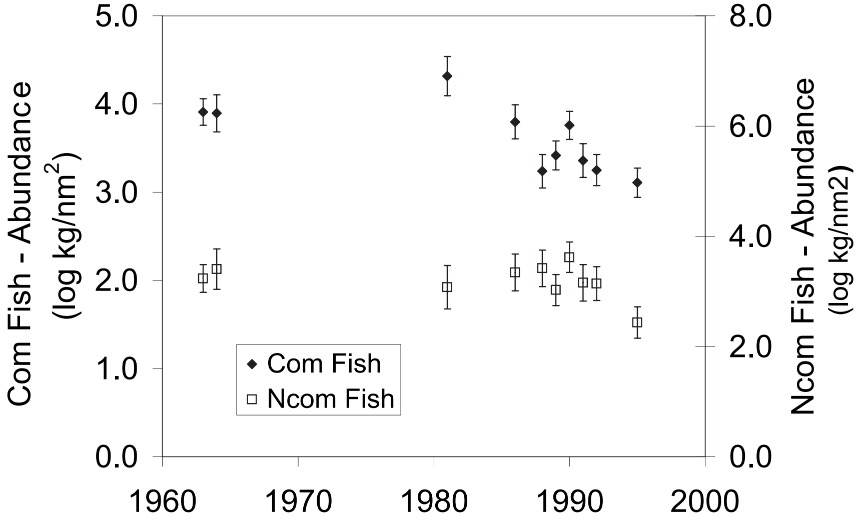


Figure 11. Mean abundance of commercial (com) and noncommercial (ncom) demersal fish over time in Guinea Bissau estimated by GLM (Table 3). For commercial fish, this was estimated as the intercept plus the appropriate year coefficient plus the mean of depth and area coefficients. The same procedure was used for noncommercial fish, but now adding the c.value coefficient plus the corresponding year:c.value interaction coefficient for each year (Table 3). Error bars represent ±SE (standard error). Note the different scales on the y-axes, which were used to avoid clutter.

Discussion

In Cape Verde, no apparent change over time was observed in the structure of demersal fish communities (Fig. 5). This can be considered consistent with a gradual increase in fishing pressure. Depth and area were expected to have an effect (e.g., Longhurst and Pauly 1987, Amorim et al. 2004), but this was not the case. This unexpected result may be related to the quality of the historical surveys in terms of species identification as well as the difficult conditions to adequately sample the shelves around the islands.

There was a decreasing trend in demersal fish abundance in Cape Verde, which is in contrast to the increasing trend of demersal fish catches (Figs. 6, 7, 8). This may be a warning signal of over-exploitation or simply an indication of a developing fishery. This issue will remain unresolved without a reference level for exploitation (e.g., MSY). Time

Table 3. GLM results for Guinea Bissau including model specification and results as well as regression coefficients for each factor level. Model = log (std. catch) ~ year + zone + d.strata + c.value + year:c.value.

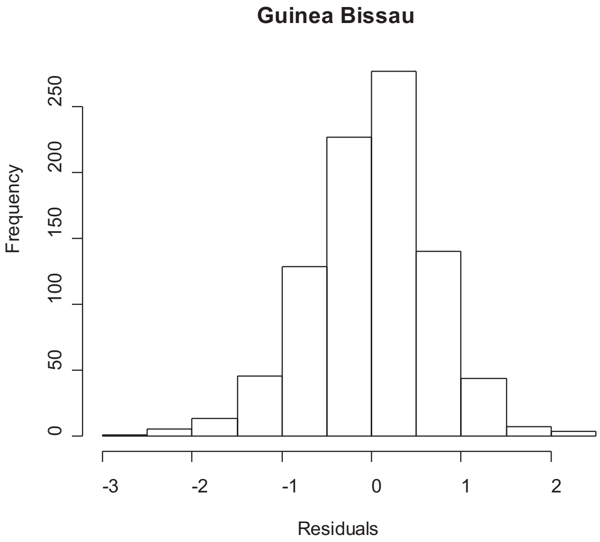
Coefficients	Estimate	Std. error	*t* value	Pr(>|*t*|)
(Intercept)	3.387	0.151	22.48	0.000
year_1964	–0.015	0.211	–0.07	0.944
year_1981	0.409	0.222	1.84	0.066
year_1986	–0.111	0.193	–0.58	0.565
year_1988	–0.671	0.191	–3.52	0.000
year_1989	–0.492	0.163	–3.02	0.003
year_1990	–0.151	0.160	–0.95	0.344
year_1991	–0.549	0.191	–2.87	0.004
year_1992	–0.658	0.178	–3.71	0.000
year_1995	–0.800	0.166	–4.83	0.000
zone_south	0.290	0.053	5.47	0.000
d.strata_depth2	0.418	0.055	7.56	0.000
d.strata_depth3	0.711	0.065	10.94	0.000
c.value_ncom	–0.675	0.202	–3.35	0.001
year_1964:c.value_ncom	0.185	0.301	0.61	0.539
year_1981:c.value_ncom	–0.568	0.327	–1.74	0.082
year_1986:c.value_ncom	0.221	0.271	0.82	0.415
year_1988:c.value_ncom	0.856	0.269	3.18	0.002
year_1989:c.value_ncom	0.283	0.229	1.24	0.216
year_1990:c.value_ncom	0.537	0.225	2.38	0.017
year_1991:c.value_ncom	0.472	0.270	1.75	0.081
year_1992:c.value_ncom	0.564	0.251	2.25	0.025
year_1995:c.value_ncom	0.002	0.231	0.01	0.993

Residual std. error: 0.6981 on 872 d.f.

Multiple *R*-squared: 0.3998; adjusted *R*-squared: 0.3847.

F-statistic: 26.4 on 22 and 872 d.f.; *P*-value: < 2.2e–16.

The model includes the factors: year (Table 1), zone (north and south), d.strata (depth1: ≤ 30 m; depth2: 30-80 m; depth3: ≥ 80 m), and c.value (commercial value: com and ncom). Also, the model includes an interaction term (year:c.value). Coefficients express the difference between each level of the factors and the first level. Mean abundance of demersal fish expressed as log kg per nm^2.



series analysis (MAFA) of catch data showed that there was a shift in the composition of fishery catches. Catches (and CPUE) of important species such as yellowfin and skipjack tuna have decreased, while they have increased for small pelagics and neritic tuna (Fig. 7).

The MAFA results were useful in providing a general pattern over time for the multispecies fisheries data in Cape Verde (Fig. 8). Also important was the fact that the correlation coefficients in relation to environmental variables were not significant, implying that fishing was driving the changes. However, it is important to point out that species such as yellowfin and skipjack tuna have ocean-wide distributions and the observed decreases in catch cannot be interpreted as the local effects of fishing in Cape Verde (e.g., ICCAT 2001). A recent modeling study showed that the decrease in abundance of important tuna predators led to less predation on small pelagics and neritic tuna, in particular, which resulted in biomass increases (Stobberup et al. 2004). The same study questions also the apparent decrease in biomass of demersal fish estimated by trawl surveys as it was not compatible with observed catches. Further research is necessary to deal with these conflicting results.

The MDS results for Guinea Bissau showed that there was no apparent change over time in the structure of demersal fish communities. In contrast, depth and area appear to play an important role in defining demersal fish community structure in terms of species composition and abundance (Fig. 9), which is consistent with a previous study covering a

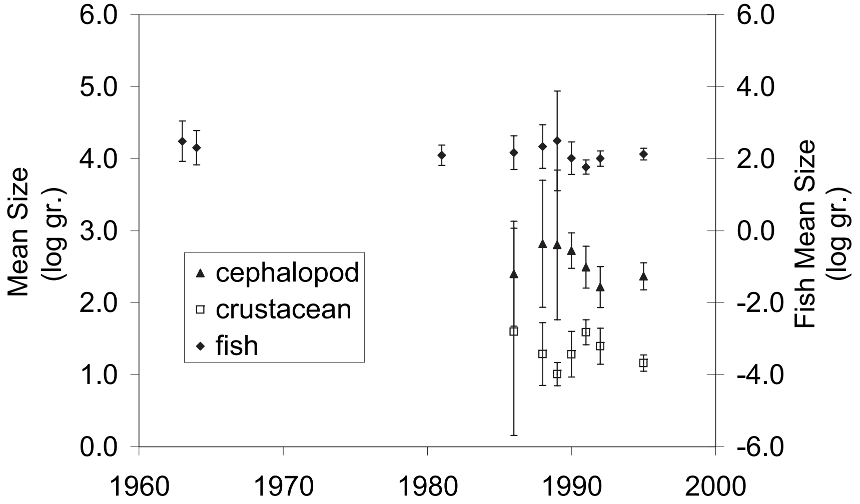


Figure 12. Mean size of commercial fish, cephalopod, and crustacean species groups, calculated as weighted averages over time. Error bars represent ±SE (standard error). Note the different scales on the y-axes, which were used to avoid clutter.

shorter period from 1988 to 1995 (Amorim et al. 2004) as well as for the region (Longhurst and Pauly 1987).

On the other hand, the decrease in abundance of commercial and noncommercial demersal fish as well as a decrease in mean size of fish was consistent in showing change for 1989-1992, particularly in the case of mean size (Figs. 11, 12). A comparison with the estimated catches for the same period indicates that this appears to be a result of intensive fishing during this period (Fig. 3).

These results were consistent with the changes observed in the *W* statistic, showing change in the relationship between abundance and biomass (Fig. 10). The increase in *W*, particularly in 1988, was the result of higher dominance by *Balistes* sp., which are relatively large individuals. This was a regional phenomenon where high abundance of *Balistes* sp. occurred during the 1980s. The decrease in *W* from 1989 to 1992 was maintained, however, even when *Balistes* sp. was excluded. Such a decrease in *W* would be the expected result of a removal of larger individuals by the fisheries as well as an increasing abundance of smaller sized individuals.

A further attempt was made to address whether these changes in abundance and mean size of species groups were related to fishing or environmental factors in Guinea Bissau. These results were rather disappointing in that there were only a few significant correlations. Most importantly, this was the highly significant negative correlation between fishing effort (industrial) and the mean size of commercial demersal fish. Thus, the effects of fishing and the environment were not clear, but it is important to point out that the lack of continuous time series is a serious limitation for this type of approach.

The effects of fishing appeared to be clearer in the case of Guinea Bissau, which is consistent with the fact that industrial trawlers operate in these waters. It was interesting to note that the perturbation signal observed in *W* and mean size disappeared when fishing intensity decreased (Figs. 3, 10, 12). However, fishing effects were also found in Cape Verde and the MAFA analysis of catches showed that an important shift has occurred in the proportions of pelagic and demersal species. The decreases in catches of tuna species has to be dealt with on the regional or ocean scale, but the effects of fishing on demersal species has to be monitored closely in order to make proper assessments of exploitation level (e.g., Marques 1999, Stobberup et al. 2004). The present study showed several approaches that can be used to study fishing effects, which become clearer when a comparative approach is used. However, the usefulness of ecosystem indicators in fisheries management is still a developing field of research and we hope to have contributed for tropical areas, in particular.

Acknowledgments

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Evaluating Marine Ecosystem Restoration Goals for Northern British Columbia

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Abstract

Using the Ecopath with Ecosim (EwE) framework, we employ historical models of northern British Columbia marine ecosystems corresponding roughly to the years 1750, 1900, 1950, and 2000 to assess them as possible restoration goals. We use a policy optimization routine to identify fishing patterns that maximize economic, social, and ecological benefits from the restored historic systems. The ecosystem models are subjected to simulated harvest under optimal fishing plans, and the most beneficial scenarios are identified through various economic, social, and ecological indices. Knowing what a restored system may be worth to stakeholders could help us to justify the costs of whole ecosystem restoration. The 1750 ecosystem emerges as the most desirable restoration goal, owing to its large biomass of valuable target species. It is able to deliver the greatest sustainable benefits in terms of fisheries rent and employment, while sacrificing less biodiversity per dollar harvested. The 1900 period is slightly less attractive in all regards. The 2000 system offers superior benefits to 1950 in terms of potential rent and jobs, though not biodiversity. Ecosystem models have data deficiencies, and parameter uncertainty can compromise optimal harvest predictions. The problem is amplified when we reach into the past, where data for even the most visible species may come from anecdotal accounts. EwE provides a capacity to deal with data uncertainty; here we test ecosystem effects of our optimal harvest policies given unsure initial biomass estimates. Poor quality input data and/or heavy exploitation rates lead to large variations in the predicted structure of the ecosystem following optimal harvests.

Introduction

Modeling the Lost Valley

Four Ecopath with Ecosim models (EwE; Walters et al. 1997, Christensen et al. 2000) have been developed to represent the marine ecosystem of northern British Columbia, Canada (Department of Fisheries and Oceans, Pacific Region, statistical areas 1-10) at different periods in the past. The models, modified from Ainsworth et al. (2002), represent a "best guess" of what the historic ecosystems may have looked like. The historic periods were chosen to represent distinct eras in the development of west coast fisheries. These are: prior to European contact (c. 1750); before the introduction of steam trawlers (c. 1900); during the heyday of the Pacific salmon fishery (1950); and the present day (2000).

Under the Back to the Future approach (BTF; Pitcher 2001, 2004), each period is evaluated as if it might represent a possible restoration goal for the future. To this stage, BTF intentionally steps over the question of how to achieve restoration and instead focuses on choosing an end goal. This avoids the contentious issue of allocation that would accompany a restoration plan and allows us to engage disparate parties, such as scientists, managers, and industry, in the necessary first step: achieving an agreed upon long-term goal. The restored periods are therefore referred to as "Lost Valley" scenarios (Pitcher et al. 2004), as if one has discovered an almost pristine system that is ready to be harvested in a sustainable and optimal way.

In this paper, we use Ecosim dynamic simulations to predict the response of historic ecosystems to optimal harvests, and use a variety of indicators to describe the trade-offs between socioeconomic benefit and ecological maintenance that are inherent in various optimal fishing plans. Comparing the sustainable benefits predicted from each historic system allows us to estimate the relative worth of these periods. Knowing what a restored system may be worth to stakeholders, in a variety of terms, may help justify the costs of restoration.

Ecosystem models always have data deficiencies, and the problem is amplified when researchers try to quantify historic systems. Models of the distant past must rely on anecdotal information (A. Erfan, University of British Columbia Fisheries Centre, Vancouver, Dec. 2002, pers. comm.) or use unconventional data sources (Ainsworth and Pitcher 2005, this volume). However, despite large data gaps, we can use EwE's mass-balance approach to infer missing information, and judge the compatibility of available data to determine whether reports of past abundances are congruent, or at least possible within the trophic constraints of the ecosystem. However, multispecies models like EwE can be sensitive to initializing parameters (Hollowed et al. 2000), and uncertainty surrounding input data can carry major implications for our harvest policy recommendations. In this report we predict a range of possible outcomes from our optimal harvest simulations of the Lost Valley ecosystems, considering uncertain input biomass information.

Methodological challenges in BTF

Constructing static models of historic ecosystems presents some difficulties (e.g., reconciling incompatible reports of predator/prey abundance, developing a historic diet matrix based on current stomach content data, estimating prehistoric human consumption). These problems are not intractable, but simplifying assumptions must be made (see review by Pitcher 2001). When we move into dynamic simulations, additional challenges arise, such as choosing an appropriate trophic flow model (consumer versus donor driven). With models of the recent past, time series catch per unit effort and biomass data can be used to tune dynamic behavior (e.g., Cox et al. 2002, Stanford 2004), but with models of the ancient past, we typically assume similar behavior to recent ecosystems.

However, the most common objection to the BTF approach is that marine ecosystems may not "rewind" to historic states because the climate has now changed. In the North Pacific and elsewhere, climate shifts affect the biomass, location, and identity of species harvested by fisheries (e.g., Ware 1995, Barenge 2002, Chavez et al. 2003) and cause changes in the species assemblage (McFarlane and Beamish 2001, Benson and Trites 2002). These are important factors to consider in the BTF restoration approach, and in forecasting the effects of climate change generally. EwE can assist us in this regard, and we are learning how to use climate forcing to challenge optimal harvest predictions made from these historic systems.

EwE can represent climatic forcing factors on primary production and other parameters (e.g., stock-recruitment relationships), and previous authors have modeled the effects of variable ocean regimes. Pitcher and Forrest (2004) challenged the 1750 northern British Columbia model with transformed temperature data from tree rings (Gedalof and Smith 2001) to show that large changes in coho salmon and juvenile halibut emulate those expected by the literature (Mantua et al. 1997, Clark et al. 1999). This work demonstrates that historic ecosystem changes can be emulated using equilibrium models, when they are driven by environmental and anthropogenic influences.

A more recent study tests predictive forecasts, like those provided by BTF policy scenarios. Effects of future climate variability were determined through an ecosystem-based population viability analysis for fish populations in Lake Malawi (T. Pitcher and E. Nsiku, University of British Columbia Fisheries Centre, Vancouver, June 2004, pers. comm.). Primary productivity was driven by randomized data obtained from biogenic silica deposits (Johnson et al. 2001). An equivalent examination of the 1750 northern British Columbia model predictions is currently under way, where future variation in primary productivity is estimated using randomized dendroclimatic records. Micropaleological evidence of fish abundance (e.g., Finney et al. 2002, Walker and Pellatt 2003) also could be used to validate predictions made from historic models, though this has not yet been attempted. Finally, temperature-induced influx of warm water species can be represented in Ecosim; for example, hake in southern British Columbia (D. Preikshot, University of British Columbia Fisheries Centre, Vancouver, June 2004, pers. comm.) and for the "Russell cycle" (Russell 1935, Southward et al. 1995) in the English Channel (Stanford and Pitcher 2004).

Major historical changes to ecosystems have been derived from fishing activities (Jackson et al. 2001, Myers and Worm 2003, Christensen et al. 2003), so driving Ecosim with both fishing and climate may emulate reality. Unfortunately, the full extent to which climate affects marine populations is not yet understood (Parsons and Lear 2001), and if the effects of climate variation on marine populations are nonlinear (e.g., Hare and Mantua 2000) then simple representations may prove to be inadequate. However, this problem is not unique to ecosystem models; it may also affect conventional stock assessment (Hofmann and Powell 1998), although most fishery quotas are thought to work reasonably well, regardless.

Methods

Ecosystem models

The historic Ecopath models are based on Ainsworth et al. (2002); they include information from archival scientific sources, archeological and historical records, and traditional ecological knowledge. To aid comparison between periods, the ecosystem models developed for this report use a similar aggregate structure, although state and rate parameters vary considerably. The earliest historical period, 1750, is modeled with the highest abundances in most commercial and noncommercial functional groups, the greatest biodiversity, and the lowest rates of production and consumption per unit biomass. The 1900 and 1950 systems are progressively depleted in biomass and biodiversity, but the 2000 system sees a slight recovery in some valuable target groups (mainly herring, some invertebrates, halibut, and other flatfish).

Ainsworth and Pitcher (2004) parameterized the models for temporal simulations in Ecosim and modified the basic structure to include an idealized fleet, called the Lost Valley fleet. We use that fleet structure for this exercise. This hypothetical fleet configuration is not based on past or present real-world fleets, but is designed according to rational criteria laid out in Pitcher et al. (2004). The Lost Valley fleet consists of traditional and contemporary fisheries, which have been modified to prevent habitat damage and permit only minimal bycatch/discards (insofar as technological amenities and responsible fishing practices could conceivably permit).

It offers an alternative to the destructive real-world fleet, which we can use to sustainably harvest the Lost Valley.

Harvest simulation

To evaluate each historic period as a restoration goal, we simulate harvests under an optimal fishing plan, and quantify benefits in diverse terms—using economic, social, and ecological indicators developed for this approach. The optimal fishing policy for each period was determined by conducting gaming scenarios in Ecosim, using the policy search routine (Walters et al. 2002). The routine iteratively varies the fishing mortality (F) per gear type to maximize simulation benefits over the time horizon. Running the harvest simulation repeatedly, and using a Davidon-Fletcher-Powell search algorithm, the routine probes an N dimensional response surface (where N is the number of gear types) for the optimal fleet configuration that maximizes the harvest objective function.

The search routine supports five harvest objectives. The economic objective maximizes profit from the system, where profit is the catch value minus the cost of fishing; the social objective maximizes employment, which is directly proportional to effort; the ecological objective increases the abundance of long-lived functional groups by maximizing biomass/production of the system (unfished ecosystems have been characterized as having many large slow-growing animals) (Odum 1969); the mixed objective combines these priorities; the portfolio log-utility function selects risk-averse policies by penalizing changes in the ecosystem far from baseline (C. Walters, University of British Columbia Fisheries Centre, Vancouver, 2002, pers. comm.).

Using optimal fishing mortalities delivered by the policy search, we conduct a 50-year dynamic harvest simulation, and then maintain the final equilibrium level of harvest for an additional 50 years. Total harvest duration is therefore 100 years. We use three socioeconomic indices to evaluate the harvest profile: intergenerational net present value (NPV), total employment, and employment diversity. Intergenerational NPV summarizes the continuous flow of benefits from the harvest profile into a single term. It uses a standard discount rate ($\delta = 0.04$) to weight early benefits more heavily than delayed benefits in the NPV term, as does the conventional discounting, but NPV also applies an intergenerational discount rate to value benefits separately that are destined for future generations (δ_{fg}) (Sumaila et al. 2001, Ainsworth and Sumaila 2003a). Total employment is measured by number of jobs, and employment diversity (Ainsworth and Sumaila 2003b) measures the evenness of effort distribution across gear types, based on the Shannon-Weaver entropy function (Shannon and Weaver 1949). The ecology of the ecosystem during and after harvest is monitored using a biodiversity indicator adapted for Ecosim by Ainsworth and Pitcher (2003): the Q-90 statistic. It is based on Kempton's Q index (Kempton and Taylor 1976), and measures the slope of the cumulative log-species abundance curve between the 10th and 90th percentiles.

Verification of the optimal policy

By default, the policy search routine in Ecosim uses baseline-fishing mortalities per gear type inherited from Ecopath to initialize the optimization. Results presented in this report have used this method, but to verify that the procedure has located the best fishing solution we conducted 25 additional runs using random *F* starting points. The occurrence of two or more clusters of optimal solutions indicates that multiple local maxima are present.

Measuring the effects of parameter uncertainty

A Monte Carlo routine in Ecosim is used to test the consequences of unsure input data on the dynamic simulation. Baseline Ecopath (input) biomass is varied for each functional group under a uniform probability distribution, according to a coefficient of variation (CV) that depends on data quality—reliable data points are varied by a smaller degree than unsure data points. We use Ecosim's data pedigree as an index of data quality (Christensen et al. 2000); it is a subjective ranking between 1 and 6, where 1 indicates poor data quality (e.g., parameter estimated internally by EwE) and 6 indicates high quality (e.g., sampling based, high precision). The CV ranges from 0.4 for poor quality data points to 0.05 for high quality data. Initial biomasses of 54 functional groups are varied 10 times for each simulation ($n = 540$). We examine the end-state ecosystem biomasses after 50 years of optimal harvest, and quantify variation in light of baseline model uncertainty.

Results

Socioeconomic benefits of restoration

Figure 1a shows the intergenerational NPV of the optimal harvest plan for all restoration periods, under each of five harvest objectives. The 1750 system produces the most valuable harvest profiles under all harvest objectives, owing to its high starting biomass. The 1900 system consistently offers the second best alternative, followed by 1950 and 2000. The rank order is significant (Table 1).

Figure 1a also demonstrates the effect of the harvest objective. To maximize total employment, the social objective increases effort by fishing uneconomically (disregarding costs of fishing). Under the social optimization the policy search tends to identify fleet configurations that support a large recreational sector. This amounts to augmenting gear types that directly or indirectly discourage predators and competitors of the recreational target species, through harvest or bycatch (mainly shrimp trawl, salmon wheel, and shrimp trap) while restricting gear types

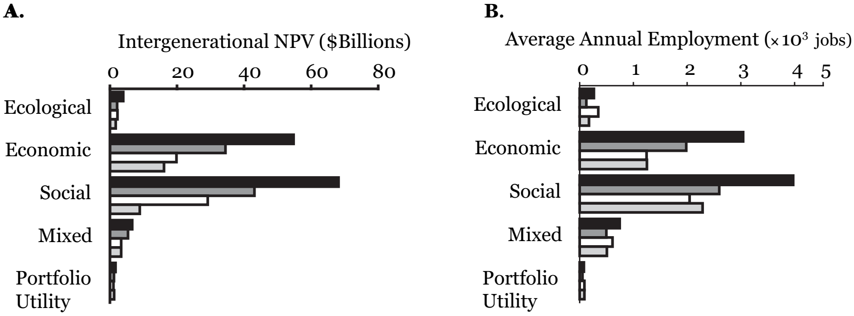

Figure 1. Economic (A) and social (B) benefits generated from each Lost Valley restoration goal over 100 years of harvest. Black bars show 1750; dark gray bars show 1900; white bars show 1950; light gray bars show 2000.

that compete with the recreational sector (primarily halibut longline). The next greatest intergenerational NPV occurs under the economic harvest objective, which seeks to maximize profit, followed by the mixed, ecological, and portfolio log-utility objectives.

In terms of the average number of jobs created per year, by restoration period and harvest objective (Fig. 1b), the social harvest objective generates the most jobs followed by the economic, mixed, ecological, and portfolio log-utility objectives (Table 1). As with the economic valuation, the 1750 system offers the best potential for employment followed by 1900. However, unlike the economic valuation, the 2000 system outperforms the 1950 system in average annual employment. While the 1950 system can generate large revenues through a massive commercial salmon industry, the 2000 system can sustain more jobs because of its large recreational sector, capitalizing on a higher biomass of halibut.

Employment diversity

Table 1 shows the rank order of employment diversity (*D*) in each optimal harvest simulation. The social and economic harvest objectives tend to concentrate fishing in a few profitable sectors, whereas the ecology and portfolio log-utility objectives spread fishing effort more evenly across sectors. The latter objectives therefore score high on the employment diversity index, but generate fewer jobs since they advocate lower harvest rates.

Profit per functional group

With 1750 as the restoration goal, our simulations suggest that we should be able to take as much as ten times the current real-world annual

Table 1. Rank order of historical periods and harvest objectives in their ability to generate economic, social, and ecological benefits. All values of Kendall concordance (*W*) are significant ($P < 0.05$). Economic valuation shows intergenerational NPV of optimal harvest plans; social valuation measures relative number of jobs and employment diversity (*D*) after 50 years of harvest; the ecological valuation measures biodiversity (*Q*-90).

		Economic valuation	Social valuation		Ecological valuation
		Intergenerational NPV	Jobs	Employment diversity	Biodiversity
Period	1750	1	1	3	1
	1900	2	2	4	2
	1950	4	4	1	3
	2000	3	3	2	4
	W ($\alpha = 0.05$)	0.17	0.17	0.17	0.17
	W^a	0.81	0.46	0.73	0.9
Objective	Ecology	4	4	1	2
	Economy	2	2	5	4
	Social	1	1	4	5
	Mixed	3	3	3	3
	Portfolio log-utility	5	5	2	1
	W ($\alpha = 0.05$)	0.24	0.24	0.24	0.24
	W^a	0.93	0.96	0.82	0.83

[a] Kendall concordance (*W*) (Kendall 1962).

revenue from the ecosystem, and this would be sustainable over the 50-year time horizon. This ceiling limit of harvest is identified by the exploitative social objective (Fig. 2). Most of the benefit of restoration would be realized in the shellfish fishery, but improved harvests of coho salmon and lingcod would also contribute to revenue. However, that high level of harvest would require a large sacrifice in biodiversity from the pristine state. With 2000 as our restoration goal (i.e., no restoration), the social optimization suggests that we could retrieve greater sustainable profits than are currently realized in ratfish, halibut, piscivorous rockfish, and lingcod. However, under that plan, harvests in most other traditional target groups would need to be sacrificed (e.g., 13% reduction in coho value from current levels; 86% reduction in chinook).

Under the intermediate harvest strategy suggested by the mixed objective function, few groups would provide any increase in profit over real-world levels, even under the abundant 1750 baseline. If conservation

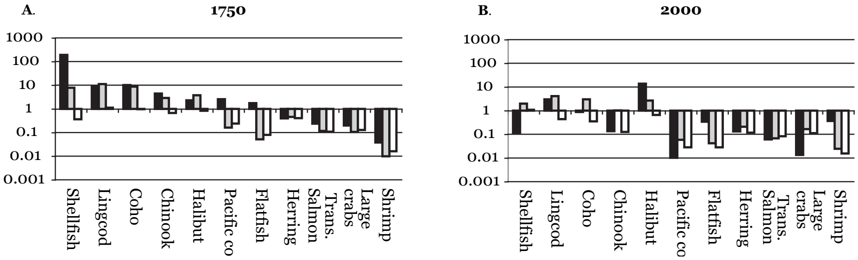

Figure 2. Profits per functional group made available by the most ambitious restoration goal (1750) and the least ambitious (2000) as a fraction of the current real-world profit taken from the study area. Profits per functional group represent the annual average. Dark bars show the upper limit on harvest value sustainable over 50 years (determined under the social harvest objective); gray bars show an intermediate solution (mixed objective); and white bars show the most conservative strategy (portfolio log-utility objective). Results from the 1900 and 1950 periods are not shown, but they are qualitatively similar to 1750 and 2000, respectively. Transient salmon includes pink, sockeye, and chum salmon.

is the priority, harvesting the restored system may continue under a conservative strategy such as the one advocated by the portfolio log-utility optimization. With 1750 as our restoration goal, only lingcod could provide a greater return than is presently realized, while the 2000 system can offer no improvement in harvest revenues. This suggests that there is currently little margin to increase real-world profits over their present levels, and maintain any semblance of sustainability. In fact, real-world profit may be expected to drop if the current harvest regime proves to be less sustainable than under the 50-year time horizon simulated by these optimal solutions.

Ecological valuation

Figure 3 illustrates ecosystem biodiversity throughout the course of the dynamic harvest simulation. The 1750 system outperforms the more depleted 1900, 1950, and 2000 systems for maintenance of biodiversity, despite allowing greater harvests under each optimal plan. The portfolio utility objective maintains biodiversity better than the ecology, mixed, economic, or social options because it advocates the lowest exploitation rates. The ecology objective (not shown) also maintains high biodiversity, although its designed purpose is to increase the abundance of long-lived species.

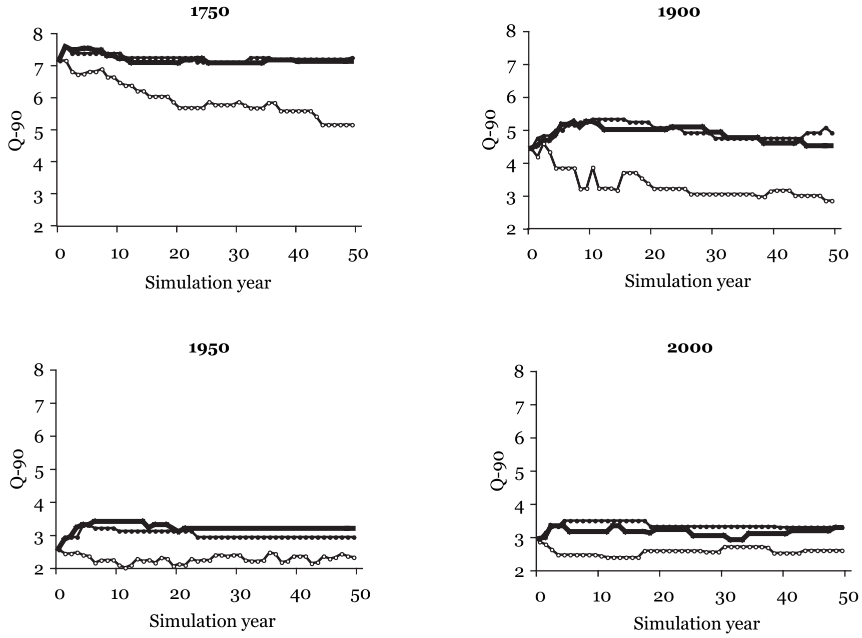

Figure 3. Ecosystem biodiversity (*Q*-90) over 50-year optimal harvest scenarios. Open circles show exploitative social objective; solid lines show moderate mixed objective; closed circles show conservative portfolio log-utility objective. High values of *Q*-90 indicate high biodiversity.

Effect of period and harvest objective on valuation indices

Table 1 shows the rank order of each period and objective in its ability to generate harvest scenarios that produce the greatest economic, social, and ecological gains. The pre-contact period has the most to offer as a restoration goal. It is able to sustain high landings, generating wealth and jobs, while preserving biodiversity after 50 years of harvest better than the other periods. However, the distant past periods (1750 and 1900) have optimal fishing strategies that include a heavy concentration of fishing in a few gear types, capitalizing on the pristine biomass of valuable species. After 50 years of harvest, the 1950 period maintains greater biodiversity than the 2000 period, despite beginning at a lower level. This is because the 1950 period permits an optimal policy that includes more diversity in fishing effort, depleting several functional groups evenly, while the 2000 system relies on a heavy recreational sector (with exploitation-minded strategies) or the invertebrate fishery (with conservation-minded strategies) that specialize in fewer target groups.

Simulations that optimize the social objective generate the greatest profit and jobs, followed by economic, mixed, ecology, and portfolio log-utility runs. However, the inverse is true for biodiversity—the conservative portfolio log-utility objective performs the best and the exploitative social objective performs the worst. Conservative objectives also support high employment diversity, whereas exploitative simulations concentrate fishing in profitable sectors.

Verification of optimal strategy

For any restoration period and harvest objective, the fleet-effort plans delivered by the policy search routine should ideally converge on a global maximum of the response surface, regardless of what fishing mortalities are used to initialize the search. However, if local maxima are present, the search may stall on a sub-optimal peak. Replicate optimizations using random *F* starting points will confirm whether we have identified the global maximum. Figure 4 uses multi-dimensional scaling (MDS) to represent response surface geometry of three example scenarios: ecological optimizations for the 1950, 1750, and 1900 periods. These examples were chosen because they represent three characteristic response surface geometries: a clearly defined peak (global maximum), several peaks (multiple local maxima), and a loosely defined peak (plateau). MDS reduces the 11 variable gear types to two dimensions; clustering of points indicates similarities in fleet-effort configuration.

In light of data and management uncertainties, the plateau response surface indicates a more robust harvest strategy than either of the two other geometries. In allocating fleet effort, we need only to approximate the optimal configuration, since falling anywhere on the surface of the plateau will generate near-maximum benefits. By contrast, if our fishing policy aims for a sharp peak on the response surface, and falls short, then much of the intended benefit will not be realized.

Kendall's test for concordance indicates that harvest objective has little effect on the clustering of random *F* solutions ($W = 0.09$; $W_\alpha = 0.24$) (Kendall 1962). The period, however, did affect the pattern of clustering ($W = 0.21$; $W_\alpha = 0.17$), with 1950 generating the tightest clusters of optima followed by 2000, 1900, and 1750. This suggests there may be fewer ways to efficiently harvest a depleted system. In recent past models (1950 and 2000), the optimal fishing pattern was always conservative to allow the system to rebuild, so the variety of possible fishing strategies was constrained by scale. In distant past models, where all functional groups are of harvestable biomass and "fair game" to the policy search, multiple fleet-structure options exist. For example, this occurs if the structure of the ecosystem can be manipulated to permit large landings on target group A or B, but not both. If these target groups have associated with them a contradictory set of supporters (e.g., prey) and detractors (e.g., predators and competitors), then the policy search must decide between mutually incompatible solutions that yield comparable benefits.

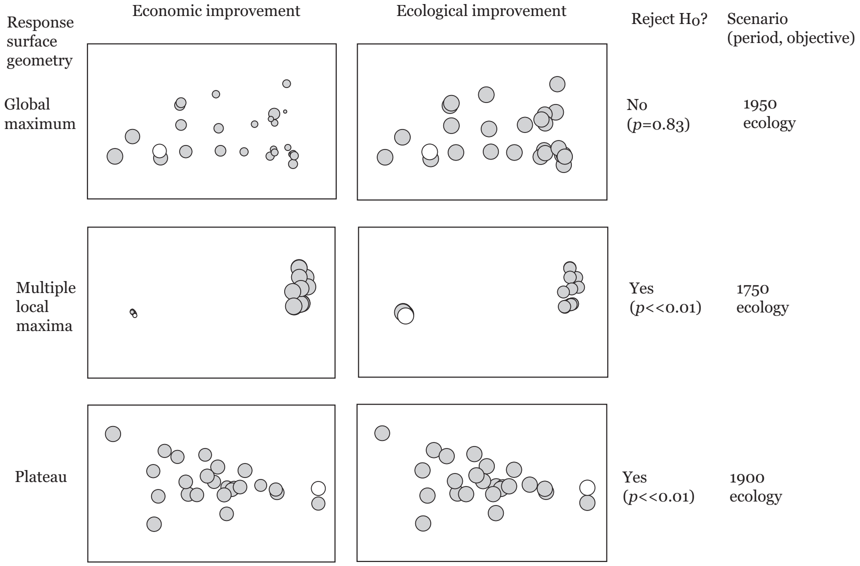

Figure 4. MDS representation of the policy search response surface showing relatedness of 25 optimal fleet configurations based on random *F* starting points. Three possible response surface geometries are illustrated: a global maximum containing a tight cluster of optimums (high ANOVA *P* value); multiple local maxima containing two or more dissimilar clusters (low *P* value); a plateau containing a loose cluster (low *P* value). Diameter of circle indicates degree of economic or ecological improvement over baseline. White circles show optimization used in the present analysis (i.e., for evaluation of harvest benefits) and gray circles show random *F* replicates. NB = an outlier was removed from the multiple local maxima example for clarity.

Examining trade-offs

Figure 5 shows the trade-off between exploitation and conservation for each base period. To the left of each graph are harvest plans (including fleets and objectives) that generate large revenue from the restored system at the expense of biodiversity: these are typically social and economic optimizations. On the right are plans that preserve biodiversity well but neglect resource users, most often portfolio log-utility and ecological optimizations. Between these extremes are policies found by the mixed objective. They provide an intermediate trade-off between money and biodiversity. The rank order of the harvest objectives along this spectrum is significant (Table 1).

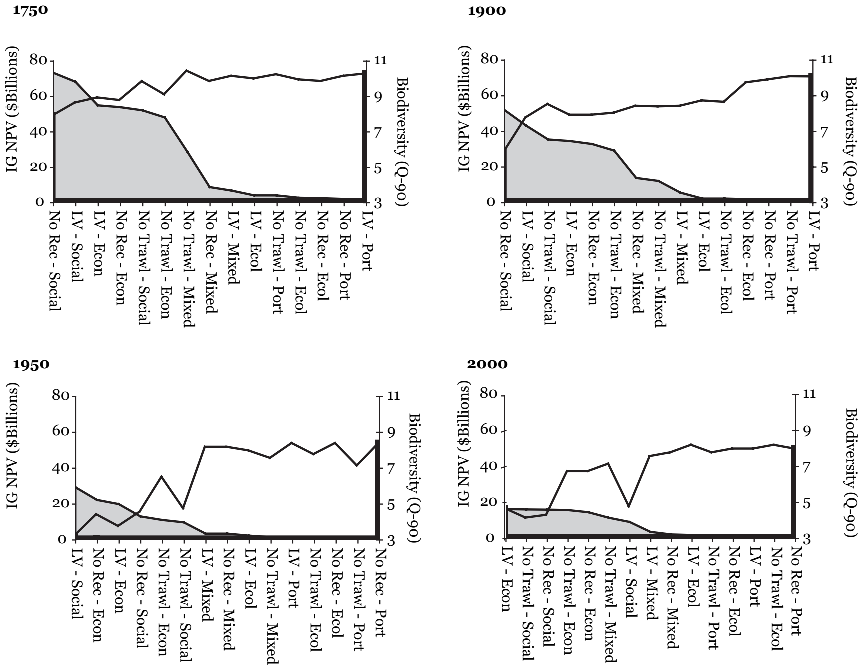

Figure 5. Economic versus ecological trade-offs in optimal harvest plans of restored ecosystems. Along the X-axis are 15 optimal harvest strategies per period, which vary harvest objective (social, economic, mixed, ecological, and portfolio log-utility) and fleet composition (LV = idealized Lost Valley fleet; No Rec = recreational gear removed; No Trawl = trawlers removed). Shaded boundary shows intergenerational NPV, black line shows equilibrium biodiversity established after dynamic harvest simulation (50 years). Policies that generate large revenues tend to sacrifice biodiversity (left side of graphs), whereas policies that maintain biodiversity provide less revenue (right side).

A similar trade-off between exploitation and conservation occurs in all periods, although the absolute level of NPV and end-state biodiversity is reduced as the restoration goal approaches the current system. Under its most lucrative harvest plan, the 1750 period generates a greater NPV than later periods ($73 billion over 100 years of harvest). However, biodiversity is sacrificed for that revenue; the end-state *Q*-90 value of the 1750 system is reduced 2.1 below the pre-contact baseline. The maximum revenue of the 1900 period is $52 billion and biodiversity drops 4.1 below baseline. The 1950 period can produce $29 billion for a biodiversity loss of 4.5, and the 2000 system can produce $16 billion for a biodiversity loss of 4.0. Large harvests can be obtained from an ecosystem with the characteristics of 1750 with a small loss in biodiversity, but depleted ecosystems must accept a greater loss in biodiversity to supply their maximum revenue.

The effect of data uncertainty on the optimal policy

Figure 6 compares the optimal fishing mortalities of each harvest objective (system average weighted by functional group biomass) with the coefficient of variation seen in the end-state biomass for the 2000 period (averaged across functional groups). Large fishing mortalities, such as those advocated by the economic objective, will result in wide variations of end-state biomass. Large variations in end-state biomass are also observed under the social objective, but its high exploitation rate drives some vulnerable functional groups to extinction (under any permutation of the initial group biomass profile). This reduces the overall variability (Fig. 6b). More conservative fishing policies, like those devised by the mixed, ecological, and portfolio-utility objectives, change the system very little from baseline. System dynamics are restrained, and ecosystem end-state after harvest becomes more predictable in the face of uncertain parameter estimates.

Figure 7 shows the variability in end-state biomass for each restoration year. The economic harvest objective was selected for this example because it generates the greatest spread in output biomass under resampling. Variations in the ecosystem end-state reflect input data quality; models of the distant past (1900 and 1750) are built on comparatively unsure biomass estimates (Fig. 7a) and yield a wide range of possible outcomes. The 1950 system shows anomalously low end-state variation across functional groups. This probably reflects a model inadequacy. The model seems too resilient to fisheries effects, although this attribute could be adjusted using Ecosim's vulnerability parameters.

Discussion

Choosing a restoration goal

In all respects, a marine ecosystem with characteristics similar to those of 1750 emerges as the most desirable restoration goal. It can deliver greater social and economic benefits than the other historical periods, while sacrificing less biodiversity. In terms of profit potential, the greatest incentive to restore the pre-contact ecosystem lies in the re-establishment of abundant bivalve populations. The same is true for 1900; social and economic optimization of fisheries requires large invertebrate harvests. Under harvest plans that stress environmental concerns, in which excessive

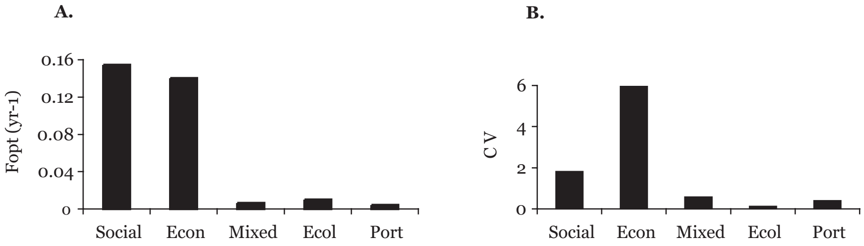

Figure 6. Variability in end-state biomass after 50 year dynamic harvest simulation for each harvest objective, given uncertain input biomass data: Monte Carlo results for the 2000 model. A. Average optimal fishing mortality identified by policy search, weighted by functional group biomass; B. Coefficient of variation (CV) of end-state biomass.

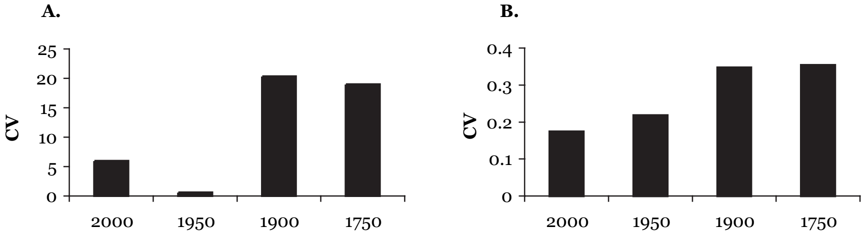

Figure 7. Variability in end-state biomass after 50 year dynamic harvest simulation for each period, given uncertain input biomass data: Monte Carlo results for the economic objective. A. Average coefficient of variation (CV) of input biomass, weighted by functional group biomass; B. CV of end-state biomass.

depletion of the system is undesirable, the 1750 and 1900 ecosystems instead become modified to sustain a large recreational sector.

In contrast, models of the recent past rely on the recreational sector to generate wealth and employment under social and economic objectives. As with the ecological harvest objective, when applied to models of the distant past, the optimal policy uses the recreational sector to maintain benefits while causing a minimum of disturbance. When large landings are acceptable, such as under the social and economic objectives with models of the distant past, then mining the resource becomes the better strategy and commercial fisheries are favored by the policy search. This is particularly true under the economic objective because of discounting; distant benefits contribute little to the NPV term. Under the conservative discount rate applied in this report, however, the economic objective appears similar to the social objective (which does not apply discounting to value future job creation). Under the ecological and portfolio log-utility objectives, the models of the recent past require a small and diverse fleet to spread fishing pressure across a large number of functional groups, and allow rebuilding.

Generally, optimal harvest strategies based on the 1750 and 1900 restoration goals involve higher fishing mortalities than those based on the 1950 or 2000 systems, particularly when policies are optimized for social or economic benefit. This strategy removes biomass from the system faster than the replacement rate. Equilibrium is reached, but only after depleting the system to a more productive state, having removed larger individuals (i.e., increased production and consumption per unit biomass of groups) and reduced stock size. In contrast, harvest strategies based on the 1950 and 2000 systems use lower harvest rates to allow the ecosystem to rebuild. This strategy is utilized by both exploitative and conservative optimizations, indicating that restoration would serve both fisheries and conservation in these two depleted systems.

Considering the risks of data deficiency

Uncertainty in Ecopath biomass estimates can affect the outcome of the optimal harvest plan. The final condition of the ecosystem after harvest is least certain for models of the distant past, which are especially troubled by data deficiency. Harvest simulations that apply large exploitation rates can likewise have an unpredictable effect on the resulting ecosystem. This constitutes a double threat to the distant past restoration goals, whose pristine biomass begs for large harvests under any solution optimized for rent or jobs. But using conservative harvest rates, the range of output variability tends to be only a few percent of group biomass even for our most unsure models. Future work will consider the effect of data uncertainty on fleet configurations recommended by the optimal solutions.

Next steps

The tools demonstrated in this report offer a framework for decision analysis for attaining a specific goal of ecosystem structure. Although we have avoided the issue of how to achieve restoration in this report, upcoming work will address this question directly. C. Ainsworth and T. Pitcher (unpubl. data) have developed a new EwE routine to generate restorative optimal harvest plans. A desired ecosystem structure is entered into Ecosim's policy search routine, and an ideal fleet-effort configuration is determined that will selectively manipulate the ecosystem toward the specified end-state. After evaluating the costs of restoration (which will be associated with fleet restructuring and loss of revenue compared to status-quo exploitation), we will be able to determine the optimal rebuilding speed, and determine how far into the past we must reach in order to maximize cost effectiveness of restoration.

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Fisheries Assessment and Management in Data-Limited Situations
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Designing Fisheries Management Policies that Conserve Marine Species Diversity in the Northern South China Sea

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Abstract

This study suggests that, under data-limited conditions, dynamic ecosystem models can be used to explore fishery management policies that aim to conserve biodiversity. The northern South China Sea is a highly diverse ecosystem where rapid expansion of fisheries and limited management has depleted resources and threatened marine biodiversity over the past five decades. Designation of marine protected areas has been suggested to be an effective and risk-averse tool to conserve and restore marine biodiversity and fishery resources. Here, a dynamic ecosystem model (Ecosim and Ecospace) was constructed using information from sporadic surveys, historical records, literature, and global databases. Effects of a two-month seasonal trawl ban set up in 1998 by the People's Republic of China were evaluated using dynamic spatial simulations; results suggest that this is insufficient to sustain fishery resources and maintain biodiversity. Our work suggests that a combination of large marine protected areas and effective effort reduction would be necessary to maintain fishery resources and biodiversity. Our model provides quantitative support for the demand for stronger fishery management actions.

Introduction

The design and evaluation of fisheries management policies using conventional assessment tools have often been limited by the availability of data. The problem becomes more apparent for tropical fisheries because basic fisheries and ecological data are particularly scarce, and their highly

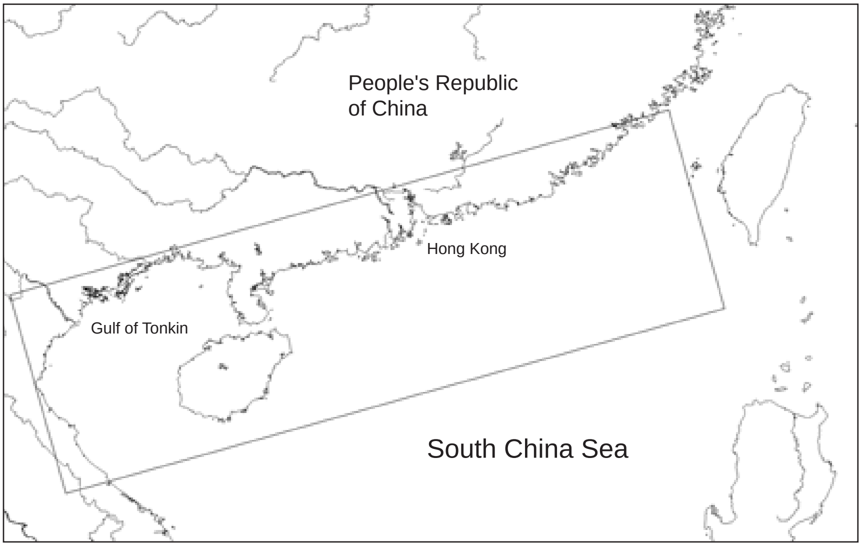


Figure 1. Map of the northern shelf of the South China Sea. The area within the rectangle was represented by the ecosystem model.

multispecies nature renders the use of conventional single species assessment tools almost impossible. Fisheries in the northern shelf of the South China Sea (NSCS) are typical examples. In these circumstances, a dynamic ecosystem modeling approach has been suggested (Pauly et al. 2000). Although the data-limited situation in the NSCS limits the accuracy of the parameters, results from an ecosystem model can be used to generate alternative hypotheses that can be tested by adaptive management experiments (Walters 1997, 2000; Walters and Martell 2004). Such models can also help identify important ecological indicators and critical information gaps, enabling more efficient use of limited resources for ecological monitoring and field studies.

Study area

We defined the NSCS as the continental shelf (less than 200 m depth) ranging from 106º53-119º48 E to 17º10-25º52 N (Fig. 1). The area falls within the Exclusive Economic Zone of the People's Republic of China and, therefore, fisheries resources were mainly exploited by Chinese fishing fleets. Over the past five decades, dramatic expansion, accompanied by mechanization and other technological advancements in fishing, has resulted in over-exploitation of the nearshore, and later, the offshore fisheries resources (Shindo 1973, Cheung and Sadovy 2004). Catch rates of Chinese trawlers in the NSCS dropped by more than 70% between 1986 and 1998 (Lu and Ye 2001). Analysis and modeling of landings data suggest a decline in trophic level of catch along the coast of NSCS (Cheung 2001, Buchary et al. 2003), agreeing with a general decline of mean trophic level of catch in the North Pacific region (Pauly et al. 1998). A range of species with high vulnerability to exploitation have been extirpated locally or regionally by fishing. For instance, the Chinese Bahaba (*Bahaba taipingensis*, Sciaenidae), endemic to the coast of China, has been rendered nearly extinct as a result of over-exploitation (Sadovy and Cheung 2003). Previously abundant red grouper (*Epinephelus akaara*, Serranidae), and other large reef-associated fishes in Hong Kong, have disappeared from commercial catches (Sadovy and Cornish 2000). Concurrently, total fishing effort has continued to increase, doubling in the past 10 years (PRC Department of Fishery 1996, 2000). Therefore, over-exploitation in the NSCS raises serious fishery management and biodiversity conservation concerns.

The Chinese fishery management authority has recognized the current poor status of fishery resources (Lu and Ye 2001), and has initiated a range of fishery management policies. These include limiting new entries to fisheries and prohibiting the use of some destructive fishing methods (He 2001). However, the degree to which regulations have been enforced has been questioned (Li et al. 1999). Since 1998, the Chinese authority has implemented a seasonal ban (June and July) on bottom trawling in the NSCS. So far, there have been no published empirical studies that evaluate the effect of the moratorium.

Marine protected areas (MPAs) have been advocated as tools to conserve biodiversity (e.g., Agardy 1994) and manage fisheries (e.g., Holland and Brazee 1996). We adopted the definition of an MPA provided by IUCN: “any area of inter-tidal or sub-tidal terrain, together with its overlying water and associated flora, fauna, historical, or cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment” (Kelleher and Kenchington 1992). Offspring produced from the protected stocks can migrate to surrounding unprotected areas and become available to fishers. Therefore, MPAs potentially can help maintain productivity and insure against stock collapse outside reserves by protecting larval production and their genetic quality, as well as protecting biodiversity and habitat (Bohnsack 1996). Small MPAs are likely effective in conserving resident species with low rates of movement (Botsford et al. 2003), and only large MPAs may help migratory stocks (Guénette and Pitcher 1999, Guénette et al. 2000). Stakeholder support is also suggested to be essential for MPAs to be effective (Agardy et al. 2003).

Currently available information generally supports the use of MPAs as management and conservation tools (Pauly et al. 2003, Lubchenco et al. 2003). Recent meta-analyses comparing the effects of existing marine reserves have suggested that density, biomass, individual size, and diversity of organisms increased after MPA establishment in historically fished areas (Halpern 2003). Empirical evidence on the effectiveness of MPAs in restoring biomass and enhancing fishery productivity elsewhere has also been documented (e.g., the Sumilon and Apo marine reserves in the Philippines; Russ and Alcala 1998, 1999; also see Roberts and Hawkins 2000 for other examples). Although an MPA's effectiveness in maximizing the sustainable yield of targeted species may be similar to more conventional fishery management tactics such as mesh size restriction (Botsford et al. 2003, Hastings and Botsford 2003), some of these conventional regulatory devices are difficult to implement in the NSCS. Considering the depleted situation of the NSCS marine ecosystem, establishing MPAs could be an effective approach for restoring fishery resources and conserving biodiversity. This has been demonstrated by a modeling study at a much smaller spatial scale for the marine ecosystem of Hong Kong (Pitcher et al. 2000, 2002; Buchary et al. 2003). However, the implications of MPAs at the larger scale of the NSCS have not been studied previously.

In this work, we explore the possible implications of a range of fishery management policy options in the NSCS marine ecosystem using Ecopath, Ecosim, and Ecospace ecosystem models (EwE: Walters et al. 1997, 1999, 2000; Christensen and Walters 2004). For depleted fisheries, such as those in the NSCS, improvement in fishery productivity for traditional food fish requires stock rebuilding (Pitcher and Pauly 1998, Pitcher 2001). At the same time, fishery management should also incorporate the conservation of nontargeted species. Therefore, here we compare the effectiveness of different policy options, including partially and fully protected areas, in achieving the policy goals of ecosystem rebuilding, maintenance of indigenous biodiversity, and reducing the risk of species loss. We focus on relative changes in the abundance of different functional groups in the study area, and the changes in biodiversity and risk of species loss as revealed by quantitative indicators. The year 1990 is used as a baseline for comparisons, primarily based on the availability of data. However, we do not propose that restoration of the 1990 ecosystem should be the policy goal for the management agencies in the NSCS but use this year only as a benchmark for comparisons.

Methods

Model structure and parameterization

A steady-state mass-balance model of the NSCS in 1990 was constructed using Ecopath. The model includes 41 groups composed of species of similar function and taxon in the ecosystem, ranging from phytoplankton, zooplankton, benthic invertebrates, demersal and pelagic fish groups, to marine mammals and birds. These functional groups in the ecosystem may be fished by seven different fishing fleets: (1) pair trawl, (2) stern trawl, (3) shrimp trawl, (4) gillnet, (5) purse seine, (6) hook and line, and (7) other gears. Model input parameters were estimated from commercial fishery catch and effort surveys, trawl surveys, empirical equations, local diet composition survey, and global databases (see Appendices A, B, and C for input parameters of the model).

We combined data from sporadic biological studies, government reports, and information from global databases to generate hypotheses on possible biological scenarios of the NSCS ecosystem. Ecosim is a dynamic simulation model which estimates changes of biomass among functional groups in the ecosystem as functions of abundance among other functional groups and time-varying harvest rates, taking into account predator-prey interactions and foraging behaviors (Pauly et al. 2000, Walters et al. 2000). Ecospace consists of spatial replicates of Ecosim simulations on a predefined base map. Each spatial cell is linked through (1) dispersal of organisms, (2) spatial movement of fishing effort in response to changes in the profitability of fishing, and (3) access to each of the seven fishing fleets as defined by the user (Walters et al. 1999, Walters 2000).

The behaviors of functional groups in the dynamic simulations are determined by sets of input parameters. One of the critical parameters is the vulnerability factor, which determines the foraging behavior of the functional groups in predator-prey interactions (Walters et al. 1997, 2000). In Ecosim, the vulnerability factor is a scaling factor describing the rate at which prey move between being vulnerable and not vulnerable to predation (Christensen and Walters 2004). Consumption rate for a given predator feeding on a prey is then calculated from the vulnerability factor, in combination with other parameters such as effective predator search rate, prey biomass, and predator abundance. Generally, when abundance of a predator is greatly reduced from its unexploited level, the vulnerability factor of its prey increases, while if predator abundance is similar to its unexploited level, the vulnerability factor will be low. As top predators in the NSCS have been greatly depleted, and the system is dominated by lower trophic level species, we have assumed the vulnerability factors of prey to be proportional to the trophic level of the predator (Cheung et al. 2002). However, we also tested the sensitivity of the simulation results to alternative settings of the vulnerability factors, including a high value (0.6) for all functional groups representing top-down trophic control, and a low value (0.3) representing bottom-up control (Christensen and Walters 2004).

Five functional groups in the ecosystem model were split into juveniles and adults using a delay-difference model to emulate large differences in ecology at different life-history stages: (1) large demersal reef-associated fishes, (2) large demersal estuary-associated fishes, (3) large demersal shelf-associated fishes, (4) large pelagic fishes, and (5) large sharks. Growth (von Bertalanffy growth parameter K) and the age at

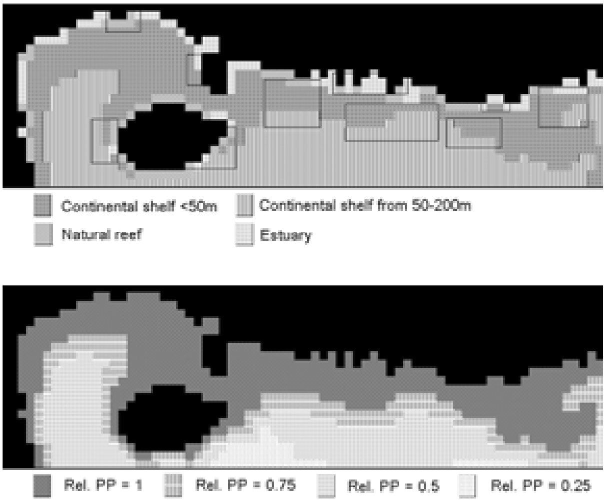


Figure 2. Base map of the northern South China Sea used for the spatial dynamic simulations in Ecospace. (a) Map of habitat types. The rectangles represent the approximate locations where MPAs were designated in the simulation. (b) Map of relative primary productivity (Rel. PP) ranging from 1 to 0.25 times the base Ecopath input.

transition to the adult group (t_k) were estimated from average values for the species making up each functional group (Froese and Pauly 2003).

We defined the model on a 72 × 22 grid map (Fig. 2a) so that each cell represented an area of 625 km^2. Four habitats were included: (1) natural reef (coral and rocky reef), (2) estuary, (3) shelf less than 50 m depth, and (4) shelf between 50 m and 200 m depth. In Ecospace, each model group was set to have different affinities for particular habitats based on their movement in relation to the preferred habitat, as determined by a habitat gradient function (Walters et al. 1999). For example, reef-associated groups, estuary-associated groups, and shelf-associated groups were set to have high affinity to natural reef, estuary, and the shelf area respectively, while pelagic groups were set to have equal association with all habitats. To represent spatial differences in primary productivity, we employed a predefined map of primary productivity. We integrated estimates of primary productivity in the NSCS (Sea Around Us Project, Fisheries Centre, University of British Columbia, Canada, unpubl. data) into four regions and assigned them to the map (Fig. 2b). Each region represents 1, 0.75, 0.5, and 0.25 times the baseline Ecopath input primary productivity value (production:biomass = 231 year^{-1}). Stochastic spatial and temporal changes in productivity were not considered.

We limited each fishing fleet to operate in their general fishing grounds: stern trawl, all habitats; pair trawl, all habitats except natural reef; shrimp trawl, only in estuary and shelf less than 50 m depth; gillnet, all except shelf of 50 to 200 m depth; hook and line, all habitats; purse seine, estuary and shelf less than 50 m depth; and other gears, all habitats except shelf of 50 to 200 m depth. The spatial distribution of fishing effort for each fleet was determined at each time step in the simulation from relative profits obtainable from each cell, based on a gravity model (Caddy 1975, Walters et al. 1999).

Lack of information forced us to estimate the dispersal characteristics of model groups in the study area. We assumed that reef-associated and small-sized fishes have lower annual dispersal rates whereas pelagic and large-sized fishes have higher dispersal rates. In addition, we considered system stability as a major factor determining dispersal parameters. We rejected parameter values if they resulted in system instability (extinction or biomass explosion of functional groups) in pilot simulations. Larval dispersal and inshore-offshore juvenile and adult migrations were not explicitly included and the effect of ocean currents also was not modeled.

Spatial simulation scenarios

Management scenarios involving changes in total fishing effort and establishment of marine protected areas (MPA) were developed. We defined a 30-year time horizon for simulations from 1990 to 2020. Based on Chinese national fishery statistics (PRC Department of Fishery 1996, 2000), we assumed that the total fishing effort on the NSCS ecosystem doubled from 1990 to 2000. Because of the unavailability of data, we extrapolated the effort from 2000 to 2003 from the trends in the past 10 years. We developed a matrix with three scenarios of fishing effort and seven scenarios of spatial fishery management (Table 1).

Data analysis

Simulated time-series of biomass of each functional group and catches by each fishing fleet from 1990 to 2020 were determined and compared among scenarios. We also compared the protected with non-protected areas within simulation scenarios.

Based on these time-series, we calculated indices tracking biodiversity and risk of local extinction. The Q-90 index is a variant of the Q-index developed by Kempton and Taylor (1976) to indicate the diversity of functional groups in an ecosystem. It is calculated from the slope of the cumulative functional group abundance curve between the 10th and 90th percentiles (Ainsworth and Pitcher 2005) and represents both functional group richness and evenness. For example, when predatory

Table 1. Scenarios of fisheries management policies for the NSCS ecosystem model (year 1990).

Scenarios	Spatial policy
1. Status quo	Nil
2. Seasonal trawl ban	Bottom trawling (pair trawl, shrimp trawl, and stern trawl) banned in June-July (2 months) from all areas 1998-2020.
3. Expanded seasonal moratorium	All fishing gear banned in June-July (2 months) from all areas 1998-2020.
4. 5% MPA[a]	5% of: (a) representative areas from each habitat type, (b) natural reef only, are fully protected from any fishing.
5. 10% MPA[a]	10% of: (a) representative areas from each habitat type, (b) natural reef only, are fully protected from any fishing.
6. 20% MPA[a]	20% of: (a) representative areas from each habitat type, (b) natural reef only, are fully protectedfrom any fishing.
7. 50% MPA[a]	50% of: (a) representative areas from each habitat type, (b) natural reef only, are fully protected from any fishing.

[a] Approximate distribution of the MPAs in the simulation model was noted in Fig. 2a.

Ten years of a baseline run were included at the start of each simulation to allow time for the functional groups to stabilize in the Ecospace model. The time frame is 30 years (from 1990 to 2020). From 1990 to 2000, we assumed a doubling in fishing effort (PRC Department of Fishery 1996, 2000). Because of the unavailability of data, we extrapolated the effort from 2000 to 2003 from the trends in the past 10 years. We included three scenarios of projected temporal changes in fishing effort: (a) annual increase of 2% from 2003 to 2020, (b) effort kept at the 2003 level from 2003 to 2020, (c) annual decrease of 5% from 2003 to 2020.

groups of an ecosystem are greatly depleted or driven to local extinction by fishing while the system is dominated by a few productive groups, the system should have a low Q-90 index. The local extinction index is an approximate indicator of the risk of loss of species within functional groups in ecosystem simulations. It is based on an empirical model that estimates the probability of species extirpation based on production to biomass ratios of species within functional groups, the rate of change of harvest rates, and the simulated biomass changes of each functional group (Cheung and Pitcher 2004).

Results

The simulations suggest that a majority of fish groups would be depleted from the 1990 baseline level under both status quo and seasonal trawl moratorium scenarios (scenarios 1 and 2 in Table 1). Without any spatial management, and with a 2% annual increase in fishing effort, all demersal fish groups were reduced by more than 70% by the year 2020, by which time the ecosystem becomes dominated by invertebrates (Fig. 3a). Reduction of modeled fishing effort reduced the level of depletion for most fish groups, except for demersal fish groups such as medium and large reef fishes, small sciaenids, and sharks (Fig. 3b,c). With a seasonal trawl moratorium, the biomass of fish groups in 2020 were still greatly reduced (Fig. 4). With a 5% annual decrease in fishing effort, changes in biomass were less dramatic for shelf-associated fishes, estuary-associated fishes, sciaenids, and pelagic fishes, whereas the biomass of benthic invertebrates increased slightly.

With a 2% annual increase in fishing effort, or effort constant at the 2003 level, closing 50% of the area to all fishing was insufficient to sustain the biomass of most fish groups (Fig. 4a,b). Reef-associated fishes were an exception and showed an increase in biomass when 50% of the area was closed to fishing. Alternatively, a general reduction in fishing effort greatly facilitated the recovery of non-reef fish groups. When fishing effort decreased by 5% annually (Fig. 4c), closing 50% of the total area to fishing allowed the biomass of most fish groups to increase slightly from the 1990 level, except for the shark and ray groups. All policy scenarios failed to restore the biomass of sharks and rays to the 1990 level, reflecting the high vulnerability to fishing of these groups (Musick et al. 2000).

Fish groups with different habitat associations showed different responses to policy scenarios. Increases in biomass with larger MPAs were more apparent for reef-associated fishes. Biomass recoveries for the estuary-associated fishes, shelf-associated fishes, sciaenids, pelagic fishes, and sharks and rays were more responsive to a general reduction of total fishing effort than to increased size of MPAs. The same effects are noticeable in scenarios in which MPAs were designated only on natural reefs and with 5% annual decreases in effort. Protection of non-reef species was only slightly less effective than establishing MPAs on all habitat types (Fig. 5).

Simulations indicated differences in total biomass and trophic structure within, on the edge, and outside MPAs. Biomasses of fish groups accumulated within protected areas (Fig. 6). When fishing effort was reduced by 5% annually, and 50% of the area reserved in MPAs, the biomass of reef fish within the MPAs was more than twice that at the immediate periphery, and over four times as much as areas farther from the MPAs (Fig. 6a). Similar biomass gradients resulted with smaller-sized MPAs (5% of total area) (Fig. 6b). However, the gradients of biomass differences between protected and unprotected areas were smaller for non-reef fish (excluding pelagic fish) groups (Fig. 6c,d). Medium to large-sized fish groups occurred in larger proportion within MPAs, whereas small-sized fish groups dominated in both the immediate surroundings and in areas further away from the MPAs. Greater accumulations of medium to large-

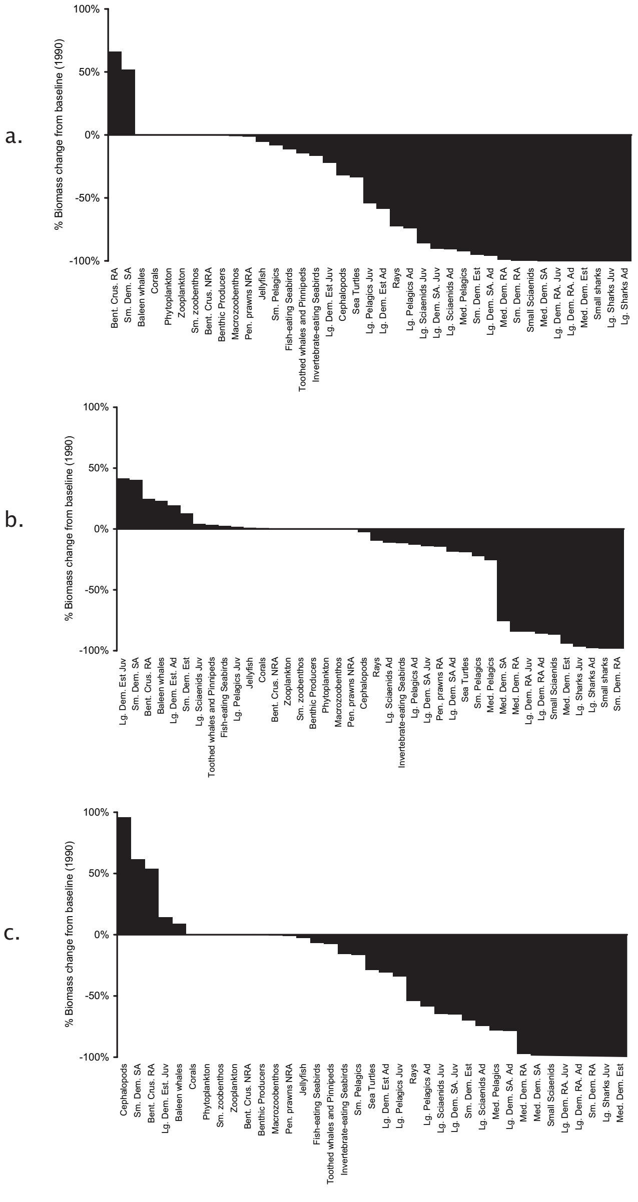


Figure 3. Relative changes in biomass (compared to 1990) of functional groups in the NSCS ecosystem model under the status quo (no management) scenario under (a) 2% annual increase in fishing effort, (b) fishing effort at 2003 level, (c) 5% annual decrease in fishing effort. Bent = benthic, Sm. = small, Med. = medium, Lg. = large, Dem. = demersal, RA = reef-associated, SA = shelf-associated, EA = estuary-associated, Ad. = adults, Ju. = juvenile.

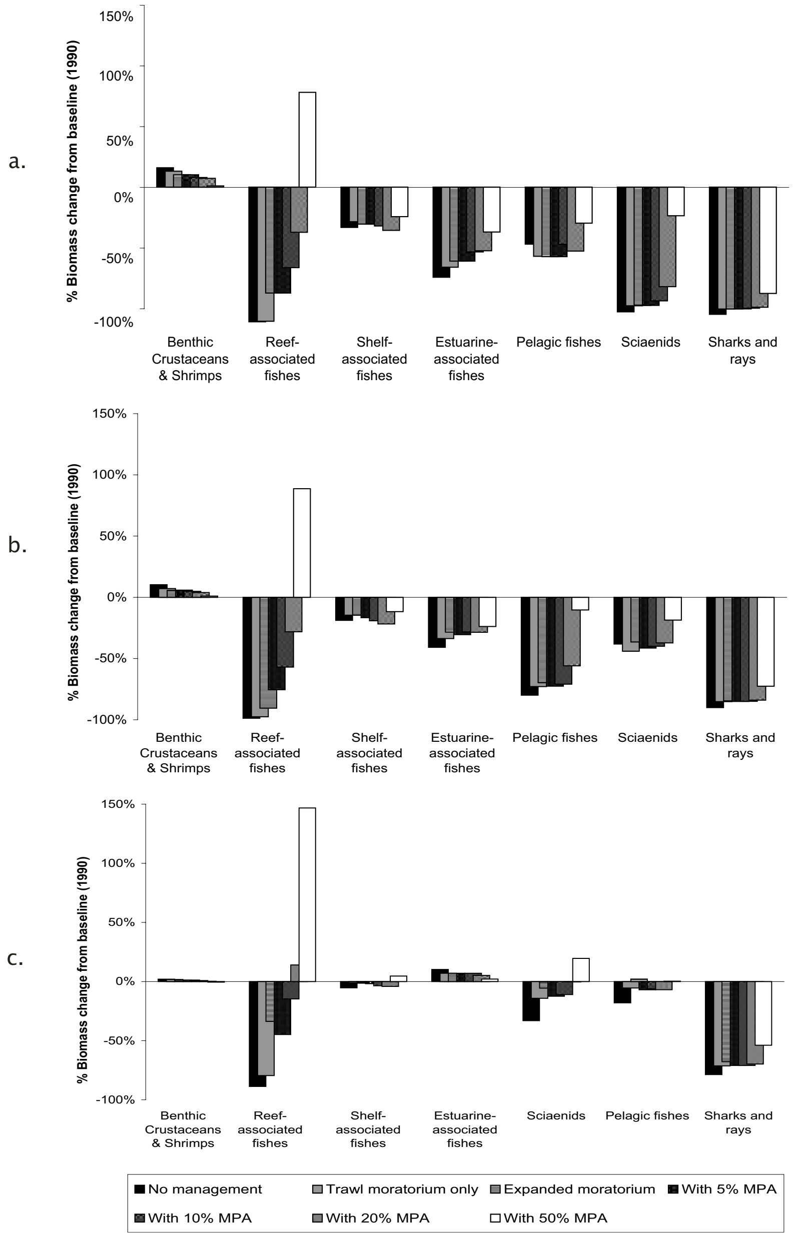


Figure 4. Changes in biomass of functional groups in the NSCS ecosystem. The bars represent (from left to right): (1) status quo (no management), (2) a two month trawl ban, (3) an expanded moratorium to all gears for two months, (4-7) 5%, 10%, 20%, 50%, respectively, of representative habitats as marine protected areas.

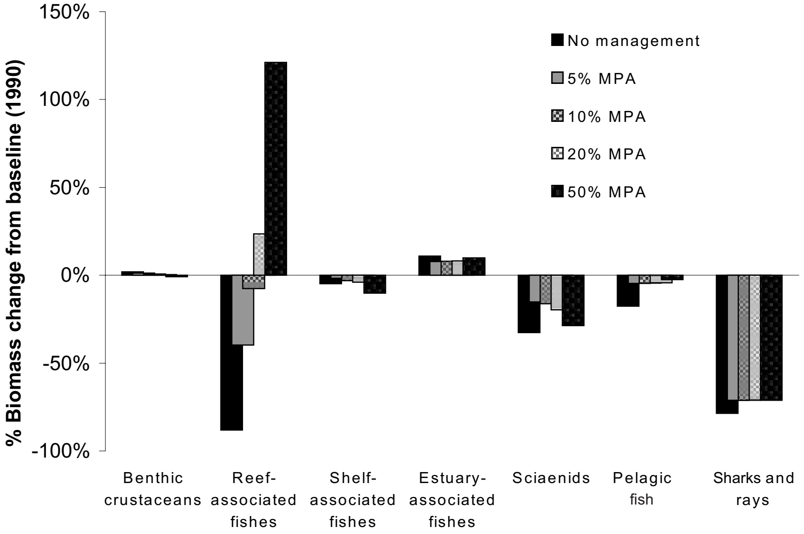
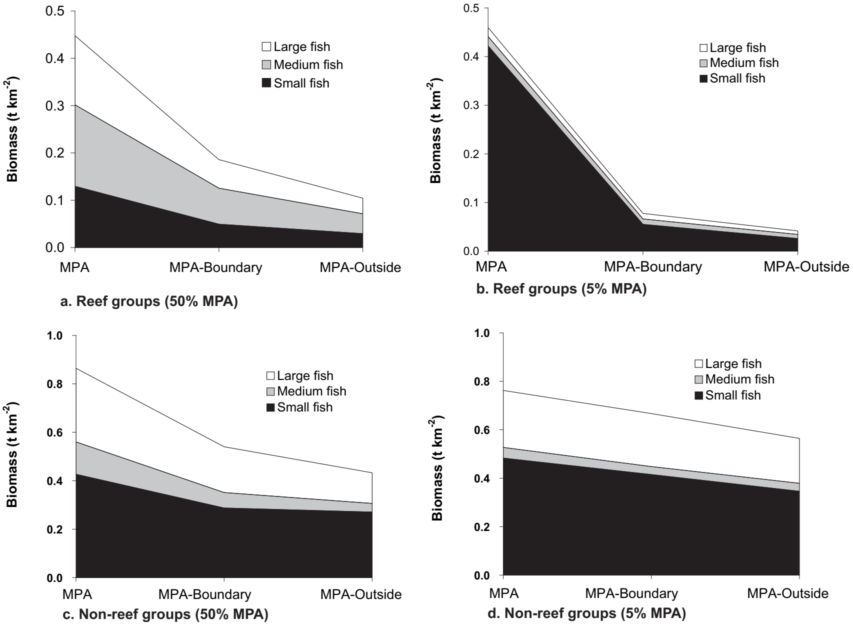


Figure 5. Changes in biomass in the NSCS ecosystem when MPAs were designed on natural reef only (assuming 5% annual decrease in fishing effort).

sized fishes within MPAs were observed when a larger proportion of the total area was protected.

Reductions in total fishing effort and an increased level of spatial management decreased the risk of local extinctions and helped maintain biodiversity. For example, mean probability of local extinction, as estimated from the average probability of 10% or more of species going extinct within each functional group, decreased from around 80% without spatial management (scenario 1) to less than 5% when 50% of the area was reserved as MPAs and with a 2% annual increase in effort (Fig. 7). Reducing fishing effort by 5% annually lowered the extinction risk to less than 1%. Biodiversity of the ecosystem in 2020, as indicated by the Q-90 index, increased from 0.7 to 1 relative to the 1990 baseline when 50% of the total area was reserved as MPAs (Fig. 8). Reducing fishing effort by 5% annually greatly reduced the size of MPAs needed to maintain biodiversity at the 1990 baseline level. Achieving this criterion would need 50% of the area in MPAs with a 2% annual increase in fishing effort, or 20% of the area in MPAs with constant 2003 level of fishing effort.



a. Reef groups (50% MPA)

b. Reef groups (5% MPA)

c. Non-reef groups (50% MPA)

d. Non-reef groups (5% MPA)

Figure 6. Gradient of size-abundance distribution of reef and non-reef fish groups between MPAs and unprotected areas.

Sensitivity analysis

Large variations in the simulated biomass of some functional groups resulted from changing assumptions about their vulnerability factors. In the scenario where MPAs occupied 20% of the total area, both low and high vulnerability factors (0.3: bottom-up control; 0.6: top-down control) resulted in less biomass recovery for most fish groups (Fig. 9a,b). Biomass of reef-associated fishes by body sizes within the MPAs also varied under different assumptions. With MPAs protecting 20% of the natural reef, and a 5% annual decrease in fishing effort, bottom-up control resulted in a smaller biomass of small reef fishes and a higher biomass of large reef fishes than with a top-down controlled system (Fig. 10). Biomass of medium-sized fishes was relatively similar for all assumptions.

Discussion

Our simulation modeling suggests that fisheries resources in the NSCS are not likely to be sustained by current levels of fishing and that the seasonal trawl ban implemented since 1998 is insufficient to reverse

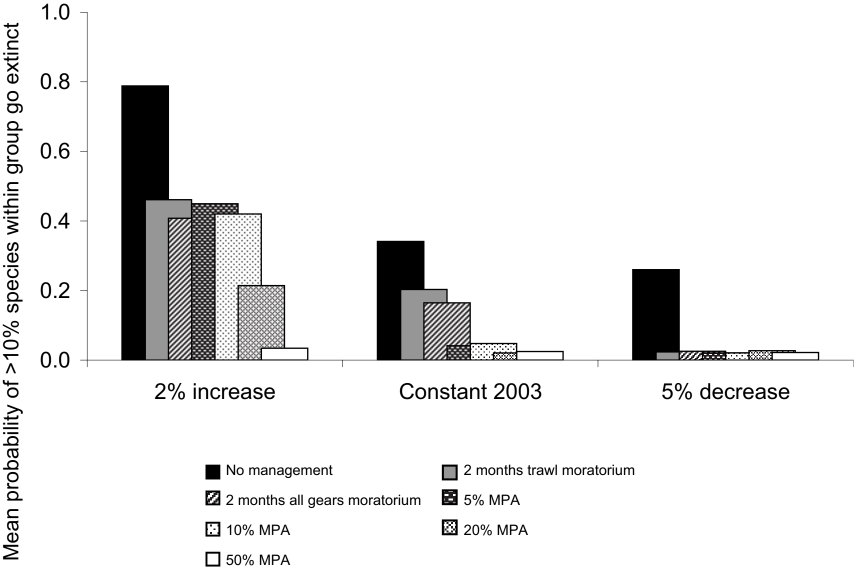


Figure 7. Mean probability of local extinction in the NSCS ecosystem under three scenarios of project fishing effort (x-axis).

these trends. Even if the Chinese management authority is successful in limiting fishing effort at current levels, large changes in species abundances could still result. This finding is alarming because the Chinese management authority appears to be relying on the seasonal trawl ban as a tool to enable sustainable exploitation of fishery resources. In particular, illegal fishing was common in Chinese waters, compromising the efforts of the government in limiting fishing effort (Wang 2001). Moreover, local anecdotal information suggests that fishers tend to spend more time fishing after the ban season, thus lowering its effect in reducing annual fishing mortality. This effect was not captured in our model, and, if incorporated, would have the effect of making our forecasts even more pessimistic about its efficacy. Although the insufficiency of the seasonal trawl ban in managing NSCS fishery resources had been suggested by a number of scholars (e.g., Wang 2000), this paper presents the first quantitative evaluation of the issue.

The consequences of maintaining the status quo are that the future ecosystem would be dominated by benthic invertebrates which are more resilient to present exploitation patterns and would be released from predation as most fish groups become depleted. As a result, many fishers would either be forced to cease fishing for economic reasons or switch

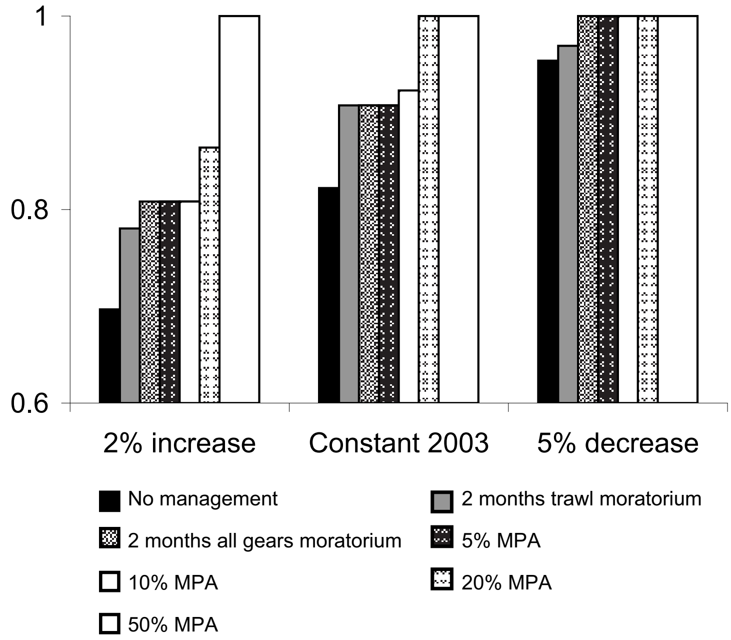


Figure 8. Q-90 biodiversity index of the NSCS ecosystem relative to the baseline (1990) level under three scenarios of projected fishing effort (x-axis).

to target benthic invertebrates. "Fishing down the food web" (Pauly et al. 1998) would become even more apparent in this region. This may already be occurring in the NSCS ecosystem in the most heavily fished coastal regions, such as the Pearl River Estuary, where commercial trawlers are supported by catches of benthic invertebrates and small fishes that are marketed as aquaculture fish feeds (Cheung and Sadovy 2004). The most vulnerable species, such as large-sized sciaenids, would have a high risk of local extinction. Thus dramatic reduction of biodiversity might result. For instance, the Chinese bahaba (*Bahaba taipingensis*), a large-bodied sciaenid distributed in the East and South China seas, already has been depleted to the verge of extinction (Sadovy and Cheung 2003). A high proportion of trawlers in Guangdong (a coastal province in southern China) were recently fishing at a loss (estimated to be 70% of total number of trawlers in 1999, Li et al. 1999). Immediate changes in fishery management regulations to enhance conservation and restore fishery productivity are therefore justified.

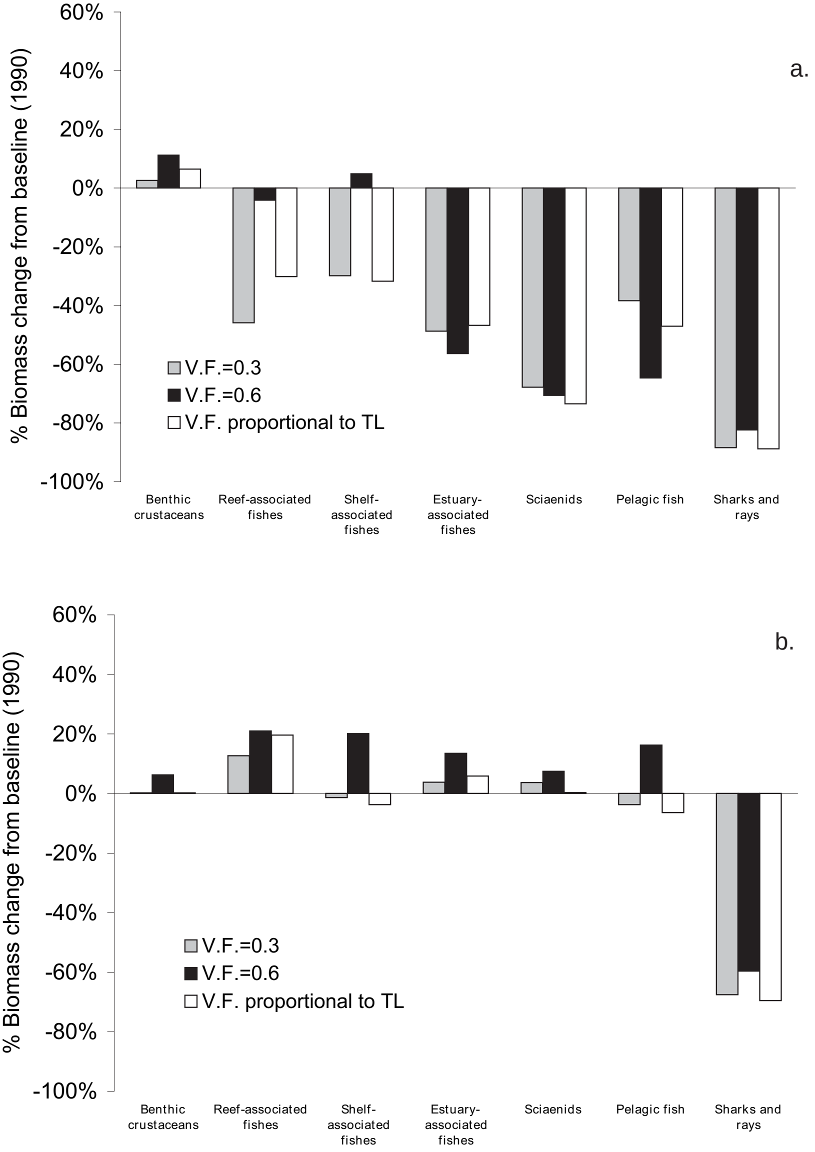


Figure 9. Changes in biomass between different fishery management scenarios under different assumptions of top-down or bottom-up ecosystem trophic control (20% of area designated as MPA). (a) 2% annual increase in fishing effort. (b) 5% annual decrease in fishing effort. V.F. = vulnerability factor of prey functional groups proportional to the trophic level of its predator (range from 0.2 to 0.5).

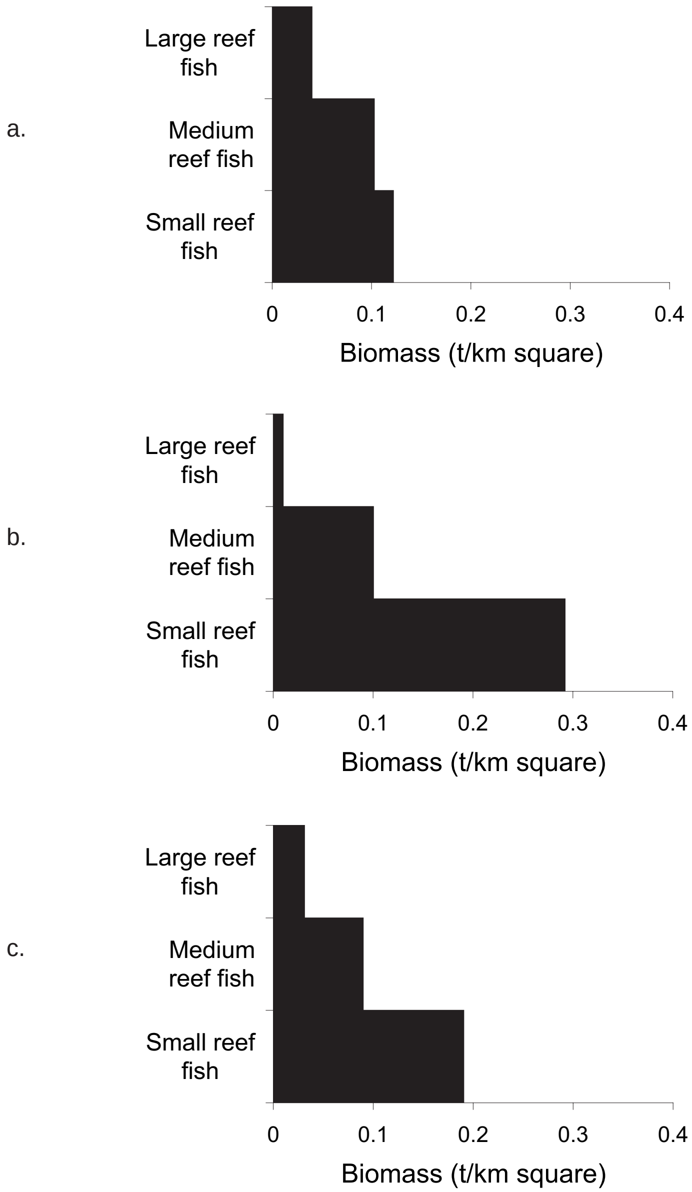


Figure 10. Differences in abundance of three size groups of fish within the MPA under different assumptions of top-down or bottom-up control (20% of area designated as MPA and 5% annual decrease in effort): (a) vulnerability factor = 0.3 (bottom-up control), (b) vulnerability factor = 0.6 (top-down control), (c) vulnerability factor proportional to the trophic level of the functional groups.

Our simulations suggest that a combination of large-scale marine protected areas and considerable reduction of fishing effort might reasonably be expected to restore the fishery resources and biodiversity in the NSCS. MPAs of sufficient size would effectively protect reef-associated species, but benefits would be less apparent for non-reef species because of their greater mobility. Production rates for reef-fish groups were higher than their net dispersal rates from the MPA to the non-protected area, allowing the biomass of reef species to accumulate inside MPAs. Our modeling shows that non-reef species are best conserved by reduction of harvest rate over the entire area. Both MPAs and a region-wide reduction of fishing effort are needed to effectively conserve the whole spectrum of species. This may be achieved by protecting ecologically critical habitats such as coral reefs together with effective management of fishing effort. However, because our objective was to compare the effects of some gross management policy options, future studies may focus on responses of functional groups to scenarios of different MPA designs, gear restriction, etc.

Assumptions about the type of trophic control in the model alter the effectiveness of policy options to restore biomass of functional groups. The vulnerability factors determine the exchange rates between refuge and foraging arenas, and can partly determine the compensatory mechanism of the stock-recruitment relationship of ontogenically split groups (Walters and Juanes 1993). For example, with a lower vulnerability setting (0.3), large demersal reef-associated fish exhibit a dome-shaped stock-recruitment relationship with a steep initial slope. As a consequence, large reef fishes become highly productive and abundances are rebuilt rapidly when they were protected by an MPA. On the other hand, a vulnerability of 0.6 results in a smaller initial slope in the stock-recruitment relationship, thus recovery of biomass in protected areas would be slower for a number of functional groups. Consequently, biomass of small reef fishes would increase rapidly as a result of the slower recovery of their predators. Assuming a vulnerability setting proportional to the trophic level of the functional groups provides results between the above two extremes. The differences could be resolved by tuning the time series of the simulation to biomass surveys or fishery VPA parameters, but these data are not available in the NSCS in the present study.

The ecosystem models described here were constructed from limited data of low precision, and many input parameter values had to be approximated. Responses of certain functional groups were sensitive to critical input parameters such as the assumptions about ecosystem trophic control and the relative dispersal rates defined for each group (Walters 2000). Thus, detailed responses of the functional groups to model scenarios should be interpreted with caution, particularly because the effects of ocean currents were excluded. Nevertheless, our simulations produced general conclusions that are robust against most of these uncertainties, i.e., maintenance of the status quo would likely result in serious resource depletion and species loss, while a combination of fishing effort control and large MPAs could possibly restore abundance of populations to the 1990 level. Moreover, we do not think data limitation should be an excuse to delay management action, given the current depleted status of the fishery resources and the obvious disastrous outcomes if management is kept at the status quo. If the suggested management policies are implemented, parameter values can be refined by monitoring carefully selected attributes, thereby improving future simulations (Walters1997).

Monitoring efforts now should be focused on documenting changes in abundance and dispersal of indicator species at each trophic level, because results suggest that effective species protection is sensitive to the type of trophic control (and thus the vulnerability factor) and species dispersal rates. While we did not analyze the effects of management policy options on fishery productivity, this, together with evaluation of uncertainty of parameter values and climate variation using Monte Carlo simulations and analysis of economic and social costs and benefits, will be examined in future studies.

Acknowledgments

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Appendix A. Basic input parameters of the NSCS Ecopath model.

Group name	Trophic level	Biomass (t km^{-2})	P/B (year^{-1})	Q/B (year^{-1})	Ecotrophic efficiency
Benthic producers	1	153.000	11.885	–	0.050
Phytoplankton	1	13.000	231.000	–	0.780
Corals	1.8	(4.252)	1.090	9.000	0.950
Zooplankton	2	14.700	32.000	192.000	(0.206)
Sea turtles	2.48	0.000	0.100	2.500	(0.409)
Jellyfish	3	1.529	5.011	25.050	(0.069)
Sm. zoobenthos	2.09	70.370	6.570	27.400	(0.824)
Macrozoobenthos	2.4	35.053	3.000	12.500	(0.950)
Bent. Crus. NRA	3.14	7.245	5.650	26.900	(0.172)
Bent. Crus. RA	2.86	(0.755)	1.850	8.350	0.950
Pen. prawns NRA	2.65	8.962	4.800	16.352	(0.950)
Pen. prawns RA	2.5	(0.155)	7.600	41.537	0.950
Cephalopods	3.64	0.039	3.100	11.970	(0.950)
Sm. Dem. RA	3.23	0.054	3.370	9.743	(0.950)
Sm. Dem. SA	3.39	0.150	2.051	8.098	(0.951)
Sm. Dem. Est.	2.88	0.045	3.598	15.218	(0.951)
Med. Dem. RA	3.72	0.071	1.380	8.502	(0.531)
Med. Dem. SA	3.42	(0.112)	1.787	7.239	0.950
Med. Dem. Est.	3.05	0.012	2.582	9.127	(0.951)
Lg. Dem. RA. Juv.	3.3	(0.036)	3.370	9.743	0.950
Lg. Dem. RA. Ad.	3.52	0.036	0.670	4.375	(0.949)
Lg. Dem. SA. Juv.	3.44	0.054	2.051	8.098	(0.950)
Lg. Dem. SA. Ad.	4.05	(0.058)	0.659	4.952	0.950
Lg. Dem. Est. Juv.	2.88	0.029	3.598	15.218	(0.951)

Appendix A. (Continued) Basic input parameters of the NSCS Ecopath model.

Group name	Trophic level	Biomass (t km^{-2})	P/B (year^{-1})	Q/B (year^{-1})	Ecotrophic efficiency
Lg. Dem. Est. Ad.	3.83	0.028	1.067	7.158	(0.950)
Sm. sciaenids	3.53	0.040	1.674	9.440	(0.950)
Lg. sciaenids Juv.	3.5	0.032	1.674	9.440	(0.852)
Lg. sciaenids Ad.	3.57	0.029	0.887	6.232	(0.950)
Sm. pelagics	2.99	(0.072)	4.965	15.702	0.950
Med. pelagics	3.22	0.029	2.572	8.977	(0.950)
Lg. pelagics Juv.	3	0.022	4.965	15.702	(0.950)
Lg. pelagics Ad.	3.74	0.018	1.617	5.230	(0.950)
Rays	3.73	(0.002)	0.437	2.187	0.824
Sm. sharks	4	(0.002)	1.345	4.560	0.921
Lg. sharks Juv.	3.93	(0.003)	0.678	3.353	0.921
Lg. sharks Ad.	4.39	0.001	1.444	5.030	(0.957)
Fish-eating seabirds	4.02	0.001	0.060	61.280	(0.950)
Invertebrate-eating seabirds	3.12	0.001	0.060	72.764	(0.950)
Baleen whales	3.37	(0.002)	0.112	4.430	0.853
Toothed whales and pinnipeds	4.32	0.005	0.045	14.768	(0.950)
Detritus	1	100.000	–	–	(0.559)

NRA = non reef-associated, RA = reef-associated, SA = shelf-associated, Bent. Crus. = benthic crustacean, Sm. = small, Med. = medium, Lg. = large, Juv = juvenile, Ad. = adult, Dem. = demersal, Pen. = penaeid, P/B = production to biomass ratio, Q/B = consumption to biomass ratio. Values in parentheses are estimated by the model based on the mass-balance assumption.

Appendix B. **Diet composition matrix of the functional groups in the NSCS Ecopath model. Diet composition was expressed as the proportion of prey (column) represented in the diet of their predators (row).**

Group no.	Prey/Predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
1	Benthic Producers					0.667			0.151	0.105	0.314		0.250		0.064	0.038	0.066	0.046		0.242	0.015
2	Phytoplankton				0.700			0.170	0.070			0.055			0.023		0.012		0.003	0.001	0.024
3	Corals								0.010				0.005					0.006			
4	Zooplanktons			0.600		0.119	1.000		0.050			0.055	0.331	0.106	0.429	0.301	0.576	0.013	0.273	0.171	0.448
5	Sea Turtles																				
6	Jellyfish					0.119									0.003	0.001	0.025	0.002	0.104		0.003
7	Sm. zoobenthos					0.048		0.080	0.312	0.369	0.002	0.111	0.130		0.130	0.192	0.197	0.047	0.147	0.167	0.135
8	Macrozoobenthos					0.048			0.005	0.232	0.419	0.337	0.015	0.235	0.012	0.030	0.011	0.020	0.013	0.009	0.012
9	Bent. Crus. NRA									0.032				0.010		0.261	0.005		0.088	0.096	0.139
10	Bent. Crus. RA										0.025			0.082	0.265	0.140	0.005	0.556	0.088	0.096	0.139
11	Pen. prawns NRA									0.210				0.000						0.068	
12	Pen. prawns RA										0.148			0.212				0.067			0.052
13	Cephalopods													0.000	0.000			0.003	0.015		0.001
14	Sm. Dem. RA													0.003	0.002			0.148			0.009
15	Sm. Dem. SA													0.028		0.004	0.000		0.086	0.014	
16	Sm. Dem. Est													0.029		0.001	0.000		0.023	0.025	
17	Med. Dem. RA																	0.005			
18	Med. Dem. SA																		0.008	0.012	
19	Med. Dem. Est																		0.003	0.003	
20	Lg. Dem. RA. Juv													0.002	0.059			0.058			0.011
21	Lg. Dem. RA. Ad																				
22	Lg. Dem. SA. Juv													0.000		0.001	0.000		0.009	0.002	
23	Lg. Dem. SA. Ad																				
24	Lg. Dem. Est. Juv													0.001		0.001	0.000		0.016	0.035	
25	Lg. Dem. Est. Ad																				
26	Small Sciaenids													0.001			0.000		0.007	0.002	
27	Lg. Sciaenids Juv													0.001			0.000		0.009	0.002	
28	Lg. Sciaenids Ad																				
29	Sm. Pelagics													0.287							
30	Med. Pelagics																				
31	Lg. Pelagics Juv																				
32	Lg. Pelagics Ad																				
33	Rays																				
34	Small sharks																				
35	Lg. Sharks Juv																				
36	Lg. Sharks Ad																				
37	Fish-eating Seabirds																				
38	Invertebrate-eating Seabirds																				
39	Baleen whales																				
40	Toothed whales and Pinnipeds																				
41	Detritus			0.400	0.300			0.750	0.402	0.053	0.093	0.442	0.269		0.012	0.031	0.103	0.028	0.106	0.056	0.013

Appendix B. (Continued).

Group no.	Prey/Predator	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39	40
1	Benthic Producers		0.034		0.066	0.020				0.032	0.065	0.032							0.177		
2	Phytoplankton				0.012	0.060				0.078	0.014	0.077	0.006								
3	Corals	0.003																			
4	Zooplanktons	0.229	0.269	0.023	0.576	0.015	0.304	0.314	0.345	0.626	0.445	0.619	0.315	0.095					0.050	0.628	
5	Sea Turtles																0.000				
6	Jellyfish	0.001	0.001		0.025		0.009	0.009	0.008	0.055	0.075	0.055	0.112							0.209	
7	Sm. zoobenthos	0.168	0.172	0.054	0.197	0.055	0.169	0.174	0.137	0.080	0.051	0.079	0.015	0.063	0.152	0.037		0.010	0.296		
8	Macrozoobenthos	0.115	0.027	0.204	0.011	0.161	0.042	0.043		0.026	0.021	0.025	0.002	0.063	0.092	0.123		0.010	0.202		
9	Bent. Crus. NRA	0.161	0.234	0.065	0.005	0.183	0.182	0.188	0.177	0.017	0.036	0.017	0.025	0.225	0.189	0.100	0.011	0.010	0.101		
10	Bent. Crus. RA	0.161	0.234		0.005		0.182	0.188	0.177	0.017	0.036	0.017	0.025	0.225	0.050	0.358	0.048	0.000			
11	Pen. prawns NRA			0.015		0.050									0.055	0.093	0.047	0.010	0.059		0.003
12	Pen. prawns RA	0.108													0.001		0.046				
13	Cephalopods	0.028		0.040		0.069				0.000	0.002	0.000	0.011		0.148	0.044	0.175	0.021	0.008	0.004	0.217
14	Sm. Dem. RA	0.014												0.018	0.006	0.010	0.015				0.196
15	Sm. Dem. SA		0.001	0.159	0.000	0.105	0.046	0.016	0.028					0.054	0.036	0.030	0.045	0.002			0.122
16	Sm. Dem. Est		0.001	0.045	0.000	0.030	0.026	0.024	0.028					0.036	0.026	0.017	0.030	0.002			0.061
17	Med. Dem. RA	0.005												0.018	0.006	0.010	0.015				0.003
18	Med. Dem. SA			0.159		0.082			0.028					0.045	0.036	0.025	0.045				0.001
19	Med. Dem. Est			0.008		0.005			0.006					0.010	0.008	0.004	0.024				0.000
20	Lg. Dem. RA. Juv	0.006												0.018	0.006	0.010	0.015				0.001
21	Lg. Dem. RA. Ad	0.001													0.003	0.004	0.015				0.001
22	Lg. Dem. SA. Juv		0.001	0.098	0.000	0.060	0.005	0.005	0.012					0.005	0.023	0.003	0.036	0.010	0.001		
23	Lg. Dem. SA. Ad			0.009		0.009			0.002						0.006	0.003	0.045				0.001
24	Lg. Dem. Est. Juv		0.001	0.045	0.000	0.030	0.032	0.035	0.020					0.021	0.025	0.011	0.030	0.072	0.004		
25	Lg. Dem. Est. Ad			0.016		0.013			0.012						0.009	0.009	0.030				0.003
26	Small Sciaenids			0.022	0.000	0.010	0.002	0.002	0.008					0.012	0.008	0.005	0.030	0.001			0.062
27	Lg. Sciaenids Juv			0.018	0.000	0.008	0.003	0.003	0.012					0.016	0.009	0.008	0.030	0.053	0.002		
28	Lg. Sciaenids Ad			0.018		0.017									0.019	0.017	0.030				0.002
29	Sm. Pelagics									0.016	0.059	0.021	0.143		0.022	0.020	0.030	0.403	0.029	0.159	0.191
30	Med. Pelagics										0.068		0.143		0.022	0.020	0.030				0.083
31	Lg. Pelagics Juv									0.016	0.060	0.022	0.143		0.022	0.020	0.030	0.395	0.029		
32	Lg. Pelagics Ad												0.060		0.022	0.020	0.030				0.047
33	Rays																0.030				0.001
34	Small sharks																0.030				0.001
35	Lg. Sharks Juv																0.030				0.001
36	Lg. Sharks Ad																0.001				0.001
37	Fish-eating Seabirds																0.000				0.001
38	Invertebrate-eating Seabirds																0.001				0.001
39	Baleen whales																0.015				0.001
40	Toothed whales and Pinnipeds																0.013				0.001
41	Detritus		0.027		0.103	0.019				0.036	0.069	0.036	0.001	0.076					0.042		

Appendix C. Fishery catches of functional group by the seven fishery sectors in the NSCS ecosystem Ecopath model.

No.	Group name	Fishery catch by sector (t km^{-2})			
		PT	ST	SHT	GN
3	Corals	0.001910	–	–	–
4	Zooplankton	–	–	0.001900	–
5	Sea turtles	–	–	0.000001	–
6	Jellyfish	–	–	0.000421	–
7	Sm. zoobenthos	0.000095	0.000012	–	–
8	Macrozoobenthos	0.000095	0.000012	0.000253	–
9	Bent. Crus. NRA	0.000095	0.000012	0.019763	0.002509
10	Bent. Crus. RA	0.000095	0.000012	0.001774	0.006857
11	Pen. prawns NRA	0.000567	0.000235	0.004307	–
12	Pen. prawns RA	0.000567	0.000235	0.010895	–
13	Cephalopods	0.026566	0.005023	0.000633	–
14	Sm. Dem. RA	0.004274	0.001204	0.002787	0.004014
15	Sm. Dem. SA	0.043567	0.004690	0.006778	0.012169
16	Sm. Dem. Est.	0.039631	0.001202	0.003389	0.010456
17	Med. Dem. RA	0.008409	0.007683	0.001013	0.005184
18	Med. Dem. SA	0.082180	0.013877	0.000190	0.007526
19	Med. Dem. Est.	0.003468	0.000619	0.000190	0.007526
20	Lg. Dem. RA. Juv.	0.008996	0.004123	–	0.009700
21	Lg. Dem. RA. Ad.	0.008898	0.001374	0.000001	0.009365
22	Lg. Dem. SA. Juv.	0.020624	0.005517	0.005553	0.004850
23	Lg. Dem. SA. Ad.	0.020526	0.001838	0.000323	0.007358
24	Lg. Dem. Est. Juv.	0.012810	0.003901	0.002776	0.002425
25	Lg. Dem. Est. Ad.	0.012712	0.001300	0.000161	0.003679
26	Small sciaenids	0.014756	0.006229	0.003389	0.001714
27	Lg. sciaenids Juv.	0.007823	0.003153	0.002776	0.002425
28	Lg. sciaenids Ad.	0.007725	0.003151	0.000161	0.003679
29	Sm. pelagics	0.010404	0.001385	0.000882	0.001840
30	Med. pelagics	0.014387	0.001410	0.000127	0.006600
31	Lg. pelagics Juv.	0.003340	0.000548	0.000127	0.009700
32	Lg. pelagics Ad.	0.003243	0.000546	–	0.012375
33	Rays	0.000120	0.000120	0.000120	0.000017
34	Small sharks	0.000188	0.000188	0.000094	0.000017
35	Lg. sharks Juv.	0.000190	0.000190	0.000094	0.000017
36	Lg. sharks Ad.	0.000324	0.000162	–	–
Sum		0.358590	0.069951	0.070880	0.131999

PT = pair trawl, ST = stern trawl, SHT = shrimp trawl, GN = gillnet, HL = hook and line, PS = purse seine, Others = other sectors.

Appendix C. (Continued.) Fishery catches of functional group by the seven fishery sectors in the NSCS ecosystem Ecopath model.

		Fishery catch by sector (t km^{-2})			
No.	Group name	HL	PS	Others	Total
3	Corals	–	–	–	0.001910
4	Zooplankton	–	0.000554	0.007565	0.010020
5	Sea turtles	–	–	–	0.000001
6	Jellyfish	–	–	–	0.000421
7	Sm. zoobenthos	–	–	–	0.000107
8	Macrozoobenthos	–	–	0.000809	0.001169
9	Bent. Crus. NRA	–	–	0.003694	0.026072
10	Bent. Crus. RA	–	–	0.014564	0.023301
11	Pen. prawns NRA	–	–	0.000275	0.005385
12	Pen. prawns RA	–	–	0.000600	0.012297
13	Cephalopods	0.000055	0.004275	0.011955	0.048506
14	Sm. Dem. RA	0.001092	0.003800	0.045610	0.062781
15	Sm. Dem. SA	0.001663	0.005086	0.026613	0.100566
16	Sm. Dem. Est.	0.001533	0.002543	0.013306	0.072061
17	Med. Dem. RA	0.002184	0.001425	0.021665	0.047564
18	Med. Dem. SA	0.002908	0.00368	0.003538	0.113900
19	Med. Dem. Est.	0.002908	0.003681	0.003538	0.021929
20	Lg. Dem. RA. Juv.	0.002648	0.009530	0.008406	0.043402
21	Lg. Dem. RA. Ad.	0.003030	0.000010	0.000013	0.022691
22	Lg. Dem. SA. Juv.	0.001324	0.001768	0.010412	0.050048
23	Lg. Dem. SA. Ad.	0.000082	0.000000	0.000856	0.030984
24	Lg. Dem. Est. Juv.	0.000662	0.000884	0.005206	0.028664
25	Lg. Dem. Est. Ad.	0.000041	0.000000	0.000428	0.018322
26	Small sciaenids	0.000130	0.002543	0.013306	0.042067
27	Lg. sciaenids Juv.	0.000662	0.000884	0.005206	0.022929
28	Lg. sciaenids Ad.	0.000041	0.000000	0.000428	0.015186
29	Sm. pelagics	–	0.043059	0.053687	0.111258
30	Med. pelagics	0.000410	0.001583	0.009887	0.034404
31	Lg. pelagics Juv.	0.001190	0.000237	0.008999	0.024141
32	Lg. pelagics Ad.	0.001190	0.000040	0.000957	0.018352
33	Rays	0.000003	–	0.000007	0.000386
34	Small sharks	0.001155	–	0.000007	0.001649
35	Lg. sharks Juv.	0.001160	–	0.000007	0.001658
36	Lg. sharks Ad.	0.001134	–	0.000007	0.001627
Sum		0.027204	0.085584	0.271549	1.015758

Ecosystem Simulations of Management Strategies for Data-Limited Seamount Fisheries

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Abstract

Traditional fisheries stock assessment requires large amounts of information, mainly from long-term data series, a requirement that is hard to apply to new or poorly documented fishing grounds. With the collapse of traditional shelf stocks and a general decline of global catches, the fishery industry has moved to alternative fishing grounds and species—a process of serial depletion. Seamounts are among those newly targeted ecosystems. In this paper we investigate if ecosystem simulations can help researchers understand the impact of fishing on pristine seamounts. Using ecosystem modeling tools, data gathered from elsewhere, and methods that search for optimal fishing policies, we explore what types of fisheries might be sustainable on seamounts. Although the analyses in this paper are not meant to describe actual fisheries for seamounts, some generalizations can be made. Simulations with policy objectives that maximize economic performance favor fleet configurations based on deepwater trawling, but entail a cost to biodiversity. Maximizing ecological performance favors fleets based on small pelagic and bottom longline fisheries, and maximizes biomass of long-lived species and biodiversity, but sacrifices total catches and jobs. The overall study suggested that sustainable seamount fisheries with tolerable ecosystem impacts appear to be closer to those found by maximizing an "ecological" objective function.

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Introduction

With global catches declining since the late 1980s (Watson and Pauly 2001), the world's fisheries resources have been characterized as seriously depleted or in danger of depletion (e.g., Jackson et al. 2001, Pauly et al. 2002, Baum et al. 2003, Christensen et al. 2003, Myers and Worm 2003), with very little evidence for any recovery (Hutchings 2000). What caused this phenomenon has been the subject of serious debate (Pitcher 2001). Indeed, poor management practices and increased fishing pressure (Ludwig et al. 1993), along with an excessive level of investment in fishing capacity, have resulted in serious stock depletion on continental shelves and have created serial depletion from new pressures on alternative fishing grounds (Pauly et al. 2002). Seamounts are among those "newly" targeted ecosystems that have been intensively fished since the second half of the twentieth century (Rogers 1994, Koslow et al. 2000).

Deepwater fisheries in general, and seamount fisheries in particular, are usually characterized by a boom and bust sequence (Koslow et al. 2000, Watson and Morato 2004), with the targeted fish stocks showing signs of overexploitation within a short period after the beginning of the fishery. For example, this has been the case with the orange roughy (*Hoplostethus atlanticus*) fishery off of New Zealand (Clark 1999, Clark et al. 2000), Australia (Koslow et al. 2000), Namibia (Boyer et al. 2001), and the North Atlantic (Branch 2001); the pelagic armourhead (*Pseudopentaceros wheeleri*) fishery over seamounts in international waters off Hawaii (Sasaki 1986); and the blue ling (*Molva dipterygia*) fishery in the North Atlantic (Bergstad et al. 2003). Seamount aggregating fish stocks are long-lived, slow growing species, with late maturity and low recruitment rates (Koslow 1997, Rico et al. 2001, Morato et al. 2004), often forming highly localized aggregations (Clark 1996). Thus, seamount fish stocks are rapidly depleted and maintenance of seamount fisheries depends on the discovery of unexploited seamounts. Moreover, many seamounts are located in international waters where no management is applied. Once depleted, seamount populations likely require decades to recover (Koslow 1997). Side effects caused by overfishing or extensive trawling on seamounts raise serious concerns: for example, damage to benthic communities dominated by corals and other fragile suspension feeders is common (Richer de Forges et. al. 2000, Koslow et al. 2001), as well as impacts on transient migratory species whose life histories rely on seamount food webs (Haney et. al. 1995, Holland et al. 1999, Weimerskirch et al. 2002). The prevention of further negative impacts on these sensitive ecosystems is now an important policy objective (Probert 1999, Roberts 2002).

There is rising concern about threats to seamount ecosystems in the Exclusive Economic Zones of coastal states and the high seas; several countries, such as Canada, Australia, New Zealand, and Portugal have begun to take action for the protection of such "fragile" ecosystems.

However, seamount ecosystems remain one of the worst cases of data-limited situations, comprising a true challenge for fishery scientists and managers who are urged to develop new fisheries under the precautionary approach of the Code of Conduct for Responsible Fisheries (FAO 1995). Little is known about seamount ecosystems in the Northeast Atlantic and elsewhere, or the impact of human activities upon them. A recent attempt to tackle this global lack of information has been made by the European Commission, which has funded the first European Seamount Project to integrate physical, biogeochemical, and biological research—the "OASIS project" (OceAnic Seamounts: an Integrated Study).

In this paper we investigate if ecosystem simulations can help in understanding the impact of fishing on pristine seamounts. By using ecosystem modeling loosely structured on North Atlantic case studies, data gathered from elsewhere, and optimization methods for policy search, we explore the types of fisheries that might be sustainable on seamount ecosystems.

Methods

Trophic model of seamount ecosystems

In this study, we used a general ecosystem model developed for North Atlantic seamounts (Morato and Pitcher 2002) based on the "Ecopath with Ecosim" approach (EwE; http://www.ecopath.org), a software for ecosystem trophic mass-balance analysis (Ecopath), with a dynamic modeling capability (Ecosim) (for details see Christensen and Walters 2004). This model was developed for a theoretical isolated seamount. Habitat covered by the model was defined by the summit, set at around 300 m, down to the base of the seamount at around 2,000 m. The model covered a small area of about 3,000 km^2 and included 37 functional groups: three marine mammal groups (i.e., toothed whales, baleen whales, and dolphins), seabirds, turtles, seven invertebrate groups (i.e., benthic filter feeders, such as corals or gorgonians, benthic scavengers, benthic crustaceans, pelagic crustaceans, seamount resident cephalopods, small and large drifting cephalopods), three zooplankton groups (i.e., gelatinous, shallow, and deepwater zooplankton), primary producers (i.e., phytoplankton), detritus, and twenty fish groups assembled according to their environmental preferences (i.e., depth and habitat: e.g., benthic, pelagic, or benthopelagic), body size, energetics, and life-history characteristics (see Appendix 1 and Morato and Pitcher 2002 for a complete description of the model).

The theoretical seamount was assumed to have a low initial level of exploitation and its fisheries were loosely based on those operating at the Azores/Mid-Atlantic Ridge comprising 6 fleets (see Appendix 1 and Morato et al. 2001): (a) demersal longline (targeting shallow water demersal and benthic fishes); (b) deepwater longline (targeting bathypelagic and bathybenthic fishes); (c) small pelagics fishery (for small pelagic fishes); (d) tuna fishery; (e) swordfish fishery; and (f) deepwater trawl (targeting seamount associated fishes, including orange roughy and alfonsinos, *Beryx splendens* and *B. decadactylus*). Landings, prices, and job estimates were loosely based on the Azores case study (Morato et al. 2001, Morato and Pitcher 2002).

Ecopath outputs are known to be very sensitive to the vulnerability parameters (see Walters et al. 1997, Pitcher and Cochrane 2002). In this study a standard value of 0.3 representing mixed predator/prey control was used. A brief sensitivity analysis of the policy simulations to different vulnerability settings was conducted by repeating simulations with vulnerabilities of 0.2, 0.3, and 0.5. Results from simulations were generally consistent between different vulnerabilities.

Model analyses

The impacts of alternative time patterns of fishing mortalities were explored using an optimization method (Walters et al. 2002, Christensen and Walters 2004), to search for patterns of relative fishing effort by fishing fleets, which would maximize one or more of the considered objectives:

1. “Economy,” or net present economic value (i.e., total present value of the catch).

2. “Jobs,” or employment (i.e., a social indicator, assumed proportional to gross landed value of catch for each fleet with a different jobs/landed value ratio for each fleet).

3. “Ecology,” or ecological “stability” (i.e., measured by assigning a weighting factor to each group based on their longevity, and optimizing for the weighted sum).

Net present economic value of landed catches was calculated as the discounted sum over all fleets and times of catches multiplied by the prices of landed fish species. A discount rate of 0.04 was used. The ecological criterion component is based on Odum’s (1971) definition of “maturity,” with mature ecosystems being dominated by large, long-lived organisms. Thus, it is intended to identify the fleet structure that maximizes biomass of long-lived organisms, defined by the inverse of their production/biomass ratios (P/B).

The objective function can be thought of as a “multi-criterion objective function,” represented as a weighted sum for the three above-mentioned criteria indicators:

$$OBJ = W_V \times \sum NPV_{ijt} + W_j \times \sum J_{ijt} + W_E \times \sum ({}^{B}\!/_{P})_{ijt} + \varepsilon$$

where W = weighting factors; V = value; J = jobs; E = ecology; I = fleets; j = species caught; t = time in years; NPV = net present value; and B/P = biomass production ratio, assumed to be proportional to species longevity and thus ecological stability, with ε a normally distributed error term.

The Davidon-Fletcher-Powell (DFP) nonlinear optimization procedure was used to iteratively optimize the three above-mentioned objectives by changing relative fishing rates (F) (Walters et al. 2002). This search procedure results in what control systems analysts call an "open loop policy"; a recommendation for what to do at different future times without reference to what the system actually ends up doing along the way (Christensen and Walters 2004). The resulting "optimum" fishing rates by year/fleet served as input for the dynamic simulation, "Ecosim," where they replaced the baseline relative efforts by fleet/gear type. Ecosim was then run for a 50-year period to simulate the effect of the optimized fishing rates and to estimate the biomass, catch, and value variation. These two scenarios were compared with a "no fishing" scenario where all the fishing rates were set to 0.

Nonlinear optimization methods, such as DFP, can be difficult to use and can be misleading. In particular, the method can "hang up" on a local maximum and can give extreme answers due to an inappropriate objective function (Walters et al. 2002). To check for false convergence to local maxima, random starting Fs were used. To search for trade-offs among objective functions, optimal scenario solutions for a range of weightings of ecological and economic objectives were accessed. Additionally, at the end of each run, ecosystem indicators such as the mean trophic level of the catch (see equation in Pauly et al. 1998) and biodiversity index (modified from Kempton and Taylor 1976, Q75; Ainsworth and Pitcher 2004) which resulted from the suggested fishing effort in each range of weighting), were estimated.

Results

Optimal fishing scenarios

The optimized fishing rates (F) for the "economy" and "ecology" objectives, expressed as proportions of the base model fishing rates are summarized in Fig. 1. Maximizing economic value led to an increase in fishing rate in all fisheries (deepwater trawl, $F_{final}/F_{base} = 97.6$; swordfish fisheries ($F_{final}/F_{base} = 18.1$); small pelagic fisheries, $F_{final}/F_{base} = 16.2$), except for the deepwater longline, where the fishing rate was reduced to 0.86 of the base model value. In contrast, maximizing "ecosystem" stability led to a large decrease in all fishing rates (swordfish fisheries. ($F_{final}/F_{base} = 0.01$; demersal longline, $F_{final}/F_{base} = 0.16$; deepwater trawl, $F_{final}/F_{base} = 0.32$).

The effects of the optimized fishing rates on biomass (i.e., percentage of biomass change from base model) after a 50-year simulation are

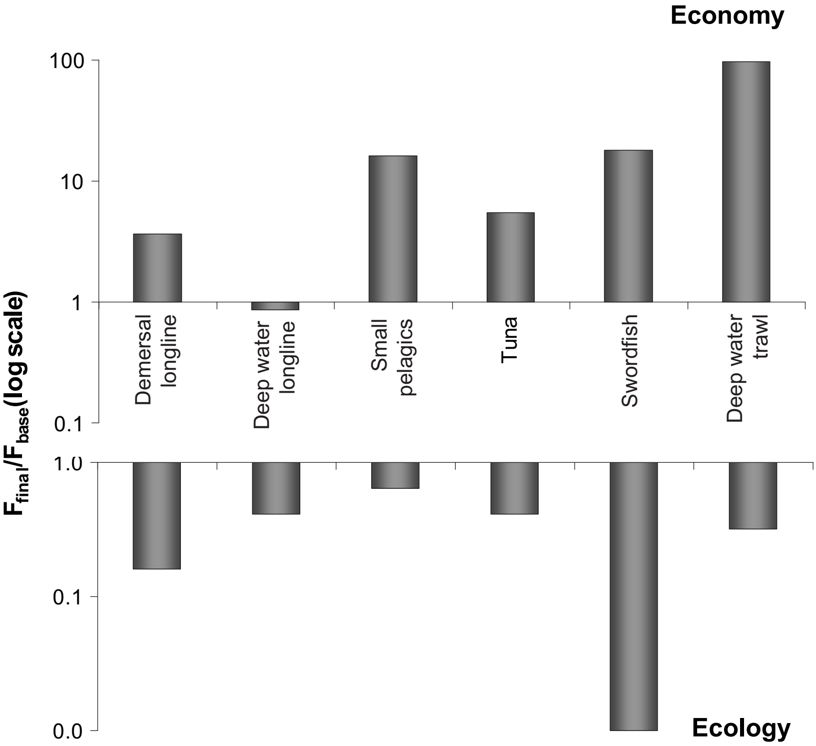

Figure 1. Optimized fishing rates (*F*), expressed as proportions of the base model, obtained by maximizing "economy" and "ecology" objectives. Note differences in scale.

presented, together with a "no fishing" scenario, in Fig. 2. Not surprisingly, the "no fishing" scenario produced a general increase in biomass for most of the groups, but particularly for sea turtles, rays and skates, and pelagic sharks. However, this was not the case for the most important prey groups of the system: mesopelagic fish and benthic invertebrates. The optimized fishing rates for the "ecology" objective function produced very similar results when compared with the "no fishing" scenario, producing a large increase in groups that have slow turnover and higher trophic levels. When economic value was maximized, a general decrease in biomass was observed associated with collapse of ten functional groups (pelagic sharks, tunas, benthopelagic sharks, seamount-associated fishes, bathypelagic fishes, sea turtles, rays and skates, alfonsinos,

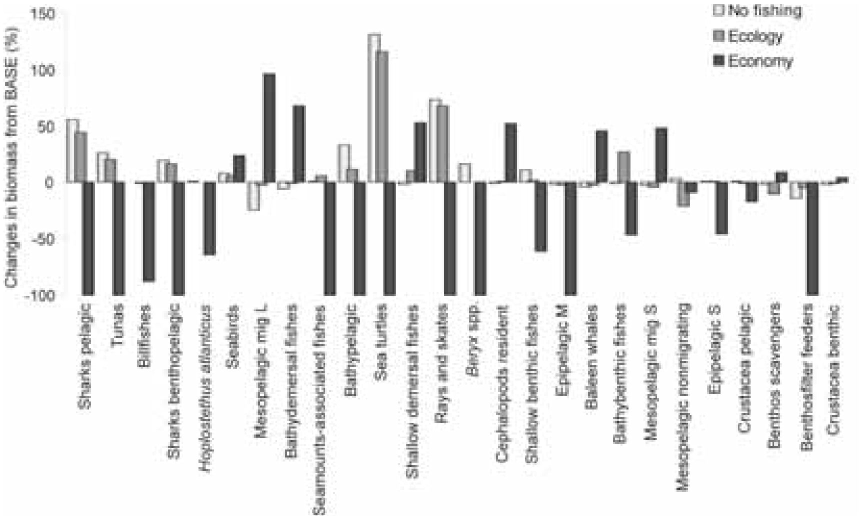

Figure 2. Changes in group biomasses (percent change of biomass from base model) under the three different fishing scenarios: no fishing, maximizing the "ecology" objective, and maximizing the "economy" objective. S = small; M = medium; L = large.

medium-sized epipelagic fish, and benthic invertebrate filter feeders, e.g., deep-sea corals).

The effects of the different fishing policies on the total landed catches are shown in Fig. 3. Maximizing the "economy" objective led to an increase in landings when compared to the base model. In this scenario, the deepwater trawl fishery was favored and had the highest contribution to the total catch. In contrast, maximizing the "ecology" objective required an overall decrease in catches and fishery operations conducted mostly by small pelagic and bottom longline fishing fleets.

Comparing the total value of the catches for the three scenarios (i.e., base model, maximizing "economy," and maximizing "ecology"; Fig. 4), maximizing "economy" generated more money than the base model and the "ecology" scenario. In all cases, deepwater trawl and bottom longline fishing fleets contributed the most to the total value.

Trade-offs

Surface plots of optimal scenario solutions for a range of weightings of ecological and economic objective functions are shown in Figs. 5-7. They show that it was not possible to maximize the performance of all three objectives (i.e., net economic value, number of jobs, and ecological

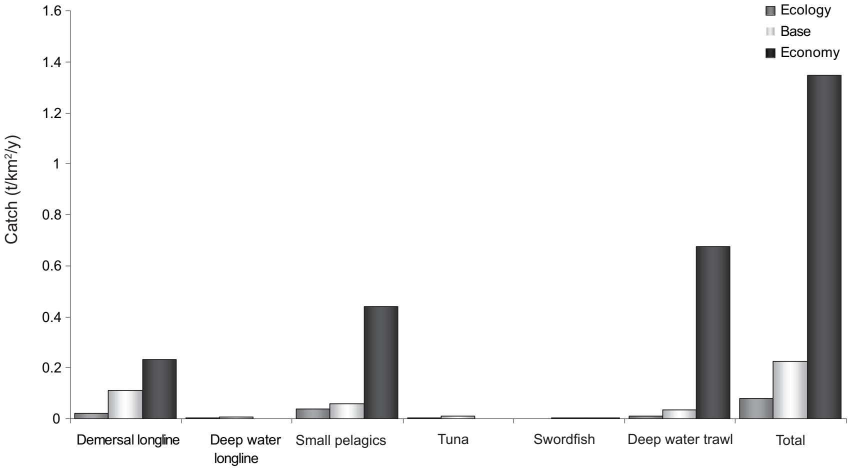

Figure 3. Catches (t km^{-2}y^{-1}) for the different fishing fleets under the base model and two fishing scenarios: maximizing the "ecology" objective function, and maximizing the "economy" objective function.

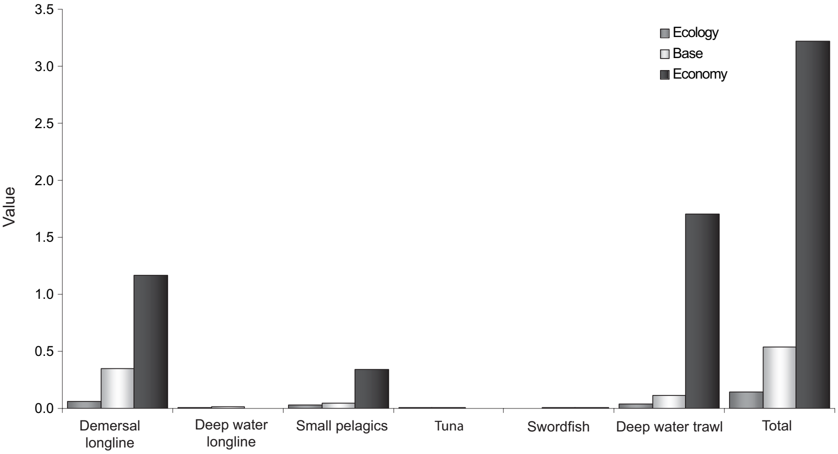

Figure 4. Value of the catches (relative value) for the different fishing fleets under the base model and two fishing scenarios: maximizing the "ecology" objective, and maximizing the "economy" objective.

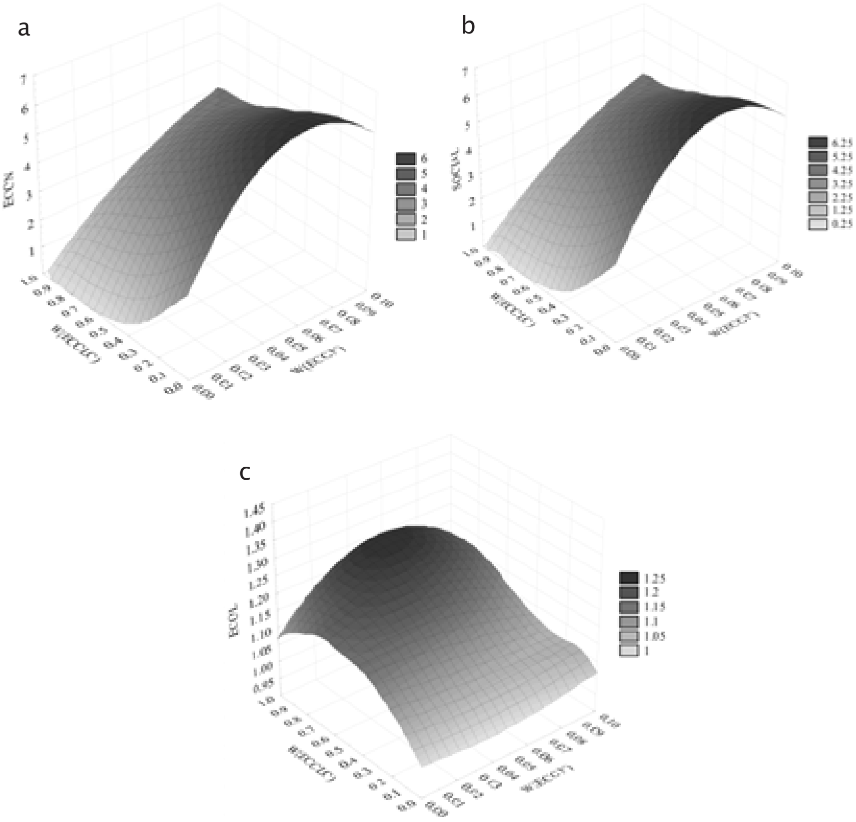

Figure 5. Surface plots showing optimal scenario solutions for a range of weightings of ecological and economic objective functions: (a) performance of "economy" objective, maximizing net economic value; (b) performance of "social" objective, maximizing number of jobs; (c) performance of "ecology" objective, maximizing ecological "stability" of the ecosystem.

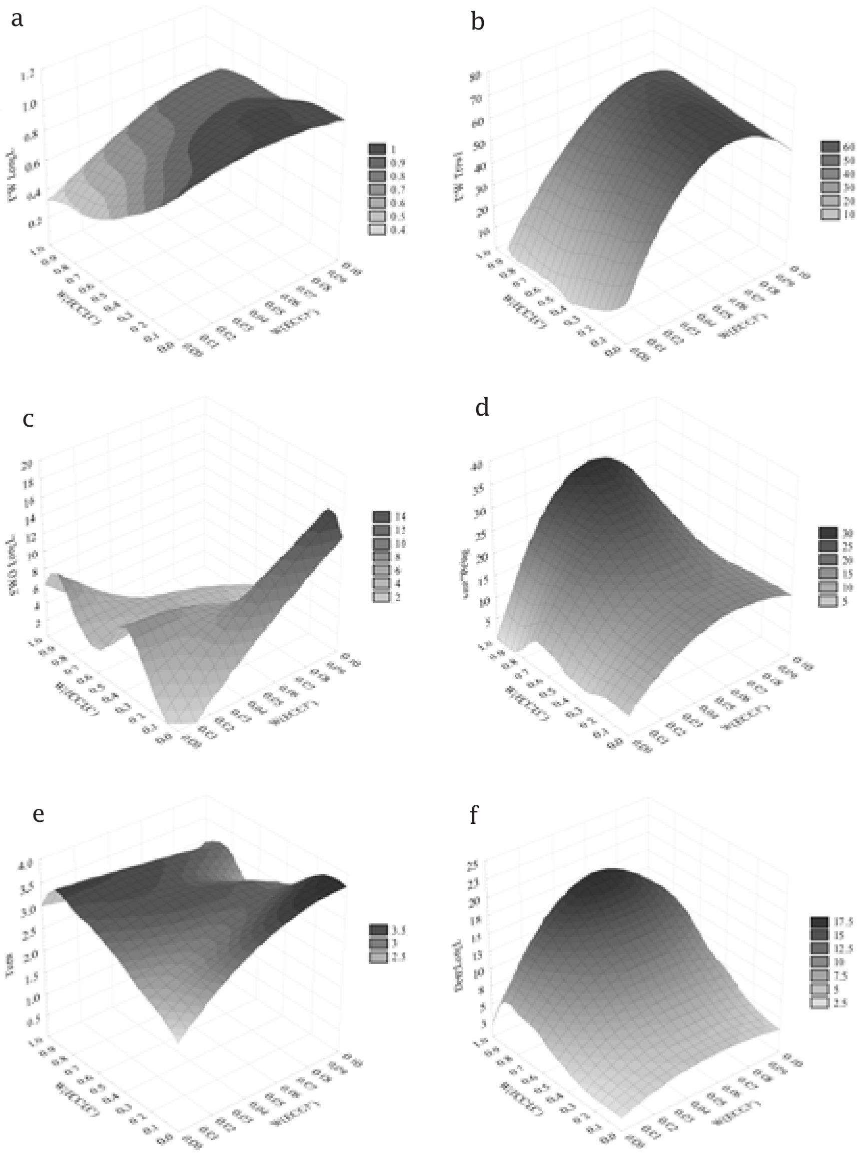

Figure 6. Surface plots showing the resulting fishing rates, as proportion of base model rates, of the optimal scenario solutions for a range of weightings of ecological and economic objective functions: (a) deepwater longline; (b) deepwater trawl; (c) pelagic longline; (d) small pelagic fishery; (e) tuna fishery; (f) demersal (bottom) longline.

"stability") simultaneously. This is true because net economic value (Fig. 5a) and number of jobs (Fig. 5b) reach a maximum with a high weighting factor on the economy objective and a small weight on ecology. This results in a decrease in the stability of the system. To the contrary, to maximize "ecosystem stability" a high weighting was assigned to the "ecology" objective (Fig. 5c). Assigning a low weighting factor to "economy" and a high weighting factor to "ecology" resulted in a decrease of net economic value and number of jobs, with a corresponding increase in the system's stability. Intermediate weightings produced, in general, intermediate performances for the three objective functions.

The fishing rates required to achieve different performances of the objective functions (i.e., fishing policies) are presented in Fig. 6. In order to maximize the net economic value of the system all fisheries, except deepwater longline (Fig. 6a), required an increase in their fishing rates. These increases were approximately 50 times the base model rate for the deepwater trawl fishery (Fig. 6b), 16 times for the swordfish fishery (Fig. 6c), 15 times for the small pelagic fish fishery (Fig. 6d), 3.5 times for the tuna fishery (Fig. 6e), and 2.5 times for the bottom longline fishery (Fig. 6f). To achieve ecological stability in the system, however, a decrease in the fishing rates of most of the fisheries was required with the exception of the tuna, swordfish, and to a lesser extent the bottom longline fisheries. The latter, along with the small pelagic fisheries, reached their highest fishing rates when a high weighting factor was assigned to the "ecology" objective and an intermediate weight to "economy."

Ecosystem indicators (i.e., mean trophic level of the catch and biodiversity) and total catches derived from the optimal fishing strategies for the overall range of weighting factors for "ecology" and "economy" are presented in Fig. 7. Total catches were maximized when weighting was high for "economy" and low for "ecology" objective functions (Fig. 7a). In contrast, the biodiversity index (Fig. 7b) was high only when a very small weight was assigned to "economy." The mean trophic level of the catch (Fig. 7c), in general, decreased with a corresponding increase in the weighting of "economy" and a decrease in the weighting of "ecology" objective functions. However, maximum trophic level was achieved with a high weighting of "ecology" and an intermediate weighting of "economy."

Discussion

The analyses in this paper are not meant to describe actual fisheries for seamounts, but rather as an exercise to explore the overall responses of seamount ecosystems to various multispecies management strategies.

The use of "open loop policy" search procedures can be unrealistic because it can entrust a fishery to fishing rates calculated at some time in the past and from data available from that time (Walters et al. 2002,

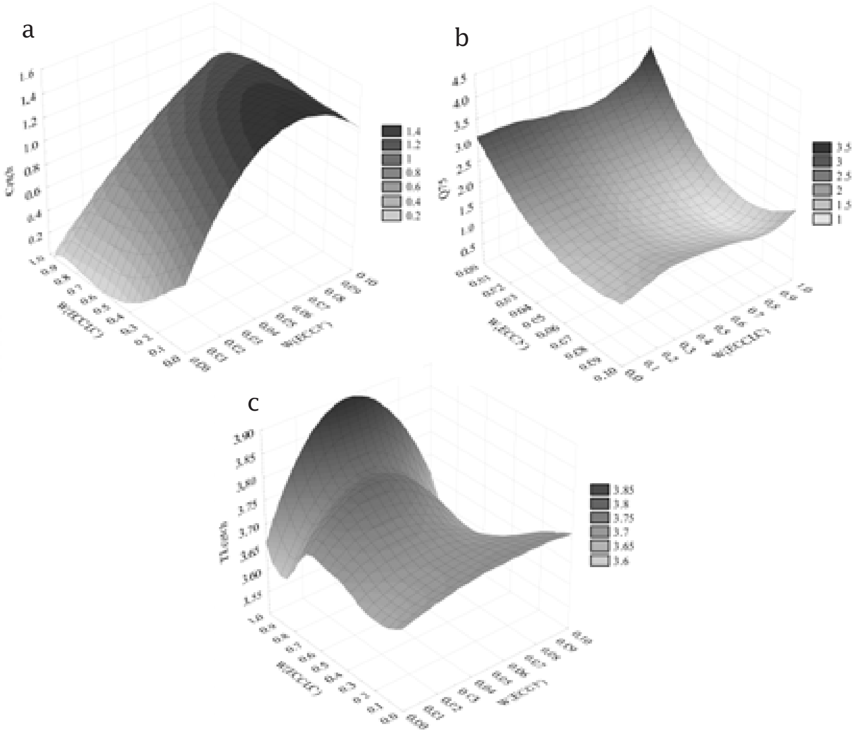

Figure 7. Surface plots showing the resulting ecosystem indicators and total fisheries catch for the optimal scenario solutions for a range of weightings of ecological and economic objective functions: (a) total catch; (b) biodiversity index (Q75) (note that this figure is shown from a different viewpoint); (c) mean trophic level of the catch.

Christensen and Walters 2004). Fisheries management needs to be implemented using "feedback policies" in which harvest goals are adjusted over time as new information becomes available and in response to unpredicted ecological changes (Christensen and Walters 2004). However, "open loop policy" calculations can give insights and directional guidance to where the system might be heading. In this study, this method appeared to be appropriate due to both the exploratory characteristic of the study and the data-limited situation of the North Atlantic seamounts.

Different extreme policy objectives for seamount fisheries may require different fleet configurations. Simulations that maximize economic performance favor deepwater trawling and require an increase in the fishing rates of all other fishing fleets, the only exception being the deepwater longline. On the other hand, maximization of ecological performance is achieved by favoring the operation of small pelagic and bottom longline fisheries. At the same time, a decrease in the fishing rates of all other fishing fleets is necessary.

Different fishing rates and fleet configurations produced different impacts on catches and consequently in the whole ecosystem. Optimizing for economics yielded six times the amount of landed catch and money than the base model scenario, and 17 and 23 times the amount of landed catches and money yielded by the scenario where ecology was maximized. This would, however, have implications to the whole ecosystem. While maximizing ecology produces an overall increase in biomass of most functional groups in the model, maximizing economics leads to a decrease and further collapse of some groups such as tuna, seamount-associated fishes, alfonsinos, as well as some charismatic species such as sea turtles and sharks. This point was well illustrated some time ago by Clark (1973) who pointed out that for populations that are economically valuable but possess low reproductive capacities (such as seamount-associated fishes, alfonsinos, and sharks), common property competitive exploitation and private property maximization of profits may lead to overexploitation and even to extinction of the population.

It is interesting to note that major collapses in deepwater fisheries, for example off New Zealand (Clark 1999, Clark et al. 2000), Tasmania (Koslow et al. 2000), and Namibia (Boyer et al. 2001), and habitat degradation (Probert et al. 1997, Koslow et al. 2001) were attributed to extensive deepwater trawling. In the Azores where no deepwater trawling is known to occur, seamount fisheries are mainly longline, handline, and pole-and-line, and are believed to be more sustainable. However, signs of stock decline are becoming apparent even in these systems (Santos et al. 1995, Menezes 2003). Thus, the question of whether deepwater (mainly trawl) fisheries are sustainable in the long term remains open (Clark 2001). Some authors (e.g., Probert 1999, Roberts 2002), agencies (World Wildlife Fund [WWF], International Union for Conservation of Nature and Natural Resources [IUCN]), and governments strongly advocate an urgent need for fishing regulations and/or the establishment of marine reserves in such areas.

It is apparent that major conflicts among stakeholders might emerge when different optimization scenarios result in completely different fishing policies (Figs. 5, 6). In addition, our results illustrate that maximizing "economy" affects biodiversity in the ecosystem and probably the trophic level of the catch, while maximizing the total landed catches (see Fig. 7). The opposite is true when "ecology" is favored; the total catch and the number of jobs are decreased in order to achieve high biomass of long-lived species and increased biodiversity in the ecosystem.

In conclusion, sustainable seamount fisheries with minimal ecosystem effects appear to be achieved when the "ecology" objective is maximized. However, more information for these fragile ecosystems and the long-term impacts of fishing and other human activities needs to be acquired. Meanwhile, the precautionary principle ought to be applied to seamount ecosystems, in order to ensure protection and sustainable management.

Acknowledgments

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Appendix 1

Modeling approach

The model parameters (Table A1), production to biomass ratios (P/B), and consumption to biomass ratios (Q/B), are calculated on a yearly basis. Biomass and catch are expressed in metric tons of wet weight per square kilometer. The Q/B ratios for fish groups were estimated using an empirical equation (Palomares and Pauly 1998). Temperature values were established as being 18ºC for the epipelagic region (0-200 m), 8ºC for the mesopelagic region (200-1,000 m), and 6ºC for the bathypelagic region (1,000-4,000 m). For some groups Q/B values were taken from other models. For most groups P/B ratios were extracted from previously constructed models, or were estimated assuming production and consumption ratio equal to 0.3 (Christensen 1996). Following Shannon and Jarre-Teichmann (1999), the proportion of food consumed and not assimilated was taken as 0.2. When no biomass estimate was available, this parameter was left to estimate by Ecopath using a value of 0.95 for ecological efficiency (EE). A preliminary diet matrix was assembled using published data, unpublished local information, or empirical knowledge. When unidentified categories were found in the literature, data were re-expressed out of 100% to exclude these groups.

The theoretical seamount was assumed to have a low initial level of exploitation. Landings (Table A2) varied from 218 t per year for shallow benthic fishes to 2.7 t per year for billfishes.

Table A1. Input parameters and estimates (in parentheses) from the theoretical non-migratory model of a seamount. P/B is production to biomass ratio, Q/B is consumption to biomass ratio, EE is ecotrophic efficiency, and TL is trophic level of the groups.

Group name	Biomass (t km^{-2})	P/B (year^{-1})	Q/B (year^{-1})	EE	TL	Landings (t km^{-2})
Toothed whales	0.000	0.020	10.270	(0.513)	5.17	
Baleen whales	0.123	0.060	5.563	(0.024)	3.56	
Dolphins	0.040	0.070	11.410	(0.050)	4.58	
Sea turtles	0.001	0.150	3.500	(0.899)	4.08	
Seabirds	0.000	0.040	84.390	(0.000)	4.36	
Tunas	0.032	0.742	16.291	(0.686)	4.58	0.011
Billfishes	0.020	0.500	4.200	(0.101)	4.54	0.001
Sharks pelagic	0.011	0.300	3.100	(0.868)	4.70	0.002
Sharks benthopelagic	0.030	0.510	6.900	(0.160)	4.40	0.002
Rays and skates	0.020	0.170	1.500	(0.678)	3.91	0.002
Large oceanic planktivores	(0.003)	0.112	2.066	0.100	3.56	
Epipelagic S	0.859	2.053	19.867	(0.567)	3.10	0.050
Epipelagic M	0.113	1.080	10.750	(0.902)	3.59	0.010
Epipelagic L	0.014	0.690	5.095	(0.487)	4.18	
Mesopelagic migrating S	2.000	1.980	8.000	(0.981)	3.37	
Mesopelagic migrating L	(0.003)	0.600	3.550	0.950	4.34	
Mesopelagic non-migrating	(1.421)	0.500	1.570	0.950	3.12	
Shallow benthic fishes	(0.820)	0.590	4.700	0.950	3.70	0.080
Shallow benthic fishes	(0.820)	0.590	4.700	0.950	3.70	
Shallow demersal fishes	(0.193)	0.660	5.200	0.950	4.04	0.020
Seamount-associated fishes	(0.890)	0.060	2.200	0.950	4.14	0.011
Hoplostethus atlanticus	(0.452)	0.048	2.000	0.900	4.39	0.010
Beryx spp.	(0.343)	0.060	2.000	0.950	3.90	0.010
Bathypelagic	(0.029)	0.500	1.477	0.950	4.12	0.006
Bathybenthic fishes	(1.143)	0.200	0.500	0.950	3.54	0.003
Bathydemersal fishes	(1.283)	0.200	0.600	0.950	4.19	0.002
Benthos filter feeders	(0.595)	0.800	9.000	0.950	2.00	
Benthos scavengers	(3.089)	1.830	13.567	0.950	2.35	
Crustacea benthic	(3.858)	1.600	10.000	0.950	2.00	
Crustacea pelagic	(5.161)	1.450	9.667	0.950	2.72	
Cephalopods resident	(0.189)	2.890	10.000	0.950	3.78	
Cephalopods drifting S	(0.175)	4.450	16.863	0.950	3.83	
Cephalopods drifting L	(0.001	2.500	10.000	(0.726)	4.33	
Gelatinous zooplankton	(8.895)	0.850	2.000	0.800	3.08	
Zooplankton shallow	16.684	(11.214)	37.379	(0.774)	2.11	
Zooplankton deep	6.849	(8.700)	29.000	(0.595)	2.23	
Phytoplankton	7.160	290.000	–	(0.358)	1.00	
Detritus	–	–	–	(0.045)	1.00	

S = small; M = medium; L = large.

Table A2. Average landings estimated for the different fisheries considered in the theoretical seamount.

	Landings by fleet (t km^{-2}y^{-1})						Total
Group name	DL	DWL	SP	T	SW	DWT	(t y^{-1})
Tunas				0.011			30.0
Billfishes					0.001		2.7
Sharks pelagic	0.001				0.001		5.5
Sharks benthopelagic	0.001					0.001	5.5
Rays and skates	0.002						5.5
Epipelagic S			0.05				136.4
Epipelagic M			0.01				27.3
Shallow benthic fishes	0.08						218.2
Shallow demersal fishes	0.02						54.6
Seamount-associated fishes		0.001				0.01	30.0
Hoplostethus atlanticus						0.01	27.3
Beryx spp.	0.005					0.005	27.3
Bathypelagic		0.005				0.001	16.4
Bathybenthic fishes		0.002				0.001	8.2
Bathydemersal fishes	0.001	0				0.001	5.5
Total t km^{-2}y^{-1}	0.11	0.008	0.06	0.011	0.002	0.029	

DL = demersal longline; DWL = deepwater longline; SP = small pelagics fishery; T = tuna; SW = swordfish; DWT = deepwater trawl; S = small; M = medium.

Minimum Stock Size Thresholds: How Well Can We Detect Whether Stocks Are below Them?

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Abstract

Management of marine fisheries in U.S. waters is based on the Magnuson-Stevens Fishery Conservation and Management Act. Rebuilding plans need to be developed for fish stocks that have been depleted to below a minimum stock size threshold, MSST. Whether a stock is below MSST is based on the results from a stock assessment. Two types of error can arise when a stock is assessed relative to MSST: (a) it can be assessed to be above MSST when it is not, or (b) it can be assessed to be below MSST when it is not. Simulation is used to assess the likelihood of making these two types of errors as a function of the true status of the resource, the stock assessment method applied, and the quality and quantity of the data available for assessment purposes. All three of the methods of stock assessment considered in this study (two age-structured methods and a production model) make the two errors, especially when the true status of the resource is close to MSST. The major factor influencing the likelihood of under- and over-protection errors is the extent of variability in recruitment, the impact of which is larger than that of data quality and quantity, at least within the range for data quality and quantity considered in this paper.

Introduction

The objectives for the management of the world's marine renewable resources generally include striking an appropriate balance between "optimum" utilization of the available resources for the benefit of the nation involved and the long-term conservation of the resources and their associated ecosystem. In the United States, the need for this balance is reflected in National Standard 1 of the Magnuson-Stevens Fishery Conservation and Management Act, viz. "Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States industry."

National Standard 1 has been made operational through a system of guidelines (e.g., Restrepo et al. 1999, Restrepo and Powers 1999). These guidelines distinguish between overfishing and being in an overfished state. "Overfishing" means that the level of fishing mortality exceeds a maximum fishing mortality threshold (MFMT), which is currently set at the rate associated with maximum sustainable yield (MSY), and "being in an overfished state" means that the current spawning output is less than a minimum stock size threshold (MSST). For many stocks, MSST has been defined to be half of S_{MSY} (the spawning output[1] corresponding to MSY). Stocks that are found to be below MSST are determined to be overfished (i.e., depleted) and there is a need for the National Marine Fisheries Service to develop a rebuilding plan to restore the stock to S_{MSY}, which is then treated as the target level of spawning output. Assessing a stock relative to some management threshold level (be it S_{MSY}, 0.5 S_{MSY}, or some proxy level) will be referred to as a status determination in this paper.

The use of S_{MSY} as the target for fisheries management can be criticized for a variety of reasons (e.g., Larkin 1997, Punt and Smith 2001). However, it remains the most common target reference point for fisheries management. For example, legislation in New Zealand dictates that management arrangements must be selected to move the resource toward S_{MSY} (Annala 1993).

Although ideally MSST should be defined in terms of S_{MSY}, the use of proxies for both the target spawning output and MSST are permitted because the data for particular species may be insufficient to estimate the shape of the relationship between spawning output and subsequent recruitment. For example, for groundfish species managed by the Pacific Fishery Management Council, the proxy for MSST has been set to 25% of the estimated unfished level of spawning output, S_0, and the target level has been set to 40% of S_0 (Pacific Fishery Management Council 2003). Figure 1 shows the control rule used by the Pacific Fishery Management Council to set optimum yields for groundfish species off the U.S. West Coast (Pacific Fishery Management Council 2003).

The ability to apply control rules such as Fig. 1 requires that it is possible to estimate a variety of quantities (current spawning output, *MSY*, and S_{MSY} or their proxies). The estimates for these quantities are derived from stock assessments. Several analytical methods are used to conduct stock assessments in the United States (e.g., National Research Council 1998). However, the bulk of the assessments are conducted using two basic approaches: ADAPT (Gavaris 1988) and Integrated Analysis (Fournier

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[1] Spawning output is variously defined as egg production or the biomass of spawning fish.

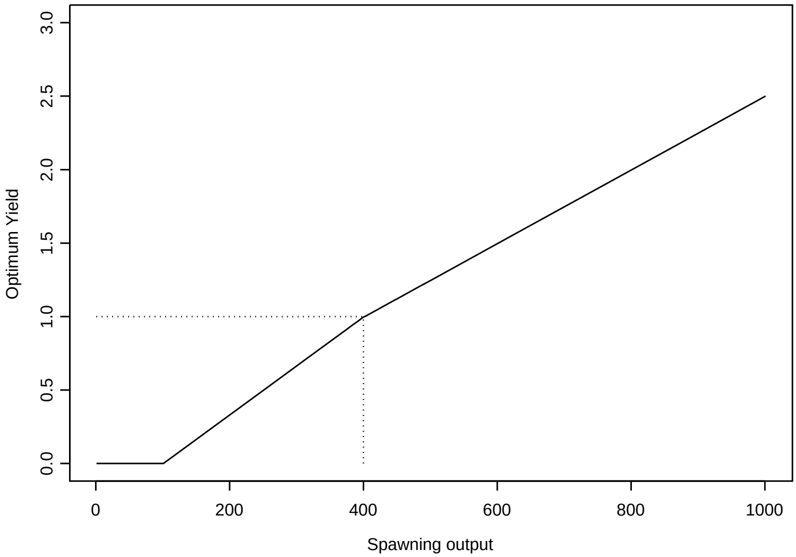

Figure 1. An example of the 40-10 control rule applied to U.S. West Coast groundfish species.

and Archibald 1982; Methot 1993, 2000). Integrated Analysis is currently the "method of choice" for assessments of the groundfish species managed by the Pacific Fishery Management Council (e.g., Jagielo et al. 2000, Williams et al. 2000, Hamel et al. 2003).

Unfortunately, it is well known that stock assessments are subject to not inconsiderable uncertainty, especially in data-poor situations. In the context of conducting evaluations relative to the application of control rules (such as that in Fig. 1), the questions that arise include: what is the probability that a stock is assessed to be above MSST when it is not (under-protection error), and what is the probability that a stock is assessed to be below MSST when it is not (over-protection error). The probability of making these two errors depends on the quality of the data available for assessment purposes and the suitability of the population dynamics model underlying the stock assessment.

Simulation is therefore used in this study to assess the likelihood of making these two types of errors as a function of the true status of the resource, the stock assessment method applied, and the quality and quantity of the data available for assessment purposes.

Methods

The most common method used to determine how well a stock assessment method is likely to perform is Monte Carlo simulation (e.g., de la Mare 1986, Patterson and Kirkwood 1995, Sampson and Yin 1998, Punt et al. 2002). Evaluation of the properties of a statistical estimation method (including its bias and precision) by simulation involves the following steps.

1. Definition of a mathematical model of the system to be assessed; this model (often referred to as the operating model) will represent the truth for the simulations.
2. Use of the operating model to generate the data sets that will be used by the assessment methods.
3. Application of a number of alternative stock assessment methods to the generated data sets.
4. Comparison of the estimates of stock status provided by the stock assessment methods with the true state of the stock as given by the operating model.

Although some of the stock assessments for West Coast groundfish species have evaluated the status of stocks relative to the target level of spawning output and MSST in probabilistic terms (e.g., Ianelli et al. 2000, Hamel et al. 2003, Cope et al. 2004), the bulk of stock assessments for these species is based on the "best" estimates of quantities such as current spawning output, S_0, etc. Although there is a need to evaluate probabilistic methods for assessing fish stocks, this study focuses on a more immediate need, namely to evaluate stock assessment methods that base their status determinations on the point estimates from a stock assessment.

Each of the 250 replicates that constitute a simulation trial therefore involves generating an artificial data set for which the true status relative to S_{MSY} and MSST are known exactly, applying each of the assessment methods under consideration to estimate the time-series of historical spawning outputs and S_{MSY}, and comparing how often the stock assessment correctly determines the status of the resource relative to the S_{MSY} and MSST. The performances of the various stock assessment methods are assessed relative to the following questions.

a. Is the stock below S_{MSY} at present?

b. Is the stock below 0.4 S_0 at present?

c. Is the stock below 0.5 S_{MSY} at present?

d. Is the stock below 0.25 S_0 at present?

The simulations therefore consider performance relative to both the target level of S_{MSY} and MSST. Consideration was given to assessing performance relative to the proxies for S_{MSY} and MSST as well as to S_{MSY} and MSST themselves (0.4 S_0 is the proxy for S_{MSY} and 0.25 S_0 is the proxy for 0.5 S_{MSY}) because of initial concern that it may prove very difficult to estimate S_{MSY} reliably (e.g., Maunder and Starr 1995) using the (noisy and sparse) data collected from the fishery.

The operating model

The operating model (see Appendix) is age-structured, relates recruitment to spawning output by means of a Beverton-Holt stock recruitment relationship, and assumes that selectivity is related to age according to a logistic curve. Allowance is provided in the model for process error by assuming that the annual deviations about the stock-recruitment relationship are log-normally distributed. The information available for assessment purposes includes catch (in mass), catch-rates, and age-composition data for the fishery catches. The latter data sources are subject to observation error (log-normal for catch-rates and multinomial for fishery catch age-compositions). The operating model has, however, many simplifications, including its assumption that natural mortality is independent of age and time, and that selectivity is time-invariant. These simplifications are necessary because examination of more complicated options would have led to excessive computational and presentational demands.

Figure 2 illustrates the three catch histories (stable, increasing, and increasing and declining) used in the simulations. The third of these ("Catch history 3" in Fig. 2) forms the "reference case" for the analyses in this paper because it most adequately reflects the catch history for most marine fish species.

Table 1 lists the values for the parameters of the model that are fixed for all 250 replicates of each simulation trial. The values in bold typeface form the reference case for the simulations. Sensitivity is evaluated by one change from the reference case set of specifications.

For each simulation trial, and for each of the 250 replicates that constitute that trial, it is necessary to select a set of values for the model parameters that are not fixed for each simulation [S_0 is the median spawning output at pre-exploitation equilibrium; α, β, and γ are the parameters of stock-recruitment relationships; and the annual recruitment residuals are $\varepsilon_y \sim N(0;\sigma_R^2)$]. This has been achieved as follows.

1. Given the values for *MSYL* (the ratio of the exploitable biomass at which *MSY* is achieved, B^e_{MSY}, to the average exploitable biomass in an unfished state, B^e_0) and *MSYR* (the ratio of *MSY* to B_{MSY}), the values

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[2] *MSYR* and *MSYL* are defined in terms of the exploitable component of the population rather than in terms of spawning output because they relate directly to the exploitation pattern of the fishery.

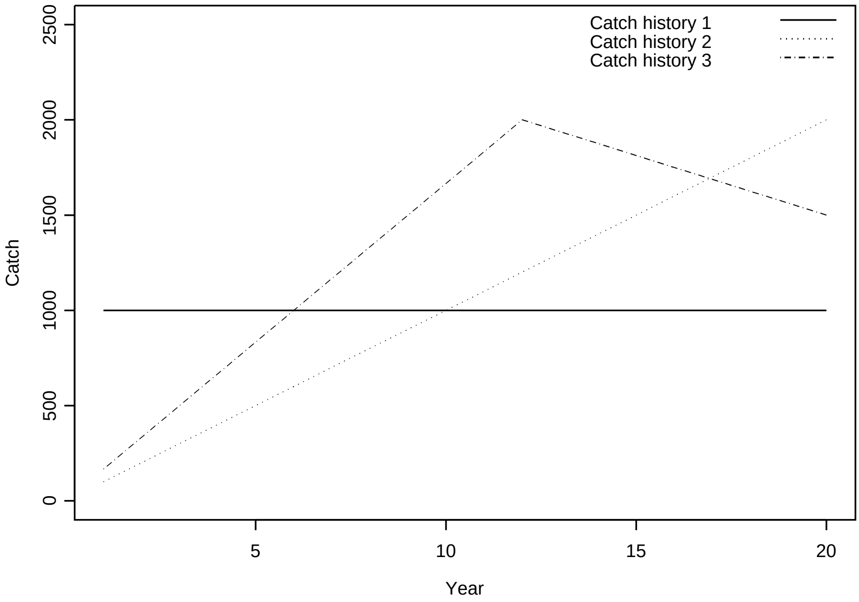

Figure 2. The three catch histories considered in this study. The simulated stocks were not harvested prior to year 1.

for α, β, and γ can be computed[2]. This involves determining the deterministic relationship between fully selected fishing mortality and yield as a function of α, β, and γ (e.g., Sissenwine and Shepherd 1987, Quinn and Deriso 1999) and solving for α, β, and γ so that *MSYL* and *MSYR* equal their pre-specified values. A byproduct of the calculation of α, β, and γ is the value for S_{MSY}, the spawning output at which the deterministic relationship between yield and spawning output is maximized.

2. The values for the ε_y for the entire 70-year period (50 years without fishing and 20 years of fishing) are generated.
3. The value for S_0 is selected so that if the model is projected from pre-exploitation equilibrium to the end of the 20-year catch series, the ratio of the spawning output at this time to S_0 equals the pre-specified current depletion[3] level (D_{init}).

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[3] The term "depletion" is used to refer to the ratio of the spawning output to S_0 (i.e., a depletion of 0.7 indicates that the spawning output is 70% of S_0).

Table 1. Values for the parameters of the operating model.

Parameter/specification	Symbol	Values
MSYR	MSY/B^e_{MSY}	0.1, **0.2**, 0.3
MSYL	B^e_{MSY}/B^e_0	0.3, **0.4**, 0.5
Current depletion	D_{init}	**0.1, 0.2, 0.3, 0.4, 0.5, 0.7**
Natural mortality	M	0.3 yr^{-1}
Age-at-maturity	a_m	2, **3**, 4
Age-at-50%-recruitment	a_r	2, **3**, 4
Extent of recruitment variability	σ_R	0.05, 0.3, **0.6**, 1

Values in bold typeface form the "reference case" for the simulations.

Table 2 lists the specifications for the data sets on which estimates of the status of the resource relative to S_{MSY} and MSST (and their proxies) are based. The scenarios range from "data-rich" to "data-poor." The values that determine the extent of variation in catchability and the sample sizes for age-composition of the fishery catches are based on the authors' experiences dealing with assessments of a wide range of species in the United States, Australia, New Zealand, and South Africa. The fourth data set type ("no-age data") examines the situation in which no catch age-composition data are available but the assessment is nevertheless based on an age-structured population dynamics model.

The stock assessment methods

Stock assessments are conducted at the end of the 20-year fishing period, and three methods of stock assessment are considered. Two of these are based on essentially the same population dynamics model as the operating model while the third is based on a surplus production model.

The age-structured stock assessment methods mimic the use of the "integrated analysis" paradigm when one is conducting assessments of even very data-poor fisheries (e.g., Cope et al. 2004) off the U.S. West Coast. These methods assume that the population was at its pre-exploitation equilibrium level at the start of the first year for which catches are available (instead of 50 years before this) and estimate the pre-exploitation equilibrium spawning output (S_0), the annual fishing mortalities, the parameters of the selectivity function, and the parameters of the stock-recruitment relationship. The two variants of the stock assessment method considered in this paper differ in that one (abbreviation "fully integrated") also estimates the annual recruitments whereas the other (abbreviation "ASPM") does not and instead assumes that recruitment is related deterministically to the stock-recruitment relationship. Only two of the parameters of the stock-recruitment relationship (α and β)

are estimated, with the third parameter (γ) being set equal to 1 (i.e., the stock-recruitment relationship is assumed to be of the Beverton-Holt form irrespective of the true form of the stock-recruitment relationship). The age-structured stock assessment methods can make use of all of the data sources (catch, catch-rate, and fishery catch age-composition data). These two methods assume that the catches and the catch-rates are log-normally distributed (the coefficient of variation for the catches is set to 0.05 to ensure that the model mimics the catch data almost exactly while the coefficient of variation for the catch-rate data is an estimated parameter) and the fishery age-composition data are assumed to be multinomially distributed. The sample size for the age-composition data is set to the minimum of the actual sample size and 100 to reflect actual practice when one is conducting assessments of West Coast groundfish species (e.g., Cope et al. 2004). The values for natural mortality, age-at-maturity, and weight-at-age are assumed, for simplicity, to be known exactly when conducting assessments.

The surplus production model (abbreviation "Schaefer model") is based on the Schaefer form of the production function and assumes that there is no error in the population dynamics equation (i.e., this is an observation error estimator). The full specifications of the surplus production model method of stock assessment considered in this paper are provided by Punt (1995).

Results

Impact of the information content of the data

Figure 3 plots percentage of simulations in which the "fully integrated" method of assessment indicates the resource to be below S_{MSY}, 0.4 S_0, 0.5 S_{MSY}, and 0.25 S_0 (i.e., below S_{MSY} and its proxy and MSST and its proxy) for the reference case operating model (for which the catch history is series 3, Fig. 2). Results are shown for actual (i.e., "true") depletion levels from 0.1 to 0.7. The solid horizontal line in Fig. 3 indicates the range of values for depletion at which the assessment should indicate the resource to be below the threshold concerned. Therefore, the ideal assessment method would provide results which are 100% for the values of depletion that are indicated by the solid horizontal line and zero for all other values. Results are shown in Fig. 3 for the four scenarios regarding data quality and quantity (Table 2).

As expected, the probability of identifying the resource to be below a threshold increases as the true value of the stock size relative to S_0 decreases. However, there are cases when this probability is substantially less than 1 when the resource is actually below the threshold and substantially larger than 0 when the resource is actually above the threshold (i.e., under- and over-protection errors). As expected, the probability of an under-/over-protection error is greatest when the actual depletion is close

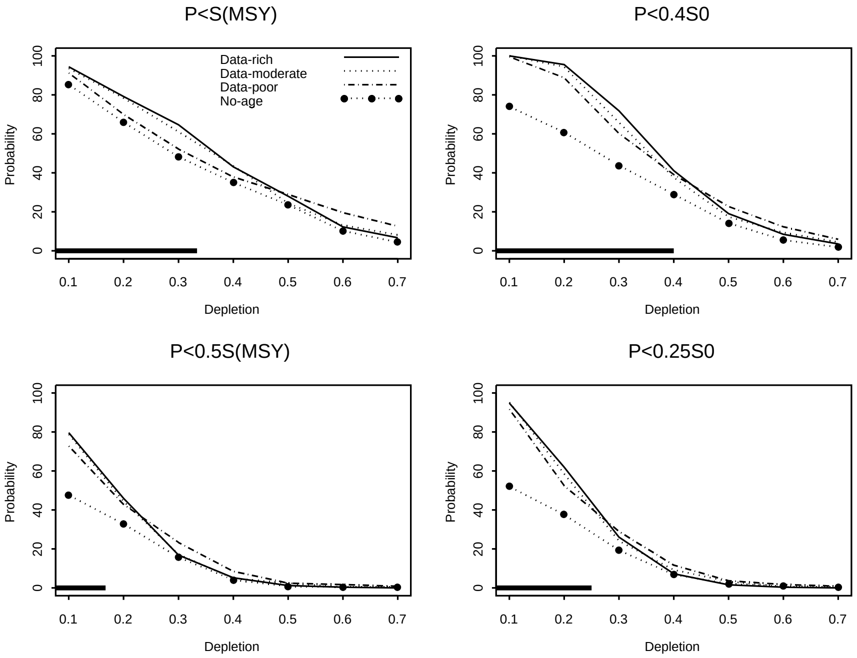

Figure 3. Percentage of simulations in which the "fully integrated" assessment method indicates that the resource is below each of the four thresholds as a function of the true value for the depletion of the resource. The results in this figure pertain to the reference case operating model (i.e., catch history 3). The horizontal line indicates the range of depletion levels that are below each true threshold level. Results are shown for the four scenarios related to data quality and quantity.

Table 2. Specifications related to data availability.

Scenario	Catch-rates	Fishery age-composition	
Data-rich	$\sigma_q = 0.15$	Every year,	$\tilde{N} = 1,000$
Data-moderate	$\sigma_q = 0.25$	Every year,	$\tilde{N} = 50$
Data-poor	$\sigma_q = 0.5$	Every 3rd year,	$\tilde{N} = 50$
No-age data	$\sigma_q = 0.25$	None	

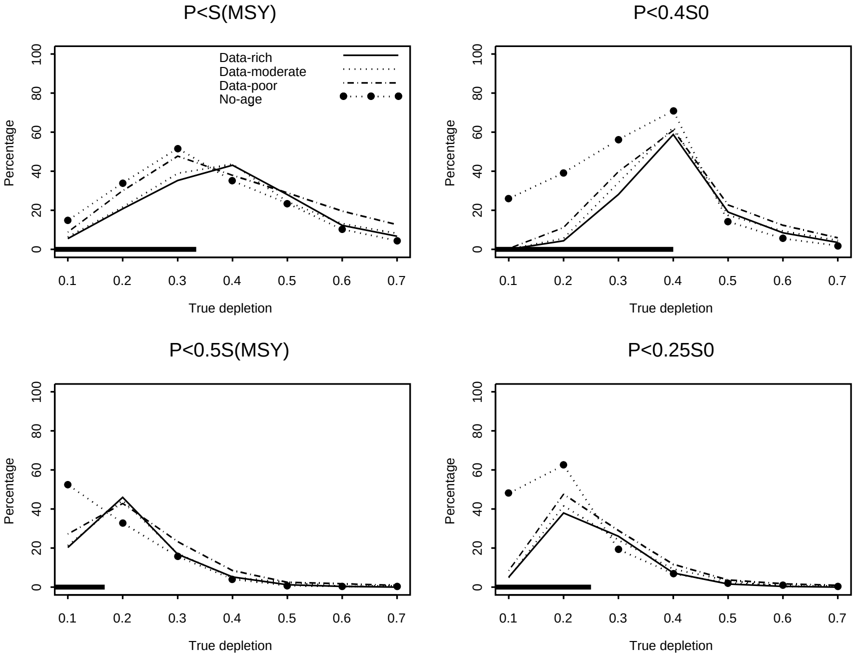

Figure 4. Percentage of simulations in which errors in status determination occur as a function of the true value for the depletion of the resource. The results in this figure relate to the use of the "fully integrated" assessment method when the data sets are generated using the reference case operating model. Under-protection errors occur for the depletion levels indicated by the horizontal line while over-protection errors occur for the remaining depletion levels.

to the threshold depletion and least when the actual depletion is much less or more than the threshold (Fig. 4). There is relatively little difference among the four data scenarios for the probability of being below S_{MSY} while the performance for the "no-age data" scenario is clearly inferior to those of the other scenarios for the other three thresholds. It is perhaps of some concern that the "no-age data" scenario leads to far higher levels of under-protection error than the other three scenarios (Fig. 4). The results for "data-rich" and "data-moderate" are essentially identical, with the performance for "data-poor" being poorer (higher occurrences of both over- and, particularly, under-protection errors).

Performance at detecting whether the resource is below the proxies 0.4 S_0 and 0.25 S_0 is superior to detecting whether the resource is below

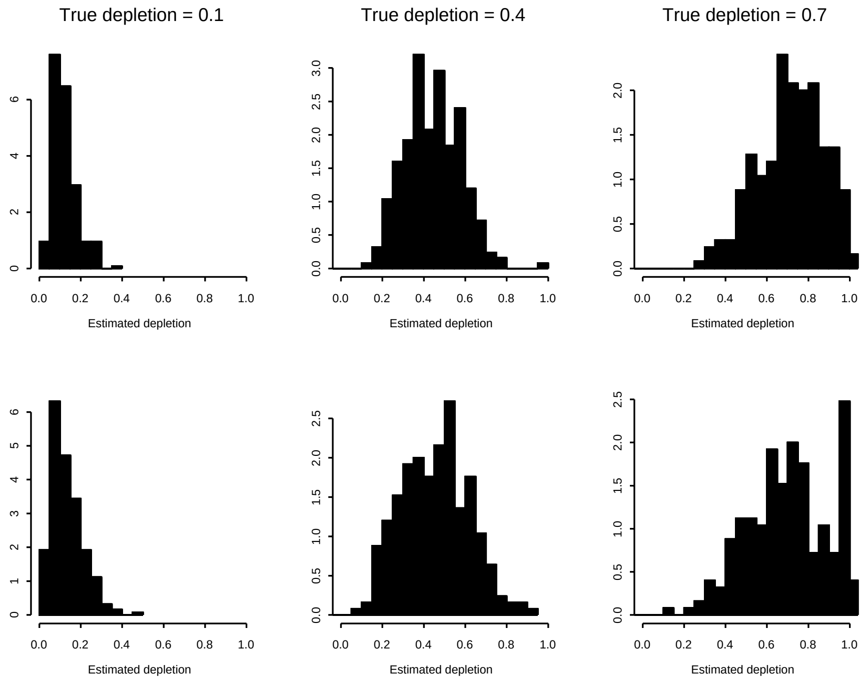

Figure 5. Distributions for the estimates of current (i.e., after 20 years of fishing) depletion from the "fully integrated" method of stock assessment for the reference case operating model and for the "data-rich" and "data-poor" scenarios (upper and lower panels respectively).

S_{MSY} and 0.5 S_{MSY} for the "data-rich," "data-moderate," and "data-poor" scenarios. This is due in part to S_{MSY} being less than 0.4 S_0 but is also due to the extra uncertainty associated with attempting to estimate the ratio S_{MSY}/S_0 rather than basing status determinations on a fixed (and pre-specified) fraction of S_0.

The distributions for the estimates of the depletion of the resource after 20 years of fishing are, as expected, wider for the "data-poor" scenario than for the "data-rich" scenario (Fig. 5). However, and expected from previous investigations into the performance of stock assessment models (e.g., Hilborn 1979), it is also the case that estimation performance is better for lower values for the actual depletion of the resource. Specifically, the performance of the stock assessment method is very poor for an actual depletion of 0.7, irrespective of the amount of data available for assessment purposes.

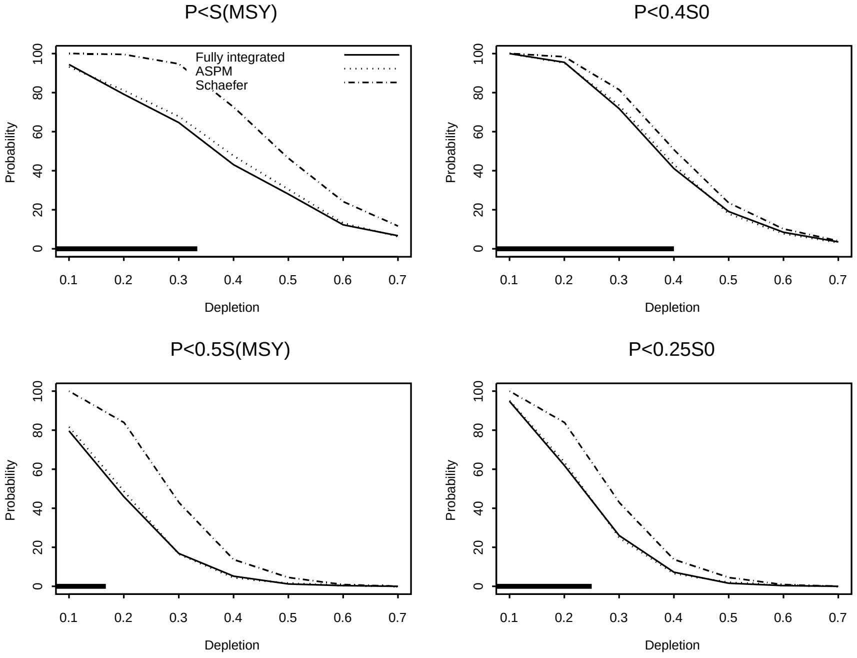

Figure 6. Percentage of simulations in which the resource is indicated to be below each of the four thresholds as a function of the true value for the depletion of the resource for the reference case operating model. The horizontal line indicates the range of depletion levels that are below each true threshold level. Results are shown for the three methods of stock assessment.

Sensitivity to the assessment method

Figure 6 contrasts the performances of the three methods of stock assessment for the reference case operating model and the "data-rich" scenario. Results are not shown for the "data-moderate" and "data-poor" scenarios because they are qualitatively identical to those in Fig. 6. The Schaefer production model assesses the resource to be below S_{MSY} much more frequently than the two age-structured stock assessment methods (resulting in less frequent under-protection errors but more frequent over-protection errors). This occurs because S_{MSY} for a Schaefer model is 0.5 S_0, i.e., substantially larger than the true S_{MSY} of 0.334 S_0. The Schaefer model estimates the resource to be below 0.4 S_0 and 0.25 S_0 more frequently than the two age-structured stock assessment methods but the effect is much smaller than is the case for S_{MSY} and 0.5 S.

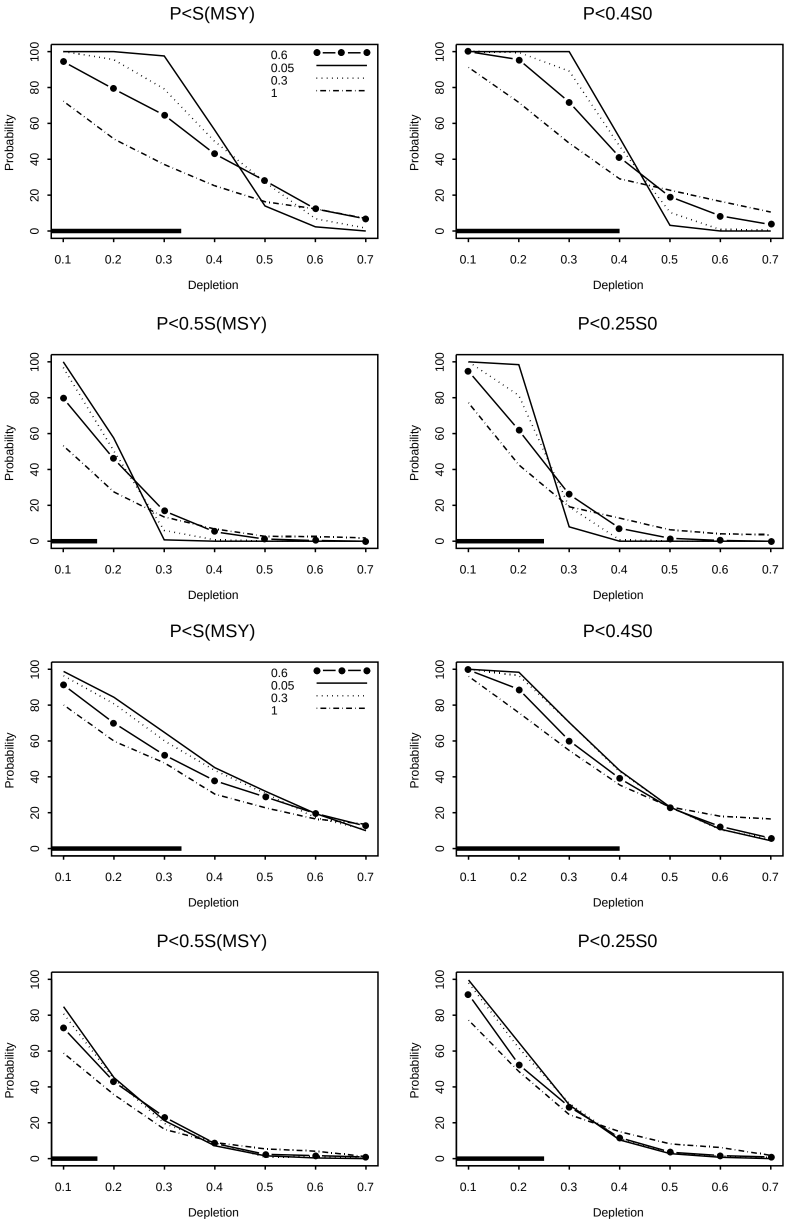

Figure 7. Percentage of simulations in which the resource is indicated to be below each of the four thresholds as a function of the true value for the depletion of the resource. The horizontal line indicates the range of depletion levels that are below each true threshold level. Results are shown for four values for σ_R and two scenarios regarding data quality.

There is little difference in the performances of the two age-structured stock assessment models. This is probably because although the "fully integrated" stock assessment method has more parameters to better capture variability in recruitment, this does not improve the ability to determine whether the spawning output (an aggregate over many age-classes) is above or below a threshold level.

Sensitivity to the specifications of the operating model

Analyses were conducted in which: (a) the values for *MSYR* and *MSYL* were varied, (b) the extent of variation in recruitment was changed, (c) the catch history was changed, and (d) the age-at-maturity and the age-at-50%-recruitment were changed (see Table 1).

The ability to detect whether the resource is below any of the thresholds is very sensitive to the value of σ_R, the extent of variation about the stock-recruitment relationship (Fig. 7). Decreasing σ_R from the reference case value of 0.6 to 0.3 and 0.05 substantially reduces the probability of both over- and under-protection errors (Fig. 7a; solid and dotted lines) while increasing σ_R to 1 leads to a greater probability of these errors. The sensitivity to the value for σ_R arises for several reasons: (a) increased variability in recruitment means that the assumption that the population was at its unfished level at the start of the first year for which catch data is available is violated to a greater extent, (b) increased variability in recruitment leads to greater errors when fitting the age-composition data for the older ages for the fishery catches (because recruitment residuals are not estimated except for the years for which catches are available), and (c) increased variability in recruitment decreases the ability to correctly identify the relationship between recruitment and spawning output (which is needed to estimate the ratio of S_{MSY} to S_0).

The impact of the different values for σ_R is case-specific, however, with much larger impacts for the "data-rich" scenario compared to the "data-poor" scenario. Specifically, there are fewer benefits of a lower value for σ_R in terms of an increased ability to correctly detect whether a stock is above or below a threshold level for the "data-poor" scenario than for the "data-rich" scenario. Lower values for σ_R reduce the impacts of the three factors above, but without informative data, it is not possible to take advantage of this.

Results (not shown here) indicate that changing *MSYL*, the age-at-maturity, and the age-at-50%-recruitment have almost no impact on the ability to correctly detect whether a resource is above or below any of the thresholds. The frequency with which the resource is found to be below all four thresholds gets lower (i.e., there is a higher probability of under-protection and a lower probability of over-protection error) if the resource is less productive (i.e., lower *MSYR*), but the size of the effect is small. The results are also largely insensitive to the catch series, although the frequency of determining the resource to be below S_{MSY} is higher for catch series 1.

Discussion

Attempts to determine whether the abundance of a marine renewable resource is above or below a threshold level are subject to both over- and under-protection error. The level of error depends on the nature of the threshold, with the error associated with making determinations related to S_{MSY} being higher than those associated with proxies for S_{MSY} such as 0.4 S_0. The difference in performance between S_{MSY} and 0.4 S_0 was, however, not very substantial for the scenarios considered in this study.

Somewhat surprisingly, the factor that influenced the sizes of the errors to the greatest extent was the true value for σ_R. This is unfortunate because, unlike the type and quality of data available for assessment purposes which can, in principle at least, be improved through additional research and monitoring, it is not possible to reduce σ_R through increased research and monitoring. The errors caused by higher values of σ_R are associated to some extent with the nature of the stock assessment method applied (e.g., that the recruitment residuals are not estimated during calculation of the age-structure of the assessed population at the start of the first year for which catches are available). Therefore, in principle, some improvement in estimation performance might be anticipated if the stock assessment method had been tailored more to specifics of the operating model. The conclusion that σ_R seems to have a larger impact on the probability of making under- and over-protection errors than other factors, including data quality and quantity, is of course case-specific. For example, had no data been available (except perhaps a catch history) there would have been no ability to even make a status determination at all.

No attempt has been made in this paper to evaluate the consequences and costs associated with making under- and over-protection errors. The costs associated with under-protection errors relate to the impact of unintended (further) depletion of the resource and the consequential impacts on its associated ecosystem while the costs associated over over-protection errors are unnecessary constraints on resource users. Both of these errors result, however, in a loss of credibility of stock assessment scientists when they are discovered.

The results should be considered to be overoptimistic regarding the ability to correctly detect whether a stock is above or below a management-related threshold. This is because the stock assessment method was provided with information (e.g., about natural mortality, weight-at-age, and fecundity-at-age) that would, in reality, be subject to error and because structurally the age-structured stock assessment method was identical to the operating model. Furthermore, the assumption that the catch-rate indices provided an index of abundance that is related linearly to abundance was correct even in the "data-poor" scenario. The presence of seven catch-rate indices over a 20-year period is probably why the "data-poor" scenario did not perform catastrophically bad, as might have been anticipated.

The approach used to determine whether a stock is above or below a threshold level uses only the point estimates of current spawning output, S_0 and S_{MSY}. No account is therefore taken of the uncertainty associated with these quantities. In principle, an approach that based status determinations on lower confidence intervals (e.g., for the ratio of current spawning output to S_0) would be more risk averse, particularly for "data-poor" situations. Future work along the lines of this paper should evaluate such approaches.

Management of fish resources is always based to some extent on a feedback control management system in which the results of a stock assessment form the basis for developing management arrangements. The assessment is then updated using new information on abundance as this information becomes available and the management arrangements modified given the results of the updated assessment. Therefore, future work along the lines of this study should examine the performance of the combinations of the stock assessment method used for status determination and the rules used to determine the management arrangements given the results of the stock assessment (e.g., Butterworth and Bergh 1993, Cochrane et al. 1998, Geromont et al. 1999, Butterworth and Punt 2003). Initial analyses along these lines have been conducted based on the rules used by the Pacific Fishery Management Council to develop management arrangements for groundfish species included in its Groundfish Management Plan (Punt 2003).

Acknowledgments

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Appendix: The operating model

The model specified below is age-structured, relates recruitment to spawner-stock size, and includes both observation- and process-error terms. The model is described first, followed by the process of setting up the simulation trials and generating the "observed" data used in the assessments.

Population dynamics

$$N_{y+1,a} = \begin{cases} N_{y+1,0} & \text{if } a = 0 \\ N_{y,a-1}\, e^{-(M+\tilde{S}_{a-1}F_y)} & \text{if } 1 \leq a < x \\ N_{y,x-1}\, e^{-(M+\tilde{S}_{x-1}F_y)} + N_{y,x}\, e^{-(M+\tilde{S}_x F_y)} & \text{if } a = x \end{cases} \qquad \text{(A.1)}$$

where	$N_{y,a}$	is the number of fish of age a at the start of year y,
	M	is the instantaneous rate of natural mortality (assumed to be independent of age and time),
	$\tilde{S}_a$	is the selectivity of harvesting on fish of age a (assumed year-invariant):

$$\tilde{S}_a = \frac{1}{1+\exp[-(a-a_r)/\delta]} \qquad \text{(A.2)}$$

a_r	is the age-at-50%-recruitment to the fishery,
δ	is the parameter which determines the width of the recruitment ogive (assumed to be 0.5 for the calculations of this paper),
F_y	is the fishing mortality on fully selected ($S_a \rightarrow 1$) animals during year y, and
x	is the plus-group (all fish in this age class are mature and recruited to the fishery, assumed to be 15 for the calculations of this paper).

Births

$$N_{y,0} = \frac{\alpha S_y}{(\beta + S_y)^{\gamma}} e^{\varepsilon_y - \sigma_R^2/2} \quad ; \quad \varepsilon_y \sim N(0;\sigma_R^2) \qquad \text{(A.3)}$$

where S_y is the spawning output at the start of year y:

$$S_y = \sum_{a=a_m}^{x} w_a N_{y,a} \tag{A.4}$$

a_m is the age-at-maturity,
w_a is the mass of a fish of age a:

$$w_a = 10 \times (1 - e^{-\kappa(a+1)})^3 \tag{A.5}$$

κ is the von Bertalanffy growth-rate parameter (assumed to be 0.3 for the calculations of this paper),

α, β, γ are the stock-recruitment relationship parameters, and

σ_R is the log-scale standard deviation of the random fluctuations in recruitment about the underlying deterministic stock-recruitment relationship.

Catches

The fully selected fishing mortality for year y (F_y) is calculated by solving the equation:

$$C_y = \sum_{a=0}^{x} w_{a+1/2} C_{y,a} = \sum_{a=0}^{x} w_{a+1/2} \frac{\tilde{S}_a F_y}{M + \tilde{S}_a F_y} N_{y,a} (1 - e^{-(M + \tilde{S}_a F_y)}) \tag{A.6}$$

where C_y is the historical catch for year y.

Initial conditions

The initial conditions (year $y = 1$) for each replicate correspond to a biomass drawn from the distribution about the average pre-exploitation level that would be expected to result from the assumed level of random recruitment fluctuation. The numbers-at-age at the start of year $y = 1$ for each of the 250 Monte Carlo replicates are generated as follows:

a. The numbers-at-age corresponding to the deterministic equilibrium are calculated.

b. The population is projected forward for 50 years with no catches, but with stochastically fluctuating recruitment (i.e., $\varepsilon_y \neq 0$; $F_y = 0$) so that it is not exactly at deterministic equilibrium at the start of the first year for which historical catches are available.

c. The resultant numbers-at-age after 50 years are taken to be the numbers-at-age at the start of year $y = 1$.

Data generated

The data available for stock assessment purposes are catches, catch-rates, and fishery age-composition data. The catches are assumed to be measured without error and the catch-rates are assumed to be log-normally distributed:

$$I_y = qB_y^e e^{\eta_y - \sigma_q^2/2} \qquad \eta_y \sim N(0;\sigma_q^2) \tag{A.7}$$

where I_y is the catch-rate for year y,

B_y^e is the exploitable biomass in the middle of year y:

$$B_y^e = \sum_{a=0}^{x} w_{a+1/2} \tilde{S}_a N_{y,a} e^{-(M+\tilde{S}_a F_y)/2} \tag{A.8}$$

q is the catchability coefficient (taken, without loss of generality, to be 1), and

σ_q is the standard deviation of the random fluctuations in catchability.

The fishery age-composition data for year y are taken to be a random (i.e., multinomial) sample of size $\tilde{N}_y$ from the fishery catch for year y, i.e., age a is selected with probability

$$C_{y,a} / \sum C_{y,a'}.$$

Evaluating Harvest Strategies for a Rapidly Expanding Fishery: The Australian Broadbill Swordfish Fishery

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Abstract

The Australian longline fishery, operating off eastern Australia, expanded rapidly in the mid- to late 1990s with swordfish catches increasing from around 50 t to over 3,000 t. Combined with New Zealand catches, the present swordfish harvest in the southwest Pacific is several times greater than historical catches by Japanese longliners that fished in the region. While comprehensive catch and effort data exist, uncertainty remains about the biology and productivity of broadbill swordfish in the region. With declines in swordfish catch rates in recent years, the Australian Fisheries Management Authority has sought advice on sustainable harvest strategies for the fishery, including total allowable effort levels.

Given the limitations and uncertainties in the available information, a management strategy evaluation (MSE) framework has been developed for swordfish in the southwest Pacific to evaluate alternative future harvest strategies. The operating models incorporate multiple fleets and areas to account for differences in targeting practices and hypotheses about seasonal swordfish movements. Catchabilities are fleet and area specific, with parameters describing changes in targeting practices over time. The model is conditioned on historical information, which includes catch and size frequency data.

The results indicate that large increases in the combined effort of both the Australian and foreign longline fleets would decrease the proportion of large fish in the catch and place the stock at a high risk of being overfished. The results were most sensitive to the assumed level of present depletion and the degree of spatial movement, the latter result highlighting the need to develop area-specific performance indicators if movement is limited. The use of an empirical decision rule to adjust effort levels slowed stock depletion, but may not allow the stock to rebuild if it is already depleted.

Introduction

Until 1995, yellowfin (*Thunnus albacares*) and bigeye (*Thunnus obesus*) tuna had remained the principal target species of the longline component of the Australian eastern tuna and billfish fishery (ETBF) off eastern Australia. However, vessels began targeting broadbill swordfish (*Xiphias gladius*) off southern Queensland in 1995 with good catches obtained around several inshore seamounts. Following this initial success, and with increased access to markets in the United States, swordfish landings increased more than tenfold to around 817 t in 1996. Due to continued expansion of the fishery, by 1999 swordfish landings reached 3,076 t, becoming the largest catch component in the longline fishery (Campbell 2002). After 1999, swordfish catch rates declined substantially and by 2003 the swordfish catch had decreased to around 2,190 t.

Although swordfish had been caught by Japanese longliners fishing off eastern Australia for many decades, the Australian swordfish fishery provides an example of a new resource development within an existing fishery. This also exemplifies "management under uncertainty" in which initial exploitation is followed by a rapid expansion of effort and catches, then possible overcapitalization as the stock becomes overexploited (Smith 1993). Because the initial size and productivity of the resource is unknown, managers must balance fishery development against overcapitalization and overexploitation.

Recent genetic studies indicate broadbill swordfish most likely possess a localized stock structure within the southwest Pacific (Reeb et al. 2000, Bremer et al. 2001). Additionally, Australian longliners take the largest catch from this stock (Campbell and Dowling 2003). However, no assessment exists for this swordfish stock, and it is unknown whether current catches of this species are sustainable. The Australian Fisheries Management Authority (AFMA) is presently finalizing a new management plan for the ETBF, to take effect in 2005. This plan will limit total allowable effort (TAE), defined as "hook days."

This paper evaluates alternative initial harvest strategies for swordfish in the ETBF. Strategies that used an empirical-based decision rule for altering the TAE were also evaluated. The approach, known as management strategy evaluation (MSE), uses an operating model to examine alternative swordfish harvest strategies under uncertainty in the population dynamics to evaluate anticipated performance relative to specified management objectives (Smith 1994, Butterworth et al. 1997, Punt et al. 2002).

Methods

Evaluation of harvest strategies: The MSE approach

The MSE approach involves the following five basic steps (Punt et al. 2001):

1. Identification of management objectives and representation of these using quantitative performance measures.
2. Identification of alternative harvest strategies.
3. Development and parameterization of alternative operating models to represent the alternative realities in the calculations.
4. Stock projections based on alternative harvest strategies.
5. The development of summary measures to quantify the performance of each harvest strategy relative to the management objectives of the fishery.

The operating models represent the population and fleet dynamics of the fishery and are used to generate observations in the form of pseudo catch, effort, and catch-at-size data sets which are then used in the management procedure.

An operating model for swordfish in the southwest Pacific

The operating model assumed a single swordfish stock in the southwest Pacific and explicitly considers the age and sex-structure of the population. Individual variability in growth (and the length-structure of the population) is accounted for by dividing each cohort into several groups (five males and five females), each of which grows according to a different growth curve. Age-specific natural mortality rates were based on those estimated for southern bluefin tuna (a similar long-lived species, A. Preece, CSIRO, pers. comm.), while the steepness parameter in the stock recruitment relation was set to 0.9, corresponding to the assumption that recruitment remained relatively insensitive to changes in parental biomass except below 20% of virgin biomass. The model updates natural and fishing mortality every quarter, with seasonal movement as a function of length applied at the end of each quarter. Spawning and recruitment are assumed to occur during the first two quarters only. It is assumed that the population is closed with respect to immigration and emigration, and unaffected by multispecies interactions.

The model is fleet-specific for the Japanese, Australian, and New Zealand longline fleets, and fishing and movement of fish occurs across five regions (Fig. 1). Data for the Japanese fleet were available since 1971, while data for the Australian and New Zealand fleets were available since 1990 and 1991, respectively. The Australian fishery occurred

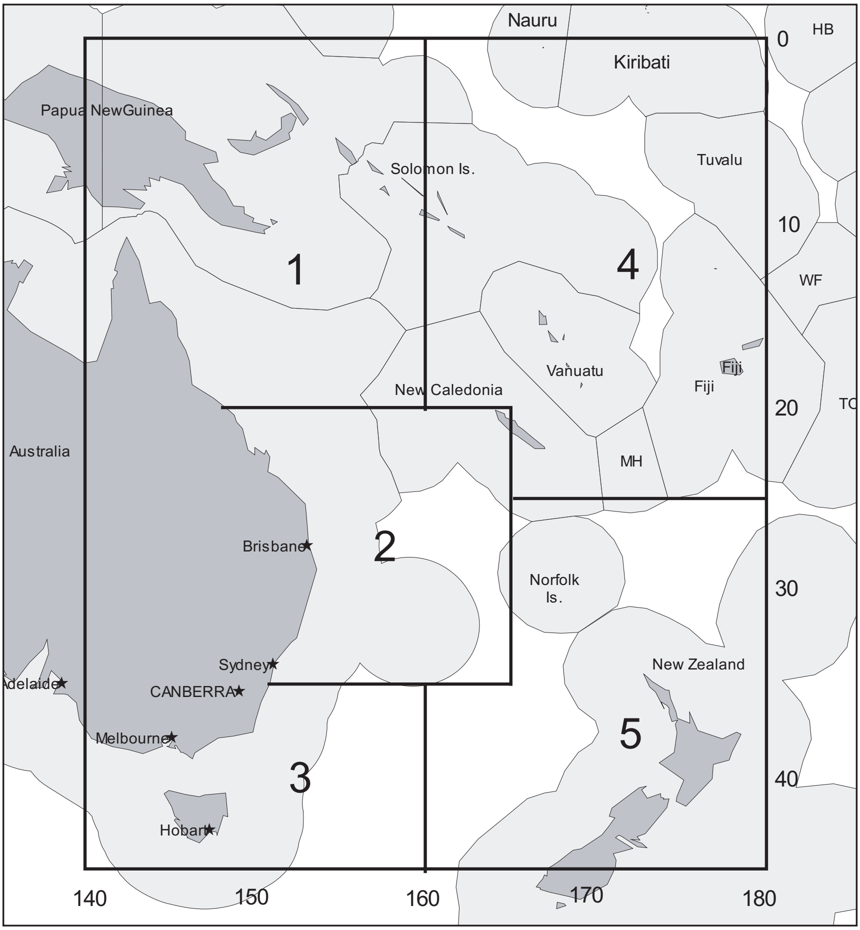


Figure 1. The five regions used to define the spatial structure of the swordfish operating model for the southwest Pacific. The light gray areas indicate exclusive economic zones of individual nation-states.

in areas 1 to 3 while the New Zealand fishery occurred in area 5. Effort for the Japanese fleet was standardized using a general linear model for changes in the spatial-temporal distribution of effort, gear configurations (hooks-per-basket), and the environment (using southern oscillation index as a proxy) in order to obtain a more appropriate measure of effort targeted at swordfish (Campbell and Dowling 2003). The Japanese catch and standardized effort data were also scaled proportionally to account for catches taken by other foreign fleets operating in the region. Gear selectivity was assumed to be the same for all fleets, though fishing efficiencies (catchabilities) were modeled separately by fleet, quarter, and area to allow for differences and changes in targeting practices. Loss of catch due to discarding and/or predation was also incorporated in the model. Campbell and Dowling (2003) further describe the operating model framework.

Model conditioning

Initial values of R_0, the number of recruits in the stock before 1971, and the parameters of the movement probability matrix were estimated. All other parameters were input as fixed values for each scenario. Observed catch by area and quarter was assumed to represent total fishing mortality. For a given R_0, the time series of fishing mortality was used with the modeled natural mortality and recruitment to project the population forward in time from 1971 to 2001. At the end of this period, features of the modeled swordfish population were then compared with the historical catch, catch rate, and size data from the three fisheries and an assumed level of depletion. This process was repeated until the values of R_0 and the movement parameters that minimized an objective function were found. At this stage, parameters describing fishery catchabilities were calculated.

Due to the lack of a stock assessment for swordfish in the southwest Pacific, the stock status at the end of 2001 remains unknown. For scenario testing, we considered a range of depletion levels, including upper and lower extremes as well as an assumed "reference" value:

Low	B(2001) = 85% B_0
Medium (reference)	B(2001) = 70% B_0
High	B(2001) = 50% B_0

These three depletion levels also act as proxies for a high, medium, or low stock productivity as more productive stocks experience less depletion for a given level of historical fishing.

Model projections

After being conditioned on the historical data, the model was projected forward 20 years under each harvest strategy. The fishery dynamics were controlled by the levels of fishing effort set for each fleet and the operating model was used to evaluate future conditions of the fishery and stock.

While the observed catch was used to calculate the fishing mortality during the conditioning process, during the projection years the catch needed to be determined from fishing effort. For this purpose, the relationship between nominal effort E_{nom} and fishing mortality F for each fleet at time t was assumed to be of the form:

$$F(t) = Q(t)E_{nom}(t)e^{\varepsilon_t - \sigma^2/2} \tag{1}$$

where Q is the catchability and ε_t is a factor to account for random variation in catchability [$\varepsilon_t \sim N(0;\sigma^2)$] . Re-parameterizing $Q(t)$ to have a time-independent q_o and a time-varying Q_t component, such that $Q(t) = q_oQ_t$, the above equation can be rewritten in terms of the effective effort E_{eff}

$$F(t) = q_oQ_tE_{nom}(t)e^{\varepsilon_t - \sigma^2/2} = q_oE_{eff}(t)e^{\varepsilon_t - \sigma^2/2} \tag{2}$$

where

$$E_{eff} = Q_tE_{nom}. \tag{3}$$

The functional form of Q_t for each fleet is given in the Appendix. The parameters describing the fleet specific catchabilities were obtained at the end of the conditioning phase by minimizing the least-squares difference between the nominal effort estimated using equation 1 with the observed effort.

Performance indicators and measures

To describe the performance of the fishery during the projection years, the following performance indicators were chosen in consultation with representatives from ETBF stakeholder groups:

Economic

1. Average annual Australian catch (metric tons).
2. Median percentage change in the Australian catch between years.
3. Average percentage of fish in the Australian catch from area 2 that are greater than 50 kg.

Conservation

1. Final spawning biomass relative to the initial spawning biomass B_0.
2. Probability that the spawning biomass drops below 30% B_0.

Collectively these indicators represent three broad management objectives for most fisheries: (1) maximized value of the catch, (2) a sustained harvest regime, and (3) industrial stability (Walters and Pearse 1996, Butterworth and Punt 1999). All harvest strategies consequently achieve some balance among these objectives, and information on trade-offs among them is needed by decision makers to make informed decisions.

Finally, performance criteria should also be easy for managers and stakeholders to interpret (Francis and Shotten 1997).

The expected value of each performance indicator under each harvest strategy was averaged across 100 Monte Carlo simulations.

Fixed effort strategies

In consultation with representatives from ETBF stakeholder groups, a range of fixed (in the sense that the annual effort for all years was predetermined) future effort strategies was identified for consideration. Hopefully these strategies would bracket future changes in both Australian and foreign effort and future increases in fishing power, also called effort creep.

Australian effort

1. Status quo: effort remains at 2001 level of 11.2 million hooks for all years.
2. Increase in nominal effort over five years to 1.5 × 2001 level (16.8 million hooks) after which time no further increase.
3. Increase in nominal effort over five years to 2.0 × 2001 level (22.4 million hooks) after which time no further increase.
4. Increase in nominal effort over five years to 2.5 × 2001 level (28.0 million hooks) after which time no further increase.

Foreign effort

1. Status quo: effort stays at average level over the last 3 years, 1999-2001.
2. Increase in effort over five years to 2.0 × status quo level, after which time no further increase.

Effort creep

1. No increase in effective effort due to effort creep.
2. Increase in effective effort due to effort creep of 2% per annum. Applied for all years to the Australian fleet and for first five years only to the foreign fleets (to account for the possible entry of new and less skilled fishing fleets).

Each of these effort strategies is shown in Figs. 2a and 2b while time series of total effort for four of the extreme strategies are shown in Fig. 2c.

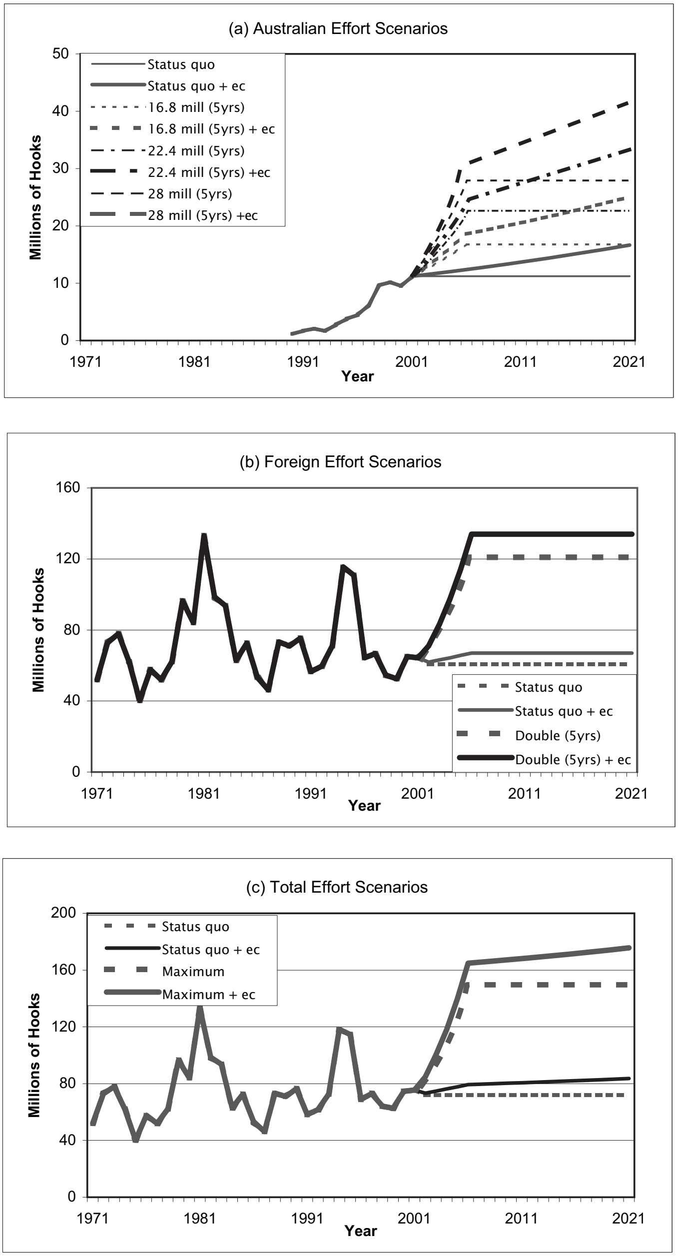


Figure 2. Time series of annual (a) Australian effort, and (b) total effort, under different future harvest strategies (ec = effort creep applied).

Table 1. List of future fixed effort harvest strategies used for evaluation purposes.

Effort strategy	Increase in Australian effort during first five years	Increase in foreign effort during first five years	Effort creep applied	Harvest strategy
1	1	1	No	Australian 11.2 m, foreign sq
2	1	2	Yes	Australian 11.2 m+ec, foreign double+ec
3	1.5	1	No	Australian 16.8 m, foreign sq
4	1.5	1	Yes	Australian 16.8 m+ec, foreign sq
5	1.5	2	Yes	Australian 16.8 m+ec, foreign double+ec
6	2	1	No	Australian 22.4 m foreign sq
7	2	1	Yes	Australian 22.4 m+ec, foreign sq
8	2	2	Yes	Australian 22.4 m+ec, foreign double+ec
9	2.5	1	No	Australian 28.0 m, foreign sq
10	2.5	1	Yes	Australian 28.0 m+ec, foreign sq
11	2.5	2	Yes	Australian 28.0 m+ec, foreign double+ec

Note: sq = status quo, m = million hooks, and ec = effort creep applied.

Table 1 lists the combination of strategies that were selected for evaluation.

Outcomes for each effort strategy were synthesized by taking a weighted average of the results across each depletion regime. Probabilities of 25, 60, and 15% were assigned to the low, medium, and high depletion regimes, respectively, guided by the relative size of the historical catches in relation to swordfish fisheries elsewhere, and the fit between the historically observed and model predicted catch-at-size data. To qualitatively assess the performance of each indicator under each of the effort scenarios, and to assist in presentation of the results to

Table 2. Indicative reference levels used to qualitatively assess the value of each economic and conservation performance indicator.

	Management option				
	Economic			Conservation	
Qualitative rank	Total Australian catch	Annual change in catch	Proportion of large fish	Final spawning biomass	Time (%) biomass < 30% B_0
Excellent	≥ 2.0			> 60%	
Very good	< 2.0	< 1.0	> 1.0	> 50%	
Good	< 1.5	< 1.1	> 0.9	> 40%	0%
Moderate	< 1.0	< 1.2	> 0.8	> 30%	< 10%
Poor		≥ 1.2	> 0.7	> 20%	< 20%
Very poor			≤ 0.7	≤ 20%	≥ 20%

the various ETBF stakeholder groups, the performance of each indicator was ranked on a qualitative scale between very poor and excellent (Table 2). In assigning ranks, we used 30% B_0 as a limit reference point for the spawning biomass and placed a high value on stabilizing the proportion of large fish in the catch.

Sensitivity to biological uncertainty

While reproductive studies have been undertaken by Young and Drake (2002), little other biological work has been undertaken on the swordfish stock in the southwest Pacific. To overcome this deficiency, a reference biological scenario was generated by using information on growth, natural mortality rates, and the stock-recruitment relationship from published research on swordfish in other oceans.

To examine the sensitivity of MSE outputs to a range of plausible biological inputs the following alternative scenarios were examined: (1) steepness in the stock-recruitment relation was changed from 0.9 to 0.65 or 0.4, (2) natural mortality-at-age was multiplied by 1.5, (3) no movement among areas was permitted, and (4) movement was random (i.e., uniform) between areas. Relative sensitivity across four different effort strategies (1, 2, 5, and 11, cf. Table 1) was examined, with all evaluations conducted under the medium depletion scenario defined earlier.

Harvest strategies involving decision rules

While an examination of stock behavior under fixed harvest strategies provides information relevant to the choice of an initial TAE for the Australian fleet, in practice it is unrealistic to expect that future effort levels could be predetermined in such a manner. Thus, we evaluated the use of a simple empirical-based feedback decision rule, based on an assessment of the performance of the fishery, to set the levels of Australian effort at predetermined time intervals.

For the first year of the projection period, an arbitrary TAE of 15 million hooks for the Australian fleet was spatiotemporally allocated by quarter and area based on the corresponding distribution of total effort from the previous year. The TAE was then updated according to the following formula (Magnusson and Stefansson 1989):

$$TAE_{t+1} = TAE_t(1 + \beta_{emp}S_t)$$

where β_{emp} is a control parameter referred to as the feedback gain factor, and S_t is the slope of a linear regression of some fishery statistic of the over the years $t - y_{emp}+1$ to t.

The fishery statistic used should preferably be some measure of exploitable biomass or fishing mortality. Here, both the annual Australian swordfish catch rate and upper 95th percentile weight of fish caught were used. Feedback gain factors of 0.1 and 0.3 were tested and regressions were performed over the past 5 or 7 years of data. Finally, the decision rule was applied either annually or every 4 years. For the two non-Australian fleets we assumed no control over management regimes and as such no management strategies or decision rules were incorporated in the model framework for these fleets. Instead, the foreign effort within each quarter/area strata for all years was set to be equal to the average level of effort over the last 3 years, 1999-2001.

The effort levels set for each fleet at the start of each year, and adjusted for the Australian fleet via the decision rule, relate to the nominal effort. However, catch rates used in the decision rule were based on the effective effort calculated using equation 3. We also compared the effectiveness of one of the harvest strategies against the alternative situation where the amount of stochastic variation in equation 1, as determined by the parameter σ, was arbitrarily fixed at a higher level (0.5) than that estimated from the conditioning process, where σ ranged from 0.2 to 0.4 across fleets and quarters.

Table 3 lists the evaluated harvest strategies. All the evaluations were conducted under the reference biological scenario defined earlier and the high depletion regime. Each harvest strategy was evaluated against a reference strategy where no decision rule was used and annual effort assigned to each fleet was constant across years.

Table 3. Alternative harvest strategies involving use of a decision rule.

Harvest strategy	Assessment time frame	Effort error σ	Empirical based strategy		
			Indicator	β_{emp}	γ_{emp}
1	Reference: Australia (15 million), foreign (status quo), $B(2001)=50\%\ B_0$				
2	Annual	Calculated	CPUE	0.1	5
3	Annual	Calculated	CPUE	0.3	5
4	Annual	0.5	CPUE	0.3	5
5	Annual	Calculated	CPUE	0.1	7
6	Annual	Calculated	95th mass	0.1	5
7	4 years	Calculated	CPUE	0.3	4

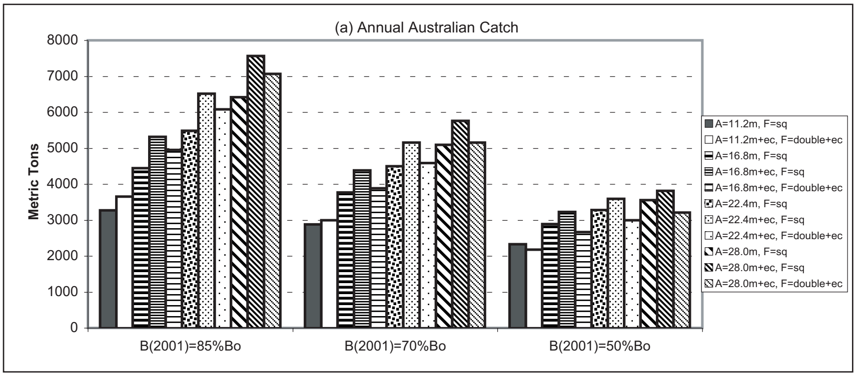
Results

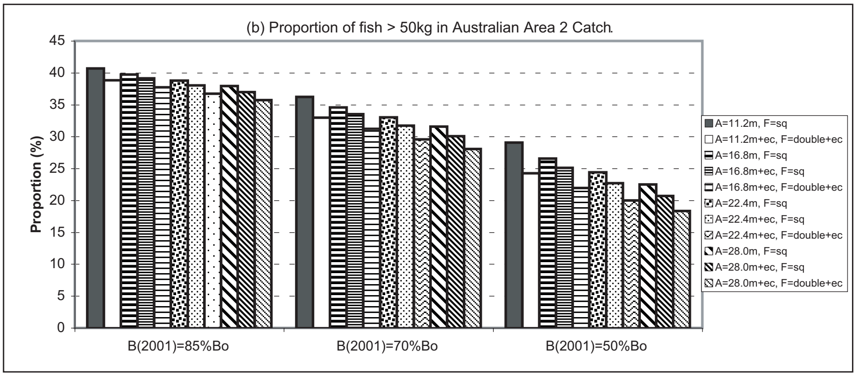
Comparison of fixed effort strategies

Table 4 shows model performance under each effort strategy across the range of assumed depletions. For comparison, the economic value indicators are expressed relative to the value obtained under the status quo strategy. Three of the results using absolute values are displayed graphically in Fig. 3.

Fishery performance improved (i.e., greater catches, lower depletion of the swordfish biomass) as the assumed depletion level at the end of 2001 was reduced. This follows since a lower depletion implies that the stock has been more resilient to historical levels of fishing effort. The relative trend of the performance indicators was generally consistent across each assumed depletion regime (Fig. 3). Status quo strategies gave the best results for all conservation indicators and for several of the economic indicators (i.e., smallest annual change in catch and highest proportion of large fish in the Australian catch), but the worst in terms of average catch. In contrast, the 28 million hook strategy with a doubling of foreign effort and effort creep produced the worst performance in terms of the conservation indicators, while yielding the second highest Australian catch.

Annual Australian catch over the projection years generally increased with greater effort. However, the relative change in catch depended on the level of depletion at the start of the projection. Under the low depletion regime, the Australian catch increased by around 130% for strategy 10 relative to the status quo, while under the high depletion regime the catch increased by only around half this level (Table 4). Australian catches were also higher when there was no corresponding increase in foreign effort,





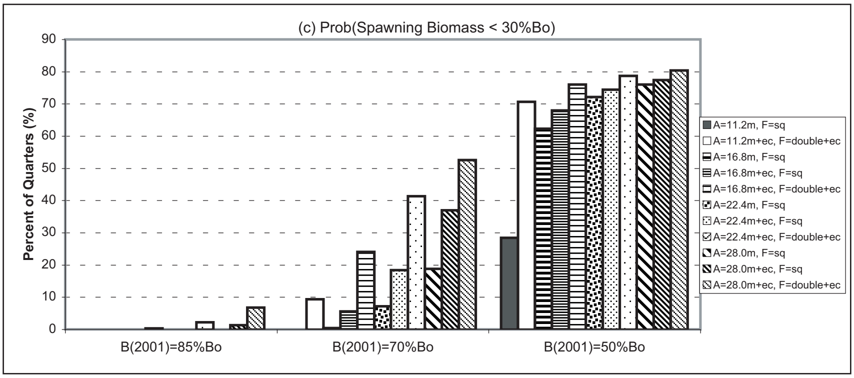


Figure 3. Mean values of three performance indicators for future effort strategies and depletion regimes (A = Australian effort, F = foreign effort, ec = effort creep, sq = status quo).

Table 4. Mean across 100 model simulations of all performance indicators, for each of the 11 future fixed effort strategies, under three alternative states of nature (spawning stock depletion at the end of 2001). Final spawning biomass is expressed as a percentage of the initial biomass B_0.

		Management options				
		Economic			Conservation	
	Harvest strategy	Australian catch[a]	Annual change[a]	Proportion large fish[a]	Final spawning biomass	Prob biomass < 30% B_0
B(2001) = 85% B_0						
1	A = 11.2 m, F = sq	1.00	1.00	1.00	71.0	0.0
2	A = 11.2 m+ec, F = double+ec	1.12	1.01	0.95	56.1	0.0
3	A = 16.8 m, F = sq	1.36	1.04	0.98	64.0	0.0
4	A = 16.8 m+ec, F = sq	1.63	1.04	0.96	57.7	0.0
5	A = 16.8 m+ec, F = double+ec	1.51	1.04	0.93	49.2	0.2
6	A = 22.4 m, F = sq	1.68	1.08	0.95	57.8	0.0
7	A = 22.4 m+ec, F = sq	1.99	1.09	0.93	50.5	0.0
8	A = 22.4 m+ec, F = double+ec	1.86	1.08	0.90	43.4	2.2
9	A = 28.0 m, F = sq	1.96	1.11	0.93	52.4	0.0
10	A = 28.0 m+ec, F = sq	2.31	1.13	0.91	44.5	1.3
11	A = 28.0 m+ec, F = double+ec	2.16	1.13	0.88	38.5	6.7
B(2001) = 70% B_0						
1	A = 11.2m, F = sq	1.00	1.00	1.00	50.4	0.0
2	A = 11.2m+ec, F = double+ec	1.04	1.01	0.91	32.8	9.3
3	A = 16.8m, F = sq	1.31	1.04	0.95	41.9	0.3
4	A = 16.8m+ec, F = sq	1.52	1.04	0.93	34.7	5.6
5	A = 16.8 m+ec, F = double+ec	1.35	1.06	0.86	26.1	24.0
6	A = 22.4 m, F = sq	1.56	1.07	0.91	35.1	7.1
7	A = 22.4 m+ec, F = sq	1.79	1.09	0.88	27.6	18.4
8	A = 22.4 m+ec, F = double+ec	1.59	1.09	0.82	21.1	41.3
9	A = 28.0 m, F = sq	1.77	1.10	0.87	29.6	18.8
10	A = 28.0 m+ec, F = sq	2.00	1.11	0.83	22.2	36.9
11	A = 28.0 m+ec, F = double+ec	1.79	1.12	0.77	17.1	52.6
B(2001) = 50% B_0						
1	A = 11.2 m, F = sq	1.00	1.00	1.00	27.9	28.4
2	A = 11.2 m+ec, F = double+ec	0.94	1.04	0.83	13.4	70.6
3	A = 16.8 m, F = sq	1.24	1.07	0.91	20.6	62.3
4	A = 16.8 m+ec, F = sq	1.39	1.05	0.86	15.0	68.0
5	A = 16.8 m+ec, F = double+ec	1.15	1.07	0.75	9.3	76.0
6	A = 22.4 m, F = sq	1.41	1.12	0.84	15.4	72.2
7	A = 22.4 m+ec, F = sq	1.54	1.09	0.78	10.3	74.4
8	A = 22.4 m+ec, F = double+ec	1.29	1.12	0.69	6.5	78.6
9	A = 28.0 m, F = sq	1.53	1.15	0.77	11.7	76.0
10	A = 28.0 m+ec, F = sq	1.64	1.15	0.71	7.3	77.4
11	A = 28.0 m+ec, F = double+ec	1.38	1.17	0.63	4.7	80.3

[a] Relative to status quo.

Note: A = Australia, F = foreign, sq = status quo, m = million hooks, and ec = effort creep applied.

being around 7% higher under the low depletion regime and between 19 and 21% higher under the high depletion regime. This suggests a negative interactive effect between the Australian and foreign fisheries.

The annual change in catch showed greater variation at higher effort levels but little variation across the depletion levels (Table 4). However, the proportion of fish larger than 50 kg caught by the Australian fleet in area 2 was more sensitive to the depletion level. Under low depletion this proportion was relatively robust to different effort levels, declining from 41% under status quo to 36% under the highest effort strategy (a relative change of 12%). However, the relative change under the highest depletion regime was 37%.

With increased effort and catches the probability that spawning biomass would fall below 30% B_0 increased. For the status quo scenario, spawning biomass never dropped below this level under either the low or medium depletion, but fell below this level an average of 28% of quarters under high depletion (Table 4). On the other hand, there was an 80% probability of the spawning biomass dropping below 30% B_0 under the highest effort strategy and high depletion regime. Indeed, for the high depletion regime this probability exceeded 50% for all strategies except the status quo. The probability that the spawning biomass drops below 30% B_0 showed the greatest variability under medium depletion, ranging from zero to greater than 50%.

Synthesis and management trade-offs

The qualitative performance of each indicator under each effort scenario after taking the weighted average across each depletion regime is given in Table 5. These results allow a qualitative assessment of the success of each alternative fixed effort harvest strategy and help to make explicit the trade-offs between achievement of the economic and conservation objectives. Only strategies 1 and 3 (low Australian effort, foreign status quo) achieved a qualitative performance of moderate or better for all five performance measures. Strategies 2, 4, 5, and 7 (all including Australian effort creep) achieved a ranking of moderate or better for four of the performance measures, but produced a poor conservation outcome with the probability of the spawning biomass falling below 30% B_0 being between 10 and 20%. The trade-off between higher catches and lower spawning biomass was more extreme for the other strategies.

Sensitivity to biological scenarios

Relative patterns of change in the performance indicators with respect to different biological scenarios were consistent across the four effort strategies considered (Fig. 4). Increasing effort resulted in higher Australian catches, a lower proportion of large fish in the Australian catch, and higher probabilities of spawning biomass falling below 30% B_0.

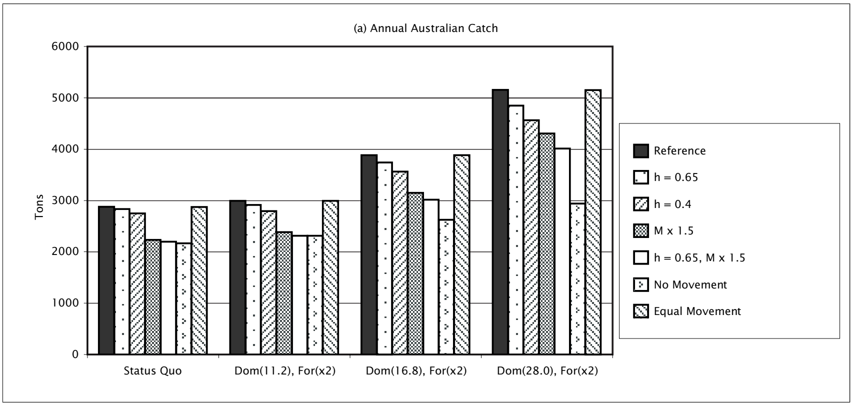
Table 5. Qualitative comparison of the economic and biological performance indicators across each fixed effort strategy after taking the weighted mean across the individual values for each of the three assumed depletion levels.

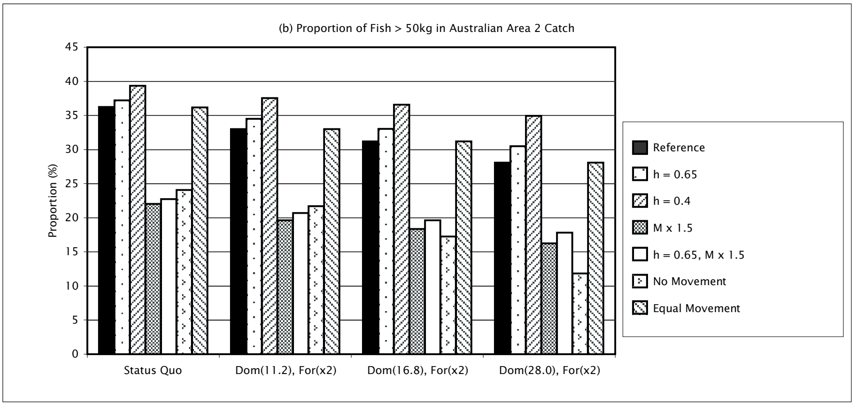
	Management options				
	Economic			Conservation	
Harvest strategy	Total Australian catch	Annual change in catch	Proportion of large fish	Final spawning biomass	Prob biomass <30% B_0
1 A = 11.2 m, F = sq	Good	Good	Good	Very good	Good
2 A = 11.2 m+ec, F =double+ec	Good	Good	Good	Moderate	Poor
3 A = 16.8 m, F = sq	Good	Good	Good	Good	Moderate
4 A = 16.8 m+ec, F = sq	Very good	Good	Good	Moderate	Poor
5 A = 16.8 m+ec, F = double+ec	Good	Good	Moderate	Moderate	Very poor
6 A = 22.4 m, F = sq	Very good	Good	Good	Moderate	Poor
7 A = 22.4+ec m, F = sq	Very good	Good	Moderate	Moderate	Poor
8 A = 22.4 m+ec, F = double+ec	Very good	Good	Moderate	Poor	Very poor
9 A = 28.0 m, F = sq	Very good	Moderate	Moderate	Moderate	Poor
10 A = 28.0 m+ec, F = sq	Excellent	Moderate	Moderate	Poor	Very poor
11 A = 28.0 m+ec, F = double+ec	Very good	Moderate	Poor	Poor	Very poor

Note: A = Australia, F = foreign, sq = status quo, m = million hooks, and ec = effort creep applied.

Reducing the steepness parameter in the stock-recruitment relationship to 0.4 resulted in a small decrease in annual Australian catch that ranged from 4% for the status quo to 11% for the highest effort strategy (Fig. 4a). This was expected, as the productivity of the stock is reduced as the steepness of the stock-recruitment relation is reduced. Furthermore, higher effort levels result in a more depleted and less productive stock. On the other hand, reducing the steepness parameter resulted in an increase in the proportion of large fish in the Australian catch in area 2 (Fig. 4b). This is due to a reduction in the number of small fish recruiting to the population as the productivity of the stock is reduced. There was a corresponding increase in the probability of spawning biomass dropping below 30% B_0, though the relative size of this change diminished with higher effort (Fig. 4c).

Increasing natural mortality reduced mean catch by an amount ranging from 23% under the status quo to 16% under the highest effort (Fig. 4a). Under high natural mortality fewer fish survive to older ages and consequently the proportion of large fish in the Australian catch declined by about 40% under all strategies (Fig. 4b). On the other hand, the probability of spawning biomass dropping below 30% B_0 decreased, most likely


(a) Annual Australian Catch


(b) Proportion of Fish > 50kg in Australian Area 2 Catch

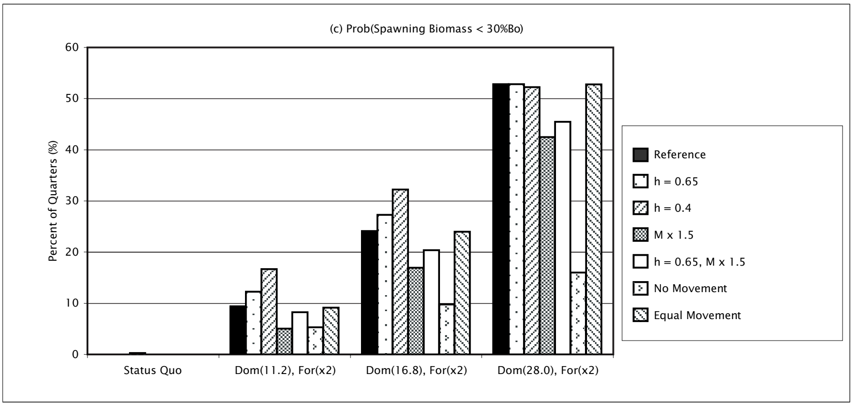

(c) Prob(Spawning Biomass < 30%Bo)

Figure 4. Three performance measures for seven alternative biologies under four future effort strategies (h = steepness, M = natural mortality).

because in order to sustain the historical time series of catches a higher natural mortality may be compensated by increased stock productivity.

When steepness was reduced and high natural mortality increased simultaneously, mean catch and the proportion of large fish in the Australian catch declined by about 23% and 37%, respectively, under all strategies. Changes in spawning biomass were between those when either parameter was changed alone, indicating compensatory impacts by these two parameters.

The assumption of equal movement had little effect on all indicators (Fig. 4a-c). However, under the scenario of no movement between areas, catch declines ranged from 25% under the status quo to 43% under high effort, while corresponding declines in the proportion of large fish ranged from 34% and 58%. This result is a consequence of localized depletion of the resource in area 2 due to the absence of resource replenishment in this area. In this regard area 2 serves as a "sink" for the swordfish population in the southwest Pacific. Finally, the probability of the spawning biomass dropping below 30% B_0 was reduced under the no movement scenario because stock depletion is mainly confined to area 2, decreasing resource vulnerability in the remaining areas.

Harvest strategies involving decision rules

Relative to the reference strategy, the annual Australian catch and the probability of spawning biomass dropping below 30% B_0 decreased while the proportion of large fish in the Australian catch increased for virtually all of the decision rule–based strategies (Fig. 5). The exception was when the effort update was conducted every fourth year instead of annually. Comparison of the performance of harvest strategy 2, which used catch rate as the performance statistic, and strategy 6, which used the upper 95th mass percentile, indicated that the latter achieved a better conservation outcome, but with a lower catch. Overall, this suggests that annual empirical assessments were effective in controlling spawning biomass declines, albeit with some economic cost.

Strategy 3, involving annual assessments and a decision rule based on CPUE regressions where $\beta_{emp} = 0.3$ and $y_{emp} = 5$ years, was the most successful in reducing the probability of spawning biomass falling below 30% B_0 (Fig. 5c). There was a corresponding increase in the estimated size of the final spawning biomass at the end of the projection period from 18 to 35%. However, average annual Australian catch was reduced to 954 t as compared to 2,024 t under the reference strategy. Applying the same harvest strategy when the error on the effective effort time series was assumed to be greater (strategy 4) gave a similar result, though with greater variability.

The proportion of large fish in the Australian catch in area 2 was not strongly influenced by application of the decision rule–based strategies.

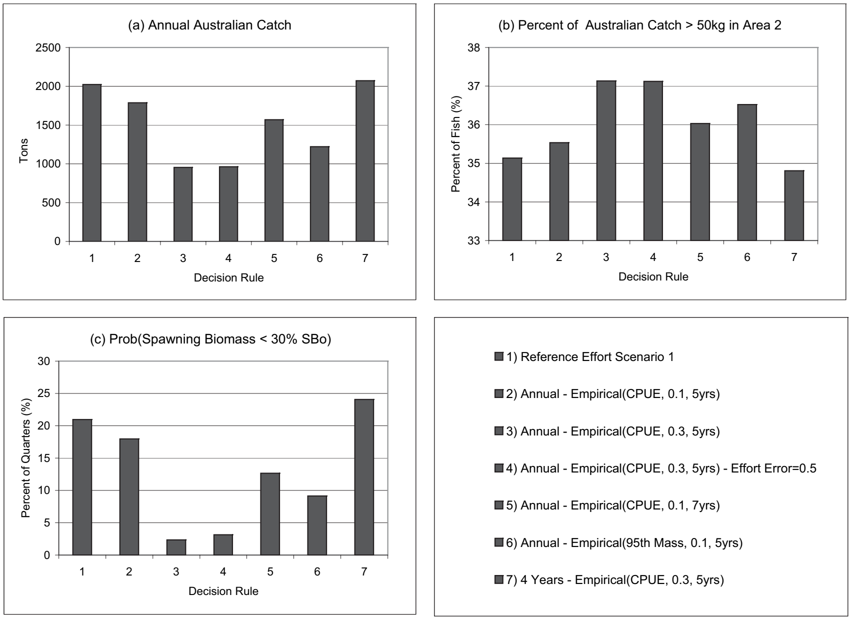


Figure 5. Comparison of performance indicators for harvest strategies under an empirical decision rule.

The greatest increase in final mass relative to the reference strategy was around 2 kg, again achieved under strategy 3 (Fig. 5b).

Decision rules using a feedback control factor $\beta_{emp} = 0.1$ gave reduced conservation outcomes as opposed to those with $\beta_{emp} = 0.3$ (Fig. 5c). A higher value of this parameter implies greater sensitivity to changes in CPUE. Better conservation outcomes were also achieved when the regression was undertaken on 7 rather than on 5 years of data. For those strategies using $\beta_{emp} = 0.3$ and $y_{emp} = 5$ years, adjusting effort every 4 years yielded poorer conservation outcomes than annual adjustments.

Discussion

Given uncertainties in both the dynamics of the resource and the fishery, the MSE approach is a valuable tool for evaluating alternative management options for the Australian swordfish fishery in the southwest Pacific. In particular, stock projections showed the effect of different effort levels on swordfish population sustainability, thus helping determine an appropriate initial TAE for the Australian fleet. Results indicated large increases in the combined effort of the Australian and foreign longline fleets would place pressure on the swordfish population in the southwest Pacific.

Annual Australian catch over the 20-year projection generally increased with greater effort. However, average fish size decreased, and the probability of the biomass falling below 30% B_0 increased. With large fish returning a higher price, a decreased proportion of large fish results in a lower unit return across the catch. Furthermore, the fishing down of the stock to lower levels may move the stock into an overfished state. Large catch increases generally resulted in poor conservation outcomes, illustrating the trade-offs between achievement of economic and conservation objectives. If one adopts 30% B_0 as a reference limit, then all scenarios where Australian effort increased above 16.8 million hooks placed the stock at high risk of being overfished. An increase to 16.8 million hooks resulted in a moderate risk.

Results were sensitive to assumptions about natural mortality, recruitment, and movement within the population, as well as to the assumed depletion level at the end of the historical time-period. The fixed effort scenarios showed that increasing future effort caused declines in conservation indicators, particularly at higher stock depletion levels. While the high (50%) depletion scenario may be considered unlikely, there may be a high risk in assuming only 15% depletion in 2001 given the size of the historical catches and the fact that the conditioning process indicated a better fit to the 30% depletion scenario.

The reference biological scenario assumed reasonable levels of movement between different regions in the southwest Pacific, and the performance of the fishery was relatively independent of the precise nature of this movement. However, under an assumption of no movement, the resource levels in areas with relatively high levels of effort, such as area 2, became depleted in the absence of replenishment from other regions. Localized depletions may also be likely if movement is limited. This result highlights the importance of area-specific performance indicators, since if only regional indicators are monitored, subregional changes in stock status may be overlooked. Indeed, declines in inshore catch rates of swordfish off eastern Australia already suggest that area-specific changes have occurred (Campbell and Hobday 2003). Given the importance of area 2 to the Australian fleet, area-specific indicators need to be included in performance monitoring of the fishery.

It is also apparent that the achievement of management objectives for any individual fleet will be dependent on the future effort levels of the other fleets. For example, when foreign effort remained at the status quo, as opposed to being doubled, the same level of Australian effort had a reduced probability of driving the spawning biomass to below 30% B_0, and the Australian fleet achieved higher catches for identical or lower levels of effort. Alternatively, when there were significant increases in both Australian and foreign effort, the likelihood of a poor conservation outcome was increased. As such, the impact of a given level of Australian effort also depends largely on the changes in effort of the foreign fleets, and on their efficiency. This reinforces the need for multilateral management arrangements for widely distributed and highly migratory stocks such as swordfish.

The empirical-based decision rules arrested spawning biomass declines by decreasing domestic effort until stability was attained in the monitored indicator variable. However, there is no guarantee that the resulting level of effort and catch would be optimal. Furthermore, previous studies have indicated that empirical approaches may not succeed for already depleted stocks (Punt et al. 2002). Ultimately, more comprehensive data and a more sophisticated stock assessment are needed to better estimate sustainable yields.

While this study focused on evaluating fixed effort strategies for establishing an initial TAE for the Australian fleet, the MSE framework should incorporate formal stock assessments for swordfish when they become available. Indeed, the MULTIFAN-CL stock assessments that have been developed for the principal tuna species caught in the western-central Pacific Ocean (e.g., Hampton and Fournier 2001) could be readily integrated into an MSE framework for evaluation of management strategies for these species in this region.

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Appendix. Fleet specific catchabilities

For the Japanese fleet, the projected fishing mortality for region A is a function of fishing effort E_t^A and time t as follows:

$$F_{t,1}^A = \tilde{Q}_{q,1}^A (E_{t,1}^A)^{\gamma_q} e^{\lambda_q t} e^{\varepsilon_{t,1}^A - \sigma_{A,1}^2/2} \tag{A1}$$

where

$\tilde{Q}_{q,1}^A$ is the catchability coefficient for area A and quarter q (corresponding to time t) for Japan (fleet 1)

γ_q is the nonlinearity factor for quarter q

λ_q is a factor to account for changes over time in effectiveness in quarter q, and

$\varepsilon_{t,1}^A$ is a factor to account for random variation in catchability

$\left[\varepsilon_{t,1}^A \sim N(0;\sigma_{A,1}^2)\right]$

This equation can be log-transformed and solved for $\tilde{Q}_{q,1}^A$, γ_q, and λ_q as a system of linear equations (ignoring the random variability term):

$$\log F_{t,1}^A = \log \tilde{Q}_{q,1}^A + \gamma_q \log E_{t,1}^A + \lambda_q \times t \tag{A2}$$

This is done for the historical time series, using the observed effort, so that the fitted values for $\tilde{Q}_{q,1}^A$, γ_q, and λ_q at the end of the historical period may be used in equation A1 for the projection time series. It should be noted that while γ_q and λ_q vary between quarter, they are similar for all regions. As such, temporal changes in catchabilities are assumed to be the same across all regions. This is based on the belief that factors which influence temporal changes in catchability impact in a synchronized manner across all regions.

For the Australian and New Zealand fleets, targeting progressively shifted onto swordfish after 1995 and fishers learned how to more efficiently target this species. Furthermore, unlike the situation for the Japanese fleet, the temporal changes in targeting practices for the Australian and New Zealand fleets were found to be different for each region. To describe these shifts, the following equation was used to describe the relationship between fully selected fishing mortality and observed effort for each fleet f for the projection time series:

$$F_{t,f}^A = \tilde{Q}_{t,f}^A \left(E_{t,f}^A\right) e^{\varepsilon_{t,f}^A - \sigma_{A,f}^2/2} \tag{A3}$$

where

$$\tilde{Q}_{t,f}^A = \begin{cases} Q_{q,f}^A & \text{if} \quad t \le t' \\ Q_{q,f}^A \left(1 + \gamma_{q,f}^A (1 - e^{-\lambda_{q,f}^A (t-t')}\right) & \text{if} \quad t > t' \end{cases} \tag{A4}$$

and t' is the time at which the increase in targeting commenced (set to first quarter of 1995 for the Australian fleet in all areas, and the first quarter of 1997 for the New Zealand fleet),

$\gamma^{A}_{q,f}$ is the amplitude of the asymptote for the catchability curve for each region, quarter and fleet,

$\lambda^{A}_{q,f}$ is the rate of learning/increase in targeting for each region, quarter and fleet, and

$Q^{A}_{q,f}$ is the baseline catchability prior to time step t', for each region, quarter and fleet.

The observed effort time series was applied in equation A3, and the least-squares difference between this and the model predicted fishing mortality was used to determine $\gamma^{A}_{q,f}$, $\lambda^{A}_{q,f}$ and $Q^{A}_{q,f}$, for each fleet, area, and quarter.

Fisheries Uncertainty: A Tropical Australian Data-Poor Fishery

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Abstract

Many world fisheries have been managed under data-limited situations with varying levels of success. Unfortunately this still occurs in many fisheries due to such factors as uncertainty during the initiation of new fisheries, lack of monitoring and enforcement resources, and vast coastlines with multiple access points. One example of a data-limited fishery is the commercial inshore gillnet fishery of northern Queensland, Australia, where one of the main species taken is the king threadfin, *Polydactylus macrochir*. Although rudimentary assessment of commercial catch and effort data from the king threadfin fishery indicates that this species currently is not overexploited, estimation of stock size using models has not been possible, and more robust assessments are hampered by limited biological data, an absence of monitoring data, unvalidated commercial logbook data, and a creep in fishing effort as technology advances. Under such circumstances a precautionary approach to management is mandated. We advocate the use of a phased approach to risk-averse management for the king threadfin fishery, and we recommend the use of marine protected areas (MPAs) to conserve threadfin aggregations as well as the use of maximum constant yield (MCY) to set a precautionary limit on annual catches. The use of such precautionary options in a phased management framework may be relevant to other fisheries similarly challenged.

Introduction

Many world fisheries have been managed under data-limited situations with varying success. In developed nations many fisheries are considered to be data-rich due to a long history of operation of fishing fleets and documentation of their harvest performance (e.g., Atlantic bluefin tuna; Fromentin 2003). In contrast, there are also data-poor fisheries that compromise managers because of such factors as inexperience during the development of new fisheries (Great Barrier Reef live fish trade, Mapstone et al. 2001; Namibia orange roughy, Boyer et al. 2001), a lack of resource monitoring and fishery enforcement (Gribble and Robertson 1998), and vast coastlines with multiple access points (see Parma et al. 2003).

However, despite having an extensive time series of catch and effort data, well-developed population models, and current biological information, many of these data-rich fisheries are over-exploited or have already collapsed (Safina 1995). With such a poor record, the application of the precautionary approach demands managers adopt conservative management strategies. This approach is called for when the level of uncertainty is high, particularly in data-limited situations. Indeed, many of the examples of fisheries failures can be attributed to the perceived need for more information delaying the need for better management (Walters 1998). The development in recent years of several management options that are considered risk-averse (e.g., Ministry of Fisheries 2002) now make it easier for managers to act responsibly and address concerns of risk and uncertainty in fisheries. This is not to say that the manager's job is an easy one, because the social and economic issues that must be considered in this process (Garcia 1994) often present conflicting objectives.

In Australia the majority of catch from domestic fisheries is exported. The Australian Commonwealth Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act) requires that all fisheries that are managed by the Commonwealth Government, or that export their product, must demonstrate ecological sustainability. As a consequence, stock assessments of target species have been initiated for a number of Australian fisheries, which in some cases represent the first formal assessment of the fishery. We carried out one such assessment on the king threadfin (*Polydactylus macrochir*) fish stocks, an important component of a valuable multispecies commercial net fishery in tropical Queensland waters (Welch et al. 2002b). This assessment identified many data deficiencies and uncertainties in the available database for the fishery. Further, the ecological relationships among the suite of species taken in the multispecies fishery were unknown. Such circumstances demand that a precautionary approach to management of king threadfin stocks be undertaken. In this paper, we examine the attributes of the king threadfin fishery that make it data-limited, and suggest precautionary management appropriate to such circumstances.

Table 1. Fishery and stock assessment attributes used to delineate data richness (derived from Restrepo et al. 1998).

	Data rich	Data moderate	Data poor
Life history characteristics	Yes	Yes	Unreliable or limited
Fishery dependent data (e.g., logbook catch and effort)	Yes	Yes	Unreliable or limited
Data time series	>20 years	Generally >20 years	Generally <20 years
Fishery independent data (e.g., monitoring)	Yes	Available or limited	No
Stock assessment	Sophisticated	Simple sophisticated	Minimal or lacking
Reliable MSY related quantities	Yes	Limited	No
Stock size estimate	Yes	Yes	No
Fishery parameters (e.g., selectivity, fishing mortality)	Yes	Yes	Unreliable or limited
Control rules	F_{msy}, B_{msy}, etc.	$F_{35\%}$, $B_{35\%}$, proxies, etc.	M, average catch, etc.
Data quality	High	High moderate	Moderate poor
Uncertainty	Accounted for	Reasonable characterization	Qualitative or lacking

What is a data-limited fishery?

In a fisheries context, a "data-limited" status is not necessarily easily defined. Of course the amount of data available for any one stock or fishery will vary but it is also likely that the perception of what constitutes "data-limited" will also vary. For example, in developed industrialized fishing nations, fisheries with good stock-recruitment relationship data may be considered "data-limited" (Robertson et al. 2002). In other fisheries the term may mean that information about catch, effort, species biology or even the species composition of the catch is lacking. It is also important to evaluate not only data quantity but also data quality. Whitworth et al. (2003) on page 113 state, "The quality of data should reflect the probability of making a serious management mistake." While these authors were referring to the collection of new fisheries data, it also applies to available and historic data. To provide clarification in terminology and application, we have adopted, and adapted, the definition of Restrepo et al. (1998) for what constitutes a data-limited fishery (Table 1). As noted by Restrepo et al. (1998), multispecies fisheries often include stocks in the species complex that are data-rich while some are data-poor.

The Queensland Gulf of Carpentaria king threadfin fishery

In northern Queensland, Australia, the inshore commercial net and recreational line fisheries target many different species on both the east coast and in the Gulf of Carpentaria (GoC). Both these regions represent vast areas that are very remote from human habitation and are largely inaccessible, except by vessel; management of these fisheries is implemented from Brisbane, the state capital, some 2,500 km away. The GoC inshore net fishery operates in tidal waters from the coastline and offshore to a 25 nautical mile limit (Fig. 1).

The main target species is the centropomid barramundi, *Lates calcarifer*, and further offshore various carcharinid sharks and scombrid mackerels. The second most valuable catch component in these inshore fisheries is the king threadfin, *Polydactylus macrochir*. The king threadfin is found in tropical and subtropical regions of Australia predominantly in shallow foreshore areas, but also in coastal rivers and estuaries (Kailola et al. 1993). King threadfin are highly prized as both table fish and sportfish, and are taken by commercial, recreational and indigenous fishers. The Queensland commercial inshore net fishery was recently valued at approximately AU$12 million gross value of product, of which the GoC king threadfin are worth approximately AU$2.2 million (Williams 2002), making this a significant contribution to the economy of coastal Queensland.

Biology and ecology

King threadfin (Fig. 2) congregate in large formations during the Southern Hemisphere autumn and spring months in foreshore areas and in tidal flats at the mouths of rivers (Garrett 2002, Garrett and Williams 2002). Garrett (1997) estimated maximum size and age of *P. macrochir* in the GoC to be 120.5 cm fork length (FL) and 14 years, respectively.

King threadfin spawn in late winter/early spring (Garrett 1997) and are protandrous hermaphrodites, changing sex from male to female during their life cycle (Griffin 1990). This has important implications for management, as the larger predominantly female fish need to be retained in the spawning biomass to ensure adequate egg production. The observed range of sizes for mature male and female king threadfins were 28-101 cm FL and 87-109 cm FL, respectively (Garrett 1997), suggesting that sex change occurs across a broad size range. Length at which 50% maturity in the female phase occurs is estimated to be 95.4 cm FL (114.9 cm total

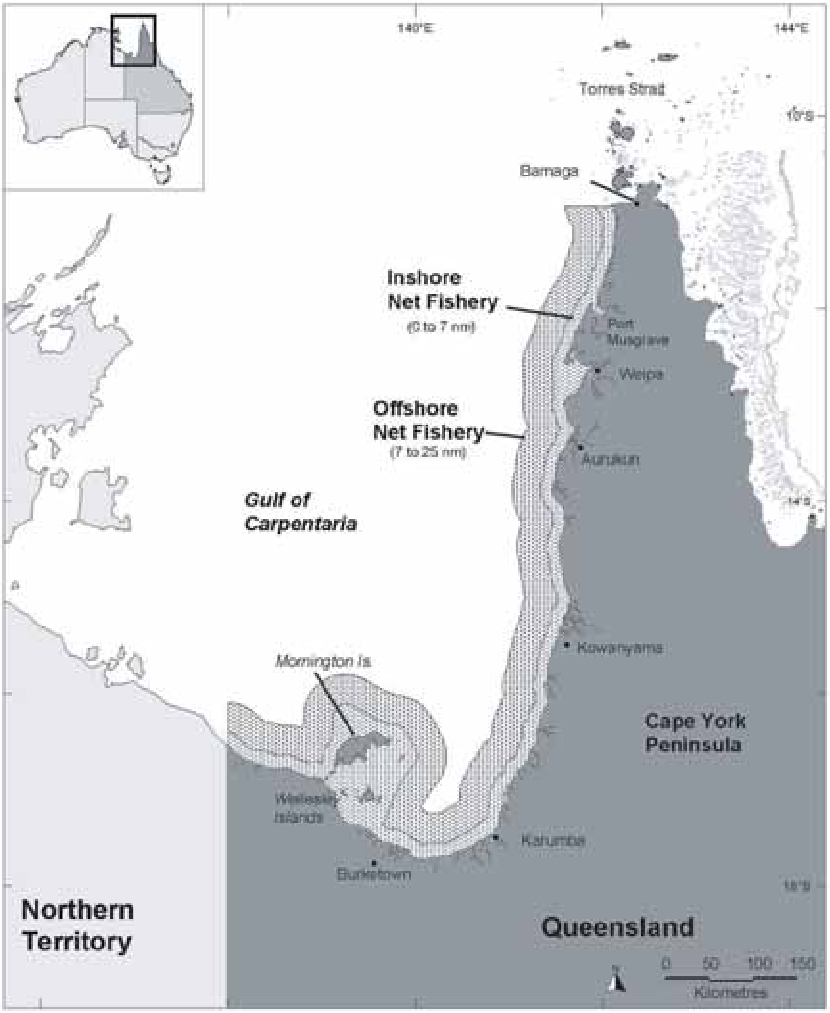

Figure 1. Map of the Gulf of Carpentaria in Queensland, Australia, showing the location of the inshore net fishery. The inshore net fishery has two components: the inner N3 fishery and the outer N9 fishery. King threadfins are almost exclusively taken in the N3 fishery.

Figure 2. A king threadfin specimen, *Polydactylus macrochir*.

length, TL) and 6-10 years old (Garrett 1997). This author estimated king threadfin reached maturity as males from 28 cm FL (34.7 cm TL).

Commercial fishing

The Queensland inshore commercial fishery targets different species depending on factors such as season, market forces, and catch rates of individual species. King threadfin are targeted when they form large concentrations in foreshore areas. Hand-hauled setnets have traditionally been used by commercial fishers in the GoC; however, the use of power-assisted net haulers (net reels) is reported to have increased in the fishery by at least 20% since 1998 (S. Peverell, fishery observer, Department of Primary Industries, Northern Fisheries Centre, Cairns, Australia, pers. comm., 26 June 2004).

Compulsory logbook recording of commercial catch and effort in its current format has been in effect in Queensland since 1988. However, the first year is considered a learning period and is not presented. The average annual catch of GoC king threadfin for the period 1989-2002 was 344 t and in 2002 catch was 444 t. Since 1989 the number of boats reporting catch of king threadfin each year in the GoC has ranged from 76 to 96 boats (Fig. 3). Effort has been relatively stable since 1994 at around 4,500 fishing days with an increase in 2001 and 2002 to around 5,400 days. Catch has increased since 1994 due to an increasing trend in catch per unit effort (CPUE) (Fig. 3). Higher catch rates in some years may be due

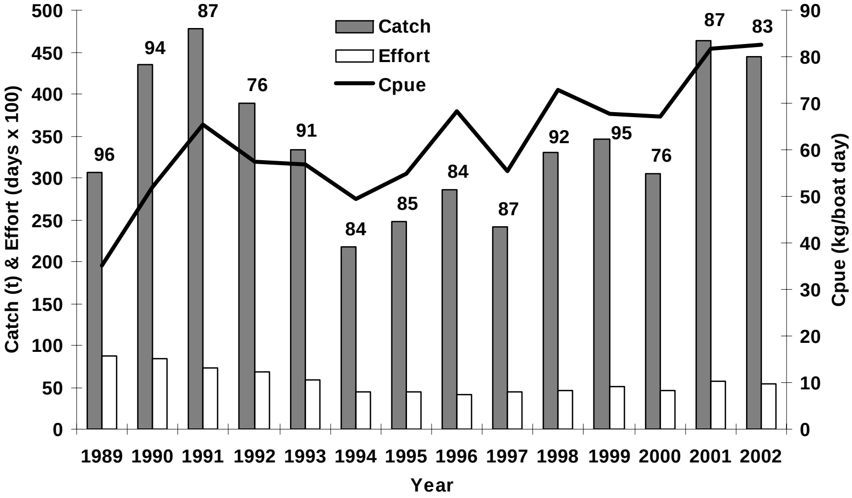

Figure 3. Catch, effort, and CPUE for king threadfin from the Queensland Gulf of Carpentaria inshore net fishery, 1989-2002. Numbers over the bars indicate the number of vessels that reported catch for that year. (Source: DPI&F CFISH database.)

to changes in fishing operations or to increased targeting of threadfins due to better market prices.

Recreational and indigenous fishing

King threadfin are a popular sportfish species for recreational anglers using hook and line, while the indigenous fishery uses hook and line, nets, traps, and spears (Garrett 1995, 2002; Garrett and Williams 2002). Historical information on catch and effort for both these fisheries is extremely limited and a recent estimate of catch was reported only at the scale of the entire Queensland state (Henry and Lyle 2003).

Resource monitoring

There are scant fisheries-independent data available for the king threadfin fishery and a current GoC fishery observer program does not collect data on the species. A statewide fisheries monitoring program currently does not collect data on king threadfin either.

Fishery management

Management decisions for the GoC inshore fishery in the past have been driven by consideration for the main target species only, the barramundi, with little regard for the multispecies nature of the fishery. Management initiatives were implemented in 1981 for the GoC fishery following concerns that barramundi stocks were in decline due to increased fishing pressure through the 1970s. The strategies employed effectively included all species taken by the inshore gillnet fishery, and included the following.

- A temporal closure each year during the barramundi's summer spawning season.
- A reduction in commercial effort by limited entry licensing and controls on allowable fishing gear.
- Compulsory monthly logbook reporting of commercial catch and effort.
- Protection of barramundi nursery habitats through area closures.

Changes to the king threadfin minimum legal size (MLS) and recreational bag limits were introduced in 1999, as well as the negotiation with industry for spatial closures through a buy-back of commercial licenses by government. Detailed spatial and temporal analysis of catch and effort has not been carried out to elucidate the effects on king threadfin, if any, of the implementation of these arrangements.

Stock assessments

Stock assessment of Queensland threadfin resources was first attempted in 1997 (Gribble 1998), and then in 2002 (Welch et al. 2002b). In order to estimate current and past trends in relative biomass of GoC king threadfin, attempts were made to fit different non-equilibrium biomass dynamic models to standardized commercial catch rate data using observation error estimation methods (Haddon 2001). A monthly production model developed for barramundi (D.J. Die, unpubl. data) at a stock assessment workshop (see Gribble 1998, Welch et al. 2002a) was used with the GoC king threadfin data. The model provided only a very poor fit to the observed data due to strong inter-monthly variability in CPUE. Subsequently, two simple models were used: an annual Schaefer model (Schaefer 1954) and an annual Pella & Tomlinson model (Pella and Tomlinson 1969). Two estimates of the catchability coefficient q were incorporated into the models to attempt to account for the increased efficiency likely to be introduced to the fishery with the reported increased use of net reels since 1998 (S. Peverell, Department of Primary Industries, Northern Fisheries Centre, Cairns, Australia, pers. comm., July 2002). In each case, the predicted index of CPUE was unreliable, with a very poor fit to the observed data. This is likely due to failure of the data rather than the models (Hilborn 1979). That is, there was not sufficient variability in CPUE and fishing effort to reliably estimate the parameters of the model (Hilborn and Walters 1992) and therefore stock size.

Uncertainties of the Queensland GoC king threadfin fishery data and parameters

In Queensland fisheries assessments, commercial CPUE is used as an indicator of trends in the status of a fish stock under the assumption that CPUE is proportional to stock size (Williams 2002). In the GoC king threadfin fishery, trends in CPUE do not show any indications that stocks are currently under threat. However, due to the aggregating nature of the species, any decline in the abundance of the stocks may not be reflected in changes in CPUE due to the efficiency of targeting individual schools (hyperstability) (Hilborn and Walters 1992). Also, fishers have reported that an increase in inter-annual variation has recently occurred in the frequency, size, and duration of threadfin aggregations on the inshore fishing grounds and that some operators now regularly target populations of adult fish located farther from the coast (Williams 2002). The increased expansion into new areas has been possible through the use of hydraulic net reels and may explain increases in catch in the last two years.

Effort creep may also be a factor affecting catches and catch rates for the GoC inshore net fishery, with the introduction of net reels and their use by a growing number of operators. Net reels greatly improve the efficiency of fishing activity in two ways: faster automated net retrieval means more sets can be employed in a given period of time, thereby improving catch rates, and product loss is likely to be significantly decreased. Using net reels, catch rates of over 1.5 t of king threadfin per hour per 600 m net have been recorded in the GoC (Gribble 2004). The introduction of net reels should increase the proportion of fish that are marketable.

Industry sources report that significant loss of product occurs mostly at the net due to predation, from deterioration of fish flesh quality during big catches and high water temperatures (> 30°C), and sometimes being left high and dry at low tides. Therefore reported catch is most likely to be an underestimate of total commercial catch, and by how much is a critical uncertainty. These factors also mean that post-release mortality is likely to be high. Uncertainty also exists in the reported levels of catch and effort because no validation of GoC logbook CPUE has been carried out (L. Williams, DPI&F Fisheries Business Group, Brisbane, Australia, pers. comm., 5 July 2002). Further, current GoC commercial net fishers logbook entry procedures do not allow any distinction in recording effort between targeted and incidental catch so that current measures of effort are not likely to be proportional to fishing mortality.

The MLS of 60 cm TL for GoC king threadfin does not allow 50% of females to spawn at least once before becoming vulnerable to the fishing gear. In particular the larger, older females recruit to the fishery well before the estimated size of 50% female maturity of 114.9 cm TL (Garrett 1997). Very few fisheries-independent age and length data of king threadfin are currently available, even for monitoring purposes, and the last dedicated sample collection program occurred a decade ago (Garrett 1997). Preliminary genetic work has indicated that separate sub-stocks may occur in the GoC (Garrett 1997). This observation, coupled with fishery targeting of king threadfin concentrations, suggests there is potential risk of localized depletions of sub-stocks. Further, a large portion of the GoC falls under a separate jurisdiction, the Northern Territory, where management arrangements for threadfins differ from Queensland. Also, from preliminary analyses, estimation of the size of the GoC threadfin stock has not been possible (Gribble 1998, Welch et al. 2002b).

Precautionary fishery management options for the GoC threadfin fishery

Based on the definitions provided in Table 1, and given the uncertainty in the available data set, we consider that the Queensland GoC king threadfin fishery is data-poor. The uncertainties present in the fishery data and parameters are many and varied. Individually, each represents a potential risk of stock depletion, but collectively this risk is increased. In such a situation the Environment Protection and Biodiversity Conservation (EPBC) Act demands the implementation of a precautionary approach to management until levels of fishery uncertainty are reduced. In recent years, several different management strategies have been promoted as being risk-averse and suitable for situations of uncertainty. Here we discuss some of these options with particular reference to their suitability for the GoC king threadfins.

Setting reference points

Setting reference points is a management strategy that has received much attention in recent years. A limit reference point represents an upper (or lower) limit for the safe exploitation of a fished stock while a target reference point represents the level of exploitation that takes into account uncertainty (Caddy and McGarvey 1996). Under this scenario, the target reference point is determined by the level of precaution deemed to be required in managing the particular stock. This approach could be applied for the GoC king threadfin, with the major limitation being that the only quantitative estimates that could be used as reference points for the fishery involve the use of catch and/or CPUE. As stated above, Queensland fisheries managers use commercial CPUE as a major indicator for the state's fisheries. However, these variables are notoriously unreliable as fishery indicators and we caution the use of catch or CPUE as reference points for GoC king threadfin stocks until a greater understanding is attained of the relationship of such indicators to stock size.

Validation of commercial logbook data is also required to reduce the current uncertainty in the use of these as fishery indicators.

Quota systems

In data-poor fisheries, a quota (total allowable catch, TAC) can be scaled downwards to create a buffer to compensate for uncertainty, and can be empirically determined or "guessed." Frederick and Peterman (1995), using allowable harvests based on point estimates, used a simulation model to estimate the optimal adjustment and showed that by incorporating uncertainty into the simulations, large adjustments to harvest levels are often needed. They also argued against arbitrary adjustments to account for uncertainty. Further, it is likely that an adaptive quota management framework that is highly responsive to trigger events will perform better in uncertain situations (Sladek Nowlis and Bollerman 2002). Individual transferable quotas (ITQs) are a preferred approach as they negate the "race for fish" attitude that can occur with a TAC, but good monitoring procedures need to be in place as management with ITQs also may encourage under-reporting of catch and high-grading (Hilborn and Walters 1992).

Currently, no quota system operates for the GoC king threadfin or any other species taken in the inshore fishery. Quota management systems usually rely on estimation of MSY or an alternative, and so assessments of stocks need to be more accurate (Walters and Pearse 1996); however, it is likely that we will never be able to reliably estimate stock size even for data-rich fisheries (Walters 1998). The New Zealand Ministry of Fisheries has developed a conservative strategy for data-limited situations by using a method for determining an alternative to MSY. Maximum constant yield (MCY) is defined as "the maximum constant catch that is estimated to be sustainable" (New Zealand Ministry of Fisheries 2002). MCY represents the average catch that can be taken from a stock taking into account the natural variability inherent in the particular stock. As long as the catch is below this range of natural variability it is considered to be sustainable. Caution with the use of MCY should be exercised, however, as using the MCY strategy at low stock sizes involves high risk because a larger proportion of the stock is being removed. Walters and Pearse (1996) state that with uncertainty, quotas need to be conservative, possibly to the point of cutting economic gain. This would certainly be the case with the GoC king threadfin fishery, as the use of the MCY strategy and its equations used to calculate catch levels would result in a reduction in catch of approximately 46% based on current reported levels.

Quota management can be a robust conservative strategy for a data-limited fishery, although implementation and enforcement in a vast and isolated region such as the GoC may make them cost-prohibitive. An expansion of the onboard fishery observer program through industry funding may provide one solution to alleviate this issue. Further, the use of quotas based on commercial catch would also necessitate the validation of logbook information.

Spatial and temporal closures

Marine protected areas (MPAs) have been increasingly used globally as a conservation and fisheries management strategy. Many authors advocate MPAs as insurance in situations of persistent uncertainty (e.g., Lauck et al. 1998). This concept has merit; however, it relies on guesswork to some extent as optimal design of MPAs is not fully understood and will likely vary for different species and different fisheries. Several authors recognize this and suggest that MPAs should be implemented in an adaptive management framework (Sainsbury 1991, Carr and Raimondi 1999). In a fisheries context, the effectiveness of MPAs has been demonstrated using model simulations. For example, Holland (2003) demonstrated that area closures could increase both the productivity and profitability for the Canadian Georges Bank multispecies groundfish fishery. In real data situations, temporal and spatial closures have been effective in protecting spawning aggregation for red hind in the U.S. Virgin Islands (Beets and Friedlander 1999) and in maintaining fisheries yield in adjacent fishing grounds (Russ and Alcala 1996). In the GoC a temporal closure in place during the spawning season of barramundi appears to have contributed to an increase in the stock size of this species (Welch et al. 2002a). A number of spatial closures have been implemented in the GoC; however, threadfins are a fairly mobile and schooling species and these closed areas are currently very small in terms of the available fishing area. Further, they are predominantly located in estuaries and river mouths to ensure protection of barramundi spawning and nursery areas. A precautionary approach to protect king threadfin stocks effectively using MPAs in the GoC would require the protection of several larger areas of foreshore habitat where threadfins are known to aggregate, and the guaranteed enforcement of these areas (see Gribble and Robertson 1998). Due to the mobility of king threadfins, the effectiveness of MPAs would need to be carefully monitored until further information is collected, and targeted protection of the fish during their spawning times may prove the most effective use of MPAs. Implemented in an adaptive framework, MPAs are likely to represent the strategy that is the least resource demanding in protecting GoC king threadfin stocks.

Other strategies

Input controls such as those already implemented in the GoC inshore net fishery, although less highlighted in the literature as precautionary measures, can be adjusted to allow for uncertainty. Examples include effort controls such as reductions in net length, changes to mesh sizes, or a reduction in the number of fishing licenses. For the GoC inshore net fishery, effort creep has been identified as a potentially problematic issue.

Therefore the banning of net reels may represent a realistic precautionary management option both from an implementation and an enforcement perspective. However, such a move would be sub-optimal economically and would not alleviate any issues relating to product loss.

When carried out against clearly stated and quantifiable objectives, simulation modeling is a useful tool in assessing different management strategies under uncertainty (Smith et al. 1999, Milner-Gulland et al. 2001, Hoyle 2002). Ideally this approach would be used for the GoC inshore net fishery. However, the data requirements currently exceed availability for king threadfin.

Future assessments

In terms of addressing data issues, it is worth noting that data-limited fisheries often arise because of certain characteristics of the fisheries themselves. Such fisheries may be new or developing, may be of low value, or the cost of data collection may be prohibitive due to the geographic spread and remoteness of the fishing grounds and a multiplicity of access points. All of these issues are relevant to the GoC king threadfin fishery. Some of these issues may be overcome through better use of the existing data (Hall 2003), greater involvement of industry in data collection and fishery monitoring (Parma et al. 2003), or by using data proxies (Hoenig 1983, Restrepo et al. 1998) and meta-analyses (e.g., Myers et al. 1999). Despite this, all Australian fisheries must now legally demonstrate ecological sustainability, and to do this for the GoC king threadfin fishery a greater degree of certainty in the current fishery data and parameters is clearly required. This would suggest increased data collection and improved fishery-dependent and fishery-independent monitoring for the GoC king threadfin fishery be required by law.

Multispecies considerations

Traditionally, management of multispecies fisheries has relied on species-specific assessments without much consideration of interspecific relationships and hence the flow-on effects of fishing for particular species. In meeting ecologically sustainable development (ESD) requirements and with the availability of greater computer power, there has been a growing impetus in recent years toward investigating ecosystem-oriented assessment and management, which attempts to account for relationships among target and non-target species (e.g., Ecopath: Christensen and Pauly 1992; Ecosim: Walters et al. 1997). Development of these methods is progressing slowly due to the lack of understanding of these complex relationships among species. Despite the challenges brought by the data-intensive requirements of multispecies approaches, consideration of the effects of fishing on all species groups is now mandatory in Australian fisheries.

An important aspect of the king threadfin fishery in the GoC is the multi-specific nature of the entire inshore net fishery, of which it is a component. The species predominantly taken in the N3 component of the inshore net fishery, which operates out to 7 nautical miles, are: barramundi, king threadfin, and four-fingered threadfin (*Eleutheronema tetradactylum*), with small quantities of grunter (*Pomadasys kaakan* and *P. argenteus*) and triple tail cod (*Lobotes surinamensis*) (Williams 2002). The separately licensed N9 component of the inshore net fishery operates in the band 7-25 nautical miles from shore, targeting tropical shark (about eight species) and grey mackerel (broadbarred king mackerel, *Scomberomorus semifasciatus*) (Williams 2002). These species are also taken by Northern Territory fisheries, managed separately under a different jurisdiction but operating in a significant portion of the GoC. The ecological relationships among all these species are not understood and to date have not received consideration. Barramundi in particular also represent an iconic species across northern Australia due to a valuable sportfishing industry, their fine table quality, and their significance in indigenous folklore. The ecological consequences of a collapse in GoC king threadfins are unknown. A possible consequence of a collapse in king threadfins would be a shift in effort to other species within the fishery. Currently, barramundi stocks are thought to be increasing under current management arrangements (Welch et al. 2002a), four-fingered threadfin stocks are considered at high risk due to uncertainty (Welch et al. 2002b), and no monitoring or assessment has been undertaken for the other species (Williams 2002). Barramundi are a high trophic level predator, and their increasing numbers could impact the other target species via recruitment processes, juvenile and adult predator-prey relationships, or competition. It is also possible that increased targeting of alternative species other than barramundi would have negative impacts on those stocks. Alternatively, a decline in total catch of any of the other target inshore species may result in increased targeting of king threadfin.

The consequent social and economic effects for the GoC fishery and the GoC community of a collapse in king threadfins are also unknown, and are likely to be significant. The barramundi is a major draw card for sportfishers from around the world for a thriving and valuable sportfish industry in the GoC, of which threadfins are a component.

Development of multispecies approaches are occurring slowly using model simulations. These methods are data-intensive and, so far, fishery-specific. Although the potential relationships among species in the GoC are certain to be complex and data-intensive, that does not mean that we should not attempt to consider the interrelationships in assessment for management. Single-species assessments are likely to be the only approach for the present for data-poor fisheries such as the GoC inshore net fishery. But we can still put assessment of this fishery into a multispecies framework, if only rudimentary. Assuming there are ecological linkages among the species targeted in the GoC, then it is likely that some basic fishery indicators will also be linked among the species. Such indicators include fisheries-dependent (catch, CPUE) and fishery-independent (size, age) information. Even in data-poor fisheries, these data types have often already been collected or can be collected relatively easily and cheaply. Fishery-independent data represent the most reliable indicators, and such data are not currently available for any species other than barramundi in the GoC inshore fishery. The inclusion of other inshore species in the monitoring program is an urgent challenge for Queensland fisheries management.

Long-term data acquisition on the ecological relationships among targeted species will help achieve the sustainable management of the fishery; however, the complexity of such relationships means that management for sustainability must proceed for the moment in the absence of this knowledge. In Australia, the EPBC Act promotes ecosystem-based management, and since many of Australia's fisheries can be considered data-poor, this approach may be prohibitive at the moment. It is more likely that simple alternatives for assessment and management will be the only options for data-poor multispecies fisheries and these alternatives should be explored and adopted to the greatest extent possible.

Conclusions and recommendations

As in other data-limited fisheries, this paper highlights a fishery lacking in information traditionally used to guide management. The king threadfin fishery in the multispecies GoC fishery, although having been in operation for over 15 years, is still in an early stage of assessment. This is due, in part, to a lack of institutional resources necessary to carry out assessments adequately. The fishery is data-poor and a high level of uncertainty is associated with the status of the stock. It may well be that, contrary to indications outlined above, king threadfin populations in the GoC are healthy and stable. But clearly, without data collection, there is no sure way to establish the real status of the resource. In the absence of information and resources necessary to carry out more formal analyses, a precautionary approach to management should be adopted until greater certainty in stock levels can be established. The GoC is an area that presents many challenges for fisheries managers due to its vastness and isolation, and any new management would require care in implementation with consideration of enforcement capabilities. We advocate the use of a phased approach to risk-averse management for the king threadfin fishery and we recommend the use of marine protected areas (MPAs) to conserve aggregations. We also recommend the concurrent use of maximum constant yield (MCY) to provide a precautionary limit on annual catches. Careful monitoring of CPUE, and the accumulation of further data that reduce the current uncertainties identified for king threadfin stocks, will allow the phasing-in of future adjustments in allowable catch levels and timing and location of MPAs.

The development of strategies to improve king threadfin fishery data quantity and quality should also be adopted, preferably within a risk assessment framework to optimize data collection in a resource-limited management environment. Immediate requirements necessary for the effectiveness of the management scenarios suggested above are the validation of commercial logbook information and the identification of areas within the entire GoC where king threadfin aggregate, and whether they represent spawning aggregations. Also critical and a major recommendation is the inclusion of king threadfin on the species list currently monitored annually using fishery-independent surveys. Knowledge of the annual age structure of the catch of king threadfin taken in different areas of the GoC would very likely provide a reliable indicator for king threadfin stocks.

These suggested approaches represent pragmatic options under the circumstances of the king threadfin fishery in the GoC, and reflect the urgent need to reduce levels of uncertainty relating to stock health. Referring back to the quote of Whitworth et al. (2003) made earlier in this paper, data collection needs to be strategic and reflect the risk of not having that information. For data collection in data-poor fisheries similar to the king threadfin fishery, this is a simple and reasonable philosophy to adopt. Ultimately, however, policy-makers need to back up legislation with resources to meet requirements that ensure fishery sustainability, including actual implementation of the precautionary approach in management.

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Canada's Staged Approach to New and Developing Fisheries: Concept and Practice

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Abstract

As participation in established fisheries in the Northeast Pacific has decreased due to declines in fish stocks, there have been increasing requests for new fisheries on unexploited species and for the expansion of fisheries on lightly exploited species. In the late 1990s Fisheries and Oceans Canada developed an approach for evaluating proposals for new fisheries and providing scientific advice to fisheries managers when little information was available on the proposed target species. This approach involved four stages: (1) compiling existing information; (2) collecting new information; (3) exploration via a small commercial fishery; and (4) a full-scale commercial fishery. To date, 24 species have been assessed in the Pacific Region using this approach, including invertebrates, small pelagic finfishes, and groundfishes. Experiences using this approach have shown that development of new fisheries can proceed in an orderly manner while preventing overcapitalization and subsequent "gold-rush" fisheries. This approach also builds partnerships between fishery assess-

ment and regulatory agencies and fishing proponents. Valuable biological information is collected prior to or at the start of a fishery, which is essential to fully understand the subsequent impacts of fishing. A multiyear time perspective must be accepted that allows industry to develop a business plan which evaluates market and environmental variability and their impacts on profitability. The biggest hurdle to the approach is the high cost of field studies without a guarantee that a new fishery will result. With the increasing and contradictory demands for access to marine resources and for conservation of these resources globally, the scientific, policy, and implementation experiences of developing new fisheries on Canada's Pacific Coast are an important contribution.

Introduction

Many fisheries are declining around the world. FAO statistics indicate that over 50% of commercial fish stocks were classified as mature or senescent in 2000 (FAO 2002). Canada has experienced recent highly publicized declines in important fish populations in the Atlantic (Atlantic cod [*Gadus morhua*] and other groundfishes) and in the Pacific (Pacific salmon). This has generated considerable interest in fishing for previously unexploited or lightly exploited species. As a result, there has been strong political pressure to find other fishing opportunities for displaced harvesters. The traditional approach to developing new fisheries has been an open fishery with few regulations or restrictions other than to report catches or landings and perhaps their general locations. This has been justified by the usually small nature of these fisheries. However, such small fisheries can explode into "gold-rush" fisheries surprisingly quickly if good markets become established for the product (Hilborn and Sibert 1988).

Attitudes have changed, however, and there is now much greater emphasis on conservation of fish populations and the need for a precautionary approach to the exploitation of living marine resources (FAO 1996). In particular, it is recognized that lack of scientific information on a species or stock is not an excuse for a lack of management actions. Here the argument becomes circular: a fishery should not be started without some type of scientific information and management control, yet what type of controls should be implemented if there is no prior information (which would usually come from a fishery)?

With this background, and within the context of recent declines of some Pacific salmon stocks, the Pacific Region of Fisheries and Oceans Canada (DFO) developed a framework for providing scientific advice for the management of new and developing data-limited fisheries (Perry et al. 1999). Although the framework was initially developed for invertebrate species, its general principles are widely applicable. In October 2001, this framework formed the basis of the national policy for the development of new fisheries and the expansion of developing fisheries in Canada. The

Table 1. Summary of the staged framework for providing scientific advice for the management of new and developing invertebrate fisheries (modified after Perry et al. 1999).

Stage 1	Information	• Literature searches to identify biological and fisheries information on target (and similar) species. • Analyses of information from local knowledge, other fisheries, and other studies. • Meta-analyses to determine distributions of parameters. • Development of alternative management approaches.
Stage 2	Experimentation	• Studies to develop new information to fill critical gaps determined from stage 1, e.g., unit stock size, spatial scale and pattern of aggregations, metapopulations, habitats, densities and abundances, biological characteristics. • Evaluate alternative management approaches and recommend regulatory reference points.
Stage 3	Exploration	• Implement selected management approaches in a closely observed fishery. • Use information from the fishery to improve analyses conducted in stage 2; modify management strategies.
Stage 4	Establishment of a new fishery	• Develop a formal fisheries management plan. • Monitor the new fishery to provide information on outstanding questions, and for analyses of the performance of the fishery and stock responses.

present paper is a report on the activities under this policy. Its purpose is to provide an overview of the approach taken in Canada's Pacific Region toward new and developing fisheries, the scientific and policy problems that it has encountered since its implementation in 1997, the interactions with industry, and its successes and failures.

Concept

Staged approach

The Canadian policy on new and developing fisheries (DFO 2001) is based upon a framework that was originally presented by Perry et al. (1999) for providing scientific advice in data-limited fisheries. The principal objective of this framework is to assure the sustainability of new and developing fisheries by identifying the scientific information required to provide appropriate management regulations. A four-stage process was developed to obtain this information (Table 1). Stage 1 is the Information Stage, the purpose of which is to gather existing information on the target species or similar species, identify critical information gaps, and identify potential management approaches (e.g., King and McFarlane 2003). Stage 2 involves Experimentation, in which experiments and/or surveys are conducted, either as part of experimental fisheries or as fishery-independent activities, to resolve the critical information gaps identified in stage 1. This is followed by an Exploratory Stage (stage 3), during which exploratory fisheries are conducted to determine whether the stock can support an economically viable fishery. Stage 4 is establishment of a new fishery with an approved Fishery Management Plan (FMP), allocation of the resource to user groups, and a plan for monitoring the fishery.

Policy implementation

The Canadian policy defines new and developing fisheries as interim fisheries permitted, in the absence of an approved Integrated Fisheries Management Plan, to target previously under-exploited, unexploited, or non-target species. Specifically, this includes species for which there is pressure to enlarge small scale harvests and to commercialize the fishery, species that are not currently harvested but for which interest in harvesting has been expressed, and bycatch species that face increased harvests by becoming target species. The policy principles are in accordance with international recommendations for precautionary approaches to all fisheries.

Prioritization model

There are many more species in Canada's Pacific marine ecosystems than are currently exploited and, as might be expected, there have been more proposals for new fisheries than can be accommodated. Government is required, at minimum, to evaluate proposals for new fisheries and to establish management regulations. The practice has been for DFO staff to conduct most of the stage 1 and 2 evaluations. To rationalize the allocation of scarce government resources between new and developing fishery proposals, a prioritization model was developed. This fisheries prioritization model includes biological, social, and economic criteria that reflect Canadian principles and priorities. An initial model was developed by a team of scientists and fishery managers in the Pacific Region responsible for a variety of invertebrate and vertebrate species. This team identified criteria for assessing new fisheries and developed the basic measures and weights of the model (Table 2). Biological criteria include biomass, stock resilience to perturbations, the distribution of the stock within its overall geographic range, the degree of gear selectivity, stock availability to a fishery, and the potential habitat impacts of commercial

Table 2. Definitions of the individual criteria for the new and developing fisheries prioritization model.

Biological criteria	
Abundance	Biomass rather than numbers so as to be comparable across species.
Resilience	Ability of a stock to resist, or to recover from, perturbations.
Location within range	Distance from the center to the edge of the range as a proxy for the likelihood of experiencing adverse environmental conditions.
Selectivity	The degree to which the target species can be captured with the proposed gear.
Availability	The proportion of the stock available to a fishery due to geographic constraints.
Habitat impacts	Potential impact of proposed gear on fish habitat.
Economic criteria	
Net income from fishing	Anticipated wholesale value of harvest (estimate of the sustainable catch × average price).
Cost of the fishery	Cost of assessment + management + enforcement.
Social criteria	
Employment	Number of jobs created by a fishery.
First Nations	Cultural interest, harvesting interest, processing ability.
Coastal communities	Economic benefits to coastal communities.

harvesting. The economic criteria assess the extent to which a prospective new fishery could be economically sustained over the long-term. Criteria include wholesale value of the fishery and the cost of the fishery including assessment, management, and enforcement. Social criteria include the number of jobs likely to be created by a fishery (including harvesting and processing sectors), First Nations issues such as harvest and cultural interest, and the potential economic spin offs that might accrue to coastal communities. Measurement scales were developed for each of these criteria based on empirical data from established fisheries. Weights for each criterion were established based on their perceived policy importance. These scores (measurement times weight) are then summed to provide an overall score for each species. Detailed information on many of these criteria may not be available for unexploited species although some information may exist from other locations and for lightly exploited species.

Practice

From 1997 to 2003, DFO evaluated twenty-four species or taxonomic complexes using this staged approach (Table 3). This included invertebrates and pelagic and demersal finfishes. Two species, venus clam (*Compsomyax subdiaphana*) and California mussel (*Mytilus californianus*), were rejected at stage 1 because of concerns over harvesting impacts to habitat. Four species, Pacific hagfish (*Eptatretus stoutii*), brown box crab (*Lopholithodes foraminatus*), purple sea urchin (*Strongylocentrotus purpuratus*), and inshore Tanner crab (*Chionoecetes bairdi*), stalled partway through the process due to industry concerns over development costs and a general lack of funding. One species, neon flying squid (*Ommastrephes bartrami*), stalled due to a lack of interest by industry once market prices declined and environmental conditions changed to make the species less available in Canadian waters. One species complex, horse clams (*Tresus capax/nuttallii*), stalled due to concerns about interactions with a lucrative existing fishery; however, these have now been resolved and stage 2 is under way. Four species, swimming scallops (*Chlamys hastata/rubida*), varnish clam (*Nuttallia obscurata*), coonstripe shrimp (*Pandalus danae*), and humpback shrimp (*Pandalus hypsinotus*), are presently in stage 2 with experiments to provide critical information on distributions and responses to fishing. The two shrimp species were originally bycatch or incidental targets in fisheries for more abundant shrimp species; they became directed targets as more stringent management regulations were placed on the larger shrimp fisheries. One species, grooved Tanner crab (*Chionoecetes tanneri*), is in stage 3 with active experiments and exploratory projects under way with joint participation from industry and DFO. Eight species/taxonomic groups, surfperch (Embiotocidae), surf smelt (*Hypomesus pretiosus*), northern anchovy (*Engraulis mordax*), Pacific octopus (*Enteroctopus dofleini* and *Octopus rubescens*), opal squid (*Loligo opalescens*), goose-neck barnacles (*Pellicipes polymerus*), and sea cucumber (*Parastichopus californicus*), are in a mixture of stages 3 and 4. These species pose a special problem as they are the subject of existing, small, and largely unregulated fisheries that are being put into this framework. Only one species, Pacific sardine (*Sardinops sagax*), is exclusively in stage 4 (a "full" fishery).

The prioritization model was developed and tested in 2001-2002, and therefore has not yet been applied to new fisheries. Trial runs of the model have produced rankings for many species that are considerably different from their current priority in the staged approach. For example, grooved Tanner crab, which is presently in stage 2, was ranked low by the prioritization model. Such differences reflect the interest of proponents for particular species over other species.

Examples

California mussels and venus clams both entered the evaluation process at stage 1 (Gillespie 1999, Lauzier 1997). The request to fish for venus clams came from a single individual, whereas the request to fish for California mussels came from a community organization. Both species were rejected after their stage 1 evaluations due to concerns over the habitat impacts of harvesting. California mussel beds are unique habitats that support over 300 species and if damaged have a long successional period to recover. There were serious concerns that harvesting this species in British Columbia at commercial levels could not be sustained without the use of fishing practices that would lead to significant habitat damage. Venus clam fisheries were not pursued due to the habitat damage that would be caused by a dredge fishery in an ecologically sensitive area such as the southern Gulf Islands of British Columbia.

Two species that were not the subject of commercial fisheries in B.C. prior to being proposed as new fisheries are the neon flying squid and the grooved Tanner crab. The stage 1 review of neon flying squid identified the lack of information and the highly migratory, oceanic, nature of this species as concerns (Gillespie 1997). A stage 2 experimental fishery was permitted with observer coverage to learn how to fish these squid using lights and jigs and where to find them in the deep waters off western Canada. Promising catch rates were achieved in 1998, but by 1999 environmental conditions had changed making these squid less available to Canadian vessels and catch rates were disappointingly low. Environmental conditions have remained unfavorable since 1999 and industry has lost interest in this species.

Exploratory fishing for grooved Tanner crab took place off B.C. in the late 1980s, but ceased when neither of the two fishermen renewed their permits (Workman et al. 2002). A decline in the Dungeness crab fishery and increased prices for *Chionoecetes* crabs in 1997 renewed interest in fishing for grooved Tanner crab. Industry organized and formed a Tanner crab fishermen's association to represent their interests. The stage 1 review (Phillips and Lauzier 1997) identified data gaps in distribution and abundance, stock structure, life history characteristics, and bycatch in existing fisheries. Subsequent stage 2 studies (Boutillier et al. 1998b; Workman et al. 2000, 2002) reported on the results from a DFO trawl survey to assess abundance, a joint DFO/industry coastwide trap survey to determine stock distributions, and analyses of life history features. As the surveys were completed, conservative commercial fishing quotas were set for limited areas of the coast. In addition, one area was closed to Tanner crab fishing as a control, and a second area was given an aggressive quota that exceeded the biomass estimate for that area, in order to test the assessment methods. This project is continuing.

The sea cucumber program includes elements of stage 2 (Experimentation), stage 3 (Exploration), and stage 4 ("full" fishery). It represents a

Table 3. Summarized information and references for new or developing fisheries assessed by the Canadian Department of Fisheries and Oceans under the staged approach, 1997-2003.

Species	Stage	Reference	Comments
Invertebrates			
Inshore Tanner crab *(Chionoecetes bairdi)*	1	Krause et al. (2001)	Unsolicited paper, process continues subject to funding; beginning phase 1.
Offshore Tanner crab *(Chionoecetes tanneri)*	1	Phillips and Lauzier (1997)	Project in stage 1; experimental fisheries to test productivity and collect biological information.
	2	Boutillier et al (1998b); Workman et al. (2000, 2002)	
Milky venus clam *(Compsomyax subdiaphana)*	1	Lauzier (1997)	Project discontinued after stage 1 due to concerns of habitat impacts of dredges.
Box crab *(Lopholithodes foraminatus)*	1	Zhang et al. (1999)	Project stalled by industry concerns over development costs.
	2	Zhang (2001)	
Sea mussels *(Mytilus californianus)*	1	Schmidt (1999)	Project discontinued prior to stage 1 due to concerns of habitat damage by fishing.
	2	Gillespie et al. (1999)	
Varnish clam *(Nuttallia obscurata)*	1	Gillespie et al. (1999)	Project ongoing in stage 2, experimental harvests to test markets, gather information on productivity.
	2	Gillespie et al. (2001)	
Neon flying squid *(Ommastrephes bartrami)*	3	Gillespie et al. (1997)	Project stalled in stage 2 due to industry indifference; reduced price and market changes rendered fishery economically unviable at present.
Coonstripe shrimp *(Pandalus danae)*	1	Dunham and Boutillier (2001)	
	2	Dunham et al. (2002)	
Humpback shrimp *(Pandalus hypsinotus)*	1	Boutillier and Nguyen (1999)	
Purple sea urchin *(Strongylocentrotus purpuratus)*	1	Workman (1999)	Project discontinued after stage 1 due to lack of funding.
Horse clam *(Tresus capax, T. nuttallii)*	1	Lauzier et al. (1998)	Project stalled at start of stage 2 due to unresolved policy decisions; now under way.
	2	Zhang and Campbell (2000)	
Swimming scallops *(Chlamys hastata, C. rubida)*	1	Lauzier and Parker (1999)	Project ongoing in stage 2; directed surveys and experimental fisheries under scientific license to gather abundance and biological information.
	2	Lauzier et al. (2000)	

Table 3. (Continued).

Species	Stage	Reference	Comments
Invertebrates (continued)			
Octopus *(Enteroctopus dofleini, Octopus rubescens)*	1	Gillespie et al. (1998)	Project ongoing in stage 2; limited conventional fishery under scientific license, additional collection of biological information through logbooks.
Opal squid *(Loligo opalescens)*	1	Walthers and Gillespie (2002)	Project ongoing; fish management proposal to limit fishery under consideration by regional management.
Sea cucumber *(Parastichopus californicus)*	1	Boutillier et al. (1998a)	Project ongoing in stages 2-4 (commercial fishery continues in limited areas).
	2-4	Hand and Rogers (1999); Bureau and Hand (2005)	
Goose barnacle *(Pollicipes polymerus)*	1	Lauzier (1999a)	Project ongoing in stage 2; directed surveys and experimental fishery to explore "turf" management and incorporate local knowledge into assessment.
	2	Lauzier (1999b)	
Marine gastropods	1	West (2002)	Scientific review recommended each species be re-examined individually. Process stalled.
Finfish			
Pacific hagfish *(Eptatretus stoutii)*	1	Leask and Beamish (1999)	Interest waned as costs were devolved to industry
	2	Benson et al. (2001b)	
Pacific sardine *(Sardinops sagax)*	1	Ware (1999)	A stock assessment process is in place, based on U.S. stock estimates and a fishery management plan is being formulated.
	4	Schweigert and McFarlane (2001)	
Elasmobranch fishes	1	Benson et al. (2001a)	
Surfperch (Embiotocidae)	1	Lane et al. (2002)	
Northern anchovy *(Engraulis mordax)*	1	Therriault et al. (2002a)	Existing fisheries are coming into the New and Developing Fisheries policy. Information gathering is being improved.
Surf smelt *(Hypomesus pretiosus)*	1	Therriault et al. (2002b)	
Bocaccio *(Sebastes paucispinus)*	1	Stanley et al. (2001)	A report was prepared under the auspices of endangered species evaluations.

special class of species, those which were previously subjected to small commercial fisheries in B.C. For such species, once the open commercial fisheries are closed, there has been strong pressure to move them into and through the staged framework. The application of the staged approach to sea cucumbers is reviewed by Bureau and Hand (2005, this volume). Although there were no signs of serious declines in this fishery, concerns over the lack of information and whether management actions were precautionary warranted its inclusion into this framework. Briefly, a ten year plan was developed which closed 50% of the B.C. coast to fishing for sea cucumbers, opened 25% of the coast to fishing under the existing quota, and developed experimental fisheries in the remaining 25% of the coast to test stock responses to varying harvest rates. The project has experienced good collaboration among DFO, the existing sea cucumber fishing industry, and First Nations, with proceeds from the experimental fisheries being used to help fund the research activities.

Pacific sardine (*Sardinops sagax*) is an example of a species that is moving into stage 4 (establishment of a new fishery). Pacific sardine is similar to neon flying squid in that its occurrence in Canadian waters is strongly influenced by environmental conditions. Unlike flying squid, however, environmental conditions have favored the presence of sardines off B.C. for the past several years (McFarlane et al. 2002). Ware (1999) and Schweigert and McFarlane (2001) outlined a process for providing scientific advice for this developing fishery in Canada, based on estimates of the biomass in U.S. waters. This U.S. preseason estimate is refined using measured ocean temperatures and in-season trawl surveys.

Discussion

There are definite benefits to having a publicized policy on new and developing fisheries in Canada:

1. *Having a clear policy* lets all participants in the process know and understand the required procedures.

2. *Opportunity for industry to develop a business plan*. A clear statement of the requirements for developing new fisheries also means that industry can establish a business plan knowing there will be a process of review, experimentation, and exploration, and that they will be required to support many of the costs.

3. *Collection of organized and detailed biological information*. Some of the most important biological information for assessing stock performance is that collected prior to and during the start-up phase of a fishery. The framework described in this paper puts a strong emphasis on collecting biological information before the fishery be-

gins. As a result, there is better information now for many of these new fisheries than for several established fisheries.

4. *Capitalization and fishery development have been controlled*. The approach has clearly prevented excessive overcapitalization and explosive growth in new fisheries.
5. *Proponents have been directly involved*. The proponents have in large part been driving the process, and there has been very good interaction between proponents and government in many of the experimental activities.
6. *Decisions are based on open information*. Assessments at all stages are reviewed in a public forum through the scientific advice review process of DFO, and published reports are readily available.

There are also, of course, several drawbacks to this approach, which have become evident through the experiences described here, including the following.

1. The slow pace of development.

The requirement for DFO to process information and to review analyses means that progress has been slow in most cases. The need for detailed fieldwork and experiments also implies that progress is likely to be slow. The sea cucumber project has planned a developmental timeline of ten years. The case of neon flying squid suggests that this slow pace of development can at times be beneficial, by forcing industry to take a longer view of potential prices and product availability due to market and environmental variability. Additional management issues that must be considered when moving to a stage 4 fishery are the number of participants to allow into the fishery, who should participate (access), the fishing gear to be used, the process by which the industry funds the directs costs of private-sector management services (observers, etc.) and the incremental costs of management by DFO.

2. Costs.

The policy requirement that development of fisheries be cost-neutral to government is a significant hurdle. It may ultimately mean that only already well-developed fisheries or industries, i.e., those with money to invest in risky new ventures, will be able to support the development of new fisheries. Most of the assessments that were conducted in the early days of this approach (Table 3), and almost all of the stage 1 reports, were supported by a special fund (the Pacific Fisheries Adjustment and Restructuring Program) provided by DFO to help relieve the economic problems of the declining Pacific salmon fisheries. This fund has now expired, and requests for stage 1 assessments of potential new fisheries have slowed, we suspect in large part because of the lack of money for this first stage. The fishing industry has generally been reluctant to contribute money to fishery development other than for the direct costs of fishing. For industry, profit drives the entire process. The neon flying squid fishery was marginally profitable until the costs associated with locating squid increased at the same time as the price dropped from $10 per kg to under $5 per kg.

3. *The process can be stopped at any time.*

The process can be stopped at any time, by either DFO (as in the cases of California mussels and venus clam) or by the proponents (as with neon flying squid). The reality is that considerable investments can be made with no guaranteed return.

4. *Unrealistic expectations by both DFO and industry.*

Expectations have in many cases been unrealistic on the part of both DFO and industry as to what can be accomplished under this framework. Can industry really bear the costs of development when the proponent may be a single individual or a small community group? The development of the sea cucumber fishery demonstrates that opportunities to continue fishing, perhaps at a reduced level that is not expected to cause damage to the stocks, provide an income stream that can be invested in experimental activities related to developing the fishery. This is more difficult to do, however, for species about which almost nothing is known or for which markets are not yet developed. The ability for an industry to invest in and support the costs of this approach also requires a high level of organization among the proponents. It may be that some proponents have viewed the experimental and exploratory stages of the process merely as opportunities to go fishing, even if under somewhat restrictive regulations. The Tanner crab experience is typical of development opportunities in British Columbia. Industry proponents were attracted to the opportunity to diversify (all are active Dungeness crab fishers), and the subsidies provided by Federal and Provincial governments ensured that costs were covered in the earliest stages. As development proceeded, subsidies decreased, and higher proportions of the true costs of exploratory and experimental work were borne by industry. Concessions were made on levels of observer coverage and other forms of fishery monitoring as industry became increasingly alarmed by the costs they were required to bear. Industry participants have indicated that the economic benefits of the fishery have been eroded by price decreases and increased value of the Canadian dollar. Federal policy is moving toward participants paying more of the costs of assessment, monitoring, and management of fisheries. Many new and developing fisheries do not enjoy strong markets and high prices, and "fisheries overhead" may render many of them economically unviable. In many developing fisheries, participants were attracted to the funded portions of the policy, and the short-term economic windfall was, in fact, the subsidies provided for fishery development, not the fishery itself.

5. *Differences between new and existing fisheries.*

Developing new fisheries has been particularly slow and expensive. Placing existing fisheries into this framework has the advantages of existing biological information, developed markets, and an organized group of fishers. When existing fisheries are closed, however, there is the disadvantage of strong pressure to re-open the fishery to continue fishing.

6. *Prioritization of DFO resources.*

This approach has required considerable investment of time by DFO staff with, at least in the early days, little sense of which species should take priority. To date the prioritization model has had little impact on planning how DFO's limited resources are allocated among competing new and developing species/fisheries. Resources were put into species such as box crab and purple sea urchin when there were existing, but relatively undeveloped, fisheries that required scientific support and that ranked higher in the model. Major problems of prioritization were created when stage 1 assessments were presented for species that ranked low in the prioritization model, but were funded by industry. The prioritization model is expected to be a major tool for ranking future requests.

7. *Apparent lack of success.*

Of the twenty-four species/taxonomic groups that were evaluated using this approach from 1997 to 2003 (Table 3), only one, Pacific sardine, can be considered to be in the process of a moving to a full fishery (stage 4). This might be considered as a very poor "success rate," and rather discouraging to industry. Of the remaining 23 species, however, it can be argued that not all should become full fisheries. Species for which there are critical habitat concerns, such as California mussels, or for which harvesting would seriously disrupt critical habitat such as venus clams, should not proceed as full fisheries. Several other species are moving through the process and are in the experimentation and exploration stages.

Conclusions

The experience of Canada's Pacific Region with new and developing fisheries demonstrates that this framework has brought order to fisheries development, prevented unmanageable overcapitalization and "gold-rush" fisheries, and produced valuable biological information, often for stocks in their unfished condition. It is hoped this will result in well-managed and sustainable fisheries. The framework requires a long time horizon, which might be considered either a good or bad characteristic depending on a regulator's or proponent's point of view. Particular drawbacks of the framework include the high costs of development and no guaranteed outcome. We suggest that this approach should be tried in other management jurisdictions to identify which are "universal" attributes and which are specific to the Canadian situation.

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Fisheries Assessment and Management in Data-Limited Situations
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The Impact of Recruitment Projection Methods on Forecasts of Rebuilding Rates for Overfished Marine Resources

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Abstract

Under the U.S. Sustainable Fisheries Act, rebuilding plans have to be developed for fish stocks that are determined to be overfished, i.e., are found to be below the minimum stock size threshold. Rebuilding plans typically include analyses to determine the minimum time, T_{MIN}, to recover to a B_{MSY} proxy and the target level of fishing mortality, F_{REC}, that is consistent with recovery to this proxy within a pre-specified time frame and with an agreed probability. Key factors that determine T_{MIN} and F_{REC} are the methods used to forecast future recruitment and to estimate the B_{MSY} proxy, here based on 40% of the size of the unfished reproductive output of the population. Several approaches to modeling future recruitment are available. For example, Monte Carlo draws from previous recruitments, from previous recruits-per-reproductive output, and from a parametric probability distribution around an estimated stock-recruitment relationship have been used for groundfish resources off the U.S. West Coast. The results of rebuilding analyses are sensitive to this choice of modeling approach. The performance of alternative approaches to modeling future recruitment, in terms of providing unbiased and precise estimates of T_{MIN} and F_{REC}, are explored by means of simulation. All of the methods examined are imprecise and most are biased. The major factors influencing the ability to make good predictions of T_{MIN} and F_{REC}

are structural, viz., the actual value of the steepness of the stock-recruitment relationship and the extent of variability about that relationship. The length and contrast of the stock-recruitment data set has a larger impact on estimation performance than its precision.

Introduction

Stock assessments of harvested fish species have long been used to estimate historical levels of abundance and fishing mortality, and to make short-term projections of abundance and yield under various levels of fishing mortality (Brodziak et al. 1998). These projections can be the technical foundation for the calculation of annual fishery quotas that will achieve predefined goals for balancing attainment of optimum yield and prevention of depletion (NMFS 2001). In the 1996 Magnuson-Stevens Fishery Conservation and Management Act (often termed the Sustainable Fisheries Act, SFA), this balance between exploitation and conservation is explicit in National Standard 1: "Conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States industry." The SFA goes on to require that fishery management plans be implemented to end overfishing and to rebuild affected stocks of fish to target levels of abundance within a specified time frame. The need to satisfy these requirements leads to a substantial increase in the expectations for technical analysis (Restrepo et al. 1999); some assessments that were once considered as having adequate data are now data-poor. In this paper, and consistent with usage by the Pacific Fishery Management Council, "overfishing" means that the level of fishing mortality exceeds that associated with maximum sustainable yield (MSY), and "being in an overfished state" means that the current reproductive output is less than a minimum stock size threshold (MSST).

The technical requirements for a rebuilding analysis involve a sequence of calculations (Jacobson and Cadrin 2002). First is determination of the current abundance relative to the minimum stock size threshold. MSST has been set at half of B_{MSY} (the biomass corresponding to maximum sustainable yield) for many stocks. Stocks found to be below MSST are determined to be overfished (i.e., depleted) and in need of a rebuilding plan to restore the abundance to B_{MSY}. Second is to determine whether the stock has at least a 50% probability of rebuilding within 10 years. The period for rebuilding is to be as short as possible, taking into consideration the biology of the stock and the needs of the fishing community, and is not to exceed 10 years, unless the biology, environmental conditions, and a few other factors dictate otherwise. Third, for stocks that cannot be rebuilt in this time period, the technical guidance provided by NMFS is that the rebuilding period should not exceed T_{MAX}, the number of years corresponding to a probability of 50% of rebuilding with no fishing (referred to as T_{MIN}) plus one mean generation time. Mean generation time is defined as the average age of the population weighted by fecundity-at-age.

The Pacific Fishery Management Council (PFMC) has had to conduct rebuilding analyses for 9 of the 82 species of rockfish, flatfish, roundfish, and sharks in its Groundfish Fishery Management Plan. These analyses incorporate the three steps outlined above. For West Coast groundfish, there generally is insufficient information on recruitment to determine B_{MSY} precisely. For these species, therefore, the MSST has been set to 25% of the estimated unfished level of reproductive output, S_0, and the target level has been set to 40% of S_0; thus this target forms the basis for the calculations of this paper. (S_0 is the average reproductive output in an unfished state; reproductive output is variously defined as egg production or the biomass of spawning fish.)

The objective of this study is to determine the effect of several factors on the ability to estimate two key variables in rebuilding analyses. These quantities are T_{MIN}, the minimum possible time for an overfished population to have a 50% probability of recovery to the target level (40% of S_0), and F_{REC}, the "recovery fishing mortality." For the purposes of this study F_{REC} is defined as the fishing mortality such that there is a 0.6 probability that the population recovers to 0.4 S_0 by T_{MAX}. A 0.6 probability of recovery has been adopted for the calculations of this paper because this value is typical of those selected by the PFMC when developing rebuilding plans for overfished groundfish species. The range of probabilities selected by the PFMC is 0.55 (cowcod *Sebastes levis*) to > 90% (darkblotched rockfish *Sebastes crameri*).

Previous studies using a stochastic biomass dynamics model (Jacobson and Cadrin 2002) found substantial and skewed sensitivity of rebuilding calculations to various sources of input variability. Here we extend this previous investigation to the situation in which assessments are based on an age-structured population dynamics model, a situation more common for groundfish species off the U.S. West Coast.

T_{MIN} and F_{REC} are determined by projecting the population ahead under various levels of fishing mortality and with stochastic recruitment. In particular, T_{MIN} is calculated by projecting the population ahead several times (each time with a different random sequence of recruitments) under zero fishing mortality and recording the year in which the population first reaches 0.4 S_0. T_{MIN} is then defined to be the median of the distribution for this year. F_{REC} is computed by projecting the population ahead under various levels of fishing mortality between 0 and 3 per year, and using the bisection method to solve for the fishing mortality for which the probability is 0.6 that the reproductive output recovers to 0.4 S_0 before or during year T_{MAX}.

There are several sources of methodological uncertainty when calculating T_{MIN} and F_{REC} from the results of a stock assessment that add to the uncertainty caused by the variability in future recruitment. Probably the most important of these are: (a) uncertainty about the current state of the population, (b) uncertainty about the value of S_0, and (c) uncertainty regarding how to generate future recruitment when conducting forecasts. This study focuses on the latter two sources of uncertainty. There are several possible ways to estimate S_0 and generate future recruitments using the estimates of historical recruitment and reproductive output from a stock assessment. This study contrasts the performance of several of these methods using simulation.

Methods

The alternative methods for estimating S_0 and generating future recruitment are evaluated by means of Monte Carlo simulation. This involves developing a model (referred to as an operating model) that will represent the "true" dynamics of the resource, and using this model to compute the "true" values for T_{MIN} and F_{REC}. This model is then used to generate a number (250 for the purposes of this study) of data sets which will be used by each of the alternative methods for estimating S_0 and generating future recruitment to estimate T_{MIN} and F_{REC}. Each of the 250 Monte Carlo replicates differs in terms of historical recruitment so that the true values for S_0, T_{MIN}, and F_{REC} differ among the Monte Carlo replicates. Analyses (not shown here) indicate that 250 simulations are sufficient to detect qualitative differences among projection methods. A larger number of simulations would have increased the precision of the results, but also would have restricted the number of options that could have been investigated.

The following two sections outline the model of the true dynamics and the alternative approaches for determining S_0 and generating future recruitment.

Calculating the "true" recovery fishing mortality

The actual ("true") values for T_{MIN} and F_{REC} are determined from projections based on the following population dynamics model:

$$N_{y,a} = \begin{cases} R_y & \text{if } a = a_{min} \\ N_{y-1,a-1}\, e^{-(M+\tilde{S}_{a-1}F)} & \text{if } a_{min} < a < a_{max} \\ N_{y-1,a_{max}-1}\, e^{-(M+\tilde{S}_{a_{max}-1}F)} + N_{y-1,a_{max}}\, e^{-(M+\tilde{S}_{a_{max}}F)} & \text{if } a = a_{max} \end{cases} \qquad (1)$$

where $N_{y,a}$ is the number of animals of age a at the start of year y,

M is the instantaneous rate of natural mortality (assumed to be independent of age and equal to 0.1 per year for the analyses of this paper),

$\tilde{S}_a$ is the selectivity for animals of age a,

F is the fully selected (i.e.,$\tilde{S}_a \rightarrow 1$) fishing mortality,

R_y is the recruitment (both sexes) during year y,

a_{min} is the lowest age class considered in the model (here set to age 0), and

a_{max} is the oldest age class considered in the model (treated as a plus-group).

The recruitment during year y is related to the reproductive output at the start of year y according to either a Beverton-Holt or a Ricker stock-recruitment relationship, i.e.:

$$R_y = \begin{cases} \dfrac{4hR_0(S_y/S_0)}{1-h+(5h-1)(S_y/S_0)}e^{\varepsilon_y-\sigma_R^2/2} & \text{Beverton-Holt} \\ R_0(S_y/S_0)\exp[-\ell n(5h)(S_y/S_0-1)/0.8]e^{\varepsilon_y-\sigma_R^2/2} & \text{Ricker} \end{cases} \tag{2}$$

where R_0 is the average recruitment in absence of harvesting (computed by dividing S_0 by the reproductive output-per-recruit in the absence of exploitation),

h is the "steepness" of the stock-recruitment relationship (the fraction of R_0 to be expected when the reproductive output is reduced to $0.2S_0$),

ε_y is the deviation about the stock-recruitment relationship during year y, $\varepsilon_y \sim N(0;\sigma_R^2)$, and

σ_R is the extent of variability about the stock-recruitment relationship.

Selectivity is assumed to be governed by a logistic equation:

$$\tilde{S}_a = \tilde{S}'_a / \max_{a'} S'_{a'}; \qquad \tilde{S}'_a = \frac{1}{1+e^{-\delta(a-a_{50})}} \tag{3}$$

where a_{50}, δ are the parameters of the logistic equation (assumed to be 3 years and 2 per year respectively).

Each simulation trial of 250 replicates involves pre-specified values for weight-at-age (Fig. 1, left panel), and fecundity-at-age (taken to be equal to weight-at-age, except that animals 3 years and younger are as-

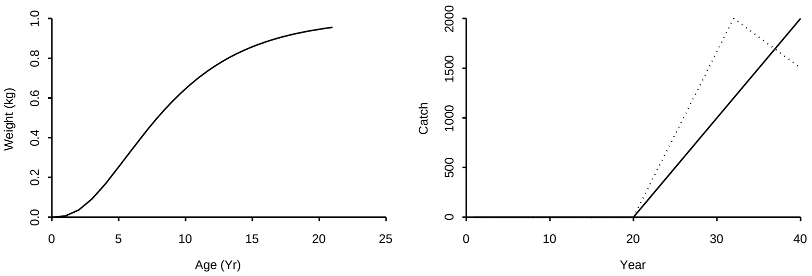

Figure 1. Weight-at-age (left panel) and the two catch series (right panel).

sumed to be immature), the time series of catches (Fig. 1, right panel), and the parameters that define the steepness and variability of the stock-recruitment relationship, the selectivity pattern, and the status of the resource when the analyses to determine T_{MIN} and F_{REC} are conducted (defined in terms of the ratio of the reproductive output after 40 years of catches to S_0, i.e., S_{41}/S_0). Table 1 lists the values examined for the parameters of the population dynamics model.

For each replicate, the values for the $\{\varepsilon_y$: y = 1,2,..., 40$\}$ are generated and the value of S_0 selected so that the ratio of the reproductive output at the start of year 41 relative to S_0 equals the pre-specified value (Fig. 2). Therefore, the value for S_0 differs among the 250 replicates that constitute a simulation trial even though the ratio S_{41}/S_0 is the same for all replicates. The projections start with a population at its pre-exploitation equilibrium level; the impact of 20 years of zero catches (see the right panel of Fig. 1) is that the age structure of the resource in the first year for which catches are available differs from an equilibrium age structure.

Estimating the minimum possible time to recovery and the recovery fishing mortality

The mathematical details of how T_{MIN} and F_{REC} are calculated are identical to those for how the "true" (i.e., operating model) values are calculated, except that the value for S_0 and the process for generating future recruitment are assumed to be based on analyses of information from a stock assessment. The population dynamics model on which the projections are based is identical to that used to define the true values for T_{MIN} and F_{REC}. Reproductive output-per-recruit in the absence of exploitation and mean generation time are assumed to be known exactly when estimating T_{MIN} and F_{REC} because fecundity-at-age and natural mortality are assumed to be known exactly.

Table 1. Values for the parameters of the operating model and the specifications related to data collection.

Parameter/specification	Symbol	Values
Stock-recruitment relationship		
Form	--	**Beverton-Holt**/Ricker
Steepness	h	0.3, **0.7**, 0.9
Extent of variability	σ_R	0.3, **0.6**, 1.0
Reproductive output in year 41	S_{41}/S_0	**0.2**, 0.1, 0.05
Catch series		**Increasing-decreasing**/ increasing only
Years for which data are available		**14-37**, 24-37

The values in bold type form the reference case for the simulations.

Estimating S_0

Two approaches to estimating S_0 (in addition to assuming it to be known exactly) are considered. The first approach involves multiplying the average recruitment for the first ten years for which estimates of recruitment and reproductive output are available (years 14-23 for the reference set of simulations) by the reproductive output-per-recruit in the absence of exploitation. The number of years of data used when estimating S_0 for this approach reflects the trade-off between increased precision (larger number of years) and avoiding biasing the estimate of S_0 because of the impact on recruitment of spawning biomass dropping over time due to the impact of fishing (fewer years). The second approach estimates S_0 by fitting a stock-recruitment relationship to the entire data set on recruitment and reproductive output.

Methods for projecting forward

The key aspect of projecting the population ahead is how the recruitment for year y, R_y, is generated based on the reproductive output for year y, S_y. The critical question is: how can the pattern of historical recruitment and reproductive output be used to generate future recruitment (both its expected value and its variability)? A variety of alternative approaches for achieving this are examined.

Sampling from the historical recruitments (non-parametric approach). The recruitment for year y is generated by selecting a range of years $[y_1, y_2]$, generating a U[0,1] random variate, X, and calculating $\tilde{y} = y_1 + (1 + y_2 - y_1)X$. The generated recruitment is that corresponding to the integer part of $\tilde{y}$.

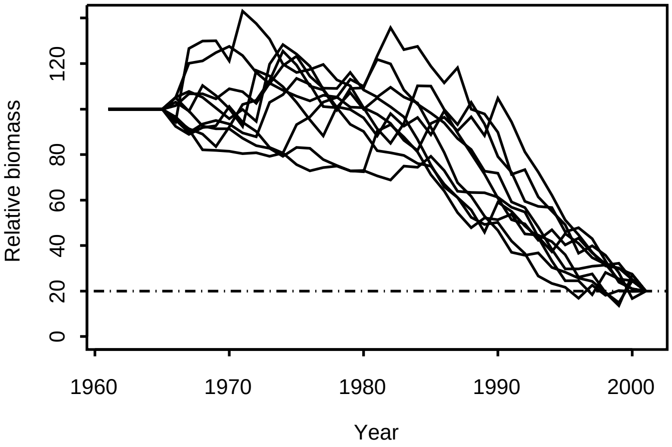

Figure 2. A subset of the time-trajectories of relative biomass for the reference case simulation trial.

Sampling from the historical recruitments (parametric approach). The recruitment for year y is generated by selecting a range of years $[y_1, y_2]$, sorting the recruitments for these years into ascending order, generating a $U[0,1]$ random variate, X, and calculating $\tilde{y} = y_1 + (y_2 - y_1)X$. The generated recruitment is computed by interpolating between the recruitments for the integer years which bound year $\tilde{y}$.

Sampling from the historical recruits-per-reproductive output ratios (non-parametric and parametric approaches). The recruitment for year y is set to the product of the reproductive output for year y and a generated recruits-per-reproductive output ratio. Two variants of this approach (parametric and non-parametric) exist. The recruits-per-reproductive output ratios are generated in the same way as when recruitment is generated by sampling from the historical recruitments.

Generating using a stock-recruitment relationship. The recruitment for year y is generated using the equation:

$$R_y = f(S_y)e^{\sigma_R \varepsilon_y - \sigma_R^2/2} \tag{4}$$

where $f(S)$ is recruitment as a function of reproductive output, either the Beverton-Holt, Ricker, or hockey-stick stock-recruitment relationship. The hockey-stick stock-recruitment relationship is R_y=$R_0 min[1,S_y/(\beta S_0)]$ where β is the fraction of S_0 at which recruitment begins to decline.

The values for the model parameters (i.e., S_0, h/β, and σ_R) are determined by minimizing an objective function. The objective function includes contributions from the fit of the stock-recruitment relationship to the data on recruitment and reproductive output, a penalty on the value for R_0, and a penalty (or "prior" distribution) on the value of the steepness parameter:

$$n\ell\text{n}\sigma_R + \tfrac{1}{2\sigma_R^2}\sum_{y=1}^{n}(\lambda_y + \sigma_R^2/2)^2 + \frac{(R_0 - \bar{R})^2}{2(\sigma_{\bar{R}}\bar{R})^2} + \frac{(h - \bar{h})^2}{2(\sigma_{\bar{h}})^2} \tag{5}$$

where n is the number of years for which information is available on recruitment and reproductive output,

λ_y is the residual for year y, i.e., $\lambda_y = \ell\text{n}[R_y/f(s_y)]$,

$\bar{R}$ is the average recruitment when reproductive output exceeds the median reproductive output,

$\sigma_{\bar{R}}$ is the coefficient of variation assumed for the penalty on the estimate of R_0 (assumed to be 0.2 for the analyses of this paper),

$\bar{h}$ is the mean of the "prior" distribution for steepness, and

$\sigma_{\bar{h}}$ is the standard deviation of the "prior" distribution for steepness.

A penalty is imposed on the average recruitment to avoid situations in which steepness is estimated to be its lowest possible value (0.2) and S_0 is consequently estimated to be essentially infinite. Equation 5 is the most general form of the objective function. The analyses discussed in this paper consider two special cases of equation 5: (a) there is no prior information on steepness (i.e. $\sigma_{\bar{h}} \rightarrow \infty$), and (b) the value for S_0 (and hence R_0) is pre-specified rather than being treated as an estimable parameter. The residuals are bias-corrected when fitting the stock-recruitment relationship; results (not shown here) indicate that the ability to adequately estimate T_{MIN} and F_{REC} deteriorates markedly if the residuals are not bias-corrected.

In summary, use of a stock-recruitment relationship asserts a convex, curvilinear relationship between recruitment and reproductive output; use of resampling from recruitments (the "Con R" approach) asserts that recruitment and reproductive output are independent (corresponding to a value for h of 1.0); and use of resampling from recruits-per-reproductive output (the "Con R/S approach") asserts a linear relationship through the origin with a slope (steepness) determined by the central tendency of the data. In principle, sampling from the historical recruits-per-reproductive output ratios and using a hockey-stick stock-recruitment relationship to generate future recruitment should be equivalent if the recruitment observations correspond to a limited range for reproductive output. We expect the stock-recruitment approach to be more satisfactory for the types of projections needed to calculate T_{MIN} and F_{REC} because the "Con R" approach may generate excessively high recruits pre-reproductive output ratios at low stock size and the "Con R/S" approach may generate excessively high recruitments as the stock rebuilds to high levels of reproductive output.

Results

Detailed results for one simulation trial and one estimator

Figure 3 shows the detailed results from the simulations for one approach to computing S_0 and generating future recruitment and one variant of the operating model. The operating model variant is that in which the value for the steepness parameter, h, is set to 0.7. The value of S_0 is calculated by multiplying the reproductive output in the absence of exploitation by the average recruitment over the first 10 years for which estimates of recruitment are available (years 14-23 in this case) while future recruitment is generated using a Beverton-Holt stock-recruitment relationship. The values for the parameters of this relationship are based on minimizing equation 5 where no prior information is assumed about steepness (i.e., $\sigma_{\bar{h}} = 100$), and in which R_0 is set equal to the average recruitment over years 14-23. The decision to estimate R_0 as the average recruitment over years 14-23 rather than from the fit of the stock-recruitment relationship was made because it eases some of the subsequent comparisons. The Beverton-Holt stock-recruitment relationship is fitted to the data for years 14-37; the data for years 38-40 are ignored because it is well known that the recruitment estimates for the last few years of a stock assessment are very imprecise (e.g., Butterworth et al. 1990).

In order to distinguish the impact of structural factors and that of the short time series for which estimates of recruitment and reproductive output are available, the calculations on which Fig. 3 and the bulk of the remaining calculations are based assume that there is no error when recruitment and reproductive output are estimated from the stock assessment. The impact of this source of uncertainty is explored in a later section.

The top row of Fig. 3 shows histograms of the 250 "true" values for S_0, the 250 estimates of S_0, and the 250 differences between the "true" and estimated values (expressed as percentages of the "true" value for S_0). The top row of Fig. 3 also plots the estimated value for S_0 against its "true" value. The second and third rows of Fig. 3 show these four plots for T_{MIN} and F_{REC}. The final row of plots in Fig. 3 shows histograms of the

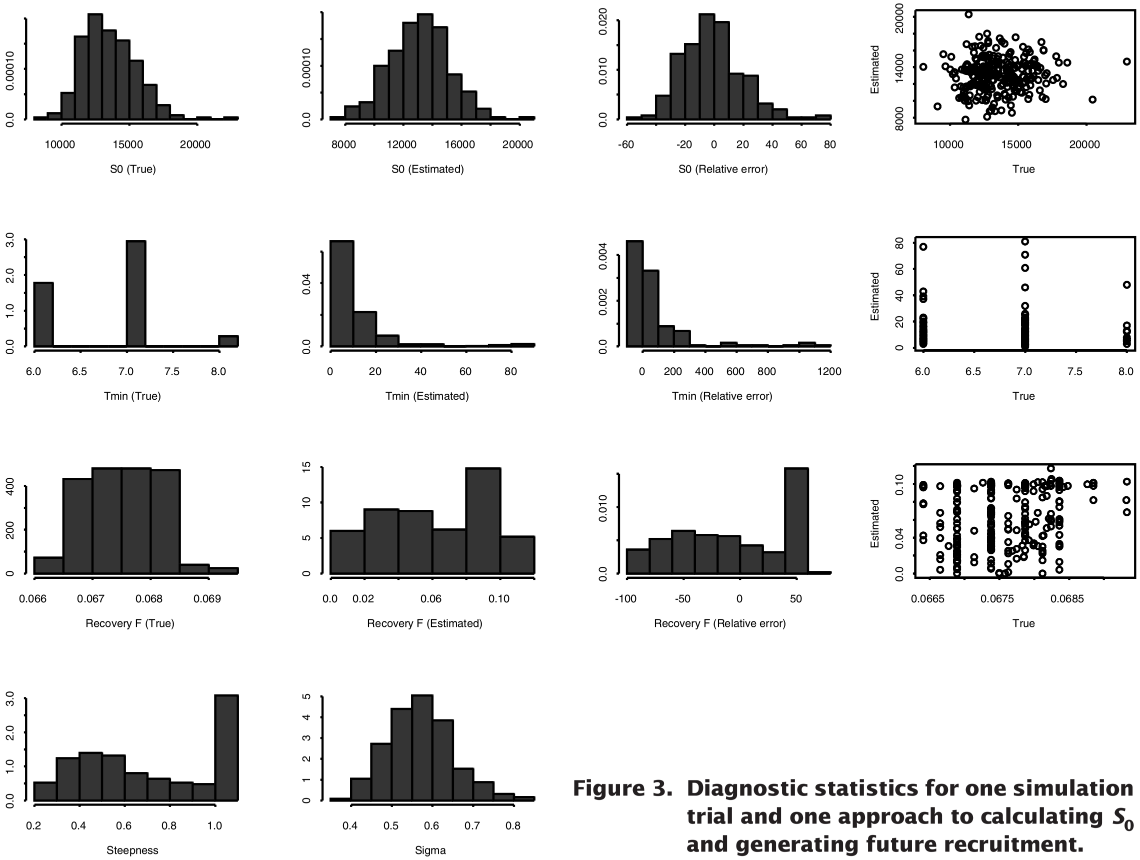

Figure 3. Diagnostic statistics for one simulation trial and one approach to calculating S_0 and generating future recruitment.

estimates of steepness and σ_R (the "true" values for these parameters are 0.7 and 0.6 respectively).

The quantity that is estimated "best" (as reflected by the range of the percentage relative errors in Fig. 3) is S_0. However, the range of percentage relative errors for S_0 still exceeds ±50% (Fig. 3). The estimates of T_{MIN} and F_{REC} can be extremely poor. For example, the range of relative errors for F_{REC} range from –100% to +50% although, on average, the estimate of F_{REC} is fairly close to the true value. One of the main reasons for the imprecise estimates of T_{MIN} and F_{REC} is that the recent recruitment data are not very informative about the value of the steepness parameter, hence about the central tendency of future recruitment (Fig. 3, lower left panel). The distribution for the estimates of steepness is bimodal with one mode at roughly 0.5 and another close to 1.

Relatively small differences in the estimate of steepness can have a profound impact on the ability to estimate T_{MIN} and F_{REC} adequately. Figure 4 plots the percentage relative errors for F_{REC} and T_{MIN} for the 250 replicates against the estimate of steepness and the percentage relative errors for F_{REC} against the percentage relative errors for T_{MIN} and S_0. It is very clear from Fig. 4 that the errors when estimating steepness are the primary factor determining the errors for both T_{MIN} and F_{REC}. There is a relationship between the errors when estimating S_0 and those when estimating F_{REC} (Fig. 4, lower left panel). This occurs because lower values for S_0 correspond to higher values for steepness and vice versa.

It should be noted, however, that it is not inability to estimate the steepness of the stock-recruitment relationship that is per se the cause of the difficulties associated with estimating T_{MIN} and F_{REC}. Rather it is that the small number of highly fluctuating recent recruitments are not very informative about the central tendency of future recruitments, whether this central tendency is parameterized using the steepness of a stock-recruitment relationship or some other method such as resampling from the observed recruits-per-reproductive output ratios. In fact, it will be shown later that the use of a stock-recruitment relationship is actually among the most robust methods for making forecasts of the expectation and variability of future recruitment.

The impact of errors when estimating T_{MIN} on the ability to estimate F_{REC} are explored in Fig. 5 which plots the same information as Fig. 4 except that the value of T_{MIN} (and hence T_{MAX}) is assumed to be known exactly. The results in Fig. 5 are essentially identical to those in Fig. 4, confirming that uncertainty about determining the central tendency of future recruitment dominates the uncertainty associated with making forecasts.

Somewhat surprisingly given the poor ability to estimate the steepness of the stock-recruitment relationship, the estimates for the extent of variability about the stock-recruitment relationship are close to unbiased.

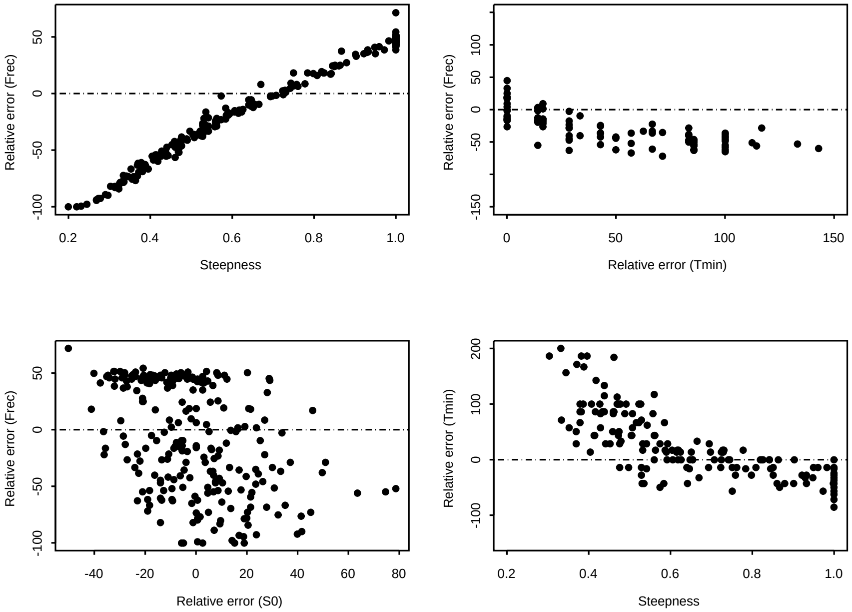

Figure 4. **Relative errors (expressed as percentages) for F_{REC} and T_{MIN} against the estimate of steepness and against the percentage relative errors for T_{MIN} and S_0. The results in this figure pertain to the variant of the reference operating model with $h = 0.7$. (Relative errors for T_{MIN} larger than 200% are excluded from the lower right panel for ease of presentation.)**

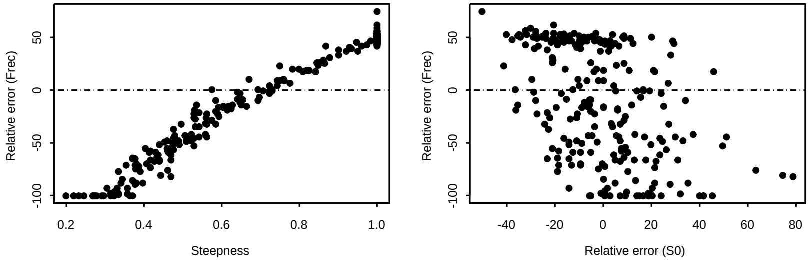

Figure 5. **Relative errors (expressed as percentages) for F_{REC} against the estimate of steepness and the percentage relative errors for S_0. The results in this figure pertain to the variant of the reference operating model with $h = 0.7$ and with T_{MIN} set equal to its "true" value.**

Comparison of various estimators for the reference simulation trials

The results of a set of simulations can be summarized fairly succinctly by box and whisker plots of the relative error distributions for S_0, T_{MIN}, and F_{REC} (e.g., Fig. 6). Figure 6 contrasts the performances of six different approaches to generating future recruitment in terms of the ability to estimate T_{MIN} and F_{REC} for the reference case operating model (steepness values of 0.3, 0.7, and 0.9). Results are not shown in Fig. 6 for S_0, because all of the estimators considered specify S_0 by multiplying the reproductive output in the absence of exploitation by the average recruitment over the first 10 years for which estimates of recruitment are available, so all six perform equally successfully. The six approaches to generating future recruitment considered in Fig. 6 are: fitting a Beverton-Holt stock-recruitment relationship (abbreviation "B-H"), fitting a Ricker stock-recruitment (abbreviation "Ricker"), fitting a hockey-stick stock-recruitment relationship (abbreviation "H-S"), generating recruitment by sampling from historical recruitments ("Con R"), generating recruitment by sampling from historical recruits-per-reproductive output ratios ("Con R/S"), and selecting between the previous two approaches depending on which of recruitment or recruits-per-reproductive output are more stable ("Con R or Con R/S").

For the approaches based on sampling from historical data rather than fitting a stock-recruitment relationship, the recruitments and the recruits-per-reproductive output ratios are computed from the estimates of recruitment and reproductive output for the entire 24 year period. This is perhaps less preferable to basing them on the data for the years when the reproductive output was estimated to be less than the target level of 0.4 S_0 but was chosen because the number of data points for the more preferable approach would be very small for some of the simulations. In any case, results not shown here indicate that the qualitative results are not particularly sensitive to whether the last three sampling-based approaches are applied parametrically or non-parameterically and whether the entire sequence of recruitments or just that corresponding to reproductive outputs less than 0.4 S_0 are used as the basis for generating future recruitment.

The ability to estimate T_{MIN} and F_{REC} depends, not surprisingly, on the "true" value for steepness. For low steepness ($h = 0.3$; upper panels of Fig. 6), T_{MIN} tends to be overestimated and F_{REC} tends to be underestimated (i.e., overprotection errors are made) by the Ricker, hockey-stick, "Con R/S," and "Con R or R/S" approaches while the opposite pattern (i.e., overprotection errors) is evident for the "Con R" approach. The same pattern is evident for $h = 0.7$ and $h = 0.9$ (middle and lower panels of Fig. 6), except that the extent of over- and under-protection gets less. Of the three stock-recruitment relationship-based approaches, that based on

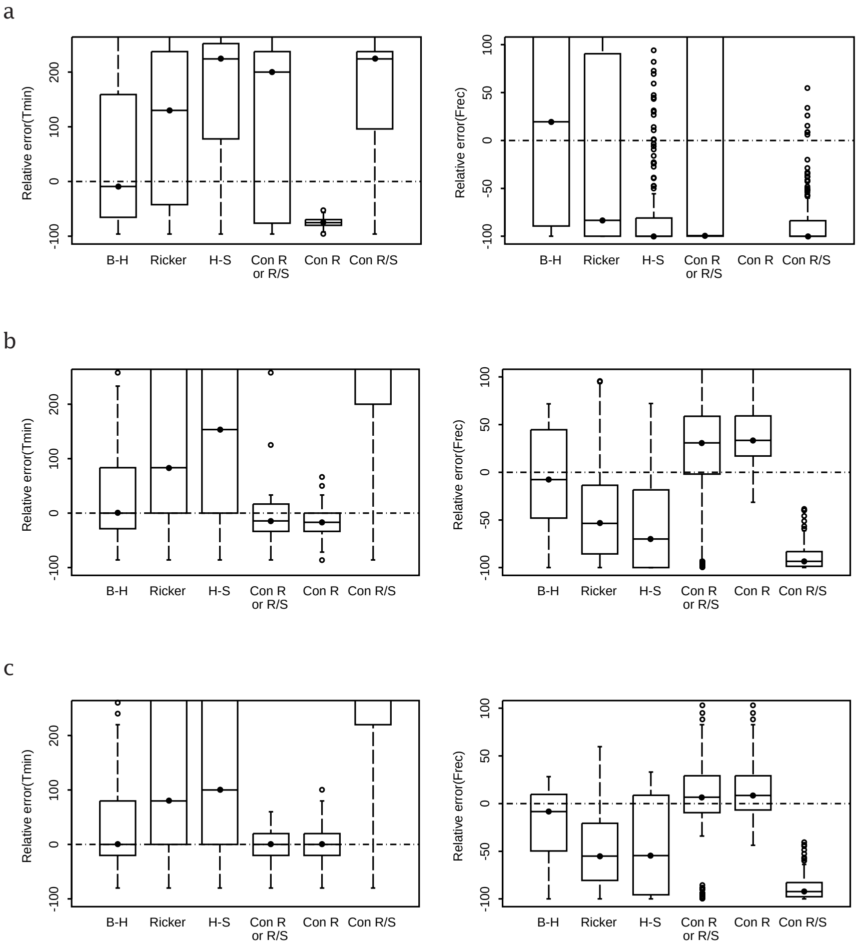

Figure 6. Relative error distributions for T_{MIN} and F_{REC} corresponding to six alternative approaches to forecasting future recruitment. Results are shown for $h = 0.3$, 0.7, and 0.9 (upper, middle, and lower panels, respectively).

the Beverton-Holt model is closest to being unbiased. The comparison between the Ricker and Beverton-Holt approaches is perhaps somewhat unfair in this case because the "true" stock-recruitment relationship has the Beverton-Holt form.

The results for the "Con R" and "Con R/S" approaches differ substantially, with the "Con R" approach always producing more optimistic predictions than the "Con R/S" approach. The approach that chooses between the "Con R" and "Con R/S" approaches is very variable (wide intervals in Fig. 6) but outperforms both the "Con R" and "Con R/S" approaches across the range of values for steepness considered. Current practice when conducting rebuilding analyses for overfished groundfish species that do not involve using a stock-recruitment relationship to predict future recruitment is to use the "Con R or R/S" approach (e.g., Hamel et al. 2003, Cope et al. 2004).

Figure 7 plots the recruitment and reproductive output data for one of the simulated data sets, the true underlying stock-recruitment relationship, and the stock-recruitment relationships implied by four of the approaches for generating future recruitment. The inability to estimate steepness precisely is readily apparent from Fig. 7.

Performance of variants of the Beverton-Holt estimator

The Beverton-Holt estimator performed among the best of the six estimators investigated in Fig. 6, although none of these estimated T_{MIN} and F_{REC} particularly precisely. The remaining calculations of this paper therefore focus on this estimator. Figs. 8-10 explore the performance of variants of this estimator in terms of the ability to estimate S_0, T_{MIN}, and F_{REC}.

Two of the estimators in Figs. 8-10 ("True S0" and "S0 Est") examine the sensitivity of the results to how S_0 is estimated compared to the base case ("B-H") in which S_0 is computed from the mean of ten early recruitments. Knowing the true value of S_0 exactly ("True S0" in Figs. 8-10) does not improve the ability to estimate T_{MIN} and F_{REC} substantially although the reduction in variability when $h = 0.7$ and $h = 0.9$ (Figs. 9 and 10) is perhaps noteworthy. The results are less variable and closer to being unbiased when S_0 is estimated along with the other parameters of the stock-recruitment relationship. This suggests that estimating S_0 from the entire data set on recruitment and reproductive output is preferable to fixing S_0 based on the average recruitment over the early years in the assessment.

Having a tight prior on h ($\bar{h}$ equal to the true value; $\sigma_{\bar{h}} = 0.1$; "h prior" in Figs. 8-10) improves the ability to estimate T_{MIN} and F_{REC} substantially. This is, however, hardly surprising given the results in Fig. 4. The performance of the "h prior" estimator is poorer for $h = 0.3$ (Fig. 8) than for $h = 0.7$ and $h = 0.9$ (Figs. 9 and 10). This is because the coefficient of variation for the steepness prior ($\sigma_{\bar{h}}/\bar{h}$) is greatest when $\bar{h} = 0.3$; the prior is therefore given less emphasis when fitting the stock-recruitment relationship when $h = 0.3$.

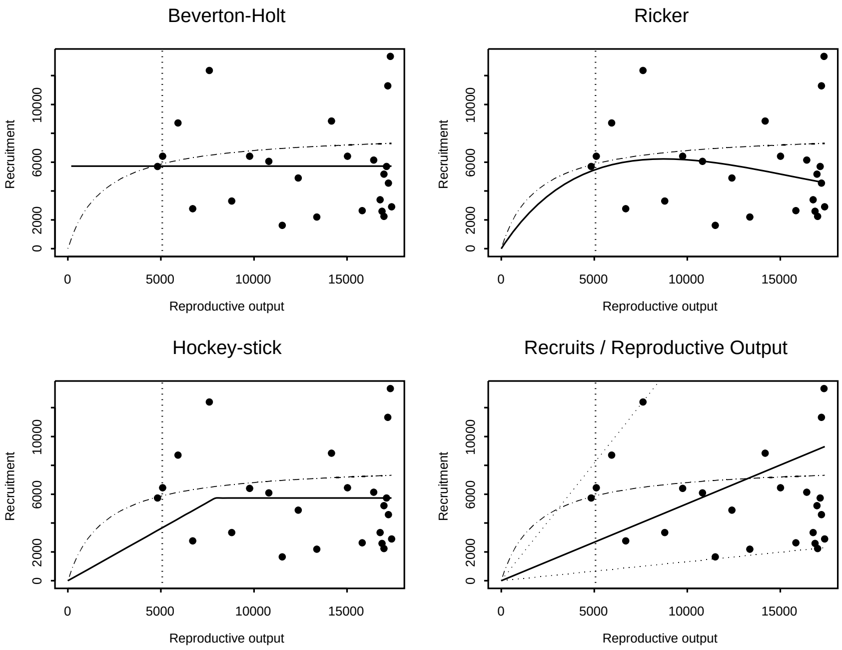

Figure 7. Stock-recruitment functions underlying four of the methods for generating future recruitment for one of the simulated data sets. The true stock-recruitment relationship is of the Beverton-Holt form and is shown by the dashed line. The vertical line indicates 40% of the estimate of S_0.

Decreasing the number of years for which estimates of reproductive output and recruitment data are available from 24 to 14 (“Less D” in Figs. 8-10) leads to a marked deterioration in the ability to estimate S_0 when $h = 0.3$ (Fig. 8). This occurs because the average recruitment on which S_0 is based comes from a period when the stock is already reduced to well below its unfished level.

Sensitivity to model structure

The results in Figs. 3-10 are based on variants of the reference operating model that examine the impact of different choices for steepness. Figure 11 shows results for the “B-H” estimator in Figs. 8-10 for a range of other variants of the operating model. Figure 11 is restricted to F_{REC} and T_{MIN} only because the estimates of S_0 are generally insensitive to changes to the specifications of the operating model.

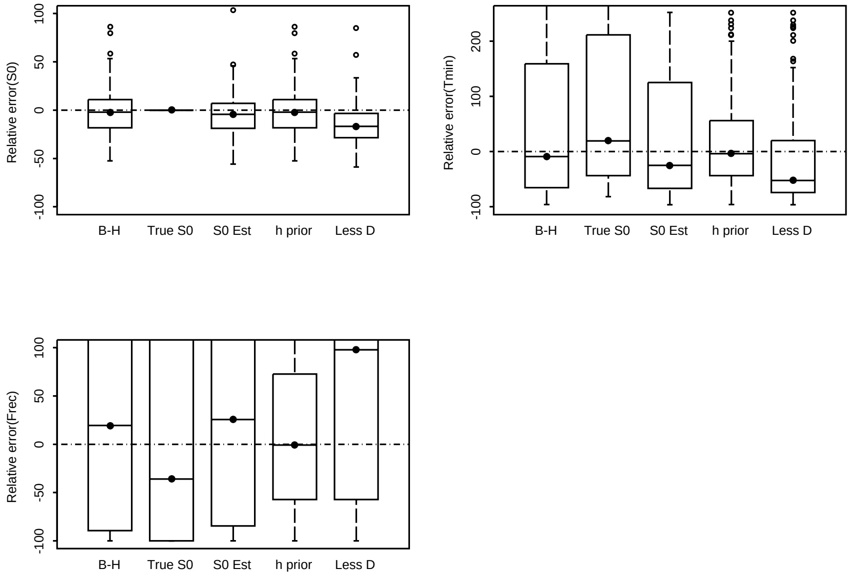

Figure 8. Sensitivity of the results for the reference case trial with *h* = 0.3 to changing the specifications for the Beverton-Holt–based method of forecasting.

The ability to correctly forecast recruitment is very sensitive to the true extent of variability in recruitment about the underlying deterministic stock-recruitment relationship. For example, the ability to estimate both T_{MIN} and F_{REC} improves substantially when σ_R is 0.3 rather than 0.6 ("SigR = 0.3" in Fig. 11) while performance deteriorates if σ_R is larger than the reference value ("SigR = 1" in Fig. 11). This sensitivity is hardly surprising given that increased variability in recruitment serves to further mask the true underlying stock-recruitment relationship. The estimates of T_{MIN} become more variable when there is inter-annual correlation in recruitment but this is ignored when the stock-recruitment relationship is fitted to the data ("Prow = 0.707" in Fig. 11).

The results are not notably sensitive to the extent of depletion when the rebuilding analyses are conducted (5, 10, and 20% of S_0; compare the results for "Dep = 0.1" and "Dep = 0.05" with those for "Base" in Fig. 11) and to the nature of the catch series. The estimate of T_{MIN} is slightly negatively biased but more precise while F_{REC} is also more precise but positively biased when the true stock-recruitment relationship is Ricker rather than Beverton-Holt.

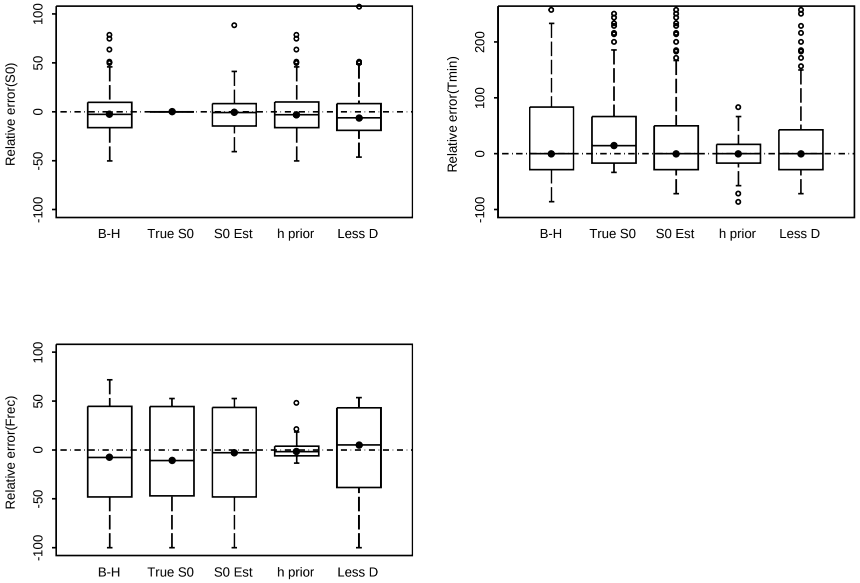

Figure 9. See Fig. 8 caption, except that the true value of *h* is 0.7.

Overall, however, the factors that impact the ability to estimate T_{MIN} and F_{REC} the most are the "true" values for the steepness of the stock-recruitment relationship and the extent of variability about the stock-recruitment relationship.

Sensitivity to error when estimating stock and recruitment

All of the analyses on which Figs. 3-11 are based assume that the information on recruitment and reproductive output is known without error. This assumption is clearly incorrect. Figure 12 therefore examines the sensitivity of the results for the "B-H" estimator for the reference operating model ($h = 0.7$) to different levels of error when estimating recruitment (CVs of 0.1 – 0.4). The data on recruitment and reproductive output used when conducting the forecasts is generated by adding lognormal error to the true recruitments, projecting each perceived recruitment forward given the true fishery catch and natural mortality rate, and computing the resultant values for annual reproductive output. The analyses reported in Fig. 12 therefore examine the impact of variability in the estimates of recruitment and not, for example, bias in these estimates. The impact of biased estimates on the estimation of T_{MIN} and F_{REC} is, however, qualitatively straightforward to determine.

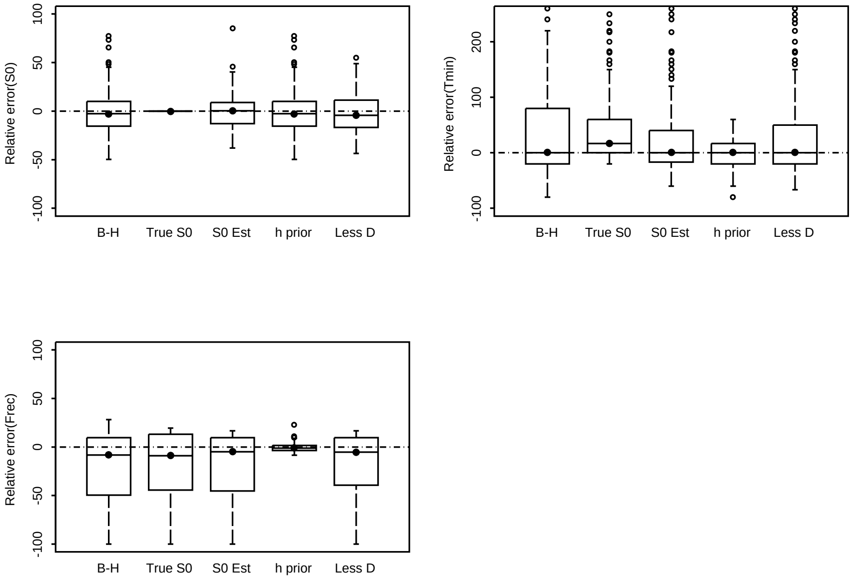

Figure 10. See Fig. 8 caption, except that the true value of *h* is 0.9.

Somewhat surprisingly, although the estimates of T_{MIN} and F_{REC} become slightly more uncertain as the coefficient of variation of the estimates of recruitment is increased, the effect is relatively small. This suggests that the major factor determining the ability to predict future recruitment is not necessarily the precision of the recruitment and reproductive output data (within reason) but rather the contrast in these data.

Discussion

The results of this paper indicate strongly that the estimates of the time it will take for overfished species to recover to target levels are quite imprecise, irrespective of the method used to specify S_0 and for forecasting of future recruitment. The difficulties associated with making these statements are associated primarily with the difficulty of estimating the central tendency of future recruitment given a short (and noisy) time series of observed recruitments and less with errors when estimating recruitment and reproductive output using assessment models.

Having accurate prior information on steepness can improve the ability to make precise forecasts markedly (see, for example, Figs. 8-10). In principle, the use of meta-analysis methods (e.g., Myers et al. 1999, 2002;

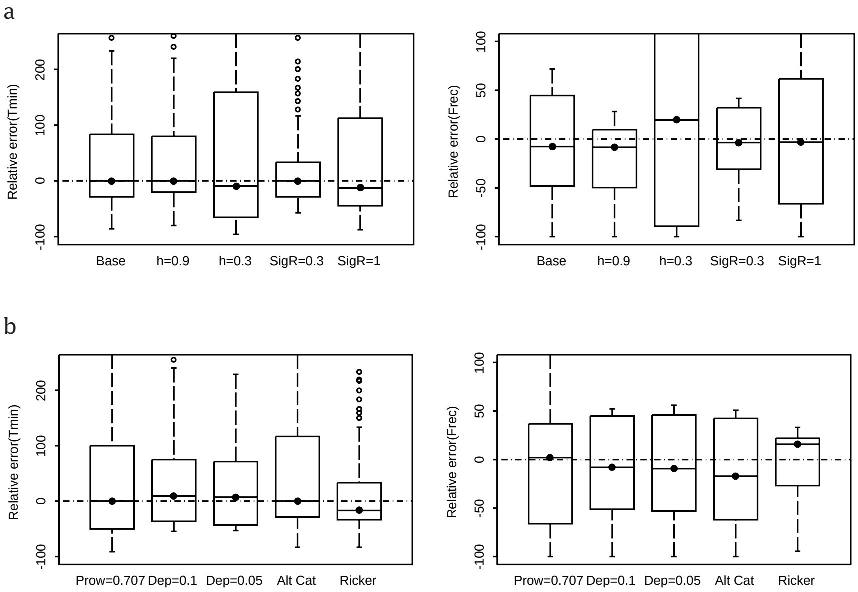

Figure 11. Sensitivity of the performance of the Beverton-Holt–based method of forecasting to changes to the specifications of the operating model.

Dorn 2002) to develop prior distributions for steepness and their subsequent use when fitting a stock-recruitment relationship can constrain the value of steepness and hence likely improve forecasting ability. The 1999 assessment of the Pacific ocean perch resource off the West Coast of the U.S. (Ianelli et al. 2000) included an informative prior distribution on steepness. Questions, of course, remain about the development and use of priors for key stock assessment parameters owing to, for example, the issue of the representativeness of the species on which such priors are based.

Although it is clearly not possible to reduce the extent of variability about the stock-recruitment relationship, attempting to account for some of the sources of this variability when fitting the stock-recruitment relationship (e.g., Maunder and Watters 2003) might lead to more accurate estimates of steepness. Inclusion of environmental factors in the analysis of the stock-recruitment data can explain some of the observed variability in recruitment and provide a mechanism for short-term forecasting of recruitment, to the extent that the environmental factors can be forecast themselves. Unfortunately, the ability to separate the effect of

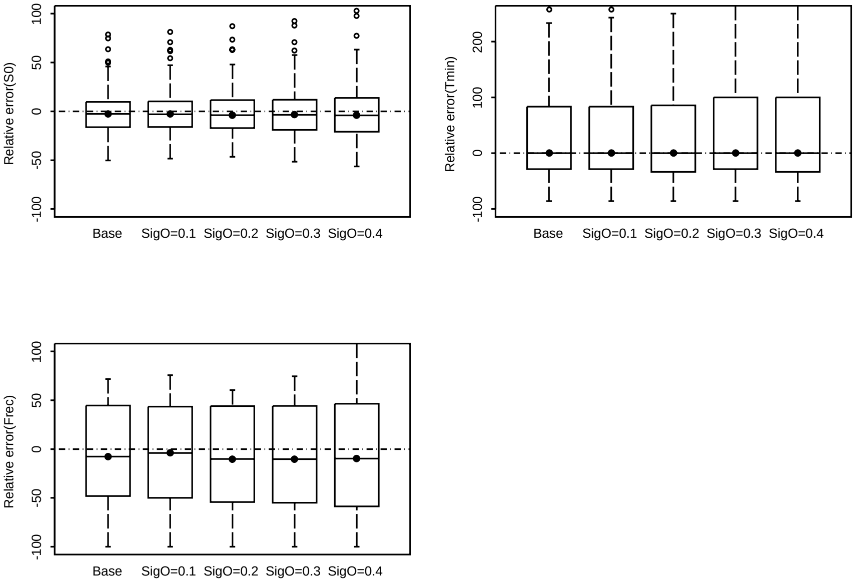

Figure 12. Sensitivity of the performance of the Beverton-Holt–based method of forecasting to the extent of error when estimating recruitment.

autocorrelated environmental factors from that of the shape of the stock-recruitment still depends upon having a sufficiently long time-series with adequate contrast in the environmental and stock effects.

None of the methods of forecasting considered in this paper clearly outperform any of the others. Some guidance regarding forecasting methods is, however, possible. First, given the sensitivity of the results to the choice of forecasting method (and the lack of a clearly superior forecasting method) it seems prudent to conduct analyses based on fitting a stock-recruitment relationship as well as using more empirical approaches (such as sampling recruitment randomly from the historical period). The results also suggest that, when using such empirical methods, applying a rule such as choosing between resampling historical recruits or recruits-per-reproductive output ratios, depending on whichever exhibits the least trend, is warranted.

In principle, the approach to calculating S_0 and generating future recruitment should be self-consistent, i.e., the long-term expected value of future stock size when estimated by generating future recruitments and no fishing mortality should equal S_0. This is achieved automatically when the forecasts are based on a stock-recruitment relationship. However, inconsistencies can arise when S_0 is based on the average recruitment over a pre-specified early period and future recruitment is based on resampling recruits or recruits-per-reproductive output ratios for a period of more recent years that have lower levels of recruitment.

The factors considered in the results section of this paper were restricted due to the need for brevity of presentation. Results (not shown here) suggest that factors such as changes over time in the fishery selectivity pattern and correlation in the residuals about the deterministic stock-recruitment relationship have a lesser impact than the factors emphasized in the results presented. Other methods for estimating S_0 and generating future recruitment warrant investigation. For example, the software package AgePro (Brodziak et al. 1998, Brodziak 2003) includes options to generate future recruitment using a Markov matrix approach. Another issue is that the scenarios examined in these simulations approximate those for West Coast groundfish in which recruitment estimates are available from a time period close to the unfished state, but few recruitment observations are available from recent, low biomass conditions. This is a marked contrast to the situation faced for New England groundfish in which there has been a long period of exploitation, little current information about unfished conditions, and much accumulated information about stock productivity under conditions of full exploitation (Brodziak et al. 2001).

Overall, the results of this paper indicate the need to highlight the extent of uncertainty associated with making forecasts based on short time-series of recruitment and reproductive output data. This uncertainty is not likely to be reduced in the short- to medium-term, although periodic assessment updates allow extension of the time series of stock-recruitment observations and refinement of the rebuilding calculations. Clearly, however, appropriately precautionary management decisions are nevertheless warranted if undesirable biological consequences are to be avoided.

Acknowledgments

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A Review of Natural Mortality Estimation for Crab Stocks: Data-Limited for Every Stock?

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Abstract

Among the four factors influencing population dynamics, recruitment, growth, instantaneous natural mortality (*M*), and harvest, *M* is the least understood. *M* estimates are not only confounded with the other three factors, but *M* is also difficult to observe directly. For hard-to-age stocks like crabs, estimating *M* is even more challenging without age information. Estimating *M* is a first step to develop an optimal harvest strategy for a stock. For stocks in data-limited situations, harvest rates are usually set based on *M* values. In this study, I reviewed *M* estimation methods and compared estimates of *M* values for crab stocks worldwide. Advantages and disadvantages of each approach were contrasted. The most significant feature of *M* estimates for crab stocks is high variation. Estimated *M* also appears to be higher for crabs than for fish with the same maximum age. Even for a crab stock with very rich data, estimates of *M* still depend on methods and fluctuate greatly over body size and time. The data-rich stock in this study, Bristol Bay red king crab (*Paralithodes camtschaticus*), appears still data-limited in terms of estimating *M* due to changes in *M* over time and confounding among *M* and other population model parameters.

Introduction

Instantaneous natural mortality (*M*) is one of the most important parameters for a large majority of stock assessment models but is not very well understood. *M* is difficult to observe directly, and its estimates are often confounded with recruitment, harvest, growth, and catchability in a model. Due to the difficulty of estimating *M*, it is common for many stock assessment studies to assume a value for *M* or guess a maximum age for estimating *M* and to examine the sensitivity of the assessment results to change in *M*.

Obtaining reliable *M* estimates is not only important for stock assessments but is also critical for developing optimal harvest strategies. Since *M* relates to growth and other life-history parameters (e.g., Alverson and Carney 1975, Pauly 1980, Hoenig 1983), *M* may also be closely related to optimal harvest rates. For data-rich stocks, *M* is needed in computer simulations or spawning biomass per recruit analysis to estimate optimal harvest rates. For data-limited stocks, *M* can be used as a proxy for fishing mortality. For example, the technical guidance of the U.S. National Marine Fisheries Service on the use of precautionary approaches to implementing National Standard 1 of the Magnuson-Stevens Fishery Conservation and Management Act suggests using $F_{msy} = M$ or $F_{msy} = 0.8\ M$ for data-limited stocks (Restrepo et al. 1998). The North Pacific Fishery Management Council assumes $F_{msy} = M$ to determine maximum sustained yield for the eastern Bering Sea crab fisheries (NPFMC 1999).

Estimating *M* is a difficult task for fish stocks and a great challenge for crab stocks without much age information. In this study, I reviewed *M* estimates for animal stocks and compared estimates of *M* reported in the literature for 14 crab stocks (*Callinectes sapidus, Cancer magister, Cancer pagurus, Cancer polyodon, Chionoecetes bairdi, Chionoecetes opilio, Lithodes aequispinus, Menippe mercenaria, Paralithodes camtschaticus,* and *Paralithodes platypus*). The relationships between *M* estimates and maximum ages for crabs, fish, mollusks, and cetaceans were also compared. The challenges for estimating *M* for crab stocks are discussed, with the example of Bristol Bay red king crab used to illustrate how a data-rich crab stock becomes a data-limited stock in terms of estimating *M*.

Approaches for estimating *M*

A variety of approaches have been used to estimate *M* for fish stocks. These approaches range from simple life-history analysis to complex mark-recapture experiments and population modeling. Quinn and Deriso (1999), Shepherd and Breen (1992), Vetter (1988), and Seber (1982, 1986) provided detailed overviews of common approaches for estimating *M*. Five common approaches are briefly outlined here.

Life-history analysis

Natural mortality is commonly related to other life-history parameters. For example, it is intuitive that a species with a long life span usually has a low *M*. Through a theoretical analysis or a meta-analysis across a variety of species and environments, relationships between *M* and other life-history parameters can be developed and used to estimate *M*. Hoenig (1983)

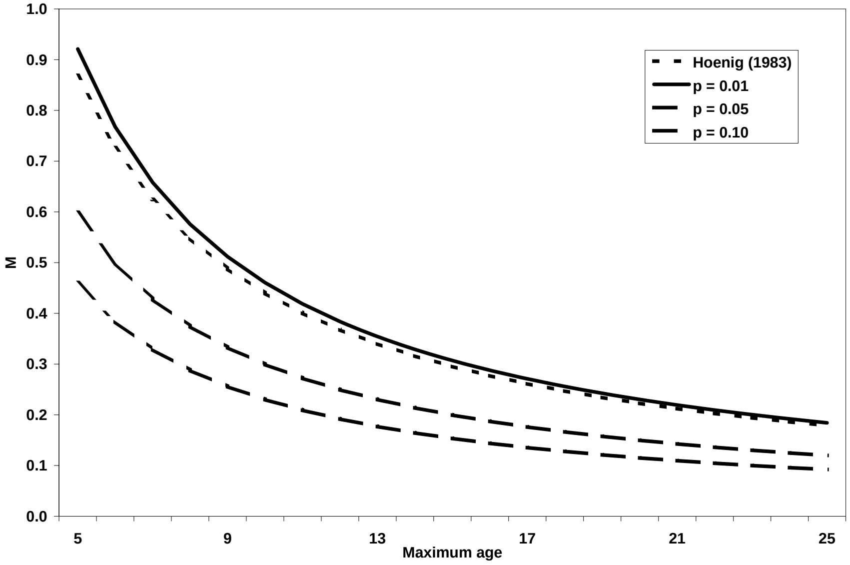

Figure 1. Relationships between instantaneous natural mortality and maximum age as a function of *p* (the proportion of animals that reach the maximum age).

empirically developed a relationship inversely relating M to longevity for 134 stocks of mollusks, fish, and cetaceans. The predictive equation is

$$M = \exp[1.44 - 0.982 \ln(t_{\max})] \tag{1}$$

where $t_{\max}$ is maximum age. Alternatively, from the exponential law of population decline for an unfished population, M is related to longevity as (Quinn and Deriso 1999),

$$M = -\ln(p) / t_{\max} \tag{2}$$

where p is the proportion of animals that reach the maximum age and is usually assumed to be 0.01 (Shepherd and Breen 1992, Clarke et al. 2003). Comparison of M estimates from equations (1) and (2) over a range of maximum ages is illustrated in Fig. 1. Equation (1) is almost identical with equation (2) with $p = 0.01$.

Other empirical equations have also been developed for fish stocks. Alverson and Carney (1975) related M to the growth coefficient of the von

Bertalanffy growth equation and the age at which an unfished cohort obtains its maximum biomass. Pauly (1980) developed a predictive relationship of *M* from maximum length (or weight), the growth coefficient of the von Bertalanffy growth equation and the mean water temperature based on data from 175 stocks covering 84 species. With data from 10 fish species, Gunderson (1980) obtained a relationship between *M* and a gonadal index (the ratio of gonad weight to total weight). The relationship was updated with data from 28 fish stocks (Gunderson 1997). Jensen (1997) derived two relationships relating *M* to age at maturity and the growth coefficient of the von Bertalanffy growth equation. Several other relationships among life-history parameters were reviewed by Vetter (1988).

Even though life-history analysis is a simple and inexpensive way to estimate *M*, there are several limitations associated with it. First, *M* estimates based on life-history analysis are often very imprecise, and the prediction errors are substantial (Pascual and Iribarne 1993). Second, the source data used to establish empirical relationships are derived from a variety of methods and stocks. If some of the estimates are biased, the relationships are also biased. Third, the relationships are good only within the life-history parameter ranges used to establish them. Fourth, the accuracy of *M* estimates is dependent on the accuracy of independent life-history parameter values in the equations. Finally, the theoretical relationships between mortality and other life-history parameters are usually derived under equilibrium assumptions (e.g., Beverton and Holt 1957).

Catch curve analysis

Based on the exponential law of population decline, the logarithm of abundance of a cohort by age over full-recruitment ages is a linear function of age (Quinn and Deriso 1999). Replacing abundance by catch and plotting the logarithm of the cohort catch by age versus age result in a catch curve (Vetter 1988, Quinn and Deriso 1999). The negative of the slope of a catch curve over full recruitment ages is *M* if the fishing mortality is negligible or is total mortality for an exploited stock. For the exploited stock, total mortality and average fishing effort over the same period can be used to estimate *M*. Under the equilibrium assumption, age compositions from one time sample can be used to approximate cohort age composition over time. But such an assumption is unlikely to hold for many animal populations.

For animals that can be aged easily, data for catch curve analysis are readily available, so it is an easy method to estimate *M*. However, the precision of *M* estimates based on this method may not be good. Vetter (1988) listed many disadvantages of this method.

Ratio method

Suppose the fishing season is short and can be approximated by pulse fishing. Abundance, N_{t+k}, in year $t+k$ can be expressed as

$$N_{t+k} = N_t \exp(-kM_t) - C_t \exp[(y-k)M_t] \tag{3}$$

where k is a time interval, y is the time interval from the time abundance is estimated in year t to the mid-point of the fishery, and C_t is catch in year t. Usually k is ≤ 1, y is $< k$, and both parameters are known for a given fishery. Given N_{t+k}, N_t, and C_t, M_t can be solved from equation (3). Equation (3) can be easily extended to the cases without the pulse fishing assumption. If $C_t = 0$ and $k = 1$, M is the negative logarithm of the ratio of abundances between two consecutive years,

$$M_t = -\ln(N_{t+1} / N_t). \tag{4}$$

Estimates of M from the ratio method are usually very variable from year to year due to measurement errors in data, so the average of M_t is often calculated. If catchability is known or $C_t = 0$, relative abundance index can be used.

The ratio method is easy to use and does not demand much data. But M estimates may be impacted greatly by measurement errors in data and in the catchability parameter. Somerton (1981), Otto (1998), and Zhang et al. (2002) employed this method to estimate M for crab stocks.

Mark-recapture experiments

A variety of techniques have been developed to estimate M based on mark-recapture experiments (e.g., Seber 1982, Quinn and Deriso 1999). The advantage of a mark-recapture approach over other approaches is that the total beginning number of animals (released) is known exactly. However, mark-induced effects on intermediate and long-term mortality, behavior, distribution, and vulnerability to capture, as well as tag loss, underreporting, or incorrect reporting may affect M estimates.

Models

A large majority of population models, including delay-difference, age-structure, catch-survey, and length-based models, include M as a parameter. Fitting these population models to the data can estimate M. Natural mortalities for herring stocks in Alaska and British Columbia are estimated through age-structured models (Schweigert 2001), and length-based models and catch-survey models are also used to estimate M for crab stocks in Alaska (Zheng and Kruse 2002; Zheng et al. 1995a,b, 1997, 1998). The advantages of a population model approach are that multiple data sets can be incorporated in the model and that M estimates can be consistent with the population data. However, M is usually confounded with other population model parameters, thus the objective function may not be very sensitive to M for some models (Quinn and Deriso 1999). Also due to confounding, it is difficult to estimate *M* and survey catchability simultaneously (Zheng et al. 1997, Zheng 2003a).

Comparison of M estimates for crab stocks

Natural mortality has been estimated for many crab stocks worldwide using a variety of methods (Tables 1 and 2). Comparison of these estimates helps us to understand the likely ranges of *M* for crab stocks, variation of estimates with different methods, and changes in estimates with body size and time. For a crab stock with limited data, the estimates of *M* from other crab stocks with similar life history parameters may serve as a starting point to establish a range of *M* for harvest strategy study. Among the methods used to estimate *M* for crab stocks listed in Tables 1 and 2, population models and mark-recapture experiments are most commonly used. Life-history analysis also has been used for several crab stocks, and all of them use longevity as the independent variable to estimate *M*. Three studies use the ratio method, and two studies use catch curve analysis (Tables 1 and 2). It is difficult to verify whether these studies represent all studies of estimating *M* for crab stocks, but they are among the studies most easily found in the literature.

The most significant feature of these *M* estimates is high variation. For Bristol Bay red king crab, estimates of *M* range from 0.02 to 1.75 (Table 1) and the large majority are from 0.02 to 1.0 (Fig. 2). This large variation reflects changes in *M* estimates over time as well as over size groups. Different methods also result in different estimates. Estimates of *M* for other crab stocks also vary greatly among different methods. For example, the ranges of *M* estimates are 0.19 to 2.04 for St. Matthew Island blue king crab, 0.27 to 2.11 for eastern Bering Sea snow crab, and 0.78 to 6.87 for Florida stone crab (Table 2).

The relationship between estimated *M* and maximum ages is difficult to be determined for crab stocks. Maximum ages of 22-24 and 15 were estimated for Alaska king and Tanner/snow crabs, respectively (NPFMC 1999). Based on growth and maximum sizes, maximum ages were assumed to be 15 for Gulf of St. Lawrence snow crab, 8 for Dungeness and Chesapeake Bay blue crabs, 23 for edible crab, 7 for Chile *Cancer polyodon* crab, and 5 for Florida stone crab for the purpose of this study (Bennett 1979, Gundersen 1979, Hankin et al. 1985, Ehrhardt 1990, Smith and Jamieson 1991, Wolff and Soto 1992, Rugolo et al. 1998). Plot of mean or best estimates of *M* for each row in Tables 1 and 2 versus estimated maximum ages shows a general trend of decreasing *M* with increasing maximum age (Fig. 3), as shown by Hoenig (1983). However, unlike the *M* estimates for mollusks, fish, and cetaceans shown in Hoenig (1983), the variation of *M* estimates for crabs is so large that a meaningful relationship between *M* estimates and maximum ages is difficult to be established. The R^2 for regression lines in Fig. 3 is 0.30 for crabs, compared to 0.78 for mollusks, 0.68 for fish, and 0.70 for cetaceans (Hoenig 1983).

Table 1. Comparison of published instantaneous natural mortality estimates for Bristol Bay red king crab.

			Mid-carapace length (mm)											
Method	Reference	Years	87	102	115	125	135	145	152	157	162	167	170	Mean
Males														
Tagging	1	1955-61					0.3-.4	0.3-.4	0.3-.4	0.3-.4	0.3-.4	0.3-.4	0.3-.4	0.35
Tagging	2	1955-61					0.07	0.06	0.21	0.36	0.63	0.86	0.93	0.45
Tagging	3	1954-61	0.51	0.37	0.57	0.30	0.09	0.24	0.46	0.57	0.93	0.63	1.22	0.54
Tagging	3	1966-68	0.06	0.12	0.12	0.08	0.08	0.15	0.52	0.66	0.75	0.81		0.34
Catch-curve	4	1968-79						0.11	0.23	0.50	0.57	0.61	0.76	0.46
Ratio	5	1969-80	0.58	0.70	0.71	0.34	0.33							0.53
Ratio	5	1981-86	1.21	0.72	0.49	0.65	1.13							0.84
Model	6	1977-80		0.48	0.08	0.07								0.21
Model	6	1981-89		0.75	0.23	0.50								0.49
Tagging	7	1990-91	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.54
Tagging	7	1990-91,93	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70
Tagging	7	1990-97	0-1.6	0-1.6	0-1.6	0-1.6	0-1.6	0-1.6	0-1.6	0-1.6	0-1.6	0-1.6	0-1.6	0.92
Life history	8					0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20
Model	9	72-79, 85-93, 98-2000	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	
Model	9	1980-84		1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Model	9	1994-97, 2001-03		0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10
		Mean for males	0.61	0.53	0.46	0.44	0.44	0.38	0.45	0.51	0.57	0.58	0.63	0.49
Females														
Model	9	1972-80		0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48
Model	9	1981-84		1.75	1.75	1.75	1.75	1.75	1.75	1.75	1.75	1.75	1.75	1.75
Model	9	1985-93, 98-2000		0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35
Model	9	1994-97, 2001-03		0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05

References: 1. Cleaver 1963; 2. Hirschhorn 1966; 3. Balsiger 1974; 4. Reeves and Marasco 1980; 5. Reeves 1988; 6. Greenberg et al. 1991; 7. Siddeek et al. 2002; 8. NPFMC 1999; 9. Vining and Zheng 2004. For reference 7, 0-1.6 range from 1990 to 1997: 0.36, 1.57, 0.52, 0.03, 1.62, 1.62, 1.62, 0.02.

Table 2. Comparison of published instantaneous natural mortality (*M*) estimates for king, Tanner, snow, Dungeness, blue, edible, *Cancer polyodon*, and stone crabs worldwide. Size is measured by carapace length (CL) or carapace width (CW).

Method	Reference	Sex and size	Years	*M* Range	Mean *M*
St. Matthew Island blue king crab					
Life history	1	Mature		0.20	0.20
Tagging	2	Males 105-139 mm CL	1982-1983	0.19-2.04	0.81
Tagging	3	80-157+ mm CL	1995	0.19	0.19
Model	4	Males >89 mm CL	1978-1998, 2000-2003	0.27	0.27
Model	4	Males >89 mm CL	1999	1.50	1.50
Pribilof Islands blue king crab					
Life history	1	Mature		0.20	0.20
Tagging	2	Males 140-169 mm CL	1978-1983	0.34-0.94	0.79
Model	4	Males >104 mm CL	1975-2003	0.31	0.31
Aleutian Islands golden king crab					
Life history	1	Mature		0.20	0.20
Tagging	3	Males 80-180+ mm CL	1997	0.38	0.38
Tagging	3	Males 80-180+ mm CL	1991	0.49	0.49
Tagging	3	Males 80-180+ mm CL	1991, 1997	0.57	0.57
Eastern Bering Sea Tanner crab					
Life history	1	Mature		0.30	0.30
Ratio	5	Males 95-110 mm CW	1973-1978	0.35	0.35
Model	5	Males >135 mm CW	1969-1979	0.13-0.28	0.25
Model	6	Males >92 mm CW	1975-1994	0.49	0.49
Model	6	Females >69 mm CW	1975-1994	0.52	0.52
Eastern Bering Sea snow crab					
Life history	1	Mature		0.30	0.30
Ratio	7	Mature males	1989-1994	1.07-2.11	1.07[a]
Model	8	Mature males	1989-2000	0.27-0.97	0.35
Model	8	Mature females	1989-2000	0.52-0.56	0.52[a]
Gulf of St. Lawrence snow crab					
Model	9	Mature males >94 mm CW	1988-2001	0.26-0.48	0.40
Model	9	Mature males <95 mm CW	1988-2001	0.53-1.02	0.75
British Columbia Dungeness crab					
Ratio	10	Nonmolting males >129 mm CW	1994-2000	0.50	0.50
Ratio	10	Molting males >129 mm CW	1994-2000	0.90	0.90
Tagging	11	Males <155 CW	1985-1986	2.3-2.8	2.50
Tagging	11	Females 135-171 mm CW	1985-1986	1.30	1.30

Table 2. (Continued.)

Method	Reference	Sex and size	Years	*M* Range	Mean *M*
California Dungeness crab					
Tagging	12	Females >154 mm CW	1981-1983	1.98-2.53	2.26
Tagging	12	Females >125-140 mm CW	1981-1983	0.69	0.69
Chesapeake Bay blue crab					
Life history	13	All crab		0.38	0.38
Southwest England edible crab					
Catch-curve	14	Males	1968-1971	0.14	0.14
Catch-curve	14	Females	1968-1971	0.06	0.06
Norwegian edible crab					
Tagging	15	All crab	1962-1966	0.24-0.30	0.28
Chile Cancer polyodon crab					
Life history	16	Males		0.74-0.99	0.87
Life-history	16	Females		0.80-1.00	0.90
Florida stone crab					
Tagging	17	Ages 1.5-2.5	1984-1985	0.78	0.78
Tagging	17	Ages 2.5-terminal age	1984-1985	1.90	1.90
Tagging	17	Ages 3.5-terminal age	1984-1985	6.87	6.87

References: 1. NPFMC 1999; 2. Otto and Cummiskey 1990; 3. Siddeek et al. 2002; 4. Vining and Zheng 2004; 5. Somerton 1981; 6. Zheng et al. 1998; 7. Otto 1998; 8. Zheng 2003a; 9. Wade et al. 2003; 10. Zhang et al. 2002; 11. Smith and Jamieson 1991; 12. Hankin et al. 1985; 13. Rugolo et al. 1998; 14. Bennett 1979; 15. Gundersen 1979; 16. Wolff and Soto 1992; 17. Ehrhardt 1990.

[a]Best estimates.

The overall estimates of *M* for crabs are higher than those for mollusks, fish, and cetaceans (Fig. 3).

Challenges to estimating *M* for crab stocks

The large variation of *M* estimates for crab stocks indicates the challenges to obtain an accurate estimate of *M* for a crab stock. Not only do the limitations of each estimation method for fish stocks apply to crab stocks, each method also has further limitations for crab stocks. Maximum age is an important parameter to determine *M*, yet with the difficulty of aging crabs, maximum age is usually an educated guess for a crab stock. Mark recapture and growth data can help to narrow the range for the maximum age, but without a good aging technique and a relatively large sample size, estimates of maximum ages for crab stocks are still a combination of art and science. It is still hard to find a crab stock that we can determine what ages to which 1% or 5% of the population survives. Among seven crab stocks for which life-history analysis was employed to estimate *M* in

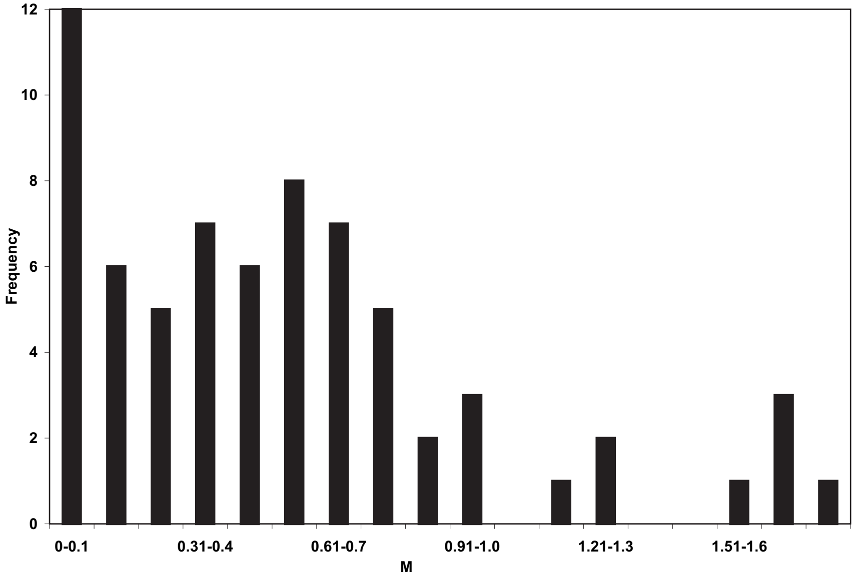

Figure 2. Frequency of published instantaneous natural mortality estimates for Bristol Bay red king crab from Table 1. Only mean values in Table 1 were used if natural mortality was assumed constant over length for the estimate.

Tables 1 and 2, equation (1) was used for five stocks with approximated maximum ages (NPFMC 1999). Equation (2) with a proportion of 0.05 was used for one stock (Rugolo et al. 1998); however, no information was provided in the paper to justify that 5% of the population survives to the assumed maximum age, and maximum ages ranging from 4 to 8 were debated (Lipton and Bockstael 2001). Both age at maturity and Pauly's formula were used to estimate *M* for one stock (Wolff and Soto 1992). Generally for fish stocks, high *M* is associated with high mean environmental temperature, fast growth rate, and small asymptotic body size as indicated by Pauly's formula. Yet for crab stocks of the same species, large asymptotic body size is usually associated with high temperature and fast growth rate. Also, based on Fig. 3, it is questionable to apply relationships established for non-crab stocks to crab stocks.

Catch-curve analysis generally requires age information. The two catch-curve analyses for crab stocks in Tables 1 and 2 (Bennett 1979, Reeves and Marasco 1980) used mean length-age relationships to convert lengths to ages. Due to length overlaps for a given age, this conversion increases errors for catch-curve analysis. The errors depend on the level

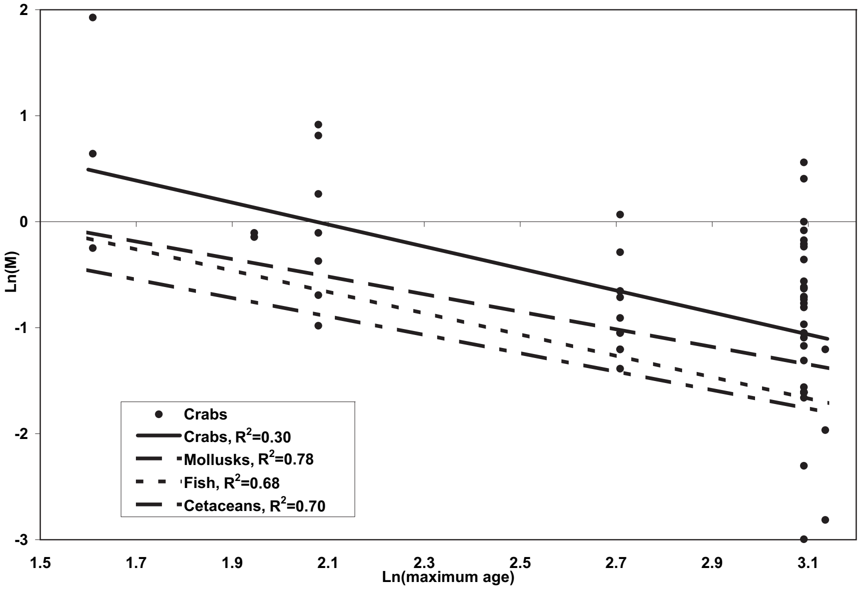

Figure 3. Comparison of estimated relationships between instantaneous natural mortality and estimated maximum age for crabs, fish, mollusks, and cetaceans. Relationships for fish, mollusks, and cetaceans are from Hoenig (1983). Mean estimates for each row in Tables 1 and 2 were used.

of length overlaps and contrast of year classes, with the large errors for old ages.

It is difficult to quantify the impacts of tagging on crabs and to estimate tag loss or underreporting. Due to molting, crabs are more difficult to tag than fish. Tagged crabs may have smaller growth increment per molt than untagged crabs for some crab stocks. For example, the growth increment per molt for male snow crab of 100 mm carapace width in Conception Bay, Newfoundland, was estimated to be 11.2 mm for tagged crab (Taylor and Hoenig 1990) and 18-20 mm for untagged crab (Hoenig et al. 1994). The impacts of tagging on crabs may affect crab catchability, which in turn influences *M* estimates.

Life-history and catch curve analyses assume a constant *M*, which may not be true for crab stocks. Vetter (1988) reviewed evidence that *M* varies with age, population density, disease, parasites, food supply, predator abundance, water temperature, sex, and size for fish stocks. As shown for abalone stocks by Shepherd and Breen (1992), *M* generally is very high during larval stages, decreases during the juvenile period, becomes relatively stable and low during most adult ages, and increases with senescence. Estimates of *M* for many crab stocks vary greatly with size and sex, with Florida stone crab being an extreme case (Tables 1 and 2). Based on survey and fishery data from 1993 to 2000, *M* for the snow crab stock in the southwestern Gulf of St. Lawrence, Canada, was estimated to be 0.11 for immature males, 0.63 for mature males with shell conditions 1 and 2 (molted within one year), 0.22 with shell condition 3 (molted 0.7-3 years ago), 0 with shell condition 4 (molted 2-5 years ago), and infinite with shell condition 5 (molted 4-6 years ago; E. Wade, Gulf Fisheries Centre, DFO, Canada, pers. comm.). Extremely high *M* associated with senescence may be common for many crab stocks, as is the case for the Gulf of St. Lawrence snow crab stock as well as the Florida stone crab stock (Table 2). Stevens (1990) suggested that high *M* for Bristol Bay red king crab during the early 1980s (Table 1) may be caused by senescence because, due to temperature-dependent growth of juvenile crab, changes in temperature in the mid and late 1970s made the length frequencies overlap for several year classes, resulting in mostly very old crab in the early 1980s.

More data may help to better understand *M*, yet even the data-rich crab stocks may still seem data limited in terms of estimating *M*. Among the 14 crab stocks in Tables 1 and 2, Bristol Bay red king crab has the most data available. Since 1972, an annual trawl survey has been conducted and catch by length and shell condition has been collected. More than 10 years of tagging data are available, which can be used to estimate growth parameters and the maximum age. Data are available for any of the five common approaches to estimate *M*. However, different approaches provide very different results (Table 1, Fig. 2). This may be caused by changes in *M* with length, age, shell condition, and sex, as mentioned above. In addition, *M* apparently varies over time (Table 1). The survey data from two periods for female Bristol Bay red king crab suggest changes in *M* over time. Only large males are allowed to be retained in the fishery, and because of low catchability for females in the fishery and relatively low bycatch mortality rate (Zheng 2003b), *M* should make up a very high proportion of total mortality for females. Mature female (≥ 90 mm carapace length) abundance decreased sharply from 1982 to 1985 and was relatively stable or increased from 1994 to 1998 despite relatively low recruitment from 1994 to 1996 and 1998 (Fig. 4). It appears that *M* was much higher from 1982 to 1985 than from 1994 to 1998, which is consistent with assessment model results (Table 1). Different *M* were estimated for different periods in a length-base model (Vining and Zheng 2004), and estimated *M* changed greatly over time (Table 1). Changes in *M* over length, age, shell condition, sex, and time make it difficult to estimate *M* even for a data-rich stock.

The challenges to estimate *M* for crab stocks explain the uncertainty of *M* estimates and indicate the need for good data. Mark-recapture

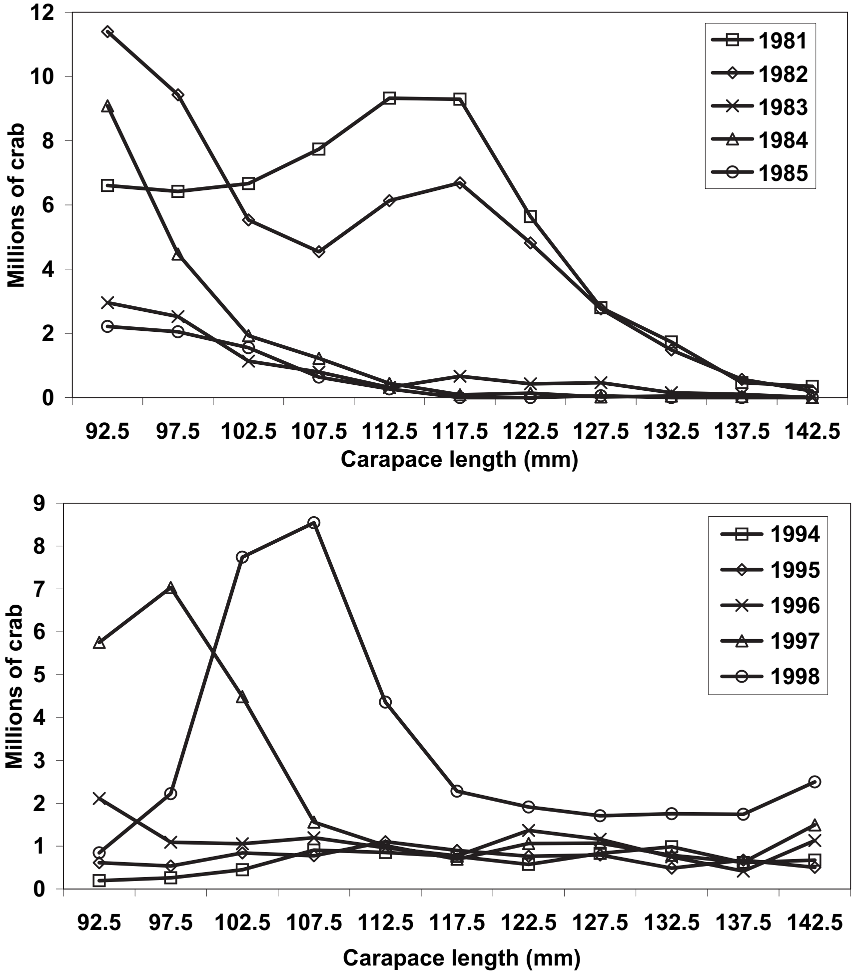

Figure 4. Abundances by length of female red king crab in Bristol Bay, Alaska, from NMFS trawl surveys during 1981-1985 (top) and 1994-1998 (bottom).

studies that are specifically and well-designed for estimating *M* would definitely help. However, if *M* varies over time as has been estimated for Bristol Bay red king crab, it is impossible to collect mark-recapture data every year for a long period to estimate *M* for any stock. Long-term survey and commercial catch data can be used to develop length-based models to estimate *M* (e.g., Fu and Quinn 2000; Zheng et al. 1995a,b, 1998; Zheng 2003a). Because of parameter confounding, some parameters in a length-based model, particularly survey catchability, have to be estimated outside of the model to obtain reliable estimates of *M* (Zheng 2003a). In the future, research efforts on estimating *M* should be focused on reducing the uncertainty of *M* estimates. Further understanding of crab life history and survey catchability will help reduce the uncertainty.

Conclusions

Several conclusions are made in this study:

1. Many approaches can be employed to estimate *M* for crab stocks, and each has its own data requirement and limitations. Mark-recapture experiments, population models, and life-history analysis are three commonly used methods for crab stocks.

2. The most significant feature of *M* estimates for crab stocks is high variation. Different approaches result in different estimates.

3. Estimated *M* appears to be higher for crabs than for fish with the same estimated maximum age. This casts doubt about using the life-history relationships established for fish stocks to estimate *M* for crab stocks.

4. *M* may change greatly over time as well as over size, which requires extensive data to get reliable estimates of *M*.

5. The data-rich stock in this study, Bristol Bay red king crab, still appears data limited in terms of estimating *M* due to changes in *M* over time and confounding among estimated parameters: *M*, survey catchability, recruitment, and molting probabilities.

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Application of a Kalman Filter to a Multispecies Stock Complex

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Abstract

Fish species are often managed as part of multispecies complexes when existing biological data and/or reported catch data are insufficient to conduct a more detailed single-species assessment. Catches of multispecies complexes may be reported for the aggregate complex, and an estimate of total catch by species can be made by applying species-specific catch proportions from fishery observer data (if available) to the total aggregated catch. However, this procedure would be expected to increase the variance of the resulting catch estimates, and possibly result in correlation in estimated catch between members of the complex. A Kalman filter is applied to such multispecies complexes to incorporate errors of this type, with the eastern Bering Sea/Aleutian Islands shortraker (*Sebastes borealis*) and rougheye (*Sebastes aleutianus*) rockfish used as an example. For this stock complex, consideration of sampling variability in observer data adds considerably to the assumed errors in catch. The Kalman filter also provides a methodology for treating the missing data that were common in the shortraker/rougheye data, and would be expected in most data-poor situations. By considering process errors, observation errors, and covariances between observations, the Kalman filter is a powerful tool for multispecies complexes that forces a consideration of sources and magnitude of the errors in the system. However, the shortraker/rougheye example illustrates that this methodology cannot make informative parameter estimates from non-informative data, and emphasizes the importance of reducing the sampling variances of input data and developing suitable priors for model parameters.

Introduction

Fish population models are often fit to observed data by assuming that deviations between model predictions and data are either the result of process error (deviations between modeled quantities [e.g., biomass] and true quantities) or observation error (deviations between measured quantities and true quantities), with observation error models generally recognized as producing more reliable parameter estimates in surplus production models in most cases (Polacheck et al. 1993). However, considerable interest exists in developing models that incorporate both process and observation errors. The Kalman filter (Kalman 1960) is a standard method of choice for these problems, and has been applied to a variety of underlying biological models, including escapement-type models (Reed and Simons 1996), the delay-difference model (Kimura et al. 1996), surplus production models (Pella 1993, Thompson 1996), and more complex length-based models (Sullivan 1992).

The ability to consider both process and observation errors motivates application of the Kalman filter to multispecies complexes in data-poor situations. One reason for managing species as part of a multispecies complex is that the biological parameters are unknown or too uncertain to support a more detailed single-species assessment. However, grouping species with perceived similar life history characteristics may result in considerable process error. Although observation errors in indices of stock size have long been recognized in fisheries modeling, observation errors in catch may become significant for new and developing fisheries where fishery sampling may be limited. The Kalman filter also provides a number of methods for addressing missing data points that are common with data-poor stocks. Finally, the Kalman filter can readily be extended from single-species models to multispecies complexes simply by extending the state and observation vectors to incorporate variables associated with additional species.

An additional characteristic occasionally seen with data-poor stock complexes is that catches are not reported by species, but rather in aggregate for the entire complex. If observer data exist that describe catch by species, an estimate of total catch by species can be made by applying the observer proportions to the total aggregated catch. However, this procedure would be expected to increase the variance of the resulting catch estimates, and possibly result in correlation in estimated catch between members of the complex. The Kalman filter provides a convenient structure for incorporating these variances and covariances in the assessment procedure.

Alaskan rockfish exhibit many of the characteristics described above. Although fishery-independent surveys exist for these species, the coefficient of variation (CV) for these surveys are quite high, often 0.25 or higher. Relatively little is known of rockfish life history characteristics, particularly in the important early stages. Difficulty in aging some species of rockfish results in little demographic information on year-class strength, and application of new methods of aging rockfish can substantially alter our perceptions of natural mortality estimates (e.g., shortspine thornyhead [*Sebastolobus alascanus*], see Pearson and Gunderson 2003). Finally, the catches of some rockfish are not reported by species, but rather as part of the multispecies complexes in which they are managed.

The purpose of this manuscript is to demonstrate how a Kalman filter can be applied to a multispecies complex, using the eastern Bering Sea/Aleutian Islands (BSAI) rougheye (*Sebastes aleutianus*) and shortraker rockfish (*Sebastes borealis*) stock complex as an example. Because published examples applying the Kalman filter to fisheries data rarely involve data-poor stock complexes, our primary focus will be to illustrate the methodology. First, the underlying biological model used, the Gompertz-Fox model (Fox 1970), is described. Second, the basic methodology of the Kalman filter is described. Third, a procedure for determining the variances and covariances of species-specific catch levels from aggregated catch data is presented, as well as treatment of missing data. Finally, an example application of the Kalman filter to the BSAI rougheye/shortraker stock complex is provided.

Methods

The Gompertz-Fox model

The Gompertz-Fox surplus production model was chosen for this analysis because it is a simple model that offers some information on population growth rate and carrying capacity, and it is easily transformed into a linear form suitable for the Kalman filter (Thompson 1996). The Gompertz-Fox model was applied to the rougheye/shortraker complex, and the Kalman filter provided a method of statistically estimating the parameter values. The Gompertz-Fox model (Fox 1970) describes the rate of change of stock size as

$$\frac{dx}{dt} = ax[\ln(k) - \ln(x)] - fx \qquad (1)$$

where a is growth rate, x is stock size, k is carrying capacity, and f is fishing mortality. The rate of catch, C, is modeled as $dC/dt = fx$, and the f level corresponding to the maximum sustainable yield (MSY) is equivalent to the growth parameter a. Equilibrium biomass (b) is

$$b = ke^{\frac{-f}{a}} \tag{2}$$

and the equilibrium stock size corresponding to MSY, b_{msy}, is ke^{-1}.

The model is mathematically equivalent to a model of individual growth developed by Gompertz, and describes a situation where stocks at low sizes would show a sigmoidal increase in stock size to an asymptote. The Gompertz-Fox model can be derived from the Pella-Tomlinson model (Pella and Tomlinson 1969) by taking the limit as the shape parameter (which controls the relative location of the peak of the production curve) approaches one. The peak of the Gompertz-Fox production curve occurs at approximately 37% of the carrying capacity, in contrast to the logistic model where the peak occurs at 50% of the carrying capacity.

The Kalman filter

A brief review of the Kalman filter is provided here, as more thorough presentations are provided in Meinhold and Singpurwalla (1983), Harvey (1990), and Pella (1993). The Kalman filter separates the system into a model of the state variables, which describes the true (but unobserved) state of nature, and a model of the observation variables, which describes how the observed data relate to the state variable. The state variable is modeled as

$$X_t = T_t X_{t-1} + c_t + R_t \eta_t \tag{3}$$

where X_t is a vector of m state variables at time t, T_t is an $m \times m$ matrix, c_t is an $m \times 1$ vector of additive deterministic components, R_t is an $m \times g$ matrix, and η_t is a $g \times 1$ vector of random process errors with a mean of zero and a covariance matrix of Q_t (Harvey 1990, section 3.1). The inclusion of the R_t vector is useful when a particular state variable is affected by more than one type of random disturbance. Note that when there is only a single state variable the problem simplifies considerably and all terms become scalars. For the shortraker/rougheye complex, the state variables at each time step are the log biomass of each species. Finally, the state variable is described with a distribution with an estimated mean α_t and variance P_t. The observation equation is

$$Y_t = Z_t X_t + d_t + \varepsilon_t \tag{4}$$

where Y_t is an $n \times 1$ vector of observed variables, Z_t is an $n \times m$ matrix, d_t is an $n \times 1$ vector of additive deterministic components, and ε_t is an $n \times 1$ vector of random observation errors with mean zero and covariance matrix H_t (Harvey 1990, section 3.1). For the shortraker/rougheye complex, the observation variables are the log survey biomass estimates and log catches for each species.

A distinct advantage of the Kalman filter is that both the process and observation errors are incorporated into the parameter estimation procedure. The method by which this occurs can be understood by invoking the Bayesian concepts of “prior” and “posterior” estimates of the state variable (Meinhold and Singpurwalla 1983). Denote α_{t-1} as the posterior estimate of X_{t-1} using all the data up to and including time $t - 1$. At time step t, a prior estimate of the state variable is made from the state equation (equation 3) and the posterior estimate from the previous step α_{t-1}. Because this prior estimate of X_t uses all the data up to time $t - 1$, it is denoted as $\alpha_{t|t-1}$. The prior estimate can be used with equation 4 to predict the observation variables at time t. Upon observation of Y_t there are now two estimates of the observed variables; the observed data Y_t and the prediction from the prior estimate $\alpha_{t|t-1.}$ The Kalman filter updates the prior by producing a posterior estimate, $\alpha_{t|t}$, that results in a value of Y_t between these two points, and the extent to which the posterior estimate differs from the prior estimate is a function of the relative magnitudes of the observation error variance and the process error variance (Harvey 1990, section 3.2). The posterior estimates are then used to create the prior estimates in the next time step to continue the recursive procedure. Parameter estimation can be obtained by minimizing the negative log likelihood of the data.

A requirement of the Kalman filter is that the model is linear in the state variable, which is satisfied for the Gompertz-Fox model by defining the state variable as log biomass. The log transformation of equation 1 is

$$\frac{dX}{dt} = a(B - X) \tag{5}$$

where $X = \ln(x)$ and $B = \ln(b) = \ln(ke^{-f/a})$, where x is biomass and b is equilibrium biomass for a given level of fishing mortality. The solution to this differential equation can be expressed as

$$X_{t+\Delta t} = e^{-a\Delta t} X_t + (1 - e^{-a\Delta t}) B_t \tag{6}$$

where annual changes in f_t result in $B_t = \ln(ke^{-f,la})$ and Δt is a discrete time period. For a single species case, defining $T_t = e^{-a\Delta t}$ and $c_t = (1 - T_t)B_t$ produces the deterministic portion of the state equation (equation 3). For the two-species shortraker/rougheye example, a version of equation 6 would exist for each species. In this case, T_t is a matrix of dimension 2 with the $e^{-a\Delta t}$ terms along the diagonal, and c_t is a vector of length 2 with each vector element corresponding to a particular species. Note that in this application the term R_t in the generic form (equation 3) is not used.

The model requires as input data the survey biomass estimates and estimates of rate of catch at the time of the survey, each on a log scale. It is assumed that the aggregated catch during the year approximates the rate of catch at the time of the survey, a reasonable approximation for BSAI rockfish because the survey occurs at the midpoint of the year. Although the observed data reflect the system at the midpoint of a year, it is expected that the instantaneous fishing mortality rate would change between calendar years; thus, a time-step of one-half year was chosen.

Treatment of missing data

In data-poor situations, it is unlikely that observations will be available for each data type at each time step of the model. This is the case for Alaskan rockfish, where biennial or triennial trawl surveys occur but estimates of fishing removals can generally be made each year. Thus, in some years only one data type is available, and there may also be some time steps where neither type of data is available (such as the beginning of the calendar year, where neither a survey biomass estimate nor an estimate of the rate of catch exists). The Kalman filter is generally robust to these situations (Harvey 1990, section 3.4.7). For years when only one data type is available, the dimension of the observation equation could be modified to reflect the reduced data. For example, in years when only catches are available, the dimension of the observation vector equals two and the terms in the observation equation (equation 4) are

$$Y_t = \begin{bmatrix} \ln(C_{re,t}) \\ \ln(C_{sr,t}) \end{bmatrix}, \quad Z_t = \begin{bmatrix} 1 & 0 \\ 0 & 1 \end{bmatrix} \text{ and } d_t = \begin{bmatrix} \ln(f_{re,t}) \\ \ln(f_{sr,t}) \end{bmatrix} \tag{7}$$

where, in a given year, $C_{re,t}$ and $C_{sr,t}$ are the aggregated catch of rougheye and shortraker, respectively, during year t and $f_{re,t}$ and $f_{sr,t}$ are the rates of removals from fishing. When both catch and observed survey biomass are available, the dimension of the observation vector equals four and the terms in the observation equations are

$$Y_t = \begin{bmatrix} \ln(S_{re,t}) \\ \ln(S_{sr,t}) \\ \ln(C_{re,t}) \\ \ln(C_{sr,t}) \end{bmatrix}, \; Z_t = \begin{bmatrix} 1 & 0 \\ 0 & 1 \\ 1 & 0 \\ 0 & 1 \end{bmatrix} \text{ and } d_t = \begin{bmatrix} \ln(q_{re}) \\ \ln(q_{sr}) \\ \ln(f_{re,t}) \\ \ln(f_{sr,t}) \end{bmatrix} \tag{8}$$

where $S_{re,t}$ and $S_{sr,t}$ are the survey biomass estimates of rougheye and shortraker in year t, and q_{re} and q_{re} are the survey catchability coefficients. Note that it is assumed that the non-logged survey biomasses are proportional to the true biomass; the proportionality terms q_{re} and q_{re} were not estimated but fixed at 1.0. The change in dimension of the observation equation between time-steps does not affect the estimation of parameters within the model other than changing the dimensions of the terms in the likelihood function. Harvey (1990) also notes that, more generally, a matrix W_t can be defined to convert the vector of all possible observation variables to the number of variables actually observed, and the example above is a particular application of this procedure. For the shortraker/rougheye complex, the parameters estimated for each species were a, k, and the time series of f_t.

Two common solutions exist for time steps when no data are available (Harvey 1990). In the first, the time-steps of the model could be varied to match the observations. In this application, most of the cases without either data type occur at the beginning of the calendar year, and creating time-steps that extend from mid-year to mid-year would not allow for changes in fishing mortality that occur at the beginning of calendar years. An alternate solution employed in this study is to skip the updating equation that relies upon new data, and set the posterior estimate of the state variable equal to the prior estimate, or $\alpha_{t|t} = \alpha_{t|t-1}$ (Harvey 1990).

Initial estimate of state variables

An initial estimate of the mean and variance of the state variable (α_0 and P_0, respectively) is required to begin the recursive calculations. Assuming P_0 equals zero corresponds to interpreting the elements of α_0 as fixed rather than described with a probability distribution, allowing them, in theory, to be estimated along with other parameters of the model. However, estimating these terms as parameters complicates the estimation procedure and is not recommended (Harvey 1990, Pella 1993). The maximum likelihood estimates (MLE) of the elements in α_0 were obtained from Rosenberg's algorithm (Rosenberg 1973, Harvey 1990), in which P_0 and α_0 were initially set to zero. The MLE estimate of α_0 can then be expressed as a function of other model parameters and the observed data, and used in a concentrated likelihood function which does not depend on α_0. Once the MLE estimates of α_0 are obtained, these estimates are then used as inputs in a second iteration of the Kalman filter to obtain the estimated values of the state variables.

Catch estimation error

Species-specific catch estimates of shortraker and rougheye rockfish are often made from application of an observed proportion of the catch (from observer sampling) to the reported aggregated catch for the species complex. This estimation procedure produces quantities that can be viewed as the product of two random variables. While overall catch data for a species or species complex are often viewed as relatively precise compared to other fisheries information, the proportions from observer sampling adds additional error. In addition, the species-specific estimates of catch are likely to be correlated because of correlations between the estimated proportions, and the dependence of each species-specific estimate upon a common variable, the total complex catch.

For shortraker and rougheye, this type of procedure is applied separately for each of several fisheries (foreign, joint venture, and domestic) and areas (eastern Bering Sea and Aleutian Islands). The total catch for each species is then a sum of several random variables:

$$\begin{aligned} C_{RE} &= \sum_i p_{RE,i} \times C_{A,i} \\ C_{SR} &= \sum_i p_{SR,i} \times C_{A,i} \end{aligned} \qquad (9)$$

where $p_{RE,i}$ and $p_{SR,i}$ are the proportions of rougheye and shortraker observed in observer sampling, $C_{A,i}$ is the aggregated catch, in biomass, of the multispecies complex, and i indexes the total number of fisheries in the summation. It is assumed that each $C_{A,i}$ is lognormally distributed, the species proportions from observer sampling follow a multinomial distribution, and these two random variables are independent. The variance of each product in the summation can be obtained from Goodman's (1960) formula for the product of independent random variables. The variances of the log of the species-specific catch estimates were set to the square of the estimated CVs of C_{RE} and C_{SR}, relying on the approximate equivalence of the CV of a lognormally distributed variable to the standard deviation of the variable on a log scale. Application of the delta method (Seber 1982) yields the covariances between $\ln(C_{RE})$and $\ln(C_{SR})$ in a given year (see appendix for derivation):

$$Cov[\ln(C_{RE}),\ln(C_{SR})] = \sum_i \left(e^{2\mu_i+\sigma_i^2}(e^{\sigma_i^2}-1)\times\frac{p_{RE,i}p_{SR,i}}{C_{RE}C_{SR}} - \frac{p_{RE,i}p_{SR,i}}{N_i}\times\frac{(C_{A,i})^2}{C_{RE}C_{SR}} \right) \tag{10}$$

where μ_i is the expected value of the ln $C_{A,i}$, σ_i is taken as the assumed CV of $C_{A,i}$, N_i is the assumed sample size for the multinomial distribution, and all random variables are evaluated at their estimated expected values. These variances and covariances populate the covariance matrix H_t corresponding to the errors of the observation equation (equation 4).

Application to BSAI shortraker and rougheye rockfish

Estimated catches from 1980 to 2003 were used for the model application (Table 1). Fisheries in U.S federal waters off Alaska were conducted by foreign and joint-venture operations until 1989, and domestic operations from 1990. Rougheye and shortraker rockfish are not target species and are taken as bycatch in other fisheries, although their catch is often retained due to their market value. Foreign catch records did not identify rougheye and shortraker rockfish by species; instead, rougheye and shortraker rockfish were reported in management categories such as "POP complex" (1980-1985, 1989; complex definition is Pacific ocean perch [*Sebastes alutus*], northern rockfish [*Sebastes polyspinus*], sharpchin rockfish [*Sebastes zacentrus*], rougheye rockfish, and shortraker rockfish) and "rockfish without POP" (1986-1988). In domestic fisheries, both the "POP complex" and the more specific "rougheye/shortraker" complex have been used to report catches, although the complexes used have differed between areas in some years.

The estimated catch variances and covariances (Table 1, Fig. 1) were computed under the assumption that the multinomial sample size N_i was 100 and σ_i (the CV of $C_{A,i}$) was 0.15. Particularly high variances for shortraker rockfish were estimated during the late 1980s when the observed proportions of shortraker rockfish in the multispecies complexes were very small. It is informative to compare these variance estimates to those that would have been obtained by assuming the CV of each catch complex remained at 0.15, but the proportions are known without error. In this case, the variance of the log catches of rougheye rockfish is shown as the bold line in Fig. 1. In most years, incorporation of error in the estimated proportions contributed substantially to the estimated variance.

Estimated biomasses of shortraker and rougheye rockfish from Aleutian Island bottom trawl surveys are shown in Table 1. Cooperative U.S.-

Table 1. Estimated catches of rougheye and shortraker rockfish (metric tons, t), variance of log catches, covariance of log catches, and survey biomass estimates t; coefficient of variation in parentheses.

	Estimated catches (t)		Variance		Covariance of	Biomass estimates (t)	
Year	Rougheye	Shortraker	ln(RE)	ln(SR)	ln(RE), ln(SR)	Rougheye	Shortraker
1980	359	455	0.098	0.041	0.0042	22,807 (0.79)	16,983 (0.20)
1981	658	573	0.055	0.052	0.0066		
1982	312	621	0.049	0.028	0.0065		
1983	111	235	0.050	0.026	0.0061	23,412 (0.37)	40,992 (0.69)
1984	114	83	0.038	0.050	0.0068		
1985	27	21	0.075	0.046	0.0077		
1986	74	30	0.068	0.114	0.0043	52,354 (0.62)	25,823 (0.28)
1987	179	64	0.062	0.135	0.0049		
1988	185	69	0.067	0.136	0.0050		
1989	585	188	0.064	0.189	0.0069		
1990	1,988	661	0.030	0.033	0.0080		
1991	250	462	0.037	0.023	0.0063	11,131 (0.45)	23,703 (0.64)
1992	1,239	364	0.023	0.044	0.0096		
1993	956	440	0.023	0.033	0.0071		
1994	778	230	0.023	0.040	0.0093	14,552 (0.26)	28,190 (0.21)
1995	394	222	0.026	0.032	0.0098		
1996	873	177	0.023	0.045	0.0076		
1997	991	164	0.022	0.047	0.0065	11,596 (0.21)	38,487 (0.26)
1998	535	176	0.024	0.039	0.0096		
1999	394	171	0.024	0.029	0.0074		
2000	274	300	0.027	0.019	0.0076	15,259 (0.21)	37,781 (0.44)
2001	624	122	0.023	0.052	0.0091		
2002	259	308	0.029	0.019	0.0084	9,613 (0.19)	16,845 (0.19)
2003	160	186	0.027	0.016	0.0068		

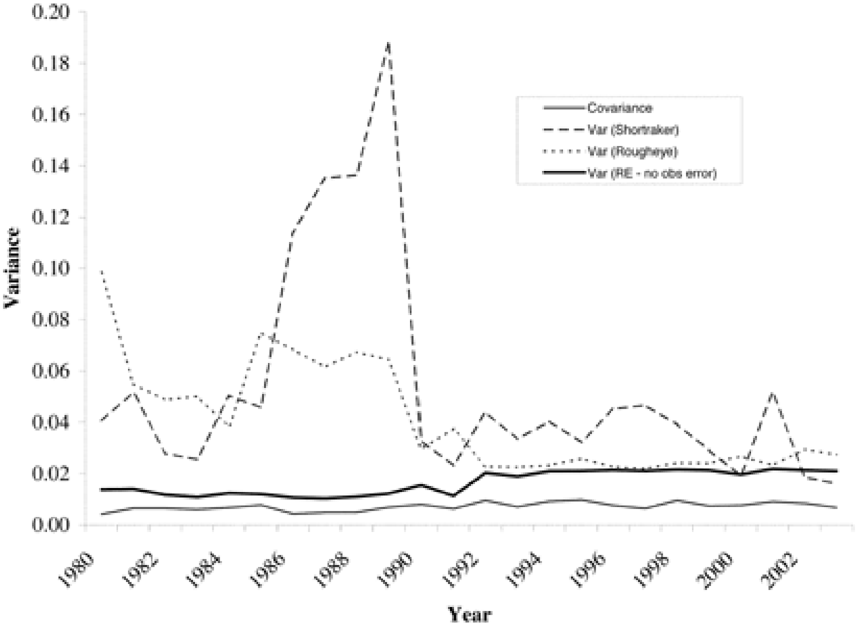

Figure 1. Estimated variances and covariances of ln(rougheye) and ln(shortraker) catches. The bold line shows the estimated variance of ln rougheye rockfish assuming that species proportions were known without error and the CVs of each aggregate complex catches were 0.15.

Japan triennial surveys were conducted from 1980 to 1986, and triennial U.S domestic trawl surveys were conducted from 1991 to 2000. Since 2000, biennial Aleutian Island trawl surveys have been conducted. A new time series of bottom trawl surveys along the eastern Bering Sea slope was initiated in 2002, and the most recent eastern Bering Sea slope survey prior to 2002 was in 1991. The variances of the log survey biomass estimates were set to the square of estimated CVs, which were obtained from the estimated sampling variances from area-swept trawl surveys. The available survey data indicate that the bulk of the BSAI rougheye and shortraker rockfish biomass exists in the Aleutian Islands (Spencer and Reuter 2002), and the lack of a substantial recent time series of eastern Bering Sea slope survey estimates motivates use of the Aleutian Islands estimates as a conservative index for the BSAI population.

A series of model fits were conducted to illustrate the effect of different relative ratios of the observation errors to process errors (Fig. 2). The input variances of the observed catch and survey data remained fixed at the levels described above, and a series of runs were made with the process error CV taking on values of 0.001, 0.05, and 0.5. The lowest level of process error variance essentially implies an observation error model, thus corresponding to many commonly used assessment models (Polacheck et al. 1993). In this case, the model fits a fairly smooth line through the observed data and does not respond strongly to the temporal pattern in the data. As the process error variance increases, the ratio of observation error variance to process error variance becomes smaller, and the temporal pattern of the individual survey data points becomes more influential. The extreme process error CVs of 0.001 and 0.5 are shown for illustrative purposes only; however, a process error CV of approximately 0.05 is expected to be appropriate for long-lived rockfish stocks and was used in the results discussed below.

The estimated fishing mortality rates allowed fairly tight fits to the observed catch data, and show similar patterns from 1980 to 1990 (Fig. 3). In the early to mid-1990s, the increased estimated fishing mortality rates for rougheye rockfish reflect the catch relative to decreased survey biomass estimates for this species. A similar decline in survey biomass estimates has not been noted for the shortraker rockfish, and the estimated fishing mortality rates for this species have been relatively stable. It should be noted that the catches for these species reflect harvest quotas for the two-species complex based upon an average of biomass estimates, and management efforts to limit bycatch have reduced rougheye bycatch and estimated fishing mortality in recent years.

The rougheye and shortraker data are relatively non-informative regarding the Gompertz-Fox model parameters (Table 2, Fig. 2). For example, unconstrained estimates for the growth parameter a were generally > 10 and thus inconsistent with perceived rougheye and shortraker life history characteristics. The model fits in Fig. 2 and Table 2 were produced by placing a lognormal prior distribution on a for each species, with the expected value equivalent to the natural mortality rate of rougheye rockfish. This choice was motivated by equivalence of a and F_{msy} in the Gompertz-Fox model, and the expectation that F_{msy} should be fairly close to the natural mortality rate (Gulland 1970). The natural mortality rate of rougheye rockfish, based upon catch curve analysis, is estimated as 0.025 (Heifetz and Clausen 1991), and a large CV of 1.0 was used for the lognormal prior so as not to unduly influence the posterior distribution of a. Marginal posterior distributions were computed from Monte Carlo Markov Chain (MCMC) integration with 1,000,000 replications (Gelman et al. 1995). The prior and posterior distributions of shortraker rockfish are similar to each other (Fig. 4), indicating a lack of information within the data. For rougheye rockfish, the mode of the posterior is similar to

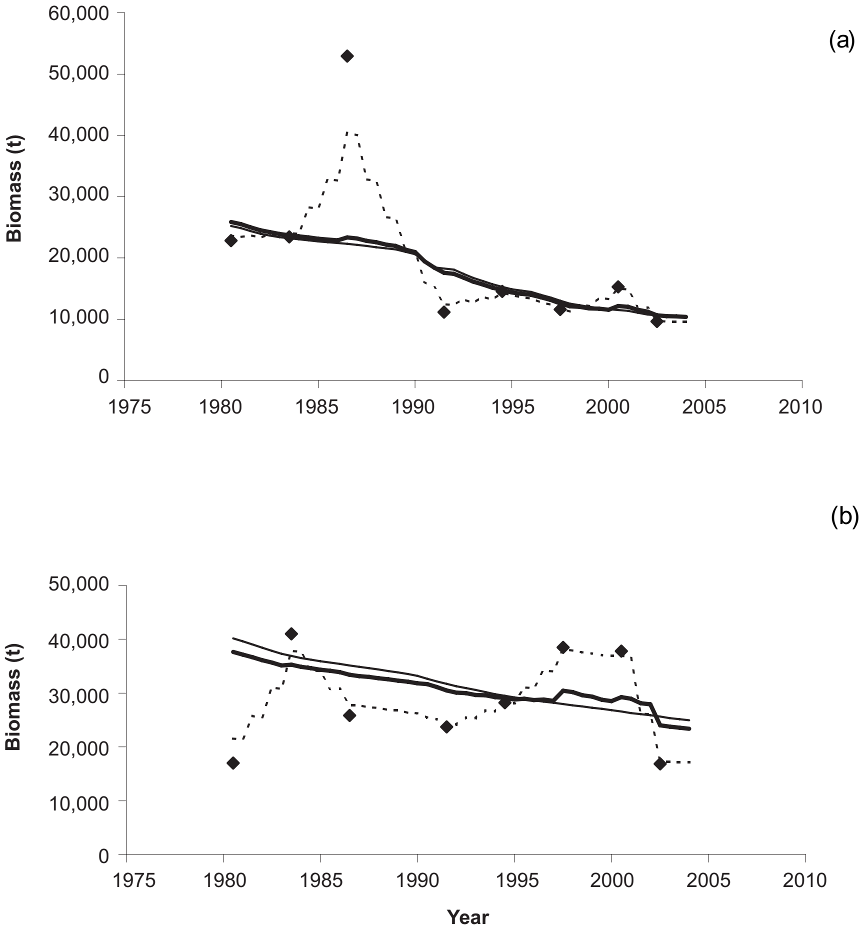

Figure 2. **Model fit of biomass (t) of rougheye (a) and shortraker (b) rockfish with the process error CV equal to 0.001 (solid light line), 0.05 (solid bold line), and 0.5 (dashed line).**

Table 2. Maximum likelihood estimates of *a* and *k* for shortraker and rougheye rockfish (process error CV = 0.05 and prior distribution on the *a* parameter), with properties of the posterior distributions obtained from Monte Carlo Markov Chain (MCMC) integration.

Species	Parameter	MLE	Mode	Mean	Median	SD	Lower 95% CI	Upper 95% CI
Rougheye	*A*	0.0183	0.0084	0.0173	0.0080	0.0245	0.0027	0.0510
Rougheye	*K*	12,154	18,644	273,271	232,611	259,557	10,798	407,622
Shortraker	*a*	0.0160	0.0125	0.0244	0.0136	0.0280	0.0033	0.1021
Shortraker	*k*	14,491	36,696	127,686	49,187	144,962	11,550	332,130

the prior, although the greater mass near the mode reduces the expected value relative to the prior. Similarly, estimates of carrying capacity *k* also show substantial uncertainty and posterior distributions with broad tails, as indicated by the estimates of the mean, mode, and median obtained from MCMC integration (Table 2).

The lack of information regarding the *a* parameter can be seen in plots of annual surplus production (ASP), defined as the change in biomass over a time period plus catch, expressed on an annual basis (Fig. 5). The *a* parameter is related to the slope of the production curve at low stock sizes, and one could imagine alternate production curves with high levels of *a* providing equally suitable fits to ASP data. Given the longevity of rougheye and shortraker rockfish, one would not expect surplus production to deviate far from zero, and this was the motivation for constraining *a* with information on the natural morality rate. Surplus production levels substantially different from zero reflect large fluctuations in estimated survey biomass over short time scales, and are generally inconsistent with perceived rougheye and shortraker life history characteristics.

Discussion

The Kalman filter provides a statistically rigorous way of addressing several problems associated with multispecies stock complexes in data-poor situations. Although the Kalman filter has typically been applied to single-species cases, the vectorized format of the Kalman filter eases application to multispecies complexes by simply increasing the dimensionality to account for additional species. By allowing both process and observation errors, the Kalman filter forces a consideration of sources and magnitude of the errors in the system. In this example, the recognition of errors in

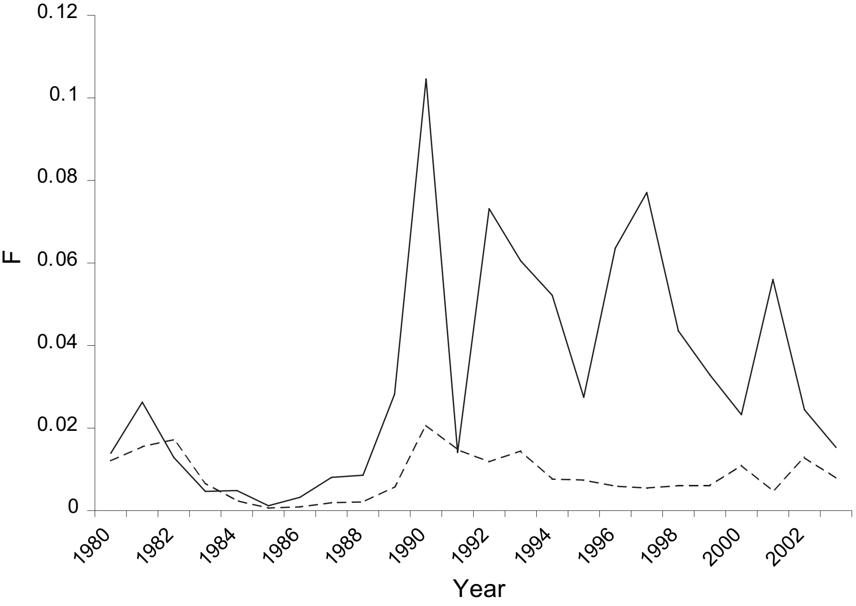

Figure 3. Estimated fishing mortality rates of rougheye (solid line) and shortraker (dashed line) rockfish.

proportions obtained from observer sampling noticeably increases our perception of the observation errors associated with single-species catch estimates (Fig. 1). The Kalman filter is relatively robust to problems of missing data that are inevitable in data-poor stocks, as illustrated by the shortraker and rougheye rockfish example. The structure of the model presented here required survey biomass estimates as input data, but fishery catch-per-unit-effort data could also be used if fishery-independent surveys are not available (Reed and Simons 1996); in this case, the fishing mortality rate f_t could be modeled as the product of a catchability coefficient q and effort E_t.

The assumptions of linearity and Gaussian distributions, often cited as limitations in applying the Kalman filter, were found not to be constraining in this application, as the available data do not clearly suggest whether a nonlinear surplus production model and/or alternative error distributions would be more appropriate. In cases where nonlinear surplus production models are required, one could apply an extended Kalman filter in which an approximation to linearity is obtained by Taylor series expansion (Harvey 1990, section 3.7.2). Alternatively, one could model state-space problems in a fully Bayesian manner, where all

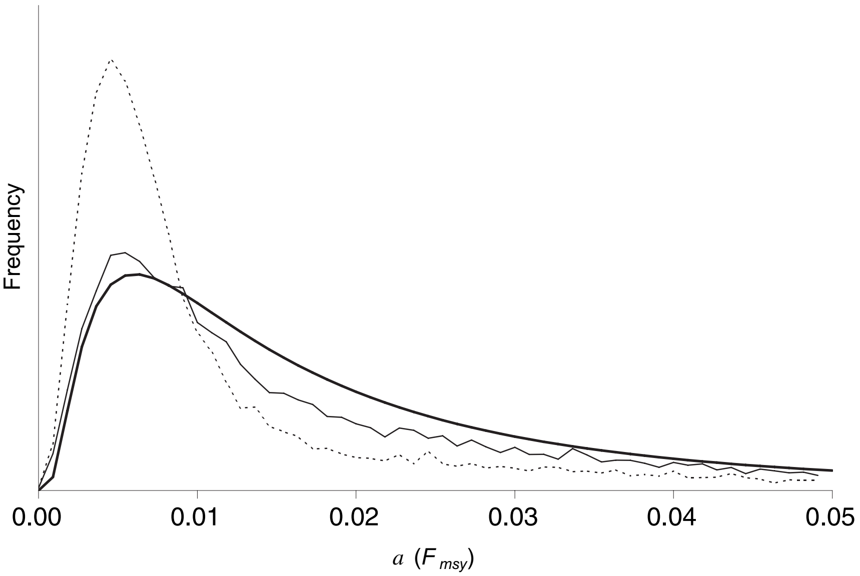

Figure 4. **Prior (bold line) and posterior distributions of Gompertz-Fox parameter a (equivalent to F_{msy}) for shortraker rockfish (light solid line) and rougheye rockfish (dashed line).**

unobserved quantities (parameters and the state variables) would have prior and posterior distributions and observed data would have some likelihood distribution (Meyer and Millar 1999). This approach requires numerical integration over a potentially highly dimensional posterior distribution. While this is computationally feasible, we felt that the added complexity would detract from our main point; that is, how a statistically reasonable approach can be developed for modeling subgroups (individual species) that fall within a management complex consisting of many species. Additionally, we introduced the use of covariances between these species composition observations (an important component of the observation-error part of the state-space model). Although the Kalman filter is somewhat more complicated than models that consider only one type of statistical error, it does provide a streamlined set of equations that allow for a more complete account of the statistical errors in the system, including covariances in the observed data. This could be especially important in emerging fisheries, where limited sampling programs could lead to significant variances in catch data with potential covariances between species.

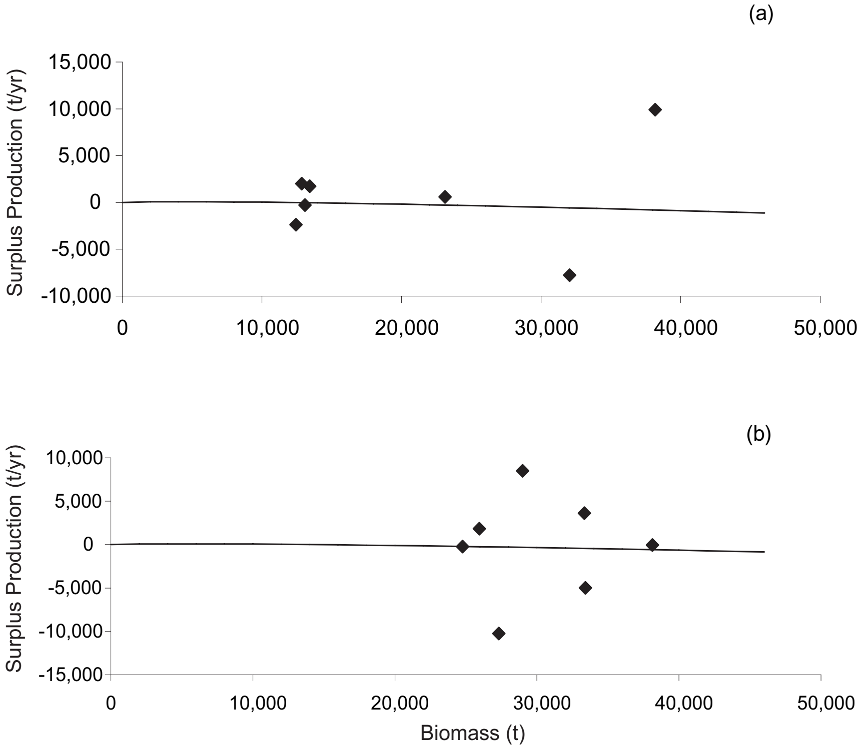

Figure 5. Annual surplus production of rougheye (a) and shortraker (b) rockfish, with estimated Gompertz-Fox production curves (based on maximum likelihood parameter estimates).

If covariances occur between the observed variables, the Kalman filter is, to our knowledge, one of the only ways to incorporate this covariance structure into the parameter estimation procedure. In the case where observer sampling is used to obtain single-species catch levels from aggregated catch data, the covariances of estimated catches is inversely related to the multinomial sample size N (equation 10), emphasizing the relationship between low sample sizes and potentially high covariances of catch estimates. Note that the multinomial sample size N refers to the number of statistically independent samples and is thus likely closer to the number of hauls rather than the number of fish measured (Pennington et al. 2002).

In estimating the variances on catches, the use of $N = 100$ and $\sigma = 0.15$ implies a greater variance on catches than is normally assumed. Even with these catch variances, the model fit the catch data closely, suggesting that the patterns in the survey biomass estimates are not strongly influenced by the catch history. Catch data are often viewed as a census rather than a statistical estimate, and either fit exactly or with very small levels of statistical error. Of course, this assumption may not hold for catch estimates from new and developing fisheries, and variance in catch estimates could have important effects in cases where the catch history is strongly indicative of the pattern in the survey biomass estimates. Shortraker and rougheye rockfish are closely related species, and errors in species identification could contribute to catch variance. Finally, the analysis here considers only statistical errors, but the effect of consistent biases on parameter estimates is an important topic for future investigation.

The Kalman filter can handle various types of statistical errors that occur with the assessment of data-poor stocks, but do not address the underlying causes of such errors; in other words, the errors do not diminish with use of the Kalman filter. Thus, some improvement could be made by increasing the precision of the input data, as the high degree of scatter in the ASP plots is thought to largely reflect sampling variability in survey biomass estimates. Estimating abundance of rockfish from trawl surveys is inherently difficult due to the patchily distributed nature of these species and their association with rough habitats, and reducing the survey sampling variability is an active area of research on Alaskan rockfish. Adaptive sampling programs have proven marginally effective in reducing the variances of biomass estimates for Pacific ocean perch, but not for shortraker and rougheye rockfish (Hanselman et al. 2001). The use of hydroacoustics to infer locations of high density rockfish patches, and the use of this information in real time to influence selection of trawl stations (Everson et al. 1996) is currently being investigated.

The shortraker and rougheye rockfish example can be classified not only as data-poor, but also information-poor, as reasonable estimates of key model parameters could not be inferred from the available data. In such cases, if prior expectations exist regarding the magnitude of these key parameters it is reasonable to use this information. However, these prior distributions must be constructed with care because of their potential influence on the results in information-poor data sets; in this example, the posterior distribution of a for shortraker rockfish was very similar to the prior distribution. The use of Gulland's (1970) familiar rule of $F_{msy} = M$ provides a useful first approximation for a prior on a, but further research could be done on this topic. For rockfish stocks with information on age structure, Bayesian meta-analyses of stock-recruitment productivity parameters have guided interpretation of harvest rates based upon spawner-per-recruit considerations (Dorn 2002). Similar extensions to produce prior distributions of productivity for stocks without substantial age-structured information remain a future research challenge.

Acknowledgments

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Appendix. Derivation of the covariance of log catches

The delta method can be used to calculate the approximate covariance of two functions $g(\boldsymbol{x})$ and $h(\boldsymbol{x})$, where $\boldsymbol{x}$ is a vector of random variables, with the formula (Seber 1982):

$$Cov[g(x),h(x)] \cong \sum_i \sum_j Cov(x_i,x_j)\frac{\partial g}{\partial x_i}\frac{\partial h}{\partial x_j} \tag{A1}$$

In this application, define the following terms:

$$x=\left[p_{RE,i},\ldots,p_{RE,n},p_{SR,i}\ldots,p_{SR,n},C_{A,i},\ldots,C_{A,n}\right]$$

$$C_{RE}=\sum_{i=1}^{n} p_{RE,i}C_{A,i} \qquad C_{SR}=\sum_{i=1}^{n} p_{SR,i}C_{A,i}$$

$$g(x)=\ln(C_{RE}) \qquad h(x)=\ln(C_{SR})$$

The chain rule is applied to obtain the partial derivatives:

$$\frac{\partial h}{\partial x_i}=\frac{\partial h}{\partial C_{SR}}\frac{\partial C_{SR}}{\partial x_i} \qquad \frac{\partial g}{\partial x_i}=\frac{\partial g}{\partial C_{RE}}\frac{\partial C_{RE}}{\partial x_i}$$

Calculation of partial derivatives yields

$$\frac{\partial g}{\partial C_{RE}}=\frac{1}{C_{RE}} \qquad \frac{\partial h}{\partial C_{SR}}=\frac{1}{C_{SR}}$$

$$\frac{\partial C_{RE}}{\partial p_{RE,i}}=C_{A,i} \qquad \frac{\partial C_{SR}}{\partial p_{SR,i}}=C_{A,i}$$

$$\frac{\partial C_{RE}}{\partial p_{SR,i}}=0 \qquad \frac{\partial C_{SR}}{\partial p_{RE,i}}=0$$

$$\frac{\partial C_{RE}}{\partial C_{A,i}}=p_{RE,i} \qquad \frac{\partial C_{SR}}{\partial C_{A,i}}=p_{SR,i}$$

Substitution of these terms into equation A1, assuming that the aggregate complex catches, $C_{A,i}$, and the proportions $p_{RE,i}$, $p_{SR,i}$ are independent of each other, yields

$$Cov[\ln(C_{RE}),\ln(C_{SR})] = \sum_{i=1}^{n}\left[V(C_{A,i})\frac{p_{RE,i}p_{SR,i}}{C_{SR}C_{RE}} + \operatorname{cov}(p_{RE,i}p_{SR,i})\frac{C_{A,i}^{2}}{C_{SR}C_{RE}}\right].$$

Under the assumption that the aggregated catches are lognormally distributed and the proportions are obtained from a multinomial distribution, the covariance is estimated as

$$Cov[\ln(C_{RE}),\ln(C_{SR})] = \sum_{i=1}^{n}\left[e^{2\mu_i+\sigma_i^2}(e^{\sigma_i^2}-1)\frac{p_{RE,i}p_{SR,i}}{C_{SR}C_{RE}} - \frac{p_{RE,i}p_{SR,i}}{N_i}\frac{C_{A,i}^{2}}{C_{SR}C_{RE}}\right]$$

where all random variables are evaluated at their estimated expected values.

Assessing the Information Content of Catch-in-Numbers: A Simulation Comparison of Catch and Effort Data Sets

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Abstract

The fishing industry provides totals for landed catch-in-weight and fishing effort in skippers' logbooks. Because this data-gathering infrastructure is in place, one potentially inexpensive source of additional information could be the catch reported in numbers of individuals landed. The performance of stock assessment models based on three logbook data sets, (1) catch-in-weight and fishing effort, (2) that of (1) plus catch-in-numbers, and (3) catch-in-weight and catch-in-numbers (no effort), was evaluated by means of simulation. Simulated data sets were generated from an individual-based model of a lobster fishery and used to test the ability of these three data sets to estimate recruitment, biomass, population numbers, and exploitation rate. The agreement of estimates from two different delay-difference models with true simulation values were quantified. With perfect knowledge of growth and natural mortality, and under nineteen simulated variations from perfect knowledge, adding catch-in-numbers to the traditional data set of catch-in-weight and effort substantially improved the precision and accuracy of the yearly population estimates.

Adding catch-in-numbers allowed the stock assessment to estimate absolute fishable population size, and track yearly changes in fishable stock size, yielding very large improvements in the estimates of both, with the greatest improvement being in estimation of absolute abundance. We recommend that methods which utilize catch-in-numbers be included in the standard stock assessment toolbox for data-poor species.

Introduction

Most commercial fisheries require skippers to record their landings (in weight) and usually also the fishing effort used to harvest that catch. Thus, a system is in place for fishers to report total catch and effort, and for it to be entered into a database. Programmers maintain, and biologists use, catch and effort data to inform management decision-making. The catch total and catch-per-unit-effort are principal (extensive and intensive) measures of fishery performance and these provide critical data input for most fishery stock assessment methods. If ways can be found to improve the information provided by fishers through logbooks, it could enhance the quality of stock assessment advice for a relatively low additional cost.

In this paper, we investigate the improvement in stock abundance estimates obtained by adding a column to the catch logs for reporting the catch in numbers of individuals landed—to count the catch as well as weigh it. Using only catch and effort totals, the methods evaluated can be used for yearly stock assessment of fisheries with no fishery-independent monitoring programs. The approaches presented may find wide application in new fisheries, or in those with small research budgets. Apart from the logbook totals of effort and catches-in-weight and catches-in-numbers, the methods presented assume prior estimates only of weights-at-age and natural mortality.

Previous research (McGarvey et al. 1997, McGarvey and Matthews 2001) found that notably accurate estimates of yearly recruitment are possible when: (1) data are available for both catch-in-weight and catch-in-numbers, (2) selectivity is independent of age above the age-at-recruitment, and (3) the stock assessment is based on a fully age-structured model. These recruitment estimates are relatively sensitive to the input weights-at-age. Previous research has, however, not considered the ability of adding catch-in-numbers to estimate yearly quantities of management interest such as biomass and exploitation rate.

The improvement in stock assessments made possible by adding information on numbers captured is quantified below by the change in the accuracy and precision of estimates of four annual time-series indicators of interest to fishery managers: recruitment, fishable biomass, fishable population numbers, and exploitation rate. We employ stock assessment methods that take catch totals as input and are simple, namely delay-difference models. These models include biological processes of growth and natural mortality, but do not explicitly represent the age-structure of the population. Fully age-structured models would have performed no better than delay-difference models because no information is given in catch and effort totals about how catchability varies with age. Therefore uniform catchability, implicit in delay-difference models, is assumed.

Two alternative formulations of the delay-difference model (referred to as “DD1” and “DD2”; see appendix) are considered. The DD1 formulation extends the delay-difference model of Fournier and Doonan (1987), which was based on a first-order growth submodel for weight-at-age. The DD2 delay-difference formulation extends the model of Deriso (1980) and Schnute (1985), which was based on a second-order model for weight-at-age. The two stock assessment models also differ in terms of whether estimation is conditioned on catch or effort, whether the errors are assumed to be normal or lognormal, and how first-year recruitment is treated. The principal aim of this paper is not to test the two assessment models but rather to evaluate the benefits of having information on catch in numbers as well as in weight. Using two different models reduces the chance that any observed improvement in the stock assessment estimates is spuriously due to some specific features of a tested model rather than to the added information content of the catch-in-numbers time series.

The performances of the two assessment models are evaluated under three scenarios regarding data availability: (A) information is available on catch-in-weight, catch-in-numbers, and effort; (B) the usual situation where information is available on catch-in-weight and effort; and (C) information is available on catch-in-weight and catch-in-numbers. The differences in results between data sets (A) (denoted “CwCnE”) and (B) (denoted “CwE”) allow the value of having information on catch-in-numbers to be assessed. Examining outputs from the data set using no effort (C) (denoted “CwCn”) tests the ability of catches in weight and numbers alone to permit reliable inferences about stock dynamics. This question is relevant for many data-poor fisheries where only catch-in-weight and an estimate of the mean weight of the catch (which can be used to compute catch-in-numbers) are available, or for other fisheries where there are no reliable effort data, including those for which CPUE does not provide a useful relative measure of abundance.

Methods

An individual-based fishery model (based on South Australian rock lobster, *Jasus edwardsii*; McGarvey et al. 1997) was developed to represent the “truth” for the simulations and as the basis for generating annual data on catch-in-weight, catch-in-numbers, and effort. This model was also used to generate the weights-at-age used by the estimation models (defined as the average weight by age for animals captured in the fishery).

Using an annual model time step, a cohort of 10,000 lobsters recruited to the stock as juveniles each simulation year. Growth followed a von Bertalanffy curve for each animal. The von Bertalanffy parameters κ and L_∞ were fixed among individuals except for one sensitivity test that examined individual variation in growth. The probability of an individual dying in each annual time step is $H + M$, the sum of the probabilities of being harvested H, and of dying of natural causes M. Note that H and M are annual probabilities rather than instantaneous rates. If an individual dies during a time step, the probability that death was by harvesting rather than natural causes is $H/(H + M)$.

The 100 data sets generated by the individual-based operating model each consisted of catch and effort totals for 17 years (nominally 1979 to 1995). The two delay-difference models were applied to each of the three data sets (A, B, and C above) to estimate quantities of management interest: recruitment, biomass, population numbers, and exploitation rate (defined as the proportion of the population harvested annually).

The estimation performance in fitting to each of the 100 replicate data sets in each year for each quantity of interest was quantified by the percentage relative error. Denoting the true and estimated values for quantity I during year t as I_t and $\hat{I}_t$ respectively, the relative error is defined as $100(\hat{I}_t - I_t)/I_t$.

Measures of bias and precision were calculated from the individual relative errors for each Monte Carlo replicate and year. Bias was quantified as the mean or median relative error over 100 simulated data sets. For each quantity of interest and fishery simulation scenario (or “case”), an overall relative error mean (OREM) statistic was calculated as the arithmetic mean of the percentage relative errors over all 100 data sets and all 17 years. Some of the (numerically minimized) estimations did not necessarily converge to the true minimum of the objective function, i.e., the Hessian matrix was not positive definite. Although results from all 100 simulated data sets were included when calculating summary statistics, the number of simulated data sets for which the Hessian matrix was not positive definite is also reported.

To simplify interpretation of the results, a “perfect-knowledge” case was defined (abbreviation “perfect-knowledge” in Table 1). This case included modest variation in annual recruitment, no error about the relationship between effort and exploitation rate, and no systematic differences between the operating model and the delay-difference models.

There are many combinations of operating model specifications and data errors that could be tested. A total of 20 cases, each based on different choices for the specifications of the operating model, were considered (Table 1). Thus, 12,000 estimation runs were summarized (20 cases, 2 models, 100 simulations, and 3 data sets from each). In Table 1, only single changes were made to the specifications of the “perfect-knowledge” case to ease interpretation.

Table 1. Specifications of the 20 cases.

Abbreviation	Recruitment variability CV	Effort error CV	Source of error	M	Recruitment scenario	Growth variability	Exploitation rate H
Perfect-knowledge	0.1	0	None	0.10 yr^{-1}	Yearly random	None	0.4
Error in M (10%)				0.11 yr^{-1}			
Underreporting (10%)			10% underreported				
Growth bias (10%)			Weights-in-age increased 10%				
H vs E; CV = 10%		0.1	Lognormal yearly variation in catchability				
Variable growth	0.6	0.25				10%	
Constant recruitment	n.a.[a]	0.25			Constant		
Spike recruitment	n.a.[a]	0.25			Spike		
Cyclic recruitment	n.a.[a]	0.25			Cyclic		
$\sigma_R = 0.3$	0.3	0.25					
$\sigma_R = 0.6$	0.6	0.25					
$\sigma_R = 1.0$	1	0.25					
$H = 0.05$	0.6	0.25					0.05
$H = 0.1$	0.6	0.25					0.1
$H = 0.8$	0.6	0.25					0.8
$\sigma_E = 0.1$	0.6	0.1					
$\sigma_E = 0.4$	0.6	0.4					
H inc-1	0.6	0.25					Effective effort increasing by 2% yr^{-1}
H inc-2	0.6	0.25					H increased yearly by 5% from 5% to 50%, then flat at 50%
H inc-3	0.6	0.25					H increased yearly by 5% from 5% to 50%, then decreased to 20%

[a] These recruitment patterns were not subject to yearly random variation (with CV = 0.1).

Eight causes (Table 1) of data or model error were considered:

1. Noise about the effort-exploitation rate relationship (quantified by the CV of the lognormal fluctuations in catchability, σ_E).
2. Variability in recruitment about its annual expected value (quantified by the CV of the yearly lognormal fluctuations in recruitment, σ_R).
3. Three temporal patterns of recruitment. The scenarios considered in the simulations include: (a) constant recruitment, (b) "spike," i.e., constant recruitment but with a single very high value (4 times the average) in 1985, and (c) cyclic variation with added noise.
4. Error in the assumed value for *M*.
5. Error in the estimates of weight-at-age.
6. Underreporting of catch and effort totals.
7. Individual variability in growth.
8. Differing levels of average exploitation rate and its trend over time.

Results

Near perfect conditions

The two assessment models yielded near-perfect estimates of all four quantities for the "perfect-knowledge" case (differing by no more than 1-2% from the "true" values) when information is available on catch-in-weight, catch-in-numbers, and effort (Table 2, Figs. 1 and 2). The DD1 estimates were marginally better than the DD2 estimates, with DD1 resulting in values of OREM of <0.1% compared to <1% for DD2. The fits to the data used by the assessment model (not shown) for this case were also nearly perfect (e.g., to within 5-6 significant places for DD1). This case is clearly unrealistic because inter alia natural mortality was known exactly, there were no errors in reported catches, no catchability variability, and no individual variation in growth. However, the case of near-perfect agreement serves as an intuitively useful origin or neutral basis for comparison when assessing the output estimation errors under more realistic scenarios. For example, if the relative error were 5% for a particular data or model error, we would know that this must have been due to the source of error considered in that case. If that error were removed, the perfect-knowledge case is again obtained. When the full CwCnE data set was used, the perfect-knowledge case had a relative error near zero for all four time series of management interest.

The results for the remaining cases in Table 2 using the three-time-series CwCnE data set suggest that error about the effort-exploitation rate

Table 2. Performances of the three data sets under perfect or 10%-error conditions from the two assessment models.[a]

	% Hessian[b]		Recruitment		Biomass		Numbers		Exploitation rate	
Case description	DD1	DD2	DD1	DD2	DD1	DD2	DD1	DD2	DD1	DD2
Data set CwCnE										
Perfect knowledge	100	100	0.0	0.1	0.0	0.3	0.1	0.3	–0.1	–0.3
Error in *M* (10%)	100	99	–2.4	–2.4	–2.4	–2.1	–2.4	–2.1	2.4	2.2
Underreporting (10%)	100	100	–10.0	–9.9	–9.9	–9.7	–9.9	–9.7	–0.1	–0.3
Growth bias (10%)	100	100	–3.4	–3.5	–17.9	–17.8	–17.9	–17.8	21.8	21.7
H vs *E*; CV = 10%	100	99	–0.2	–0.1	–2.0	0.1	–1.1	0.2	2.4	–0.1
Data set CwE										
Perfect knowledge	2	79	10.2	213.1	–25.8	818.2	–14.1	621.2	35.3	–53.7
Error in *M* (10%)	1	75	9.9	74.9	–32.0	727.6	–18.9	521.9	47.7	–55.3
Underreporting (10%)	10	81	10.5	–17.9	–51.3	112.4	–34.5	43.3	85.7	–47.8
Growth bias (10%)	15	85	10.5	–17.7	–44.7	133.9	–33.3	44.6	81.6	–48.6
H vs *E*; CV = 10%	5	78	323.6	504.3	5,431.2	1,272.4	1,981.6	822.3	–12.6	–4.9
Data set CwCn[c]										
Perfect knowledge	100	85	2.5	2,764.6	2.5	1,755.2	2.5	1,735.0	–2.2	–5.2
Error in *M* (10%)	100	78	0.4	6,639.4	0.4	4,317.1	0.5	4,229.1	–0.1	–17.8
Underreporting (10%)	100	83	–4.9	6,443.5	–4.9	4,127.9	–4.8	4,075.5	–4.5	–21.0
Growth bias (10%)	100	80	–0.9	11,040.9	–15.7	6,652.8	–15.6	6,584.3	19.0	–17.8
H vs *E*; CV = 10%	100	89	2.4	2,207.8	2.4	1,381.4	2.4	1,363.2	–2.1	–3.1

[a] See Table 1 for the detailed specifications of the cases.

[b] Percentage of 100 estimations that successfully converged to a positive definite Hessian matrix.

[c] Model DD1 was modified to handle the CwCn data set; see appendix.

Values in the last eight columns give overall relative error means (OREM) expressed as a percentage, which quantify the observed deviation from true for estimates of the four quantities of management interest. OREM is the relative error averaged over the 100 simulated data sets fitted for each case.

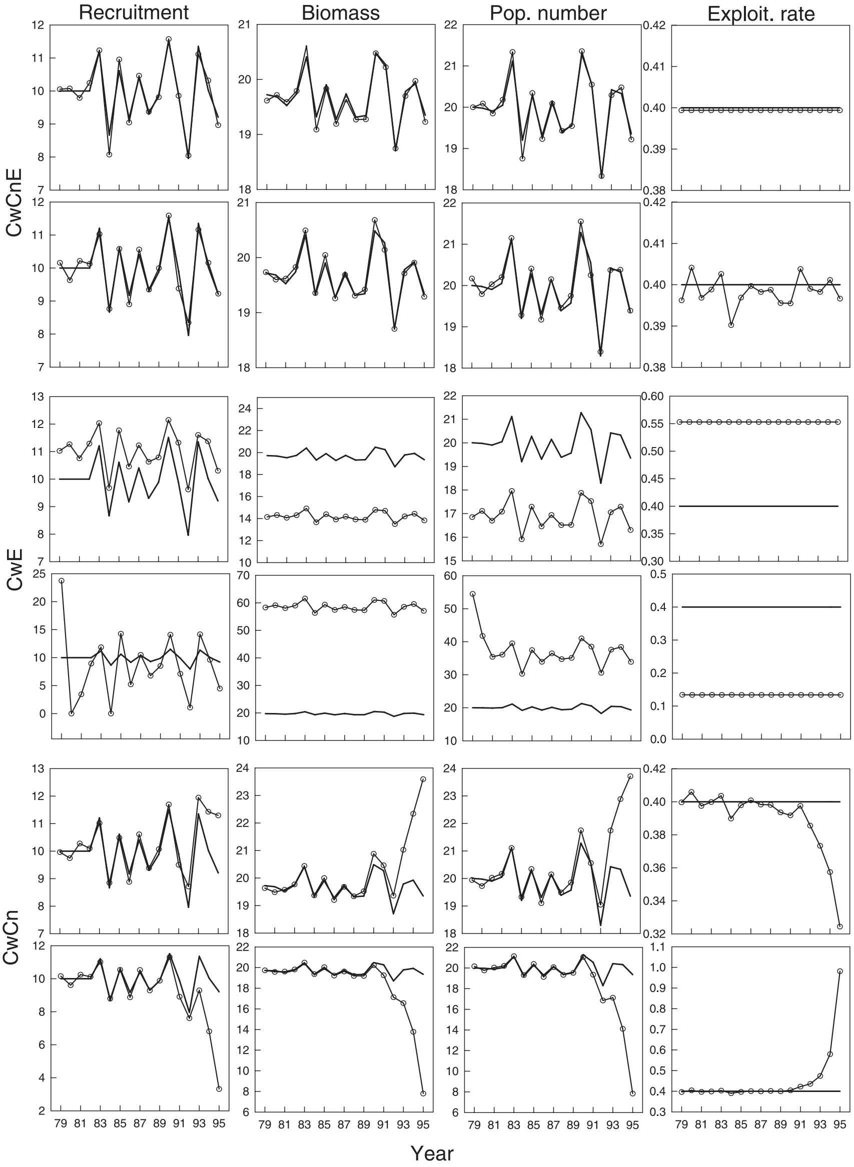

Figure 1. Model estimates compared with true simulated values from one replicate of the "perfect-knowledge" case for the four quantities of management interest, three catch and effort data sets, and two assessment models. The true values are shown by the solid lines and the estimates by the open circles (model DD1, rows 1, 3, and 5; model DD2 rows 2, 4, and 6).

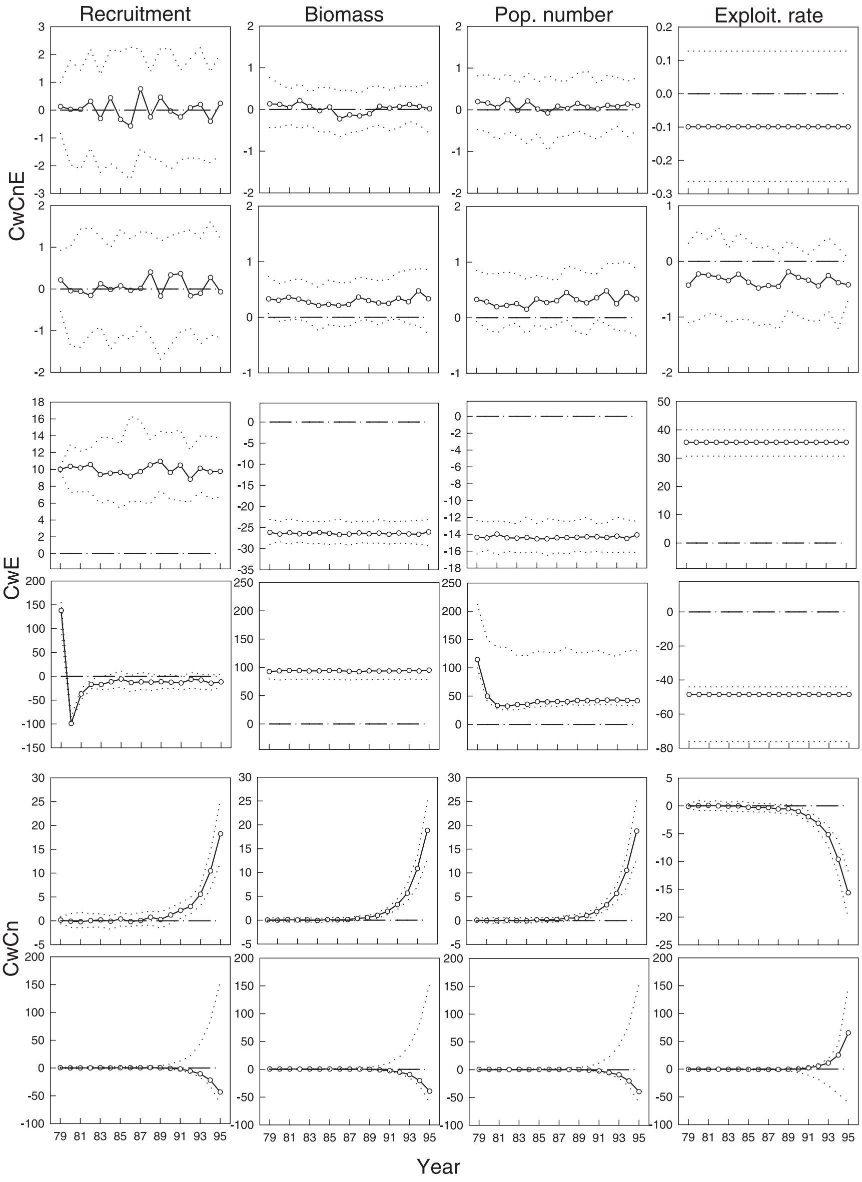

Figure 2. **Time-series of percentage relative errors based on 100 replicates of the "perfect-knowledge" case for four quantities of management interest, three data sets, and two assessment models. The open circles represent the medians of the errors for the year concerned while the dotted lines show the 25% and 75% quantiles (model DD1, rows 1, 3, and 5; model DD2 rows 2, 4, and 6).**

relationship of 10% and an error of 10% in *M* are largely inconsequential in terms of bias (values for OREM <5%). The main impact of error in the effort-exploitation rate relationship was that the precision of the two assessment models deteriorated markedly (wider inter-quartile intervals; dotted lines of Fig. 3). The size of the effect was different for the two assessment models with DD2 being very precise until year 15 when the results become considerably less precise and DD1 being equally imprecise for all years. It is not surprising that DD1 and DD2 differ in terms of how they respond to error in the effort-exploitation rate relationship because DD1 assumes this relationship to be exact whereas DD2 allows for error in this relationship.

In contrast, with the CwCnE data set, the impact of catches (and effort) being underreported by 10% led to corresponding underestimates of recruitment, biomass and numbers of very nearly the same 10%, while for exploitation rate this bias cancels out, yielding essentially no estimation error (Table 2). Underreporting did not, however, impact the ability to track yearly changes in biomass, recruitment, and numbers as the impact of the 10% underestimation bias was the same in all years (not shown graphically).

Of all the sensitivity tests considered for the CwCnE data set (Table 2), overestimating weight-at-age by 10% had the largest impact. Specifically, biomass and numbers were underestimated by close to 20% (Fig. 4) and exploitation rate was overestimated by slightly more than 20%.

Impact of excluding catch-in-number or effort data

Even under the "perfect-knowledge" case, removing information on catch-in-numbers led to substantial errors, with nearly all values of OREM larger than 10%, sometimes much larger (data set CwE in Table 2). The DD1 model outperformed the DD2 model for all cases except for "*H* vs *E*; CV = 10%" in terms of OREM (Table 2), probably because it (here correctly) assumed that effort is exactly proportional to exploitation rate.

Both DD1 and DD2 are generally accurate for the CwCn data set until the last 4-5 years when the estimates diverge from the true values (Figs. 2, 3, and 4). Additional analyses (not shown) based on a 37-year rather than a 17-year data set confirm that the behavior evident in Figs. 2-4 for the CwCn data set is an "end effect" because the divergence from the truth occurs in the last 4-5 years for the 37-year as well as for the 17-year data set. This type of behavior is consistent with how the results from virtual population analysis behave in the absence of effort data (e.g., Butterworth et al. 1990). For the assessment models analyzed above, having information on the catch-in-numbers as well as on catch-in-weight performed better than having effort in addition to catch-in-weight. The version of the DD1 model which had been modified to fit to a data set with no effort (see appendix) performed markedly better for the CwCn data set than

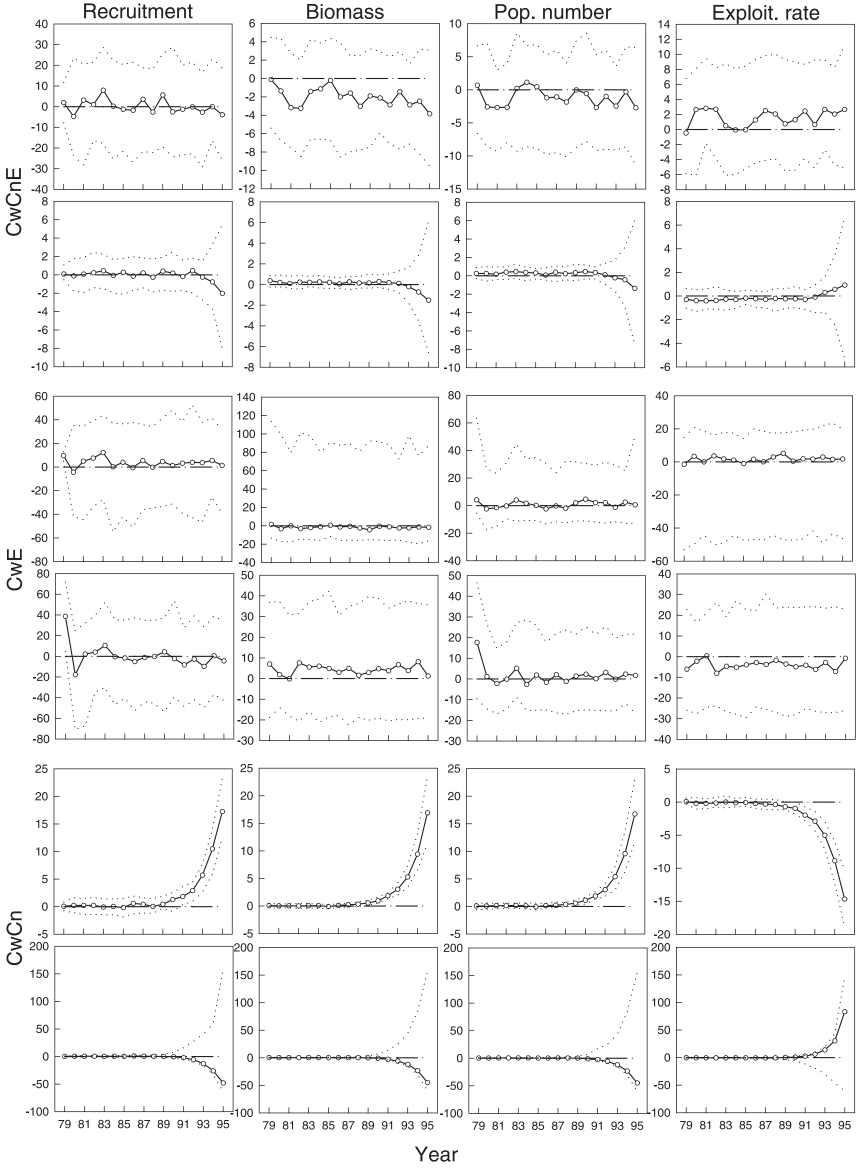

Figure 3. **Time-series of percentage relative errors based on 100 replicates of the "*H* vs *E*; CV = 10%" case for four quantities of management interest, three data sets, and two assessment models. The open circles represent the medians of the errors for the year concerned while the dotted lines show the 25% and 75% quantiles (model DD1, rows 1, 3, and 5; model DD2 rows 2, 4, and 6).**

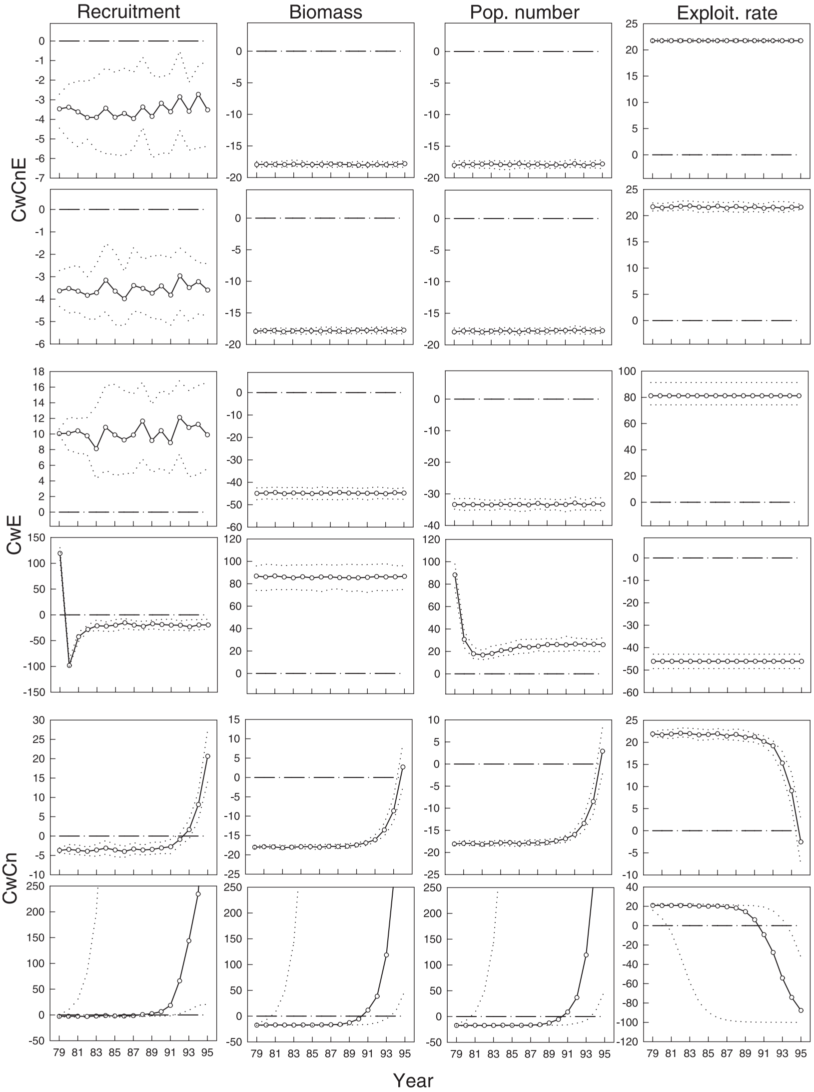

Figure 4. Time-series of percentage relative errors based on 100 replicates of the "growth bias (10%)" case for four quantities of management interest, three data sets, and two assessment models. The open circles represent the medians of the errors for the year concerned while the dotted lines show the 25% and 75% quantiles (model DD1, rows 1, 3, and 5; model DD2 rows 2, 4, and 6).

the standard DD1 model for the CwE data set, frequently yielding OREM values <10%.

The less reliable convergence of the CwCn data set and particularly of the CwE data set is reflected by the lower (much lower for DD1) percentage of simulations for which the Hessian matrix was positive definite (Table 2).

Other sensitivity tests

It is clear from Table 2 and Figs. 1-4 that the performances of the assessment models are improved substantially by having information on catch-in-numbers as well as on catch-in-weight and effort. However, the scenarios considered in Table 2 are not particularly challenging. Table 3 therefore investigates the sensitivity of the two assessment models for the CwCnE data set to a wider range of errors (see Table 1 for details).

In general, both DD1 and DD2 perform adequately for the cases in Table 3. The only cases where performance (as measured by OREM) was notably poor are when the variability in recruitment is high ($\sigma_R = 1.0$), the noise about the effort-exploitation rate relationship is high ($\sigma_E = 0.4$), and when the exploitation rate equals the two lowest values considered ($H = 0.05$ and $H = 0.1$). The poorer performance for the latter two cases is hardly surprising. Increasing σ_E from 0.25 to 0.4 reduces the information content of the effort data, while it is well-known (e.g., Hilborn 1979) that it is difficult to estimate quantities of management interest from data sets in which the exploitation rate is low and the stock size consequently does not change much over time, except due to exogenous variation in recruitment (i.e., there is a lack of "contrast" in exploitation level). Error in effort data does not of course affect estimates based on the CwCn data set.

The DD2 model outperformed the DD1 model in terms of OREM for the bulk of the cases (Table 3) and was also much more precise (except over the last few years) (Fig. 5). The likely reason for superior performance of the DD2 model for the cases in Table 3 is that there was noise about the true effort-exploitation relationship for all of these cases.

Discussion

Previous simulation studies evaluating methods of stock assessment have examined the relative merits of different choices of model formulation. The objective of this study was instead to determine which of several data sets yielded better estimates, and by how much. Thus, the information content of different data sets that might be available for data-poor fisheries, rather than the power of different estimators, is assessed. For that reason, to enhance confidence that the results were robust, two different stock assessment models were examined. The results were not ambiguous—in both precision and accuracy, very large improvements

Table 3. Sensitivity of the estimates based on the CwCnE data set to a wide range of data and model errors.[a]

	% Hessian[b]		Recruitment		Biomass		Numbers		Exploitation rate	
Case description	DD1	DD2	DD1	DD2	DD1	DD2	DD1	DD2	DD1	DD2
Recruitment scenarios										
Constant recruitment	92	100	0.9	1.0	−1.7	1.2	−0.4	1.2	8.9	−0.7
Spike recruitment	99	100	0.7	0.4	−10.0	0.4	−5.5	0.4	13.9	0.1
Cyclic recruitment	99	100	0.0	0.6	−8.9	0.7	−4.9	0.7	12.8	−0.2
$\sigma_R = 0.3$	98	99	1.1	0.8	−10.8	0.9	−6.1	0.9	14.7	−0.4
$\sigma_R = 0.6$	99	100	0.6	0.4	−6.6	0.5	−3.6	0.6	12.8	0.1
$\sigma_R = 1.0$	95	100	5.4	1.1	−11.1	0.9	−6.4	0.9	16.5	−0.4
Effort scenarios										
$H = 0.05$	100	100	−25.4	−10.8	−45.8	−19.8	−41.5	−19.8	89.0	27.9
$H = 0.1$	100	100	−2.1	−2.4	−29.6	−7.8	−24.3	−7.8	43.8	9.5
$H = 0.8$	99	89	2.0	−0.6	6.9	2.0	2.3	0.4	8.3	−0.7
$\sigma_E = 0.1$	100	100	−0.3	0.2	−1.8	0.4	−1.0	0.4	2.2	−0.3
$\sigma_E = 0.4$	99	100	5.6	1.5	−15.9	1.1	−9.0	1.2	30.8	0.2
H inc-1	100	99	2.9	4.1	−6.9	4.8	−2.5	4.8	11.6	−3.7
H inc-2	99	100	5.8	2.9	−11.6	−0.1	−4.4	−0.1	12.8	0.7
H inc-3	98	100	3.8	3.8	−11.1	−1.0	−4.6	−1.0	13.0	1.5
Variable growth	100	99	4.6	4.9	23.8	26.1	23.8	26.2	−18.8	−20.2

[a] See Table 1 for detailed specifications of the cases.

[b] Percentage of 100 estimations that successfully converged to a positive definite Hessian matrix.

Values in the last eight columns are overall relative error means (OREM), expressed as a percentage, of estimates of the four quantities of management interest from the two assessment models.

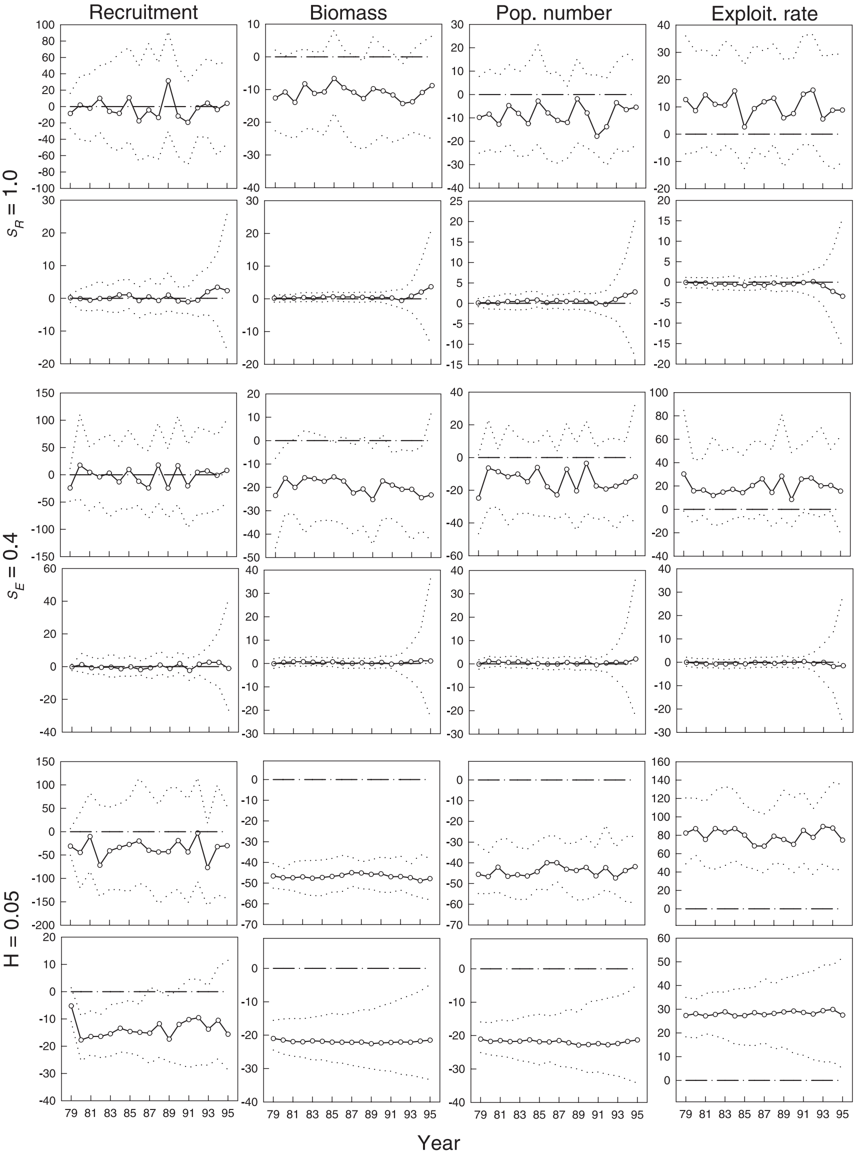

Figure 5. Time-series of percentage relative errors based on 100 replicates for three of the cases in Table 3 for four quantities of management interest, and two assessment models. The results in this figure are based on the CwCnE data set. The open circles represent the medians of the errors for the year concerned while the dotted lines show the 25% and 75% quantiles (model DD1, rows 1, 3, and 5; model DD2 rows 2, 4, and 6).

in performance were obtained from both models by including catch-in-numbers along with catch-in-weight and effort data in the assessment.

A question that arises is why adding catch-in-numbers can improve performance so markedly. The answer is that having catch-in-numbers as well as catch-in-weight gives a highly precise (100% sample) measure of the mean weight of an individual animal in the catch. Annual mean weight in the catch, combined with estimated mean-weights-at-age, in turn permit inference on the mean age of the population above the age-at-recruitment. Beverton and Holt (1956) first showed that a measure of the mean length of legal-size fish, together with growth parameters, was sufficient to estimate total mortality. Therefore, information on the size of animals in the catch, together with assumed values for natural mortality and growth, allow the exploitation rate (H), here quantified as the yearly fraction harvested, to be estimated. Absolute fishable numbers are inferred from the estimate of exploitation rate (as Cn/H).

A surprising result is that, even with no effort data, information on catch-in-numbers and catch-in-weight alone permit estimates of both absolute abundance and relative changes in abundance, for some of the simulations. This performance outcome was more evident for the DD1 model (Table 2, data set CwCn). The second version of DD1, which freely estimated all yearly values of R and H, was used to analyze CwCn. Thus, under at least some model formulations, and for all but the last few years of the time series, it is possible to track yearly changes in biomass accurately using only catch totals given in weight and in numbers.

How a time series of annual mean weights can shed light on the population dynamics of a population and why precision decreases markedly in the last few estimation years with the CwCn data set can be illustrated by considering the example of a drop in mean weight. Mean weight could decline due to increased recruitment (of younger thus lighter fish) or increased fishing mortality (gradually removing more older fish). However, the subsequent time trends under these two scenarios differ. If the drop in mean weight is due to increased recruitment, then mean weight should subsequently rise fairly quickly as the pulse of recruited fish grows. In contrast, a rise in mean weight is not expected when the drop in mean weight is due to increased levels of exploitation. In the last few years of a data time series, the subsequent effect of a change in mean weight is not yet observed and this ambiguity between possible changes in recruitment or in exploitation level remains unresolved. This interpretation would explain the observed divergence in estimation precision at the end of the time series for the CwCn data sets, when no measure of effort was available.

The ambiguity between increased recruitment and increased fishing mortality is, however, resolved with the addition of effort data, which brings an index of abundance as CPUE. Because changes in recruitment and fishing mortality alter exploitable population size in opposite directions, they predict opposite effects for change in CPUE. For example, in the case where mean weight dropped, if in addition CPUE was observed to rise, the model would infer an increase in recruitment rather than higher exploitation.

For most cases and for years prior to the last few years of the fishery time series, analyses based on the CwCn data set outperformed those based on the CwE data set. It therefore appears that, in cases where effort data are unreliable as indicators of exploitation rate, it should nevertheless be possible to carry out a preliminary assessment without recourse to (or in addition to) fishery-independent surveys, which are generally costly to conduct. When CPUE is accepted as an informative indicator of change in stock size, basing an assessment on catch-in-weight and catch-in-numbers can provide an auxiliary check on CPUE as a measure of abundance, at least for the earlier years of the assessment period.

The applicability of including catch-in-numbers in assessments is limited mainly by the practical feasibility of counting the catch. For species where the numbers landed by a boat usually do not exceed a few hundred (e.g., high-value species like lobsters and abalone), or for large organisms (e.g., sharks), it should be feasible to obtain a complete count of the catch. This information is already being collected in some fisheries. For example, in South Australia, the numbers of rock lobster landed have been reported in logbooks since 1970, originally for compliance reasons.

When a complete census of catch numbers is not feasible, as is the case for most net fisheries, onshore sampling of landed bins provides an alternative method for estimating mean weight. If researchers, or appointed representatives who could probably be fishers or processors, were able to count all the fish in a sample of bins, and if each of these bins was weighed individually at weigh-in, this sample estimate of mean weight of the catch would contain most of the information in a total count of numbers landed. Similarly, if not all fishers agreed to report numbers captured, as long as this decision was not correlated with the size of animals in their catch, the mean weight obtained from participating fishers would usually provide a sufficiently precise measure of mean weight.

DD1 and DD2 differ in several ways (see appendix). Perhaps the most important of these relates to whether the model is conditioned on catch or on effort. The results in Table 3 suggest strongly that conditioning on catch leads to less bias when there is considerable uncertainty in the relationship between effort and exploitation rate. Essentially the same conclusion was drawn by Polacheck et al. (1993) for assessments based on biomass dynamic models. Although models conditioned on effort have been popular historically (e.g., Schnute, 1977, Hilborn 1979, Fournier and Warburton 1989), the recent trend is toward conditioning on catch (e.g., Methot 1993, Prager, 1994). Reasons for this include the results of simulation studies such as that of Polacheck et al. (1993) and the ability to deal with situations in which effort (but not catch) is missing for some years. For cases where yearly effort did provide a reliable index of fishing mortality, the effort-conditioned model (DD1) was more accurate and precise. Although beyond the scope of this study, methods that allow for error in both catch and effort (for example, by using a Kalman filter) may perform better than either DD1 or DD2.

In conclusion, the sensitivity analyses suggest that assessments based on catch-in-numbers and catch-in-weight as well as effort data provide estimates with (relatively) low bias and (relatively) high precision and that are quite robust to a wide range of fishery errors. The main sources of error (as reflected by the values for OREM in Tables 2 and 3) are errors in weight-at-age, high variability in recruitment and catchability, and (particularly) low exploitation rate. The data requirements for the assessment models considered in this paper are low and we recommend that methods such as those considered in this paper be included in the standard toolbox for data-poor species.

Acknowledgments

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Appendix: Estimation models

Two models ("DD1" and "DD2"), respectively first- and second-order delay-difference models with annual time steps, are considered. Both models represent the population in two stages: recruits to the fishable stock and all older ages combined. Both make predictions of the annual recruitment, the annual exploitation rate, the annual catch-in-weight, and the annual catch-in-numbers.

Model variables are as follows:

R_t the number of recruits at the start of year t,

N_t the total number in the fishable population at the start of year t,

B_t the fishable biomass at the start of year t,

H_t the exploitation rate (fraction harvested) during year t,

$\hat{C}_t^n$ the model-predicted catch-in-numbers during year t, $\hat{C}_t^n = N_t H_t$,

$\hat{C}_t^w$ the model-predicted catch-in-weight during year t, $\hat{C}_t^w = B_t H_t$,

C_t^n the observed catch-in-numbers during year t,

C_t^w the observed catch-in-weight during year t, and

λ_t the survival rate during year t

where $\lambda_t = 1 - M - H_t$, and $M = 0.1$, the constant yearly discrete natural mortality proportion.

The DD1 and DD2 models were both implemented using the AD Model Builder package (http://otter-rsch.com/admodel.htm). The two models differ in terms of whether the dynamics are modeled using a first-order or a second-order delay-difference model, how the annual exploitation rate is defined, the error model assumed when fitting the data, and how the recruitment for the year before the first year for which catches are available is defined.

First-order delay-difference model

The first-order delay-difference model (DD1) uses the following first-order delay-difference equations for numbers and biomass:

$$N_{t+1} = N_t \lambda_t + R_{t+1} \tag{A.1}$$

$$B_{t+1} = \alpha \lambda_t N_t + \beta \lambda_t B_t + R_{t+1} w_r . \tag{A.2}$$

The first-order dynamic equations for N and B were derived from the first-order weight-difference equation describing growth, in similar fashion to Fournier and Doonan (1987) who fitted to catches by weight and effort, and a sample of mean length:

$$w_a = \alpha + \beta w_{a-1}. \tag{A.3}$$

The value for w_r (the mean weight of a recruit) was set to the observed mean weight of recruits, while the values for the parameters β and α were determined by a weighted linear regression of equation A.3 with weightings by age (a) of $(0.5)^{a-r}$. The lower weighting of older animals (this weighting declining by 50% for each year of increased age) reflects the fact that there would be fewer older animals in the catch. The 0.5 factor is based on the average decline in abundance with age for the "perfect-knowledge" case. Thus, a better fit of mean weight-at-age is obtained for the younger fish that are correspondingly more numerous in the reported catch totals. This weighting scheme improved the ability of the stock assessment models to estimate the true operating model values for the "perfect-knowledge" case.

The annual exploitation rates for the DD1 model were calculated by multiplying the annual reported efforts by catchability, $H_t = qE_t$, i.e., model DD1 is "conditioned on effort." This approach assumes that catchability does not vary over time and effort is measured without error. The remaining parameters of the population dynamics model are the initial biomass and the initial population numbers (B_0 and N_0 respectively), the (assumed steady-state) exploitation rate for years prior to data (H_0), and the recruitments for years 1 to m (where m is the number of years of catch data in the logbook time series). The values for N_0 and B_0 are computed from an estimate of the numbers in the first age-class, R_0 (R_0 is assumed to be the same as R_1), and an estimate of the survival rate for the years prior to the first year for which catches are available $[1 - (M + H_0)]$.

A negative log-likelihood involving catch-in-weight and catch-in-numbers data (because the model mimics the effort data exactly) and assuming normal errors, is minimized to find the estimates for the model parameters:

$$-\ell n(L) = m\ell n(\sigma_n) + m\ell n(\sigma_w) + \frac{\sum_{t=1}^{m}\left(C_t^n - \hat{C}_t^n\right)^2}{2\sigma_n^2} + \frac{\sum_{t=1}^{m}\left(C_t^w - \hat{C}_t^w\right)^2}{2\sigma_w^2}. \tag{A.4}$$

The observation error standard deviations, σ_n and σ_w are assumed to be proportional to the mean catches by number and weight in the data, i.e., $\sigma_n = \sigma_c \bar{C}^n$ and $\sigma_w = \sigma_c \bar{C}^w$, where σ_c is freely estimated.

When the data for catch-in-numbers are not used (i.e., data set CwE), the DD1 model is implemented by omitting the term related to catch-in-numbers from equation A.4. However, in the absence of effort data (i.e., data set CwCn) the model requires further modification because it is not possible to set the exploitation rate equal to qE_t. Therefore, for data set CwCn the annual exploitation rates are treated as estimable parameters, H_0 is set equal to H_1, and R_0 is set to equal to R_1 and the objective function minimized is

$$O=\sum_{t=1}^{m}\left(\frac{C_t^n-\hat{C}_t^n}{\hat{C}_t^n}\right)^2+\sum_{t=1}^{m}\left(\frac{C_t^w-\hat{C}_t^w}{\hat{C}_t^w}\right)^2. \tag{A.5}$$

Equation A.5 has 2 m terms while there are also 2 m parameters (the m recruitments and m exploitation rates). Thus, there is a unique set of values for the model parameters that minimize equation A.5.

Second-order delay-difference model

The second-order delay-difference model (DD2) is based on the following second-order delay difference equations for numbers and biomass (Deriso 1980, Schnute 1985):

$$B_{t+1}=(1+\rho)\lambda_t B_t-\rho\lambda_t\lambda_{t-1}B_{t-1}+w_r R_{t+1}-\rho\lambda_{t-1}w_{r-1}R_t \tag{A.6a}$$

$$N_{t+1}=N_t\lambda_t+R_t+1. \tag{A.6b}$$

The growth equation for DD2 is also second-order (Deriso 1980, Schnute 1985):

$$w_{a+1}=(1+\rho)w_a-\rho w_{a-1} \tag{A.7}$$

where w_a is the mean weight of a fish of age a, ρ is the Brody growth coefficient, and the values for ρ, w_{r-1} and w_r are determined by weighted linear regression of equation A.7 using the same age-specific weightings as DD1.

The values for the annual survival rates are calculated using the equation $\lambda_t = (M + C_t^w/B_t)$, i.e., model DD2 is "conditioned on catch" (i.e., model DD2 is based on the assumption that the annual catches-in-weight are measured without error). The remaining parameters of the population dynamics model are R_0 (from which N_0 and B_0 can be calculated), H_0, and recruitments for years 1 to m. Unlike model DD1, R_0 is treated as a free parameter of the model given the assumption $B_{-1} = B_0$.

The negative log likelihood minimized to estimate the model parameters contains terms related to the effort and catch-in-numbers data (because the model mimics the catch-in-weight data exactly) and assumes lognormal error:

$$-\ell\text{n}L = m\,\ell\text{n}(\sigma'_q) + m\,\ell\text{n}(\sigma'_q) + \frac{\sum_{t=1}^{m}\left(\ell\text{n}E_t - \ell\text{n}(qH_t)\right)^2}{2\sigma'^2_q} + \frac{\sum_{t=1}^{m}\left(\ell\text{n}(C^n_t) - \ell\text{n}(\hat{C}^n_t)\right)^2}{2\sigma'^2_n}. \tag{A.8}$$

The square of the observation error coefficient of variation for the catch-in-numbers data, σ'^2_v is assumed to be 5% of that for the effort data, σ'^2_q. Assuming that $\sigma^2_n = 0.05\sigma^2_q$ forces the model to give much more emphasis to fitting the catch-in-numbers data.

The only modifications to the DD2 model needed to handle situations in which no effort or no catch-in-numbers data are available involves removing the terms related to the data source concerned from the likelihood function (equation A.8).

FISMO: A Readily Generalized Fisheries Simulation

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Abstract

Assessing tropical fisheries is often problematic due to the fact that the population parameters of natural stocks are poorly understood or unknown. Even when there is evidence of stock depletion (e.g., reduced catch and declining abundance) occurring within the fishery, accurate diagnosis and assessment of actual stock condition is difficult or impossible. Therefore, a semi-automated, age-structured simulation model (FISMO) was developed to allow reconstruction of the age structure and population size of a fishery to assess the stock and some important reference points useful for decision-making and management. The model carries out assessments using parameter values of targeted, or similar, stocks that are available in FishBase and catch records from the FAO Web site. Reconstruction of cohorts is made as numbers and biomass per age group. The number of age groups explicitly considered is limited to 20. Population parameter values (growth rate, length-weight, age at maturity, age at first catch [t_c], catchability), total catch value, costs of fishing operations, number of fishing days per year, fleet size, and 15 years of catch data are used as input. The model also estimates natural mortality, longevity, ϕ', asymptotic weight, the benefit/cost ratio, and total profits. The stock-recruitment relationship is determined so that simulated cohorts can be linked over time and, using the yield equation, the model estimates fishing mortality (F) for each year. Yield and profits are simulated by assuming that t_c and fishing gear selectivity patterns have remained unchanged over time. Graphic displays allow identification of biological and economic optimum yields as a function of F and t_c. The model fits recorded catch, and then simulated yields and profits as functions of F and t_c (both chosen by the user) are model outputs and can be used as an

interactive management tool. This model version can be applied to most exploited stocks and can be modified for particular cases.

Introduction

Fish stock assessment models have become a common tool for fishery biologists. Now that more fish stocks fall into the categories of depleted and overexploited, estimating abundance-at-age through modeling to explore management alternatives is a permanent concern (Ludwig et al. 1993, Jackson et al. 2001, Pauly et al. 2002). This is an incentive for the development of more and better methods to minimize human impact on fisheries resources and to guarantee their sustainable exploitation.

Based on this premise, a simulation model (FISMO) was developed in an Excel spreadsheet file. Its basic structure was briefly described by Chávez (1996), based on principles contained in more comprehensive literature (Megrey 1989, Hilborn and Walters 1992, ICES 1993, Funk et al. 1998), and after estimates taken from other published sources such as FishBase (Froese and Pauly 1997, 1998). It runs originally with data belonging to another fishery that serves as a template. To fit data from a study case to the model, parameter values and statistical records of the new problem of study are required from independent estimates. These estimates can be based on analysis of length-frequency samples, e.g., using Fisat (Gayanilo et al. 1996).

Population parameters

Parameter values to be used as inputs and other parameters estimated by the program are in Table 1.

Inputs

Inputs are region or area of the fishery; scientific name; commercial name of the exploited stock; parameter of the von Bertalanffy growth model (vBgm); the rate parameter K; asymptotic length L (parameter of the vBgm); age equivalent to zero length or t_0 (parameter of the vBgm); condition factor or coefficient a of the length-weight relationship (aL^b); exponent b of aL^b, or isometric parameter, approaching 3; catchability coefficient q (used here only to fit the benefit/cost ratio); value per kg of fish landed; cost per daily trip of fishing operations of a typical or normal fishing boat and an approximate number of fishing days per season; and a recent estimate of the number of fishing boats. With any new value, growth curves in length and weight are immediately displayed in the first screen, as shown in Fig. 1.

Parameter estimates from the program are: asymptotic weight W (parameter of the vBgm); longevity, estimated as $3/K$; instantaneous rate of natural mortality M, as a Beverton invariant, as 1.5 K (Jensen 1996,

Table 1. Input data required and first estimated parameter values.

Region of fishery	Baja California
Scientific name	*Panulirus interruptus*
Commercial name	Spiny lobster
Bertalanffy	
$K =$	0.1164 per year
$L =$	64 cm
$W =$	5,124 g
$T_0 =$	–0.059
$a =$	0.024 (length-weight)
$b =$	2.95 (length-weight)
Age of 1st catch	4 years
Maturity age	5 years
Longevity	26 years
$M =$	0.1746 per year
$\phi' =$	2.7
Catchability	0.00001532
Value/kg	3.7 US$ (2001)
Cost/day	63 US$ (2001)
Boats	1,250 (estimate)
Benefit/cost	8.67
Trips	2,528 (last year)
Profits	3,341,445 US$ (1993)

W, natural mortality (M), longevity, ϕ', benefit/cost, and profits are estimated by the program. Other values are model inputs. In the same screen, a graph (Fig. 1) describes growth rate in length and weight of the stock, whose tendency can be modified by the user changing the von Bertalanffy growth parameters (L, K, t_0), and the values of A and B of the length-weight relationship.

1997); ϕ' ($\phi' = \log K + 2 \log L$), a length-based index of growth performance (Munro and Pauly 1983); and benefit/cost ratio of current fishing operations.

Matching statistical records with age structure

Statistical catch records are shown in Table 2. The model works with fifteen years of catch records. Annual data is substituted first. The model estimates catch and corresponding F values. This is done with a macro, and a graph with stock and exploited biomass, as well as recorded and estimated yield (Fig. 2), shows these results. A macro is an Excel subroutine,

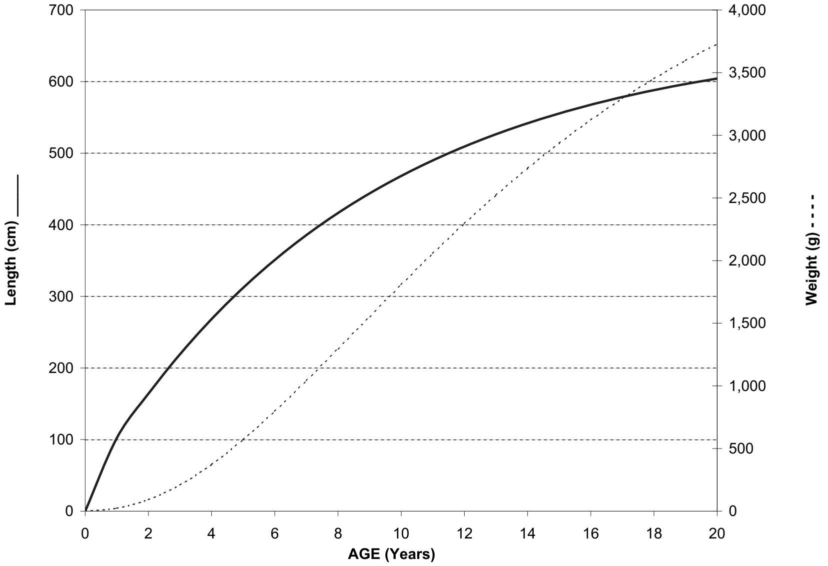

Figure 1. Growth curves in length and weight of the study case chosen. By changing growth parameter values, the graph immediately displays the new tendencies.

usually in Quick Basic, seen only through the tools heading that the Excel user defines with a few instructions. It is saved within the structure of the file, and can be invoked each time it is required through the model fitting process, with a key defined as a virtual button on the screen.

Key points

- *The highest yield.* From the catch-data series, the maximum value is chosen. This pivotal value will set the highest recruitment level and is associated with the maximum catch allowed by the model after the potential yield is estimated. It defines the initial age structure of the stock and its size (Table 2). This pivotal value will set the highest recruitment level and is associated with the maximum catch allowed by the model after the potential yield is estimated.
- *Fishing mortality.* A fishing mortality value for the year when the highest yield occurred is adopted as $F = M$; however, a different value can be chosen by the user.

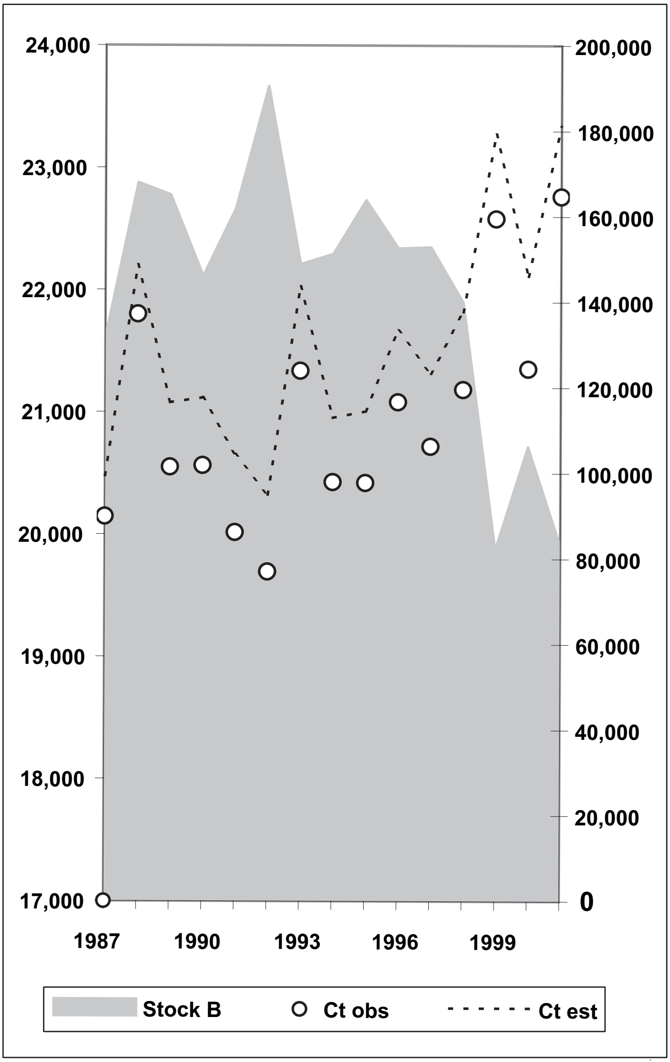

Figure 2. Estimation of stock biomass, exploited stock, and initial fitting of recorded and simulated yield over the historical record. Here, recorded catch data are not fitted to simulated data.

Table 2. Yield records and other important data of the fishery of a fifteen-year series chosen as an example. Output for 30 years of simulation are added in deterministic mode.

Year	Catch (t)	Mortality F	Millions of recruits	Millions of adults	No. of boats	Carrying capacity (t)
1981	877.9	0.10	0.20	0.26	31	107
1982	902.0	0.30	0.20	0.23	36	119
1983	715.5	0.30	0.20	0.19	44	114
1984	704.7	0.30	0.19	0.18	42	132
1985	846.0	0.50	0.19	0.13	38	137
1986	829.7	0.30	0.19	0.13	30	124
1987	814.5	0.20	0.19	0.15	40	125
1988	969.5	0.20	0.18	0.17	43	108
1989	1,085.2	0.20	0.19	0.18	41	101
1990	848.0	0.20	0.19	0.19	35	103
1991	762.0	0.20	0.19	0.19	35	106
1992	735.0	0.10	0.19	0.21	29	108
1993	947.0	0.10	0.20	0.24	33	115
1994	1,175.0	0.10	0.20	0.26	33	128
1995	1,300.0	0.11	0.20	0.27	33	128
1	1,609.4	0.35	0.20	0.22	41	135
2	1,379.2	0.35	0.20	0.18	41	135
3	1,191.5	0.35	0.19	0.16	41	135
4	1,043.2	0.35	0.18	0.14	41	135
5	927.8	0.35	0.20	0.13	41	135
6	837.0	0.35	0.19	0.13	41	135
7	764.9	0.35	0.19	0.12	41	135
8	715.0	0.35	0.19	0.12	41	135
9	678.8	0.35	0.19	0.12	41	135
10	653.1	0.35	0.19	0.12	41	135
11	634.2	0.35	0.19	0.12	41	135
12	620.0	0.35	0.19	0.12	41	135
13	610.1	0.35	0.19	0.11	41	135
14	602.9	0.35	0.18	0.11	41	135
15	597.8	0.35	0.18	0.11	41	140
16	594.1	0.35	0.18	0.11	41	140
17	591.2	0.35	0.18	0.11	41	140
18	589.0	0.35	0.18	0.11	41	140
19	587.0	0.35	0.18	0.11	41	140
20	585.3	0.35	0.18	0.11	41	140
21	583.8	0.35	0.18	0.11	41	140
22	582.5	0.35	0.18	0.11	41	140
23	581.4	0.35	0.18	0.11	41	140

Table 2. (Continued)

Year	Catch (t)	Mortality F	Millions of recruits	Millions of adults	No. of boats	Carrying capacity (t)
24	580.5	0.35	0.18	0.11	41	140
25	579.7	0.35	0.18	0.11	41	140
26	579.0	0.35	0.18	0.11	41	140
27	578.3	0.35	0.18	0.11	41	140
28	577.8	0.35	0.18	0.11	41	140
29	577.4	0.35	0.18	0.11	41	140
30	577.0	0.35	0.18	0.11	41	140

- *Length-weight*. Parameter values (a and b) must give expected and logical outputs. If not, changing the order of magnitude of parameter a by adding or reducing zeros (one at a time) will provide logical output, as displayed in Fig. 1).
- *Age of first catch and age of sexual maturity.* Reasonable values estimated directly from the fishery or obtained from published sources must be provided.

Economic variables

The model estimates the benefit/cost (B/C) ratio and total profits of fishing activities throughout the entire period of analysis as compared to current effort f, costs of fishing operations, catch value at landing, and assuming constant values throughout the whole series of catch records. When the B/C ratio is not logical, i.e., too high or lower than 1, its value can be changed by using the relationship $F = qf$. Cost replaces its corresponding effort; here, the value of the catchability coefficient q can be determined with the aid of the “goal seek” function of Excel by choosing an acceptable B/C target value. The goal seek function is a routine of Excel that searches for a given value of a formula until the expected value is found. This is done by changing other values in cells containing a formula or a reference to calculations.

Model fitting process

Recruitment

Catch estimates by year are made with the aid of the goal seek function. The program gives an estimate of the 1-year-old recruits and age structure of the stock in numbers and biomass. For the fifteen years of records, the model provides values estimated year by year. The Beverton and Holt stock-recruitment (S-R) model (1957), slightly modified by adding a parameter α, the initial slope, is used to establish a relationship between adult numbers and the number of 1-year-old recruits the year after. The parameter α can be modified by the user, but was left constant here ($\alpha = 0.25$). The only one that should be modified is β. The estimate of this parameter is based on the initial age structure previously determined, and it is made by using another virtual button that triggers a subroutine which estimates the maximum number of recruits determined by the β value. This number is used to define the initial age structure (Fig. 3A,B) that is used as a framework. The criterion used to define this relationship is based on the principle that the number of adults of initial age structure required are defined by the recruit number surviving over time and by assuming that the stock is in a steady state. Here total mortality is $M + F$ where $F = M$, arbitrarily defined as initial condition. The modified equation of the Beverton and Holt stock-recruitment model is

$$R = (\beta N_{ad} R_{max})/[N_{ad} + (\alpha R_{max})]$$

where

N_{ad} = number of adults in the previous year,

R_{max} = maximum number of recruits in data series, and

α, β are as explained above.

Age structure

The age composition of the stock, through historic records and simulation is in Fig. 4. The time series shows numbers per age group over the fifteen years of catch records and thirty additional years of simulation. Collapse of the stock when undesirably high F values are simulated is clearly evident.

Uncertainty

The model may work in either a deterministic or a stochastic version. Recruit number is multiplied by a random variable with normal distribution. Twenty percent of the coefficient of variation (CV) is the most commonly used value; however, the CV of recruitment is an estimate given by the model.

Fishing mortality and yield

To fit recorded yields with simulations, a button key triggers the subroutine that finds an F value for each year such that recorded and simulated catch values are equal. This F value is used in the catch equation where

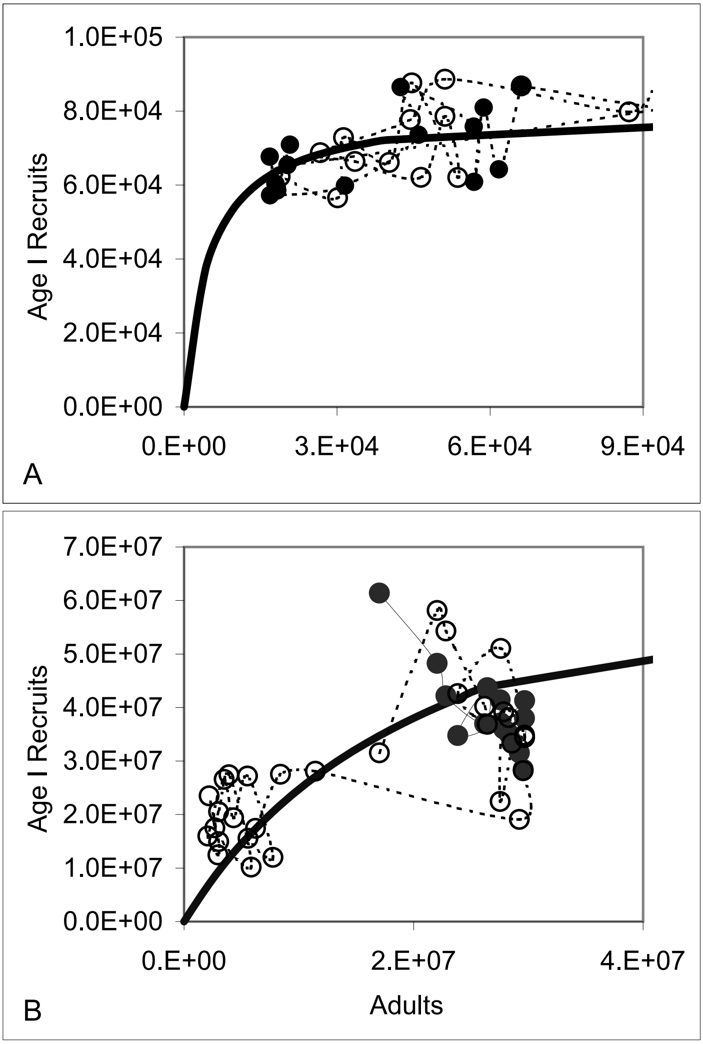

Figure 3. Stochastic stock-recruitment (Beverton and Holt) relationship. Dots = historic. Circles = simulated. A. Stable stock. B. Overexploited. Display shows uncertainty of this process by multiplying recruit number by a CV = 0.20 with normal distribution.

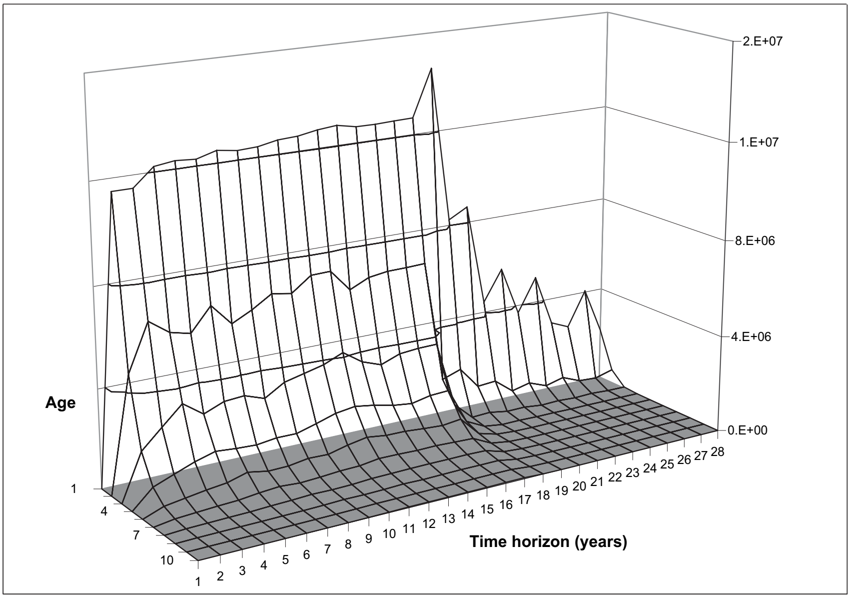

Figure 4. Age structure of the stock over time. A 20% recruitment uncertainty was applied. The first fifteen years correspond to the historical period and additional ones are simulated. The example displayed here shows an overexploited strategy leading the fishery to stock depletion.

numbers and weights per age each year simulate expected values and where simulated catches are matched with historical catch records (Table 1). The subroutine runs and estimates an *F* value for each year of the series. In some cases the model may not be able to find a value, because the method of approximation used by the goal seek routine sometimes searches out of the logical range of values. When this occurs, it is advised to modify the particular *F* given as root value and to try the fitting process again. When completed, the program displays a graph showing yields from the historical records and thirty additional years of simulation. The mean values of catch and net profits of the last five years of simulation are expressed numerically beside the graphic output, and can be used as an interactive tool. It is useful for managing the fishery chosen as a study case because any change made to *F* and t_c used as constants through the 30 years of simulation is shown graphically and numerically.

Assessment of fishing strategies

Economic information

Just as the value per kilogram of fish landed has a cost per trip (= the benefit/cost ratio), estimates of the number of fishing days and the fleet size are complementary economic variables useful for defining criteria to choose fishing strategies. With these values, the program estimates net profits. Since any economic activity has a $B/C > 1$, it must be above the economic equilibrium limit ($B/C = 1$). Economically feasible options must yield zero or positive profits (Wilmann and García 1985). Most fisheries operate with a reasonable margin of profit (usually less than 50% above current costs). For this reason, if accurate values are not available a clue for making an educated guess is given by using the criterion of a value such that $1 < B/C < 1.75$. An estimate of q must be given to get further estimates of fishing effort over the time period of analysis and simulations from the model. In addition, fine-tuning of the B/C value by changing q will improve estimates of the initial B/C ratio in the simulation when economic performance of the fishery is explored.

Fishing strategies

Yield and net profits through historic catch records and along the simulation are displayed as in Fig. 5. Fishing mortality value and age of first catch can be changed by the user, enabling the model to be used as a management tool. The graphic display shows the yield and profits through the fifteen years of records where F and t_c were assigned previously; it also shows the simulation output where F and t_c were chosen by the user. Numerical values of the output as a result of a combination of F and t_c values chosen to simulate the stock response are shown on the right side of the screen, as seen in Table 3.

Choosing management options

To choose suitable management strategies, numerical values of variables such as catch, B/C ratio, and net profits are displayed. They correspond to mean values of the last five years of the simulation period, and were chosen to give a picture of the most probable path of the fishery. The graph (Fig. 5) and the values shown in the screen beside it (Table 3) are used as biological and economic indicators of the performance of the fishery. This part of the output was designed to allow the model to be used to aid managers to define a criterion for the exploitation policy in the next fishing season, if management is adaptive and management policy is evaluated annually.

Potential yield and potential profit

Trend lines describing curves of yield and profits are displayed in Fig. 6. They are obtained as a result of stock response through simulation and

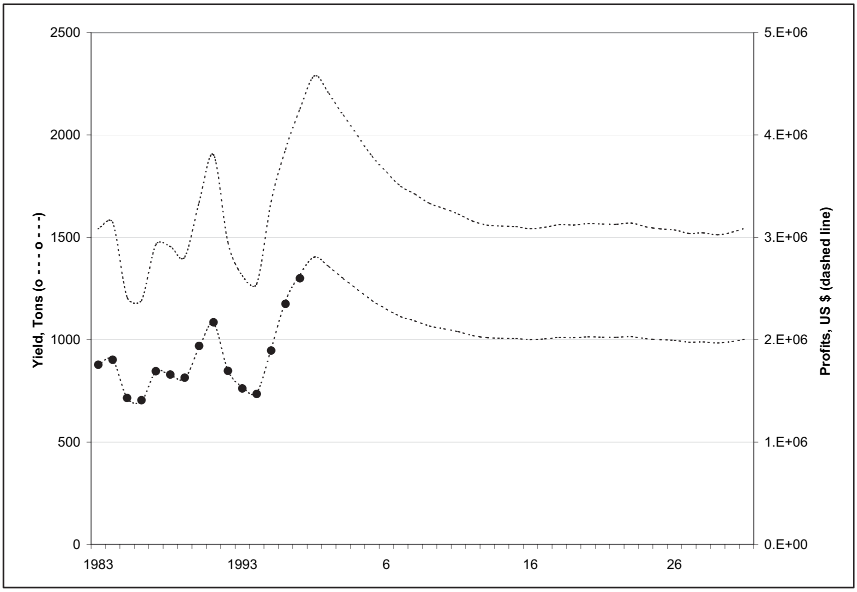

Figure 5. Yield and profits through historic records and the simulation period after fitting the model to catch data. *F* and age of first catch (t_c) can be modified by the user for simulation, and these results are displayed on the graph and shown in a time scale of 30 years after the catch records. This screen can be used as an interactive tool for choosing management strategies. The figure illustrates the case of a sustainable harvesting policy, exploiting the stock near its MSY level. The simulated yield and profits are associated to a recruitment with CV = 0.2.

depend on the parameter values and recruitment level, and therefore are specific for a given combination of those values. The output is obtained for each t_c value by testing different *F*s and with the aid of the table function, whose partial values are displayed in the table on the right side of the screen (not shown here). With values in the graphic display, detailed analysis for decision-making can be made by identifying the best *F* and t_c for evaluating reference points, such as maximum yield, threshold of overfishing, maximum profits, economic equilibrium limit, and other measures of fishing intensity that may threaten the sustainability of the fishery.

A response-surface graph displays expected yields under a combination of different fishing intensity values and age of first catch (Fig. 7). Expected profits from fishing activities under the effect of the same two

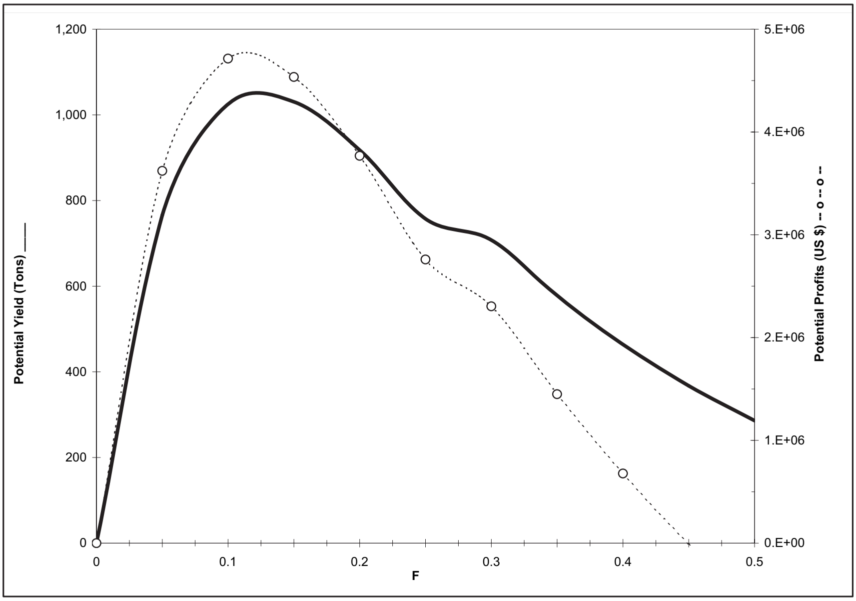

Figure 6. Potential yield and profits as an effect of a given combination of *F* and t_c values.

Table 3. Numerical values displayed beside the graphic output (Fig. 5), used to show the quantitative performance of the fishery in the long term.

F	0.35 per year
Exploitation rate	0.42
Yield	577.5 t
Trips	22,850 per year
Boats	41
Capacity	51.2 t
Benefit/cost	1.48
Profits	2,136,633 US$
Profits per boat	888 US$
Age of first catch (t_c)	4 (1-12)

F and t_c values are inputs chosen by the user and can be changed at any time.

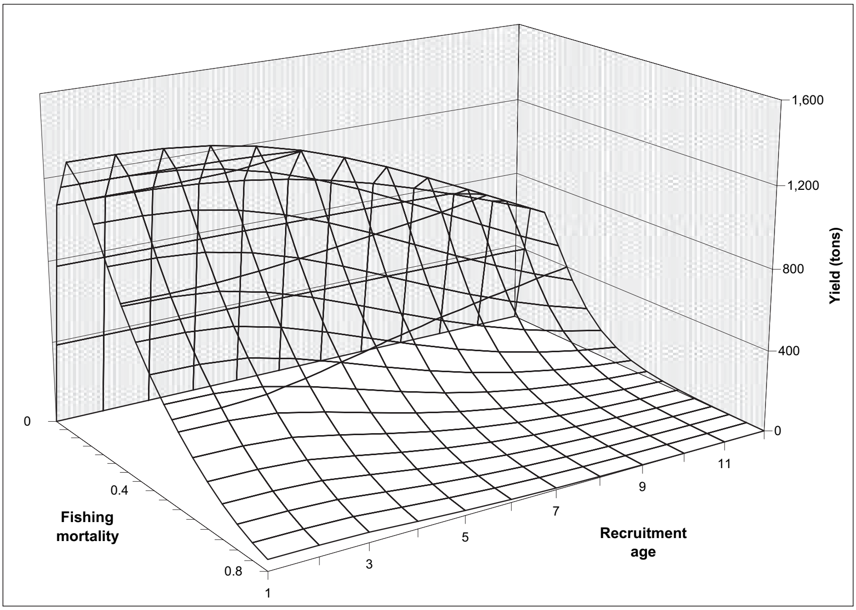

Figure 7. Response-surface display showing the effect of changes in *F* and age of first catch (t_c) on yield.

variables are shown in Fig. 8. These two outputs may be valuable tools in the planning stage of fishing activities. Maximum yields and maximum profits are displayed at the top of these graphs, showing that they can be attained within a range of *F* and t_c values rather than as a single combination of these two variables, allowing a range of possibilities in the process of decision-making. Another observation is that maximum yields and maximum profits are not usually attained with the same combination of *F* and t_c; in most fisheries, maximum profits are obtained at lower *F* values than maximum profits.

Optimizing the model

The model permits one to maximize the fitting of observed and expected catch values in several stages of its development. At the initial stage the corrections of the *a* parameter of the length-weight relationship are crucial, and just a little experience and common sense are required by watching the performance of the growth curves.

A second step important to optimize is initial number of juveniles of age 1 or recruits. A subroutine integrated in a macro of the spreadsheet is

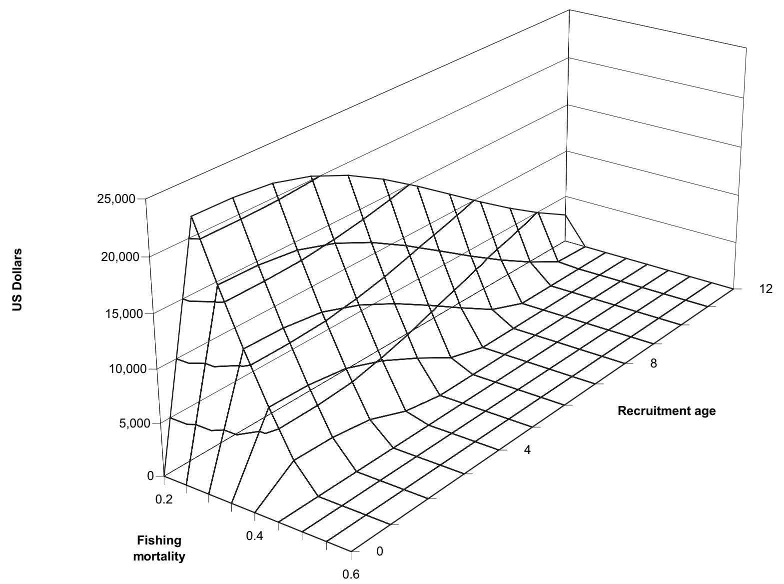

Figure 8. Response-surface display showing the effect of changes in *F* and age of first catch (t_c) on profits. Real currency values are usually displayed.

used to determine the number required to match with calculations. The third important step is the estimation of *F* values for each fishing year, aided by another subroutine activated by a virtual button. The age of first catch t_c is assumed to be constant over the records and can be changed for the simulation.

The goal seek function is used again to estimate one of the two parameter values of the parent-recruit analysis. To fit the β parameter of the reconstructed stock size of the year when the maximum yield was obtained and after the *F* values were estimated, a small bias results from the difference of recorded and simulated catch. It is corrected by minimizing the sum of residuals, i.e., the differences between observed and recorded catch, using the goal seek function and making the sum of differences equal to zero. A random variable with a normal distribution may be used to multiply the estimated number of recruits over the simulation.

Finally, a graph shows real and simulated catch-data values, and the goodness of fit between these two series is depicted by their closeness to a straight line (Fig. 9A). A bar graph (Fig. 9B) shows the sequence of recorded and simulated catch residuals. Any departure or bias from the algebraic sum is corrected with the aid of the goal seek routine by making its sum equal to zero, as described in the former paragraph.

Discussion

The actual structure of a model imposes limits (Haddon 2001). The model developed here is minimally complex in structure and data requirements, the details of which may be found elsewhere (Hilborn and Walters 1992, Quinn and Deriso 1999). This approach is a move toward simpler rules for setting harvest levels as pointed out by Hilborn (2002). Therefore, by sharpening Occam's Razor, this model integrates the basic processes of population dynamics of exploited stocks and the fisheries they belong to into a tool that may be useful for decision-making. The model has been applied to several fisheries (Table 4).

Advantages of the model

The model allows

- Assessment of the impact of fishing activities on exploited resources.
- Examination of the details of stock structure, e.g., the number of recruits over time and the immediate effects of any change in age of first catch (as result of an implicit change of mesh size of fishing gears), the age structure, and the catch.
- Determination of the effects of several exploitation options on yields and on catch composition.
- Evaluation of the effects of several exploitation options on the stock, on the economic variables, and on the level of employment.
- Easy modification for the analysis of other variables like fleet size.
- Recommendations for the conservation of fish resources and sustain ability.

Table 4. Application of FISMO to eleven Mexican fisheries. Earlier versions of the model were used for assessing the fisheries analyzed in the 1990s.

Stock	Area	Goal	Source
Queen conch, *Strombus gigas*	NWC	SA	Chávez 1994a, Chávez and Arreguín-Sánchez 1994
Spanish mackerel *Scomberomorus maculates*	SGM	SA	Chávez 1994b
King mackerel, *Scomberomorus cavalla*	SGM	SA	Chávez and Arreguín-Sánchez 1995
Three stocks of snapper	WM	SA	Cruz-Romero 1996
Octopus, *Octopus maya*	CB	AOHS	Chávez 1998
Pacific sardine, *Sardinops caeruleus*	GC	AOHS	Nevárez-Martínez et al. 1999
Rock shrimp stock, *Sicyonia penicillata*	GC	APY	López-Martínez et al. 2000
Pink shrimp, *Penaeus duorarum*	CB	AOC	Ramírez-Rodríguez et al. 2000
Blue abalone, *Haliotis fulgens*	WBC	AOHS	Ponce-Díaz et al. 2000
Spiny lobster, *Panulirus argus*	MBRS	AOHS	Chávez 2001
Purple conch, *Plicopurpura pansa*	WM	BE AOHS	Michel-Morfín et al. 2002
Skipjack tuna, *Katsuwonus pelamis*	WM	SAFC	Chávez 2003
Red snapper, *Lutjanus peru*	SGC	AOHS	Díaz-Uribe et al. 2004

Areas: CB = Campeche Bank; GC = Gulf of California; MBRS= Meso-American Barrier Reef System; NWC = Northwestern Caribbean; SGC = South Gulf of California; SM = South Gulf of Mexico; WBC = West Baja California; WM = West Mexico.

Goals: AOC = Assessment of causes of collapse; AOHS = Assessment of optimum harvesting strategies; APY = Assessment of potential yield; BE AOHS = Bio-economic assessment of optimum harvesting strategies; SA = Stock assessment; SAFC = Assessment of stock and fleet capacity.

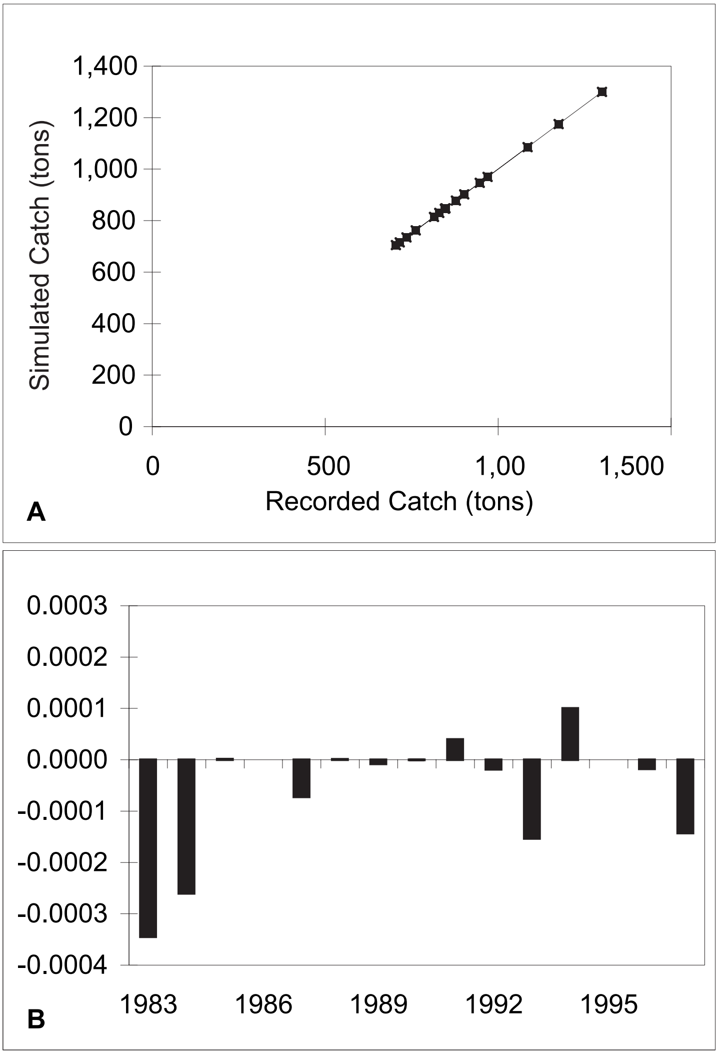

Figure 9. Results of the validation process. (A) Relationship between estimated and recorded yield are shown. (B) Tendency of residuals (tons) over time. When there is any bias in this tendency, a fine-tuning of the model can be done by making the mean value of residuals equal to zero by modifying the initial number of recruits with the aid of the goal seek function.

Limitations of the model

- When catch data display a declining tendency over time, the model tends to overestimate stock biomass of the remaining years of the series, because stock-recruitment parameter values depend on the highest yield. This can be controlled by having this factor in mind and re-estimating recruitment parameters based on yields of the last few years of the series.
- It cannot be applied to special considerations as sequential fisheries like shrimp, or to situations where closed seasons or closed areas are involved.
- It does not explicitly consider situations such as significant differences in sex ratio.

Acknowledgments

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Appendix: Subroutines in macros

FISMO uses three macros that were developed to provide a semi-automated procedure of calculation to fit the model into particular study cases.

1. Initial age structure

At initial stage, dummy values perform the structure of a symbolic stock, where numbers per age group are already assigned, and where the age of first maturity t_m and the age of first catch are known.

Here, $N_t = N_1 \exp^{(M+F)t}$ and initial F value was arbitrarily considered as $F = M$; t is expressed in time units, usually years, and recruits $R = N_1$.

The breeding stock S is the sum of individuals in the population number from maturity age t_m, to the oldest age group in the stock, or $t = 3/K$.

Let the catch be C', estimated from population parameter values and expressed through the catch equation from the age of first catch t_c to the last or longevity age group t_λ as:

$$C' = \sum_{tc}^{t\lambda} N_t \times W_t \times (F/(M+F) \times [1 - \exp^{-(M+F)}]$$

Here, all data are linked in vectors, so that any change in parameter values will affect C.

At initial age structure, the problem consists of finding the value of R, such that C becomes equal to the highest catch of the data series. Therefore, the goal seek function allows one to find that value, and a macro was defined to carry out that procedure.

2. Initial recruitment

Here, the stock-recruitment equation by Beverton and Holt (1957), slightly modified, is applied as follows:

$$R_t = \beta A_d A_{max} / [A_d + (\alpha A_{max})],$$

where

α = 0.25 is the slope of the curve; the value was assigned arbitrarily, but it can be modified by the user.

β = Unknown.

A_d = Number of adults in a certain year.

A_{max} = Maximum number of adults estimated for the initial age structure.

R_t is the number of recruits at the beginning of each following year, to be estimated through the S-R equation and thus linking each generation with the one in the following year over the catch data series and through the simulation. The goal seek function allows one to find that value and another macro was defined to carry out that procedure.

3. Yield

The catch equation is used here, where C_t is the recorded catch for the year t and F is an unknown for that year. Therefore, to find the F value for the year t, the goal seek function integrated in another macro is used.

For the thirty years of simulation, t_c and F values are provided directly by the user and this way many harvesting strategies can be tried.

Fisheries Assessment and Management in Data-Limited Situations
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Three-Stage Catch-Survey Analysis Applied to Blue King Crabs

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Abstract

Catch-survey analysis (CSA) is a stock assessment method that is applicable in data-limited situations. It requires time series of catch and relative abundance divided into two categories: recruits and post-recruits. This method is particularly useful for invertebrate populations (e.g., crabs) that cannot be aged but can be classified by their size distributions. In this study we extended the existing two-stage CSA to include a third stage, pre-recruits. An additional equation describes the transition of pre-recruits to the recruit stage as a function of survival and molt probability. We applied the three-stage CSA to two blue king crab (*Paralithodes platypus*) stocks: St. Matthew Island and the Pribilof Islands in the eastern Bering Sea. Sensitivity analyses and Monte Carlo tests were performed on data from the St. Matthew Island stock. The final version of the model includes bootstrapped confidence intervals on the estimated population abundances. Abundance of both stocks declined following high harvest rates in the early 1980s and the fisheries have been closed since 1999. Three-stage catch-survey analysis has several advantages over the two-stage CSA. It uses more data and therefore has a higher ratio of observations to parameters. Three-stage CSA also provides estimates of mature and legal male crabs, both of which are used for management. By accounting for observation errors for an additional stage, it provides more accurate estimation of abundance in the most recent year, which is critical for setting harvest guidelines.

Introduction

Many fisheries agencies conduct standardized research surveys of relative abundance as a cornerstone of stock assessments of exploited fish and shellfish populations. To convert relative to absolute estimates of population size, survey gear must be calibrated with a survey catchability coefficient, *q*, defined as the proportion of the population sampled with one unit of survey effort. A limitation of research surveys is the requirement to assess vast areas with a limited number of observations. Random sampling can ensure unbiased abundance estimates, but can't eliminate large observation errors that arise from patchy spatial distributions and other chance elements of sampling.

Collie and Sissenwine (1983) developed catch-survey analysis (CSA) as one of the first stock-assessment methods to calibrate fishery-independent surveys while accounting for observation errors. This method was designed for populations with limited age information, either because of the difficulty or high cost of age determination. It requires only that the population be divided into two categories, recruits and post-recruits, which can often be distinguished by their size distributions. The size or age distribution of the commercial catch is not required, though it can be used to calculate the selectivity of the fishing gear. This simple formulation produces abundance estimates in close agreement with those of age-structured methods (Collie and Sissenwine 1983), length-based models (Zheng et al. 1996), and production models (Cadrin et al. 1999).

Catch-survey analysis has been applied to a variety of demersal fish (Collie and Sissenwine 1983), mollusk (Conser 1991), and crustacean species (Conser and Idoine 1992, Cadrin et al. 1999). It is particularly well suited to invertebrate species for which routine age determinations do not exist. If gear size selectivity is known, it can be used to partition survey catches into recruit and post-recruit categories (Cadrin et al. 1999). King crabs are particularly good candidates for CSA because, as adults, they molt, at most, once per year. Although absolute age determination is impossible, legal-sized crabs can be classified into recruits (legal size for less than one year) and post-recruits (legal size for more than one year) based on the size and relative age (i.e., new-shell/old-shell) of their carapace. On this basis, Collie and Kruse (1998) applied CSA to two red king crab (*Paralithodes camtschaticus*) stocks in Alaska.

The original catch-survey analysis estimated process errors (also known as equation errors) and observation errors (also known as measurement errors) (Collie and Sissenwine 1983). Process errors arise from mis-specifying the population dynamics equation; for example, variations in a natural mortality rate, *M* and *q* that are assumed constant. However, the mixed-error formulation requires the estimation of unobserved, latent variables, which increases the risk of overparameterization and non-convergence to a unique solution. Several studies since then included simulations to test alternate model formulations and the sensitivity of the method to the magnitude of errors commonly encountered in survey data. A general result is that *q* can be estimated with low bias (<8%) with realistic levels of observation error in the survey data (Collie and Kruse 1998). Estimates of *q* are highly dependent on *M*, which must therefore be specified as an input parameter.

A version of CSA that assumed only observation errors, ignoring process errors, performed well when tested with data containing various levels of correlated and uncorrelated observation errors, and low levels of process error (Collie and Kruse 1998). With fewer parameters to estimate (no latent variables), this version reduces the risk of overparameterization. With high levels of process error (higher variance than in most real data sets) the all-observation error model resulted in more highly biased parameter estimates and some Monte Carlo trials failed to converge. Derived quantities of interest, such as the abundance of legal male crabs, were estimated with low bias (<4%). Monte Carlo simulations provided bootstrapped confidence intervals for the quantities used in decision making (e.g., absolute abundance, harvest rate).

Further sensitivity analyses have been conducted with data generated from a known fully age-structured model (Cadrin 2000, Mesnil 2003). Absolute abundance estimates are sensitive to relative catchability of one stage to the other, which must be specified externally. If this relative catchability is size dependent, its value can vary with time due to changes in the age structure of the population caused by fishing and/or recruitment variability (Mesnil 2003). Erroneously treating this relative catchability as constant can therefore introduce process errors in the estimation. Notwithstanding these potential sources of error, Mesnil (2003) found CSA to be “a very valuable method to support management advice in data limited contexts, in which age data are lacking or uncertain.”

Populations of the blue king crab (*Paralithodes platypus*) occur in the eastern Bering Sea near St. Matthew Island and the Pribilof Islands. As there is no evidence of migration between the two populations, they are assessed and managed as two separate stocks (Otto and Cummiskey 1990). Whereas juveniles strongly prefer shallow-water habitats with shell hash that provides refuge from predators (Armstrong et al. 1985), adults are generally thought to occupy soft bottoms (mud-sand) adjacent to coarse bottoms (gravel-cobble-boulder-rock) at depths of 48-75 m.

Although subadult and adult males from both areas have the same mean growth increment (~14 mm carapace length, CL), molting probability declines sharply after attaining size at maturity, which is 105 mm CL for St. Matthew Island and 120 mm CL for the Pribilof Islands. Likewise, the mean size of landed blue king crabs is much larger for the fishery off the Pribilof Islands (3.5-3.6 kg) than off St. Matthew Island (1.8-2.2 kg). Harvest guidelines are based on a harvest rate applied to abundance estimates of mature male crabs, but only legal males (≥120 mm CL for St. Matthew Island and ≥135 mm CL for the Pribilof Islands) may be retained.

In the past, stock assessments were not based on population models; instead crab abundance was estimated by the area-swept method in which survey catches were expanded from the area swept by trawl tows to the entire region encompassed by the stratified systematic survey design (Stevens et al. 2002). Catch-survey analysis is an attractive assessment tool for blue king crabs because there is limited length-frequency data to support more data-intensive methods. A two-stage CSA has been used to assess the St. Matthew Island and Pribilof Islands blue king crab stocks (Zheng et al. 1997). The estimates of survey catchability were uncertain, partly because crab habitats around these islands include rugged, untrawlable bottoms with uncertain crab densities. To address this uncertainty, Zheng et al. (1998) incorporated an index of commercial catch per unit effort into the CSA, because commercial crab pots (traps) are deployed over a wider range of bottom types than survey trawls. As a result, the spatial distributions of the survey and the fishery do not fully overlap.

The objective of this study was to improve the performance of CSA by refining the estimate of legal blue king crab abundance and by including the estimation of mature crab abundance in the analysis. Though previous applications were promising, the output of CSA was not ideally suited to crab management, because mature male crabs were not included in the analysis. Mature crabs include pre-recruit males that have not yet molted into the legal size range. We therefore added a pre-recruit stage to the standard two-stage CSA. A three-stage CSA would make more complete use of the survey data but would also require the specification of additional parameters associated with the pre-recruit stage. In this study we develop a three-stage version of CSA and apply it to the estimation of blue king crab abundance. Monte Carlo simulations are used to evaluate the performance of the three- versus two-stage CSA.

Methods

Model formulation

The three-stage catch-survey analysis uses a simple population model to calibrate relative abundance indices from research surveys with the commercial catch in numbers C_t. The procedure estimates q, which can then be used to estimate a time series of absolute population abundance. The stages are defined as pre-recruit N, recruit R, and post-recruit P. As in the two-stage model, the recruit category is defined such that all individuals become post-recruits the following year and the post-recruit category is a "plus group" (Collie and Kruse 1998). The population model for true (absolute) abundance is

$$P_{t+1} = \left(R_t + P_t\right)e^{-M_t} - C_t e^{-M_t(1-T_t)},$$

where P_t and R_t are the respective post-recruit and recruit numerical abundances in year t, M_t is the instantaneous natural mortality rate, and T_t is the time lag from the survey to the midpoint of the fishery. Most of the variability in the time lag is due to historical changes in fishery start dates and duration. Recruit abundance is a function of pre-recruit abundance N_t, in the previous year:

$$R_{t+1} = mN_t e^{-M_t},$$

where m is the transition probability between the pre-recruit and recruit stages. Pre-recruits that don't molt remain as pre-recruits the following year, which adds the constraint

$$N_{t+1} \geq (1-m)N_t e^{-M_t}.$$

Only the male component of the population is modeled, because females cannot be legally retained by the fishery. Size limits were set to allow males one opportunity to mate before achieving legal size. So, the pre-recruit class includes mature males that are one molt increment shy of recruitment into the commercial fishery. The transition probability m is the probability that a pre-recruit crab will molt in a given year. Recruits are defined as crabs that entered the legal size class in that year. This determination is based on two criteria: they are "new-shell" meaning they molted less than one year before being caught in the survey and they are less than one molt increment greater than minimum legal size. The post-recruit size class includes all crabs greater than legal size that are not new recruits (Collie and Kruse 1998). Owing to differences in growth and maturity schedules between St. Matthew Island and the Pribilof Islands, size classes differ for each stock as defined in the next section.

As in the two-stage CSA, true (unknown) relative abundance can be written as the product of a catchability coefficient and absolute abundance:

$$n_t = q\,\theta\,N_t,$$

$$r_t = q\phi R_t, \text{ and}$$

$$P_t = q\,p_t,$$

where n_t, r_t, and p_t are the respective pre-recruit, recruit, and post-recruit relative abundances in year t, ϕ is the catchability of recruits relative to post-recruits, θ is the catchability of pre-recruits relative to post-recruits, and q is the survey catchability coefficient for post-recruit crabs. Substituting these relationships into the population equations and solving for post-recruit and recruit relative abundance results in:

$$p_{t+1} = \left(r_t/\phi + p_t\right)e^{-M_t} - qC_t e^{-M_t(1-T_t)}$$

$$r_{t+1} = mn_t \frac{\phi}{\theta} e^{-M_t}.$$

Observed (survey) relative abundance indices, denoted below with the tilde (~), are assumed to have lognormally distributed observation errors such that their logarithms can be written as:

$$ln(\tilde{n}_t) = ln(n_t) + \gamma_t,$$

$$ln(\tilde{r}_t) = ln(r_t) + \kappa_t, \text{ and}$$

$$ln(\tilde{p}_t) = ln(p_t) + \varepsilon_t,$$

where γ_t, κ_t, and ε_t are either correlated or uncorrelated normal deviates (Collie and Sissenwine 1983). The model parameters are estimated by minimizing the sum of squared differences between the observed and predicted relative abundance indices, with the objective function:

$$SSQ = \sum_{t=1}^{y-1} w_{pre}^2 [\ln(\tilde{n}_t + \delta) - \ln(n_t + \delta)]^2 + \sum_{t=1}^{y} w_{rec}^2 [\ln(\tilde{r}_t + \delta) - \ln(r_t + \delta)]^2$$

$$+ \sum_{t=1}^{y} [\ln(\tilde{p}_t + \delta) - \ln(p_t + \delta)]^2,$$

where δ is a small constant added to ensure the equation remains defined, w_{pre} and w_{rec} are the weights of pre-recruits and recruit errors relative to post-recruit errors, and y is the number of years of data. The parameters ϕ, M_t, and T_t are assumed to be known independently; the model estimates θ, q, r_1, p_1, and pre-recruit relative abundance n_t, for each year but the last. When fit to y years of data, the model has $y + 3$ parameters to estimate, $3y - 1$ residual errors, and $2y - 4$ degrees of freedom.

Estimated relative abundances are converted to absolute population estimates using

$$N_t = \frac{n_t}{q\theta}, \; R_t = \frac{r_t}{q\phi}, \text{ and } \; P_t = \frac{p_t}{q}.$$

The number of mature crabs each year is estimated as $N_t + R_t + P_t$, and the number of legal crabs each year is $R_t + P_t$. The three-stage formulation allows legal crab abundance to be estimated in all years and mature crab abundance to be estimated in all years but the last, for which the observed pre-recruit index must be used. Historical harvest rates h are calculated relative to both legal and mature crab abundance:

$$h(legal)_t = \frac{C_t}{\left(R_t + P_t\right)e^{-M_t T_t}}$$

$$h(mature)_t = \frac{C_t}{\left(N_t + R_t + P_t\right)e^{-M_t T_t}} .$$

Data sources

Abundance indices for blue king crabs were derived from the National Marine Fisheries Service (NMFS) eastern Bering Sea trawl survey (Stevens et al. 2002). This fishery-independent bottom-trawl survey samples a fixed 20 nm by 20 nm grid of stations to monitor the abundance of crabs and demersal fish. Survey strata adjacent to St. Matthew Island and the Pribilof Islands were used by NMFS to derive area-weighted estimates of blue king crab abundance. Though these survey data have been calibrated for the effective width of the trawl using measurements from electronic sensors, here we treat them as relative abundance estimates. Commercial catch data and timing of the commercial catch came from Alaska Department of Fish and Game (ADFG) annual management reports. We set M equal to 0.3, typical of assessments for these stocks (Zheng et al. 1997).

Input data files include time series of pre-recruit, recruit, and post-recruit relative abundance (survey) indices, commercial catch in numbers, timing of the commercial catch for each year the fishery was open, estimates of M for pre-recruits and recruits/post-recruits, and pre-recruit molt probability (Table 1). The pre-recruit and recruit size classes were 105 to 119 mm and 120 to 133 mm for the St. Matthew stock and 120 to 134 and 135 to 148 mm for the Pribilof stock, respectively. The recruit size class includes only new-shell crabs, such that all recruits become post-recruits the following year. For simplicity we used only the trawl-survey data and did not include auxiliary catch-per-unit-effort data.

Initial development of the three-stage model was based on data from 1980 to 1997 and initial baseline runs and simulation tests were conducted with this 18-year time series. For the final runs reported in this paper, we updated the data sets to 2003 and included several years of pre-1980 data that recently became available. For this analysis, we fixed $\phi = 1.0$, because the survey trawl catches recruit and larger crabs equally well. For the baseline runs the offset parameter in the sum of squared residuals term δ was set to 0.001. All error weights (w) were fixed at 1, though in principle they could be estimated from the within-year CV of the survey data.

Table 1. Input files for the three-stage catch-survey analysis.

(a) St. Matthew Island blue king crabs

Year	Abundance indices			Commercial catch (millions)	Timing of commercial catch	Mature harvest rates
	Pre-recruit	Recruit	Post-recruit			
1978	2.175	1.187	0.587	0.436	0.068	0.115
1979	1.793	1.455	0.333	0.053	0.060	0.011
1980	2.588	1.699	1.197	0.033	0.068	0.005
1981	1.480	1.195	1.648	1.046	0.052	0.133
1982	2.615	3.617	3.263	1.936	0.068	0.274
1983	1.639	1.399	1.956	1.932	0.121	0.405
1984	0.500	0.788	0.762	0.841	0.096	0.361
1985	0.431	0.541	0.708	0.485	0.137	0.350
1986	0.425	0.164	0.185	0.220	0.137	0.148
1987	0.757	0.492	0.292	0.235	0.137	0.117
1988	0.703	0.417	0.411	0.302	0.137	0.120
1989	1.235	0.940	0.954	0.248	0.134	0.081
1990	0.957	0.954	1.164	0.391	0.140	0.104
1991	1.636	1.353	0.889	0.727	0.178	0.161
1992	1.582	1.338	1.247	0.545	0.142	0.125
1993	1.994	1.605	2.000	0.630	0.178	0.138
1994	1.350	1.246	1.120	0.827	0.178	0.197
1995	1.321	0.993	0.902	0.667	0.178	0.153
1996	1.970	1.950	1.331	0.661	0.178	0.150
1997	2.319	2.213	1.853	0.940	0.178	0.254
1998	1.843	1.397	1.766	0.612	0.178	0.265
1999	0.215	0.179	0.436	0.000	NA	0.000
2000	0.310	0.323	0.488	0.000	NA	0.000
2001	0.527	0.345	0.681	0.000	NA	0.000
2002	0.266	0.186	0.576	0.000	NA	0.000
2003	0.330	0.188	0.379	0.000	NA	NA

Table 1. (Continued.) Input files for the three-stage catch-survey analysis.

(b) Pribilof Islands blue king crabs

	Abundance indices			Commercial catch (millions)	Timing of commercial catch	Mature harvest rates
Year	Pre-recruit	Recruit	Post-recruit			
1975	6.201	4.144	4.873	0.315	0.200	0.026
1976	0.647	1.412	2.147	0.856	0.200	0.084
1977	1.275	3.379	8.472	0.807	0.200	0.093
1978	3.525	1.684	3.918	0.797	0.250	0.087
1979	1.075	1.285	2.282	0.816	0.250	0.111
1980	1.158	1.284	3.767	1.497	0.252	0.259
1981	0.716	0.686	3.170	1.202	0.249	0.321
1982	0.299	0.596	1.657	0.588	0.175	0.281
1983	0.597	0.359	1.037	0.276	0.173	0.181
1984	0.215	0.218	0.335	0.040	0.142	0.037
1985	0.155	0.138	0.227	0.078	0.192	0.083
1986	0.018	0.098	0.394	0.037	0.233	0.057
1987	0.071	0.068	0.731	0.095	0.274	0.182
1988	0.000	0.034	0.164	0.000	0.000	0.000
1989	0.000	0.000	0.211	0.000	NA	0.000
1990	0.658	0.234	0.094	0.000	NA	0.000
1991	0.754	0.577	0.584	0.000	NA	0.000
1992	0.730	0.374	0.741	0.000	NA	0.000
1993	0.640	0.318	0.658	0.000	NA	0.000
1994	0.314	0.183	0.651	0.000	NA	0.000
1995	0.842	0.332	1.669	0.174	0.175	0.108
1996	0.938	0.145	1.071	0.124	0.175	0.076
1997	0.261	0.282	0.540	0.069	0.175	0.056
1998	0.192	0.099	0.739	0.069	0.175	0.067
1999	0.145	0.105	0.347	0.000	NA	0.000
2000	0.190	0.017	0.531	0.000	NA	0.000
2001	0.068	0.000	0.404	0.000	NA	0.000
2002	0.018	0.033	0.161	0.000	NA	0.000
2003	0.016	0.082	0.218	0.000	NA	NA

Estimation of molt probability

Blue king crab pre-recruit molt probabilities were estimated from analysis of tag-recapture data collected by NMFS on the St. Matthew Island and Pribilof Islands blue king crab stocks and by ADFG on the St. Matthew Island stock (Blau 1996). A logistic equation that describes molt probability as a function of carapace length was fitted to these data:

$$prob_L = \frac{1}{1+ae^{bL}} \quad ,$$

where $prob_L$ is the probability a crab of carapace length L will molt in any given year and a and b are parameters. Carapace length is the distance from the posterior margin of the right eye orbit to the posterior carapace margin. Logistic regression is the appropriate statistical model for these recapture data because each data point is weighted equally and the molt probability is a binomially distributed random variable (the crab did or did not molt), with an expected molt probability associated with the crab's release length (Neter et al. 1996). The logit transformation of the logistic molt probability function is

$$ln\left(\frac{prob_L}{1-prob_L}\right) = ln\left(\frac{1}{a}\right) - bL \quad \cdot$$

We estimated the pre-recruit molt probability m by averaging the fitted equation over the length of the pre-recruit size-class for each stock.

For the Pribilof Islands stock, we followed the procedure of Otto and Cummiskey (1990). There were 188 recaptures of crabs that were at large between 1 and 1.5 years, which ensures the crabs had the opportunity to molt once and only once. A crab was presumed to have molted if the carapace length increased more than 7 mm and to have not molted if the recapture carapace length was within 3 mm of the release length. Four crabs with growth increments between 4 and 7 mm inclusive were excluded from the analysis as it could not be determined whether these crabs molted or their shells were measured with error.

For the St. Matthew Island stock, we pooled the NMFS and ADFG data and used recapture information from crabs that were at large between 1 and 1.5 years and between 2 and 2.5 years. As there were only 81 recaptures between 1 and 1.5 years, we included data from 22 crabs at large between 2 and 2.5 years to increase the sample size. We determined whether the crabs at large between 2 and 2.5 years molted in the first and/or second year by considering their growth increment and shell condition at recapture by the following logic. None of the crabs in this sample had carapace length increases of 7 mm or less. The average growth per molt for the St. Matthew Island stock follows a normal distribution with an approximate mean of 14.1 mm and variance of 9.6 mm (Otto and Cummiskey 1990). The symmetric interval from 8 to 20 mm about the mean approximates the 95% confidence interval for mean growth per molt. So, we designated crabs that molted once as those whose carapace length increased by 20 mm or less. Of these crabs, those recaptured with old shells were assumed to have molted in the first year and those with new shells in year two. Those recaptured crabs whose carapace length increased by more than 20 mm were assumed to have molted twice, regardless of the recapture shell condition. Only the first-year molt frequencies were used to calculate molt probabilities, so that each crab would be counted only once.

Simulation tests

We tested the sensitivity of the three-stage model to errors in input parameters and compared the bias and precision of the three-stage and two-stage model formulations with data from the years 1980 to 1997 for the St. Matthew Island stock only. Two types of analyses were performed. First, we looked at the sensitivity of the model's output parameters θ and q to mis-specification of the input parameters, including pre-recruit and recruit catchability, pre-recruit mortality, and pre-recruit molt probability. Collie and Kruse (1998) performed an analogous exercise with the two-stage model.

Second, we performed Monte Carlo simulations of the three-stage model. The Fortran program (a) starts with a data set that fits the model perfectly (i.e., model estimates of relative abundance output from the baseline run), (b) accepts as input, the standard deviation and correlation of observation errors, and the standard deviation of process errors, (c) simulates the uncorrelated process errors and assumes these new relative abundance indices represent the true population, (d) simulates correlated or uncorrelated observation errors on the true abundance indices from step (c), (e) fits the simulated observed data with the three-stage model, and (f) repeats the simulation as many times as desired. Identical Monte Carlo simulations with the two-stage model were performed for comparison. The simulated data had the same state dynamics expected by the estimation method, except for the addition of process errors and correlated observation errors.

Process errors exist when the true population dynamics are not adequately described by the population model. We simulated the process errors by applying lognormally distributed errors to the input (fitted) survey indices with the following calculations, assuming that the new abundance indices represent the true population:

$$r_{t+1} = \left[n_t m \frac{\phi}{\theta} e^{-M_t} \right] e^{z\sigma - \sigma^2/2},$$

$$p_{t+1} = \left[\left(r_t / \phi + p_t \right) e^{-M_t} - qC_t e^{-M_t(1-T_t)} \right] e^{z\sigma - \sigma^2/2},$$

where z is a random standard normal deviate, σ is the standard deviation, and σ^2 the variance of the process errors. In the simulations examining process error only, these simulated "true" relative abundance indices are equivalent to the observed indices.

Correlated observation errors were placed on these indices with the following methodology. The 3×3 matrix V contains the input variances and covariances, which in this case were assumed to be the same for all stages. If z is a vector of independent standard normal deviates, and L is the lower triangular matrix from the Cholesky decomposition of v, then $l = Lz$ is a vector of cross-correlated random deviates (Kennedy and Gentle 1980). In each scenario, we simulated 1,000 replicates with the following error structures: (1) uncorrelated observation errors with standard deviation of 0.3, 0.5, or 0.7, (2) correlated observation errors with standard deviations as in (1) and a correlation of 0.5, and (3) a combination of correlated and uncorrelated observation errors and process error standard deviations of 0.1 and 0.3.

Confidence intervals on the estimates of population abundance for both stocks were estimated by parametric bootstrapping. The bootstrapping program first fit the input data to the model and calculated the standard deviation of the residuals. Unless otherwise specified (as in the Pribilof Islands simulation), the bootstrap used this standard deviation to simulate lognormally distributed errors on the relative abundances calculated in the initial run. Observation errors were simulated in the same manner as in the Monte Carlo analysis with uncorrelated errors. We performed 1,000 replicates and calculated the mean, standard deviation, and 95% empirical confidence intervals of the bootstrapped fits.

Results

Molt probabilities

Chi-square and deviance goodness of fit tests indicated that both of these data sets did not significantly differ from the logistic model (Neter et al. 1996). The logistic regression for the Pribilof Islands stock resulted in a pre-recruit molt probability of 0.75, which was used in all subsequent analyses (Fig. 1a). Using Otto and Cummiskey's (1990) parameter estimates resulted in a pre-recruit molt probability of 0.82. For the St. Matthew stock, the logistic regression resulted in a molt probability of 0.63, which was used for all the runs presented here (Fig. 1b). The reduced data set, consisting of 81 crabs at liberty between 1 and 1.5 years, resulted in

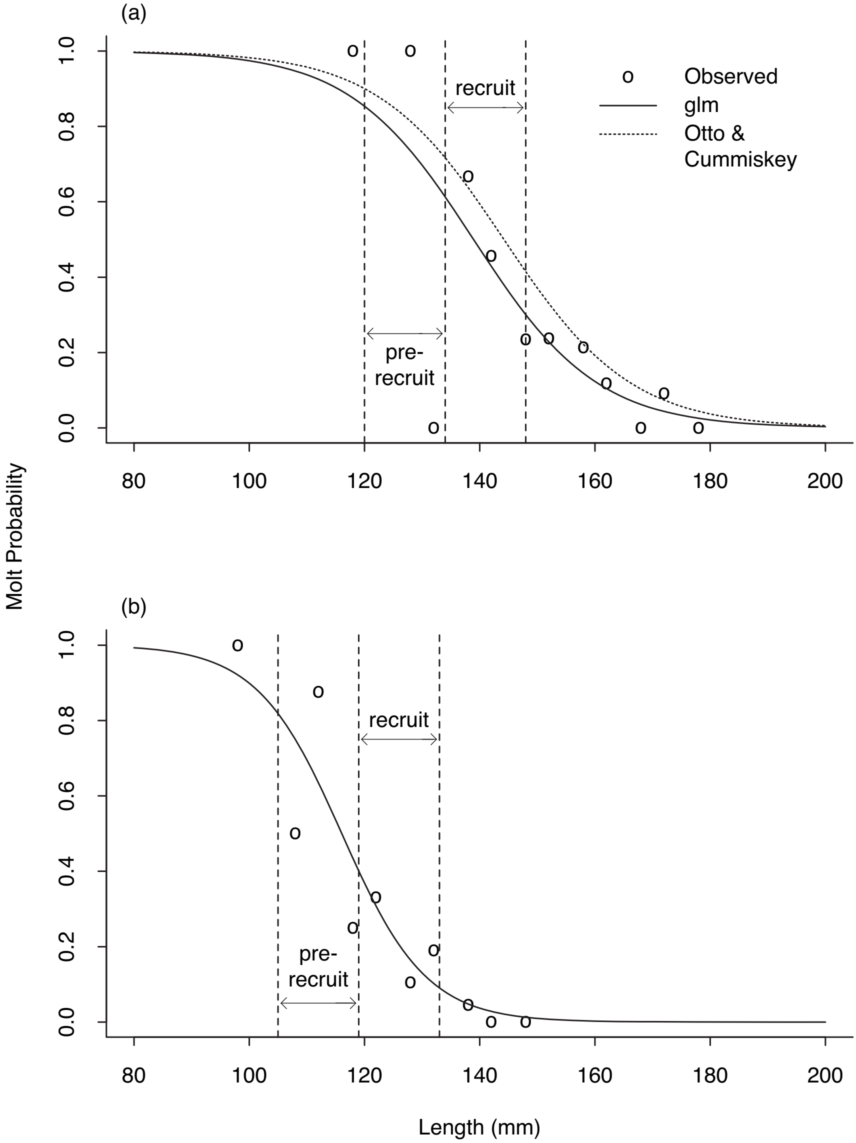

Figure 1. Comparison of molt probability curves for the Pribilof Islands (a) and St. Matthew Island (b) blue king crab populations. Vertical lines indicate range of the pre-recruit and recruit size classes for each population. The solid curves are the generalized linear model fits (glm) to the binomial data with logit transformation.

Table 2. Residual analyses of the baseline runs.

Statistic	St. Matthew	Pribilof Islands[a]
Number of years	18	18
Catchability coefficient (q)	1.105	1.405
Standard error of q	0.363	0.857
Catchability of pre-recruits (θ)	0.516	0.625
Standard error of θ	0.357	0.900
Sum of squared residuals	4.063	18.070
Standard deviation	0.356	0.751
Prob of normal distribution	0.925	0.055
Pre-recruit to recruit		
Correlation	0.702	–0.118
Prob correlation = 0	0.002	0.674
Prob of equal variances	0.955	0.871
Correlation (pre-recruit lag-1)	–0.585	–0.930
Prob correlation = 0	0.014	0.000
Pre-recruit to post-recruit		
Correlation	0.588	0.454
Prob correlation = 0	0.013	0.089
Prob of equal variances	0.834	0.114
Correlation (pre-recruit lag-1)	–0.407	–0.364
Prob correlation = 0	0.105	0.181
Recruit to post-recruit		
Correlation	0.613	–0.052
Prob correlation = 0	0.007	0.842
Prob of equal variances	0.876	0.151
Correlation (pre-recruit lag-1)	–0.380	0.626
Prob correlation = 0	0.132	0.009

[a] Zero survey indices were observed in 1988 and 1989 for the Pribilof stock. Zero indices are problematic because of the lognormal error structure. The sum of squares and standard deviation were calculated and the residual analyses performed with the zero survey indices omitted.

Prob = probability.

slightly lower molt probabilities and a less significant fit. These estimates are lower than the value 0.76, obtained by Otto and Cummiskey (1990), by lumping together all crab recaptures after 1 year at liberty.

Baseline runs

The distribution of residuals from the baseline runs (Table 2) was examined with histograms and quantile-quantile plots (not shown). Residual distributions for both stocks were skewed but were not significantly different from a normal distribution according to chi-square tests. The variances of the residuals from the different size classes of crabs were tested for homogeneity with an *F*-ratio test. There were no significant differences between size classes for either stock.

The standard deviation of residuals was 0.36 for the St. Matthew stock and 0.89 for the Pribilof stock. The Pribilof stock had four large positive residuals associated with zero abundance indices, in which case the offset added to prevent undefined logarithms substantially increased the residual standard deviation (Table 2). When the three residuals for the zero indices were removed, the standard deviation for the Pribilof stock was reduced to 0.75. The pre-recruit, recruit, and post-recruit residuals each year tended to be positively correlated. This might occur because all three indices come from the same survey and the total survey catch is subject to a common observation error. The correlations were stronger for the St. Matthew Island stock, and weaker for the Pribilof Islands stock, with negative, insignificant pre-recruit/recruit and recruit/post-recruit correlations.

First-order autocorrelation of the residuals was weak. However, one-year lagged correlations were sometimes significant (Table 2). For St. Matthew, the lag-1 correlation between pre-recruits and recruits was negative and significant. A correlation involving the pre-recruit residual lagging the recruit residual by one year would occur when there is a low survey year followed by a high survey year; the pre-recruit residual would be positive and the recruit residual negative to “split the difference.” The lag-1 correlation between recruits and post-recruits and between pre-recruits and post-recruits were also negative but not significant. For the Pribilof stock, the lag-1 correlation between pre-recruits and recruits was negative and significant and between recruits and post-recruits was positive and significant.

In summary, the residuals were normally distributed and homoscedastic, as assumed in fitting the model and in creating the bootstrapped standard deviations of the abundance estimates. The main features of the residuals that were not accounted for in fitting the model are positive covariances and negative lagged covariances. This residual structure could be addressed in future extensions of catch-survey analysis.

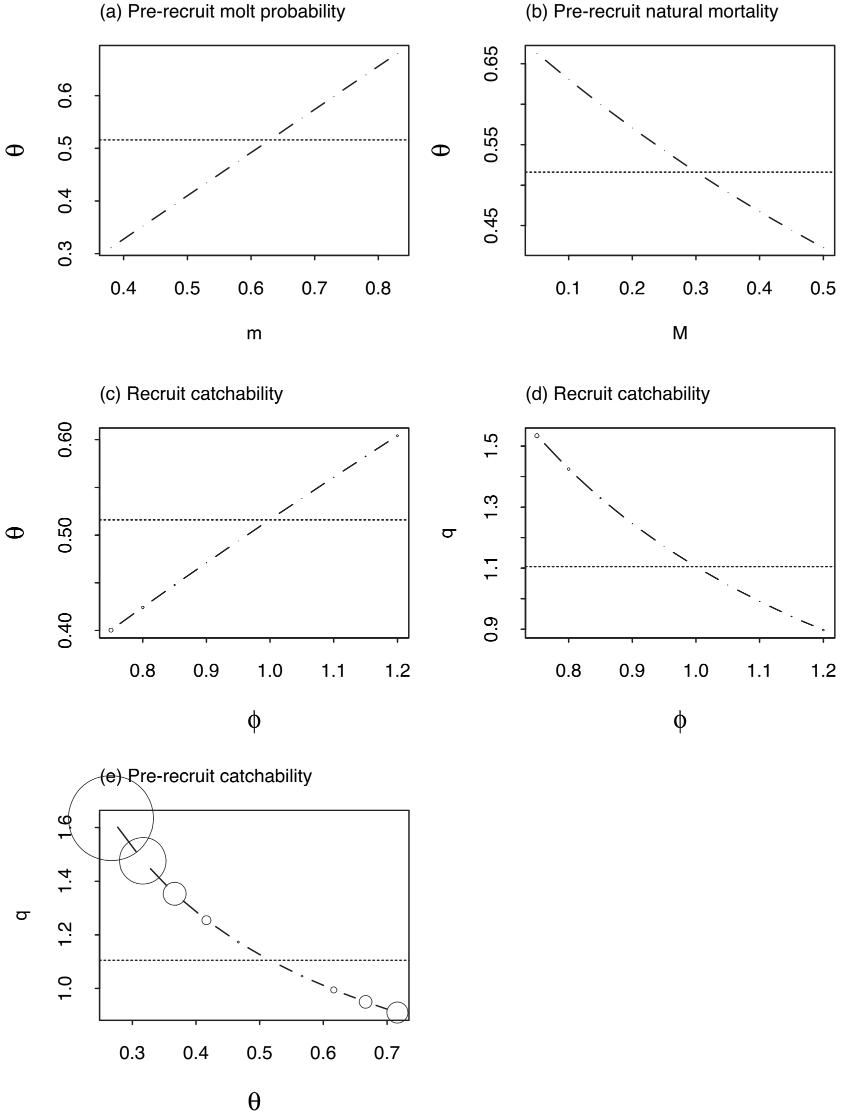

Figure 2. **Sensitivity of the estimated post-recruit catchability coefficient (*q*) and pre-recruit catchability (θ) to errors in specifying the input values of pre-recruit molt probability (m), natural mortality (*M*), recruit catchability (ϕ), and θ. The reference values were q = 1.105, θ = 0.516, *m* = 0.63, *M* = 0.3, and ϕ = 1. In each subplot, the value of the abscissa was varied over the range of points shown, and the value of the ordinate re-estimated with three-stage CSA. The areas of the circles are proportional to the increase in the sum of squared residuals.**

Sensitivity tests

The St. Matthew stock was used as a representative example for sensitivity tests and simulation analyses. We examined the sensitivity of the estimated post-recruit q, and pre-recruit θ, catchability coefficients to errors in specifying the input values of pre-recruit molt probability m, natural mortality M, recruit catchability ϕ, and pre-recruit catchability θ (Fig. 2). The absence of an increase in the sum of squared residuals indicates that pre-recruit catchability θ, pre-recruit molt probability, and natural mortality are structurally confounded (Fig. 2a,b). It is therefore possible to estimate only one of these three parameters in the model fit, in this case pre-recruit catchability. Pre-recruit catchability and natural mortality are negatively correlated. The way the three-stage model is formulated, molt probability of pre-recruits is analogous to natural survival and thus inversely related to M. Thus, estimates of θ should be positively related to molt probability (Fig. 2a). A small fraction of pre-recruits may molt directly into the post-recruit size range without being recruits. However, this is offset by a small fraction of recruits whose molt increment is insufficient to put them in the post-recruit stage.

Errors in molt probability or pre-recruit natural mortality directly influence pre-recruit abundance estimates. For example, if we assume that pre-recruit mortality equals recruit and post-recruit mortality (when in fact it is higher for smaller crabs), θ would be overestimated and pre-recruit abundance would be underestimated. Thus, accurate estimates of the absolute abundance of pre-recruits depend on good estimates of molt probability and natural mortality. Estimates of both pre-recruit catchability and recruit catchability are inversely related to estimates of post-recruit catchability (Fig. 2d,e). However, as the three-stage model compensates for errors in recruit catchability by changing pre-recruit catchability (Fig. 2c), errors in pre-recruit catchability increase the sum of squared residuals much more than errors in recruit catchability (Fig. 2e).

Monte Carlo simulation analyses

Using the St. Matthew Island fit to data from 1980-1997 as the “true” population, we performed the same simulations with the two-stage and three-stage models. We performed Monte Carlo simulations assuming correlated and uncorrelated observation errors, process errors, and mixed process and observation errors. Generally, we expect a combination of both measurement and process errors. Table 3 contains details of the errors used in the simulations and a comparison of the bias and standard deviation of q, the average percent bias of the abundance estimates, average magnitude (absolute value) of the percent bias of the abundance estimates, the average coefficient of variation, of the abundance estimates and the number of converged replicates out of 1,000 for the two-stage and three-stage models. About 98% of the simulations converged in all

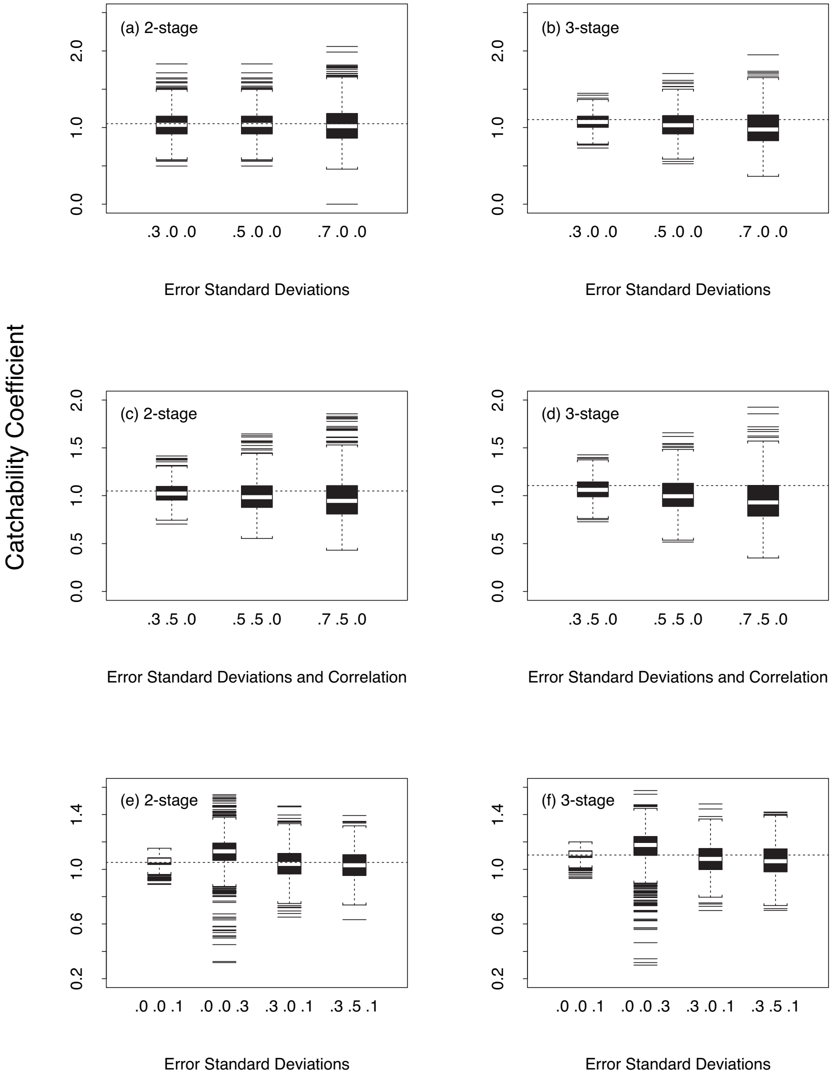

Figure 3. Distributions of the catchability coefficients (*q*) estimated in the Monte Carlo simulations of the 2- and 3-stage CSA. Each box plot summarizes the distribution of 1,000 *q* values. The white bar in the interior of the box is the median. The height of the box is equal to the interquartile distance (IQD). The whiskers extend 1.5 times the IQD, or to the extreme range of the data, whichever is less. Individual lines beyond the whiskers are outliers. The three numbers under each box indicate the standard deviation of the observation errors, the correlation of observation errors, and process error standard deviation (from left to right).

Table 3. Results of the Monte Carlo simulations of the 2- and 3-stage CSA.

Standard deviation of measurement error	Correlation of measurement errors	Standard deviation of process errors	Number of replicates that converged		Bias of q estimates		Standard error of the q estimates	
			2-stage	3-stage	2-stage	3-stage	2-stage	3-stage
0.3	0.0	0.0	999	999	–0.01	–0.03	0.11	0.11
0.5	0.0	0.0	998	1,000	–0.01	–0.06	0.18	0.18
0.7	0.0	0.0	995	998	–0.02	–0.10	0.25	0.24
0.9	0.0	0.0	995	983	–0.02	–0.14	0.31	0.30
0.3	0.5	0.0	1,000	1,000	–0.02	–0.04	0.11	0.11
0.5	0.5	0.0	999	1,000	–0.06	–0.09	0.17	0.18
0.7	0.5	0.0	997	995	–0.08	–0.15	0.23	0.23
0.9	0.5	0.0	979	985	–0.10	–0.21	0.29	0.29
0.0	0.0	0.1	988	983	0.01	0.00	0.04	0.04
0.0	0.0	0.3	989	931	0.07	0.05	0.14	0.14
0.3	0.0	0.1	998	993	–0.01	–0.03	0.12	0.12
0.3	0.5	0.1	1,000	995	–0.02	0.00	0.11	0.04

Table 3. (Continued). Results of the Monte Carlo simulations of the 2- and 3-stage CSA.

Standard deviation of measurement error	Correlation of measurement errors	Standard deviation of process errors	Legal abundance estimates					
			Average % bias		Average absolute % bias		Average CV	
			2-stage	3-stage	2-stage	3-stage	2-stage	3-stage
0.3	0.0	0.0	–0.6	0.3	0.7	0.2	15.29	13.22
0.5	0.0	0.0	–2.2	–0.2	2.5	0.8	26.72	24.06
0.7	0.0	0.0	–4.9	–2.5	5.3	2.8	38.36	37.35
0.9	0.0	0.0	–9.0	–6.4	9.7	6.3	51.02	54.94
0.3	0.5	0.0	0.7	1.4	0.6	1.2	14.71	13.42
0.5	0.5	0.0	0.8	2.4	1.4	2.4	25.66	24.66
0.7	0.5	0.0	–0.9	1.1	2.8	4.5	37.96	39.37
0.9	0.5	0.0	–4.3	–2.2	6.1	7.0	50.93	59.55
0.0	0.0	0.1	–0.3	–0.3	0.4	0.4	11.00	12.83
0.0	0.0	0.3	–3.2	–5.2	3.2	5.2	46.69	58.94
0.3	0.0	0.1	–0.1	0.5	0.4	0.6	19.56	19.23
0.3	0.5	0.1	0.6	1.3	0.5	1.3	19.28	19.41

cases except for one; only 93% of the runs using the three-stage model and process errors with standard deviation of 0.3 converged.

The simulation results from both models were qualitatively similar (Fig. 3, Table 3). According to a chi-square goodness of fit test, the distribution of the logarithm of the simulated q and θ values from the three-stage model did not differ from the normal distribution, indicating that these parameters are lognormally distributed. In all simulations, the bias of q tended to be smaller with the two-stage than with the three-stage model, but the standard errors of q were similar between the two models. The standard error of q increased as the standard deviation of the observation errors was increased. Adding correlated observation errors did not affect the standard error of q, but caused an increase in the bias of q in both models. In the simulations with process errors only, the bias of q was large and positive with both models. However, with a combination of correlated and uncorrelated measurement and small process errors, the bias of q became very small in both models (Fig. 3).

The main interest for stock assessment is the ability of the models to estimate population abundance accurately. Perhaps the best comparison of the two models is the average percent bias of legal abundance, average magnitude of the bias of legal abundance, and the CV of legal abundance from the two models (Table 3). In the runs with uncorrelated observation errors, the output from the models indicates that the average bias, average magnitude of the bias, and the CV of the abundance estimates are smaller with the three-stage model than with the two-stage model. With uncorrelated observation errors, the three-stage formulation appeared better able to smooth this variability than the two-stage model.

The average bias of the legal abundance estimate in each year from the two-stage model increased and became more negative as the standard deviation of observation errors increased (Fig. 4a,b,c). The largest bias occurred during 1982-1985 when the population abundance was decreasing to very low levels. The average yearly bias of the three-stage model also increased and became more negative as the standard deviation of the observation errors increased. However, in comparison with the two-stage model, the magnitude of the bias with the three-stage model was greater during the period 1982-1985 and less during the remainder of the time series.

In general, one may expect a slightly positive bias in the abundance estimates because the simulated lognormal process errors cause some large positive deviations from the fitted input data. However, we simulated the lognormal observation errors to have mean 1.0, which causes the median of the observation errors to be less than 1.0. As the standard deviation of the observation errors increases, the median shifts to a lower and lower value (0.94 with a standard deviation of 0.3, and 0.81 with a standard deviation of 0.7). Furthermore, with a standard deviation of 0.3 and 0.5, there was a significant correlation ($P < 0.07$ that the correlation

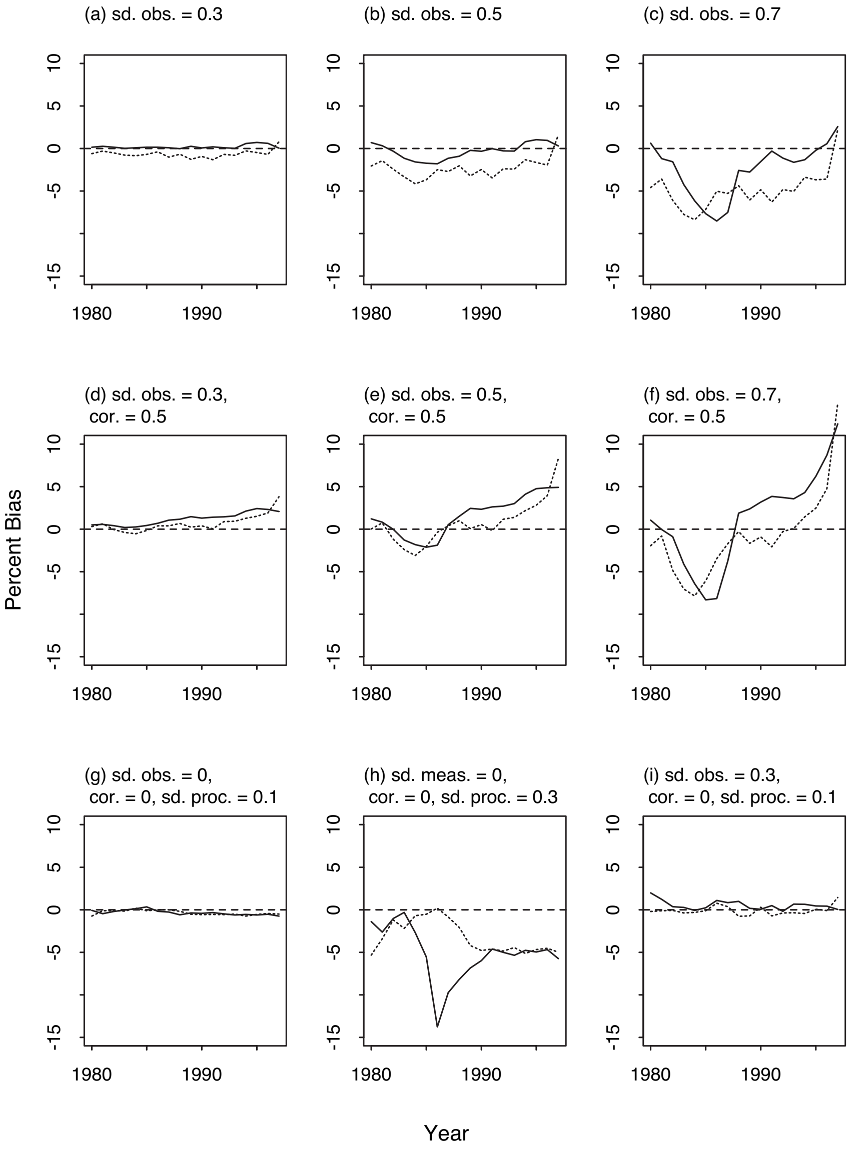

Figure 4. Percent bias by year of legal abundance from Monte Carlo simulations with the 3-stage (solid line) and 2-stage (dotted line) models with different combinations of observation and process errors. The abbreviations indicate the standard deviation of observation errors (sd. obs.), the correlation of observation errors (cor.), and the standard deviation of process errors (sd. proc.).

is zero) between the median of the observation error for one run and the average percent bias of the population estimates of the run. As a result, as the standard deviation increased, the bias became more negative.

The percent bias of legal abundance was quite large during the period 1982-1985 (Fig. 4). During the intermediate years (1982-1987), the legal abundance decreased and became quite low in comparison to the earlier and later years. The percent bias was most likely amplified during the years of low abundance because of the logarithmic transform in the objective function. The simulated relative abundance estimates could become very small for these intermediate years, especially as the standard deviation of the observation error increased. The logarithm in the objective function places greater emphasis on the residuals during the years of low abundance and the model tries hard to fit these small values. This bias may be more extreme with the three-stage model than with the two-stage model because there are three residuals associated with the small cohorts rather than two.

In the simulations with correlated observation errors, the two-stage model appeared to out-perform the three-stage model. It also appeared to do better than the two-stage model with uncorrelated observation errors (Table 3). This counter-intuitive result is opposite from the findings of Collie and Kruse (1998) in their simulations with the two-stage model applied to the Bristol Bay red king crabs and from our simulations of correlated and uncorrelated errors with the three-stage model. However, over the range of expected observation errors (standard deviation of 0.3-0.5), the average magnitude of the percent bias of legal abundance was 1.4% and 2.4% from the two- and three-stage models, respectively, a particularly small bias. The most extreme biases occurred during the period 1982-1985 (Fig. 4d,e,f) as with the uncorrelated observation errors, and for the same reasons.

With the process-error assumption, the three-stage output had slightly larger bias in the legal abundance estimates than the output from the two-stage model (Table 3). One possible reason for this is that the process error is compounded over two stages in the three-stage model, recruit and post-recruit. The difference was most apparent in the run with just process error with a standard deviation of 0.3 (Fig. 4h). Again, the three-stage model had the largest bias during the intermediate years when the population abundance was very low. The three-stage model also had a notably larger CV of legal abundance in this particular run.

St. Matthew Island blue king crab

The final runs were made with data from the period 1978-2003, with bootstrapped confidence intervals. Estimates of legal crab abundance from the three-stage model followed those from the two-stage CSA closely, although the three-stage model predicted slightly higher legal abundances for all but 5 years (Fig. 5a). As we set the instantaneous natural mortality

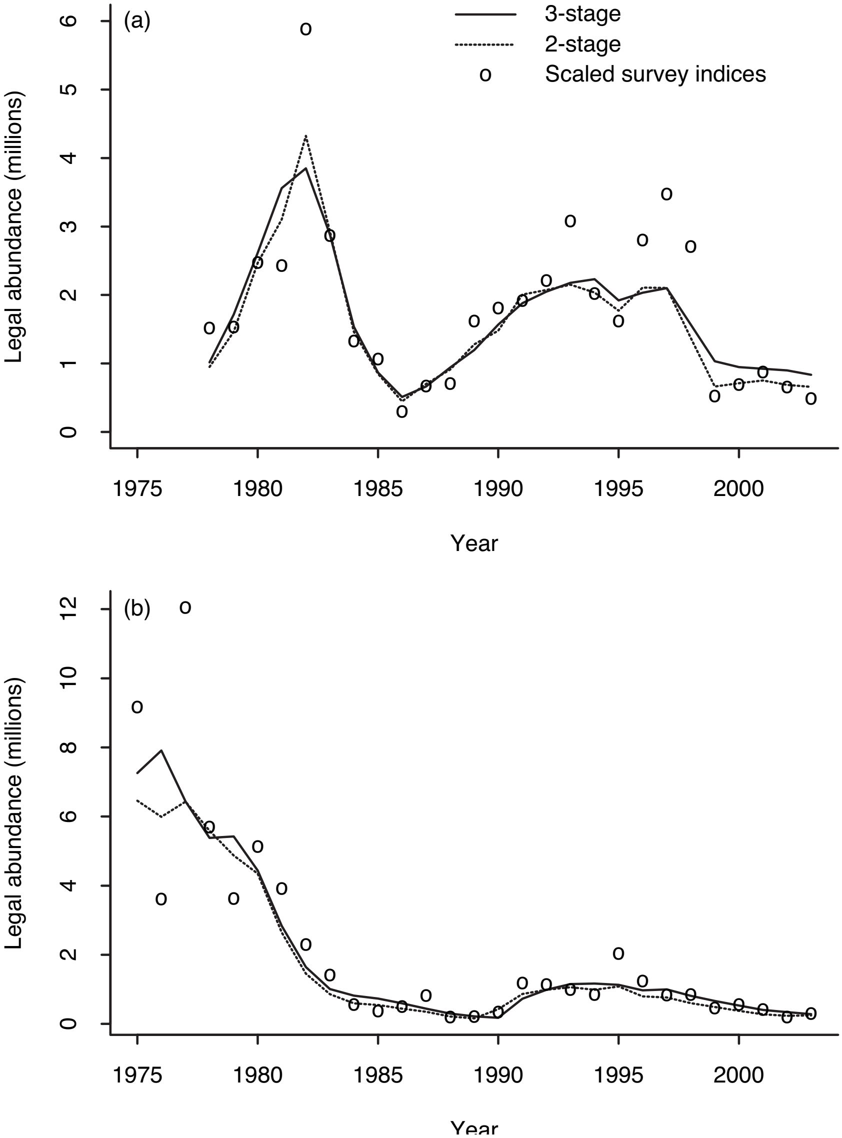

Figure 5. Comparison of the 2- and 3-stage catch survey analyses for (a) St. Matthew and (b) Pribilof Islands blue king crab stocks.

Table 4. Comparison of the two- and three-stage CSA output for the final runs with all years.

	Pribilof Islands		St. Matthew Island	
	Two-stage	Three-stage	Two-stage	Three-stage
Catchability coefficient	1.130	0.984	1.280	1.170
Standard error of q	0.488	0.968	0.436	0.428
Degrees of freedom	27	54	24	48
Sum of squared residuals	5.445	46.919	4.604	10.883
Mean square error	0.202	0.869	0.192	0.227

rate to 0.3 for all years, these results are consistent with the results of Zheng et al.'s (1997) two-stage CSA. In comparison with the two-stage model, the three-stage model smoothed and moderated abundance levels through time with the peak in 1982 reduced from about 4.5 to 3.9 million crabs. The estimated catchability of pre-recruits relative to post-recruits was estimated to be 0.60. The catchability coefficients and mean square errors from the two- and three-stage CSAs were comparable (Table 4).

The bootstrap runs resulted in a very small (–0.073) bias in the catchability coefficient and a considerably smaller (0.004) bias in the mean estimate of the catchability of pre-recruits relative to post-recruits (θ). The bootstrapped average percent bias of mean legal abundance was 0.29 and the average coefficient of variation of legal abundance was 19.5. The bootstrapped abundance means coincided almost exactly with the model estimates of legal and mature crab abundance, indicating low bias in the estimation. The 95% confidence intervals are wider at high abundance because of the lognormal error structure and they are narrower during periods of high exploitation because the model more heavily constrains the abundance estimates (Fig. 6). After increasing in the 1990s, the St. Matthew stock decreased abruptly in 1999 and the fishery has been closed since then. Zheng and Kruse (2000) attributed this abrupt decrease to high natural mortality in 1999.

Pribilof Islands blue king crab

Our updated analysis used data for years 1975-2003 (Table 1b). The residual standard deviation from this fit was 0.93, which is quite high in comparison with the other stock. For the bootstrap simulation we used an input standard deviation of 0.36, which was calculated by removing the four largest residuals associated with the zero survey abundance estimates. The catchability coefficient was slightly negatively biased (–0.026) and the catchability of pre-recruits relative to post-recruits was slightly positively biased (0.010) in the bootstrap fits. The bootstrap average percent bias of mean legal abundance was –0.87 and the average coeffi-

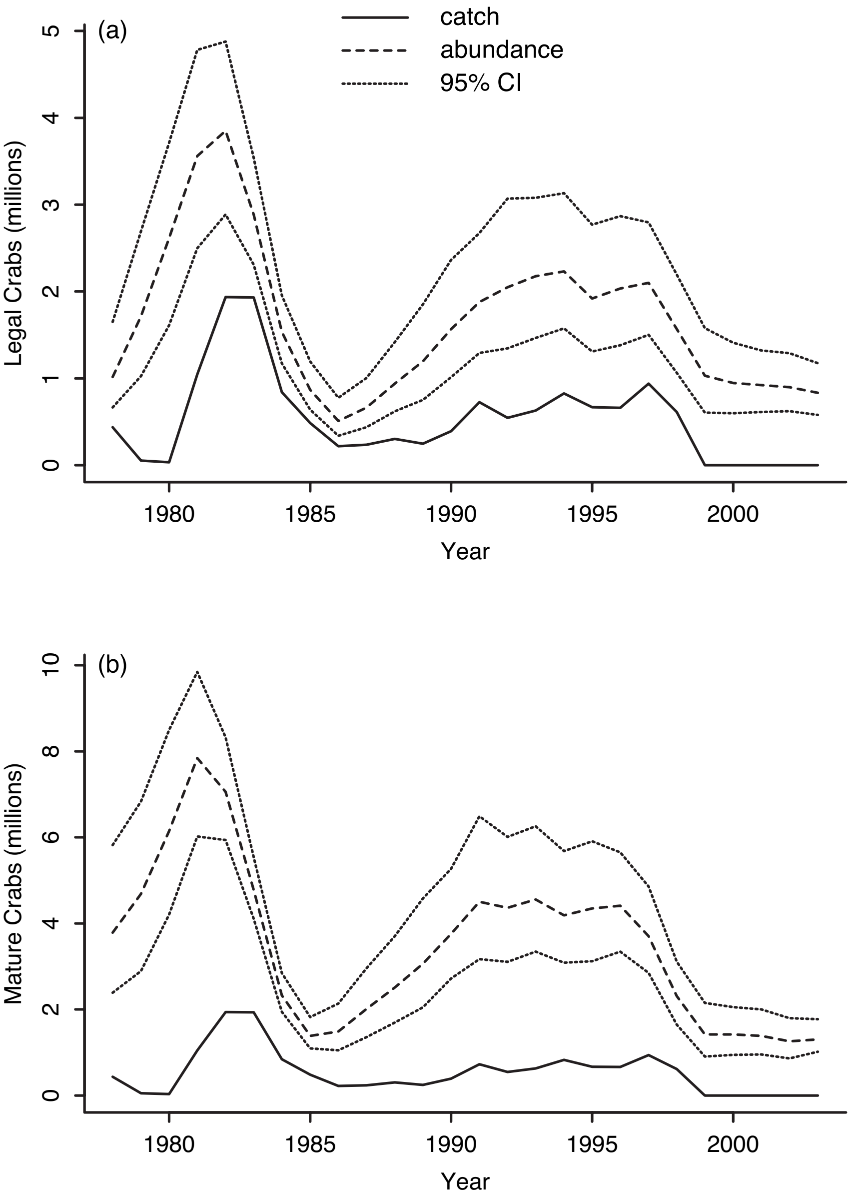

Figure 6. Legal and mature crab abundances and bootstrapped 95% confidence intervals from the 3-stage model for the St. Matthew Island blue king crab stock.

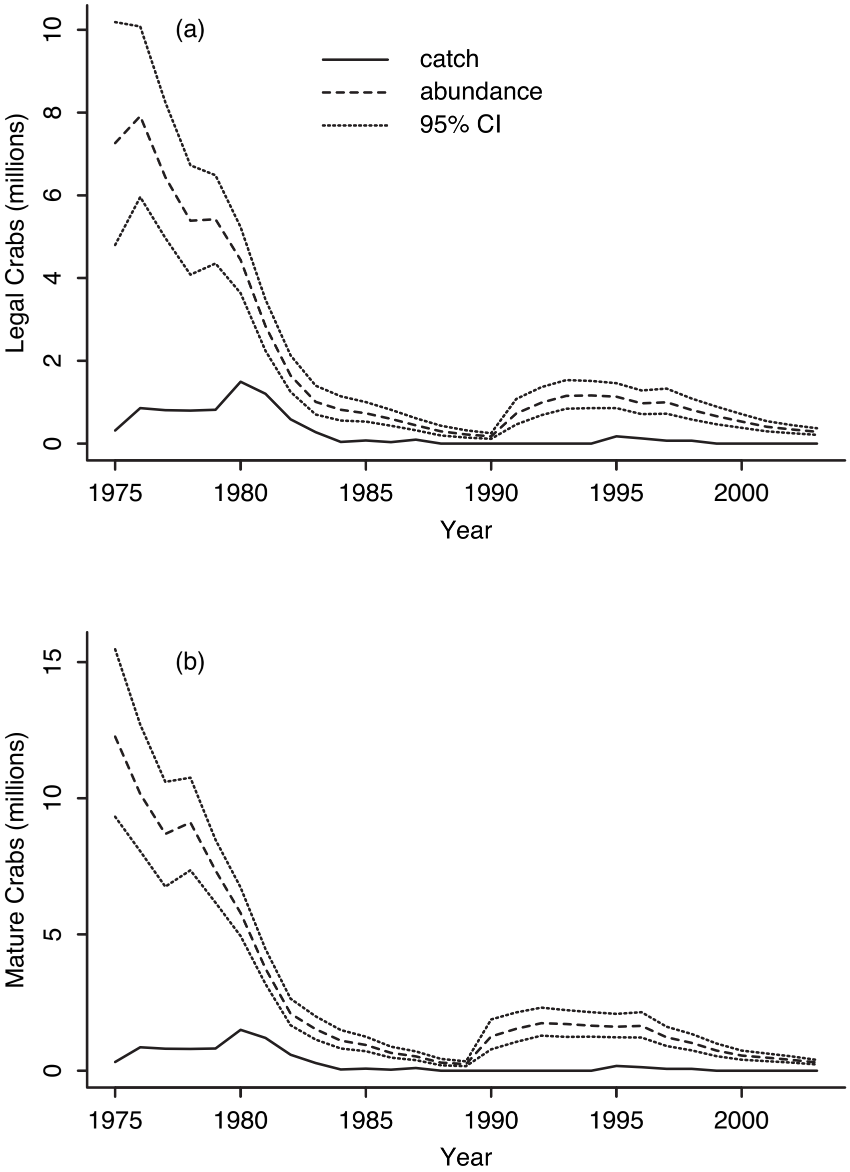

Figure 7. **Legal and mature crab abundances and bootstrapped 95% confidence intervals from the 3-stage model for the Pribilof Islands blue king crab stock.**

cient of variation of legal abundance was 15.71. The model and bootstrap estimates of legal and mature crab abundance and the corresponding bootstrap 95% confidence intervals are plotted in Fig. 7a,b. After a modest recovery in the early 1990s the abundance of Pribilof Islands blue king crabs declined and the fishery has been closed since 1999.

There were a few large residuals from the three-stage model fit for the Pribilof Islands stock. These large residuals are associated with the cohorts with very low abundance as described in the residual analysis section of this paper. We found that the standard error of the residuals is sensitive to the offset value δ, used in the objective function; offset values of 0.1, 0.01, 0.001, and 0.0001 yielded residual standard deviations of 0.45, 0.69, 0.93, and 1.29, respectively. Woodby (1994), in his estimation of the Southeast Alaska red king crab population size, used the smallest offset value that allowed the minimization subroutine to converge. Comparing the effect of the offset value on the estimation of population abundance, we found that for the wide range of offset values listed above there was a relatively small difference in population estimates.

For the Pribilof Islands stock, the three-stage CSA results were consistent with the results from the two-stage CSA except in 1975 and 1976 (Fig. 5b). The three-stage model predicted higher legal abundance because of the high pre-recruit index in 1975. During the periods from 1980 to 1989 and 1993 to 2003, the legal abundance estimates from the three-stage model were slightly higher than those from the two-stage model. The two-stage model predicted slightly higher abundances in 1990, 1991, and 1992. The catchability of pre-recruits relative to post-recruits at 0.81 was larger than the value estimated for St. Matthew, perhaps due to the larger size of this class. The catchability coefficient was estimated to be 0.98 in the three-stage model and was slightly lower than the q from the two-stage model (Table 4). The sum of squared residuals at 46.9 was quite high in the three-stage model, with the 1989 pre-recruit cohort contributing 36% of the sum of squared residuals and the 2000 pre-recruit cohort contributing 26%. In years with zero pre-recruit or recruit abundance indices, the model predicts a small, positive value to be consistent with non-zero indices in the previous and following year. The residuals were accentuated for these low indices due to the logarithm in the objective function.

Discussion

The three-stage catch survey analysis appears to have several benefits over the two-stage CSA as it applies to king crabs. First, a smoothed value of legal abundance can be estimated for the most recent year with the three-stage model. The two-stage CSA is unable to estimate recruit abundance in this last year; thus the unsmoothed survey index must be used. The three-stage model can also be used to estimate the abundance of mature male king crabs, because the size of pre-recruits corresponds approximately to the size at maturity of male king crabs (Somerton and MacIntosh 1983). This is important for fishery managers because the commercial quota is based upon estimates of both legal and mature crab abundance. However, the three-stage model cannot estimate pre-recruit abundance in the terminal year, so model estimates of legal crabs must be combined with the survey index of pre-recruits to estimate mature abundance in the last year.

The three-stage model uses more data than the two-stage model and therefore has a higher ratio of observations to parameters. For the three-stage model only two more parameters need to be estimated, one relative abundance estimate and θ. The bootstrap simulations indicate that θ is well defined in this model. The coefficient of variation of θ from the simulations was 13.3% for St. Matthew Island and 9.5% for the Pribilof Islands, which resulted in quite narrow confidence intervals for θ. For the St. Matthew Island stock, the mean square error of the three-stage model was comparable to the two-stage model (Table 4). In the Pribilof Islands case, the high mean square error resulted from the zero abundance indices.

The sensitivity analyses revealed that the three-stage model is able to smooth uncorrelated observation errors better than the two-stage formulation. As a result, the bootstrapped confidence intervals about the legal abundance estimates should be narrower with the three-stage than with the two-stage model. This was the case for Bristol Bay red king crab legal abundance confidence intervals when we compared the three-stage model (not shown) to the two-stage model of Collie and Kruse (1998). More detailed analysis will be necessary to diagnose the reason why the two-stage model with correlated observation errors performed better than the three-stage model with corresponding error structure. Also, the two-stage formulation is slightly more robust than the three-stage model with regard to process errors, which makes sense because the process errors are compounded over two time steps with the three-stage model as opposed to one time step with the two-stage model. Sensitivity analysis showed that estimates of pre-recruit molt probability and natural mortality are critical to the estimates of pre-recruit absolute abundance and, consequently, the mature crab abundance estimates.

The main purpose of this study was to develop and test a three-stage catch-survey analysis. The blue king crab examples were meant more to illustrate the method than as definitive assessments of these two stocks. However, CSA is used to set harvest guidelines for some Alaska king crab stocks. The harvest rate of blue king crabs is set on a sliding scale from 10% to 20% depending on mature crab abundance (Zheng and Kruse 2000). Below a minimum abundance threshold the fishery is closed, as has been the case for both stocks since 1999. This harvest strategy underscores the importance of accurate abundance estimates and of filtering out observation errors in the most recent year to the extent possible.

Additional extensions to three-stage catch-survey analysis are possible. Jie Zheng (ADFG, pers. comm.) pointed out that pre-recruits that don't molt can survive as pre-recruits the following year. This information could add additional structure to the three-stage model, especially if the shell age were known for the pre-recruit size class. Even without observations of shell age, the predicted number of old-shell pre-recruits (surviving pre-recruits that didn't molt the previous year) should be less than or equal to the predicted number of total pre-recruits for each year.

We examined the output from the baseline runs of the three-stage model to see if this constraint was met. In every case but one the total pre-recruit estimate exceeded the prediction of old-shell pre-recruits. The exception occurred in 1988 for the Pribilof Islands stock, for which the pre-recruit index was 0.0 and the predicted old-shell pre-recruits was 0.0075, a very small difference. For these king crab stocks, the proportion of pre-recruits not molting and surviving is small, around 0.2. Thus the extra constraint is unlikely to be binding. However, partitioning the pre-recruit size class into new and old shell categories could provide additional model structure to help estimate confounded parameters.

In some applications it is necessary to convert the harvest rates h to the corresponding fishing mortality rates F, for comparison with biological reference points. Given the assumption of a discrete harvest, the most consistent approximation is $F \approx -\ln(1 - h)$. However, the reference point should be calculated in the same way, and for stocks that are assessed with CSA, it may be preferable to express the reference points as harvest rates.

In future examinations of the three-stage model, one might investigate the effect of autocorrelated process errors, as might be expected because of environmental shifts. Further, it would seem appropriate to explore lognormal observation errors for the aggregate survey index and multinomial classification of errors within each year. In their original formulation of CSA, Collie and Sissenwine (1983) estimated process and observation errors. Our Monte Carlo simulations with two- and three-stage models confirm that the all observation error version is robust to realistic levels of measurement and process error. Cadrin (2000) used Monte Carlo simulations to evaluate the performance of the mixed-error two-stage CSA; similar studies are needed for the three-stage CSA. More rigorous and realistic assumptions about the error distributions could be incorporated in a full likelihood model (Punt 2003).

Recently, Zheng and Kruse (2000) applied a four-stage CSA to the estimation of blue king crabs. The size of the fourth stage is one molt below the pre-recruit stage. By incorporating additional data into the estimation it is possible to obtain model estimates of mature abundance in the most recent year. Adding more stages to the model is a logical progression toward a full length-based model with stochastic growth (Zheng et al. 1995). Yet a full length-based model is not possible for these blue king crab stocks owing to limited size-structured data stemming from difficulties conducting trawl surveys in these rugged areas. Optimal model complexity depends, in a large part, on how the results are used. For setting harvest guidelines, a simpler model can often perform as well as a more complicated model. In the case of king crabs, incorporating additional stages is clearly important for setting harvest guidelines. However, this comes at the cost of adding potentially noisy data that can decrease model precision. Therefore model performance needs to be evaluated with simulation studies, such as we have performed here, on a case-by-case basis.

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Bias of Equilibrium-Based Estimators under Biological and Fishery Disequilibria

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Abstract

Steady state or equilibrium assumptions have been recurrent temptations in fishery science to circumvent the problem of limited data and to provide managers with biological parameters and population status estimates in data-poor situations. Historically, fishery scientists and ecologists have used equilibrium approaches to produce estimates of key population parameters (e.g., mortality rates and abundance) with applications ranging from small-scale artisanal to large-scale industrial fisheries. Unfortunately, these methods rely on many restrictive hard-to-test assumptions, especially for those related to the lack of representation of the underlying dynamics. Size (mostly length) frequency data are among the basic statistics collected in most fisheries. Length-based cohort analysis and length-converted catch curve analysis (LCCCA) are part of the methodological toolkit available to modelers to obtain abundance estimates and mortality rates. Ingenious variations and extensions of these methods have been proposed but the uncertainty in model structure has virtually not been addressed. We evaluate the performance of equilibrium-derived estimators using two different methods. We implemented a dynamic operating model to show the lack of robustness of Jones' length-based cohort analysis to trends in fishing mortality rates. We also evaluated the performance of LCCCA in estimating natural mortality rates of the Patagonian scallop (*Zygochlamys patagonica*) by a cross comparison of these results to the estimates from a fully integrated statistical dynamic model. Substantial biases in parameter estimates suggest the importance of formally incorporating dynamics into the model structure. We recom-

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mend that mortality rates and abundance estimates derived from these methods should be cautiously accepted or not used at all. Indeed, scientists and managers should take more proactive measures to collect new data and develop a better understanding of the status and productivity of the stocks.

Introduction

Developing and small-scale artisanal fisheries often lack appropriate data time series to implement formal stock assessment analyses. These data constraints have historically tempted scientists to use simple alternative modeling approaches in an effort to provide fishery managers with reference points and/or stock status information (Sparre and Venema 1998). Many of these methods are based on restrictive assumptions that cannot be validated with the available data. Included among these methods are equilibrium-based techniques, which have a long tradition in ecological data analysis and fisheries stock assessment (Table 1).

Terms like "equilibrium," "steady-state," or "pseudo-equilibrium conditions" have been widely used in population biology to describe different concepts. Caddy (1996) points to alternative ways in which equilibrium assumptions can be introduced into stock assessment, for example by using stationary model parameters (e.g., natural mortality), implementing fitting procedures (e.g., Gulland-Fox equilibrium approximation methods) and using historical trend information to generate model predictions (most models). In a strict sense, many dynamic estimation procedures, which we commonly use from simple aggregated models to more complex age/length/stage structured models, can harbor equilibrium assumptions, i.e., in the initial conditions (Punt 1990, Hilborn and Walters 1992). This situation differs greatly from rigid equilibrium assumptions used to generate length-based equilibrium estimators, which are popular in tropical and artisanal fish stock assessments (Pauly and Morgan 1987).

Sparre and Venema justify the use of simple equilibrium methods through the following rationale: "We are often in a situation which forces us to make assumptions, which are known to be crude approximations to reality. It often happens that only by making such assumptions we are able to carry out an analysis of available data, and it is better to do a crude analysis than none at all." (Sparre and Venema 1998, p. 133). This line of thinking reflects a proactive attitude toward resource management in data-poor situations, but can often create several additional problems: (a) produces very biased results and therefore bad management advice; (b) induces managers to develop an unrealistic confidence in those data and results (given the quantitative nature of these estimates); (c) leads managers to inappropriately address the problems of lack of data and research funding in one particular fishery, and therefore discourage the

Table 1. Some equilibrium methods/assumptions traditionally used in ecological and fisheries data analysis.

Type of analysis	Type of data	Estimation
Age-based catch curve analysis	*AFD*	Abundance-at-age/*Z*
Size converted catch curve analysis	*SFD/GP*	Abundance-at-size/*Z*
Van Sickle's method	*SFD/GP*	Total mortality
Jones' length cohort analysis	*SFD/GP/M*	Abundance-at-size/*Z*-at-size
Yield-per-recruit	*GP/M*	Age of entry/F_{max}
Dynamics age/length structured model	*AFD/SFD/GP/M/Idx*	Initial age and length structure
Fitting a biomass dynamic model	*C/f*	Stock productivity

LFD = length frequency data, *AFD* = age frequency data, *GP* = growth parameters, Idx = index of abundance, *M* = natural mortality, *Z* = total mortality, *C* = catch, *f* = effort.

implementation of more costly approaches (e.g., tagging); and (d) deters the development of alternative simpler robust management methods.

Analytical and computational problems of dealing with highly nonlinear models also historically promoted this venue of scientific inference (Table 1). Development of stock assessment techniques, aided by a large increase in computational power and availability of longer time series, has determined an increasing trend in the use of dynamic statistical nonlinear estimation techniques in many major fisheries (Megrey 1989, Hilborn and Walters 1992, Methot 2000, Ernst 2002). Nevertheless, stock assessments in small-scale artisanal fisheries often rely on equilibrium derived estimators, partially due to data limitations as well as the technicalities involved in the dynamic statistical model fitting approach.

There are several equilibrium methods readily available to estimate mortality rates and/or assessing stock status using size-frequency data and additional input parameters (Sparre and Venema 1998). Two of these approaches are Jones' length cohort analysis (JLCA) and length-converted catch curve analysis (LCCCA). Both methods assume that length-frequency data accurately describe a harvested population at steady state. The use of both techniques is still popular worldwide and implementations of these methods often appear in peer-reviewed and nontraditional journals (Table 2). Despite the generalized use of these techniques, especially in small-scale fisheries, few analyses have been aimed at evaluating the robustness of output statistics derived from these models to departure from equilibrium assumptions (Ernst 2002). Jones (1987) tested the performance of his estimator on cohort growth variability, but did not test the equilibrium assumption. Lai and Gallucci (1988) used analytical tech-

Table 2. Applications of Jones' "length cohort analysis" or "length converted catch curve analysis" on finfish, crustacean, and mollusk fisheries worldwide.

Species	Location	Source
Nephrops norvegicus	Mediterranean Sea	Sarda et al. 1998
Paneus stylirostris	Costa Rica	Gonzalez-Pajuelo et al. 1996
Sardinella aurita	Fujian Coast	Xu et al. 1995
Dosichus gigas	Gulf of California	Pierce and Guerra 1994
Sprattus sprattus	Bulgaria, Black Sea Coast	Daskalov and Prodanov 1994
Aristeus antennatus	Northwestern Mediterranean Sea	Demestre et al. 1993
Panulirus argus	Jamaica	Haughton and King 1992
Gadus morhua	Baltic Sea	Sullivan 1991
Scomber japonicus	Tsushima Current	Limbong et al. 1988
Paneus indicus	Tanzania	Wakwabi 1990
Theragra chalcogramma	Bering Sea	Honkalehto 1989
Protothaca staminea	Garrison Bay, Washington	Lai and Gallucci 1988
Strongylocentrotus pallidus	Northern Barents Sea	Bluhm et al. 1998
Schilbe intermedius	Cross River, Nigeria	Etim et al. 1999
Periophthalmus barbarus	Imo River estuary, Nigeria	Etim et al. 2002
Crassostrea rhizophorae	Santa Marta, Colombia	Mancera and Mendo 1996
Pristipomoides filamentosus	Mahe Plateau, Seychelles	Mees 1993
Zygochlamys patagonica	Argentinean Shelf	Lasta et al. 2001

niques to evaluate Jones' method for its sensitivity to input parameters, and indicated that abundance estimates were very sensitive to natural mortality estimates. Despite the relevance of local sensitivity analysis, it does not capture the effect of key model assumptions on potential biases of model outputs. Tests on other equilibrium length-based models have also been conducted. Somerton and Kobayashi (1991) implemented numerical experiments to test the Wetherall length-based regression estimator to departure from population equilibrium, and whose results indicated a serious lack of robustness. Ehrhardt and Ault (1992) presented a sensitivity investigation on Beverton and Holt's length-based total mortality model (Beverton and Holt 1956), which demonstrated a lack of robustness under biased input estimates of exploitable life span of the species considered in their analysis.

We think that the current published literature does not sufficiently warn fishery analysts about the potential risks associated with length-based equilibrium models used for yield management parameters and/or stock status information. The objective of the present work is to evaluate the impact of departure from equilibrium assumptions on parameter estimates produced by (a) Jones' length cohort analysis (Jones 1984) and (b) length converted catch curve analysis (Pauly 1984, Sparre and Venema 1998) using two different approaches. First, Monte Carlo simulation methods were applied to assess the effect of trends in fishing mortality on abundance estimates from JLCA, and second, the bias in LCCCA's natural mortality estimates for the Patagonian scallop was compared with results of a statistical integrated dynamic age-structured model. These examples aim at illustrating problems associated with these equilibrium estimators under common non-equilibrium conditions.

Material and methods

Jones' length cohort analysis

Estimation model

Jones' length cohort analysis (JLCA) was developed to produce stock size and total mortality estimates using very basic population data (Jones 1984). The method follows a formulation similar to standard age cohort analysis (Megrey 1989), but with the added complication of dealing with length as opposed to age. The main model assumptions can be summarized as follows: (a) an exponential decay function determines the demographic changes of a cohort, (b) continuous recruitment, (c) constant fishing mortality rate operating on the stock, (d) individual growth given by a deterministic von Bertalanffy growth model, and (e) a stationary set of fishing and biological parameters (F_{Term}, M, L_∞, k). A set of equations allows the computation of numbers-at-length based on fisheries' length frequency data (LFD) and four fishing and biological parameters (Jones 1984). The abundance in the last length interval is computed by a modification of the catch equation and estimation proceeds backward using a modified version of Pope's approximation (Jones 1984, 1987). Corrections are necessary to compute the average number of individuals in the population at each size interval. This procedure avoids biases based on the dependence of abundance estimates to the chosen length interval width of the LFD.

Operating model and data

The operating model was based on a dynamic age-structured model (see Appendix I for details of model equations and Appendix II for parameter values). The abundance at age was propagated using standard demographic equations and the simulation horizon was 20 years. This provided a reasonable temporal framework to test different exploitation trends. Fishing mortality was divided into temporal and age components

(Megrey 1989). Logistic models were used to describe the selectivity and maturity ogives, and recruitment was driven by a Beverton-Holt relationship. The operating model was parameterized by data from the squat lobster population (*Pleuroncodes monodon*) off central Chile (Appendix II). This is a temperate zone galatheid that inhabits the continental shelf off central Chile and has a medium life span of about 7 years. Other important parameters that were used are: natural mortality = 0.5 yr^{-1}, k = 0.197 yr^{-1}, L_∞= 5.045 cm, number of size intervals = 35. Sampling from the simulated population was performed using Baranov's catch equation. Numbers-at-age from the catch were assigned to the different equally spaced length intervals using a probability of length-given-age matrix (Fournier et al. 1998; see Appendix I for details). After constructing the deterministic length distribution of the catch, numbers collected at each length category were obtained by independently sampling the length intervals in a lognormal fashion (σ_{obs}, observation error levels of 0, 0.5, and 0.2 were used for scenarios A, B, and C-F, respectively). All the length frequency samples were taken at the beginning of each month and averaged to obtain an annual length frequency distribution. This was implemented to mimic the sampling process in a year-round fishery.

Non-equilibrium experiments were implemented as trends in fishing mortality rates (Fig. 1). Different scenarios included an equilibrium base case situation with a constant fishing mortality of 0.1 (scenario A), linear increasing trends in fishing mortality rate (to mimic a developing fishery) and linear increase in the first 10 years followed by a linear decrease (scenarios C-F). These trends are calculated by

$$F_{t+1} = F_t + F_{Step}$$

in which F_t is the fishing mortality rate in year t.

The simulation scheme is presented in Fig. 2 and considers 100 Monte Carlo trials for each case. Stock status was assessed by the equilibrium estimator in each time step using only LFD for that particular year; therefore there is no buildup of information along the time series.

Even though the mean square error is a more comprehensive statistic for evaluating estimator performance (Casella and Berger 1991), we focused on the bias component introduced by the equilibrium assumptions of the estimator. We used a relative error (ε_θ) as a measure of performance represented by

$$\varepsilon_\theta = \frac{(\theta_t^{Est} - \theta_t^{True})}{\theta_t^{True}}$$

where θ_t^{Est} and θ_t^{True} are the estimated (from JLCA) and true (from operating model) values respectively for the abundance at time t for one particular simulation run.

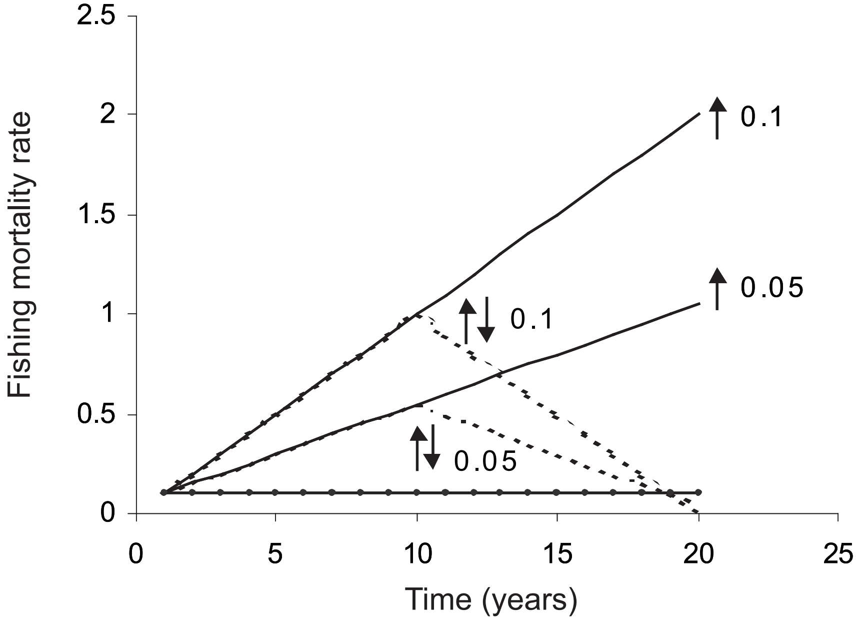

Figure 1. **Simulation of perturbation histories determined by trends in fishing mortality rates. Horizontal line corresponds to equilibrium scenarios with a constant fishing mortality rate of 0.1.**

Length converted catch curve analysis

Our second case study is presented to illustrate serious biases that arise in natural mortality estimation if we ignore the recruitment history and somatic growth dynamics of benthic resources. By reanalyzing and taking full advantage of the data we showed that equilibrium assumptions could be relaxed in favor of a statistical, integrated, and dynamic estimation procedure and we highlight bias levels in length-based estimates that arise under several data combinations aided also by simulation analysis.

Early uses of this approach are reviewed in Ricker (1975). This method assumes exponential mortality (constant), and therefore plotting the logarithms of the number of individuals caught (or sampled) against age results in a linear model with a slope equal to the total mortality (Z). This method was extended to versions that use length data, for cases where age data are not available (Pauly 1984, see Sparre and Venema 1998). Basically, lengths are mapped into ages using available growth

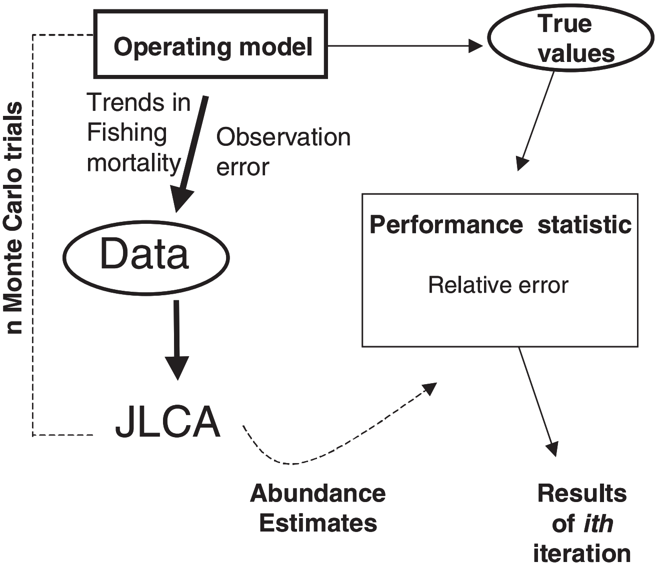

Figure 2. Simulation scheme for assessing the performance of Jones' length cohort abundance estimator. One hundred simulation trials were performed and the observation error was incorporated on the catch-at-length data in a log-normal fashion (CV = 0.2 for the last four panels of Fig. 3).

information. Key assumptions of this method include population at equilibrium, constant recruitment, and invariance of growth parameters across cohorts. Lasta et al. (2001) used this approach to estimate natural mortality using a combined size frequency distribution (SFD) from 1995 to 1998, by pooling bimonthly samples from an area closed to fishing. Here, we reconstructed their analysis and obtained further natural mortality estimates using size frequency distributions from each year independently (1995, 1996, 1997, and 1998) instead of pooling the data and discarding valuable information on the dynamics. In addition to the analysis of the raw data, we performed simulation analysis to assess the performance of an LCCCA equilibrium estimator. Simulated distributions were obtained from an age-structured operating model with known constant mortality, growth parameters, and either constant or variable recruitment (parameter estimates taken from Lasta et al. 2001 and Valero 2002). We organized the analysis into 11 scenarios (Table 3) that represent the estimation of natural mortality under different data conditions.

Table 3. Mortality estimates and associated asymptotic 95% confidence limits using different scenario inputs.

Scenario	Method	M yr^{-1}	CI 95%
1	Integrated model (IM)	0.31	0.21-0.42
2	Simulated data equilibrium ($M = 0.31$)	0.29	0.22-0.37
3	1996	0.55	0.39-0.71
4	Pooled 1995-2000 SFD recruits predicted from IM	0.67	0.61-0.74
5	Lasta et al. (2001)	1.04	Not reported
6	Pooled 1995-1998 SFD data	1.04	0.94-1.14
7	1995	1.19	1.02-1.36
8	Pooled 1995-2000 SFD data	1.30	1.14-1.45
9	Pooled 1995-1998 SFD predicted from IM	1.34	1.20-1.49
10	1998	2.10	1.80-2.39
11	1997	2.43	2.15-2.71

SFD = length frequency data.

Scenario 1 corresponds to natural mortality estimation using real data and the statistical integrated age-structured size-based model described in the following section. In scenarios 3, 7, 10, and 11 only a single year of SFD was used and for scenarios 5, 6, and 8 averaging of SFD over several years was implemented (see Table 3). These seven scenarios were also based on real data. Simulation testing was introduced in scenarios 2, 4, and 9, where mortality estimates from the equilibrium estimator were based on simulated SFD data either under constant recruitment (scenario 2) or recruitment history as estimated by the integrated model (scenarios 4 and 9). Confidence bounds were estimated following Sparre and Venema (1998).

Statistical dynamic age-structured model and data

We implemented an integrated age-structured size-based model largely following Fournier et al. (1998), in order to describe monthly dynamics of Patagonian scallop (*Zygochlamys patagonica*) abundance and shell growth. The model consisted of exponential survival equations, which account for natural mortality (the only source of mortality in this area), cohort-specific individual growth and variability of size-at-age and a logistic selectivity ogive of the sampling gear (Valero 2002). This model was fitted to data derived from bimonthly surveys in the *Reclutas* bed (central position: 39º24S, 55º56W, Argentina) in the northern Atlantic distribution of *Z. patagonica* from January 1995 to December 1998. Total bed area is approximately 1,020 km^2 (Valero 2002). The data set contains biological (i.e., size frequency distributions, total individual weights, and weights of specified body fractions) as well as survey fisheries data (e.g., survey catch per unit effort, catch composition). Since this section of the *Reclutas* bed is closed to commercial fishing we assumed that the only source of mortality was related to natural events described as an exponential decay of cohorts as a function of a time invariant natural mortality parameter.

The integrated modeling approach allowed the estimation of cohort-specific growth parameters and an overall natural mortality rate in a dynamic and integrated way. Maximum likelihood theory (Fournier et al. 1998) was employed to derive model parameter estimates and their associated uncertainty using nonlinear optimization procedures. This model simulates population dynamics and allows for simultaneous estimation of natural mortality, research vessel selectivity, cohort specific shell growth parameters, and year class strength. The maximum likelihood estimate of natural mortality constituted scenario 1, and the associated overall parameterization (model and parameter values) was used to simulate SFD for scenarios 2, 4, and 9.

Results

Jones' length cohort analysis and length converted catch curve analysis yielded biased results in cases that incorporated dynamic systems. JLCA was tested under trends in fishing mortality rates, simulating a developing fishery, and failed to accurately describe the abundance dynamics. On the other hand the performance of LCCCA, tested under variable recruitment (characteristic of sessile, highly spatially structured benthic populations), was poor and the size-based estimator overestimated natural mortality rates in all non-steady state scenarios. All these results assume a perfect knowledge of additional input parameters required to perform the analysis.

A 21% negative relative error was detected between the JLCA estimates and the operating model under equilibrium conditions (upper two panels of Fig. 3). This was attributed to underlying differences in model structure between the simulator and the equilibrium estimator, such as variability in length-at-age considered by the operating model versus deterministic growth assumed by JLCA. Observation error did not induce additional bias in the abundance estimates and only increased the dispersion around median values. Non-equilibrium scenarios mediated by fishing mortality rate trends strongly affected JLCA's ability to produce reliable abundance estimates. Increasing trends (Fig. 3, C and E) in fishing mortality induced a severe positive bias in the abundance estimates, and decreasing trends (Fig. 3, D and F) caused a negative bias in the abundance estimates. These changes were detected for low and moderate changes in fishing mortality rates (Fig. 3).

Natural mortality estimates derived from the integrated model and LCCCA are summarized in Table 3 and Fig. 4. Table 3 shows mortality es-

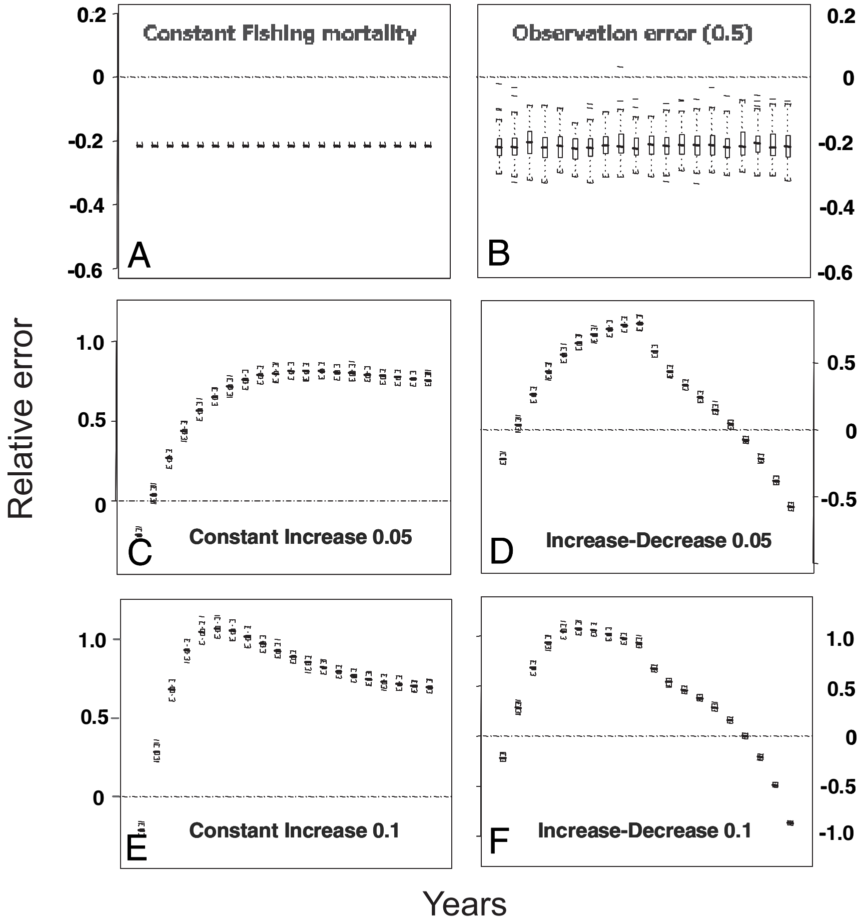

Figure 3. Time series (20 years) of relative error of abundance estimates (in numbers) derived from JLCA. First two panels correspond to equilibrium situations with (right) and without (left) observation error. Bottom four panels represent different perturbation histories induced by positive and/or negative trends in fishing mortality rates.

timates and associated asymptotic 95% confidence limits under different scenarios and data inputs. Only the scenario with constant recruitment (scenario 2) yielded results comparable with those obtained with the integrated model (scenario 1, Table 3 and Fig. 4). The mortality estimate of the integrated model is lower by a factor of three than previously reported by Lasta et al. (2001) (scenario 5, Table 3, Fig. 4). Mortality estimates obtained through LCCCA are greater than estimated by the integrated model by as much as a factor of 6 (scenarios 2-11, Table 3, Fig. 4). Simulation analysis showed that LCCCA lacks robustness to recruitment disequilibria and the confidence bounds derived from this approach are unrealistically small, not containing the true parameter value (Fig. 4).

Discussion

Equilibrium-based estimates provided by both methods were seriously biased under realistic departures from model assumptions. Validations of these methods have traditionally involved mostly local sensitivity analysis to input parameters, therefore largely have not evaluated the fundamental model assumptions from these analyses. Non-equilibrium conditions are clearly expected in incipient fisheries, due to the characteristic increase in fishing effort during the developing phase. The results reported for JLCA can be generalized to other species. Ernst (2002) showed similar trends for two finfish species.

We showed that LCCCA produced serious biases when estimating natural mortality for the Patagonian scallop. This method is broadly used worldwide to provide total mortality estimates which are often used as proxies of stock productivity during the quota recommendation process. This fishery, as with many other fisheries in the world (i.e., Zheng et al. 2002), uses natural mortality to approximate target reference points, and therefore the serious bias levels reported in this work could have a negative impact in the appropriate fishery management.

Model testing using complex operating models has proven useful in model comparisons (Punt 1988), or to assess the effect of strong restrictive model assumptions. Evaluation of model performance and robustness under non-equilibrium scenarios calls for the implementation of simulation analysis (e.g., Monte Carlo approach). Parrack (1990) indicated that the list of unknown parameters might be so lengthy for some specific statistical models that determination might be a problem, and therefore Monte Carlo tests should be implemented to analyze an estimator's performance. Simulation analysis has been implemented in a number of stock assessment applications, ranging from models of similar structural complexity to comparative analysis across different models or methodologies (e.g., Hilborn 1979, Ludwig and Walters 1985, Butterworth 1988, Punt 1997, Maunder 2001, Ernst 2002).

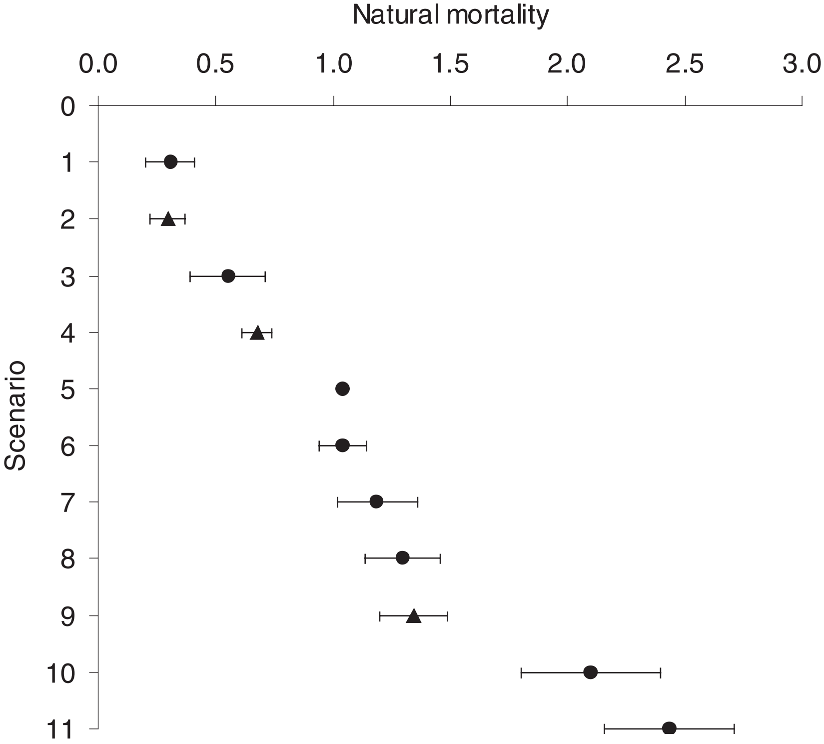

Figure 4. Natural mortality estimates derived from LCCCA for all scenarios described in Table 3. MLE was obtained from the statistical calibration of the integrated dynamic age-structured model. Solid triangles represent mortality estimates derived from simulated data; solid circles were obtained from observed data; horizontal lines represent confidence bounds.

Jones (1987) used an equilibrium simulator to assess the performance of his method, and therefore ignored the effect of steady-state assumptions on parameter estimates. Our approach includes a more generic formulation of the state of nature and addresses deficiencies in model performance. Evaluation of model performance of other equilibrium size-based estimators through simulation analysis has been conducted in the past. Somerton and Kobayashi (1991) developed a simulation model to test the robustness of the Wetherall length-based method for recruitment variability and fishing effort trends. *Z/K* (ratio of total mortality rate to individual growth rate) estimates derived from this approach presented large bias levels throughout their 15 year time series. Their results indicated that the Wetherall size-based estimator was very sensitive to recruitment and fishery disequilibria, a situation that is also present in both of our model testing results, and only recommended this approach under strict equilibrium conditions.

Temporal and/or spatial variability in recruitment is characteristic of small-scale, sedentary stocks, and spatially structured systems, nowadays called S-fisheries (Orensanz et al. 2005). Equilibrium methods are often used to assess the status of these populations due to lack of data or appropriate knowledge of alternative methods. It is unlikely that steady-state assumptions hold for these particular systems due to underlying dynamics or the sampling process scale. Caddy (1996) warned about the increasing use of equilibrium length-based methods and questioned the validity of some of their results, given that the underlying untested equilibrium assumptions probably are not met.

In addition to potential biases in abundance or mortality estimates associated with a model structure simplification, there is also an underutilization and lack of integration of input data. In order to match the data requirements of these methods, valuable auxiliary information is often ignored (Valero 2002). Additionally, the quantitative nature of the parameter estimates often creates overconfidence in managers, and discourages the development of further research. We disagree with Caddy (1996) and do not believe that managers are always critical and skeptical about the results of these assessments and their results often permeate directly into management.

Consequently scientists and managers can be misled and let the already available equilibrium models wrongfully dictate the kind of data collection or analysis required, instead of letting the available data dictate the nature of the models to be used to capture the dynamics and underlying uncertainty. Due to biases inherent to these methods, alternative robust approaches should be encouraged in data-limited situations.

Equilibrium assumptions will continue to play a crucial role for exploring and understanding dynamic systems behavior and this practice should be differentiated from attempting to estimate key population parameters under steady state assumptions (Caddy 1996). In light of the current results we do not advocate the use of either of these two equilibrium methods to assess stock status and/or estimate key demographic rates (perhaps only when equilibrium can be demonstrated); but rather encourage scientists to critically review the conditions under which these models are applied (Somerton and Kobayashi 1991).

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Appendix I. The population dynamics model

The simulation model considered in this appendix is based on a dynamic age-structured model. A probability matrix of length-given-age was used to represent the variability around the mean length-at-age. Yearly samples were collected and passed to the JLCA abundance estimator to assess stock size during a simulation framework of 20 years.

A. Basic population dynamics

The abundance at age was propagated using the following equations

$$N_{a+1,t+1} = N_{a,t} \exp\{-(M + F_t s_a)\} \tag{I.1}$$

$$\begin{aligned} N_{A,t} = {} & N_{A-1,t-1} \exp\{-(M + F_{t-1} s_{A-1})\} \\ & + N_{A,t-1} \exp\{-(M + F_{t-1} s_A)\} \end{aligned} \tag{I.2}$$

where

$N_{a,t}$ is abundance at age a and time t ,

F_t is instantaneous fishing mortality rate at time t,

M is instantaneous natural mortality rate,

A is the age of the plus group,

s_a is selectivity at age a, represented by the following logistic model:

$$s_a = \frac{1}{1 + \exp\left(-\left[\frac{\ln(19)(a - A_{50})}{(A_{95} - A_{50})}\right]\right)} \tag{I.3}$$

A_{50} and A_{95} are age of 50 and 95% of recruitment to the fishing gear.

B. Recruitment

Population renewal was modeled after a Beverton-Holt recruitment model

$$R_{t+1} = \frac{S_t}{\alpha + \beta S_t} \tag{I.4}$$

where

R_t is recruitment in year t,

S_t is spawning biomass at time t,

$$S_t = \sum_a w_a \Phi_a N_{a,t} \tag{I.5}$$

w_a is average weight of an individual at age *a* computed from the mean length-at-age and allometric parameters [= $\exp(\delta_1)(L_a)^{\delta_2}$],

ϕ_a is the maturity at age computed by the following logistic model:

$$\phi_a = \left\{ \frac{1}{1+e^{a_1+a_2 a}} \right. \tag{I.6}$$

α_1 and α_2 are parameters of the logistic maturity ogive.

The parameters of the Beverton-Holt model were re-parameterized as virgin recruitment (R_0), virgin spawning stock size (S_0) and steepness (z) by using the following equations:

$$\alpha = S_0 \frac{1-z}{4z \;\; R_0} \tag{I.7}$$

$$\beta = \frac{5z-1}{4zR_0} \tag{I.8}$$

C. Initial conditions

Initial conditions allowed for a historic equilibrium exploitation rate

$$N_{a,1} = R_{2-a} \prod_{i=1}^{a-1} \exp[-(M+F_{init}s_i)] \qquad a>1 \tag{I.9}$$

$$N_{A,1} = \frac{R_{2-A} \prod_{i=1}^{A-1} \exp[-(M+F_{init}s_i)]}{\{1-\exp[-(M+F_{init}s_A)]\}} \tag{I.10}$$

D. Length frequency samples

Baranov's catch equation was used to compute numbers-at-age in the catch for each time interval,

$$C_{a,t} = N_{a,t} \frac{F_t s_a}{F_t s_a + M}(1-\exp\{-[F_t s_a + M]\}) \tag{I.11}$$

Numbers-at-age were assigned to the different equally-spaced length intervals by using a probability of length given age matrix, parameterized with a mean length-at-age (μ_a) and a variance of length-at-age (σ_a^2):

$$P(l\,|\,a) = \frac{(2\pi\sigma_a^2)^{-1/2}\exp\{-[\frac{(l-\mu_a)^2}{2\sigma_a^2}]\}}{\sum_{l=l_1}^{l_n}(2\pi\sigma_a^2)^{-1/2}\exp\{-[\frac{(l-\mu_a)^2}{2\sigma_a^2}]\}} \tag{I.12}$$

$$\mu_a = L_\infty(1-e^{-k(a-a_0)})$$

$L\infty$, k and a_0 are standard von Bertalanffy growth parameters. In order to constrain the parameter space, σ_a^2 were computed based on σ_1^2 and σ_A^2 and a linear relationship with age.

Numbers collected at each length category were obtained by independently sampling the length intervals in a lognormal fashion

$$N_{l,t} = \left[C_{a,t}P(l\,|\,a)\right]\exp\{\varepsilon_{l,t}^{obs} - \frac{\sigma_{obs}^2}{2}\} \tag{I.13}$$

where

$\varepsilon_{l,t}^{obs}$ is a normal deviate with mean zero and variance σ_{obs}^2.

Appendix II. Model parameters

The following list contains all model parameters used in the equations of Appendix I.

Simulation parameter	Data set
Upper length interval	4.4
Lowest length interval	1.0
Step size for length interval	0.1
M	0.5
k	0.197
L_∞	5.045
a_0	–0.51
A	7
δ_1	–8.642
δ_2	3.31
R_0	10^7
Z	0.7
α_1	13.65
α_2	–0.502
σ_1	0.17
σ_A	0.27
A_{50}	3.1
A_{95}	4.0
F_{init}	0.1
Number of length categories	35

Using a Spatially Structured Model to Assess the Tasmanian Fishery for Banded Morwong (*Cheilodactylus spectabilis*)

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Abstract

Fisheries can be regarded as data poor if insufficient information is available to produce a defensible stock assessment. Data-limited fisheries can be (1) new fisheries with no time-series of information; (2) low-value fisheries for which little data are collected; (3) bycatch fisheries, which are often ignored; and (4) spatially structured fisheries where data collected may not be representative of the whole stock. This last category is especially problematic because when little data are available, stock assessments are generally limited to simple models. Unfortunately, the impact of fishing and our ability to assess a fishery can be greatly affected when the dynamics of a stock or stocks are influenced by some spatial structure. In the instance studied here, there are similarities to studies of fished areas adjacent to marine protected areas closed to fishing. A simulation study of a spatially structured fishery was undertaken, using the Tasmanian banded morwong fishery to mimic such a data-limited situation. It was found that the characteristics of the catch (age-structure and sex ratio), which were available through a relatively focused research program, could only be fitted using a model that included a spatially structured population, with fished and unfished components (based on depth) and movement in between. Based on such a fishery assessment, the stock was seen to be declining. This outcome contrasted greatly with the outcome of considering only the catch and effort data typically available for such inshore reef fisheries in Australia. The catch and catch rate information, when modeled as coming from a spatially homogeneous single population, gave the appearance of relative stability and sustainability. Fisheries for species likely to have some spatial structuring, such as widely distributed reef dwelling species, may possibly be far more complex and difficult to manage than the classic theory of homogeneous fish populations would suggest. Some form of explicit spatial management would be a more appropriate approach to ensuring long-term sustainability in such fisheries.

Introduction

A fishery is data poor if insufficient information is available to produce a defensible stock assessment. This definition moves away from a simple statement about the quantity of data and focuses instead on the purpose for which data are collected. Insufficient information for an assessment can arise because (1) a fishery is new or developing and a time series of information has yet to be collected; (2) sufficient specific information is unavailable in low-value fisheries irrespective of how long they have operated (research is expensive and extensive data collection is difficult to justify in such circumstances); (3) data collection tends to focus on the target species so bycatch fisheries constitute another common data-poor category, even though they may constitute a significant proportion of a target fishery's catch; and (4) fisheries that are spatially or otherwise structured where an assumption of spatial homogeneity has been adopted. Where a stock has spatial structure, simple data collections may not be representative of the whole and thus could not be used validly to assess the whole (e.g., Orensanz and Jamieson 1998, Parma et al. 2003). A spatially structured fishery seems likely to cause extra problems in data-poor circumstances because when little data are available, stock assessments are generally limited to simple models. Unfortunately, our ability to assess a fishery can be greatly affected when the dynamics of a stock or stocks are influenced by some spatial structure (Punt 2003). Spatial structuring of a stock can occur in many ways, making it difficult to produce a simple solution to the complexities present in the instance studied here. We present an example from a spatially structured (both along coast and onshore and offshore), low-value temperate reef fish fishery. The dynamics of the fishery, which involves onshore fished areas and offshore unfished areas, imply there are similarities to studies of fished areas adjacent to marine protected areas closed to fishing (Beverton and Holt 1957, Botsford and Hobbs 1986, Quinn et al. 1993, Haddon et al. 2003, Gerber et al. 2003). The objectives of this present work were to illustrate problems that can arise when one assumes that available data are representative of the dynamics of a spatially homogeneous stock when it is in reality spatially structured (Punt 2003), and to suggest a simple strategy that can be adopted in data limited situations to reduce the risk of generating inappropriate management advice.

Many reef fish populations demonstrate spatial structuring, with limited movement between reefs after settlement (e.g., McCormick and Choat 1987, Jones 1988, Newman and Williams 2001). Banded morwong (*Cheilodactylus spectabilis*), is such a species in New Zealand and southeastern Australia. Principally in Tasmania, banded morwong supports a small-scale fishery that services domestic live-fish markets. The fishery has a short history, being first developed in the early 1990s and producing a current annual catch of about 50 t (see Ziegler et al. 2005). The only data generally available for the scalefish fisheries in Tasmania are the legally required commercial catch and effort data. Fortunately, banded morwong has, in addition, been the subject of a previous research project (Murphy and Lyle 1999), and of recent fishery-independent catch samples, which have provided sex ratio, age-structure, and location data relating to the catch. Banded morwong exhibit an unusual combination of very fast initial growth, an early age of maturity and a long life expectancy, with maximum ages greater than 80 years for both sexes (Murphy and Lyle 1999). In addition, growth rates and maximum sizes are distinctly different for the two sexes. Because of the availability of these research data, banded morwong was chosen as an example reef fish fishery with which to test a range of fishery performance indicators for data-poor fisheries that usually only have commercial catch and effort statistics, sometimes of questionable integrity, plus limited availability of biological information.

To test a range of fishery performance measures (see Ziegler et al. 2005), a detailed model description of the stock was developed—an operating model—in order to generate artificial data sets for the comparisons. However, in the process of developing this detailed description, it proved difficult to fit a relatively simple model to all the observed biological characteristics of the catch. Catch rates (CPUE) and verbal reports concerning the fishers' impressions of the status of the fishery, did not exhibit any recent trend (Fig. 1). However, the observed age-structure of the catch, which included many relatively old fish of both sexes of 50 years and above, suggested that while some stock depletion had occurred since the advent of the fishery, there was still an accumulation of older fish available. The numbers of animals above 15 years of age had been reduced but the age-structure of these older fish was still relatively flat. Despite this, the observed change to the sex ratio from 70% at the start of the fishery to just below 50% females in the catch in the most recent years of the fishery implied significant stock depletion, at least in some areas. The change in sex ratio came about both through a reduction in the proportion of older females (>15 years old) from about 65% to about 30% in the catch and an increase in the proportion of younger fish in the catches. Whether the increases in younger fish were a result of stock depletion allowing more younger fish to be seen or were due to increases in absolute recruitment success was unknown. Whatever the explanation, a change

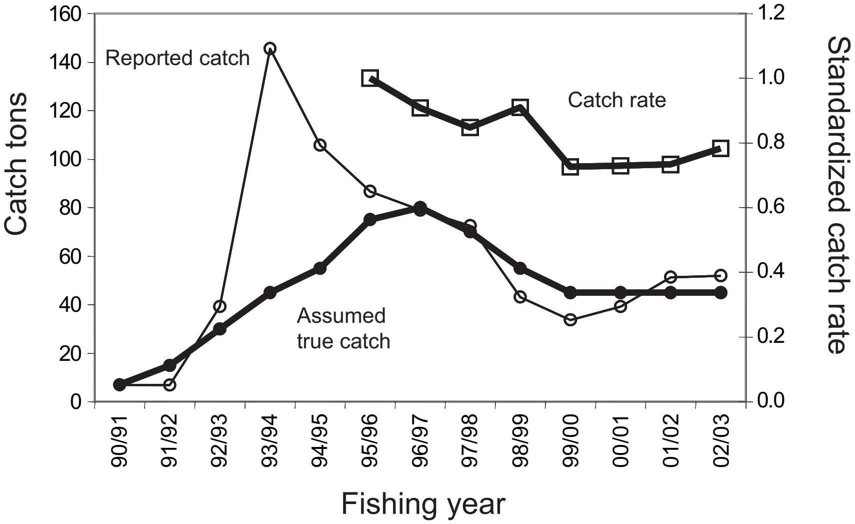

Figure 1. The reported catch compared to the catch time series used in all the simulations. The extremes reported in 1993/1994 and 1994/1995 were reduced to remove false catch returns. The time series of assumed true catches is meant to represent the typical development of a fishery. The reported catch rates (CPUE) were standardized relative to 1995/1996.

in the dynamics of the fishery had occurred and a simple model was unable to provide an adequate description of the available data. Before the structure of the operating model could be fully developed, these apparent inconsistencies between the various data sources had to be resolved.

The quality of the catch and effort statistics from the fishery's early period is highly questionable (Ziegler et al. 2005) and catch rate data are only available from 1995/1996 onward, so the early part of the fishery is poorly known (Fig. 1). Limited entry by license was introduced in 1998 with reported catch history used in the license allocation process. This use of catch history had been anticipated by fishers prior to allocation and led to some of them generating large amounts of fictitious "paper" fish, artificially inflating the catch statistics (Fig. 1).

The fishery is currently managed by limited entry licenses, a two-month spawning season closure (March and April), and a keyhole size limit currently between 360 and 460 mm fork length. Given the selectivity of the usual fishing gear (gillnets), and sex-based growth rate characteristics (Murphy and Lyle 1999), this size limit range implies that almost all female fish remain in the fishery after they reach legal size, whereas

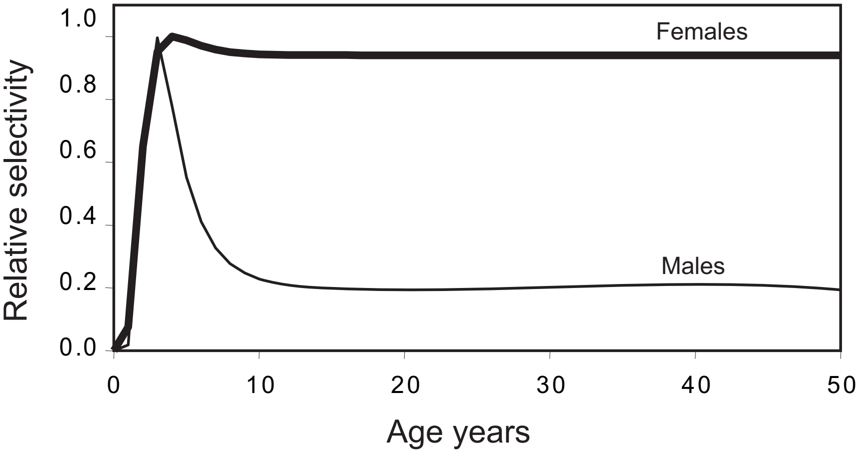

Figure 2. A comparison of the relative selectivity by age for male (fine line) and female (thick line) banded morwong from the east coast of Tasmania (in 137 mm gillnets). Lines are derived from translating selectivity by length into selectivity by age.

the males mainly grow beyond the upper size limit by the age of 11 to 12 (Fig. 2). This keyhole size-limit regulation combined with the growth characteristics of the species enable the use of sex ratio as a performance indicator for this fishery, with an expectation of a predominance of females in the catch, which certainly occurred early in the fishery. The keyhole size limit gives rise to discarding with the potential for some discard mortality. However, discard mortality is assumed to be trivially low because fishers set their nets in shallow waters and continuously retrieve fish soon after they are caught in an effort to maintain high quality for the live fish market.

The changes to the age-structure and changes to the sex ratio, observed in research sampling, suggested different levels of depletion. Finding a simple explanation or model that fitted or was consistent with the observations did not appear possible when only a homogeneous population was considered. A somewhat more complex explanation and model was that a proportion of the fish population lived in an unfished depth refuge and there was some movement between the two sub-populations. Thus, older individuals were supplied to the open population through leakage out of the unfished area. This seems plausible because the known depth distribution of banded morwong extends well below the depths that are fished. Fishers generally restrict their activities to depths less than 20 to 25 meters to avoid problems of barotrauma in this live-fish trade. Exploration was made of the implications of subdivided populations for simple performance indicators.

Methods

A standard age-structured population model was constructed (Quinn and Deriso 1999, Haddon 2001) that subdivided the available biomass between a fished portion and an unfished portion with movement occurring between. This was used to investigate the effect of subdivision and movement on simple catch statistics and other fishery performance measures.

For the simulations, the proportion of the total suitable habitat used by each of the sub-populations was used as a proxy for the initial distribution of the virgin biomass. Thus, 50:50 implies that half the initial biomass lies in each of the two sub-populations, while 40:60 implies 40% is fished and 60% is unfished. These proportions were also used to determine the distribution of the recruits, and thus, effectively, of the initial spawning biomass. In addition, the relative area available to each sub-population influenced the movement dynamics. Each of the two sub-populations had its own dynamics but recruitment was generated as a function of the total spawning biomass combined. Movement between the sub-populations was a combination of mobility rate and the relative proportion of suitable habitat into which the animals can move. A further assumption was that the age-structure and sex ratio was identical in the fished and unfished areas when the fishery began. The base case, against which everything else was compared, was one with a mobility of 1.0 (completely mobile) or of habitat ratio of 100:0. This is equivalent to the special case where the fished population is equivalent to the whole population, as with a non-spatial model.

Growth is described in terms of length at age and weight at length by the standard von Bertalanffy equation:

$$L_{s,t} = L_{s,\infty}\left(1 - e^{-K_s(t - t_0^s)}\right) + \varepsilon \tag{1}$$

where the $L_{s,t}$ is the length of sex s at age t, $L_{s,\infty}$ is the average maximum length for the species for each sex s, K_s is the Brody growth coefficient by sex s, t_o^s is the age at a hypothetical length of zero for each sex s, and ε is a normal random residual. The weight at length relationship for each age t, for each sex s, is described by:

$$W_t^s = aL_{s,t}^b \tag{2}$$

where the a and b coefficients define the relationship between length and weight.

A logistic model is used to describe maturity at age for females, which is used to determine the mature or spawning biomass:

$$\mu_t = (1 + e^{c+dt)})^{-1} \tag{3}$$

where μ_t is the proportion of age class t that is sexually mature, and c and d are the maturity parameters. Given these, and a knowledge of numbers in each population and sex, then the mature or spawning biomass in year y can be determined using:

$$B_{S,y} = \sum_{t=0}^{t_{\max}} \sum_{p=1}^{2} \mu_t W_t N_t^{p,s} \tag{4}$$

where $B_{S,y}$ is the spawning biomass in year y, $N_t^{p,s}$ is the number of fish in population p of age t where the sex s is female, t_{max} is the maximum age (plus-group). The numbers at age 0 at the start of year y are a function of the total mature or spawning biomass across both fished and unfished populations at the start of year y:

$$R_{0,y} = \frac{B_{S,y}}{(\alpha + \beta B_{S,y})} e^{\varepsilon_y} \tag{5}$$

where α and β are the parameters of the Beverton-Holt stock recruitment relationship, $B_{S,y}$ is the total spawning biomass in both the fished and unfished populations in year y, and ε_y is the recruitment residual for year y. The values for α and β are determined from the steepness of the stock recruitment relationship (h) and the pre-exploitation biomass (B_0) using Francis' (1992) method (Haddon 2001). For the purposes of this work, which was primarily about searching for parameter combinations that gave rise to realistic outputs, recruitment variability was set to zero so that recruitment was deterministic.

Numbers at age under equilibrium conditions of no fishing mortality and constant average recruitment are described by the standard equation modified by an expression summarizing movement between populations:

$$N_t^{p,s} = \begin{cases} \pi_p \bar{R}_{s,o} & t = 0 \\ N_{t-1}^{p,s} e^{-M} & 1 \le t \le t_{max} - 1 \\ N_{t_{\max}-1}^{p,s} e^{-M} / (1 - e^{-M}) & t = t_{max} \end{cases} \tag{6}$$

where $N_t^{p,s}$ is the numbers of fish in population *p*, of sex *s*, and of age *t*, π_p is the proportion of available habitat and hence of recruits to be found in population area *p*, $\bar{R}_{s,0}$ is the average recruitment of sex *s* and age 0, t_{max} is the maximum age modeled (the plus-group), and *M* is the instantaneous rate of natural mortality, assumed constant across all ages. Recruitment variability has been omitted from equation 6. The symbol t_{max} is referred to as the plus age-group because it combines age's t_{max} and all older ages that are not modeled explicitly. Equation 6 is combined with the movement dynamics to initiate the populations.

Movement is modeled as occurring at the end of each year and is restricted to mature fish (all females mature before entering the fishery). Movement between populations is a combination of mobility *m*, defined as the proportion of the mature population that becomes vagrant or mobile, and is capable of shifting from each population to adjoining populations, combined with π_p, the proportion of habitat/biomass in each population *p* into which the animals can move (also used to subdivide the recruits between populations). Thus, the movement rate from population 1 into population 2 can be represented as $m\pi_1$. The dynamics of movement can be represented formally for each age class as:

$$N_t^{s,p} = (1 - m\,\pi_{p+1})\mu_t N_t^{s,p} + m\,\pi_p \mu_t N_t^{s,p+1} \tag{7}$$

Thus, population *p* retains $1 - m\pi_{p+1}$ of its total but gains $m\pi_p$ of the neighboring population, $p + 1$. With only two populations, *p* and $p + 1$ are the only possibilities. If the proportion of habitat is equal (i.e., $\pi_p = 0.5$) then the movement rate equals the mobility, however, if the proportional distribution of the population deviates from 50:50 then the movement rates will become asymmetric. This approach to describing movement was selected because it represents the degree of mobility in the population and determines where the fish will finally settle by the total area over which it can spread (which includes the area in which it began to move). Thus, a fish may begin to move and its probability of settling in one of the available areas is related to the relative area inhabited by the two populations.

The keyhole legal size range combined with the early rapid growth has a strong effect on selectivity at age. The gillnet mesh selectivity is best described using a gamma distribution (Millar and Holst 1997):

$$V_t^s = \left(\frac{l_t}{\gamma k n}\right)^{\gamma} e^{\left(\gamma - \frac{l_t}{kn}\right)} \tag{8}$$

where V_t^s is the selectivity of age class *t* for sex *s*, l_t is the length of age class *t*, *n* is the mesh size of the nets used, and γ and *k* are the selectivity parameters (different for each sex). This gillnet mesh selectivity is modified at the legal limits for each sex. Together with the different growth pattern of the males and females this leads to a large difference in their selectivity characteristics (Fig. 2). The females remain vulnerable to the fishery for much longer than the males.

The fishing mortality rate for each age class is defined in terms of the fully selected instantaneous fishing mortality rate F_y in year *y*, combined with the selectivity V_t^s for each age class *t* and sex *s*:

$$F_{y,t}^{s} = V_t^s F_y \tag{9}$$

The catch in each year *y* is defined as the sum of the predicted catch at age multiplied by the weight at age:

$$C_y = \sum_{s=1}^{2} \sum_{t=0}^{t_{max}} W_t \frac{F_{y,t}^s}{F_{y,t}^s + M} N_t^{p,s} \left(1 - e^{-\left(M + F_{y,t}^s\right)}\right) \tag{10}$$

while the dynamics of the two populations are described by the combination of:

$$N_{t,y+1^*}^{p,s} = \begin{cases} \pi_p \bar{R}_{s,0,y+1} & t = 0 \\ N_{t-1,y}^{p,s} e^{-\left(M+F_{y,t}^s\right)} & 1 \le t \le t_{max} - 1 \\ N_{t_{\max}-1,y}^{p,s} e^{-\left(M+F_{y,t}^s\right)} + N_{t_{max},y}^{p,s} e^{-\left(M+F_{y,t}^s\right)} & t = t_{max} \end{cases} \tag{11}$$

remembering that fishing mortality only acts on one population *p*. Equation 11 is combined with the equation for movement, with movement acting at the end of each year to generate the final numbers at age in each population.

$$N_{t,y+1}^{s,p} = (1 - m\pi_{p+1})\mu_t N_{t,y+1^*}^{s,p} + m\pi_p \mu_t N_{t,y+1^*}^{s,p+1} \tag{12}$$

The exploitable biomass is defined as the fishable biomass before any mortality in a given year had been imposed:

$$B_y^E = \sum_{s=1}^{2} \sum_{t=0}^{t_{max}} W_t V_t^s N_t^{p,s} \tag{13}$$

The model was fit to the available data by setting the initial unfished biomass, the mobility rate, and the relative proportion of fished and unfished populations and then pursuing an iterative search for the set of fully selected fishing mortality rates that generated the observed catches. The sex ratio, proportion of fish in the plus group, the exploitable biomass and total spawning biomass in each populations, through the years of the fishery were saved, graphed, and inspected.

The effect of spatially distributing the spawning biomass was investigated by generating sub-populations at different distribution ratios (fished to unfished ratios) and growing the population for 13 years (the approximate age of the fishery) with different mobility levels. The outputs of interest, in effect representing fishery performance indicators, were the sex ratio of the catch and the proportion of the female catch of 15 years and older (a simple proxy for age-distribution). These were suggested by the changes observed in both sex ratio and the proportion of older fish in the research sampling conducted on the east coast of Tasmania.

Because the model was over-parameterized for the available data, instead of directly fitting the model to data, a single catch history was used for all simulations and many different combinations of initial biomass, mobility rates, and fished : unfished biomass ratios were examined. The catch history used in all simulations (assumed catch in Fig. 1) was similar to that observed in the fishery (reported catch in Fig. 1) with the extreme catches, most likely to be based on false data, removed. In each case, the model was conditioned on catch by searching for the schedule of annual fishing mortality rates required with each combination of parameters that would give rise to the catch time-series used in each simulation. This conditioning on catch, under different combinations of initial biomass, mobility, and fished : unfished ratios, led to different degrees of depletion, different age-structures, and different sex ratios of the catch following the 13 years of the fishery prosecuted upon the open population. The model outcomes for the final year were plotted as contour maps relative to the mobility and proportion of the stock fished (e.g., Fig. 3). The combined contour maps for both the measure of sex ratio and of age-structure enabled the optimal combinations of mobility and proportion of stock fished to be identified for each level of initial biomass (e.g., Fig. 5). The predicted values from each combination of model parameters were also compared (using minimal squared residuals) across all years where observed values were available. The combinations that led to the closest fit to the final year's values were also those that fitted best to all available data.

As no estimate of initial or unfished biomass was available the simulations were run with three different levels: 1,000 t, 1,200 t, and 1,500 t. These bracketing values were chosen because they enabled the model to predict sex ratios of 70:30 in the first year as well as the predicted levels (~65%) of females in the 15+ age group. The levels of mobility that were included ranged from zero to 1.0 (no mobility to completely mobile).

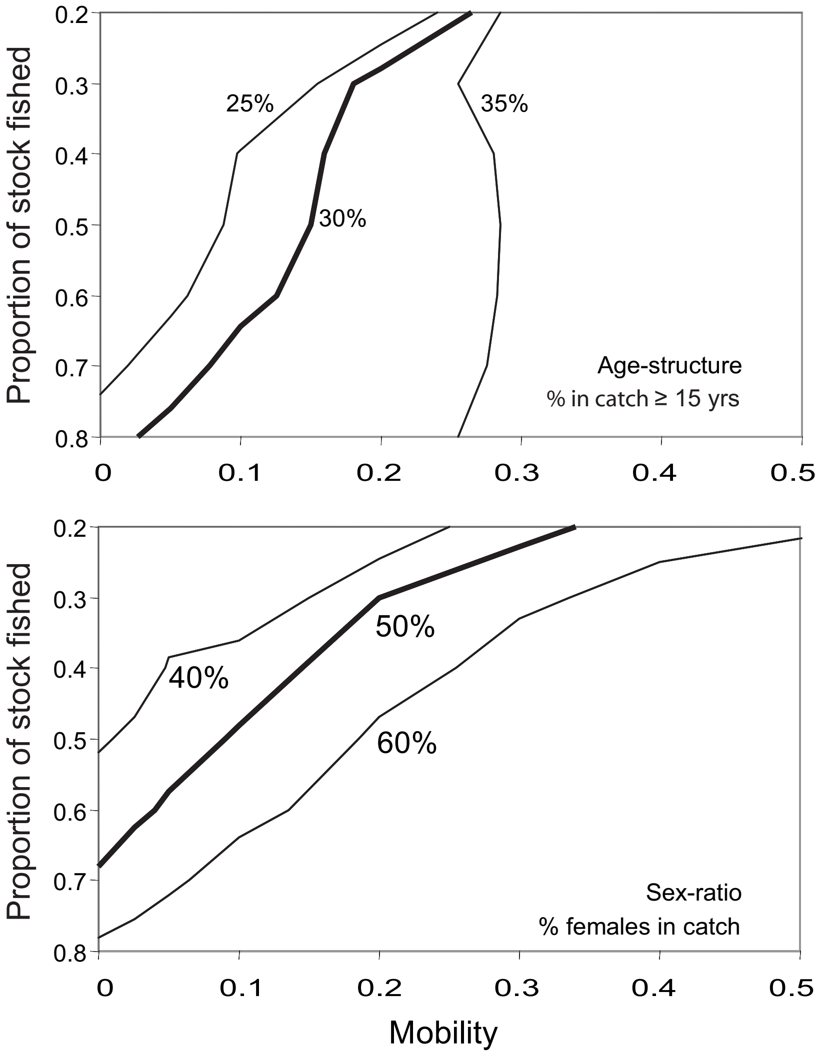

Figure 3. Simulations with 1,200 t of unfished biomass. Contours of the percent of females ≥15 years of age (top panel) and of the percent of females in the catch or sex ratio (bottom panel). The approximate levels observed in research sampling are indicated by heavy lines.

Table 1. Proportion of females (sex ratio) in the catch after 13 years of fishing under different combinations of mobility and fished proportion of the total stock.

Fished proportion	Mobility 0.00	0.05	0.10	0.20	0.30	0.40	1.00
0.20					0.484	0.532	0.593
0.30				0.480	0.543	0.566	0.599
0.40		0.410	0.469	0.533	0.563	0.580	0.602
0.50	0.386	0.460	0.507	0.555	0.577	0.588	0.604
0.60	0.459	0.509	0.539	0.571	0.587	0.594	0.604
0.70	0.517	0.545	0.563	0.583	0.593	0.598	0.606
0.80	0.557	0.573	0.581	0.593	0.599	0.602	0.606

The observed sex ratio was approximately 50%. The unfished biomass in all cases was 1,200 t. Empty cells were combinations for which realistic solutions were not possible.

Finally, the proportion of the available biomass exposed to fishing ranged from 0.2 to 0.8 (20-80%).

Results

By conducting trials of many combinations of the parameters (initial biomass, mobility, relative size of the two populations) it was possible to tabulate predicted values of two performance indicators, the sex-ratio and the proportion of the catch aged 15 years or greater (e.g., Table 1). From such tables, contour diagrams of different levels of the two performance indicators were constructed (Figs. 3 and 4). In each case, the proportion of the stock that was fished had less effect upon the proportion of the females ≥15 years of age than upon the sex ratio. This is shown by the contours being more near to vertical in the age-structure graph than with the sex ratio (Figs. 3 and 4).

The contour lines relating to the final values in the research sampling after the 13-year fishery, i.e., 50% females in the sex ratio and 30% females ≥15 years of age, are highlighted in the graphs. Combinations of unfished biomass, mobility, and proportion of the stock exposed to fishing that would give rise to the observed population characteristics of sex ratio and age-structure (Fig. 5) were identified by where the contour lines overlap. As the unfished biomass was increased to 1,500 t, the overlap of the two types of contours occurred at decreasing levels of both the mobility and the proportion of the stock fished. Which particular combination is the most realistic would require more information. Under no combinations

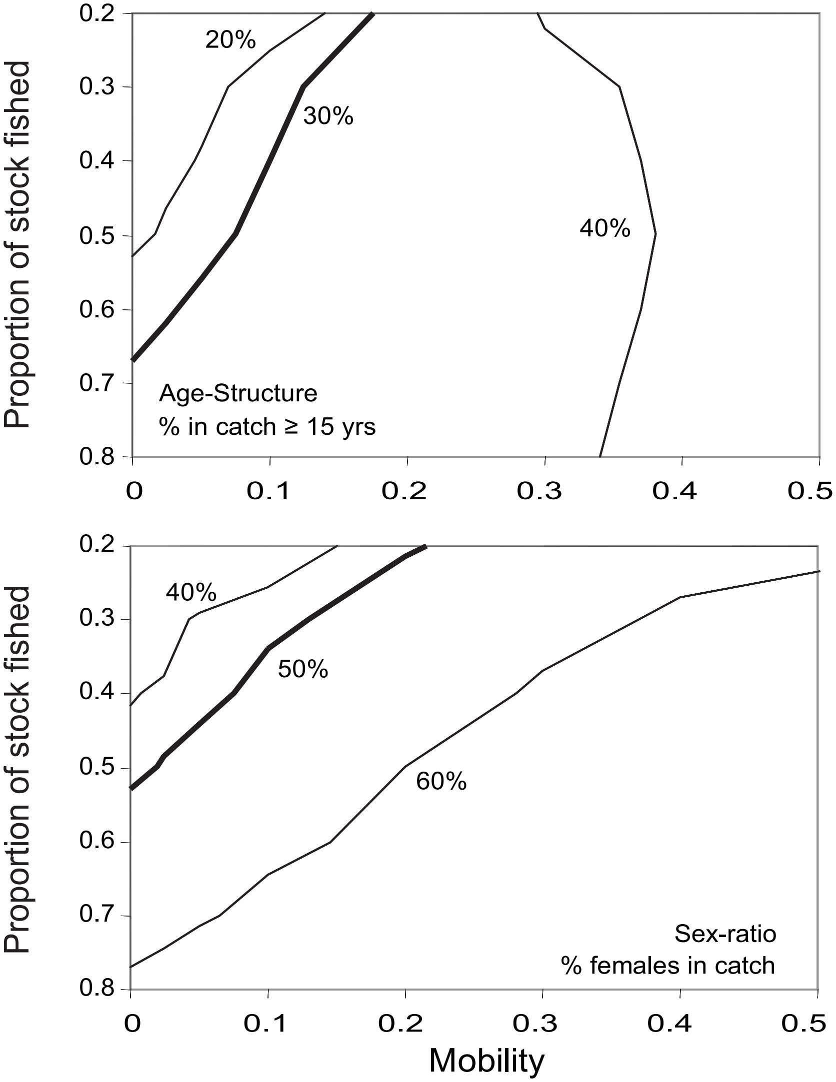

Figure 4. Simulations with 1,500 t of unfished biomass. Contours of the percent of females ≥15 years of age (top panel) and of the percent of females in the catch or sex ratio (bottom panel). The approximate levels observed in research sampling are indicated by the heavy lines.

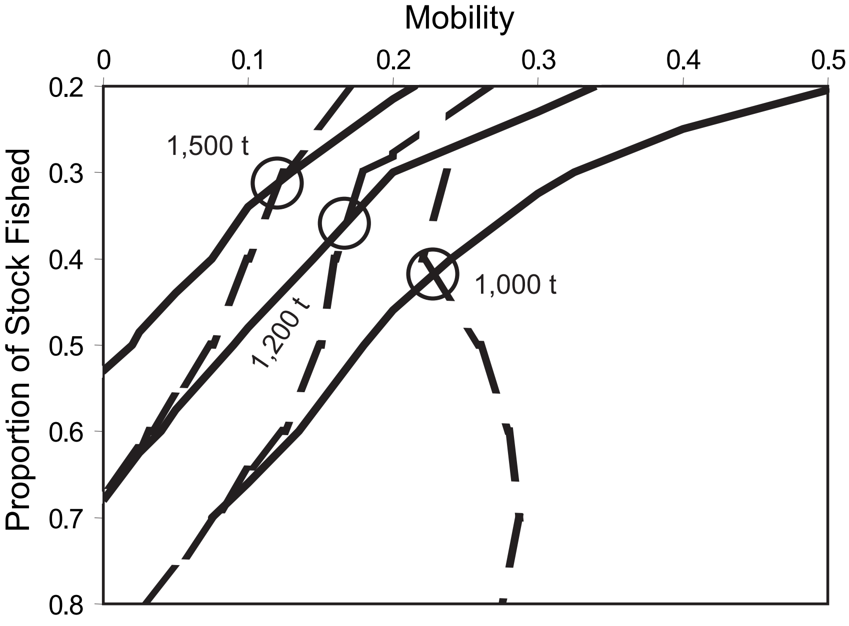

Figure 5. The overlap of the 30% contours for the percent of females ≥15 years of age (dashed lines) and of the 50% contours of females in the catch or sex-ratio (solid lines) for three different levels of unfished biomass (1,000 t, 1,200 t, and 1,500 t). The circles are centered over the points of overlap.

were the required conditions through time of sex ratio and of age-structure obtained without some degree of spatial structuring of the population and some degree of mobility between them.

In the case where the initial unfished biomass was 1,200 t and with the mobility level and proportion of stock fished that gave rise to the observed values of sex ratio and age-structure, the exploitable biomass in the fished population steadily declined until it appeared to be relatively stable at a low level after the 8th year of the fishery. The exploitable biomass in the unfished population only began to decline in the 5th year of the fishery after which it declined at a steady rate (Fig. 6). Assuming a constant catchability in the fishery, this pattern of depletion would also be reflected in accurately reported catch rates.

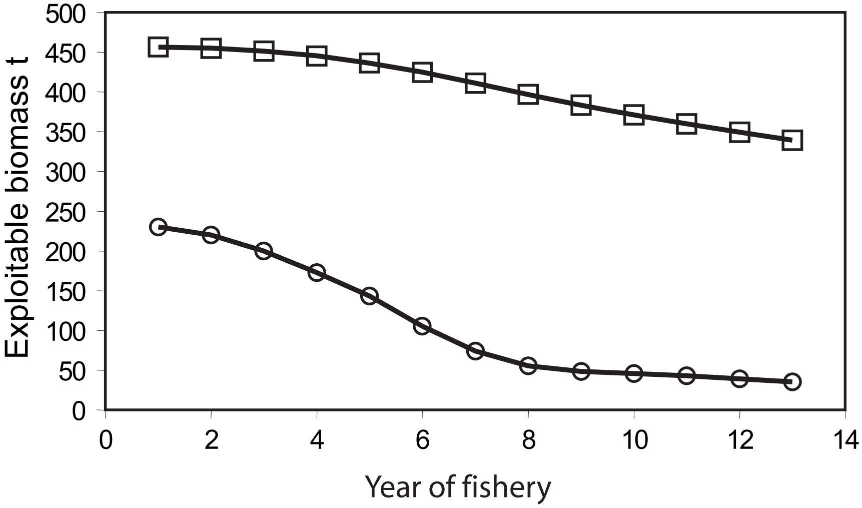

Figure 6. Exploitable biomass in metric tons against year of the fishery, in the population open to fishing (circles) and that unfished (squares). This simulation was with an unfished biomass of 1,200 t, a fished proportion of 34% and a mobility of 0.1575, conditions that lead to a sex-ratio of approximately 50% females and 30% females of age 15 years of greater.

Discussion

Data-poor fisheries are not uncommon. Unfortunately, because the effects of spatial structuring of fished stocks are rarely taken into account when collecting data, many such stocks could also be considered to be data poor. In this context, even if extensive data collections are available, data-poor fisheries may thus be even more common than generally believed. In essence, the assumption that any data that are collected are representative of a complete population is called into doubt when a stock is spatially structured. Walters (2003) pointed out that large biases (up or down) may arise in stock assessments based on catch rate (CPUE) data when the spatial distribution of effort and of the stock is ignored. The fishery for the Tasmanian banded morwong is an extreme example of spatial structuring of a fished stock and of the information about that stock. Walters's (2003) comments, when applied to spatially structured data-poor fisheries, are not encouraging. The fishery for the Tasmanian banded morwong, being a site attached reef dwelling species, provided an opportunity for demonstrating the effects of spatial structuring on our ability to assess an exploited population. Even though insufficient fishery data were available to properly fit a spatially explicit assessment model, it was possible to explore the impacts of parameters such as the mobility of the species, the initial unfished biomass, and the proportion of the stock exposed to fishing.

It was found that spatially structured populations, with movement between the populations, can generate fisheries data (catch rates, age structure, and sex ratios) that appear to be inconsistent with each other when using a non-spatially structured assessment model. This would appear to be a general conclusion to any spatially structured stock with fished and unfished areas. Significant leakage of larger, older animals into a fished area can obscure the depletion of older age-classes. In the case of banded morwong, significant numbers of 50-plus-year-old female fish among numerous young fish being caught, led initially to the idea that the stock was not depleted. It appeared to suggest that there had been a large accumulation of biomass that was in the early stages of being fished down. However, this did not tally with the expectation of low productivity from such a long-lived species or with a closer inspection of biological data (large changes to the sex-ratio). If a fished stock is spatially structured such that a significant proportion is effectively unfished, but there is some movement between fished and unfished populations, then clearly, standard data collection and assessment methods based on simple catch statistics that ignore spatial structuring would lead to an invalid assessment (Walters 2003, Walters and Bonfil 1999). Orensanz and Jamieson (1998) emphasize that for relatively sessile species the assumptions of most assessment models are inappropriate. In particular the assumptions that a unit stock is closed to immigration and emigration and that the effects of the fishing process and the distribution of the fished stock are both homogeneous within the region occupied by the stock, are all inappropriate. Such models and assumptions preclude the possibility of serial or localized depletion, which, in a spatially structured stock is a real possibility. Unfortunately, many published studies of spatially structured fisheries (Booth 2000, Berkeley et al. 2004, Smith and Rago 2004) are clear in the requirement for large amounts of detailed data from across the range of the fishery. We explored the implications of spatial structuring in the absence of such information.

In the model developed here, the amount of movement out of the fished and unfished areas at the start of fishing, before any depletion had occurred, was in proportion to their relative areas (hence in equilibrium). However, if the overall mobility is low enough, the fished part of the stock is quickly depleted and the fishery soon becomes dependent upon the growth of recruits in the fished part of the stock and the movement of larger fish out of the unfished area and into the fished area. In this case, the apparent relative stability arising from a consideration of catch and effort data obscures continued depletion of the total stock, at least in the short-term. This result is not novel as it has similarities to the outcomes of modeling marine protected areas where there is a fished and an unfished structure to a stock (Guénette et al. 1998, Guénette and Pitcher 1999, Walters and Bonfil 1999). However, this is a novel result in terms of being expressed on a reef-based fishery. If a stock assessment was conducted in ignorance of the spatial structuring and its consequences, the present effort levels and their associated catch might be deemed sustainable. If projected forward in time, this situation would continue until, from the point of view of an assessment assuming a homogeneous fished population, the fishery would inexplicably collapse. If reef-based fish stocks have a greater tendency to be spatially structured this problem seems likely to be relatively common.

The simplest conclusion to draw from these simulations is that ignorance of the biology and behavior of a fished species can lead to risk-prone fisheries management. If insufficient weight is given to the implications of the biology of a fished species, then any assessment is likely to produce bad management advice. It would appear to be stating the obvious to suggest that a review of the known biology and other fisheries for a particular species be undertaken in data limited situations. However, care needs to be taken in such meta-analyses to ensure that there are at least some close similarities between the species being compared. King and McFarlane (2003) explore the possibility of categorizing species into five different life history strategies in a manner analogous to the classic *r*- and *K*-strategies but identifying strategies that relate more to the marine than the terrestrial environment. This approach holds the promise of being more reliable than simply assuming a related species will be similar, but it would require at least some biological investigation. Unfortunately, it would appear that investigating spatial structuring is essential for the stock assessment of all fisheries, including data-poor situations. Tagging studies in combination with even crude habitat mapping have great value in providing insights into the potential distribution of fished species for comparison with reports of where fishing occurs (habitat maps could be of use to many fisheries in an area). Of course, if fisheries are data poor because of a lack of resources then habitat mapping is unlikely to occur. Even with such information available the question remains how to include spatial information into an assessment. In the particular case of spatially structured populations some form of precautionary spatial management suggests itself. While managers learn about the fishery and its effects, which might take decades, it would be risk averse to require the fishery to be focused in a delineated area leaving significant proportions of any known habitat unfished for the species in question. This is not a recommendation for the creation of marine protected areas as fishing for other species could occur in the areas closed to, say, banded morwong fishing. Rather, it is a way of implementing the precautionary approach (Garcia and Grainger 1997) while allowing some fishing to occur, thus enabling managers to learn about the productivity of the species and how it reacts to fishing. In addition, management plans need to be developed that account for the potential spatial structuring of the fishery. Some form of adaptive management is required, especially in a relatively new fishery (Collie and Walters 1991). Using the fishery to learn about the stock implies having some form of data gathering available, at least about landings and possibly effort.

Management objectives concerning sustainability are commonly so general that they provide little guidance in cases where there is spatial structuring in fisheries. Like most current stock assessment models, such objectives often reflect an implicit assumption of spatial homogeneity. For example, does sustainable exploitation mean sustainable in each subpopulation or sustainable in the context of the complete species distribution? These two options imply different things and would lead to different management. If it is desired to exploit small-scale, low valued, spatially structured fish stocks, then management objectives that recognize the possibility of stocks being spatial structured must evolve that reflect the risk management strategies being implemented.

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Sensitivity of, and Bias in, Catch-Survey Analysis (CSA) Estimates of Stock Abundance

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Abstract

Catch-survey analysis (CSA) is an assessment method designed to estimate absolute stock abundance from total catch and relative indices, by filtering the latter through a simple two-stage population dynamics model. It qualifies as a tool for data-limited contexts since it does not require data on the full age composition, which are the main impediment to using conventional age-based approaches, either because age data are too costly to obtain routinely, or because some species cannot be reliably aged. This paper focuses on three issues that are not clearly settled in the sparse literature dealing with CSA:

i. The sensitivity of stock size estimates to the ratio of recruits to fully recruited survey catchabilities, which needs to be set by the user based on external information; this study shows that sensitivity is not dependent on the magnitude of this parameter.

ii. The biases due to changes in catchability, which are to be expected when commercial CPUE data are used as indices; as with other methods, CSA estimates are biased high when increasing trends in the ratio between indices and abundance are not accounted for.

iii. The effects of occasional missing indices, which are an element of data limitations; in this example, these effects are found to be small and ephemeral.

Investigation of these questions was carried out using artificial data generated with a different, length- and age-structured model, providing true population states for comparisons with CSA estimates.

Introduction

It has been a customary practice in many fisheries management systems throughout the world, and progressively a legal obligation in many of them, that proposals for management action should be underpinned by scientific assessments of the current state and expected development of the stock. For temperate fisheries at least, stock assessments are predominantly carried out with age-based methods. Since age-dependence is known to occur in such key biological processes as growth, reproduction, natural mortality, and vulnerability to fishing gears, models assuming age structure in the associated parameters are presumed to be more realistic, and they enable to evaluate a broad range of management measures. Considerable efforts have thus been spent over the years to refine age-based approaches. However, there are many stocks for which they are not applicable. The sequential population analysis (SPA, in the generic sense suggested by Megrey [1989]) methods generally used to estimate the parameters of age-based assessment models require a sufficient time series of catch-at-age data, and sometimes other age-disaggregated data. Such data may be lacking for reasons of cost, considered to be too uncertain for use in a decision context, or even impossible to obtain for some species like crustaceans that are notoriously difficult to age. Lack of age data is but one facet of so-called "data-limited situations," and this study is mostly confined to this meaning of the phrase. However, two aspects of data deficiencies regarding abundance indices are also considered.

Several reasons make it important to give serious consideration to other approaches than SPA. First, as the stocks of major species are being depleted, fishermen are turning to species that were formerly less important but now make up a significant fraction of their income. Scientists may expect that management advice on these stocks will soon be requested, but if they condition their response on the availability of sufficient series of fully age-disaggregated data for SPA, clients will need to be very patient. Second, on a similar topic, many institutes have conducted regular surveys for many years but the information used quantitatively for advice has mostly concerned the major species, in association with SPA. This implies that those expensive surveys have been underutilized for a range of other species caught in the surveys, due to the lack of appropriate methods to turn this information into suitable advice. Third, setting aside species that are impossible to age reliably, a large fraction of the cost incurred in assembling assessment data, apart from surveys, is due to the processing of otoliths or scales for setting up age composition data. Some institutes face difficulties to mobilize enough resources to handle the large amount of material required each year. The situation may worsen if the list of species to assess continues to grow. Fourth, even for species that have been assessed for decades, it may happen that, for various reasons, an SPA cannot be run in some years, which often puts scientific bodies in serious trouble to formulate advice. A case in point is the Advisory Committee on Fishery Management of ICES (International Council for the Exploration of the Sea) which has often been paralyzed by its inability to offer managers any useful advice when SPA assessments were not available, or when those available were not deemed reliable. Many fisheries management institutions have embraced the precautionary approach, which provides that management should adopt cautious measures until sufficient data are available to assess the state and sustainability of the fishery, but practice appears to be still far remote from the good intentions. The most likely consequence of having no defensible bases for advice is that decisions leading the stocks to undesirable states are taken.

For species that are genuinely difficult to age, even obtaining a robust estimate of the overall growth coefficient may be impossible. This is an impediment to using some alternative methods, like length-based techniques or delay-difference models, in which this coefficient is a key parameter. For some stocks, however, examination of length-frequency distributions may reveal a distinct recruits component, particularly when data are compiled over short time periods, even though the picture may be blurred for older ages. This is the situation where catch-survey analysis (CSA) finds its place. This assessment method is based on the two-stage model proposed by Collie and Sissenwine (1983) and does not require that the catch data be subdivided by age. In addition to total catch, it requires time series of abundance indices, from fishery CPUE or preferably from surveys, and only these need to be split into a "recruits" stage, and a "fully recruited" stage in which all older ages are lumped together. It is apparent that the demand in data is considerably reduced compared to age-based approaches, which qualifies the method for data-limited situations in the sense mentioned above. CSA is regularly used, notably for shellfish assessments, in the Northwest Atlantic (e.g., Conser and Idoine 1992, Cadrin et al. 1999, Helser et al. 2002) and in the North Pacific (e.g., Collie and Kruse 1998; Zheng et al. 1997, 1998). No record has been found of its use for advice on European stocks and fisheries.

Three particular issues regarding the capabilities of CSA for assessing stock status are examined in this study. The first one follows from Mesnil (2003) who found that CSA can do very well to trace relative trends in biomass and recruitment, but that the absolute value of estimates is very sensitive to one parameter, the catchability ratio between recruits and fully recruited in the survey, which has to be set by the analyst. Two questions arose then. One was whether the observed sensitivity was dependent on the magnitude of this parameter, which was fairly low in the examples treated. The other was whether auxiliary information, such as length composition of the survey catch and knowledge of the selection curve of the survey gear, could aid in setting this parameter objectively. These questions have been addressed by simulating a length- and age-based population and a length selective survey with three possible mesh sizes. The next two issues address other facets of data limitation, regarding the survey data. CSA needs to use relative abundance indices, which should preferably come from research surveys. However, in many regions, the availability of sufficient time series of well-standardized surveys may be viewed as an unimaginable luxury. Users may then be inclined to use commercial CPUE with all the associated troubles, such as trends in efficiency, that have caused so much concern in recent ICES assessments. The bias in CSA estimates due to unaccounted catchability trends is the second issue treated in this paper. Even in reputed data-rich situations, some data may be lacking in occasional years (typically, cancellations of surveys due to bad weather or mechanical failure) and this can pose a challenge to the best assessment techniques. The effects of such deficiencies on CSA estimates are also examined.

Methods and data

The two-stage model

Following Collie and Sissenwine (1983), it is assumed that the population consists of two stages: the recruits (belonging to a single year class), and the fully recruited animals. The time step is annual, with years defined either on a calendar basis or as the interval between regular surveys. Population dynamics is described by a discrete difference model:

$$N_{t+1} = (N_t + R_t)e^{-M} - C_t e^{-(1-\tau)M} \qquad (1)$$

where N_t is the population size, in number, of fully recruited animals at start of year t, R_t is the population size, in number, of recruits at start of year t, C_t is the total catch in number during year t, M is the instantaneous rate of natural mortality (same for both stages here), and τ is the fraction of the year when the catch is taken.

Estimating the time-series of N_t and R_t knowing the catches and M is the basic task of all assessments but, as with other methods, this requires additional information in the form of relative indices n_t and r_t of abundance for each stage, typically from surveys. The indices are assumed to be proportional to absolute population sizes N_t and R_t and are considered to be measured with lognormal observation error:

$$n_t = qN_t \exp(\eta_t) \;\; ; \; t = 1,\, T \qquad (2)$$

$$r_t = sqR_t \exp(\delta_t) \;\; ; \; t = 1,\, T\text{–}1 \qquad (3)$$

where q is the fully recruited catchability coefficient, supposed to be constant with time, s is the ratio of recruits catchability to fully recruited q, and η and δ are normally distributed random variables. Note that q and s apply to the survey, not to the fishery. In principle, the s ratio is estimable together with the other parameters but, in fact, it is strongly (negatively) correlated with q (Conser 1994). In practice, it has to be fixed using external information, and procedures to do this are discussed in this study.

Parameter estimation

All early works about this method considered fitting the model in the context of a mixed error structure, where process error in the dynamics (an error term appended to equation 1) is considered on top of observation (measurement) error on the indices. There is a growing recognition, however, that considering process error unduly complicates matters and that assuming observation-error only in fitting the model is preferable (Collie and Kruse 1998; see also Polacheck et al. 1993 in the context of surplus-production models). Experience so far indicates that estimates obtained with either route are fairly close anyway. The all-observation-error approach enables to estimate the absolute abundance R and N directly, by minimizing the sum of squared residual errors:

$$SS(\theta) = \sum_{t=1}^{T} \eta_t^2 + \lambda_\delta \sum_{t=1}^{T-1} \delta_t^2 \tag{4}$$

where λ_δ is the (user defined) weight of the observation error on recruits, relative to the observation error on the fully recruited, and θ is the set of parameters. It is often difficult to find objective justifications for the choice of weight (the variance estimate based on the sampling design may not be sufficient to appraise the quality of an index series) and it is appropriate to evaluate the effects of changing it through a sensitivity analysis. By default, λ_δ was assumed to be unity throughout this study. Minimization of the nonlinear function SS with respect to the parameters can be done with any suitable NLLS (nonlinear least squares) algorithm. Note that the catch and the recruitment index in the terminal year are not used in fitting the model.

Given time series of catches $\boldsymbol{C}$ and of abundance indices $\boldsymbol{r}$ and $\boldsymbol{n}$ for T years, T+1 parameters are estimated: $q + \{R_1 \ldots R_{T-1}\} + N_1$, all other Ns $\{N_2 \ldots N_T\}$ being derived by projection using equation 1. Since the abundance of both stages is "known" in each iteration and the corresponding indices are input data, an advantage of this approach is that the survey catchability can be computed as:

$$q = \exp\left[\frac{1}{W}\left(\sum_{t=1}^{T-1}\lambda_\delta \ln\frac{r_t}{s\hat{R}_t} + \sum_{t=1}^{T}\ln\frac{n_t}{\hat{N}_t}\right)\right] \tag{5}$$

where W is the sum of weights, that is $W = [(T\text{–}1) \times \lambda_\delta + T \times 1]$ in the case where no index data is missing (otherwise, the weight, λ_δ or 1, of each missing index is discounted). Adding this equation removes one free parameter from the NLLS search (Polacheck et al. 1993) and releases an extra degree of freedom (T–1 in total). All results presented hereafter have been obtained with this option.

A further advantage of the all-observation-error approach, compared to the mixed-error structure, is that the assessment is feasible when abundance indices are missing in some years, as may be expected in data-limited situations. The number of missing data must be kept reasonably low, however. In addition, it enables to make use of several sources of indices, but this option is probably irrelevant in data-limited contexts.

The procedure above yields the time-trajectories of stock abundance by stage, except recruitment in the terminal year. In this study the latter was estimated by dividing the observed index in the final year (r_T) by s times the estimated q but other procedures can be contemplated. In most real assessments, stock biomass is the primary quantity of interest, in which case CSA users have to provide estimates of mean weights by stage (but note that these play no role in fitting the model). Total stock biomasses are obtained as:

$$B_t = R_t \bar{W}_{R,t} + N_t \bar{W}_{N,t} \tag{6}$$

We will leave aside exploitable biomass, which is the conventional indicator of stock state in some management arenas (notably in North America), as this requires knowledge about the selectivity in the fishery. This is particularly complex when several gear types are involved. A measure of fishing intensity is also often required for management or for forecasts. Several indicators are suggested in the literature, but the simple harvest rate (catch divided by stock abundance in each year) is probably sufficient. If the catch can be subdivided by stage, e.g., using the same criterion as for the indices, harvest rates can also be computed for each stage.

Data generation

The basic data required by CSA are time series of total catch in number, and of "survey" indices for the recruits and for the fully recruited. These data were generated from an artificial age- and length-structured population of known characteristics, comprising 15 age groups. That is, the data generation model is much more detailed than the two-stage model used for estimation. The main options used for data generation are outlined in this section and more details are given in Appendix A. The population was simulated over 40 years of exploitation using an age-based projection, with recruitments derived from spawning biomass the year before through a Beverton-Holt stock-recruitment relationship with random noise and autocorrelation. The exploitation history roughly echoed that of some North Sea stocks, with the fishery's nominal F increasing gradually to peak at 0.7, decreasing to 0.4, and increasing again to 0.6 in the final year. A desired feature was that F should be well above natural mortality (assumed 0.2) over most of the period, in contrast with the data set used by Mesnil (2003). F also largely exceeds F_{MSY} (about 0.17). Length structure was then applied to the surviving population in each year. Lengths at age were assumed to be normally distributed about the mode given by a von Bertalanffy growth curve, with variable (arbitrary) standard deviations depending on age. The relative distributions of lengths within ages were considered to be the same in all years. Only the data for the final 20 years were retained for use in CSA.

The selection patterns of the survey and of the fishery were independent. The survey was supposed to sample the population in a length-selective manner. "Cod-like" selection factor (SF = 0.4) and ratio of selection range (L75-L25) to L50 (about 0.4) were assumed for the survey trawl, which could be equipped with meshes of 20, 50, or 80 mm. The rationale for considering three mesh sizes was that these result in distinct magnitudes for the s ratio: high (near 1.0), medium (near 0.5), or low (near 0.1), while other characteristics of the population and fishery stood the same. The simulations yielded survey catches at length. Plots of these length compositions indicated a cut-off point between ages 1 and 2 at about 25 cm, whatever the mesh size. Indices for the recruits were thus constructed by pooling survey catches over all length classes less than 26, while all larger fish contributed to the indices for the fully recruited.

A normal CSA assessment also requires data on mean weights by stage to translate stock numbers into biomass. It was assumed that, in a typical data-limited situation, the only source of information would be the length composition of the survey catch. Mean weights for each stage were thus computed as weighted averages of weights at length, weighted by numbers caught at each length in surveys, for sizes below or above 26 cm. For recruits notably, the perceived weights diverge from the true ones in the population as mesh size is increased, thus adding to the bias in estimated biomasses. In order to leave this problem aside, most comparisons with the true population will focus on stock numbers.

An additional scenario was considered in which the "survey" would have experienced a catchability trend of +5% per annum during the last

10 years, which the unaware analyst is unable to take into account. Since such a phenomenon is more likely to affect a commercial fleet, it was only simulated in conjunction with the largest mesh (80 mm). The program generating the data for all 4 scenarios always used the same random seed for recruitment noise, so that the sequences of true population states and fishery's catches were identical in all cases. The only differences were in the survey indices, mean weights, and recruits to fully recruited catchability ratios. Actually, for the 80 mm mesh, the latter were not changed when the q-trend was assumed.

All CSA runs assumed $M = 0.2$, as in the data generation, and $\tau = 0.5$, consistent with the Baranov catch equation used in the simulator. All results considered hereafter are based on exact values of catches and of indices obtained from deterministic data generations (perfect information). The choice of using "clean" data is motivated by the fact that we are still in the process of exploring intrinsic properties of CSA. When "noisy" input data are used and peculiar results are observed, one may risk blaming the latter on errors in the data whereas they are due to inherent features of the method. Such ambiguities are avoided with clean data.

Setting the catchability ratio *s*

When data are simulated, the true s ratios by year can be easily computed since the generator "knows" the true population and the survey indices by stage (and even by length and age) in each year. Dividing the latter by the former gives the true catchabilities by stage, whose ratio gives the vector of annual s. Unfortunately, this ability to know s is out of reach in real applications. Two procedures to "objectively" set s were explored.

The first procedure uses information that would normally be calculable from the survey data (the mean length of catches in each stage) in conjunction with the selection curve of the survey gear. The selection parameters may have been estimated in specific gear experiments, or inferred from species of similar shape and behavior. This procedure consists of taking the mean lengths of both stages, looking up the corresponding retention rates on the selection curve, and taking the ratio. The generator was coded to provide these estimates, one for each year, which will be called "length-based" estimates of s.

The time trajectories of true and length-based s ratios are shown in Fig. 1. The first thing to note is that, although the selection pattern of the survey is assumed constant (for a given mesh size), there are significant variations in s from year to year (except for the smallest mesh). This occurs because the age- and length-structure in the underlying population varies in response to fluctuations in recruitment and fishing mortality. This means that s is not related solely to gear selection, but also to the differential availability of recruits and older fish. Figure 1 also indicates that the highs and lows in the length-based estimates of s tend to coincide roughly with those in the true s, but that the absolute values differ

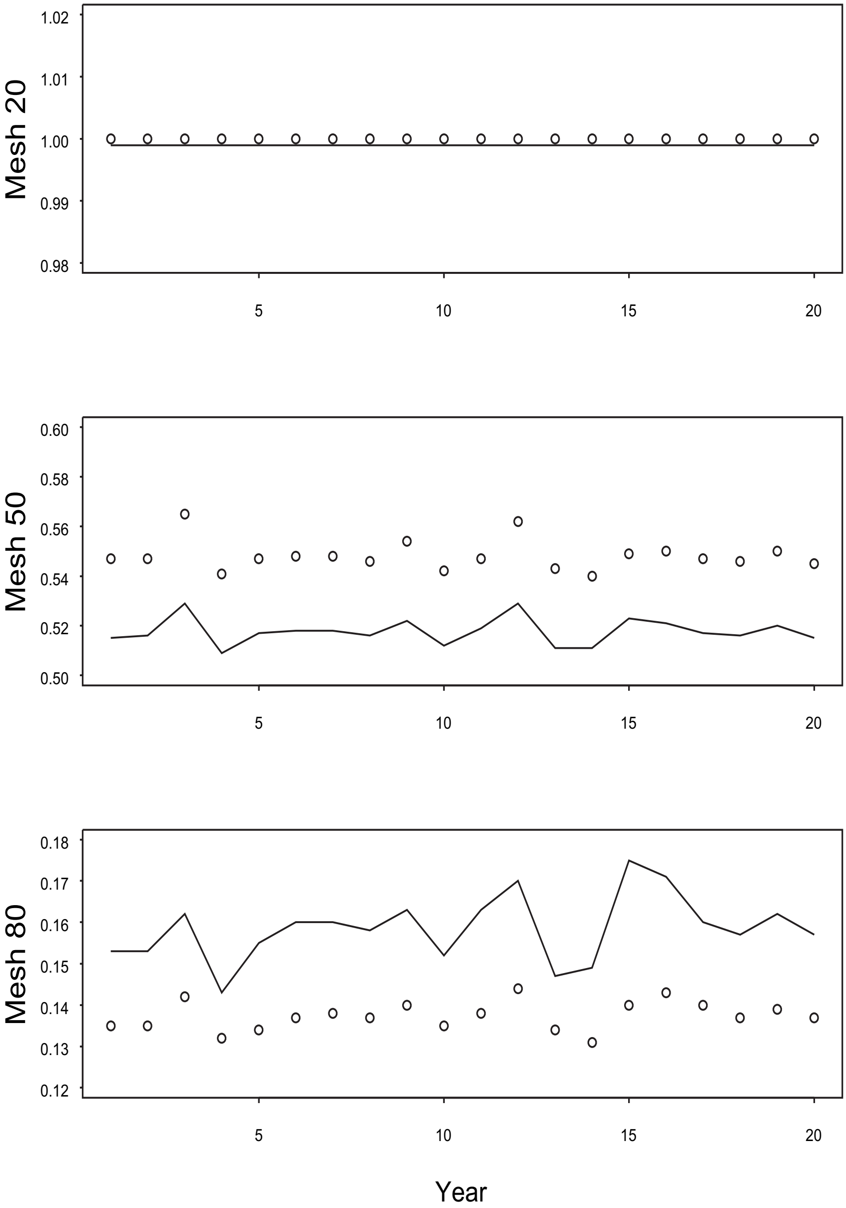

Figure 1. True (–) vs. length-based (o) estimates of annual catchability ratios *s*.

somewhat except in the case of the smallest mesh (in this instance, nearly all fish are retained by the gear and the catchability ratio is bound to be very close to unity). The differences with the true *s* may look trivial, but the effect on estimates may be significant in some instances.

The other procedure is purely "formulaic," as it does not even pretend to make any reference to the reality of the stock and survey. It consists in running CSA assessments for a range of *s* values, and retaining the value that gives the smallest sum of squares *SS* (equation 4). This will be called the "profile minimum" estimate of *s*. In most (though not all) examples where this has been tried, the *SS* profile effectively went through a minimum for a specific value of *s* (but the range scanned has to be carefully set, to exclude unfeasible *s* values). Contrary to the other procedure, it can only give a single *s* value for the whole period.

Results

Sensitivity to assumed *s* ratios

With each set of survey data, comparative CSA runs were made under various options regarding the input *s*. Typically, these involved using the true values in each year (label "Tru-Ann" in Table 1); a constant *s* set at the mean of the true values ("Tru-Mean"); the annual length-based estimates ("Lbas-Ann"); a constant *s* equal to the mean of these ("Lbas-Mean"); and the *s* value giving the lowest *SS* ("Prof-Min"). Deviations from the true values were summarized in terms of the average relative error (ARE, mean of 100 × (estimated minus true)/true over the 20 years), which is primarily a measure of bias, but also a useful indication of relative error across the different cases. The ARE statistics are shown in Table 1 together with the estimates of catchability ($q \times 10^3$) obtained with each option. Note that the AREs for biomass *B* include the estimation error on *R* and *N*, and the error made in deriving the mean weights by stage from the length composition of the survey catch (while the true biomasses provided by the simulator were computed as the sum of contributions by the 15 age groups in the stock, each with its true mean weight).

Comparisons of CSA stock size estimates, in number of recruits (*R*) and fully recruited (*N*) between a selection of runs and with the true states in the simulated population are shown graphically in Fig. 2, which shows that CSA estimates track the variations in abundance of the true population quite well, both for recruitment and for the number of fully recruited. All three surveys give a consistent picture of the historical trajectory of the stock. The absolute estimates of abundance are uncertain, however, as they are sensitive to the assumed *s*; the higher *s* is set, the higher the abundance estimates of *R* and *N*, and the lower the *q* estimates are (Table 1). Changing *s* moves all trajectories up or down, but the general profiles remain the same. Overall, at least in these examples, CSA estimates recruitment more precisely than *N*s, as confirmed by the

Table 1. Deviations between CSA estimates of recruitment, fully recruited abundance, or total biomass and true values, as measured by the average relative errors, for a range of s options. The estimate of catchability *q* obtained under each option is also shown.

20 mm mesh					
Option	Tru-Ann	Tru-Mean	Lbas-Ann	Lbas-Mean	Prof-Min
s value	0.999	0.999	1.000	1.000	1.035
ARE R	–1.46	–1.46	–1.40	–1.40	0.65
ARE N	–5.36	–5.36	–5.22	–5.22	–0.16
ARE B	–4.30	–4.30	–4.16	–4.16	0.86
$q \times 10^3$	1.0464	1.0464	1.0448	1.0448	0.9904
50 mm mesh					
Option	Tru-Ann	Tru-Mean	Lbas-Ann	Lbas-Mean	Prof-Min
s value	var[a]	0.518	var[a]	0.548	0.535
ARE R	–1.56	–1.33	1.83	2.10	0.58
ARE N	–5.48	–5.22	3.10	3.27	–0.48
ARE B	–2.84	–2.57	5.81	5.98	2.20
$q \times 10^3$	1.0274	1.0245	0.9398	0.9380	0.9743
80 mm mesh					
Option	Tru-Ann	Tru-Mean	Lbas-Ann	Lbas-Mean	Prof-Min
s value	var[a]	0.158	var[a]	0.137	0.170
ARE R	–3.02	–2.63	–9.73	–9.37	1.70
ARE N	–7.05	–6.48	–23.80	–23.63	4.47
ARE B	11.60	12.26	–8.14	–7.96	25.15
$q \times 10^3$	0.8100	0.8067	0.9956	0.9941	0.7200

[a] var: variable s depending on year.

lower AREs for R in Table 1. This table also indicates that there is often no benefit in using year-specific s values rather than a mean level, and that it is more important to set the magnitude right in the first place. Efforts to fine-tune the annual variations in s may not be worth the investment.

It is troubling that CSA estimates of R and N are biased (low in these examples) even when the true annual values of s are input (first column of Table 1) and other data are exact. In a personal communication Jeremy Collie (Univ. Rhode Island, Narragansett, Dec. 2002) showed that this bias arises because the model assumes that catchability of the fully recruited is constant, whereas it actually fluctuates from year to year in the simulated population depending on changes in the age structure, as

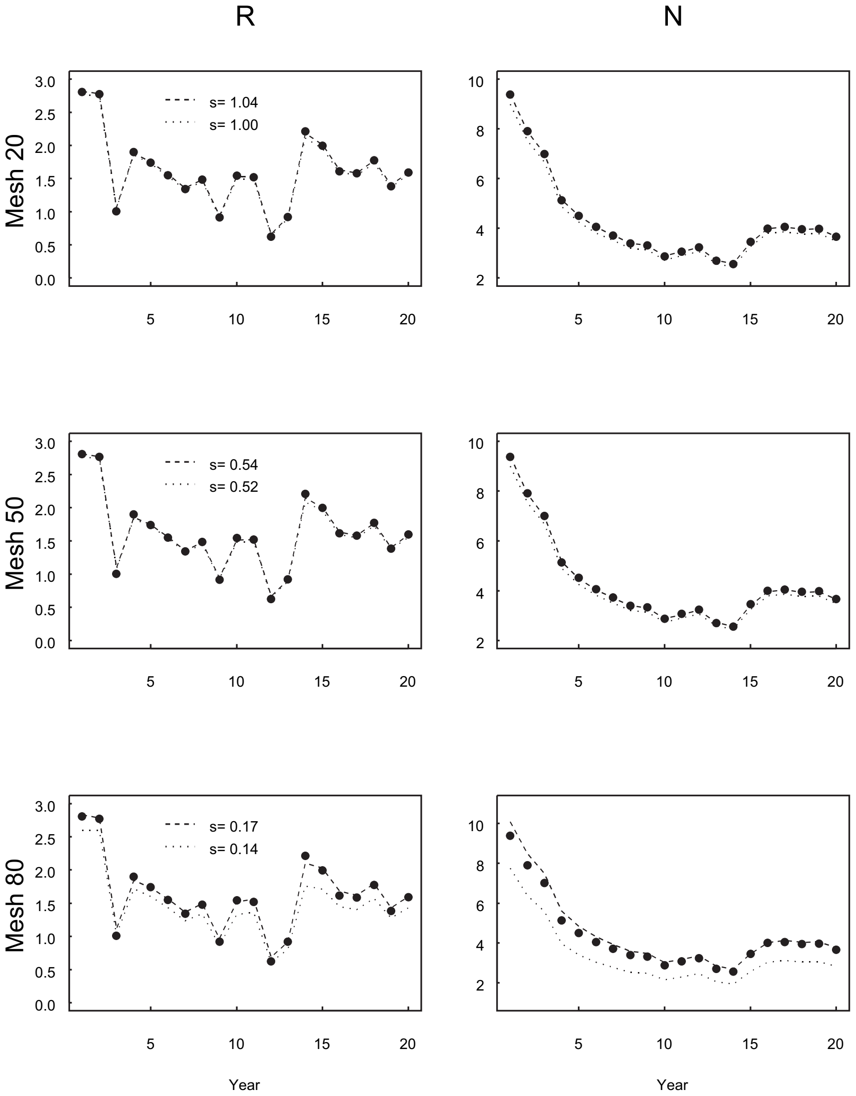

Figure 2. **CSA estimates of *R* and *N* vs. true values (solid circles) under two *s* options for each survey mesh size.**

shown in Fig. 1. One may expect that such bias would be present in real applications, but the sign and magnitude of the bias would depend on the specifics of the population and survey and cannot be predicted.

The procedures that were tried to set the *s* ratio, given information at hand, produced mixed results. The length-based estimates (annual or mean) can be about right in some cases, but too high or too low in others. Here, it is problematic for the 80 mm survey, causing errors of about –10% on recruits and –25% on post-recruits, which is not negligible. The procedure based on the *SS* profile did amazingly well in these examples, yielding the smallest AREs overall (last column of Table 1). But it can produce bizarre things. In the 20 mm mesh case, for example, it yielded an *s* greater than 1.0, implying that recruits are more vulnerable to the gear than post-recruits, which would be hard to explain. Moreover, this is inconsistent with the true *s* in the simulation for this mesh, which are all less than unity. It can also be seen in Table 1 that this option amplified the error in the biomass estimates for the 80 mm mesh data. Thus, none of these procedures can be recommended as a general safe recipe. If used with care, they may be useful nevertheless, as a starting point to give a rough indication of where *s* is likely to stand.

The design with three survey mesh sizes was set up to explore whether the sensitivity of CSA results depends on the magnitude of *s*. This issue was investigated by looking at the sensitivity coefficients (i.e., relative changes in output divided by relative changes in input) of results obtained with each survey for small changes (±1 or 5%) in *s*. For each survey, the base run was with a constant *s* close to the true value. The sensitivity coefficients obtained for *R*s and *N*s were not equal across years, but the variations were fairly small (note: with CSA there is nothing like the magic convergence property of virtual population analysis (VPA); estimates for the earlier years are not necessarily more stable than those for recent years). The coefficients for each quantity were thus averaged over the 20 years; those for biomasses are not shown since they are practically the same as those for *N*s. The sensitivity coefficients shown in Table 2 are roughly equal across cases, indicating that sensitivity is not dependent on the actual level of *s*. In these examples CSA amplifies errors for the fully recruited but not for the recruits; setting *s* too high by 10% (say) would result in a 5-6% overestimation of recruitment, and a 14-15% overestimation of fully recruited abundance associated with a relative error of the same magnitude, but of opposite sign, in *q*.

Much larger coefficients had been obtained in previous simulations with similar specifications, but lower fishing mortality. A simple exercise, only involving the 50 mm survey, was thus carried out to check whether there is a causal link between the sensitivity to errors in *s* and the fishing intensity. A special data set ("Low *F*") was created with nominal *F* in years 11-40 reduced by 0.3 compared to the base set. However, since the simulation involves a stock-recruitment relationship (*S–RR*), lower *F*s resulted

Table 2. Sensitivity coefficients of CSA estimates: ratios between relative change in output (*q*, *R*, or *N*) and relative change in input *s* (first column) around the base values.

Mesh	20 mm			50 mm			80 mm		
base *s*	0.98			0.52			0.15		
$\Delta s/s$	*q*	*R*	*N*	*q*	*R*	*N*	*q*	*R*	*N*
–5%	–1.60	0.55	1.44	–1.62	0.57	1.46	–1.55	0.51	1.40
–1%	–1.54	0.56	1.47	–1.56	0.58	1.49	–1.49	0.52	1.43
+1%	–1.51	0.57	1.49	–1.53	0.59	1.51	–1.46	0.52	1.44
+5%	–1.45	0.58	1.52	–1.47	0.60	1.54	–1.41	0.53	1.47

in more survivors, larger SSBs, and hence larger recruitments ("High *R*") and it became difficult to compare things on equal footing. An extra set was thus generated in which the *S-RR* was switched off and the time series of recruits was imposed to be the same as that used in the base case ("Same *R*"). For both recruitment options, the true *s* were nearly identical to those obtained above with the 50 mm survey, and the sensitivity analysis was also conducted about the base value of 0.52. With the "Low *F*-High *R*" data, sensitivity coefficients for estimated recruits and post-recruits population sizes were clearly larger, about 1.4 and 2.3, respectively. However, with the "Low *F*-Same *R*" data, they were only marginally larger: 0.8 and 1.7. It is well known (Pope 1972) that VPA-based methods behave poorly when catch is not the dominant cause of population decrease (i.e., when $F < M$), and perhaps this limitation also applies to CSA. Although tentative, this exercise indicates, however, that the degree of sensitivity of CSA estimates to errors in *s* involves complex, case-dependent interactions of factors, fishing intensity being but one of them. A practical implication is that sensitivity analyses should be systematically carried out in each application, to check whether the effects of plausible errors in *s* on CSA estimates are worth being concerned about.

In response to a question from the audience at the symposium additional runs were made using data generated with dome-shaped selection patterns in the survey, in the fishery or both, instead of the logistic, flat-topped patterns assumed in the base case. Both patterns had the same ascending limb and input *s* was set by the profile minimum procedure in all cases. The change in selection did not affect the results; deviations from the truth, as measured by the AREs, and sensitivity coefficients remained essentially the same. Thus, CSA is sensitive to the relative selection of recruits and post-recruits, but not to the pattern within the latter.

Effects of catchability trends

When time series of coherent survey data are missing, have gaps, or are too short, it may be tempting to use commercial CPUEs instead. We have learned to know, however, that the efficiency of fishing fleets does improve gradually as fishers adopt modern technology, and that this can lead to severe blunders in stock assessments when it is not taken into account in the compilation of effort data for deriving abundance indices (e.g., ICES 2002). CSA makes the same strong assumption of a single constant catchability parameter as most surplus production models and tuned SPA methods. It is thus likely to suffer from the same problem. The issue was investigated by assuming an efficiency creeping of (arbitrarily) 5% per annum over the last 10 years in simulating data for the 80 mm survey. This trend implies that catchability of both stages in the terminal year is about 60% larger than in years 10 and earlier. The underlying population remains the same as in the no-trend case and all data except the survey indices are unchanged.

Results of a CSA run with these data and with *s* set at 0.154 (using the profile minimum procedure) are shown against the true values in Fig. 3. Clearly, the results diverge from those in the no-trend run with a similar *s* and steadily overshoot the true values in recent years. Like with SPA, a CSA assessment using data uncorrected for trends in catchability is bound to be overly optimistic regarding the recent state of the stock. Moreover, a bizarre property of CSA in this instance is that it would have estimated a lower overall *q* (0.686) than when there was no trend in the simulated data (0.720) if both runs had used the same *s* of 0.17; decreasing *q* is the only mechanism by which this method is able to derive a larger stock from a given index.

With SPA, retrospective analyses are commonly used for diagnostics and often show divergence of estimates as the data sets are augmented when there are trends in *q* (e.g., Mohn 1999, ICES 2002). They are straightforward to implement with CSA and two sets of retrospective runs are shown in Fig. 4 for the 80 mm survey with or without catchability trend. It is noticeable first that in both instances the year-on-year revisions are fairly small; this has been observed in applications with both real and artificial data and it indicates that CSA estimates tend to be quite stable. But the most remarkable feature compared to what has been seen in many SPA assessments is that the retrospective pattern does not scatter in recent years when there is an unaccounted *q* trend. Actually, the retrospective pattern is not worse when there is a *q* trend than when there is none. This is in fact worrisome because it implies that retrospective CSA analyses cannot be relied upon as a diagnostic tool to detect departures from the model's basic assumption(s). Actually, many of the conventional diagnostics associated with current implementations of CSA are inadequate to detect problems, when they are not simply misleading (Mesnil 2003). This issue urgently warrants further research.

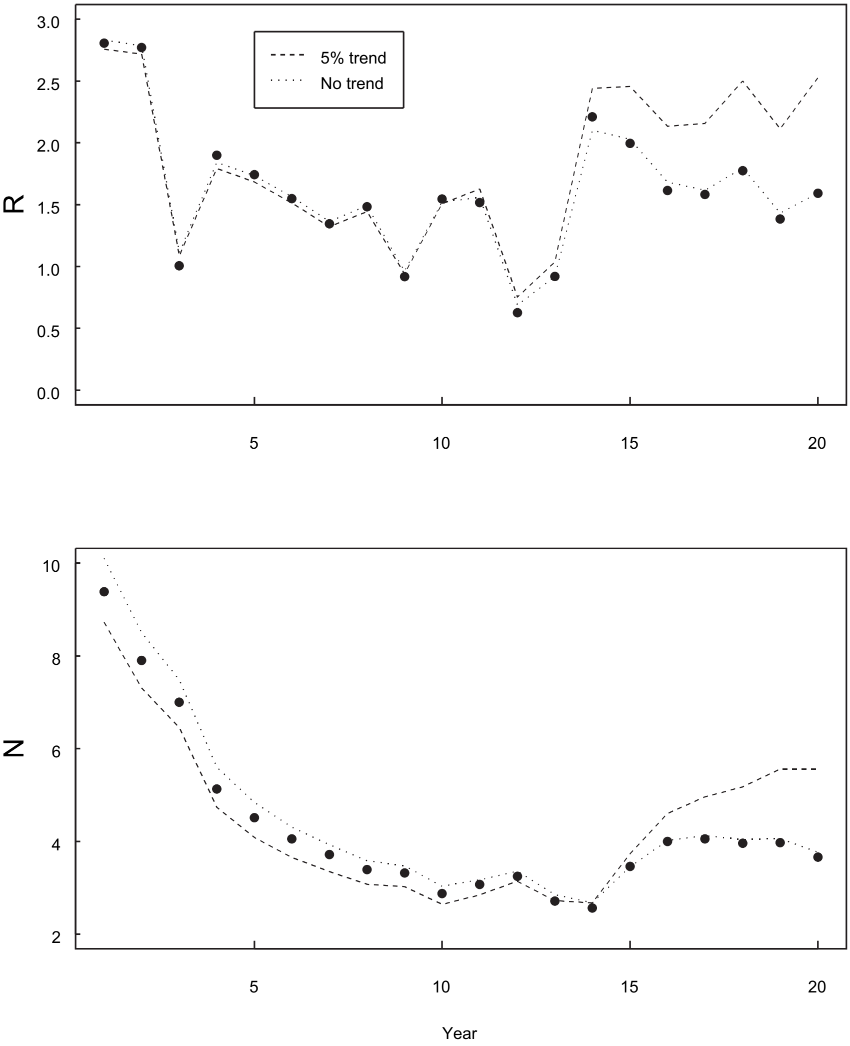

Figure 3. Effects of unaccounted catchability trend in 80 mm mesh survey upon CSA estimates. Solid circles denote the true values.

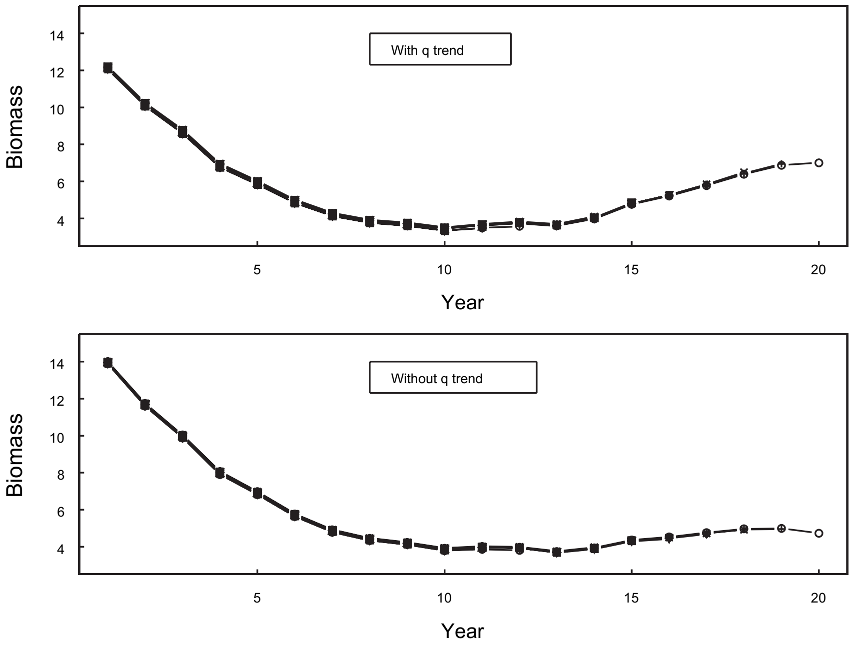

Figure 4. Retrospective CSA estimates of biomass with vs. without a *q* trend.

Effects of missing survey data

Abundance indices obtained from (properly designed) fishery-independent surveys are now recognized to be a key ingredient for reliable assessments (e.g., NRC 1998). However, even the best managed sea survey programs are likely to experience failures to obtain usable (if any) indices in one year or an other, due to bad weather or other reasons. Such data deficiencies may jeopardize the ability to assess the stock status and to provide advice, depending on the vulnerability of the assessment method used. The latter may also depend on when the deficiencies occur in the time series, and how frequent they are. Investigating the consequences of all possible combinations is clearly challenging, and only a small number of cases were examined, using data from the 20 mm mesh survey and assuming $s = 1.0$.

In order to have contrasting cases, it was supposed that survey data could have been missing, for one or two consecutive years, in the terminal period (years 19, or 20, or 19 and 20), in the recent period (years 15, or 15 and 16), by the middle of the series (years 11, or 11 and 12), early in the series (years 6, or 6 and 7), or in the first year. In each case, indices for the two stages were supposed absent. Like in the sensitivity analysis above, the effects on stock size estimates were summarized with the average relative error (ARE) over the 20 years, compared with the true population sizes. In a "normal" assessment, however, special attention is given to estimates for the final year, since these are the initial states for forecasts based on which recommendations of allowable catches (or other management measures) are made. Bearing this in mind, relative errors on stock estimates in year 20, compared with the truth, and on catch in weight predicted for year 21 under status quo, compared with no missing data, were thus considered as well. Status quo is defined here as application of harvest rates by stage equal to the means of the estimates over the recent three years, setting aside years with missing data in the terminal period.

Surprisingly, the effects were fairly small (hardly visible on plots, hence no graph is provided), and the AREs were often smaller than with the complete data set. As expected, the cases where data were missing for two consecutive years were somewhat more pathological, but only for estimates in those specific years with little or no persistence into subsequent years. Contrary to expectations, the estimates for the terminal year were not necessarily more affected when data were missing in proximate years; the (comparatively) largest errors on R_{20} and N_{20} occurred when indices were absent for years 6, or 6 and 7, that is, well in the past. Nevertheless, it was the absence of indices in year 20 that had the largest effect on the catch predicted for year 21 (part of this may be due to the way in which input data for predictions were set up; in a real application, a clever analyst might have done better than using a default procedure).

Although this example indicates that CSA may be robust to missing abundance indices, exploration of a broader variety of data sets is needed before firm conclusions are drawn, as consequences are likely to be case specific. In any case, it is prudent to treat with care, or even discount, the stock estimates for those years in which indices are missing.

Discussion

A primary virtue of CSA is that it makes limited demand in data compared to age-based methods. The main requirement is that a recruits stage can be distinguished from older fish in the survey data, e.g., when a clear cut-off size is visible for the youngest age component in length distributions or from sparse age data. Like most fish stock assessment methods it also needs time series of abundance indices, preferably from scientific surveys. The indices must be subdivided by stage, but not the catch.

In these examples, involving no error in the data, CSA performed quite well to track relative changes in stock abundance and recruitment of a hypothetical multi-age population, although it considers a very simplified population structure with only two stages. In many instances where the current state of knowledge is insufficient even to assert with reasonable confidence whether a stock is going up or down, a CSA assessment that enabled advice on stock status to be cast at least in terms of relative trends would be a major step forward. Fisheries management is generally characterized by the phrase "too little, too late," where "too late" is often because scientists were not in a position to provide defensible evidence of stock decline in due time. A method enabling to give warnings without requiring a wealth of data has obvious advantages in this context.

However, absolute CSA estimates are often problematic since they are very sensitive to one key parameter, the recruits to post-recruits catchability ratio, which must be set by the user. The sensitivity to s is a critical issue that has been largely overlooked in the literature dealing with CSA, and this is why this study was focused on the problem. Most management agencies require advice framed in absolute quantities, and it is therefore of utmost importance that methods be devised for determining s as objectively as possible given the data commonly available. One of the procedures tried here, based on mean lengths by stage and the selection curve of the survey gear, is acceptable to indicate year-to-year variations in this parameter, if one wants to take these into account, but should be viewed as an approximation. This may be in part because, although s is primarily defined as a function of gear selection, it is influenced by other processes, such as the relative availability of the many age-groups lumped into the post-recruits stage as shown in Fig. 1. Moreover, the retention rate derived at the mean length is generally a biased estimate of the retention rate for the corresponding stage. The other procedure is a computational contrivance that is commonly used in similar problems where some parameters are confounded. It often works remarkably, but failures were experienced in trials where indices had large measurement errors. It has the advantage of requiring no additional data, but there is no guarantee that the s value so obtained is "the best" in all circumstances. Both procedures remain useful to give a first approximation to where the "true" s is likely to lie, which is enigmatic when one is tasked to analyze a new data set and has no alternative assessment to offer guidance. Clearly, more work is needed on this topic, so that more confidence can be placed on CSA estimates of absolute stock size.

The sensitivity of CSA to s is quite an annoying feature, but the method is not unique in this respect. Indeed, many current assessment methods (e.g., all those reviewed by ICES 2003) are sensitive to selectivity parameters that need to be constrained based on external or a priori knowledge. This is clearly the case of methods assuming separability of fishing mortality into age and year components, which can produce significantly different assessments depending on whether asymptotic or dome-shaped selectivity is assumed (e.g., NRC 1998). Choice of either form may be difficult, particularly when dealing with mixed fisheries involving a variety of gear types. CSA has the advantage that the selectivity assumption relates to the survey, rather than the fishery, which should be simpler to deal with (in theory at least, the protocol is under scientists' control). Even with traditional VPA, users have to somehow constrain the *F* on the oldest age, relative to that on younger ages, although the assumption made may not be critical when conditions of fast convergence are met ($F >> M$). It seems a pervasive problem in fisheries science, that we are asked to estimate more things than we have "hard" data for.

The setup used in this study, with the three surveys arranged to involve distinct levels of *s*, enabled to clarify that the degree of sensitivity is not dependent on the actual magnitude of *s*. Note, however, that the particular sensitivity coefficients shown in Table 2 are specific to this example, and not representative of CSA behavior in general; quite different values were obtained in trials with other simulated populations. The crude test reported here was inconclusive to show (or discount) a simple relation with fishing intensity. There are other factors that make each data set unique. These findings imply that any CSA assessment should include a sensitivity analysis, to check whether absolute estimates are sufficiently reliable given the uncertainty in the assumed *s* in the context of the particular application. Actually, the current trend in stock assessment is to devise procedures, be they Bayesian or otherwise, that explicitly incorporate uncertainty from data through advice and implementation (e.g., Berkson et al. 2002). Frameworks of that kind would be ideal to formally treat uncertainty in *s* and in other CSA parameters or inputs, and need to be developed.

There are situations, however, that are favorable for CSA applications, namely when it can be safely assumed that recruits and post-recruits are equally catchable by the survey gear. In such instances, the *s* ratio is bound to be fairly close to unity and, even if the sensitivity coefficients are found to be large, the errors in estimated stock size due to assuming $s = 1.0$ may be acceptable. This suggests that the method should preferably be used when abundance indices are obtained from small-mesh research surveys.

In contrast, potential users who have no other choice than resorting to commercial CPUE data, in absence of adequate research surveys, should be very cautious about the use of CSA. Like a number of assessment methods that assume some stationarity in *q*, CSA will give an over-optimistic perception of stock states when there is an increasing trend in catchability, due to gains in fleet efficiency, which is not accounted for in the compilation of effort data. A difficulty is that most diagnostic tests recommended in the literature, including retrospective analyses, are unable to reveal a *q* trend (ICES 2003). Such failure occurred here with clean data, and things would be worse with real data plagued with all kinds of noise. When one is dealing with a commercial fleet, the null hypothesis should be that its efficiency has improved, and will continue to do so. Assessments using CPUE based on nonstandardized effort data are bound to be misleading.

Previous works in the literature used a fitting approach that did not allow for missing survey data. The issue has been addressed rather superficially here, and in any case it would very difficult to establish general laws regarding the impact of missing indices on CSA stock estimates. Absence of data may have different effects when it occurs at one time or another in the trajectory of the stock, and everything is case specific. Nevertheless, this exercise gave some surprising results. First, the effects of missing data were quite small and "local," in the sense that estimates for subsequent years quickly returned on track once indices were available again. But estimates for the particular year(s) when data are missing must be treated with caution. Second, the errors in any year of interest (e.g., the terminal year) were not necessarily worse when data were missing in a year close by vs. in a year well back in time.

A clear limit of this evaluation is that it used exact catches and indices, and a population with good contrast. Simulations with noisy data, and mimicking the dynamics of the real stock of interest together with the characteristics of the associated survey, as done by Cadrin (2000), should be conducted to confirm some of the apparently nice behaviors of the method emerging from this study. However, if errors in the CSA input data (catches, indices, M) were contemplated, then a fair basis for comparison would be with stock estimates provided by more complex age-based methods as these would also depart from the truth in similar ways, at least qualitatively, due to noise in data. This is certainly the case for errors in catches (e.g., due to misreporting) that scale CSA and SPA results alike. As established since Collie and Sissenwine (1983), overestimation of natural mortality M results in underestimation of catchability q and overestimation of stock size (and vice versa), which is also what happens with SPA. It is less immediate to conclude generally on whether both classes of methods react in the same fashion to errors in the indices, but it is known for sure that both give similarly biased results when there is a trend in the relationship between indices and stock abundance (ICES 2003). Moreover, SPA results may suffer from age determination errors in catches or indices for any age, including old ones, a problem that is largely avoided by CSA which is only concerned with errors in allocating fish to the first age (all others being lumped into the post-recruits), and this only for the survey data, not for the catches. Note, however, that CSA requires catches in number whereas catches are generally reported in mass; converting from one to the other must be done properly and may be arduous in some instances.

In brief, CSA appears to offer assessment scientists facing data limitations, in the sense of lacking extensive age data, a chance to do a fairly decent job in documenting changes in fish stock abundance. Estimating absolute stock abundance is also feasible, subject to care in setting some input parameters and/or validation of results through sensitivity analyses for each specific application. CSA can challenge more complex methods, and provide bases for equally useful management advice while requiring less expensive data. The dominant persuasion is that consideration of detailed age data is a prerequisite to doing a legitimate stock assessment. If refined with a bit more research work, CSA may prove this thinking to be ill-founded.

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Carl O'Brien (CEFAS, Lowestoft, UK) suggested the "profile-minimum" procedure to set the catchability ratio *s*. It was a pleasure to incorporate the judicious comments made on an earlier draft by Steven Cadrin (NMFS, Woods Hole) and André Punt (University of Washington, Seattle). The two anonymous referees are also thanked for their help.

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Appendix A.
Specifications of the data simulations

The data generation procedure used is an adaptation of that used by Restrepo et al. (2000). It considers an age structured population comprising 15 ages (1-15, no plus-group; contributions of ages 16 and older are ignored). The population structure in the first year is generated under equilibrium, with a recruitment of 200,000 animals. This population is then simulated forward over 40 years, with nominal fishing mortality increasing gradually to peak at 0.7 in year 27, reduced to 0.4 in year 34, then increasing again to reach 0.6 in the final year. The fishery has a specific age-dependent selection pattern, which is constant over the period. Recruitment in each year depends on spawners' biomass the year before through a Beverton-Holt stock-recruitment relationship, with stochastic noise moderated by autocorrelation. The specifications are summarized in Table A.1.

It is further assumed that the population is length-structured. Modal lengths by age follow a von Bertalanffy growth schedule. Lengths within age are assumed normally distributed about the mode, with standard deviations (SD) varying (arbitrarily) with age. The distributions extend ±3 SD about the mode for each age. The relative length composition within each age does not change in time (truncation of length distribution under high *F*s is ignored).

The survey is length selective with a logistic selection curve. The selection factor (SF) is 0.4 and the ratio of range to L50 is about 0.4 ("cod-like"); the selection range thus increases (flatter curve) with mesh and L50 (= SF × mesh). Three instances of this survey were simulated, supposing that mesh was 20, 50, or 80 mm. The survey's nominal catchability is 0.001 for the fully selected animals. The latter is increased by 5% per annum during the last 10 years in the q-trend scenario.

For the CSA runs, only the final 20 years are retained (re-coded 1 to 20) and data are aggregated in line with the two-stage structure. Annual fishery's catches are the sums of catch numbers at age. Recruit indices include all fish $\leq$length class 25 caught in the survey, while the indices for the fully recruited comprise all larger sizes. Mean weights for each stage are calculated from the survey catches as weighted means, taking account of catch number at each length, given a fixed length-weight relationship.

An example of CSA input data is shown in Table A.2. Table A.3 shows the true values of key variables in the population (valid for all surveys) with only the last three columns specific for this instance (50 mm mesh) of the survey. True catchabilities by stage are the survey indices divided by the true population numbers in each year; their ratio gives the true s.

Table A.1. Specifications of the age-structured population and fishery simulation used to generate the CSA input data.

Natural mortality	$M = 0.2$, all ages and years
Growth	$K = 0.15$; $L_{inf} = 100$; $t_o = -0.5$
Length weight	$a = 0.00001$; $b = 3$
Maturity at age	4*0.0, 0.3, 0.5, 0.7, 0.9, 0.95, 6*1.0[a]
Recruitment:	
Type	Beverton-Holt: $R = S/(\alpha+\beta S)$
Parameters	$\alpha = 0.7295$; $\beta = 7.599\text{E}{-5}$ (steepness = 0.7)
Variability	Log-normal; CV = 0.6; autocorrelation $\rho = 0.5$
Selectivity at age	
Fishery	0.05, 0.1, 0.3, 0.7, 0.9, 10*1.0[a]
Survey	Variable depending on assumed mesh size
Survey nominal q	0.001

[a] The notation "n*value" means that the value applies to the next n ages.

Table A.2. Example of input data: 50 mm survey.

	Catches (thousands)[a]		Survey indices		Mean weight (kg)	
Year	Recruits	Full-rec	Recruits	Full-rec	Recruits	Full-rec
1	83.8	2,229.8	1.463	9.154	0.091	1.177
2	79.1	1,861.6	1.444	7.691	0.091	1.171
3	28.3	1,557.1	0.566	6.784	0.095	1.133
4	55.1	1,334.6	0.968	5.049	0.090	1.265
5	52.8	1,133.2	0.911	4.370	0.091	1.205
6	48.3	923.2	0.812	3.922	0.091	1.098
7	39.6	793.3	0.705	3.599	0.091	1.035
8	41.0	707.6	0.769	3.294	0.091	1.034
9	23.8	638.8	0.492	3.209	0.092	1.010
10	38.0	575.4	0.789	2.806	0.090	1.087
11	35.3	534.8	0.791	2.954	0.091	1.015
12	13.4	497.4	0.347	3.124	0.094	0.973
13	16.5	435.5	0.473	2.656	0.090	1.140
14	42.6	460.5	1.122	2.498	0.090	1.218
15	41.1	476.6	1.045	3.294	0.091	0.963
16	36.1	569.2	0.850	3.845	0.092	0.898
17	36.8	687.9	0.824	3.925	0.091	0.959
18	43.6	756.1	0.920	3.839	0.091	1.019
19	35.9	769.0	0.731	3.840	0.092	1.012
20	42.7	750.6	0.825	3.559	0.091	1.044

[a] Here, catches have been subdivided by stage but this is not required by CSA.

Table A.3. Corresponding true states.

		Stock size				Survey *q*		
Year	Total biomass	Recruits	Full-rec	SSB	Harvest rate[b]	Recruits[a]	Full-rec[a]	*s* ratio[a]
1	11,065.4	2,805.1	9,381.5	3,773.7	0.190	0.507	0.984	0.515
2	9,286.4	2,770.2	7,902.3	2,923.7	0.182	0.507	0.983	0.516
3	7,817.5	1,006.3	6,996.9	2,504.1	0.198	0.519	0.981	0.529
4	6,565.9	1,899.0	5,130.4	2,099.4	0.198	0.503	0.989	0.509
5	5,439.2	1,741.1	4,508.7	1,755.2	0.190	0.507	0.981	0.517
6	4,464.9	1,549.8	4,053.8	1,481.8	0.173	0.508	0.979	0.518
7	3,862.2	1,343.7	3,716.7	1,087.5	0.165	0.508	0.980	0.518
8	3,552.2	1,482.1	3,395.7	921.2	0.153	0.506	0.981	0.516
9	3,341.6	918.3	3,321.6	856.3	0.156	0.511	0.979	0.522
10	3,193.0	1,543.4	2,876.1	830.0	0.139	0.504	0.984	0.512
11	3,149.2	1,517.5	3,067.5	826.8	0.124	0.507	0.977	0.519
12	3,117.3	625.2	3,241.6	855.3	0.132	0.517	0.978	0.529
13	3,115.0	921.1	2,706.6	842.0	0.125	0.504	0.987	0.511
14	3,240.6	2,211.0	2,563.4	957.0	0.105	0.502	0.983	0.511
15	3,370.9	1,995.6	3,456.5	1,057.1	0.095	0.508	0.971	0.523
16	3,619.8	1,613.0	3,998.2	981.7	0.108	0.509	0.976	0.521
17	3,923.0	1,580.2	4,049.7	881.7	0.129	0.507	0.980	0.517
18	4,084.8	1,774.0	3,958.1	949.3	0.140	0.506	0.981	0.516
19	4,032.6	1,384.8	3,974.6	1,010.1	0.150	0.509	0.979	0.520
20	3,871.9	1,591.9	3,665.2	999.3	0.151	0.506	0.981	0.515

[a] Only the figures in the three rightmost columns are specific to the 50 mm mesh survey.

[b] Total catch divided by total stock size, both in number.

A Modified Catch-Length Analysis Model for Golden King Crab (*Lithodes aequispinus*) Stock Assessment in the Eastern Aleutian Islands

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Abstract

The golden king crab (*Lithodes aequispinus*) has provided a valuable fishery in the Aleutian Islands region during the past two decades. Despite their economic importance, the Aleutian Islands golden king crab stock has not been surveyed annually, biological data are limited, and assessment models are lacking. As an initial assessment attempt, we applied a modified version of the catch-length analysis method to observer data of males in association with total catch and effort data to estimate total legal male (≥ 136 mm carapace length, CL) and total male recruit (≥ 101 mm CL) abundances, and legal male harvest rate. We used the 1996-2001 data from catches in the eastern Aleutian Islands to determine these parameters. The estimated male recruit abundance fluctuated while estimated legal male abundance increased from 1996 to 2000 and slightly decreased in 2001 for the best fit model. The estimated legal male harvest rate at the time of the fishery decreased from 99% in 1996 to 52% in 2001. These findings should be treated cautiously because of the short history of data sets and lack of independent and reliable estimates of some vital biological parameters. The modeling procedure, however, demonstrated

the feasibility of using observer data in stock assessment when no annual stock surveys are available.

Introduction

Due to the deepwater and rough-bottom habitat, golden king crabs are not surveyed by trawl gear and annual stock-abundance estimates are not provided for this species from National Marine Fisheries Service (NMFS) surveys. In comparison with other Bering Sea crab stocks, such as Bristol Bay red king crab (*Paralithodes camtschaticus*) and Bering Sea snow crab (*Chionoecetes opilio*), stock assessment models have not been developed to determine appropriate harvest levels because of limited biological and stock survey information. The Alaska Department of Fish and Game (ADFG) divides the Aleutian Islands golden king crab (*Lithodes aequispinus*) fishery into eastern and western districts at 174ºW (ADFG 2002). The stocks in the two areas are managed with a mostly constant annual guideline harvest level or total allowable catch (3.0 million pounds [about 1,500 tons] for the eastern Aleutian Islands based on 1991/92-1995/96 mean adjusted catch; and 2.7 million pounds for the western Aleutian Islands based on 1990/91-1994/95 mean catch). Additional management measures are a male-only fishery, a minimum legal size limit (152.4 mm carapace width [CW], approximately 136 mm carapace length [CL], which is at least one annual molt increment higher than the 50% maturity length 120.8 mm CL [Otto and Cummiskey 1985]), and in-season monitoring of fishery performance by daily catch and catch per unit effort determination. These guideline harvest levels have yet to be shown appropriate for sustainable management.

The life history characteristics of golden king crab pose problems to development of appropriate stock assessment models. Golden king crab larvae are lecithotrophic and not known to rise to the upper water layer to feed, suggesting that the spring bloom is an unlikely cue for spawning and the spawning period is protracted (Otto and Cummiskey 1985, Shirley and Zhou 1997). Tagging and laboratory studies indicated most crabs in old-shell condition will molt within the next year, whereas the probability of molting within the next year of males in new-shell condition is negatively correlated with size (Paul and Paul 2001, Watson et al. 2002).

Modeling the complicated dynamics of golden king crab is feasible, but limited stock information prevents developing a comprehensive model at this time. Under these circumstances, we followed a simpler approach and modeled the dynamics of only male golden king crabs from the eastern Aleutian Islands fishery. We employed a modified version of the catch-length analysis (CLA) method (Sullivan et al. 1990) to estimate stock and recruit abundances, harvest rate, and a number of stock assessment model parameters.

Materials and methods

The primary data required by the modified CLA model are a time series of male CPUE (number of crabs per pot lift) by length, and commercial catch and effort. The commercial catch is defined as the number of legal crabs landed and effort is defined as the total number of pot lifts in the fishery. In addition, the model requires the annual growth increment per molt at each size, instantaneous natural mortality (M), and handling mortality rate of discards (h) as input parameters.

Regulations stipulate 100% observer coverage in the golden king crab fishery. Observers randomly select a predetermined number of pots daily and examine the entire pot contents for catch composition, including carapace lengths and shell condition. This is referred to as "bycatch sampling." The number of pots sampled accounts for 5-7% of the total pot lifts in the eastern Aleutian Islands fishery (Moore et al. 2000, Barnard et al. 2001, Neufeld and Barnard 2003). Detailed observer sampling procedures are given in Neufeld and Barnard (2003). Observer data have been collected since 1988, but due to a change in the management area boundaries in 1995 and the increased reliability of latter seasons' data, we selected a shorter time series for the period 1996-2001 to fit the model.

Length-specific CPUE data collected by at-sea observers provide information on a wider size range of the stock than does the commercial catch length frequency data on only legal-sized males. Monthly mean length frequency data were constructed from 1996-2001 observer samples. The mean CPUE for retained legal male crabs ($\geq$ 136 mm CL), discarded legal male crabs, and sublegal male crabs (< 136 mm CL) were estimated for each month. We considered a restricted size range, 101-185 mm CL, to input some externally estimated population parameters to fit the CLA model (e.g., the estimate of the β parameter of the annual growth increment distribution model based on tagging data is applicable only to a size range 91-185 mm CL). The total male CPUE for each month was estimated by adding each male CPUE category (retained legal, discarded legal, and sublegal) and the monthly length frequency was used to split the total CPUE into monthly length-specific CPUE. If the fishery exceeded one month, a weighted average (weighted by the catch) of the monthly length-specific CPUE was determined for the season. The length-specific CPUEs were determined separately for new-shell and old-shell crabs for each season.

The annual catch and effort for the eastern Aleutian Islands fishery were estimated from the ADFG landing records (fish tickets). The total annual catch was used to estimate the predicted effort in the model (see the model section below). The fishing time δ_t (as a fraction of a year), and the elapsed time from an arbitrary mid-survey date of July 1 to the date of fishing season opening T_t (as a fraction of a year) were estimated for each year (ADFG 2002). The fishing and elapsed times for different fish-

Table 1. Elapsed time from July 1 (an arbitrarily set mid-survey time) to start of the fishery T_t, and fishing period δ_t for the eastern Aleutian Islands golden king crab fishery, 1996-2001.

Fishing season	T_t (years)	δ_t (years)
1996	0.1699	0.3151
1997	0.1699	0.2301
1998	0.1699	0.1863
1999	0.1699	0.1507
2000	0.1260	0.1096
2001	0.1260	0.0712

ing seasons (Table 1) were used to determine predicted CPUE and effort during the fishery for fitting the model.

Watson et al. (2002) estimated a mean annual growth increment of 14.4 mm CL for combined shell types, new-shell and old-shell, of tagged male crabs recaptured 12-15 months after release. The release size range was 91-185 mm CL. Reanalysis of their tagging data indicated that the linear relationship between annual growth increment and pre-molt length was not significant ($P > 0.05$). Thus, we computed a mean annual growth increment and the variance from their original data to be applicable to the entire length range considered in the analysis.

The North Pacific Fishery Management Council (NPFMC) considers an M of 0.2 for all king crab harvest management strategies (NPFMC 1999). Siddeek et al. (2002), based on tag-recapture data analysis, estimated an M of 0.38 for Aleutian Islands male golden king crab. We considered both these M values and a middle M value of 0.3 for sensitivity analysis.

Scant information is available on the level of handling mortality as a result of capture and release of unmarketable crabs although a large number of sublegal males (and females) are captured and released in the fishery (Blau et al. 1996, Neufeld and Barnard 2003). We used an arbitrary 20% handling mortality rate (h) on discarded males, which was borrowed from the red king crab literature (Kruse et al. 2000, Siddeek 2002). We also used 0% and 50% handling mortality rate to investigate handling effects on other parameter estimates.

The modified CLA model

The modified CLA model used for golden king crab data analysis was an adaptation from models presented in Sullivan et al. (1990), Zheng et al. (1996), and Lai and Bradbury (1998) to analyze length specific CPUE with catch and effort data. Following Sullivan et al. (1990), Zheng et al. (1996) used the catch and effort data of red king crab, whereas Lai and Bradbury (1998) used the diver survey size frequency data with selectivity information from commercial landings of red sea urchin (*Strongylocentrotus franciscanus*) in assessing the respective stock parameters by CLA.

The molt probability (m_i) for a given length class i was described by the logistic function:

$$m_i = \frac{1}{1+e^{-a+bi}} \quad , \tag{1}$$

where a and b are parameters to be determined by the model.

The expected proportion of molting crabs ($P_{i,\,j}$) growing from length class i to length class j during an year was described by the gamma distribution as follows:

$$P_{i,\,j} = \frac{\int_{j_1-\tau_i}^{j_2-\tau_i} gamma(x/\alpha_i,\beta)dx}{\sum_{j=1}^{n}\int_{j_1-\tau_i}^{j_2-\tau_i} gamma(x/\alpha_i,\beta)dx} \tag{2}$$

where

$$gamma(x/\alpha_i,\beta) = \frac{x^{\alpha_i-1}e^{-\frac{x}{\beta}}}{\beta^{\alpha_i}\Gamma(\alpha_i)}$$

and where x is the growth increment, α_i and β are parameters, j_1 and j_2 are lower and upper limits of the receiving length interval j, τ_i is the midpoint of the contributing length interval i, and n is the total number of receiving length intervals. The summation in the denominator is a normalizing factor for the discrete gamma function.

The mean annual growth increment per molt is equivalent to $\alpha_i\beta$ and the variance is equivalent to $\alpha_i\beta^2$ (Sullivan et al. 1990). Mean annual growth increment (14.423 mm CL) and the variance (8.337) estimated from 1997 golden king crab tagging experiments in the Aleutian Islands were used to estimate β (variance/mean = 0.578) and α_i (mean/β = 24.95) as fixed input parameter values to the CLA model. Although α_i has a subscript *i* to represent a length-specific value, it was treated as a constant because of lack of a linear relationship found between annual growth increment and pre-molt length for this stock.

The total number of annual recruits (R_t) to the sublegal and legal categories (101-185 mm CL) was distributed to different length intervals (*i*) by a fixed proportion (P''_i). Thus,

$$R_{i,t} = R_t P''_i \tag{3}$$

where P''_i was estimated using a gamma distribution model as follows:

$$P''_i = \frac{\int_{i_1}^{i_2} gamma(x/\alpha_r,\beta_r)dx}{\sum_{i=1}^{n}\int_{i_1}^{i_2} gamma(x/\alpha_r,\beta_r)dx} \tag{4}$$

where *x* is the length, α_r and β_r are parameters, i_1 and i_2 are lower and upper limits of the receiving length interval *i*, and *n* is the total number of receiving length intervals.

The annual abundances of male crabs by shell condition were determined separately for fishery pre-recruit length groups (101-135 mm CL) and fishery recruit and post-recruit length groups (136-185 mm CL), and then summed up to obtain the overall abundances by each length group. The annual total recruits to the legal and sublegal groups were assumed to fall into new-shell length intervals.

The abundances of crab growing into the same or any larger length intervals were determined by,

$$\mathrm{N}_{j,t+1} = \sum_{i=1}^{j}[(N_{i,\mathrm{t}} + O_{i,\mathrm{t}})m_i \mathrm{P}_{i,j}]\mathrm{e}^{-(\mathrm{M}+u_{i,t}\,\delta_t)} + R_{j,t+1} \tag{5}$$

$$O_{j,t+1} = (N_{j,t} + O_{j,t})(1-m_j)\mathrm{e}^{-(\mathrm{M}+u_{j,t}\,\delta_t)} \tag{6}$$

where

$u_{j,t} = HM_{j,t}$ for $j \le 135$ mm CL

$u_{j,t} = F_{j,t}$ for $j \geq 136$ mm CL

where N and O are numbers of new and old shell crabs, respectively; $F_{j,t} = qf_t s_j$ is the size-specific annual instantaneous fishing mortality, q is the commercial pot gear catchability, f_t is the annual fishing effort, and s_j is the length-specific pot selectivity given by the following logistic model:

$$s_j = \frac{1}{1+e^{-c(j-d)}} \tag{7}$$

where c and d were estimated by fitting the model.

The annual instantaneous handling mortality $HM_{j,t}$, at length j, is a variable and is defined as a function of $F_{j,t}$, h, and δ_t, ignoring M during the brief fishing period as follows:

$$1-e^{-HM_{j,t}\delta_t} = h(1-e^{-\mathrm{F}_{j,t}\delta_t}).$$

Therefore,

$$HM_{j,t} = -\frac{1}{\delta_t}\ln[1-h(1-e^{-F_{j,t}\ \delta_t})] \tag{8}$$

The predicted new-shell catch per unit effort (NCPUE) was determined by:

$$NCPUE_{j,t} = \frac{q\, s_j}{F_{j,t}+M} N_{j,t}\, e^{-M\,T_t}(1-e^{-(F_{j,t}+M)\,\delta_t}) \tag{9}$$

The old-shell catch per unit effort (OCPUE) was predicted by:

$$OCPUE_{j,t} = \frac{q\, s_j}{F_{j,t}+M} O_{j,t}\, e^{-M\,T_t}(1-e^{-(F_{j,t}+M)\,\delta_t}) \tag{10}$$

The annual effort was predicted by:

$$f_t = \frac{C_t}{q \displaystyle\sum_{j=legal}^{n} \frac{s_j e^{-M\,T_t}(1-e^{-(F_{j,t}+M)\,\delta_t})}{(F_{j,t}+M)}(N_{j,t}+O_{j,t})} \tag{11}$$

where C_t is the annual total catch in number of crabs.

Even though some undesirable legal-sized males (those excessively covered with epifauna, have severe shell rot, too many missing or regenerating legs, or have leather shell) are discarded in the fishery, for simplicity we assumed their numbers to be minimal, and a 136 mm CL cutoff point was used to separate the length range into two groups to apply two different total mortality in the stock dynamics formulas: $HM_{j,t} + M$ to crabs < 136 mm CL (sublegal males) and $F_{j,t} + M$ to crabs ≥ 136 mm CL (legal males).

Following Quinn et al. (1998), a root normal distribution was assumed to construct an objective function for parameter estimation. We considered additive measurement/observation error, but not process error (i.e., no errors either on the parameter values that were fixed or model specification) to formulate the minimization function, which is a combination of residual sum of squares (SSQ) of square root of length-specific CPUE and square root of annual effort. Although square root transformation is likely to reduce the contrast in the magnitude of different data sources (Fu 2000), we further reduced the contrast by scaling down the effort by a factor of 10^{-5}.

$$\begin{aligned} SSQ &= \sum_{t,j}\left(\sqrt{NCPUE_{j,t}} - \sqrt{N\hat{C}PUE_{j,t}}\right)^2 \\ &+ \sum_{t,j}\left(\sqrt{OCPUE_{j,t}} - \sqrt{O\hat{C}PUE_{j,t}}\right)^2 \\ &+ \lambda \sum_{t,j}\left(\sqrt{f_t} - \sqrt{\hat{f}_t}\right)^2 \end{aligned} \tag{12}$$

where the "hat" sign refers to predicted values. The λ is the weight that describes the degree of belief in the effort estimates compared to CPUE (Sullivan et al. 1990, Zheng et al. 1996). High λ values indicate high confidence in the effort observation compared to CPUE. We varied λ and compared the parameter estimates for different weighting factors. We used the IMSL Fortran subroutine DBCNLS (IMSL 2000) to estimate parameters of the model by setting non-negative bounds to all parameters.

Starting from the second year, annual abundances by length and shell condition were estimated recursively from (1) the abundances by length and shell condition in the preceding year, (2) annual recruitment to the new-shell category, and (3) mortality and growth (annual molt probability and growth increment probability) parameters. The 1996 observer relative length frequency data were used to distribute the first-year total new-shell and old-shell abundances (which are parameters to be estimated by the model) into length groups. To reduce the number of parameters to be estimated from the limited data sets, we fixed some of the model parameters (M, h, β) at their plausible values. We estimated 14 parameters from 210 quantities by minimizing SSQ: a and b of the molt probability model; c and d of the pot selectivity model; α_r and β_r of the recruitment distribution model; catchability q; total new-shell abundance in the first year (i.e., in 1996) N_{96}; total old-shell abundance in the first year O_{96}; and total number of male recruits for each year, except the first year, R_{97}, R_{98}, R_{99}, R_{00}, R_{01}.

Results

In general, as the value of M was increased, estimates of the initial new-shell and old-shell male crab total abundances and annual total number of male recruits increased. The initial abundance of old-shell crabs was small (Table 2). Closer model fits (i.e., lower SSQ values) were realized for $\lambda = 0$ followed by 1 for the three M values (0.2, 0.3, and 0.38). Higher λ values (5 and 10) produced larger SSQ values than those for $\lambda = 0$ and 1. In some cases of higher λ values, the objective function did not converge and required an addition of a small constant (0.05 or 0.1) to each component of the function to induce convergence (e.g., the scenarios, $M = 0.2$ and $\lambda = 10$, and $M = 0.38$ and $\lambda = 5$). There was a tendency for SSQ to decrease with increasing M and decreasing λ (Table 2). The lowest λ value (= 0) increased the precision of fits for all selected M values. Thus, we chose the parameter estimates at $M = 0.3$, $h = 20\%$, and $\lambda = 0$ as plausible for this stock. Further justifications for this choice are given in the subsequent sections.

Table 3 provides the description and estimates of parameters for the chosen fit. Figure 1 shows a close fit of the gamma distribution model with the input β of 0.578 to relative frequencies of observed growth increments estimated from tagging data. Figure 2 illustrates the pot selectivity curve for the estimated parameters under this fit. The 50% selectivity length, 125 mm CL, was 11 mm lower than the minimum legal size, 136 mm CL. The commercial pot catchability was 0.000145.

Parameter estimates at $\lambda = 0$ for the three M values, zero and 50% handling mortality, and for a scenario when 90% of old-shell crabs had a molting probability of 1 (a conclusion made by Watson et al. 2002) are provided in Table 4. As the handling mortality was increased to 50%, the initial new-shell male total abundances slightly decreased, and the total number of male recruits and catchability increased. The different molt probability assumption on old-shell males produced only slight changes in initial male stock and total recruit abundances, and catchability.

The time series of predicted versus observed length-specific CPUE of new-shell and old-shell crabs for the best fit (for $M = 0.3$, $h = 20\%$, $\lambda = 0$) and the next best fit (for $M = 0.3$, $h = 20\%$, $\lambda = 1$) scenarios are shown in Fig. 3. The two curves are almost identical indicating that either one may adequately describe the data, but we chose the scenario $\lambda = 0$ based on the lowest SSQ value. For new-shell crabs, the trends in predicted

Table 2. Parameter estimates by the modified catch-length analysis model for eastern Aleutian Islands male golden king crab for different sets of natural mortality (*M*) and effort weighting factor (λ) values at a fixed handling mortality rate (*h*) = 20%.

M	0.2	0.2	0.2	0.2	0.3	0.3	0.3	0.3	0.38	0.38	0.38	0.38
λ	0	1	5	10	0	1	5	10	0	1	5	10
a	18.8393	18.8142	18.5324	25.3473	18.2810	18.2546	18.0421	18.9258	17.8494	17.8206	20.4747	18.8098
b	0.1236	0.1234	0.1209	0.1713	0.1199	0.1197	0.1178	0.1248	0.1171	0.1168	0.1357	0.1245
c	0.0250	0.0250	0.0250	0.0420	0.0250	0.0250	0.0295	0.0419	0.0250	0.0250	0.0366	0.0412
d	125.0000	125.0000	125.0000	134.9572	125.0000	125.0000	125.0000	125.3827	125.0000	125.0000	130.4090	130.8745
β	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780
α_r	47.0864	47.3457	50.3351	37.5726	50.8264	51.1484	52.7932	47.6984	53.5171	53.8750	49.6782	48.7893
β_r	2.6640	2.6497	2.4954	3.3038	2.4794	2.4642	2.3823	2.6248	2.3619	2.3468	2.5300	2.5669
N_{96}	1.7990	1.8170	1.9530	1.8821	1.8849	1.9044	2.0498	2.0357	1.9627	1.9836	2.0430	2.0958
O_{96}	0.0335	0.0340	0.0369	0.0457	0.0350	0.0354	0.0380	0.0368	0.0363	0.0368	0.0403	0.0374
R_{97}	1.4118	1.4238	1.5671	1.5319	1.5599	1.5770	1.7856	1.8684	1.6887	1.7100	1.8638	2.0452
R_{98}	1.7668	1.7681	1.7073	1.4462	2.0039	2.0086	1.9712	1.7635	2.2121	2.2202	2.0626	1.9540
R_{99}	1.1527	1.1627	1.2333	1.1145	1.3409	1.3550	1.4532	1.3722	1.5141	1.5322	1.5435	1.5510
R_{00}	1.8999	1.9335	2.3088	1.8760	2.2212	2.2665	2.7353	2.5697	2.5080	2.5641	2.8008	2.8191
R_{01}	0.4925	0.5127	0.8815	1.5114	0.6130	0.6436	1.1494	2.0547	0.7305	0.7712	1.3007	2.4087
$q \times 10^6$	160.8427	158.7354	139.3914	203.9854	145.4598	143.1815	121.8830	123.3007	133.8935	131.5759	138.7146	127.7336
SSQ	1.1824	1.2239	1.8528	1.9604	1.0693	1.1171	1.8166	2.6788	0.9980	1.0510	1.5068	2.6839

SSQ = minimum total sum of squared errors between observed and expected values. N = millions of new shell crabs, O = millions of old shell crabs, R = total male recruits to 101-185 mm CL size groups in millions of crabs. The subscripts 96, 97, 98, etc. refer to 1996, 1997, 1998, etc.

Table 3. Description of parameters and selected estimates at effort weighting factor $\lambda = 0$, $h = 20\%$, M = 0.3, and $\beta = 0.5780$ from the modified catch-length analysis model for eastern Aleutian Islands male golden king crab.

Parameter	Description	Estimate	Remarks
λ	Effort weighting factor in SSQ	0	Fixed input value
M	Annual instantaneous natural mortality	0.3	Fixed input value
h	Percentage of discarded males and females died due to handling and discarding at sea	20%	Fixed input value
a	Logistic male molt probability model parameter	18.2810	Model fitted value
b	Logistic male molt probability model parameter	0.1199	Model fitted value
c	Logistic pot selectivity model parameter	0.0250	Model fitted value
d	Logistic pot selectivity model parameter	125.0000	Model fitted value
β	A parameter in the gamma distribution to describe variation in annual growth increment	0.5780	Fixed input value estimated from tagging data
α_i	A parameter in the gamma distribution to describe variation in annual growth increment, independent of size (i) for this stock.	24.9517	Fixed input value estimated from tagging data
α_r	A parameter in the gamma distribution to describe variation in recruitment by length class	50.8264	Model fitted value
β_r	A parameter in the gamma distribution to describe variation in recruitment by length class	2.4794	Model fitted value
q	Catchability coefficient of commercial pot gear	0.0001	Model fitted value
N_{96}	Total new-shell male abundance (millions of crabs) in 101-185 mm CL size groups in 1996	1.8849	Model fitted value
O_{96}	Total old-shell male abundance (millions of crabs) in 101-185 mm CL size groups in 1996	0.0350	Model fitted value
R_{97}	Total male recruit abundance (millions of crabs) in 101-185 mm CL size groups in 1997	1.5599	Model fitted value
R_{98}	Total male recruit abundance (millions of crabs) in 101-185 mm CL size groups in 1998	2.0039	Model fitted value
R_{99}	Total male recruit abundance (millions of crabs) in 101-185 mm CL size groups in 1999	1.3409	Model fitted value
R_{00}	Total male recruit abundance (millions of crabs) in 101-185 mm CL size groups in 2000	2.2212	Model fitted value
R_{01}	Total male recruit abundance (millions of crabs) in 101-185 mm CL size groups in 2001	0.6130	Model fitted value
SSQ	Minimum total sum of squared errors between observed and expected with d.f. = 196	1.0693	Model fitted value

Table 4. Parameter estimates by the modified catch-length analysis model for eastern Aleutian Islands male golden king crab for different sets of natural mortality (*M*) and handling mortality rate (*h*) values at fixed effort weighting factor (λ) = 0.

M	0.2	0.2	0.3	0.3	0.38	0.38	0.2	0.3	0.38
h (%)	0	50	0	50	0	50	20	20	20
Molt prob.	Size spec.	Size spec.	Size spec.	Size spec.	Size spec.	Size spec.	New-shell: size spec. Old-shell: 0.9	New-shell: size spec. Old-shell:0.9	New-shell: size spec. Old-shell: 0.9
a	19.0829	17.9199	18.5194	17.4021	18.0885	16.9920	19.5402	18.9663	18.5203
b	0.1245	0.1183	0.1209	0.1148	0.1181	0.1121	0.1288	0.1251	0.1221
c	0.0250	0.0250	0.0250	0.0250	0.0250	0.0250	0.0250	0.0250	0.0250
d	125.0000	125.0000	125.0000	125.0000	125.0000	125.0000	125.0000	125.0000	125.0000
β	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780	0.5780
α_r	41.8248	55.3318	46.6460	57.5987	50.0593	59.2916	47.0897	50.8239	53.5181
β_r	2.9495	2.3077	2.6680	2.2192	2.4997	2.1574	2.6645	2.4801	2.3624
N_{96}	1.8063	1.7948	1.8990	1.8757	1.9816	1.9506	1.7967	1.8822	1.9593
O_{96}	0.0337	0.0334	0.0353	0.0348	0.0366	0.0360	0.0331	0.0345	0.0357
R_{97}	1.2920	1.6116	1.4546	1.7407	1.5952	1.8562	1.4101	1.5574	1.6857
R_{98}	1.6239	2.0035	1.8883	2.2088	2.1171	2.3952	1.7612	1.9969	2.2037
R_{99}	1.0487	1.3671	1.2410	1.5434	1.4201	1.7065	1.1528	1.3401	1.5124
R_{00}	1.7924	2.0936	2.1426	2.3822	2.4528	2.6470	1.8941	2.2130	2.4978
R_{01}	0.4537	0.5911	0.5726	0.7178	0.6872	0.8383	0.4881	0.6080	0.7242
$q \times 10^6$	151.0557	170.2569	137.0611	153.1366	126.5565	140.1409	161.7836	146.3555	134.7492
SSQ	1.3002	1.1086	1.1375	1.0449	1.0371	1.0017	1.1893	1.0764	1.0053

SSQ = minimum total sum of squared errors between observed and expected values, Old-shell: 0.9 = 90% of old-shell crabs have a molting probability of 1. N = millions of new-shell crabs, O = millions of old-shell crabs, R = total male recruits to 101-185 mm CL size groups in millions of crabs. The subscripts 96, 97, 98, etc. refer to 1996, 1997, 1998, etc.

values were close to those observed during 1996-2001, except 2000. On the other hand, old-shell crab fits were not good, providing acceptable fits only for a few years: 1996 and 2000, and to a lesser extent 1998 and 1999. This was not surprising because of small number of old-shell crabs caught in the fishery.

The trends in predicted versus observed effort at $M = 0.3$, $h = 20\%$ for different λ values, are illustrated in Fig. 4. The trends in the predicted effort for $\lambda = 0$ and 1 are similar and closer, but these two did not provide better fit to effort data compared to higher λ (5 and 10) values.

The trends in estimated legal male abundance at the time of the fishery and estimated total number of male recruits at $M = 0.3$, $h = 20\%$ for different λ values, are depicted in Fig. 5. The trends in total recruits and total legal male abundance are almost identical for $\lambda = 0$ and 1. At these two λ values, legal male abundance steadily increased during 1996-2000 and slightly dropped in 2001. The same trends in legal male abundance were observed for $\lambda = 5$ and 10, but the 2001 drop started to straighten up and increase with increase in the λ value. On the other hand, the total recruit abundance fluctuated during this period for all λ values and sharply increased in 2000 and dropped in 2001. The terminal year estimates cannot be taken very seriously because of inherent errors in the forward projection by CLA with a short time series of data.

The estimated harvest rate of legal males (observed legal male catch divided by estimated legal male abundance at the time of the fishery) systematically declined from the initial year very high value: 99% in 1996 and 1997, 70% in 1998, 61% in 1999, 47% in 2000, and 52% in 2001.

Discussion

Using observer sampling, catch and effort data from the commercial fishery, and growth information from tagging data, we developed a CLA model for eastern Aleutian Islands golden king crabs to estimate annual total abundances of male recruits and legal males, and annual harvest rates. The legal male abundance has systematically increased over time, indicating no adverse effect of the current harvest policy of prescribing a constant guideline harvest level for the fishery. The legal male harvest rate has systematically decreased over time. However, the total number of recruits has fluctuated and sharply declined in 2001, which signals a warning to pursuing a constant catch policy. A constant fishing mortality rate strategy is preferred to a constant catch strategy because the former takes into consideration high and low abundances and adjusts the yield accordingly to conserve the population (Braumann 2001). A constant harvest rate strategy is used in other Bering Sea crab stock management (NPFMC 1999). We intend to expand this study with a longer time series of data from diverse sources (fishery, resources survey, and tagging ex-

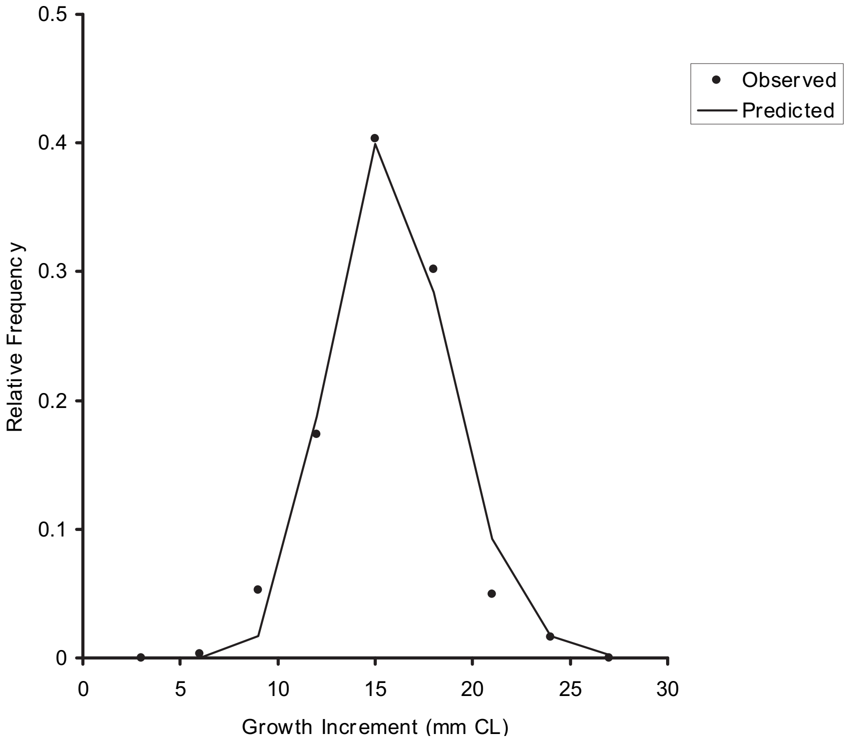

Figure 1. Comparison of observed and predicted relative frequencies of annual growth increment for eastern Aleutian Islands male golden king crab. The observed estimates are from the 1997 tagging experiment in the Aleutian Islands (Watson et al. 2002) while the predicted estimates are from fitting a gamma distribution with a β parameter of 0.578.

periments) to determine an optimum fishing mortality rate to replace the constant catch management policy for this stock.

The parameter estimates were more sensitive to higher λ values than 1. The best fit of the model was obtained for the lowest λ value (= 0), which ignores the independent information on annual effort in fitting the model. However, the goodness of fit statistic (SSQ) at $\lambda = 1$ was much closer (<5.5% difference) to the lowest value realized at $\lambda = 0$. The estimated parameters at both λ values produced similar fits to observed CPUE and effort data, and estimated recruit and legal male abundances. As the λ value was increased, more emphasis was placed on effort data rather then CPUE data and hence the fit to effort data became better.

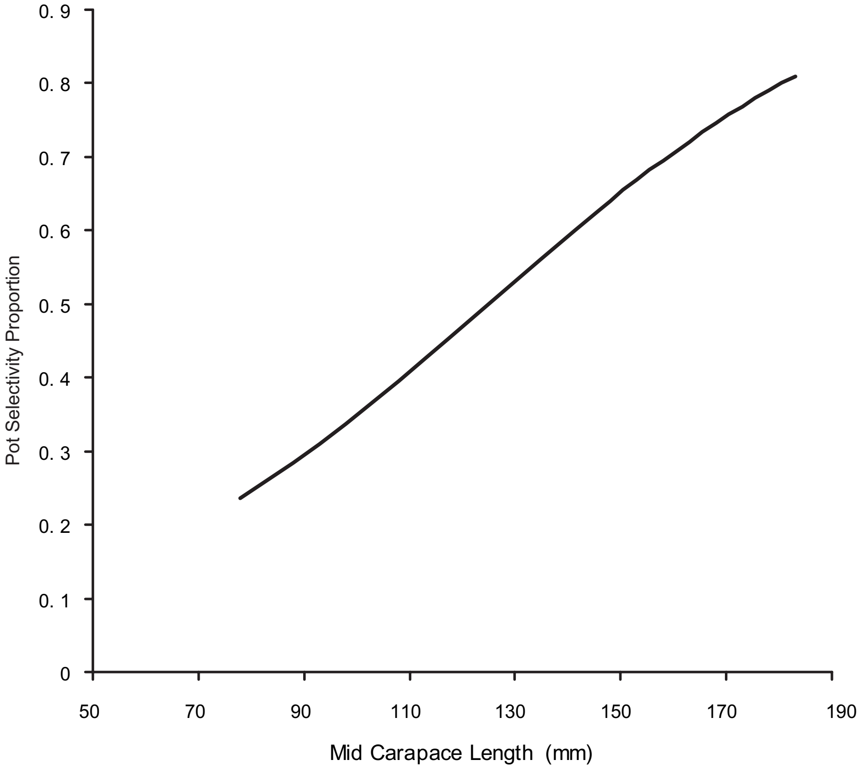

Figure 2. Commercial pot selectivity curve with estimated selectivity parameters, c = 0.025 and d = 125 mm CL for eastern Aleutian Islands male golden king crab. The model fitting procedure considered a zero weighting factor ($\lambda = 0$) for effort; M = 0.3; and h = 20% to estimate the parameters.

The CLA method is very useful for assessing data poor stocks lacking fishery-independent survey data. The Aleutian Islands golden king crab stock is not surveyed annually. However, a pilot pot survey with tagging was conducted in 1991 (Blau and Pengilly 1994) and triennial pot surveys and tagging experiments were performed in 1997, 2000, and 2003 (Blau et al. 1998, Watson and Gish 2002, Watson 2003). Continuation of the triennial survey would provide a long time series of data for any rigorous stock assessment. For now, we resorted to methods suitable for catch and effort data alone, which are regularly collected from the fishery. The Aleutian Islands golden king crab fishery has had observer coverage since 1988, which produced comprehensive data of the retained

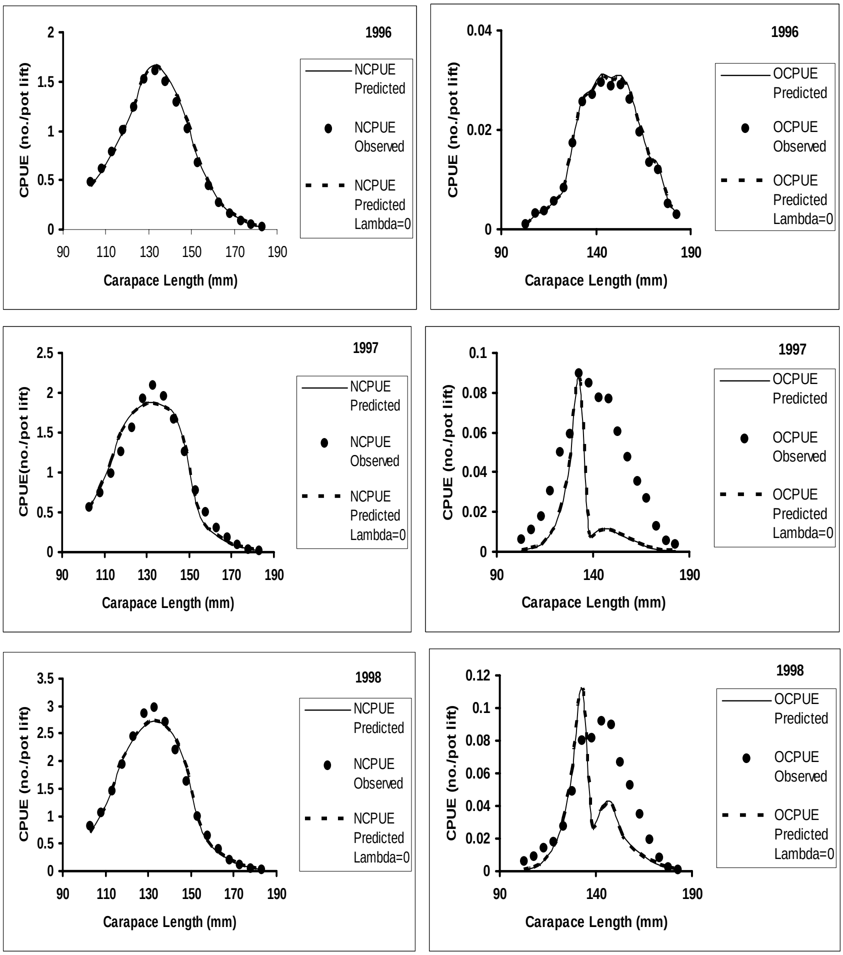

Figure 3. **Comparison of observed and estimated CPUE by length of new-shell (left) and old-shell (right) eastern Aleutian Islands male golden king crab, 1996-2001. The model fitting procedure considered two weighting factors for effort, lambda (λ) = 0 (dashed line) and 1 (solid line); M = 0.3; and h = 20%.**

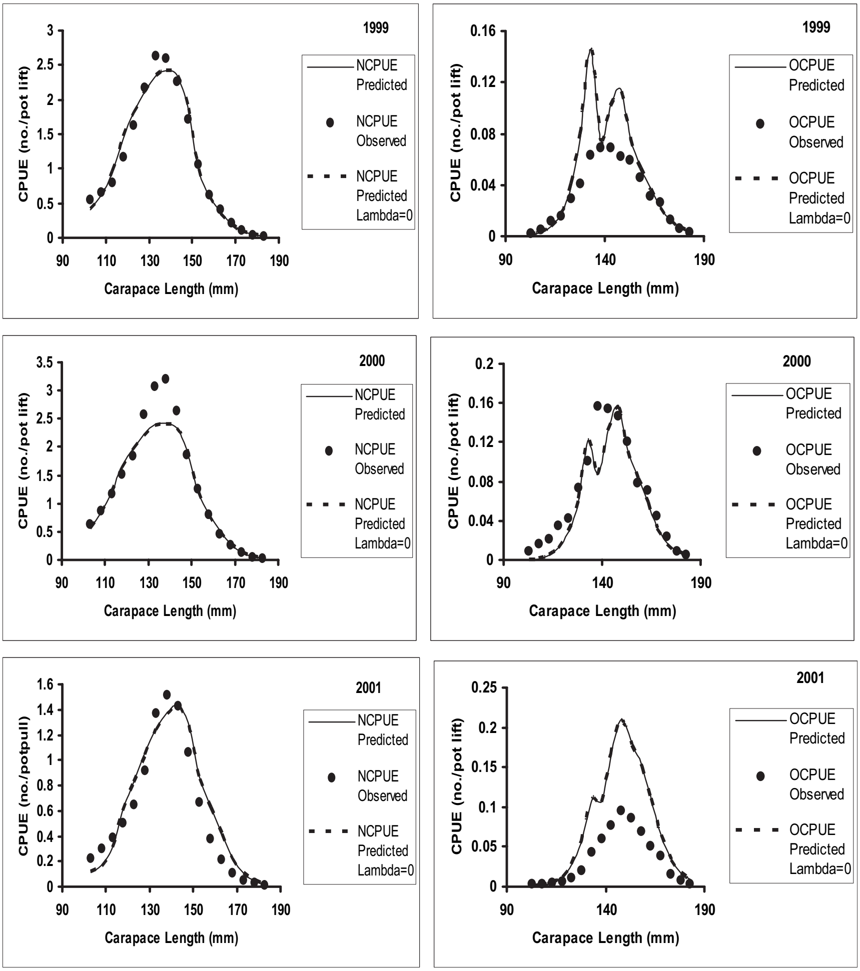

Figure 3. (Continued).

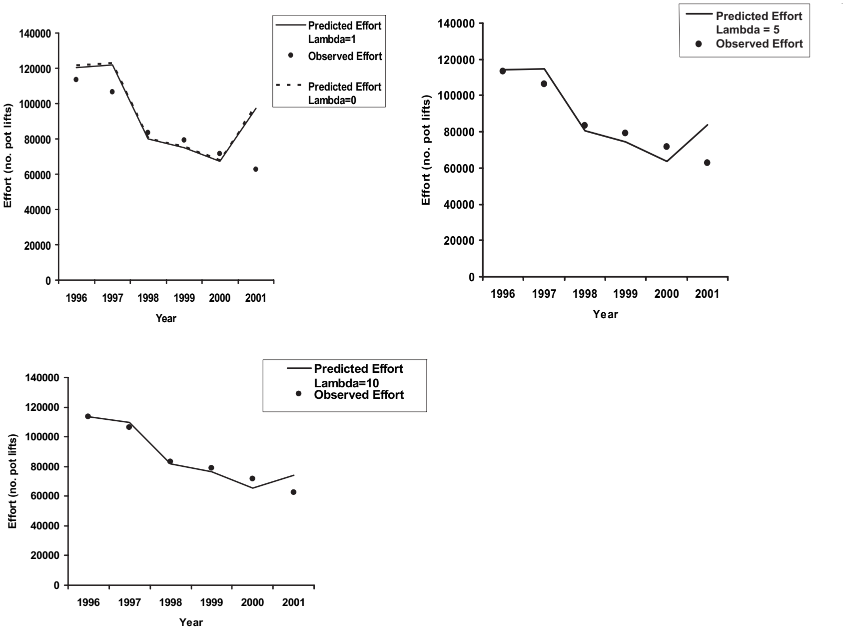

Figure 4. Comparison of observed and estimated fishing effort of eastern Aleutian Islands male golden king crab, 1996-2001. The model fitting procedure considered a range of weighting factors for effort, lambda (λ) = 0 (dashed line), 1, 5, and 10 (solid lines); M = 0.3; and h = 20%.

catch and all sizes of crabs retained by the pot gear. However, observer data suitable for modeling was only available in the late 1990s, and we selected a data set from 1996 to 2001. As far as we know, this is the first time that observer-sampling data have become a major component of crab population model fitting.

Even though square root transformation was more suitable than log transformation to handle different magnitudes of data in the objective function (Quinn et al. 1998), the model did not provide close fits to CPUE data, especially those for old-shell crabs. Apart from short time series of data considered in this analysis, very small magnitudes of length-specific CPUE, especially those of old-shell crabs, may have contributed to this discrepancy. Nevertheless, the modified CLA model captured the general trends in size-specific abundance (in terms of CPUE) reasonably well for new-shell crabs under these constraints.

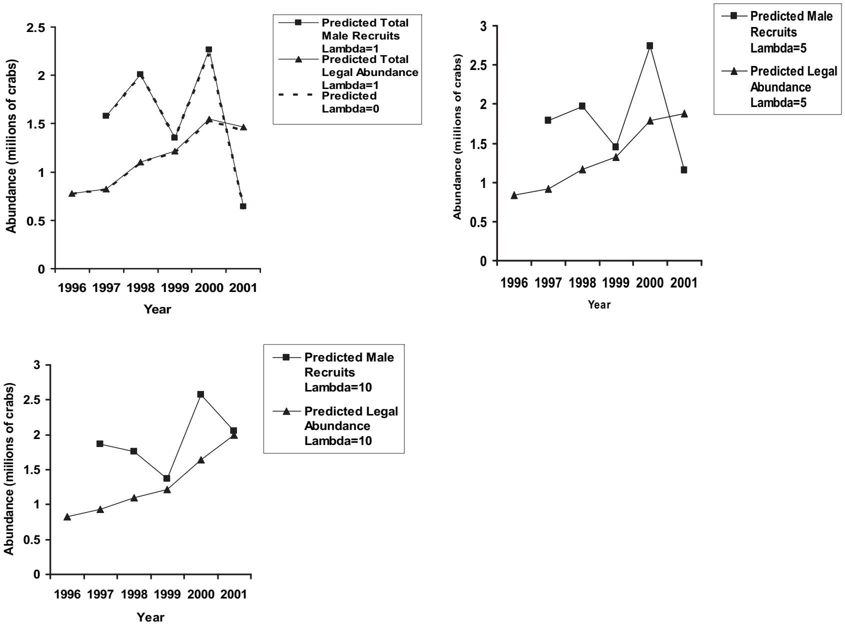

Figure 5. Trends in estimated annual total male recruit and total legal male abundances of eastern Aleutian Islands golden king crab, 1996-2001. The model fitting procedure considered a range of weighting factors for effort, lambda (λ) = 0 (dashed line), 1, 5, and 10 (solid lines); M = 0.3; and h = 20%. The total male recruit abundances are new arrivals to fishery pre-recruit, recruit, and post-recruit length groups, 101-185 mm CL.

The legal male harvest rates estimated from this study during the 1996-2001 fishing seasons were very high initially and sharply declined as time progressed. This was not surprising because the guideline harvest level was fixed at a constant level while the estimated legal male abundance continued to increase and observed effort continued to decrease. The reported commercial CPUE for this region also increased during this time period (ADFG 2002), supporting this finding. Because the commercial fishing fleet may or may not have covered the entire stock distribution in the eastern Aleutian Islands, a definite conclusion on the harvest level can be made only after analyzing more historical data and identifying the distribution of the fishing fleet in relation to stock distribution, which is beyond the scope of this paper.

One can argue that observer-sampling data may be biased because the data come from the commercial fishery and hence are not a representative random sample from the whole population. This is true for any modeling exercise using only commercial catch and effort data. The results are strictly applicable to that part of the stock available for the commercial fishery.

Because of limited data we fixed some of the important parameters at values that could be either estimated externally or guessed from related stock parameter values for optimization. We estimated the β parameter from tagging data and borrowed a value from the red king crab literature for h. We fixed the value of M at 0.3 for a number of reasons: (1) it produced lower SSQ value than that for an M of 0.2 in the current analysis; (2) the 1997 Aleutian Islands golden king crab tag-recapture data analysis produced an M of 0.38 (Siddeek et al. 2002), which may be an overestimate due to tag related errors; and (3) values for M higher than 0.2 have been considered in other king crab stock analyses (e.g., M of 0.3 for the Bristol Bay and Kodiak red king crabs [Kruse et al. 2000], M of 0.37 as well as variable M for St. Matthew Island blue king crab [Vining and Zheng 2004], and M of 0.28 for males and 0.3 for females for Pribilof blue king crab [Siddeek 2004]).

We simplified the CLA model by applying handling mortality to only sublegal male lengths <136 mm CL. However, undesirable legal-sized males are also discarded in the fishery. The model can be improved by specifying separate retained and discarded crab selectivity functions. However, we did not consider two selectivity curves in the current analysis to reduce the number of parameters to be estimated using limited data sets.

Apart from the β estimate, other input parameter values were educated guesses. Further investigations are required to confirm M and β values. In the golden king crab fishery, many sublegal males and females are caught and thrown overboard (Blau et al. 1996, Neufeld and Barnard 2003) and the effect of handling mortality on population abundance is largely unknown. The estimated selectivity curve was not steep near the minimum legal size with a small slope (0.025) and a smaller 50% selectivity length (125 mm CL) than the minimum legal size (136 mm CL), which explained the retention of a significant number of sublegal males in the pot fishery. A laboratory and field investigation similar to that conducted for snow crab (Warrenchuk and Shirley 2002) is suggested for handling mortality rate estimation. Pot selectivity experiments are needed to determine selectivity parameters. We restricted our investigation to a sub-region with the assumption that estimated recruits came from the same exploited stock through growth and mortality. However, there is a possibility that additional recruitment can occur as a result of immigration from neighboring stocks; our modeling analysis did not consider this possibility. Analysis of historical tagging data and, if necessary, independent large scale tagging experiments on sublegal male crabs are required to address this problem.

The estimates of recruit and legal male abundances, and harvest rate from this study, should be treated cautiously because of the uncertainty in a number of the biological parameters and the short time history of data used in fitting the model. However, the modeling procedure demonstrated the utility of observer data in stock assessment when no annual stock abundance surveys are conducted.

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Using Reproductive Value to Evaluate Surplus Production Models for Sharks

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Abstract

The data limitations common to shark populations have led to the use of surplus production models for the stock assessment of these species. This is despite their generally long life spans and late ages of maturity, traits that suggest their dynamics could be better described by age-structured models. Identification of factors that influence the fit of surplus production models will improve understanding of the limitations and the appropriate uses of these models, and can guide the allocation of resources for data collection. The correspondence between the functions for weight and for reproductive value is proposed as such a factor. This proposition is tested by a simulation procedure that fit a Schaefer model to biomass indices generated from a density-dependent, age-structured matrix model. Greater correspondence between weight and reproductive value is found to reduce the bias and variability of the Schaefer model estimates of virgin biomass under the condition that the biomass index is a measure of all ages in the population. A generalized linear model applied to the simulation results is used to compare the influence of several factors on the fit of the Schaefer model. The use of surplus production models to describe the dynamics of shark populations is discussed in general.

Introduction

The data available for many shark species are insufficient for stock assessment models structured by age, stage, or length. Age-determining methods have been validated for only a few species and sampling effort is often low. The data required for length or stage-based models are more common, but for many species available data are limited to catch and an index of abundance. Recent assessments of shark populations have used complex models (Punt and Walker 1998, Cortés 1999, Simpfendorfer et al. 2000, and others discussed in Cortés 2004) but data remain scarce for most shark fisheries. This data-poor situation has led to new uses of the Schaefer (1954) model in stock assessments for sharks (McAllister et al. 2001, Cortés 2002b). Although these recent applications are Bayesian analyses, there remains a concern that the long life span and late age of maturity common to many sharks can give rise to population dynamics that may not be well approximated by the Schaefer model. A better understanding of the limitations of this model can guide its use or inspire the use of alternative models. The objective of this study is to investigate a potential indicator of the bias and variability of estimates of population parameters from the Schaefer model when it is fit to a biomass index from an age-structured model.

The basis of surplus production models is the assumption that population growth is a specific function of biomass. However, in a population with a late age of maturity, the changes in biomass are highly dependent on the age-distribution of the population. Delay-difference models (Quinn and Deriso 1999) take into account the lag between birth and maturity but require the specification or estimation of an age of recruitment.

One quantity more closely linked to population growth than biomass is reproductive value v_x, which is a measure of the contribution of individuals of different ages to long-term population change (Caswell 2001). This quantity, developed by Fisher (1930) as a measure of the genetic contribution of individuals to a population in continuous models, can be calculated from a discrete life-table or Leslie matrix. For an individual of age x, v_x is the expected number of future offspring, discounted by the rate of change of the population. Reproductive potential, a related measure, is defined here as the sum of all individuals in the population weighted by their reproductive value. This function has also been called total reproductive value (Charlesworth 1980; Leaman 1987, 1991). One property of reproductive potential that suggests some utility in measuring a population's growth is its exponential increase over time in the absence of density-dependence. This exponential change is independent of the age distribution of the population (Fisher 1930, Leslie 1948). In contrast, biomass and population size exhibit damped oscillations as the age distribution stabilizes. However, if a species has a high correspondence between the reproductive value and the weight at each age, then reproductive potential and biomass will be measuring similar quantities and change in similar ways. In this case, biomass may be a good measure of the potential for population growth and surplus production models may more accurately describe the dynamics of that species. These patterns lead to the following proposition, which we evaluate in this paper:

> Proposition: a high correspondence between the reproductive value and the average weight at each age of a given species increases the ability of a Schaefer surplus production model to simulate the biomass dynamics of that species.

We examine this proposition both through analytical methods and the use of simulations. The results support the proposition under certain conditions, and thus demonstrate a way in which knowledge of patterns in growth and maturity can guide the choice of modeling methods for data-poor species.

Methods

Overview

A discrete Schaefer surplus production model was fit to a relative biomass index generated from an age-structured model. The biomass index is a time series proportional to annual total biomass values generated from a deterministic, density-dependent, age-structured matrix model. This fitting process was repeated 500 times for each of seven shark species. The survival and fecundity estimates in the projection matrix for each species were based on published demographic variables (Cortés 2002a).

Fishing pressure was simulated by generating a stochastic time series of annual removals from the age-structured population. The average annual catch and age of entry into the fishery were stochastic variables, as was the length of the time series of biomass values. The stochastic variables allowed the creation a large set of biomass indices representing a broad range of fishing scenarios.

For each biomass index, maximum likelihood estimates (MLEs) of the Schaefer model parameters for virgin biomass B_0, maximum population growth rate r, and catchability coefficient q, were computed. The ability of the Schaefer model to replicate the time-series of biomass was evaluated by comparing the distribution of the MLE for B_0 to the true value used in the age-structured model. The statistical properties of this MLE distribution were compared to a mean square difference measure of the correspondence between reproductive value and the von Bertalanffy (LVB) value for weight at each age. The degree to which species with high correspondence between reproductive value and weight also had lower bias and variability in the distribution of parameter estimates was used to evaluate the proposition above.

Leslie matrices

For each species, a Leslie matrix $\mathbf{M}_t$ was formulated as a function of B_t, the biomass in year t. Demographic parameters for survival, fecundity, age of maturity, and maximum age were taken from Cortés (2002a). In cases where Cortés presented a normal or uniform distribution for a parameter,

Table 1. Demographic and growth parameters used for each species.

Scientific name	Common name	Survival	Max. age	Maturity[a]	m_x[b]	K	t_0
Carcharhinus brachyurus	Bronze whaler shark	0.93	30	19.5	15	0.038	–3.477
Lamna nasus	Porbeagle shark	0.87	22	14	4	0.061	–5.90
Mustelus californicus	Grey smoothhound shark	0.71	9	2.5	3.5	0.22	–1.032
Rhizoprionodon taylori	Australian sharpnose shark	0.56	7	1	[c]	1.013	–0.455
Sphyrna lewini	Hammerhead shark	0.87	17	15	35	0.073	–2.2
Squalus acanthias	Spiny dogfish	0.95	80	35.5	[d]	0.0437	–3.557
Triakis semifasciata	Leopard shark	0.87	30	17	[e]	0.073	–2.74

[a] Age of maturity.

[b] Mean number of pups per mature female per year.

[c] $m_x = 0.19\ length(x) - 7.919$.

[d] $m_x = 0.2\ length(x) - 13.24$.

[e] $m_x = 22.64 - (7592)(0.4208)^{weight(x)}$.

the mean of that distribution was chosen for this analysis. Where Cortés presented a triangular distribution, the modal value was chosen. Parameters used in the LVB growth function were chosen from the following sources: *Carcharhinus brachyurus* from Walter and Ebert (1991), *Lamna nasus* from Natanson et al. (2002), *Mustelus californicus* from Yudin and Cailliet (1990), *Rhizoprionodon taylori* from Simpfendorfer (1999), *Sphyrna lewini* from Branstetter (1987), *Squalus acanthias* from Saunders and McFarlane (1993), and *Triakis semifasciata* from Cailliet (1992). Additional fecundity information for *Rhizoprionodon taylori*, *Squalus acanthias*, and *Triakis semifasciata* was drawn from Simpfendorfer (1999), Ketchen (1972), and Cailliet (1992), respectively. All chosen parameters are shown in Table 1.

Three forms of density-dependence were considered: density-dependent changes in survival, in fecundity, or in both survival and fecundity. In each case, the simplifying assumption was made that the survival and fecundity estimates from the literature were maximum levels, corresponding to minimum biomass. This assumption may lead to an overall underestimation of the productivity of all species, but estimation of the level of depletion to which the survival and fecundity estimates correspond was beyond the scope of this paper. The results should therefore not be directly applied in the management of any specific species. Density-dependence was implemented through an exponential reduction in survival and/or fecundity at higher biomass.

The annual growth of a population was approximated by λ, the dominant eigenvalue of the Leslie matrix. This quantity is frequently used as a measure of population growth in matrix models (Caswell 2001, Frisk et al. 2002, and Cortés 2002a) because it represents the annual multiplicative change in population size when the population is at a stable age distribution (SAD). In the density-dependent model used here, the dominant eigenvalue will be a function of the biomass in each year: $\lambda(B_t)$. Without the influence of fishing, the model has two steady states: virgin biomass $B_t = B_0$, and extinction $B_t = 0$. At virgin biomass $\lambda(B_0) = 1$, while the limit as the biomass approaches 0 is $\lambda(0) = \lambda_{max}$, where λ_{max} is the maximum annual increase in the population (at the appropriate SAD). The corresponding instantaneous rate of population growth is calculated as

$$r = \log(\lambda_{max}). \tag{1}$$

The Leslie matrix used to project the population from year t to year $t + 1$ in this analysis has the form:

$$\mathbf{M}_t = \begin{pmatrix} f_0 e^{-gB_t} & f_1 e^{-gB_t} & \cdots & f_{n-1} e^{-gB_t} & f_n e^{-gB_t} \\ s e^{-hB_t} & 0 & & 0 & 0 \\ 0 & s e^{-hB_t} & & 0 & 0 \\ \vdots & & \ddots & & \\ 0 & 0 & & s e^{-hB_t} & 0 \end{pmatrix}. \tag{2}$$

where s is the survival probability for all ages (Table 1). The fecundity is given by $f_x = sm_x$ where m_x is the average number of female pups per female of age x. The m_x are either constant, or increase with length or weight depending on the species (Table 1). The abundance index is assumed to be post-breeding (Caswell 2001) so the first age class is age 0 individuals that have just been born. The parameters g and h control the density-dependence according to one of the following three forms:

a. Density-dependent survival: $g = 0$ and h is set using an iterative search so that at $\lambda(B_0) = 1$.

b. Density-dependent fecundity: $h = 0$ and g is set using an iterative search so that at $\lambda(B_0) = 1$.

c. Both survival and fecundity are density-dependent:

$$g = h = \frac{-\log(1/\lambda_{max})}{B_0},$$

> which leads to $\lambda(B_0) = 1$. In this last case, λ_{max} is initially computed at $B_t = 0$ when the exponential terms in the Leslie matrix drop out.

This formulation of density-dependence is a special case of the general model of Liu and Cohen (1987).

Under the assumption that the catch is removed before natural mortality or reproduction, the projection equation takes the form,

$$\mathbf{N}_{t+1} = \mathbf{M}(B_t)\,[\mathbf{N}_t - \mathbf{C}_t] \tag{3}$$

where $\mathbf{M}$ is the projection matrix, which is a function of the biomass in year t, $\mathbf{N}_t$ is the vector of numbers in the population at each age in year t, and $\mathbf{C}_t$ is the vector of the catch at each age in year t.

The initial population vector of numbers-at-age in each simulation experiment was in a SAD corresponding to the matrix $\mathbf{M}(B_0)$. The projection equation (3) was used to project this population forward 50 years in each simulation. Projections with no catch were used to confirm that the initial population for each species was at equilibrium.

Weight and reproductive value

Weight was assumed to follow the LVB model (Gallucci and Quinn 1979),

$$w_x = w_\infty(1 - e^{-K(x-t_0)})^b \ , \tag{4}$$

where the exponent b was assumed to be 3 in all cases except *Triakis semifasciata* where it was 3.05 (Cailliet 1992). The asymptotic weights w_∞ are not reported because the virgin biomass used for all projections was arbitrarily chosen, making weight at age only meaningful as a relative quantity. The biomass of the population in year t is given by

$$B_t = \sum_{x=0}^{max.\,age} w_x N_{x,t} \ , \tag{5}$$

where $N_{x,t}$ is the number of individuals at age x in year t and *max. age* is the maximum age of the species from Table 1.

The vector of reproductive values $\mathbf{v}$ was computed as the left eigenvector of the Leslie matrix (Caswell 2001). This is the vector that satisfies:

$$\mathbf{v}^{\mathbf{T}}\mathbf{M} = \lambda\mathbf{v}^{\mathbf{T}}, \tag{6}$$

where $\mathbf{v}^{\mathbf{T}}$ denotes the transpose of $\mathbf{v}$ and λ is the dominant eigenvalue corresponding to $\mathbf{M}$. In order to have the vector of reproductive values

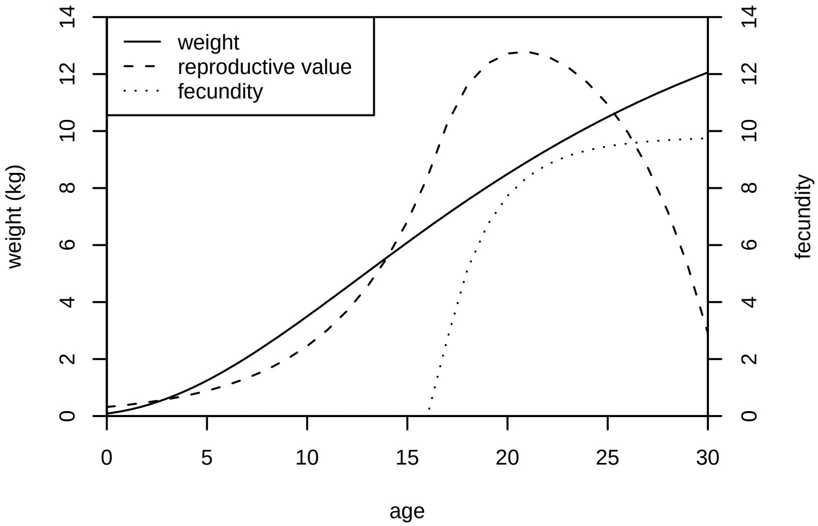

Figure 1. Functions for weight, reproductive value, and fecundity for *Triakis semifasciata*. Reproductive value has been scaled to have the same mean as weight. Fecundity values are the f_x used in the first row of the Leslie matrix (2).

independent of the form of density-dependence, **v** was computed when survival and fecundity were at the assumed maximum values (at $B_t = 0$). Any scalar multiple of **v** will satisfy the eigenvector-eigenvalue relationship (6), so **v** was scaled to make the mean reproductive value across all ages equal to the mean weight across all ages.

Reproductive value differs significantly from fecundity because it takes into account expected contributions at future ages (Charlesworth 1980, Caswell 2001) rather than measuring only reproductive output at the current age. Thus, even if fecundity were proportional to weight, reproductive value and weight would still differ. Figure 1 shows the values for weight, reproductive value and fecundity at each age for *Triakis semifasciata*. Reproductive value increases steadily as individuals approach the age of maturity, reflecting the increasing probability of surviving to have offspring, while fecundity is 0 prior to the age of maturity. Reproductive value also declines as the maximum age is approached while the fecundity function for this species increases monotonically. The decline in reproductive value is due to the decreasing number of future ages. Adding a plus-group to the Leslie matrix (2) results in a reproductive value that increases monotonically.

Reproductive potential is the sum of the reproductive values of all individuals in the population. P_t is defined here as the reproductive potential of $\mathbf{N}_t$, the population in year t. P^*_t is defined as the reproductive potential of $\mathbf{C}_t$, the catch that is removed from $\mathbf{N}_t$. These two scalar quantities were calculated as the product of a row and a column vector:

$$P_t = \mathbf{v}^{\mathbf{T}}\mathbf{N}_t\,, \tag{7a}$$

$$P^*_t = \mathbf{v}^{\mathbf{T}}\mathbf{C}_t\,. \tag{7b}$$

These two equations are equivalent to the summations,

$$P_t = \sum_{x=0}^{max.\,age} v_x N_{x,t}\ , \tag{8a}$$

$$P_t^* = \sum_{x=0}^{max.\,age} v_x C_{x,t}\ , \tag{8b}$$

where v_x is the reproductive value of individuals at age x, and $C_{x,t}$ is the number of age x individuals removed by the harvest in year t.

The degree of correspondence between the functions for reproductive value and weight was measured using the mean square difference on the log scale,

$$MSD = \frac{1}{max.\,age + 1} \sum_{x=0}^{max.\,age} [\log(v_x) - \log(w_x)]^2\ . \tag{9}$$

The log scale for *MSD* was chosen to reduce the influence of differences occurring at older ages with higher weight and reproductive value relative to differences at younger ages where the two measures are smaller. Low values of *MSD* correspond to high correspondence between the functions for weight and reproductive value.

Changing the function for weight

The effect of the degree of correspondence between weight and reproductive value on the fit of a surplus production model was investigated by comparing the performance of the Schaefer model across species, each of which had a different *MSD*. Additionally, simulations at different values of *MSD* for a single species were compared by adjusting the function for the weight of that species. The LVB weight function for *Triakis semifasciata* was replaced with a new function w^*_x, which was a combination of the “true” LVB value w_x and the reproductive value v_x,

$$w^*_x = (1 - d)w_x + dv_x, \tag{10}$$

where d is a parameter that scales the *MSD*. Thus, for all x, $w^*_x = w_x$ at $d = 0$, and $w^*_x = v_x$ at $d = 1$. When the function for weight was changed using (10), *MSD* was computed with w^*_x instead of w_x in (9). This change in the weight function does not have biological meaning, but it allowed a direct analysis of the effect of the correspondence between weight and reproductive value on the distribution of population parameter estimates from the Schaefer model.

Biomass indices

Two types of indices of biomass were considered, one proportional to the biomass of all ages in the population (like a small mesh trawl survey), and the other proportional to the biomass of the ages selected into the fishery (like a catch per unit effort index). Thus,

$$I_t = \begin{cases} q\sum_x w_x N_{x,t} & \text{for index of all ages} \quad (11a) \\ q\sum_x S_x w_x N_{x,t} & \text{for index of selected ages} \quad (11b) \end{cases}$$

where q is the catchability coefficient (arbitrarily set to 0.001 in the simulations), S_x = fraction of age x selected into the fishery, w_x is the average weight of individuals at age x, and $N_{t,x}$ is the number at age x in year t. It is acknowledged that in general, biomass indices include some type of selectivity function, but it is not necessarily always equivalent to that of the fishery.

No observation error was added to this index. The difference between Schaefer model estimates and the parameters used in the age-structured model are therefore the result of model differences alone, rather than sampling error. Under this assumption of “perfect data” the resulting Schaefer model estimates are the best possible fit to the biomass dynamics of the age-structured model.

Stochastic elements in projections

The biomass index (11a) or (11b) for each simulation was computed over a subset of the 50 year projection. All indices included the final year, but the initial year of the index t_{start} was drawn randomly from the first 40 years, with all years given equal probability. All years from t_{start} to 50 were included in the biomass index. Thus, the longest biomass indices were 50 years, while the shortest covered only the final 10 years of the projection.

For simplicity, selectivity was assumed knife-edged, which could be described by a single stochastic parameter, the age of entry into the fishery. The age of entry was uniformly distributed, where the lower limit was age 0 and the upper limit was the age corresponding to two-thirds of the asymptotic length in the LVB growth function. The upper limit was chosen to allow fisheries that targeted only adults, but to avoid the simulation of fishing patterns where only a tiny fraction of the population was selected by the fishery. This is a simplification used in the absence of more detailed data required for the use of the gamma distribution to represent stochastic age of entry (Sullivan et al. 1990).

The average catch C_{mean} was drawn from a uniform distribution between 0 and the value that led to a maximum reduction of the biomass selected by the fishery to 1% of the initial level. The total catch in each year was calculated as lognormal (CV = 0.3) deviations around one of four functions defined over the interval $1 \le t \le 50$:

(a) constant, $C'_t = 1$;

(b) increasing, $C'_t = \frac{2t}{51}$;

(c) decreasing, $C'_t = 2\left(1 - \frac{t}{51}\right)$;

(d) increasing then decreasing, $C'_t = \frac{2t}{26}$ for $1 \le t \le 25$, and $C'_t = 2\left(1 - \frac{t-25}{26}\right)$ for $25 < t \le 50$.

Each of these four functions was chosen to make the mean of the equal to 1.

The total catch in each year was computed as

$$C_t = C_{mean} C'_t \varepsilon_t \Big/ \sum_{i=1}^{50} \varepsilon_i, \tag{12}$$

where ε_t is a lognormal random variable with mean 1 and CV = 0.3. In each projection one of the four functions (a)-(d) was chosen at random with equal probability. Figure 2 shows an example of a series of total catch for each function. The total catch C_t given by (12) is used to form a vector $\mathbf{C}_t$ of the numbers removed from each age in (3) by having an equal fraction removed from each age selected into the fishery,

$$C_{x,t} = h_t N_{x,t}, \tag{13}$$

where h_t is the fraction harvested from each selected age.

Fitting the Schaefer model

The discrete Schaefer model,

$$B_{t+1} = B_t + rB_t(1 - B_t/B_0) - C_t, \tag{14}$$

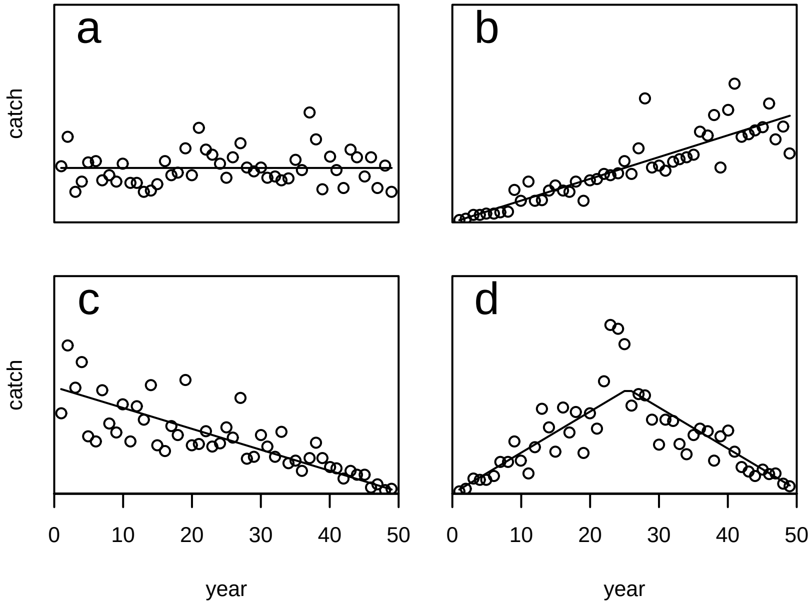

Figure 2. Examples of annual total catch (circles) shown with expected trend (line) for each of the four forms of catch series: (a) constant, (b) increasing, (c) decreasing, and (d) increasing then decreasing.

(Quinn and Deriso 1999) was fit to the biomass index from the age-structured model by solving for the values of B_0, r, and q that maximized the likelihood,

$$L(B_0,r,q)=\sum_{t=t_{start}}^{50}[\log(I_t)-\log(qB_t)]^2, \tag{15}$$

where I_t is the biomass index from (11a) or (11b) for year t, B_t is the Schaefer model estimate of biomass in year t from (14), and q is the catchability coefficient. The difference between the observed and expected quantities was assumed to have a lognormal distribution (Punt and Walker 1998) in this likelihood. The maximum likelihood for q was estimated analytically by solving the partial derivative of (15),

$$\frac{\partial L(B_0,r,q)}{\partial q}=0\text{, to get}$$

$$\log(\hat{q}) = \frac{1}{51 - t_{start}} \sum_{t=t_{start}}^{50} \log(I_y / B_y)\,. \tag{16}$$

MLE values $\hat{B}_0$ and $\hat{r}$ were found using the Nelder-Mead simplex algorithm implemented through the "optim" function in the software package R (R Development Core Team 2004). Four initial starting values were used in the search: (a) both B_0 and r at the true values from the age-structured model, (b) B_0 at double the true value and r at half the true value, (c) B_0 at half the true value and r at double the true value, (d) both B_0 and r at double the true value. The true values of B_0 and r from the age-structured model are unlikely to be the MLEs for the Schaefer model, but are used to get the estimation into a range of parameter space that may be close to the MLEs. From the four solutions to the optimization algorithm, the parameters associated with the highest likelihood were taken to be the MLEs. Simulations that resulted in estimates of r below 0 or greater than 1, a range assumed to include all biologically realistic values for sharks, were considered outliers and not included in the results.

A comparison between the estimate of virgin biomass and the true value was used as the primary measure of fit of the surplus production model. Virgin biomass is less important from a management standpoint than the estimate of maximum sustainable yield (MSY) or the corresponding fishing mortality, but in an age-structured model these fishery based measures depended on the selectivity of the fishery, which was a stochastic parameter in this analysis. The maximum growth rate r differs between species, making comparisons of bias and variability difficult. Virgin biomass was constant across all simulations so a comparison between the true and estimated values of this parameter revealed problems with the fit of the Schaefer model most clearly.

Comparing the influence of different variables on model fit

A generalized linear model (GLM) was used to estimate the importance of all the stochastic variables used in the simulations on the fit of the Schaefer model. The response variable y used to describe fit of the Schaefer model was the presence/absence of each estimate of virgin biomass within 10% of the true value:

$$Y_i = \begin{cases} 1 & \text{if } \left|\hat{B}_0 / B_0 - 1\right| < 0.1 \\ 0 & \text{otherwise.} \end{cases} \tag{17}$$

This variable was treated as the response in a binomial regression. The binomial GLM assumed that the probability that the Schaefer model estimate fell within 10% of the true value was a linear function of the regression covariates after a logit transformation. The regression covariates included the following: duration of the biomass index, age of entry into the fishery as a fraction of maximum age, depletion at the end of the projection (final biomass as a fraction of B_0), mean catch as a fraction of B_0, the type of catch trend (constant, increasing, etc.), as well as *MSD*, the measure of correspondence between weight and reproductive value. The catch trend was treated as a categorical variable, while all other variables were treated as continuous in the regression. Analysis of deviance (type II) was used to determine which of these stochastic variables were most influential in the fraction of MLEs falling within 10% of the true value.

Results

Analytical results

The changes over time in reproductive potential in a density-independent age-dependent model can be described by an age-independent model:

$P_{t+1} = \mathbf{v}^{\mathbf{T}}\mathbf{N}_{t+1}$ definition of reproductive potential (7a)

$P_{t+1} = \mathbf{v}^{\mathbf{T}}\mathbf{M}[\mathbf{N}_t - \mathbf{C}_t]$ replacing $\mathbf{N}_{t+1}$ using (3)

$P_{t+1} = \lambda\mathbf{v}^{\mathbf{T}}[\mathbf{N}_t - \mathbf{C}_t]$ using the eigenvector-eigenvalue relationship (6)

$P_{t+1} = \lambda\mathbf{v}^{\mathbf{T}}\mathbf{N}_t - \lambda\mathbf{v}^{\mathbf{T}}\mathbf{C}_t$ distributing

$P_{t+1} = \lambda[P_t - P^*_t]$ definitions of P_t and P^*_t (7a) and (7b) (18)

Equation 18 is an exponential, density and age-independent function relating the reproductive potential in year t+1 to that in year t after removals by the fishery. This is an extension of the exponential growth observed by Fisher (1930) and Leslie (1948) to include catch.

Another characteristic of reproductive potential is its formal correspondence with the functions for biomass, viz. (8a) corresponds to (5). Thus, B_t and P_t are both weighted sums of the age structured population, and are scalar quantities that fluctuate over time. How closely changes in their dynamics resemble each other depends on the degree of correspondence between weight w_x and reproductive value v_x. Thus, (18) may also describe the change in biomass over time in the absence of density-dependence, still depending on the degree of correspondence between w_x and v_x.

When density-dependence is added to the age-structured model, changes in reproductive potential no longer follow (18). In fact, whereas the time series of biomass from a density-dependent age-structured projection (with no harvest) can have large oscillations (dotted curve, Fig. 3), the time series of reproductive potential from the same projection can be a much smoother, logistic-like curve. The plausibility of logistic-like shape was evaluated by fitting a logistic model (Schaefer, with no harvest) to the

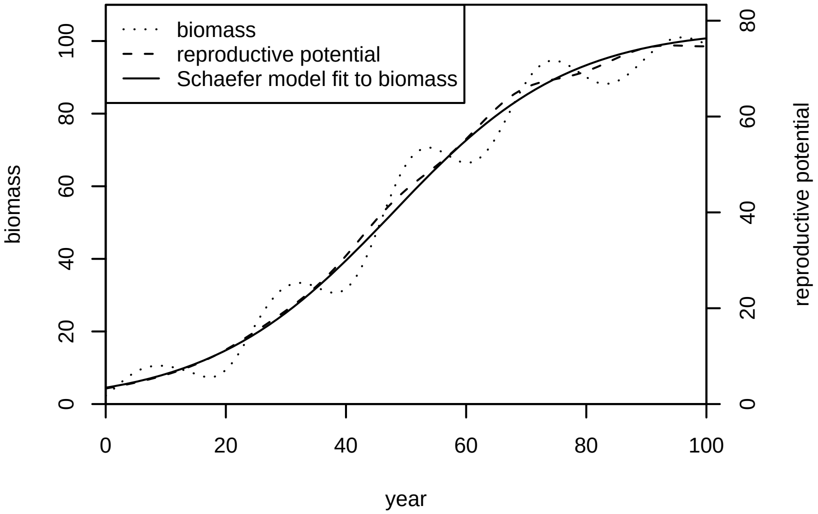

Figure 3. Plots of biomass and reproductive potential in a 100-year projection of the density-dependent age-structured model, along with the Schaefer model fit to the biomass time-series. The initial population in the age-structured projection is a single depleted cohort of *Triakis semifasciata* at age 0. Both survival and fecundity are density-dependent.

biomass values, resulting in the smooth solid curve that closely follows the dashed curve for reproductive potential in Fig. 3. Thus, the Schaefer model may describe changes in the time-series of reproductive potential (8a) better than it describes changes in the time-series of biomass (5), at least for a long-lived, late-maturing species of shark. This suggests that while the Schaefer model is traditionally applied to biomass or numbers rather than reproductive potential, it might be interpreted as a density-dependent extension of the reproductive potential function (18).

However, applying the Schaefer model to a time-series of reproductive potential is not necessarily useful because the computation of reproductive potential (8a) requires knowledge of the age-distribution $\mathbf{N}_t$ in each year. Knowledge of the age-specific time-series of abundances, $\mathbf{N}_t$, would be sufficient to fit a more complex, age-structured model, but for shark populations such data are rare. Reasoning by analogy, the Schaefer model can describe the biomass dynamics of an age-structured population if it can describe changes in reproductive potential and if the correspondence between weight and reproductive value is high. The Schaefer model could be used with more success, if such reasoning holds, on some of the biomass indices that are often one of the few sources of information on the dynamics of many data-poor shark species.

Since density-dependence complicates the analytical relationships for P_t derived above in (18), simulations were used to investigate the influence of correspondence between weight w_x and reproductive value v_x on the fit of the Schaefer model to biomass indices and thus evaluate the proposition in the introduction.

Simulation results when changing the function for weight

Fitting the Schaefer model (14) to biomass indices from the age-structured model (3) indicated that increasing the correspondence between weight and reproductive value using (10) improved the estimation of population parameters, but only when all ages are included in the biomass index, i.e. (11a) is used rather than (11b), and survival is density-dependent. Figure 4 shows boxplots of the ratio of the Schaefer model estimate of virgin biomass over the true value used in the age-structured model, $\hat{B}_0/B_0$, at three values of the parameter d in (10), which changes the function for weight. The first three boxplots ($d = 0$) are based on simulations with weight given by the LVB growth function, the last three boxplots ($d = 1$) are based on weight equal to reproductive value, and the three in the middle ($d = 0.5$) have the weight at each age equal to the mean of the LVB function and the reproductive value. Table 2 shows the fraction of $\hat{B}_0$ within 10% of the true value (16) for each value of d, type of biomass index, and form of density-dependence.

The variability in the estimates of virgin biomass $\hat{B}_0$ from the surplus production model decreased as d was changed to decrease *MSD* when the biomass index was based on all ages in the population (11a) and survival was density-dependent (Fig. 4a). For the two cases that met these conditions, the fraction of within 10% of the true value went from 0.42 to 1 as the correspondence between weight and reproductive value was increased (Table 2). However, when the biomass index included selectivity (11b) and (Fig. 4b) or when fecundity was the only density-dependent parameter, little change in the distribution of $\hat{B}_0$ resulted from changes in d. A second notable difference between the results for the two biomass indices is that the distributions of $\hat{B}_0$ when selectivity was included in the biomass index (Fig. 4b) have higher variability and a consistent negative bias. In general, the results of simulations in which the function for weight was changed using (10) supported the proposition (repeated below) under the conditions that the biomass index included all ages in the population and survival was density-dependent.

> Proposition: a high correspondence between the reproductive value and the average weight at each age of a given species increases the ability of a Schaefer surplus production model to simulate the biomass dynamics of that species.

Table 2. Results of simulation tests for *Triakis semifasciata* when the weight function is changed.

d parameter in function for weight (10)	*MSD*	Fraction of $\hat{B}_0$ within 10% of true value					
		Index of whole population (11a)			Index of selected population (11b)		
		d.d. survival[a]	d.d. fecundity	d.d. survival & fecundity	d.d. survival	d.d. fecundity	d.d. survival & fecundity
0	0.16	0.42	0.64	0.42	0.09	0.03	0.07
0.5	0.05	0.62	0.65	0.63	0.08	0.02	0.07
1	0	1.00	0.56	1.00	0.06	0.03	0.08

[a] d.d. = density dependent.

Simulation results comparing species

Comparing the variability of $\hat{B}_0$ across the 7 shark species revealed strong inter-species differences in the bias and variability of the parameter estimates from the Schaefer model, but again supported the proposition under certain conditions. Table 3 gives the mean square difference (*MSD*) from (8) used to measure the degree of correspondence between weight and reproductive value for each species. Table 3 also shows the fraction of $\hat{B}_0$ within 10% of B_0 (16) for each species, type of biomass index, and form of density-dependence. Figure 5 shows boxplots of $\hat{B}_0/B_0$ for the seven species ordered from left to right by increasing correspondence between weight and reproductive value as measured by decreasing *MSD*. Again, the cases of a biomass index based on all ages (Fig. 5a) and on selected ages (Fig. 5b) are both considered.

Differences in life span and other demographic traits confound the effort to separate the effect of the correspondence between weight and reproductive value from other factors that might influence the fit of the Schaefer model. However, when the biomass index is based on all ages (11a), a noticeable effect remains. The porbeagle *Lamna nasus* is the species with the lowest *MSD* and also has the highest fraction of $\hat{B}_0$ within 10% of B_0. The two species with the highest *MSD*, *Mustelus californicus* and *Sphyrna lewini*, showed a strong upward bias and had the lowest fraction of $\hat{B}_0$ within 10% of B_0 when the biomass index included all ages (11a). This pattern of lower variability in $\hat{B}_0$ associated with lower *MSD* holds for both biomass indices, but was stronger when the biomass index included all ages (11a). All species had a lower median estimate of B_0 when the selectivity was included in the biomass index (11b). Thus, the two species with a strong upward bias for the index of all ages, *Mustelus californicus* and *Rhizoprionodon taylori*, which are both short lived, were the only species that had less bias when selectivity was in the biomass index (Fig. 5b). In general, as in the analysis with changing weight, the

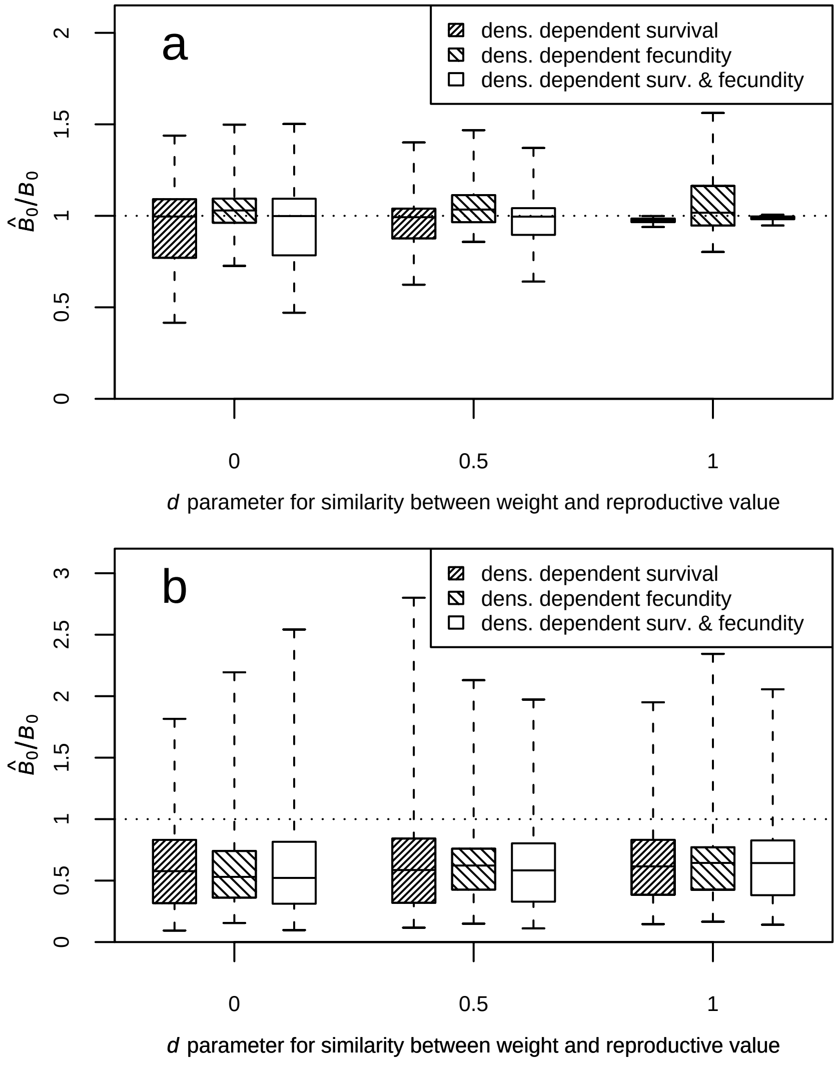

Figure 4. Boxplots of the estimates of virgin biomass from Schaefer model as a fraction of the true value used in the age-structured model, $\hat{B}_0/B_0$, for a series of weight functions for *Triakis semifasciata*. The function for weight was shifted between the true von Bertalanffy weight and the function for reproductive value by the parameter *d* in (10). The upper boxplots (a) have the biomass index based on all ages in the population (11a) while the lower (b) has the biomass index based on ages selected by the fishery (11b). Boxplot whiskers extend to 0.025 and 0.975 quantiles.

Table 3. *MSD* and fraction of estimates of virgin biomass within 10% of true value resulting from simulation for seven shark species.

		Fraction of within 10% of true value					
		Index of whole population (11a)			Index of selected population (11b)		
Species	*MSD*	d.d. survival[a]	d.d. fecundity	d.d. survival & fecundity	d.d. survival	d.d. fecundity	d.d. survival & fecundity
Carcharhinus brachyurus	0.13	0.43	0.32	0.44	0.19	0.04	0.13
Lamna nasus	0.03	0.68	0.60	0.72	0.04	0.04	0.04
Mustelus californicus	1.09	0.11	0.20	0.13	0.15	0.13	0.15
Rhizoprionodon taylori	0.42	0.17	0.16	0.14	0.17	0.23	0.16
Sphyrna lewini	0.23	0.46	0.55	0.46	0.17	0.12	0.25
Squalus acanthias	0.20	0.40	0.47	0.42	0.05	0.03	0.05
Triakis semifasciata	0.16	0.42	0.64	0.42	0.09	0.03	0.07

[a] d.d. = density dependent.

simulation results across species indicated that when the biomass index is based on all ages, a greater correspondence between weight and reproductive value, as measured by *MSD*, is associated with better estimates of B_0 in the Schaefer model.

Comparing the influence of different variables on model fit

A generalized linear model (GLM) applied to the simulation results for the seven species showed that *MSD* was a significant factor in explaining differences in the fraction of the Schaefer model estimate of virgin biomass $\hat{B}_0$ falling within 10% of B_0 (16). Analysis of variance (type II) for the GLM indicated that the age of entry into the fishery as a fraction of the maximum age was the most significant factor in determining how often $\hat{B}_0$ fell within 10% of B_0. Biomass indices from simulations with more ages selected into the fishery were better fit by the Schaefer model than those with fewer ages selected into the fishery, regardless of whether selectivity was included in the biomass index. *MSD*, the measure of correspondence between weight and reproductive value, was the second most influential factor when the biomass index was based on all ages. When the biomass index was based only on ages selected into the fishery, *MSD* was still a significant factor, but less so than the age of entry into the fishery, the length of the biomass index, the final level of depletion, and the mean catch.

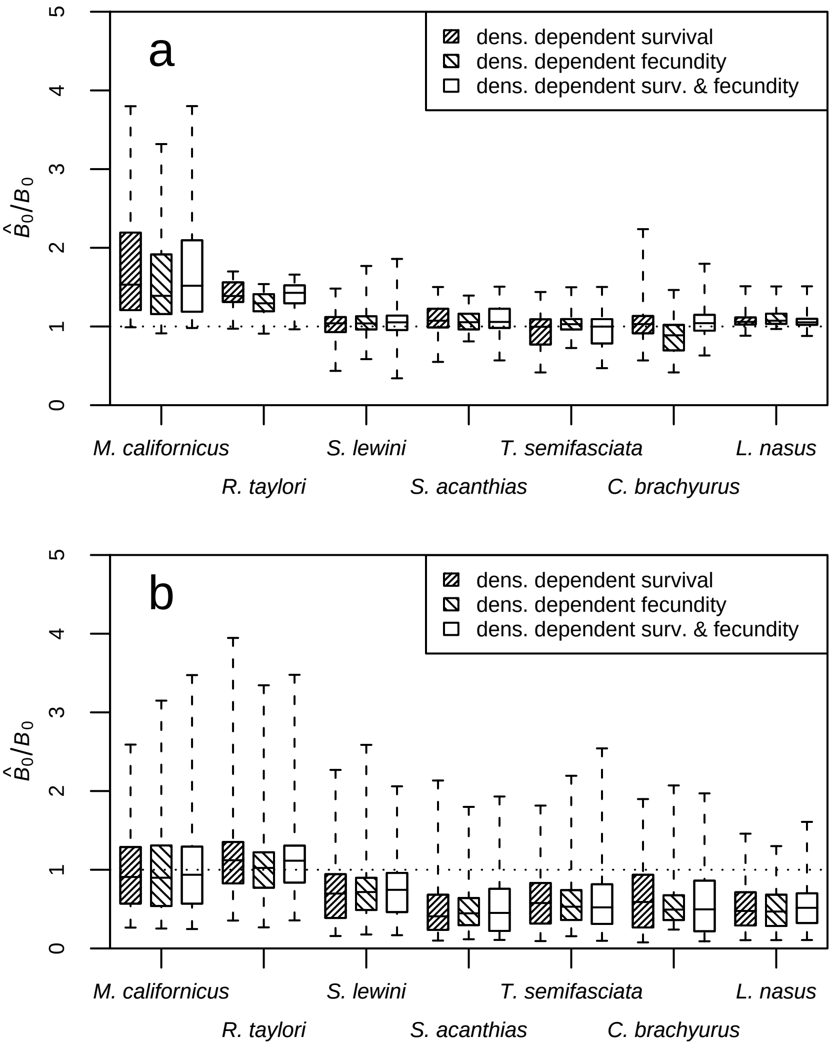

Figure 5. Boxplots of the estimates of virgin biomass from Schaefer model as a fraction of the true value used in the age-structured model, $\hat{B}_0/B_0$, for seven shark species. Species are ordered from left to right by increasing correspondence between weight and reproductive value, as given by the mean square difference between the functions for weight and reproductive value (9). The upper boxplots (a) have the biomass index based on all ages in the population (11a) while the lower boxplots (b) have the biomass index based on ages selected by the fishery (11b). Boxplot whiskers extend to 0.025 and 0.975 quantiles.

Discussion

The proposition presented in the introduction (repeated below) is supported by the results, subject to the conditions that the biomass index is based on all ages in the population and survival is density-dependent.

> Proposition: a high correspondence between the reproductive value and the average weight at each age of a given species increases the ability of a Schaefer surplus production model to simulate the biomass dynamics of that species.

The analytical results show that reproductive potential can be described by an age-aggregated model in the absence of density-dependence. Even when density-dependence is added, the Schaefer model may approximate changes in reproductive potential better than changes in biomass in many cases. The proposition is supported by simulations with density-dependent models, in which changes in the function for weight to make it correspond more closely to reproductive value decreased in variability of estimates of virgin biomass from the Schaefer model.

The variability of the surplus production model estimates was less correlated with the correspondence between weight and reproductive value when selectivity is included in the biomass index (11b), or when fecundity is the only density-dependent parameter. Increases in the biomass index do not immediately follow increases in births when the biomass index depends only on selected ages. Density-dependent changes in fecundity can also take much longer to propagate through the population than changes in survival, especially for long lived sharks such as *Squalus acanthias* which has an age of 50% maturity estimated at 35.5 years (Saunders and McFarlane 1993). The Schaefer model is not able to simulate such delays, and for this reason delay-difference models (Quinn and Deriso 1999) are frequently used as an alternative. Factors that influence the fit of delay-difference models should be the subject of future research. We note that the analyses in this paper could also be extended to include the more general Pella-Tomlinson model, which Maunder (2003) suggested should always be chosen over the Schaefer model.

This study was based on the assumption that survival and fecundity parameters from the literature described populations growing at rates that would only be reduced by density-dependence. Deterministic recruitment was also assumed, but this is probably acceptable because the size at birth of many shark species is believed to make these populations less sensitive to environmental variability compared to teleost fish (Cortés 2004).

This is the first attempt to relate reproductive value and biomass, an association ripe for further investigation. The contrast between the precision and accuracy of estimates of the virgin biomass $\hat{B}_0$ for *Lamna nasus* (Fig. 5 and Table 3), which had a high correspondence between its functions for weight and reproductive value, and the biased and highly variable estimates for the species that had the least correspondence between weight and reproductive value, supports the association.

The bias and variability of estimates of surplus production model parameters has been examined already (Punt 1989, Bonfil 1996), but the impacts on these statistical properties of differences in demographic traits between species were not considered. The two shortest lived species had biased estimates of virgin biomass from the Schaefer model indicating that concern about the fit of surplus production models should not be associated only with long-lived sharks. The GLM indicated that the fraction of the population selected into the fishery is more important in determining the fit of the Schaefer model than the correspondence between weight and reproductive value, but that this correspondence does have some utility in determining the appropriateness of the Schaefer model. Unfortunately, the influence of the association between weight and reproductive value was weakest for the more common, CPUE-like biomass indices that include selectivity.

The Schaefer model parameter estimates are often poor, even without observation error in the model. For most species, less than half of the best fit estimates of virgin biomass fell within the chosen criterion of 10% of the true value, even when the biomass index was based on all ages in the population (Table 3). No case had more than one quarter of the estimates within 10% of the true value when the biomass index included selectivity. This implies that the effect of error in model specification associated with surplus production models can be considerable.

The results of this study indicate that for some species of sharks, the estimates of population parameters from the Schaefer model are not sufficient to successfully manage a population. The a priori determination of which species are appropriate for using this model depends in part on estimation of reproductive value.

Acknowledgments

We are grateful to Randall Peterman for his thorough and insightful review. André Punt, Rick Methot, and the students in the University of Washington Shark Lab provided valuable advice and support. The research of IGT was supported by NOAA grant NA07FE0473. VFG received partial support from NOAA grant NA96FX0124 to the North Pacific Marine Science Foundation.

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Fisheries Assessment and Management in Data-Limited Situations
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Evaluation of Scuba Assessment Methods for Shallow-Water, Nearshore Black Rockfish (*Sebastes melanops*)

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Abstract

Management of black rockfish, *Sebastes melanops,* in Southcentral Alaska currently relies on historical fishery indicators, resulting in uncertainty on the population status. Because of the lack of fishery-independent data, an index to monitor population trends is needed. This study evaluated the utility of using strip transect sampling techniques to estimate black rockfish density within specific study areas. Four scuba transect surveys were conducted during the three-year study. Inter-annual density index estimates were highly variable, with coefficients of variation ranging from 22 to 33%. In addition, between-observer estimates differed by 60%. Responsive movement of black rockfish to the presence of divers was detected during the first survey, prompting the development of a separate experiment designed to further assess responsive movement. During the experiment, black rockfish counts increased significantly with the time divers spent at stationary locations, while there was not a significant between-observer effect. Thus, strip transect sampling for black rockfish will likely yield estimates that are highly variable, not repeatable, and biased high. As an alternative to strip transect sampling, timed counts are recommended for indexing black rockfish abundance because of the less subjective counting method.

Introduction

Black rockfish, *Sebastes melanops,* support commercial, recreational, and subsistence rockfish fisheries by shore-based fleets in Southcentral Alaska (Meyer 2000, Trowbridge et al. 2001, Berceli et al. 2002). However, biological and fishery-independent survey data are limited, and consequently the management of black rockfish in this region relies on historical harvest patterns (Bechtol 1992). While commercially viable quantities are easily located and harvested, the relatively low productivity of black rockfish populations requires that management be cautionary. Without fishery-independent survey data, black rockfish resources may be overfished or, conversely, too conservatively managed, leading to a loss of opportunity for fishermen.

Traditional groundfish survey methods are not always practical for black rockfish due to their patchy distribution, association with rocky substrates, and physoclistic swim bladder, which may cause barotrauma upon rapid depressurization. Therefore, novel survey methods are needed to assess population status.

Underwater visual censuses can be used to estimate fish density when other survey methods are not applicable (Richards et al. 1985). Transect methods using scuba have been used to estimate rockfish, *Sebastes,* densities for temporal and among habitat comparisons along the Pacific west coast (Matthews 1989, Fox et al. 1998, VenTresca et al. 2002) and for regional comparisons within Alaska waters (Rosenthal et al. 1981). Following development by Brock (1954), strip transect sampling has become a preferred method for underwater visual censuses of fish populations (Thresher and Gunn 1986). More recently, stationary point count methods have been developed (Bohnsack and Bannerot 1986, Bortone et al. 1989). Despite widespread use and acceptance, underwater visual census methods have limitations. Studies assessing precision and bias in visual counting methods have emphasized underestimating the number of fish species and individuals due to their cryptic behavior (Brock 1982, Sale and Sharp 1983, Smith 1988, Willis 2001). For larger mobile species, the situation may be quite different. For example, when movement is non-random, density may be overestimated depending on the fishes' direction of approach (Watson et al. 1995).

Strip transect or fixed radius sampling assumes that 100% of all animals are detected within the area sampled. Generally, detections of fish tend to decrease with distance from the observer (Sale and Douglas 1981, Cheal and Thompson 1997) and the rate of decrease can be species- or size-specific (Sale and Sharp 1983, Kulbicki 1998). For example, large mobile species are more likely to be detected at wider sighting distances than small species (Thresher and Gunn 1986). For schooling species, large schools may have a higher detection rate at greater distances than small schools (Kulbicki 1998). Distance sampling methods are superior to strip transect or fixed radius sampling if the detection of animals within the fixed sampling area is less than 100% (Burnham and Anderson 1984). However, distance data is difficult to collect on underwater surveys and only rarely are collected (but see Thresher and Gunn 1986, O'Connell and Carlile 1993, Ensign et al. 1995). Therefore, if strip transect or fixed radius sampling is used, the sighting distance should be carefully chosen to suit the objective (Bohnsack and Bannerot 1986).

Distance data can also be useful for examining fish behavior and detecting responsive movement of fish. Diver presence can alter the natural behavior of fish, causing some species to hide or flee or others to swim toward the observer (Chapman et al. 1974, Brock 1982, Sale and Sharp 1983, Davis and Anderson 1989, Kulbicki 1998). Density will be overestimated if fish move toward the observer prior to detection, or conversely underestimated if fish are repelled by the presence of the observer. Thus, to minimize the bias induced by responsive movement, observers should record animals at the time of first sighting (Buckland et al. 2001).

Observer effects also can contribute to precision and bias of underwater visual assessment methods. Managers can overcome these problems by training observers (Sale and Sharp 1983) and adhering to sampling protocols (Bohnsack and Bannerot 1986). While some observer bias and variation is expected, density estimates may only be meaningful if the contributing bias is low and the variation does not contribute significantly to the comparisons being made.

Commonly, the objective of an underwater visual census is to monitor density or abundance of a fish population over time or for comparison between areas. An estimate of absolute density may be needed for ecological studies, while an index of abundance may be acceptable for management, where the objective is to monitor population change over time. Therefore, even if an assessment method is biased, as long as the bias is consistent and variance is low enough to detect population changes, it can still be a valuable management tool.

The goal of this three-year study was to develop a sampling approach to estimate density of the shallow-water component of the black rockfish population along several sections of the outer Kenai Peninsula in the northern Gulf of Alaska. The objective was to assess the utility of strip transect sampling techniques for estimating black rockfish density. Specifically, the study evaluated the validity of statistical assumptions, investigated between-observer precision and accuracy, and compared the variability between annual density estimates. An experiment to assess the responsive movement of black rockfish to the presence of divers was also performed to further investigate violations in the assumptions of transect sampling.

Materials and methods

Study area

Five surveys were conducted along the north gulf coast of the Kenai Peninsula between June 2001 and May 2003. The initial survey was a pilot study intended to identify specific study areas, refine survey methods, and collect encounter rate and school size data for determining sample

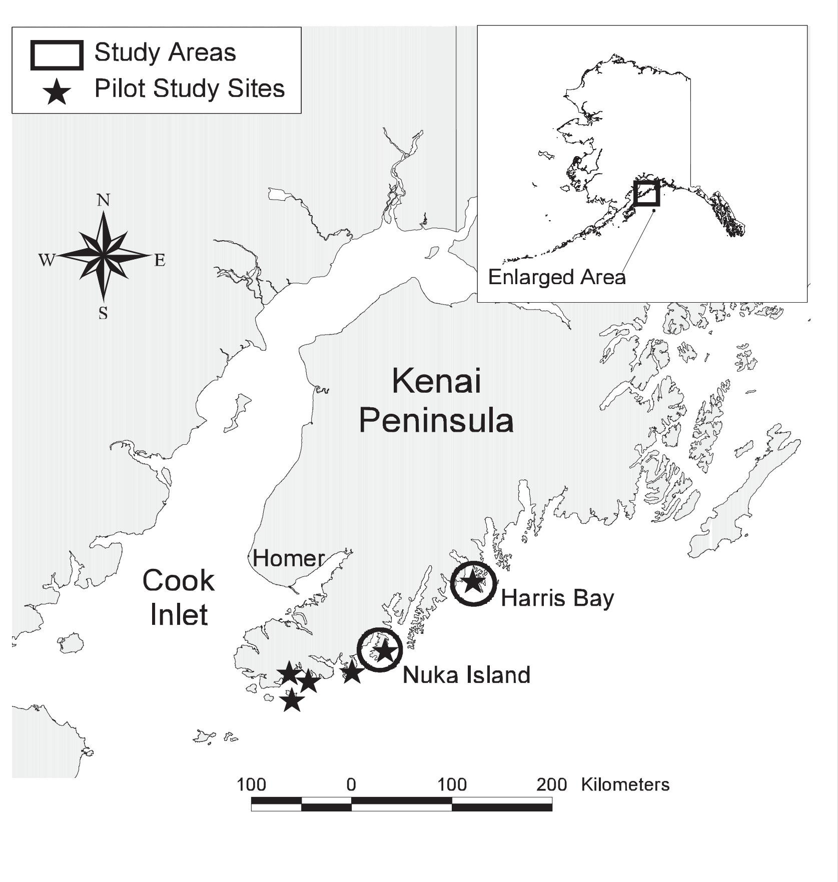

Figure 1. Study areas for the nearshore rockfish assessment surveys.

size (Fig. 1). Eight areas were sampled during the pilot study with two areas, Nuka Island and Harris Bay, chosen for subsequent surveys based on the relative densities of black rockfish encountered and on historical commercial and recreational fishery performance. Both areas are exposed to high-energy waves from the Gulf of Alaska, but also contain both moderately exposed and protected embayments. Substrates in both areas include soft bottom habitats, but are composed primarily of rocky substrates ranging from cobble to high relief basement and boulder.

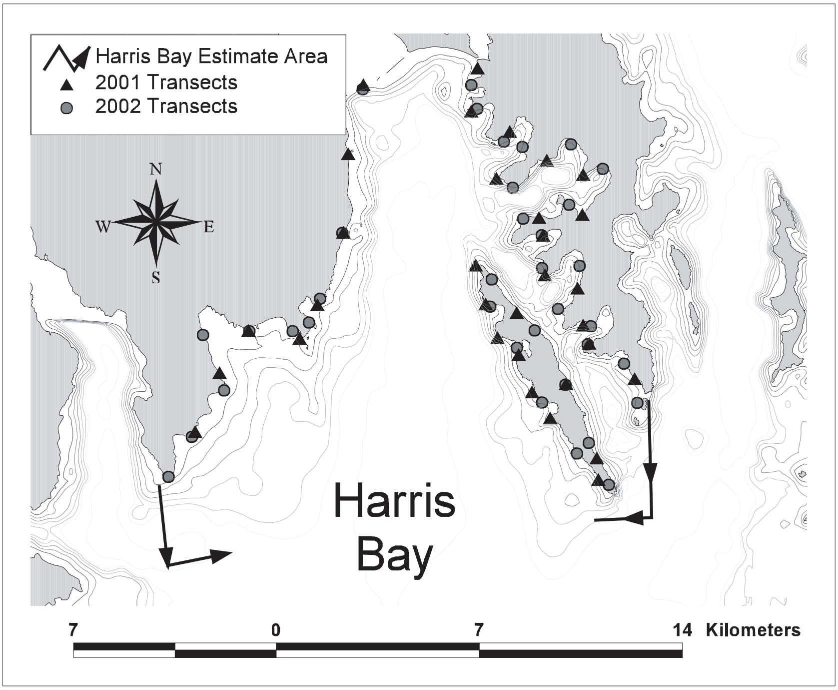

Figure 2. Harris Bay study area with transect sites.

Field procedures

Strip transect surveys

Strip transect surveys were conducted at discrete locations along the entire 49 km of shoreline of Harris Bay in 2001 and 2002, while at Nuka Island approximately 20 km of shoreline were surveyed in 2002 and 9 km in 2003 (Figs. 2 and 3).

Transects were assigned systematically with a random starting point. Single transects oriented perpendicular to the shoreline were traversed at Nuka Island. Due to a steep bottom slope, transects in Harris Bay consisted of outbound and inbound legs oriented 90 degrees to each another and 45 degrees to the shoreline. At each transect, a dive team descended at the intertidal area and swam down the transect line following a set compass course to a terminal depth of approximately 25 m. At Nuka Island, divers ascended to the surface after reaching the terminal depth. At Harris Bay, divers swam the outbound leg to the terminal depth, swam a course parallel to shoreline for a distance that was greater than twice

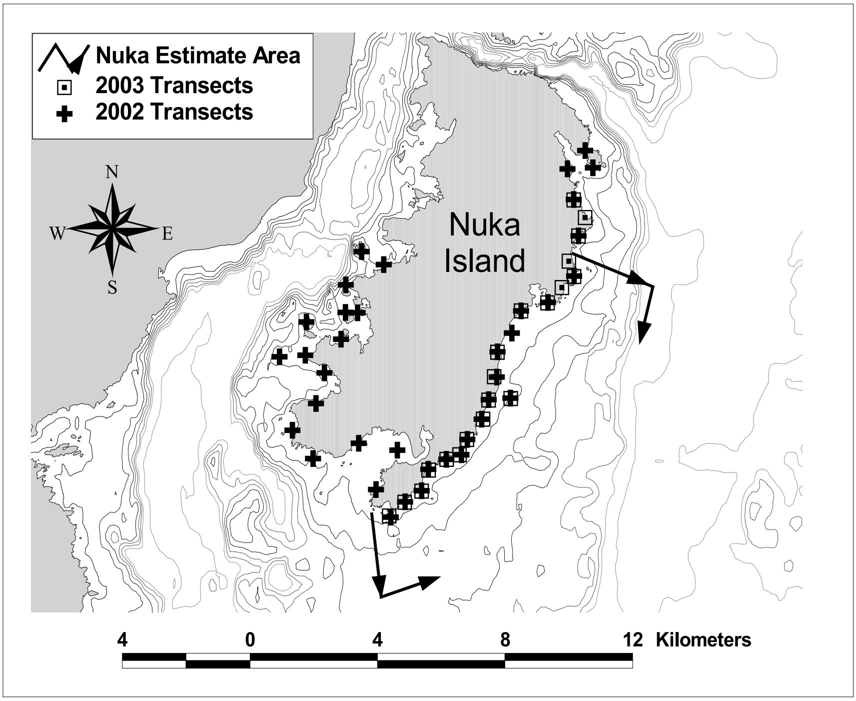

Figure 3. Nuka Island study area with transect sites.

the observed visibility, and then swam the inbound leg to shore. Transect legs at each site in Harris Bay were pooled for the analysis.

Strip transect methods were used for estimating black rockfish density within each survey area, though perpendicular sighting distances were also collected to evaluate detectability and responsive movement. A dive team consisted of a fish observer who recorded all species of fish encountered along the transect line, and a habitat observer who recorded habitat data at quadrats spaced at 10 m intervals (planar distance). To calibrate each diver for estimating the interval distance, divers repeatedly swam a 10 m line laid on the bottom at a constant depth prior to each survey. At each quadrat, the habitat observer recorded primary and secondary vegetation type and percent cover, substrate type within the quadrat, depth (feet), and visibility (nearest meter). For each black rockfish school, the fish observer estimated the number of individuals and the perpendicular distance (nearest meter) from the transect line to the geometric center of the school. To minimize the potential of double counting, only fish from in front of the divers to 90 degrees either side of the dive team were recorded. Care was taken to record fish sightings as soon as individuals or schools came into view. Fish that were judged to have followed or to have been attracted by the divers were not counted. To assess between-observer bias and variance in fish density estimates, both observers recorded fish counts in 2003. To maximize the time the habitat observer spent counting fish, only substrate type was recorded during this survey. The habitat observer's primary task was to enumerate fish and secondarily record the substrate type. Though dark rockfish, *Sebastes ciliatus*, occurred in both study areas, divers were proficient at distinguishing them from black rockfish. Identifying characteristics used to separate the two species were mottling on body sides, ventrum color, and prominence of the symphyseal knob.

Responsive movement experiment

It became evident during the 2001 pilot survey that black rockfish move in response to the presence of divers, thereby possibly introducing bias into density estimation. To address this concern, a responsive movement experiment was conducted following the Nuka Island survey in 2002. Experimental sites were located either at previously sampled transect sites where black rockfish were encountered or at nearby sites where sonar observations of fish schools or rocky bottom structure suggested the presence of black rockfish. At each site, a dive team descended directly to the bottom in 10 to 20 m of water. The lead diver signaled the other diver to start counting and both divers independently counted all individual black rockfish visible in a 360 degree circle. Subsequent counts were made every minute for a total of eight minutes. After the eight-minute period, the dive team ascended and moved to another site. A total of 17 trials were conducted.

Analytical methods

Strip transect surveys

Histograms of the sighting data were plotted for each survey to examine the shape of the distributions and to examine violations of the assumptions of transect sampling. Although line transect estimation was not performed, perpendicular sighting distance data were used to assess whether detections decreased within the strip width and to assess the responsive movement of fish to the presence of divers. Kendall's τ was used to determine if visibility was correlated with the occurrence of black rockfish schools (Sokal and Rohlf 1995). Least-squares regression was used to determine if school size was dependent on distance from the transect line. Because responsive movement was detected and imposed a violation in the assumptions of strip transect sampling, a consistent positive bias was assumed due to the positive responsive movement. In addition, because of the subjective nature of observers determining whether to count fish that may have followed or been attracted to the divers, all density estimates were treated as an index rather than an estimate of true density. Transect width was calculated for each survey as the minimum water visibility that occurred in at least 95% of the aggregate length of all transects for that survey. All black rockfish schools sighted beyond this distance were excluded from the analysis. For estimating the index of black rockfish density for each survey area, let

w = the width of the transect in meters,

k = the number of transects sampled within the survey area,

l_i = the length in meters of the ith transect, $i = 1,\ldots,k$,

n = the number of schools of black rockfish observed, and

s_i = the number of individual black rockfish observed in school n_i.

Black rockfish average school size $\bar{s}$, school density $\hat{D}_s$, and density $\hat{D}$ for each survey area was estimated as:

$$\hat{D} = \hat{D}_s \, \bar{s}$$

$$\hat{D}_s = \left(\frac{n}{2wL} \right)$$

where $L = \sum_{i=1}^{k} l_i$

and $\bar{s} = \sum \frac{s_i}{n_i}$

A bootstrap method was used to estimate variance in density by re-sampling the transects 1,000 times (Buckland et al. 2001). Upper and lower 95% confidence intervals were calculated using the percentile method (Efron 1981). Since the duration of individual surveys were 10 days or less, immigration and emigration of black rockfish in the survey area were assumed negligible. In addition, to account for possible seasonal movements when making inter-annual comparisons, surveys were conducted at approximately the same time each year.

Responsive movement experiment

To examine the effect of fish behavior due to the presence of divers, the number of fish observed was compared to time including site and fish counter as random effects, using a linear mixed effects (LME) model (Pinheiro and Bates 2000). The full model used was

$$y_{ijk} = \beta_0 + \beta_1 t_{ijk} + b_{0,i} + b_{1,i} t_{ijk} + b_{0,ij} + b_{1,ij} t_{ijk} + \varepsilon_{ijk}$$

with

$$\varepsilon_{ijk} \sim N(0, \sigma^2),\ b_{0,i} \sim N(0, \tau^2_{01}),\ b_{1,i} \sim \mathrm{N}(0, \tau^2_{11}),\ b_{0,ij} \sim \mathrm{N}(0, \tau^2_{02}),\ b_{1,ij} \sim \mathrm{N}(0, \tau^2_{12})$$

where
y_{ijk} = the number of fish counted at site i by observer j at time period k, where ($i = 1,\ldots,17$), ($j = 1,2$), ($k = 0,\ldots,8$),

β_0 = the intercept for all sites and counters,

β_1 = the slope (increase in the number counted per increase in time unit) for all sites and observers,

$b_{0,i}$ = random effect of site i on the intercept,

$b_{1,i}$ = random effect of site i on the slope,

$b_{0,ij}$ = random effect of fish counter j site i on the intercept,

$b_{1,ij}$ = random effect of fish counter j site i on the slope, and

ε_{ijk} = independent random error.

The observer effect was nested in the site effect.

The parameters were estimated using restricted maximum likelihood in order to avoid the bias of maximum likelihood estimation (Pinheiro and Bates 2000). The significance terms in the LME model were tested using a likelihood ratio test (LRT) (Hogg and Craig 1978). The LRT was compared to a mixed chi-square distribution (Stram and Lee 1994). The residuals from the fitted model were examined for autocorrelations within sites and observers.

Results

Strip transect surveys

Ninety-nine percent of the aggregate transect length in the 2001 Harris Bay survey had visibilities of at least 4 m, while greater than 96% had visibilities of at least 3 m for each of the remaining surveys. Therefore, transect widths were truncated at 8 m (4 m either side of the transect line) for Harris Bay in 2001, and 6 m for Harris Bay in 2002 and Nuka Island in 2002 and 2003. Only the lengths of transect that met these widths were included in the analysis.

Harris Bay surveys included 32 transects totaling 4,760 m in 2001, and 35 transects totaling 4,550 m in 2002 (Table 1). The initial Nuka Island survey design in 2002 included 30 transects. However, no black rockfish were observed along the low and moderately exposed south and western shoreline of the island (Fig. 3), resulting in a low sample size for the remaining study area. Therefore, the Nuka Island study area was redefined in 2002 to include only the outside portion of the island, and transects were added equidistant between the existing transects to increase the sample size and maintain the systematic design (Fig. 3). Sampling during the 2003 survey was conducted within the 2002 redefined

Table 1. Statistics for strip transect surveys for Harris Bay and Nuka Island.

	Harris Bay		Nuka Island	
	2001	2002	2002	2003
Date surveyed	Aug. 2-9	Aug. 7-14	June 4-13	June 27-30
Strip width (m)	8	6	6	6
Number of transects	32	35	16	16
Transect spacing (km)	1.37	1.26	0.51	0.51
Total transect length (m)	4,760	4,550	2,900	2,280
Encounter rate	0.036	0.024	0.025	0.0371
Coefficient of variation	18%	17%	22%	17%

area. The resulting effort for the Nuka Island surveys was 16 transects totaling 2,900 m in 2002 and 2,280 m in 2003 (Table 1).

Distributions of the sighting data indicated that fish observations decreased with distance off the transect line (Fig. 4). Although expected with transect sampling techniques, the large proportion of fish schools reported directly on the transect line illustrated the behavioral response of black rockfish to the presence of the divers. There was a marginally positive correlation between perpendicular sighting distance and visibility during the Harris Bay 2002 survey (Kendall's $\tau = 0.28$) while no correlation was present in the other surveys (Kendall's τ for Harris Bay 2001 = 0.09, for Nuka Island 2002 = –0.09, and for Nuka Island 2003 = 0.10). There was no significant relationship between school size and distance from the transect line for each survey. Therefore, an average was used to estimate school size within year and survey area.

Index estimates of black rockfish school density were similar for both years in Harris Bay, ranging from 3,946 schools per km^2 in 2002 to 4,460 schools per km^2 in 2001 (Table 2). Estimated school size was smaller in 2002 than in 2001, resulting in a lower index of density estimate in 2002 of 31,092 black rockfish per km^2 compared to 48,612 black rockfish per km^2 in 2001. Although both the fish observer and the habitat observer enumerated black rockfish during the 2003 Nuka Island survey, only estimates from the designated fish observer were used for inter-annual comparisons. Index school density and size estimates for Nuka Island were larger in 2003 than in 2002. Estimated black rockfish density for Nuka Island in 2002 was 33,521 fish per km^2 compared to the 2003 estimate of 70,000 fish per km^2. The coefficient of variation for estimates of individual density for all surveys ranged from 22 to 33%. There was no detectable increase in precision with a greater amount of transect line traversed. Due to the large variation, confidence intervals for density estimates overlapped for both survey area inter-annual comparisons.

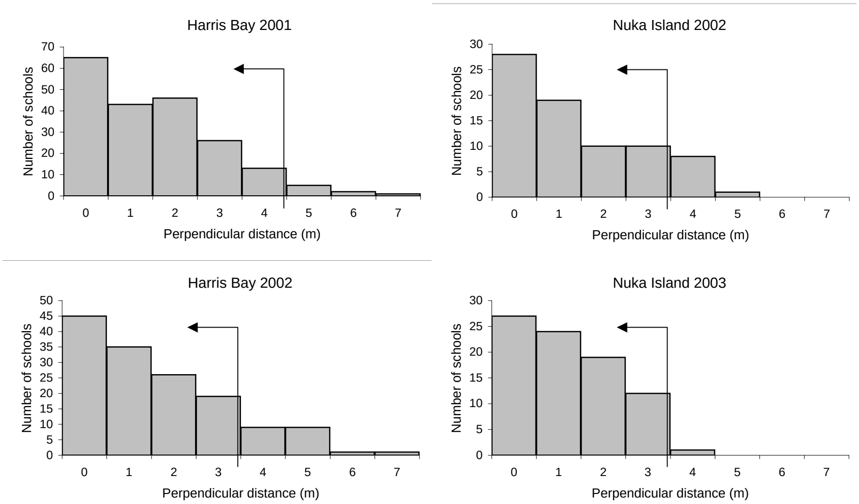

Figure 4. Black rockfish sighting data for each survey area and year. Perpendicular distances are in meters for the number of black rockfish schools sighted from the transect line. Arrows indicate transect widths based on minimum water visibility.

Table 2. Strip transect survey estimates for Harris Bay and Nuka Island by year.

	Study area	Year	Estimate	CV	LCL	UCL
Average school size	Harris Bay	2001	10.9	11%	8.7	13.7
		2002	7.9	16%	5.8	10.7
	Nuka Island	2002	8.0	16%	5.9	10.9
		2003	11.3	17%	8.2	15.7
No. schools/km^2	Harris Bay	2001	4,460	19%	2,870	6,193
		2002	3,946	17%	2,640	5,230
	Nuka Island	2002	4,198	22%	2,446	6,020
		2003	6,181	17%	4,069	8,200
No. individuals/km^2	Harris Bay	2001	48,612	22%	28,949	72,148
		2002	31,092	33%	14,710	55,033
	Nuka Island	2002	33,521	32%	15,747	56,241
		2003	70,000	28%	34,197	108,399

Average school size = average number of individual black rockfish per school; CV = coefficient of variation; LCL and UCL = lower and upper 95% confidence intervals, respectively.

Table 3. Strip transect survey estimates for between recorder comparisons.

	Fish recorder	Habitat recorder	Difference
Average school size	11.3	9.1	1.2
CV	17%	13%	
Number of schools per km^2	6,181	4,861	1.3
CV	17%	18%	
Number of individuals per km^2	70,000	44,167	1.6
CV	28%	26%	

Average school size = average number of individual black rockfish per school; CV = coefficient of variation.

Table 4. Test statistics and *P*-values for random effects from the responsive movement experiment.

Coefficient	Likelihood ratio test statistic	Degrees of freedom for mixed chi-square distribution[a]	*P*-value	SE	*t*-statistic
$b_{0,I}$	91.7148	0,1	<0.0001		
$b_{1,I}$	26.9714	1,2	<0.0001		
$b_{0,ij}$	0.000085	2,3	0.9999		
$b_{1,ij}$	0.000077	3,4	1.0000		
β_0 (intercept)			<0.0001	1.84	5.68
β_1 (time)			<0.0001	0.55	6.29

[a] A mixture of chi-square distributions was used to test the random effects; SE = standard error.

The coefficients of variation for estimates of school density, school size, and individual density were similar for the between-diver comparison, although black rockfish density estimates differed by 60% between observers (Table 3). The designated fish observer estimated both a greater number of black rockfish schools and a larger average school size than the habitat observer.

Responsive movement experiment

The number of fish observed during the timed count trials consistently increased for the first four counts, after which no discernable pattern was

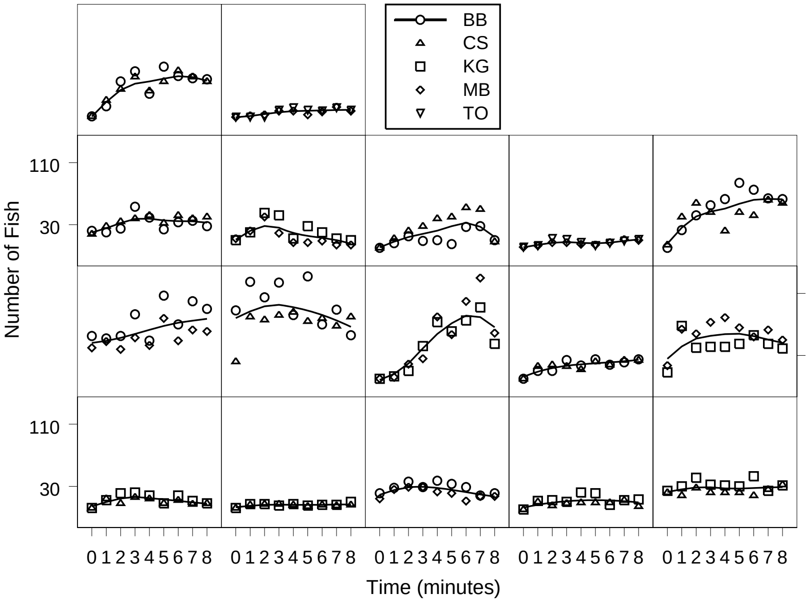

Figure 5. Graphs of observed black rockfish counts over time. Each plot is from a separate sampling site or trial. Symbols represent different fish counters. The line was generated using a Loess smoothing function.

evident (Figs. 5 and 6). Therefore, the analysis was restricted to evaluate the increase in fish over the first four counts (0 to 3 minutes). Fish counts were log transformed since the variance increased with the number of fish observed.

Significant site and time effects were detected while the observer effect was not significant (Table 4). The significance of the random effect of site on the intercept means that each site had a different count at time 0. The significance of the random effect of site on the slope means that the rate of increase from time 0 to time 3 is also dependent. The non-significant fish observer random effects indicate that there was no between-observer effect. Finally, the analysis showed that the number of fish observed increased significantly from time periods 0 to 3 minutes.

Discussion

The decay in sightings from the transect line was likely due more to responsive movement than to a decrease in detections with distance

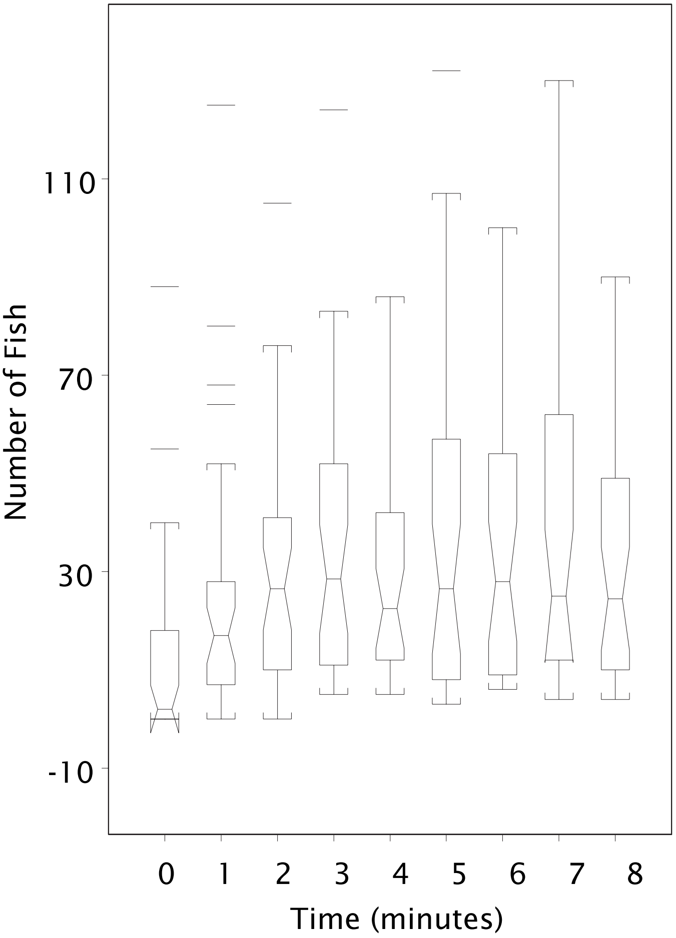

Figure 6. Box plot of black rockfish counts against time in minutes for all trials combined. Median (line inside box), 25th and 75th percentile (lower and upper ends of box), the span = 1.5 [25th and 75th percentile].

from the transect line. Fish counters also may have lumped observations toward the transect line, due to the difficulty of accurately determining the geographic center of fish schools. Since black rockfish are primarily pelagic, they are easily detected in the water column out to distances permitted by water visibility. Having strip widths of six to eight meters should have maintained near 100% detection within the strip with reasonable certainty.

The responsive movement experiment indicated that strip transect surveys for black rockfish will have problems with assumption violations, particularly related to fish movement. Specifically, transect methods assume that animals do not respond negatively or positively to the observer. This assumption may be violated, since the experiment showed that black rockfish move toward the observer. With respect to the strip transect surveys, it is likely that black rockfish moved within the strip in response to the presence of the observer prior to the observer detecting the fish. This positive responsive movement will result in counts that are biased high, causing overestimation of density and abundance. Further, responsive movement may not only introduce a positive bias, but might also cause a possible violation of independence. As fish are attracted to an observer, the size of the school can grow significantly, as was shown in the responsive movement experiment.

It is unclear whether selectively counting fish that have moved in response to observers can adequately adjust for the positive bias. Strip transect survey field methods attempted to address the responsive movement problem by (1) counting schools of fish as soon as they were sighted; (2) ignoring fish that were attracted as the school moved toward the divers; and (3) ignoring fish that followed, as the divers proceeded along the transect. From plots of the sighting data, however, it appeared that significant numbers of fish were still moving toward the observers prior to detection.

Responsive movement is a common, though understated, problem with underwater visual censuses. In an exhaustive study of the behavioral response of fish to the presence of divers, Kulbicki (1998) described four fish behaviors: neutral, shy, curious, and secretive. Behaviors were classified by the shapes of detection functions for each species, as determined from distance data collected at the time of first sighting. While the type and magnitude of the response varied by species, some type of responsive movement was detected in 58 of the 105 species studied. Kulbicki (1998) suggests that new methods be developed to assess density for those species where responsive movement biases estimates. Field and analytical methods have been developed to address the bias introduced by the responsive movement of animals on transect density estimates (Turnock and Quinn 1991, Buckland and Turnock 1992, Palka and Hammond 2001). All these methods require auxiliary data, either independent observations or data on animal orientation. The development of similar methods for black rockfish is beyond the scope of this paper, but it is unlikely that an efficient method could be developed.

The appropriateness of using an index of strip transect density estimates for black rockfish is also unknown. Bias in a population index is acceptable for monitoring the trend in a population, as long as the bias is consistent. The 60% discrepancy in between-diver density estimates from Nuka Island in 2003 showed that qualitatively determining whether fish were attracted to or following divers is difficult. It is likely that individual observers made dissimilar determinations on whether to count fish or not, thus decreasing the accuracy of the counts. Observer error in estimating fish density can be significant, though the magnitude can vary with the method used and the species enumerated (Bohnsack and Bannerot 1986, Thresher and Gunn 1986, Bortone et al. 1989). Part of the discrepancy in between-observer estimates during the observer comparison survey may also be attributed to the designated fish observer having more time available to enumerate fish, with the habitat observer possibly missing fish while recording habitat data. This was likely less of a contributing factor, since only substrate type was collected , which required very little time to categorize and record. An additional confounding aspect is individual diver knowledge and experience. Authors have stressed that training and experience will affect accuracy and precision in underwater visual censuses (Sale and Sharp 1983, Thompson and Mapstone 1997). Though two of the divers participated in all of the surveys, the remainder of the dive crew was not consistent. Finally, the field of view may not have been the same for individuals in the dive teams.

Even if the bias and accuracy issues with strip transect sampling could be overcome, the low precision in the estimates may still not allow the detection of changes in population size. Though the inter-annual Nuka Island density estimates differed from, by more than two times, the confidence intervals around those estimates, they still overlapped.

As an alternative to transect sampling, timed counts used during the responsive movement experiment may be a quantitative method for indexing the abundance of black rockfish. Stationary point counts have been used successfully for estimating fish density of reef fishes (Bohnsack and Bannerot 1986) and larger mobile species (Thresher and Gunn 1986). Point count estimates for large carangid species were more consistent and had lower variation than transect methods, and were the most efficient in terms of field effort (Thresher and Gunn 1986). In a simulation study, Watson and Quinn (1997) found that point counts can out-perform transect sampling in terms of achieving higher precision with lower effort. The bias with timed counts would be lower than with strip transects since the method accepts the fact that black rockfish are attracted to divers. In this study, there was no significant between-observer effect during the responsive movement experiment. This was likely due to the fact that observers simply counted all the fish present and did not have to make a decision whether to count a fish or not. Thus, the accuracy of timed counts should be higher than strip transects since there is less subjectivity in counting. The analysis showed that fish counts increased significantly in the first 3 minutes of observation, irrespective of school size. The trend followed no discernable pattern after the initial increase, although counts for all trials peaked at varying times within the full 8-minute counting period. With black rockfish likely attracted from an undetermined distance, a peak count may serve as an index of abundance for an area. This method would be analogous to baited stations where bait is set out to attract fish from the surrounding area while a diver or camera records the number of fish (Gotshall 1987, Ellis and DeMartini 1995, Willis and Babcock 2000). Due to the attraction of black rockfish to divers, the diver would serve as the bait.

In summary, strip transect methods can be problematic due to the attraction of black rockfish to divers and at most, estimates should be treated as an index. The responsive movement experiment indicated that the aggregation of fish around divers significantly increased for the first three minutes of observation. Within the duration of 8-minute trials, the number of fish increased to a peak count, although the rate of the increase and timing of the peak varied among trials. Counting methods were less subjective in the timed count experiment than during transect surveys. There was no significant between-diver effect during the responsive movement experiment, while between-observer density estimates during the transect survey comparison differed by 60%. Thus, timed counts may have potential as a quantitative method to index black rockfish abundance. These results are based on a relatively small sample size ($n = 17$), however, and more research is needed to explore the influence of factors such as visibility and habitat influences on stationary timed fish counts. As the behavior of fishes differs, so may the success of the methods used to assess them. Hence, the appropriateness of transect methods for estimating fish density will be species specific.

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Sampling Designs for the Estimation of Longline Bycatch

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Abstract

Unbiased estimates of longline bycatch on data-limited species are critical, especially when bycatch data may be the main source of information on such species. Estimates might be affected both by the spatial distribution of animals along the longline as well as by the sampling design used. We compared bycatch estimates from four sampling designs over six spatial distributions of bycatch using simulated data. The four sampling designs were simple random sampling, systematic sampling, partial sampling, and systematic sampling of longline sections. Spatial distributions included random distribution of bycatch and distribution of bycatch with periodicity. The methods were also used to estimate shark bycatch from a subset of the International Pacific Halibut Commission's 2002 Pacific Halibut Stock Assessment Survey data. Both spatial pattern and sampling design were found to affect the estimates of bycatch from longlines. All sampling methods gave unbiased estimates when bycatch was random, aggregated, or rare. The estimates of longline bycatch were most affected by the presence of periodicity in bycatch distribution along the line. Systematic sampling and partial sampling provided unbiased and the most precise bycatch estimates in most cases, and thus are the preferred sampling designs. Of these two, partial sampling is logistically the easiest in the field.

Introduction

Longline bycatch has long been an issue of concern, especially in the case of charismatic species like billfish, seabirds, sea turtles, and certain pelagic shark species (Klaer and Polacheck 1998, Goodyear 1999). A large body of literature exists on mitigating longline bycatch of endangered and charismatic species (Melvin et al. 2001, Loekkeborg and Robertson 2002, etc.). Unfortunately, data availability is very limited on bycatch of noncommercial and non-charismatic species on longlines. Thus the assessment of the effects of bycatch (and potential bycatch mortality) on such species is largely unexplored.

Generally, bycatch information is collected from logbooks, fishery-independent surveys, or observer data (Diamond 2003). Usually, if the observer coverage is not complete, a part of the fishing effort (fishing vessels and fishing gear) is sampled and the results are extrapolated to the entire fishery to estimate total bycatch (Diamond 2003). Statistical estimators such as mean catch-per-unit-effort or ratio estimators are used for the extrapolation process (Diamond 2003). Studies have been conducted on the effect of the use of different statistical estimators on the final estimate of total bycatch from trawl fisheries (Ye 2002, Diamond 2003). However, no such study has been published on estimation of bycatch from longline fisheries. Additionally, little research has been done on sampling designs to estimate longline bycatch. Fisheries bycatch data are used for several purposes. First, the principal use of bycatch data, in the case of commercially important species, e.g., Pacific halibut, is to incorporate bycatch levels and associated bycatch mortality in fish stock assessments (Leaman 1995). Second, bycatch estimates are used to make comparisons to some national standard, e.g., the potential biological removal (PBR) for a marine mammal stock (NMFS 2003). Third, it is used for development of bycatch mitigation plans (NMFS 2003). Fourth, bycatch data are sometimes used in ecosystem dynamics studies that analyze trophic relationships (Cox et al. 2002). Fifth, for data-limited species, bycatch data themselves might be the main source of information on the species. Thus, it is evident that unbiased and precise estimates of bycatch are a critical issue. This paper compares bycatch estimates from different sampling designs over different spatial patterns of bycatch on longlines. Bycatch estimates are compared for bias and precision. An estimate is unbiased if the 95% confidence intervals of the difference between estimated and true bycatch proportion includes zero. Precision is measured by the variance of the bycatch estimate, where lower variance reflects higher precision. This paper describes alternative sampling designs to collect unbiased and precise estimates of bycatch. No attempt is made to draw conclusions about the total population.

Estimates of longline bycatch might be affected both by the spatial distribution of species on the line as well as the sampling method involved (Murphy and Elliott 1954). Species that tend to be highly aggregated on the line could lead to different bycatch estimates as compared to species that are more solitary along the line. A sampling design that fails to capture the dynamics of the spatial distribution of species along the line could lead to biased estimates of bycatch. We analyzed the performance of four sampling designs in estimating longline bycatch over six spatial distributions of bycatch along the longline, using simulated longline data. The four sampling techniques used for the analysis were (1) simple random sampling (Cochran 1977), (2) systematic sampling, (3) partial sampling, and (4) systematic sampling of longline sections (Fig. 1 shows the latter three). The four sampling designs were applied to six spatial distributions: random distribution of bycatch, aggregated distribution of bycatch, rare bycatch, and bycatch distribution with periodicity. Three different frequencies of bycatch occurrence were used with respect to periodic data. We tested the hypothesis that each sampling design provides an unbiased estimate of total bycatch, for each spatial pattern. The same four sampling methods were then used to estimate bycatch on real longline data sets obtained from the International Pacific Halibut Commission's (IPHC) 2002 Pacific Halibut Stock Assessment Survey, conducted in the northeast Pacific Ocean.

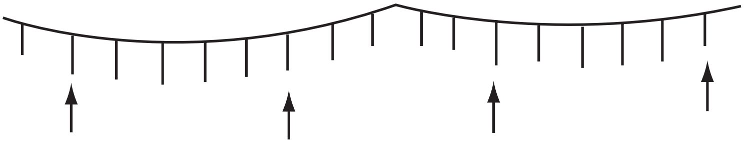
Methods

Experimental design: Data simulation, sampling, and resampling techniques

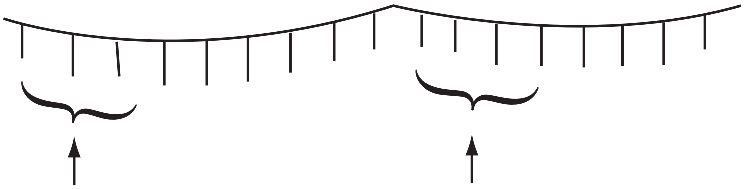
Simulated longline data sets were used for the analysis due to the lack of sufficient real longline data sets with hook-by-hook catch information. Each simulated longline was assumed to have the standard IPHC hook size, hook spacing, and bait species. Total number of hooks in a longline was 500 and was uniform for all simulated data sets. Each 100-hook section is called a skate of gear. The simulated data consisted of seven species (one target, six bycatch) and empty hooks. Generally the target species is landed and does not require sampling designs for estimation of its total catch. Hence for this study, we analyzed only one species of bycatch and did not consider the target species in our analysis. The total bycatch on the simulated longlines ranged from 10 to 250 animals per 500 hooks and were selected randomly. All data simulation was carried out in the *R* statistical language.

A total of 50 different longline data sets (total hooks per longline, $N = 500$) were simulated for each of the six spatial distributions of bycatch along the longline. Each of the 300 data sets was then sampled using the four different sampling methods to estimate bycatch. Sample size was $n = 100$ hooks for each of the four sampling methods, resulting in four samples per longline. The proportion of hooks with the bycatch species (p) was determined for each of the four samples per longline data set. Each of the 300 longline data sets was then re-sampled j times ($j = 500$) without replacement. The total number of animals in each re-sample was the same as the original longline, but their order of occurrence changed with each re-sample. Thus each re-sample of the longline was a permutation of the original longline. The four sampling methods were applied

a) Systematic sampling - sampling interval is fixed, starting point is random



b) Partial sampling - a sub-section is sampled, starting point is fixed, e.g., the first couple of hooks in a longline set



c) Section sampling - a sub-section is sampled, starting point is random

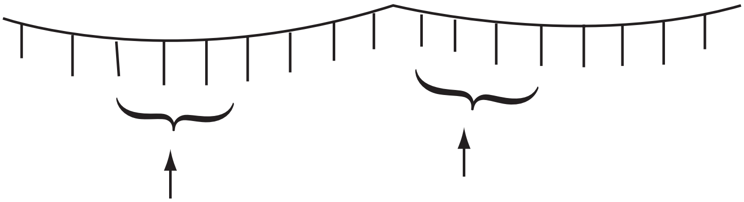


Figure 1. Schematic of systematic sampling, partial sampling, and section sampling designs.

Table 1. Details of the simulated data and the experimental design.

Simulated longline data	
Total number of simulated longlines	60
Total hooks per longline	500
Categories of bycatch species (including target species and empty hooks)	8
Total number of bycatch species simulated	25-250
Experimental design	
Total number of spatial patterns of bycatch	6 = Random, aggregated, rare, periodic (with three frequencies of occurrence)
Total number of sampling designs	4 = Random, systematic, partial and section sampling
Sample size per sampling design	100 hooks
Sampling fraction per longline	0.20
Total number of replications per longline	100

to each permutation of bycatch resulting in a sampling distribution of bycatch estimates for each of the original 300 longlines. From the sampling distribution, the mean and variance of the estimated bycatch was calculated, per sampling method per spatial pattern of bycatch. Each mean proportion was then compared to the true proportion of bycatch in each of the 300 data sets and tested for significant difference. If the sampling design is an unbiased technique, the confidence intervals of the difference between the estimated and true proportions should contain zero, at least 95% of the time (Zar 1999). The experimental design has been tabulated in Table 1. A schematic of the experimental design for the random spatial distribution is presented in Fig. 2.

The bycatch data obtained from each permutation of the data set was in the form of proportions, p_i. The binomial distribution is the underlying probability distribution of proportions (Zar 1999). Since the data sets per spatial distribution per sampling technique were large (50), the normal approximation to the binomial distribution was used in the analysis. The 95% confidence intervals of the difference between estimated and true proportion were also based on the normal approximation of the binomial (Cochran 1977).

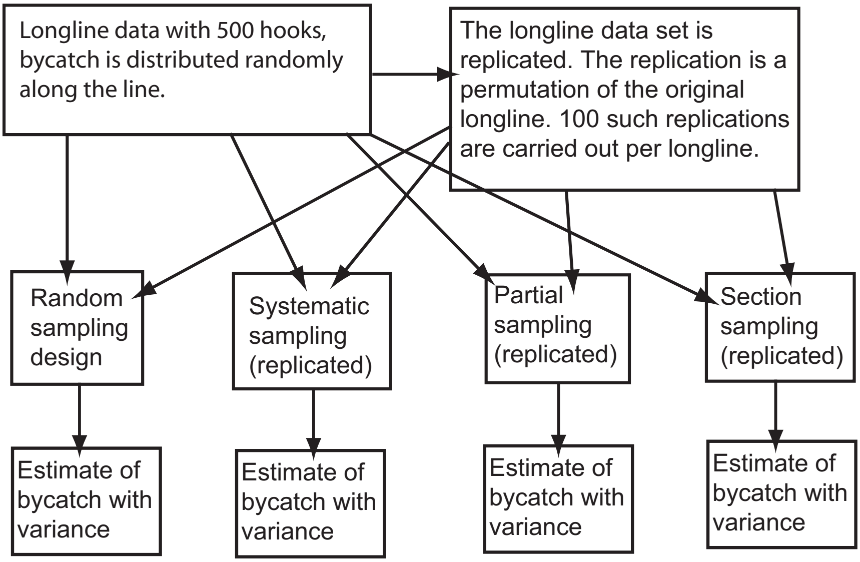


Figure 2. Schematic of experimental design for one spatial pattern of bycatch (random).

Sampling statistics

Four sampling designs were used to estimate bycatch from the simulated and real longline data sets. The sampling designs are discussed in this section.

Simple random sampling

A random sample of size $n = 100$ was taken from each simulated data set. The proportion of hooks, p, with the bycatch species on it was determined. The sample proportion, p, is an estimate of the true bycatch proportion, P.

$$p = \frac{1}{n}\sum_{i=1}^{n} y_i \tag{1}$$

is the proportion of bycatch species in the sample where,

y_i = catch of the ith hook (= 1 if the bycatch species is hooked, otherwise 0)

n = sample size = 100

i = hook number, ranges from 1 to 100

$$V\hat{a}r\left(\hat{P}\right) = \left(1 - \frac{n}{N}\right)\frac{p(1-p)}{(n-1)} = \text{Variance estimate of } p \tag{2}$$

where,

N = total number of hooks = 500

Systematic sampling

Every Kth hook of the longline was sampled until total sample size equaled n ($n = 100$). The starting point of each sample was selected randomly from the first Kth units ($K = 5$). This is the only random part in the sampling technique. K was estimated by $K = \frac{N}{n}$. Replicated systematic sampling was carried out to enable estimation of variance. Single systematic samples do not provide unbiased and consistent estimates of variance. The empirical variance of the replicate samples is an estimate of total variance provided the mean of all possible replicates follows a normal distribution (Kish 1965). The estimate of bycatch proportion is

$$p_{sysrep} = \frac{\sum_{r=1}^{c} p_r}{c} \tag{3}$$

where,

p_r = proportion of bycatch in the rth systematic sample

c = total number of replicate samples = 2 for our study

r = sample replication number = 1, 2, …, c

$$V\hat{a}r\left(\hat{P}\right) = \frac{\left(1 - \frac{nc}{N}\right)}{c(c-1)}\sum_{r=1}^{c}(p_r - p_{sysrep})^2 = \text{The estimate of variance of } p \tag{4}$$

Partial sampling

Partial sampling is a subsample of the longline. From a fixed starting point, a predetermined number (L) of consecutive hooks was sampled. The sampling interval was fixed and predetermined. The number of sections to be sampled was given by $K = \frac{N}{n} = 5$, as before, and length $L = \frac{n}{k}$ = 20 hooks.

Thus our partial samples consisted of the first 20 hooks of every skate of the longline. Replicate sampling was used to estimate variance. The two replicate samples used were the first 20 hooks and the middle 20 hooks of every skate. The proportion of bycatch and variance were estimated as in equations 3 and 4.

Systematic sampling of sections of the longline

Instead of individual hook, sections of the longlines were sampled systematically. The total number of sections to be sampled was determined by $K = \frac{N}{n}$, as before.

The starting point of the first section was determined randomly from the first Kth numbers ($K = 5$). The length of each section was calculated as $L = \frac{n}{k} = 20$ hooks.

Replicate sampling was used to get an estimate of variance. The proportion of bycatch and variance were estimated as in equations 3 and 4.

Spatial distributions of bycatch along the longline

Data were generated over six types of spatial distributions of the bycatch species along the longline. The bycatch species could be (1) randomly distributed, (2) rare, (3) aggregated, or (4) periodic (three different frequencies used). The hook positions for random bycatch were drawn from a random uniform distribution, without replacement. When the bycatch species was rare (caught on less than 10% of total hooks), a predetermined number of the species was randomly inserted into the simulated longline data set. Aggregated data were simulated such that the bycatch species occurred in clusters. For periodic data, the bycatch species was inserted into the data set at specific points based on a pre-set frequency of occurrence.

Three types of bycatch frequencies were simulated for the periodic data. In each simulation, the bycatch was split into two equal sized groups (group 1 and 2). In the first type of frequency (type I), bycatch group 1 occurred randomly in 20 hooks (hook 1 to hook 20) at the beginning of every longline skate (one skate = 100 hooks). Group 2 of the bycatch occurred randomly in the remaining 400 hooks. The second kind of simulated frequency (type II) had half of group 1 randomly distributed in 20 hooks at the beginning (hooks 1 to 20) and the other half in the middle (hooks 51 to 70) of each skate. Group 2 of the bycatch was randomly distributed in the remaining 300 hooks. In the third type of frequency (type III), a starting point was chosen randomly from the first five hooks. Then every fifth hook from this starting hook was selected (hook set 1). Group 1 of the bycatch species was randomly assigned to hook set 1. Group 2 of bycatch was randomly assigned to the remaining hooks. Type I bycatch frequency was one-fourth that of the partial and section sampling frequency. Type II bycatch frequency was half of the partial and section sampling frequency. Type III bycatch frequency was one-fourth the systematic sampling frequency. Though such animal distributions are probably highly artificial, the aim was to model the worst scenarios that could occur during sampling, i.e., most or all hooks with bycatch would be the hooks that were included in the sample. The purpose is to illustrate only one extreme of the sampling spectrum, which is possible only with simulated data.

Analysis of real longline data

Real longline data were also used in the study to corroborate the results obtained from the simulated data. Hook-by-hook data from a subsection of the IPHC 2002 Pacific Halibut Stock Assessment Survey was used for our analysis. The survey covers offshore waters from Oregon to the Bering Sea, in the northeast Pacific Ocean. The longline gear used for the survey is standardized gear (further details in the IPHC 2002 Stock Assessment Survey Manual). Bycatch was estimated for two shark species, the spiny dogfish (*Squalus acanthias*), and sleeper shark (*Somniosus pacificus*) for the available data sets using the sampling designs mentioned in the previous sections. Sample size remained the same as before, $n = 100$ hooks. No permutation of the longline data sets was conducted in this case. Estimated bycatch proportions were compared with true proportions to determine any bias in the estimates.

Results

Random distribution of bycatch along the longline

Bycatch was randomly distributed along the longline and the total bycatch (one species) per longline ranged from 52 to 200 animals per 500 hooks, i.e., the particular species of bycatch ranged from 10% to 40% of the total number of hooks. Section sampling gave unbiased bycatch estimates when bycatch was random. Both random sampling as well as systematic sampling methods overestimated bycatch. Partial sampling tended to underestimate bycatch. None of the methods were biased since the 95% confidence interval for the difference between estimated and true proportions contained zero, for all the sampling methods (Table 2). Partial and systematic sampling methods were the least variable methods whereas random sampling was the most variable method. The results are presented as a box plot in Fig. 3.

Rare distribution of bycatch along the longline

Bycatch of the particular species was rare along the longline and ranged from one animal to 50 animals per 500 hooks (0.02% to 10% of total hooks). Both random sampling and partial sampling gave unbiased estimates of bycatch. Systematic sampling tended to underestimate bycatch

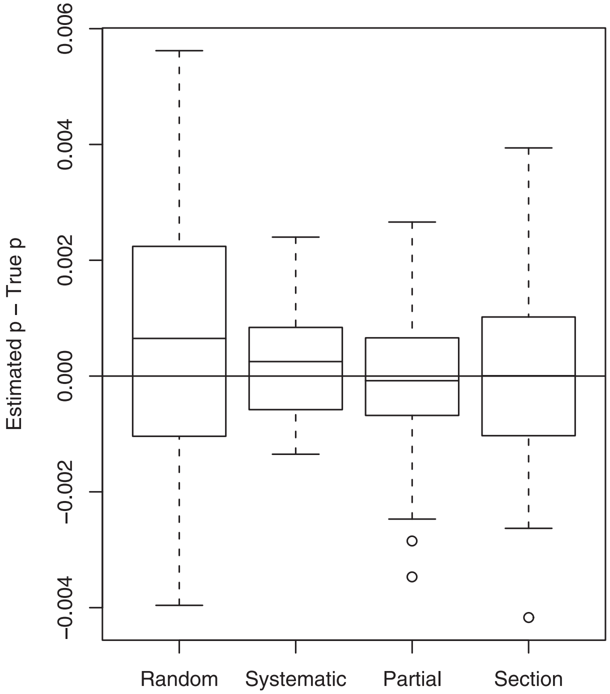


Figure 3. **Box plot of difference of estimated and true bycatch proportions from four sampling methods when bycatch is random. The horizontal line represents the zero line. Top and bottom borders of the boxes represent the 25th and 75th percentiles.**

and section sampling tended to overestimate bycatch. None of the methods were biased since zero fell within the 95% confidence interval for the difference between estimated and true proportions, for all the sampling methods (Table 2). Again, partial and systematic sampling methods were the least variable with random sampling being the most variable method. The box plot of the results is presented in Fig. 4.

Aggregated distribution of bycatch along the longline

Bycatch was simulated that the particular bycatch species always occurred in clusters. The bycatch of the species ranged from 50 to 250 animals per 500 hooks (10% to 50% of total hooks). Random sampling gave unbiased bycatch estimates. Systematic and partial sampling tended

Table 2. 95 % confidence intervals for the difference between estimated and true bycatch proportions.

Spatial distribution/ sampling design	Random sampling	Systematic sampling	Partial sampling	Section sampling
Random distribution	± 0.004	±0.002	±0.002	±0.003
Rare distribution	±0.002	±0.001	±0.001	±0.002
Aggregated distribution	±0.004	±0.002	±0.002	±0.003
Type I periodic distribution	±0.003	±0.002	(0.0197, 0.003)	(0.465, 0.005)
Type II periodic distribution	±0.003	±0.003	(0.107, –0.006)	(0.089, –0.005)
Type III periodic distribution	±0.004	(0.29, 0.01)	(0.011, –0.002)	(0.029, –0.005)
Real longline data analysis				
Pacific sleeper shark	(0.03, –0.013)	(0.01, –0.009)	(0.01, –0.008)	± 0.01
Spiny dogfish	±0.08	±0.03	(0.06, –0.04)	(0.06, –0.04)

to underestimate bycatch whereas section sampling tended to overestimate bycatch (Fig. 5). None of the methods were biased since zero fell within the 95% confidence interval for the difference between estimated and true proportions, for all the sampling methods (Table 2). Partial and systematic sampling were the least variable and random sampling was the most variable of all sampling designs (Fig. 5).

Periodic distribution of bycatch along the longline

Type I periodicity

As described earlier, here the frequency of bycatch along the longline was one-fourth of the partial sampling frequency, i.e., one-fourth of all hooks sampled by partial sampling would have bycatch on them. The number of the bycatch species per 500 hooks ranged from 10 to 180 (2% to 36% of total hooks). As expected, both partial and section sampling were positively biased (Fig. 6). Zero was not included in the 95% confidence interval of the difference between estimated and true proportions for these methods (Table 2). Systematic sampling was unbiased and random sampling tended to underestimate bycatch (Fig. 6). Section and random sampling were the most variable methods and systematic sampling was the least variable of all methods.

Type II periodicity

Here bycatch was simulated such that half of the hooks sampled by partial sampling design would have the bycatch species on them. Bycatch of the species studied ranged from 10 to 200 animals per 500 hooks (10% to 40% of total hooks). Again as expected both partial and section sampling

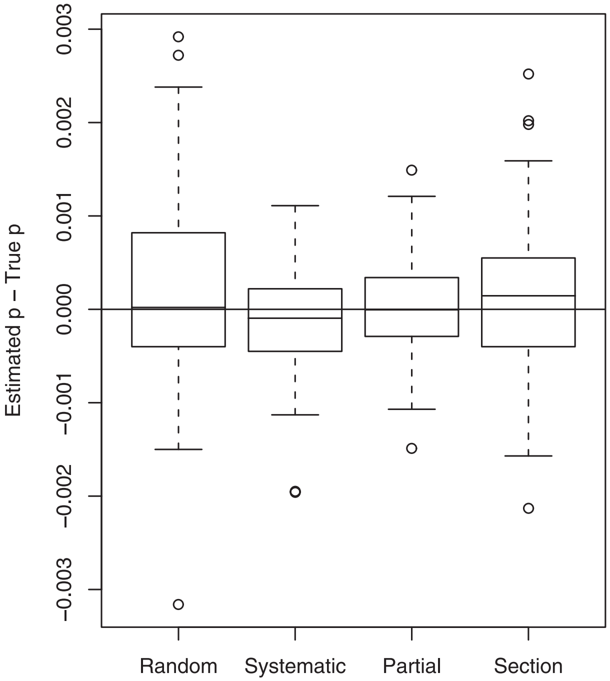


Figure 4. Box plot of difference of estimated and true bycatch proportions from four sampling methods when bycatch is rare. The horizontal line represents the zero line.

appeared positively biased (Fig. 7) but this was not reflected in the confidence interval values since the 95% CI for both these sampling designs included zero (Table 2). Random and systematic sampling were unbiased. Partial and section sampling designs were also the most variable; random and systematic sampling designs were the least variable.

Type III periodicity

Here bycatch was simulated such that half of the hook sampled by the systematic sampling design would contain the bycatch species. The bycatch species ranged from 14 to 180 animals per 500 hooks (2.8% to 36%). As expected systematic sampling was positively biased (Fig. 8). Random sampling was unbiased, whereas both partial and section sampling tended to overestimate bycatch but were not biased (Table 2). Random and partial sampling were the least variable whereas systematic sampling was the most variable of all methods.

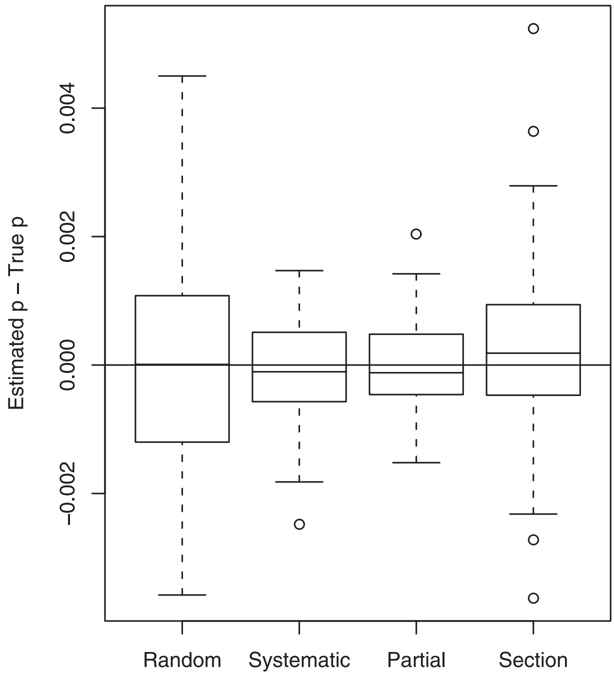


Figure 5. Box plot of difference of estimated and true bycatch proportions from four sampling methods when bycatch is aggregated. The horizontal line represents the zero line.

Analysis of spiny dogfish and sleeper shark bycatch

Analyses of real longline data sets were conducted to corroborate the results from simulated data. Hook-by-hook data were collected from twenty-one longline sets (one set = 100 hooks) July 1-10, 2002, during the 2002 IPHC Pacific Halibut Setline Stock Assessment Survey in central Gulf of Alaska. Spiny dogfish was the most common species, caught on 17.12% of the hooks. Pacific sleeper sharks were very rare and caught only on 0.2% of the hooks (for further details, see Menon 2004).

A runs test (Siegel and Castellan 1988) conducted on the real longline data sets did not reject the null hypothesis of dogfish and sleeper shark being randomly distributed on them (*p* value > 0.05). A chi-square test based on the geometric distribution was used to test for randomness in the distribution of dogfish and sleeper sharks (Skud and Hamley 1978).

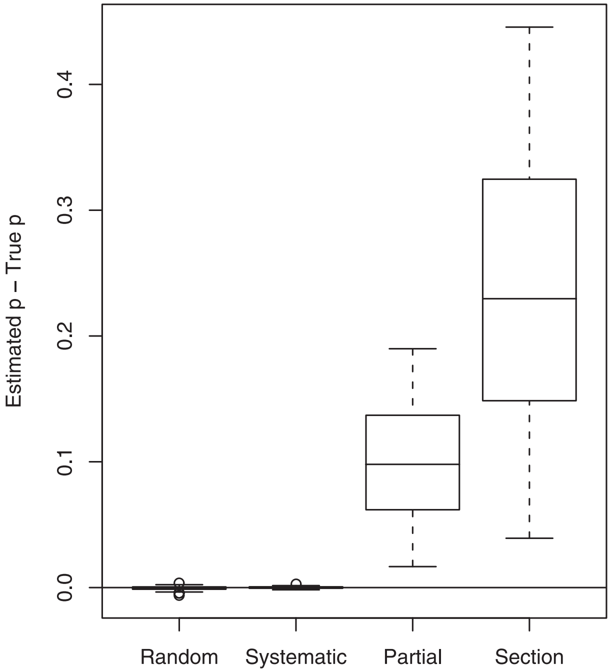


Figure 6. Box plot of difference of estimated and true bycatch proportions from four sampling methods when periodicity of bycatch is one-fourth of partial frequency. The horizontal line represents the zero line.

Again the null hypothesis of random distribution was not rejected for these two shark species, for the data available (p value > 0.05).

The results for the sleeper shark bycatch estimation are shown in Fig. 9. Sleeper shark is an example of rare bycatch in the longline data sets available for analysis. Partial sampling performed the best among all sampling designs, providing both unbiased as well as least variable bycatch estimates of sleeper sharks. Systematic and section sampling underestimated while random sampling overestimated sleeper shark bycatch. The confidence interval values (Table 2) indicate that none of the sampling designs are biased though this might be an artifact of the low number of longline data sets with positive sleeper shark bycatch.

The results of the spiny dogfish bycatch estimation are shown in Fig. 10. Both Figs. 9 and 10 are box plots over the available longline data sets. Spiny dogfish are a more common bycatch than sleeper sharks in

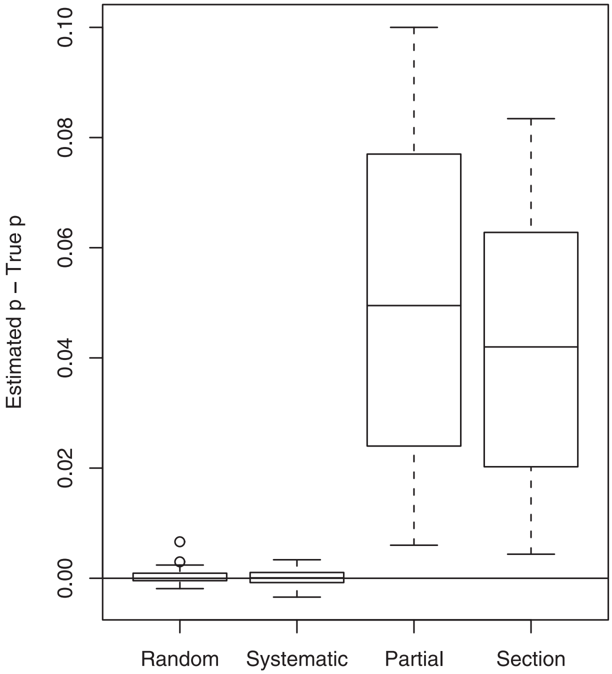


Figure 7. **Box plot of difference of estimated and true bycatch proportions from four sampling methods when periodicity of bycatch is half of the partial sampling frequency. The horizontal line represents the zero line.**

the longline data sets available for the analysis. Systematic sampling performed best in the case of dogfish providing unbiased and least variable estimates of bycatch. Partial and section sampling tended to overestimate dogfish bycatch whereas random sampling underestimated bycatch. Random and section sampling were the most variable sampling designs. None of the methods were biased (Table 2).

Discussion

Simulated longline data were used to test the performance of four sampling designs to estimate bycatch over six spatial patterns of occurrence along the longline. The objective was to compare the robustness of alternate sampling designs to variations in spatial patterns of catch. The results indicate that both sampling design, as well as the nature of the

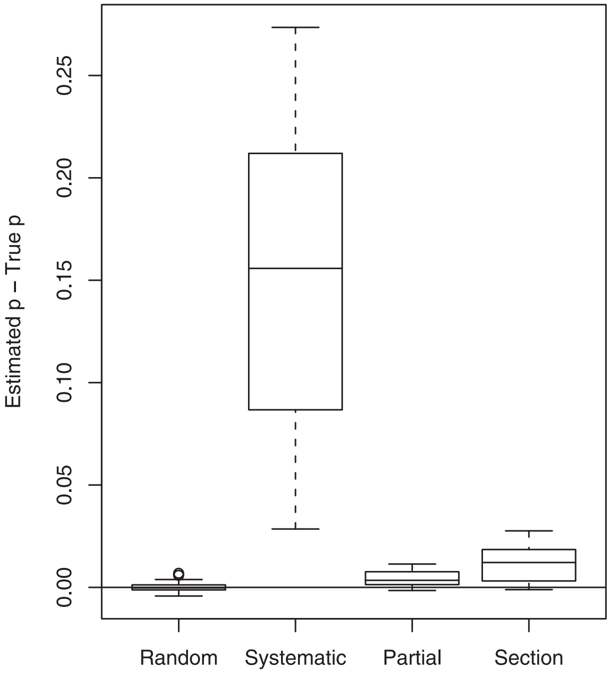


Figure 8. Box plot of difference of estimated and true bycatch proportions from four sampling methods when periodicity of bycatch is half of the systematic sampling frequency. The horizontal line represents the zero line.

distribution of bycatch along the longline, affect the bias and variability of bycatch estimates. In the absence of periodicity in bycatch occurrence, random sampling gave most precise but most variable bycatch estimates. Systematic and partial sampling tended to underestimate bycatch but these methods gave the least variable results. Section sampling generally tended to overestimate bycatch with high variability.

In the presence of periodicity, large fluctuations were seen in the results based on the frequency of bycatch occurrence along the longline. Random sampling gave unbiased estimates with low variability in most cases (with periodicity). Systematic sampling gave unbiased and less variable estimates depending on the frequency of bycatch. Both partial and section sampling tended to overestimate periodic bycatch with the resulting variability of estimates being high.

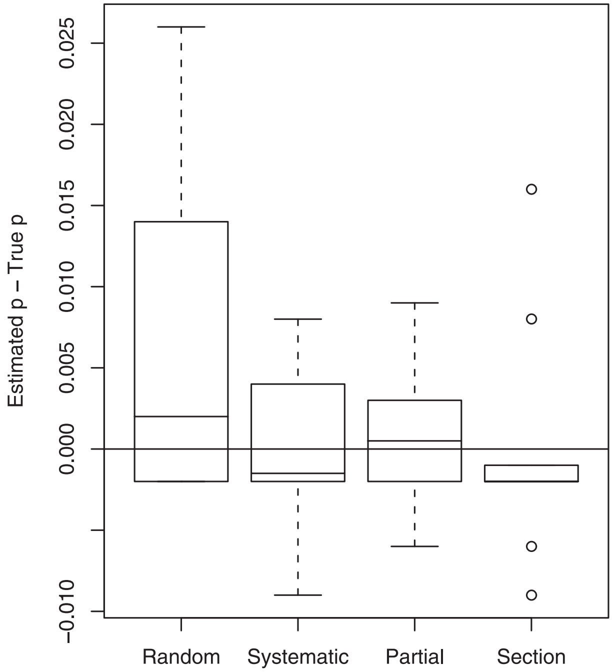


Figure 9. Box plot of the difference of the estimated and true sleeper shark bycatch proportions for the four sampling methods. The horizontal line represents the zero line.

The presence of periodicity as simulated here is likely rare in real longline data sets. The periodicities were included to obtain a full spectrum of possible bycatch distributions, which is possible only with simulated data. Real longline bycatch data are probably a mixture of all the spatial patterns described here. Very few studies have been conducted on the pattern of fish caught along the longline, and the few that have been done concentrated on target species like yellowfin tuna and Pacific halibut (Murphy and Elliott 1954, Maeda 1960, Skud and Hamley 1978). The tuna studies indicate a strong clustering pattern and clumping along the line (Murphy and Elliott 1954). The halibut study also indicated a nonrandom distribution of halibut along the line (Skud and Hamley 1978). Thus, it appears that clustering or aggregation is common along longlines. Assuming that real longline data sets rarely show periodicity, it is expected that both systematic and partial sampling would perform effectively to estimate bycatch, i.e., provide unbiased and precise estimates. Of these

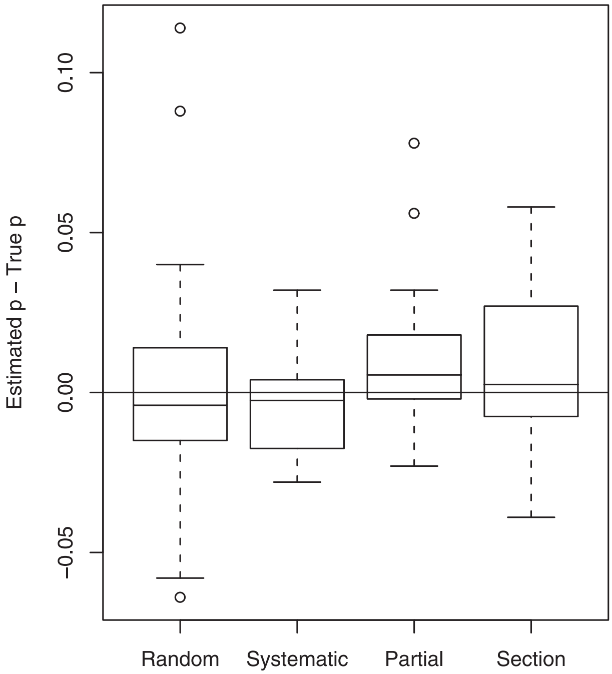


Figure 10. Box plot of the difference of the estimated and true spiny dogfish bycatch proportions for the four sampling methods. The horizontal line represents the zero line.

two methods, partial sampling is logistically the easiest in the field. It is easier for the observer to sample the first *x* hooks of a longline skate or sample a block of time during the hauling operation, both of which are examples of partial sampling. Thus with partial sampling the observer does not have to be present on the deck during the entire process of hauling to get a good estimate of bycatch. Twenty percent of the hooks were observed in our study (40% with replication). The effects of changes in sample size on the subsequent estimates were not studied. Determining the actual number of hooks to be sampled needs further research and is contingent on several issues, including the research/management objectives and the costs associated with sampling.

Though several sampling designs are presently being used to estimate longline bycatch (e.g., IPHC surveys), no studies have been published on the effects of the sampling design on bycatch estimates. Studies have been carried out to determine the best estimators of total bycatch from shrimp trawl data as well as to determine the ratio of bycatch-to-shrimp estimates (Ye 2002, Diamond 2003). Our study was the first to conduct a systematic study of the effects of different sampling designs on longline bycatch estimates. Partial sampling is commonly used in estimating longline bycatch (e.g., IPHC surveys) due to its logistical ease. Our study validates the method as being unbiased and precise for the estimation of longline bycatch. Getting unbiased and precise estimates of bycatch is critical since such data are put to several uses, including fish stock assessments (Leaman 1995), compliance with fishery regulations, design of bycatch reduction methods, etc.

There is no control on the pattern of bycatch along the longline. Suitable sampling methods could limit the variability and bias of bycatch estimates. This study provides the comparisons of different sampling designs to estimate unbiased and precise longline bycatch, which is especially important in the case of data-limited species, where the major source of information about the species is through bycatch data.

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Accounting for Climate Variability in Forecasting Pacific Salmon in Data-Limited Situations

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Abstract

Poor understanding of the major sources of environmental influence on Pacific salmon precludes quantitative forecasting in data-limited situations. Since 1997, low numbers of returning chum salmon to the Kuskokwim and Yukon rivers of Alaska have resulted in low harvests and significant negative economic and social impacts to rural residents of the region. The causes of these recent declines are unclear; however, poor ocean conditions for survival are thought to be important. No formal forecast has been available for this region, as estimates of the population size necessary to derive a quantitative forecast of returns were lacking. We recently generated abundance and escapement estimates for these two river systems. Our objectives in this study were to describe the spawner-recruit dynamics of this system and to identify important environment-recruit relationships. We explored a set of variables with plausible mechanistic relationships to five biological processes: freshwater survival, early marine survival, early marine predation, open-ocean survival, and open-ocean competition. We winnowed variables in these life history categories through an exploratory phase, and then used formal model selection procedures on those remaining variables under restricted combination scenarios. Our best models implied strong environmental effects and explained 89% and 81% of the variability in the data in the Kuskokwim and Yukon respectively. Cross validation estimates of forecast error were much smaller for models containing environmental covariates, confirming their utility. We also performed stepwise variable selection on the same set of reduced predictor variables. Results were similar to the previous models, yet identified the most influential life history stage rather than the broad scope of the restricted category-based models. We recommend managers use both forms of model selection and concentrate future research efforts on processes that confirm the mechanisms implied by the best predictor variables. We caution managers to be conservative when applying these models to management decisions and to consider simulation analyses to incorporate uncertainty in the reported estimates. The procedures developed here are applicable to other data-limited salmon systems.

Introduction

Fisheries management is often hindered by sparse data. These data-limited situations are typical of new and developing fisheries in which the practical constraints of a large geographic management area or a small local economy limit data collection. Managers are responsible for developing appropriate regulations for fisheries, even when conventional estimates of stock and recruitment are poor or unavailable. A typical approach is to produce qualitative assessments in the form of harvest outlooks or informal run projections. Unfortunately, these measures are limited in scope and predictive capability. To produce a reliable quantitative forecast for a given region, it is necessary to have some measure of stock productivity and a forecasting procedure suited to the system's conditions (Chatfield 1984, Haddon 2001).

Management of chum salmon of the Kuskokwim and Yukon rivers (Fig. 1) provides two examples of such data-limited situations. Chum salmon (*Oncorhynchus keta*) of these systems support important subsistence and commercial fisheries for rural area residents. Currently, however, there are no rigorous quantitative procedures for forecasting and managing chum salmon of these rivers because historic estimates of escapement (spawner abundance) and total run size have been considered unreliable or inadequate for developing such procedures (Bergstrom et al. 2001, Burkey et al. 2002). Instead, the Alaska Department of Fish and Game (ADFG) produces informal run outlooks for these rivers based on available escapement estimates, age composition, anticipated harvest, and recent abundance trends given current management regulations (Eggers 2002). The fisheries are managed in-season based on assessments from subsistence reports, test-fish catches, commercial catch statistics, main stem sonar, and tributary escapement projects such as aerial surveys, counting towers, weirs, and tributary sonar counts (Eggers 2002). The primary management objective for chum salmon in the Kuskokwim and Yukon rivers is to ensure adequate spawning escapements (JTC 2002), which are typically set in accordance with historical escapement levels (Burkey et al. 2002). Recently escapement goals were developed for two early run

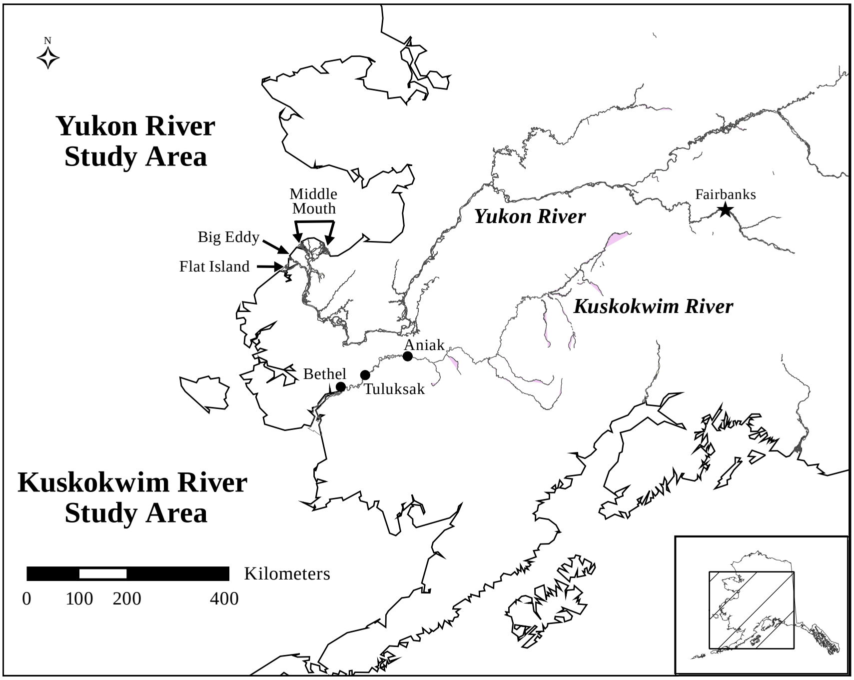


Figure 1. Study area for Kuskokwim and Yukon rivers. Locations on map show areas where age data were collected.

chum salmon stocks in the Yukon River basin based on spawner-recruit relationships; however, these analyses rely on broad assumptions about stock composition in the commercial harvest (Clark and Sandone 2001).

There have been severely decreased harvests of chum salmon in the Kuskokwim and Yukon rivers since 1997. The average annual chum salmon harvest recently dropped from 2 million fish (1980-1996) to 0.3 million (1997 to 2001), and was generally coupled with low chum salmon escapements. Recent declines prompted the governor to issue formal declarations of economic disaster for this region. The low harvests coupled with declines in market value of chum salmon (Buklis 1999, Eggers 2002) have resulted in significant negative social and economic impacts in many rural communities along the Kuskokwim and Yukon rivers.

Poor ocean conditions for salmon survival are among the list of plausible causes for the recent declines in Kuskokwim and Yukon river chum salmon (Geiger and Hart 1999). The declines were observed across widely separated river systems, indicating poor conditions affecting fish that share a common marine environment (Kruse 1998). In addition, oceanic conditions in the eastern Bering Sea since 1997 have been highly variable beginning with fluctuations in the physical environment as warm sea surface temperature (SST) anomalies occurred in conjunction with the 1997-1998 unusually intense and early El Niño event (Niebauer 1999, Stabeno and Hunt 2002). Unusually weak winds, anomalous mixing events, and rapid melting of the ice edge led to observations of aquamarine waters extending as far south as Bristol Bay and into the Chukchi Sea in 1997-1998. Such chalky-type waters often indicate a predominance of coccolithophores in the phytoplankton bloom. These events persisted through 2001 and reappeared again in 2003, suggesting relatively nutrient-deficient oceanic conditions (Stabeno and Hunt 2002). The phytoplankton biomass in the Bering Sea is typically dominated by diatoms and the persistence of a large coccolith bloom implies a reorganization of the food web and potentially more steps to transfer energy through this system (Olsen and Strom 2002). These extreme and persistent conditions in the Bering Sea were associated with unprecedented changes throughout the ecosystem, including declines of zooplankton, salmon, northern fur seals, Steller sea lions, and seabirds and substantial increases of jellyfish and baleen whales (Stabeno and Hunt 2002).

Many studies have related environmental variables to fluctuations in Pacific salmon data (e.g., Hare and Francis 1995, Gargett 1997). In conjunction with these process oriented papers, some investigations employed time series analysis to develop models that identified important relationships between simple environmental measures, such as sea surface temperature, and salmon catches (Quinn and Marshall 1989, Downton and Miller 1998). Some investigations have considered the spatial covariation between environmental variables and survival rates of salmon to better understand the scale on which environmental processes affect salmon (e.g., Mueter et al. 2002b). Other studies considered formal model selection on a variety of environmental predictors to improve forecast accuracy of salmon returns (Adkison et al. 1996, Adkison and Peterman 1999) or to understand the relative influence of regional and large-scale effects on survival rate (Mueter et al. 2002a). However, few studies emphasize the importance of defining the environmental predictors to focus on the influential life history periods of the particular salmon stock being modeled. Recently, Logerwell et al. (2003) developed a conceptual model of four time periods throughout the early life history of Oregon coho (*Oncorhynchus kisutch*). They used a general additive model with environmental data representing the four stages to explain a large proportion of variability in the coho smolt-to-survival estimates. When combined with forecasting techniques, these mechanism-based studies have large potential for developing biologically meaningful predictions more than one year into the future (Logerwell et al. 2003).

Our study objective was to develop rigorous, mechanistically based models to improve forecasts of Kuskokwim and Yukon river chum salmon and to better understand the potential effects of environmental conditions on their survival and recent declines. As a first step, Shotwell and Adkison (2004) proposed a new methodology for deriving estimates of total escapement and run size of summer chum salmon in each of these data-limited systems. This nonstandard technique combined the information from multiple data sources that alone were not sufficient to develop the necessary estimates. In this paper, we develop stock-recruitment models that combine those abundance estimates with environmental variables that may be useful predictors of recruitment.

Because the time series of available stock-recruitment data for Kuskokwim and Yukon River chum salmon are short (roughly 20 years in each case), and because numerous candidate environmental variables were considered (over 30 in each case), we undertook a three-stage approach designed to maintain biological realism and reduce the chance of spurious relationships. In data-limited situations, it is well known that putative environment-recruitment relationships often break down as new data become available (e.g., Myers 1998). This is more likely to occur when investigating environmental indices that poorly reflect the temporal and/or spatial scales of biologically reasonable relationships. For example, analyses of spatial covariation in survival rates of salmon suggest that important environmental effects occur primarily at local or regional scales less than approximately 1,000 km (e.g., Pyper et al. 2002). Thus, in our first stage of model development, we attempted to select environmental variables with temporal and spatial scales consistent with their potential effects (either direct or indirect) on recruitment. We then organized the variables into five categories related to the timing and nature of those potential effects by life history mechanism (freshwater survival, early marine survival, early marine predation, open-ocean survival, and open-ocean competition), and considered only one or two variables from a given category in each alternative model. For each biological mechanism, we selected environmental factors from appropriate locations and time periods, given our understanding of chum salmon life history.

In the second stage, we conducted an exploratory analysis to limit the number of candidate variables to those that exhibit at least some relationship with the residuals of the basic spawner-recruit model. We used a conservative correlation measure to eliminate variables and inspected cross-correlations between variables to eliminate redundancy within a given biological category. Models were then developed based on the remaining variables and evaluated using a formal model-selection criterion to further limit the set of models to a relatively small group. Following this we used regression diagnostics and prediction error to inspect competing models for predictive ability and potential violation of assumptions. We also examined the biological realism of the models by analyzing the magnitude and direction of parameter coefficients across alternative models. We then produced forecasts for the best model with estimates of forecasting error and compared these results with those of the base model that did not include environmental predictors. Our approach to model development and selection should produce a more accurate and reliable forecast than model selection without the life history categories and identify indicators of chum salmon variability for future management decisions in these data-limited systems.

Methods

Salmon data

Chum salmon have one of the broadest ocean distributions of any Pacific salmon species, ranging in Asia from Korea to the Laptev Sea in the Arctic Ocean and in North America from California to the Mackenzie River in the Beaufort Sea (Salo 1991). In addition, some populations of Yukon River chum salmon travel more than 2,000 miles upriver to spawn. A given river system often has two genetically distinct groups of chum salmon that differ in their run timing (early and late, Salo 1991). In this study, we concentrate on the early runs of chum salmon to the Kuskokwim and Yukon rivers. The early run population of chum salmon begins to enter the Kuskokwim River from the ocean in early June, with numbers peaking in early July and diminishing through early August (Molyneaux 1998). Spawning typically occurs between mid July and late August (D. Molyneaux, Alaska Dept. of Fish and Game, pers. comm.). In the Yukon River, the early run population (referred to as “summer chum”) enters freshwater beginning in mid June, peaks in late June to early July, and diminishes through late July (T. Lingnau and D. Molyneaux, Alaska Dept. of Fish and Game, pers. comm.). These summer chum typically spawn in the lower 500 miles of the Yukon and Tanana river drainage (Bergstrom et al. 2001).

Following emergence, chum salmon immediately begin their seaward migration, beginning in April to May in the Kuskokwim and Yukon rivers (D. Molyneaux, Alaska Dept. of Fish and Game, pers. comm.). Chum fry typically feed on larval and adult insects during their early life history but it is unknown to what extent they feed during downstream migration in the Kuskokwim and Yukon rivers. Often major predators of chum salmon during the early life stages are coho salmon, cottids, and trout (Healey 1982, Salo 1991). However, there is no information on predation mortality for summer chum along the Kuskokwim and Yukon rivers (C. Zimmerman, U.S. Geological Survey, pers. comm.). The early marine distribution of chum salmon is also relatively unknown for these river systems. An early study found that chum juveniles in the Yukon River were widely distributed throughout the delta region from June through late July. The juveniles were thought to move rapidly through the delta following ice breakup (Martin et al. 1986). Chum juveniles originating from the Kuskokwim and Yukon rivers are thought to move out of the

Bering Sea to the North Pacific by late fall or winter. They typically spend the next few years feeding primarily on gelatinous zooplankton (e.g., pteropods, appendicularians, jellyfish) and crustaceans (e.g., euphausiids, copepods, amphipods) in the Gulf of Alaska (Tadokoro et al. 1996) and return primarily in June of their maturing year through the central Aleutian passes (Salo 1991). Chum salmon mature after two to six years of age, with four- and five-year-olds dominating in the Kuskokwim and Yukon rivers (D. Molyneaux, Alaska Dept. of Fish and Game, pers. comm.).

We used annual estimates of escapement and total abundance for the Kuskokwim (1976-2000) and Yukon rivers (1975-1999) from Shotwell and Adkison (2004) and were able to update the Kuskokwim values through 2002. We used percent-at-age data for each river to derive estimates of recruitment by brood year (i.e., the abundance of recruits corresponding to a given brood-year escapement). For the Kuskokwim River, numbers-at-age data were available from samples taken in the commercial fishery from 1976-1979, 1981-1992, 1994-1996, and 1998. These data were weighted by the commercial catch at the time the sample was taken and expressed as percent-at-age. For the remaining years we used percent-at-age estimates from available projects throughout the river. Specifically, we used data from the Aniak River sonar for 1980 and 1997-1999, the Tuluksak River weir for 1993, and the Bethel Test Fishery for 2000-2002 (D. Molyneaux, Alaska Department of Fish and Game, pers. comm.).

For the Yukon River, we obtained age data from the ADFG Mark, Tag and Age Laboratory. We used test fishery 5.5 inch (0.14 m) mesh samples (Flat Island test fishery from 1975-1978, Big Eddy test fishery from 1979-1983 and 1986-1999, and Middle Mouth test fishery for 1985) with the exception of one year, 1984, where we used Big Eddy 8.5 inch (0.22 m) mesh samples of chum because no other age samples were available for that year. Age composition is thought to change throughout the duration of the run in any given year (D. Molyneaux, Alaska Dept. of Fish and Game, pers. comm.). However, due to limited data, samples were pooled across the season in order to compute the age proportions for a given year. We consider ages three, four, five, and six for both rivers over all years.

Recruits by brood year were then calculated by first multiplying the percent-at-age by the abundance for a given year and then offsetting the recruits for that brood year by the age. Total recruits were then just the sum of the individual ages for a given brood year (Quinn and Deriso 1999). Since six-year-olds constitute on average just 0.01% and 2.0% of the total population for the Kuskokwim and Yukon rivers respectively, we included a final year where returns were only available through age five. This resulted in spawner-recruit data by brood year from 1976 to 1997 for the Kuskokwim River and 1975 to 1994 for the Yukon River.

Environmental data

We selected environmental variables for each river that reflected conditions during the life history stages potentially most influential on summer chum salmon survival (Table 1). Variables that pertain to the freshwater stage were generally lagged 1 year from spawning to represent conditions during emergence or outmigration. Early marine residence variables were lagged 1 to 2 years from spawning to represent the general state of the Bering Sea during entry or after one year of ocean residence. We defined open-ocean competition as occurring between 2 and 4 years after spawning, during the period of co-residence with the major populations of Japanese chum and pink salmon (*Oncorhynchus gorbuscha*). Where appropriate, variables were averaged over the months considered (Table 1) and then all variables were standardized by their respective mean and standard deviation.

Model development

We used the linear form of the generalized Ricker stock-recruitment curve as our model for estimating survival rates ($\hat{R}/S$) of summer chum:

$$\ln(\hat{R}/S) = \ln(\hat{\alpha}) - \hat{\beta}S + \hat{\gamma}_1 X_1 + \ldots + \hat{\gamma}_p X_p + \varepsilon, \quad \varepsilon_y \sim N(0,\sigma_\varepsilon^2). \tag{1}$$

Here $\hat{R}$ is recruits, S is spawners, $\hat{\alpha}$ is the productivity parameter that determines recruits per spawner at low stock levels, and $\hat{\beta}$ is the density-dependent parameter or the decrease in survival rate as S increases, $X_1 \ldots X_p$ are the environmental predictor variables for a given model and $\hat{\gamma}_1 \ldots \hat{\gamma}_p$ are the associated scaling parameters (Quinn and Deriso 1999, Haddon 2001). Recruits-per-spawner is generally assumed to have lognormal error; therefore, we express $\hat{R}/S$ with normal random error ε_y having a mean zero and standard deviation σ_ε (Hilborn and Walters 1992). As our base model, we used the simple linear Ricker model; i.e., the model above without any environmental predictors. The Ricker model is traditionally used for examining Pacific salmon data; however, other models, such as the Beverton-Holt, may also be appropriate (Quinn and Deriso 1999). In this study we did not consider other stock-recruitment models because we were concentrating on the utility of the environmental information. A recent study on the spatial covariation of survival rates in pink salmon suggested that there was relatively little difference between rates derived from the Ricker and Beverton-Holt models across multiple stocks in the Northeast Pacific (Pyper et al. 2001).

Initially, we collected a total of thirty-four and thirty-six potential variables for the Kuskokwim and Yukon rivers, respectively. This is a fairly extensive set to consider for model selection (Burnham and Anderson 1998). We therefore conducted an initial exploratory analysis to limit

Table 1. First stage environmental variables for both rivers.

Category[a]	Variable[b]	Locations[c]	Name[d]	Months[e]	Lag[f]	Sign[g]
Freshwater (1)	Air temperature[a]	Bethel, Tanana, and Nome station	ATPF	AM	+1	N
			ATSR	JJ	+4	N
			ATNF	JJAS	+1	P
	Precipitation[a]	Bethel, Tanana, and Nome station	PRPF$_{K}$	AM	+1	~P,P
			PRSF$_{Y}$	JJAS	+1	N,0
	Streamflow[i]	Crooked Creek, Pilot station, and Tanana	STPF	AM	+1	N
			STSR	JJ	+4	P
	Iceout Yukon[j]	Yukon Delta	IYDA$_{Y}$	Date	+1	P,0
Early marine residence (2)	Winter surface air temperature[k]	St. Paul Island	SATF$_{K}$	DJFM	+1	0,P
			SATS	DJFM	+2	P
	Favorable winds[k]	Mooring 2 57°N 164°W	FWMF$_{Y}$	MJJ	+1	N,~N
			FWMS$_{K,Y}$	MJJ	+2	0,0
	Strong winds[k]	Mooring 2 57°N 164°W	SWMF	MJJ	+1	0,P
			SWMS$_{K}$	MJJ	+2	P
	Along-peninsula component of wind stress[k]	Unimak Pass 54°N 165°W	WUPWF$_{Y}$	NDJFMA	+1	P,0
			WUPWS$_{K,Y}$		+2	~P
			WUPSF$_{Y}$	MJ	+1	N,~P
			WUPSS$_{Y}$		+2	P,~P
	Sea level pressure[k]	Bering Sea 55°-65°N 170°-160°W	SLP$_{K,Y}$	AMJ	+1	0,0
	Ocean sea ice cover[k]	Bering Sea, various	ICF$_{K}$	Winter	+1	~P,N
			ICS	Winter	+2	N
	Sea surface temperature[k,l]	Mooring 2, Bering Sea	MSSTF	JFMA	+1	P
			ERSSTF$_{Y}$	MJ	+1	N,~N
	Zooplankton[k]	Bering Sea	ZOOP$_{K,Y}$	JJ	+1	0
Early marine predators (3)	Western Alaska coho[m]	Kuskokwim, Delta rivers	WAKCF	Season	+1	N
			WAKCJ$_{Y}$	Season	−1	N,0
Open-ocean (4)	Arctic oscillation[n]	North of 20°N	AOWF	DJF	+1	N
			AOSF	JJAS	+1	N
	Pacific decadal oscillation[k]	North of 20°N	PDOWF$_{K,Y}$	DJF	+1	~P
			PDOWR	DJF	+4	P
	Siberian-Alaskan index[k]	Siberia, Alaska-Yukon	SAIF$_{K,Y}$	DJFM	+1	~P, ~N
			SAIS	DJFM	+2	N
Open-ocean competition (5)	Japanese chum salmon[o]	Japan harvest	ACS	Season	+2	N
			ACD		+3	N
			ACR$_{Y}$		+4	N,~N
	Japanese pink salmon[o]	Japan harvest	APS	Season	+2	P,0

(a) Categories are the five major periods of survival. (b) Variables are the types of information we acquired. (c) Locations are the areas where data was obtained. (d) Names are the acronyms for each variable. Subscripts following the acronyms indicate a variable eliminated in the exploratory phase, K = eliminated for Kuskokwim, Y = eliminated for Yukon. (e) Months are the months which we averaged data or over which the index was based (J = January, F = February, etc.). (f) Lag was the number of years a particular index was lagged from the spawner-recruit data. (g) Sign was the direction of the *r*-value between the variables and the residuals from the base model in both regions (Kuskokwim, then Yukon; N = negative, P = positive, ~ = slightly, 0 = none). (h) WRCC 1998. (i) USGS No date. (j) Bergstrom et al. 2001. (k) NOAA No date a. (l) IRI No date. (m) Burkey et al. 2002. (n) NOAA No date b. (o) NPAFC No date.

the number of candidate variables, then used a formal model selection criterion to arrive at our final best model(s) for each river system.

Exploratory phase

We calculated Pearson product moment correlations (*r*-value) between each of the predictor variables and the estimated residuals from our base model. We then ran a simple *F*-test on these correlation values to determine which variables were significantly correlated with the spawner-recruit data (Kleinbaum et al. 1998). We used a conservative (in terms of classical statistics) reduction approach, limiting the potential variables to those with *r*-values of at least ±0.2, which approximately corresponded to *p*-values of 0.35. This served to eliminate variables that would add very little to explaining the variability in the data and still retain at least some variables from each life history category. Following this exploratory phase we examined cross-correlations among environmental variables to check for redundancy within a particular life history category.

Selection phase: Category-based models

Using a combination of AD Model Builder (version 5.01, Otter Research) and S-Plus software (version 6, release 2, Copyright 1988-2001 Insightful Corp.), we examined a family of models based on the remaining variables from the exploratory phase. We allowed models with up to five environmental variables and included all nested models (i.e., models with fewer than five variables). The initial five variable models were created from one of two scenarios. First, we examined all possible combinations in which one variable was selected from each of the five life history categories (Table 1). Second, we created models where we allowed two variables from the early marine residence category and three additional variables with only one variable selected from each of the remaining four categories. We explored the second option because Pyper et al. (2002) suggested that the early marine phase might be the most important in determining survival. We compared model fits using the Akaike Information Criteria (AIC_c) for small sample size (Burnham and Anderson 1998):

$$AIC_c = n\ln(\hat{\sigma}^2) + 2K\left(\frac{n}{n-K-1}\right) \tag{2}$$

$$\text{where, } \hat{\sigma}^2 = \frac{\sum \hat{\varepsilon}_i^2}{n}, \text{ (the MLE of } \sigma^2\text{)} \tag{3}$$

where n is the number of observations, K is the number of parameters, and $\hat{\varepsilon}_i$ are the estimated residuals for a particular candidate model (Burnham and Anderson 1998). For each candidate model there were at least three parameters, $\hat{\alpha}$, $\hat{\beta}$, and $\hat{\sigma}$ plus $\hat{\gamma}_1 \ldots \hat{\gamma}_p$, depending on the model. We computed the differences (Δ_i) between the lowest AIC_c value and the AIC_c values of all other candidate models in the set. Our "best" or preferred model was the model with the lowest AIC_c. However, a model for which the Δ_i was small ($\Delta_i < 3$) has substantial support and can be considered to fit the data almost equally well (Burnham and Anderson 1998, Mueter et al. 2002a). Therefore, we present all models that were selected as plausible ($\Delta_i < 3$) with their respective parameter estimates and predictor variables.

Model diagnostics and cross-validation

Following model selection, we assessed the consistency and reliability of the plausible models through regression diagnostics and estimates of prediction error. We checked all parameter coefficients for statistical significance and for large changes in magnitude between models as this is an indicator of a potential spurious relationship. We examined the sign of each parameter to determine if it made biological sense and inspected diagnostic plots of the residuals for the presence of non-normality, outliers, and autocorrelation. We then calculated the multiple *r*-squared (R^2) value of all models and several sub-models to determine the relative contribution of key variables to the improvement in fit.

We calculated estimates of forecast error for the equally plausible models by cross-validation (Efron and Tibshirani 1993, Adkison and Peterman 1999):

$$\mathrm{CVFE} = \sqrt{\frac{\sum_{i=1}^{n}(y_n - y_{n-i})}{n}}\,. \tag{4}$$

Here CVFE is the cross-validation forecast error, y_n is the sum of squared residuals over all years (n), and y_{n-i} is the sum of squared residual over all years minus the squared residuals of the left out year (i). We calculated CVFE values (equation 4) for each of the competing models and generated 80% confidence intervals (CIs) around the estimate of recruits for each model (Adkison and Peterman 1999). We reserved the final year of each data set to allow comparison of the actual observation for this final year with the forecasted value. We compared models for consistency in CVFE and whether the real value for the final observed year fell within the 80% CIs for each model. We then constructed new estimates of CVFE using all the data and generated forecasts one year into the future for summer chum returns in both the Kuskokwim and Yukon rivers. We present yearly forecasts with 80% CIs for the best model and the base model.

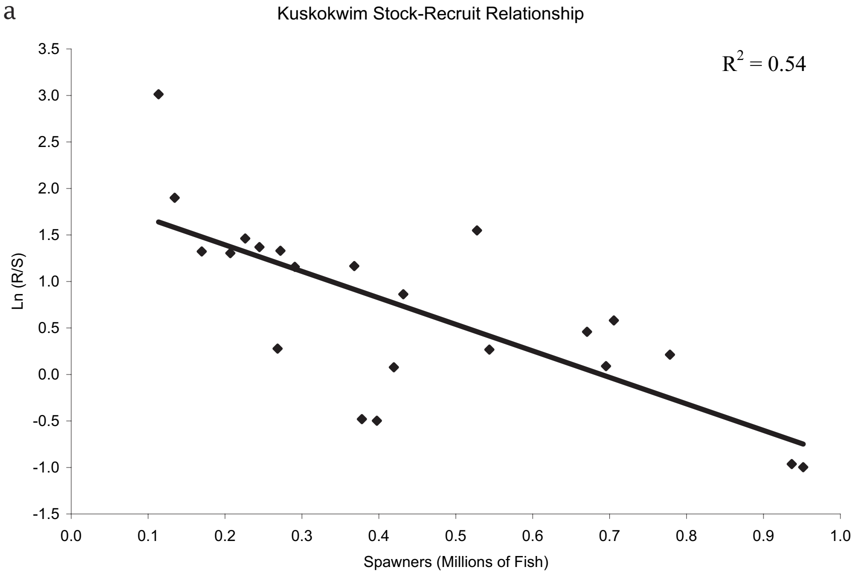
Influence of biological categories: Stepwise models

Our restricted scenarios that constrained all possible models to selecting one or two (early marine residence only) variables from each of the life history categories potentially mask other models with higher explanatory power. We developed the scenarios to reduce the total number of models examined following the exploratory phase. This method also limits the potential for spurious relationships arising from the interaction between variables and pinpoints the most influential variables within each category. It may be that one category is completely dominant and explains more variability than several different categories. We investigate this possibility through stepwise variable selection where variables incorporated into the model are reexamined at every addition or deletion (step) of another candidate variable (Kleinbaum et al. 1998). Again we used S-Plus software (version 6, release 2, Copyright 1988-2001 Insightful Corp.) to apply the step function to the set of variables for each river following the exploratory phase. The process begins with the base model and then performs single variable additions and deletions based on the Akaike Information Criteria (*AIC*) statistic (Burnham and Anderson 1998). If the *AIC* value is lower than the current model, a variable is added or deleted. We report the variables selected and multiple R^2 value for the "best" step model and compare results with that of the category-based model selection.

Results

Initial plots of the observed data suggest a strong stock-recruitment relationship for the Kuskokwim and more of a shotgun pattern for the Yukon (Figs. 2a and 2b). The exploratory phase of our model selection eliminated 10 variables from the Kuskokwim model and 15 variables from the Yukon model (Table 1). This left 6, 9, 2, 4, 4 variables in the Kuskokwim and 6, 7, 1, 4, 3 variables in the Yukon for the freshwater, early marine residence, early marine predators, open-ocean, and open-ocean competition categories, respectively. The second phase of model selection drew different models from these 25 and 21 remaining variables.

The cross-correlation matrix of the variables following the exploratory phase did exhibit some high *r*-values ($0.5 \leq r \leq -0.5$) within a given life history category. In general, high correlations existed between successive lags of a particular variable, namely Asian chum salmon (ACS, ACD, ACR) and coho predators (WAKCF, WAKCJ). Other high correlations occurred between variables in the freshwater and early marine residence categories. This was found between springtime air temperature (ATPF) and streamflow or Yukon Delta ice-out (STPF, STSR, IYDA) and between ocean sea ice cover (ICF, ICS) and winter surface air temperature, strong winds, or sea surface temperature (SATF, SATS, SWMF, SWMS, MSSTF). In the Yukon, there was also a high correlation between winter surface air temperature (SATF) and sea surface temperature (MSSTF) as well as



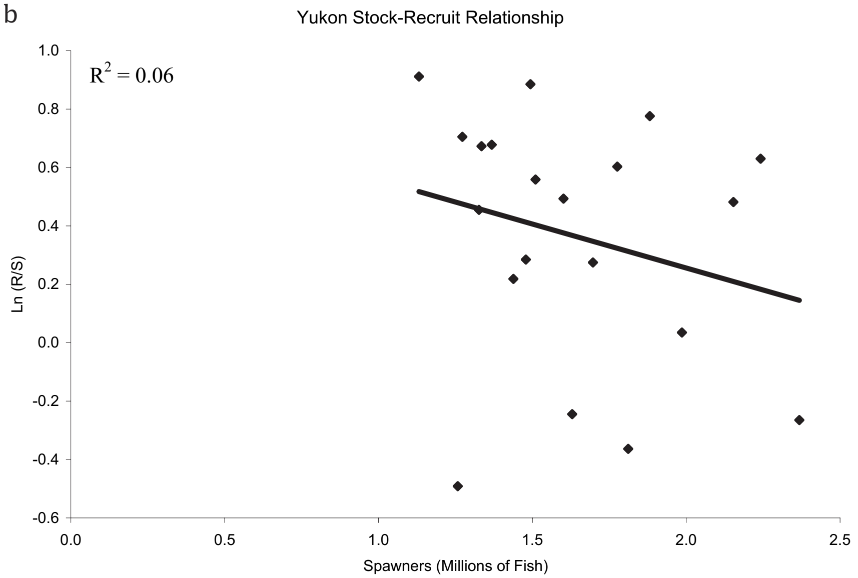


Figure 2. Ln(Recruits per Spawner) base model for Kuskokwim (a) and Yukon (b) rivers. Trendline included is simple-linear regression (equation 1) with no environmental prediction and associated R^2.

between Asian chum salmon (ACD) and Asian pink salmon (APS). These high correlations indicate some redundancy between predictor variables; however, we chose to retain all variables following the exploratory phase. We did this because the most appropriate lag for determining survival in summer chum salmon is unknown. Also mechanisms driving more complex variables such as ocean sea ice cover invariably originate from interactions between simpler physical variables such as air temperature or winds. We expect these somewhat "nested" variables to be highly correlated but again it is unknown which is best for explaining changes in survival.

The set of all plausible category-based models (AIC_c results where $\Delta_i < 3$) are presented in Tables 2 and 3 for the Kuskokwim and Yukon rivers, respectively. For comparison, we also report the base model with no environmental predictors. There were eleven equally plausible models for the Kuskokwim and four models for the Yukon. The "best" AIC_c-selected model included three environmental parameters for both rivers (Tables 2 and 3, model 1). Predictor variables for the "best" Kuskokwim model were the spring Bethel air temperature during the freshwater stage (ATPF), the spring along-peninsula component of wind stress during early marine residence (WUPSF), and the number of Kuskokwim adult coho salmon as early marine predators (WAKCF). The three variables for the "best" Yukon model were the spring precipitation at Tanana station during the freshwater stage (PRPF), the late-spring/summer strong winds at M2 station in the eastern Bering Sea during early marine residence (SWMF), and the summer arctic oscillation lagged 1 year in the open-ocean stage (AOSF).

Model diagnostics and cross-validation

Twelve different predictor variables were contained in the set of plausible models for the Kuskokwim and six for the Yukon; however, we report only on environmental predictor variables that were significant in the models. We present parameter estimates for both rivers along with their respective standard errors and significance values (Tables 2 and 3). All parameter coefficients were consistent in direction across models for a given river, as were the coefficients for variables shared among rivers. There were no significant outliers or trends in the residuals of the plausible models in either river. We discuss possible trends in the predictor variables, differences in the magnitude of parameters, biological realism of parameter values, deviations from assumptions about the distribution of residuals, and autocorrelation in residuals for each river separately.

Kuskokwim model

There was significant autocorrelation in the spawner and Japanese chum (ACS) data at the 95% confidence level. Weak but not significant autocorrelation also existed in the log recruits per spawner (ln(*R*/*S*), coho predators (WAKCF), and winter and summer arctic oscillation (AOWF and AOSF) data. The same three variables were included in every plausible model: spring air temperatures (ATPF), spring wind stress (WUPSF), and coho predators (WAKCF). The parameter estimates for these variables were significant in all models (Table 2) but additional parameter estimates (models 2 through 11) were not significant. The additional environmental variables chosen were primarily in the early marine residence category (Table 1). These were winter surface air temperatures (SATS, model 6), strong winds (SWMS, model 7), winter wind stress (WUPWF, model 8), spring wind stress second year (WUPSS, model 9), and sea surface temperature (MSSTF, model 10). The remaining model variables were in the open-ocean and open-ocean competition categories (Table 1). These were the winter and summer arctic oscillation (AOWF and AOSF, models 2 and 3, respectively), the Pacific decadal oscillation (PDOWR, model 4), and Japanese chum salmon harvest (ACS, model 5). Model 11 contained five environmental variables (primary three, WUPWF, and AOSF). Biologically implausible signs on parameter values were estimated for the winter wind stress in the first year (WUPWF) and summer wind stress in the second year (WUPSS), which appear in models 8, 9, and 11. The base model and models 3, 8, and 11 exhibited slight deviations from normality in their residuals. Slight to significant negative autocorrelation existed in all models at lag 5, with the exception of the base model with slight positive autocorrelation at lag 2. After inspecting Cook's distance plots of residuals, we conclude that the base model and models 1, 4, 8, and 11 fit the most recent years relatively well and other models fit earlier years better.

The relative contribution of each model to the improvement in fit from the base model is shown by the absolute increase in R^2 between the competing 11 models and the base model (Table 2). It is clear from the base model R^2 that the basic spawner-recruit model is fairly informative, explaining 54% of the variability in the data. On average, the percent increase in variance explained by the eleven competing models was 36% from the base model, regardless of the number of parameters. Also, the percent increase in R^2 for the four- and five-parameter models was only 1% to 3% from the best AIC_c three-parameter model (model 1). Sub-models of the best AIC_c-selected model identified the relative contribution of the three key variables: spring air temperatures (ATPF), spring wind stress (WUPSF), and coho predators (WAKCF). Variables ATPF and WAKCF seem to contribute the most alone ($R^2 = 0.63$ and 0.66, respectively). The CVFE across models was very consistent (average = 0.41) with the exception of the base model, for which CVFE was much larger (0.72). The actual observation for 1997 recruits fell within the CIs for the base model and competing models 1, 3, 4, 8, and 11.

Model 1 was the best or preferred model because it had the lowest AIC_c. This model was also the most parsimonious model; had no trends in residuals or deviations from assumptions; had significant, consistent and biologically reasonable parameter estimates; and fit the most recent

Table 2. Summary of equally viable category-based models for the Kuskokwim River.

Kuskokwim parameters	Base	Model 1 (best)	Model 2	Model 3	Model 4	Model 5	Model 6	Model 7	Model 8	Model 9	Model 10	Model 11
Δ AIC_c	20.76	0	2.88	0.50	2.77	2.35	2.45	1.39	2.65	2.86	1.58	2.56
R^2	0.539	0.888	0.895	0.906	0.896	0.898	0.897	0.902	0.896	0.895	0.901	0.918
|>| (%) of base R^2		34.8	35.6	36.6	35.6	35.8	35.8	36.3	35.7	35.6	36.2	37.8
Estimates												
ln(α)	1.96	2.48	2.47	2.40	2.48	2.41	2.38	2.35	2.50	2.48	2.48	2.41
– β	–2.8 E–06	–3.6 E–06	–3.5 E–06	–3.4 E–06	–3.6 E–06	–3.4 E–06	–3.4 E–06	–3.4 E–06	–3.6 E–06	–3.6 E–06	–3.7 E–06	–3.4 E–06
ATPF (1)		–0.40	–0.37	–0.39	–0.38	–0.36	–0.39	–0.35	–0.36	–0.39	–0.37	–0.35
WUPSF (2)		–0.41	–0.44	–0.41	–0.40	–0.41	–0.39	–0.34	–0.43	–0.37	–0.41	–0.44
WAKCF (3)		–0.54	–0.52	–0.47	–0.52	–0.43	–0.51	–0.46	–0.51	–0.50	–0.49	–0.44
Standard error												
ln(α)	0.298	0.199	0.198	0.193	0.197	0.203	0.213	0.210	0.197	0.198	0.192	0.186
– β	5.9 E–07	3.7 E–07	3.7 E–07	3.6 E–07	3.7 E–07	3.8 E–07	3.9 E–07	3.9 E–07	3.7 E–07	3.7 E–07	3.7 E–07	3.5 E–07
ATPF (1)		0.090	0.092	0.085	0.090	0.093	0.088	0.092	0.093	0.089	0.088	0.086
WUPSF (2)		0.090	0.095	0.085	0.090	0.088	0.090	0.097	0.092	0.094	0.087	0.085
WAKCF (3)		0.096	0.097	0.097	0.097	0.130	0.098	0.105	0.098	0.101	0.097	0.097
***p*-value (>[t])**												
ln(α)	**	***	***	***	***	***	***	***	***	***	***	***
– β	**	***	***	***	***	***	***	***	***	***	***	***
ATPF (1)		**	**	**	**	*	**	*	*	**	**	**
WUPSF (2)		**	**	**	**	**	**	*	**	*	**	**
WAKCF (3)		**	**	**	**	*	**	**	**	**	**	**

Δ AIC_c is change in AIC_c from the lowest (best) AIC_c value; |>| (%) of base R^2 is the absolute increase (in percent) from the base model R^2. Only significant variables are presented. Asterisks denote significant values; * = *p*-value ≤ 0.05, ** = *p*-value ≤ 0.0001, *** = *p*-value ≤ 0.0000001. Model 1 is the best AIC_c-selected model. Categories for each environmental predictor variable are shown in parentheses following the variable acronym. Freshwater = 1, early marine residence = 2, early marine predators = 3, open-ocean = 4, and open-ocean competition = 5.

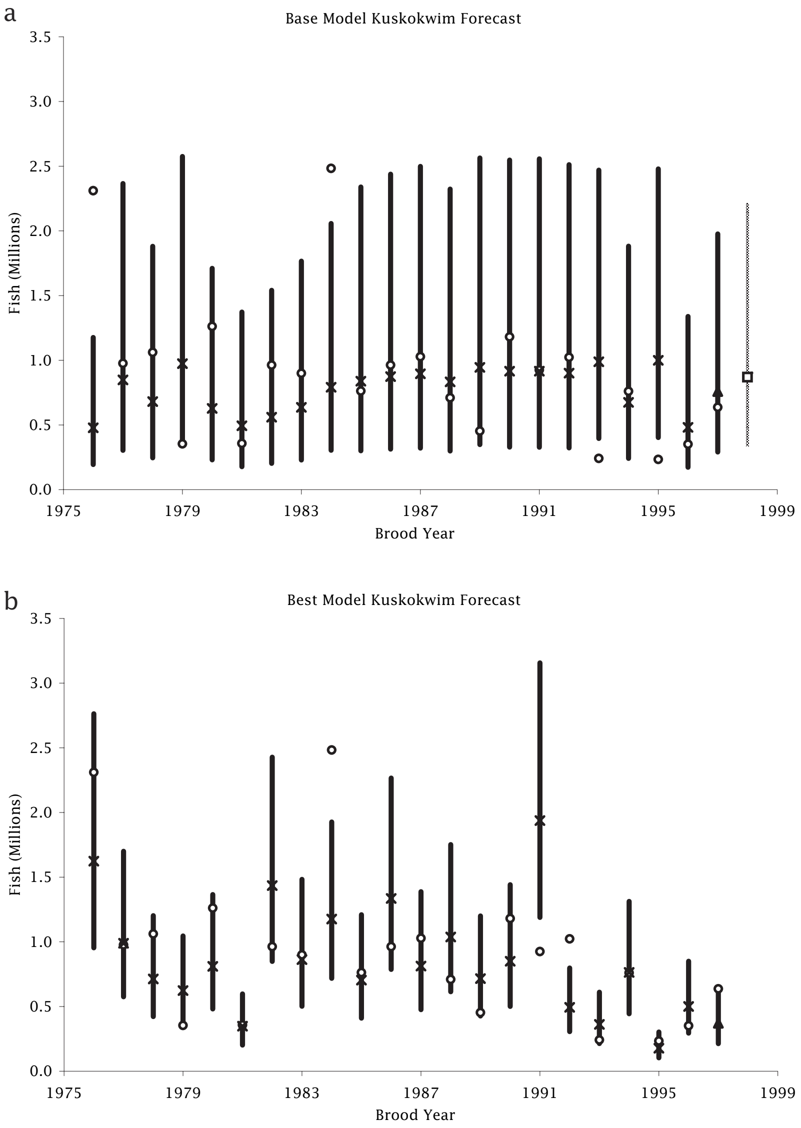


Figure 3. **Kuskokwim forecasts of recruits with predictions based on leave-one-out cross validation technique. Eighty percent forecast error confidence intervals are provided for both the base model (a) and best model (b). Crosses are the predictions, circles are the observations, and triangles are predictions for an observed year without using that year of data in the cross-validation. Squares for the base model are predictions for the unobserved year. Best model 1998 prediction was out of bounds and is not shown here.**

years moderately well. Figures 3a and 3b show yearly forecasts from 1976 to 1996 and include the 1997 forecast in both the best and base model. For the most part, CIs are substantially smaller in the best model versus the base model. We did not, however, include the 1998 forecast in the best model plot (Fig. 3b) because the estimate was well above the previous predicted and observed abundances. We discuss likely reasons for the anomalous 1998 estimate below.

Yukon model

Significant autocorrelation existed only in the Japanese chum and pink (ACS and APS) data, while weak but not significant autocorrelation existed in the strong winds (SWMF) data. For the competing Yukon models, two variables were selected in all models except the base model: spring precipitation (PRPF) and summer arctic oscillation (AOSF). Two other variables were also selected more than once in the competing models from the early marine residence and open-ocean competition category (Table 1). These were strong winds (SWMF, models 1 and 2) and Japanese chum lagged 2 years (ACS, models 2 and 3). The final variable chosen was Japanese pink salmon in the open-ocean competition category (APS, model 4). Parameters of the first three variables were significant in all models (Table 3), while ACS was not significant in either model but very close in one model at 0.056 (Table 3, model 3). An implausible sign for a parameter estimate existed only for Japanese pink (APS) in model 4. The base model and model 4 exhibited slight deviations from normality and the residual-fit (r-f) plot of the base model showed a much greater spread in the residuals than the fitted data. There was only slight (not significant) negative autocorrelation in the residuals of models 2 and 4 at a lag of 5. Plots of Cook's distance demonstrated that all models including environmental parameters explained the most recent years fairly well.

The base model was uninformative, explaining only 6% of the variability in the data (Table 3). Percent increase in variance explained by the four competing models was substantial with an average of 77% from the base model. However, percent increase for the four parameter models was only 2% to 3.5% relative to the best AIC_c three-parameter model (model 1). We generated several sub-models of the variables chosen most often in the AIC_c selection process: spring precipitation (PRPF), summer arctic oscillation (AOSF), strong winds (SWMF), and Japanese chum (ACS). The summer arctic oscillation (AOSF) and Japanese chum (ACS) variables explain the most variability alone ($R^2 = 0.343$ and 0.403, respectively). Values of CVFE were very consistent across all plausible models (average = 0.26), while that of the base model was nearly twice as large (0.45). The actual observation for 1994 recruits fell within the CI's for only model 2, but was very close to the lower bound for models 1 and 3.

Model 1 was the best model because it had the lowest AIC_c. Again, this model was also the most parsimonious, had no trends in residuals or

Table 3. Summary of equally viable category-based models for the Yukon River.

Yukon parameters	Base	Model 1 (best)	Model 2	Model 3	Model 4
Δ AIC_c	20.74	0	0.83	2.77	2.78
R^2	0.061	0.808	0.843	0.827	0.827
|>| (%) of base R^2		74.7	78.2	76.6	76.6
Estimates					
ln(α)	0.86	1.52	1.61	1.63	1.63
– β	–3E–07	–7E–07	–7.6E–07	–7.8E–07	–7.7E–07
PRPF (1)		0.42	0.37	0.39	0.43
SWMF (2)		0.15	0.12		0.16
ICF (2)				–0.12	
AOSF (4)		–0.48	–0.44	–0.48	–0.46
ACS (5)			–0.10	–0.11	
Standard error					
ln(α)	0.47	0.25	0.24	0.25	0.26
– β	2.79E–07	1.49E–07	1.43E–07	1.51E–07	1.57E–07
PRPF (1)		0.084	0.084	0.090	0.083
SWMF (2)		0.047	0.047		0.048
ICF (2)				0.055	
AOSF (4)		0.094	0.092	0.098	0.095
ACS (5)			0.054	0.055	
***p*-value (>[*t*])**					
ln(α)	0.082	**	**	**	**
– β	0.295	**	**	**	**
PRPF (1)		**	**	**	**
SWMF (2)		*	*		*
ICF (2)				*	
AOSF (4)		**	**	**	**
ACS (5)				0.056	

Δ AIC_c is change in AIC_c from the lowest (best) AIC_c value; |>| (%) of base R^2 is the absolute increase (in percent) from the base model R^2. Only significant variables are presented, with the exception of the ACS variable in one model and the base model. Asterisks denote significant values; * = p-value ≤ 0.05, ** = p-value ≤ 0.0001, *** = p-value ≤ 0.0000001. Model 1 is the best AIC_c-selected model. Categories for each environmental predictor variable are shown in parentheses following the variable acronym. Freshwater = 1, early marine residence = 2, early marine predators = 3, open-ocean = 4, and open-ocean competition = 5.

deviations from assumptions, had significant, consistent and biologically reasonable parameter estimates, and fit the most recent years very well. Figures 4a and 4b illustrate the improvement in forecast error between the base and best models from 1975 to 1993 with the 1994 and 1995 predictions.

Influence of biological categories: Stepwise models

The "best" selected model from the stepwise procedure again included three environmental parameters for both rivers. Parameter estimates for both $\hat{\alpha}$ and $\hat{\beta}$ were similar to that reported for the category-based models. However, only two predictor variables for each river were the same as those in the equally plausible category-based models. For the Kuskokwim step model, the best predictor variables were strong winds in the early marine residence (SWMS, model 7), Asian chum salmon in the second year (ACS, model 5), and Asian chum salmon in the third year (ACD, new variable). Parameter estimates for SWMS and ACS were similar to that reported in the category-based models. The R^2 value was slightly lower than the best category-based model at 0.877. In the Yukon, two predictor variables were also chosen in the category-based best model, spring precipitation in freshwater (PRPF, all models) and summer arctic oscillation in the open-ocean (AOSF, all models). The third variable was streamflow at Tanana station in freshwater (STSR, new variable). Parameter estimates for the PRPF and AOSF were similar to the category-based model. The R^2 value for this step model was the same as the category-based model at 0.808.

Discussion

Our structured model-development and selection procedure provided "best" models for the Kuskokwim and Yukon rivers that explained 89% and 81% of the variability in survival rates, respectively. These are good improvements over the base models, which explained 54% of the variability in the Kuskokwim and only 6% in the Yukon. These increases in explanatory power suggest that adding environmental variables to the stock-recruitment relationships of summer chum salmon in these two river systems may dramatically improve future forecasts of recruitment. Additionally, these models were developed using a process that focused on biologically reasonable links between recruitment and environmental conditions. At a minimum, managers may consider the simple trends in these key variables to formulate some idea of potential fluctuations in returns from year to year. The best predictor variables for each system differed somewhat; however, all seem to show changes in recent years that can account to some extent for the decreased returns to these river systems.

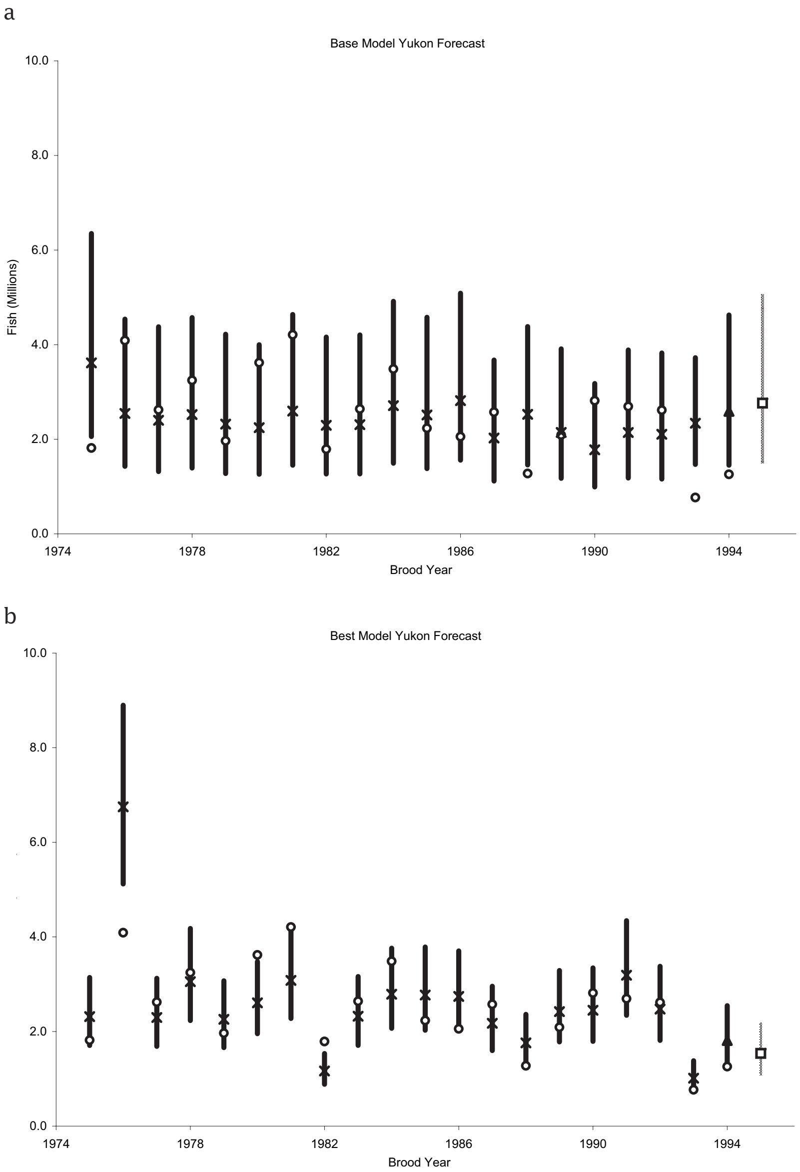


Figure 4. **Yukon forecasts of recruits with predictions based on leave-one-out cross validation technique. Eighty percent forecast error confidence intervals are provided for both the base model (a) and best model (b). Crosses are the predictions, circles are the observations, and triangles are predictions for an observed year without using that year of data in the cross-validation. Squares are predictions for the unobserved year.**

The best category-based predictors for Kuskokwim chum salmon were springtime air temperature during the freshwater life stage (ATPF), along-peninsula wind stress (WUPSF) during early marine residence, and Kuskokwim coho adult predation (WAKCF) during early marine residence. The relationships between these variables and survival rate were all negative. We expect survival to decrease due to increased predators; however, the other negative relationships may not be obvious. To the extent that air and water temperatures during the freshwater life stage are related, increased temperature may be associated with decreased survival due to increased metabolic rates of salmon fry and decreased dissolved oxygen (Salo 1991). Alternatively, given that monthly air-temperature anomalies are often highly autocorrelated, the time period over which we defined the air temperature index may simply represent a different process that occurred slightly earlier or later in the year. An opposite effect of increasing air temperature is found in the ocean environment. Ocean surface air temperature is typically highly correlated with sea surface temperature, which was found to exhibit a positive relationship with survival rates of Alaska salmon as warm temperatures in the North Pacific Ocean often imply good feeding conditions (Mueter et al. 2002a). We also found a positive correlation between residuals of the base model and several of the temperature-related variables we considered (St. Paul air temperatures (SATF), and Mooring 2 sea surface temperatures (MSSTF), Table 1), but these did not show up in the competing models. Therefore, air temperatures will influence survival of chum salmon in altering directions depending on the life history categories. Additionally, fluctuations in freshwater air temperatures are more drastic than ocean air temperatures thereby producing a more direct and stronger influence on chum salmon survival.

Negative values of the along-peninsula wind stress are related to strong winds from the east. These are associated with northward transports through Unimak Pass that are thought to pulse nutrient-rich water from Bering Canyon into the inner front or “green belt” region of the Bering Sea continental shelf (Stabeno and Hunt 2002). We expect a negative relationship with this index and survival as negative values would imply more nutrient rich water transported into the eastern Bering Sea and allow more primary and secondary production. The summer index (WUPSF) has moderate interannual variability but does contain one recent exceptionally positive year, 1998, where anomalous strong west winds forced southward transports through Unimak Pass. This variable may be explaining recent changes in the Bering Sea relatively well as concerns Kuskokwim chum salmon and, therefore, may be an important indicator of survival rates.

The best category-based Yukon predictors were springtime precipitation in freshwater (PRPF), strong winds at station M2 in the Bering Sea (SWMF), and the summertime arctic oscillation index (AOSF) again all in the first year. The directions of relationships for the first two predictors were positive while the AOSF sign was negative. We expect strong winds to mix new nutrients into the euphotic zone and so produce better feeding conditions for salmon smolts. We may also expect the arctic oscillation relationship as positive values are associated with relatively mild, warm conditions and net melting of the Arctic Sea ice pack (Rigor et al. 2002), reflecting current trends in the Bering Sea when survival of summer chum salmon was low. We may expect a negative or positive relationship with precipitation and survival rates. High precipitation may mean increased streamflow and more scouring and destruction of redds, therefore decreasing survival. However, precipitation and streamflow indices are only moderately related in our variables and this makes sense in a dry continental climate like the Yukon interior where there is little overall precipitation. In this case, we may expect a positive relationship as more precipitation could imply increased nutrients in the streams through runoff and perhaps more turbid waters that allow predator avoidance.

An important application of the environmental models is that they help explain the lowered survival of the past few years much better than the base spawner-recruit model. Recent trends in the predictor variables suggest warm, mild conditions in the Bering Sea. Freshwater temperatures were warmer (except 1999), anomalous 1998 southward transport, fairly high coho predators (up until 1996, then decreasing to the anomalously low year of 1999). Precipitation and wind speeds were lower, and there were large swings in the summertime arctic oscillation. Asian chum salmon have increased systematically over time, implying increased potential for competition. These indicators all suggest lowered survival rates in these two river systems in recent years.

In conjunction with this better fit, the CVFE for the best models in both systems was much smaller than that of the base model. This is useful to managers as there are tighter bounds about the point estimate implying a better understanding of the stock-recruitment relationship. The prediction for 1995 brood year returns to the Yukon was 1.5 million fish, in line with recent returns. The predicted return for the Kuskokwim was anomalously high. We attribute this to an extremely low return of coho predators that corresponded to an approximate 93% reduction from the average coho harvest over the set of years we considered (1975-2001) and out of the previously observed range of data. Other variables in that year were also indicative of good survival conditions and the spawner biomass was fairly large. This resulted in a very large (and likely unrealistic) prediction. Walters (1989) found through numerical simulation that short-term forecasts explaining less than 60-80% of recruitment variation should basically be ignored because a conservative adaptive management program would be just as effective. In the Kuskokwim case, the more simplified base model, which alone explained a large proportion of the variation in recruitment, may be preferable for forecasting.

In contrast, the Yukon base model explained almost no variation, and the large improvement with the environmental predictors suggests that the environmental model should not be abandoned. When considering the utility of environmental information in the recruitment models we emphasize the need for a precautionary approach to modeling particularly under extremely anomalous conditions. Regardless of the situation, in-season management will rely to some degree on the quality of the pre-season forecast and the estimates may be used as a guideline for long-term productivity changes (Walters 1989).

There is some concern for the potential of this modeling procedure to include spurious relationships due to the high number of models evaluated. There were 34 and 36 initial variables considered for predicting chum salmon survival and the number of permutations of these variables is intractable. We reduced the total number of candidate models by developing restricted scenarios; however, this did not decrease the chance of developing a spurious relationship from any one variable. We, therefore, chose our variables based on ecologically plausible relationships of summer chum salmon survival. The restricted scenarios substantially reduced the possibility of spurious relationships arising from the interaction between variables. We can see the result of no constraints by inspecting the "best" model selected for the Kuskokwim through the stepwise variable selection procedure. Two of the three predictor variables were Asian chum salmon at the 2 and 3 year lags. These two variables are highly autocorrelated. Even though this model contains similar explanatory power as the category-based model, it lacks in biological significance since there is high redundancy in the predictor variables. Our restricted scenarios do buffer against this occurrence; however, we anticipate that the explanatory power of our best model is overstated. Our suggestion is to simulate the environmental model selection procedure to gain a more reliable understanding of explained variability.

We also feel that the stepwise selection procedure may be very informative as a straightforward methodology when variables chosen are biologically meaningful. The best step model for the Yukon where two freshwater variables were chosen (precipitation and streamflow) had equal explanatory power as the best category-based model. In this case, these variables were only moderately positively correlated and may reflect different environmental pressures on chum salmon survival. We suggest exploring both types of methodologies. The category-based procedure explicitly defines major environmental pressures throughout the life history of chum salmon producing a general idea of important predictor variables to monitor. The stepwise procedure will identify the most influential life history stages where future studies could concentrate research efforts.

Finally, it is important to consider the various sources of error in the spawner and recruit indices, the age data, and the environmental data.

Often one expects environmental data to be autocorrelated, and in the plausible models evaluated Asian chum salmon, Asian pink salmon, and Kuskokwim spawners exhibited significant autocorrelation. Inflation of significance in parameter estimates occurs when both dependent and independent variables contain significant autocorrelation (Chatfield 1984). However, there was no significant autocorrelation in the dependent variable of log recruits per spawner; therefore, we do not expect a change in the explained variability of the different models with respect to influence of autocorrelation in the independent variables. Autocorrelation in the residuals of a particular model may be explicitly accounted for by including a *p*th order autoregressive term in the parameters (Quinn and Deriso 1999). We found strong negative autocorrelation in the model residuals at a 1 year lag in all of the competing Kuskokwim models. If one were to suspect this trend to be real, models could include an autocorrelation term such as an ARIMA model to account for this variation. Additionally, the autocorrelation in Kuskokwim spawners may indicate that sibling catches from year to year are another potential variable to consider in this modeling framework. Geiger and Hart (1999) found that last year's catch was fairly useful in predicting summer chum returns for a guideline in the south peninsula June sockeye fishery.

The estimates of abundance and escapement have inherent error associated with them from the estimation process (Shotwell and Adkison 2004). Specifically, measurement error in the escapement data may have a large effect on the reliability of these estimates depending on the properties of the multiple data sources used in the analysis. The age composition data was fairly difficult to obtain and estimates were made from very heterogeneous data due to limited sampling. These sources of error should be considered when developing biological escapement goals or harvest strategies from these estimates. We report standard errors for the model parameter estimates. However, direct use of these values for productivity and density-dependent parameters can be complicated when calculating management parameters such as optimal stock size and optimal harvest rates (Hilborn and Walters 1992). Annual estimates of precision through mark-recapture studies, errors in variables approaches, and mixed error models such as the Kalman filter are all potential methods for considering measurement and process error in these models (Hilborn and Walters 1992, Quinn and Deriso 1999). Simulation analyses of measurement and process error that replicate the fishery and environmental conditions of these two river systems should also be conducted to fully understand potential biases in estimating management parameters (Mundy et al. 2001). Managers should use a precautionary approach to these model results and incorporate adaptive management (Mundy et al. 2001) by consistently reevaluating the environmental predictor variables, model selection, and forecasting estimates when new data becomes available.

The environmental variables identified in both the category-based and stepwise models pinpoint key areas for future research. Coho predation was a large predictor of chum salmon survival in the Kuskokwim model and coho have been shown to feed on juvenile chum salmon along their seaward migration (Orsi et al. 2000). Almost nothing is known about the major predators of Kuskokwim and Yukon river summer chum juveniles in the early marine environment (C. Zimmerman, U.S. Geological Survey, pers. comm.). Investigations that sample regularly in the delta region of both rivers could record timing of ocean entry for smolts and characterize major predators and primary diet. Clearly fluctuations in the freshwater environment are very important in determining summer chum salmon survival, particularly for the Yukon. There is relatively little known about the particular characteristics of the spawning grounds or incubation habitat of summer chum salmon along the Kuskokwim and Yukon rivers (C. Zimmerman, U.S. Geological Survey, pers. comm.). Studies concentrating on water quality and food availability throughout index streams or development of remote sensing analysis would be very useful for identifying specific freshwater variables for continued monitoring. Finally, other variables might be considered useful for inclusion in these models that are not necessarily life-history based. Mundy et al. (2001) suggest the use of all available information to supplement the results of formal modeling approaches such as presented in this paper. The focus of this study was to identify key variables that influence summer chum survival, and to factor these variables into run abundance forecasting models. Our forecasting model may be improved by also considering the influence of interception fisheries, such as the south peninsula June sockeye fishery (False Pass or area M).

Given these considerations, we find inclusion of appropriately defined environmental variables to be useful for stock-recruitment analysis and forecasting even in this data-limited situation. We were able to apply many environmental indices to this system. Most of these variables are fairly easy to acquire and are constantly updated. In situations like the Kuskokwim and Yukon rivers, where populations seem to be dwindling for unknown reasons, it is important to be able to generate an understanding of the potential mechanisms involved and to formulate a modeling procedure that accounts for these changes.

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Surf Smelt (*Hypomesus pretiosus*) in Burrard Inlet, British Columbia: A Limited Data Assessment to Address Concerns about Potential Recreational Overharvesting

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Abstract

Surf smelt (*Hypomesus pretiosus*) occur throughout temperate coastal regions of the Northeast Pacific. Our understanding of the biology, distribution, and abundance of this species is poor. Within Burrard Inlet, adjacent to metropolitan Vancouver, small local fisheries that have operated for over a century continue. During the early 1900s most smelt were taken in small, commercial fisheries for local consumption. Gradually, commercial fisheries diminished, and were replaced by a rapidly expanding recreational fishery that peaks during spring and summer months on surf smelt spawning beaches. Because of the many uncertainties associated with the biology and fisheries for this species, some managers have expressed concern about fishery sustainability and potential recreational overharvesting. Surf smelt stocks in British Columbia previously have not been assessed. In this paper, we develop simple methods to (1) estimate spawning stock biomass based on measurements of spawn deposition and (2) estimate recreational catches based on favorable fishing times, as creel surveys are not available. Also, we examine surplus production models to estimate fishery parameters. These analyses indicate that recent annual surf smelt catch in Burrard Inlet may remove up to 40% of the potential spawning biomass, a high level for a short-lived, iteroparous species, especially as the estimated annual natural mortality rate is around 0.45. In general, results from our analyses indicate that the Burrard Inlet surf smelt population is not being overharvested but some

concerns regarding recreational harvest levels might be justified and verification of actual recreational catches should be a priority.

Introduction

The precautionary approach to fisheries management attempts to conserve stocks by using scientific advice to evaluate harvest strategies (FAO 1995). Thus, management plans need clear objectives and include efforts to monitor and assess the effects of harvesting on a stock. This precautionary approach poses a potential dilemma for fisheries focused on underexploited stocks where data often are limited (as extensive data are normally a prerequisite for providing scientific advice). For mature or senescent fisheries, scientific advice usually is based on two general model types, although others exist. A common assessment tool for many fish stocks is a virtual population analysis (VPA) or age-structured model (ASM) (e.g., Fournier and Archibald 1982). Alternatively, when age composition data are unavailable, an escapement model that relates spawn to spawning stock biomass (SSB) can be used. To be applied appropriately, any assessment method requires biological understanding of the target species and detailed population and catch data for modeling. For small fisheries, our biological understanding is minimal and detailed population data often do not exist. Thus, an alternative assessment strategy that is consistent with the precautionary approach is needed. In this paper, we describe such a strategy for a small surf smelt (*Hypomesus pretiosus*) fishery in British Columbia.

Surf smelt are small, silvery, pelagic schooling fish that are important prey for larger predatory fish, marine mammals, and birds (Penttila 1995). They occur throughout coastal regions of the eastern Pacific Ocean from California to Alaska (Hart and McHugh 1944, Hart 1973) and are targeted by both commercial and recreational fisheries in British Columbia that coincide with spawning during summer months at spawning beaches. The largest commercial catches occurred during the early 1900s with catches exceeding 200 t, but since then the fishery has steadily declined such that current commercial catches rarely exceed 5 t (Fig. 1). The recreational harvest is unknown but anglers are allowed to harvest 20 kg per day under the British Columbia Tidal Waters Sport Fishing Guide with no annual catch limit. Thus, the recreational harvest could be substantially greater than the commercial harvest given increased interest by recreational anglers. This is especially true for metropolitan areas such as Vancouver, British Columbia, adjacent to the Burrard Inlet surf smelt stock that has supported local fisheries for over a century (Anderson 1880). It is unclear whether current harvest levels are sustainable in British Columbia since there has been little research and no formal assessment to estimate current catch or spawning biomass.

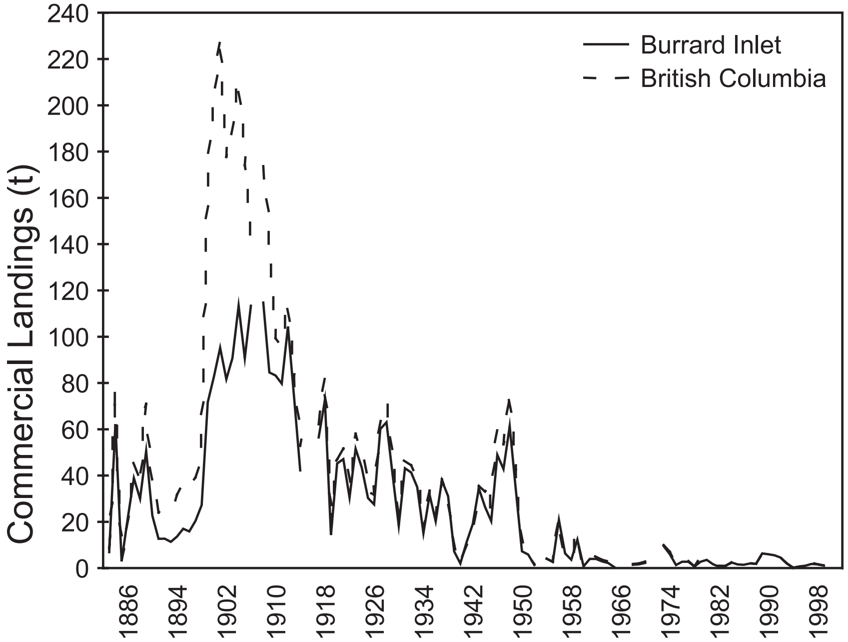

Figure 1. Commercial surf smelt catches for Burrard Inlet (DFO statistical management areas 28 and 29) and British Columbia (all management areas) between 1886 and 2002.

Kilambi (1965) hypothesized that coastal ocean and estuarine populations from Puget Sound were genetically distinct based on glacial movements and resultant division and distribution of osmerid populations over 13,000 years ago. Spawning time, parasite incidence, meristic and morphological characteristics, and serological analyses each support this hypothesis (Kilambi et al. 1965, Kilambi and DeLacy 1967). Thus, surf smelt stocks in Burrard Inlet are potentially similar to those in Puget Sound and different from other populations in coastal British Columbia such as those from San Juan Inlet on the west coast of Vancouver Island, and Prince Rupert Harbour in northern British Columbia.

In British Columbia, surf smelt are harvested both commercially and recreationally. Thus, a measure both of commercial and recreational harvest is needed. Commercial surf smelt fisheries have a logbook requirement but there is no similar information for recreational resource users. In this paper we develop a method to estimate recreational harvest when data are limited (or nonexistent). Further, we develop an egg production method based on spawn deposition to estimate stock size.

Commercial fisheries

Landings from British Columbia commercial fisheries between 1886 and 2002 vary over time. Catches increased during the late 1800s and early 1900s with a maximum catch of just over 230 t in 1904 (Fig. 1). Since this peak, the fishery has steadily declined, most notably since the mid-1950s. A combination of increased fishing pressure and habitat loss due to increased human population and industrialization (i.e., oil refineries, mills) have contributed to the reduction of surf smelt around the Lower Mainland, especially English Bay and Burrard Inlet, since the 1920s (Fig. 1; Motherwell 1922). Also, the percentage of smelt landed from the Vancouver area has changed over time. Early in the fishery, large quantities of surf smelt were landed from areas other than Vancouver, but between the 1920s and present almost the entire catch has come from this one part of the coast (Fig. 1).

Since 1984, logbook submissions have been a required licensing regulation. However, since this regulation was implemented, compliance has averaged 36.9%. It is unclear if fishers who did not submit a logbook decided not to fish or simply did not comply with the licensing regulation. Thus, we suspect the actual commercial catch is underestimated, but only slightly, as we believe most fishermen who did not submit a logbook decided not to use their commercial license. One requirement of the logbook is to record the number of hours spent fishing (a measure of effort). Unfortunately, this requirement often was not met, as many fishermen submitted logbooks that had no measure of fishing time. Most fishermen reported their harvest by day (as required) so a measure of effort was available, although such an estimate is less desirable than the number of hours spent fishing. The timing of most surf smelt fishing corresponds to high tide events so we assumed that the number of hours spent fishing is likely relatively consistent among fishermen and days, such that effort measured as fishing days is a reasonable proxy for hours of fishing effort.

We estimated both the average and the median CPUE based on logbook data to establish a standardized time series for the surf smelt fishery in Burrard Inlet. The recreational harvest of surf smelt is suspected of being substantially greater than the commercial harvest (see below) so using only commercial logbook data will provide a conservative estimate for estimated parameters, including MSY. Time trends were similar between the two measures of CPUE (Fig. 2) and good contrast was observed in trajectories of mean and median CPUE versus fishing effort (Fig. 3). These contrasts suggest that a surplus production model could be used to estimate stock parameters. We used available commercial data for surf smelt landings to estimate parameters for a biomass dynamic model by transforming equations into linear form. Hilborn and Walters (1992) provide the basic relationships:

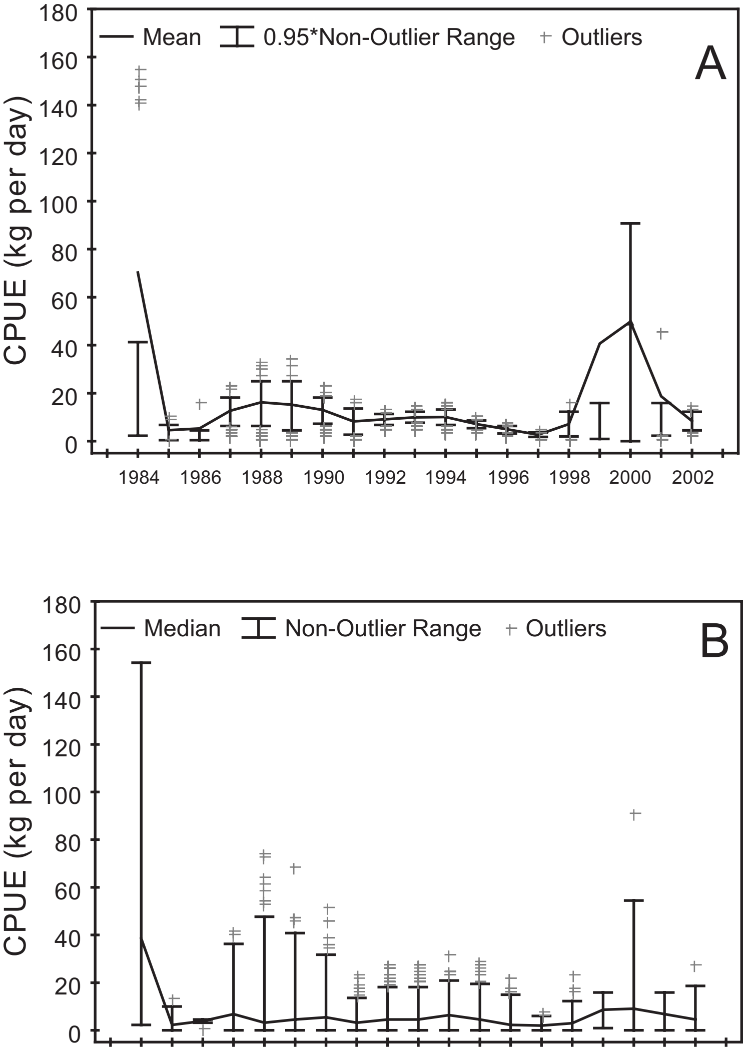

Figure 2. Comparison of mean (A) and median (B) CPUE by fishing year including outliers and non-outlier ranges (whiskers) for the Burrard Inlet surf smelt fishery between 1984 and 2002.

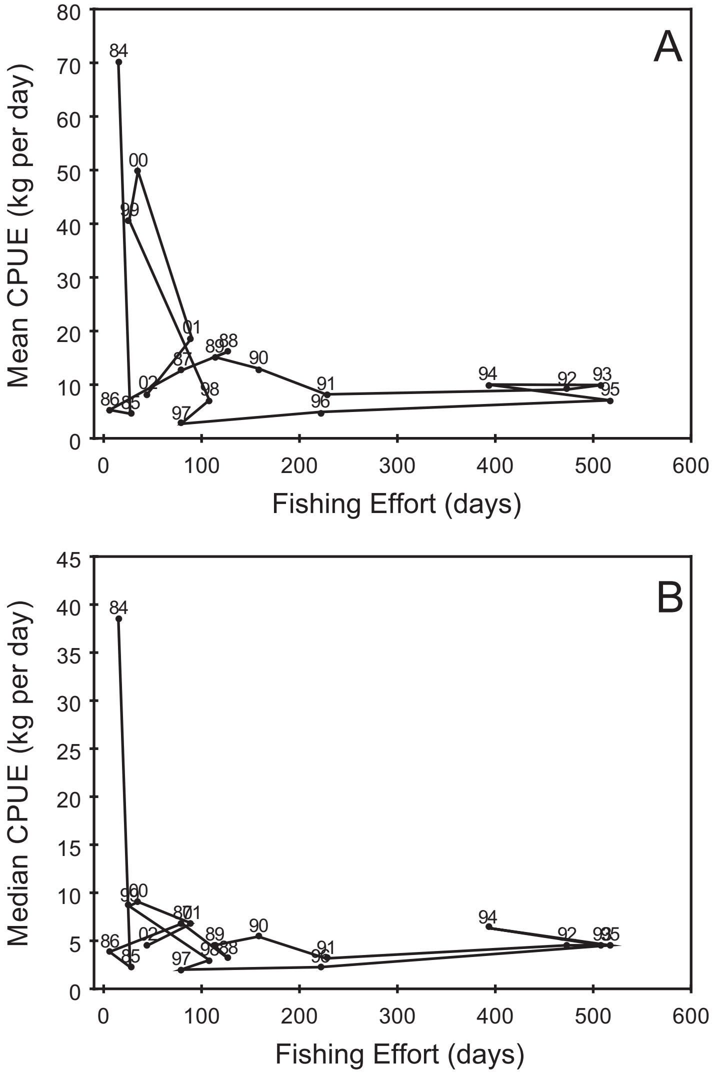

Figure 3. **Trajectories of the mean (A) and median (B) CPUE versus fishing effort for the Burrard Inlet surf smelt fishery between 1984 and 2002.**

$$B_{t+1} = B_t + rB_t\,[1 - (B_t \times k^{-1})] - qE_tB_t \tag{1}$$

and

$$B_t = U_t \times q^{-1} \tag{2}$$

where B is biomass (kg), U is CPUE (kg × days^{-1}) and assumed proportional to stock abundance, E is fishing effort (days), r is the intrinsic rate of population growth, q is catchability, and k is the unfished equilibrium stock size. By substitution and re-arrangement of the above relationships the equation becomes:

$$(U_{t+1} \times U_t^{-1}) - 1 = r - \{[r \times (kq)^{-1}] \times U_t\} - qE_t \tag{3}$$

By solving the multiple regression equation corresponding to equation (3), we estimated r, q, k, and MSY ($rk/4$) for each estimate of CPUE (Table 1). Further, equation (2) can be solved for B_t for each fishing year based on U_t and q estimated in equation (3). Using this estimate of biomass and the reported catch, an exploitation rate was determined (Exploitation Rate = [catch/biomass] × 100). Exploitation peaked in the mid-1990s based on both mean and median estimates of CPUE (Fig. 4). During this period, overharvesting was possible but exploitation rates have since declined. The estimates of stock carrying capacity were 13 and 14 t using the two estimates of CPUE. However, it is important to note that presently this population is heavily exploited so current stock biomass is assumed to be less than carrying capacity shown in Table 1.

Estimates of spawning biomass

An alternative to surplus production models is an estimate of spawner biomass based on estimates of spawn deposition. Models based on egg production have been used widely in fisheries science (e.g., Gunderson 1987). We develop simple methods for escapement estimation that could be used for future assessments of surf smelt spawning biomass. The model combines three independent estimates: (a) spawning area; (b) egg density; and (c) relative fecundity (number of eggs per gram of spawning fish) to determine stock biomass. This method was used to provide the first relative measure of stock biomass for Burrard Inlet surf smelt. Where multiple sources of data exist, we report the mean and range (minimum and maximum) for each variable used in the model. To estimate spawning area we used information from commercial fishermen and published reports (i.e., Levy 1985) to identify spawning beaches and the corresponding length of spawning shoreline around Burrard Inlet (DFO statistical management areas 28 and 29). Also, we used data from biophysical surveys of Burrard Inlet to estimate the approximate width

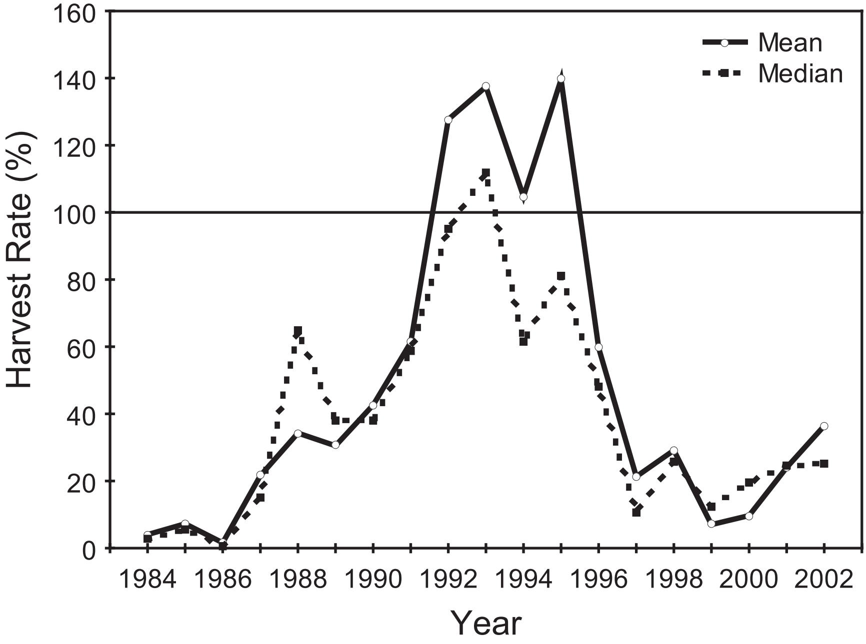

Figure 4. Exploitation rates for the Burrard Inlet surf smelt fishery between 1984 and 2002. Exploitation rates were calculated based on biomass estimates from the surplus production model using both mean and median CPUE data.

of spawning locations with suitable spawning substrate (Casher and Roberts 1992). The estimated total potential spawning area is 29,838 m^2 and represents the maximum area available. This estimate of spawning area was corroborated with a Geographical Information Systems (GIS) analysis within 20%, a range we will use in our calculations. Also, studies in Puget Sound suggest surf smelt only use approximately 20% of what researchers perceive to be available substrate (D. Penttila, Washington Department of Fish and Wildlife, pers. comm.). Therefore, the estimate of actual spawning area used in any year would be about 5,968 m^2 (or 20% of 29,838 m^2) ranging between 4,774 m^2 and 7,161 m^2. Based on Washington state surveys, the estimated egg deposition depth is about 0.025 m so the corresponding volume of spawning substrate would be 152 m^3 (5,968 $m^2 \times 0.025$ m). However, eggs might be deposited shallower or deeper depending on actual beach conditions, beach slope, tidal conditions, etc. Thus, we assume a range between 0.013 m and 0.038 m, a range that allows eggs deposited too shallow to die due to limited protection from the elements, and those deposited too deep to die due to physiological stress. The corresponding volume of spawning substrate then ranges between 61 m^3 and 273 m^3. Egg density surveys have not been conducted for British Columbia populations, but in Puget Sound Wildermuth (1993) observed an egg density of 1.24 eggs cm^{-3} at Ross Point. Penttila (1978) reported much higher densities in other areas, ranging between 15 and 150 eggs cm^{-3}. Due to changes in spawning activity over time we assume the mean density of eggs to be 75 eggs cm^{-3} ranging between 1.24 and 150 eggs cm^{-3}. It should be noted this variable introduces considerable uncertainty due to the wide range of measured egg densities reported. The corresponding egg deposition would be 1.14×10^{10} eggs, ranging between 7.5×10^7 eggs (based on smaller available area, shallower egg deposition depth, and minimum egg density) and 4.1×10^{10} eggs (based on larger available area, greater deposition depth, and maximum egg density).

Relative fecundity for Fraser River surf smelt (adjacent to Burrard Inlet) was estimated as 556.5 eggs g^{-1} for females with a range between 455 and 671 eggs g^{-1} for females. Assuming a 1:1 sex ratio and approximately equal weights among sexes, the fecundity data corresponds to 278.25 eggs g^{-1} for both sexes with a range between 227 and 335.5 eggs g^{-1} for both sexes (spawning fish). Therefore, the estimated spawn deposition would correspond to a spawning biomass of 41 t (1.14×10^{10} eggs/278.25 eggs g^{-1}), ranging between 0.2 t (minimum number of eggs deposited and maximum relative fecundity) and 180 t (maximum number of eggs deposited and minimum relative fecundity).

Recreational harvest

Recreational fishing for surf smelt has increased significantly over the last decade, especially on beaches of the Lower Mainland. In Canada, the laws governing foreshore rights provide the public unlimited access to most beaches in British Columbia. Easy access and ample fishing opportunities make gillnetting for surf smelt a popular recreational fishery. Since the current management plan does not consider recreational harvest, likely a significant portion of surf smelt landings, we provide a working estimate of the recreational harvest using some general assumptions. As with estimating spawning biomass, this is a methodological approach and parameters must be measured to provide reasonable estimates of the recreational harvest. High evening and weekend tides attract the greatest number of fishermen (D. Levy, Hatfield Consultants Ltd., pers. comm.). There are 77 evening and weekend high tide events (Monday to Friday, 3:00 pm–8:00 pm, Saturday and Sunday 8:00 am–8:00 pm), between mid-May and the end of September in Burrard Inlet. Of those 77 fishing opportunities, 27 fall during the fishery conservation closure leaving 50 possible fishing opportunities. Weather also affects the ability and desire to fish. Assuming that an additional 25% of these opportunities will be lost due to weather, 37.5 fishing opportunities remain. There is an estimated 13,800 m of shoreline used by recreational fishermen. On a good night, nets are set every 2 m (D. Levy, pers. comm.) but not every location is used equally, as some locations are very popular while others are less popular. Thus, we assume one fisherman every 50 m. Using an average catch of 56 fish per trip (D. Levy, pers. comm.) and an average weight of 22.67 g per fish, the estimated recreational harvest would be 13.2 t. It is important to note that this estimate does not take into account several important elements. First, average catches used in this estimate come from an area known to be one of the most productive for recreational fishing and it is unknown whether this level of fishing success would be equaled in all areas. This assumption would tend to overestimate recreational catches if fishing success was lower in other areas. Also, it is likely many users are fishing outside of the preferred fishing areas used in our harvest estimate, an assumption that would tend to underestimate the actual recreational harvest. Finally, this estimate assumes no fishing opportunities during the conservation closure and there are many reports by fishermen, the Greater Vancouver Regional District, and Conservation and Protection (DFO) that fishing during the closure is common, so again, estimated recreational landings are likely underestimated.

We are unaware of any data on sources or rates of mortality for surf smelt in British Columbia. Robson and Chapman (1961) proposed a simple method of estimating total annual mortality for a population based on age-frequency distributions. The length-frequency distribution for Burrard Inlet surf smelt suggests two distinct age-classes corresponding to age-1 and age-2 fish, respectively (Fig. 5). These observed size modes correspond closely to size modes observed for another smelt, the eulachon (*Thaleichthys pacificus*) sampled from offshore waters (Hay and McCarter 2000). The total annual mortality for Burrard Inlet surf smelt using the Robson and Chapman method is 0.85. Based on the generalized spawn deposition (escapement) model estimated biomass of 41 t and a total harvest of 15 t (recreational 13 t plus commercial 2 t), the fishing mortality would be 0.37 (15 t/41 t). Assuming that the fishery occurs over a relatively short period allows the assumption that the interaction term between fishing mortality and natural mortality becomes zero. Thus, the estimate of natural mortality is simply the difference between total mortality and fishing mortality or about 0.48. Alternatively, the carrying capacity estimated by the more conservative surplus production model (median CPUE) is 13 t and if we assume the recreational harvest is double the commercial harvest (since we believe the recreational harvest is substantially greater than the commercial harvest), we can estimate fishing mortality. The average commercial harvest between 1984 and 2002 was 1.8 t, so the total harvest would be 5.4 t. Thus, a 5.4 t harvest of an estimated 13 t stock corresponds to an exploitation rate of 0.42 and a corresponding natural mortality rate of 0.43. However, since we believe

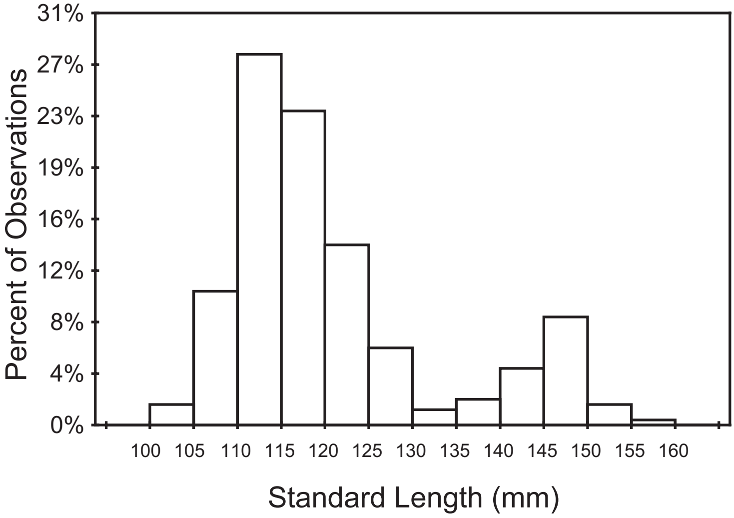

Figure 5. Length-frequency distribution for surf smelt collected from Burrard Inlet in fall 2001 showing a higher proportion of age-1 surf smelt (left mode) compared to age-2 surf smelt (right mode).

the stock is below carrying capacity due to exploitation, fishing mortality is likely underestimated and natural mortality overestimated. Regardless, the average exploitation rate derived from the surplus production models between 1984 and 2002 was 0.47 (mean) and 0.39 (median) and in good agreement with the other estimates of exploitation.

Discussion

Our knowledge of surf smelt in British Columbia is data limited. To make credible science-based assessments for surf smelt, more basic knowledge and biological data are required. Deficiencies in biological understanding include the limited information on distribution, pre-spawning and spawning biomass, fishing, and natural mortality rates. Deficiencies for management-related issues include our ignorance about the impact of commercial and recreational fishery gear and the bycatch of non-target species (i.e., salmon, surfperch, herring, rockfish, etc.). The most extensive surf smelt fisheries in British Columbia occur in Burrard Inlet (Vancouver) where harvester impacts are unknown. Because of the limited understanding about this species (and its fishery) we advocate the adoption of a precautionary management plan both for commercial and recreational users (FAO 1995). Such a plan should include strict enforcement of regulations, limited effort and catches for both commercial and recreational users, and the inclusion of a biologically based sampling program as recommended by Perry et al. (1999) for new and developing fisheries.

Limited (or poor) data do not have to prevent an initial assessment for a stock where relatively little is known. For example, egg production methods have been widely used in fisheries research in various forms (Gunderson 1987). Here, we developed an escapement methodology based on hydrographic charts and basic biological information about the species (relative fecundity). This methodology easily could be adapted for other beach spawning species such as Pacific sand lance (*Ammodytes hexapterus*). Further, until recently, an escapement method was used for the stock assessment of Pacific herring (*Clupea pallasi*) in British Columbia (Schweigert 1993).

The estimation of catch for some recreationally exploited fish has proven problematic. For example, a creel survey for Burrard Inlet surf smelt was initiated in 2003 only after Therriault et al. (2002) identified possible concerns regarding the status of the stock. Our estimate of recreational harvest is based on assumptions about the potential desire for fishermen to go fishing and could be modified for a variety of recreationally harvested species. For example, by identifying preferred fishing times, locations, dates, method of harvest, etc., it may be possible to estimate useful measures of effort. When combined with some measure of harvest per unit of effort, the total recreational harvest can be estimated. Of course variability would increase with increasing uncertainty associated with estimated parameters but nevertheless could be useful in providing a crude index of exploitation (e.g., low, medium, high).

Based on reported commercial catches, the Burrard Inlet stock has decreased substantially since its peak in the early 1900s: landings averaged less than 2 t in recent years (Fig. 1). Further, our simple calculations provide an estimated annual fishing mortality (commercial plus recreational) of about 0.40 and a corresponding annual natural mortality around 0.45. If we assume that $F_{MSY} = M$ (cf. Hilborn and Walters 1992), then it appears that current catch rates of surf smelt in Burrard Inlet are substantially lower than catch rates at MSY and concerns about overfishing are not supported. Further, if correct, this natural mortality estimate is similar to another pelagic species, Pacific herring, in British Columbia although the fishery mortality is considerably lower for herring, less than 0.20 (Schweigert 2002). Thus, a fishing mortality around 0.40 could be high for an iteroparous species.

Although we acknowledge limitations associated with the data, we evaluate this fishery with respect to some generally accepted fishery reference points. One comparison is the current stock size with the biomass expected at MSY (here $k/2$). Based on the surplus production models, the

Table 1. Estimates of the intrinsic rate of population growth (*r*), catchability (*q*), unfished equilibrium stock size (*k*), and maximum sustainable yield (MSY) based on surplus production models of commercial surf smelt fishery data (mean and median CPUE).

Estimated parameter	Mean CPUE	Median CPUE
r	1.358 (0.574)	0.6046 (0.239)
q	0.0027 (0.002)	0.001 (0.0008)
k (t)	14	13
MSY (t)	4.8	2

Standard errors of estimates are in parentheses.

biomass expected at MSY would be between 6.5 and 7 t (Table 1). Our estimate of current stock size based on the escapement methodology is 41 t. Based on these data it appears the population is not being exploited at MSY and concerns of overfishing might not be justified. A second comparison that could identify possible overharvesting is between the current fishing mortality rate and the fishing mortality rate expected at MSY (here $r/2$). Based on the surplus production models, the fishing mortality rate at MSY would range between 0.3 and 0.68 (Table 1). Assuming an annual fishing mortality rate of 0.4 (see above) the corresponding instantaneous rate would be 0.51, within the range estimated by the surplus production models. Again, concerns of overharvesting appear not to be justified. A third comparison, and perhaps the most informative, is simply the comparison of the current harvest to the estimate of MSY (here $rk/4$). Based on the surplus production models MSY ranges between 2 and 4.8 t (Table 1). In recent years, the commercial harvest alone has averaged 2 t. Based on our fisher model, the estimated recreational harvest is 13 t. Even if the estimate from this model is high, the recreational harvest is assumed greater than the commercial harvest. If so, overharvesting surf smelt in Burrard Inlet is a possibility. However, in order to verify this, a refined estimate of the recreational harvest is needed.

Because the available data are so limited it is unclear if the drastic decline in surf smelt landings reflects a decrease in biomass, decrease in fishing effort, unreported catches, or a combination of these factors. Available commercial catch data suggest CPUE has not decreased over time or with effort and that the Burrard Inlet surf smelt fishery has not experienced a "one-way trip" (cf. Hilborn and Walters 1992). Since the recreational harvest accounts for almost 90% of the total fishery harvest (13 of 15 t), if overharvesting is occurring, it is in the recreational rather than commercial surf smelt fishery. Although there seems to be a perception that there is limited commercial demand for surf smelt there may be a high incidence of illegal fishing (both recreationally and commercially), which suggests demand actually is high. The smelt fishery in British Columbia went through a similar cycle 50 years ago when catches decreased at a time when demand was high, probably due to reduced stock biomass (Hart and McHugh 1944). Currently, the commercial surf smelt fishery has unlimited entry and no catch limits. Enforcement measures are limited, there is no functional management of bycatch, and compliance to the harvest log submission requirement is poor (average 37%). While the commercial catch data alone do not suggest over-utilization of the resource, when combined with the estimated recreational harvest and estimated spawner biomass for Burrard Inlet, potential overutilization of the resource could be occurring. We acknowledge that these conclusions are tentative so we recommend caution in the development of appropriate management decisions. Nevertheless, the present limited management of this species appears to conflict with the precautionary approach to fisheries management. In accordance with DFO's National Policy for New Emerging Fisheries (DFO 2001) there are not sufficient, science-based data to reasonably manage this resource. For example, there is no biological basis to support an unlimited entry, unlimited quota fishery where biological data are sparse and formal science-based assessments are not possible at this time.

In contrast to the relatively small commercial fishery in British Columbia (Burrard Inlet) a substantially larger commercial fishery exists in Washington state with approximately 45 t of surf smelt harvested per year with most landings from inside Puget Sound (WDFW 1998). Comparable harvests have not been made in British Columbia since the mid-1950s (Fig. 1). In Puget Sound, approximately 322 km of surf smelt spawning beaches have been identified (Penttila 1997, 2001; WDFW 1998) compared to 13.8 km of spawning beaches around Burrard Inlet. Thus, based on length of spawning beaches identified, it is possible that Puget Sound surf smelt stocks are ten times larger than those in Burrard Inlet. Also, it is possible that Burrard Inlet surf smelt are part of a larger stock that use spawning beaches elsewhere in the Strait of Georgia and the estimated biomass from the escapement (spawn deposition) model underestimates the actual stock biomass. A similar argument can be made for the surplus production model since commercial catch and effort data only apply to a small part of a potentially much larger stock. However, if the stock was considerably larger than estimated here, then we might have expected greater interest by commercial fishermen. Such an expectation assumes that markets for surf smelt are available. Also, we would have expected more reports of surf smelt occurring as bycatch in other fisheries operating in the southern Strait of Georgia, but such records are relatively rare. Further, it is possible, but unlikely, that surf smelt spawning habitat has been lost or not identified. Such a possibility is especially unlikely in

Burrard Inlet, which is adjacent to metropolitan Vancouver. Beaches in that area are intensely scrutinized, so substantial unreported spawning activity is not likely. On the other hand, there is competition between conflicting pressures for urban development and habitat preservation (e.g., Levings and Jamieson 2001). Additional habitat loss could occur indirectly. Surf smelt prefer to spawn on fine to coarse gravel (Schaefer 1936; Penttila 1978, 2001), which is subject to erosion and sediment sloughing. Since surf smelt eggs are susceptible to sedimentation (Morgan and Levings 1989), preservation of spawning habitat should be a priority for conservation of this stock. It is conceivable that considerably more unidentified spawning habitat exists in areas adjacent to, but outside of Burrard Inlet. This assertion is based on experience in Washington state where it took a considerable amount of time and effort to map surf smelt spawning beaches (i.e., Penttila 2001). A comparable level of effort has not been undertaken in British Columbia.

Acknowledgments

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Acoustical Estimations Confirm the Good Health of Gulf of Riga Herring

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Abstract

The Baltic herring (*Clupea harengus membras*) is one of the most important commercial species of the Baltic Sea. The herring stocks have decreased remarkably in the Baltic proper in recent decades. At the same time, due to favorable environmental conditions and successful management the annual landings of local gulf population of Baltic herring have grown in recent years. The assessments are indicating the increasing trend of the abundance and biomass of gulf herring in the Gulf of Riga.

In order to obtain catch-independent tuning data for the gulf herring stock assessments the joint Estonian-Latvian acoustic surveys were initiated in the whole Gulf of Riga in 1999. The results show that the Gulf of Riga as a semi-enclosed basin with favorable bottom topography and the settled character of gulf herring stock has offered favorable conditions for successful implementation of the results of acoustical surveys into the assessment of gulf herring stock. Acoustically estimated number and biomass of gulf herring in the Gulf of Riga have been in good agreement with VPA estimates. This suggests high reliability of the abundance and biomass estimates and confirms the good health of gulf herring stock.

Introduction

The Gulf of Riga is situated in the eastern part of Baltic Sea (Fig. 1) and connected with Baltic proper through two sounds. The area of the gulf at the long-term mean water level is approximately 16,330 km^3 with the volume being 424 km^3, which constitutes 3.9% of the total area and 2.1% of the total volume of the Baltic Sea, respectively (Berzinsh 1995).

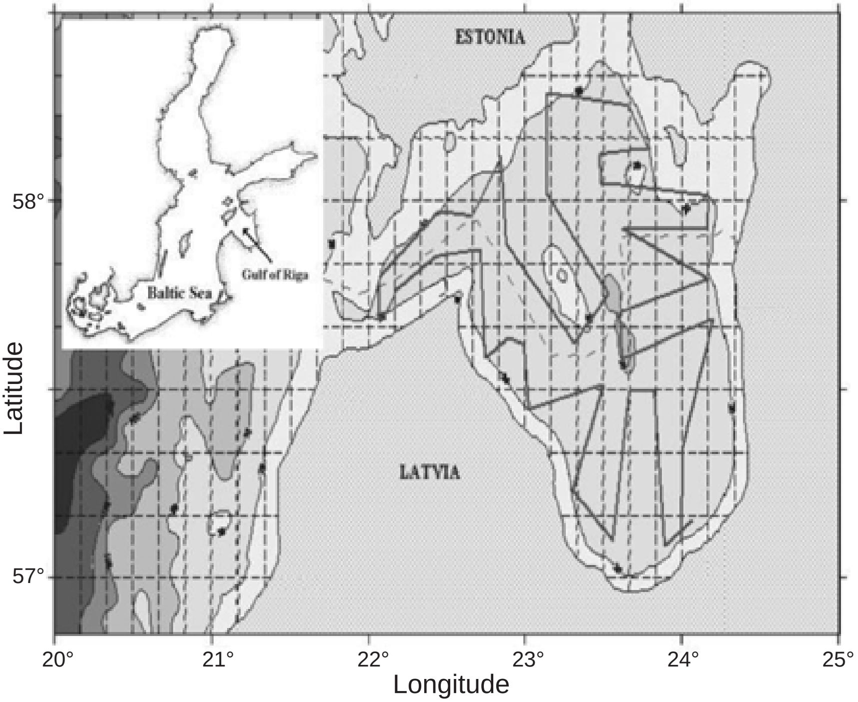

Figure 1. Typical track of acoustic survey in the Gulf of Riga.

Two Baltic herring (*Clupea harengus membras*) populations—sea herring and gulf herring—can inhabit the Gulf of Riga. The open sea herring (Central Baltic herring) enters the gulf only for spawning time (in May) while the gulf herring population resides there year-round.

The present international fisheries regulation system in the Baltic Sea relies on the presumption that the weight-based catch quotas (total allowable catch), if properly followed, ensure the biologically safe exploitation of fish stocks. However, several cases have shown the decline of stocks despite all assessment and regulation efforts. An example of such a case is the Central Baltic herring. At the same time, some herring stocks, like the Gulf of Riga herring, have demonstrated quite different developments during the same period. The abundance and biomass of Gulf of Riga herring have shown an increasing trend since the late 1980s. The spawning stock biomass (SSB) has doubled since the 1980s, reaching the level of 130,000 t in 2002. Accordingly, the annual landings of gulf herring have grown from 15,000 t in the beginning of the 1990s to close to 40,000 t in recent years. At the same time, the estimates of fishing mortality for gulf herring have been higher than for Central Baltic herring.

Until 1999, there were no catch-independent tuning data available for the Gulf of Riga herring stock assessments. The Gulf of Riga herring stock was assessed using an analytical model—virtual population analysis (VPA), tuned against catch per unit of effort (CPUE) data from pound-net fisheries. In order to obtain catch-independent tuning data for the assessments and identify the peculiarities of distribution of the gulf herring, joint Estonian-Latvian acoustic surveys were initiated in the whole Gulf of Riga in 1999.

Materials and methods

Estonian-Latvian joint acoustic surveys were carried out with F/V *Zane* (engine power 300 hp, vessel length 25 m) in the Gulf of Riga during the second half of July from 1999 to 2002 (Fig. 1). Calibration of equipment, acoustic measurements, fish sampling, and data analyses were made according to the methodology recommended by ICES (International Council for the Exploration of the Sea) Baltic International Fish Survey Working Group (ICES 2001).

The acoustic equipment used was a Simrad EY500 portable sounder system. A 38 kHz split beam Simrad transducer ES38-12 was employed in a towed body. To avoid errors in acoustic abundance estimates caused by diurnal behavior of herring, all investigations were performed during daytime, when 70% of herring biomass is based in the 10-20 m depth layer. The effect of avoidance reaction of herring was not investigated during the survey and not taken into account in acoustic abundance estimates. The acoustic measurements were averaged by 1 nautical mile interval along the survey track. During the acoustic integration, which covered in total 3,142 square nautical miles (10,777 km^3), the vessel speed was 6-8 knots. The fish samples were taken using a commercial midwater trawl with a 20 mm stretched mesh size in the cod-end. The trawling speed was 2.2-3.0 knots, and haul duration 30-60 minutes. Number of acoustic samples (length of survey track in nautical miles), trawl hauls, and herring sampled for the age-length-key in the Gulf of Riga in 1999-2002 are shown in Table 1.

Herring total number (*N*) were estimated using the formula

$$N = \frac{S_A}{4\pi \times \sum_j f_j \times 10^{[(a+b\times\log Lj)/10]}} \times A \qquad (1)$$

where S_A is nautical area scattering coefficient (m^2 per NM^2), A is integration area (area in NM^2 below 10 m depth), L_j is the midpoint of the *j*th length-class (cm), and f_j is the mean portion of length-class *j* of herring

Table 1. Number of acoustic samples, trawl hauls, and herring sampled in the Gulf of Riga in 1999-2002.

Year	Acoustic samples	Trawl hauls	Herring measured	Herring aged
1999	633	12	2,400	1,100
2000	381	15	2,404	633
2001	428	12	2,388	1,248
2002	445	12	2,623	1,072

in the fishing samples. Values for constants *a* and *b* were taken from the Manual for the Baltic International Acoustic Survey (ICES 2001).

Gulf and sea herrings were distinguished on the basis of otoliths and discriminated in abundance calculations. The gulf herring otoliths clearly differ from sea herring otoliths by their smaller summer zones (especially beginning with the third zone), and by a larger number of additional rings and sulci, etc. (Ojaveer and Gaumiga 1995).

Based on estimates of herring abundance by age groups in two subsequent years, the fishing mortality was estimated as:

$$F = \ln N_t / N_{t+1} - M \qquad (2)$$

N_t and N_{t+1} are number of the year-class in two subsequent years; and M is natural mortality, assumed equal to 0.2.

A total of four VPA runs were made for the period of 1980-2002 using different tuning data packages. The first VPA run (Darby and Flatman 1994) was made without tuning (simple cohort analysis). For the other three runs XSA (extended survivors analysis) was used (Darby and Flatman 1994) with three different series of tuning data:

1. Age structure of gulf herring in the trap-net catches at age 2-9 and the catch unit (number) of trap-nets.
2. Abundance and age structure of gulf herring estimated with acoustic methods.
3. Combined catch statistics of trap-nets and acoustic assessments.

The catch statistics of trap-nets originates from the materials reported to ICES Baltic Fisheries Assessment Working Group by Estonia and Latvia (ICES 2003).

Table 2. Estimated abundance (millions) of gulf herring by age groups and total biomass (t) in the Gulf of Riga in July 1999-2002.

Age	1999	2000	2001	2002
1	5,292	4,486	7,567	3,998
2	4,363	4,012	2,004	5,994
3	1,343	1,791	1,447	1,068
4	1,165	609	767	526
5	457	682	206	221
6	319	336	296	87
7	208	151	56	165
8	61	147	66	34
9+	37	97	106	97
Sum	13,245	12,311	12,516	12,190
Biomass	209,778	213,871	189,698	243,523

Results

Survey results have shown that July is a very favorable time for gulf herring research since there are few open sea herring (< 3%, by numbers) in the Gulf of Riga.

Estimated herring stock abundance by age groups in the survey period is presented in Table 2. The total estimated stock size in numbers showed a rather stable pattern, varying from 13,245 million in 1999 to 12,190 million in 2002 (by 8%). The biomass estimates showed a slightly higher variation—a difference between 243,523 t in 2002 and 189,698 t in 2001. Standard deviation and coefficient of variation of the total biomass estimates of the Gulf of Riga herring from acoustic surveys and VPA is presented in Table 3.

The higher difference in biomass estimates was obviously caused by higher number of young fish and low abundance of older age groups in control hauls in 2001. One possible explanation for that phenomenon can be connected with the late and prolonged spawning of herring. Due to the low water temperatures in 2001, spawning of herring was prolonged into July and part of older herrings being on the spawning grounds was not accounted for by the acoustic survey.

The comparison of acoustic and XSA estimates (tuned with trap-net data) of stock abundance and total biomass are presented in the Fig. 2. Comparison reveals that the stock estimates from the surveys and the respective analytical assessments were sufficiently close. However, small yet systematic differences were observed: both the abundance and total

Table 3. Standard deviation and coefficient of variation of the total biomass estimates of the Gulf of Riga herring from acoustic surveys (1999-2002) and VPA (1980-2002) calculated using different tuning data packages (catch statistics of trap-nets, acoustic assessments, combined catch statistics of trap-nets, and acoustic assessments, not tuned).

	Acoustic	VPA			
	survey	Not tuned	Trap-nets	Acoustic	Acoustic + trap-nets
SD	22,210	48,020	45,436	40,360	44.671
CV	0.10	0.38	0.36	0.33	0.36

biomass estimates from the acoustical surveys exceed the respective estimates from analytical assessments. So, the acoustic abundance has been higher by 22% in 1999, 20% in 2000 and 2001, and 8% in 2002. The differences in total biomass estimates decreased from 33% in 1999 to 7% in 2001 and increased again to 33% in 2002. Comparison of the respective estimates by the age groups show that the main source of the differences is the younger age groups (1 and 2), which, particularly age group 1, is not yet fully recruited to the fishery and therefore underestimated by the XSA (Kaljuste et al. 2002). Despite the differences mentioned, the age composition estimates of the stock from the surveys and the respective analytical assessments were sufficiently close, particularly keeping in mind that the analytical assessments express the situation in the beginning of the year, while the acoustical surveys were performed a half year later, in July.

Comparative estimations of three relevant parameters describing the population dynamics (abundance of recruits, spawning stock biomass, and fishing mortality) calculated using VPA with different tuning data packages are presented in Fig. 3. For spawning stock biomass (SSB) the best agreement was found between VPA estimates with no tuning and VPA estimates tuned with catch statistics of trap-nets. For fishing mortality the best agreement was found between VPA estimates with no tuning and VPA estimates tuned with the combined data package.

In general, the tuning with acoustic data only gave lower spawning stock biomass estimates and higher fishing mortality estimates than tuning with trap-net data, particularly for most recent years. The obtained estimates of fishing mortality were relatively high throughout all periods regardless of tuning data, indicating high fishing pressure on the stock.

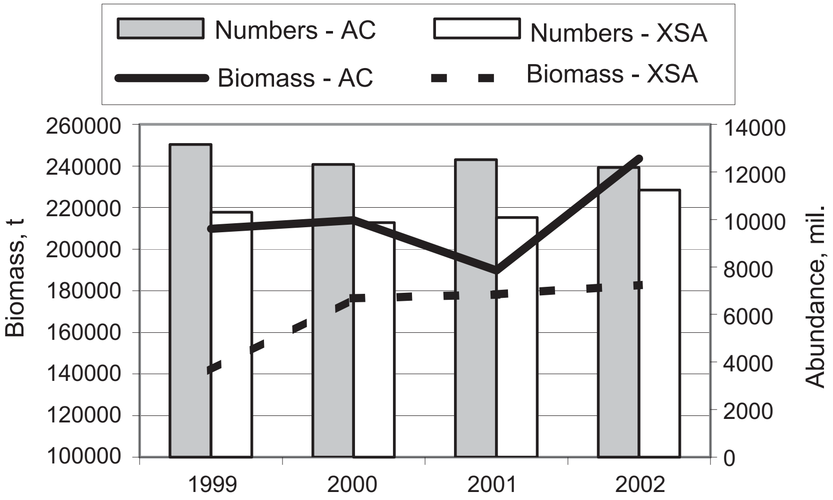

Figure 2. Gulf of Riga herring: abundance and total biomass estimates from acoustic surveys and XSA (ICES 2000, 2001, 2002, 2003) tuned with data from the trap-net fishery.

Discussion

The Central Baltic herring (ICES Sub-divisions 25-29 and 32), including several local populations, has been assessed and managed as one unit stock since 1990 (ICES. 1990). The spawning stock biomass (SSB) of that complex has shown a decrease by approximately 70% from 1974 to 2001. After 1994, the decrease in SSB has been particularly steep. Landings have decreased from 300,000 t to below 200,000 t. Fishing mortality estimates have been in the range of 0.2-0.3, which were above the defined F_{PA}(F_{PA} = 0.19, ICES 2002) until 1994, increasing thereafter up to almost 0.5 in 2000. The only clearly pronounced peak in F was observed in mid-1980s.

The Gulf of Riga herring population has been assessed by the ICES Baltic Fisheries Assessment Working Group separately from the Central Baltic herring stock before 1990 and then again since 1994. VPA estimates (tuned with the combined data package) show that, variously from the Central Baltic herring, the abundance and biomass of Gulf of Riga herring increased from the late 1980s to the mid-1990s, decreased after that somewhat, and has been stable since 1998. The SSB doubled since the 1980s, reaching the level of 100,000 t in 2002. Accordingly, the annual landings of gulf herring have grown from 15,000 t in the beginning of the 1990s to close to 40,000 t in recent years. At the same time, the estimates of fishing mortality have been higher than in the Central Baltic herring.

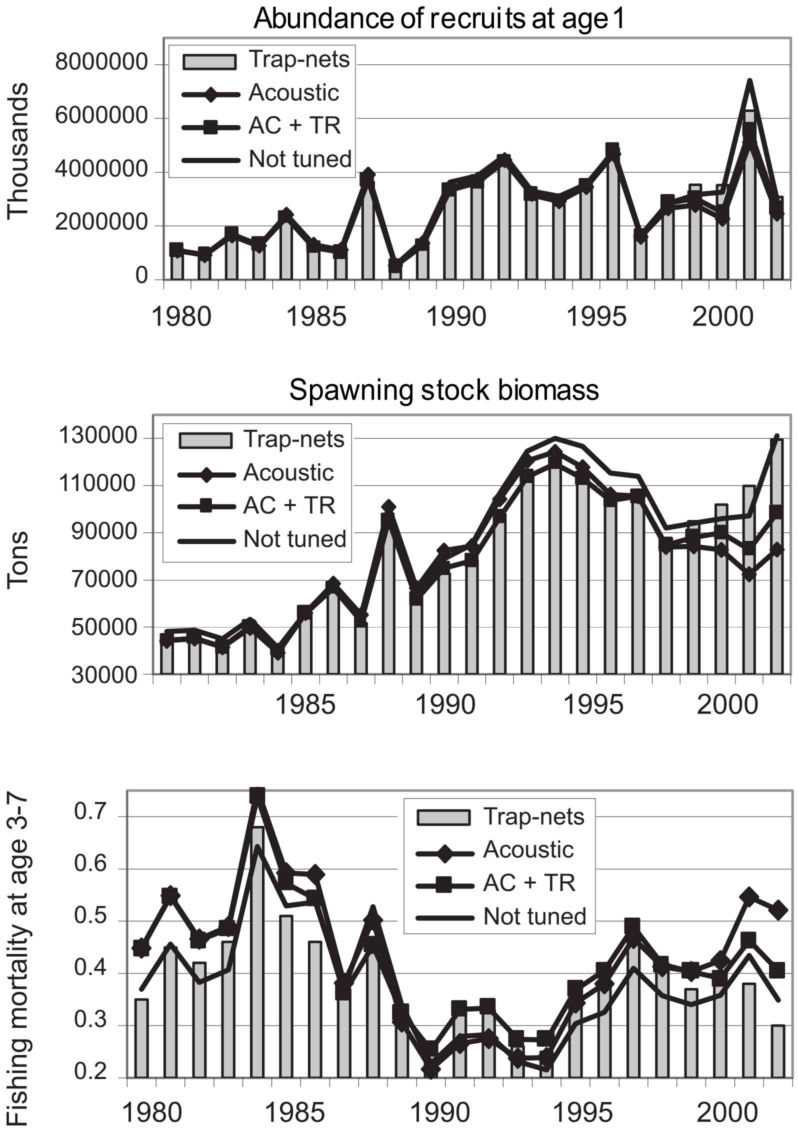

Figure 3. Some parameters of Gulf of Riga herring population in 1980-2002 calculated using VPA with different tuning data packages: catch statistics of trap-nets, acoustic assessments, combined catch statistics of trap-nets, and acoustic assessments, not tuned.

Until 2003, the Gulf of Riga herring stock was assessed by the ICES Baltic Fisheries Assessment Working Group using XSA tuned against CPUE data from pound-net fisheries. According to these assessments the fishing mortality estimates have been mainly below the defined F_{PA}($F_{PA} = 0.4$, ICES 2003). Such a high F_{PA} value for the Gulf of Riga herring stock has been set due to two reasons:

1. Gulf of Riga herring has practically no natural enemy and its natural mortality is very low (ICES 2004).
2. The abundant herring year-classes have produced very good recruitment each year since 1998 (ICES 2004).

So, despite the high fishing pressure the Gulf of Riga herring stock can be characterized as in good health.

Availability of certain basic information and measure taking are fundamental in order to secure the sustainable management of any exploited fish stock. Among those, the presence of following data sets and measures are crucial:

- Distribution pattern and stock structure (age structure, growth pattern, etc.).
- Catch structure and fishing effort data.
- Regular assessment of stock developments using appropriate methods, tuned with survey results and/or other appropriate data.
- Regular and flexible catch advice taking into account natural fluctuations in populations.
- Following of advice by fishery.

The above have been the cornerstone in international stock assessment and advice procedures on herring stocks in the Baltic.

The Baltic Sea has been faced with an extended stagnation period since the major inflows of the late 1970s. The altered salinity conditions together with a series of mild winters in the Northern Baltic have favored good recruitments of gulf herring populations in the 1990s (Ojaveer 1991, Ojaveer and Järvik 1996). Additionally, some other factors affecting the dynamics of the Gulf of Riga herring and the Central Baltic herring stocks can be outlined. A part of those are probably connected to the different nature of the Central Baltic herring and the Gulf of Riga herring stocks: while the Gulf of Riga herring stock includes just one population, the Central Baltic herring stock is a complex of several sub-stocks (populations), differing from each other, e.g., in mean weights at age as well as in other markers and dynamics (Ojaveer 1991). Catch statistics of gulf herring landings follows the catch advice rather well in the Gulf of Riga (Raid and Kaljuste unpubl.). Due to low salinity the number of another pelagic species—Baltic sprat—is negligible in the Gulf of Riga and there are no serious difficulties in getting adequate information on species composition in catches.

The Gulf of Riga has proved to be ideal for performing acoustic surveys because of its configuration and bottom topography, allowing coverage of almost all areas of herring distribution. The ultimate test of the acoustic survey results is to compare the estimates of the same stock obtained by the acoustic and other methods (MacLennan and Simmonds 1992). In the case of the Gulf of Riga herring, we compare the acoustical results with the results of analytical assessment. The respective time series as a result of XSA extends back to1970 (ICES 2002). The comparison of the acoustical and VPA results has been performed for many stocks, e.g., for Icelandic summer spawning herring (Jakobsson 1983), cod and haddock of the northeast Arctic (Nakken and Ulltang 1983), North Sea herring (Bailey and Simmonds 1990), and others. However, in many cases the result of the comparison has not been strictly valid, because the VPA has been tuned with the same acoustical data (MacLennan and Simmonds 1992). In the case of the Gulf of Riga herring that potential shortcoming is excluded since the XSA has been tuned with data from the trap-net fishery, not against the acoustical survey results.

As a result of the study it was also found that selection of the data packages for tuning of the VPA model can substantially influence the retrospective estimation of the population dynamics, especially at the end of the investigated time interval. While the main input for the prognostic models is the status quo of the population, the prognostication of the population dynamics is directly influenced by the chosen tuning method. So, for analytical assessment of the Gulf of Riga herring population, the traditional VPA model should be tuned using additional information from the catch statistics of trap-nets (for older age groups) and from acoustical estimations (for younger age groups).

Conclusions

Due to favorable environmental conditions and successful management of the Gulf of Riga herring the abundance and biomass of gulf herring has increased since the late 1980s. Accordingly, the annual landings of gulf herring have grown.

In general very good agreement between the results of the acoustic surveys of 1999-2002 with the results of analytical assessments allow conclusion that the implementation of the acoustical survey as an additional source of information for herring stock assessment in Gulf of Riga herring has been successful. Topography and the settled character of the gulf herring stock probably have been the key factors contributing to the favorable conditions for carrying out acoustic surveys. The continuation of acoustic surveys will not only provide additional estimates of stock abundance and biomass of the Gulf of Riga herring, but also form an additional data set for tuning the analytical models in the assessment process. This allows us to be more confident in the reliability of the abundance and biomass estimates and confirms the good health of the gulf herring stock.

The stock assessment and management of the Gulf of Riga herring has been successful in past decade due to several reasons:

- The distribution area of the population is well-defined with the semi-enclosed gulf.
- The herring belonging to the population can be easily distinguished by the otolith structure.
- The results of analytical assessments have been verified by the extensive acoustic surveys.
- Commercial landings are well recorded and sampled.
- Low abundance of sprat in the Gulf of Riga eliminate the possibility of misreporting.
- The fishery has not failed to follow the catch advice.

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