

A General Approach for Making Short-Term Stochastic Projections from an Age-Structured Fisheries Assessment Model

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Abstract

Catch-age analysis is a powerful tool for assessing the status of fisheries resources and catch-age analyses are routinely conducted for many commercially exploited fish stocks around the world. In this paper, we illustrate a general approach for making short-term stochastic projections from the ADAPT age-structured assessment model. Our approach uses the standard statistical techniques of bootstrapping and Monte Carlo simulation to project performance measures such as landings, discards, spawning biomass, and recruitment under alternative management policies. The key idea is to propagate variability in estimates of initial stock size forward in stochastic projections of future possibilities. We use bootstrap replicates of current population size from an age-structured ADAPT model combined with a stochastic stock-recruitment relationship to simulate population trajectories through the projection horizon. The approach is illustrated for a commercially important New England groundfish, Georges Bank yellowtail flounder, *Pleuronectes ferrugineus*. Although hypothetical, this illustration provides some general insight for the rebuilding of the Georges Bank yellowtail flounder stock.

Introduction

Catch-age analysis is a powerful tool for assessing the status of fisheries resources and catch-age analyses are routinely conducted for many commercially exploited fish stocks around the world. Diverse methods of catch-age analysis exist (Megrey 1989) and these include, but are not limited to, tuned virtual population analysis (Laurec and Shepherd 1983, Parrack 1986, Gavaris 1988), statistical catch-at-age analysis (Fournier and Archibald 1982, Deriso et al. 1985, Hilborn and Walters 1992), stock synthesis (Methot 1989, 1990), as well as state-space (Mendelsohn 1988, Schnute 1994) and Bayesian approaches (Hilborn et al. 1994, McAllister and Ianelli 1997). Despite underlying differences in model assumptions and statistical framework, these diverse approaches to age-structured assessment generally provide estimates of fishing mortality and stock size at age through the assessment time horizon. To provide quantitative advice to fisheries managers, these estimates of fishing mortality and stock size are often compared to biological reference points or to threshold stock sizes to evaluate whether current exploitation rates exceed target or limit reference points and whether stock size has been maintained above threshold levels (Caddy and Mahon 1995). However, while such analyses provide historic and current estimates of stock size and fishing mortality to evaluate previous harvest policies, they do not address the management question of projecting what would happen in the future under alternative harvest policies.

In this paper, we illustrate a general approach for making short-term stochastic projections from the ADAPT age-structured assessment model. Our approach extends the work of Mohn (1993), who used bootstrapping and simulation to project uncertainties in catch projections from ADAPT, and Smith and Gavaris (1993), who compared several methods of evaluating the accuracy of projected catch estimates from ADAPT, in several ways. First, in our approach, simulation is systematically applied to the entire bootstrap distribution of initial population sizes to propagate uncertainty. Second, our approach permits several types of future harvest policies to be evaluated including quota policies, target fishing mortality rate policies, or any mixture of these two. Third, we provide a standard set of performance measures, including spawning biomass, recruitment, landings, discards, and the probability of achieving spawning biomass thresholds, to evaluate the potential benefits of alternative harvest policies. Since 1994, this approach has been used to project the consequences of alternative management measures for several age-structured assessments in the northwest Atlantic including Atlantic mackerel, Georges Bank cod, haddock, and yellowtail flounder.

Our approach uses the standard statistical techniques of bootstrapping and Monte Carlo simulation to project performance measures such as landings, discards, spawning biomass, and recruitment under alternative management policies. The key idea is to propagate variability in esti-

mates of initial stock size forward in stochastic projections of future possibilities. We use bootstrap replicates of current population size from an age-structured ADAPT model combined with a stochastic stock-recruitment relationship to simulate population trajectories through the projection horizon. As a consequence, uncertainties in both initial population abundance and future recruitment are directly incorporated into management advice. The implications of management decisions can be quantified and compared using empirically derived sampling distributions of catch, landings, discards, spawning biomass, recruitment, and, in the case of management under fixed catch quotas, fishing mortality. Estimates of the probability of exceeding biological reference points or achieving management targets are also quantified. The approach is illustrated for a commercially important New England groundfish, Georges Bank yellowtail flounder, *Pleuronectes ferrugineus*. This example is based on analyses of Amendment 7 to the Northeast Multispecies Fishery Management Plan (New England Fishery Management Council 1996) which was implemented to rebuild depleted New England groundfish stocks.

Materials and Methods

Initial Population Abundance

Initial population abundance is the first of two key sources of randomness in this approach to making stochastic projections, where $\mathbf{N}(1)$ is the vector of stock size at age in the current year which is also the first year of the projection horizon. Regardless of the age-structured assessment model used to estimate $\mathbf{N}(1)$, it is common practice to determine the sampling distribution of model estimates of current population size and other key outputs such as fishing mortality and spawning biomass. Uncertainties in key model outputs are often described with confidence intervals or probability profiles which express the chance that an output falls below a biological reference point (Rosenberg and Restrepo 1994).

The ADAPT method of sequential population analysis provides the initial abundance estimates for the projection approach described in this paper (Parrack 1986, Gavaris 1988, Conser and Powers 1990, Mohn and Cook 1993). This age-structured assessment approach combines deterministic virtual population analysis with a nonlinear least squares (NLS) objective function to estimate model parameters (θ) such as stock size at age through time. As generally implemented, the ADAPT method is a measurement error model in which observed indices of relative abundance (I_{iy}) are modeled as random deviations from the true values of the abundance indices. In particular, it is often assumed that the log-transformed measurement errors are independent and identically distributed random variables with zero mean and constant variance. The NLS objective function [$SS(\theta)$] typically consists of the weighted sum of squared differences between model predictions of indices of stock size at age and observed

survey indices of stock size at age in natural logarithmic scale. That is, the objective function $SS(\theta)$ commonly has the form

$$SS(\theta) = \sum_{i=1}^K \sum_{y=1}^{Y_i} \lambda_{iy} \left(\log_e I_{iy} - \log_e I_{iy}^* \right)^2 \quad (1)$$

where I_{iy} is the observed value of the i th survey index in year y , I_{iy}^* is the predicted value of the i th survey index in year y , K is the number of age-specific survey indices, Y_i is the number of years of abundance data for the i th survey index, and λ_{iy} is the relative weighting factor for the i th survey and y th year term. In the absence of prior information on the accuracy and precision of survey indices, the λ_{iy} are often set to 1. Given initial estimates of model parameters and fixed weighting factors, an iterative minimization algorithm, such as the Marquardt algorithm, is used to compute the NLS estimate of model parameters, θ_{NLS} ; this NLS estimate can be interpreted as a maximum likelihood estimate (conditioned on the relative weighting factors) if the measurement errors are independent and identically distributed lognormal random variables with zero mean and constant variance.

We use the nonparametric bootstrap (Efron 1982) to empirically determine the sampling distribution for the estimator of $\mathbf{N}(1)$. To accomplish this, the set of residuals $\{\varepsilon_{iy}\}$ from the observed and predicted values of survey indices of the best ADAPT model formulation are generated for resampling

$$\varepsilon_{iy} = \log_e I_{iy} - \log_e I_{iy}^* \quad (2)$$

The set of residuals is randomly resampled with replacement a large number of times (B) to create a family of new input data sets of relative abundance indices. In particular, the b th bootstrap data set $\{I_{iy}^{(b)}\}$ is constructed from the b th set of resampled residuals $\{\varepsilon_{iy}^{(b)}\}$ as

$$\log_e I_{iy}^{(b)} = \log_e I_{iy}^* + \varepsilon_{iy}^{(b)} \quad (3)$$

This resampling is conditioned on the correctness of the fitted model (Punt and Butterworth 1993) where it is assumed that residuals are independent and identically distributed. Parameters of the age-structured assessment model are re-estimated for each of the new data sets to generate a set of bootstrap replicates of initial population abundance, $\{\mathbf{N}^{(1)}(1), \mathbf{N}^{(2)}(1), \dots, \mathbf{N}^{(B)}(1)\}$. This set gives the empirical (bootstrap) distribution function of the estimator of $\mathbf{N}(1)$ conditioned on the model. Bootstrap confidence intervals are derived from percentiles of the bootstrap distribution of $\mathbf{N}(1)$ (see Efron and Tibshirani 1993, Chapter 13) and point estimates of parameter bias can also be computed (Efron 1982, Smith and Gavaris 1993).

Age-Structured Population Model

An age-structured population model is the basis for making short-term projections. The model describes the dynamics of a closed, iteroparous fish population whose abundance changes due to recruitment, natural mortality, and fishing mortality. Recruitment to the population is measured at the beginning of each year. Population abundance changes throughout the year due to the concurrent forces of natural and fishing mortality so that the fishery is classified as a type-2 fishery (Ricker 1975).

Population Abundance, Survival, and Spawning Biomass

Population abundance is measured by the number of survivors within each recruited age class at the beginning of the year and is denoted by the vector $\mathbf{N}(t)$ with elements $N_a(t)$ where “ a ” indexes age class and “ t ” indexes year in the projection horizon. The youngest age class comprises the recruits and the age of recruitment (R) is typically age-1 or age-2. The oldest age class is a plus-group comprised of all fish that are at least as old as a chosen cutoff age (A).

Population survival at age from year to year is calculated in the standard manner using instantaneous mortality rates. To describe annual survival through mortality, $M_a(t)$ is the instantaneous natural mortality rate and $F_a(t)$ is the instantaneous fishing mortality rate on age- a fish. Population size at age changes according to

$$\begin{aligned} N_a(t) &= N_{a-1}(t-1) \times e^{-M_{a-1}(t-1) - F_{a-1}(t-1)} \text{ for } a = R+1 \text{ to } A-1 \\ \text{and } N_A(t) &= N_A(t-1) \times e^{-M_A(t-1) - F_A(t-1)} + N_{A-1}(t-1) \times e^{-M_{A-1}(t-1) - F_{A-1}(t-1)} \end{aligned} \quad (4)$$

Typically, age-specific values of natural mortality are not estimable due to a lack of data and natural mortality is parsimoniously modeled as a constant value across age classes. We adopt this convention and assume that natural mortality (M) is constant in what follows. Recruitment is modeled as a stochastic process and is not determined by equation (4).

Annual spawning biomass is determined from the population size vector $\mathbf{N}(t)$ and mortality rates as well as additional information concerning fish maturity and size at age. Population abundance at the start of the spawning season depends on the level of fishing and natural mortality prior to spawning; the number of age- a fish alive at the start of spawning ($N_{s,a}$) is

$$N_{s,a}(t) = N_a(t) e^{-T(t)[M+F_a(t)]} \quad (5)$$

where $T(t)$ is the fraction of total annual mortality that occurs from January 1 to the start of the spawning season. This fraction may need to be modeled as time-varying if seasonal patterns in fishing effort are altered

by management measures. Because this approach focuses on short-term projections, growth and maturation rates are assumed to be independent of stock density during the projection horizon. Spawning biomass [$SB(t)$] is the sum over all age classes of the weight of mature female and male fish at the start of the spawning season

$$SB(t) = \sum_{a=R}^A W_{S,a} P_a N_a(t) e^{-T(t)[M+F_a(t)]} \quad (6)$$

where P_a is the average fraction of age- a fish that are mature and $W_{S,a}$ is the average spawning weight of an age- a fish.

Catch, Landings, and Discards

In this model, the entire population is available to the fishery. Annual catch at age [$C_a(t)$] is given by the standard catch equation (Baranov 1918)

$$C_a(t) = \frac{F_a(t)}{M + F_a(t)} (1 - e^{-[M+F_a(t)]}) N_a(t) \quad (7)$$

To account for age-specific discarding of fish, let $D_a(t)$ be the annual fraction of age- a fish that are discarded and die and $W_{L,a}$ and $W_{D,a}$ are the average weights of age- a fish that are landed and discarded, respectively. Here the age-specific discarding pattern may vary if fishery regulations are expected to change during the projection horizon. Total landed weight [$L(t)$] is

$$L(t) = \sum_{a=R}^A C_a(t) [1 - D_a(t)] W_{L,a} \quad (8)$$

and the total discarded weight [$D(t)$] is

$$D(t) = \sum_{a=R}^A C_a(t) D_a(t) W_{D,a} \quad (9)$$

Population Harvest

There are three options for determining the level of population harvest in each year of the projection horizon. The first option is a time series of annual fishing mortality rates (F -based management) and the second is a time series of annual landings quotas (quota-based management). Any mixture of the F -based and quota-based options can be combined so that F -based management is applied in some years and quota-based management in the other years. This mixed option can be useful for projecting stock status in periods between assessments. For example, if a full stock assessment is only conducted every 2 years while catch biomass data are updated annually, then population status can be monitored in years be-

tween assessments using the quota-based option to project the effects of an additional year of catch data.

Under F -based management, catch at age is determined by setting $F_a(t)$ in equation (7). Separability is assumed and fishing mortality for age- a fish is the product of fully recruited fishing mortality $[F(t)]$ and partial recruitment for age- a fish $[S_a(t)]$.

$$F_a(t) = F(t) S_a(t) \quad (10)$$

Typically, values of partial recruitment are based on recent estimates from the assessment and are modeled as constant unless fishery regulations are expected to change during the projection horizon.

Under quota-based management, the landings quota $[Q(t)]$ corresponds to a fully recruited fishing mortality rate that depends on population size, partial recruitment pattern, and discard pattern, if applicable. This quota depends on the level of fully recruited fishing mortality, $Q = L(F)$, where L is given by equation (8), and the solution to $Q - L(F) = 0$ is determined numerically. After solving for the F that produces the landings quota, total catch at age and discard biomass are determined from equations (7), (9), and (10).

Stock-Recruitment Relationship

In many fisheries, stock-recruitment data are highly variable due to intrinsic variability in factors governing survival and measurement error in estimates of recruitment and spawning biomass. In our approach, the stock-recruitment relationship is the second key source of randomness for population projections because it ultimately defines the sustainable level of harvest and its variability over time. This follows from the model assumption that growth, maturation, and natural mortality are density-independent and adequately represented by expected (e.g., constant) values through the projection horizon.

In the application of this approach, several stochastic recruitment models have been used and two commonly used models are briefly described below. Because no single model will be suitable in all applications, we recommend that alternative stock-recruitment relationships be explored by the assessment practitioner. The first model is the empirical recruits per spawning biomass model. This is a non-parametric, density-dependent approach. In this model, random values of recruits per spawning biomass are generated from a uniform (0,1) variate by applying the probability integral transform to the empirical cumulative distribution function of recruits per spawning biomass from an appropriate time period. Here the choice of time period is important because this model requires stationarity in the distribution of recruits per spawning biomass. Recruitment is modeled as the product of spawning biomass and the random level of recruits per spawning biomass. That is, $N_R(t) = SB(t - R)\xi$ where ξ is a randomly chosen value of recruits per spawning biomass taken from

the distribution of estimates of recruit per spawning biomass during the chosen period. This model exhibits no compensatory population response because average recruitment does not eventually decline as spawning biomass increases. The second model is the stock-recruitment curve with multiplicative lognormal error term. In this model, recruitment is determined as the output of a stock-recruitment curve multiplied by a random realization from a lognormal distribution. That is, $N_R(t) = g[SB(t - R)]\exp(\xi)$ where $g[\cdot]$ is a parametric stock-recruitment curve (typically a Beverton-Holt, Ricker, or Shepherd curve) and ξ is a normal random variable with zero mean and constant variance. This parametric model is stochastic, density-dependent, and exhibits compensatory population response due to the choice of stock-recruitment curve.

The age of recruitment determines whether additional information is needed to begin the projections with a density-dependent recruitment model. In particular, if $R \geq 2$, then assessment estimates of spawning biomass in the $R-1$ years prior to the first year of the projection horizon are needed for input to the recruitment model. These lagged estimates of spawning biomass must be available for each bootstrap replicate of initial population size, $N^{(b)}(1)$ to use this approach to making stochastic projections.

Application to Georges Bank Yellowtail Flounder

Georges Bank yellowtail flounder (*Pleuronectes ferrugineus*) is a highly productive flatfish stock with a long-term potential yield of 16,000 t at a spawning biomass level of about 65,000 t (NEFSC 1995). This stock is currently at a low level of abundance due to overfishing (Fogarty and Murawski 1998). Based on the most recent assessment (NEFSC 1997), fishing mortality rates on Georges Bank yellowtail flounder exceeded the overfishing level ($F_{20\%} = 0.64$) during 1980-1994 (Fig. 1). Reductions in fishing effort achieved under Amendment 7 have reduced fishing mortality in recent years and spawning biomass levels have increased. A threshold level of spawning biomass of 10,000 t has been established as an interim rebuilding target for Georges Bank yellowtail flounder (New England Fishery Management Council 1996).

Georges Bank yellowtail flounder has been assessed with the ADAPT age-structured assessment model since 1991 (Conser et al. 1991). In the 1997 assessment (Cadrin et al. 1997), relative abundance indices taken from four research surveys on Georges Bank were used to calibrate a virtual population analysis for estimating population size at age for 6 age groups ($A = 6$) consisting of ages 1 to 5 and age 6+. The four surveys were the Northeast Fisheries Science Center (NEFSC) spring (age 1 to age 5+, 1968-1996) and autumn (age 1 to age 5+, 1963-1996) bottom trawl surveys, the NEFSC scallop survey (age 1 to age 4+, 1982-1996), and the Canadian spring bottom trawl survey (age 2 to age 5+, 1987-1996). Given the eighteen time series of age-specific indices of relative abundance, the objective function for the ADAPT assessment model was the sum of the

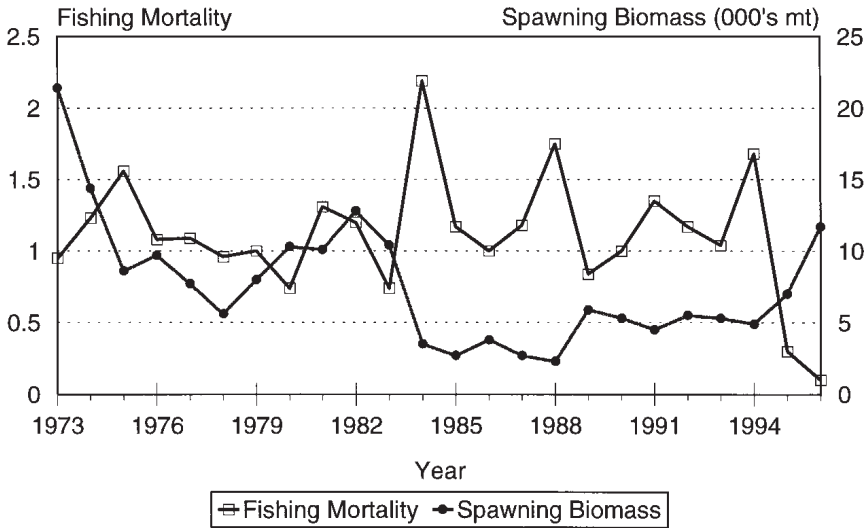


Figure 1. Fishing mortality (open square) and spawning biomass (solid circle) of Georges Bank yellowtail flounder, 1973-1996. Data from NEFSC (1997).

squared differences between predicted and observed survey indices as in equation (1) with $\lambda_{iy} = 1$ (NEFSC 1997). The non-parametric bootstrap was applied to resample the residuals of the fitted indices with replacement to generate a set of 200 bootstrap replicates of population abundance in 1996. This set determined the bootstrap distribution of population abundance and was used to characterize the sampling distribution of fishing mortality and spawning biomass in 1996. Bootstrap results gave 80% confidence intervals (CI) for $F(1996)$ of 0.08 to 0.14 (Fig. 2A) and for $SB(1996)$ of 9,800 to 14,600 t (Fig. 2B). In addition, bootstrap results indicated there was zero probability that $F(1996)$ exceeded the overfishing level of 0.64. Further the probability that $SB(1996)$ was below the interim rebuilding threshold of 10,000 t was about 0.12 (see Fig. 4) while the probability that $SB(1996)$ was below the long-term spawning biomass target of 65,000 t was 1.

A stochastic stock-recruitment relationship has been derived for Georges Bank yellowtail flounder based on a Beverton-Holt curve with a multiplicative lognormal error term (Beverton and Holt 1957). In this parametric model, survival to recruitment age is density independent and recruitment is subject to stochastic variation through

$$N_R(t) = \frac{a \times SB(t-R)}{b + SB(t-R)} \times e^w \quad (11)$$

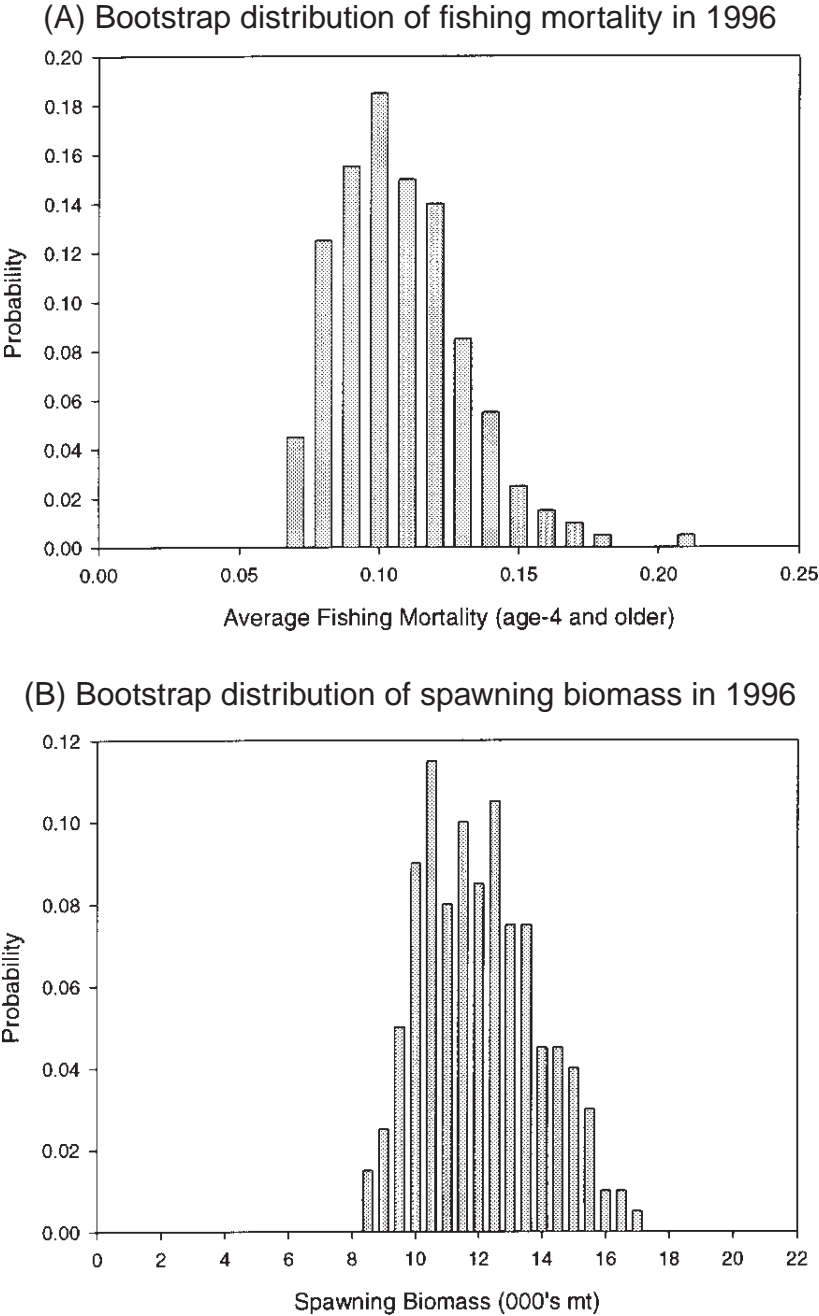


Figure 2. Bootstrap distributions of estimated fishing mortality and spawning biomass of Georges Bank yellowtail flounder in 1996.

where a and b are model parameters and where w is a normal random variable with zero mean and variance σ_w^2 . Based on the most recent assessment of Georges Bank yellowtail flounder, Overholtz et al. (1997) estimated parameters of equation (11) with nonlinear regression and tested model residuals for normality and autocorrelation. Results indicated that an assumption of normally distributed errors could not be rejected and that autocorrelation was not significantly different from 0 for lags of 1 to 6 years. Point estimates of the parameters a and b were 50,090 and 10,737 while the residual variance σ_w^2 was 0.42.

Projections were conducted for 1997-2001 under three F -based management scenarios to illustrate the general approach to making stochastic projections. The three F -based scenarios were: constant F equal to the point estimate of fully recruited fishing mortality in 1996 (F_{1996}), constant F equal to $F_{0.1}$ (Gulland and Borema 1973), and constant F equal to F_{MAX} (Beverton and Holt 1957). These three alternative harvest policies represent a status quo scenario (F_{1996}), a target reference point scenario ($F_{0.1}$), and a limit reference point scenario (F_{MAX}). Values of F_{1996} , $F_{0.1}$, and F_{MAX} for yellowtail flounder were 0.10, 0.24, and 0.61, respectively. The three projections for 1997-2001 used biological and fishery parameters reported in Cadrin et al. (1997), and Overholtz et al. (1997). A total of 100 simulations were conducted for each of the 200 bootstrap replicates of initial population abundance to characterize the empirical distribution of spawning biomass, the probability of achieving the threshold level of spawning biomass, and the empirical distributions of recruitment, landings, and discards.

Projection results indicated that under the status quo scenario, median spawning biomass of Georges Bank yellowtail flounder in the year 2001 and its 80% CI would be 34,300 t with an 80% CI of 25,200 to 49,900 t (Fig. 3A). Under the target reference point scenario, median spawning biomass in 2001 would be 22% lower than under the status quo (Fig. 3B). In contrast, under the limit reference point scenario, median spawning biomass in 2001 would be 55% lower than under the status quo (Fig. 3C). Overall, levels of spawning biomass would likely be higher in 2001 under the status quo and target reference point scenarios in comparison to the limit reference point scenario, although the uncertainty in spawning biomass, as measured by its 80% CI, would also be greater. Further, the spawning biomass threshold of 10,000 t would be reached in 2001 under both the status quo and the target reference point scenarios and would have a good chance of being achieved under the limit reference point scenario (Fig. 4). In contrast, it was unlikely that the long-term spawning biomass target would be achieved in 2001 under any scenario. In particular, the probabilities of achieving the long-term spawning biomass target under the status quo, the target reference point, and the limit reference point scenarios were 0.02, 0.004, and 0, respectively.

In terms of recruitment, landings and discards, the status quo scenario would lead to median recruitment of about 38 million age-1 fish with an 80% CI of 16 to 88 million age fish (Fig. 5A). Median recruitment in 2001

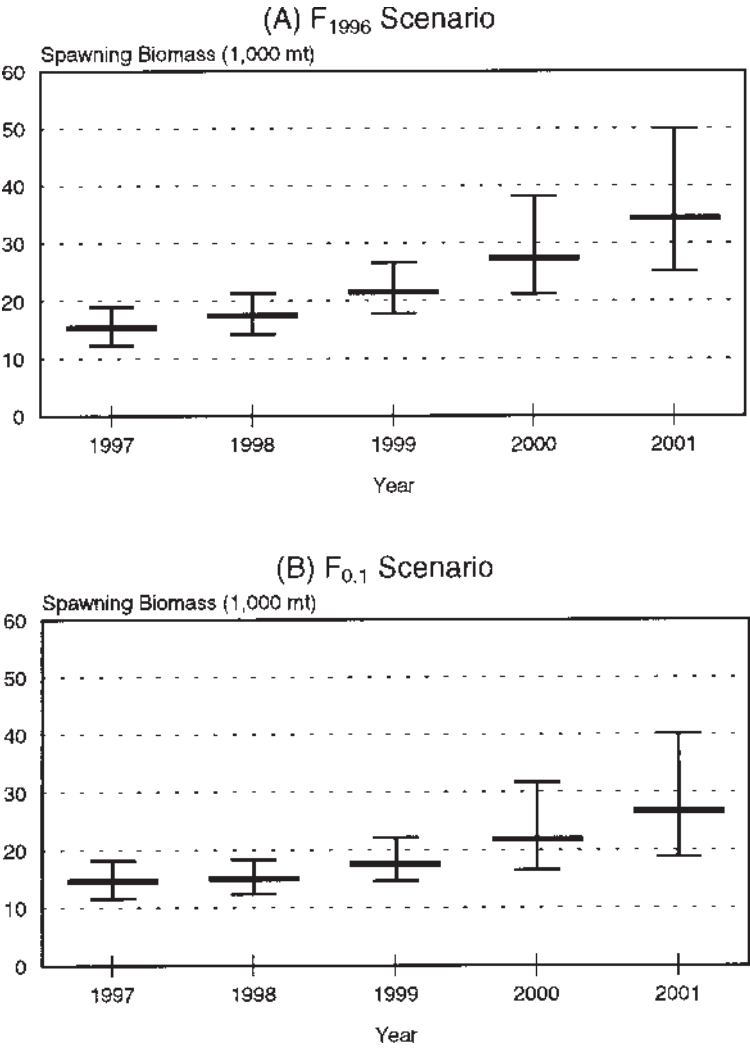


Figure 3. Projections of median spawning biomass of Georges Bank yellowtail flounder and its 80% confidence interval during 1997-2001 under (A) F_{1996} , (B) $F_{0.1}$, and (C) F_{MAX} management scenarios.

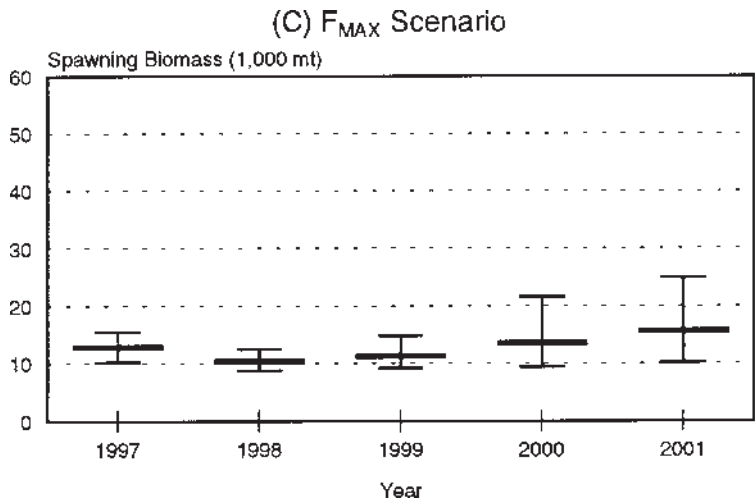


Figure 3. (Continued.)

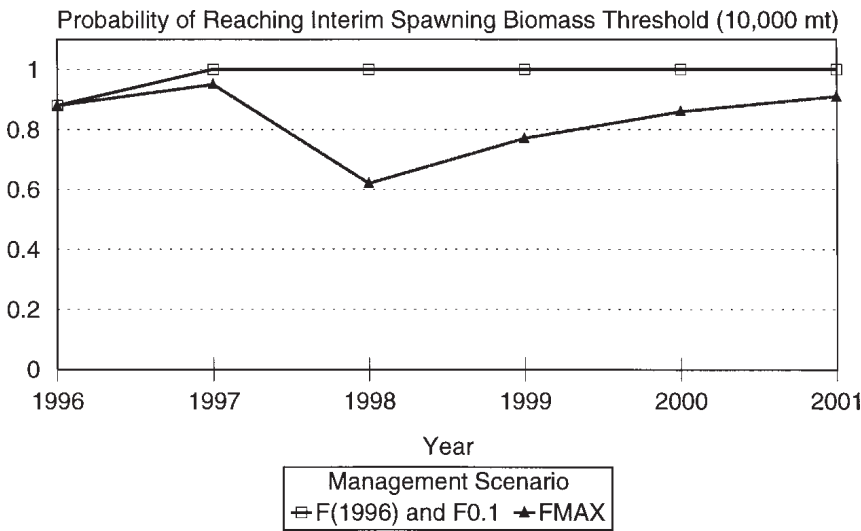


Figure 4. Probabilities of reaching the interim spawning biomass threshold for Georges Bank yellowtail flounder during 1996-2001 under F_{1996} , $F_{0.1}$, and F_{MAX} management scenarios.

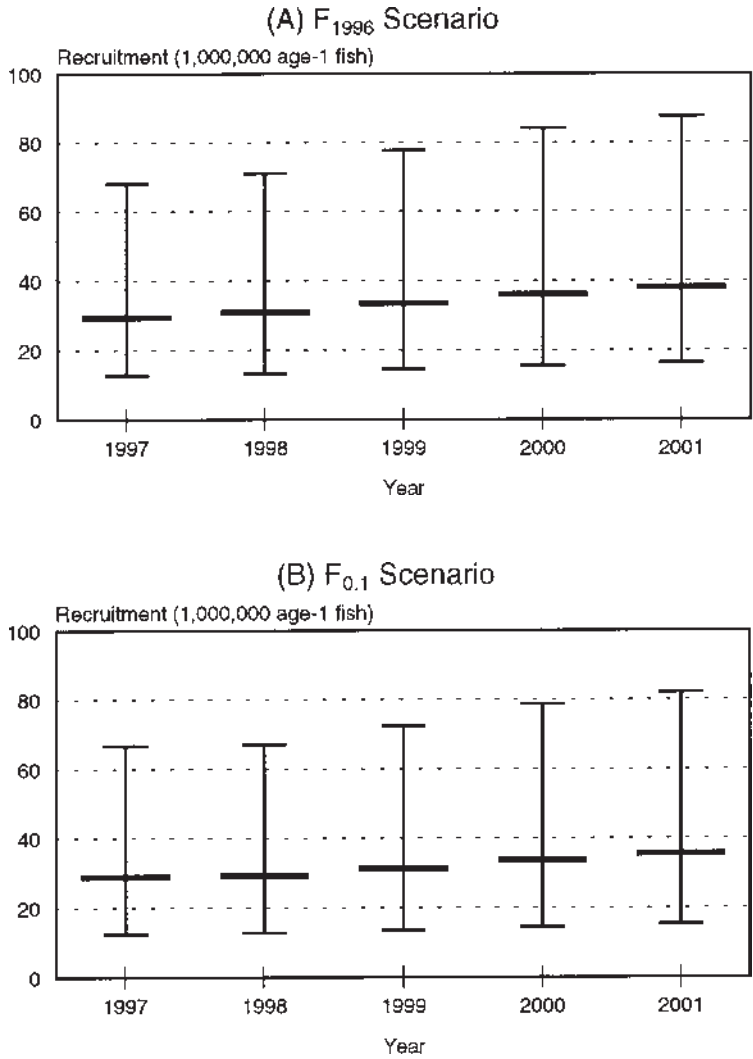


Figure 5. Projections of median recruitment of Georges Bank yellowtail flounder and its 80% confidence interval during 1997-2001 under (A) F_{1996} , (B) $F_{0.1}$, and (C) F_{MAX} management scenarios.

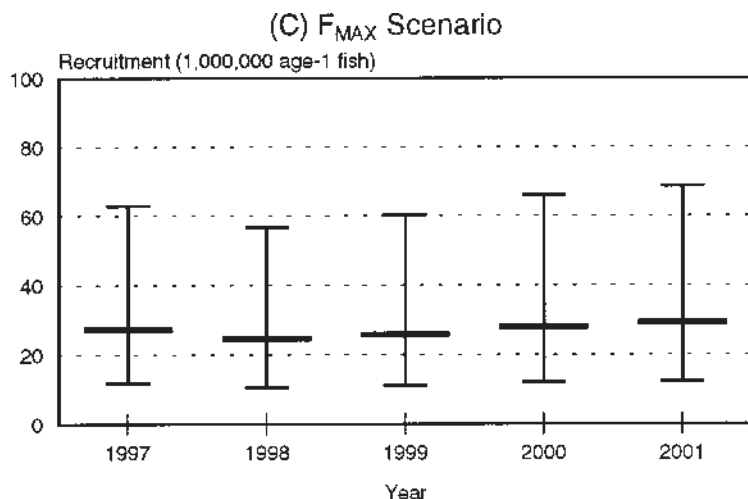


Figure 5. (Continued.)

would be about 6% lower under the target reference point (Fig. 5B) and 23% lower under the limit reference point (Fig. 5C). The status quo scenario would lead to median landings of 2,500 t in the year 2001 with an 80% CI of 1,900 to 3,500 t for yellowtail flounder (Fig. 6A). In comparison, median landings in 2001 under the target reference point would be 72% higher (Fig. 6B). Under the limit reference point scenario, median landings in 2001 would be 128% higher (Fig. 6C). Median discards of Georges Bank yellowtail flounder in 2001 would be 50 t under the status quo scenario with 80% CI of 30 to 80 t (Fig. 7A). In comparison, median discards in 2001 would be 114% and 314% higher under the target (Fig. 7B) and limit reference point scenarios (Fig. 7C), respectively. Overall, levels of landings and discards would likely be greater under the limit reference point scenario, but the uncertainty in these outputs, as measured by the 80% CI, would also be greater than under the status quo and target reference point scenarios. In contrast, recruitment would likely be lower under the limit reference point scenario but would have less uncertainty than under the status quo or target reference point scenarios.

Discussion

Although hypothetical, this illustration of the projection approach provides some general insight for the rebuilding of the Georges Bank yellowtail flounder stock. First, application of the limit reference point F_{MAX} as a harvest rate target is not precautionary for stock rebuilding in the short term and would not likely improve the reproductive potential of this depressed stock. Second,

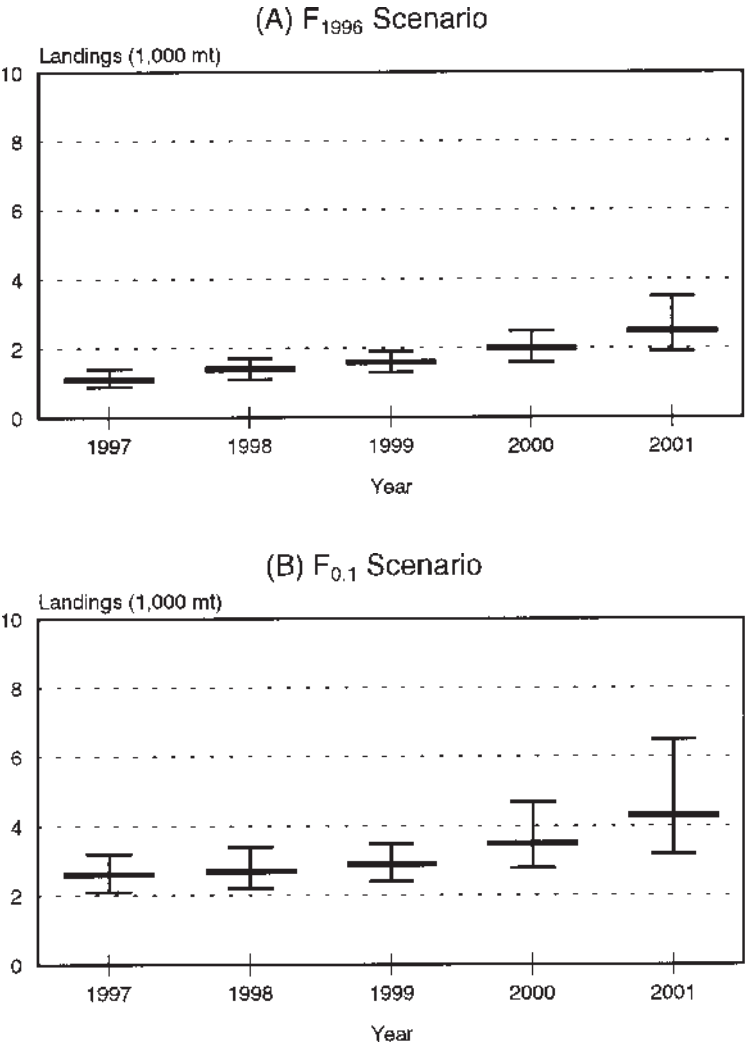


Figure 6. Projections of median landings of Georges Bank yellowtail flounder and its 80% confidence interval during 1997-2001 under (A) F_{1996} , (B) $F_{0.1}$, and (C) F_{MAX} management scenarios.

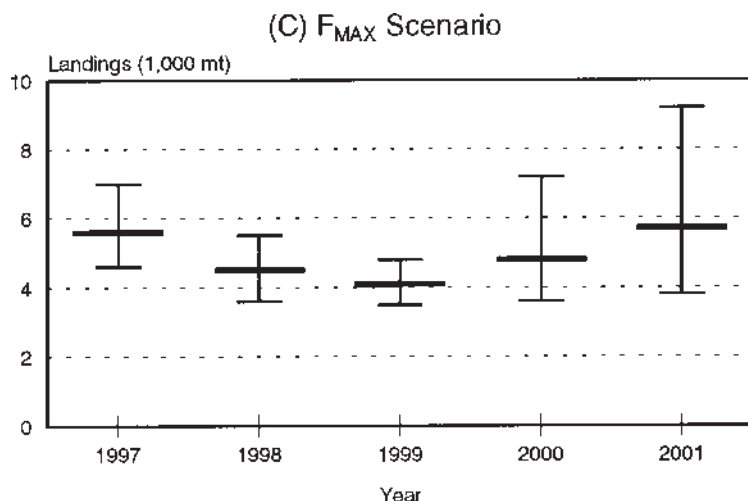


Figure 6. (Continued.)

projections showed that the expected level of variability in spawning biomass would be higher under the status quo scenario but variability in landings would be lower. Conversely, uncertainty in spawning biomass would be lower under the limit reference scenario but uncertainty in landings would be higher. Third, loss of yield due to discarding would be lowest under the status quo scenario and highest under the limit reference point scenario. Last, status quo levels of fishing effort appear sufficient to achieve the interim rebuilding target for Georges Bank yellowtail flounder during 1997-2001. Nonetheless, it is highly unlikely that the long-term target spawning biomass target would be achieved during the projection horizon, although the status quo scenario would lead to the greatest increase in spawning potential. Rebuilding the spawning potential of Georges Bank yellowtail flounder and other depressed New England groundfish stocks through effort control or other conservation measures is an important management consideration because rebuilding would lead to increased benefits (Edwards and Murawski 1993, Overholtz et al. 1995).

The projection approach presented here is not completely general because it does not consider model uncertainty or implementation uncertainty which may be important considerations in developing harvest policy (Rosenberg and Restrepo 1994). In some cases, for example, the model assumption that growth and maturation rates are constant through the projection horizon may not be tenable and density-dependence in growth, maturation, and natural mortality rates may need to be explicitly modeled (Overholtz 1993; Helser and Brodziak 1998). However, whether additional sources of uncertainty will affect the determination of harvest policy is

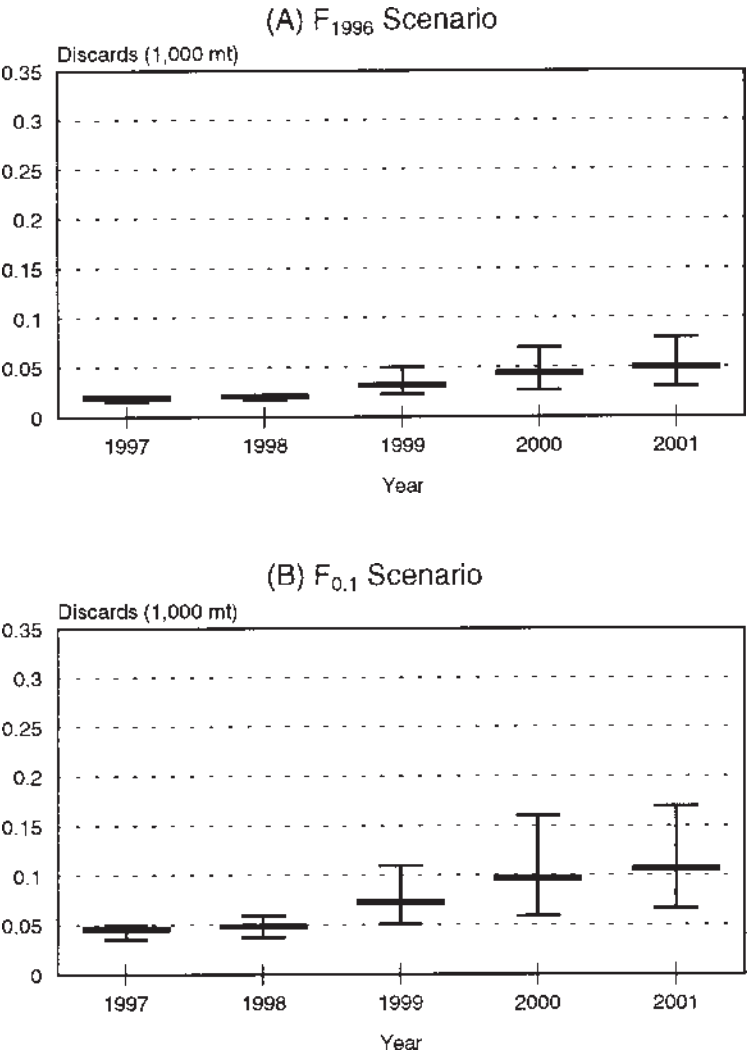


Figure 7. Projections of median discards of Georges Bank yellowtail flounder and its 80% confidence interval during 1997-2001 under (A) F_{1996} , (B) $F_{0.1}$, and (C) F_{MAX} management scenarios.

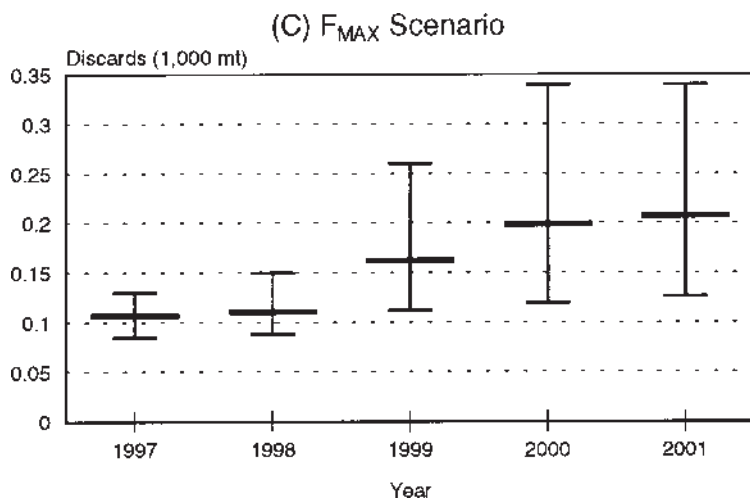


Figure 7. (Continued.)

application specific and dependent upon management objectives (Frederick and Peterman 1995). Although additional sources of random variation can readily be incorporated into this approach, uncertainty in initial population abundance and future recruitment will generally be primary factors to characterize for management advice.

Nonetheless, our approach to making short-term projections is a relatively simple algorithm based on the computationally intensive techniques of bootstrapping and Monte Carlo simulation. Others have used similar approaches to include uncertainty in initial conditions in projections (Mohn 1993, Smith and Gavaris 1993) and our purpose here is to emphasize the generality and flexibility of the approach as a way to evaluate the potential consequences of future management actions. In practice, this approach can readily be applied in any situation where residuals are independent and identically distributed (IID) errors conditioned on model structure or have a relatively simple composite error structure consisting of several IID components (cf. Mohn 1993). Furthermore, the approach is not restricted to age-structured assessment models. For example, this approach has been applied to analytical assessments of surf clams in the northwest Atlantic based on the modified DeLury assessment model (Conser 1994, Weinberg et al. 1995).

Parametric approaches to projecting the consequences of uncertainty in initial population abundance are, of course, also appropriate (Gavaris 1993) and may be computationally expedient in comparison to our resampling approach. However, any parametric approach requires an additional assumption to define the statistical distribution of errors and such assumptions are

another potential source of error in projecting the consequences of alternative harvest policies. In many applications, the nonparametric bootstrap can be recommended because it is flexible and parsimonious and also because it has the desirable theoretical property of providing exact inference for nonparametric maximum likelihood estimation (Efron and Tibshirani 1993).

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References

- Baranov, F.I. 1918. On the question of the biological basis of fisheries. *Nauchn. Issled. Ikhtologicheskii Inst. Izv.* 1:81-128.
- Beverton, R.J.H., and S.J. Holt. 1957. On the dynamics of exploited fish populations. *Fish and Fisheries Series* 11, Chapman and Hall, New York. 533 pp.
- Caddy, J.F., and R. Mahon. 1995. Reference points for fisheries management. *FAO Fish. Tech. Pap.* 347. 83 pp.
- Cadrin, S.X., W.J. Overholtz, J.D. Neilson, S. Gavaris, and S.E. Wigley. 1997. Stock assessment of the Georges Bank yellowtail flounder stock for 1997. *Northeast Fisheries Science Center Reference Document*, Woods Hole, MA.
- Conser, R.J. 1994. Stock assessment methods designed to support fishery management decisions in data-limited environments: Development and application. *Ph.D. dissertation*, Univ. of Washington, Seattle.
- Conser, R.J., and J.E. Powers. 1990. Extension of the ADAPT VPA tuning method designed to facilitate assessment work on tuna and swordfish stocks. *Int. Comm. Cons. Atl. Tunas, Coll. Vol. Sci. Pap.* 32:461-467.
- Conser, R.J., L. O'Brien, and W.J. Overholtz. 1991. An assessment of southern New England and Georges Bank yellowtail flounder stocks. *Northeast Fisheries Science Center Reference Document* 91-03, Woods Hole, MA.
- Deriso, R.B., T.J. Quinn II, and P.R. Neal. 1985. Catch-age analysis with auxilliary information. *Can. J. Fish. Aquat. Sci.* 42:815-824.
- Edwards, S.F., and S.A. Murawski. 1993. Potential economic benefits from efficient harvest of New England groundfish. *N. Am. J. Fish. Manage.* 13:437-449.
- Efron, B. 1982. The jackknife, the bootstrap, and other resampling plans. *SIAM (Soc. Ind. Appl. Math.) CBMS (Conf. Bd. Math. Sci.) Monograph* 38. 92 pp.
- Efron, B., and R. J. Tibshirani. 1993. *An introduction to the bootstrap*. Chapman and Hall, New York. 436 pp.
- Fogarty, M.J., and S.A. Murawski. 1998. Large-scale disturbance and the structure of marine systems: Fishery impacts on Georges Bank. *Ecol. Appl., Suppl.* 8(1):S6-S22.

- Fournier, D.A., and C.P. Archibald. 1982. A general theory for analyzing catch at age data. *Can. J. Fish. Aquat. Sci.* 39:1195-1207.
- Frederick, S.W., and R.M. Peterman. 1995. Choosing fisheries harvest policies: When does uncertainty matter? *Can. J. Fish. Aquat. Sci.* 52:291-306.
- Gavaris, S. 1988. An adaptive framework for the estimation of population size. *Can. Atl. Fish. Sci. Adv. Comm. (CAFSAC) Res. Doc.* 88/29.
- Gavaris, S. 1993. Analytical estimates of the reliability of projected yield from commercial fisheries. In: S.J. Smith, J.J. Hunt, and D. Rivard (eds.), *Risk evaluation and biological reference points for fisheries management*. *Can. Spec. Publ. Fish. Aquat. Sci.* 120:185-191.
- Gulland, J.A., and L.K. Boerema. 1973. Scientific advice on catch levels. *Fish. Bull.* 71:325-336.
- Helser, T.E., and J.K.T. Brodziak. 1998. Impacts of density-dependent growth and maturation on assessment advice to rebuild depleted U.S. silver hake (*Merluccius bilinearis*) stocks. *Can. J. Fish. Aquat. Sci.* 55:882-892.
- Hilborn, R., and C.J. Walters. 1992. *Quantitative fisheries stock assessment*. Chapman and Hall, New York. 570 pp.
- Hilborn, R., E.K. Pikitch, and M.K. McAllister. 1994. A Bayesian estimation and decision analysis for an age-structured model using biomass survey data. *Fish. Res.* 19:17-30.
- Laurec, A., and J.G. Shepherd. 1983. On the analysis of catch and effort data. *J. Cons. Cons. Int. Explor. Mer* 41:81-84.
- McAllister, M.K., and J.N. Ianelli. 1997. Bayesian stock assessment using catch-age data and the sampling-importance resampling algorithm. *Can. J. Fish. Aquat. Sci.* 54:284-300.
- Megrey, B.A. 1989. Review and comparison of age-structured stock assessment models from theoretical and applied points of view. In: E. Edwards and B. Megrey (eds.), *Mathematical analysis of fish stock dynamics: Reviews and current applications*. *Amer. Fish. Soc. Symp.* 6:8-48.
- Mendelssohn, R. 1988. Some problems in estimating population sizes from catch-at-age data. *Fish. Bull. U.S.* 86:617-630.
- Methot, R.D. 1989. Synthetic estimates of historical abundance and mortality for northern anchovy. In: E. Edwards and B. Megrey (eds.), *Mathematical analysis of fish stock dynamics: Reviews and current applications*. *Amer. Fish. Soc. Symp.* 6:66-82.
- Methot, R.D. 1990. Synthesis model: An adaptable framework for analysis of diverse stock assessment data. *Int. North Pac. Fish. Comm. Bull.* 50:259-277.
- Mohn, R.K. 1993. Bootstrap estimates of ADAPT parameters, their projection in risk analysis and their retrospective patterns. In: S.J. Smith, J.J. Hunt, and D. Rivard (eds.), *Risk evaluation and biological reference points for fisheries management*. *Can. Spec. Publ. Fish. Aquat. Sci.* 120:173-184.
- Mohn, R.K., and R. Cook. 1993. Introduction to sequential population analysis. *Northwest Atl. Fish. Organ. Sci. Coun. Studies* 17.

- New England Fishery Management Council. 1996. Amendment #7 to the Northeast Multispecies Fishery Management Plan. New England Fishery Management Council, Saugus, MA.
- Northeast Fisheries Science Center. 1995. Status of the fishery resources off the northeastern United States for 1994. NOAA Tech. Memo NMFS-NE-108.
- Northeast Fisheries Science Center. 1997. Report of the 24th Northeast Regional Stock Assessment Workshop. Northeast Fisheries Science Center Reference Document, Woods Hole, MA. 290 pp.
- Overholtz, W.J. 1993. Harvesting strategies and fishing mortality reference point comparisons for the northwest Atlantic stock of Atlantic mackerel (*Scomber scombrus*). Can. J. Fish. Aquat. Sci. 50:1749-1756.
- Overholtz, W.J., S.F. Edwards, and J.K.T. Brodziak. 1995. Effort control in the New England groundfish fishery: A bioeconomic perspective. Can. J. Fish. Aquat. Sci. 52:1944-1957.
- Overholtz, W., S.A. Murawski, P.J. Rago, W.L. Gabriel, and M. Terceiro. 1997. Ten-year projections of landings, spawning biomass, and recruitment for five groundfish stocks considered at SAW 24. Northeast Fisheries Science Center Reference Document, Woods Hole, MA.
- Parrack, M.L. 1986. A method of analyzing catches and abundance indices from a fishery. Int. Comm. Cons. Atl. Tunas, Coll. Vol. Sci. Pap. 24:209-221.
- Punt, A.E. and D.S. Butterworth. 1993. Variance estimates for fisheries assessment: Their importance and how best to evaluate them. Can. Spec. Publ. Fish. Aquat. Sci. 120:145-162.
- Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bull. Fish. Res. Board Can. 191.
- Rosenberg, A.A., and V.R. Restrepo. 1994. Uncertainty and risk evaluation in stock assessment advice for U.S. marine fisheries. Can. J. Fish. Aquat. Sci. 51:2715-2720.
- Schnute, J.T. 1994. A general framework for developing sequential fisheries models. Can. J. Fish. Aquat. Sci. 51:1676-1688.
- Smith, S.J., and S. Gavaris. 1993. Evaluating the accuracy of projected catch estimates from sequential population analysis and trawl survey abundance estimates. In: S.J. Smith, J.J. Hunt, and D. Rivard (eds.), Risk evaluation and biological reference points for fisheries management. Can. Spec. Publ. Fish. Aquat. Sci. 120:163-172.
- Weinberg, R.J., S.A. Murawski, R. Conser, J. Brodziak, L. Hendrickson, H.-L. Lai, and P. Rago. 1995. Analytical assessment of surfclam populations in the Middle Atlantic region of the United States. Northeast Fisheries Science Center Reference Document 95-05, Woods Hole, MA.

A Simulation Study of the Effects of Aging Error and Sample Size on Sustained Yield Estimates

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Abstract

A Monte Carlo simulation model of an exploited age-structured fish population was constructed to evaluate the effects of sampling and aging the catch on estimates of population parameters from catch-age analysis and resultant estimates of sustained yield. Aging error, error in the annual catch estimates, and error in the annual survey exploitable abundance estimates were included in the simulation. Three sets of simulation runs were conducted among small (100), medium (300), and large (900) sample sizes to estimate the catch-at-age composition. The simulation included five reader types with differing aging abilities resulting in a total of five catch-at-age compositions evaluated for each program replication. Sustained yield estimates from 1,000 replications of each reader type were compared to true sustained yield. Aging bias and imprecision had dramatic effects on estimated sustained yield: positive aging bias and imprecision generally caused underestimation of sustained yield, while negative aging bias caused overestimation of sustained yield. These results suggest that this component in stock assessment deserves greater attention in planning and practice. In addition, increases in sample size resulted in increased precision in sustained yield estimates only if the catch-at-age composition was estimated using a precise and unbiased reader type. While this evaluation was based on the Arctic grayling (*Thymallus arcticus*) population within the Gulkana River in Alaska, the simulation model can be

used for a variety of sport fish populations; a diskette and user manual are available.

Introduction

Fisheries management should be rooted in sound knowledge of all factors contributing to the dynamics of the exploited fish population (Summerfelt 1987). These factors include, but are not limited to, understanding the basic life history of the species, exploitation patterns of the prosecuting fisheries, and environmental factors that influence the abundance and distribution of the species. Once these basic biological parameters have been uncovered, various mathematical constructs to model population dynamics exist which can be used to define patterns and levels of exploitation to achieve management objectives. One model of particular utility in estimating sustained yield was proposed by Quinn and Szarzi (1993). Quinn and Szarzi's model (QS) assumes a constant fishing mortality harvest policy and relies on the results of catch-age analysis with auxiliary information (Deriso et al. 1985, 1989) to generate estimates of sustained fishing mortality and subsequent estimates of sustained yield in a Leslie matrix (Getz and Haight 1989) framework.

The concept of managing fisheries to achieve sustained yield is a management goal that is widely embraced but rarely explicitly defined. Various management objectives can result in a sustained yield policy but can have different implications with regard to the productivity of the stock, the expected age composition of the catch, and many other population parameters (Deriso 1987, Sissenwine and Shepard 1987, Clark 1991). Our study addressed one of the many management objectives which result in sustained yield, that is, the sustained yield (SY_{ST}) that results from applying a constant fishing mortality rate (F_{ST}) that tends to force the population to a steady long-term equilibrium abundance and age composition (Quinn and Szarzi 1993). This specific objective, conceptually equivalent to F_{rep} (Sissenwine and Shepard 1987), is applicable for many sport fishery management programs where the prevailing goal is "to maintain the historic size and age composition of the exploited population." Translating this broad goal into a specific management objective that is both quantifiable and defensible allows for evaluation of the successes and failures associated with management policies aimed at maintaining fish stocks in equilibrium. The QS model provides a structure to estimate F_{ST} and resultant SY_{ST} associated with a specific exploited fish population.

Like many complex age-structured analyses, the QS model relies on catch-at-age data which are subject to introduced error from sampling and age determination (aging error). Aging error occurs when a reader responsible for estimating the age of a structure assigns an age, called an age reading, that is different from the true age. Aging error can occur as a result of aging imprecision, defined as the variability among multiple readings of a single structure, aging bias, defined as the difference between

the expected value of the observed age and the true age, or a combination of these two (Kimura and Lyons 1991).

Catch-age analysis with auxiliary information (CAGEAN) is the cornerstone to the QS model. CAGEAN belongs to a class of models generally known as age-structured stock assessment models (ASA; Megrey 1989). The effects of sample size and aging error on model estimates have been dealt with by various researchers in both ASA type models and in yield-per-recruit or biomass based models (Lai and Gunderson 1987, Bradford 1991). However, the effect of sample size and aging error has not been examined as related to sustained yield estimates from a combined ASA and Leslie matrix model like the QS model. The intent of this paper is to investigate the performance of the QS model given various sample sizes and types of aging error.

Methods

A FORTRAN computer program was constructed to model an exploited age-structured fish population and evaluate the effect of the processes of sampling and aging the catch on estimates of population parameters from catch-age analysis and resultant estimates of sustained yield. The program (AGEERR) contains four procedures: (1) generation of an exploited age-structured fish population with associated true catch- and abundance-at-age; (2) construction of observed catch-at-age data by incorporating measurement error due to sampling, aging error, and variability in the total catch; (3) estimation of population parameters through catch-age analysis with auxiliary information based on observed catch-at-age and survey data; and (4) estimation of sustained yield using the population parameter estimates from catch-age analysis with auxiliary information. Descriptions of the formulations of each of the procedures as well as the methods used to conduct the case study using Gulkana River grayling, an important sport fish population in Alaska, are presented in the following sections. The program can be used for a variety of sport fish populations; a diskette and user manual are available (Coggins 1997).

Generation of the True Population

To simulate the population under investigation, a time series of true catch- and abundance-at-age was generated using the typical recursion and Baranov catch equations (Baranov 1918). The relevant formulae are given as:

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

$$N_{A+,t+1} = N_{(A+)-1,t} e^{-Z_{(A+)-1,t}} + N_{A+,t} e^{-Z_{A+,t}}, \quad (2)$$

$$C_{a,t} = \mu_{a,t} N_{a,t}, \quad (3)$$

$$\mu_{a,t} = \frac{F_{a,t}}{Z_{a,t}} [1 - e^{-Z_{a,t}}], \quad (4)$$

$$Z_{a,t} = F_{a,t} + M, \quad (5)$$

$$F_{a,t} = s_a f_t, \quad (6)$$

where $N_{a,t}$ is the true abundance of age a fish in year t , $Z_{a,t}$ is the total instantaneous mortality rate, $A+$ is an aggregate age group, $C_{a,t}$ is the true catch, $\mu_{a,t}$ is the true exploitation rate, M is the true instantaneous natural mortality rate, $F_{a,t}$ is the true fishing mortality rate, s_a is the true gear selectivity coefficient, and f_t is the true full recruitment fishing mortality (Deriso et al. 1985, 1989). Equation (2) is a generalization of (1) appropriate when an aggregate age class ($A+$) is used. The common assumption of separable fishing mortality (Doubleday 1976, Pope 1977, Fournier and Archibald 1982, Pope and Shepherd 1982, Deriso et al. 1985) is implicit in equation (6).

Observed Catch-at-Age Data

Observed catch-at-age was calculated as the product of the observed catch-at-age proportion and the observed total catch as:

$$C'_{a,t} = C'_t \theta'_{a,t}, \quad (7)$$

where $C'_{a,t}$ is the observed catch of age a fish during year t , C'_t is the observed total catch, and $\theta'_{a,t}$ is the observed catch-at-age proportion. Variability was included in both the observed total catch and the observed catch-at-age composition. To mimic the uncertainty in the estimates of total catch, it was assumed that the observed total catch followed a normal distribution with expected value equal to the true total catch and a constant coefficient of variation (cv_C):

$$C'_t \sim N(C_t, \sigma_{C(t)}^2), \quad (8)$$

$$\sigma_{C(t)}^2 = (cv_C C_t)^2, \quad (9)$$

where C_t is the true total catch in year t , $\sigma_{C(t)}^2$ is the variance of the observed total catch, and cv_C is the constant coefficient of variation of the observed catch. This formulation allows one to specify the precision of the observed catch simply by an assumed constant coefficient of variation. Other forms of variance based on empirical sampling results could also be considered (Crone and Sampson 1998).

The observed age composition of the catch was generated by sampling and aging the true catch and incorporating error in the aging process. Catch sampling was conducted randomly without replacement such that the probability of selecting a fish of a particular age was equal to the true proportion of that age remaining in the catch.

Modeling Aging Error

The simulation uses multiple reader types to generate a set of aging error distributions that can be compared objectively. If a is the observed age, b is the true age, c is the constant bias of the inaccurate reader, σ is the constant standard deviation (SD), and $\sigma(b)$ is the SD as a function of true age, the five reader types are described as follows. Reader 0 (R0) is a perfect reader without aging error [$a = b$]. Reader 1 (R1) is characterized as the ideal “real life” reader, being both accurate (no bias) and precise (low variability) [$a \sim N(b, \sigma)$]. Reader 2 (R2) is accurate but less precise than reader R1 [$a \sim N[b, \sigma(b)], \sigma \leq \sigma(b)$]. Reader 3 (R3) is inaccurate but precise [$a \sim N(b+c, \sigma)$]. Reader 4 (R4) is inaccurate and imprecise [$a \sim N[b+c, \sigma(b)]$]. Readers R1 and R3 always have identical constant and high precision relative to readers R2 and R4. Readers R2 and R4 have identical and variable low precision as a function of true age that is always defined to be less than or equal to the constant precision of readers R1 and R3.

Classification matrices were used to specify aging precision for each reader type. Richards et al. (1992) present the following formulation for constructing a classification matrix that is central to modeling aging error in this study. The classification matrix is a function of the parameter vector $\Phi = (\sigma_r, \sigma_A, \alpha)$ and specified as:

$$\sigma(b) = \begin{cases} \sigma_r + (\sigma_A - \sigma_r) \frac{1 - e^{-\alpha(b-r)}}{1 - e^{-\alpha(A-r)}}; & \alpha \neq 0 \\ \sigma_r + (\sigma_A - \sigma_r) \frac{b-r}{A-r}; & \alpha = 0 \end{cases} \quad (10)$$

$$\chi_{ab}(\Phi) = \frac{1}{\sqrt{2\pi} \sigma(b)} e^{-\frac{1}{2} \left[\frac{a-b}{\sigma(b)} \right]^2}, \quad (11)$$

$$q(a|b, \Phi) = \frac{\chi_{ab}(\Phi)}{\sum_{a=r}^A \chi_{ab}(\Phi)}, \quad (12)$$

where σ_r is the SD associated with recruit-age fish, σ_A is the SD associated with the oldest age fish, α is a parameter that governs the non-linearity of $\sigma(b)$, and $\chi_{ab}(\Phi)$ is the discrete normal probability density function of a

given b . The preceding three equations define the classification matrix $[Q(\Phi)]$ made up of the row vectors q , which are probabilities that a fish of true age b is assigned an observed age a .

There are two assumptions regarding the formulation of the classification matrix. The first assumption is:

$$q(b | b, \Phi) \geq q(a | b, \Phi), a \neq b.$$

This is the “modal” probability assumption and asserts that fish of true age b are assigned an observed age a equal to b with higher probability than any other observed age. The second assumption is:

$$\sum_{a=r}^A q(a | b, \Phi) = 1 \text{ for each } b.$$

This assumption assures that the probabilities of all the possible observed ages for any given true age sum to one.

Although the modal assumption assures that observed age a equals true age b with highest probability, observed age a cannot be said to be unbiased. This is because the observed age must lie within the lifespan of the species (or the true age range recruited to a fishery) and truncation of the observed age distribution occurs near the bounds of the true age range. For example, if the youngest fish of a given species ever captured had a true age of 3, the probability of assigning an observed age $a < 3$ would be zero. Therefore, the age of fish with a true age of 3 would tend to be overestimated. A similar argument can be made regarding the underestimation of the age of older fish.

Therefore, the simulation also allows for the inclusion of systematic bias by specifying a constant bias c . This was accomplished by adding the bias after the true age was modified with imprecision so that the imprecision was specified based on the true age. A negative value of the bias c caused under-aging of a structure while a positive value caused over-aging. It should be noted however, that the bias will not cause the observed age to traverse the true age range. For example, application of a negative bias will still result in an observed age no smaller than 2 if the lower bound of true age is equal to 2. Alternate formulations such as including bias as a function of true age may also be appropriate to consider in future investigations.

Catch-Age Analysis with Auxiliary Information

Catch-age analysis with auxiliary information was performed using a variant of CAGEAN (Deriso et al. 1985) called CAGEM. In CAGEM, a multinomial-like measurement error structure is assumed, appropriate given aging error in the observed catch-at-age data. We refer to measurement error as the deviation of an observed quantity from its true value in the course of

sampling and aging, as is common in stock assessment (Deriso et al. 1985). The relevant formulae for the objective function for parameter estimation and survey exploitable abundance are:

$$\min \left\{ \sum_{a,t} \frac{[C'_{a,t} - \hat{C}_{a,t}]^2}{\hat{C}_{a,t}} + \lambda_s \left[\ln(SUR_t) - \ln(\hat{EN}_t) \right]^2 \right\}, \quad (13)$$

$$\hat{EN}_t = \sum_a \hat{N}_{a,t} \hat{s}_a, \quad (14)$$

where $C'_{a,t}$ is the observed catch of age a fish in year t , $\hat{C}_{a,t}$ is the estimated catch, SUR_t is the total exploitable abundance from an independent survey, \hat{EN}_t is the estimated total exploitable abundance, λ_s is a weighting factor for the auxiliary survey information, $\hat{N}_{a,t}$ is the estimated abundance, and \hat{s}_a is the estimated selectivity-at-age. The catch $\hat{C}_{a,t}$, abundance $\hat{N}_{a,t}$, and selectivity \hat{s}_a were estimated within program CAGEM using equations (1)-(6).

For the simulation model, variability in the survey exploitable abundance was specified with a coefficient of variation as:

$$SUR_t \sim N[EN_t, \sigma_{s(t)}^2], \quad (15)$$

$$\sigma_{s(t)}^2 = [cv_s(EN_t)]^2, \quad (16)$$

where EN_t is the true exploitable abundance of age a fish in year t , $\sigma_{s(t)}^2$ is the variance of the survey total exploitable abundance around the true exploitable abundance, and cv_s is the assumed constant coefficient of variation of the survey total exploitable abundance.

Estimation of Sustained Yield

The estimation of sustained yield (SY_{ST}) assumes that the population is exploited with a constant fishing mortality (F_{ST}) and that early life survival, natural mortality and fecundity-at-age are constant. The management objective associated with SY_{ST} seeks to find F_{ST} that will take the population to its steady long-term equilibrium abundance and age composition. This was accomplished by finding F_{ST} that caused the net reproductive value of an r year-old fish (age at recruitment to the fishery) to equal 1. A simple interpretation of this equilibrium condition is that every fish recruited to the fishery must on average produce 1 recruit at age r . In the course of finding F_{ST} , it is necessary to estimate early life survival. This was accomplished by taking the mean of the annual estimates of early life

survival. Once F_{ST} was found, SY_{ST} in the year following the last year of the analysis was estimated by projecting the abundance using fecundity-at-age, average early life survival, and survival-at-age. The relevant formulae for the estimation of F_{ST} and SY_{ST} are found in Quinn and Szarzi (1993).

Gulkana River Grayling Case Study

To examine the effect of sample size and aging error on estimates of sustained yield, AGEERR was used to simulate the estimation of sustained yield for the Arctic grayling (*Thymallus arcticus*) population of the main-stem Gulkana River (Bosch 1995). The population parameters estimated by Bosch (1995) were used to simulate the population among 24 runs of AGEERR, performed according to a factorial design with three sample sizes and eight levels of aging error (Table 1). Each of the runs from AGEERR used 1,000 Monte Carlo replications.

Bosch (1995) estimated sustained fishing mortality and yield (F_{ST} , SY_{ST}) using the QS model. The estimates of SY_{ST} , F_{ST} , and fecundity-at-age estimated by Bosch are reported in Table 2. Note that since Bosch used an aggregate age group of 7+ but considered the population to have a longevity of age 10, the plus group abundance was apportioned to the contributing age classes and the fecundity of the plus group was estimated (Coggins 1997). The parameter cv_s was set equal to the average annual coefficient of variation (22%) of the exploitable abundance estimates from mark-recapture studies of Gulkana River grayling (Bosch 1995). An estimate of cv_c was not available so an arbitrary value of 5% was used.

For the purposes of this study, we assumed that it was possible to select a single perfectly random sample of fish from the entire catch. Furthermore, three sample sizes (100, 300, and 900) were used to evaluate the effect of sample size on the sustained yield estimates from the QS model. The sample sizes were picked to generally reflect sampling rates which would be below, approximately equal to, or above acceptable limits to achieve age-class proportion estimates that were within 5% of the true value with 80% confidence assuming a strict multinomial error structure (Thompson 1987).

Two levels of aging precision (low and high cases) and 4 levels of bias were used to incorporate aging error in the simulations (Table 1). To specify aging precision germane to the R2 and R4 type readers [$\sigma(b)$] in what we call the low precision case, a data set was constructed containing estimated ages of scales collected from grayling before and after a known time-at-large (i.e., after capture and recapture in a mark-recapture experiment). These data were from grayling collected at Fielding Lake, and the Chatanika, Chena, Salcha, and Gulkana rivers in Alaska (Merritt and Fleming 1991, Bosch 1995). To estimate the standard deviation of observed age given a true age, we assumed that the first reading (at initial capture) was correct and the expected age of the second reading (at recapture) was the sum of the first reading and the time-at-large. The data were then

Table 1. Input parameters for each simulation run of AGEERR.

Run	Sample size	Aging precision	Aging bias (c)	σ_r^a	σ_A^b	α^c	σ^d	cv_s^e	cv_c^f	Age at full selectivity	Age at full maturity (m)	Pooling age
1	100	High	-1	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
2	100	High	-2	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
3	100	High	1	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
4	100	High	2	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
5	100	Low	-1	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
6	100	Low	-2	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
7	100	Low	1	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
8	100	Low	2	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
9	300	High	-1	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
10	300	High	-2	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
11	300	High	1	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
12	300	High	2	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
13	300	Low	-1	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
14	300	Low	-2	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
15	300	Low	1	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
16	300	Low	2	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
17	900	High	-1	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
18	900	High	-2	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
19	900	High	1	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
20	900	High	2	0.41	0.64	-0.25	0.41	0.22	0.05	3	6	7
21	900	Low	-1	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
22	900	Low	-2	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
23	900	Low	1	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7
24	900	Low	2	0.82	1.28	-0.25	0.82	0.22	0.05	3	6	7

^a σ_r = Standard deviation of observed age given recruit age fish.
^b σ_A = Standard deviation of observed age given oldest age fish.
^c α = Parameter governing the non-linearity of observed age as a function of true age.
^d σ = Constant standard deviation of reader types R1 and R3.
^e cv_s = Coefficient of variation of the survey data.
^f cv_c = Coefficient of variation of the total catch data.

sorted by expected age and the standard deviation of observed age was estimated for each expected age class. Both formulations of equation (10) were then fit to the data to estimate σ_r , σ_A , and α . Although the three parameter case was not found to have a significantly better fit than the two parameter case ($p = 0.57$) using an F -test proposed by Schnute (1981), the three parameter case was used throughout the simulations so that precision of readers R2 and R4 could be specified non-linearly. The constant aging precision of readers R1 and R3 (σ) in the low precision case was set equal to the parameter estimate for σ_r .

As described above, the parameters of the function $\sigma(b)$ in equation (10) in the low precision case were estimated from available data, and the low precision case of σ set equal to the estimated σ_r . For the high precision case, the σ_r and σ_A parameters of the function $\sigma(b)$ were set equal to

Table 2. Estimated sustained yield (SY_{ST}), sustained fishing mortality (F_{ST}), and population parameters for the grayling stock of the Gulka-na River in Alaska either contained in, or estimated from, Bosch (1995). Selectivity and fecundity are assumed to be constant over the time of the simulation.

Sustained fishing mortality	$F_{ST} = 0.402$
Sustained yield (number of fish)	$SY_{ST} = 29,867$
Projected abundance in 1992 (number of fish)	Proj. Abu. = 153,133

Fishing mortality	
Year	f_t
1986	0.330
1987	0.232
1988	0.190
1989	0.154
1990	0.059
1991	0.081

Abundance (number of fish)

Year	Age					
	2	3	4	5	6	7+
1986	34,253	30,311	23,874	4,540	2,331	81
1987	41,271	23,852	16,127	12,702	2,416	1,283
1988	22,390	29,274	14,007	9,471	7,459	2,172
1989	46,004	16,010	17,946	8,587	5,806	5,905
1990	90,218	33,117	10,175	11,405	5,457	7,443
1991	16,940	66,100	23,125	7,105	7,964	9,008

	Age					
	2	3	4	5	6	7+
Selectivity	0.187	1	1	1	1	1
Net fecundity (eggs)	124	469	975	1,616	2,739	4,140

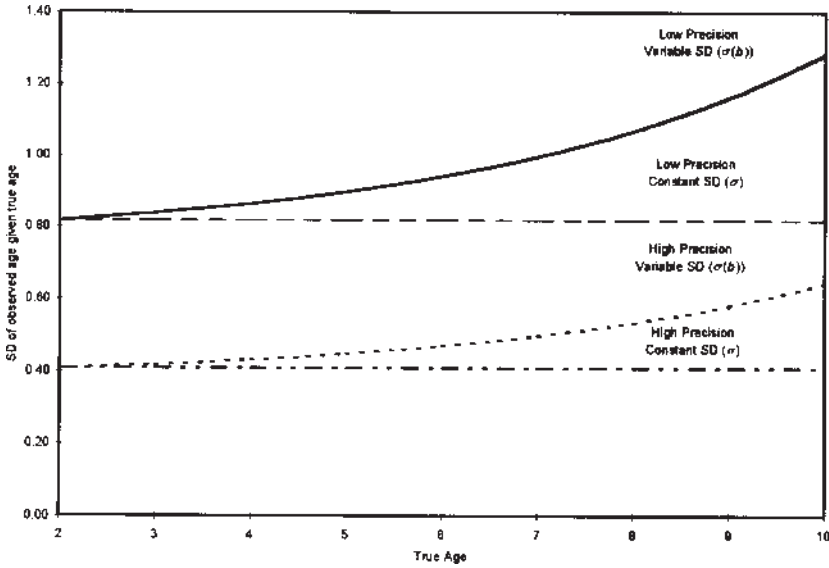


Figure 1. Standard deviation (SD) of observed age a given true age b for both the high and low precision cases of variable $[\sigma(b)]$ and (σ) constant SD.

one half the values in the low precision case, and the constant value of σ was set equal to σ_r (Table 1 and Fig. 1). The bias levels were set equal to -1, -2, 1, and 2.

Lambda (λ_s), the weighting parameter that specifies the amount of influence the survey data has in parameter estimation (equation 13), was specified as 1,300 based on simulations of CAGEM during its development. To test the performance of $\lambda_s = 1,300$, a simulation set was run with no specified error in any of the input parameters. The parameter estimates given the R0 reader type and a sample size of 1,000 differed from the true parameter estimates by a negligible amount.

To evaluate the process of estimating sustained yield with the QS model and the variable input data detailed above, two statistics were examined. The first was the relative error, an indicator of bias, between the parameter estimate and the true value of that parameter. Relative error is calculated as:

$$RE = \frac{\hat{\theta} - \theta}{\theta}, \quad (17)$$

where RE is the relative error, θ is the true value of the parameter, and $\hat{\theta}$ is the parameter estimate. In a Monte Carlo framework, the parameter estimate

is equal to the mean of the estimates of that parameter among the Monte Carlo replications.

Following Hightower (1996), the second statistic was the proportion of replicate parameter estimates among Monte Carlo replications that were within 10% of the true value. This statistic, like relative error, is an indicator of bias but has the additional property of revealing the precision of the process by estimating the probability of obtaining an “accurate” estimate (relative error < 10%).

Results

Overview of Sustained Yield Estimation

The estimation of sustained yield relies directly on estimates of sustained fishing mortality, projected abundance, and selectivity of age-2 fish. In general, an increase in any one of these parameters caused an increase in sustained yield. Moreover, negatively biased estimates of sustained fishing mortality and projected abundance resulted in negatively biased estimates of sustained yield. The exceptions occurred when negative aging bias caused extreme overestimation (relative error > 750%) of the selectivity of age-2 fish and offset the effect of underestimation from sustained fishing mortality and projected abundance. To show these general results, we averaged over all scenarios and expressed results in terms of relative error (Fig. 2).

The R0 reader type produced highly accurate (relative error < 5%) but slightly negatively biased estimates of SY_{ST} , F_{ST} , and projected abundance, as well as positively biased estimates of the selectivity of age-2 fish (relative error < 10%; Fig. 2). Under high aging precision, the R1 and R2 reader types produced accurate estimates of SY_{ST} (relative error < 10%), and fairly inaccurate estimates of SY_{ST} (relative error < 40%) under low aging precision. In all cases of the R1 and R2 reader types, F_{ST} and projected abundance were negatively biased while the selectivity of age-2 fish was positively biased.

Under both the high and low cases of aging precision, positive aging bias resulted in extreme underestimation of SY_{ST} (nearly 100% relative error) from both the R3 and R4 reader types. This was primarily a result of extreme underestimation of F_{ST} (roughly 100%), but also a result of underestimation of projected abundance. In addition, the positive aging bias caused the exclusion of age-2 (+1 bias) and age-2 and age-3 fish (+2 bias) from the analyses and therefore, selectivity of age-2 fish was not estimated. Under negative aging bias, SY_{ST} was overestimated for both high and low aging precision using the R3 and R4 reader types. This was a result of moderate overestimation of F_{ST} (relative error < 12%) and huge overestimation of the selectivity of age-2 fish (relative error > 750%). The low precision case produced more accurate estimates of SY_{ST} than the high precision case from the R3 and R4 reader types under negative aging bias.

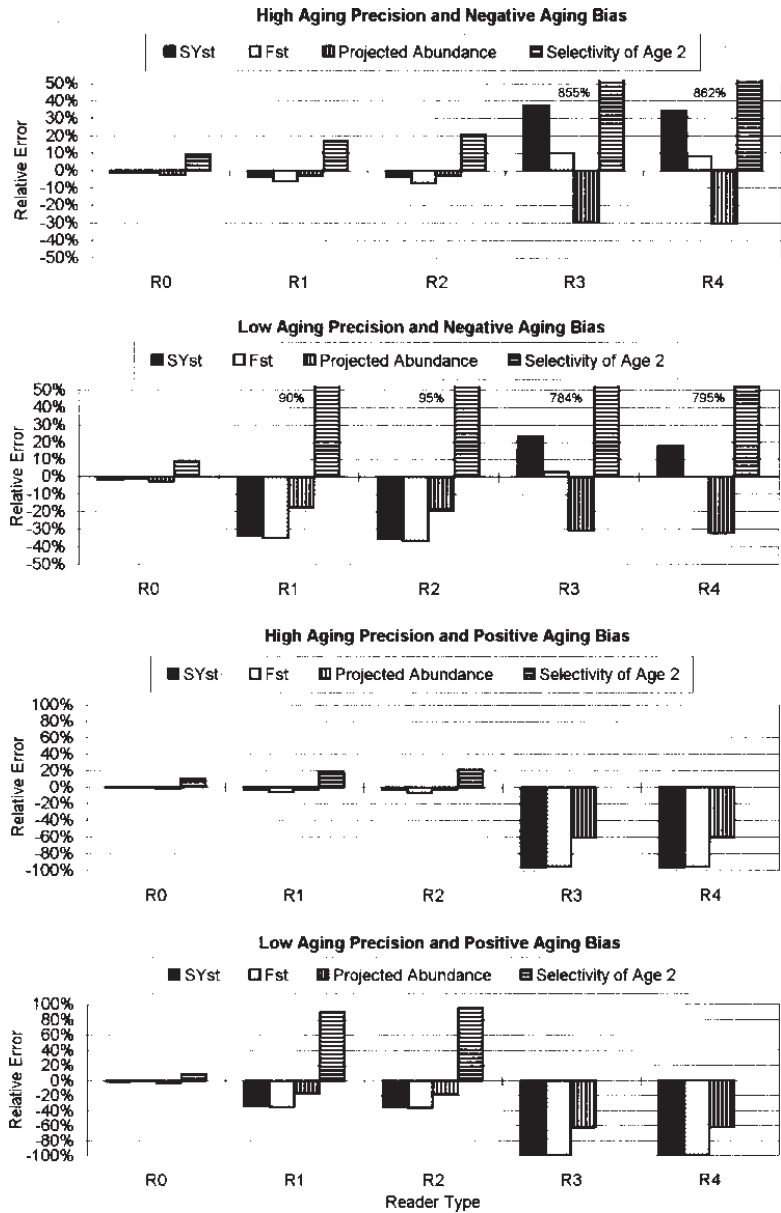


Figure 2. Average relative error in estimates of sustained yield (SY_{ST}), sustained fishing mortality (F_{ST}), projected abundance, and selectivity of age-2 fish under high and low aging precision and negative (two upper panels) and positive (two lower panels) aging bias, among all samples sizes and reader types (see Modeling Aging Error for descriptions of reader types R0-R4).

Sustained Yield by Reader Types and Sample Size

In this section, detailed examination of sustained yield estimates by reader type and sample size is described. The estimated replacement sustained yield in 1992 ranged from a low of 0 fish to a high of 42,959 fish corresponding to relative errors of -100% and 44%, respectively (Fig. 3). The R0 reader type produced estimates of SY_{ST} nearly identical to the true sustained yield but biased slightly low. This slight underestimate was due to the method of back transformation of the lognormal distribution used in AGEERR; which does not use the classic back-transformation defined as $\hat{\theta} = \exp(\mu + \sigma^2/2)$, but instead conducts the back-transformation as $\hat{\theta} = \exp(\mu)$. This method of back transformation inherently leads to slightly biased estimates, but is justified because there is one fewer parameters to be estimated. Under high precision, the SY_{ST} estimates from the R1 and R2 reader types are essentially equal to the R0 reader type and therefore, only slightly negatively biased. Under low precision, the R1 and R2 estimates of SY_{ST} are more negatively biased (relative error $\approx 35\%$) than under the high precision case. The R3 and R4 reader types produced estimates of SY_{ST} that are positively biased under negative aging bias and negatively biased under positive aging bias, regardless of aging precision. However, under low aging precision and negative aging bias, the magnitude of the bias in SY_{ST} is less than under high aging precision. Finally, the level of bias (i.e., relative error) is essentially unchanged for each reader type, level of aging precision, and direction/magnitude of aging bias among the three sample sizes.

While sample size did not affect the bias in estimates of sustained yield, it did influence the variability of the sustained yield estimates. This is shown by the proportion of replicate sustained yield estimates that are within 10% of the true sustained yield and is essentially the probability of obtaining an "accurate" estimate. The average proportion of replicate estimates of SY_{ST} that were within 10% of the true value ranged between 0% and 36% among all levels of specified aging error, reader types, and sample sizes (Fig. 4). As expected, the highest proportions were obtained with reader R0, with proportions ranging from approximately 24%, 31%, and 35% among sample sizes of 100, 300, and 900, respectively. For readers R1 and R2, the proportions were essentially identical to the R0 reader type under high aging precision, but dropped substantially under low aging precision. The proportions of accurate replicate estimates from the R3 and R4 reader types were uniformly less than or equal to the R0 reader type, identically equal to zero under positive aging bias, and greater than the R1 and R2 reader types only under low aging precision and negative bias. The general trend supported by the results is that increasing sample size results in increasing the probability of obtaining an accurate estimate only if the catch sample is aged by an accurate and precise reader.

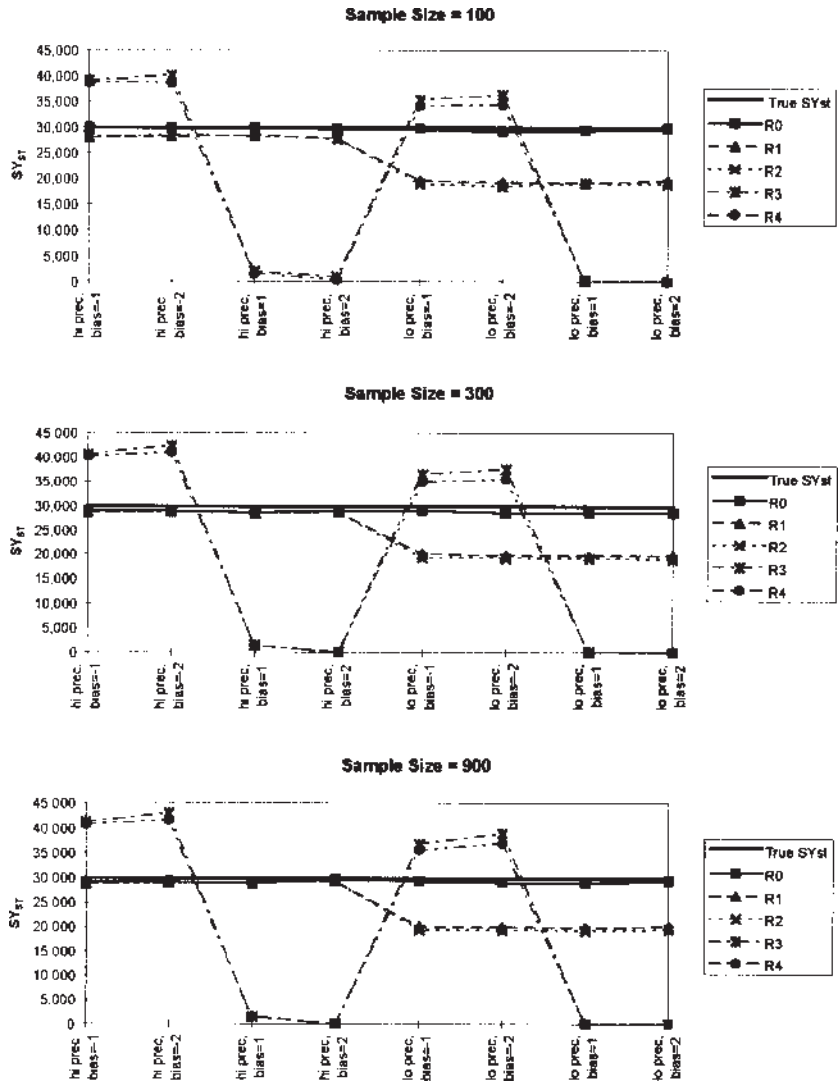


Figure 3. Estimated value of sustained yield (SY_{ST}) among reader types R0-R4 as compared to true sustained yield under different combinations of aging precision and bias for sample sizes of 100, 300, and 900 (see Modeling Aging Error for descriptions of reader types R0-R4).

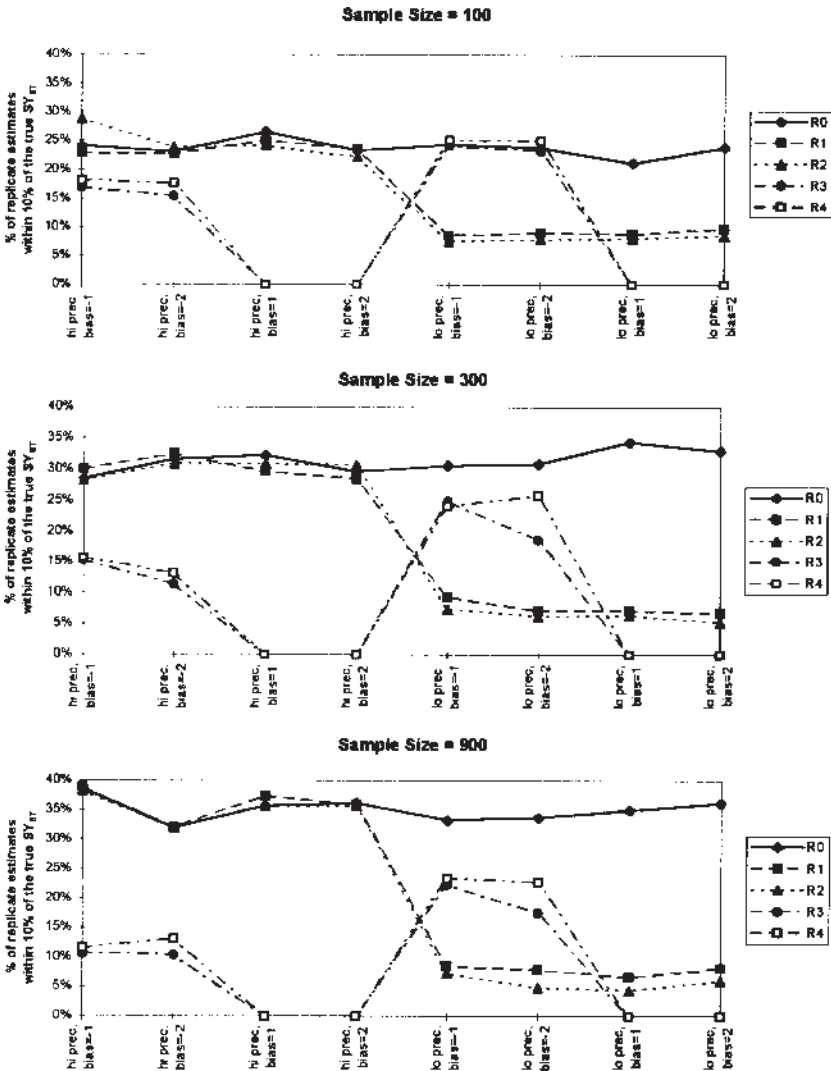


Figure 4. Proportion of Monte Carlo replicates that produced estimates of sustained yield (SY_{ST}) that were within 10% of the true value under different combinations of aging precision and bias for sample sizes of 100, 300, and 900 (see Modeling Aging Error for descriptions of reader types R0-R4).

Discussion

Effects of Sample Size on Estimates of Sustained Yield

The effect of sample size on estimates of SY_{ST} was of little consequence as demonstrated by the observation that estimates by reader type and aging error combination varied little among different sample sizes. This result suggests that attempting to mitigate bias in sustained yield estimates resulting from poor reader performance by increasing sample size is not an effective tactic.

In contrast, sample size does play a major role in determining the variability of sustained yield estimates. The probability of obtaining an estimate of sustained yield that is within 10% of the true value given a particular Monte Carlo replicate generally increases with increasing sample size. Kimura (1990) reported a similar trend in the precision of biomass estimates from ASA models as a function of sample size. This result was uniformly true among the R0 reader type and the R1 and R2 reader types under the high aging precision case. However, for the R1 and R2 reader types under the low precision case, the probability is either unchanged or decreasing as a function of increasing sample size. Similarly for the biased R3 and R4 reader types, an increasing sample size tended to have either no effect or a degrading effect on the probability of obtaining an accurate estimate.

The unexpected result that increasing sample size does not always increase the probability of obtaining an accurate estimate is probably also due to the influence other types of information have in catch-age analysis, namely estimates of relative or absolute abundance. The weighting procedures used for this other information can have a large effect on abundance estimation and, hence, obviate the influence of the aging information. Furthermore, the more imprecise the aging information is, the more deleterious an effect it may have with larger sample size, being in competition with the other abundance information.

Effects of Aging Error on Estimates of Sustained Yield

In comparison to the effects of sample size on the estimated value of SY_{ST} , the effects of aging error are more pronounced. Furthermore, aging bias tends to have greater effect on estimates of sustained yield than aging precision. In general, negative aging bias tended to produce positively biased estimates of SY_{ST} and positive aging bias tended to produce negatively biased estimates of sustained yield. This result is similar to the findings published by other researchers studying the effects of aging error on ASA and yield-per-recruit models (Rivard 1983, Lai and Gunderson 1987, Tyler et al. 1989, Kimura 1990). Also, this suggests that using bony structures that have a demonstrated tendency to negatively bias the age of fish (i.e. scales, whole otoliths; Pikitich and Demory 1988, Wilson and Boehlert 1990, Secor et al. 1995), could lead to ostensibly optimistic estimates of SY_{ST} . Furthermore, species that are extremely long lived and tend

to approach asymptotic growth in mid-life, may also be subject to substantial negative aging bias.

The effect of decreasing aging precision on the expected value of SY_{ST} was to decrease the expected value of SY_{ST} . This result was particularly evident given the unbiased R1 and R2 reader types among all scenarios. This is a heartening result for fisheries managers, since it suggests that decreasing aging precision results in more conservative management strategies. Furthermore, in the case of the biased R3 and R4 reader types, the effect of aging imprecision caused the estimated value of SY_{ST} to be less biased given negative aging bias in the low precision case versus the high precision case, because of the offsetting trends of negative aging bias and low aging precision. However, these results should not be used to imply that managing fisheries with imprecise data is desirable.

Effects of Model Structure

By definition, the structure of a model affects output and, hence, implications drawn from its use. Recall that in AGEERR the introduction of aging error was a two step process: (1) incorporating imprecision as a function of true age using a classification matrix and (2) given a biased reader type, adding a constant bias. When incorporating aging error due to imprecision, the process prohibited the resultant observed age from traversing the true age range. Furthermore, when subsequently incorporating aging bias, the resultant observed age was again restricted to the true age range.

As a result, under negative aging bias, the biased R3 and R4 reader types produced age compositions ranging from 2 to 9 (bias of -1) or 2 to 8 (bias of -2). After the incorporation of the plus group, the age ranges considered in the catch-age analysis were 2 to 7+ for both -1 and -2 biases, and the resultant catch-at-age was highly skewed towards age-2 fish. The net effect on parameter estimates was to grossly overestimate the selectivity of age-2 fish (relative error $> 750\%$), and to underestimate the projected abundance.

Under positive aging bias, the biased R3 and R4 reader types produced age compositions ranging from 3 to 10 (bias of $+1$) or 4 to 10 (bias of $+2$). After the incorporation of the plus group, the age ranges considered in the catch-age analysis were 3 to 7+ ($+1$ bias) and 4 to 7+ ($+2$ bias). Furthermore, since the simulation assumed that all ages > 2 were fully recruited, the catch-age analysis did not estimate any selectivity coefficients. The net effect was to cause even larger underestimation in the projected abundance than generated under negative aging bias.

Other strategies to incorporate aging bias could also be used, and we encourage their investigation. An alternative method for modeling aging bias is to specify bias directly in the transition matrix (Rivard 1983, Tyler et al. 1989, Bradford 1991). In addition, one could specify a bias as a function of age. A third strategy could be to specify a constant bias throughout the age range but allow observed age to traverse the age range. One could then disregard all observed ages outside the true age range. We

suspect that all these methods would produce similar trends in the relative error of sustained yield, because the same changes in the distribution of ages would occur. The results using the methods incorporated in our study may be more dramatic than what would be found using alternate methods of modeling aging error. However, the focus of our study is to point out the relative trends caused by sampling and aging error rather than to predict absolute error. Indeed, to model all the vagaries found in estimating the age of fishes is nearly impossible. Moreover, the conclusions drawn from our study do provide fishery resource managers baseline information regarding how the inherent variability (uncertainty) surrounding age-composition data can influence fishery-related parameters routinely estimated in fish stock assessments.

Our study emphasizes the importance of careful validation of aging techniques. Common problems encountered in determining the age of fish are: (1) measurement error due to imprecision caused by poor aging judgment and ability (e.g. readers failing to recognize all the extant annuli in structures collected from older fish); and (2) process error due to annuli not being formed (e.g. missing first year annuli in some rainbow trout [Lentsch and Griffith 1987]). Since imprecision and bias can act in different directions, the resultant effect on estimates of sustained yield may be unclear. Careful age validation and development of aging protocols is needed to validate estimates of sustained yield when aging error is present.

Acknowledgments

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References

- Baranov, F.I. 1918. On the question of the biological basis of fisheries. Nauchn. Issled. Ikhtologicheskii Inst. Izv. 1:81-128. (In Russian.)
- Bradford, M.J. 1991. Effects of aging errors on recruitment time series from sequential population analysis. *Can. J. Fish. Aquat. Sci.* 48:555-558.
- Bosch, D.E. 1995. Population dynamics and stock assessment of arctic grayling in the Gulkana River drainage, Alaska. Master's thesis, University of Alaska Fairbanks. 190 pp.
- Clark, W.G. 1991. Groundfish exploitation rates based on life history parameters. *Can. J. Fish. Aquat. Sci.* 48:734-750.
- Coggins Jr., L.G. 1997. The effects of sample size and aging error on estimates of sustained yield. Master's thesis, University of Alaska Fairbanks. 193 pp.

- Crone, P.R., and D.B. Sampson. 1998. Evaluation of assumed error structure in stock assessment models that use sample estimates of age composition. In: F. Funk, T.J. Quinn II, J. Heifetz, J.N. Ianelli, J.E Powers, J.F. Schweigert, P.J. Sullivan, C.-I. Zhang (eds.), *Fishery stock assessment models*. Alaska Sea Grant College Program, University of Alaska Fairbanks, AS-SG-98-01 (this volume).
- Deriso, R.B. 1987. Optimal $F_{0.1}$ criteria and their relationship to maximum sustained yield. *Can. J. Fish. Aquat. Sci.* 44(Suppl. 2):339-348.
- Deriso, R.B., T.J. Quinn II, and P.R. Neal. 1985. Catch-age analysis with auxiliary information. *Can. J. Fish. Aquat. Sci.* 42:815-824.
- Deriso, R.B., P.R. Neal, and T.J. Quinn II. 1989. Further aspects of catch-age analysis with auxiliary information. In: R.J. Beamish and G.A. McFarlane (eds.), *Effects of ocean variability on recruitment and an evaluation of parameters used in stock assessment models*. *Can. Spec. Publ. Fish. Aquat. Sci.* 108:127-135.
- Doubleday, W.G. 1976. A least squares approach to analyzing catch at age data. *Int. Comm. Northw. Atl. Fish. Res. Bull.* 12:69-81.
- Fournier, D.A., and C.P. Archibald. 1982. A general theory for analyzing catch at age data. *Can. J. Fish. Aquat. Sci.* 39:1195-1207.
- Getz, W.M., and R.G. Haight. 1989. *Population harvesting: Demographic models of fish, forest, and animal resources*. Princeton University Press, Princeton, NJ. 391 pp.
- Hightower, J.E. 1996. Comparison of reliability of catch-at-age analyses using auxiliary survey estimates of relative or absolute recruitment. *Can. J. Fish. Aquat. Sci.* 53:70-79.
- Kimura, D.K. 1990. Approaches to age structured separable sequential population analysis. *Can. J. Fish. Aquat. Sci.* 47:2364-2374.
- Kimura, D.K., and J.L. Lyons. 1991. Between-reader bias and variability in the age-determination process. *Fish. Bull. U.S.* 89:53-60.
- Lai, H.L., and D.R. Gunderson. 1987. Effects of aging errors on estimates of growth, mortality, and yield per recruit for walleye pollock (*Theragra chalcogramma*). *Fish. Res.* 5:287-302.
- Lentsch, L.D., and J.S. Griffith. 1987. Lack of first-year annuli on scales: Frequency of occurrence and predictability in trout of the western United States. In: R.C. Summerfelt and G.E. Hall, (eds.), *Age and growth of fish*. Iowa State University Press, Ames, IA, pp. 177-188.
- Megrey, B.A. 1989. Review and comparison of age-structured stock assessment models from theoretical and applied points of view. *Amer. Fish. Soc. Symp.* 6:8-48.
- Merrit, M.F., and D.F. Fleming. 1991. Evaluations of various structures for use in age determination of arctic grayling. Alaska Department of Fish and Game, Fishery Manuscript No. 91-6. Anchorage, AK. 21 pp.
- Pikitch, E.K., and R.L. Demory. 1988. Assessment of scales as a means of aging dover sole. *Trans. Am. Fish. Soc.* 117:345-349.

- Pope, J.G. 1977. Estimation of fishing mortality, its precision and implications for the management of fisheries. In: J.H. Steele (ed.), *Fisheries mathematics*. Academic Press, New York, pp. 63-76.
- Pope, J.G., and J.G. Shepherd. 1982. A simple method for the consistent interpretation of catch-at-age data. *J. Cons. Cons. Int. Explor. Mer* 40:176-184.
- Quinn II, T.J., and N.J. Szarzi. 1993. Determination of sustained yield in Alaska's recreational fisheries. In: G. Kruse, D.M. Eggers, R.J. Marasco, C. Pautzke, and T.J. Quinn II (eds.), *Proceedings of the International Symposium on Management Strategies for Exploited Fish Populations*. University of Alaska Sea Grant, AK-SG-93-02, Fairbanks, pp. 61-84.
- Richards, L.J., J.T. Schnute, A.R. Kronlund, and R.J. Beamish. 1992. Statistical models for the analysis of aging error. *Can. J. Fish. Aquat. Sci.* 49:1801-1815.
- Rivard, D. 1983. Effects of systematic, analytical, and sampling errors on catch estimates: A sensitivity analysis. In: W.G. Doubleday and D. Rivard (eds.), *Sampling commercial catches of marine fish and invertebrates*. *Can. Spec. Publ. Fish. Aquat. Sci.* 66:114-129.
- Schnute, J. 1981. A versatile growth model with statistically stable parameters. *Can. J. Fish. Aquat. Sci.* 37:241-247.
- Secor, D.H., T.M. Trice, and H.T. Hornick. 1995. Validation of otolith-based aging and a comparison of otolith and scale-based aging in mark-recaptured Chesapeake Bay striped bass, *Morone saxatilis*. *Fish. Bull. U.S.* 93:186-190.
- Sissenwine, M.P., and J.G. Shepard. 1987. An alternative perspective on recruitment overfishing and biological reference points. *Can. J. Fish. Aquat. Sci.* 44:913-918.
- Summerfelt, R.C. 1987. Preface. In: R.C. Summerfelt and G.E. Hall (eds.), *Age and growth of fish*. Iowa State University Press, Ames, IA, pp. xii-xiv.
- Tyler, A.V., R.J. Beamish, and G.A. McFarlane. 1989. Implications of age determination error to yield estimates. In: R.J. Beamish and G.A. McFarlane (eds.), *Effects of ocean variability on recruitment and an evaluation of parameters used in stock assessment models*. *Can. Spec. Publ. Fish. Aquat. Sci.* 108:27-35.
- Thompson, S.K. 1987. Sample size for estimating multinomial populations. *American Statistician* 41:42-46.
- Wilson, C.D., and G.W. Boehlert. 1990. The effects of different otolith aging techniques on estimates of growth and mortality for the splitnose rockfish, *Sebastes diploproa*, and canary rockfish, *S. pinniger*. *California Fish and Game* 76(3):146-160.

The Effects of Artisanal Fisheries on Penaeid Shrimp Stocks in the Gulf of Mexico

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Abstract

In the Gulf of Mexico, exploitation of the three main commercial penaeid species (brown shrimp *Penaeus aztecus*, white shrimp *P. setiferus* and pink shrimp *P. duorarum*) by artisanal fisheries is carried out in the nursery areas inside coastal lagoons. In addition, a new artisanal fishery on adult white shrimp has been developed. Present data on the offshore industrial fishery for white and pink shrimp shows a strong decrease to less than 20% of the mean maximum yield attained during 1970-1980 (1,700 and 7,000 metric tons whole weight, respectively). The sustained increase of artisanal fishing effort is one of the main factors that affected shrimp production. Available fishery catch-effort data of offshore fisheries showed a normal decline in catch associated with decreasing effort. However, this does not consider artisanal fishing effort on shrimp stocks causing growth overfishing on the three penaeid shrimp and recruitment overfishing on white shrimp. The lack of adequate artisanal fishing effort records does not allow assessment of its magnitude correctly. Both artisanal and industrial fisheries, besides affecting yield, may reduce shrimp reproductive potential and lead stocks to a collapse. Simulation exercises based on age-structured models are used to assess the effect of different fisheries and define management strategies. A multistage model connecting a series of stock-recruitment curves over main life-history stages could be adequate for defining broad strategies to keep penaeid shrimp stocks in good condition.

Introduction

Penaeid shrimp is a valuable resource, exploited mostly in tropical seas over a wide range of species (García 1989). Worldwide demand of this product is high and not satisfied. On the contrary, its demand has been growing steadily in the last decades. Due to the high price per unit of weight and the consumption of different sizes and ages, wild penaeid shrimp populations have been subjected to high exploitation in their juvenile and adult stages.

The worldwide shrimp industry, which started in the 1950s, developed very quickly and is presently at its maximal production limits, as wild shrimp populations seem to have reached their maximal production stage, and unexploited shrimp stocks are not likely to be found. Shrimp aquaculture has been considered a viable and important alternative to meet the demand deficit, and at the same time relieve some of the pressure exerted on the wild populations. However, sometimes shrimp aquaculture has exerted an additional pressure on life cycle stages not traditionally exploited due to the postlarval demand for pond stocking. Postlarval extraction plus sequential fishing of juvenile and adults has caused some species, such as the Pacific white shrimp *Penaeus vannamei*, to be exploited during almost their whole life cycle. The future of shrimp populations subjected to such intense fishing pressure seems uncertain (Gracia and Vázquez Bader, in press).

The objectives of our study are to examine the interactions of artisanal and industrial fisheries for white (*P. setiferus*) and pink shrimp (*P. duorarum*) in the Gulf of Mexico. We also examine the potential implications of exploitation of shrimp postlarvae for use in aquaculture. In our analysis we consider the diverse stages of the shrimp life cycle and examine the implications on the recruitment process and reproductive potential.

Catch History

The shrimp fishery in Mexican waters of the Gulf of Mexico (Fig. 1) started in the 1950s and, just as most penaeid fisheries, developed quickly, reaching its peak production in the 1970s. During this period, catches of over 30,000 t (total weight) per year, with a maximum of 39,500 t in 1972, were obtained with an associated fishery effort of 1,100 trawler boats (Fig. 2). Later on, in the 1980s, total annual production of shrimp declined to almost 20,000 t (Gracia and Vázquez Bader, in press). Coincidentally, during this period the fishing effort of trawler boats also declined approximately 40%; however, an increase of inshore artisanal fishing occurred. In the last decade, shrimp production was below the maximum recorded, and a negative tendency has been observed for some species (e.g., white *P. setiferus*, Fig. 3; and pink *P. duorarum* shrimps, Fig. 4).

The white shrimp, *P. setiferus*, is mostly fished in the southwestern Gulf of Mexico, in the Campeche Sound and the adjacent coastal region, mainly the Términos Lagoon. This fishery reached its maximum produc-



Figure 1. Fishing areas of shrimp in the Mexican Gulf of Mexico.

tion levels during the 1970s when catches ranged from 1,200 to 2,200 t (total weight). After this period, white shrimp catches declined to levels below 20% of the average maximal production (Fig. 3). According to Gracia (1989a), the maximum sustainable yield of this species was estimated in 1,630 t with an optimal fishery effort of 33,406 ship days.

Traditionally pink shrimp, *P. duorarum*, was the most important species for shrimp fishery at the Mexican coast of the Gulf of Mexico. Catches obtained at the Campeche Sound during the 1970s represented between 70% to 90% of the total shrimp catch in the south of the Gulf of Mexico. After this period, catch and fishing effort decreased, leveling off from 1980 to 1986. Afterward, catches presented an acute negative tendency

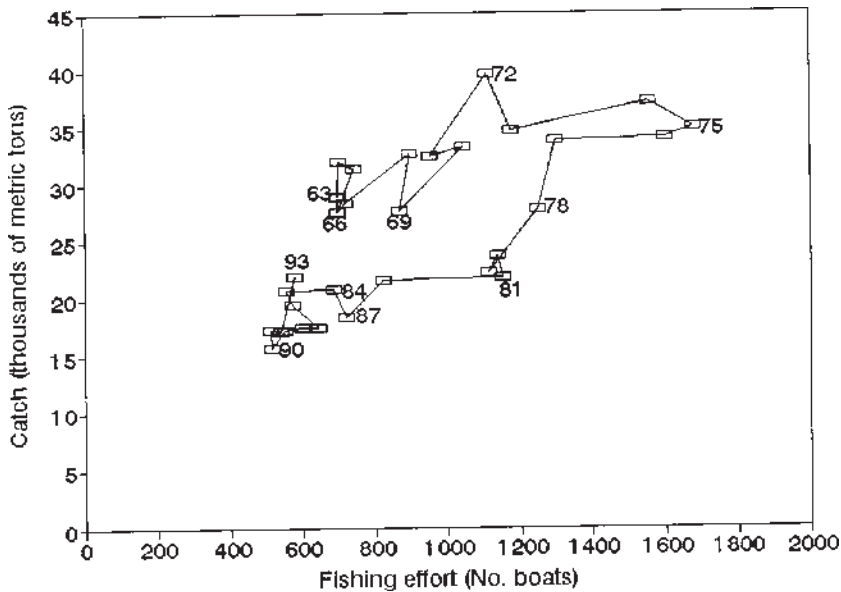


Figure 2. Diagram of catch and effort data of the shrimp fishery in the Gulf of Mexico (1963-1993).

that correlated with low abundance of the species. Pink shrimp yield recorded in one of the most important fishing ports (Carmen City) declined to almost 20% of the average maximal production reached in the 1970s (Fig. 4). According to estimations made by several authors (Arreguín-Sánchez and Chávez 1985, Navarrete and Uribe 1993), maximum sustainable yield varies from 4,300 to 11,000 t.

Artisanal fisheries target the three main shrimp species, *P. setiferus*, *P. duorarum* and *P. aztecus*, and include fishing in lagoons, estuaries, and coastal areas. These artisanal fisheries affect mainly juvenile stages, although recently a new artisanal fishery with drift nets was developed, which focuses mainly on the exploitation of the adult white shrimp in marine waters.

Material and Methods

Statistical data of the shrimp fisheries in the Gulf of Mexico were considered for the study. Analysis of shrimp industrial fisheries was based on catch and effort statistics for white and pink shrimp in the main fishing areas during 1973-1991 and 1973-1990, respectively. This information was available at government agencies (Fisheries Department of Mexico). Total catch per each of nine categories (10-15, 16-20...61-70 tails per pound)

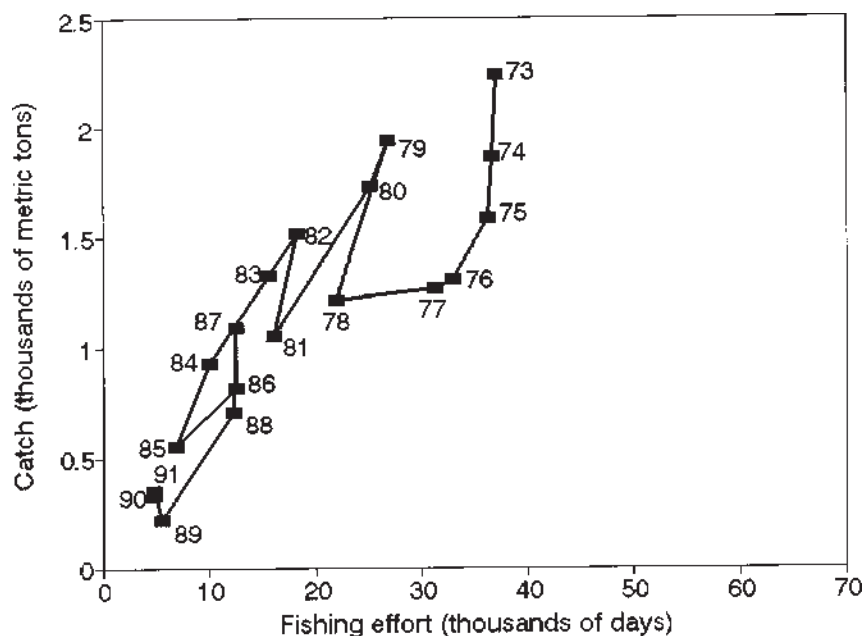


Figure 3. White shrimp fishery trajectory based on catch-effort data of Campeche Bank from 1973 to 1991 (redrawn from Gracia 1996).

by month were analyzed to estimate indices of shrimp abundances in open sea waters. Numbers of individuals by size and sex were estimated by the procedure of Gracia (1989a, 1991), based on sex ratio and the mean tail weight for each market category. Numbers were calculated by using a computer spreadsheet and assuming a normal distribution. Information of artisanal fishery total catch was also compiled (1985-1990), when available, from government statistics. Size composition and monthly shrimp abundance indices of the artisanal fishery were obtained from periodic samplings carried out in these fisheries (Gracia 1989a, 1991, 1995, 1996).

Population and fishery parameters were obtained from the literature or were estimated from the available statistics (Table 1). Size was converted to age according to the von Bertalanffy growth curve for each species. Age-specific fishing mortality was obtained through age-based virtual population analysis for white shrimp and pseudocohort analysis for pink shrimp of the composition of monthly shrimp catches (Sims 1985, Mesnil 1988). A mean life span of 16 months for the shrimp was considered based on size population structure and shrimp growth curves.

Analysis of the interaction among the diverse fisheries was based on a yield per recruit (Y/R) age-structured Thompson and Bell type model.

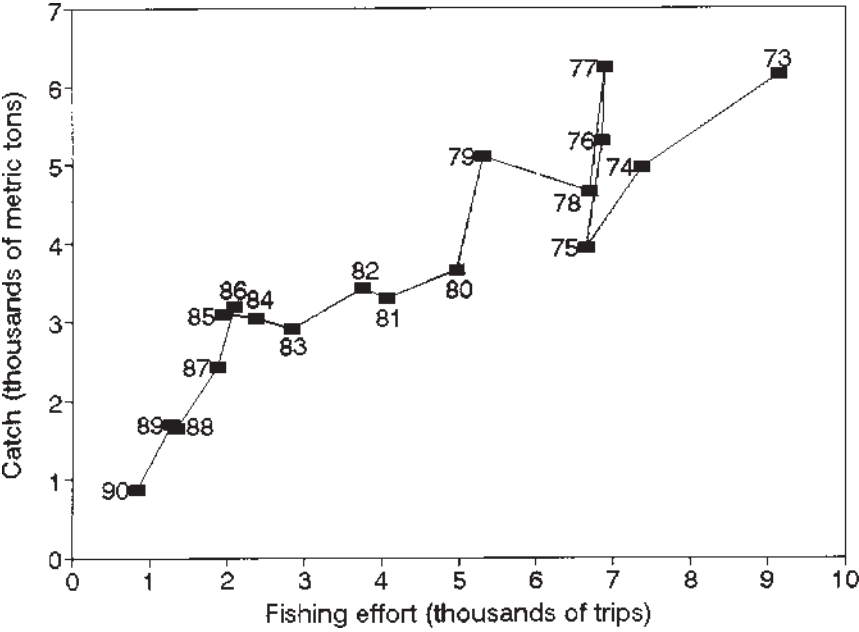


Figure 4. Pink shrimp fishery trajectory based on catch-effort data of Campeche Bank (1973-1990).

This model provides a simple description of non-steady state situations and has been used for sequential shrimp fisheries earlier by Willmann and García (1985). Basic equations used in the model are as follows:

$$Z_i = M + F_i$$

$$N(t_i + dt) = N(t_i) \exp(-Z_i dt)$$

$$C_i = [N(t_i) - N(t_i + dt)] F_i / Z_i$$

$$Y_i = C_i w_i$$

$$B_i = Y_i / [F_i dt]$$

Where Z_i is instantaneous coefficient of total mortality at age i , M is instantaneous natural mortality rate (monthly), F_i is instantaneous fishing mortality rate at age i , N_i is shrimp population size at the end of the age i , C_i is catch in numbers at age i , w_i is average shrimp weight, Y_i is yield, B_i is biomass, i is the age in months, t is time and dt is the time spent in the interval.

Table 1. White and pink shrimp parameters.

	White shrimp ^a	Pink shrimp ^b
Growth parameters		
L_{α}	209 mm	203 mm
W_{α}	72 g	68 g
K	0.2016	0.2167
t_0	0.3267	0.466
Estuarine natural mortality	0.6	0.7
Offshore natural mortality	0.25	0.27
Marine fishing mortality		
Age	F	F
4	0.012	0.018
5	0.063	0.053
6	0.090	0.089
7	0.178	0.125
8	0.245	0.225
9	0.492	0.338
10	0.355	0.388
11	0.409	0.393
12	0.412	0.445
13	0.386	0.464
14	0.454	0.306
15	0.364	0.366
16	0.330	0.315

Source: ^aGracia 1989a, ^bthis study.

For simplicity, it was assumed that recruitment, growth, and fishing pattern was constant during the life span of shrimps and were known without error. An average single-cohort, age-structured model was built for each species and was assumed that, at equilibrium, this yield was equivalent to that one resulting from all the cohorts present in the fishery during the year. To analyze the effect of inshore artisanal fisheries on the offshore industrial fishery different levels of artisanal mortality were used while maintaining offshore fishing rates constant. Simulation results were compared and expressed as percent change of the shrimp total yield.

Results and Discussion

White Shrimp

The proportion of white shrimp in inshore artisanal fisheries varies with the geographic location of the lagoons. According to fishery statistics, the white shrimp represents from 5% to 15% of the total artisanal shrimp catch in the coastal lagoons of the northeastern Gulf of Mexico. Juveniles are highly concentrated in Términos Lagoon in Campeche Sound (Gracia 1989a). This fact led to the development of fishery activities in this region during the 1970s. At present, this activity is prohibited but is performed clandestinely. Due to the illegal character of this activity, there are no formal statistical records of the effort and catch rates of white shrimp by the artisanal fishery; nonetheless, the available information allows analysis of the main aspects of its relationship to industrial fishery (Gracia 1989a).

Size composition of artisanal catch obtained using otter trawls ranges from 3.5 to 14.0 cm total length (TL), corresponding to 2- to 4-month-old organisms. However, the highest proportion is represented by organisms aged under 3 months with an average monthly size of 7.0 to 10.0 cm TL (Fig. 5) corresponding to a market category of over 70 tails per pound.

Artisanal catches vary with seasonal fluctuations in shrimp abundance in the nursery areas. According to Gracia (1989a), in Términos Lagoon they have high and low abundance periods. In general, the high abundance period goes from the end of spring to the fall (rainy season) and, on average, the maximal abundance of juveniles occurs between June and July. The lowest abundance usually occurs from December to March. The artisanal fishery adjusts to these variations; therefore the invested effort depicts proportional fluctuations to the availability of white shrimp in the nursery areas. These seasonal fluctuations in the abundance of juveniles and migrating sizes was taken into account for the simulation of interactions among white shrimp sequential fisheries based on the yield per recruitment model.

Analysis of the interaction among sequential fisheries in the Términos Lagoon–Campeche Sound area (illegal inshore–industrial offshore) based on population parameters, estimated for all stages of *P. setiferus* indicates that the decline in total production due to the illegal artisanal

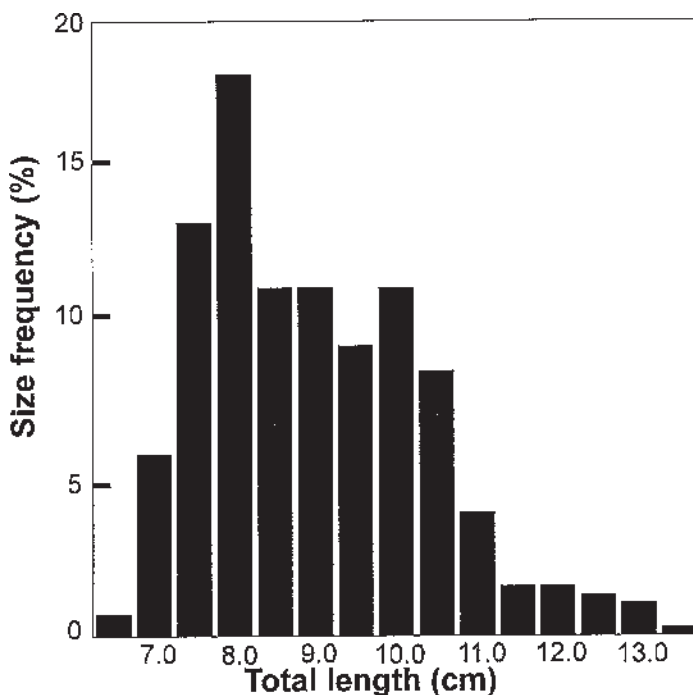


Figure 5. White shrimp size composition obtained by the inshore artisanal fishery using otter trawls.

fleet—estimated in 1984 to range from 30 to 300 boats (Smith 1985)—could be close to 20%. Based on this data, the estimated artisanal catch could only contribute between 3% to 6% of the total catch, resulting in a net reduction of shrimp production offshore. Although the lack of available statistics of artisanal fishing effort does not allow precise estimates, it is appreciated that the effect on total shrimp catch is directly related.

If the fishing effort in Términos Lagoon is increased to levels causing a monthly fishing mortality of 0.44-0.5, equivalent to using fixed fishing gears (Gracia and Soto 1986), the increase in artisanal catch would correspond to 14% of the total biomass, and the industrial fishery would decline 40% (Fig. 6). The effect is even more noticeable if it is analyzed in terms of the gross value of the catch, since the artisanal fishery catches smaller sizes of less market value. In this case gross value of the catch will reduce about 40% and the artisanal catch value will only represent about 2% of the total.

Interaction between growth rates and natural mortality, and even fishing mortality, results in a larger biomass of *P. setiferus* after migration,

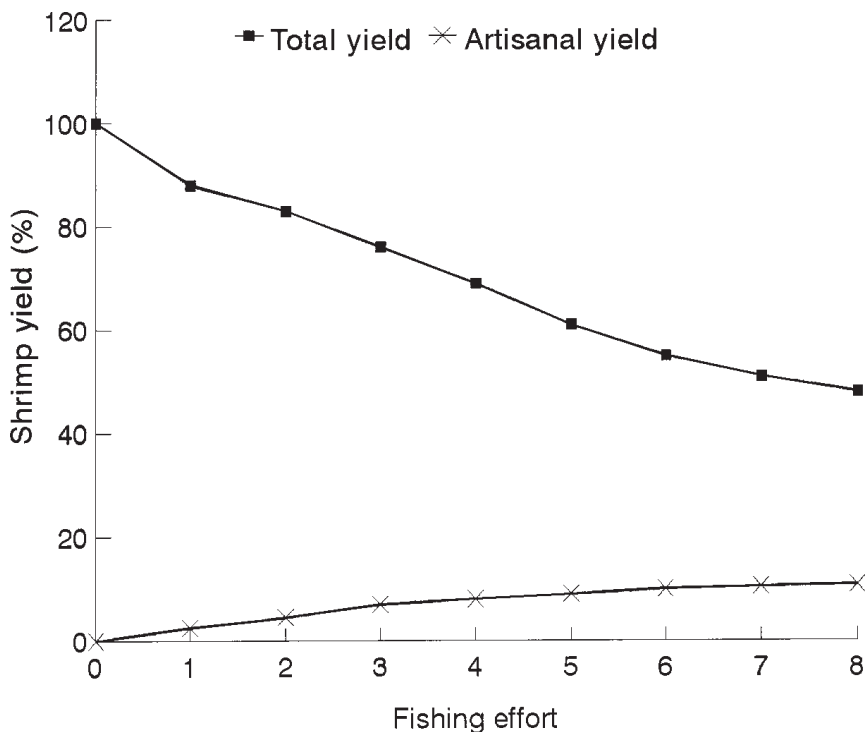


Figure 6. Impact of inshore artisanal fishery on the total yield of white shrimp.

once they escaped artisanal exploitation in the coastal lagoon environment. Each kilogram of white shrimp caught in the estuary decreases marine catch in a 1:2.8 proportion. The cumulative residual loss due to artisanal exploitation leads to a global reduction in the catch that correlates with increasing fishing effort.

Therefore, the magnitude of the impact caused by inshore artisanal fisheries on the catch levels of the following fisheries is directly proportional to the fishing effort exerted on the estuarine populations and has a negative exponential relationship with marine catch in terms of biomass (Fig. 11). If the number of organisms is considered this relationship is linear and directly proportional to the fishing effort.

Pink Shrimp

The greatest artisanal exploitation of pink shrimp is being taken in the protected coastal waters of the states of Campeche and Yucatán, which are the most important nursery areas of the pink shrimp in Mexican waters of the Gulf of Mexico.

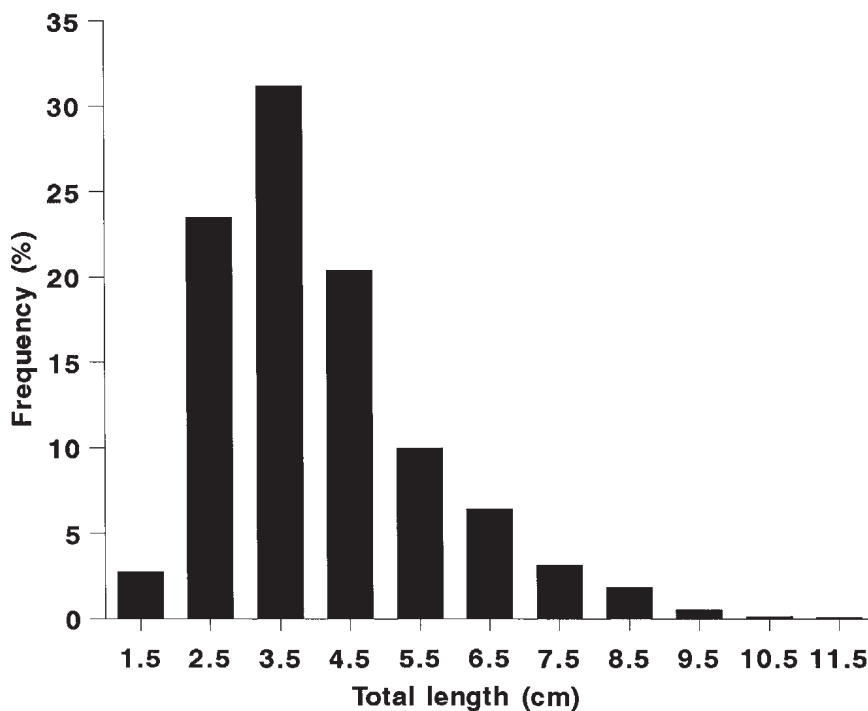


Figure 7. Pink shrimp size composition exploited by inshore artisanal fishery.

Artisanal fishery of pink shrimp in the coastal areas of Campeche is accomplished through the use of different types of fishing gears (Alarcón-Fuentes and Arreguín-Sánchez 1992). Two of them are fixed gears placed in the channels to catch shrimps during their migration to the sea. The other two are mobile and are operated manually in shallow waters near the coastline. Catches are basically made up of pink shrimp (more than 96%), although some palemonid shrimps have also been recorded in small proportion (4%).

The size range caught by this artisanal exploitation corresponds to very small shrimps, and although sizes go from 1.2 to 11.5 cm TL, the mode of the catch corresponds to 3.0-4.0 cm TL (Fig. 7). Most (85%) of the organisms caught through this artisanal fishery are only 25 to 60 days old (Fig. 7).

According to information gathered in the Campeche area, artisanal fishing of juvenile pink shrimp has been going on for more than 20 years, but exploitation used to be on a small scale just for household subsistence, i.e., catches were small and not recorded. At the beginning of the 1980s, exploitation increased to commercial levels and the first data of

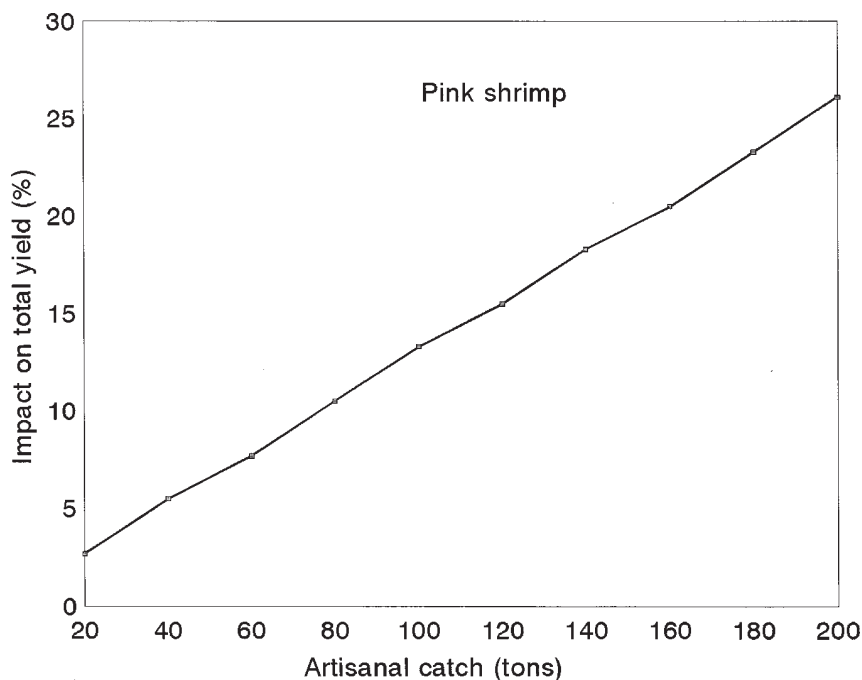


Figure 8. Impact of artisanal fishery on the pink shrimp total yield.

juvenile pink shrimp catches along the coast of Campeche was recorded. It is worthwhile mentioning that these records are variable and do not necessarily reflect the level of artisanal exploitation. The recorded catches of this juvenile stage revealed a steady increase from the first years of data recording until this activity was prohibited in 1992.

The average artisanal exploitation, according to the available statistical data, reveals a constant activity year-round with monthly variations that could coincide with abundance fluctuations of juveniles. The available records show that the maximum average catch was obtained in the months of March, May, September, and October, which agrees with periods of high abundance of juveniles in the area (Gracia and Soto 1990). However, it should be noted that no knowledge exists on the fluctuations in artisanal fishing effort that might introduce bias in the relationship of abundance-catch levels; for example, in the months of January and February, when shrimp abundance is low and juvenile catches do not follow this pattern (Gracia 1995).

The effect of artisanal fishery of juvenile pink shrimp on the industrial fishery is also directly related to the extracted volume (Fig. 8). The

relationship is similar to that of the white shrimp, except that for the pink shrimp the impact is even greater on the total production. The loss of total biomass is greater since shrimp caught by the artisanal fishery are smaller. The exploitation of juvenile *P. duorarum* of such small sizes results in a waste of global biomass of 9 kg for each kilogram of juvenile organisms extracted from the nursery areas. In terms of commercial value, this proportion increases to an average of 28:1, since the value per unit weight increases with shrimp size.

The effect on the adult population, estimated through simulation models, indicates that a fishery activity similar to the one recorded could cause a 10%-20% decline in offshore production. The effect could be even greater if we consider that records of an artisanal fishery are usually incomplete. The annual production of artisanal pink shrimp catches in the nursery area of Yucatán state from 1985 to 1990 show fluctuations from 25 to 230 tons. Inclusion of precise catches in the model could account for most of the decline in pink shrimp production.

Other factors have been pointed out as causing the decline in pink shrimp production (Gracia 1995). Among them are (a) decrease in the fishing effort due to deterioration of the fishing fleet, (b) loss of fishing areas due to the expansion of the oil industry, (c) deterioration of the nursery areas, and (d) expansion of the artisanal fishery during the earlier stages of the pink shrimp. However, according to the results, the artisanal fishery seems to be the most important factor responsible for the decline in production. None of the other factors seem to be critical for the decline, since fishing effort has recovered in some cases without improving production, and the relationship between deterioration of nursery areas and decline in catches has not been proven.

Exploitation of Shrimp Postlarvae

At present there is no extraction of postlarval organisms to be used as "seeds" for shrimp culture in the Gulf of Mexico, since there is no aquaculture development with native species in the area. However, due to the problems posed by some diseases and the potential risk of exotic viruses transmitted by foreign species (NMFS 1997), culture of native species seems to be a good option for increasing shrimp production. Considering the potential involved in aquaculture, a postlarval fishery could be started with native species of the Gulf of Mexico. Nonetheless, this analysis could also be a reference point for other species, such as *P. vannamei* from the Pacific, where postlarvae are already being exploited.

Literature regarding the effect of postlarval extraction on shrimp population is scarce. Gracia (1989b) assessed such an activity for the white shrimp *P. setiferus*, a potential aquaculture species that occupies the third place in pond productivity after *P. vannamei* and *P. stylirostris* (Lawrence 1984). The impact varies according to the age of juveniles caught. The impact is lower when the postlarval organisms are caught before entering

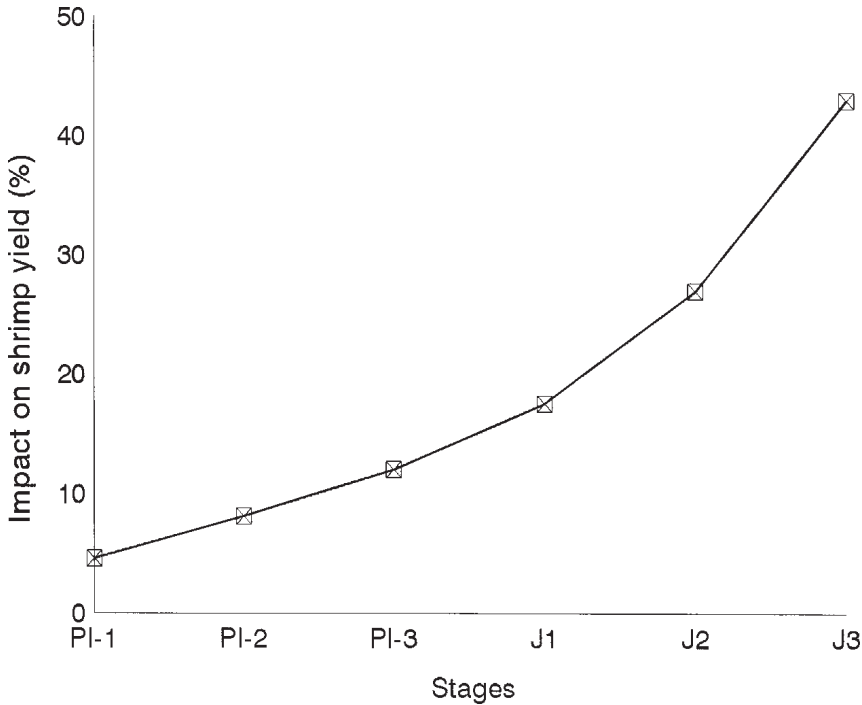


Figure 9. Impact of postlarvae exploitation of different ages on total white shrimp yield.

the coastal lagoons and increases exponentially with increasing age of the shrimp (Fig. 9). Indeed, catching of already settled postlarval-juvenile organisms in nursery areas is a variant of juvenile artisanal fishing as described for the pink and white shrimps with similar effects.

The impact on shrimp production increases proportionally with the exploitation level on postlarvae (Fig. 10). However, there is a marked difference if the catch is performed before the settlement of postlarval organisms in the estuarine benthos, or if the catch includes already established stages. This difference is due to the fact that postlarval settlement in nursery areas represents a critical point in the life history of the shrimp and in the regulation of the population size. The carrying capacity of the nursery areas, determined by diverse environmental factors (type of bottom, nutrients, salinity, depth, vegetation, etc.), is to a great extent responsible for the recruitment level in marine populations (García 1989). This explains why the curve slope elevation of postlarval exploitation-production impact is greater when organisms that have already surpassed this critical stage are exploited.

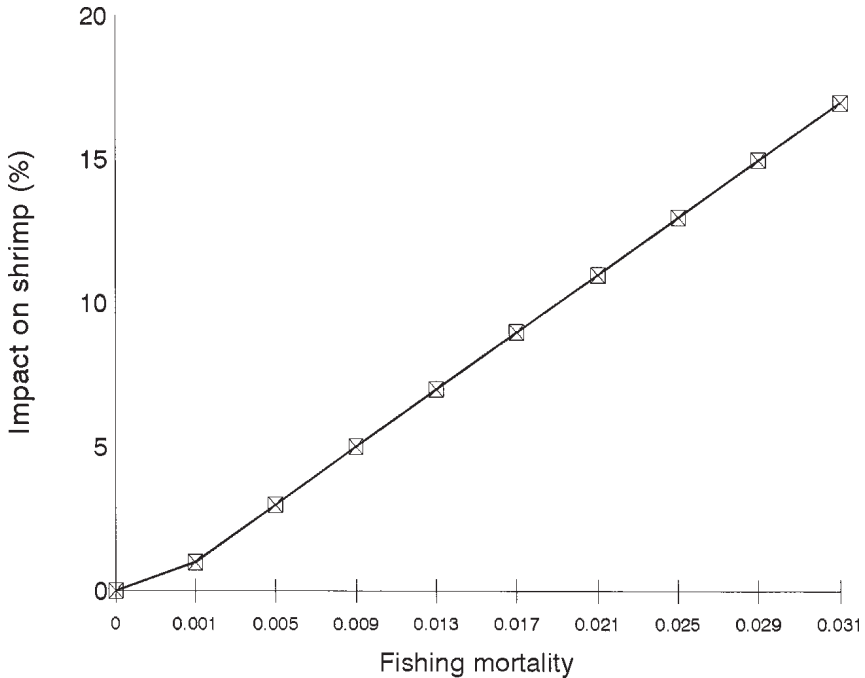


Figure 10. Relationship between postlarvae fishing mortality and shrimp yield decline.

Relationship among the Different Life Cycle Stages

White shrimp and pink shrimp have a similar life history. Both of them have an estuarine-dependent stage. Differences lie in the time they spend in the nursery areas. Pink shrimp tend to migrate out of the lagoons sooner than white shrimp. However, similarities allow common management strategies for both.

According to the performed analyses and the recorded data in the literature (Gracia 1989a, 1989b, 1991, 1997) it can be assumed that the relationships among the different stages of one shrimp generation subjected to a potential fishery vary between potential and linear, as shown in Fig. 11. In the presented model, it is assumed that there is a linear relationship between the number of spawners and the number of produced larvae. However, when relating spawners to settled postlarvae two weeks later (8-12 mm total length), a potential relationship is expected. Density dependent mortality mainly occurs during the estuarine stage, therefore the relationship between larvae and juveniles would be potential. The relationship between the number of resulting juveniles and the number of

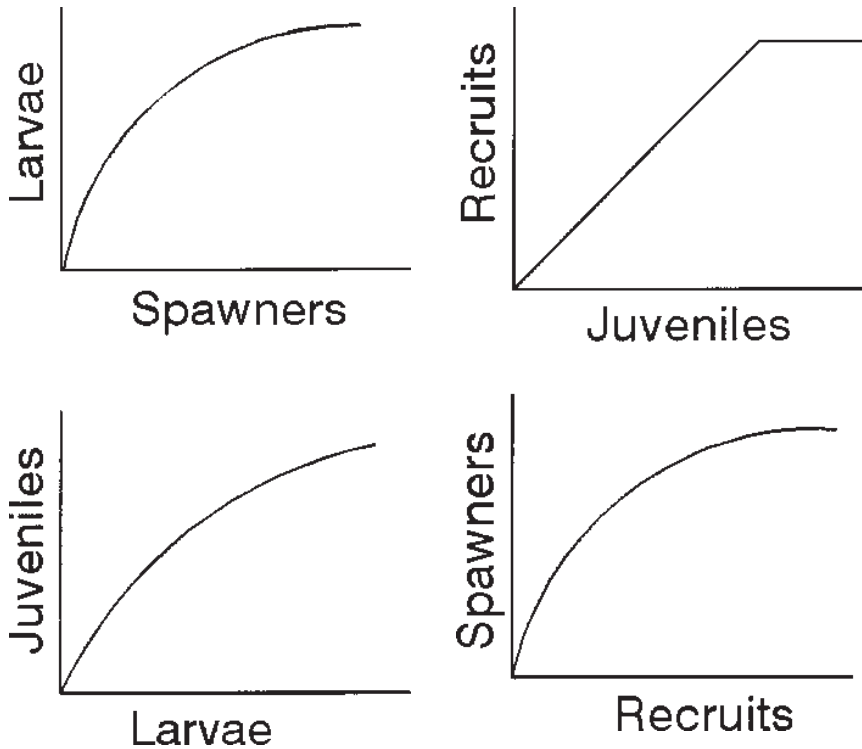


Figure 11. Relationship among the different shrimp life cycle stages.

recruits to the marine population is linear, although a limit would be expected for this relationship at very high abundance levels of juveniles in the nursery areas. The duration of this stage varies between an average of 2-3 months (6.5-8.5 cm TL) for pink shrimp and 3-4 months (8.5-9.5 cm TL) for white shrimp. In the marine environment the relationship between the number of recruits (3-4 months old) and spawners would have a potential form due to density-dependent factors. A stock-recruitment relationship of this type has been established for white shrimp by Gracia (1991).

These relationships can be affected by environmental changes, and could be represented by diverse levels on the curves limiting an area where different abundance values for each stage can be encountered. These values are influenced by the fishing mortality derived from the fishery over the different stages of the life cycle, and its impact is reflected on the subsequent stages, and eventually on the total production of shrimp.

General Discussion

Along the history of shrimp fisheries, small-scale artisanal fisheries were conducted before the industrial fishery developed, since exploiting this resource in coastal regions was easy. Profitability of the industrial fishery caused a shift of fishing efforts toward offshore fishing. During the 1980s, artisanal fishery efforts on several species of the Gulf of Mexico strongly increased, which in some instances even inverted the relationship of artisanal versus industrial fishing effort.

This increase in artisanal fishery effort directed to early shrimp stages of those species of greater market value caused overfishing at this growth stage, since exploitation occurs before the optimal biomass per organism is achieved. On the other hand, increases in artisanal fishery with drift nets has caused a recruitment overfishing of white shrimp (Gracia 1992a,b, 1996). Both factors led to a decline in total available biomass of the three species, affecting the optimal profitability of the resource and the reproductive potential of the species.

The critical problem in defining precisely the impact of artisanal fisheries on the exploitation of the main shrimp species and to differentiate this impact from other factors (such as environmental effects or deterioration of the nursery areas) is the lack of adequate records on artisanal fishing effort. Despite their importance, artisanal fishing effort is not given their due weight in assessing shrimp populations in the Gulf of Mexico. Information available at this moment suggests that most of the deterioration in shrimp populations is due to the uncontrolled increase in artisanal fisheries plus the already existing industrial fishery effort, which has already reached maximum allowable levels.

Fishing mortality in the juvenile and pre-adult stages must be added to the fishing mortality caused by industrial fishing, on which evaluations of shrimp stocks have traditionally been based. As a result, we now have total fishing efforts that had never been reached before in the history of shrimp exploitation. Disregard of the artisanal effort also hinders evaluation of the total accumulated fishing efforts in each of the exploitation stages, and impairs taking adequate measures to insure a healthy management of this resource.

The main characteristic of the artisanal fishery is in providing income to a relatively large number of fishermen with low-cost investments. This makes this activity quite attractive and fostered an uncontrolled increase in fishing effort. To a certain point, Mexican artisanal fisheries in the Gulf of Mexico are almost freely accessed without any control adversely affecting the resource. The established objective for shrimp fishing in the area are (1) that there must be a healthy balance between artisanal and industrial fisheries to allow for optimal exploitation of the resource without negatively affecting the reproductive population, and (2) to generate a global benefit. In this sense, it is necessary to limit the artisanal effort. This is particularly important for the present conditions of the resource,

where (1) there is no possibility to increase catches of natural populations above the already reached levels, (2) global fishing effort have increased markedly, and (3) world aquaculture is in an expansion stage, which will increase supply of small shrimps competing with the shrimp obtained by artisanal fisheries.

An important point to be considered is the impact caused by all fisheries on the reproductive potential. According to the analyses made, the recruitment level can be affected, which in turn will affect the reproductive biomass. Garcia and Van Zalinge (1982) and Garcia (1989) emphasize that fecundity per recruit at a given exploitation level can be increased by protecting the juvenile population. Accumulation of fishing efforts on different stages of the shrimp's life cycle can exert an opposite effect and eventually lead to recruitment overfishing.

In most worldwide fisheries, recruitment overfishing has been questioned, since stock-recruitment (S-R) relationships are not apparent for most penaeid shrimps. However, these S-R relationships have been demonstrated in some penaeid species (Penn and Caputi 1986, Gracia 1991) and recruitment overfishing has been documented in some species (e.g., in *P. setiferus* resulting from the additional artisanal effort on reproductive populations, Gracia 1996).

Traditional management to prevent recruitment overfishing consists of maintaining a healthy relationship between the spawner stock and the recruitment produced by this stock, considering the different stages of the shrimp's life cycle as a black box. Such is the case for the white and pink shrimp in the Gulf of Mexico, although some protection is aimed to the juvenile stage. According to Hilborn and Walters (1992) the reasons to group the different life stages are (1) the spawning stock is the one that can be managed and not the other stages of the life cycle, and (2) it is important to have direct measurements of the consequences of this control. Although management through such a general relationship can be very useful, in the case of penaeid shrimps the multiple fishery conditions can lead to recurrent stock reductions, shortening of the fishing season, and/or depletion of the fishery. An increasing effort in artisanal fishery would tend to decrease the spawning stock independently of the protection measures taken at this stage. Reducing fishing effort only on spawners would affect mainly the offshore industrial fishery, which is at the end of the exploitation chain, and the results would not be very tangible. Eventually, the fishery could reach levels of smaller production due to growth overfishing and, if the fishing effort is excessive, become unprofitable at the marine stage.

To avoid damage to the fishery, it is necessary to implement adequate management strategies based on escape levels in each stage. Gracia (1996), based on a study in white shrimp, suggests some practical reference points for the management of the fishery in relation to the critical levels of the reproductive stock. In this case management strategies can be directed especially to the main annual cohorts. Based on the analysis of

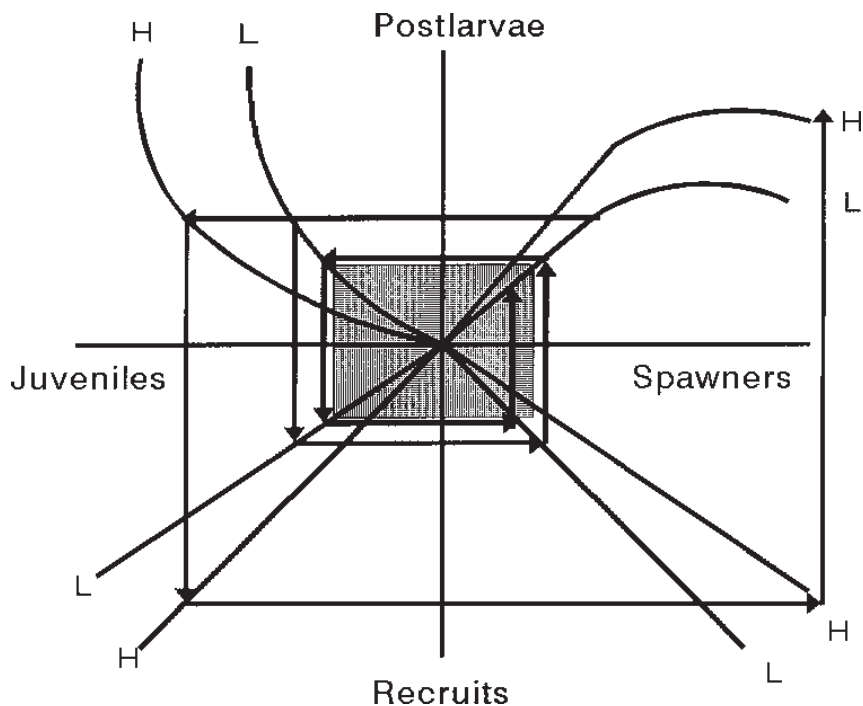


Figure 12. Base diagram to analyze the trajectory of one shrimp generation subjected to exploitation in its different life cycle stages. Shadow area represents the risk area for population renewal, H and L are "good" and "bad" environmental conditions. Arrows represent hypothetical population trajectories along varying conditions.

the S-R relation of the main *P. setiferus* cohorts, Gracia (1996) suggests that to preserve the population lying on the flat section of the stock-recruitment curve, the stock must be maintained at 25%-30% of the highest reproductive biomass value recorded in the fishery. This means that the spawning stock must be kept at levels higher than 17%-20% of virgin biomass.

These reference points can be linked with information on the different stages (Fig. 12) of the shrimp's life cycle through a multiple stage model as proposed by Paulik (1973). To design management strategies based on this type of models, it is necessary to have information on recruitment both in the growth areas and in the marine environment, and on effort levels in the artisanal and industrial fisheries, as well as to have critical reference points on the stock-recruitment relationship. Basically, escape of organisms in the different stages can be insured through catch-

ing quotas, reducing fishing effort by setting spatial and temporal closures, or monitoring catch per effort unit in the diverse fisheries. This should aim to leave a sufficient number of migrating organisms to allow for an adequate level of spawners and, thus, avoid deterioration or collapse of recruitment. Such a management strategy could be applied not only to penaeid shrimp of the Gulf of Mexico, but to the those which are subjected to similar exploitation levels. Establishing an escapement-based management strategy may, in first instance, increase total shrimp yield and gross fishery value by reducing growth overfishing, which would mainly benefit offshore fishery. An example of this can be found in the brown shrimp seasonal closure of the Gulf of Mexico directed to reduce growth overfishing which enhanced cohort yield by more than 300% (Gracia 1997). Benefits to the artisanal fisheries could be expected in the long term by increasing shrimp spawning potential and increasing average shrimp size in the nursery areas. However, the most important point will be that this strategy could allow the coexistence of the different fisheries while assuring high profitability in their exploitation.

Acknowledgments

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References

- Alarcón-Fuentes, T., and F. Arreguín-Sánchez. 1992. La pesca de camarón rosado en Sabancuy y Champotón, Campeche. JAINA Boletín Informativo 3(4):20.
- Arreguín-Sánchez, F., and E.A. Chávez. 1985. Estado del Conocimiento de las pesquerías de camarón en el Golfo de México. Investig. Mar. CICIMAR 2(2):23-44.
- García, S. 1989. The management of coastal penaeid shrimp fisheries. In: J.F. Caddy (ed.), Marine invertebrate fisheries: Their assessment and management. Wiley and Sons, New York, pp. 281-306.
- García, S., and N.P. Van Zalinge. 1982. Shrimp fishing in Kuwait: Methodology for a joint analysis of artisanal and industrial fisheries. In: Assessment of the shrimp stocks off the west coast of the gulf between Iran and the Arabian Peninsula. FAO/UNDP Proj. FI/DP/RAB/80/015, pp. 119-142.
- Gracia, A. 1989a. Ecología y pesquería del camarón blanco *Penaeus setiferus* (Linnaeus, 1767) en la Laguna de Términos-Sonda de Campeche. Tesis doctoral, Universidad Nacional Autónoma de México. 127 pp.
- Gracia, A. 1989b. Impacto de la explotación de postlarvas sobre la pesquería del camarón blanco *Penaeus setiferus* (Linnaeus, 1767). An. Inst. Cienc. del Mar y Limnol., Universidad Nacional Autónoma de México 16(2):255-262.
- Gracia, A. 1991. Spawning stock-recruitment relationships of white shrimp in the southwestern Gulf of México. Trans. Am. Fish. Soc. 120:519-527.

- Gracia, A. 1992a. Explotación y manejo del recurso camarón. *Ciencia y Desarrollo* 18(106):82-95.
- Gracia, A. 1992b. La sobrepesca del reclutamiento en las pesquerías de camarón. *JAINA Boletín Informativo* 3(1):11.
- Gracia, A. 1995. Impacto de la pesca artesanal sobre la producción de camarón rosado *Penaeus farfantepenaeus duorarum* Burkenroad 1939. *Cienc. Mar.* 21(3):343-359.
- Gracia, A. 1996. White shrimp (*Penaeus setiferus*) recruitment overfishing. *Mar. Freshw. Res.* 47:59-65.
- Gracia, A. 1997. Simulated and actual effects of brown shrimp *Penaeus aztecus* closure in Mexico. *Marine Fisheries Review* 59(2):18-24.
- Gracia, A., and L.A. Soto. 1986. Estimación del tamaño de la población, crecimiento y mortalidad de los juveniles de *Penaeus setiferus* (Linnaeus, 1767) mediante marcado-recaptura en la Laguna de Chacahito, Campeche, México. *An. Inst. Cienc. del Mar y Limnol., Universidad Nacional Autónoma de México* 13(3):217-230.
- Gracia, A., and L.A. Soto. 1990. Population study of the penaeid shrimp of Términos Lagoon, Campeche, México. *An. Inst. Cienc. del Mar y Limnol., Universidad Nacional Autónoma de México* 17(2):241-255.
- Gracia, A., and A.R. Vázquez Bader. In press. Shrimp fisheries in the south of the Gulf of Mexico: Present state and future management alternatives. In: *The Gulf of Mexico large marine ecosystem: Assessment, sustainability and management*. Blackwell Scientific Publications, Oxford.
- Hilborn, R., and C.J. Walters. 1992. Quantitative fisheries stock assessment: Choice, dynamics and uncertainty. Chapman and Hall, New York. 570 pp.
- Lawrence, A. 1984. Marine shrimp culture in the Western Hemisphere. In: P.C. Rothlisberg, B.J. Hill, and D.J. Staples (eds.), *Second Australian National Prawn Seminar*, Cleveland, Australia, pp. 327-336.
- Mesnil, B. 1988. Logiciels pour l'évaluation des stocks de poissons. ANACO: Logiciel d'analyse des données de capture par classe d'âge sur IBM PC et compatibles. *FAO Doc. Tech. Pêches* 101(Suppl. 3):1-73.
- NMFS. 1997. An evaluation of potential shrimp virus impacts on cultured shrimp and wild shrimp populations in the Gulf of Mexico and Southeastern U.S. Atlantic Coastal waters. A report to the Joint Subcommittee on Aquaculture prepared by the JSA shrimp virus work group. NMFS-APHIS-NCEA-USFWS. 65 pp.
- Navarrete, A., and J. Uribe. 1993. Evaluación de la pesquería industrial del camarón de altamar del Puerto de Campeche, México en el periodo de 1981 a 1990. Centro Regional de Investigaciones Pesqueras de Campeche, Camp., Instituto Nacional de la Pesca, pp. 1-15.
- Paulik, G.J. 1973. Studies of the possible form of the stock and recruitment curve. *Rapp. P.-V. Reun. Cons. Int. Explor. Mer* 164:302-315.
- Penn, J.W., and N. Caputi. 1986. Spawning stock-recruitment relationships and environmental influences on the tiger prawn (*Penaeus esculentus*) fishery in Exmouth Gulf, Western Australia. *Aust. J. Mar. Freshw. Res.* 37:49-51.

- Sims, E. 1985. Selected computer programs in FORTRAN for fish stock assessment. FAO Fish. Tech. Pap. 259:1-189.
- Smith, K. 1985. Investigación de pesquerías clandestinas y de las zonas y épocas de reproducción del camarón blanco en la zona de Cd. del Carmen. Centro Regional de Investigaciones Pesqueras de Cd. del Carmen, Instituto Nacional de la Pesca, SEPES, pp. 1-17.
- Willmann, R., and S.M. García. 1985. A bioeconomic model for the analysis of sequential artisanal and industrial fisheries for tropical shrimp (with a case study on Suriname shrimp fisheries). FAO Fish. Tech. Pap. 270:1-49.

Fisheries Management Based on Reproductive Potential: An Example for Chub Mackerel

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Abstract

One of the main purposes of stock assessment is to evaluate the spawning stock size. Spawning stock biomass (SSB) has been generally used as an indicator of spawning stock size. However, SSB is a short-term indicator of spawning stock size, which represents the magnitude of the spawning in the coming spawning season. Because SSB cannot account for the long-term reproductive ability of the stock, fisheries management based on SSB may result in overfishing in the future. Katsukawa (1997) suggested the use of total reproductive value of the standing stock as an indicator of spawning stock size. We call this the reproductive potential. The reproductive potential can be used to evaluate the long-term reproductive ability of the standing stock, considering the biological parameters and the age composition. In this study, we have examined the performance of a fisheries management strategy based on reproductive potential. We compared the performance of constant reproductive potential strategy and that of constant SSB strategy. By deterministic analysis, we showed that fisheries management based on reproductive potential is more robust to the errors in the stock size than that based on SSB. A stochastic computer simulation also supports this result.

Introduction

One of the main purposes of fisheries management is to avoid recruitment overfishing. Spawning stock biomass (SSB) has been generally used as an indicator of the spawning stock size. Many reference points in fisheries management are based on SSB. For example, a level of 20% virgin or unexploited biomass has historically been adopted as a threshold for recruitment overfishing (Beddington and Cooke 1983). Myers et al. (1994)

and Zheng et al. (1993) examined a variety of methods for estimating the spawning stock biomass level as a threshold for recruitment overfishing.

Although a large number of studies has been done on the level of SSB needed to avoid recruitment overfishing, SSB presents some problems as an indicator of spawning stock size, especially for long-lived species. SSB is an indicator of short-term reproductive ability, representing only the magnitude of spawning in the coming spawning season. For long-lived species, the standing stock must sustain not only the next spawning but also the spawning of subsequent years. Ensuring sufficient SSB may be useful for avoiding recruitment overfishing in the coming spawning season. However, as SSB cannot account for spawning beyond the upcoming spawning season, a fisheries management strategy based on SSB may result in overfishing in the future. For example, immature fish have some potential value due to their future reproduction, which is not accounted for when using SSB.

In order to ensure the potential spawning ability of the stock, we must take into account the value of future spawning. Katsukawa (1997) developed an indicator of long-term spawning ability of the standing stock, called reproductive potential (*RP*). The objective of this paper is to examine the effectiveness of reproductive potential as a reference point of fisheries management. We examined the difference between reproductive potential and SSB by using an example of chub mackerel. We also compared the performance of fisheries management strategies based on *RP* (constant *RP*) and SSB (constant SSB).

Methods

Fisher's Reproductive Value

We could use the concept of Fisher's reproductive value (*RV*) for effectively evaluating the long-term productivity of the individual (Fisher 1930). *RV* can be expressed as follows:

$$RV_t = \sum_{i=t+1}^{\infty} e^{-r(i-t)} E_i A_i$$

$$= \sum_{i=t+1}^{\infty} \left[e^{-r(i-t)} E_i \exp \left\{ - \sum_{j=t}^{i-1} (F_j + M_j) \right\} \right] \quad (1)$$

where RV_t is reproductive value of the individual at age t (just after the spawning season of age- t), E_i is fecundity of the individual at age i , A_i is survival rate till the spawning season at age i , F_j is the coefficient of fishing mortality at age j , and M_j is the coefficient of natural mortality at age j , and r is the instantaneous growth rate of the population size.

The instantaneous growth rate r is important when the stock size is exponentially increasing or decreasing. For example, when we think about

Table 1. Estimation of body weight (g), rate of maturity, selectivity, reproductive value (RV) (g), RV per weight by age, and stock number-at-age in 1970 (10^5 individuals) of chub mackerel as summarized from Wada et al. (1996).

Age	Body weight (w_i)	Rate of maturity (m_i)	Selectivity (q_i)	RV		RV per weight		Number-at-age in 1970
				$f = 0$	$f = 1.73$	$f = 0$	$f = 1.73$	
1	252	0.0	0.25	1,120	147	4.29	0.583	52,766
2	434	0.2	0.44	1,584	251	3.55	0.578	31,833
3	610	1.0	0.62	1,753	191	2.80	0.313	14,411
4	672	1.0	0.79	1,944	160	2.83	0.238	5,370
5	811	1.0	1.00	2,089	126	2.53	0.155	3,328
6	912	1.0	1.00	2,204	146	2.37	0.160	2,560
7+	1,084	1.0	1.00	2,204	146	2.00	0.135	838

a stock rebuilding program, spawning of this year is more valuable than that of the next year. In this paper, we will discuss the management strategy to fix stock size at one level. Therefore, we assumed $r = 1$. In this case, RV of an age- t individual is the expected spawning of that individual from age t until the end of the individual's life.

Comparison between Reproductive Value and Body Weight

In order to examine if we can use body weight as an approximation of RV, we compared the reproductive value (RV) and the body weight of chub mackerel (*Scomber japonicus*). The population parameters for chub mackerel were described by Wada et al. (1996) and they are summarized in Table 1. We assumed the natural mortality (M) = 0.4. The fishing mortality at age i (F_i) is expressed as the product of the selectivity at age i (q_i) and fishing effort (f) [$F_i = q_i f$]. The fecundity at age i (E_i) is expressed as the product of the rate of the maturity at age i (m_i) and the weight at age i (w_i) [$E_i = m_i w_i$].

When $f = 0$, RV and the body weight increase with age. As the increase in RV is much slower than that of the body weight, younger individuals have higher RV per body weight than older ones. This implies that young fish have higher future productivity than old fish with the same weight. If there is no fishing mortality, 1 kg of one-year-old chub mackerel will, with respect to long-term spawning, have an equal contribution as a 2.2 kg seven-year-old chub mackerel. Therefore, the reproductive ability of young fish is underestimated if we use biomass as an indicator of spawning stock size.

In Table 1, we show the *RV* corresponding to the average fishing effort between 1980-1989 ($f = 1.73$). An increase in fishing effort leads to a decrease in *RV*. The decreasing rate of *RV* is different for different ages. Because the selectivity of old individuals is higher than that of young ones, old individuals are generally more vulnerable to fishing pressure than young ones. One-year-old individuals are the only exception. Only 20% of two-year-olds are mature, and one-year-olds therefore take yet another year to fully mature. If the fishing pressure is high, the one-year-olds hardly survive next two years. Although one-year-olds have higher growth rate and lower fishing mortality than two-year-olds, they are more sensitive to fishing pressure. One-year-old individuals are as sensitive as three-year-old individuals.

Because the future production of the individual is affected by the age and fishing pressure, body weight is not a sufficient indicator to evaluate the future production of the individual. If we have enough biological information, the future production of an individual should be estimated by *RV*.

Reproductive Potential

Katsukawa (1997) suggested to use total *RV* of the standing stock as an indicator of spawning stock size. We have called this the reproductive potential (*RP*). *RP* can be expressed as follows:

$$\begin{aligned}
 RP &= \sum_{i=1}^{\infty} N_i RV_i \\
 &= \sum_{i=1}^{\infty} \left\langle N_i \sum_{j=i+1}^{\infty} \left[e^{-r(j-i)} E_i \exp \left\{ - \sum_{k=i}^{j-1} (F_k + M_k) \right\} \right] \right\rangle \\
 &= \sum_{i=1}^{\infty} \left\langle N_i \sum_{j=i+1}^{\infty} \left[e^{-r(j-i)} E_j \exp \left\{ - \sum_{k=i}^{j-1} (q_k f + M_k) \right\} \right] \right\rangle
 \end{aligned} \tag{2}$$

where N_i is stock number of age i , q_k is selectivity at age k , and f is fishing effort. The reproductive potential can be used to evaluate the future production of the standing stock, considering the biological parameters and the age-composition of the stock.

We can use reproductive potential not only as an indicator for stock assessment but also as a reference point for fisheries management. If the number-at-age (N_i), fecundity (E_j), selectivity (q_k), and natural mortality (M_k) are estimated, reproductive potential (*RP*) can be expressed as a monotonic decreasing function of fishing effort (f). Therefore, we can change the value of *RP* by adjusting f . The maximum Reproductive potential is obtained when $f = 0$. As f increases, *RP* decreases toward zero. Consequently, we can obtain any *RP* value between $RP_{f=0}$ (the maximum *RP*) and zero by controlling f . Therefore, targeting a certain *RP* level will show us

the f value needed to obtain that RP level. If the target level is higher than $RP_{f=0}$, no yield can be allowed because stock size is too low to achieve the target level. Otherwise, we can calculate the unique f needed to obtain the target reproductive potential level from equation (2). When the RP threshold is set, the f needed to obtain that threshold can be considered as the maximum acceptable f .

Let us think about the constant reproductive potential strategy (CRP). We used chub mackerel population as an example. We fixed the age-distribution as that in 1970. The stock size (α) is expressed relative to the stock size in 1970. Namely number-at-age i ($N_{i,t}$) equals $\alpha N_{i,1970}$. We employed relative stock size of $\alpha = 1$ (stock size in 1970) as the minimum stock size needed to reach the target level. In this case, fishing effort (f) is determined from the RP of the standing stock equal to the target level ($RP_{f=0,1970} = 1.55 \times 10^7$ t).

$$\sum_{i=1}^{7+} (N_{i,t} RV_i) = \sum_{i=1}^{7+} \left(\alpha N_{i,1970} \sum_{j=i+1}^{\infty} \left[W_j m_j \exp \left\{ - \sum_{k=i}^{j-1} (q_k f + M) \right\} \right] \right) \quad (3)$$

$$= 1.55 \times 10^7 \text{ t}$$

where $N_{i,t}$ is the number of age- i individuals in year t , α is the relative stock size $N_{i,t}/N_{i,1970}$, $N_{i,1970}$ is the number of age- i individuals in 1970. Figure 1 shows the relationship between f and relative stock size (α).

Figure 1 also contains f determined by the constant escapement strategy based on SSB (CSB). The target level of SSB is determined as the stock size in 1970 ($\alpha = 1$). The f of the CSB is determined as the SSB equals to the target level.

$$\sum_{i=1}^{7+} \left\{ \alpha N_{i,1970} W_{i+1} m_{i+1} \exp(-q_i f - M) \right\} = 3.00 \times 10^6 \text{ t} \quad (4)$$

No fishing is allowed when $\alpha \leq 1$, because stock size is too low to achieve the target stock size. On the other hand, when $\alpha > 1$, we can obtain an unique f that reaches the target level. The higher f is allowed as the stock size increases. When the relative stock sizes (α) is the same between CSB and CRP, the f of CSB is always higher than that of CRP. This is due to the difference in the time scale of the indicators. For the calculation of RP , we use the f value in the subsequent years. RP is calculated on the assumption that the stock will be exploited under the same fishing pressure throughout the remainder of its lifetime. By this assumption, we can determine the fishing pressure (f) that considers the future spawning of the standing stock. On the other hand, CSB allows exploitation of all surplus stock by the next spawning season. Let us think about a stock with low natural mortality. If the stock size is double of the target, CSB allows exploiting almost half of standing stock within one year. This exploitation

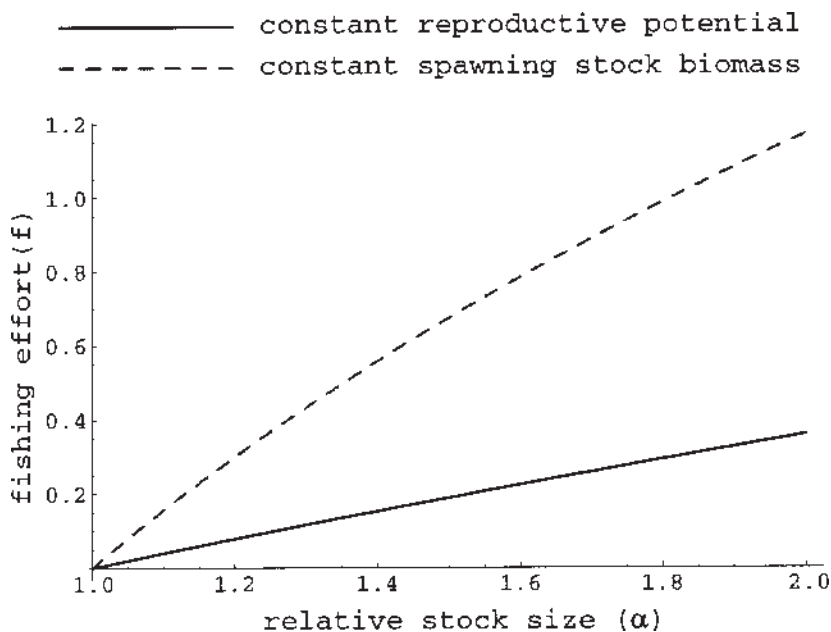


Figure 1. The fishing effort (f) determined by constant reproductive potential strategy and constant SSB strategy, for chub mackerel. We fixed age composition as that in 1970. Stock size (α) is expressed as the ratio to the stock size in 1970. We employed relative stock size of $\alpha = 1$ (stock size in 1970) as the minimum stock size needed to reach the target level.

rate will decrease the stock size one forth in two years and one eighth in three years. Therefore, the future spawning of the standing stock is decreased to less than half. The f determined by CSB is not allowable from the viewpoint of long term productivity.

Simulation

By using a computer simulation, we compared the performances of the constant reproductive potential strategy (CRP) and the constant SSB strategy (CSB). We used chub mackerel for our simulation. This population is known to have considerable recruitment variation. Therefore, age composition is unstable. We can consider the age composition in light of reproductive potential. Since this stock is important for the Japanese fishing industry, the information needed for the calculation of RP is available.

We used a Beverton-Holt stock-recruitment relationship. The parameters of the Beverton-Holt curve were estimated from the stock-recruitment

relationship during the period 1970-1991, by the maximum likelihood estimation to give

$$R = \frac{1.16 \times 10^{14} S}{5.32 \times 10^{15} + S} e^{N(0,1.01)} \quad (5)$$

where R is the number recruits (10^5 individuals), S is SSB (tons), and $N(0, 1.01)$ is the normal distribution, with mean 0 and standard deviation 1.01.

We used the number-at-age in 1970 as the initial stock size of the simulation. We used the same target levels as the deterministic analysis above. If the estimated stock size is lower than that in 1970, no fishing is allowed. We used a normal distribution with mean 1 and standard deviation 0.5 for the estimation error of the number-at-age. We assumed independent estimation error for each age class. Namely, not only the total stock size, but also the age composition were estimated with error. When an estimated number-at-age become negative, we used zero for the estimated number-at-age. We used the same target level of reproductive potential ($RP_{\text{target}} = 1.55 \times 10^7$ t) as the deterministic analysis above. The fishing effort (f) under CRP is determined as the RP of the standing stock equals to the target level of RP :

$$\sum_{i=1}^{7+} \left(\varepsilon_i N_i \sum_{j=1}^{\infty} \left[W_j m_j \exp \left\{ - \sum_{k=i}^j (q_k f + M) \right\} \right] \right) = 1.55 \times 10^7 \text{ t} \quad (6)$$

where ε_j is the estimation error of the number-at-age j , W_j is weight at age j , and m_j is proportion mature at age j . Fecundity at age j (E_j) is expressed as the product of body weight at age j (W_j) and maturity at age j (m_j). If the estimated stock size is too low to reach RP_{target} , we cannot satisfy equation (5). Therefore, in that case, f is set at zero.

The fishing pressure of CSB is determined as the SSB of the standing stock equals to the target level:

$$\sum_{i=1}^{7+} \{ \varepsilon_i N_i W_i m_i \exp(-q_i f - M) \} = 3.00 \times 10^6 \text{ t} \quad (7)$$

In this simulation, we updated f annually by the equations (6) and (7), using newly estimated number-at-age. The simulated time is 50 years. We repeated this simulation 100 times.

Results

The performance criteria of constant reproductive potential (CRP) and constant SSB (CSB) are presented in Table 2. Figure 2 shows fishing effort, SSB, and yield of one realization of the hundred replications.

Table 2. Performance criteria of the constant reproductive potential (CRP) and constant SSB (CSB).

Performance criteria	CRP	CSB
Target level (10^6 t)	15.5	3.00
Mean yield (10^6 t)	4.36	2.17
Variance in yield (10^{13} t ²)	4.08	1.18
Mean fishing effort (f)	0.76	0.82
Variance in fishing effort (f)	0.42	0.93
Mean RP level (10^7 t)	5.56	2.70
Mean SSB level(10^6 t)	6.20	2.99
Minimum RP (10^7 t)	1.69	0.96
Minimum SSB (10^6 t)	2.93	1.71
Overfishing years	0.69	29.96
Moratorium years	2.31	13.14

All criteria are expressed by the average of the 100 replications.

The average of fishing effort (f) of CSB was slightly higher than that of CRP, but the variance of f of CSB was almost four times bigger than that of CRP. Figure 2a shows one realization of fishing effort over time. As we used the same time series of estimation error and recruitment variation, the trends of fishing effort of both strategies were similar. But the peaks of CSB were sharper and higher than that of CRP. The f -curve for CSB is steeper than that for CRP (Fig. 1). This implies that the value of f determined by CSB suffer higher fluctuation by the error of stock size estimation than that determined by CRP. Numbers of the moratorium years were significantly different. The moratorium years of CSB was more than 25% of the simulated time. Such frequent moratorium is not desirable for the fishing industry.

Figure 2b shows the time series of SSB. SSB level of CRP was higher and more variable than that of CSB. As we used a highly variable stock-recruitment curve, there were some years with high recruitment. CSB exploit all of the surplus production within one season. Therefore, SSB level of CSB fluctuated near the target level. On the other hand, as CRP exploited the abundant cohort gradually, SSB level of CRP increased after recruitment success. This is the reason why SSB level of CRP was higher and more variable than that of CSB. The average minimum RP was higher than the target level. If the stock is productive enough, the target level of RP works as the lower limit. CRP may be effective for conservation of the stock.

More than half of the simulated time, CSB decreased SSB level below the target.

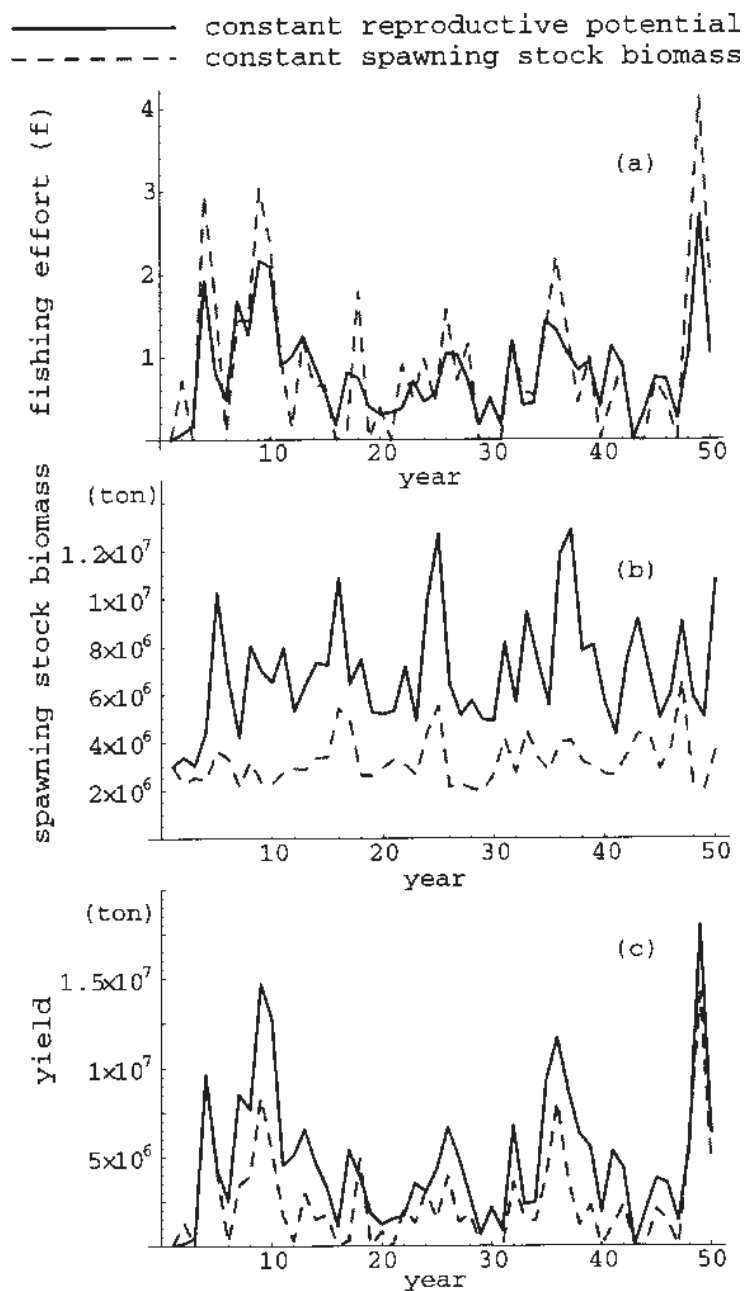


Figure 2. The time series of fishing effort, SSB and yield for constant reproductive potential and constant SSB. This represents one of the 100 replications.

As we used a stock with high productivity, the newly recruited cohort often compensated the overfishing. This is the reason why there was a moratorium on fishing only 13 years while overfishing occurred more than 30 years. The minimum SSB level of CSB is about half of the target level. For a stock with low productivity, such overfishing may result in long successive moratorium.

We examined the robustness of two strategies by a deterministic analysis. We assumed the same estimation error (ε) for all age classes. In this case, f is determined by the biased relative stock size ($\varepsilon\alpha$) instead of true relative stock size (α). Figure 3a shows the stock size after one year exploitation at f determined by CRP. Here we expressed the stock size as the maximum future spawning of survived stock, which is attained by the moratorium starting from the next year. The darker area indicates the smaller stock size. The areas surrounded by the dotted line indicates overfishing. In this area, the standing stock cannot achieve the target lifetime spawning (target RP), even if the standing stock will not be exploited from next year. Otherwise, we can continue exploitation and achieve RP_{target} by tuning f . Under CRP, overestimating the stock size results in a fishing plan that leads to overfishing. But, we exploit only a part of surplus yield within one fishing season. Even when the stock size has been overestimated, we may still reach the target RP by decreasing f the next year.

Figure 3b shows the SSB after one year of exploitation at f determined by CSB. Because CSB exploits all surplus stock within a year, overestimating the stock number inevitably results in overfishing. The darkest areas indicates severe overfishing. In these areas, the escapement size is less than 75% of the target level. The darkest area of CSB is wider than that of CRP. Consequently, by applying CRP, we can reduce not only the probability of overfishing but also the intensity of overfishing.

By setting SSB at the most productive level, we can achieve the maximum sustainable yield (Reed 1979). As the recruitment of the next year relates to SSB, the fisheries management based on SSB has an advantage in terms of the maximizing recruitment. In this simulation, yield of CRP is about the double of CSB. This is mainly due to the difference in the biomass level. As the variance of SSB level is smaller for CSB, CSB may maximize yield if the target level is properly set. On the other hand, as RP represents the long-term productivity of the stock, the fisheries management based on RP is effective for the conservation of the stock.

The fisheries management based on RP and that based on SSB have different strengths depending on the nature of the indicators. Therefore, it is essential to clarify the objective of the fisheries management and to choose the appropriate indicator in view of that objective. SSB should be used for the maximizing yield, while RP should be used for the conservation of the stock. We can use the advantage of both SSB and RP by setting the target level of SSB and the threshold level of SP at the same time. If the stock level is low, we can protect the stock by using the RP indicator.

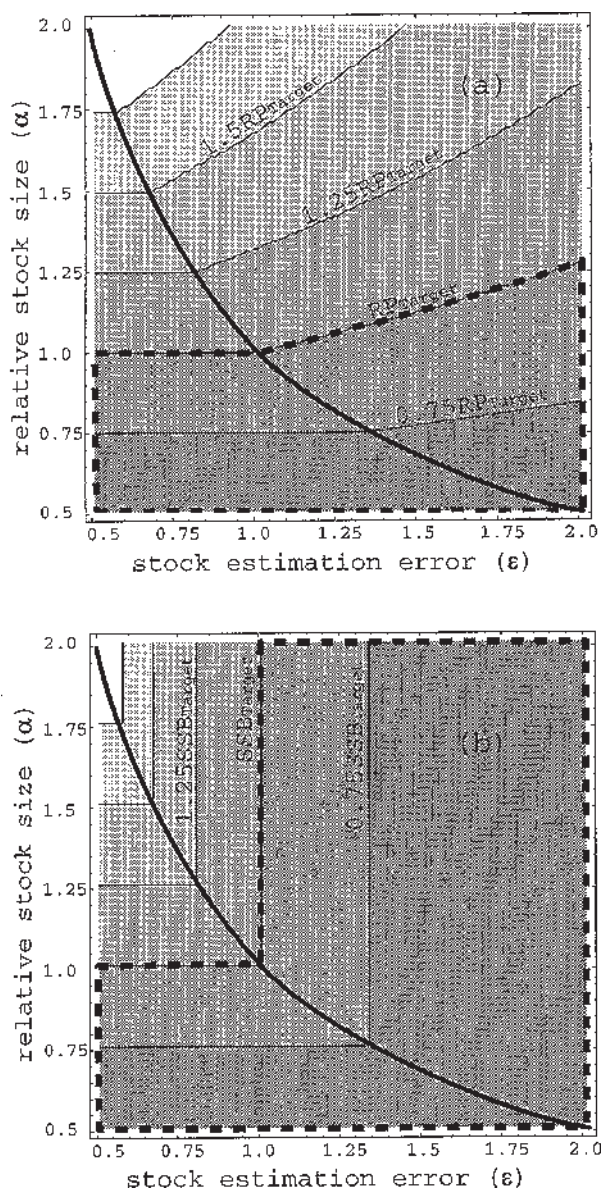


Figure 3. The distribution of escapement stock numbers after one year's exploitation under the constant escapement strategy based on the reproductive potential (a) and that based on SSB (b). The areas below the curved line in both diagrams represents the areas of moratorium on fishing. The area surrounded by the dotted line indicates overfishing.

Conversely, if the stock level is high enough, we can maximize the number recruits by using the SSB indicator.

Discussion

Feedback Management

In Rosenberg et al. (1994), 67% of the overfishing definitions are fishing mortality rate-based. The remaining 33% of the definitions are based on SSB. Although fishing mortality based management is widely used, sustainable fishing mortality rate is usually uncertain. Feedback from stock size is effective to avoid the risk from this uncertainty. Even if sustainable fishing mortality rate is overestimated, fishing mortality rate is automatically decreased by the feedback from stock decline. Feedback process works even when we don't have enough knowledge about the mechanism of the population dynamics.

Although theoretically appealing, constant escapement strategy is not widely used, because of highly vulnerability to the stock estimation error. Instead of CES, SSB threshold policy is commonly used (Sigler and Fujioka 1993, Ianelli and Heifetz 1995). Under threshold policy, harvesting occurs at a constant rate but reduced or ceased when stock size drops below a threshold. The objective of threshold policy is mainly conservation of the stock. Then *RP* is an appropriate indicator for threshold policy, because *RP* has advantages to evaluate long-term spawning ability. Furthermore, as we have seen, we can decrease risk of overfishing due to stock size estimation error by using a *RP* indicator.

Comparison between Reproductive Potential and Spawning per Recruit

The calculation method of reproductive value (*RV*) is similar to that of spawning per recruit, which is often used in fisheries management (Gabriel et al. 1989, Clark 1993, Goodyear 1993, Mace and Sissenwine 1993). Spawning per recruit is equivalent to the *RV* of an individual that has just been recruited. Although the basic equation of *SPR* and reproductive potential is related, the management concept of both are completely different. The objective of *SPR* analysis is to obtain the exploitation rate that will lead the stock to the sustainable equilibrium. *SPR* analysis is a tool for the constant harvesting rate strategy, under which the fishing rate is determined independent of the stock size. On the other hand, the objective of the management based on *RP* is directly escaping enough spawning stock. We can use *RP* as the target level or threshold level of the stock size. In this case, fishing pressure is determined by the estimated stock size.

Alee Effect

Alee (1931) pointed out that a population may fail to recover if the stock decreases below a certain threshold level. Some stocks have been suggest-

ed to exhibit such behavior, in the sense of experiencing a severe decline and subsequently failing to recover despite a reduction in the fishing mortality rate (Thompson 1993). In this study, CRP always led to a higher minimum stock level than CSB. Thus, fisheries management based on reproductive potential (*RP*) may effectively prevent the stock from falling below an undesirable point of no return.

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References

- Allee, W.C. 1931. *Animal aggregations*. University of Chicago Press, Chicago. 431 pp.
- Beddington, J.R., and J.G. Cooke. 1983. The potential yield of fish stocks. *FAO Fish. Tech. Pap.* 242. 50 pp.
- Clark, W.G. 1993. The effect of recruitment variability on the choice of a target level of spawning biomass per recruit. In: G. Kruse, D.M. Eggers, R.J. Marasco, C. Pautzke, and T.J. Quinn II (eds.), *Proceedings of the International Symposium on Management Strategies for Exploited Fish Populations*. University of Alaska Sea Grant, AK-SG-93-02, Fairbanks, pp. 233-246.
- Fisher, R.A. 1930. *The genetic theory of natural selection*. Oxford, U.K. 291 pp.
- Gabriel, W.L., M.P. Sissenwine, and W.J. Overholtz. 1989. Analysis of spawning stock biomass per recruit: An example for Georges Bank haddock. *N. Am. J. Fish. Manage.* 9:383-391.
- Goodyear, C.P. 1993. Spawning stock biomass per recruit in fisheries management: Foundation and current use. In: S.J. Smith, J.J. Hunt, and D. Rivard (eds.), *Risk evaluation and biological reference points for fisheries management*. *Can. Spec. Publ. Fish. Aquat. Sci.* 120:67-81.
- Ianelli, J.N., and J. Heifetz 1995. Decision analysis of alternative harvest policies for the Gulf of Alaska Pacific Ocean Perch Fishery. *Fish. Res.* 24:35-63.
- Katsukawa, T. 1997. Introduction of spawning potential: Improvement in the threshold management theory. *Rev. Fish Biol. Fish.* 7:285-289.
- Mace, P.M., and M.P. Sissenwine. 1993. How much spawning per recruit is enough? In: S.J. Smith, J.J. Hunt, and D. Rivard (eds.), *Risk evaluation and biological reference points for fisheries management*. *Can. Spec. Publ. Fish. Aquat. Sci.* 120:101-118.
- Myers, R.A., A.A. Rosenberg, P.M. Mace, N. Barrowman, and V.R. Restrepo. 1994. In search of thresholds for recruitment overfishing. *ICES J. Mar. Sci.* 51:191-205.

- Reed, W.J. 1979. Optimal escapement levels in stochastic and deterministic harvesting models. *J. Environ. Econ. Manage.* 6:350-363.
- Rosenberg, A., P. Mace, G. Thompson, G. Darcy, W. Clark, J. Collie, W. Gabriel, A. MacCall, R. Methot, J. Powers, V. Restrepo, T. Wainwright, L. Botsford, J. Hoening, and K. Stokes. 1994. Scientific review of definitions of overfishing in U.S. fishery management plans. NOAA Tech. Mem. NMFS-F/SPO-17.
- Sigler, M.F., and J.T. Fujioka. 1993. A comparison of policies for harvesting sablefish, *Anoplopoma fimbria*, in the Gulf of Alaska. In: G. Kruse, D.M. Eggers, R.J. Marasco, C. Pautzke, and T.J. Quinn II (eds.), *Proceedings of the International Symposium on Management Strategies for Exploited Fish Populations*. University of Alaska Sea Grant, AK-SG-93-02, Fairbanks, pp. 7-19.
- Thompson, G.G. 1993. A proposal for a threshold stock size and maximum fishing mortality rate. In: S.J. Smith, J.J. Hunt and D. Rivard (eds.), *Risk evaluation and biological reference points for fisheries management*. *Can. Spec. Publ. Fish. Aquat. Sci.* 120:303-320.
- Wada, T., C. Sato, and Y. Matsumiya. 1996. Fisheries management for the Pacific stock of chub mackerel, *Scomber japonicus*, based on spawning per recruit analysis. *Bull. Jpn. Soc. Fish. Oceanogr.* 60:363-371. (In Japanese.)
- Zheng, J., T.J. Quinn II, and G.H. Kruse. 1993. Comparison and evaluation of threshold estimation methods for exploited fish populations. In: G. Kruse, D.M. Eggers, R.J. Marasco, C. Pautzke, and T.J. Quinn II (eds.), *Proceedings of the International Symposium on Management Strategies for Exploited Fish Populations*. University of Alaska Sea Grant, AK-SG-93-02, Fairbanks, pp. 267-289.