

**Abstract**—Many modern stock assessment methods provide the machinery for determining the status of a stock in relation to certain reference points and for estimating how quickly a stock can be rebuilt. However, these methods typically require catch data, which are not always available. We introduce a model-based framework for estimating reference points, stock status, and recovery times in situations where catch data and other measures of absolute abundance are unavailable. The specific estimator developed is essentially an age-structured production model recast in terms relative to pre-exploitation levels. A Bayesian estimation scheme is adopted to allow the incorporation of pertinent auxiliary information such as might be obtained from meta-analyses of similar stocks or anecdotal observations. The approach is applied to the population of goliath grouper (*Epinephelus itajara*) off southern Florida, for which there are three indices of relative abundance but no reliable catch data. The results confirm anecdotal accounts of a marked decline in abundance during the 1980s followed by a substantial increase after the harvest of goliath grouper was banned in 1990. The ban appears to have reduced fishing pressure to between 10% and 50% of the levels observed during the 1980s. Nevertheless, the predicted fishing mortality rate under the ban appears to remain substantial, perhaps owing to illegal harvest and depth-related release mortality. As a result, the base model predicts that there is less than a 40% chance that the spawning biomass will recover to a level that would produce a 50% spawning potential ratio.

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## A catch-free stock assessment model with application to goliath grouper (*Epinephelus itajara*) off southern Florida

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The last decade has witnessed considerable interest in the so-called precautionary approach to resource management, where human activities are curtailed to prevent further environmental degradation without the burden of proving that these activities are to blame. Fisheries applications of the precautionary approach typically hinge on the notion that fishing pressure should be reduced in a pre-determined fashion as certain “limit” reference points are approached (FAO, 1995; Caddy, 1998; Restrepo et al., 1998). In the United States, the Magnuson-Stevens Fishery Conservation and Management Act (Public Law 94-265) mandates the development of fishery management plans (FMPs) that specify criteria for determining when a stock is overfished and the remedial measures necessary to ensure a timely recovery. The National Standard Guidelines developed by the National Marine Fisheries Service to implement the Act require each FMP to include an “MSY control rule” that comprises two reference points, known as the maximum fishing mortality threshold (MFMT) and the minimum stock size threshold (MSST). When the abundance of the stock dips below the MSST, special provisions must be made to rebuild the stock to the level that would support the maximum sustainable yield within a time frame that is as short as possible and that is commensurate with the intrinsic productivity of the stock and the needs of the fishing community.

Many modern stock assessment methods provide the machinery for determining limit reference points as well as for appraising where the stock is in relation to them and how quickly it can be rebuilt. However, these methods typically require data on total catch or absolute abundance, which are not always available. In the case of goliath grouper (*Epinephelus itajara*), for example, a recent review panel concluded that the catch statistics were unreliable and not useful for assessment purposes (Anon.<sup>1</sup>). Several *ad hoc* control rules have been developed to accommodate such “data-poor” situations. One of the more common is simply to define the MSST in terms of historical indices of abundance that supposedly represent a desirable stock condition (Annala, 1993; Cadrin et al., 2004). An advantage of this type of approach is that it is model-free, and nothing is assumed concerning the recovery rate of the stock. Being model-free, however, is a disadvantage with respect to the requirements of the Magnuson-Stevens Act, inasmuch as the recovery time cannot be estimated. Moreover, there may be other types of information that could influence the perception of the status of the stock, and it would be useful to integrate that information formally into the assessment.

<sup>1</sup> Anon. 2003. Goliath grouper data workshop report. SEDAR3-DW-1, 11 p. South Atlantic fishery Management Council, 1 Southpark Circle, Charleston SC 29406.

The purpose of this article is to introduce a model-based framework for estimating reference points, stock status, and recovery times in situations where catch data and other measures of absolute abundance are unreliable. The specific estimator developed in this study is essentially an age-structured production model recast in terms relative to pre-exploitation levels. A Bayesian estimation scheme is adopted to allow the incorporation of pertinent auxiliary information such as might be obtained from meta-analyses of similar stocks or anecdotal observations. The approach is applied to the population of goliath grouper off southern Florida, which is believed to have been severely depleted during the 1980s and has been protected from all harvest since 1990 (GMFMC<sup>2</sup>).

**Materials and methods**

**Population dynamics model**

The study period begins when the stock is believed to be near virgin levels, such that the relative abundance  $N$  of each age class  $a$  at the beginning of the first year is given by

$$N_{a,1} = \begin{cases} 1 & a = a_r \\ N_{a-1,1}e^{-M_{a-1}} & a_r < a < A \\ N_{A-1,1}e^{-M_{A-1}} / (1 - e^{-M_A}), & a = A \end{cases} \quad (1)$$

where  $a_r$  = the youngest age class in the analysis;  
 $A$  = a “plus-group” representing age classes  $A$  and older; and  
 $M$  = the natural mortality rate.

The relative abundance at the beginning of subsequent years ( $y$ ) is modeled by the recursion

$$N_{a,y} = \begin{cases} r_y & a = a_r \\ N_{a-1,y-1}e^{-F_{y-1}v_{a-1}-M_{a-1}} & a_r < a < A \\ N_{A-1,y-1}e^{-F_{y-1}v_{A-1}-M_{A-1}} + \\ N_{A,y-1}e^{-F_{y-1}v_A-M_A}, & a = A \end{cases} \quad (2)$$

where  $r_y$  = the annual recruitment to age class  $a_r$  relative to virgin levels;  
 $F$  = the fishing mortality rate on the most vulnerable age class; and

$v_a$  = the relative vulnerability of the remaining age classes (which implicitly includes factors such as gear selectivity, size limit regulations, and the fraction of the stock exposed to the fishery).

Relative recruitment ( $r$ ) is modeled as a first-order log-normal autoregressive process,

$$r_y = \mu_r e^{\varepsilon_y} \quad (3)$$

$$\varepsilon_y = \rho_r \varepsilon_{y-1} + \eta_{r,y},$$

where  $\mu_r$  = the median expectation  
 $\rho_r$  = the correlation coefficient; and  
 $\eta$  = normal-distributed random variates having mean 0 and standard deviation  $\sigma_r$  (ostensibly representing the effect on recruitment of fluctuations in the environment).

The median is modeled by the Ricker or Beverton-Holt spawner-recruit functions recast in terms of the maximum lifetime reproductive rate  $\alpha$  and relative spawning biomass  $s$ :

$$\mu_r = \begin{cases} s_{y-a_r} \alpha^{1-s_{y-a_r}} & \text{Ricker} \\ \frac{\alpha s_{y-a_r}}{1 + (\alpha - 1)s_{y-a_r}} & \text{Beverton and Holt} \end{cases} \quad (4)$$

$$s_y = \sum_{a=a_r}^A E_a e^{-(F_y v_a + M_a) s} N_{a,y} / \sum_{a=a_r}^A E_a e^{-M_a t} N_{a-1},$$

where  $E$  = an index of the per-capita number of eggs produced by each age class ( $E$ ); and  
 $t_s$  = the fraction of the year elapsed at the time of spawning.

The shapes of these two curves are essentially the same as the conventional relationships (Fig. 1); however their domain is implicitly limited to the interval  $0 \leq s \leq 1$  (see Appendix 1 for a derivation).

The fishing mortality rate on the most vulnerable age class  $F$  is also modeled as a first-order lognormal autoregressive process,

$$F_y = \mu_F e^{\delta_y} \quad (5)$$

$$\delta_y = \rho_F \delta_{y-1} + \eta_{F,y},$$

where  $\mu_F$  = the median level;  
 $\rho_F$  = the correlation coefficient; and  
 $\eta$  = normal-distributed random variates having mean 0 and standard deviation  $\sigma_F$ .

The median annual is generally assumed to be proportional to an index of fishing effort  $f$ :

$$\mu_F = \phi f_y, \quad (6)$$

<sup>2</sup> GMFMC (Gulf of Mexico Fishery Management Council). 1990. Amendment number 2 to the fishery management Plan for the reef fish fishery of the Gulf of Mexico, 31 p. Gulf of Mexico Fishery Management Council. 3018 North US Highway 301, Suite 1000, Tampa, FL 33619.

where  $\phi$  can vary among three eras of exploitation: a “prehistoric” period, during which little data are available; a “modern” period, when presumably there are some data on abundance or mortality rates; and a “future” period, when fishing mortality rates are controlled (input). The absence of data during the “prehistoric era” generally precludes the estimation of annual deviations in recruitment ( $\epsilon$ ) or fishing mortality rate ( $\delta$ ) during that period.

The average weight or fecundity of the plus group is expressed as a function of the average age of the plus-group. Initially, it is assumed that the age composition of the plus-group is in equilibrium consistent with Equation 1, in which case the average age of the plus-group at the beginning of the first year is approximately

$$\bar{a}_{A,1} = A + \frac{e^{-M_A}}{1 - e^{-M_A}}. \quad (7)$$

Subsequently, the age of the plus-group is updated as

$$\bar{a}_{A,y+1} = \frac{AN_{A-1,y}e^{-F_y v_{A-1} - M_{A-1}} + (\bar{a}_{A,y} + 1)N_{A,y}e^{-F_y v_A - M_A}}{N_{A,y+1}}. \quad (8)$$

### Reference points

Equations 1–4 describe the relative dynamics of a population apart from its absolute abundance. As such they are suitable for developing management plans where the fishing mortality rate is controlled directly (e.g., by reducing effort) and the biomass reference points are expressed on a relative scale. When the virgin spawning biomass itself is used as the reference point, the estimated value of  $s_y$  is a direct measure of the status of the stock. For example, if the management goal is to maintain spawning biomass at or above 50% of the virgin level, then estimates of  $s$  below 0.5 may trigger some action to reduce fishing pressure.

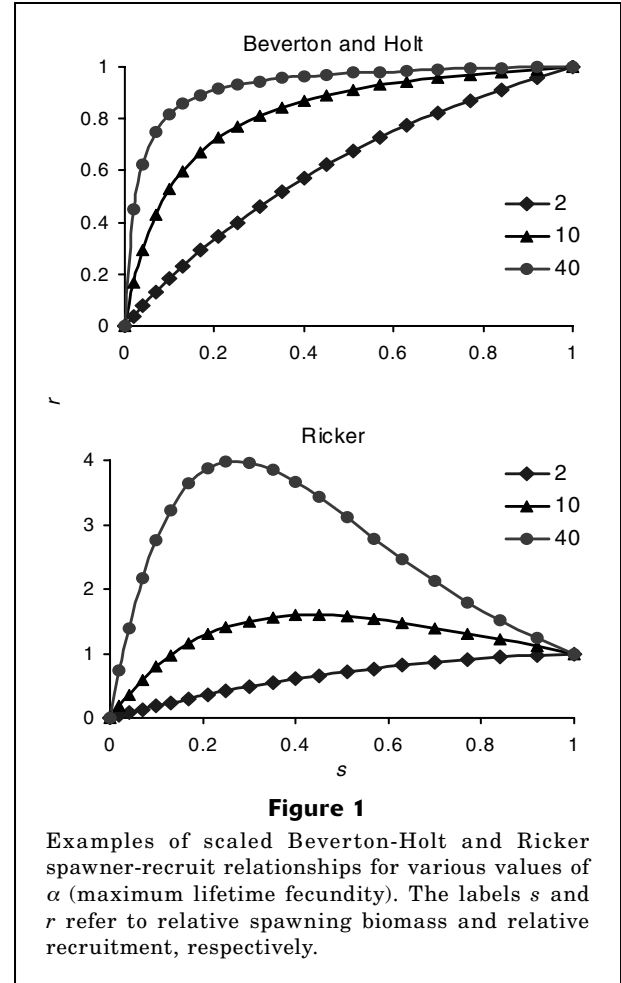
A related reference point is the equilibrium spawning potential ratio (Goodyear, 1993), defined as the expected lifetime fecundity per recruit at a given  $F$  ( $\psi_F$ ) divided by the expected lifetime fecundity in the absence of fishing ( $\psi_0$ ):

$$p = \frac{\psi_F}{\psi_0} \quad (9)$$

$$\psi_F = \sum_{a=0}^A E_a e^{-(F_a + M_a)t_s} e^{-\sum_{i=0}^{a-1} F v_i + M_i}.$$

As shown in Appendix 2, the corresponding equilibrium level of relative spawning biomass (denoted by a tilde) may be computed as

$$\tilde{s}_p = \begin{cases} 1 + \frac{\log_e p}{\log_e \alpha} & \text{Ricker} \\ \frac{\alpha p - 1}{\alpha - 1} & \text{Beverton and Holt} \end{cases} \quad (10)$$



Note that  $\tilde{s}_p$  is independent of the vulnerability vector  $v$ . Accordingly, MSST definitions based on  $\tilde{s}_p$  will have the desirable property of being insensitive to changes in fishery behavior.

Other management plans employ reference points such as  $F_{max}$  or  $F_{0.1}$ , which are based on the yield per recruit statistic

$$\left(\frac{Y}{R}\right) = \sum_{a=0}^A w_a F v_a \frac{1 - e^{-(F v_a + M_a)}}{F v_a + M_a} e^{-\sum_{i=0}^{a-1} F v_i + M_i}, \quad (11)$$

where  $w_a$  is some measure related to the average weight of the catch. Inasmuch as there are no terms involving the absolute abundance of the stock, the calculation of such statistics poses no special problems for the relative framework presented in the present study. Prescriptions based on the maximum sustainable yield ( $MSY$ ) are slightly more complicated because equilibrium yield is the product of equilibrium recruitment  $\tilde{R}$  and equilibrium yield per recruit:

$$\tilde{Y} = \tilde{R}_F \sum_{a=0}^A w_a F v_a \frac{1 - e^{-(F v_a + M_a)}}{F v_a + M_a} e^{-\sum_{i=0}^{a-1} F v_i + M_i}. \quad (12)$$

However, the fishing mortality rate that maximizes Equation 12 also maximizes Equation 12 divided by the virgin recruitment  $R_0$  (a constant). Thus,  $F_{MSY}$  may be obtained from

$$\max_F \left\{ \frac{F \bar{s}_p}{p} \sum_{a=0}^A w_a v_a \frac{1 - e^{-(F v_a + M_a)}}{F v_a + M_a} e^{-\sum_{i=0}^{a-1} F v_i} \right\}. \quad (13)$$

where  $\bar{s}_p/p$  has been substituted for  $\tilde{R}_F/R_0$ .

The values of  $p$  and  $\bar{s}_p$  corresponding to  $F_{max}$ ,  $F_{0.1}$ , or  $F_{msy}$  may be calculated by means of Equations 9 and 10, respectively. Note however, that  $p$  is no longer the target value specified by management, but a derivative of the targeted values of  $F$ . This means that MSST definitions based on  $\tilde{s}_{max}$ ,  $\tilde{s}_{0.1}$ , and  $\tilde{s}_{msy}$  will vary somewhat with the behavior of the fishery. In some cases this could lead to risk prone situations where the perception of stock status changes simply because the fishery targets different age groups (i.e., the definition of MSST changes rather than the abundance of the resource). In the case of MSY, a more stable alternative is to define the MSST in terms of a "spawn at least once" policy where mature animals are regarded as fully vulnerable to the fishery and immature animals area regarded as invulnerable.

### Parameter estimation

The equations above include numerous "unknowns" representing the processes of reproduction, mortality, and growth. In the case of "data-poor" stocks like goliath grouper, there are insufficient data to estimate all of these unknown parameters with an acceptable level of precision. However, it is often possible to increase the precision of the estimates through the use of Bayesian prior probability densities constructed to reflect expert opinion (e.g., Wolfson et al., 1996; Punt and Walker, 1998) or based on meta-analyses involving similar species (e.g., Liermann and Hilborn, 1997; Maunder and Deriso, 2003).

The Bayesian approach to estimation seeks to develop a "posterior" probability density for the parameters  $\Theta$  that is conditioned on the data  $D$ ,  $P(\Theta | D)$ . By application of Bayes rule it is easy to show that

$$P(\Theta | D) \propto P(D | \Theta) P(\Theta). \quad (14)$$

where  $P(D | \Theta)$  is the sampling density (likelihood function); and  $P(\Theta)$  is the prior density (in this case the analyst's best guess of the probability density for  $\Theta$ ).

Estimates for  $\Theta$  may be obtained by integrating the posterior (the classical Bayes moment estimator; cf. Gelman et al., 1995)

$$\bar{\theta}_j = \int \theta_j P(D | \Theta) P(\Theta) d\theta_j, \quad \theta_j \in \Theta \quad (15)$$

or by minimizing its negative logarithm (the highest posterior density estimator; Bard, 1974)

$$\min_{\Theta} \{-\log_e P(D | \Theta) - \log_e P(\Theta)\}. \quad (16)$$

In the present model, a prior needs to be specified for the parameters reflecting recruitment ( $\alpha$  and  $\varepsilon_y$ ), mortality ( $M$ ,  $\phi$ ,  $\delta_y$ ,  $v_a$ ), fecundity ( $E_a$ ), and growth in weight ( $w_a$ ). It is assumed in the present study that the parameters are statistically independent with respect to prior knowledge, such that the joint prior is merely the product of the marginal priors for each parameter. The exceptions are the process error functions for the annual recruitment and fishing mortality rate deviations,  $\varepsilon_y$  and  $\delta_y$ . These are assumed to be autocorrelated lognormal variates with negative- log density functions of the form

$$-\log P(\varepsilon) = \frac{1}{2\sigma_r^2} \left[ \varepsilon_1^2 + \sum_y (\varepsilon_{y+1} - \rho_r \varepsilon_y)^2 \right] + 1 \text{ g } \sigma_r, \quad (17)$$

where  $\rho_r$  = the correlation coefficient; and  $\sigma_r^2$  = the variance of  $\log_e \eta_r$ .

For stability reasons, it is assumed that  $\varepsilon_0 = 0$ .

It is possible, at least in principle, to conduct an assessment based on prior specifications alone. However, it may be difficult to develop sufficiently informative priors for some of the parameters, particularly for the fishing mortality rates. The preferred approach, of course, is to condition the estimates on data. With the present model it is assumed that catch data are either unavailable or unreliable, otherwise a standard age-structured production model (cf. Restrepo and Legault, 1998) would be more appropriate. However, time series of catch-per-unit-of-effort data or fishery-independent surveys are often available even when total catches are not. To the extent that changes in these data ( $c$ ) are proportional to changes in the abundance of the population as a whole ( $N$ ), they may be modeled as

$$c_{i,y} = q_i \sum_a v_{i,a} e^{-(F_y v_a + M_a)} e^{\gamma_{i,y}} \quad (18)$$

$$\gamma_{i,y} \sim \text{Normal}(0, \sigma_{c,i}),$$

where  $i$  indexes the particular survey time series;

$q$  = the proportionality coefficient scaling the time series to the relative abundance of the population;

$t_i$  = the fraction of the year elapsed at the time of the survey;

$\sigma_{c,i}$  = the standard deviation of the fluctuations in  $\log_e c_i$  owing to observation errors or changes in the distribution of the stock; and

$v_a$  and  $v_{i,a}$  = the relative vulnerability of each age class to the fishery and the  $i^{\text{th}}$  survey, respectively.

The corresponding negative logarithm of the sampling density is

$$-\log P(c|\Theta) = \sum_i \sum_y \left\{ \frac{1}{2\sigma_{c,i}^2} \left( \log_e(c_{i,y}) - \log_e \left( q_i \sum_a v_{i,a} N_a e^{-(F_y v_{i,a} + M_a) \Delta} \right) \right)^2 + \log \sigma_{c,i} \right\}. \quad (19)$$

Anecdotal observations may be treated in similar fashion. For example the perceptions of constituents on the abundance of the resource relative to virgin levels ( $n$ ) can be modeled as

$$n_y = \frac{\sum_a \lambda_a N_{a,y} e^{-(F_y v_{i,a} + M_a) \Delta}}{\sum_a \lambda_a e^{-M_a \Delta} e^{-\sum_{i=0}^{a-1} M_i}} e^{\gamma_{n,y}} \quad (20)$$

$\gamma_{n,y} \sim Normal(0, \sigma_n)$ .

Here  $\lambda_a$  is the relative contribution of each age class in forming the perception of total abundance (e.g., fishermen may never encounter very young fish),  $\Delta$  is the time of the year most reflective of the period upon which the perceptions were based (e.g., the peak of the fishing season), and  $\sigma_n$  is the standard deviation of the fluctuations in  $\log_e n_y$ .

It is not generally possible to obtain consistent estimates for all of the elements of the covariance matrix  $\mathbf{V}$  (i.e.,  $\rho_F$ ,  $\sigma_{F_y}^2$ ,  $\rho_r$ ,  $\sigma_r^2$ ,  $\sigma_{c,i}^2$  and  $\sigma_n^2$ ). In the case of survey data, the variances associated with sampling variability are often estimated extraneous to the population model (e.g., during the standardization procedure). However, there may be additional variance owing to fluctuations in the distribution of the stock in relation to the survey area (IWC, 1994). To accommodate such possibilities, the survey variance parameters are modeled as

$$\begin{aligned} \sigma_{c,i,y}^2 &= \chi_{c,i,y}^2 + \beta_{c,i} \sigma^2 \\ \sigma_{n,y}^2 &= \chi_{n,y}^2 + \beta_n \sigma^2, \end{aligned} \quad (21)$$

where the  $\chi_{c,i,y}^2$  and  $\chi_{n,y}^2$  = the annual observation variances (estimated outside the model);

$\sigma^2$  reflects some overall process variance (estimated within the model); and

$\beta$  = constant multipliers (usually fixed by the analyst based on a careful consideration of the inherent variability of the underlying processes).

The recruitment variance and correlation coefficient are generally inestimable without a good index of recruitment and may have to be fixed to some moderate values (say  $\sigma_r=0.4$  and  $\rho=0.5$ ).

**Table 1**

Anecdotal reports from nine individuals concerning the abundance of adult-size goliath grouper (*Epinephelus itajara*) in 1990 in relation to their abundance in the 1950s (expressed as percent reduction).

Interviewee	Method	% reduction
1	diver	60
2	angler	70
3	diver	75
4	angler	90
5	diver	90
6	diver	95
7	diver	95
8	diver	97
9	diver	98

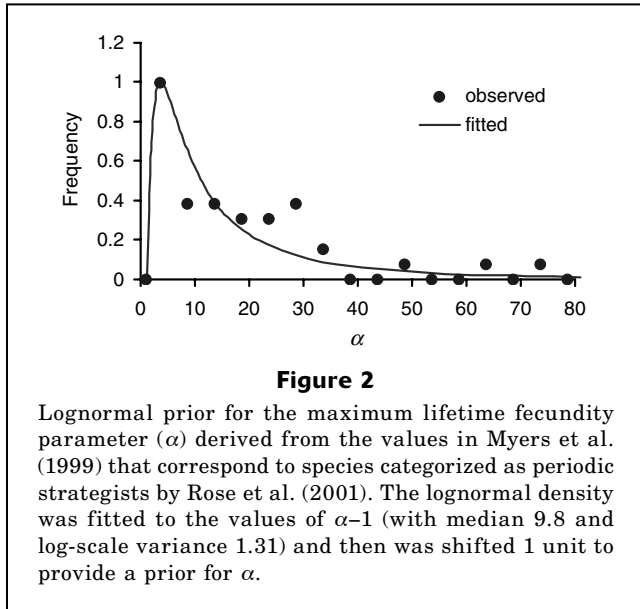
The model outlined above was implemented by using the nonlinear optimization package AD Model Builder (version 4.5, Otter Research Ltd., Sidney, Canada), which provides facilities for estimating the mode and shape of the posterior distribution. Confidence intervals for the probability of recovery were generated directly from the posteriors approximated by the likelihood profile method (the accuracy of which was checked by replicating the prior distributions without data and by comparing the modes of the posterior with the HPD estimates). For some quantities confidence limits were computed by using normal approximations centered at the HPD estimate with variances obtained by inverting the Hessian matrix. This approach reduced computing time considerably, but the approximations were poor for confidence intervals broader than 80 percent owing to the thick tails and skewed nature of the posterior distributions.

**Application to goliath grouper**

Goliath grouper are large, long-lived predators found predominantly in the tropical western Atlantic and Caribbean Sea. They are among the least wary of reef fishes, easily approached by spearfishers and readily caught in traps or by hook and line gear. Not surprisingly, they have declined considerably throughout much of their range (Sadovy and Eklund, 1999). Although there are few data on the historic abundance of these animals in southern Florida, anecdotal reports suggest that they were much more abundant during the 1950s and 60s than they are now (Table 1). Concerns of overfishing prompted regulators in the U.S. to impose a moratorium on the harvest of goliath grouper that has remained in effect since 1990. To date, the duration of the moratorium has not been specified owing to the paucity of information on their potential recovery rates.

**Spawner-recruit relationship** There does not appear to be any reliable information on the nature of the

spawner-recruit relationship for any grouper species. A Beverton and Holt model was assumed in this study because it is difficult to envision a mechanism for the strong density dependence in mortality rates required by the Ricker model. A prior for the value of  $\alpha$  (Fig. 2) was constructed from a subset of the values collected by Myers et al. (1999) that corresponds to larger, highly fecund fishes with long life spans (the ‘periodic’ strategists of Rose et al., 2001).



**Fecundity and growth** To date there are insufficient data for estimating a fecundity-at-age relationship for goliath grouper. We followed Legault and Eklund<sup>3</sup> and substituted the weight at age relationship:

$$E_a = \begin{cases} 0 & a < 6 \\ w_a = 1.31 \times 10^{-5} l^{3.056} & a \geq 6 \end{cases} \quad (22)$$

$$l = 200.6(1 - e^{-0.12(a+0.4)})$$

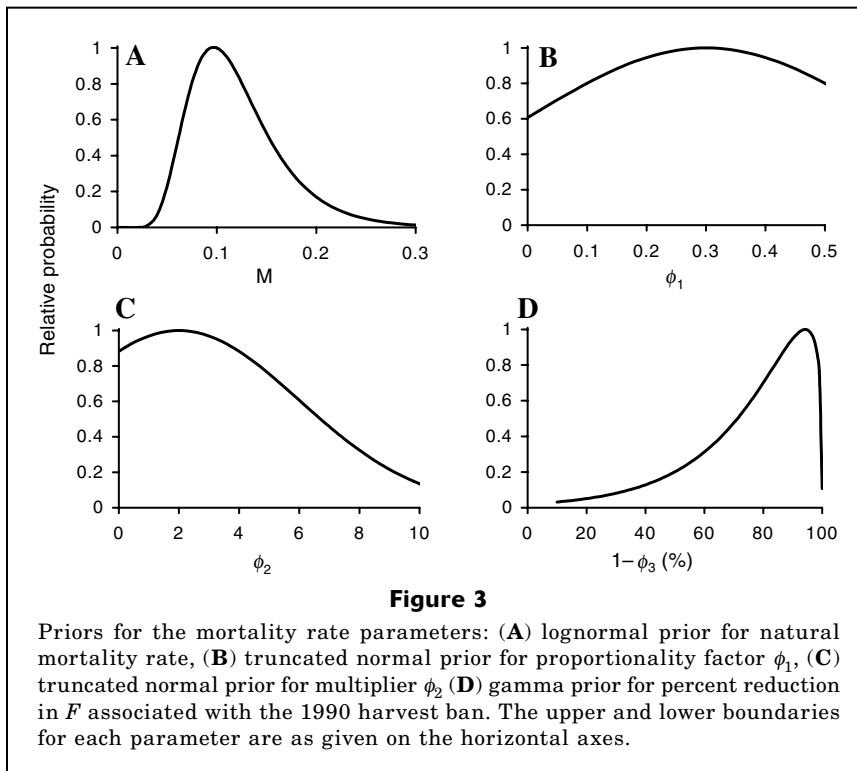
where  $w$  = weight in kg; and  
 $l$  = length in cm expressed as a von Bertalanffy function of age (see Bullock et al., 1992).

**Natural mortality** The maximum observed age of 37 years (Sadovy and Eklund, 1999) suggests a value for  $M$  of about 0.11/yr according to the method of Hoening (1983). Legault and Eklund<sup>5</sup> suggested a plausible range of 0.037 yr to 0.19/yr (midpoint 0.11) based on an analysis of the fraction surviving to various maximum ages. To reflect this uncertainty, a lognormal prior with a median of 0.11 and CV of 0.4 was used (Fig. 3A).

**Fishing mortality rate and relative vulnerability** A large fraction of the recreational landings of goliath grouper appear to come from the Ten Thousand Islands area in Southwest Florida, where most of the animals caught have been between the ages of one and five years. However, large animals were often targeted by commercial and recreational fishermen in other areas. Accordingly, we assumed the vulnerability of goliath grouper generally increased with age according to the sigmoid-shaped logistic curve

$$v_a = \frac{1}{1 + e^{-(a-a_{50})/d}} \quad (23)$$

Estimates for the parameters  $a_{50}$  and  $d$  were obtained by fitting the curve (weighted by cumulative mortality at age) to the relative frequency of ages in two different data sets. The first data set included mostly juveniles animals between the ages of 0 and 5, obtained during creel censuses of recreational catches in the Ten Thousand Islands area of the Everglades National Park (see Porch et



<sup>3</sup> Legault, C. M., and A.-M. Eklund. 1998. Generation times for Nassau grouper and jewfish with comments on M/K ratios. Sustainable Fisheries Division Contribution SFD-97/98-10A, 5 p. Southeast Fisheries Science Center, 75 Virginia Beach Drive, Miami, Florida 33149.

al.<sup>4</sup>). The second data set included mostly adult animals obtained opportunistically from recreational and commercial catches in the eastern Gulf of Mexico (Bullock et al., 1992). The SEDAR stock assessment review panel based their advice on models that used the former selection curve (Kingsley<sup>5</sup>); however the effect of using the latter curve was examined as a sensitivity analysis. The two curves are contrasted in Fig. 4A.

The fishing mortality rate on the most vulnerable age class was modeled as follows:

$$F_y = \begin{cases} \phi_1 f_y & 1900 \leq y \leq 1979 \\ \delta_y \phi_2 F_{1979} & 1980 \leq y < 1990 \\ \phi_3 \bar{F}_{1980-89} & 1990 \leq y \end{cases} \quad (24)$$

where  $f_y$  = a time series of historical effort; and  $\phi_1, \phi_2, \phi_3, \delta_y$  are parameters to be estimated.

In the present study, effort was assumed to track the U.S. Census<sup>6</sup> for the number of people living in South Florida coastal counties between 1900 and 1980. From 1980 to 1989 this assumption was no longer required owing to the availability of several time series of relative abundance (see below). Instead, interannual variations in fishing mortality were modeled according to Equation 5 with median  $\phi_2 F_{1979}$ , log-scale variance  $\sigma_F^2=0.15$  and correlation coefficient  $\rho_F=0.5$ , which essentially amounts to a mild constraint on year-to-year changes in  $F$ . The nonzero correlation coefficient is intended to reflect the momentum in effective fishing effort from one year to the next that arises from a combination of market demands and the tendency of many fishermen to target only the species they are most adept at catching. Even so, the relatively large variance term admits substantial inter-annual variations if the data warrant them. Moreover, runs with  $\rho_F=0.0$  (no year-to-year momentum) did not produce substantially different results.

The effect of the harvest moratorium was modeled as a percentage  $\phi_3$  of the average fishing mortality rate in the 1980–89 period. Relatively uninformative priors were used for  $\phi_1$  and  $\phi_2$  (Fig. 3, B and C). A somewhat

more informative prior with bounds between 0.01 and 0.5 was used for  $\phi_3$ , based on the opinions of members of the SEDAR panel (Fig. 3D).

**Survey information** Porch and Eklund (2004) have developed relative indices of abundance from two visual surveys: the personal observations of a professional spearfisher (DeMaria<sup>7</sup>) and a volunteer fish-monitoring program administered by the Reef Education and Environmental Foundation (REEF 2000). In addition, Cass-Calay and Schmidt<sup>8</sup> have standardized catch rate data collected in the Ten Thousand Islands area by the Everglades National Park (ENP). The two visual surveys are assumed to reflect the abundance of mature fish ages 6 and older (based on diver reports of size). The ENP catch rate index, on the other hand, is assumed to reflect the relative abundance of juveniles with relative vulnerabilities given by the dome-shaped gamma function (normalized to a maximum of 1):

$$v_{ENP,a} = \left( \frac{a}{a_{100\%}} e^{1-a/a_{100\%}} \right)^{cv^{-2}-1}, \quad (25)$$

where  $a_{100\%}$  = the most vulnerable age; and  $CV$  = the coefficient of variation.

Estimates for  $a_{100\%}$  (3.47) and  $CV$  (0.34) were obtained by fitting the mortality-weighted gamma curve to the frequency of ages 0–7 in the Ten Thousand Islands data mentioned earlier (for more detail see Porch et al.<sup>4</sup>). The resulting curve is shown in Figure 4B.

**Anecdotal impressions of stock status** Johannes et al. (2000) pointed out that local fishermen often disagree with the conclusions drawn by scientists in data-poor situations and suggest that many times additional data will prove the fishermen correct. As mentioned earlier, expert judgements about the relative abundance of a stock can be treated as data or represented by a “prior.” We collected information on the value of  $s$  at the time moratoriums began (1990) by interviewing fishermen and divers who had been active in southern Florida since the early 1960s or before. Specifically, interviewees were asked to state their perception of the percent reduction in goliath grouper populations from the time they began diving to the time the moratorium on catch was imposed (1990). The average percent reduction reported for large goliath (approximately age 6 and older) was 86% (standard deviation of about 13%, Table 1). This information was modeled as data in accordance with Equation 20.

<sup>4</sup> Porch, C. E., A-M. Eklund and G. P. Scott. 2003. An assessment of rebuilding times for goliath grouper. Sustainable Fisheries Division Contribution SFD-2003-0018. Southeast Fisheries Science Center, 75 Virginia Beach Drive, Miami, Florida 33149. 26 p.

<sup>5</sup> Kingsley, M. C. S., ed. 2004. The Goliath Grouper in southern Florida: assessment review and advisory report. Report prepared for the South Atlantic Fishery Management Council, the Gulf of Mexico Fishery Management Council, and the National Marine Fisheries Service, 17 p. South Atlantic fishery Management Council, 1 Southpark Circle, Charleston SC 29406.

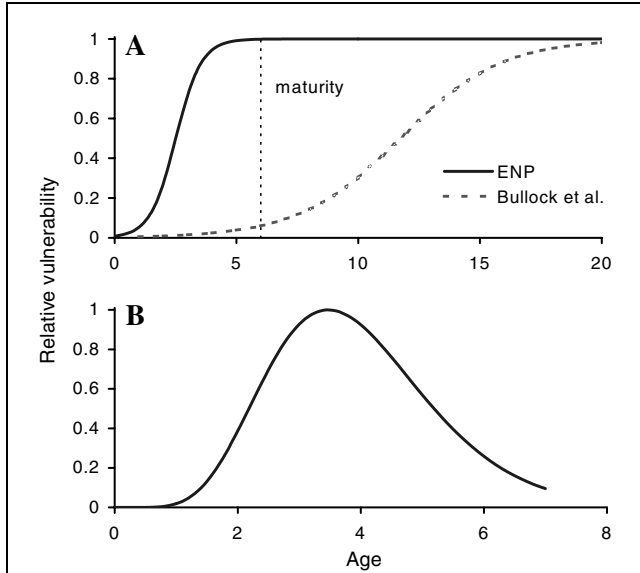
<sup>6</sup> Population of Florida Counties by Decennial Census: 1900 to 1990, 4 p. 1995. Compiled and edited by Richard L. Forstall. Population Division, U.S. Bureau of the Census. Washington, DC 20233

<sup>7</sup> DeMaria, D. 2004. Personal. observ. P.O. Box 420975, Summerland Key, FL 33042.

<sup>8</sup> Cass-Calay, S. L., and T. W. Schmidt. In review. Standardized catch rates of juvenile goliath grouper, *Epinephelus itajara*, from the Everglades National Park Creel Survey, 1973–1999.

**Results**

The model was able to fit the ENP index of juvenile goliath grouper very well but could not reconcile the conflicting trends indicated by the DeMaria and REEF indices



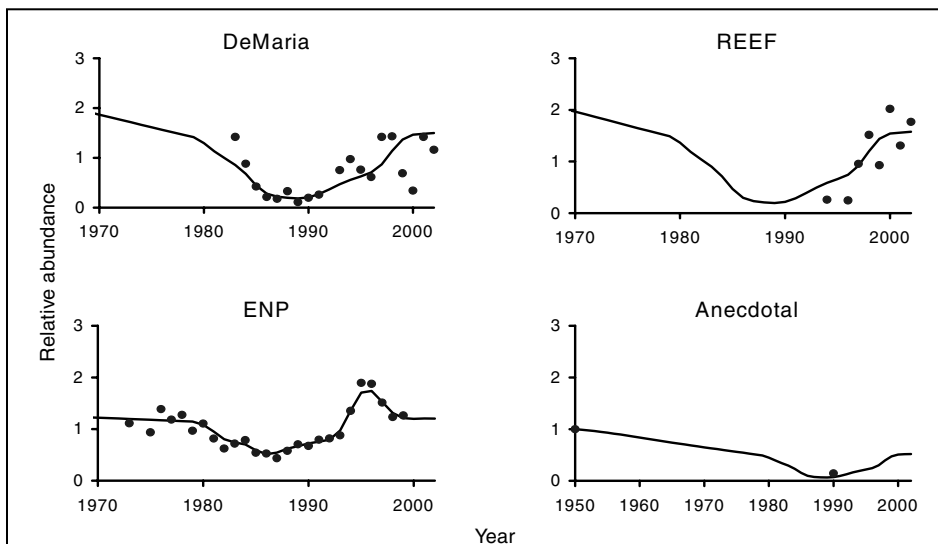
**Figure 4**

Selection curves used to represent the vulnerability of goliath grouper (*Epinephelus itajara*) to (A) the overall fishery and (B) Everglades National Park anglers. The logistic curves shown in (A) were fitted to either age-composition data derived from the Everglades National Park (ENP) creel census or opportunistic samples from offshore fishing trips (Bullock et al., 1992).

for adult goliath grouper (Fig. 5). The estimated trends in spawning biomass were rather uncertain (Fig. 6A), but nevertheless indicated a rapid decline to about 5% of virgin levels by the time the harvest ban was imposed in 1990, followed by a significant increase. The estimates of fishing mortality were also somewhat uncertain, but generally indicated a gradual increase in fishing mortality to moderate levels during the 1970s followed by a rapid increase during the 1980s (Fig. 6B). The harvest moratorium was estimated to have been about 83% effective in reducing fishing mortality, nevertheless losses owing to human activities (e.g., illegal harvest and release mortality of animals caught at depth) were still estimated to be substantial ( $F = 0.05/\text{yr}$ ). If, in accordance with the Gulf of Mexico Management Council's generic Sustainable Fisheries Act amendment, the limit reference point is taken to be the equilibrium spawning biomass corresponding to a spawning potential ratio of 50%, then the model indicates that current fishing mortality rates are near  $F_{50\%}$  and that there is less than a 50% chance the stock will recover within 15 years (Fig. 7).

Sensitivity runs were conducted to examine the implications of 1) dropping one or more of the indices, 2) increasing the assumed minimum age represented in the REEF and DeMaria indices from 6 to 10, 3) assuming that the historical period began in 1950 rather than 1900 and using the anecdotal information as a tuning index and (4) using the alternate fishery selection curve that was fitted to the data from Bullock et al. (1992), where adult animals were much more vulnerable to the fishery than were juveniles. Of these, the results were most sensitive to removal of the DeMaria index—the projected trends being much more optimistic (Fig. 8). This is because the DeMaria index indicates that the adult population increased rapidly

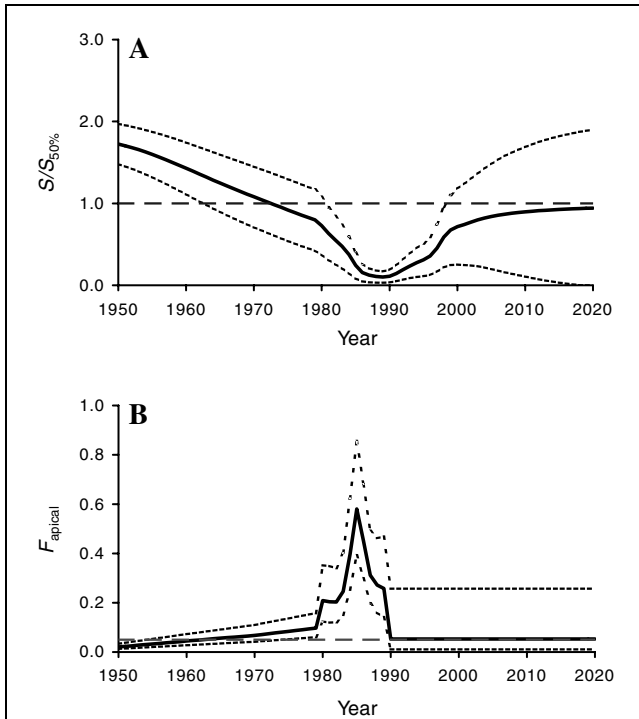
during the first few years of the harvest ban, but then suffered a set back in 1999 and has since leveled off. In contrast, the REEF index indicates that the population continued to increase during that time. Thus, when the DeMaria index is removed, the model allows for a faster postmoratorium increase in the adult population by estimating a low fishing mortality rate of about 0.01/yr (i.e., a harvest ban that is 97% effective). The fishing mortality rate estimates for the 1980s are also lower without the DeMaria index inasmuch as the DeMaria index indicates a more precipitous decline during that time than the ENP index (the REEF index does not begin until 1994).



**Figure 5**

Base model fitted to the four indices of abundance for goliath grouper (*Epinephelus itajara*) in southern Florida.





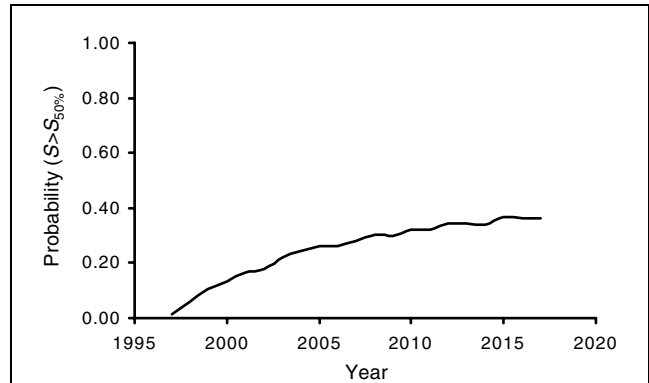
**Figure 6**

Base model predictions of (A) spawning biomass of goliath grouper (*E. itajara*) in southern Florida in relation to the equilibrium level associated with a spawning potential ratio of 50%,  $s/s_{50\%}$ , and (B) the apical fishing mortality rate on goliath grouper (*E. itajara*),  $F_{apical}$ . The lines with small dashes represent approximate 80% confidence limits and the dashed horizontal lines represent the levels associated with an SPR of 50%.

The sensitivity run with the alternate selection curve also produced more optimistic results (Fig. 8). Inasmuch as the model now attributes most of the fishing mortality to age classes well beyond the age at first maturity (see Fig. 4), the spawning stock biomass is estimated to have been reduced to a lesser extent (to about 10% of virgin levels by 1990 as compared to 5%). Thus, other things being equal, recovery requires less time. The level of  $F_{50\%}$  increased with the alternate selection curve because fewer age classes are affected by fishing.

**Discussion**

All of the model formulations examined depicted the same qualitative patterns: escalating fishing mortality rates and rapidly declining spawning biomass, particularly during the 1980s, followed by a sharp decrease in fishing mortality and strong recovery in spawning biomass after the 1990 harvest ban. These trends are remarkably consistent with the anecdotal observations shown in Table 1 and Figure 5 as well as with the expert testimony given during the SEDAR stock assessment



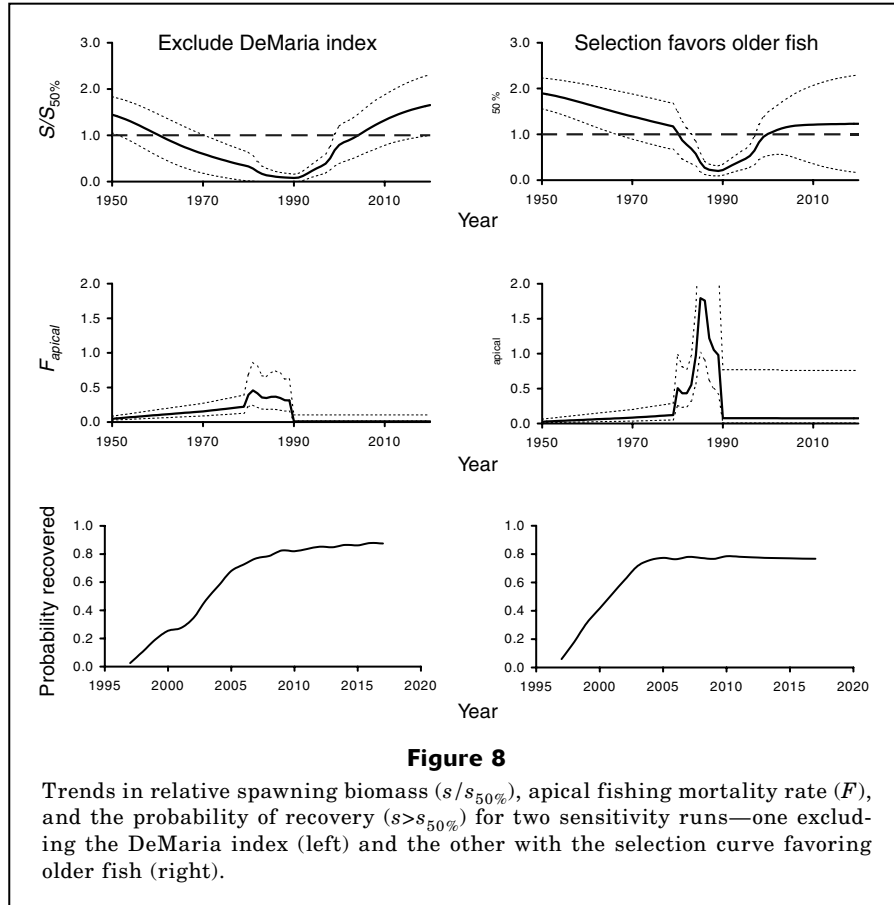
**Figure 7**

Predicted probability that the stock of goliath grouper (*E. itajara*) in southern Florida will have recovered to levels exceeding the equilibrium spawning biomass associated with a spawning potential ratio of 50%.

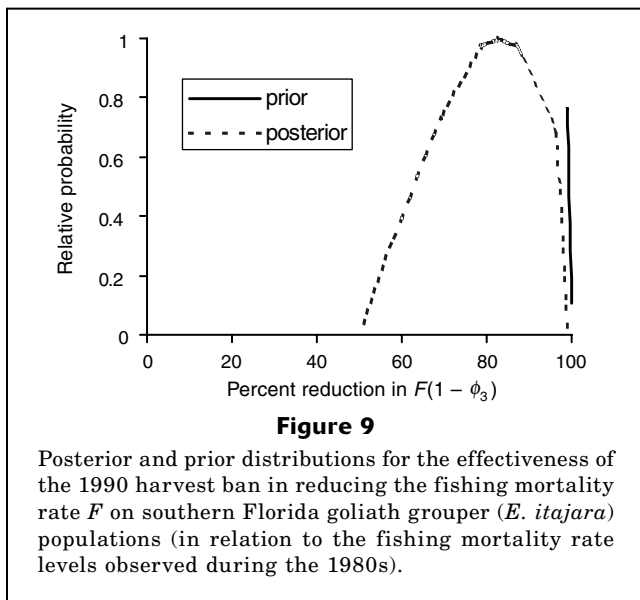
review. The estimated rapid increase in fishing mortality during the 1980s appears to reflect a real increase in effort that occurred due to elevated demand and selling prices (Sadovy and Eklund, 1999), as well as the widespread use of the LORAN-C navigational system (which made it easier for fishermen to relocate productive offshore shipwrecks). Thus, it seems safe to conclude that the population was overfished at the time the harvest ban was imposed and is currently undergoing a substantial recovery. Less clear is the extent to which the population has recovered since the harvest ban.

Using the base model, we estimated that the harvest ban has reduced fishing pressure by more than 50%, but probably less than 90% (Fig. 9). Thus, there is a strong chance that the current fishing mortality rate, although greatly reduced as compared to the 1980s, remains greater than  $F_{50\%}$  (i.e., above 0.05/yr). This in turn translates into less than a 40% chance that the population will recover to levels above  $s_{50\%}$  within the next 15 years. Several fishermen have testified that the harvest ban is probably less than 90% effective because goliath grouper are still taken illegally in places and because animals caught and released in deeper water often do not survive<sup>7</sup>; therefore this result does not appear unrealistic.

More optimistic results, implying a 70% to 80% chance of recovery within 15 years, were obtained when the DeMaria index was excluded or when selection was oriented more towards older animals. There does not appear to be a strong *a priori* case for excluding the DeMaria index in favor of the REEF and ENP indices. Although the coverage is rather limited, the trends of the DeMaria index are consistent with those of the ENP index (with a suitable time lag) and with anecdotal accounts of the trends in other areas.<sup>7</sup> The issue of selection is more vexing. It can be argued that the age-composition data from the ENP creel census adequately reflects the composition of the juvenile catch inasmuch



as it comes from the center of juvenile abundance; however most adults were caught outside this area of abundance. Thus, the relative contribution of juveniles and adults to the overall catch is unclear and the directional bias in the fitted logistic selection curve is uncertain.



The only other age composition information that has come to light comes from the study by Bullock et al (1992), which was not designed to provide a random sample of the catch and is probably biased towards larger animals caught on offshore wrecks. In principle, one could reflect this uncertainty more formally either by developing a prior for the selectivity parameters or else by weighting the results from the two selection models. The scientists on the SEDAR stock assessment review panel based their advice on the selection curve derived from the ENP data,<sup>7</sup> which is equivalent to placing negligible weight on the curve derived from the Bullock et al. (1992) data; however they recognized the selection curve as an important source of uncertainty that is difficult to address without adequate data.

It is important to emphasize that the Bayesian approach adopted in the present study allows one to explicitly model the uncertainty about parameters such as  $M$ , for which no data may exist, but a prior distribution covering the plausible range of values may be specified. There is, of course, the potential for introducing bias when one or more of the priors are based on expert opinion or otherwise subjective information. However, the same sorts of bias can be introduced by conducting sensitivity analyses where the unknown parameters are fixed to various values selected by the analysts. Furthermore, if unbiased data are in short supply, analyses

based on completely uninformative priors will be useless for generating advice because the range of plausible outcomes is too large. Accordingly, we view the use of subjective priors primarily as a vehicle for providing more realistic limits on uncertainty and prefer to express the model outcomes in terms of probability statements. For example, the point estimate from the base model indicated that the population would never recover to  $s_{50\%}$  because the fishing mortality rate under the harvest ban was still slightly above  $F_{50\%}$ . However, consideration of the uncertainty led to the conclusion that the chance of recovering to  $s_{50\%}$  within 15 years was nearly 40%.

Some sources of uncertainty have not been adequately accounted for in the above assessment. For example, the relationship between fecundity and age is unknown. We used weight-at-age as a proxy for the relative fecundity-at-age in our analysis, but it is often the case that fecundity increases with age faster than weight. If this is true for goliath grouper, then our projections would be too optimistic. It should also be remembered that the results apply strictly to the goliath grouper population in southern Florida. It is believed that the center of abundance for the population in U.S. waters is off southern Florida, particularly in the Ten Thousand Islands area, but goliath grouper are known to have occurred throughout the coastal waters of Gulf of Mexico and along the east coast of Florida, and on up through the Carolinas. Inasmuch as goliath grouper are not highly migratory, it is possible it may take some additional time for the species to fully occupy its historical range, thus delaying the overall recovery of the U.S. population.

The primary advantage of the catch-free assessment model proposed in the present study is that it does not require knowledge of the total number of removals. In this light it is worth noting that 623 of the 905 stocks included in the 2000 annual report to Congress on the Status of Fisheries were listed as having unknown status, often because catch data were either unavailable or deemed unreliable. Thus we expect the proposed method will become increasingly useful as fishery scientists are asked more and more to develop FMPs for poorly monitored fisheries. The fact that the model estimates the population's relative abundance, rather than its absolute abundance, is of little consequence when, as is often the case, adjustments to the target fishing mortality rate or catch quota are made in relation to the levels in previous years (Caddy, 2004). Moreover, certain biases tend to cancel out when dimensionless quantities like relative abundance are used. If, for example, only a consistent fraction of the population were sampled, then the absolute estimates of abundance would be biased but the relative estimates would not (Prager et al., 2003).

The greatest drawback of the catch-free method is probably its inability to provide direct estimates of the equilibrium catch levels associated with particular reference points (e.g., MSY). This situation could perhaps be ameliorated by obtaining estimates of absolute

abundance from a comprehensive short-term survey covering the entire range of the animal, in which case the relative outputs from the model (including relative catch) could be appropriately scaled. Alternatively, a long-term monitoring program at select sites located throughout the known range of the animal could be established to detect changes in relative abundance under various closely monitored trial levels of catch.

## Acknowledgments

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## Appendix 1: reparameterized spawner-recruit relationships

The number of young fish recruiting to a population ( $R$ ) is often related to the aggregate fecundity of the spawning stock ( $S$ ) by using one of two functional forms:

$$R = \begin{cases} aSe^{-bS} & \text{Ricker} \\ \frac{abS}{b+S} & \text{Beverton and Holt} \end{cases} \quad (\text{A.1})$$

The parameter  $a$  is the slope of the curve at the origin and the parameter  $b$  controls the degree of density dependence. Notice that the domain of both functions extends from zero to infinity, whereas in practice there must be some limitation on  $S$  and  $R$  even in the absence of fishing owing to environmental constraints (call them  $S_0$  and  $R_0$ , respectively). This being so, we obtain

$$a \frac{S_0}{R_0} = \begin{cases} e^{bS_0} & \text{Ricker} \\ 1 + S_0/b & \text{Beverton and Holt} \end{cases} \quad (\text{A.2})$$

The ratio  $S_0/R_0$  represents the maximum expected lifetime fecundity of each recruit and  $a$  represents the survival of recruits in the absence of density dependence. Accordingly, the product  $\alpha = aS_0/R_0$  may be interpreted as the maximum possible number of recruits produced by each spawner over its lifetime (Myers et al., 1999).

The dimensionless character of  $\alpha$  makes it useful for interspecies comparisons, or for borrowing values from species with similar life history strategies. Solving for  $b$  in terms of  $\alpha$  one obtains

$$b = \begin{cases} \log_e \alpha / S_0 & \text{Ricker} \\ S_0 / (1 - \alpha) & \text{Beverton and Holt} \end{cases} \quad (\text{A.3})$$

Substituting Equation A.3 into Equation A.1 gives

$$R = \begin{cases} aS\alpha^{-S/S_0} & \text{Ricker} \\ \frac{aS_0}{1 + (\alpha - 1)S/S_0} & \text{Beverton and Holt} \end{cases} \quad (\text{A.4})$$

and, since  $a = \alpha R_0 / S_0$ ,

$$R = \begin{cases} R_0 \frac{S}{S_0} \alpha^{1-S/S_0} & \text{Ricker} \\ R_0 \frac{\alpha S / S_0}{1 + (\alpha - 1) S / S_0} & \text{Beverton and Holt} \end{cases} \quad (\text{A.5})$$

Dividing through by  $R_0$  and defining  $s$  as  $S/S_0$  gives Equation 4.

**Appendix 2: formula for equilibrium spawning biomass**

The spawning potential ratio ( $p$ ) is defined as the number of spawners produced by each recruit at equilibrium with a given fishing mortality rate  $F$  divided by the number

of spawners per recruit under virgin conditions ( $F=0$ ). This may be written

$$p = \frac{\psi_F}{\psi_0} = \frac{\tilde{S}_F / \tilde{R}_F}{\tilde{S}_0 / \tilde{R}_0} = \frac{\tilde{S}_F / S_0}{\tilde{R}_F / \tilde{R}_0} = \tilde{s} / \tilde{r}, \quad (\text{A.6})$$

where the tilde signifies equilibrium values.

At equilibrium we also obtain from Equation 4

$$\tilde{r} = \begin{cases} \tilde{s} \alpha^{1-\tilde{s}} & \text{Ricker} \\ \frac{\alpha \tilde{s}}{(1 + \tilde{s}(\alpha - 1))} & \text{Beverton and Holt} \end{cases} \quad (\text{A.7})$$

Dividing both sides of Equation A.7 by  $\tilde{r}$ , substituting  $p$  for Equation A.6, and solving for  $\tilde{s}$  gives Equation 10.