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[Article]

# Stock Assessment of Scalloped Hammerheads in the Western North Atlantic Ocean and Gulf of Mexico

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*Abstract*.—The status of the western North Atlantic Ocean population of scalloped hammerheads *Sphyrna lewini* (Sphyrnidae [hammerhead sharks]) was assessed from 1981 through 2005 by using Schaefer (logistic) and Fox surplus-production models. The population declined rapidly before 1996 but began rebuilding in the late 1990s as fishing pressure decreased. The Akaike information criterion for small sample sizes—a test of goodness of fit for statistical models—indicated that the Fox model provided a slightly better fit to the data. Bootstrapped parameter values showed that in 2005 the probability of the scalloped hammerhead's being overfished was greater than 95% (the population was estimated to be 45% of that which would produce the maximum sustainable yield [MSY]) and a 73% probability that overfishing was occurring (fishing mortality was approximately 129% of that associated with the MSY). The size of this population was estimated to be 17% of what it had been in 1981, that is, it has been depleted by about 83% from the virgin stock size. Monte Carlo simulation predicted that the population had a 58% probability of rebuilding in 10 years if the 2005 catch level (4,135 individuals) were maintained and an 85% probability of rebuilding if the 2005 total catch were halved. Sensitivity analyses showed that the stock assessment results were most sensitive to removing the University of North Carolina longline survey index of relative abundance, the method of weighting indices, and excluding fishery-dependent indices of relative abundance.

Scalloped hammerheads *Sphyrna lewini* (Sphyrnidae [hammerhead sharks]) are globally distributed and occur in coastal and adjacent pelagic waters (Compagno 1984). Scalloped hammerhead fins are highly valued in the Asian shark fin trade for shark fin soup (IUCN 2006). Like many shark species, scalloped hammerheads have a high potential for overexploitation; they are characterized by late age at maturity, relatively small reproductive output, and long lifespan (Piercy et al. 2007). Estimates vary widely by location, but males are sexually mature at lengths of 1.5–2.3 m and females mature at 2–2.5 m, which corresponds to about age 15 (Compagno 1984; Branstetter 1987; Chen et al. 1990; Cortés 2000; Piercy et al. 2007). In the western North Atlantic Ocean and Gulf of Mexico, Piercy et al. (2007) estimated maximum ages of 30.5 years for both sexes. After a 9–10-month gestation period, scalloped hammerheads give birth to 10–40 live pups every other year (Branstetter 1987; Liu and Chen 1999). Unlike most other sharks, scalloped hammerheads exhibit schooling behavior, which makes them vulnerable to being caught in large numbers.

In the USA, large coastal sharks (Carcharhinidae, Sphyrnidae, and Ginglymostomatidae) have been managed as an aggregate group comprising 11 species. This approach to management is potentially risky because the decline in abundance of one species can be masked by the increase of more or equally productive species. In 2006, the National Marine Fisheries Service (NMFS) completed the eleventh Southeastern Data, Assessment and Review (SEDAR 11), in which the 11-species aggregate of large coastal sharks (including scalloped hammerheads) was determined not to be overfished; the estimated population size was larger than the size needed to produce maximum sustainable yield (NMFS 2006a). For all species but sandbar sharks *Carcharhinus plumbeus* and blacktip sharks *C. limbatus*, data were too limited to conduct a species-specific assessment. However, the review panel recommended that the NMFS conduct species-specific assessments of all large coastal sharks as data permit (NMFS 2006a).

To date, no comprehensive assessment of stock status for scalloped hammerheads has been made.

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TABLE 1.—Summary of relative abundance indices (standardized observations or samples taken over time to estimate the number or biomass of fish) and associated model scenarios. Geographic coverage abbreviations are as follows: SA = South Atlantic Ocean, GOM = Gulf of Mexico, and NA = North Atlantic Ocean; FD indicates a fishery-dependent survey and FI a fishery-independent survey. Positive hauls is the proportion of hauls that included at least one scalloped hammerhead. The first six indices were included in the BASE scenario.

| Index | | Geographic coverage | Fishery dependence | Years | Positive hauls (%) | Reference |
|---|---|---|---|---|---|---|
| Code | Name | | | | | |
| CSFOP | Commercial shark fishery observer program | SA and GOM | FD | 1994–2005 | 21 | Cortés et al. (2005) |
| GNOP | Shark drift gill net observer program | SA (Georgia, Florida) and GOM (Florida) | FD | 1994–2005 | 39 | Carlson et al. (2005) |
| PLLOP | Pelagic longline observer program | Western NA | FD | 1992–2005 | 9 | Beerkircher et al. (2002) |
| NMFS LL SE | NMFS Mississippi bottom longline survey | SA and GOM | FI | 1995–2005 | 9 | Ingram et al. (2005) |
| PCGN | NMFS Panama City gill-net survey | Northeastern GOM | FI | 1996–2005 | 23 | Carlson and Bethea (2005) |
| NCLL | University of North Carolina longline survey | Onslow Bay, North Carolina | FI | 1972–2005 | 6 | Schwartz et al. (2007) |
| GACP | Georgia Coastspan | Georgia estuaries | FI | 2000–2005 | | McCandless and Belcher (2007) |
| NMFS LL NE | NMFS Narragansett longline survey | SA (Florida to Delaware) | FI | 1996–2005 | | Natanson (2005) |

Baum et al. (2003) estimated an 89% decline in stocks of scalloped hammerheads in the western North Atlantic Ocean. Conclusions derived from Baum et al. (2003) were contentious because the authors limited the scope of the assessment to a single relative abundance index (the pelagic longline logbooks), ignored data sets that would have produced different conclusions, and disregarded factors that possibly biased results (Baum et al. 2003; Burgess et al. 2005). Largely on the basis of the Baum et al. (2003) paper, the World Conservation Union (IUCN) recently changed the global status of scalloped hammerheads from "Near Threatened" to "Endangered" (IUCN 2002, 2006). The present study is the first species-specific assessment of scalloped hammerheads in the western North Atlantic Ocean and Gulf of Mexico, and it synthesizes all available data.

## Methods

Surplus-production or age-aggregated models are commonly used when only total catch and relative abundance data (catch-per-unit-effort [CPUE] data) are available, as is the case with scalloped hammerheads. We investigated the goodness of fit of surplus-production models with two productivity curves: Schaefer (1954), or logistic, and Fox (1970) using the Akaike information criterion for small sample sizes ($AIC_c$; Akaike 1974; Bedrick and Tsai 1994). Surplus-production models may outperform more complex models by estimating fewer parameters, thus minimizing uncertainty associated with each parameter (Ludwig and Walters 1985; Prager 1994). Multiple scenarios were constructed to test the influence of relative abundance indices (Table 1), the weighting scheme for those indices, and initial population size.

*Catch data*.—Annual catch data from the western North Atlantic Ocean and Gulf of Mexico (Table 2) were recorded by the NMFS, starting in 1981. Although some catches probably were taken before 1981, the data are insufficient to estimate those catch values. Initial model runs assumed that no catch took place to 1981 (i.e., the population starts at carrying capacity); however, this assumption was tested through sensitivity analyses, described below.

Recreational catches dominated the early fishery (Figure 1), largely in response to the release of the movie "Jaws." Recreational catch data (Cortés and Neer 2005; NMFS 2006a) were collected through three surveys: the NMFS Marine Recreational Fishery Statistics Survey, the NMFS Headboat Survey, and the Texas Parks and Wildlife Department Marine Sport-Harvest Monitoring Program. These data were available only in numbers, no reliable average weight information being available. With no way to estimate recreational catch in biomass, this assessment was conducted in numbers.

Commercial landing data (Cortés and Neer 2005; NMFS 2006a) on weight were collected by the NMFS Southeast Fisheries Science Center (SEFSC) from the Pelagic Dealer Compliance program and by the SEFSC and Northeast Regional Office from the Accumulated Landings System, in which dealers report directly to the individual states. The annual catch was converted into numbers by dividing the weight by an average weight of individual animals measured in the Commercial Shark Fishery Observer Program (Cortés et al. 2005).

TABLE 2.—Number of scalloped hammerheads caught by year and fishery sector. Estimated discards are given in parentheses; they were included in the BASE scenario but excluded from scenario NoDC. Asterisks indicate relatively high reported catch values that were estimated by averaging adjacent years in the CATCH scenario.

| Year | Recreational | Commercial | Discards | Total |
|---|---|---|---|---|
| 1981 | 5,880 | 0 | (1,487) | 7,367 |
| 1982 | 48,138 | 1 | (1,487) | 49,626* |
| 1983 | 20,962 | 365 | (1,487) | 22,814 |
| 1984 | 7,003 | 0 | (1,487) | 8,490 |
| 1985 | 44,042 | 0 | (1,487) | 45,529* |
| 1986 | 5,321 | 0 | (1,487) | 6,808 |
| 1987 | 6,372 | 0 | 1,228 | 7,600 |
| 1988 | 4,518 | 2 | 1,674 | 6,194 |
| 1989 | 6,191 | 0 | 1,389 | 7,580 |
| 1990 | 18,373 | 12 | 1,151 | 19,536 |
| 1991 | 8,935 | 4 | 1,221 | 10,160 |
| 1992 | 7,325 | 67 | 2,257 | 9,649 |
| 1993 | 21,723 | 91 | 516 | 22,330 |
| 1994 | 3,886 | 301 | 368 | 4,554 |
| 1995 | 3,695 | 1,479 | 567 | 5,741 |
| 1996 | 882 | 1,479 | 290 | 2,652 |
| 1997 | 3,905 | 1,041 | 938 | 5,884 |
| 1998 | 1,083 | 642 | 234 | 1,959 |
| 1999 | 545 | 386 | 344 | 1,275 |
| 2000 | 6,350 | 68 | 277 | 6,695 |
| 2001 | 1,112 | 1,152 | 339 | 2,602 |
| 2002 | 6,113 | 1,180 | (431) | 7,724 |
| 2003 | 2,859 | 2,606 | (431) | 5,896 |
| 2004 | 803 | 1,351 | (431) | 2,585 |
| 2005 | 803 | 2,901 | (431) | 4,135 |

Dead discard data (Beerkircher et al. 2002; Cortés et al. 2005) were obtained from the SEFSC, which uses the pelagic longline observer program (PLLOP) and dealer weigh-out data to produce annual estimates. Because discard estimates for scalloped hammerheads were not available before 1987 and scalloped hammerheads no longer appeared as a distinct category after 2001 (being lumped into a larger category in dealer reports), estimates for 1982–1986 and 2002–2005 were based on the average discards in 1987–1992 and 1993–2001, respectively (NMFS 2006a). The years used to estimate discards were split on the basis of regulatory actions (e.g., commercial quotas, recreational bag limits) implemented in 1993 (NMFS 2006a).

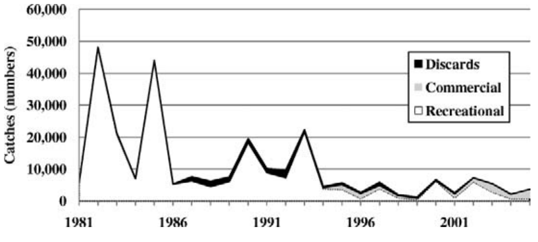


FIGURE 1.—Commercial and recreational catches and discards of scalloped hammerheads, 1982–2005. No discard estimates were available for 1981–1986 or 2002–2005 because of changes in dealer reporting.

*Indices of relative abundance*.—Fishery-dependent indices utilize catch and effort data provided by the commercial fishery through logbooks and observer programs (Table 1). This study followed the NMFS (2006) recommendations to use observer data (as opposed to logbook data) in the assessment when available. Fishery-dependent relative abundance indices (Tables 1, 3) include the commercial shark fishery observer program (Cortés et al. 2005), the gill net observer program (GNOP; Carlson et al. 2005), and the pelagic longline observer program (PLLOP; Beerkircher et al. 2002). A two-part standardization approach (Cortés et al. 2007) derived from Lo et al. (1992) was applied to each index before the assessment was made. This standardization technique is especially useful for handling the large number of zeros in the data (NMFS 2006a); therefore, it is often used to standardize shark abundance indices, including those presented here (Cortés et al. 2007). Relative abundance indices were standardized by the corresponding lead authors, except PLLOP (standardized by E. Cortés, NMFS). All available fishery-dependent abundance indices were included in the BASE model.

Fishery-independent surveys (Tables 1, 3) are often considered less biased indices of abundance than fishery-dependent data because samples are taken from randomly selected stations and, in contrast to fishing vessels, they do not target concentrated areas of fish.

TABLE 3.—Indices of the relative abundance of scalloped hammerheads (see Table 1), standardized by the Lo method (Lo et al. 1992) and normalized to their own means. Blanks indicate that no data were available.

| Year | CSFOP | GNOP | PLLOP | NMFS LL SE | PCGN | NCLL | GACP | NMFS LL NE |
|---|---|---|---|---|---|---|---|---|
| 1981 | | | | | | 1.329 | | |
| 1982 | | | | | | 0.816 | | |
| 1983 | | | | | | 1.174 | | |
| 1984 | | | | | | 1.438 | | |
| 1985 | | | | | | 0.344 | | |
| 1986 | | | | | | 0.719 | | |
| 1987 | | | | | | 0.886 | | |
| 1988 | | | | | | 1.223 | | |
| 1989 | | | | | | 0.154 | | |
| 1990 | | | | | | 0.049 | | |
| 1991 | | | | | | 0.076 | | |
| 1992 | | | 2.736 | | | | | |
| 1993 | | | 1.378 | | | 0.216 | | |
| 1994 | 0.183 | 0.979 | 0.745 | | | 0.102 | | |
| 1995 | 0.344 | 4.218 | 1.162 | 1.055 | | | | |
| 1996 | 0.362 | | 0.285 | 0.404 | 0.127 | 0.206 | | 0.060 |
| 1997 | 0.429 | | 1.091 | 0.567 | 0.541 | | | |
| 1998 | 0.601 | 1.129 | 1.201 | | 0.265 | 0.112 | | 0.386 |
| 1999 | 0.161 | 0.158 | 0.449 | 0.947 | 0.742 | 0.987 | | |
| 2000 | 0.012 | 1.151 | 0.613 | 1.322 | 1.000 | 0.277 | 0.358 | |
| 2001 | 0.421 | 0.365 | 0.886 | 1.244 | 0.912 | 0.141 | 0.686 | 1.954 |
| 2002 | 0.825 | 0.274 | 0.454 | 1.347 | 0.819 | 0.147 | 2.381 | |
| 2003 | 1.000 | 0.240 | 0.708 | 1.530 | 0.596 | 0.187 | 0.456 | |
| 2004 | 0.773 | 0.755 | 0.458 | 0.584 | 0.436 | 0.216 | 1.265 | 1.599 |
| 2005 | 0.452 | 0.730 | 0.507 | | 0.459 | 0.392 | 0.855 | |

The more statistically rigorous methods of fishery-independent surveys are assumed to more accurately reflect population abundance (NMFS 2006a).

Fishery-independent surveys included in the BASE model of the assessment were NMFS longline southeast (NMFS LL SE; Ingram et al. 2005), Panama City gill-net survey (PCGN; Carlson and Bethea 2005), and North Carolina longline survey (NCLL; Schwartz et al. 2007). Georgia Coastspan (GACP; McCandless and Belcher 2007), and NMFS longline northeast (Natanson 2005; NMFS LL NE) surveys had few data points (Table 3); they were excluded from the BASE scenario but included in the sensitivity run, ALL. All fishery-independent surveys were standardized by corresponding lead authors, except NMFS LL NE and NCLL, which were standardized by C. McCandless of NMFS.

*Assessment models*.—Surplus-production models have been used in many shark stock assessments, including the NMFS assessments and International Commission for the Conservation of Atlantic Tunas assessments (Babcock and Pikitch 2001; Cortés 2002; Cortés et al. 2002; Babcock and Cortés 2005; NMFS 2006a, 2007). These models are useful in cases such as scalloped hammerheads, for which only catch and relative abundance data are available (Prager 1994). Prager and Goodyear (2001) found production models to be robust to mixed-metric data, as was the case for scalloped hammerheads, where catch was in numbers and some of the indices were in biomass. Simpler production models can sometimes outperform more intricate age-structured models (Ludwig and Walters 1985; Ludwig et al. 1988).

This study analyzed two forms of the surplus-production model: logistic (Schaefer 1954) and Fox (1970). Both variants assume that the maximum sustainable yield (MSY) or maximum surplus production occurs at some population size below carrying capacity. Surplus production increases as individuals are removed from the population to a point (population size associated with maximum sustainable yield, $N_{\mathrm{MSY}}$) below which surplus production begins decreasing. The logistic model assumes $N_{\mathrm{MSY}}$ is half of the unfished population size ($K$), whereas the Fox model assumes $N_{\mathrm{MSY}}$ occurs at $K/e$, or approximately 37% of $K$. Model goodness of fit was compared through AIC corrected for small sample size ($\mathrm{AIC}_c$), which provides an unbiased order of model choice and is recommended for use regardless of sample size (Bedrick and Tsai 1994; Burnham and Anderson 2004).

The basic surplus-production model used for this study was

$$N_{t+1} = N_t + G_t - C_t, \tag{1}$$

where $N_t$ is the population size at time $t$; $G_t$ is the population growth or surplus production; and $C_t$ is the catch at time $t$.

Fishing mortality ($F_t$) was estimated by

$$F_t = \frac{C_t}{N_t}. \quad (2)$$

For this study, we compared model performance of two production curves by using $AIC_c$ as follows:

$$\text{Logistic}: G_t = rN_t[1 - (N_t/K)], \quad (3)$$

$$\text{Fox}: G_t = rN_t\{1 - [\log_e(N_t)/\log_e(K)]\}, \quad (4)$$

where $r$ is the intrinsic population growth rate and $K$ is the unfished (virgin) population size.

Initial population size, $N_0$, was set equal to $K$ or a proportion of $K$ (Punt 1990). The parameters $r$ and $K$ were estimated by applying the observation error estimator (Polacheck et al. 1993). Assuming a lognormal error structure,

$$I_{i,t} = q_i N_t e^{\varepsilon_{i,t}}, \quad (5)$$

where $I_{i,t}$ is the abundance index $i$ at time $t$; $q_i$ is the parameter that scales the population abundance to that of the index $i$, also termed the catchability coefficient; and $\varepsilon_{i,t}$ is normally distributed ($N[0, \sigma^2]$) observation error associated with index $i$ at time $t$.

Following the recommendations of NMFS (2006), equal weight was used for all scenarios except one (inverse variance weighting; INCV) and the objective function minimized (Prager 1994; MATLAB vers. 7.5.0.342) was

$$\sum_i \lambda_i \sum_t [\log_e(I_{i,t}) - \log_e(\hat{I}_{i,t})]^2, \quad (6)$$

where $\lambda_i$ is the weight of index $i$.

Punt (1990) found that setting the initial population size equal to $K$ outperforms models where it is estimated separately. For comparison, this study also looked at how some initial depletion (i.e., $N_0 = 0.7K$) would affect the results. Because $q_i$ are nuisance parameters, a generalized linear model approach (Jiao and Chen 2004) was used to estimate the catchability coefficient $q$. The algorithm for this approach has two stages when searching for maximum likelihood estimates (MLE) over the parameters $r$, $K$, and $N_0$ (Jiao and Chen 2004). In the first stage, population abundance is projected on the basis of the population dynamic equation (1), the productivity equations (3) and (4) and the gridded parameters $r$, $K$, and $N_0$. In the second stage, the catchabilities are estimated by application of a generalized linear model to fit the observed abundance and the projected population abundance.

*Sensitivity analysis*.—Sensitivity analyses are often conducted to determine how the model is driven by certain data sets or even data points. In this study, model sensitivity to the removal of abundance indices, discard estimates, and anomalous catch data points was tested (Table 4). Scenario BASE included available abundance indices recommended by NMFS (2006; Table 1). Scenario ALL included all available indices; INDY included only fishery-independent abundance indices; BASE – NCLL included BASE indices, except NCLL; INCV tested the sensitivity of the model to inverse variance weighting of the abundance indices; IDEP explored how the results vary when a 30% initial depletion is assumed ($N_{1981} = 0.7K$). In scenario CATCH, two years (1982 and 1985) of catch data that were twice the magnitude of any other year were estimated by averaging the reported catch of the year before and after (Table 2). Scenario NoDC excluded discard estimates for the years with missing data (Table 2).

*Parameter estimation from bootstraps*.—Median values and confidence intervals of estimated parameters were produced through the nonparametric bootstrap method (Hilborn and Walters 1992). Lognormal residuals were randomly sampled with replacement and added to the fitted log abundance indices to produce a new log abundance index. This newly generated index was treated as a new independent sample and applied to the model to generate new parameter estimates. We ran the simulation for 5,000 iterations, producing probability distributions for each parameter and management reference points (Haddon 2001).

*Population projections in Monte Carlo simulation*.—The effect of various fishing regimes on population rebuilding was tested by using the probability distributions produced through the bootstrap approach. In assessing the potential for rebuilding at various fishing levels, parameter values were randomly selected from the probability distributions and projected 10, 20, and 30 years into the future in a process known as Monte Carlo simulations. These 5,000 simulated populations were subjected to 0, 50, 69, 100, and 150% of 2005 catch levels (4,135 individuals).

## Results

### Model Selection

The Fox model slightly outperformed the logistic model ($AIC_c$ = 172.6 and 173.6, respectively). The parameters estimated in the Fox model implied a less productive population than did those in the Schaefer model (Figure 2), so the Fox model was used for population projections. Although $N_{MSY}$ occurs at a smaller proportion of $K$ in the Fox model, the intrinsic

TABLE 4.—Results of logistic and Fox surplus-production models under eight scenarios. BASE includes the six relative abundance indices noted in Table 1; ALL includes all available abundance indices; INDY includes only fishery-independent surveys; INCV uses inverse variance weighting; IDEP sets the 1981 population equal to 0.7 times the unfished population; BASE–NCLL includes all BASE indices except NCLL; CATCH estimates the catches in 1982 and 1985 by averaging the years before and after; and NoDC excludes unreported discard estimates. Other abbreviations are as follows: $AIC_c$ = the Akaike information criterion corrected for small sample size; $r$ = the intrinsic annual population growth rate; $K$ = the size of the unfished population (thousands); MSY = the maximum sustainable yield (thousands); $F$ = the annual fishing mortality rate; and $N$ = the size of the actual population (thousands).

| Variable or statistic | BASE | ALL | INDY | INCV | IDEP | BASE–NCLL | CATCH | NoDC |
|---|---|---|---|---|---|---|---|---|
| | | | | **Logistic results** | | | | |
| $AIC_c$ | 173.6 | 220.1 | 130.6 | 256.1 | 173.9 | 112.5 | 174.7 | 173.9 |
| $r$ | 0.29 | 0.36 | 0.39 | 0.55 | 0.31 | 0.72 | 0.37 | 0.30 |
| $K$ | 142 | 129 | 125 | 106 | 159 | 93 | 83 | 134 |
| MSY | 10.4 | 11.6 | 12.1 | 14.5 | 12.5 | 16.8 | 7.6 | 10.0 |
| $F_{MSY}$ | 0.15 | 0.18 | 0.19 | 0.27 | 0.16 | 0.36 | 0.18 | 0.15 |
| $N_{MSY}$ | 71 | 65 | 62 | 53 | 80 | 47 | 42 | 67 |
| Depletion (%) | 83 | 82 | 78 | 91 | 86 | 91 | 79 | 82 |
| $F_{2005}/F_{MSY}$ (%) | 114 | 99 | 78 | 157 | 122 | 140 | 128 | 102 |
| $N_{2005}/N_{MSY}$ (%) | 35 | 36 | 44 | 18 | 27 | 18 | 43 | 36 |
| | | | | **Fox results** | | | | |
| $AIC_c$ | 172.6 | 219.5 | 129.3 | 248.7 | 129.4 | 110.6 | 172.7 | 172.9 |
| $r$ | 0.11 | 0.13 | 0.15 | 0.17 | 0.14 | 0.20 | 0.16 | 0.12 |
| $K$ | 169 | 162 | 150 | 142 | 182 | 133 | 104 | 162 |
| MSY | 7.1 | 7.6 | 8.5 | 9.0 | 9.7 | 10.0 | 6.2 | 7.0 |
| $F_{MSY}$ | 0.11 | 0.13 | 0.15 | 0.17 | 0.14 | 0.20 | 0.16 | 0.12 |
| $N_{MSY}$ | 62 | 60 | 55 | 52 | 67 | 49 | 38 | 59 |
| Depletion (%) | 83 | 84 | 80 | 92 | 85 | 93 | 81 | 83 |
| $F_{2005}/F_{MSY}$ (%) | 129 | 125 | 89 | 210 | 103 | 220 | 130 | 114 |
| $N_{2005}/N_{MSY}$ (%) | 45 | 44 | 55 | 22 | 41 | 19 | 52 | 47 |

growth rate—and corresponding resilience to fishing pressure—was estimated to be a smaller value in the Fox model ($r$ = 0.11) than in the Schaefer model ($r$ = 0.29).

### *Model Fit*

Interannual variability in the indices of relative abundance was high in some cases, such that the model had trouble fitting some trends very well (Figure 3).

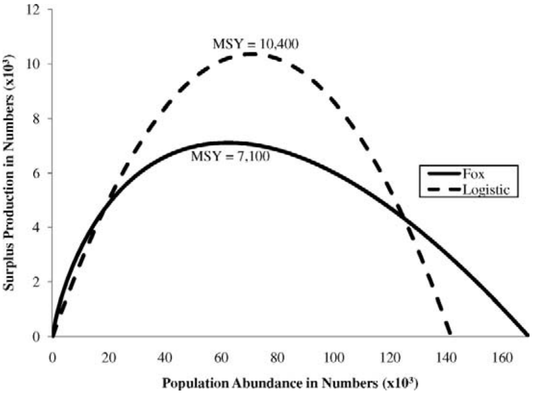


FIGURE 2.—Surplus-production curves obtained by fitting the logistic and Fox models in the BASE scenario.

Both models best fit the NMFS LL SE index according to visual inspection of model fit and residuals. The NMFS LL SE has the greatest geographical distribution and should more closely reflect population abundance than smaller surveys can. The PCGN and GNOP are examples of spatially limited surveys that resulted in poor fits. The model was also very sensitive to the NCLL survey (longest time series).

### *Population Status*

Though the nominal catch was highest in the early 1980s, the fishing mortality rate peaked in the early 1990s (Figure 4). By the late 1990s, fishing pressure was reduced and population decline slowed (Figure 5). Although we did see some evidence that a recovery of the population may have begun, scalloped hammerheads were currently (for 2005) overfished—that is, current stock size was below the population size that produces MSY—in all combinations of inputs and models we investigated. Overfishing, or rate of fishing greater than that associated with MSY, most probably occurred in 2005; however, some scenarios indicated that fishing levels were below $F_{MSY}$ in 2005 (Figure 6).

When the logistic model was applied to the BASE scenario, the population was both overfished and experiencing overfishing (Table 5; Figure 6). The estimated population size in 2005 was 35% (95%

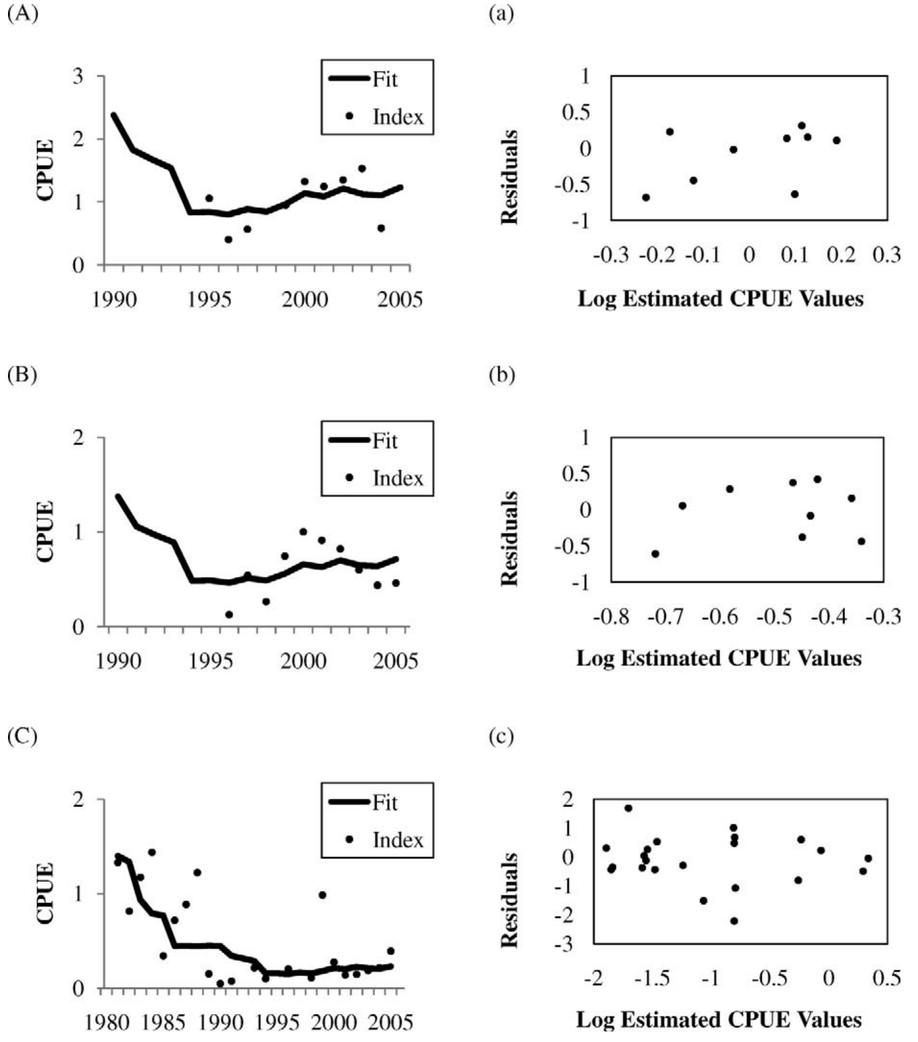


FIGURE 3.—Fox model fits for **(A)** the NMFS longline southeast (NMFS LL SE), **(B)** Panama City gill-net (PCGN), **(C)** North Carolina longline (NCLL), **(D)** commercial shark fishery observer program (CSFOP), **(E)** gill net observer program (GNOP), and **(F)** pelagic longline observer program (PLLOP) relative abundance indices (see Table 1) under the BASE scenario. Panels **(a)–(f)** show the corresponding residuals of the abundance indices.

confidence interval [CI] = 19–87%) of $N_{MSY}$, the estimated fishing mortality was 114% (95% CI, 43–397%) of $F_{MSY}$, and estimated depletion relative to 1981 ($N_{current}/N_{1981}$) was 83% (95% CI, 53–90%). The Fox model led to very similar conclusions. In the BASE scenario, scalloped hammerheads were likely overfished and subject to overfishing (Figure 6). In 2005, the estimated population size was 45% (95% CI, 18–89%) of $N_{MSY}$, fishing mortality was estimated to be 129% (95% CI, 54–341%) of $F_{MSY}$, and depletion was very similar to that of the logistic model: 83% (95% CI, 67–93%). Given the uncertainty associated with the data, confidence intervals are wide, particularly when one is trying to determine whether overfishing is occurring ($F > F_{MSY}$). However, the stock is probably overfished (i.e., > 95% probability that $N < N_{MSY}$).

*Sensitivity Analyses*

Model sensitivity to the NMFS LL NE and GACP relative abundance series in scenario ALL was minimal (Figure 6). The model was, however, sensitive to the removal of fishery-dependent abundance indices in scenario INDY. When only fishery-independent abun-

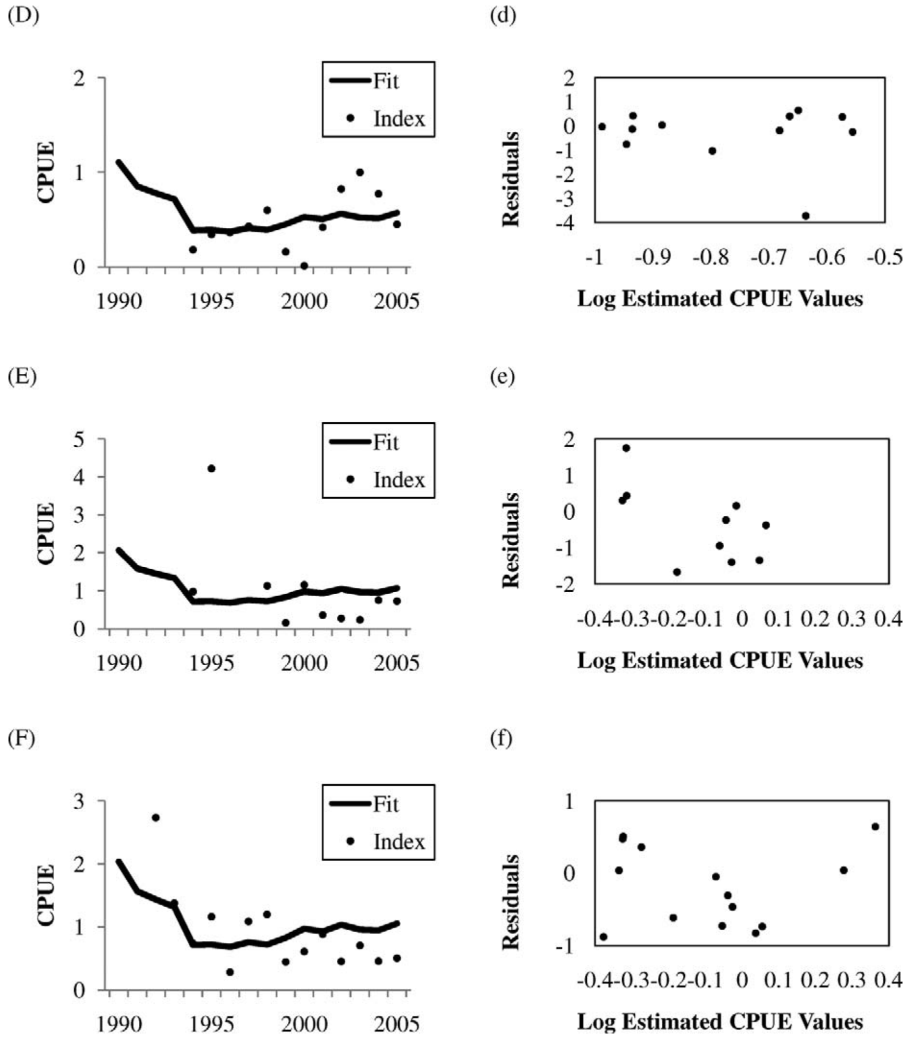


FIGURE 3.—Continued.

dance indices were used, overfishing no longer occurred. The population was still overfished, though to a lesser degree. The model was sensitive to the removal of NCLL, and results were more pessimistic when NCLL was excluded from the BASE scenario.

When the BASE abundance indices were weighted by the inverse of the variance in scenario INCV, population status became more pessimistic (Figure 6). The same was true if 30% initial depletion (relative to 1981) was included (scenario IDEP), though this had little effect on results. Scenario CATCH, which tested the sensitivity to catch in years 1982 and 1985, showed little change in results when catch data of those years were included. Similarly, missing discard estimates—scenario NoDC—had little effect on the model.

*Population Projections and Alternative Catch Level Evaluation*

Managers could set constant catch levels based on target probability of recovery. In 95% of the simulated populations with no fishing, $N_{MSY}$ was reached within 10 years (Table 6). When a constant catch of 2,068 fish (half of the 2005 Catch; 50% $C_{2005}$) was projected, 85% of the simulated populations reached $N_{MSY}$ within 10 years. To achieve the acceptable level of risk (>70% probability of $N > N_{MSY}$ within 10 years; NMFS 2006b) would involve removing 69% of $C_{2005}$, or 2,853 fish, annually. If 100% $C_{2005}$, or 4,135 fish, were removed annually for 10 years, the simulation study predicted a 58% probability of recovery (reaching $N_{MSY}$). Only 20% of simulated populations

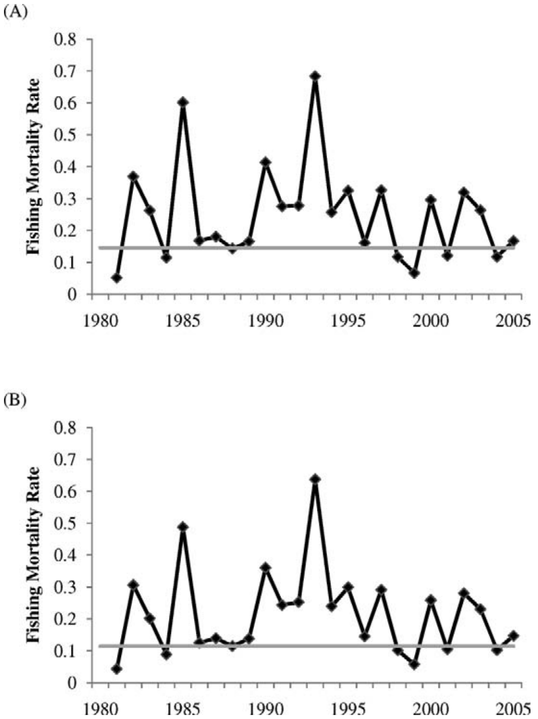


FIGURE 4.—Estimated fishing mortality rates of scalloped hammerheads derived from the **(A)** logistic and **(B)** Fox models for the period 1981–2005. The gray horizontal lines represent the fishing mortalities associated with the maximum sustainable yields from the two models.

subjected to a constant catch of 6,202 fish (150% $C_{2005}$) recovered in 10 years. However, a longer time horizon increased the probability of recovery (Table 6).

## Discussion

Surplus-production models are being used less frequently in stock assessments in favor of age-

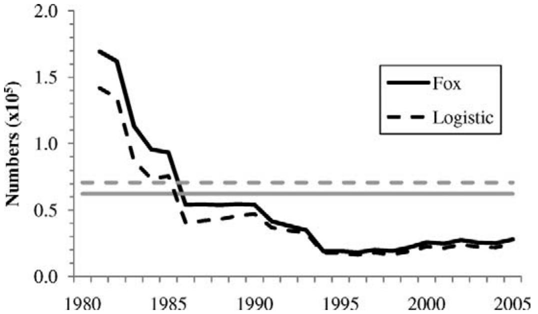


FIGURE 5.—BASE scenario abundance estimates derived from the logistic and Fox models for the period 1981–2005. The gray horizontal lines represent the populations associated with the maximum sustainable yields from the two models.

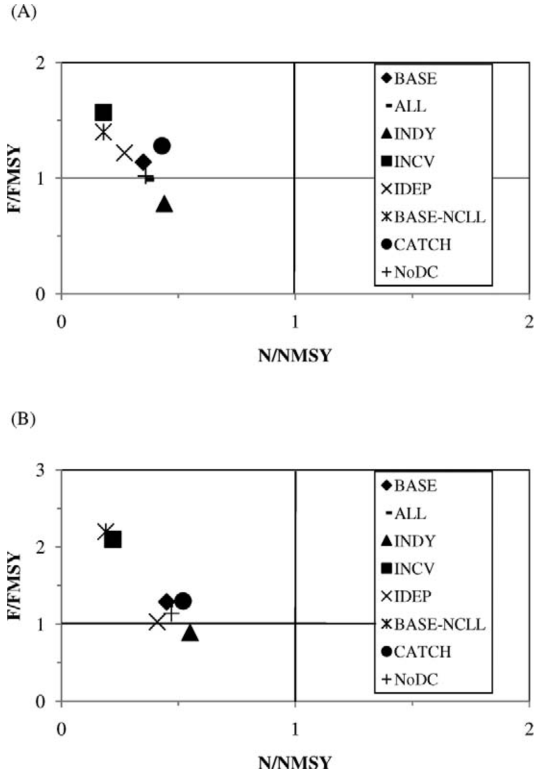


FIGURE 6.—Phase plots of the population size in 2005 relative to that associated with the maximum sustainable yield ($N_{MSY}$) and the fishing mortality rate in 2005 relative to that associated with the maximum sustainable yield ($F_{MSY}$) as derived from the **(A)** logistic and **(B)** Fox models. The BASE Scenario included the CSFOP, GNOP, PLLOP, NMFS LL SE, PCGN, and NCLL indices (see Figure 3); ALL included all available indices; INDY included the fishery-independent indices; INCV weighted the BASE scenario indices by the inverses of their variances; IDEP included 30% initial depletion; BASE – NCLL was the BASE scenario with NCLL removed; CATCH included the 1982 and 1985 catch estimates (averaged years before and after); and NoDC excluded discard estimates for missing data.

TABLE 5.—Biological reference points derived from the logistic and Fox models with the BASE scenario. See Table 4 for additional details.

| Variable | Logistic | Fox |
|---|---|---|
| $r$ | 0.29 (0.05–0.45) | 0.11 (0.06–0.23) |
| $K$ | 142 (116–260) | 169 (126–218) |
| MSY | 10.4 (4–13) | 7.1 (5–10) |
| $F_{MSY}$ | 0.15 (0.03–0.23) | 0.11 (0.06–0.23) |
| $N_{MSY}$ | 71 (58–130) | 62 (47–80) |
| Depletion (%) | 83 (53–90) | 83 (67–93) |
| $F_{2005}/F_{MSY}$ (%) | 114 (43–397) | 129 (54–341) |
| $N_{2005}/N_{MSY}$ (%) | 35 (19–87) | 45 (18–89) |

TABLE 6.—Probability (%) that the stock of scalloped hammerheads will rebuild (i.e., attain a final population size greater than $N_{MSY}$) in 10, 20, and 30 years under several constant-catch scenarios (relative to the catch in 2005) using the BASE scenario with the Fox surplus-production model.

| Time frame | No catch | Percent of 2005 catch (number) 50 (2,068) | 69 (2,853) | 100 (4,135) | 150 (6,203) |
|---|---|---|---|---|---|
| 10 years | 95 | 85 | 70 | 58 | 20 |
| 20 years | 99 | 96 | 92 | 86 | 50 |
| 30 years | 99 | 98 | 96 | 91 | 63 |

structured models, which are more biologically realistic (Simpfendorfer et al. 2000). The balance between biological reality and parsimony is indeed a part of model selection for any stock assessment. For this assessment, data availability was the driving factor in selecting the surplus-production model. Though age-structured data are not currently available for an age-structured model of scalloped hammerheads, it would be important in the future to investigate how the incorporation of age structure affects status estimates.

Given the similar performance of the logistic and Fox models and the similar estimates of status, the precautionary approach (Garcia 1994; Richards and Maguire 1998) may be a factor in deciding which model to use for management purposes. The precautionary approach is a management strategy that is applied to reduce risk when scientific information is incomplete (Garcia 1994). In the case of scalloped hammerheads, the Fox model produced the lower estimates of MSY and $F_{MSY}$ as a result of estimating a smaller population growth rate. If management objectives are to rebuild the stock quickly, the Fox model should be used because it estimated a slower rate of increase and would be expected to be more risk-averse. However, the performance (AICc), stock status, and the implications for future management recommendations are similar for the two models.

By utilizing a surplus-production model, this study has implicit assumptions that should be addressed as more data become available in the future. First, this model does not distinguish between immature recruits to the fishery and mature adults. The annual variation in proportions of these two groups will have an effect on the overall population growth rate; a declining proportion of mature adults could lead to stock collapse, particularly in this viviparous species. Second, the indices of abundance are assumed to be proportional to population size, a relationship assumed to be constant over time. However, fishing practices probably will have changed over time as a result of the acquisition of better equipment, which could have increased catchability. This would probably mask declines in the population, because CPUE would be kept artificially high in fishery-dependent indices. Third, catch data are assumed to be known perfectly. Catch levels drive the magnitude of population abundance estimates; that is, if all catch data are underestimates, the population is probably larger than the model would suggest. Finally, this model assumes an evenly distributed population. Indices with small geographical coverage are given representation equal to those that cover larger areas. Scalloped hammerheads, however, are most probably not evenly distributed, a result of life history constraints such as foraging and reproductive needs.

The decline in catch seems to have given this population the opportunity to begin rebuilding. The NMFS (2006) found that the 11-species complex declined from the 1970s through the mid-1990s. However, both the complex and the scalloped hammerhead population within it stabilized just after the 1994 fishery management plan (NMFS 2006a). Scalloped hammerheads, which are among the faster growing species in the complex, have a relatively high probability of recovering quickly. Despite its slow life history characteristics, this scalloped hammerhead population appears to have a 58% or greater probability of recovery within a decade if the 2005 catch is maintained or decreased. Note, however, that surplus-production models are often overly optimistic in estimating rebuilding times (NMFS 2006a). The results of the latest sandbar shark assessment (NMFS 2006a) may lead to reduced quotas for all large coastal sharks in the USA. If implemented, this reduction could potentially decrease the time necessary for the western North Atlantic Ocean population of scalloped hammerheads to reach $N_{MSY}$.

Species-specific assessments are important if fisheries managers aim to protect all species from stock collapse. The recent NMFS (2006) assessment estimated that the stock size of the large coastal shark aggregate (including scalloped hammerheads) in 2004 was 125% of $N_{MSY}$ and fishing mortality was 61% of $F_{MSY}$ (NMFS 2006a). This exemplifies the problem of performing assessments on stock complexes, wherein some highly productive species probably mask the decline of less productive species, such as scalloped hammerheads. The level of population depletion (relative to 1981) found in the present study (83%) is similar to that found by Baum et al. (2003), who estimated an 89% decline in the western North Atlantic Ocean population of scalloped hammerheads during 1986–2000, based on pelagic longline logbook data. Species-specific assessments, such as the one presented here, improve our understanding of a stock's status and provide a sounder basis for future management.

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