American Fisheries Society Symposium 27:45–62, 2002

Incorporating Uncertainty into Stock Assessments: A Case Study of Atlantic Striped Bass

GARY R. SHEPHERD
National Marine Fisheries Service, Northeast Fisheries Science Center, 166 Water Street Woods Hole, Massachusetts 02543, USA

Abstract.—A stock assessment of Atlantic striped bass *Morone saxatilis* was presented to illustrate potential sources of uncertainty in application of an age-based population model. Erroneous conclusions in stock assessment can result from incorrect model selection, input data that are not representative of the target population, and improper configuration of the selected model. Influence of incorrect input data and model configuration was investigated using striped bass catch-at-age data analyzed with a tuned virtual population analysis model (ADAPT VPA). Variations in model configurations were explored in addition to sensitivity to input parameters such as natural mortality. Violation of the assumption of constant natural mortality-at-age had a significant influence on the resulting estimates of F and stock size. Discard losses, particularly from the commercial fishery, were the largest source of uncertainty in the catch-at-age. Uncertainty due to process error in the VPA model was characterized by bootstrap realizations of the nonlinear least-squares estimates of fishing mortality. The implications associated with fishing at various Fs were also examined using a stochastic projection model. A comparison of fishing mortality estimates derived from two independent models, an age-structured population model and a tag-recovery model, indicated that both methods produced equivalent results. Evaluation of the striped bass stock assessment demonstrates that uncertainty could result from a variety of sources but this variability was only partially captured within the model framework. Understanding the possible sources of uncertainty and implications in interpreting model results should benefit the analyst in providing assessment advice to managers.

Introduction

The decline in marine fish stocks in the United States over the last decade has been well documented (Clark 1998; NMFS 1999). Despite increasing efforts to adopt more stringent management measures, relatively few examples exist where marine fish populations have recovered to predecline levels (Mace 1997). The Atlantic striped bass *Morone saxatilis* is one exception. Through aggressive management of commercial and recreational fisheries, improved habitat and good fortune, striped bass abundance has returned to levels comparable to the predecline period of the 1960s. By 1995, the Atlantic States Marine Fisheries Commission (ASMFC) Striped Bass Management Board declared, based on the recommendation of the ASMFC Striped Bass Technical Committee, that spawning stock biomass was at high enough levels that striped bass should be considered recovered. Restoration of striped bass, particularly the Chesapeake Bay stock, was an important accomplishment for fishery managers. Maintaining stock biomass at a self-sustaining level in the face of increasing fishing pressure on the resource may be even more challenging. This formidable task is dependent on adequate stock assessment information and an understanding by managers of uncertainty in those results as an element of decision making.

Coastal migratory Atlantic striped bass originate in the Chesapeake Bay and its tributaries, the Hudson River and the Delaware River. The stock from the Roanoke River/Albermarle Sound is believed to contribute insignificant numbers to the coastal migratory group at the present time. Despite many attempts to develop methods for stock delineation within the mixed coastal stock, stock identification techniques have not provided a high degree of correct categorization (Waldman and Fabrizio 1994; Waldman et al. 1997; Wirgin et al. 1997). Consequently, the fishery management plan and stock assessment considered the entire mixed stock migratory group as a single unit stock.

Striped bass have been subjected to management regulations since the time of the early settlers in New England. Concerns about overfishing along the New England coast were expressed as early as the 1700s (Bigelow and Schroeder 1953). More recently, declining striped bass abundance in the 1930s was the impetus for development of the ASMFC in 1942. The commission was organized to provide a compact among Atlantic coastal states to cooperatively manage coastal migratory stocks. The collapse of striped bass stocks in Chesapeake Bay during the late 1970s marked the beginning of state and federal efforts to restore the stocks to sustainable levels. These efforts culminated in the closure of striped bass fisheries in Maryland in 1985 and most other jurisdictions by 1989 (Richards and Deuel 1987). However, the fishery management plan (FMP) was structured with a regulatory mechanism that allowed a reopening of the fishery if above average recruitment occurred. The time-series average in the Maryland juvenile index (the mean number per tow of age-0 striped bass from a seine survey of Maryland estuaries begun in 1954) was exceeded in 1989, which led to a reopening of the fishery in 1990. However, strict management regulations were implemented that maintained large minimum sizes and a target instantaneous fishing mortality rate (F) at one half F_{msy} (0.38) (Field 1998). Populations of striped bass continued to rebuild and in 1993 and 1996 produced the two largest year classes in the time-series of juvenile indices.

A virtual (sequential) population analysis (VPA) of Atlantic striped bass was developed in 1997 as part of the ongoing work of the ASMFC Striped Bass Technical Committee (Shepherd and Lazar 1998). Previously, conclusions about the status of the stock were based on tagging conducted by state agencies and coordinated through the U.S. Fish and Wildlife Service (Young-Dubovsky et al. 1996). Although the tagging results provided useful information on survival rates, the type of information resulting from tagging studies was not sufficient to support various management requirements. Analysis of catch-at-age data with a VPA integrated catch, length and age data into year specific mortality and abundance estimates, beginning in 1982. These results provided the necessary information for managers to begin quota-based regulations of the entire coastal stock, as recommended in the FMP (ASMFC 1995).

Development of the VPA began with a compilation of available aging data, catch data and indices of abundance for the area between Maine and North Carolina. The assembled catch-at-age matrix was evaluated using the ADAPT method for virtual population analysis (Conser and Powers 1990; Gavaris 1988). The results of the analysis were reviewed and accepted during the 26th Northeast Fisheries Science Center Stock Assessment Workshop (NEFSC 1998). More detailed results are contained in the ASMFC Striped Bass Source document and the proceedings of the 26th Stock Assessment Workshop (Shepherd and Lazar 1998; NEFSC 1998).

The level of uncertainty in stock assessments depends on model selection and the quality of input data (Hilborn 1997). There are a variety of age-structured population models appropriate for assessing fish stocks, each with its own inherent suite of biases and assumptions (Megrey 1989). The intent of this paper is to explore the sources of uncertainty in the striped bass assessment associated with development of input data and model configuration within the ADAPT VPA framework.

Data Input

Commercial catch

Accuracy of the catch data were the first consideration in analysis of the catch-at-age matrix. Striped bass commercial landings data were least reliable during the period 1982–1990 when landings were collected by the National Marine Fisheries Service (NMFS) from records of fish dealers. Commercial striped bass fisheries were primarily small day-boat operations and were more likely to be missed or under-reported in the NMFS reporting system compared with landings in major ports. Since 1990, state agencies were responsible for monitoring striped bass landings and the methods more effectively sampled small dealers. In most states, licensed commercial fishermen were issued a limited number of locking jaw tags and these tags were required on each fish sold. This system provided essentially a census of the total number of striped bass landed. In states without tagging systems, reporting by licensed fishermen was required and strictly enforced. Landings were reported as numbers of fish and eliminated the error associated with converting landed weight to number using length frequency data and length-weight conversions. The strict regulations, limited quotas, and census-type reporting methods for landings increased the likelihood that commercial striped bass landings were relatively precise components of the catch matrix in recent years.

Commercial discards were the area of greatest uncertainty in the catch-at-age matrix. Limited quotas and increasing striped bass abundance raised the likelihood that a variety of fisheries would catch but discard striped bass. Sea sampling data were limited and expansion of sea sample data to total discard estimates required information on effort or some other measure across all fleets that may intercept striped bass. The problem was compounded by the large and diverse set of fleets that potentially catch striped bass owing to the dynamics of striped bass migrations. An alternative source of information available since 1987 was tag return data that contained information on the disposition of the fish. Total commercial bycatch discards were estimated using the ratio of commercial to recreational discards determined from tag returns. The ratio was expanded using total recreational discards estimated by the Marine Recreational Fisheries Statistics Survey (MRFSS) program. Tag reporting rate was assumed equivalent, although it is likely commercial reporting was lower. Total commercial discards were extrapolated for the period prior to the tagging programs using the trend in ratios from adjacent years. The proportion of discards by gear type was derived from tag recoveries, and gear specific survival rates were applied to discarded fish. Total estimated commercial discards followed expected trends over the time series. In the years with small minimum sizes and no quotas, discards were low but as the regulations were tightened and stock abundance increased, total discards also increased. Since 1994, higher discards followed years with large recruiting year classes. Several state marine fishery agencies had previously attempted to estimate discards in local fisheries using a combination of sea sample data and survey data. The sum of these estimates was comparable to the expanded tagging estimates. There were no expectations that the discard estimates would have a high degree of accuracy. However, since commercial bycatch was known to exist, an effort was made to incorporate that source of mortality to reduce a potentially significant source of bias.

Recreational catch

The MRFSS produced estimates of striped bass recreational catch and proportional standard errors since1982. Estimates were derived from expansion of survey results rather than the census approach used for commercial landing estimates. Total recreational landings declined steadily from 1982 to 1989. Despite the consistent trend in the total landing estimates, state specific landings were variable among years. For example, Virginia showed no recreational striped bass landings in 1982, 1983, and 1985 while the neighboring state of Maryland had significant landings. Landings in other year/state combinations, such as New Jersey in 1983, were three to four times greater than adjacent years. Associated with the reopening of fisheries in 1990 was a responsibility by states to collect additional recreational data. States with substantial recreational landings were required under the FMP to achieve a proportional standard error in their MRFSS estimate of 20% or less, which was accomplished by increasing the sample size of interviews. Consequently, total catch estimates became more precise and consistent since 1990. Recreational fisheries in the Hudson River were not included in the MRFSS program and were estimated by surveys conducted by the state of New York. The use of state specific landings rather than regional estimates in the development of a landings-at-age matrix prior to 1990 likely increased the variance in the analysis.

The MRFSS program was also the source for estimates of recreational discards. An assumption in the total discard estimate was that each release represented a unique fish. It was evident from examining tagging data that multiple recaptures of some fish occur during the year. The extent of double counting in the discard estimate is currently being investigated. Field experiments concluded that an average of 8% of released striped bass die due to hooking mortality (Diodati and Richards 1996). Although it has been shown that discard mortality varies by water temperature, terminal gear type, handling time, and a variety of other factors, the average rate (8%) was applied to total annual discard estimates.

Length data

Expansion of catch into length categories required representative samples by time, area and fishing gear. Commercial landings were divided into semiannual periods and length frequencies matched to fisheries by time and area cells whenever possible. Generally, the length data were adequate, with an average of 5% of landed fish measured. Smaller fisheries, such as fyke nets, tended to be poorly represented but since they contributed little to the total catch the consequences were minimal. The opportunity for collecting length frequency data from landings improved after 1995 as commercial quotas increased.

Length data for commercial discards were generally inadequate. Lengths of discarded striped bass in directed and nondirected commercial fisheries were expanded based on limited sea sampling or length frequencies of sub-legal size fish collected in fisheries independent surveys using comparable gear.

The number of length samples from recreational fisheries increased in recent years as states added larger numbers of samples to the intercept portion of the MRFSS survey. Since 1990, the MRFSS data have also been supplemented by information collected by volunteer fishermen. The length distributions of fish collected by volunteer anglers were assumed to be representative of recreational catches in that time and area. Length data to characterize discards were provided by anglers participating in a tag and release program managed by the American Littoral Society (Boreman and Lewis 1987). Since these fish were tagged and released by recreational anglers, they were by definition equivalent to the discarded (B2) fish in the MRFSS. Additional length data of discards were collected by volunteer fishermen from several states. The overall number of lengths available for recreational fisheries far exceeded the commercial data. In 1996, the recreational catch was expanded based on lengths of 11,240 landed and 19,303 discarded striped bass.

Catch at age

Uncertainty in the catch-at-age data are not incorporated into the ADAPT VPA model, however sensitivity to error in the catch data can be evaluated. Bias in commercial landings data are almost always a result of underestimating rather than overestimating the catch. Recreational catch estimates from survey results could be either under- or overestimated. The error in the striped bass catch-at-age matrix was likely most severe in the 1982–1989 period when sampling was not dictated by the FMP. Annual catch estimates for this period were varied ±50% to examine the influence on terminal year estimates of abundance and fishing mortality (Table 1). If catch were underestimated by 50%, the terminal year estimate of fishing mortality would be underestimated by 12% (0.03) and abundance by 12% (4.8 million). In contrast, if catch were overestimated by 50%, the terminal year F would be overestimated by 15% (0.05) and abundance by 13% (5.2 million). Numerous combinations of catch error could be explored but it is clear that catch for the first half of the time series would have to be severely misestimated to significantly impact F and N in the terminal year.

Both the catch data and tuning indices were categorized into ages using age-length keys. Semiannual age keys were compiled and divided into five geographic areas. All ages were determined from scales, which may have introduced error into the analysis. Aging error for scales increase beyond age 12, with scales biased toward underestimation of true age (Secor et al. 1995). However, an evaluation of scale ages using known age fish captured in New York (hatchery fish recovered with coded wire tags) showed a 90% agreement up to age 12 (Vecchio and O'Riordan 1999). A catch-at-age matrix with a 15+ category reduced the impact of aging error in older fish but undoubtedly some element of age error contributed to uncertainty in the assessment results.

The catch-at-age matrix from 1982 to 1997 is provided in Table 2.

Model Development

The ADAPT model framework was chosen for the striped bass assessment because of the flexibility it provided in evaluating a large set of abundance indices. This model assumes error in the indices is greater than in the catch-at-age matrix. Since the tuning indices were from multiple stocks and a variety of sampling designs, the assumption seemed reasonable. Initial configuration of the model included a matrix of 16 years × 15 ages and 108 potential age specific tuning indices. The catch-at-age matrix was restricted to 14 age groups with a 15+ category. During the late 1980s and early 1990s the minimum size in some coastal states was 91 cm (36″). Therefore any truncation of the matrix below age-14 would have incorporated a large percentage of the catch from the coastal recreational fisheries into the plus category, the age group that tends to be most poorly estimated in the model.

Tuning indices

The ADAPT model requires auxiliary abundance data for estimating population size. The striped bass assessment was unusual in the amount of information available, particularly juvenile indices. Each of the three stocks had long time series of surveys designed to measure juvenile abundance. The indices in the Hudson River were strongly correlated to abundance-at-age one measured in surveys of Long Island nursery areas (McKown 1992). Juvenile indices for the Maryland portion of the Chesapeake Bay were validated using relative abundance-at-age in subsequent commercial landings (Goodyear 1985). Juvenile surveys were also available from the Delaware River (Rago et al. 1995) and the Virginia portion of Chesapeake Bay (Rago et al. 1995). Although annual variance estimates were available for all these indices they were not incorporated into the tuning procedure.

Time series of fishery independent surveys designed to measure relative abundance of adult striped bass were available for the Maryland portion of Chesapeake Bay and coastal New York. The Maryland survey used a multiple meshsize gill net and the CPUE was adjusted for selectivity in different panels of the net (Helser et al. 1998; Helser and Waller 1999). The New York survey used a commercial haul seine deployed from beaches in

TABLE 1. Relative effect of changes in striped bass catch for 1982–1989 on estimates of age 1+ abundance (>000) and fishing mortality (age 4–13) from virtual population analysis.

Relative change in catch	Year																
	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Percent change in age 1+ abundance (000s)																	
−50%	−37%	−32%	−25%	−16%	−12%	− 8%	− 6%	− 4%	− 3%	− 4%	− 5%	− 6%	− 8%	−10%	−11%	−12%	−13%
−25%	−18%	−16%	−12%	− 8%	− 6%	− 4%	− 3%	− 2%	− 2%	− 2%	− 3%	− 3%	− 4%	− 5%	− 6%	− 6%	− 6%
−10%	− 7%	− 6%	− 5%	− 3%	− 2%	− 2%	− 1%	− 1%	− 1%	− 1%	− 1%	− 1%	− 2%	− 2%	− 2%	− 2%	− 3%
0	5231	6513	7345	8427	9661	11459	13984	17091	22226	24289	25526	28047	37738	38692	37528	42578	40237
+10%	7%	6%	5%	3%	2%	2%	1%	1%	1%	1%	1%	1%	2%	2%	2%	2%	2%
+25%	18%	16%	12%	8%	6%	4%	3%	2%	2%	2%	3%	3%	4%	5%	6%	6%	6%
+50%	37%	32%	25%	16%	12%	9%	7%	5%	4%	5%	6%	7%	8%	9%	11%	11%	12%
Percent change in age 4−13 fishing mortality																	
−50%	−28%	−27%	−31%	−33%	−38%	−39%	−44%	−49%	1%	3%	3%	4%	6%	7%	10%	15%	
−25%	−12%	−11%	−13%	−13%	−18%	−18%	−22%	−22%	0%	1%	1%	2%	3%	3%	5%	8%	
−10%	−4%	−4%	−6%	−4%	−6%	−7%	−9%	−9%	0%	0%	1%	1%	2%	1%	1%	3%	
0	0.24	0.21	0.12	0.12	0.12	0.07	0.13	0.09	0.17	0.20	0.17	0.25	0.22	0.24	0.28	0.32	
+10%	3%	3%	3%	5%	6%	7%	9%	8%	−1%	−2%	−1%	0%	0%	−2%	−2%	−3%	
+25%	7%	9%	8%	14%	14%	18%	18%	21%	−1%	−2%	−2%	−2%	−2%	−4%	−5%	−6%	
+50%	14%	15%	18%	25%	26%	29%	36%	43%	−2%	−3%	−4%	−4%	−5%	−7%	−10%	−12%	

TABLE 2. Striped bass catch-at-age (000s); 1982–1997.

	Age															
Year	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15+	Total
1982	1.8	105.6	256.8	221.0	58.5	19.2	24.3	16.8	11.7	10.6	11.0	13.7	3.4	4.1	8.1	2300.4
1983	3.6	110.4	178.4	193.3	150.3	39.4	18.7	4.1	2.9	3.7	4.6	5.7	4.9	4.1	4.6	2185.8
1984	5.6	543.1	303.0	82.7	60.6	51.9	18.4	4.7	2.1	2.1	0.7	0.3	2.2	4.3	4.7	3259.2
1985	1.3	72.6	102.1	40.6	58.8	43.2	43.6	17.3	6.4	3.4	1.0	0.8	0.5	0.9	8.9	1204.4
1986	11.4	21.1	64.0	133.0	50.0	32.1	20.4	24.1	9.2	5.3	3.4	1.6	0.9	2.5	6.7	1157.1
1987	1.4	11.1	38.0	51.8	67.6	25.2	13.3	6.6	6.5	3.0	1.5	2.0	3.4	2.1	7.7	723.2
1988	2.6	30.8	42.5	63.9	106.3	97.4	40.6	24.6	14.0	5.8	3.7	3.2	2.4	3.0	4.1	1334.6
1989	0.8	36.8	80.7	69.0	105.7	95.9	46.1	21.2	10.5	3.8	3.3	2.0	1.9	1.6	5.4	1454.4
1990	2.7	54.4	135.9	203.4	181.6	174.3	111.9	75.6	24.3	8.6	6.0	4.0	4.9	4.2	7.6	2998.2
1991	1.9	77.9	152.7	217.4	165.7	103.4	95.3	87.6	62.9	25.7	14.8	3.0	2.8	3.5	17.7	3097.3
1992	3.3	51.1	216.7	201.3	188.2	113.8	65.7	73.6	64.3	49.4	10.2	4.5	1.9	5.5	10.4	3180.0
1993	0.3	76.1	197.6	344.0	299.9	194.0	90.1	71.5	89.6	83.2	45.0	10.2	5.0	1.3	13.0	4562.1
1994	5.7	146.1	350.5	291.7	368.8	233.0	135.8	87.1	100.4	81.4	36.2	22.4	3.4	1.5	9.8	5621.3
1995	3.7	414.7	446.9	438.1	385.7	461.9	201.2	186.1	147.9	86.4	50.8	16.1	9.3	1.9	3.3	8561.9
1996	0.5	98.8	659.5	666.4	552.0	476.9	457.2	217.7	143.3	71.7	44.2	48.3	13.2	4.7	2.6	10371.3
1997	2.5	287.3	487.3	849.5	618.7	598.6	415.6	382.6	207.9	125.7	61.8	30.9	13.0	7.9	5.1	12283.6

southeastern Long Island. A source of uncertainty in this and all surveys is whether the survey gear captures fish proportional to stock size. It has been suggested that haul seine catches were biased toward small fish, but there were no data available to evaluate this potential bias. Other tuning indices in the model were derived from fishery dependent data. These included Hudson River shad gill-net CPUE, hook-and-line CPUE from Massachusetts commercial fisheries, and CPUE of Connecticut volunteer recreational fishermen.

Selection of the final tuning indices was based on the diagnostics from the VPA including CVs of N, mean square residuals, and partial variance estimates of indices. Indices with relatively high CVs were rejected. Time trended patterns in the residuals from stock specific indices were not considered in the diagnostics because of the mixed stock nature of the catch data. If a residual pattern existed, it may have been the result of changes in the contribution of that stock component over time relative to combined stock abundance. A total of 57 age-specific indices were chosen for inclusion into the model (Table 3; Appendix I).

Model configuration

The ADAPT software used in the striped bass assessment allowed evaluation of various model configurations that had important implications for the results. Some important choices included the use of constant or age-specific natural mortality (M); the assumption of a flat-topped or dome-shaped partial recruitment (PR) vector; use of reweighting algorithms for tuning indices; whether indices are tuned to population size at 1 January or 1 July; the

TABLE 3. Indices used in tuning the striped bass virtual population analysis. Age-0 indices advanced 1 year and 1 age in tuning procedure.

	Age								
Data Source	0	1	2	3	4	5	6	7	8 to 15+
MA commercial CPUE									X
CT recreational CPUE			X	X	X	X	X	X	X
Hudson River seine survey	X								
Hudson shad fishery CPUE							X	X	X
Long Island seine survey			X						
NY ocean haul seine survey						X	X	X	X
New Jersey seine survey		X							
Delaware seine survey	X								
MD spawning stock survey			X	X	X	X	X	X	X
Maryland seine survey	X	X							
Virginia seine survey	X								

TABLE 4. Effect of variable age-1 natural mortality (M) on striped bass virtual population analysis abundance at age estimates (000s) for 1998.

	M			
Age	0.2	0.3	0.45	0.6
1	7767	8583	9972	11585
2	11605	11605	11605	11605
3	4833	4833	4833	4833
4	4567	4567	4567	4567
5	6633	6633	6633	6633
6	1843	1843	1843	1843
7	735	735	735	735
8	773	773	773	773
9	847	847	847	847
10	453	453	453	453
11	228	228	228	228
12	140	140	140	140
13	78	78	78	78
14	81	81	81	81
15+	31	31	31	31

TABLE 5. Effect of variable age-1 natural mortality (M) on striped bass virtual population analysis estimate of fishing mortality at age for 1997.

	M			
Age	0.2	0.3	0.45	0.6
1	0.0	0.0	0.0	0.0
2	0.05	0.05	0.05	0.05
3	0.09	0.09	0.09	0.09
4	0.11	0.11	0.11	0.11
5	0.27	0.27	0.27	0.27
6	0.56	0.56	0.56	0.56
7	0.40	0.40	0.40	0.40
8	0.35	0.35	0.35	0.35
9	0.35	0.35	0.35	0.35
11	0.41	0.41	0.41	0.41
11	0.34	0.34	0.34	0.34
12	0.31	0.31	0.31	0.31
13	0.14	0.14	0.14	0.14
14	0.33	0.33	0.33	0.33
15+	0.33	0.33	0.33	0.33

criteria used to identify age at full recruitment to the fishery; and the calculation of F using a weighted or unweighted average across ages groups.

As with all assessments, natural mortality was one of the more elusive parameters in the model. The null hypothesis was an M that resulted in approximately 5% of a cohort surviving to maximum age. Maximum age was defined as 20 years, although some evidence suggested a higher maximum age was possible (Secor et al. 1995). Striped bass were assessed using a constant M of 0.15. An alternative hypothesis examined was the use of age specific values of M. Some evidence from field studies suggested natural mortality on juvenile and age one fish could be higher due to predation (Buckel et al. 1999). Beyond that age, the size of striped bass likely reduced their vulnerability to predation. Increasing the value of M on age-1 increased the abundance estimates of recruits but had no effect on estimates of fully recruited fishing mortality (ages 4–13) (Tables 4–6).

The consequences of misspecifying M in a sequential population analysis have been well documented (Mertz and Meyers 1997; Hilden 1988; Lapointe et al. 1989; Sims 1984). The impact of alternative Ms of 0.1, 0.2, 0.25, or 0.3 on the striped bass VPA results were examined. An M of 0.1 resulted in a decreased abundance estimate and an increased F in the terminal year while higher Ms resulted in higher estimates of N and lower Fs (Figures 1 and 2). The bias created by misspecification of M also increases with greater variability in F (Mertz and Meyers 1997). A change in M caused a similar directional change in the biological reference point (i.e., higher M resulted in higher value for F_{msy}). Therefore, if the constant M used in the assessment is greater than the "true" value there may be an increased risk of exceeding the biological reference point.

Identification of the age at full recruitment to the fishery has important implications in defining fully recruited F and development of biological reference points. One approach to calculation of partial recruitment to the fishery is the relationship between the highest F among ages and age specific Fs. For

TABLE 6. Effect of variable age-1 natural mortality (M) on striped bass virtual population analysis abundance at age estimates (000s) for 1982–1998.

	M			
Year	0.2	0.3	0.45	0.6
1982	1452	1605	1864	2166
1983	2888	3192	3708	4307
1984	2534	2800	3253	3779
1985	3267	3611	4195	4874
1986	2913	3219	3739	4343
1987	3626	4007	4656	5409
1988	4646	5134	5965	6930
1989	5727	6330	7354	8544
1990	8388	9270	10770	12513
1991	6311	6974	8103	9414
1992	5894	6513	7567	8791
1993	7440	8223	9553	11099
1994	15777	17436	20257	23534
1995	8409	9293	10797	12543
1996	7237	7998	9292	10795
1997	14178	15668	18203	21148
1998	7767	8583	9972	11585

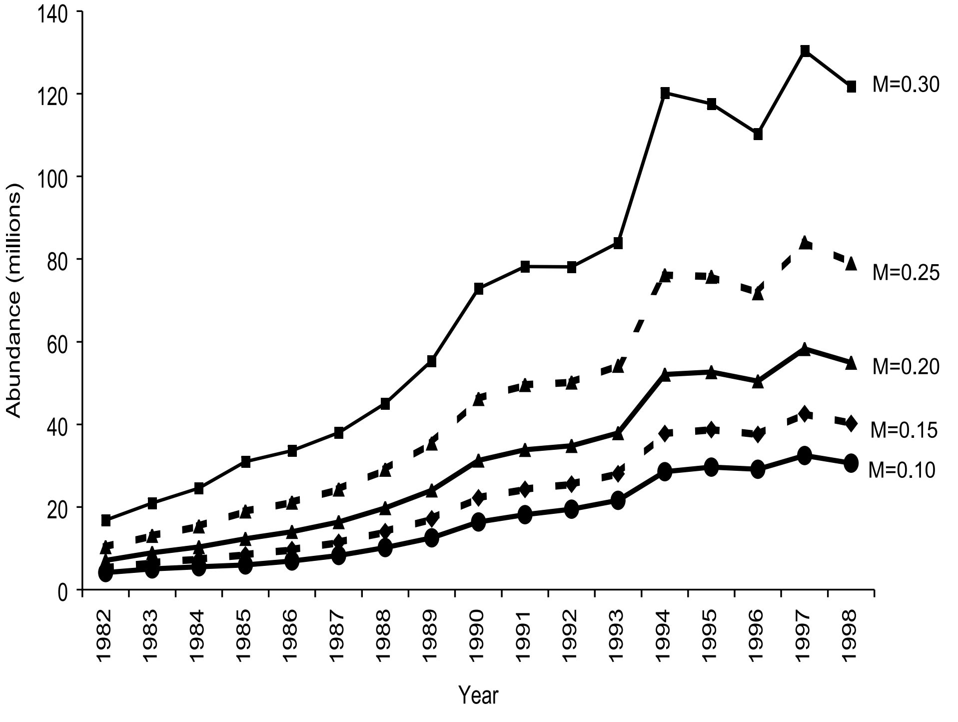

FIGURE 1. The impact of varying levels of instantaneous natural mortality (M) on virtual population analysis estimates of striped bass abundance (millions, age 1+).

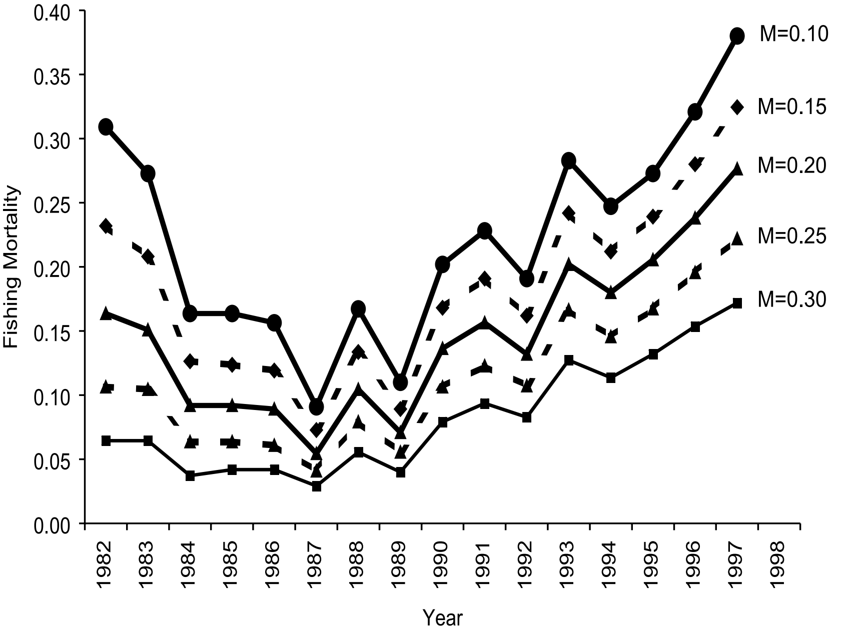

FIGURE 2. The impact of varying levels of instantaneous natural mortality (M) on estimate of average (unweighted) age 4–13 fishing mortality from virtual population analysis estimate for striped bass.

instance, if F at age 10 were the highest at 0.3 and F at age 4 was 0.15, then the partial recruitment at age 10 would be 1.0 and age 4 would be 0.5. An alternative approach was to use the age of maximum catch in the annual catch vectors. Depending on year class strength the maximum catch could occur at a significantly different age than the model calculation. For striped bass, the age at maximum catch was chosen to characterize age at full recruitment. The model calculations of partial recruitment reflected changes in the fishery regulations throughout the time series but were not used in calculation. The selection of age at full recruitment particularly affected calculation of fully recruited F in those years where the maximum catch occurred at ages 3 or 4 while the model PR equaled 1.0 at ages 8–10.

The shape of the partial recruitment curve also has important implications in the estimate of biological reference points. In the striped bass assessment a flat-topped PR at age vector was used in calculation of F_{msy}. The ADAPT VPA model does not use the partial recruitment values if the entire suite of Fs at age in the terminal year are estimated, as was the case in striped bass. However, if the true selectivity pattern was best defined by a dome shaped partial recruitment vector, the estimate of F_{msy} would be incorrect. The consequence could be allowable levels of fishing mortality that would not maintain an optimal level of spawning stock biomass.

Fully recruited F can also be calculated in several ways. Average fishing mortality could be an average of Fs on fully recruited fish or an average weighted by abundance or biomass. With a flat topped PR, all age classes were assumed to be fully recruited, therefore the associated Fs contributed equally to the total fishing mortality. The unweighted average F in 1997 is 0.33. The alternative was an average weighted by abundance-at-age; the fully recruited F in 1997 weighted by N was 0.24. The two approaches gave different results due to the presence of some large year classes that experienced low fishing mortality. The influence of the 1993 year class resulted in an overall weighted F lower than if the assumption were made that all fully recruited Fs were equivalent. The two estimates of F have different implications in comparison to the target fishing mortality of 0.31. A weighted average would downweight several older cohorts that were at or above the target F. An increase in mortality to attain the target F would subject these cohorts to an increased risk of overfishing. An unweighted average F was chosen to represent the fully recruited fishing mortality in striped bass. The use of this estimate may reduce the risk of increased mortality on cohorts already at or above the target.

Tuning indices in the model can be defined to reflect population abundance at either the beginning or middle of the year. Since the indices were not measured on 1 January or 1 July, the assumption was made that mortality was insignificant between the time the indices were collected and the tuning date. For instance, if indices were collected in October and a major fishery occurred in November, different levels of mortality could create an annual index that may not reflect population abundance as of 1 January. In the striped bass model all indices were tuned to population abundance on 1 January.

Iterative reweighting of the tuning indices is an option within the ADAPT framework and was used for the striped bass model. Re-weighting is done by weighting the relative contribution of an index by the inverse of its partial variance in minimizing the models objective function. Ideally, stock identification methods would distinguish among stocks in the coastal fisheries and the relative contribution of each stock could be used to weight stock specific indices. Since stock information was not available and the stock contributions were assumed to be unequal, the model was allowed to weight the indices based on relative fit to the total catch-at-age data. If annual production in all stocks was equivalent and reflected by the indices, then reweighting would have little impact. In striped bass, the relative contributions of the Chesapeake stock were generally highest. Therefore, the reweighting served to emphasize the stronger year classes produced by this stock. Although there was uncertainty associated with the reweighting scheme, it likely produced more accurate results in the mixed stock model than an assumption of equal contributions among stocks.

Characterizing the Uncertainty

A feature of the ADAPT software is the ability to characterize the uncertainty in model fit using a bootstrap resampling of the residuals from the observed-predicted tuning indices (Mohn 1993). In the nonlinear least-squares (nlls) estimate of a solution, estimates of population abundance were chosen that provided the best fit to the tuning indices. The residuals of that fit were bootstrapped 500 times and new values of N produced. The distribution of the associated Fs provided an indication of variation and the bias (bootstrap mean-nlls mean). Results of the bootstrap indicated an 80% probability that terminal values (1997) of F in the model ranged from

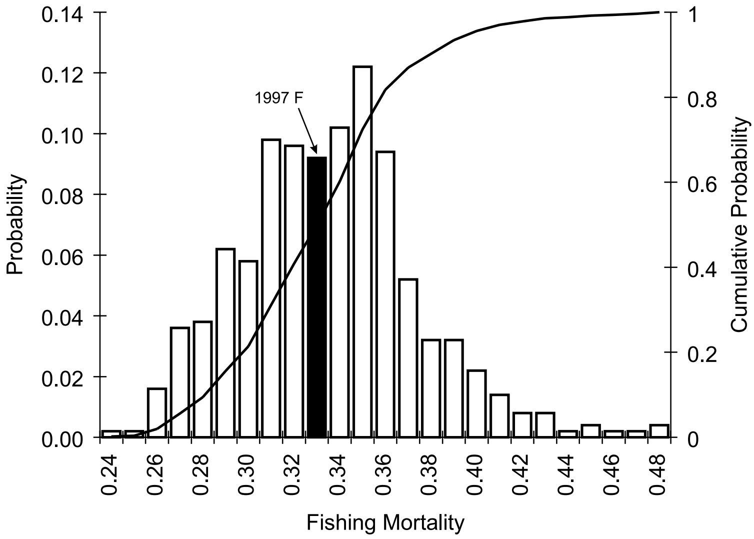

FIGURE 3. Bootstrap results of 1997 fishing mortality estimate for striped bass from virtual population analysis; $N = 500$.

0.30 to 0.36 (Figure 3), and bias in the F and N estimates was 2%. The model output also provided an estimate of the CV at age for N that ranged from 0.19 to 0.34.

The characterization of the uncertainty in the results based on the bootstrap estimates or CVs reflect the variability associated with the model fitting procedure. The bootstrapping did not account for most of the issues previously mentioned that could contribute to uncertainty in the assessment results. The model assumes that the catch-at-age matrix was measured without error. Therefore, any error associated with landings data, length data, and age data was not included. The error in the model configuration was assumed to occur in the time-series of the survey indices. Variances in the point estimates of the tuning indices were also disregarded although they were reduced by using a $\log_e$-$\log_e$ model in the fitting process. Another assumption in the bootstrapping was the correct configuration of the final model. If the model criteria, such as age at full recruitment and M, were incorrect it would not be reflected in the bootstrap results.

A common problem in virtual population analysis is bias associated with the terminal year estimates of fishing mortality and abundance (Mohn 1993). The striped bass VPA results were analyzed to determine if any retrospective patterns existed in the parameter estimates. The tendency of the model was to underestimate fishing mortality in the terminal year and over estimate recruitment of age-1 fish (Table 7). However, the retrospective pattern was not consistent among all years as there were also instances where the model over estimated the terminal year F. Estimates from the striped bass VPA model were not corrected for any retrospective patterns.

Implications

A series of issues pertaining to input data and model selection have been identified that contributed to uncertainty in the results. An important concern was the impact this had on the conclusion about the stock status. One method to assess the implication of this uncertainty is to compare estimates of F with an independently derived estimate. Such a comparison was possible with striped bass because there has been an extensive coast wide tagging program designed to estimate survival. The U.S. Fish and Wildlife Service (FWS) has coordinated the release of nearly 250,000 tagged striped bass by state fisheries agencies between North Carolina and Massachusetts since 1987 (Smith and He 1998). Annual estimates of survival were produced from analysis of tag returns from two programs sampling the mixed coastal stock; the New York ocean haul seine survey and a FWS tagging program off the coast of North Carolina. The results represented survival of fish 28 in and greater, fish approximately age 6 and older. Survival estimates from tag return data were adjusted to account for bias resulting from tag recoveries of fish that were released alive (Smith et al. 2000). The number of tag releases and recoveries were assumed to be proportional to abundance; therefore, results

TABLE 7. Retrospective analysis of striped bass virtual population analysis.

Terminal year	Year																
	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Estimates of abundance at age 1 (000s)																	
1992	1851	4856	4152	3729	2396	3756	5290	5589	7514	5603	4380	8270					
1993	1691	4321	3801	3562	2402	3223	4513	4952	6600	4983	4168	6039	16434				
1994	1591	3969	3524	3490	2814	3465	4597	5212	6472	4921	4737	5855	15021	8800			
1995	1382	3583	3176	3286	2804	3628	4785	5440	6686	4952	4714	6647	13804	7591	6578		
1996	1372	2746	2812	3109	2688	3346	4321	5279	7111	4953	4810	6747	13720	7055	6299	17885	
1997	1381	2747	2411	3108	2772	3449	4419	5448	7979	6003	5606	7078	15008	7999	6884	13486	7388
Estimates of fully recruited fishing mortality																	
1992	0.18	0.16	0.09	0.08	0.07	0.04	0.07	0.05	0.10	0.09	0.08						
1993	0.20	0.18	0.11	0.10	0.09	0.05	0.10	0.06	0.12	0.12	0.10	0.13					
1994	0.21	0.19	0.11	0.11	0.10	0.06	0.11	0.07	0.13	0.15	0.11	0.16	0.14				
1995	0.22	0.20	0.12	0.11	0.11	0.06	0.12	0.08	0.10	0.17	0.13	0.19	0.16	0.20			
1996	0.23	0.21	0.12	0.12	0.12	0.07	0.13	0.09	0.17	0.19	0.16	0.24	0.21	0.25	0.29		
1997	0.23	0.21	0.12	0.12	0.12	0.07	0.13	0.09	0.17	0.20	0.17	0.25	0.22	0.24	0.28	0.33	

were compared with the VPA average F weighted by abundance. The results showed a close comparison in estimates of total fishing mortality (Figure 4). The comparison was particularly encouraging because the estimates were made using different data input and models. There is extensive literature documenting the sources of error in tag survival estimates that include incomplete mixing of tagged fish into the population, tag loss, tag induced mortality, biased sampling and nonreporting of tag recoveries. Nevertheless, this type of independent verification of model results provided additional support that the VPA was correctly identifying the trend and relative magnitude in fishing mortality despite the different assumptions of both models.

Comparison to biological reference points

An overfishing definition for striped bass was defined as F_{msy} (0.38), calculated using a modified Thompson-Bell yield per recruit model and a Shepherd stock-recruitment model (Shepherd and Lazar 1998). A fishing mortality of 0.31 was chosen as the target fishing mortality in the FMP. Consequences of fishing at the target F were evaluated using a stochastic projection model incorporating the

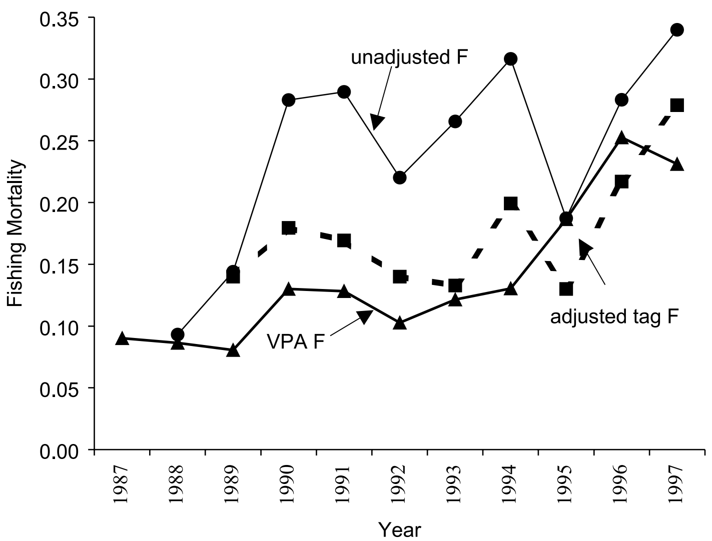

FIGURE 4. Comparison of tagging estimates of striped bass fishing mortality and VPA estimates of fishing mortality weighted by abundance.

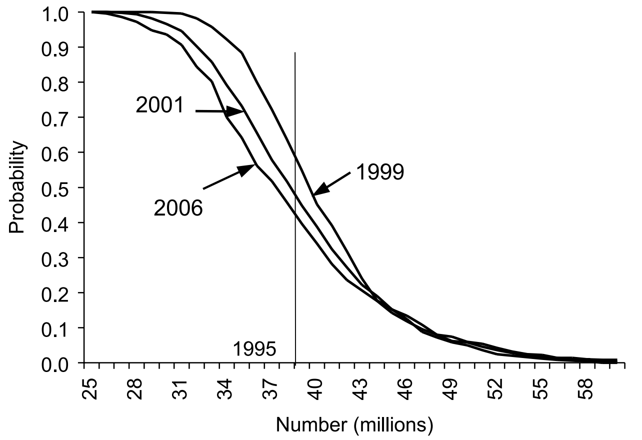

FIGURE 5. Probability of remaining at or above the 1995 level of striped bass population abundance if fishing mortality were held equal to 0.31.

bootstrapped terminal year VPA stock size and variability. Incoming annual recruitment was randomly chosen from log-normal recruitment distributions bounded by the levels occurring under moderate to high levels of spawning stock biomass. Starting stock size vectors were randomly chosen from a matrix of bootstrap realizations of population abundance. A Monte Carlo routine decremented stock size-at-age by the designated F and M and the 1997 exploitation pattern. Mean annual abundance and standard error were calculated from 500 abundance-at-age estimates for a ten-year projection period. The frequency distributions from the simulation were calculated to determine the probability of exceeding the benchmark level of 1995, the year the stock was declared restored (Figure 5).

The results provide insight into the short-term sustainability of the stocks at various levels of fishing mortality. Stock abundance would be expected to remain near current levels if fished at the target F of 0.31. The probability of remaining at or above the 1995 point estimate of abundance level was 55% in 1999, 47% in 2001, and 43% by 2006 (Figure 5). The projections at Fmsy followed a similar trend with a 53% probability of remaining at or above the benchmark level by 1999, 42% by 2001, and 34% by 2006 (Figure 6). Since large fish are targeted in several fisheries, abundance of fish greater than

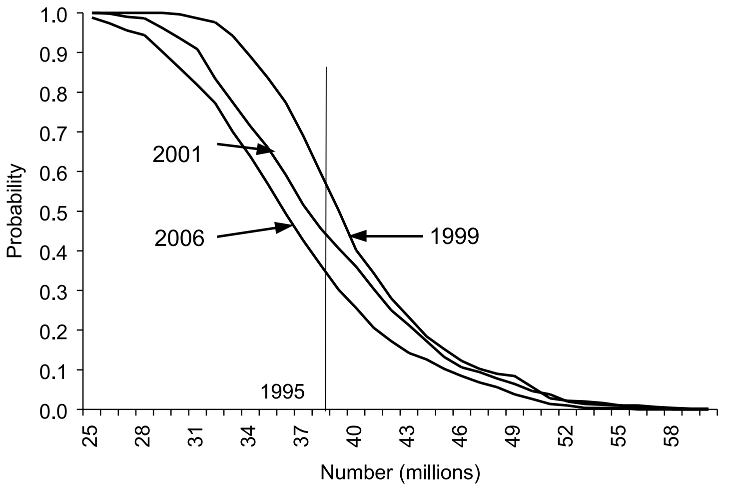

FIGURE 6. Probability of remaining at or above the 1995 level of striped bass population abundance if fishing mortality were held equal to F_{msy} (0.38).

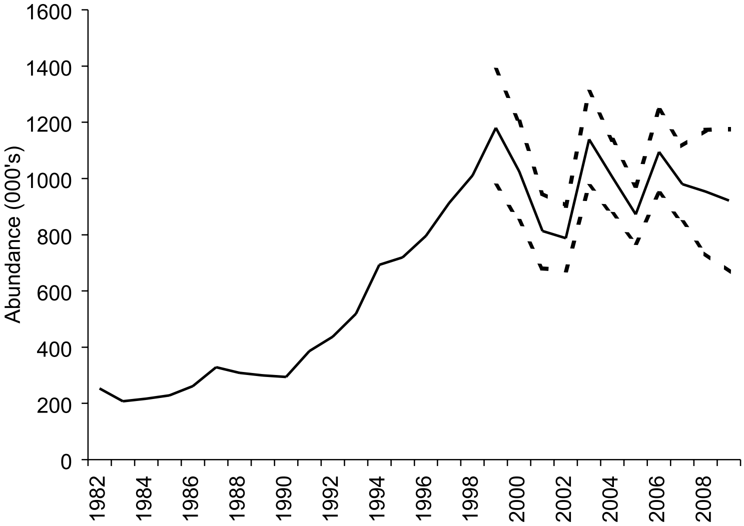

FIGURE 7. Projected abundance of age 10+ striped bass with fishing mortality equal to 0.31. Mean abundance and 80% prediction intervals with virtual population analysis estimates for 1982–1998.

age 10 was also examined. The presence of large year classes already in the population helped maintain the abundance of older fish, at least over the short term (Figure 7). However, increasing fishing mortality to F_{msy} resulted in a decline in the abundance of large older fish despite the influence of strong year classes (Figure 8). For the striped bass fishery, the level of acceptable risk and the long-term expectations of management are still being debated.

Conclusions

The striped bass assessment was presented as an example of an analysis with a variety of fishery dependent and independent data, reasonable catch

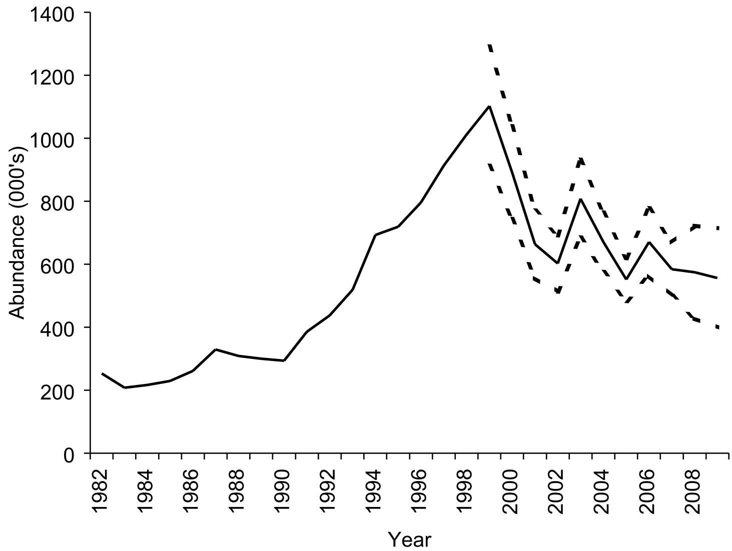

FIGURE 8. Projected abundance of age 10+ striped bass with fishing mortality equal to 0.38. Mean abundance and 80% prediction intervals with virtual population analysis estimates for 1982–1998.

data, and a clear contrast in population abundance during the time series. Beginning in the mid-1980s, stocks of striped bass increased dramatically and produced a consistent upward trend in abundance indices and landings. With such obvious signals from input data, most population models would reach similar conclusions. The bootstrap estimates of the VPA run provided some indication of the uncertainty in the model fit and for striped bass the results suggested the conclusions of the model were relatively precise. Yet, the uncertainty was evaluated relative to the model and did not fully incorporate variance surrounding the input data. Despite the uncertainty in the VPA, tagging models not dependent on catch matrices, produced comparable results. This type of redundancy shows that either both methods produced reasonably accurate results or were similarly biased.

As the population growth of striped bass begins to stabilize or decline, the implications of uncertainty in the results will become more critical. The difference between the target fishing mortality and the overfishing definition is only 0.07. The data and population models available for stock assessment are not adequate to estimate fishing mortality with that level of resolution. Although point estimates of F may be on target, the probability distribution from the bootstrap procedure may also exceed Fmsy. Consequently there will be some probability that overfishing is occurring in addition to risk associated with whatever variance was unaccounted for in the bootstrap.

The uncertainty in stock assessment results, as illustrated with the striped bass example using an age-structured model, can occur at several levels throughout the analysis; with the input data, the model selection and the configuration of the chosen model. The basis for any assessment model is representative sampling of the removals from a population. As that basic catch data is divided into sub-components of length and age, there becomes increased risk of introducing error into the model. Some model structures can incorporate information associated with error in the catch and age data, however, the analyst must have some knowledge of that uncertainty. Model selection can also contribute to the uncertainty in the result but this source of error is often not evaluated. Corroborating the assessment results using alternative models can provide some helpful insight regarding the conclusions. Another often overlooked aspect of the uncertainty in assessments is the effects of density changes on population parameters such as growth, maturity and natural mortality (Rosenberg and Restrepo 1994). Despite recent work demonstrating some strong population responses to density (Rochet 1998; Trippel et al. 1997), few assessment models directly account for these responses as a function of changes in fishing mortality (Trippel 1999). Knowledge of the plasticity of life history traits should be considered in evaluating possible sources of uncertainty.

When choosing the target mortality, fisheries managers should define the acceptable level of risk that they will exceed the overfishing level. It is the responsibility of the assessment scientists to characterize as much uncertainty in the results as possible and the risk of overfishing associated with various management approaches. Throughout the development of an assessment, scientists make both subjective and objective decisions that will affect the results in some fashion. A better understanding of the implications of those decisions should improve the chances that the correct choices are made.

Acknowledgments

The striped bass stock assessment was the result of cooperative work by members of the ASMFC striped bass stock assessment sub-committee, technical committee and tagging working group. I acknowledge John Field from ASMFC for his efforts in coordinating the striped bass program and Victor Vecchio of NYDEP for supplying tagging results. The manuscript benefited from comments from several reviewers, particularly Steven Murawski. The ADAPT software used in the analysis was developed by Laura Shulman of the NEFSC.

References

ASMFC. 1995. Amendment #5 to the Interstate Fishery Management Plan for Atlantic Striped Bass. ASMFC Fisheries Management Report No. 24, Washington, D.C.

Bigelow, H. B., and W. C. Schroeder. 1953. Fishes of the Gulf of Maine. U.S. Fish Wildlife Service Fish Bulletin 74(53): 1–577.

Boreman, J., and R. R. Lewis. 1987. Atlantic coastal migration of striped bass. Pages 331–339 *in* M. J. Dadswell, R. J. Klauda, C. M. Moffitt, R. L. Saunders, R. A. Rulifson, and J. E. Cooper, editors. Common strategies of anadromous and catadromous fishes. American Fisheries Society, Symposium 1, Bethesda, Maryland.

Buckel, J. A., D. O. Conover, N. D. Steinberg, and K. A. McKown. 1999. Impact of age-0 bluefish (*Pomatomous saltatrix*) predation on age-0 fishes in the Hudson River estuary: evidence for density-dependent loss of juvenile striped bass (*Morone saxatilis*). Canadian Journal of Fisheries and Aquatic Sciences 56:275–287.

Clark, S. H., editor. 1998. Status of the Fishery Resources off the Northeastern United States. NOAA Technical Memorandum NMFS-NE-115.

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. International Commission for the Conservation of Atlantic Tunas, Collected Volumes of Scientific Papers 32:461–447.

Diodati, P. J., and R. A. Richards. 1996. Mortality of striped bass hooked and released in salt water. Transactions of the American Fisheries Society 125(2):300–307.

Field, J. D. 1998. Atlantic striped bass management: Where did we go right? Fisheries 22(7):6–8.

Gavaris, S. 1988. An adaptive framework for the estimation of population size. Canadian Atlantic Fisheries Scientific Advisory Committee (CAFSAC) Research Document 88/29. Dartmouth, Nova Scotia.

Goodyear, C. P. 1985. Relationship between reported commercial landings and abundance of young striped bass in Chesapeake Bay, Maryland. Transactions of the American Fisheries Society 114(1):92–96

Helser, T. E., J .P. Geaghan, and R. E. Condrey. 1998. Estimating gillnet selectivity using non-linear response surface regression. Canadian Journal of Fisheries and Aquatic Sciences 55(6):1328–1337.

Helser, T., and L. Waller. 1999. Selectivity of experimental gill nets for Chesapeake Bay Striped Bass (*Morone saxatilis*): direct and indirect estimates. Report to the ASMFC Striped Bass Technical Committee, Baltimore, Maryland.

Hilborn, R. 1997. Uncertainty, risk and the precautionary principle. Pages 100–105 *in* E. K. Pikitch, D. D. Huppert, and M. P. Sissenwine, editors. Global trends: fisheries management. American Fisheries Society, Symposium 20, Bethesda, Maryland.

Hilden, M. 1988. Errors in perception in stock and recruitment studies due to wrong choices of natural mortality rate in virtual population analysis. Journal du Conseil, Conseil International pour l'Exploration de la Mer. 44:123–134.

Lapointe, M. F., R. M. Peterman, and A. D. MacCall. 1989. Trends in fishing mortality rate along with errors in natural mortality rate can cause spurious time trends in fish stock abundances estimated by virtual population analysis (VPA). Canadian Journal of Fisheries and Aquatic Sciences 46:2129–2139.

Mace, P. M. 1997. Developing and sustaining world fisheries resources: the state of the science and management. Commonwealth Scientific and Industrial Research Organization, Collingswood, Australia.

Megrey, B. A. 1989. Review and comparison of age-structured stock assessment models from theoretical and applied points of view. Pages 8–48 *in* E. F. Edwards and B. A. Megrey, editors. Mathematical analysis of fish stock dynamics. American Fisheries Society, Symposium 6, Bethesda, Maryland.

Mertz, G., and R. Meyers. 1997. Influence of errors in natural mortality estimates in cohort analysis. Canadian Journal of Fisheries and Aquatic Sciences 54:1608–1612.

McKown, K. 1992. Validation of the Hudson River young-of-the-year striped bass indices. New York Department of Environmental Conservation, Division of Marine Resources, Stony Brook, New York.

Mohn, R. K. 1993. Bootstrap estimates of ADAPT parameters, their projection in risk analysis and their retrospective patterns. Pages 173–184 *in* S. J. Smith, J. J. Hunt, and D. Rivard, editors. Risk evaluation and biological reference points for fisheries management. Canadian Special Publication of Fisheries and Aquatic Sciences 120.

NMFS. 1999. Our Living Oceans. Report on the status of U.S. living marine resources, 1999. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-F/SPO-41 Northeast Fisheries Science Center. 1998. 26th Northeast Regional Stock Assessment Workshop: stock assessment review committee (SARC) consensus summary of assessments. NEFSC Reference Document 98–03.

Rago, P. J., C. D. Stephan, and H. M. Austin. 1995. Report of the juvenile abundance indices workshop, ASMFC Special Report 48.

Richards, R. A., and D. G. Deuel. 1987. Atlantic striped bass: stock status and the recreational fishery. Marine Fisheries Review 49(2):58–66.

Rochet, M. J. 1998. Short-term effects of fishing on life history traits of fishes. ICES Journal of Marine Science 55: 371–391.

Rosenberg. A. A, and V. R. Restrepo. 1994. Uncertainty and risk evaluation in stock assessment advice of U.S. marine fisheries. Canadian Journal of Fisheries and Aquatic Sciences 51:2715–2720.

Secor, D. H., T. M. Trice, and H. T. Hornick. 1995. Validation of otolith-based ageing and a comparison of otolith and scale-ageing in mark-recaptured Chesapeake Bay striped bass, *Morone saxatilis*. Fisheries Bulletin 93: 186–190.

Shepherd, G. R., and N. Lazar, editors. 1998. Source Document to Amendment 5 to the Interstate Fishery Management Plan for Atlantic Striped Bass. ASMFC Fisheries Management Report 34.

Sims, S. E. 1984. An analysis of the effect of errors in the natural mortality rate on stock-size estimates using virtual population analysis (cohort analysis). Journal du Conseil, Conseil International pour l'Exploration de la Mer 41:149–153.

Smith, D. R., and X. He. 1998. An overview of methods to estimate annual mortality of Atlantic Striped Bass from tag-recovery data. In G. R. Shepherd and N. Lazar, editors. Source Document to Amendment 5 to the Interstate Fishery Management Plan for Atlantic Striped Bass. ASMFC Fisheries Management Report 34.

Smith, D. R., K. P. Burnham, D. M. Kahn, X. He, C. J. Goshorn, K. A. Hattala, and A. W. Kahnle. 2000. Bias in survival estimates from tag-recovery models where catch-and-release is common, with an example from Atlantic striped bass (*Morone saxatilis*). Canadian Journal of Fisheries and Aquatic Sciences 57:886–897.

Trippel, E. A., M. J. Morgan, A. Frechet, C. Rollet, A. Sinclair, C. Annand, D. Beanlands, and L. Brown.

1997. Changes in age and length at sexual maturity of Northwest Atlantic cod, haddock and pollock stocks, 1972–1995. Canadian Technical Report of Fisheries and Aquatic Sciences 2157.

Trippel, E. A. 1999. Estimation of stock reproductive potential: history and challenges for Canadian Atlantic gadoid stock assessments. Journal of Northwest Atlantic Fisheries Science 25:61–81.

Vecchio, V. J., and H. O'Riordon. 1999. A study of striped bass in the Marine District of New York state-ocean haul seine study. NMFS Completion Report, 1997–1998. New York Department of Environmental Conservation, Division of Marine Resources, Stony Brook, New York.

Waldman, J. R., and Fabrizio, M. C. 1994. Problems of stock definition in estimating relative contributions of Atlantic striped bass to the coastal fishery. Transactions of the American Fisheries Society 123(5):766–778.

Waldman. J. R., R. A. Richards, W. B. Schill, I. Wirgin, M. C. Fabrizio. 1997. An empirical comparison of stock identification techniques applied to striped bass. Transactions of the American Fisheries Society 126(3):369–385.

Wirgin, I. J. R. Waldman, L. Maceda, J. Stabile, and V. J. Vecchio. 1997. Mixed-stock analysis of Atlantic coast striped bass *(Morone saxatilis)* using nuclear DNA and mitochondrial DNA markers. Canadian Journal of Fisheries and Aquatic Sciences 54(12):2814–2826.

Young-Dubovsky, C., G. R. Shepherd, D. R. Smith, and J. Field. 1996. Striped Bass Research Study. Report for 1994 U.S. Department of Commerce Report to Congress. Washington, D.C.

Appendix I. Indices of abundance used in tuning the striped bass VPA

Table A1. Young of year indices (lagged to January 1).

Year	Hudson River	Delaware	New Jersey	Maryland	Virginia
1982	8.86	–	0.00	0.59	1.57
1983	14.17	–	0.12	3.54	2.71
1984	16.25	–	0.03	0.61	3.40
1985	15.00	–	0.29	1.64	4.47
1986	1.92	–	0.02	0.91	2.41
1987	2.92	–	0.28	1.34	4.74
1988	15.90	–	0.41	1.46	15.74
1989	33.46	–	0.35	0.73	7.64
1990	21.35	0.42	1.03	4.87	11.23
1991	19.05	0.11	1.00	1.03	7.34
1992	3.60	0.18	0.47	1.52	3.76
1993	11.43	1.13	1.19	2.34	7.32
1994	12.59	1.14	1.78	13.97	18.12
1995	17.64	0.19	0.96	6.40	10.48
1996	16.23	0.42	1.98	4.41	5.45
1997	8.90	1.36	1.70	17.46	23.05
1998	22.30	0.14	1.01	3.91	9.35

Table A2. Long Island ocean haul seine abundance indices (number per haul).

	Age										
Year	5	6	7	8	9	10	11	12	13	14	15+
1988	8.71	7.71	2.89	1.13	0.28	0.15	0.00	0.01	0.00	0.01	0.00
1989	4.86	4.49	2.65	0.90	0.45	0.13	0.07	0.02	0.00	0.07	0.10
1990	1.27	2.03	1.42	1.18	0.32	0.09	0.11	0.02	0.01	0.01	0.05
1991	4.38	1.93	2.12	1.60	1.28	0.47	0.19	0.04	0.00	0.01	0.01
1992	5.12	1.64	0.77	1.05	1.46	0.80	0.36	0.09	0.08	0.02	0.10
1993	3.58	1.93	0.62	0.41	0.70	0.63	0.41	0.16	0.09	0.01	0.08
1994	6.65	2.75	1.80	0.74	0.46	0.57	0.45	0.31	0.12	0.04	0.05
1995	3.22	3.15	1.91	1.29	0.56	0.56	0.58	0.28	0.32	0.06	0.08
1996	2.34	0.70	0.76	0.39	0.21	0.14	0.16	0.14	0.05	0.06	0.02
1997	7.70	2.97	0.96	0.83	0.37	0.16	0.12	0.01	0.11	0.01	0.01
1998	34.96	5.40	1.57	0.34	0.36	0.37	0.17	0.06	0.13	0.00	0.05

Table A3. Maryland spawning stock gillnet survey CPUE (number per set).

	Age													
Year	2	3	4	5	6	7	8	9	10	11	12	13	14	15+
1985	72.83	243.05	41.79	19.02	8.88	8.25	1.44	1.83	2.19	0.39	1.74	1.31	0.31	7.01
1986	62.72	164.74	467.30	7.10	4.44	3.16	2.63	0.94	0.73	0.00	0.00	0.94	0.65	2.22
1987	60.93	204.10	128.14	335.33	3.72	2.95	3.48	0.12	0.00	0.00	0.00	0.00	7.25	4.94
1988	32.21	67.75	73.47	72.33	107.36	2.16	0.00	0.00	0.73	0.00	0.02	0.00	0.08	1.86
1989	15.52	121.59	100.51	71.51	91.10	59.62	0.38	0.00	0.37	0.00	0.19	0.00	0.00	0.34
1990	25.63	182.33	204.18	88.67	68.95	67.02	52.92	0.46	0.18	0.02	0.24	0.26	0.05	0.39
1991	40.31	186.40	72.20	68.43	40.60	38.94	35.44	14.97	0.43	0.30	0.00	0.11	0.10	0.45
1992	17.40	240.64	199.49	63.24	84.36	59.62	41.81	19.13	8.79	0.15	0.00	0.03	1.09	0.72
1993	33.40	130.16	222.42	98.53	60.37	57.34	46.52	22.28	7.92	3.27	0.33	0.31	0.46	0.35
1994	11.12	37.90	67.64	98.17	37.34	20.90	30.06	12.22	3.34	0.63	0.36	0.09	0.00	0.05
1995	42.90	110.10	71.81	72.13	56.29	49.63	33.55	41.38	17.83	24.86	8.21	2.14	0.00	0.34
1996	8.38	510.92	140.80	47.65	93.05	109.70	85.01	66.80	34.79	16.59	5.05	1.66	0.00	0.00
1997	33.46	27.73	181.71	63.84	29.59	32.17	41.79	32.57	22.26	8.80	6.69	3.38	0.44	0.00
1998	19.98	31.89	209.97	112.30	36.61	20.82	25.17	21.87	15.90	16.45	4.79	2.49	0.47	0.00

Table A4. Massachusetts CPUE (number per trip) from commercial hook and line fishery.

	Age							
Year	8	9	10	11	12	13	14	15+
1991	0.56	2.30	0.92	0.43	0.28	0.12	0.44	1.95
1992	0.86	3.51	4.99	0.92	0.31	0.31	0.07	2.04
1993	0.38	3.17	5.89	4.78	0.51	0.22	0.06	0.99
1994	0.19	1.97	6.41	8.59	5.33	0.86	0.17	0.50
1995	0.43	3.74	9.74	6.26	2.18	1.03	0.10	0.53
1996	1.13	5.62	9.13	6.75	2.84	1.08	0.27	0.11
1997	0.90	4.81	6.12	5.58	4.68	2.47	0.75	0.41
1998	1.28	9.48	10.36	7.68	5.40	3.00	1.80	1.00

Table A5. Connecticut volunteer angler CPUE (number per trip).

	Age													
Year	2	3	4	5	6	7	8	9	10	11	12	13	14	15+
1982	0.22	0.32	0.16	0.14	0.11	0.06	0.03	0.02	0.02	0.01	0.00	0.00	0.00	0.00
1983	0.33	0.21	0.11	0.09	0.08	0.04	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00
1984	0.40	0.19	0.08	0.04	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1985	0.12	0.33	0.23	0.14	0.05	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1986	0.06	0.31	0.22	0.12	0.09	0.04	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00
1987	0.08	0.20	0.47	0.45	0.18	0.05	0.01	0.05	0.02	0.00	0.00	0.00	0.00	0.01
1988	0.03	0.24	0.34	0.20	0.14	0.06	0.04	0.03	0.03	0.01	0.00	0.00	0.00	0.00
1989	0.02	0.52	0.28	0.18	0.15	0.12	0.05	0.03	0.01	0.00	0.00	0.00	0.00	0.00
1990	0.27	0.48	0.47	0.16	0.18	0.13	0.09	0.03	0.01	0.01	0.00	0.00	0.00	0.00
1991	0.17	0.58	0.55	0.27	0.12	0.13	0.15	0.13	0.05	0.02	0.01	0.01	0.00	0.01
1992	0.15	0.67	0.43	0.35	0.14	0.07	0.09	0.13	0.09	0.03	0.01	0.00	0.00	0.00
1993	0.17	0.48	0.57	0.29	0.23	0.11	0.09	0.16	0.15	0.09	0.02	0.00	0.00	0.00
1994	0.07	0.70	0.62	0.49	0.28	0.22	0.09	0.08	0.11	0.10	0.05	0.01	0.00	0.01
1995	0.21	0.61	0.88	0.46	0.57	0.35	0.23	0.16	0.19	0.14	0.07	0.05	0.01	0.00
1996	0.60	1.20	1.34	0.59	0.59	0.32	0.18	0.19	0.19	0.12	0.05	0.02	0.01	0.00
1997	0.47	1.09	2.39	0.90	0.84	0.37	0.59	0.37	0.23	0.10	0.08	0.10	0.02	0.02
1998	0.18	1.11	1.28	1.64	0.58	0.31	0.23	0.21	0.12	0.06	0.07	0.13	0.03	0.04

Table A6. Hudson River commercial shad gillnet CPUE (number per haul).

	Age		
Year	6	7	8
1982	0.012	0.023	0.010
1983	0.147	0.273	0.116
1984	0.443	0.110	0.005
1985	0.085	0.069	0.026
1986	0.248	0.080	0.031
1987	0.413	0.234	0.055
1988	0.536	0.327	0.164
1989	1.278	0.562	0.309
1990	0.856	0.682	0.365
1991	0.477	0.382	0.382
1992	0.707	0.200	0.354
1993	2.172	0.790	0.559
1994	1.755	1.047	0.277
1995	0.869	0.448	0.263
1996	0.000	0.000	0.000
1997	0.485	0.164	0.154

Table A7. Age-1 indices (data lagged to January 1).

Year	W. Long Island	Maryland
1982	–	0.02
1983	–	0.02
1984	–	0.32
1985	–	0.0
1986	0.61	0.15
1987	0.30	0.03
1988	0.21	0.05
1989	0.77	0.06
1990	1.73	0.15
1991	0.37	0.33
1992	1.24	0.19
1993	1.34	0.11
1994	0.72	0.19
1995	1.37	0.76
1996	1.26	0.12
1997	1.52	0.07
1998	0.99	0.26