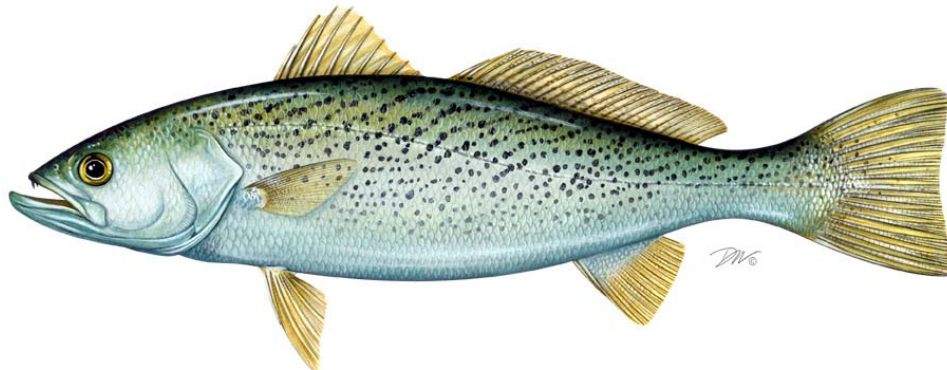


Atlantic States Marine Fisheries Commission

2006 Weakfish Stock Assessment

Weakfish Stock Assessment Report
Terms of Reference and Advisory Report
Technical Committee Supplemental Material



December 2006



*Working towards healthy, self-sustaining populations for all Atlantic coast fish species
or successful restoration well in progress by the year 2015*

ACKNOWLEDGEMENTS

The Atlantic States Marine Fisheries Commission (ASMFC) thanks all of the individuals who contributed to the development of the weakfish stock assessment and the peer review of the assessment. The Commission extends its appreciation to the Weakfish Stock Assessment Peer Review Panel for its efforts in evaluating the stock assessment and developing the Terms of Reference and Advisory Report: Dr. Brian Rothschild, Panel Chair, University of Massachusetts - Dartmouth; Dr. Steven Cadrin, NOAA/UMass Cooperative Marine Education and Research Program; Dr. Steven Martell, University of British Columbia; and Dr. Ransom Myers, Dalhousie University.

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PREFACE

This document contains the following reports:

Section A – Stock Assessment

Weakfish Stock Assessment Report for Peer Review (Part 1)
(ASMFC Weakfish Stock Assessment Subcommittee, February 2006)

Weakfish Stock Assessment Report for Peer Review (Part 2) – An Ecological Assessment of Weakfish: Examination of Fishing and Trophic Effects on the Recent Stock Decline
(ASMFC Weakfish Stock Assessment Subcommittee, February 2006)

Section B – Peer Review

Terms of Reference and Advisory Report to the Weakfish Stock Assessment and Peer Review
(ASMFC Weakfish Stock Assessment Peer Review Panel, April 2006)

Section C – Supplemental Material Requested by the Weakfish Management Board

Implementation of 40th SARC Recommendations
(ASMFC Weakfish Technical Committee, June 2006)

Summary of Weakfish Technical Committee (TC) agreements and disagreements with the 2006 peer review and the record of implementing recommendations of the 40th SARC
(ASMFC Weakfish Technical Committee, July 2006)

Executive Summary: How Much Additional Discarding and Under-Reporting are Necessary to Cause the Decline in Weakfish?
(Uphoff, J., June 2006)

Executive Summary: Estimating at-sea discards of weakfish using NEFSC observer database
(ASMFC Weakfish Technical Committee, July 2006)

The weakfish population structure along the Atlantic coast: a literature review
(Munyandorero, J., July 2006)

Overview of the Weakfish Resource
(Uphoff, J., August 2006)

What's Next with Weakfish? Fishery Status & Board Requests (presentation)
(ASMFC Weakfish Technical Committee, August 2006)

EXECUTIVE SUMMARY

The first peer review of a weakfish stock assessment was conducted in 1996 by the Stock Assessment Review Committee (SARC) at the 26th Northeast Regional Stock Assessment Workshop (SAW). The Review Committee did not approve the assessment and recommended that future assessments be conducted using ADAPT VPA. (Assessments up to that time had employed CAGEAN and Extended Survivor VPA, among other catch-at-age models.) The subsequent assessment, including data through 1998, was peer reviewed at the 30th SAW/SARC, which concluded that results of the ADAPT VPA could be used to calculate biological reference points and that figures illustrating the expanded size and age composition of weakfish would be useful for developing management advice. However, the Review Committee did find a noticeable retrospective bias in the ADAPT results which overestimated stock size and underestimated fishing mortality in the last few years.

A stock assessment update was conducted in 2002 (with data through 2000) using the SARC-approved methodology. Although this assessment was not peer reviewed, the Weakfish Technical Committee (TC) expressed concern about a strong retrospective bias that may have resulted from some shortcomings in the assessment data and model.

In 2003, the Weakfish Stock Assessment Subcommittee (SAS) began preparation for a 2004 peer review through the 40th SAW. Model results using the previously approved SARC methodology still exhibited a strong retrospective pattern, and results from both ADAPT VPA and a biomass dynamic model portrayed the stock as at very high levels, carrying capacity in the case of the biomass dynamic model, with very low fishing mortality. In light of the fishery landings having dropped to record low levels, the model results were not deemed credible.

For these reasons, the SAS deemed the ADAPT VPA methodology as insufficient to characterize the weakfish resource and proceeded to investigate alternative assessment methods. Although the revised weakfish assessment was incomplete at the time of the peer review, the SARC agreed to review the work and provide guidance on issues that were impeding the progress of the assessment, such as the inconsistency between survey indices, some of which portrayed an increase in abundance, and fishery-dependent indices of abundance and catch-at-age, which had declined significantly.

The Review Committee agreed with the SAS that the results of the work in progress, although using the same approach as the SARC-approved assessment in 1999, were not suitable for management at present. The Review Committee indicated that they felt the problem was conflicting data, and expressed skepticism about the reliability of some survey indices, especially the Northeast Science Center Fall Survey. Recommendations from the SARC proved to be useful, and some were incorporated into the stock assessment. The assessment was also expanded to include some alternative approaches previously explored by the SAS in the 2002 update process.

The stock assessment was completed in February 2006 (See Section A). The assessment was submitted to the Commission and was evaluated through the Commission's external peer review process on March 20-22, 2006 in Providence, RI. The Peer Review Panel consisted of four

fisheries biologists with expertise in population dynamics and stock assessment methods. The stock assessment was reviewed relative to the following Terms of Reference:

1. Characterize commercial and recreational catch including landings and discards.
2. Review adequacy and uncertainty of fishery-independent and dependent indices of relative abundance.
3. Review the appropriateness of constant and variable natural mortality (M) estimates in the assessment.
4. Review the estimates of fishing mortality (F), spawning stock biomass, and total stock biomass for 1981-2004, and characterize the uncertainty of these estimates.
5. Review the estimated biological reference points, as appropriate.
6. Review stock projections.
 - Review the projection of impacts on the stock of recent estimated rise in fishing mortality.
 - Review the projection of stock response to reductions in fishing mortality given the estimated increase in natural mortality.
7. Make research recommendations for improving data collection and assessment.

The Panel reviewed the 2006 Weakfish Stock Assessment and did not endorse the recommendations within the assessment report regarding stock status of weakfish along the Atlantic coast (see Section B). The Panel identified several issues that required additional work or attention by the Weakfish TC before they would support its use for management purposes. In particular, the Panel had concerns regarding stock structure, age composition data, and fishery discards.

The Weakfish Management Board directed the Weakfish TC to address the issues identified by the Review Panel. Specifically, the Board tasked the TC with furthering investigating stock structure and discards; determining agreements and disagreements among the assessment report, the peer review panel report, and the 40th SARC report; and providing an account of the implementation of recommendations from the 40th SARC.

Final Results

In August 2006, the Weakfish TC provided a response to the Board tasks (see Section C). Based on these responses, the TC's analyses, and significant evidence, **the Board accepted the following five points for management use:**

- 1) the stock is declining,
- 2) total mortality is increasing,
- 3) there is not much evidence of overfishing,
- 4) something other than fishing mortality is causing the decline in the stock, and
- 5) there is a strong chance that regulating the fishery will not, in itself, reverse stock decline.

WEAKFISH STOCK ASSESSMENT REPORT FOR PEER REVIEW (PART 1)

February 2006

BY

THE ASMFC WEAKFISH STOCK ASSESSMENT SUBCOMMITTEE

Desmond M. Kahn, Chair
Jim Uphoff, Vice-Chair
Victor Crecco
Douglas Vaughan
Brian Murphy
Jeffrey Brust
Robert O'Reilly
Lee Paramore

Note: an external peer review panel did not endorse the recommendations within this assessment report. For more information see the Peer Review Report (Section B), and the Weakfish Technical Committee's responses to that report (Section C).

ACKNOWLEDGEMENTS

Brad Spear of ASMFC has been helpful in many ways. We appreciate the substantial work done by Harry Rickabaugh of Maryland DNR and Patrick Kilduff of the ASMFC staff in developing the catch at age and weight at age estimates. We thank Robin Cook, Peter Shelton, Norman Hall, John Casey and Terry Smith of the 40th SARC for their helpful comments and analysis. Alan Seaver of the North East Fisheries Science Center has been very helpful on ADAPT software issues.

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EXECUTIVE SUMMARY

This assessment covers the period of 1982 through 2003 for catch-at-age analysis and biomass dynamic modeling and 1982-2004 or, conditionally, 2005, for some other approaches. The last peer-reviewed assessment covered 1982-1998 (NEFSC 2000). In 2002, an update was conducted which covered 1982 – 2000.

Recreational landings have been in decline since 2000, while commercial landings have been in decline since 1998. From 2000 through 2003, commercial landings were the lowest in the period. The 2003 recreational landings were the lowest in the period; the previous low point was in 1993. These declines suggest that abundance has declined. One or two of the research trawl surveys, however, contradict the picture that abundance has declined to very low levels, but those surveys show evidence of large year-to-year effects in weakfish availability, which reduces their value as indicators of trends in relative abundance. All variants of the recreational CPUE indices have been declining since about 1998.

Cohort catch curves of the catch-at-age matrix indicated that mortality had been relatively high for cohorts from the 1980s, declined to low levels for year classes born in the early 1990s, but then increased to very high mortalities for year-classes produced from the mid-late 1990s. Since Z was about 1.4 for recent year classes, subtraction of an assumed constant $M = 0.25$ indicated that F has increased greatly for these cohorts to levels of 1.15 or so. This pattern in Z of a decline in the early 1990 year classes followed by an increase was significantly correlated with the pattern found in cohort catch curve estimates of the Delaware trawl survey indices of relative abundance.

As in past assessments using ADAPT, retrospective bias was present in all ADAPT runs, meaning estimates of variables such as F and stock size were unstable for recent years, changing as additional years of data are added or subtracted. These biases occurred in opposite directions when either survey indices or recreational CPUE indices were employed. Model configurations using recreational CPUE indices of relative abundance as tuning indices, however, showed more recent stability than runs without these indices. Depending on the tuning indices selected, stable estimates of F and biomass are available from 1982 through about 1997 to 2000. Assuming $M = 0.25$, estimated fully-recruited F dropped sharply in 1995. For run #14 using trawl indices for tuning, F dropped to 0.26 in 1995, then rose to 0.40 in 1996 and about 0.36 in 1997 and 1998. After that, retrospective bias makes estimates unstable. The estimates of F decline to 0.03 in 2003, however. For run #20 using the recreational CPUE indices, fully recruited F dropped to 0.32 in 1995, then increased to 1.06 in 2000, the last year with relatively stable estimates. For this run, in contrast to the run with trawl survey indices, estimated F climbs to 1.6 – 1.5 in 2002-2003. Estimated SSB from ADAPT climbed above the SSB threshold estimated in the currency of the 2002 assessment update of 14,400 metric tons in 1994 with the trawl survey indices (run #14) and increased to 31,628 metric tons in 1997, higher than any previous year. Later year's estimates are unstable, but climb to 125,843 mt by 2003. For the run tuned to the recreational CPUE indices (run #20), SSB approached the threshold from below in 1994 and 1996, but did not exceed it until 1996. SSB climbed to a peak in 1997 of 18,752 mt, then declined to 6,260 mt in 2000. Estimates for late years are unstable, but declines by 2003 to 2,573 mt, lowest by far in

the time series. The lack of stable recent estimates from ADAPT considerably reduces its value for management.

When trawl survey exploitable biomass per tow was used as an index of stock biomass, a biomass dynamic model indicated that the stock reached very high levels of 37,000 – 39,000 mt by 1997 and remained there through 2003, near carrying capacity. Estimated biomass-weighted F from this model was very low in recent years (0.11 – 0.17 between 1994-2000), then dropped rapidly to 0.03 by 2003.

Reviewers from the 40th SARC reviewed the assessment as work in progress prior to development of the ADAPT runs with the recreational CPUE indices. They concluded that the models tuned to the trawl surveys revealed problems with the survey data and recommended extensive analysis of this data. Following their recommendation, we subjected each index to cohort catch curve analysis and found that two of the surveys produced negative Z estimates for recent cohorts.

We were forced to investigate alternative approaches due to the instability of recent years' estimates of F and SSB from ADAPT, and the unrealistic estimates from the biomass dynamic model and ADAPT tuned to survey indices. These models estimated recent extremely high biomass and very low F , despite severe declines in fishery catches. As a result, we estimated trends in fishing mortality as relative F , calculated as annual catch divided by an index of relative abundance averaged over two years. The index was a version of recreational catch per trip in weight (including discards), which had the advantage of great geographical coverage in contrast to most of the survey indices. One valuable attribute of relative F analysis is that it does not make assumptions about the amount or constancy of natural mortality. The general trends in relative F were confirmed when trawl survey exploitable biomass indices were used in place of the recreational CPUE index.

Biomass-weighted relative F estimates from 1982-2004 closely followed the trend in ages 1-5 biomass-weighted F from the converged portion of the ADAPT VPA models. We then converted these relative F estimates to absolute values of F by scaling them to the VPA F estimates for 1987-1991. Biomass weighted fishing mortality (FWt) estimates (ages 1+) on weakfish rose steadily from about 0.41 in 1982 to peak levels in 1988 (FWt = 1.08). The FWt estimates remained relatively high (FWt range: 0.60 to 0.81) from 1989 to 1992 (Table 2), then declined steadily to below the 0.40 level in most years from 1993 to 2005. The 2004 and 2005 FWt estimates were around 0.25.

With time series of F and catch, we estimated stock biomass as catch/ F . Biomass was high initially (34,010 mt in 1982), remained relatively high in the mid-1980s, then declined sharply in 1989 and remained below 10,000 mt through 1993. Stock biomass then began to rebuild, reached a secondary peak in 1998 (28,858 mt) and declined steadily to levels similar to those of the early 1990s, with the lowest level of the time series in 2004 (5,739 mt). We then developed a time series of surplus production estimates which indicated some recent years with negative production.

An external Gompertz production model provided a good fit ($R^2 = 0.68$) to the biomass data with statistically significant ($P < 0.05$) r and K parameter estimates. The resulting overfishing thresholds (F_{msy} , B_{msy}) for weakfish of 0.32 and 25,259 mt, respectively, were similar in magnitude to previous estimates. However, this production model consistently over predicted weakfish biomass from 1998 to 2005 by 10 to 50%. A full Steele-Henderson (S-H) production model including striped bass as a major predator was fitted by nonlinear least squares regression and also by iterative re-weighted least squares regression. Both S-H models provided very good fits ($R^2 = 0.88, 0.94$) to the biomass and predation data with statistically significant ($P < 0.05$) r , K , c and A parameter estimates. Overall, the S-H model was not only a better fit to weakfish biomass than the basic Gompertz model, but it also estimated the r and K parameters with much higher precision, particularly from iterative re-weighting.

The resulting weakfish overfishing thresholds (F_{msy} , B_{msy}) from the nonlinear least squares S-H model for weakfish were 0.59 and 21,179 mt, respectively. The overfishing thresholds (F_{msy} , B_{msy}) based on the iterative re-weighted S-H model were 0.55 and 23,400 mt, respectively and had higher precision than the nonlinear least squares fit. Moreover, unlike the severe residual pattern evident from 1998 to 2004 in the Gompertz model fit, there is little if any systematic residual pattern from the S-H model fitted by iterative re-weighting. The biomass-weighted F estimates after 1995 were all well below the overfishing definition ($F_{msy} = 0.55$) from the S-H model. Weakfish biomass has been well below $B_{msy} = 23,400$ mt since 2000 despite low and stable fishing mortality rates from 1995 to 2005.

Total mortality (Z) on Atlantic coast weakfish has risen steadily since about 1998, whereas fishing mortality (F) rates based on relative F were relatively low and stable during this period. Since total mortality (Z) each year was the sum of fishing and natural mortality, the recent rise in Z and ensuing failure in weakfish surplus production were more likely due to a recent increase in natural mortality. Most of the statistical evidence given herein supports the predation hypothesis as the most reasonable explanation for the recent rise in M and failure in weakfish productivity. Although results from regression and production models alone do not demonstrate causality, recent empirical evidence is consistent with a predation hypothesis involving striped bass. In addition, the apparent emergence of a weakfish recruitment bottleneck between ages 0 and 1 makes stock rebuilding via the implementation of additional management measures a difficult task.

We developed projections of weakfish stock biomass under a range of F levels, using an external production modeling approach that did not require an assumption of ecological stability. Z was considered to be the negative counterpart of the intrinsic rate parameter (r') and the Schaefer biomass dynamic model was reparameterized with the intrinsic rate of increase parameterized as ($r' - Z$) to project biomass into the future under current F levels, a 50% reduction and a complete moratorium. Results indicated that stock decline would continue even with a moratorium. However, continued current levels of F were projected to drive the stock to extinction within 10 years, while a 50% reduction would merely extend the time to extinction. A moratorium would eliminate extinction, but only a decline in M would allow stock rebuilding.

Part 2 of this assessment contains analyses that further examine the role of finfish predation and inter-specific competition on the recent rise in weakfish natural mortality. Several hypotheses

are offered to better interpret the intricate trophic pathways between weakfish and several candidate finfish species of importance in the mid-Atlantic region, including Atlantic menhaden, spot and bay anchovy, Atlantic croaker and striped bass. Our findings demonstrate that the potential impacts of predation and interspecific competition on weakfish and other exploited finfish stocks should be integrated into fisheries models and rigorously tested as a potential alternative explanation to the overfishing hypothesis.

TERMS OF REFERENCE

1. Characterize commercial and recreational catch including landings and discards.
2. Review adequacy and uncertainty of fishery-independent and dependent indices of relative abundance.
3. Review the appropriateness of constant and variable natural mortality (M) estimates in the assessment.
4. Review the estimates of fishing mortality (F), spawning stock biomass, and total stock biomass for 1981-2004, and characterize the uncertainty of these estimates.
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7. Make research recommendations for improving data collection and assessment.

INTRODUCTION

The management unit for weakfish is the Atlantic Coast. The fisheries for weakfish were largely unregulated until the 1990s, although some states had their own regulations. Weakfish are subjected to estuarine and near-shore fisheries from a variety of gears as they moved into and out of their estuarine spawning grounds. The largest landings, however, have historically occurred in the North Carolina fisheries targeting the overwintering aggregations off the Carolina coast. The Atlantic States Marine Fisheries Commission developed a Fishery Management Plan in 1985, which was first amended in 1992, but these plans were not mandatory. In 1993, Delaware and New Jersey instituted some management measures prior to most other states, but coastwide management measures were not implemented until 1995 when Amendment 2 to the Weakfish Fishery Management Plan was implemented under a mandatory basis governed by the Atlantic Coastal Cooperative Fisheries Management Act. Amendment 4 was passed in 2002 and made some relatively minor changes to regulations.

Commercial management measures include a minimum size of 12” (certain gears in certain states excepted), minimum mesh sizes to achieve the minimum size limits and effort restrictions designed to achieve a 32% reduction in fishing mortality from a baseline time-period (1990-1992) representing an unregulated fishery. All states but North Carolina have seasonal closures. North Carolina has an area closure (waters south of Cape Hatteras) instead. The plan also mandated bycatch reduction devices for the South Atlantic shrimp fishery, which required testing to demonstrate a 40% reduction in bycatch by number or a 50% reduction in bycatch mortality.

For the recreational sector, Amendment 3 included a varied set of recreational minimum size and creel limit combinations designed to provide incentives for conservation, based on equivalency in the currency of spawning stock biomass. For example, current regulations allow a 6 fish creel limit if a state has a 12” minimum size, an 8 fish creel limit if the minimum size is 13” (33 cm), a 10 fish creel limit if the minimum size is 14” (35.6 cm), etc. Northerly areas of the range tend to have larger fish, on average. Whether these larger fish are simply the product of northern habitat, or are larger, older fish from southerly areas that migrate north in the summer is not clear. Minimum sizes in New York and southern New England are 16” (40.64 cm) for both recreational and commercial fisheries, while states to the south have 13” minimums for recreational and 12” minimums for commercial, although some states have varied their recreational regulations within the suite of options offered in the FMP.

Weakfish have historically waxed and waned, according to landings records. For example, Bigelow and Schroeder (1953) report that during the latter part of the eighteenth century they were “...well known in Massachusetts Bay. But they vanished so completely sometime prior to 1800 that when a stray specimen was taken at Provincetown in June 1838, it was sent to Boston for identification. In the second half of the 1800s, they again became abundant off southern Massachusetts by 1870. Around 1900, a significant fishery developed in the southern Gulf of Maine with landings of 45 mt, in which 2 kg weakfish were abundant. Bigelow and Schroeder stated that “This marked the commencement of a period of local abundance, which was entirely unexpected (for nothing like it had been experienced since the settlement of the country), and which (with its equally sudden eclipse) is perhaps the most interesting event in the history of the local fisheries...during the summer of 1903 that the traps at North Truro alone reported 280,000 pounds (116 mt).” In 1906, when landings had begun to decline, “...the Cape Cod Bay traps (excluding Barnstable, Chatham, Yarmouth and Dennis) reported almost half a million pounds for that year (208 mt). By 1910 only 907 pounds were reported from the Gulf of Maine. Weakfish were completely absent in most years since that time in landings reports from the Gulf through the period covered by their 1953 report. They discuss a hypothesis that “weakfish are plentiful when bluefish are scarce...”

Joseph (1972) reported that as of 1967, “...weakfish stocks have been at record lows for the past few years.” All the states north of Cape Hatteras collectively landed only 416 metric tons in 1967, whereas the “all-time high catch” was 14,969 mt in 1945. In the 1970s and early 1980s, large catches and large weakfish were common. Catch-per-unit effort estimates from Virginia data showed a large peak in 1936, not in 1945. This analysis indicated that weakfish had been continuously declining from 1945 to 1967. Joseph also cites Perlmutter et al. (1956) who stated that prior to 1910 the bulk of the weakfish catch was obtained north of Chesapeake Bay and were predominantly fish aged 3 and older, but by the 1920s, Virginia was taking more than half the

total catch. Virginia, however, as today, “has always depended on smaller and younger fish. This means that in 1945 the number of weakfish captured may have been as much as twice the number caught in 1908 when the landings were approximately the same by weight.” Joseph then raises the possibility that this shift to higher numbers of younger fish could have led to recruitment overfishing. He also reported that the abundance of weakfish larvae in ichthyoplankton tows in lower Chesapeake Bay in 1959-1963 was less than one per tow, whereas Pearson (1941) had collected 25-67 larvae per tow in 1929-1930. Joseph advanced a hypothesis that the decline in weakfish stocks was caused by the³ advent of widespread use of DDT beginning in 1945. It was used for mosquito control in salt marshes as well as for agriculture.

Several Mid-Atlantic states had thousands of weakfish per year entered in their recreational tournaments, where the minimum sizes were as high as 4.4 kg. In 1981, coastwide landings were 19,000 metric tons (Table 2). From 1982 through 1988, landings fluctuated from 10,000 to 14,000 tons. Landings declined beginning in 1989, reaching a nadir of 3600 tons in 1993 and 1994. Landings then climbed to a second peak during 1982-1988, declined rapidly in the early 1990s, rebuilt slightly through 1998, and finally declined to an all-time low by 2003.

Previous assessments

The stock assessment of weakfish was last updated in 2002 (Kahn 2002b) with data through 2000. That report focused on results from tuned virtual population analysis using ADAPT. The Weakfish Stock Assessment Subcommittee had also in hand several alternative approaches that we were exploring, including a relative exploitation analysis (Crecco 2002), biomass dynamic model (Uphoff 2002a) and Integrated Catch at Age model (separable VPA: De Silva 2002). The ASMFC Weakfish Management Board, however, requested that we update the ADAPT model, since that had been used for the 1999 stock assessment, which had been approved by the Stock Assessment Review Committee meeting at the Northeast Fisheries Science Center in late 1999 (NEFSC 2000).

The 2002 assessment update had a serious problem with severe retrospective bias in the ADAPT results. This problem was uncovered by retrospective analysis, where the model was run with the most recent year of data removed; this was done repeatedly going back for several years. The estimates of fully-recruited fishing mortality and spawning stock biomass for a given year were then compared over the runs to see if they changed as additional years of data were included. While it has often been stated that the most recent year’s estimates from VPA are the least certain, the retrospective analysis of the 2002 model showed that the uncertainty extended back for several years. A distinct pattern was apparent; as additional years of data were added, the estimate of F for a given recent year increased substantially. Conversely, estimated SSB declined by 50%. For example, when 1996 was the terminal year (all data after 1996 deleted), the estimate of fully recruited $F_{1996} = 0.17$. After the addition of four years of data through 2000, the estimate of F_{1996} rose to 0.32, an 88% increase. The estimate of $SSB_{1996} = 40,379$ metric tons (mt) when 1996 was the terminal year. When data through 2000 was included, the estimate of SSB_{1996} declined to 27,134 mt, which was a 33% decline. This bias extended back 7 years to the 1993 estimates. The estimated $SSB_{1993} = 12,164$ mt when 1996 was the terminal year. When data through 2000 was included, this estimate declined by 27% to 8,884 mt.

The assessment approved by the SARC had a similar retrospective bias which was discussed in the report (NEFSC 2000). Essentially the retrospective bias means that estimates of management parameters for recent years are unstable and hence unreliable. As part of the 2002 assessment process, Uphoff (2002b) presented a bias correction approach for the retrospective pattern, but such corrections are dependent on the assumption that the bias pattern in estimates of several years back will continue into the new terminal year estimates. Previous assessments had used untuned VPA, Cagelan and Extended Survivors VPA. The 26th SARC reviewed an earlier assessment in 1998 but did not approve the catch at age modeling effort in that assessment. The 26th SARC recommended that ADAPT be used for the catch-at-age analysis.

Review of work in progress by the 40th SARC

The 40th SARC meeting in November 2004 reviewed the current assessment as a work in progress, after we made it clear that we did not have a completed assessment in November 2004. One of the SARC reviewers commented on development of the catch-at-age matrix after discussing the need to estimate catch at age from different states with different minimum sizes and from the recreational data obtained from NMFS that, “Because of the nature of this fishery, this is a complex undertaking.” (Shelton 2004). The SARC agreed that “the current status of the assessment was insufficient as a basis for providing advice.” (Cook undated). The SARC found there were conflicting signals from the research trawl surveys which tended to indicate higher abundance versus the catch or catch-based indices, which indicated declining abundance. The SARC recommended analysis of the various surveys and exclusion of any that are not internally consistent or that portray unrealistic variation or strong year-to-year variation. The SARC also recommended that the assessment proceed by stating various hypotheses about stock status with implied management action, then evaluating which evidence and analyses support the various hypotheses. This approach of exploring alternative hypotheses dealing with conflicting data is also recommended by Hilborn and Walters (1992, pp. 491, 536) and Schnute and Hilborn (1993).

A decision table for weakfish management lays out 3 basic hypotheses: stock increase, stock stability and stock decline (Table 1A). The table lays out potential likely stock responses for each hypothesis to three general management approaches: restricting harvest, status quo and liberalizing harvest. In general, in the case of stock decline, restriction would pose the best chance of reducing or stemming the decline. In the case of stock stability, status quo seemed reasonable, and in the case of stock increase, liberalization would be reasonable, although not required. In contrast, liberalization or status quo may increase the chance of further stock decline, if that hypothesis is correct. We attempt to summarize the evidence supporting each hypothesis in Table 1B.

LIFE HISTORY

Weakfish are an important sciaenid species of the Atlantic coast with its primary range from North Carolina in the upper southeast through southern New England. It is also encountered south to Florida. Weakfish are primarily estuarine and inshore oceanic inhabitants who migrate seasonally. In the autumn, they migrate south to North Carolina or Virginia coastal waters, where they overwinter. In spring they move northward and inshore into estuarine spawning grounds including Pamlico Sound in North Carolina, Chesapeake and Delaware Bays and smaller coastal

estuaries up through Gardiners Bay on eastern Long Island, New York and as far north as Long Island Sound and Narragansett Bay in Rhode Island. The ASMFC Fisheries Management Plan considers weakfish to be a unit stock, based on genetic analysis (Graves et al. 1992). Recent work however, using geochemical composition of otoliths, found weakfish homed to their natal estuaries (Thorrold et al. 2001). This was the first published example of an estuarine spawner exhibiting homing behavior. Earlier tagging work performed in the 1930s also indicated that weakfish returned to their natal estuary (Nesbit 1954). These findings are not consistent with the unit stock hypotheses, and it should be pointed out that the genetic analysis cannot detect separate stocks if there is even a very minimal amount of mixing among stocks.

Surveys along the Atlantic coast indicate that estuaries provide feeding areas and spawning grounds for adult weakfish and are equally as important as nursery areas for juveniles. Adult weakfish are often found near the periphery of eelgrass beds, perhaps because weakfish feed on shrimp, other crustaceans, and small fish that are found near these grass beds. Spawning occurs in late May through the summer in the latitude of Delaware Bay and occurs inside the Bay. Spatial and temporal variation in juvenile weakfish diet has been observed in studies conducted in the Delaware Bay and Chesapeake Bay (Hartman and Brandt 1995, Greccay and Targett 1996, Latour et al. *in review*). In Delaware Bay, Greccay and Targett (1996) found mysid shrimp to dominate the diet of juvenile weakfish collected in 1986, while the bay anchovy dominated the diet of juvenile weakfish collected in the Chesapeake Bay in the early 1990s. Latour et al. (*in review*) examined the diet of weakfish from the Chesapeake Bay from 2002 to 2004 and found that mysid shrimp were an important component of the diet not only in juvenile weakfish but also for adults in contrast to earlier diet studies of Chesapeake Bay weakfish (Hartman and Brandt 1995). Part 2 of this report examines changes in reported weakfish diet in detail.

Older weakfish typically have been shown to become increasingly piscivorous with age, with Atlantic menhaden or other clupeids comprising a significant portion of the diet of older weakfish (Merriner 1975, Hartman and Brandt 1995). Recently, Latour et al. (*in review*) found mysids and bay anchovy to comprise a significant portion of the diet of all age weakfish in Chesapeake Bay, with Atlantic menhaden comprising only a small portion of the diet of age-5+ weakfish. Differences in the two studies were attributed to different sampling methods or temporal changes in the abundance of prey items between the time periods of the two studies. The low prevalence of other sciaenids, spot and croaker, in light of high commercial landings of those species was also noted (Latour et al. *in review*).

Maximum age recorded using otoliths was 17 years. The ASMFC Weakfish Technical Committee considers T_{max} to be 12 years. World record weight is 7.75 kg. Growth rate has slowed since 1990, to the point that mean lengths at age of adults are several inches shorter than they were about 1990 (Lowerre-Barbieri et al. 1995, Kahn 2002a). Weakfish weight at age plummeted in the 1990s by nearly half for 3-5 year-old weakfish. Ninety per cent of weakfish are mature at age one. Weakfish are indeterminate batch spawners (Lowerre-Barbieri et al. 1996). The 26th SARC recommended that $M = 0.25$ be used in modeling constant natural mortality.

FISHERY DESCRIPTION

Landings from 1950-2003 peaked in 1981 (Figure 1A). During the period from 1982-2004, landings were highest at the beginning, totaling about 12,500 metric tons in 1982. Landings declined in 1989 through 1993. An increase began in 1995, the year Amendment 2 was approved. Landings then peaked at above 5,000 metric tons in 1998. A decline began in 1999, which continued through 2004, which was a record low level of less than a thousand metric tons (Figure 1). Landings of weakfish by the commercial and recreational sectors have followed similar trends of strong declines in recent years (Table 2; Figures 1, 1A, and 2). This fact is strong evidence that the stock is declining. If recreational landings, for example, had declined at a much higher rate than commercial landings, we could suppose that recreational fishers had reduced efforts directed at weakfish, or that weakfish had changed behavior to make them less available to recreational anglers, specifically. Since landings by both sectors have declined in parallel, changes in directed effort or availability are much less likely reasons.

In the winter, overwintering aggregations off North Carolina are harvested by gill nets and fly net trawlers, and these landings are large enough that they cause North Carolina to exceed other states' commercial landings (Table 2). Other major states are Virginia and New Jersey. When weakfish move north and inshore into estuaries from Pamlico Sound, North Carolina through Narraganset Bay, Rhode Island, they are harvested by a variety of estuarine gears, including gill nets, pound nets and floating traps, seines and hand lines. When weakfish move south on the fall migration along the coast, the aggregations are targeted with trawls and gill nets. In 1999, for instance, gill nets caught the largest share of the commercial landings, 48%, while trawl landings were second at 27%, followed by pound nets at 20% and haul seine at 5% (Vaughan 2000).

Patterns in recreational total catch and harvest changed after imposition of minimum sizes and creel limits in 1995 (Figure 2a). After this time, total catch ($A + B1 + B2$ of the MRFSS) became significantly larger than harvest. This suggests that the regulations had a significant impact on recreational harvest, causing a large increase in live releases (discards).

There have been significant changes in reporting requirements for commercial landings in two major states, Virginia and North Carolina, during the period covered by this assessment, as well as a federal change. In 1993, Virginia implemented a commercial fishermen mandatory reporting system, whereby each harvester reports daily activity (harvest, effort) on a monthly basis. The previous system in Virginia had been a voluntary buyer-reporting system. North Carolina in 1994 mandated trip-level reporting by licensed dealers for all commercial landings. This program requires dealers to complete a trip ticket for each transaction with a fisherman. These reports must be submitted on a monthly basis to the North Carolina Division of Marine Fisheries (NCDMF). Previously, the state had a voluntary reporting system from 1978-1993 obtained through a NMFS/NCDMF cooperative statistics program. In addition, NMFS in 2004 implemented a mandatory electronic reporting system for all federally-permitted seafood buyers. The combined effect of these improvements in these reporting systems is a probable increase in accuracy in landings estimates.

North Carolina landings from four estuarine gears show declines over the period 1994-2003 (Figure 3a). Catch-per-unit effort indices of abundance from these gears suggest that the landings decline is from a decline in abundance, not a decline in effort (Figure 3b). Virginia estuarine pound net CPUE exhibits a similar decline. In contrast to the estuarine gears, CPUE of two

oceanic gears employed on the overwintering aggregations off North Carolina do not show a declining trend, although CPUE declined in the most recent years (Figure 3c). Because these gears target aggregations, such fisheries can maintain CPUE in the face of stock declines if fishers can locate remaining aggregations, particularly mobile gears such as fly net trawlers. Therefore, lack of a declining trend in these latter two indices should not be taken as indicative of stable abundance. In contrast, estuarine fisheries would usually be targeting weakfish in less aggregated distributions.

INDICES OF RELATIVE ABUNDANCE

Recreational catch per unit effort indices

Indices of relative abundance from recreational catch-per-unit effort were reinstated in this assessment. While tables of similar indices appeared in the last peer-reviewed assessment report (NEFSC 2000), the indices had not been used in the final run because the SARC advised that fishery dependent indices were less desirable than fishery independent indices. Problems have come to light with many of the available fishery independent indices during the course of this assessment (see below). An advantage of the recreational indices is that they have very broad geographic coverage, whereas most of the fishery independent indices are limited to the waters of one state. By using only part of the recreational fishery for the indices, the exact correspondence to the catch at age matrix is reduced.

We have developed a measure of effort that seems improved over previous estimates. Previously, the effort was termed directed effort, defined as trips that either caught weakfish or for which anglers stated in interviews that they were targeting weakfish. This approach has the danger of exhibiting hyperstability (Hilborn and Walters 1992), because as weakfish catches decline, the number of trips defined as directed will also decline. The resulting index tends to remain relatively high because the divisor (number of directed trips) falls off rapidly. Consequently, the index may not decline at the rate that abundance declines. Our new indices are based on either total catch or harvested catch per trip, where the trips are those that caught one of a suite of species that are significantly correlated with weakfish in anglers catches. If weakfish abundance declines, anglers will continue to fish for the co-occurring species. Consequently, the number of trips will not decline due to a decline in weakfish catches, and the indices are unlikely to exhibit hyperstability.

The indices are based on catches and trips in the private-rental boat mode in state waters in the Mid-Atlantic region. This component, while comprising a majority of the catch in the Mid-Atlantic region, which is the dominant region for recreational catch, is still only a portion of the total recreational catch. One index is the total catch per trip, including discards. Such discards are not aged, so this index is considered an age-aggregated index over ages 2 – 6+. We have also developed harvest-at-age indices which consist of the total recreational harvest ($A + B1$) of the private-rental boat mode in Mid-Atlantic state waters, divided by the total number of trips that caught one of the suite of co-occurring species. The total harvest was divided into harvest at age by multiplying it times the proportion at age of the total Atlantic coast recreational harvest per year.

Recreational indices of abundance show relatively high but erratic levels in the 1980s, a decline into the early 1990s, then some recovery into the mid-late 1990s, particularly by the total catch per trip. Following the imposition of recreational minimum size limits and bag limits in the early-mid 1990s, the total catch per trip began to increase and diverged from the total harvest per trip, presumably due to a change in selectivity caused by the new regulations. Selectivity of the harvest per trip age-structured indices presumably changed in the early-mid 1990s. After the increase of indices through about 1997, they leveled off and began to decline in 1997 to 2000, depending on the index (Figure 3e). Indices declined to very low levels by 2003. Figure 3d portrays harvested weight per trip. Fig 3e portrays the tuning indices used in ADAPT. Total harvest at age was lagged forward when used in ADAPT to tune abundance the following January 1. Total catch per trip, including discards was not lagged. A third set of recreational tuning indices was developed by Crecco (2005) and used to estimate relative abundance for relative F calculation (Figure 3f). These indices are from MRFSS data on private boats trips in the Mid-Atlantic, either total weight caught including discards (discard mean weight assumed equal to harvested mean weight) per trip, using total trips, or total number caught, including discards per trip.

Research trawl surveys of relative abundance

After the 30th SARC recommended that the weakfish assessment use only data from the core area of abundance (NEFSC 2000), defined as New York through North Carolina, the SAS pared down the number of age-structured surveys employed to four: the NMFS North East Fishery Science Center (NEFSC) fall inshore survey which has run continuously since 1975, the New Jersey Division of Fish, Game and Wildlife ocean trawl survey (NJ) conducted from 1989 through the present, the Delaware Division of Fish and Wildlife's 30 ft. trawl survey of Delaware Bay (DE) conducted intermittently from 1966 but used here from 1982-1984 and 1991-2003, and the South East Monitoring and Assessment Program (SEAMAP) conducted from 1989 through the present (Table 3). We also use six indices of age 0 abundance which we lag and tune to age 1 abundance that are conducted by the New York Department of Environmental Conservation in Peconic Bay, the Delaware Division of Fish and Wildlife's juvenile trawl survey in Delaware Bay, one conducted by the Maryland Department of Natural Resources in Chesapeake Bay and one conducted in the Maryland Coastal bays, one conducted by the Virginia Institute of Marine Sciences in the Chesapeake Bay and one conducted by the North Carolina Division of Marine Fisheries in Pamlico Sound. We also use a survey of 1 year old abundance conducted by the North Carolina Division of Marine Fisheries in Pamlico Sound that we lag and use to tune abundance estimates of age 2 (Table 4). Shelton in his report from the 40th SARC in 2004, stated that "a very important point in the context of the 2004 weakfish assessment" was made in Crecco's (2002) relative exploitation analysis using data through 2000, that "tuning indices that display high frequency fluctuations in relative abundance (large year effects) are likely prone to excessive measurement error, poor reliability and low information content about stock variability". Because of these concerns, we developed an objective screening and evaluation analysis for the four trawl surveys, namely catch curve estimation of year class total mortality. The criteria for inclusion was that a survey did not display negative Z, that is, that a survey not indicate that year classes grow in abundance throughout their life. In fact, year classes can only decline in number throughout their life.

Here we focus on the four age-structured surveys, as these have the major influence on abundance and F estimates for both the ADAPT model and the biomass dynamic model based on survey indices of relative abundance (Uphoff 2005a). The only survey that was conducted continuously from 1981-2003 is the NEFSC fall inshore survey. While this survey covers the region from Cape Cod to Cape Hatteras, the NEFSC has recently revised the weakfish index, using only tows inshore from New Jersey through Cape Hatteras. The mean total catch per tow and exploitable biomass per tow (Uphoff 2005a) from 1981-2003 peaked in 2003 (Figure 4). This peak in 2003 also appears in the indices for ages 1-3 (Figure 5). Due to lack of precision in fitting indices of older ages, the SAS uses only ages 1-4 of this survey. As part of the assessment, we used catch curve analysis to estimate the instantaneous total mortality for each year class appearing in each survey. The NEFSC survey Z estimates showed biologically impossible results, namely that total mortality was negative, indicating a year class gained members during its lifetime. Therefore, while we used this survey during our initial ADAPT run to be comparable to the 2002 ADAPT run, we excluded it from further runs as biased and biologically implausible. The probable cause of this bias is large effects of year-to-year changes in availability of weakfish to the survey. This survey comes down the coast once in the fall. If it coincides with the fall migration of weakfish, catches are likely to be large, and vice versa. The migratory nature of weakfish produces serious difficulties for a coastal survey that is run once a year, such as the NEFSC survey.

The New Jersey coastal trawl survey index has exhibited saw-toothed fluctuations in recent years with no trend (Figure 6). The indices by age indicate that the relatively high 2004 value is primarily due to age 1, but age 3 and age 4 are relatively high also. When the survey indices were analyzed as year class catch curves, the survey had a recent year class, 2000, that showed negative Z, like the NEFSC survey. The 2000 year class had only 2 ages in the survey, ages 1 and 2, but they also show a positive slope. This survey then, in recent years, also exhibits estimates that are biologically impossible. On that basis, we eliminated the New Jersey survey from the second set of ADAPT runs.

Moving down the coast, the Delaware survey shows a different pattern than the NEFSC and the New Jersey surveys. The total catch per tow and the index of exploitable biomass peaked in 1996 and have generally declined since then, with some fluctuation (Figure 8). The indices by age have been high recently in age 1, followed by age 2 (Figure 9). Older ages have declined, however. In fact, age truncation is occurring in the indices: Age 6 declined to 0 in 2002 and 2003, while age 5 declined to 0 in 2003. This truncation is the reverse of the build-up of the age structure that occurred in the early 1990s. In 1991, the oldest age present was age 3. Older ages appeared gradually until Age 8 appeared in 1999. By 2001, though, the age structure began to truncate. Total mortality estimates by catch curve analysis found no year classes displaying negative Z in this survey (Table 3). The year class mortality estimates show a decline to a nadir for the 1993 year class, which was present from 1994-2000. Total instantaneous mortality then increased to levels of 1.5 or higher for the 1996-1999 year classes, declining slightly for the 2000 year class.

The final age-structured survey is the SEAMAP survey, from which we use tows along the North Carolina coast only. This survey's mean catch per tow increased to high levels in the 2003 and 2004 indices (lagged 2002 and 2003 fall indices). The age-structured indices for ages 1 and 2 are

the source of the increase in the last 2 years, with ages 3 and 4 showing very low and relatively flat values (Figure 9a). Ages 5 and 6 from this survey have been eliminated due to low precision, as for the NEFSC survey. Catch curve estimates of total instantaneous mortality from the SEAMAP survey found no instances of negative Z (Table 3). Since only 4 ages were available, little information about trends and precise estimates of Z can be obtained from this survey. Since all estimates of Z were positive, however, this index was judged to be biologically plausible to some extent and was included in the second cut of ADAPT runs.

DEVELOPMENT OF CATCH-AT-AGE AND WEIGHT AT AGE MATRICES

Prior to the 1990s, most ageing of weakfish was based on scales. Otoliths were used increasingly in the 1990s. They were judged to be clearer to read. The Technical Committee conducted comparative studies using otoliths and scales from the same fish and determined that different age patterns of age structure were obtained with the otoliths versus scales. The committee decided to convert older scale-based ages to otolith equivalents, using a procedure akin to age-length keys. So for a given scale age, certain proportions were assigned to several otolith ages, based on scale-otolith data. For some years now, only otolith ages have been accepted.

The catch-at-age matrix for 2001-2003 was developed using the same general procedure outlined in Kahn (2002b). Four age-length keys were developed for each year, following the procedure used by Vaughan (2000), a north early (first six months), north late (second six months), south early and south late. All age-length data available from commercial samples or research surveys was pooled into these keys. Sample sizes are presented in Table 4. The overall number of samples per metric ton landed is higher for commercial than for recreational landings. The distribution of samples among regions, however, shows that the northern areas of the range have been deficient in sampling. Few commercial samples have been conducted north of Delaware. In recent years, Rhode Island has obtained length and some age samples from its floating trap fishery; these samples indicated a striking difference from the size structure of the more southerly landings (Figure 3d). In New York and southern New England, the minimum size for recreational or commercial harvest is 16" (40.6 cm), while New Jersey and south have minimum sizes of 13" (33 cm) for some New Jersey gears some of the year, or 12" (30.5 cm) for commercial catches in New Jersey's other gears and all states south of New Jersey (Virginia and North Carolina have some estuarine gears with no minimum size for landings). No state south of New York has a 16" minimum size for recreational harvest, because many fish are caught below this size limit. The Rhode Island commercial length frequencies and the higher minimum sizes in the north indicate that northern areas tend to produce larger fish, whether due to migration of larger southern fish north or for some other reason. Consequently, substitution of southern length frequency distributions for northern commercial harvest may bias the estimated catch at age towards younger fish. While we truncated the lfd's at the 16 inch minimum size for years 2000-2003, this procedure was not followed for previous years.

Recreational catch at age was developed as in Vaughan (2000), where for each year, early and late catches were divided into geographic regions, a north and south region. Therefore four length frequency distributions were obtained from the Marine Recreational Fishery Statistical Survey (MRFSS) for each year: north early, north late, south early and south late. The corresponding age-length keys were applied to each of the length frequency distributions to

convert length frequencies into proportions at age. These proportions at age were multiplied by the estimated total recreational losses. These losses were estimated as the MRFSS estimated total number harvested ($A + B1$) plus 20% of the estimated discard numbers ($B2$ portion of the MRFSS estimates). The Weakfish Technical Committee has in the past estimated recreational discards to undergo a mortality rate of 20%. Recent discussions have concluded that the committee may change this mortality rate estimate to 10% based on recent experimental results, but we have not implemented this change yet. Although discarded weakfish may tend to be smaller than harvested weakfish since 1995 due to imposition of minimum size limits, we currently have no data with which to estimate lengths of discarded weakfish. Consequently, we assume the discards have the same length distribution as harvested weakfish.

Commercial catch at age was developed by half year, state and gear due to different regulations among states and different size selectivity among gears. Gears employed fall into five main categories: trawl, gill net, pound net, haul seine and long haul seine and hook and line (handlines). Gears were sometimes combined or substituted, depending on availability or lack of sample data for a particular gear. For the north region, length data from Virginia, Delaware and Rhode Island were developed into gear-specific length frequency distributions for gill net, haul seine and pound net/floating trap gear. Maryland contributed trawl length samples for late 2003 and some pound net lengths. Virginia pound net data was used for trawl length frequency distributions in many states, due to severe lack of sample data from trawl landings. Some trawl length frequency distributions were obtained from the NMFS At-Sea Observer database. MRFSS recreational length frequency distributions were used to estimate commercial hook and line catch lengths. For New Jersey and New York catches, Virginia, Delaware and Rhode Island length frequencies by gear were used, but we truncated them to account for 16 inch minimum size limits in New York and the 13 inch minimum size for portions of the New Jersey landings. Catch at age matrices were developed for each gear in each state (where applicable) for each half of the three years 2001-2003.

To allocate landings among lengths, lengths were converted to weight using a length-weight equation for each half-year, developed from length-weight data supplied by the states (H. Rickabaugh, MD DNR personal communication). The frequencies at length then became the frequencies for each mean weight. Frequencies at weight were then converted to proportions at each length-weight combination. The total landings in pounds for a gear-state-season were multiplied by these proportions to give the landings in pounds at each length-weight combination. Landings at each length were then divided by the mean weight at that length to convert from pounds into numbers of fish at length, giving catch at length in numbers. The catch at length was then converted into the catch at age by means of an age-length key.

For the first time, the catch at age matrix included estimates of commercial discards (de Silva 2004). The Subcommittee learned that data on weakfish discards was included in some of the NMFS at sea observer data, which began in 1994. These data were developed into several different matrices of estimated discards using different methods and assumptions. The primary gear that produced weakfish discards was trawl gear. Smaller numbers of discards were produced by gill net. Small sample sizes and spotty coverage in many years made estimation tenuous in many cases. After due consideration, the committee chose a method that scales up the ratio of discards to harvest of a reduced core suite of target species. That is, the ratio of at sea

observations of the number of discarded weakfish to pounds of harvested target species by gear (trawl and gill net) was scaled up to total discard estimates based on total landings of the target species. For example, if 50 weakfish were discarded per 1000 pounds of bluefish landed by trawl gear, we would get the total coastwide landings of bluefish by trawl gear, divide those landings by 1000, and multiply the quotient by 50 to estimate the total number of weakfish discards due to bluefish harvest with trawl gear for that year. For trawl gear, the most common target species were summer flounder, Atlantic longfin squid, horseshoe crabs, butterfish, bluefish and weakfish. For gillnet catches, the most common target species harvested with weakfish discards were weakfish, bluefish, Atlantic croaker, butterfish, spot, Atlantic menhaden and spiny dogfish. For a full explanation of the procedures used and problems encountered, see de Silva (2004). The primary reason for discarding weakfish in most years was regulations; the second most common reason was market conditions. For 1998-2000, however, market conditions were the most common reason for discarding weakfish.

We also explored an alternative to the ratio approach, which was a regression approach; however, it produced poorer estimates than the ratio method. We also explored an alternative to scaling up observed discards by total target species landings. This alternative is estimating discards per trip and scaling up to total discards by the total number of trips. Unfortunately, commercial trip data was not available for much of the range of weakfish.

One pitfall with our approach is the possibility of overcounting discards. For example, if weakfish discards occurred on a trip that harvested croaker and bluefish, we did not want to scale up these discards by both the bluefish landings and the croaker landings, for that would be double counting. We used a reduced suite of target species to avoid overcounting of discards. A principal components analysis determined species associations among the target species. That is, the analysis determined which groups of species were often harvested together (on the same trip). Then we could scale up the observed discards per unit harvest for one species from each group to avoid overcounting discards. We thus selected a reduced suite of core species by selecting only one target species from each association. For trawl gear, four target species groups were identified: 1) Atlantic long-fin squid, 2) butterfish, 3) weakfish, Atlantic croaker and bluefish and 4) summer flounder. For gill net, four groups were also identified: 1) Atlantic menhaden and weakfish 2) Atlantic croaker and butterfish 3) spiny dogfish and 4) bluefish and spot.

Since the observer database does not cover years prior to 1994, estimated discards were hindcast, assuming that regulatory discards only occurred after imposition of regulations in the early 1990s. That is, the proportion of discards that occurred in the database for market reasons was assumed to be occurring back to 1982. Thus estimated discards were lower prior to 1994. Completed commercial discards at age are most commonly ages 0, 1 and 2 (Table 4, from Table 47 of de Silva (2004)). Since the VPA does not analyze catches of age 0, the addition of commercial discards to the catch at age tends to primarily increase the catch of ages 1 and 2. Figure 10 shows the catch at age estimate for 2002 with and without commercial discards as an example.

The total catch at age matrix including recreational and commercial landings and discards is in Table 5. Because commercial discard data was only available through 2002, we averaged the commercial discard estimates for 2001 and 2002 and used these for 2003.

For our update of the input data from 2001-2003, the estimates of the catch weights at age were developed using the process suggested by Quinn and Deriso (1999, p. 305) for unbiased estimates, employing the age-length keys and an average weight-at-age that is weighted by the frequencies at length within ages. This involved constructing spreadsheets for every component of the catch that we had developed catch at age matrices for, namely semi-annually for each significant gear in each of the nine states with commercial landings, plus 4 recreational catch at age matrices per year. For the 3 years we updated, the total number of spreadsheets developed exceeded two hundred. The report of the 30th SARC (NEFSC 2000) listed the following suggestion as a research recommendation: “Obtain mean weights at age corresponding to the catch-weighted mean weight from the catch at age estimation process.” Previously, catch weights at age had been estimated with a von Bertalanffy model using all available age-length data, assuming catch occurred in the middle of the year. The problem with the old procedure is that it failed to take selectivity of the gear into account. It appears that we have fulfilled this SARC recommendation.

Catch weights at age for recent years showed a geographical trend, especially for younger ages. Mean weights at age tended to be larger in northern states (Figures 11-13). This could be due to higher minimum sizes in more northerly states, in part. The stock weights at age, or January 1 biomass weight at age, in contrast to the catch mean weight at age, is intended to depict the mean weight at age of the total stock, without fishery selectivity effects. The trend in these mean weights has been higher levels in the 1980s, when estimates are based on scale ages, and low points about 1995-1998, with recent increases (Figure 14).

On a related note, the 30th SARC report also made a research recommendation to “Investigate source of the relatively large sum of products correction factor.” It is unclear exactly what this refers to, but it may have referred to the product of the estimated catch at age and the estimated mean catch weight at age. The sum over ages of this product should match the total landings. There does not appear to be a large discrepancy currently except for two years in the 1980s (Figure 15).

CATCH-AT-AGE ANALYSIS

Cohort catch curves

The most basic analysis of the catch at age matrix consisted of catch curves by year class or cohort catch curves. The cohort approach avoids violations of the assumption of constant recruitment that occur when catch curve analysis is performed across the catches at age of one year (Ricker 1975). The standard methodology was followed, where the log of the catch at each age was plotted, and analysis included all ages to the right of the modal age (Table 6; B. Murphy, RI DMF personal communication). Figure 16 plots the resulting estimates of total instantaneous mortality, Z , and estimates of F based on the assumption that $M = 0.25$, which the Weakfish Technical Committee has assumed in the past. Since the Z and F estimates are for the lifespan of

a cohort, they are not values that apply to only one year, as in a VPA. Rather, they estimate mortality over the range of years analyzed, as stated in Table 6. The general pattern is that year classes of the 1980s had relatively high Z estimates, fluctuating around 0.8. A nadir was reached with the year classes of the early 1990s, where Z for the 1992 year class was 0.3. This year class would have reached age 3 when Amendment 2 took effect in April 1995. However, Z climbed abruptly for the 1994 year class and reached a plateau for the remaining cohorts, fluctuating without trend from 1.3 to 1.5. The general trend of Z estimates here is supported by a significant correlation with year class Z estimates from an independent source, the catch curve analysis of the Delaware trawl survey (Figure 17). If we assume $M = 0.25$, F would be 1.05 to 1.25. However, these F levels are not supported by other analyses. The fishery regulations were relatively stable during this whole period. This suggests that the increase in Z indicated here could have been due to an increase in M, rather than an increase in F in the late 1990s and early 2000s.

Gulland's cohort analysis

The catch at age matrix was subjected to a Gulland cohort analysis (Haddon 2001), which is essentially an untuned virtual population analysis. This can be seen as an analysis of the catch data. Because the analysis proceeds from the most recent years and moves backwards, the most recent years' estimates are least reliable, since it assumes that the ages in the most recent year have no survivors. Therefore, the last three to four years' estimates should not be considered reliable. Starting four or five years back, prior to 2000, however, the estimates have more validity. As opposed to catch curves, cohort analysis gives estimates of F at age and year, assuming $M = 0.25$ (Table 7), and estimates of stock size (Table 8, Figure 18).

We consider 1999 to be the most recent year of reliable estimates. Figure 18 indicates that stock size was relatively high in the early and mid-1980s, peaking in 1986 at about 120 million fish. The stock then declined to about 50 million from 1989 – 1992. The stock began to increase in 1993, reached a recent peak in 1994-1996 at about 75 million, then declined to about 30 million in 1999. If we assume M was constant at 0.25, fully-recruited F estimates (ages 4 and 5) are erratic in the 1980s, fluctuating greatly between 1.0 and 2.5. From 1989 through 1993, F estimates are more stable, ranging between about 1.50 and 1.80. The estimate of F declines in 1994 and 1995, to about 0.75 in 1993 and fluctuated between 0.75 and 1.00 without trend until 1999. The pattern of average Z for ages 1-5 is also plotted in Figure 18. This estimate fluctuates erratically from 1982-1988, then stabilizes from 1989 – 1993 at about 1.25. Estimated Z declines in 1994 and 1995, and then fluctuates around 0.75 through 1999. This decline from 1995-1999 seems consistent with a reduction in F due to implementation of Amendment 2, in April 1995. The increased Z estimates for the 1995-1998 year classes seen in the two catch curve analyses (Delaware survey and catch at age matrix) are not detectable here, but no reliable estimates of Z are available after 1999 from the cohort analysis, which covers much of the relevant period for the later year classes in the catch curve analysis.

The estimates of fully-recruited F for the period since Amendment 3 was passed are considerably higher than some other estimates, fluctuating between 0.75 and 1.00. Most of the shifts in F and Z seem consistent with changes in fishery management. During the period of the unrestrained fishery in the 1980s and early 1990s, F and Z estimates were high. Some states, primarily New

Jersey and Delaware, imposed restrictions beginning in the early 1990s. Amendment 2, passed in July 1994, went into full force in April 1995.

ADAPT results

ADAPT was employed to analyze the catch at age data, in conjunction with survey and fishery dependent indices of relative abundance to tune terminal year estimates (actually, ADAPT estimates abundance in the terminal year + 1 and F in the terminal year). The first ADAPT run presented here, run 24, is an update of the model configuration recommended by the 30th SARC and implemented in the 2002 update (Kahn 2002b). Run 24 differs from the earlier models in that it includes estimated commercial discards in the catch-at-age matrix, but it uses the same set of tuning indices: all four age-structured surveys discussed above, plus age 1 indices from New York, Delaware, Maryland (2 such surveys), Virginia (VIMS survey) and North Carolina (Table 12), plus an age 2 survey from North Carolina. The run results (Table 9A) showed strong retrospective bias, similar to those of earlier years, and as such, is not considered reliable for recent years. The retrospective pattern is that F is underestimated (Figure 19, Table 9B) and spawning stock biomass is overestimated (Figure 20, Table 9B). When only data through 1998 is entered the fully-recruited F estimate for the terminal year, 1998, was 0.14. Once five additional years of data were input through 2003, the 1998 F estimate rose by 207% to 0.43. Conversely, the 1998 SSB estimate declined by more than half from 59,102 MT with 1998 data to 22,599 MT with 2003 data. With this amount of bias, the results of the VPA do not appear to be reliable after the early 1990s. The estimated F in 1991 increases by only 2.5% with the addition of 5 years of data, while estimated SSB for 1991 declined by 3%. Basically, in this run, the survey tuning indices indicate the stock is relatively high, so the estimated stock size in the terminal year +1, 2004, is estimated as a high number, 206 million (Table 9A). The stock size is then back-calculated to the terminal year, 2003, assuming $M = 0.25$ and adding in the catch. Since the catch was low and estimated losses due to natural mortality are moderate, the estimated stock size in 2003 is not much bigger than that of 2004. That result indicates total mortality was low, so when Z was estimated, it is relatively low. The assumed constant value of $M = 0.25$ was subtracted from that low Z and F is then estimated as extremely low, only .012 for age 4 in 2003, for instance. The combination of a high estimate of stock size in 2004 due to the survey indices, the low catch in 2003, and the assumption that $M = 0.25$ produce high terminal year estimates of stock size and F.

The 40th SARC panel that reviewed this assessment as work in progress agreed that the combination of high survey indices and low catches produced inflated estimates of N and low estimates of F. The reviewers indicated that the conflict between the high trawl indices and the low catch produced the retrospective bias observed. They were critical of some survey indices, particularly the NEFSC indices. They recommended critical review and removal of surveys that did not give biologically plausible results. Following their review, we conducted the catch curve analysis of the surveys described above and eliminated the NEFSC and NJ surveys on the grounds that they showed negative mortality, or “spontaneous generation” (growth of a year class in numbers during its life, as opposed to constant decline in numbers). The next run, run 14, used only the two remaining age-structured surveys, the Delaware and SEAMAP (Figures 21, 22, Table 9A). This model run showed a more severe retrospective pattern than run 24, however, with the terminal year F estimate with data through 1998 ($F_{1998} = 0.08$), increasing by 450% with

the addition of five years of data to 0.44 (Table 9B). The estimated SSB for 1998 declined by 77% from the run with 1998 as the terminal year to the run with 2003 as the terminal year. Again, this indicates that F is probably greatly underestimated for recent years in run 14 and SSB is probably greatly overestimated.

Our next change was to include fishery-dependent indices of abundance as tuning indices. The Weakfish SAS had used recreational catch at age per unit effort as tuning indices in VPA prior to the 30th SARC. That committee recommended against using any fishery-dependent indices “because the SARC believed sufficient fishery-independent surveys were available” (NMFS 2000). However, we added recreational CPUE indices because the surveys did not give the model tuning information for estimation of terminal year abundance consistent with the fishery experience of drastically declining catches, except for some aspects of the Delaware survey. The 40th SARC agreed with this decision. These recreational CPUE indices are declining over the last several years.

Run 18 used the Delaware and SEAMAP age-structured indices and the age 1 and 2 tuning indices included in runs 24 and 13, plus recreational fishery harvest per trip for ages 3-6 (Table 9A). Since harvest changed due to minimum size and bag limits over the time series, we also added the unaged total recreational catch per trip (including discards – B2 component of MRFSS) as an aggregated index tuned to ages 2-6. The retrospective pattern in F and SSB of run 18 is similar to the previous two runs (Figures 23A, B), except that the final estimate of SSB for 1998 was approximately 2/3 of the final 1998 estimate of the two previous runs (Table 9B). Conversely, the final F estimate for 1998 was 50% higher than the estimates from the previous two runs. Not until we look back at the 1992 estimates have the estimates of F and SSB closely converged among these three runs, although they become roughly similar by 1995.

The DEDFW survey has a distinctive residual pattern in Run 18, however (Figure 24A). This survey had negative residuals in the early 1980s for younger ages, consistent with statements in reports from that period that recruitment was low. The survey resumption in 1990 began with negative residuals, followed by consistently positive residuals in the mid-1990s followed by negative residuals in recent years. The SEAMAP survey residuals are fairly random, but positive in 2003 and 2004 (Figure 24B). Recruitment indices residual patterns are consistent with a change in fishery selectivity around 1995 when mandatory compliance was instituted under Amendment 2. Residuals were largely negative prior to 1995 and then became positive, with several again becoming negative by 2002 (Figure 24C). The opposite pattern occurs with the recreational CPUE residuals (Figure 24D). In this case, residuals of the age-structured harvest per trip indices switched from positive to negative in 1995. This is consistent with a regulation-driven change in recreational selectivity, as minimum sizes and creel limits caused an increase in discarding, changing the catchability of the recreational fishery and consequently of these indices. Large negative residuals occurred for these indices for ages 4 and 5 in 2004. The exception to the general pattern is the total catch index pooled over ages (Figure 24D). That index has a fairly random pattern, as would be expected since total catch (including discards) should not be affected by creel and size limits.

Run 18 has considerably higher F estimates and lower SSB estimates than runs 13 and 24, such that the SSB estimates for 1999-2001 are below the biomass threshold in Amendment 4 of

14,421 MT. However, the estimated SSB for 2002 and 2003 increased to 23,567 MT by 2003 (Figure 23B, 2003 data). Again, however, this estimate is biased high and presumably will decrease substantially with additional data. The estimated F for 1996-2000 is much higher in run 18 than in runs 13 and 24. In the former, the F estimates range from 0.50 in 1996 up to 0.75 in 2000. In the latter two runs, these estimates range from about 0.3 up to 0.44. The estimated F for 1995 is almost always lower than that of 1996-2000. For run 18, $F_{1995} = 0.31$, while for run 14, $F_{1995} = 0.28$. That year saw the mandatory implementation of Amendment 2.

Run 20 is the final ADAPT run presented here. It used only the recreational CPUE indices for tuning of aged 2-6 as described for run 18 (Table 9A). This completed a combination of runs with all survey indices (run 24), a reduced group of trawl survey indices screened for biological plausibility (Run 14), a run with these two surveys plus the recreational CPUE indices (run 18) and the recreational CPUE indices alone (run 20). No index tuned the estimate of age 1. Run 20 also shows a strong retrospective bias, but in the opposite direction from the other runs (Figures 25, 26, Table 9B). This pattern is reversed for estimates when data through 2002 only is entered. Here F is overestimated in recent years and SSB is underestimated. When 1998 is the terminal year, $F_{1998} = 2.05$, but with data through 2003, it declined to 0.70, for a 66% decline. The estimate of SSB when 1998 is the terminal year was only 5,468 mt, but it increased to 12,674 with the addition of data through 2003. Again, by about 1993, the estimates have converged to rough similarity with those previous runs. Distinct patterns appear in the residuals for this run (Figure 26B). For the age-structured harvest-per-trip indices, initially residuals were positive. They became mostly negative in the early 1990s and remained so until 2002, when ages 3 and 4 became positive, followed in 2003 by ages 5 and 6. Large negative residuals did not appear in this run, with none exceeding an absolute value of 2.0. The residuals for the total catch including discards (ages 2:6) were negative from 1989-1997 and positive from 1998-2003. Some of these residual patterns are clearly related to the change in selectivity in the middle 1990s, but this relationship is not as clear-cut as in Run 18.

The reason that the choice of tuning indices affects the estimates back approximately 10 years to 1993 has to do with the way ADAPT calculates the F on the oldest true age. Because ages 6 and older are pooled into the so-called “plus group”, age 5 is termed the oldest true age in these analyses. VPA calculations move backwards through each cohort or year class, from more recent years when each cohort was older towards years farther back when each cohort was younger. In the case of the oldest true age, however, it is not correct to back calculate from a plus group, here age 6, to the oldest true age, here age 5. That is because the plus group contains members of more than one cohort, and to back calculate from plus group to oldest true age would inflate the age 5 numbers. Consequently, ADAPT uses the F estimate from the other fully-recruited ages as the F estimate for the oldest true age. In this case, we have only one other fully recruited age, age 4. Because we have only two fully recruited ages, either option in ADAPT to calculate this F on the oldest true, the average method and the Heincke method, arrive at the same number. So the F calculated for the cohort that is age 4 in a given year is used as the estimate of age 5 F, which age belongs to the previous cohort. This F estimate is used to calculate the abundance of age 5 with the catch data and catch equation (Anonymous 2003), $N = ZC/FA$, where A is total annual mortality as a percentage. Here Z is estimated assuming $M = 0.25$, so $F + 0.25 = Z$. Now that the abundance of the oldest true age in that year has been determined, the program back calculates the abundance of the same cohort in the previous year as age 4. The F estimated for age 4 is used

for the oldest true age of the previous cohort. Consequently, the effect of the tuned estimate of N in year $T + 1$ and F in year T propagates backward through a whole series of cohorts, in this case for about a total of 10 years.

The net result of the ADAPT models is that we have stable estimates from 1982 through about 1992 or 1993. More recent years have unstable estimates of fishing mortality and SSB which change dramatically as additional data is added. In previous assessments, this pattern was noted and discussed, but perhaps not emphasized as it should have been. For management purposes, we consider ADAPT results unreliable for parameters and trends after 1993. The estimates of F from 1982 through 1991 have proved useful, however. Crecco (2005) used them to scale relative F estimates to absolute F estimates. A cautionary note here is that the fact that the 1982-1993 estimates are stable does not necessarily mean they are accurate. Simulation work by Sinclair et al. (1991) found that the stable, converged estimates of VPA were not always accurate estimates. Since they created simulated data, they knew the correct estimates. In our case, we do not have that luxury. The combination of scanty data in the 1980s, even compared with more recent years, and the conversion of scale ages to otolith ages indicates considerable uncertainty in the estimated catch at age matrix during that decade.

In terms of the three biological hypotheses presented above, which are supported by the catch at age analyses presented above? Unfortunately, the ADAPT models are unhelpful in discerning recent stock trends. The catch curves by year class indicate that total mortality declined greatly for cohorts originating in the early 1990s, but then climbed up to high levels of about 1.4 since the 1995 year class (Figure 16). This estimate is a 75% increase over the Z estimates from the early 1980s. These estimates are averaged, essentially, over all ages in a year class, so they tend to be quite lower than the fully-recruited F estimates from VPA for the 1980s. The total mortality of 1.4 indicates that either fishing or natural sources of mortality have greatly increased in the later 1990s and early 2000s. This level of Z could produce stock decline and could be considered to support that hypothesis. These estimates are supported by the independent Z estimation from the catch curves developed from the Delaware survey indices, and that considerably strengthens the case for a recent large increase in total mortality (Figure 17). The cohort analysis assumes that instantaneous natural mortality is constant at 0.25. Given that assumption however, the estimated stock size through 1999 declines steeply from the 1994-1996 plateau (Figure 18). Although that is not very recent, it is consistent with a stock decline at least over the period 1997-1999. The estimated decline occurred despite the fact that the estimate of F declined significantly after 1994. The stock decline depicted, then, does not seem to be caused by an increase in F .

SIZE AND AGE STRUCTURE ANALYSIS

Amendment 4 requires that descriptors of age or size structure be reported to the Management Board as part of each stock assessment. Proportional stock density (PSD) is a standardized method for analyzing length-frequency data that quantifies size structure into categories of sizes of interest to recreational anglers (Uphoff 2005b). The Quality+ PSD (PSD Q+) equaled the proportion of weakfish greater than or equal to 210 mm (8.3 inches) that were 340 mm (13.4 inches) or larger in DE, NJ, and NEFSC trawl surveys. Sample sizes in the DE and NJ surveys

were large enough for precise estimates of PSD Q+, but sample sizes in the NEFSC survey were not and these data were excluded from analysis.

The PSD Q+ size quality indices for NJ (1989-2003) and DE (1966-1971, 1979-1984, and 1990-2003) indicated that size quality of weakfish in recent years was quite poor (Figure 27). A recovery in size quality after Amendment 3 (1996) faltered after 1998. Weakfish PSD Q+ has the potential to be a good bit higher than the peak observed in 1998. Values in the early 1980s were twice as high as this peak.

The DE PSD Q+ index was significantly and positively associated with recreational fishing quality (trophy citations) over a broad (DE, MD, VA) area, commercial and recreational yield along the Atlantic Coast, recreational harvest per trip in the mid-Atlantic (VA-NY), and the proportion of recreational harvest outside of 3 miles. Proportion of recreational harvest in bays and sounds was negatively associated with DE PSD Q+. These associations indicated that this size quality index was a highly relevant measure of population and fishery status that should meet the needs of the Weakfish Management Board. This analysis of size structure of the weakfish stock seems to be consistent with a stock in decline.

Our analysis of age structure assumes that the percentages at age from ADAPT output are not affected by retrospective bias. This assumption seems reasonable based on retrospective analysis of the 2002 ADAPT output (Kahn 2002). The mean bias for age 1, age 2, and age 3 weakfish in that final run was 0.88 for age 1, 0.83 for age 2, and 0.85 for age 3. Results of the current Run 20, the run preferred by the Technical Committee, show there has been a steep decline in the percentage of older weakfish since 1999 in all three runs (Figure 27a). Older weakfish comprised 5-8% of the estimated population during 1999 and this percentage fell to about 1% by 2003. Clearly, age structure is contracting rather than expanding toward “that necessary for restoration of the fishery”.

BIOMASS DYNAMIC MODELING

Biomass dynamic models (also known as surplus production models) are the simplest analytical methods that provide a full stock assessment (Haddon 2001). They are relatively simple to apply because they pool the overall effects of growth, mortality, and recruitment into a single production function. Their data needs are small - minimum data needed are an index of relative abundance and landings (both in weight). The stock is considered as undifferentiated biomass and age, size, and sex structure are ignored (Haddon 2001). Hilborn and Walters (1992) stated that if both methods are applied and give different answers, then assessment scientists should try to understand why the answers are different and analyze their management implications. This biomass dynamic model has been developed to supplement and corroborate the age-structured techniques employed elsewhere in the assessment.

Trawl survey indices of relative abundance were formulated as kg per tow (Figure 28; Uphoff 2005a). These indices were also employed to estimate relative F, where harvest in MT was divided by the average of the indices for that year plus the following year. The index is then harvest as a proportion of average biomass for a year and is more similar to F than to exploitation. These estimates indicate that F reached a peak in 1987, then declined gradually to a

low level in the mid 1990s (Figure 9c). A small rise in relative F occurred in some of the estimates in 1997-1998, then levels were very low through 2003.

The time-series fitting method was used to estimate production model parameters (Hilborn and Walters 1992; Haddon 2001). We used a spreadsheet version of the discrete form of the biomass dynamic model based on the logistic function:

$$U_t = U_{t-1} + rU_{t-1} (1 - (U_{t-1} / sK)) - sC_{t-1} + \varepsilon;$$

where U_t was the index of abundance in year t ; U_{t-1} = index of abundance the previous year, r = intrinsic rate of population increase; s = scalar for the abundance index; K = maximum population biomass; C_{t-1} = harvest (commercial and recreational) in the previous year; and ε is measurement error (Hilborn and Walters 1992). A genetic algorithm super solver (Evolver, Palisade Corporation) was used to estimate parameters that minimized observation error ε (observed $\log_e U_t$ - predicted $\log_e U_t$)² (Prager 1994). The spreadsheet version of the model combined with the genetic algorithm provided a great deal of flexibility for trying different model variations.

Biomass of the exploitable stock in year (EB_t) was estimated as predicted U_t / s (Hilborn and Walters 1992). Instantaneous annual fishing mortality rate in year t was estimated as

$$F_t = C_t / [(EB_t + EB_{t-1}) / 2] \text{ (Ricker 1975);}$$

F_t in 2003 was approximated as C_t / EB_t (Haddon 2001). Relative biomass was calculated for sensitivity analyses as EB_t / B_{msy} in some cases or EB_t / K in others; relative F was estimated equivalently.

Residuals were examined for normality and serial trends. Bootstrapping (Efron and Gong 1983) and jackknifing ("inflated" jackknife; Efron and Tibshirani 1993) sample reuse techniques were used to describe variability of parameters in the preferred run.

Several deterministic approaches were used to investigate sensitivity of preferred model parameters (r , K , s , and U_0) and estimates of F and EB . Discard estimates (recreational and commercial; de Silva 2004) were included in one run. Deterministic runs were made to test sensitivity of the preferred trawl survey based model to z -transformation (untransformed indices were modeled), aggregation of indices (mean $z+1$), addition or omission of discards, and omission of each z -transformed survey index time-series. An extended $z+1$ transformed trawl survey time-series (1975-2003) was created and compared to the base time-series. Delaware survey data were also split into long (1981-2003) and short (1990-2003) time-series, reflecting survey vessel changes (Michels and Greco 2003) and one DE model was developed with scalars for 1981-1989 and 1990-2003. Production model parameters r and K , and relative F (F/r) and relative biomass (EB/K) time-series were calculated for each model. Time-series sensitivity was explored by sequentially removing time-blocks of up to five years from the end of the time-series and up to seven years from the beginning.

All three time-series of z -transformed + 1 biomass indices generally agreed that EB was low during 1985-1993, and likely rising during 1994-1996 (Figure 28); EB indices were mixed from low to high during 1981-1984. Trawl survey EB $z+1$ indices indicated the stock slowly declined through the early 1990s and then underwent rapid growth in the mid-1990s. In any given year after 1996, trawl indices would lie between lows observed during 1981-1991 and values more

than four-times that level (Figure 28). Trawl based indices were positively correlated with each other. Correlations of EBNJ with EBFSC (Pearson correlation coefficient or $\rho = 0.69$) and EBDE with EBNJ ($\rho = 0.78$) were strongest and both were significant at $P < 0.005$. EBDE and EBFSC were not as strongly correlated ($\rho = 0.45$, $P = 0.054$).

The fit of the preferred run was modest ($r^2 = 0.42$; Figure 28). The overall mean of residuals was near zero (-0.09); residuals were positively skewed with a negative tail. EBFSC (N = 23) contributed 46% to total SSQ; EBNJ (N = 15), 31%; and EBDE (N = 18), 23%. A sequential trend was not apparent in the residuals as a whole. Residuals of EBFSC tended to be slightly positive overall (mean = 0.06) while residuals of EBNJ and EBDE tended to be negative (means = -0.23 and -0.17, respectively).

The intrinsic rate parameter (r) equaled 1.32 and K was estimated at about 41,000 mt; EB index in 1981 was estimated as 0.98 and the scalar was $4.932 \cdot 10^{-5}$. Precision of all bootstrapped (N = 200) parameters was excellent and estimates were not noticeably different between bootstrapped and jackknifed estimates.

Estimates of EB fell from 20,000 mt to 5,000 mt during 1981-1990, steadily climbed to 37,000 mt by 1997 and then leveled off between 37,000 and 39,000 mt (Figure 29). When changes in biomass were slight (1981-1990 and after 1996), bootstrapped estimates were very precise. Modest variation in estimates was observed when growth in biomass was rapid (1990-1996). Estimates of biomass have been very close to K since 1997 (Figure 29).

Estimates of F were near or exceeding the level that would have collapsed the stock ($F_{\text{collapse}} = r = 1.32$) during 1981-1991 (Figure 29). A rapid drop in estimated F followed and F was 0.17 by 1994. Estimates of F fell between 0.11 and 0.17 during 1994-2000 and F was all but extinguished by 2003 ($F = 0.03$; Figure 29). Trends in precision of F were the same as those for biomass; there was virtually no variation when changes were gradual, with some variation evident when transition was rapid.

The addition of discard estimates did little to change estimates in the directed harvest model (Figure 29). Parameter r rose slightly to 1.35, K fell to 39,910 mt, s increased slightly to $5.848 \cdot 10^{-5}$, and U_{1981} increased to 1.45. Estimates of EB were slightly lower and F estimates were slightly higher when discards were included (Figure 29).

Twelve additional trawl survey based biomass dynamic models were developed for the 1981-2003 time-series, as was one extended time-series (1975-2003) model (Table 9C). Generally, the $z+1$ transformation resulted in stable parameter estimates in their eight scenarios; r ranged between 1.10-1.44 and K fell between 36,000 and 47,000 mt. Removal of each time-series of $z+1$ transformed surveys did not affect parameter estimates appreciably. Extending the three $z+1$ index time-series to 1975 resulted in a large drop in r (to 0.33), while K remained within the range of the estimates made with the shorter $z+1$ time-series. Biomass dynamic models using untransformed indices individually produced lower and more variable estimates of r (0.33-1.12) and higher estimates of K (some very much higher). DE survey models were very sensitive to length of time-series and whether a single scalar was used or scalars were developed for about each half of the time-series. Estimates of r for the DE treatments ranged between 0.33 and 1.08

and K estimates fell between 82,000 and $1.54 \cdot 10^8$ mt. NEFSC or NJ survey models featured r 's of 0.83 and 1.12, respectively, while K was estimated as 54,000 and 57,000 mt. A biomass dynamic model using all three untransformed indices (necessitating an additional two scalars and two initial year survey parameters) estimated r as 1.80 and K as 42,000 mt (Table 9C).

Relative biomass (EB/K) estimates from these different trawl survey model formulations were in general agreement and indicated that biomass has been high and stable since 1996. Biomass estimates approached K in the most recent years. Relative biomass was generally lowest in the 1980s and increased rapidly during the early to mid-1990s. Timing and trajectories of this increase in biomass were more variable in the different models.

All trawl survey model formulations of relative F (F/r) indicated that F has been very low for the past decade. Relative F was high through 1989 and fell rapidly by 1993. The scale of the 1975-2003 scenario was generally over twice as high as all 1981-2003 scenarios, but the trajectory was similar over the years in common. Relative F was high (F at approximately r) and rising through 1980 in this extended time-series; it then followed the previously described trajectory for the 1981-2003 models.

When up to five years of recent data were removed from the preferred model, parameters, F , and EB did not detectably change. Removal of up to seven years from the beginning of the time-series had a profound effect on estimates of K and s , and induced some variation in r and U_0 . As up to seven years were sequentially removed from the beginning, K steadily dropped from about 41,000 mt to 22,000 mt; the scalar steadily rose. K and s were significantly correlated ($r = -0.85$, $P = 0.015$). Sequential removal of years from the beginning of the time-series generally caused estimated EB to fall and F to rise in the more recent years of the time-series, while preserving the general trend observed (low EB to high EB and high F to low F) when all years were retained.

Biomass dynamic models of trawl survey indices portrayed an Atlantic Coast weakfish stock that is now at very high biomass and subject to little F . These results seem contradictory to landings that have continuously declined since 1998 and are now the lowest in the time-series, unless the stock is now comprised of individuals too small to be of interest to recreational and commercial fisheries (a stunted population) or fishers have become disinterested in weakfish.

These results are supportive of results obtained in the previous assessment (1981-2000) with biomass dynamic models, ADAPT VPA and ICA VPA (Kahn 2002b; de Silva 2002; Uphoff 2002a, 2002b). All survey based biomass dynamic models in this analysis indicated that biomass was low in the 1980s and then experienced a rapid recovery in the early to mid-1990s; estimated F was excessive in the 1980s, dropped precipitously in the early 1990s, and stabilized at a minimally low level afterwards. Biomass may now be near carrying capacity. Significant reductions in F and biomass recovery would have occurred prior to what were considered significant management measures in Amendment 3 to the Weakfish FMP (Lockhart et al. 1996).

Biomass dynamic models of trawl survey indices portrayed an Atlantic Coast weakfish stock that is now at very high biomass and subject to little F (Figure 29). These results are supportive of results obtained in the previous assessment (Kahn 2002b). All survey-based biomass dynamic models in this analysis indicated that biomass was low in the 1980s and then experienced a rapid recovery in the early to mid-1990s; estimated F was excessive in the 1980s, dropped

precipitously in the early 1990s, and stabilized at a low level afterwards. Results of these models indicate that biomass may now be near carrying capacity. According to model results, significant reductions in F and biomass recovery occurred prior to what were considered significant management measures. These results seem contradictory to landings that have continuously declined since 1998 and are now the lowest in the time-series, unless the stock is now comprised of individuals too small to be of interest to recreational and commercial fisheries (a stunted population) or fishers have become disinterested in weakfish.

RELATIVE AND SCALED-UP ABSOLUTE F, STOCK BIOMASS, SURPLUS PRODUCTION AND SURPLUS PRODUCTION MODELS.

Biomass-based Relative Abundance Indices

A time series (1982-2004) of weakfish relative biomass indices were derived as a ratio between annual recreational weakfish catches (MRFSS: A + B1 +B2) in weight and total fishing effort data from the private boat sector based on the MRFSS annual surveys:

$$\text{RelWt} = \text{MidWt} / \text{Effortt} , \quad (1)$$

Where RelWt denotes a biomass-based relative abundance index, MidWt is the recreational catches in weight from the private/rental boat mode of the MRFSS for the Mid-Atlantic region (New York through Virginia) in year t and Effortt is the annual total trips of the private/rental boat mode in the Mid-Atlantic region in year t.

The resulting age aggregated (ages 1+) indices (Table 10) were confined to the private boat recreational catch and effort data from the Mid Atlantic subregion (New York to Virginia). This approach seemed justified because private boat catches from this subregion have, in most years, comprised a large (> 60%) percentage of the coast-wide recreational catches. Also, the private boat catch and effort estimates from the Mid Atlantic subregion have been measured with relatively high precision (relative CV < 0.13). Moreover, the private boat fishery is highly mobile and, unlike most trawl surveys, capable of catching larger (> 60 cm.) weakfish throughout their range. Since the MRFSS has monitored weight (kg) data from only the harvest (A, B1), the catch in weight of released weakfish (B2) was derived indirectly by assuming that the average weight (kg) of released fish (B2) was the same as the average weight of harvested fish. Although released weakfish caught in the recreational fishery are likely to weigh less than harvested fish, particularly after the imposition of minimum size limits in 1994, there is no known time series (1982-2004) of weight measurements on released fish with which to verify our assumption. Also, if recreational harvest (A, B1) in weight was used instead of total catch in weight (A, B1, B2) in the numerator of equation (1), the resulting indices after 1993 would be underestimated. Beginning in 1994, minimum size limits and daily creel limits were imposed coast-wide on the weakfish recreational fishery, thereby reducing weakfish recreational harvest from 1994 to 2005. Note that our indirect approach of estimating the B2 catch in weight in this context has also been used to estimate the B2 discards in weight for the current ADAPT VPA.

The abundance indices via equation (1) were based on total private boat fishing effort from the Mid Atlantic subregion rather than from directed fishing effort on weakfish (Table 10). In a

previous analysis (Crecco 2005d), the time series (1982-2004) of directed fishing effort from the Mid Atlantic private boat fishery was used to estimate weakfish abundance indices. Since the direct and total fishing effort time series was highly correlated (Pearson $r = 0.96$, $P < 0.0001$), the trend in abundance indices derived from total and directed effort was nearly identical and therefore the use of directed fishing effort would not have altered this analysis.

While this index is fishery dependent, problems of autocorrelation between it and coast-wide landings should be relatively minor for three reasons. First, commercial landings and discards have comprised, in most years, at least 65% of the total weakfish harvest. Thus, the Mid Atlantic private boat indices from 1982-2004 utilizes only about 20-30% of the total annual harvest in the index including the commercial harvest, thereby minimizing autocorrelation to the coast-wide landings. Second, the indices were based on the Mid Atlantic private boat catches (A, B1 B2) that include released fish instead of the harvest (type A and B1). This should further minimize the degree of autocorrelation with the total coast-wide harvest. Finally, the weakfish indices were based on the ratio of private boat recreational catch to fishing effort. This extra step of dividing catch by effort should reduce the probability of autocorrelation since the time series of coastwide harvest was inversely related (Pearson $r = -0.51$, $P < 0.006$) with private boat fishing effort from the Mid Atlantic region.

Fishing Mortality, Stock Biomass and Surplus Production

Relative fishing mortality estimates were also derived from 1982 to 2004. Relative F was estimated as a ratio of annual landings (commercial and recreational plus discards in mt) of ages 1+ weakfish in year t to the average of relative abundance indices in years t and t+1:

$$\text{RelFWt} = \text{HARVt} / [(\text{RelWt} + \text{RelWt}+1)/2], \quad (2)$$

where RelFWt is the biomass-based relative F for year t, HARVt is the total harvest in metric tons including estimated weight of discards for year t, RelWt is the relative abundance index for year t and RelWt+1 is the index for year t+1 (Table 10). Since the index is not yet available for 2006, an estimate of relative F for 2005 cannot be derived via equation (2). Instead, an approximate estimate for 2005 was derived as a ratio of the preliminary 2005 landings to the 2005 index. Note that the 2005 weakfish recreational landings and discards (mt) are available from the MRFSS, but the 2005 commercial landings and discards are not yet available. A preliminary 2005 coast-wide harvest estimate (commercial and recreational plus discards) was derived by dividing the 2005 recreational coast-wide harvest and discards by the 0.45, which was the ratio of 2004 recreational harvest to the 2004 total harvest.

The next step in this analysis was to transform relative F estimates into units of absolute biomass-weighted fishing mortality of ages 1+ weakfish. This transformation was made by multiplying the relative F estimates values from 1982 to 2005 by a scalar consisting of the ratio of the average biomass weighted F (ages 1+) from run 20 of ADAPT VPA for 1987-1991 to the average the relative F estimate from 1987-1991 (Crecco 2005b). The VPA F estimates were confined here to the 1987 - 1991 period because the 1987-1991 F estimates from the VPA were fully converged and therefore robust to retrospective bias noted from more recent (1999-2003) F estimates (Table 10).

Average stock biomass (mt) of ages 1+ weakfish was estimated from 1982 to 2004 as a ratio of coastwide harvest to the scaled biomass weighted F:

$$\text{Biot} = \text{HARVt} / \text{FWt}, \quad (3)$$

where Biot is the stock biomass for year t, HARVt is the harvest in year t in metric tons, including discards, and FWt is the biomass-weighted F for year t (Table 11).

A time series (1982-2004) of weakfish surplus production estimates from 1982-2004 was derived by subtracting weakfish stock biomass in year t from stock biomass in year t+1 followed by the addition of annual total landings (mt) in year t:

$$\text{SURPt} = \text{Biot}_{t+1} - \text{Biot}_t + \text{HARVt}, \quad (4)$$

where SURPt is surplus production in year t, Biot_{t+1} is the stock biomass in year t + 1, Biot_t is stock biomass in year t and HARVt is harvest in year t (mt) including discards (Table 11). Surplus production estimates have been used to monitor trends in per capita stock productivity for exploited finfish populations (Jacobson et al 2002).

Biomass weighted fishing mortality estimates on weakfish rose steadily from about 0.41 in 1982 to peak levels in 1988 (FW₁₉₈₈ = 1.08) (Table 10). The F estimates remained relatively high (FW_t range: 0.60 to 0.81) from 1989 to 1992, then fell steadily to below the 0.40 level in most years from 1993 to 2005. The 2004 and 2005 FW_t estimates were around 0.25.

Weakfish biomass levels remained relatively high between 15,000 and 34,000 mt from 1982 to 1988 then fell consistently below 10,000 mt from 1989 to 1993 (Table 11). Biomass rose abruptly to above 20,000 mt from 1994 to 1999, then weakfish biomass fell steadily thereafter to the lowest level in the time series in 2004 (5,700 mt). The preliminary 2005 biomass level of 7,300 mt is about 28% higher than the 2004 estimate, but the 2005 level is the third lowest in the time series (Table 11).

The time series of weakfish surplus production from 1982 to 2004 generally followed the trend in stock biomass (Table 11). Weakfish surplus production remained relatively high from 1982 to 1986 and again in 1993 and 1994, but levels fell greatly after 1997 and remained low thereafter despite relatively low and stable fishing mortality (Table 10). Weakfish surplus production estimates were negative in 2000 and 2002, indicating the absence of sustainable harvest in those years. Surplus production estimates were positive in 2003 and 2004, but were below the estimates prior to 1999 (Table 11).

Biomass Weighted Natural Mortality (MWt)

Most of the current evidence (Uphoff 2005b; Crecco 2005c) suggests that the recent decline in weakfish surplus production is likely due to a recent rise in weakfish natural mortality (M) linked to enhanced predation and perhaps inter-specific competition. It would be useful to estimate a time series of age aggregated (ages 1+) natural mortality (M) rates and relate the M time series

statistically (stepwise regression model) to the candidate finfishes and abiotic factors. Statistical support for the Predation Hypothesis would be evident if the time series of one or more of the candidate finfish predators were positively correlated ($P < 0.05$) to the trend in weakfish natural mortality. Standard output from the ADAPT VPA includes biomass weighted fishing mortality (F_{vpa}) rates of ages 1+ weakfish from 1982 to 2003. Recall that the natural mortality rate (M_{vpa}) in all VPA runs was assumed to be constant at 0.25 across all ages and years. As a result, biomass weighted total mortality (Z_{Wt}) rates from 1982 to 2003 were easily derived by adding 0.25 to each of the aggregated age 1+ F values from the VPA run #20 that included MRFSS recreational cpue as the sole tuning index. The Weakfish Stock Assessment Subcommittee chose run #20 as the preferred VPA model run. The degree of retrospective bias in recent F and weakfish stock size estimates from run #20 was much less extensive than the bias for other model runs that were tuned by either trawl survey indices alone, or a combination of trawl survey and MRFSS indices. Note that scaled biomass (ages 1+) weighted fishing mortality (FWt) have been estimated independently from the VPA via equation 2 (Table 10). As a result, a time series of biomass weighted natural mortality rates (MWt) of ages 1+ weakfish were calculated from 1982 to 2003 by subtracting the biomass weighted FWt estimates (Table 10, Figure 34) from total mortality estimate (Z_{Wt}) estimates based on the VPA run #20.

Although natural mortality of age 1+ weakfish was assumed to be constant in previous Yield-per-Recruit and VPA model runs, the trend in annual biomass weighted natural mortality (MWt) from 1982 to 2003 does not support this assumption (Figure 34). Annual (MWt) estimates for ages 1+ weakfish were highly variable with systematic trends occurring from 1982 to 2003 (Table 10). Weakfish MWt levels from 1982 to 1986 remained relatively high, ranging from 0.44 to 0.82. Weakfish MWt levels fluctuated greatly between 1987 and 1995 from a high of 0.64 in 1988 to a low of 0.01 in 1989 (Figure 34). After 1995, however, weakfish natural mortality (MWt) rose steadily from 0.31 in 1996 to a high of 1.09 in 2002. Not only was annual weakfish MWt levels not constant from 1982 to 2003, but the long-term (1982-2003) average MWt level derived here (mean $MWt = 0.47$, $SE = 0.06$) was significantly higher (t-statistic = 3.67, $P < 0.01$) than the assumed constant $M = 0.25$.

Age 0 Mortality

One major problem in quantifying predation or other trophic responses on weakfish is pinpointing the period in the life history where the highest predation risk takes place. A temporal shift in predation mortality can occur across many weakfish ages (ages 0+) or may be confined mainly to a single age group (i. e. age 0 mortality). Since age 0 weakfish rarely exceed 18 cm TL, this early stage is particularly susceptible to a heightened risk of mortality from a vast array of potential finfish predators. Several recent predation studies (Beck 1997; Wahle 2003) have shown that size dependent mortality during the juvenile stage may lead to a demographic bottleneck which can inhibit the flow of recruitment to older ages. If this bottleneck is severe and persists over time, prey abundance will eventually cascade downward, resulting in a stock collapse emanating from the youngest to the oldest ages (i. e. bottom-up effect). To examine whether or not a demographic bottleneck may occur for age 0 weakfish, a time series of relative weakfish juvenile mortality (Z_0) estimates was derived for the 1982 to 2003 year-classes. The Z_0 estimates were expressed by a log ratio between coast-wide age 1 abundance (N_{1t+1}) in year $t+1$ to the mean coast-wide juvenile abundance index (N_{0t}) in year t :

$$Z_0 = -\log(N_{1t+1} / N_{0t}) . \quad (5)$$

Weakfish juvenile abundance (N_0) has been monitored along the Atlantic coast from Rhode Island to North Carolina including the South Atlantic (SEAMAP) from 1982-2004. Nine juvenile surveys (Table 12) were used to construct an average coast-wide index (N_0) from 1982 to 2004 (see Section for more details). The abundance indices were expressed as the geometric mean catch per tow or geometric mean catch per seine haul, resulting in indices of vastly different magnitude. As a result, before the indices from the nine surveys were combined into a coast-wide average index (N_0), the relative abundance values for each survey had to be standardized to equivalent abundance units. Equivalent units were established in a three-step process. First, the long-term (1982-2004) geometric mean abundance index was derived annually for the recruitment time series of each of the nine surveys. Second, a scalar for each data set was derived as a ratio of the long-term average index to the long-term average New Jersey index. Each annual index from each data set was then multiplied times the respective scalar, thereby transforming the magnitude of the eight indices to units of the New Jersey indices (Table 13). Finally, the coast-wide geometric mean index (N_0) was derived as the grand mean across the scaled indices from 1982-2004 (Table 13).

A brief description of each of the nine juvenile trawl surveys is given below. The Rhode Island Division of Fish and Wildlife has conducted a juvenile finfish trawl survey in Narragansett Bay from 1982 to 2004. Juvenile weakfish abundance was expressed annually by the geometric mean catch/tow. The Connecticut DEP trawl survey has employed a stratified random design based on 120 tows per year. Tows have been conducted biweekly with a 42 ft otter trawl throughout Long Island Sound from April to June and from September to October. The survey data on weakfish are available from 1984 to 2004. Juvenile and age 1 weakfish (geometric mean catch/tow) have been primarily sampled by this trawl survey. The New York Division of Fish, Wildlife and Marine Resources has conducted a juvenile trawl survey from 1985 to 2004 using a 16 ft. trawl targeting juvenile estuarine finfish in the Peconic Bay of Eastern Long Island. Juvenile weakfish abundance was expressed annually by the geometric mean catch/tow. New Jersey has conducted an ocean trawl program from 1989 to 2004 using a stratified random design. Relative abundance (geometric mean catch/tow) of juvenile and ages 1+ weakfish have been monitored annually from August and October based on 78 tows/year. The Delaware Division of Fish and Wildlife conducts a juvenile trawl survey in Delaware Bay with a 16 ft. trawl from 1982 to 2004. Juvenile weakfish abundance was expressed annually by the geometric mean catch/tow. The Maryland Department of Natural Resources has conducted a juvenile trawl survey in Chesapeake Bay from 1980 to 2004 using a 16 ft. otter trawl. A time series of juvenile weakfish abundance was expressed annually by the geometric mean catch/tow. The Virginia Institute of Marine Science (VIMS) has conducted a trawl survey in lower Chesapeake Bay from 1987 to 2004. A time series of juvenile weakfish abundance (geometric mean catch/tow) was derived annually using August-October tows from three river tributaries. The North Carolina Division of Marine Fisheries has conducted a juvenile trawl survey in Pamlico Sound from 1987 to 2004. A time series of juvenile weakfish abundance was expressed annually by the geometric mean catch/tow. The Southeast Area Monitoring and Assessment Program (SEAMAP) has conducted trawl surveys of juvenile and adult (ages 1+) weakfish since 1989 which sample from

Cape Hatteras to Cape Canaveral. A time series (1989-2004) of juvenile abundance has been expressed as the geometric mean catch/tow.

A time series (1982-2004) of weakfish age 1 abundance estimates (N_1 in millions of fish) used in the numerator of equation (5) was taken directly from run #20 from the VPA (Table 13). The N_1 estimates from run #20 were derived independently of the relative juvenile abundance estimates (Table 12) since none of the juvenile indices were used to tune run # 20. Note that the juvenile abundance indices in the denominator of equation (5) are expressed in relative units. Thus, the juvenile total mortality rates (Z_0) are relative mortality estimates in contrast to the ages 1+ natural mortality estimates (M_{Wt}) from the previous section that are absolute mortality estimates based on total abundance data from run #20 of the VPA. Also, note that the current VPA generated age 1 abundance estimates ($t+1$) for the 1981 to 2002 year-classes during the years 1982 to 2003. Since there is currently no 2004 age 1 index that represents the 2003 year-class, age 1 abundance for the 2003 year-class was assumed to be the same as the age 1 abundance estimate for the 2002 year-class (Table 13). Given the steady decline in age 1 abundance (N_1) from the VPA from 1997 to 2002, this assumption seemed reasonable. The age 1 abundance (N_1) values from the VPA were occasionally larger than the magnitude of the coast-wide juvenile index (Table 13) resulting in some negative values of mortality (Z_0) via equation (5). In order to generate mortality rates (Z_0) as positive integers, all age 1 abundance data (N_1) were scaled down by dividing the N_1 values by an arbitrary value of 9.0.

Weakfish juvenile indices varied greatly across the nine surveys for the 1982 to 2004 year-classes (Table 12). In the Rhode Island survey, juvenile indices remained relatively high from 1982 to about 1999, then juvenile year-class strength after 1999 dropped quickly to low levels. Juvenile indices from other state survey, however, exhibited a general rise in year-class recruitment after 1995 (Table 12). When the nine juvenile data sets were standardized and combined into a mean coast-wide index (N_0) (Table 13), the coast-wide weakfish juvenile recruitment showed a fairly persistent rise after 1994 (Figure 32). The persistent rise in juvenile recruitment after 1998 coincided with a pronounced drop in weakfish ages 1+ biomass (Table 11), indicating the presence of compensatory survival from the larval to juvenile stage. Abundance of age 1 weakfish (N_{1J}) based on run #20 of the VPA varied without trend from 1982 to about 1995, then age 1 abundance fell steadily from 1996 to 2003 (Figure 33) in sharp contrast to the general rise in juvenile recruitment after 1995. The resulting weakfish juvenile mortality (Z_0) estimates increased in magnitude for the 1995 to 2003 year-classes (Figure 33). These findings strongly suggest that recent natural mortality on juvenile weakfish increased systematically since 1995 consistent with the recent emergence of a trophic bottleneck that may have interfered with the flow of recruitment from age 0 to ages 1+.

PREDATION HYPOTHESIS

Coastwide weakfish biomass (mt) has fallen steadily by 65% since 1999 to levels well below the estimated B_{msy} threshold. Total mortality (Z) estimates from both the VPA (run #20) and cohort catch curves have risen sharply since 1996 but biomass weighted F estimates (ages 1+) during this period have remained low and steady (Crecco 2005a). Moreover, annual changes in average annual F (lagged or unlagged) from 1982 to 2004 exhibited no statistical relationship (Pearson $r = 0.22$, $P < 0.31$) to changes in weakfish surplus production (Crecco 2005b),

suggesting that the recent failure in weakfish productivity is not related to fishing effects. As a result, the sudden and unexpected drop in weakfish productivity since 1998 is thought to be due to a systematic rise in weakfish natural mortality (M). This increase in M is likely due to one or a combination of biotic and abiotic factors including a rise in finfish predation, enhanced interspecific competition for food and perhaps a major temporal shift in key environmental variables that could play a role in affecting weakfish recruitment.

Statistical and empirical evidence gathered thus far supports a Predation Hypothesis involving striped bass (*Morone saxatilis*) as a primary predatory finfish on weakfish (Uphoff 2005a, 2005b; Crecco 2005a, 2005c). Striped bass abundance along the Atlantic coast has recently risen to record high levels after 1998 (ASMFC 2005) coincident with the recent failure in weakfish productivity. Although statistical and empirical evidence linking striped bass to the recent failure in weakfish productivity is persuasive, there is yet no direct evidence that ties the recent increase in weakfish M directly to striped bass predation. Also, the weakfish age(s) at which the risk of predation is highest has yet to be established. Since predation is generally regarded as a major force structuring marine fish communities (Bax 1991), it would be useful in this context to present an analytical model that merges the population dynamics of weakfish with the foraging characteristics of striped bass. In addition, to provide a more thorough examination of the Predation Hypotheses, other candidate finfish predators on weakfish such as bluefish, summer flounder and spiny dogfish need to be considered as well as potential competitors of weakfish. Finally, systematic changes in finfish productivity may respond to high frequency climatic changes over broad geographical areas (Scheffer et al. 2001). Thus, a more complete examination of temporal shifts in climatic variables, such as water temperature and long-term deviations in the North Atlantic Oscillation Index, need to be considered as an alternative to the Predation hypothesis.

In this section, we used the age-aggregated Steele and Henderson (1984) (S-H) production model to further examine the joint effects of fishing and predation on Atlantic coast weakfish. The S-H model has extensive theoretical appeal since it incorporates the compensatory stock dynamics of the prey (weakfish) with fishing effects plus a sigmoid foraging response by the predatory finfish that may lead to critical depensation at low prey abundance (Spencer and Collie 1997). Since the S-H model can also accommodate climatic variables (Spencer 1997), this modeling approach represents a modest but straightforward attempt at ecosystem modeling. In this section, the S-H model was used to re-estimate overfishing thresholds (F_{msy} , B_{msy}) for weakfish in the presence of finfish predation. In addition, more robust and precise estimates of the overfishing thresholds (F_{msy} , B_{msy}) from the S-H model were attempted through iterative reweighted least squares regression. Model validation and an analysis of process error were also undertaken by an examination of residual patterns. As first indicated by Uphoff (2005c), the predation exponents estimated directly from S-H model can be used to derive weakfish natural mortality rates (M_p) associated with striped bass predation. In this section, a time series (1982-2004) of M_p values was compared to the magnitude and trend in age aggregated annual M_{Wt} estimates (ages 1+) derived independently from run # 20 of the VPA (i. e. $M = Z-F$) combined with biomass weighted fishing mortality rates. A direct causal link between the rise in weakfish natural mortality and striped bass predation would be established, if the 1982-2004 predator-related M_p estimates derived empirically from the S-H model closely track the weakfish ages 1+ M estimates. Finally a stepwise multiple linear regression model was used to identify additional

explanatory variables that may be linked to changes in weakfish surplus production, weakfish natural mortality (MWt) and weakfish juvenile total mortality from 1982 to 2004. These explanatory variables included other finfish predators (i.e. summer flounder, bluefish and spiny dogfish), a potential finfish competitor of weakfish (Atlantic croaker) and selected climatic variables (mean annual water temperatures (C) and annual deviations in the North Atlantic Oscillation index).

Statistical Analysis

A stepwise multiple regression model (SAS 2002) was used to identify several key biotic and abiotic factors that may be linked statistically ($P < 0.05$) to changes in weakfish surplus production (SURPt), weakfish natural mortality (MWt) and weakfish juvenile mortality (Z0) from 1982 to 2004. The stepwise analysis was conducted separately using surplus production (SURPt), age aggregated (ages 1+) natural mortality (MWt) and age 0 mortality (Z0) as dependent variables in the three stepwise models. Explanatory variables in the three model runs included biomass weighted fishing mortality (FWt), as well as stock abundance of several candidate finfish predators on weakfish such as striped bass, bluefish, summer flounder and spiny dogfish (Table 14). Striped bass is regarded as a voracious predator from the Mid and North Atlantic on menhaden, gizzard shad and herring (Hartman 1993). Larger (> 60 cm) striped bass, however, have been reported to undergo an ontogenic shift in prey preference from herring and small menhaden to spot and weakfish in Chesapeake Bay (Hartman and Brandt 1995; Walter and Austin 2003). Bluefish (*Pomatomous saltatrix*) also prey upon a variety of finfishes including weakfish throughout the Atlantic coast (Bowman et al. 2000). Larger (> 50 cm) summer flounder (*Paralichthys dentatus*) (Bowmann et al 2000) also prey on a variety of finfishes including smaller (< 20 cm) weakfish from mostly the North and Mid Atlantic. By contrast, spiny dogfish (*Squalus acanthias*) are found coast-wide and are regarded as a primary finfish predator of river herring, capelin, and sand lance (Bowmann et al 2000), although Rountree (1999) reported that spiny dogfish also prey upon smaller weakfish.

A time series of Atlantic croaker (*Micropogonias undulatus*) abundance was also included as a potential competitor of weakfish (Table 14). The ecological role of Atlantic croaker as a potential competitor or predator on weakfish is poorly understood but croaker trophic status is thought to be somewhat below that of weakfish (Miller et al 2004). Thus croaker may compete with or prey upon small weakfish particularly in Mid Atlantic waters during spring when both species overlap. Potential weakfish predators and competitors are thought to undermine weakfish surplus production by enhancing weakfish natural mortality or by constricting the flow of age 0 recruitment (i. e. trophic bottleneck). As a result, one or more the candidate finfishes in the model would be identified as a potential stressor on weakfish surplus production if the respective regression slope(s) associated with each finfish in the stepwise regression was negative and statistically significant ($P < 0.05$). Conversely, in the analyses with weakfish ages 1+ natural mortality (MWt) and juvenile mortality (Z0) as dependent variables, support for the Hypothesis would be given if the slope(s) for one or more of the candidate finfishes were positive and statistically significant ($P < 0.05$). Fishing mortality estimates (FWt) were also included as an explanatory variable to further test the overfishing hypothesis. If, for example, fishing mortality (FWt) was positively correlated ($P < 0.05$) to the recent rise in age aggregated (ages 1+) natural

mortality (MWt), this might imply that discard losses due to fishing have been severely underestimated in the catch-at-age matrix used in the VPA model.

In this analysis, annual abundance of four out of five candidate finfish (striped bass, summer flounder, bluefish, and spiny dogfish) was taken from the most current assessment of that species (Table 14). Coast-wide biomass (mt) of spiny dogfish was expressed annually as thousands of metric tons by the area swept approach based on the NEFSC spring trawl survey (Matt Gates, ASMFC spiny dogfish Technical Committee pers. comm.). Coast-wide striped bass, summer flounder (fluke), and bluefish stock abundance estimates from 1982 to 2004 were taken from the most recent ADAPT VPA runs (ASMFC web site, 2005). Fluke abundance from 1982 to 2004 was expressed as spawning stock biomass in mt based on the 2005 VPA. Bluefish abundance was expressed each year as millions of pounds of ages 1+ fish based on the ASAP model. In the case of the Atlantic croaker, stock size estimates from the VPA are only available from 1982 to 2002. For this reason, the 1982-2004 total recreational catch of croaker (A, B1, B2) from the MRFSS was used as a coast-wide index in the stepwise regression (Table 14). Croaker abundance each year was represented by the total catches (A, B1 and B2) from the Mid Atlantic subregion. Use of croaker recreational catches as a stock index seemed justified, given that the 1982-2002 recreational catches were highly correlated (Pearson $r = 0.80$, $P < 0.0001$) to the 1982-2002 VPA abundance estimates. Striped bass abundance was expressed as millions of ages 8+ fish from the 2005 VPA. The striped bass VPA tended to overestimate recent (2002-2004) F levels and therefore underestimate recent (2002-2004) stock sizes. As a result, a second time series of ages 7+ stripers (millions of fish) was also used in the stepwise model based on the ratio of ages 7+ landings to tagged-based F estimate of ages 7+ stripers (Kahn 2005b). Given that recent (2002-2004) tag-based F estimates were 20% to 40% lower than the corresponding F estimates from the VPA, the tag-based abundance estimates exhibited a more pronounced rise from 2002 to 2004 (Table 14).

Changes in large-scale climatic conditions could have also influenced the recent decline of weakfish surplus production by interrupting the formation of dominant year-classes. A time series of average annual surface water temperature (Temp, C) and deviations in the winter North Atlantic Oscillation (NAO) index (Table 14) were also included in the stepwise model as potential environmental forcing variables on weakfish surplus production. Since all female weakfish reach sexual maturity by age 2 (Wilk 1979), the Temp and NAO variables were lagged t-1 and t-2 years to coincide with weakfish recruitment to the adult stock. A time series (1980-2002) of average annual water temperatures was taken from a continuous temperature recorder located at the Millstone Nuclear Power Station, Waterford CT. Annual deviations in the NAO indices from 1980 to 2002 were taken from a recent American lobster manuscript (Fogarty and Gendron 2004).

The time series of weakfish surplus production (SURPt) from 1982 to 2004 (Table 11) was regressed in a stepwise model against the nine explanatory variables (Table 14). The stepwise model selected striped bass abundance based on the VPA (Strip) as the most significant ($P < 0.001$) explanatory variable that was inversely related to weakfish surplus production (Table 15). No other explanatory variable was selected at the $P < 0.05$ level in the stepwise model. The correlation matrix (Table 16) indicated that weakfish surplus production was also inversely related ($P < 0.05$) to striped bass abundance based on tagging (Strip2) and to Atlantic croaker

(Croak). The time series of tag-based striper abundance (Strip2) and croaker abundance (Croak) from 1982 to 2004 closely followed (co-linear) the trend in VPA-based striper abundance (Strip) (Table 14). Thus, due to the presence of substantial co-linearity, the stepwise model was unable to select either Strip2 or Croak as a second or third significant variable. None of the other six explanatory variables including fishing mortality (FWt) were negatively and significantly related ($P < 0.05$) to weakfish surplus production (Table 16). Statistical evidence given here strongly suggests that the recent failure in weakfish productivity was linked to predation and inter-specific competition with striped bass and perhaps Atlantic croaker.

The time series of biomass weighted natural mortality estimates (MWt) on ages 1+ weakfish from 1982 to 2004 (Table 10, Figure 34) were used as a response variable in a second stepwise model against the nine explanatory variables (Table 14). The stepwise model selected fluke (summer flounder) as the most significant ($P < 0.001$) explanatory variable (Pearson $r = 0.79$, $P < 0.0001$) that was closely tied to the trend in weakfish ages 1+ natural mortality (Table 15). No other explanatory variable was selected at the $P < 0.05$ level in the stepwise model. The correlation matrix (Table 16) revealed that weakfish natural mortality (MWt) was also positively related ($P < 0.05$) to both striped bass abundance estimates (Strip, Strip2). Since the time series of striped bass abundance from 1982 to 2004 closely tracked (co-linear) the trend in fluke abundance (Table 14), the stepwise model was unable to select either Strip or Strip2 as a second significant ($P < 0.05$) variable. None of the other six explanatory variables were positively and significantly related ($P < 0.05$) to weakfish ages 1+ natural mortality (Table 16). Statistical evidence given here strongly suggests that the recent rise in weakfish ages 1+ natural mortality (MWt) was linked to enhanced predation from fluke and striped bass.

The time series of juvenile weakfish mortality rates (Z0) from 1982 to 2004 (Table 13, Figure 33) were used as a response variable in a third stepwise model against the nine explanatory variables (Table 14). The stepwise model selected striped bass abundance from the VPA as the most significant ($P < 0.001$) explanatory variable that was closely tied (Pearson $r = 0.91$, $P < 0.0001$) to the rise in weakfish juvenile mortality (Z0) (Table 15, Figure 35). No other explanatory variable was selected at the $P < 0.05$ level in the stepwise model. However, striped bass abundance based on tagging (Strip) (Pearson $r = 0.81$, $P < 0.0001$) and Atlantic croaker abundance (Pearson $r = 0.85$, $P < 0.0001$) were both closely tied to the pronounced rise in weakfish juvenile mortality (Table 16). The correlation matrix (Table 16) also revealed that weakfish juvenile mortality (Z0) was, to a lesser extent, positively related ($P < 0.05$) to fluke (Pearson $r = 0.46$, $P < 0.03$). Since the time series of Atlantic croaker abundance from 1982 to 2004 closely tracked (co-linear) the trend in striped bass (Table 14), the stepwise model was unable to select croaker as a second significant ($P < 0.05$) variable. None of the other five explanatory variables were positively and significantly related ($P < 0.05$) to weakfish ages 1+ natural mortality (Table 16). Statistical evidence given here strongly suggests that the recent failure in weakfish productivity was due primarily to the emergence of a age 0 recruitment bottleneck linked to enhanced predation with striped bass and perhaps Atlantic croaker and fluke.

Steele-Henderson (S-H) Model

The age aggregated Steele-Henderson (S-H) production model (Steel and Henderson 1984) was used to re-estimate the overfishing thresholds (Fmsy, Bmsy) for weakfish with and without the

presence of a significant ($P < 0.05$) predator-prey response. The S-H model assumes the existence of compensatory density-dependent mortality for finfish populations, a position widely held by most fish population ecologists (Wahle 2003). All of the population dynamics processes (somatic growth, natural mortality and recruitment) in the S-H model are subsumed in the intrinsic rate of population increase (r) and to a lesser extent in the carrying capacity (K) parameters. Like all production models, successful fitting (precise and robust parameter estimates) from the S-H model requires a high degree of contrast in the time series (1982-2004) of weakfish biomass. Previous production modeling (Uphoff 2005a; Crecco 2005a) has thus far indicated that precise estimates (r , K) were obtained based on the 1982-2004 time series of weakfish landings and biomass estimates. The S-H model was originally expressed as a logistic production model with an added sigmoid function that reflected the foraging response by the predator. Previous weakfish production modeling indicated that the gompertz asymmetrical model was a better fit to the weakfish biomass data than the logistics model (Crecco 2005a, 2005b). As a result, the surplus production portion of the S-H model was converted from the logistics to the Gompertz form:

$$\text{Biot}+1 = \text{Biot} + \log(K) * r * \text{Biot} * (1 - (\log(\text{Biot}) / \log(K))) - \text{catch} - [(c * \text{Predt} * (\text{Biot})^{**2}) / (A^{**2} + (\text{Biot})^{**2})] \quad (6)$$

where: $\text{Biot}+1$ = weakfish biomass (mt) in year $t+1$;

Biot = weakfish biomass (mt) in year t ;

Predt = abundance of the predator in year t ;

K = estimated carrying capacity of weakfish (mt);

r = intrinsic rate of population increase;

c = per capita consumption rate of the predator;

A = weakfish biomass (mt) where predator satiation takes place.

In the Gompertz model, F_{msy} is expressed solely by the intrinsic rate estimate (r), whereas B_{msy} is expressed as $K/2.72$, where $2.72 = \exp 1$ (Quinn and Deriso 1999). All parameter estimates (r , K , c and A) from the S-H model (equation 6) were derived from the NLIN procedure (Marquardt algorithm) contained in the Statistical Analysis System (SAS 2002). The S-H model was fitted to weakfish biomass (Biot , $\text{Biot}+1$) (Table 11) and one or more of the candidate finfish predators (Predt) (Table 14) by nonlinear least squares regression methods. Note that the choice of one or more of the candidate finfish predators (Predt) (Table 14) was conditional upon results from the stepwise regression analysis.

Given the potential for moderate to severe measurement errors in the input data, the S-H models was also fitted as a nonlinear robust regression using the iterative reweighted least squares method outlined by Holland and Welsch (1978). The algorithm and rationale for this approach is described in the current SAS (2002). This re-weighting scheme allowed the down weighting of data from certain years in the model where model residuals (regardless of direction) exceeded a previously defined threshold level. As indicated by Holland and Welsch (1978), the choice of a threshold is subjective and always represents a trade-off between minimizing the variances around the parameters (r , K , c and A) and at the same time generating globally converged parameter estimates. As suggested by Holland and Welsch (1978), a range of threshold estimates were used initially and the final threshold value was selected that satisfied the trade-off between global convergence of all parameter estimates and parameter estimates with maximum precision (minimum variance). The two step re-weighting approach always produced converged estimates (global estimates) that were within 10% of the parameter estimates (r , K , c and A)

derived by the ordinary least squares approach. However, the standard errors about the estimates based on iterative re-weighting were always 30 to 45% lower, resulting in much narrower confidence limits about the overfishing definitions (Fmsy, Bmsy) and the predation parameters (c, A).

As indicated by Steele and Henderson (1984), the S-H model (equation 6) is a conventional discrete time surplus production model:

$$\text{Biot}+1 = \text{Biot} + \log(K) * r * \text{Biot} * (1 - (\log(\text{Biot}) / \log(K))) - \text{catcht}, \quad (7)$$

that is merged with a type 3 predator functional response of the type:

$$[(c * \text{Predt} * (\text{Biot})^{**2}) / (A^{**2} + (\text{Biot})^{**2})] \quad (8)$$

In previous weakfish surplus production modeling (Uphoff 2005a), the discrete logistics model in the absence of predation effects (equation 7) was shown to seriously over predict recent (1998-2003) weakfish biomass, leading to a clear trend in recent residuals. This clear residual pattern indicated that the conventional production model (equation 7) was highly uncertain especially for short-term predictions. Model uncertainty is usually caused by the failure to include one or more informative parameters, or by choosing the wrong form of the model. Note that a model with low levels of uncertainty will produce no systematic residual pattern over time. To further examine for potential model uncertainty, the discrete component (equation 7) of the S-H model (i. e. without predation effects) was fitted to the 1982-2004 weakfish biomass estimates (Biot, Biot+1) (Table 11). If a systematic pattern should emerge in the residuals based on the recent (1998-2004) observed and predicted weakfish biomass (Biot+1) estimates, then severe process errors would again be manifested in the model. Statistical support for the Predation Hypothesis would be evident, if the addition of the predator component (equation 8) to the S-H model (equation 6) not only improved the model fit, but also greatly reduced model uncertainty that had previously plagued the discrete production model (equation 7).

In an earlier weakfish report, Uphoff (2005c) noted that if the predation parameter estimates (c, A) from the S-H model are sufficiently precise, then a time series of weakfish biomass estimates (Dt) consumed annually by the predator (Predt) can be derived in the form:

$$\text{Dt} = [(c * \text{Predt} * (\text{Biot})^{**2}) / (A^{**2} + (\text{Biot})^{**2})] . \quad (9)$$

Once Dt estimates were obtained via equation (9), the instantaneous natural mortality rate (biomass weighted) ascribed to predation (Mp) can be derived:

$$\text{Mp} = \text{Dt} / [(\text{Biot} + \text{Biot}+1) / 2]. \quad (10)$$

Further empirical support for the Predation Hypthesis would be given if the biomass weighted natural mortality due to predation (Mp) on weakfish from 1982 to 2004 closely followed the trend in age 1+ weakfish natural mortality (MWt) estimates based on run #20 from the VPA.

The Gompertz surplus production model (equation 7) was fitted by nonlinear least squares regression to weakfish stock biomass in year t and $t+1$. This model did provide a good fit ($r^2 = 0.68$) to the biomass data with statistically significant ($P < 0.05$) r and K parameter estimates (Table 17). The resulting overfishing thresholds (F_{msy} , B_{msy}) for weakfish of 0.32 and 25,259 mt, respectively, were similar in magnitude to previous estimates (Table 17). However, this production model consistently over predicted weakfish biomass from 1998 to 2005 by 10 to 50% (Table 18), indicating that the Gompertz model was plagued by substantial model uncertainty. Although fitting the model (equation 7) by iterative reweighting least squares substantially improved the fit ($r^2 = 0.82$) and enhanced precision around the parameter estimates (r , K) and overfishing definitions (F_{msy} , B_{msy}) (Table 17), a severe residual pattern nevertheless persisted from 1998 to 2005 (Table 19). Severe model uncertainty, as indicated by a systematic trend in the residuals, usually indicates the absence of an informative parameter(s) to the model.

The full S-H production model (equation 6) was fitted by nonlinear least squares regression and iterative re-weighting to weakfish stock biomass in year t and $t+1$ (B_{iot} , B_{iot1}) and striped bass (Strip) as the potential predator (Tables 11 and 14). Both S-H model provided very good fits ($r^2 = 0.88, 0.94$) to the biomass and predation data with statistically significant ($P < 0.05$) r , K , c and A parameter estimates (Table 20). Overall, the S-H models was not only a better fit to weakfish biomass than the basic Gompertz model, but the r and K parameters, particularly from iterative re-weighting, were estimated with much higher precision. The resulting weakfish overfishing thresholds (F_{msy} , B_{msy}) from the nonlinear least squares S-H models for weakfish was 0.59 and 21,179 mt, respectively (Table 20). The overfishing thresholds (F_{msy}, B_{msy}) based on the iterative re-weighted S-H model was 0.55 and 23,400 mt, respectively (Table 11). The S-H model fitted by iterative re-weighting exhibited clear improvements in precision about the parameters (r , K , c and A) over the traditional least squares S-H model. Moreover, unlike the severe residual pattern evident from 1998 to 2004 based on the traditional gompertz models (Tables 18 and 19), there is little if any systematic residual pattern from the S-H model fitted by iterative re-weighting (Table 21). This indicated little if any uncertainty for the S-H model fitted by iterative re-weighting least squares. Finally, the time series (1982-2003) of predation-based natural mortality (M_p) from striped bass was derived via equations (9) and (10) (Table 22). The predatory mortality rates (M_p) rose in magnitude after 1997 coincident with a steady drop in weakfish biomass. This inverse relationship between M_p and weakfish biomass is consistent with the presence of depensatory density-dependent predation mortality which, if persistent over time, could be highly destabilizing to future weakfish population growth. Finally, the time series (1982-2004) of predatory natural mortality (M_p) was significantly correlated (Pearson $r = 0.61$, $P < 0.003$) to the age 1+ natural mortality (M_{Wt}) values estimated independently (Table 22, Figure 36). These findings strongly suggest that the recent rise in weakfish ages 1+ natural mortality (M_{Wt}) coupled with the failure in weakfish productivity ($SURPt$) are tied directly to the increase in striped bass predation mortality (M_p).

Most of the statistical evidence given herein supports the Predation Hypothesis as the most reasonable explanation for the recent failure in weakfish productivity. Although results from the stepwise model identified striped bass, Atlantic croaker and fluke as primary weakfish predators (Table 15), striped bass was selected first over the other two in two ($SURPt$, Z_0) out of three model runs. In the case of the stepwise model with ages 1+ natural mortality (M_{Wt}) as the response variable, striped bass was chosen second only to fluke as a major predator of adult

weakfish (Tables 15 and 16). Annual changes in fluke abundance from 1982 to 2004 were closely coupled to annual changes in weakfish ages 1+ MWt (Tables 10 and 14). Nevertheless, it is doubtful that this statistical relationship suggests a plausible causal mechanism. The mouth gape of even large (> 75 cm) fluke is relatively small (Olla et al. 1972) as compared to the mouth gape of other inshore apex predators such as striped bass and bluefish. The relatively small mouth size of fluke is primarily adapted to the consumption of invertebrates and small (< 25 cm) finfish prey. Therefore, larger fluke could conceivably prey heavily on age 1 weakfish (length range: 18-25 cm). However, it would be highly unlikely that even the largest fluke could prey effectively enough on ages 2+ weakfish to have caused the close positive fit (Pearson $r = 0.79$, $P < 0.0001$) given here between weakfish ages 1+ natural mortality and fluke abundance (Table 16). Therefore, we regard the promising statistical relationship given here between weakfish ages 1+ MWt and fluke abundance as a statistical artifact.

The close statistical coupling between the recent rise in weakfish juvenile mortality (Z_0) and the parallel rise in striped bass and croaker abundance from 1982 to 2004 (Figure 35) implies that the recent emergence of a recruitment bottleneck at age 0 was due to predation. If this bottleneck persists over time, the flow of weakfish age 0 recruitment to older ages may become severely constricted, causing the weakfish stock to cascade downward from the bottom up. The contribution of croaker as a competitor or predator responsible for the weakfish recruitment bottleneck is difficult to establish at this time. Dietary studies of croaker are few in number (Miller et al 2004), but current research indicates that croaker, due to their relatively small size, eat mostly invertebrates (Overstreet and Heard 1978). The diet of croaker does overlap that of smaller (< 40 cm) weakfish, and like weakfish, larger (> 33 cm) croaker apparently undergo an ontogenic shift in diet from invertebrates to mainly fish (Miller et al 2004). Nevertheless, there is currently no empirical evidence that large croaker prey on weakfish of any size. On the other hand, since the diet of juvenile weakfish and croaker overlap, the strong statistical relationship given here between weakfish age 0 mortality and croaker abundance might indicate a rise in inter-specific competition for food between croaker and weakfish. If the outcome of recent inter-specific competition has favored croaker as indicated by the recent rise in croaker abundance (Table 14), somatic growth rates of age 0+ weakfish may have recently fallen, rendering juvenile and ages 1+ weakfish more susceptible to predation. This potential interaction between weakfish growth and predation mortality is directly supported by growth studies by Uphoff (2005b), who reported a steady decline in the coast-wide percentage of 13 in. + weakfish from 45.0% during the mid -1980's to about 1.0% in 2003. Although the strong statistical relationship between weakfish age 0 mortality and croaker abundance cannot be ruled out at this time, there is no empirical evidence thus far that Atlantic croaker are a major predator of juvenile weakfish.

The preponderance of statistical evidence given here supports the Predation Hypothesis involving striped bass as the primary predator of weakfish. Statistical evidence in support of the Predation Hypothesis consists of a strong inverse correlation (Pearson $r = -0.66$, $P < 0.0004$) between weakfish surplus production and striped bass abundance from 1982 to 2004 (Table 16). The strong correlation (Pearson $r = 0.91$, $P < 0.0001$) between weakfish juvenile mortality (Z_0) and striped bass abundance (Figure 35) also suggests that the emergence of a weakfish recruitment bottleneck at age 0 was largely due to striped bass. Finally, striped bass mortality rates (M_p) on ages 1+ weakfish derived empirically from the S-H model from 1982 to 2003 were

closely correlated (Pearson $r = 0.61$, $P < 0.002$) to ages 1+ natural mortality (MWt) rates derived independently from run #20 of the VPA.

Although results from regression and production models alone do not demonstrate causality, recent empirical evidence is consistent with the Predation Hypothesis involving striped bass. Striped bass are known to consume finfish prey up to 60% of their own body length (Manooch 1973). Unlike large fluke and Atlantic croaker that prey mainly on small (< 22 cm) fish, large (> 76 cm) striped bass can easily prey on larger (> 40 cm) weakfish. In addition, the recent drop in weakfish productivity after 1997 was shown to have a spatial component that matched the distribution of striped bass. Weakfish landings and surplus production fell rapidly after 1998 from the Mid and North Atlantic subregions, but surplus production from the South Atlantic subregion had remained steady (Crecco 2005c). It so happens that anadromous striped bass are very abundant along the Mid and North Atlantic coast throughout the year, but are seldom encountered in any numbers south of Cape Lookout, NC. The predator assemblage south of Cape Lookout, NC is largely composed of Gulf Stream predators such as king mackerel and cobia etc. This well-defined spatial match between prey (weakfish) and predator (striped bass) clearly supports the validity of the Predation Hypothesis. Further empirical support of the hypothesis includes recent food habit studies of striped bass (Walter and Austin 2003; Ruderghausen et al 2005), indicating that weakfish and other sciaenids (spot) were primary food items of larger (> 60 cm) striped bass in Chesapeake Bay and Albemarle Sound. Moreover, a recent rise in striped bass abundance has been linked empirically to the steady decline of blueback herring in the Connecticut River and Albemarle Sound (Savoy and Crecco 2004; Rudershausen et al 2005), as well as with the menhaden decline from Chesapeake Bay (Uphoff 2003; Walter and Austin 2003). These findings suggest that the recent rise in striped bass production to record levels from 1997 to 2004 is largely responsible for the recent decline in weakfish productivity and sustainable yield.

The strong inverse relationship between striped bass mortality (M_p) rates (Figure 36, Table 22) and weakfish biomass since 1997 (Figure 37, Table 11) is consistent with the presence of depensatory density-dependent mortality. This phenomenon plus the apparent emergence of a weakfish recruitment bottleneck between ages 0 and 1 (Figures 32 and 33) makes stock rebuilding via the implementation of fisheries management measures an exceedingly difficult task. As indicated by Spencer and Collie 1997), fish stocks that are subject to moderate to severe depensatory predatory mortality, often undergo a sudden and persistent drop in surplus production over time even when fishing mortality rates have remained low for several years. Note that the biomass weighted fishing mortality (FWt) on ages 1+ weakfish have been below the Steele-Henderson F_{msy} of 0.55 (Table 20) since about 1993 (Figure 36). Under depensatory predation, the weakfish stock is also expected to remain low and unresponsive to favorable climatic events and to further fishery management restrictions. Note that weakfish stock biomass has fallen steadily since 1999 to a 2005 stock biomass level of 7400 mt which is about 60% below the new B_{msy} level of 23,400 mt (Table 20, Figure 37). The phenomenon of depensatory mortality, if driven largely by striped bass predation, could lead to a persistent and perhaps irreversible failure in weakfish productivity unless striped bass productivity in the next few years reverts back to pre 1998 levels.

The pessimistic outlook regarding the future of the Atlantic coast weakfish due to depensatory predation may be tempered somewhat by our findings of compensatory density-dependent survival during or before the juvenile stage. Compensatory mortality was manifested here by the recent and persistent rise in the coast-wide juvenile recruitment (N_0) after 1995 (Figures 32) despite the pronounced drop in weakfish biomass after 1998 (Figure 37, Table 11). It is notoriously difficult to sort out the ramifications on future weakfish stock growth when both depensatory and compensatory mechanisms are operating simultaneously. However, the added stock resiliency due to compensatory processes at the juvenile stage may persist over several more years. If so, it may in fact overcome or at least balance out the adverse effects of depensatory predation, allowing the weakfish stock to achieve equilibrium at the current low levels for the foreseeable future.

The most restrictive management measures, such as a coast-wide moratorium to weakfish harvest, would reduce the 2004 F of 0.25 and landings to zero. A coast-wide moratorium would also reduce the high total mortality (Z) levels on ages 1+ weakfish by about 30% based on the 2004 ratio of fishing (F) to total mortality (F/Z). In a sense, the F/Z ratio is a relative measure of leverage that fishery managers can exert in order to enhance the chances of rebuilding depleted stocks. From 1982 to 1995, the F/Z ratios for weakfish were, in most years, well above 0.75, indicating the presence of relatively high leverage and thus a high probability that management measures if implemented then would have lead to stock rebuilding. As natural mortality (M) on ages 1+ weakfish increased after 1999 (Table 22), however, the F/Z ratios fell quickly below 0.35, thereby greatly reducing the likelihood that management measures imposed after 2005 would eventually lead to a weakfish stock recovery. This relationship between the probability of stock rebuilding via management action and the F/Z ratio was recently addressed by (Uphoff 2005d) by forecasting weakfish stock abundance over the next 20 years from the logistic production model. If M was allowed to remain at the 2003 level, he reported little if any future weakfish stock rebuilding following a 50% reduction in fishing mortality (F) if the current weakfish fishing mortality rates (F) on weakfish comprised less than 40% of total mortality (Z). The problem of weakfish stock rebuilding is made even worse by the emergence of a recruitment bottleneck at age 0 (Figure 33). Even a coast-wide moratorium to weakfish harvest would have little if any impact on the recently emergent recruitment bottleneck, where age 0 weakfish are not susceptible to direct harvest.

Like virtually all finfish stock assessments conducted along the Atlantic coast, natural mortality (M) of age 1+ weakfish was assumed to be constant ($M = 0.25$) in previous Yield-per-Recruit and VPA model runs. We found that the trend in annual biomass weighted natural mortality (MWt) from 1982 to 2003 did not support the constant M assumption (Table 10, Figure 34). Annual (MWt) estimates for ages 1+ weakfish rose steadily from 0.31 in 1996 to a high of 1.09 in 2002 (Figure 34). The management consequences of assuming a constant M when M actually rises systematically over time can be serious (Mertz and Myers 1997). As noted in this weakfish assessment, by holding M constant, the resulting ages 1+ fishing mortality rates (F) on weakfish would have risen steadily to around 0.85 in 2003 based on the VPA run #20. If the constant M assumption and ensuing VPA results were accepted without qualification, we would have concluded falsely that the recent failure in weakfish productivity was due to overfishing. In future assessments here and elsewhere, the assumption of constant M for ages 1+ fish needs to be critically examined. In addition, the impacts of trophic and environmental effects on exploited

finfish stocks should be integrated into fisheries models and rigorously tested as a potential alternative hypothesis to the overfishing hypothesis.

STOCK PROJECTIONS

The weakfish assessment has evolved into aggregated biomass modeling after the failure of ADAPT VPA to provide accurate estimates of recent stock parameters (Crecco 2005a; Kahn et al. 2005). Crecco (2005) used external surplus production modeling (Jacobson et al. 2002) to estimate overfishing thresholds for weakfish. Surplus production models can be used as a basis for projections (Prager 1994), but the underlying assumptions of ecological stability (Spencer and Collie 1997) and dependence of changes in biomass on F in the model would render their utility doubtful in a situation where M is changing and likely the dominant factor driving biomass dynamics.

The current weakfish assessment provides the pieces for an external production modeling approach that does not require an assumption of ecological stability. Estimates of full time-series (1981-2003) of F and biomass (B) were provided in Crecco (2005) and Z was estimated with minimal or no retrospective bias through 2000 (Kahn et al. 2005). Therefore, biomass-based estimates of Z ($F + M$) were available through 2000. If Z is considered the negative counterpart of the intrinsic rate parameter (r' ; Jordan and Coakley 2004), the Schaefer biomass dynamic model can be parameterized as

$$1) B_t = B_{t-1} + (r' - Z) \cdot B_{t-1} \cdot (1 - (B_{t-1} / K)).$$

This estimate of r' may not be equivalent to the standard estimate of r derived by production modeling because M can vary within the estimate of Z . In the discrete time-step form, an initial estimate of biomass, (B_{1981}), r' and K are estimated. The latter two parameters are critical for projecting biomass under assumed levels and trajectories of annual F and M . This model was used to project biomass into the future under assumed levels of F_t and M_t .

Methods

Development of projections was a two-step process. First, r' , K , and B_{1981} were estimated using the 1981-2000 estimates of B_t from Crecco (2005) and Z from Kahn et al. (2005; VPA with MRFSS tuning only). Estimates of B_t and Z_t were ultimately based on MRFSS private boat catch per effort as a biomass index and catch estimates that included discards (Crecco 2005a; Kahn et al. 2005). Equation 1 was fit in an Excel spreadsheet using Solver to minimize $\sum (\ln \text{observed } B_t - \ln \text{predicted } B_t)$. This was considered the base model for projections.

Instantaneous annual fishing mortality rate in year t was estimated from predicted biomass as

$$(2) F_t = H_{t-1} / [(B_t + B_{t-1}) / 2] \text{ (Ricker 1975);}$$

where B is estimated from the equation 1 and H is harvest (including estimated discards).

The second step completed the estimates of Z_t through 2003. Crecco's (2005) estimates of B_t for 2001-2003 were assumed to be accurate. Goalseek in Excel was used to estimate Z_t necessary to produce B_t given the parameters estimated by equation 1. With estimates of Z_t complete, M_t could be estimated as $Z_t - F_t$ through 2002. Since F_t in equation 2 was harvest based, F_{2003} could not be estimated without projecting biomass forward and F_{2002} represented the

last estimate in the time-series. An estimate of F_{2003} was available from Crecco (2005), so M could be estimated for 2003 in this case.

Bootstrapping ($N = 250$) indicated precision of r' , K , B_{1981} , B_t , and F_t of the base model for 1981-2003 data. To investigate model sensitivity, estimates of r and K from previous production models were used as a starting point ($r = 1.17$ and $K = 49,500$ mt; $r = 0.58$ and $K = 57,700$ mt; and $r = 0.90$ and $K = 64,028$ mt; predator-prey, Schaefer, and Gompertz, respectively; Uphoff 2005a; Crecco 2005a), held K constant, and estimated only r' and B_{1981} .

Projections of biomass through 2025 used base model r' , K , B_{2003} and mean estimates of F_t and M_t for 2001-2002 as a starting point. It was assumed that cuts in F would not occur until 2006, so biomass trajectory during 2004-2005 was based on a continuation of average conditions. Cuts were instituted in 2006 and maintained until 2025. Biomass trajectory was estimated for Z minus $F \cdot 0$ (no action), $F \cdot 0.25$, $F \cdot 0.5$, $F \cdot 0.75$, $F \cdot 1.0$ (complete moratorium with no discard loss), $F \cdot 1.5$, and $F \cdot 2.0$. The latter two cuts implied instant reductions in M as well as F .

The Technical Committee (TC) requested simulations that depicted a gradual decline in M to a baseline of $M = 0.25$. This level of M equaled the constant estimate of M used in previous SARC reviewed weakfish stock assessments (NEFSC 1998; 2000) and in the ADAPT VPA in the current assessment (Kahn et al. 2005). Use of a baseline M other than 0.25 in projections may have required rerunning VPA and rescaled relative F analyses (V. Crecco, CT DEP, personal communication).

The model was modified to portray declines in M over time to a base level of $M = 0.25$. Scenarios were run of cuts in F (0, 25%, 50%, 75%, and 100%) with constant declines in M_t to this baseline level. A decrease in M after 2006 until $M_t = 0.25$ was simulated as

$$(3) M_{t+1} = M_t \bullet \beta.$$

Once $M_t \bullet \beta = 0.25$ was reached, M_t was held constant at 0.25.

To gain insight on how rapidly M could decrease, the trajectory of the increase in M during 1996-2001 (M increased from 0.29 to 1.13) was reversed to mimic a rate of maximum decline (β). The estimate of M in 2002, 0.77, was not included because it was lower than the maximum estimated M in 2001 (1.13). Solver in Excel was used to estimate the initial M and β in equation 1 that minimized the sum of squared differences between the observed and estimated M . The model was further constrained so that M in the final year was 0.25. This model predicted the change in M well ($r^2 = 0.99$) and estimated β as 0.76. This estimate of β was rounded to 0.8. Values of β less than 0.8 tended to produce very similar results in previous projection exercises to $\beta = 0.8$ (Uphoff 2005a; 2005b).

Results and Discussion

The external production model fit the 1981-2000 biomass estimates well ($r^2 = 0.84$, $P < 0.001$; Figure 39). There was some serial patterning of residuals, with three to four year periods of over- or underestimation (Figure 39). Model parameters were estimated as $r' = 0.90$, $K = 86,409$ mt, and $B_{1981} = 40,692$ mt. Bootstrapping indicated these estimates were extremely precise, with a maximum difference of no more than 0.02% among these parameters. This precision carried

over to estimates of biomass and other parameters derived from biomass (F_t and M_t). Base model estimates of Z_t were 1.49 during 2001, 1.22 during 2002, and 1.45 during 2003.

Estimated F_t was high during 1981-1991, typically 0.6 or greater (Figure 40). Fishing mortality fell to about 0.2 by 1999 and then began a steady rise to 0.4 by 2002. Estimates of M_t were higher than F_t during 1982-1984 and then plunged to near zero by 1987. M_t varied from 0.2-0.5 during 1988-1998 and then increased to 0.8-1.1 during 2000-2002 (Figure 40).

Estimates of r' were insensitive to values of K used in the external production models (Table 23). Even though estimates of K varied between 49,500 and 86,400 mt, estimates of r' fell between 0.893 and 0.903 (Table 23). Estimates of r' increased linearly as K increased. High precision in estimating r' was important because the early phases of recovery would be dependent on it. Estimates of B_{1981} varied somewhat among the model conditions and all were less than estimated in Crecco (2005; Figure 41). Biomass estimated by external production models with varying assumptions about K converged with each other by 1985 and were very similar through 2000. These estimates diverged somewhat from those in Crecco (2005; Figure 41). This pattern of agreement was exhibited with estimates of F_t (Figure 42) and M_t (Figure 43) as well. The increase in M since the late 1980s exhibited here corresponded with trends in M_t from the Schaefer biomass dynamic model with a striped bass predation term (Uphoff 2005a).

Projections of biomass started in 2003 at 6,843 mt, $F_{\text{current}} = 0.41$, and $M_{\text{current}} = 0.95$. Modeled stock biomass dropped to about 2,000 mt before management actions were implemented in 2006 (Figure 44); this represented about 7% of mean 1998-2000 biomass that served as a target for recovery. Recovery was indicated once the mean 1998-2000 biomass (29,875 mt) was reached. The TC chose this value because it had been attained in the recent past; harvests and other attributes such as size quality were considered to be nearly recovered at that time. A value equal to 30% of K was considered as a target as well as a proxy for Amendment 4's 30% maximum spawning potential (30% MSP). This target was rejected because of substantial uncertainty of estimates of K among methods tried (Table 23) and the potential for it to change unexpectedly and drastically under the regime shift concept (Spencer and Collie 1995). Estimation of 30% MSP required equilibrium assumptions (constant growth and M) that could no longer be supported.

Weakfish biomass fell continuously in all cases when only F_{current} was cut and M_{current} was unchanged (Figure 44). In the two scenarios where current F was maintained or cut by 25%, weakfish became "extinct" (exploitable biomass = 0) after 2020 and 2025, respectively. Only the scenario of a completely effective moratorium, where discard mortality did not occur, kept biomass from falling to less than 100 mt by 2025. Under a completely effective moratorium, biomass was projected to fall to 875 mt in 2025 (Figure 44). Stemming the decline in weakfish biomass by managing the fishery alone seems unlikely based on these simulations unless the amount of Z attributed to F has been seriously underestimated (for instance, F is 0.8 instead of 0.4).

Biomass increased only when some depreciation of M_{current} was part of the reduction in Z_{current} (Figure 45; Table 23). Biomass slowly increased when Z_{current} was reduced by $1.5 \cdot F_{\text{current}}$

starting in 2006. Recovery did not occur until 2025 with the $1.5 \cdot F_{\text{current}}$ cut (Figure 45). A cut in Z_{current} equal to $2.0 \cdot F_{\text{current}}$ resulted in recovery by 2015 (Figure 45).

Without cuts in F_{current} , a decline to a baseline M of 0.25 when $\beta = 0.8$ did not allow biomass to approach the level of recovery through the modeled horizon (2006-2025; Figure 46). If F_{current} was cut by 25%, biomass recovered by 2024. When a 50% cut in F_{current} was instituted, recovery occurred by 2021 and recovery to target biomass occurred in 2018 when F_{current} was cut by 75% (Figure 46).

In light of the recent declines in landings, questions have been raised regarding the effectiveness of Amendment 3 of the FMP, imposed in 1995. The Amendment, bolstered by the federal Atlantic Coast Cooperative Fisheries Management Act, which made ASMFC management plans mandatory along the coast, included creel limits, minimum sizes, minimum commercial mesh sizes and restrictions on commercial effort. Following this, landings and relative abundance began to increase. A striking divergence appeared between the MRFSS estimates of total catch including discards (B2 component) and the estimated recreational harvest (A + B1 component) (Figure 2A). The former began rising well above estimates of the latter. This was the apparent conservation effect of minimum size limits and creel limits on the fishery. Previously, the two estimates had been almost identical. Estimates of scaled-up relative F have been relatively low and stable since 1995. ADAPT estimates of SSB showed increases in 1996 and 1997 before beginning the recent decline in 1998, according to Run 20. Abundance of the plus group began to increase after 1995 through 1999. Although this improvement in stock status did not persist, the technical committee strongly supports a continuation of at least the current fisheries conservation measures, because they had an apparent positive effect during the middle 1990s.

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TABLES

Table 1A. Decision Table for Weakfish Management. Table entries are possible or probable results of the given management action, given a correct biological hypothesis.

BIOLOGICAL HYPOTHESIS	MANAGEMENT ACTION		
	Restrict	Status Quo	Liberalize
Stock Increasing	Probably lower yield; possible further stock growth	higher yield	much higher yield
Stock Stable	Reduced yield; stock growth	stable yield	higher yield at first, but stock could decline
Stock Declining	Even lower yield ; reduction in total mortality and possible reduction in stock decline; including possible stabilization or even increase	low and declining yield, continued stock decline(?)	Possible temporary increase in yield, then greater stock decline (?)

TABLE 1B. Evidence in support of the three hypotheses of stock status and trends in Table 1A.

1. Analyses and data supporting increasing stock hypothesis:

- NMFS and NJ survey Total CPT
- results from ADAPT with trawl surveys as tuning indices (severe retrospective bias invalidates these results, though)
- ADAPT tuned to trawl surveys plus recreational catch per unit effort indices (again severe retrospective bias largely invalidate this)

2. Analyses and data supporting stable stock hypothesis:

- Biomass dynamics model with trawl survey tuning indices: stable and high

3. Analyses and data supporting declining stock hypothesis:

- decline in New Jersey age-structured trawl survey exploitable biomass catch per tow
- decline in Delaware age-structured trawl survey exploitable biomass catch per tow
- age structure truncation in Delaware trawl survey catch
- decline in recreational catch
- decline in commercial catch
- decline in recreational harvest per trip
- decline in recreational total catch per trip
- biomass dynamic models tuned to recreational fishery cpue indices
- results of ADAPT tuned to only recreational catch per unit effort as indices (again severe retrospective bias invalidates this result, however)
- decline in untuned cohort analysis through 1999
- length frequency analysis of Delaware and New Jersey survey catches (proportional stock densities) shows a decline in size structure
- stock size estimated by $N = \text{catch}/F$ from scaled up F estimates.
- decline in surplus production through 2002

Table 2. Atlantic coast harvest of weakfish from 1981 through 2003, by sector and total.

TOTAL YEAR	RECREATIONAL HARVEST			COMMERCIAL LANDINGS	COMMERCIAL PROPORTION OF HARVEST IN POUNDS	TOTAL LANDINGS	
	TOTAL CATCH, INCLUDING DISCARDS (A+B1+B2)	HARVEST IN NUMBERS (A + B1)	HARVEST IN POUNDS	POUNDS		POUNDS	METRIC TONS
1981	9,629,309	9,344,461	16,105,028	26,363,607	0.62	42,468,635	19,260.15
1982	2,044,670	1,854,090	8,285,326	19,478,274	0.70	27,763,600	12,591.2
1983	5,916,269	5,642,950	11,730,619	17,475,003	0.60	29,205,622	13,245.18
1984	3,769,040	3,520,811	7,013,781	19,773,587	0.74	26,787,368	12,148.47
1985	2,775,824	2,419,670	5,489,026	16,953,357	0.76	22,442,383	10,177.95
1986	10,973,586	8,664,122	10,141,786	21,187,973	0.68	31,329,759	14,208.51
1987	5,719,807	4,871,532	6,749,890	17,072,159	0.72	23,822,049	10,803.65
1988	6,446,383	5,626,268	6,331,649	20,526,402	0.76	26,858,051	12,180.52
1989	1,674,568	1,495,391	2,177,237	14,163,008	0.87	16,340,245	7,410.54
1990	1,671,808	1,232,253	1,347,260	9,438,260	0.88	10,785,520	4,891.39
1991	2,601,480	1,812,691	2,130,563	8,692,760	0.80	10,823,323	4,908.54
1992	1,667,809	960,151	1,398,980	7,453,788	0.84	8,852,768	4,014.86
1993	2,218,559	1,079,275	1,102,340	6,853,579	0.86	7,955,919	3,608.13
1994	4,928,951	1,826,495	1,795,517	6,190,520	0.78	7,986,037	3,621.79
1995	5,696,423	1,588,079	1,855,548	7,098,831	0.79	8,954,379	4,060.94
1996	7,306,298	2,269,330	2,925,392	6,940,578	0.70	9,865,970	4,474.36
1997	6,832,363	2,815,654	3,692,716	7,297,859	0.66	10,990,575	4,984.39
1998	5,697,395	2,386,345	4,044,974	8,424,725	0.68	12,469,699	5,655.19
1999	4,477,989	1,651,554	3,143,427	6,903,940	0.69	10,047,367	4,556.63
2000	6,727,052	2,089,202	4,154,794	5,372,672	0.56	9,527,466	4,320.85
2001	5,233,868	1,526,583	2,722,630	4,988,718	0.65	7,711,348	3,497.21
2002	3,289,765	1,171,889	2,192,607	4,767,232	0.68	6,959,839	3,156.39
2003	2,098,056	497,571	864,962	1,970,291	0.69	2,835,253	1,285.83
2004	2,602,057			1,178,582			
2005							

Table 2A. Age and length sample intensity by half-year and state used for the catch-at-age calculations since the 30th SAW.NMFS (2000), which reviewed the weakfish assessment from 1982-1998. Age samples for the SEAMAP fall and NEFSC fall inshore surveys were produced by Dr. Charles Wenner, South Carolina Department of Natural Resources.

YEAR	SEASON	STATE	AGED	MEASURED FROM	COMM LANDINGS	COMM	MEASURED FROM	RECREATIONAL	REC
				COMM LANDINGS	MT	LENGTHS / MT	REC LANDINGS, MRFSS	LANDINGS, MT	LENGTHS / MT
1999	EARLY	MA	0	0	0.2	0.0	0	0.00	0
		RI	0	0	18.3	0.0	2	0.78	2.55
		CT	0	0	2.7	0.0	1	0.94	1.06
		NY	0	0	39.3	0.0	10	17.74	0.56
		NJ	0	0	115.3	0.0	48	188.49	0.25
		DE	335	4256	163.8	26.0	96	88.28	1.09
		MD	1	43	28.5	1.5	20	28.26	0.71
		VA	159	1182	345.6	3.4	56	93.43	0.60
		NC	309	9995	991.4	10.1	135	33.55	4.02
		SC	0	0	0.3	0.0	1	0.27	3.73
		GA	102	0	0.3	0.0	0	0.00	0
		FL	0	43	4.5	9.6	7	17.01	0.41
		SEAMAP	SURVEY	229					
	TOTAL	1,135	15,519	1,710	9.1	376	468.8	0.80	
1999	LATE	MA	0	0	1	0.0	0	0.00	0.00
		RI	0	0	38.9	0.0	2	1.88	1.06
		CT	0	0	7.5	0.0	2	1.95	1.03
		NY	0	0	182.8	0.0	15	10.86	1.38
		NJ	0	0	470.4	0.0	152	434.84	0.35
		DE	0	70	36.3	1.9	134	131.34	1.02
		MD	226	556	72.7	7.6	242	218.71	1.11
		VA	374	2128	414.2	5.1	61	87.82	0.69
		NC	348	5752	186.5	30.8	123	39.61	3.11
		SC	0	0	0	0	3	1.72	1.74
		GA	129	0	0	0	2	0.73	2.73
		FL	10	13	3.5	3.7	81	27.65	2.93
		SEAMAP	SURVEY	255					
NEFSC	SURVEY	373							
	TOTAL	1,715	8,519	1,414	6.0	817	957	0.85	
2000	EARLY	MA	0	0	0.0		0	0.00	0.00
		RI	0	46	17.1	2.7	1	0.87	1.15
		CT	0	0	1.0	0.0	3	2.92	1.03
		NY	0	0	57.8	0.0	7	59.53	0.12
		NJ	0	0	173.5	0.0	45	299.31	0.15
		DE	481	1995	106.8	18.7	54	177.02	0.31
		MD	113	113	22.4	5.0	22	31.05	0.71
		VA	165	1219	258.8	0.0	117	142.25	0.82
		NC	424	11816	608.7	4.8	51	18.80	2.71
		SC	0	0	0.0	0	0	0.00	0.00
		GA	0	0	0.0	0	1	0.20	5.13
		FL	25	25	1.9	4.8	47	24.12	1.95
		SEAMAP	SURVEY						
	TOTAL	1,208.0	15,214.0	1,248.0	12.2	348.0	756.1	0.46	
2000	LATE	MA	0	0	0.0		0	0.00	0.00
		RI	0	132	58.6	2.3	0	0.00	0.00
		CT	0	0	2.2	0.0	12	13.00	0.92
		NY	0	0	76.3	0.0	2	15.10	0.13
		NJ	0	0	312.4	0.0	132	569.83	0.23
		DE	965	224	21.1	10.6	20	111.17	0.18
		MD	99	99	60.7	1.6	221	284.95	0.78
		VA	390	1885	365.7	5.2	77	82.83	0.93
		NC	192	6426	232.4	27.7	71	21.08	3.37
		SC					9	2.86	3.14
		GA					3	1.40	2.15
		FL		0	2.1	0.0	29	26.33	1.10
		SEAMAP	SURVEY						
NEFSC	SURVEY								
	TOTAL		8,766.0	1,131.5	7.7	576.0	1,128.6	0.5	

Table 2A continued

YEAR	SEASON	STATE	AGED	MEASURED FROM COMM LANDINGS			RECREATIONAL		REC LENGTHS/MT
				COMM LANDINGS	MT	COMM LENGTHS/MT	REC LANDINGS, MRFSS	LANDINGS, MT	
2001	EARLY	MA	0	0	0.0	0.0	0	0.00	0.00
		RI	178**	178**	10.8	16.5	0	0.00	0.00
		CT	34	0	0.3	0.0	0	0.00	0.00
		NY	0	0	39.9	0.0	9	39.04	0.23
		NJ	0	0	108.0	0.0	43	114.82	0.37
		DE	300	370	75.0	4.9	69	50.69	1.36
		MD	0	8	27.3	0.3	5	15.61	0.32
		VA	152	758	249.9	3.0	82	107.63	0.76
		NC	328	9,747	723.3	13.5	19	2.44	7.78
		SC					0	0.00	0.00
		GA			0.0		1	0.12	8.06
		FL		0	2.4	0.0	6	10.42	0.58
		SEAMAP SURVEY	99	na	na		na		
		TOTAL	992	11,306	1,237	9.1	234	341	0.7
		2001	LATE	MA	0	0	0.1	0.0	0
RI	178**			178**	38.9	4.6	0	0.00	0.00
CT	69			0	2.7	0.0	1	2.22	0.45
NY	0			372***	222.9	1.7	3	29.72	0.10
NJ	0			0	271.8	0.0	362	452.70	0.80
DE	861			0	2.7	0.0	59	27.77	2.12
MD	193			261	68.2	2.8	294	241.87	1.22
VA	420			1806	280.6	6.4	106	61.66	1.72
NC	220			3199	158.8	21.6	161	69.42	2.32
SC							0	0.00	0.00
GA							2	1.22	1.64
FL				0	2.5	0.0	17	7.63	2.23
SEAMAP SURVEY	151			na	na		na		
NEFSC SURVEY	617			na	na		na		
TOTAL	2,699			6,003	1,049.3	5.7	1,005	894	1.1
2002	EARLY	MA	0	0	0.2	0.0	0	0.00	0.00
		RI	50	50	30.4	1.6	0	0.00	0.00
		CT	22	0	2.6	0.0	0	0.00	0.00
		NY	0	0	45.0	0.0	4	9.48	0.42
		NJ	0	0	92.4	0.0	101	228.98	0.44
		DE	561	1,179	54.9	21.5	201	81.06	2.48
		MD	20	21	11.0	1.9	12	10.80	1.11
		VA	328	2,399	325.8	7.4	110	39.92	2.76
		NC	231	9,121	691.8	13.2	47	7.92	5.93
		SC					0	0.18	0.00
		GA					0	0.00	0.00
		FL		0	1.6	0.0	17	21.17	0.80
		SEAMAP SURVEY	122*	na	na		na		
		CHESMAF SURVEY	141	na	na		na		
		TOTAL	1,478	12,770	1,255.74	10.2	492	400	1.2
2002	LATE	MA	0	0	0.2	0.0	0	0.00	0.00
		RI	0	0	25.2	0.0	2	1.72	1.16
		CT	42	0	2.0	0.0	1	5.12	0.20
		NY	0	0	188.1	0.0	8	17.11	0.47
		NJ	0	0	299.0	0.0	164	321.67	0.51
		DE	760	0	23.5	0.0	58	29.23	1.98
		MD	44	216	44.8	4.8	58	68.16	0.85
		VA	318	4170	211.2	19.7	141	94.08	1.50
		NC	281	3642	130.8	27.8	59	29.61	1.99
		SC					7	22.56	0.31
		GA					1	0.31	3.22
		FL		0	1.1	0.0	21	5.66	3.71
		SEAMAP SURVEY	153	na	na		na		
		CHESMAF SURVEY	550	na	na		na		
		NEFSC SURVEY	692	na	na		na		
TOTAL	2,838	8,028	925.9	8.7	520	595	0.9		

Table 2A, continued

YEAR	SEASON	STATE	AGED	MEASURED FROM	COMM LANDINGS	COMM	MEASURED FROM	RECREATIONAL	REC
				COMM LANDINGS	MT	LENGTHS / MT	REC LANDINGS, MRFSS	LANDINGS, MT	LENGTHS / MT
2003	EARLY	MA	0	0	0.2	0.0	0	0.00	0.00
		RI	0	0	8.4	0.0	0	0.00	0.00
		CT	4	0	4.6	0.0	0	0.00	0.00
		NY	0	0	30.6	0.0	1	11.55	0.09
		NJ	36	104	70.4	1.5	22	58.20	0.38
		DE	580	944	38.4	24.6	14	4.77	2.94
		MD	0	7	3.2	2.2	0	0.00	0.00
		VA	350	1900	100.5	18.9	41	49.83	0.82
		NC	269	5227	267.9	19.5	42	15.64	2.68
		SC					0	0.00	0.00
		GA					0	0.00	0.00
		FL		0	1.0	0.0	9	4.50	2.00
		SEAMAP SURVEY	219*	na	na		na		
		CHESMAP SURVEY	78	na	na		na		
		TOTAL	1500		8,182	525.2	15.6	129	144
2003	LATE	MA	0	0	0.0	0.0	1	0.40	2.53
		RI	211	343	20.3	16.9	1	1.08	0.93
		CT	22	0	0.3	0.0	1	1.60	0.62
		NY	0	0	34.7	0.0	4	5.28	0.76
		NJ	29	0	83.6	0.0	38	93.16	0.41
		DE	372	0	3.1	0.0	23	21.48	1.07
		MD	202	276	11.1	24.8	17	11.20	1.52
		VA	323	2226	108.1	20.6	49	47.93	1.02
		NC	220	3523	114.1	30.9	89	57.60	1.55
		SC					2	1.95	1.02
		GA					3	0.60	4.98
		FL		0	0.2	0.0	13	5.56	2.34
		SEAMAP SURVEY	0	na	na		na		
		CHESMAP SURVEY	595*	na	na		na		
		NEFSC SURVEY	0						
TOTAL	1975		6368	375.5	17.0	241	247.9	1.0	

* MANY AGED WERE AGE 0

**EARLY AND LATE SEASONS POOLED

***NMFS AT-SEA OBSERVER DATA FROM NY-BASED TRAWLS

Table 3. Results of catch curve analysis of the four age-structured research trawl surveys. Each year class represented in each survey was subjected to catch curve analysis. For each survey, the table presents the number of ages represented for each year class, the statistical probability that the linear regression is due to chance alone and the resulting estimate of Z. Entries in bold are year classes present in the terminal year of the survey when they function as tuning indices in the estimation of stock size in the terminal year plus one. Estimates of Z that are negative are highlighted. Negative Z indicates a cohort gained animals instead of lost them as time proceeded, because the catch curve has a positive slope. These results are biologically nonsensical, and are grounds for disqualifying a survey if those year classes are present in the terminal year.

YEAR CLASS	NEFSC			NEW JERSEY			DELAWARE			SEAMAP		
	NUMBER OF AGES	P	Z	NUMBER OF AGES	P	Z	NUMBER OF AGES	P	Z	NUMBER OF AGES	P	Z
1981	4	<i>ns</i>	0.81									
1982	4	<i>ns</i>	0.57									
1983	4	0.02	0.94									
1984	4	0.083	1.83									
1985	4	0.13	1.5									
1986	4	<i>ns</i>	0.54									
1987	4	<i>ns</i>	0.43									
1988	4	0.055	1.37	6	0.003	1.09						
1989	4	0.016	1.59	6	0.019	1.76						
1990	4	0.023	1.7	6	0.009	1.06	6	0.003	1.12	4	0.104	2.01
1991	4	0.067	0.9	6	0.096	0.37	6	0.027	0.70	4	<i>ns</i>	0.42
1992	4	<i>ns</i>	0.57	6	<i>ns</i>	0.29	6	0.009	0.78	4	<i>ns</i>	0.72
1993	4	<i>ns</i>	-0.12	6	0.047	0.77	6	0.02	0.56	4	0.024	2.14
1994	4	0.076	1.18	6	0.0035	0.74	6	0.0004	0.88	4	0.12	1.48
1995	4	0.013	1.43	6	0.0005	0.95	6	0.0045	1.23	4	<i>ns</i>	0.35
1996	4	0.102	0.82	6	0.037	0.75	6	0.013	1.51	4	0.11	1.79
1997	4	<i>ns</i>	-0.06	6	0.075	0.66	6	0.18	1.49	4	<i>ns</i>	1.23
1998	4	0.053	0.6	6	<i>ns</i>	0.51	5	0.062	1.90	4	0.10	2.18
1999	4	0.13	0.63	5	0.053	0.60	4	0.11	1.68	4	<i>ns</i>	1.13
2000	4	<i>ns</i>	0.38	4	0.079	-1.80	3	<i>ns</i>	1.19	4	0.04	1.34
2001	3	0.08	-0.52	3	<i>ns</i>	0.10	2	--	Pos. Z	3	<i>ns</i>	0.91
2002	2	--	Neg. Z	2	--	Pos. Z				2	--	Pos Z

Table 4. Commercial discard at age matrix for Mid-Atlantic based on a reduced suite of target species by year for all gears combined using the NMFS web site non-seasonal landings. This is table 47 in de Silva (2004).

Year	Age 0	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6+	Total
1973	496,272	1,798,471	432,026	10,992	1,353	328	63	2,739,505
1974	358,872	1,276,077	537,044	97,007	29,789	6,756	1,310	2,306,855
1975	327,205	1,675,847	477,224	128,566	34,446	7,857	1,529	2,652,674
1976	398,670	1,794,891	838,439	139,509	41,493	9,461	1,845	3,224,308
1977	118,146	857,487	815,590	138,242	21,875	6,298	52	1,957,690
1978	530,594	1,726,687	1,030,398	115,699	2,054	404	65	3,405,901
1979	1,405,045	2,936,122	1,205,998	179,724	75,242	17,090	3,314	5,822,535
1980	432,452	2,474,408	1,490,736	204,834	52,510	15,268	278	4,670,486
1981	1,119,410	2,150,565	1,139,045	242,816	4,434	899	117	4,657,286
1982	1,231,231	2,020,770	1,173,279	54,421	4,153	1,405	270	4,485,529
1983	338,196	1,573,161	769,150	120,319	4,590	1,090	204	2,806,710
1984	1,431,711	2,911,017	844,951	83,804	5,562	1,335	303	5,278,683
1985	664,091	1,363,018	572,220	112,636	5,630	1,315	260	2,719,170
1986	339,109	1,578,321	778,022	127,113	7,668	1,799	328	2,832,360
1987	226,608	1,279,953	809,783	93,259	6,879	1,685	356	2,418,523
1988	66,507	1,209,939	698,020	89,845	34,294	8,202	1,505	2,108,312
1989	434,524	1,459,189	508,949	27,032	4,681	1,125	233	2,435,733
1990	107,696	615,116	353,646	81,365	4,603	957	168	1,163,551
1991	112,805	692,218	365,542	64,104	18,979	4,420	914	1,258,982
1992	280,006	873,586	364,007	35,170	3,343	815	159	1,557,086
1993	126,917	1,205,400	586,812	53,935	5,533	1,179	207	1,979,983
1994	219,550	378,474	934,132	355,722	127,660	2,953	945	2,019,436
1995	810,598	576,647	421,332	125,037	42,341	1,329	455	1,977,739
1996	1,026,321	643,909	403,408	608,121	138,314	28,882	1,418	2,850,373
1997	81,776	488,903	550,869	229,738	318,110	35,324	8,578	1,713,298
1998	262,115	300,532	296,292	309,302	79,709	70,208	26,904	1,345,062
1999	2,347,445	416,099	104,082	123,654	183,637	15,922	15,080	3,205,919
2000	766	224,967	453,005	276,424	84,084	27,765	3,605	1,070,616
2001	1,872,576	342,137	432,406	201,192	86,170	17,497	10,359	2,962,337
2002	1,739,612	747,214	153,041	75,843	16,362	5,765	1,809	2,739,646

Table 5. The catch-at-age matrix through 2003, including commercial and recreational harvest and discards.

	1	2	3	4	5	6+
1982	9,914	12,967	5,473	2,778	722	639
1983	8,004	12,869	5,823	2,780	568	424
1984	10,444	14,737	6,521	3,045	485	255
1985	14,153	11,262	3,246	1,171	213	55
1986	18,611	15,778	4,942	1,824	264	52
1987	16,256	14,343	4,347	1,485	145	11
1988	8,162	16,141	10,545	6,092	1,051	71
1989	3,705	5,305	4,334	2,922	626	85
1990	9,510	4,890	2,094	1,205	591	89
1991	9,796	5,826	2,750	1,374	463	57
1992	5,179	6,046	2,211	1,255	528	65
1993	4,975	6,357	2,180	1,139	401	48
1994	3,544	3,810	3,357	1,490	202	39
1995	4,047	3,517	3,504	1,617	197	54
1996	2,126	2,456	4,682	3,094	1,363	99
1997	1,459	2,104	2,792	5,355	1,505	406
1998	1,136	2,005	3,844	1,983	2,897	897
1999	1,221	1,252	2,200	3,241	718	1,138
2000	1,159	1,499	1,939	1,838	1,851	466
2001	549	2,279	1,255	819	431	370
2002	1,142	700	2,451	808	391	217
2003	857	1,066	742	554	104	130

Table 6. Catch curve analysis results for the catch at age matrix, analysed by year class. Number of ages indicates the number of ages used in the regression, P indicates the statistical significance of the regression, R^2 indicates the proportion of variance explained and F value is the F statistic of the regression. Note that Fishing mortality = $Z - M$, where $M = 0.25$. 1993 data was not amenable to catch curve analysis, because the modal age was too old.

Year Class	# Ages	Z	Fishing Mortality	P	R^2	F value
1982	3	1.55	1.30	0.22	0.77	7.59
1983	3	0.77	0.52	0.20	0.82	9.82
1984	3	0.97	0.72	0.42	0.25	1.65
1985	4	1.08	0.83	0.04	0.90	26.87
1986	4	1.19	0.94	0.002	0.99	389.89
1987	4	0.73	0.48	0.001	0.97	98.79
1988	3	0.96	0.71	0.068	0.98	87.57
1989	4	1.07	0.82	0.020	0.94	48.57
1990	4	1.06	0.81	0.042	0.88	22.20
1991	3	0.45	0.20	0.219	0.77	7.78
1992	4	0.29	0.04	0.115	0.68	7.25
1993	-	-				
1994	8	0.96	0.71	0.0003	0.89	57.61
1995	5	1.40	1.15	0.002	0.96	90.48
1996	4	1.31	1.06	0.006	0.98	159.89
1997	3	1.51	1.26	0.174	0.85	12.77
1998	3	1.26	1.01	0.220	0.77	7.69

Table 7. Estimates of F, instantaneous fishing mortality by age and year from Gulland's cohort analysis performed on the catch at age matrix.

YEAR	AGE						FULLY-RECRUITED F (average of ages 4 and 5)
	1	2	3	4	5	6+	
1982	0.29	0.70	0.81	1.24	0.88	---	1.06
1983	0.25	0.79	0.86	1.60	1.04	---	1.32
1984	0.31	1.07	1.47	2.24	2.08	---	2.16
1985	0.29	0.70	0.79	1.47	1.45	---	1.46
1986	0.36	0.66	0.84	1.95	2.95	---	2.45
1987	0.42	0.55	0.40	0.71	0.98	---	0.85
1988	0.44	1.05	1.16	2.00	2.39	---	2.20
1989	0.23	0.62	1.02	1.50	1.89	---	1.70
1990	0.51	0.56	0.58	1.01	2.22	---	1.61
1991	0.50	0.75	0.78	1.05	1.90	---	1.48
1992	0.21	0.71	0.78	1.17	2.27	---	1.72
1993	0.19	0.46	0.65	1.46	2.20	---	1.83
1994	0.10	0.23	0.51	1.53	1.39	---	1.46
1995	0.23	0.14	0.36	0.52	0.96	---	0.74
1996	0.09	0.22	0.29	0.66	1.32	---	0.99
1997	0.12	0.13	0.45	0.69	0.87	---	0.78
1998	0.13	0.24	0.40	0.71	1.14	---	0.92
1999	0.17	0.21	0.49	0.75	0.66	---	0.71
2000	0.13	0.34	0.62	1.12	1.63	---	1.37
2001	0.14	0.42	0.56	0.62	0.97	---	0.79
2002	0.50	0.28	1.25	0.97	0.74	---	0.85
2003	---	---	---	---	---	---	---

Table 8. Stock size by year and age estimated by Gulland's Cohort Analysis of the catch at age matrix. Newton's method was used for the final estimates (Haddon 2001).

YEAR	AGE						TOTAL
	1	2	3	4	5	6+	1 - 6+
1982	44,849	28,833	10,985	4,309	1,369	665	91010.2
1983	40,777	26,254	11,204	3,816	971	442	83464.2
1984	43,793	24,748	9,300	3,690	602	266	82399.5
1985	62,727	24,972	6,582	1,669	305	59	96314.5
1986	69,393	36,470	9,676	2,315	299	56	118208.5
1987	53,490	37,783	14,701	3,260	257	12	109504.1
1988	25,546	27,472	16,944	7,655	1,250	75	78942.4
1989	20,487	12,777	7,488	4,129	803	90	45773.4
1990	26,468	12,709	5,341	2,096	719	94	47426.6
1991	28,034	12,330	5,643	2,338	593	61	49000.6
1992	30,535	13,299	4,553	2,013	638	69	51105.9
1993	32,680	19,241	5,112	1,631	489	51	59204.4
1994	43,206	21,090	9,443	2,088	296	42	76164.6
1995	22,418	30,535	13,087	4,431	354	58	70881.7
1996	27,043	13,914	20,695	7,130	2,043	105	70929.0
1997	14,991	19,193	8,684	12,021	2,866	423	58178.8
1998	10,849	10,394	13,100	4,328	4,715	931	44318.1
1999	8,910	7,452	6,339	6,846	1,650	1,180	32377.8
2000	10,829	5,868	4,706	3,020	2,522	661	27607.6
2001	4,789	7,416	3,260	1,980	771	386	18602.2
2002	3,257	3,248	3,787	1,447	830	228	12796.5
2003	878	1,542	1,917	845	429	307	5918.0

Table 9A. Summary of selected 1982-2003 ADAPT VPA runs for wekfish.

Indices	24 All Trawl Surveys	14 DE, SEAMAP surveys YOY Surveys	18 DE, SEAMAP surveys Rec. Harvest/Trip at age Total Rec Catch/trip	20 Rec. Harvest/Trip at age Total Rec Catch/trip	32 Rec. Harvest/Trip at age Total Rec Catch/trip
M	constant 0.25	constant 0.25	constant 0.25	constant 0.25	constant 0.25 1982-1995 increasing 1995 - 2001 constant 2001-2003 M = 0.6
Number of indices used	27	17	19	5	5
MSR	1.29	1.192	1.445	0.687	0.87
2004 N; CV					
	1 68,477; 0.39	57,287; 0.0019	37922; 0.47	not estimated	not estimated
	2 50,912; 0.32	49,403; 0.0022	31,019; 0.40	not estimated	not estimated
	3 33,654; 0.29	35,203; 0.0031	16,599; 0.38	2227.8; 0.72	2629; 0.80
	4 11,775; 0.27	18,728; 0.0058	5,671; 0.39	258.3; 0.80	320; 0.86
	5 39,909; 0.18	60,381; 0.0018	8,741; 0.29	164.6; 0.76	127; 0.86
6+	1,592; 1.18	3,163; 0	80; 0.12	64.1 0.85	73; 0.96
2003 F					
	1 0.1	0.02	0.02	0.19	0.17
	2 0.03	0.03	0.05	0.35	0.26
	3 0.05	0.03	0.1	1.24	0.96
	4 0.01	0.01	0.05	1.36	1.38
	5 0.06	0.04	0.12	1.59	1.41
6+	0.06	0.04	0.12	1.59	1.41
2003 F ₄₋₅	0.03	0.03	0.09	1.48	1.39
zero indices treated as	0.01	0.01	missing	missing	missing
Method for full F year T	Average Method	Average Method	Average Method	Average Method	Average Method

Table 9B. Retrospective bias in four ADAPT runs: 24, 13, 18 and 20. Output estimates of F and SSB when only data through 1998 is input, and when data through 2003 is input and resulting per cent change in estimates.

RUN NUMBER:	24	14	18	20
F ₁₉₉₈ with 1998 data	0.10	0.10	0.30	2.05
SSB ₁₉₉₈ with 2003 data	59,102	85,561	36,482	5,468
F ₁₉₉₈ with 2003 data	0.43	0.36	0.64	0.70
SSB ₁₉₉₈ with 2003 data	22,599	27,551	14,156	12,674
% change in F ₁₉₉₈	+207	+260	+113	-66
% CHANGE IN SSB ₁₉₉₈	-62	-68	-61	+132

Table 9C. Comparison of biomass dynamic model sensitivity analyses: estimates of intrinsic rate (r) and unfished stock biomass (K , mt of 25 cm+ weakfish). Model is a general description of indices and treatments; Indices is the number of indices included in the sensitivity run, q 's is the number of scalars estimated; and Discards indicates whether discard estimates were included with landings. NE = Northeast Fisheries Science Center Survey. All surveys use trawls. Regression coefficients were not calculated for every model.

Model	Years	Indices	q 's	Discards	r^2	N	$(SSQ/N)^{0.5}$	r	K
NE, DE, NJ Z transformed	1981-2003	3	1	No	0.41	56	0.66	1.32	41912
NE, DE, NJ Z transformed	1981-2003	3	1	Yes		56	0.49	1.35	39910
Mean of NE, DE, NJ Z indices	1981-2003	1	1	No	0.58	23	0.44	1.10	47260
Mean of NE, DE, NJ Z indices	1981-2003	1	1	Yes		23	0.23	1.44	41950
Multiple q 's, NE, DE, NJ	1981-2003	3	3	No	0.56	56	0.77	1.80	41912
NEFSC	1981-2003	1	1	No	0.41	23	0.78	0.83	53581
NJ	1989-2003	1	1	No	0.26	15	0.61	1.12	56889
DE	1990-2003	1	1	No	0.48	14	0.52	0.85	1.54E+08
DE	1981-1984, 1990-2003	1	1	No	0.33	18	0.51	0.33	251708
DE, 2 q 's for years	1981-1984, 1990-2003	1	2	No	0.49	18	0.46	1.08	82043
DE&NJ Z indices	1981-2003	2	1	No	0.48	38	0.56	1.37	40850
NE&NJ Z indices	1981-2003	2	1	No		41	0.54	1.34	35926
NE&DE Z indices	1981-2003	2	2	No		33	0.65	1.35	36141
NE, DE, NJ Z transformed	1975-2003	3	1	No	0.33	64	0.36	0.33	38313

TABLE 10. TOTAL HARV+DISCARDS (HARVDIS, mt), MID-ATLANTIC PRIVATE BOAT FISHING EFFORT (TRIPS * 1000),MID-ATLANTIC PRIVATE BOAT REC CATCH (MIDHARV2,MT), PRIVATE BOAT CPUE (RelWt) AND RelWt+1 IN T+1, AVERAGE AVRelWt IN YEARS T AND T+1, RELATIVE F AND SCALED FWt AND BIOMASS WEIGHTED M (MWt), 1982-2005

YEAR	HARVDIS	teffort	midharv2	RelWt	relwt1	avrelwt	RelFwt	FWt	MWt
1982	13823	6028	2805.90	0.46548	0.50041	0.48294	28622.46	0.40644	0.57
1983	14426	8472	4239.46	0.50041	0.31305	0.40673	35468.49	0.50365	0.67
1984	13338	7881	2467.12	0.31305	0.27831	0.29568	45110.00	0.64056	0.82
1985	11181	6994	1946.49	0.27831	0.41802	0.34816	32114.40	0.45602	0.49
1986	15590	10003	4181.41	0.41802	0.35225	0.38513	40479.53	0.57481	0.44
1987	11691	8554	3013.15	0.35225	0.30444	0.32835	35605.73	0.50560	0.23
1988	13090	8862	2697.96	0.30444	0.12821	0.21633	60510.07	0.85924	0.64
1989	8136	6756	866.21	0.12821	0.08615	0.10718	75909.95	1.07792	0.01
1990	5440	7881	678.91	0.08615	0.10483	0.09549	56970.88	0.80899	0.06
1991	5610	8713	913.38	0.10483	0.13552	0.12017	46682.86	0.66290	0.34
1992	4793	6904	935.60	0.13552	0.08764	0.11158	42955.58	0.60997	0.39
1993	4448	8750	766.89	0.08764	0.18681	0.13723	32413.81	0.46028	0.33
1994	5395	9087	1697.51	0.18681	0.28561	0.23621	22840.19	0.32433	0.25
1995	5977	8581	2450.79	0.28561	0.37890	0.33226	17989.19	0.25545	0.22
1996	6879	8844	3351.02	0.37890	0.34514	0.36202	19001.69	0.26982	0.31
1997	7206	9725	3356.46	0.34514	0.44201	0.39357	18309.24	0.25999	0.37
1998	7809	8630	3814.51	0.44201	0.37756	0.40978	19056.48	0.27060	0.47
1999	6219	7935	2995.92	0.37756	0.41587	0.39671	15676.32	0.22260	0.56
2000	6126	11324	4709.30	0.41587	0.23894	0.32741	18710.67	0.26569	0.81
2001	5369	11982	2863.04	0.23894	0.23951	0.23923	22443.23	0.31869	0.61
2002	4434	9551	2287.53	0.23951	0.11906	0.17929	24731.48	0.35119	1.09
2003	2660	11286	1343.76	0.11906	0.05831	0.08869	29993.56	0.42591	0.76
2004	1483	11146	649.89	0.05831	0.10468	0.08150	18197.39	0.25840	.
2005	1937	11866	1242.18	0.10468	.	.	18503.36	0.26275	.

**TABLE 11. WEAKFISH AVERAGE BIOMASS ($BIOT_t$, MT) IN YEAR T, BIOMASS IN T+1 ($BIOT_{t+1}$, MT) TOTAL WEAKFISH HARVEST (HARVDIS) AND WEAKFISH SURPLUS PRODUCTION ($SURP_t$), 1982-2005
NOTE THAT $SURP_t$ ESTIMATE IN 2005 NOT POSSIBLE WITHOUT A 2006 BIOMASS ESTIMATE**

YEAR	BIOT	BIOT1	HARVDIS	SURPt
1982	34010.03	28642.76	13823	8455.73
1983	28642.76	20822.34	14426	6605.58
1984	20822.34	24518.42	13338	17034.08
1985	24518.42	27122.04	11181	13784.62
1986	27122.04	23122.96	15590	11590.92
1987	23122.96	15234.34	11691	3802.38
1988	15234.34	7547.86	13090	5403.52
1989	7547.86	6724.46	8136	7312.60
1990	6724.46	8462.86	5440	7178.39
1991	8462.86	7857.77	5610	5004.91
1992	7857.77	9663.76	4793	6598.99
1993	9663.76	16634.26	4448	11418.49
1994	16634.26	23398.25	5395	12158.99
1995	23398.25	25494.39	5977	8073.15
1996	25494.39	27716.32	6879	9100.93
1997	27716.32	28857.87	7206	8347.55
1998	28857.87	27937.54	7809	6888.67
1999	27937.54	23056.82	6219	1338.28
2000	23056.82	16846.89	6126	-83.93
2001	16846.89	12625.75	5369	1147.86
2002	12625.75	6245.47	4434	-1946.28
2003	6245.47	5739.10	2660	2153.63
2004	5739.10	7372.09	1483	3115.99
2005	7372.09	.	1937	.

TABLE 12. OBSERVED WEAKFISH JUVENILE INDICES FROM RHODE ISLAND TO THE SOUTH ATLANTIC

YCL	RI	CT	NY	NJ	DE	MD	VIMS	NC	SEA
1982	9.80	.	.	.	6.0	0.24	6.0	.	.
1983	7.00	.	.	.	11.5	0.22	10.9	.	.
1984	5.70	1.00	.	.	4.5	1.30	10.9	.	.
1985	2.20	6.19	.	.	6.7	0.13	6.1	.	.
1986	6.20	13.16	1.5	.	9.3	1.70	37.0	.	.
1987	1.80	0.63	0.3	.	12.8	0.44	4.6	.	.
1988	3.20	3.49	0.3	.	5.8	0.40	17.8	12.1	.
1989	10.10	8.69	0.1	25.7	4.7	0.20	21.8	102.0	.
1990	2.80	5.56	0.6	43.4	11.1	0.20	21.3	14.2	.
1991	16.50	11.95	0.3	14.7	8.7	0.80	30.0	50.2	2.1
1992	24.00	3.05	4.4	27.1	20.1	0.50	15.3	37.0	0.9
1993	5.60	4.08	1.2	6.0	14.7	2.30	15.9	43.7	0.2
1994	5.70	11.19	0.4	23.9	14.8	1.10	15.4	8.7	15.3
1995	15.60	5.22	1.7	37.1	11.5	1.50	7.0	68.1	43.8
1996	2.20	15.23	0.9	77.5	13.5	6.10	11.0	38.2	3.6
1997	38.60	12.38	4.7	46.3	12.1	5.10	7.4	72.4	20.3
1998	25.80	5.02	2.7	21.8	15.4	7.30	14.8	32.8	0.7
1999	4.20	30.93	0.5	3.0	11.4	16.00	9.9	70.4	4.5
2000	0.01	63.31	2.2	25.3	13.5	2.10	16.3	100.0	24.5
2001	0.01	40.09	3.9	0.1	14.1	6.50	11.1	63.0	9.0
2002	0.30	41.35	3.2	21.7	7.6	7.50	11.5	30.3	5.6
2003	0.20	49.41	2.6	39.3	6.0	3.90	8.6	22.0	44.3
2004	3.10	58.98	1.4	72.7	10.9	1.80	5.4	23.9	22.2

TABLE 13. JUVENILE INDICES SCALED IN MAGNITUDE TO THE NEW JERSEY INDICES WITH THE COASTWIDE MEAN JUVENILE INDEX (N0), THE AGE 1 INDEX (NIJ) AND THE JUVENILE MORTALITY (Z0) RATE

YCL	RI1	CT1	NY1	NJ	DE1	MD1	VIMS1	NC1	SEA1	N0	NIJ	Z0
1982	36.260	.	.	.	16.80	2.496	13.20	.	.	17.19	4.55556	1.32798
1983	25.900	.	.	.	32.20	2.288	23.98	.	.	21.09	4.90000	1.45956
1984	21.090	1.600	.	.	12.60	13.520	23.98	.	.	14.56	7.04444	0.72604
1985	8.140	9.904	.	.	18.76	1.352	13.42	.	.	10.32	7.87778	0.27004
1986	22.940	21.056	26.70	.	26.04	17.680	81.40	.	.	32.64	6.04444	1.68640
1987	6.660	1.008	5.34	.	35.84	4.576	10.12	.	.	10.59	2.92222	1.28757
1988	11.840	5.584	5.34	.	16.24	4.160	39.16	7.865	.	12.88	2.31111	1.71795
1989	37.370	13.904	1.78	25.7	13.16	2.080	47.96	66.300	.	23.14	2.96667	2.05412
1990	10.360	8.896	10.68	43.4	31.08	2.080	46.86	9.230	.	18.07	3.26667	1.71048
1991	61.050	19.120	5.34	14.7	24.36	8.320	66.00	32.630	4.41	26.21	3.92222	1.89948
1992	88.800	4.880	78.32	27.1	56.28	5.200	33.66	24.050	1.89	35.58	3.97778	2.19106
1993	20.720	6.528	21.36	6.0	41.16	23.920	34.98	28.405	0.42	20.39	5.31111	1.34524
1994	21.090	17.904	7.12	23.9	41.44	11.440	33.88	5.655	32.13	21.62	2.50000	2.15733
1995	57.720	8.352	30.26	37.1	32.20	15.600	15.40	44.265	91.98	36.99	3.18889	2.45098
1996	8.140	24.368	16.02	77.5	37.80	63.440	24.20	24.830	7.56	31.54	1.68889	2.92719
1997	142.820	19.808	83.66	46.3	33.88	53.040	16.28	47.060	42.63	53.94	1.11111	3.88251
1998	95.460	8.032	48.06	21.8	43.12	75.920	32.56	21.320	1.47	38.64	0.90000	3.75965
1999	15.540	49.488	8.90	3.0	31.92	166.400	21.78	45.760	9.45	39.14	1.18889	3.49413
2000	0.037	101.296	39.16	25.3	37.80	21.840	35.86	65.000	51.45	41.97	0.38889	4.68142
2001	0.037	64.144	69.42	0.1	39.48	67.600	24.42	40.950	18.90	36.12	0.71111	3.92777
2002	1.110	66.160	56.96	21.7	21.28	78.000	25.30	19.695	11.76	33.55	0.56667	4.08102
2003	0.740	79.056	46.28	39.3	16.80	40.560	18.92	14.300	93.03	38.78	0.56667	4.22589
2004	11.470	94.368	24.92	72.7	30.52	18.720	11.88	15.535	46.62	36.30	.	.

TABLE 14. TIME SERIES OF ATL. CROAKER, TAG-BASED STRIPED BASS, VPA-BASED STRIPED BASS SUMMER FLOUNDER, SPINY DOGFISH AND BLUEFISH ABUNDANCE. AVERAGE ANNUAL WATER TEMPERATURE (LAGGED T-2) AND DEVIATIONS IN THE NORTH ATL OSCILLATION INDEX (LAGGED T-2), 1982-2004.

YEAR	croak	STRIP	strip2	FLUKE2	SHK2	BLUE2	TEMP	NAO
1982	361	1395	218	17.5	695	229.1	11.1	31
1983	4143	1619	253	17.5	250	212.5	11.0	112
1984	2597	1734	271	18.8	265	182.2	11.1	44
1985	3698	1498	234	16.1	1057	162.9	12.0	187
1986	4903	1734	271	15.0	278	170.3	12.0	87
1987	4828	1210	357	13.9	550	154.9	12.0	-34
1988	4854	2849	348	14.4	708	119.6	11.9	27
1989	3400	3827	501	8.1	464	103.7	11.8	-41
1990	6434	2075	997	5.2	939	91.8	11.1	39
1991	15348	1844	1278	7.5	547	83.7	11.3	277
1992	9401	1994	1840	6.0	736	71.2	12.1	216
1993	13542	2486	2022	7.3	566	72.0	12.6	56
1994	17127	4027	2373	9.2	515	67.0	11.5	179
1995	11599	3486	2832	12.0	461	67.7	11.7	146
1996	10973	5201	3129	15.6	769	65.6	11.6	165
1997	17522	4893	3606	15.9	516	64.7	12.5	216
1998	16569	3877	4120	15.7	355	70.7	10.6	-206
1999	15952	4256	4072	17.8	480	73.0	10.9	-11
2000	22407	8280	4592	16.5	355	80.3	12.1	39
2001	21186	9907	5960	19.4	305	87.7	12.9	93
2002	20772	13066	6621	25.5	472	88.2	12.2	153
2003	18773	13672	6056	29.4	469	92.2	12.5	49
2004	17600	17099	6712	36.7	225	104.1	12.7	112
2005

Table 15. Three stepwise regression models with weakfish surplus production (SURPt), biomass weighted (ages 1+) natural mortality (MWt) and juvenile mortality (Z0) used as dependent variables. The independent variables included abundance of striped bass, bluefish, croaker, fluke and spiny dogfish as well as mean annual water temperature, deviations in the North Atlantic Oscillation Index and fishing mortality (FWt). Given below are the independent variables that were selected by the stepwise model. The slope (b) of the regression its standard error (SEb), Pearson correlation coefficient (r) and the probability (P) level for model fit are included.

Significant Variable	<i>Dependent Variables</i>											
	SURPt				MWt				Z0			
	b	SEb	r	P	b	SEb	r	P	b	SEb	r	P
Strip	-0.83	0.21	-0.66	<0.001								
Fluke					0.03	0.006	0.79	<0.0001				
Strip2									0.0005	0.00005	0.91	<0.0001

Table 16. Pearson correlation (r) matrix relating the three weakfish dependent variables (SURPt, MWt, Z0) with each of the nine potential explanatory variables. The P value indicates the level of statistical significance.

<i>Explanatory Variable</i>	<i>Dependent Variable</i>					
	SURPt		MWt		Z0	
	r	P	r	P	r	P
Strip	-0.62	0.001*	0.55	0.009*	0.84	0.0001*
Strip2	-0.66	0.001*	0.44	0.04*	0.91	0.0001*
Fluke	-0.37	0.07	0.79	0.0001*	0.46	0.03*
Croaker	-0.60	0.003*	0.30	0.17	0.85	0.0001*
Bluefish	0.33	0.13	0.24	0.29	-0.61	0.002**
Spiny Dog	0.23	0.29	-0.36	0.10	-0.36	0.10
Temperature	-0.26	0.23	0.12	0.61	0.27	0.23
NAO	0.13	0.56	-0.002	0.99	-0.10	0.65
FWt	0.22	0.31	-0.32	0.14	-0.20	0.33

- * indicates a statistically significant ($P < 0.05$) relationship with dependent variable.
- ** indicates a statistically significant ($P < 0.05$) relationship with dependent variable but in the wrong direction.

Table 17. Parameter estimates (r, K) and weakfish overfishing thresholds (Fmsy, Fcoll, Bmsy) derived from the basic (see equation 7) Gompertz production model (no predation effects) fitted by nonlinear least squares and by iterative reweighting least squares, 1982-2004. The standard error (SE) is given for each parameter estimate, as well as the coefficient of determination (r2).**

Parameters	<i>Least Squares</i>		<i>Iterative Reweighting</i>	
	Mean	SE	Mean	SE
r	0.32	0.11	0.36	0.07
K	68,704 mt	25,618 mt	62,072 mt	11,989 mt
r**2	0.68		0.82	
Overfishing Thresholds				
Fmsy	0.32	0.11	0.36	0.07
Fcoll	0.87	0.30	0.98	0.19
Bmsy	25,259 mt	9,412 mt	22,821 mt	4,417 mt

TABLE 18. OBSERVED AND PREDICTED WEAKFISH BIOMASS AND RESIDUALS FROM GOMPERTZ PRODUCTION MODEL WITH NO ITERATIVE REWEIGHTING

YEAR	BIOL	pred22	RESID22
1982	28643	27664.87	978.13
1983	20822	21879.39	-1057.39
1984	24518	14896.08	9621.92
1985	27122	20953.48	6168.52
1986	23123	19198.01	3924.99
1987	15234	18990.86	-3756.86
1988	7548	8887.38	-1339.38
1989	6724	4223.48	2500.52
1990	8463	5785.83	2677.17
1991	7858	7979.14	-121.14
1992	9664	7986.35	1677.65
1993	16634	10713.85	5920.15
1994	23398	18196.42	5201.58
1995	25494	24993.03	500.97
1996	27716	26258.62	1457.38
1997	28858	28177.34	680.66
1998	27397	28709.05	-1312.05
1999	23057	28847.07	-5790.07
2000	16847	24486.57	-7639.57
2001	12626	18465.05	-5839.05
2002	6245	14438.58	-8193.58
2003	5739	7894.16	-2155.16
2004	7372	8350.52	-978.52
2005	7372	10182.53	-2810.53

TABLE 19. OBSERVED AND PREDICTED WEAKFISH BIOMASS,RESIDUALS (RBI) AND WEIGHTING COEFFICENTS FROM GOMPertz PRODUCTION MODEL WITH ITERATIVE REWEIGHTING

YEAR	BIOL	pred11	RBI	_WEIGHT_
1982	28643	27779.62	863.38	0.98168
1983	20822	22102.22	-1280.22	0.95994
1984	24518	15228.40	9289.60	0.00000
1985	27122	21242.02	5879.98	0.32851
1986	23123	19446.97	3676.03	0.69417
1987	15234	19297.73	-4063.73	0.63381
1988	7548	9252.63	-1704.63	0.92954
1989	6724	4540.42	2183.58	0.88574
1990	8463	6088.12	2374.88	0.86559
1991	7858	8309.66	-451.66	0.99497
1992	9664	8308.20	1355.80	0.95513
1993	16634	11058.38	5575.62	0.37971
1994	23398	18557.72	4840.28	0.50518
1995	25494	25296.45	197.55	0.99904
1996	27716	26533.12	1182.88	0.96575
1997	28858	28416.35	441.65	0.99519
1998	27397	28928.01	-1531.01	0.94296
1999	23057	29091.47	-6034.47	0.30298
2000	16847	24794.26	-7947.26	0.04851
2001	12626	18825.48	-6199.48	0.27616
2002	6245	14802.12	-8557.12	0.00922
2003	5739	8186.74	-2447.74	0.85753
2004	7372	8631.84	-1259.84	0.96119
2005	7372	10496.55	-3124.55	0.77347

Table 20. Parameter estimates (r, K, c, A) and weakfish overfishing thresholds (Fmsy, Fcoll, Bmsy) derived from the Steele-Henderson Production Model (see equation 6) with striped bass predation effects fitted by nonlinear least squares and by iterative reweighting least squares, 1982-2004. The standard error (SE) is given for each parameter estimate, as well as the coefficient of determination (r2).**

Parameters	<i>Least Squares</i>		<i>Iterative Reweighting</i>	
	Mean	SE	Mean	SE
r	0.59	0.16	0.55	0.09
K	57,607 mt	13,620 mt	63,649 mt	11,683 mt
c	1.52	0.53	1.71	0.41
A	10,364 mt	3,966 mt	11,541 mt	3,406 mt
r**2		0.88		0.94
<i>Overfishing Thresholds</i>				
Fmsy	0.59	0.16	0.55	0.09
Fcoll	1.60	0.43	1.50	0.25
Bmsy	21,179 mt	5,001 mt	23,400 mt	4,305 mt

TABLE 21.OBSERVED AND PREDICTED WEAKFISH BIOMASS,RESIDUALS (RBI) AND REWEIGHTING COEFFICIENTS FROM STEELE-HENDERSON MODEL WITH STRIPED BASS AND ITERATIVE REWEIGHTING

YEAR	BIOL	pred11	rbi	_WEIGHT_
1982	28643	29747.52	-1104.52	0.95083
1983	20822	24388.75	-3566.75	0.54815
1984	24518	17986.22	6531.78	0.01672
1985	27122	24078.38	3043.62	0.65763
1986	23123	21720.70	1402.30	0.92135
1987	15234	22626.86	-7392.86	0.00000
1988	7548	11005.06	-3457.06	0.57168
1989	6724	6284.94	439.06	0.99215
1990	8463	8680.47	-217.47	0.99807
1991	7858	11122.99	-3264.99	0.61222
1992	9664	11007.51	-1343.51	0.92768
1993	16634	13462.74	3171.26	0.63164
1994	23398	18843.16	4554.84	0.33247
1995	25494	25479.17	14.83	0.99999
1996	27716	24037.78	3678.22	0.52402
1997	28858	26027.77	2830.23	0.69978
1998	27397	27863.24	-466.24	0.99115
1999	23057	27675.86	-4618.86	0.31879
2000	16847	18461.01	-1614.01	0.89650
2001	12626	12240.82	385.18	0.99395
2002	6245	7231.76	-986.76	0.96065
2003	5739	6248.72	-509.72	0.98942
2004	7372	6039.48	1332.52	0.92884
2005	7372	.	.	.

**TABLE 22. WEAKFISH FISHING MORTALITY (FWt), PREDATOR NATURAL MORTALITY (MP)
FROM EXPONENTS OF THE S-H MODEL AND NATURAL MORTALITY (MWT), 1982-2004**

YEAR	FWT	MP	MWT
1982	0.410	0.06828	0.57
1983	0.500	0.09630	0.67
1984	0.640	0.10006	0.82
1985	0.460	0.08121	0.49
1986	0.570	0.09993	0.44
1987	0.510	0.08637	0.23
1988	0.860	0.27173	0.64
1989	1.080	0.27474	0.01
1990	0.810	0.11842	0.06
1991	0.660	0.13512	0.34
1992	0.610	0.12328	0.39
1993	0.460	0.13325	0.33
1994	0.320	0.23224	0.25
1995	0.260	0.19613	0.22
1996	0.270	0.27743	0.31
1997	0.260	0.25208	0.37
1998	0.270	0.20320	0.47
1999	0.220	0.24501	0.56
2000	0.270	0.56747	0.81
2001	0.320	0.78241	0.61
2002	0.350	1.29008	1.09
2003	0.430	0.88369	0.76
2004	0.260	0.88427	.
2005	0.263	.	.

Table 23. Model conditions used for projections and to test for sensitivity. Starting model = Base 3 Parameter is the base model run where r , K , and B_{1981} were estimated simultaneously; Schaefer, Gompertz and Predator-prey used previous surplus production models as a starting point and held K constant (fixed). Recent F and Z are the two year averages for 2001-2002, with the exception of the 3 Parameter model with F fixed at values estimated in Crecco (2005); this run used the average for 2001-2003.

Starting model	Base 3 Parameter	Crecco (2005)	Schaefer ¹	Gompertz ¹	Predator- prey ²
F	Estimated	Fixed ¹	Estimated	Estimated	Estimated
K	Estimated	Estimated	Fixed	Fixed	Fixed
Estimated K (mt)	86,409	86,409	57,700 ¹	64,028 ¹	49,500 ²
Estimated r ?	0.903	0.903	0.896	0.897	0.893
Recent F	0.41	0.35	0.41	0.41	0.41
Recent Z	1.36	1.39	1.43	1.38	1.42
Recent F/Z	0.30	0.25	0.28	0.29	0.29
Z - (F*0.5)	1.15	1.21	1.23	1.18	1.22
Z - (F*1.0)	0.95	1.04	1.02	0.98	1.02
Z - (F*1.5)	0.75	0.86	0.82	0.77	0.81
Z - (F*2.0)	0.54	0.69	0.62	0.57	0.61

1 = Crecco (2005).

2= Uphoff (2005).

FIGURES

**Figure 1.
Total Weakfish Landings 1981-2004**

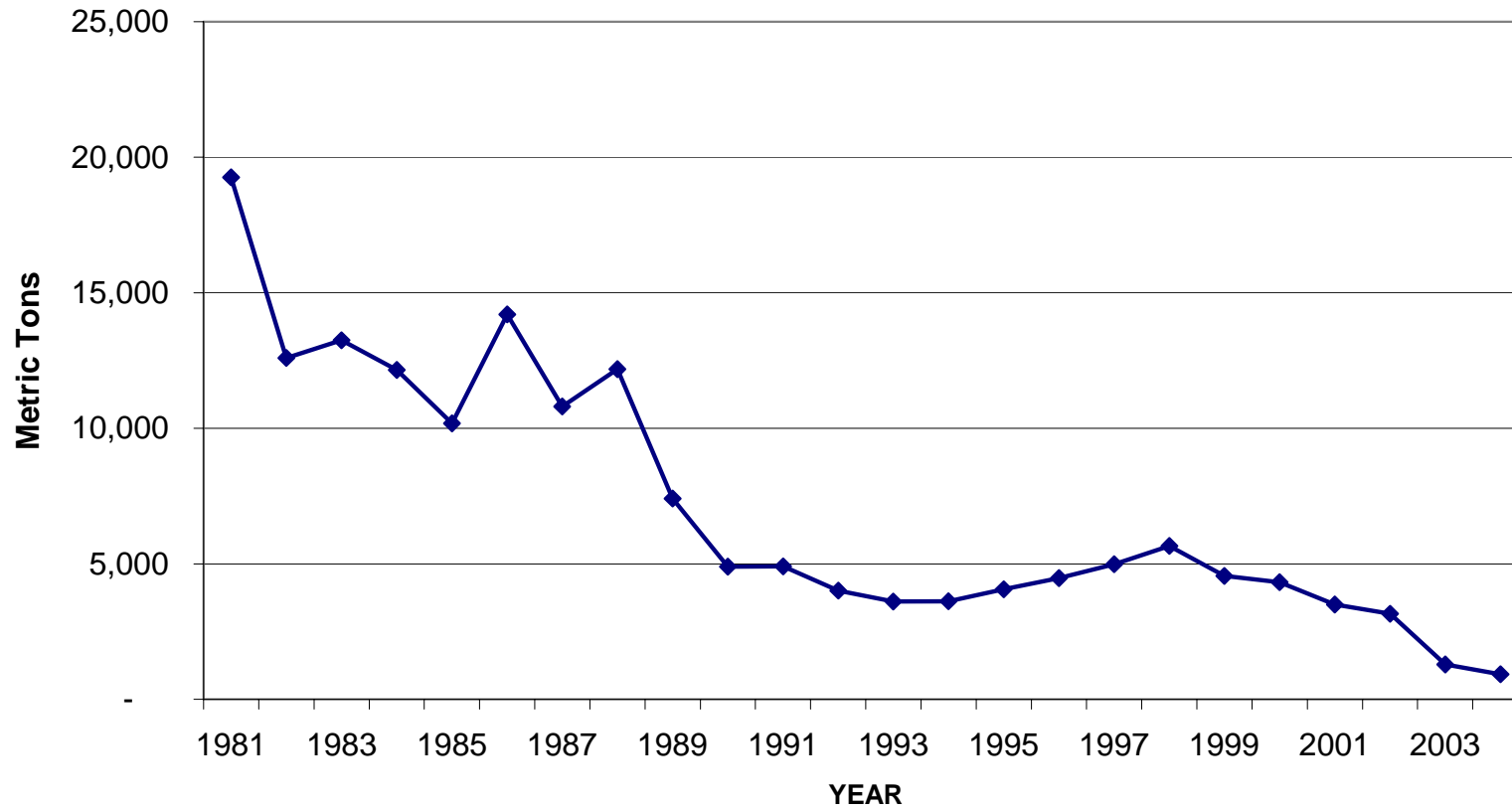


Figure 1A. Atlantic coast 1950-2003 commercial and 1981-2003 recreational landings of weakfish.

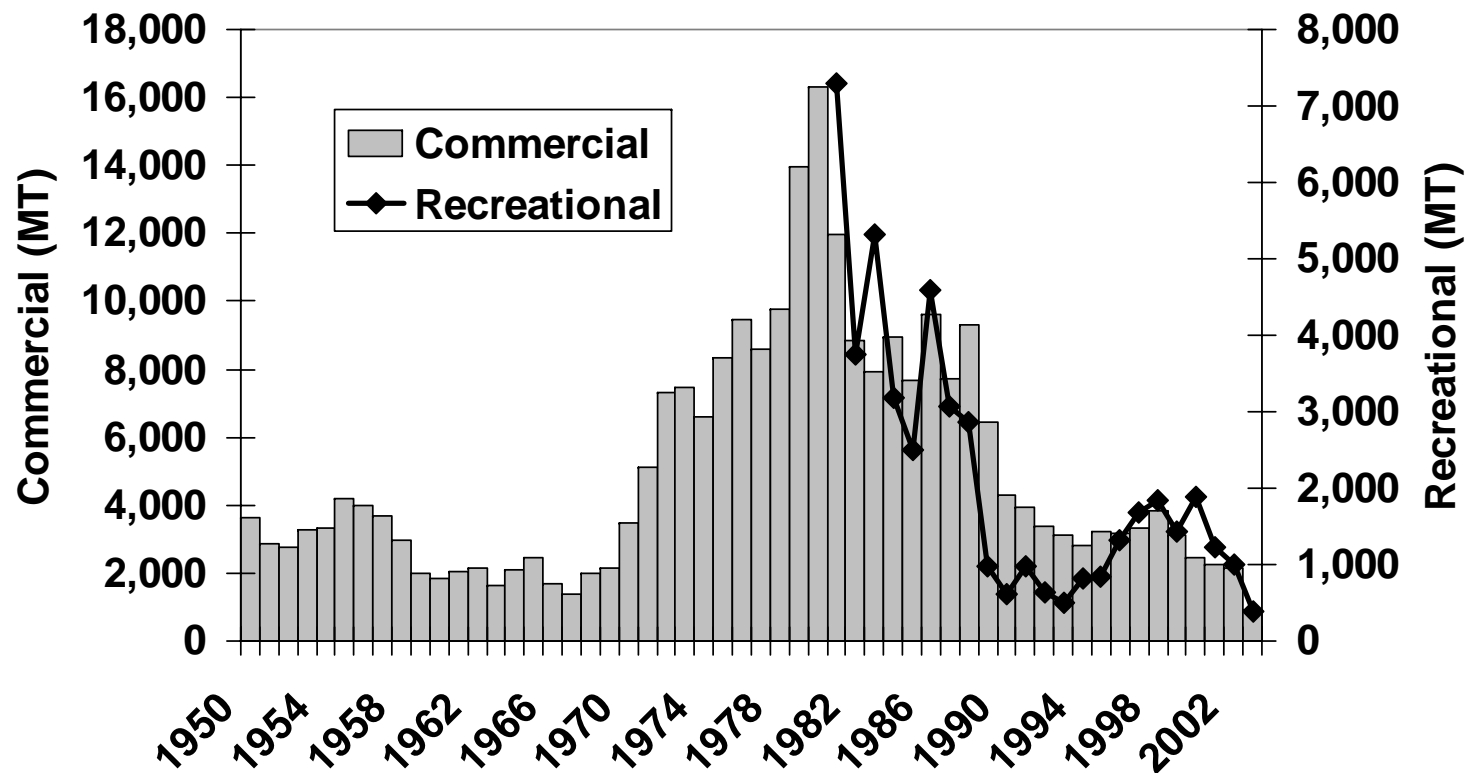


Figure 2. WEAKFISH LANDINGS BY SECTOR, 1981-2004

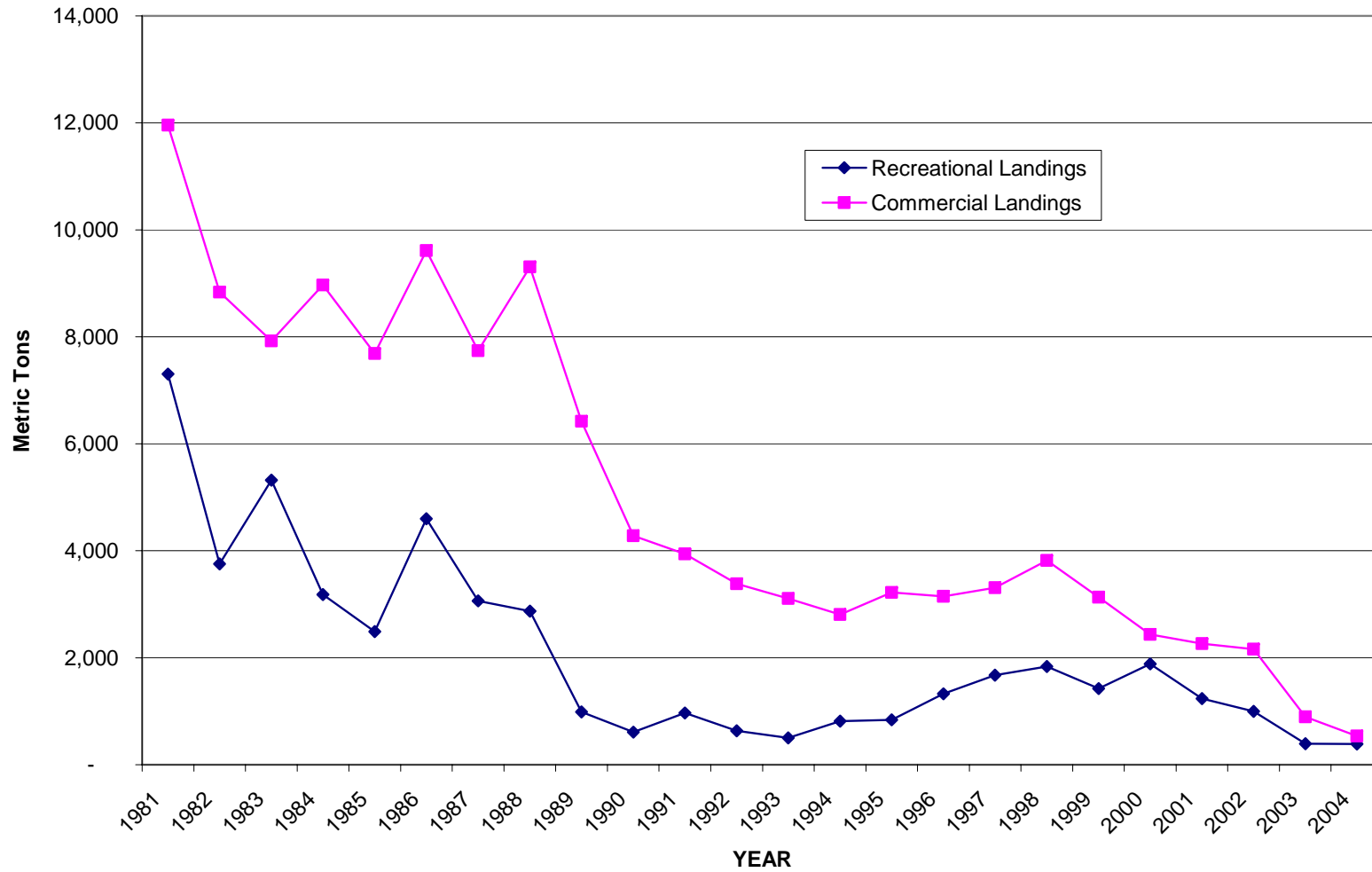


Figure 2a. Coastwide Weakfish Recreational Catch and Harvest Through 2004

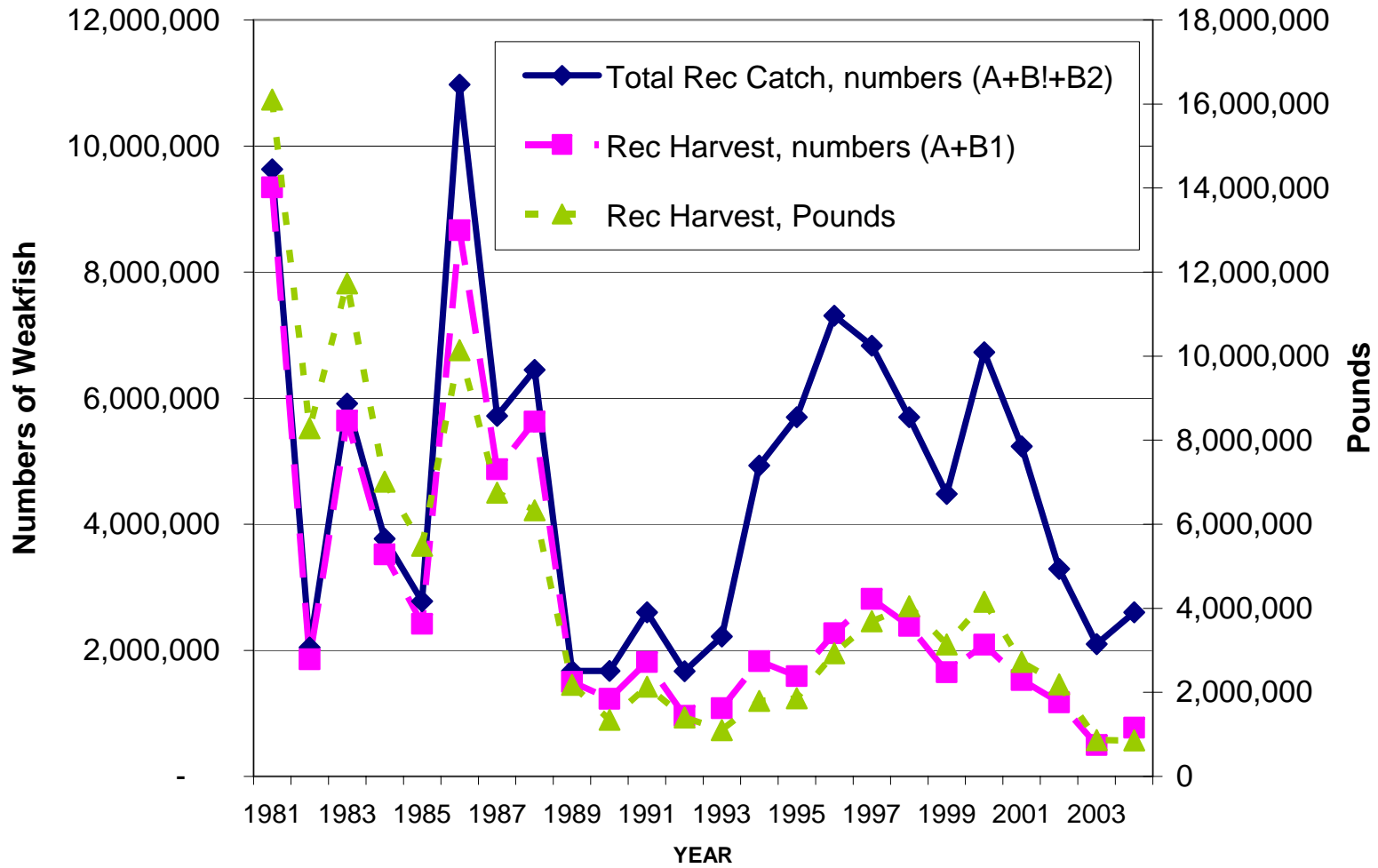


Figure 3. Commercial proportion of Coastwide Landings

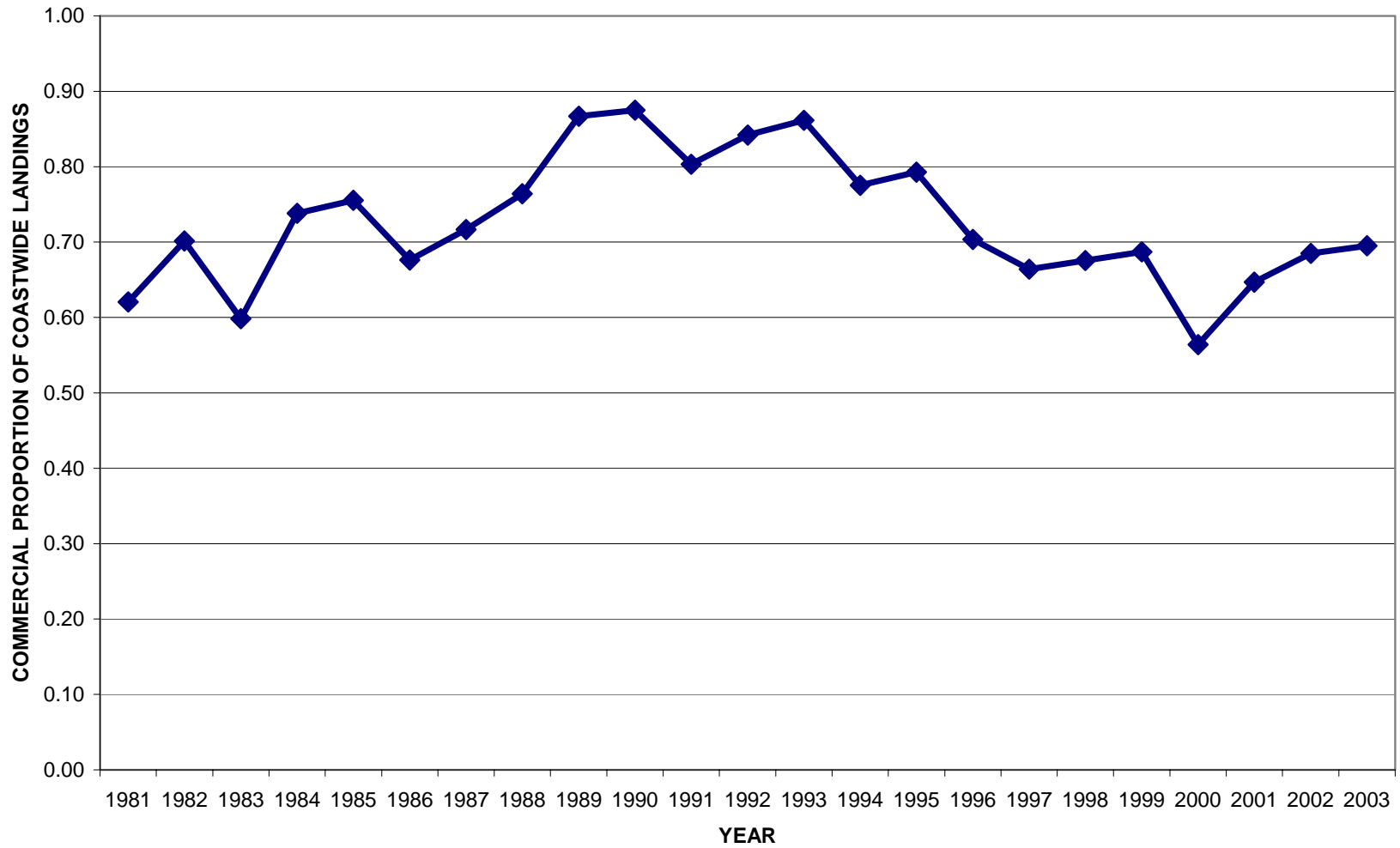


Figure 3a. NC Commercial Weakfish Landings for Estuarine Gears

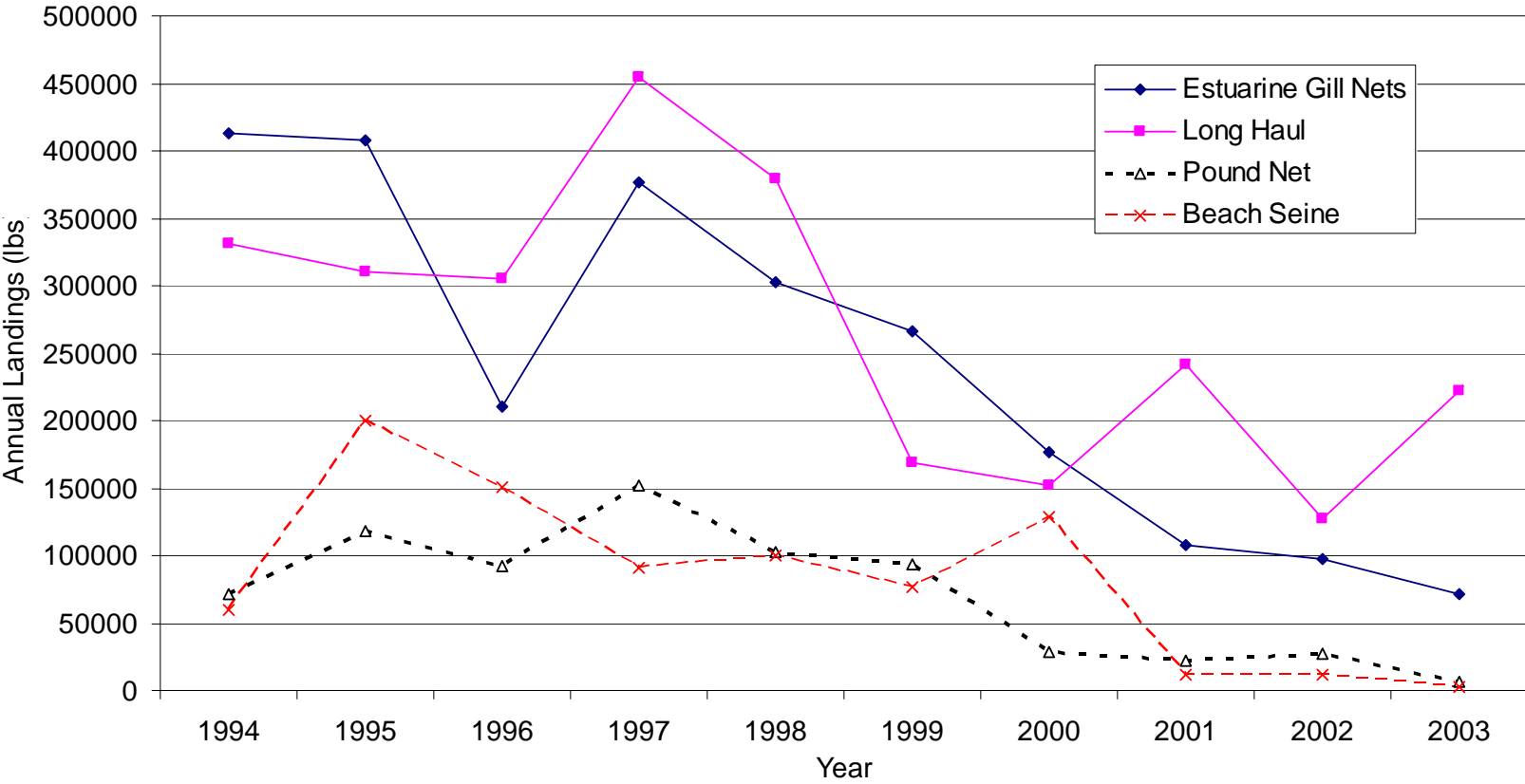


Figure 3b. NC Estuarine Commercial Gears CPUE Index

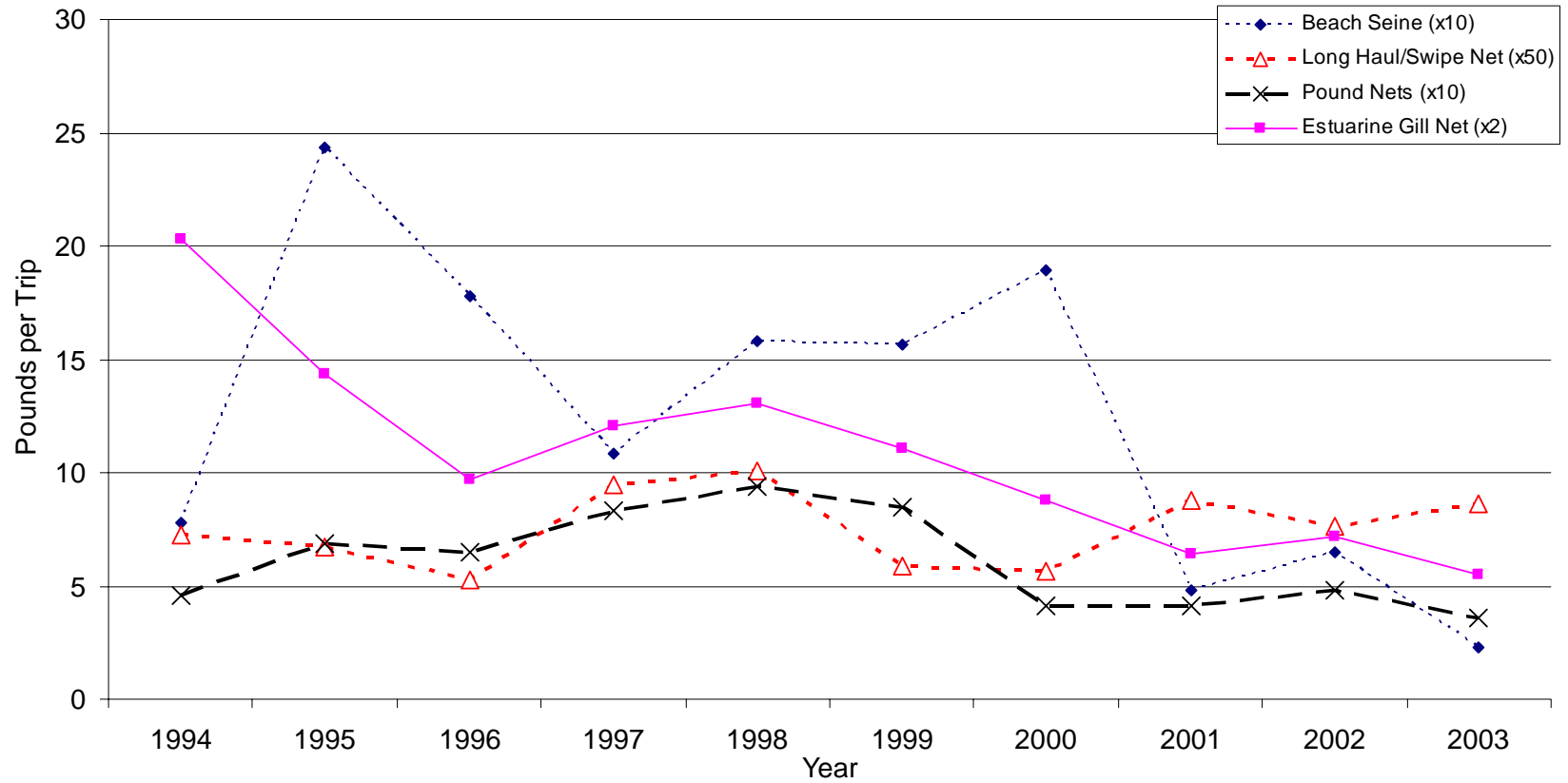
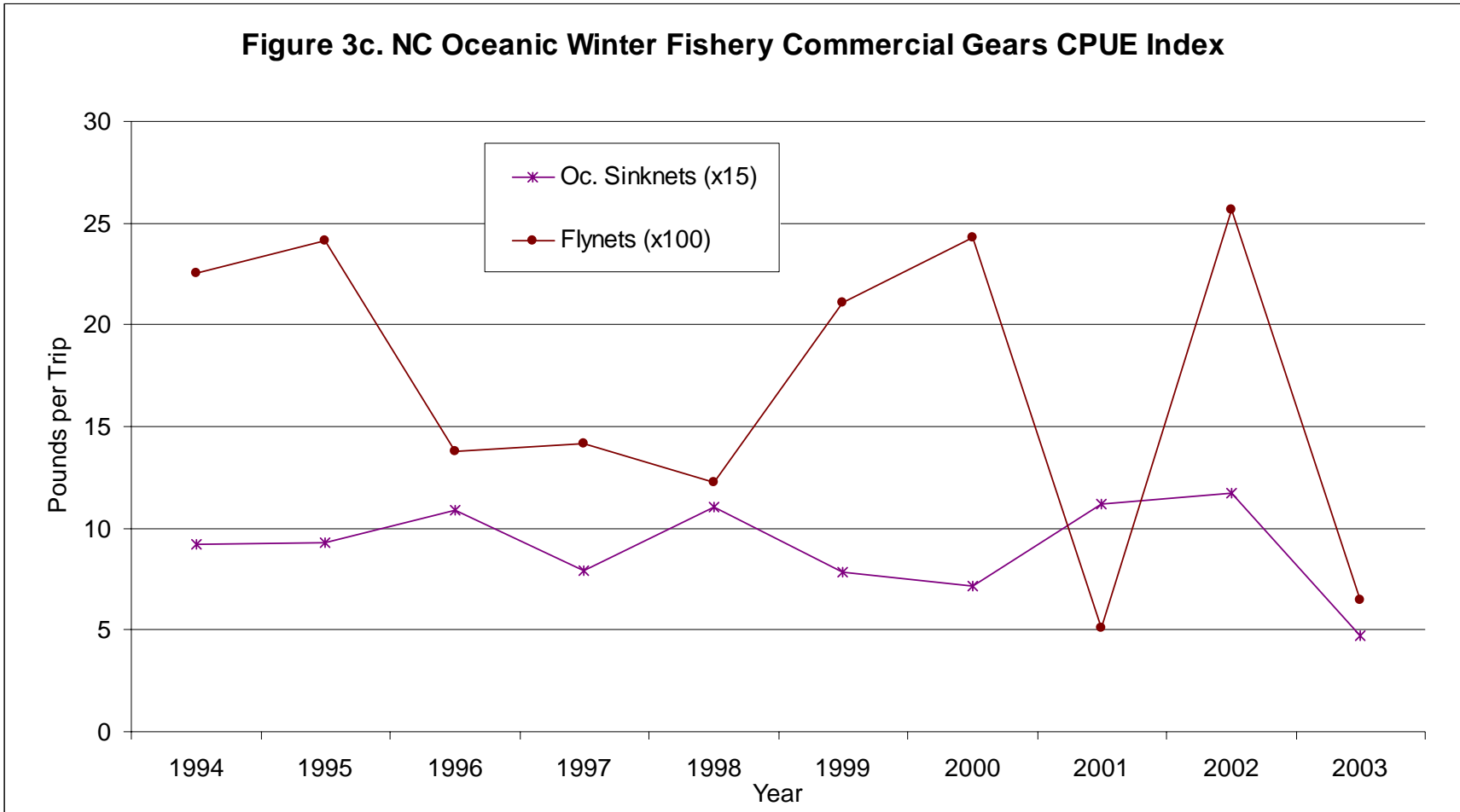


Figure 3c. NC Oceanic Winter Fishery Commercial Gears CPUE Index



**Figure 3d. Recreational fishery Catch-per-trip
Indices of Relative Abundance**

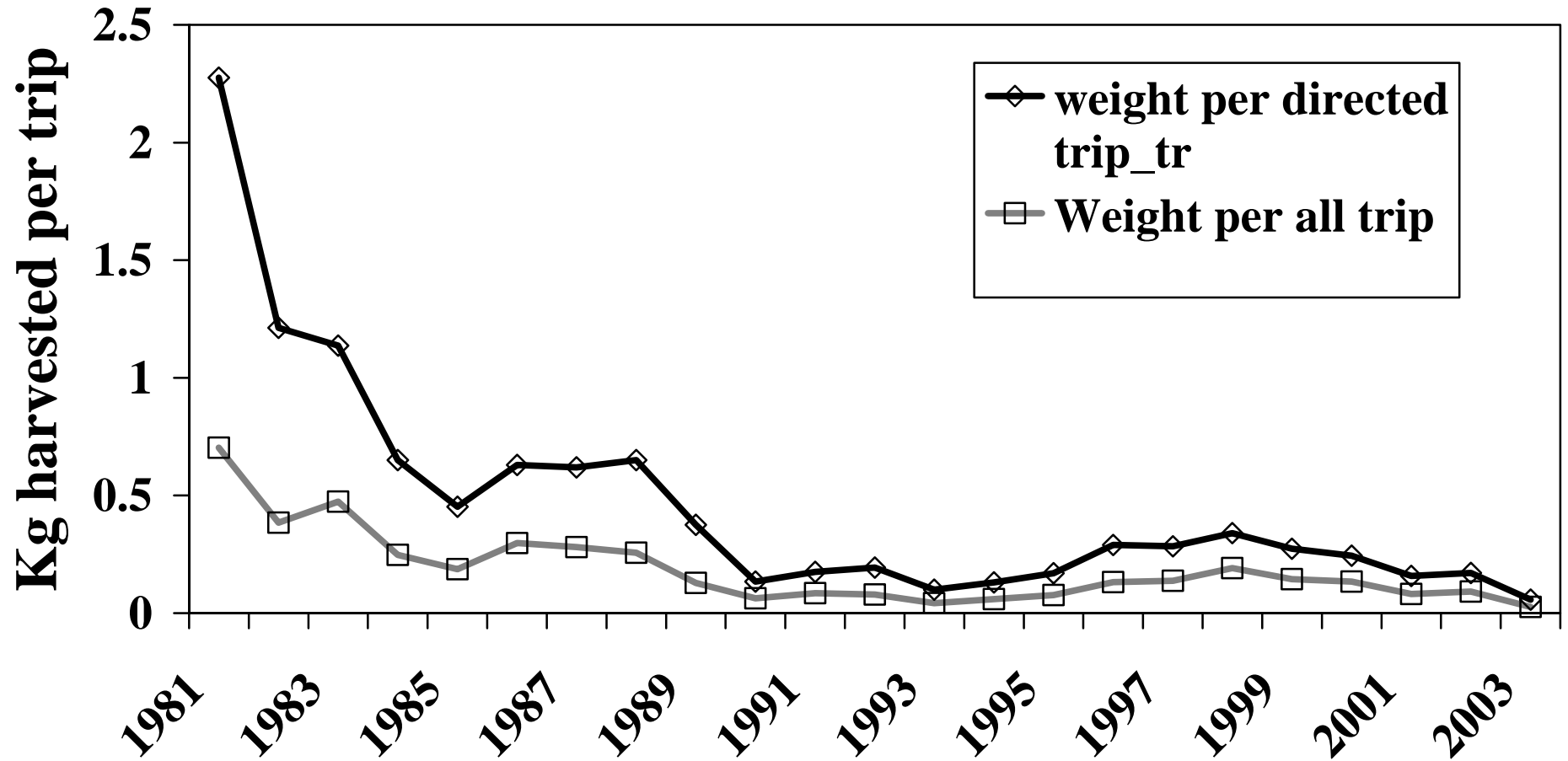


Figure 3e. Recreational harvest per trip age-structured tuning indices and total catch per trip aggregated for ages 2-6

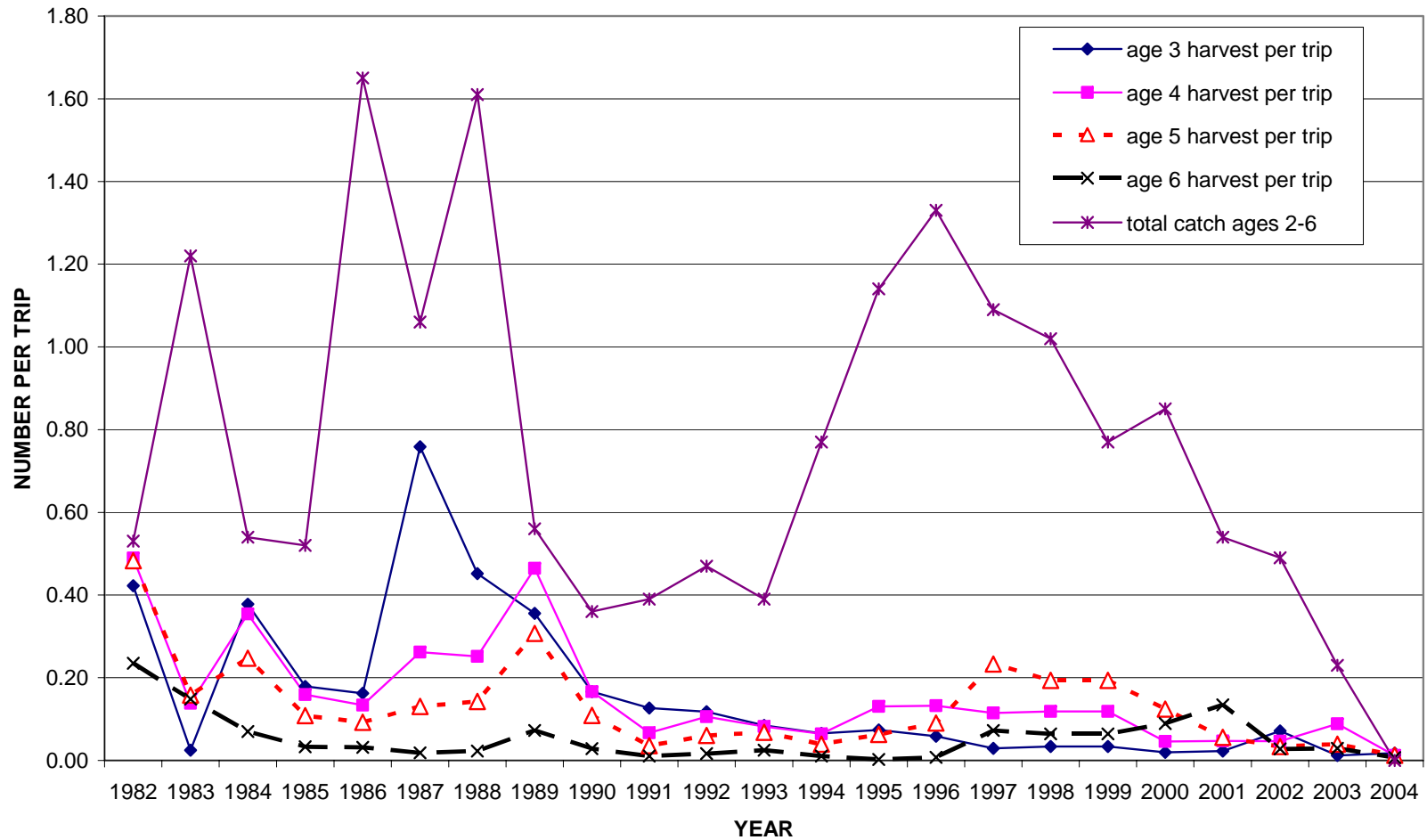


Figure 3f. Relative Abundance from Mid-Atlantic Private Boat Total Catch and Total Trips



FIGURE 4. NEFSC FALL INSHORE SURVEY TOTAL CATCH PER TOW

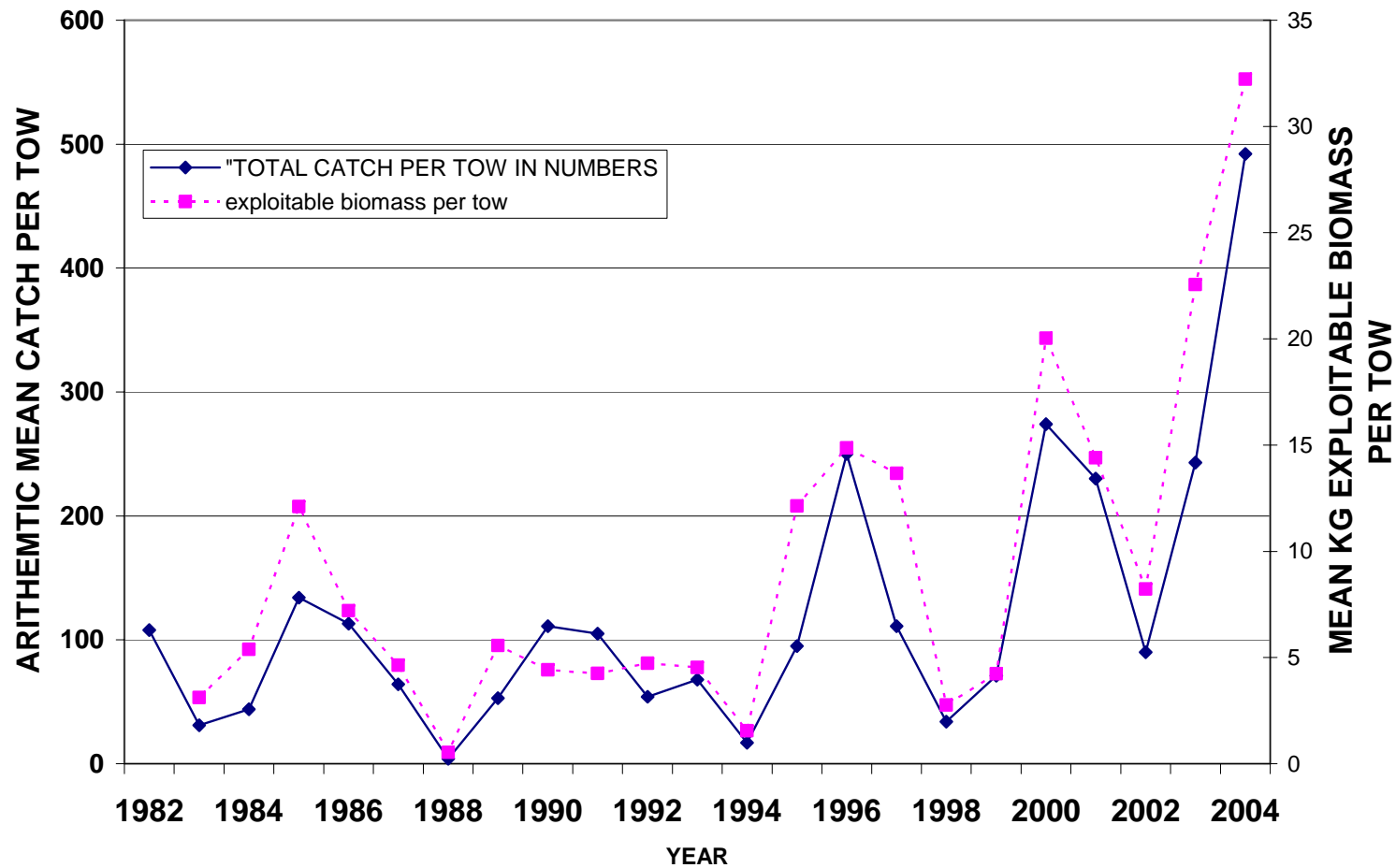


FIGURE 5. NEFSC SURVEY INDICES BY AGE

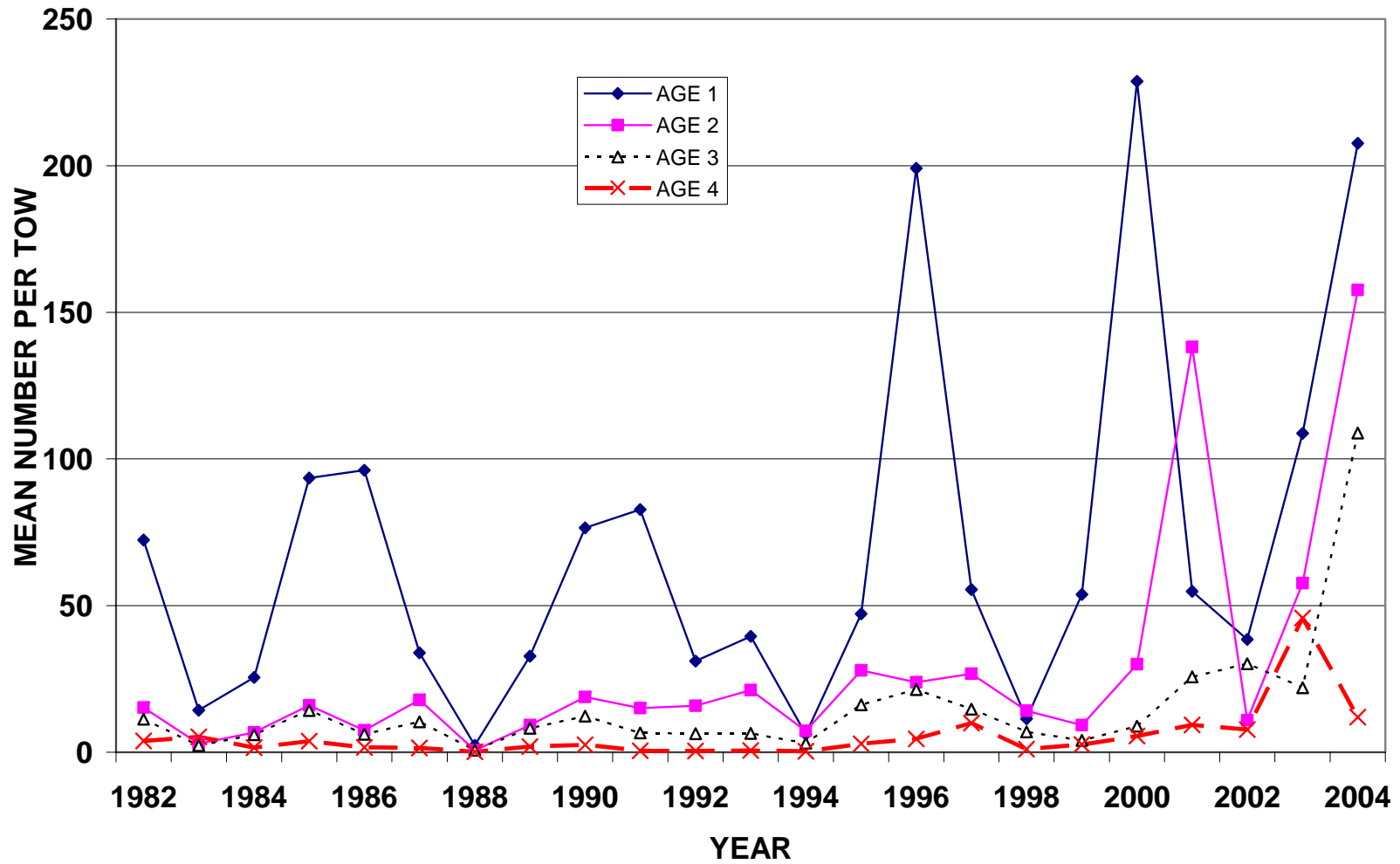


Figure 6. NJ Survey Total Catch Per Tow

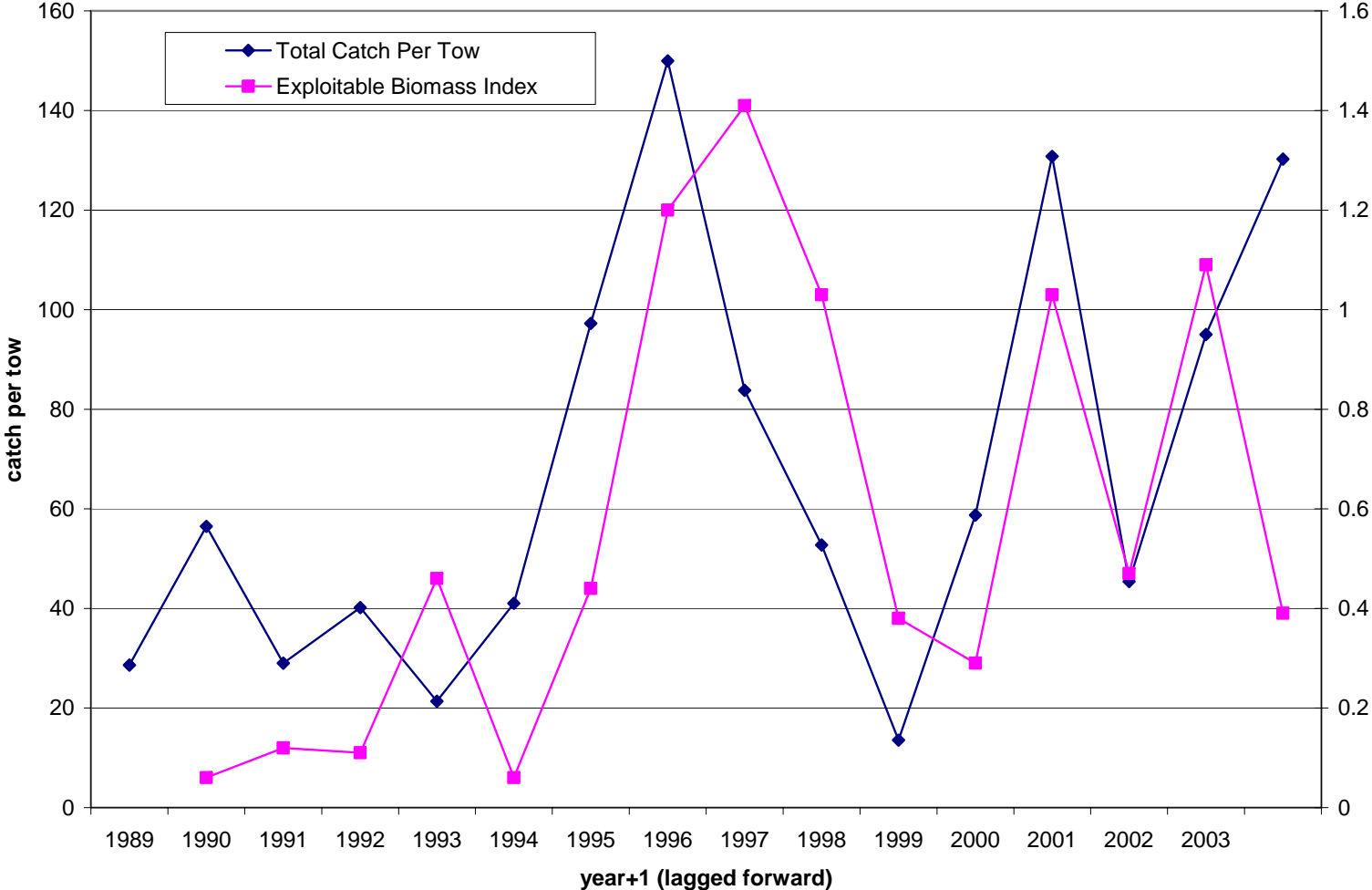


FIGURE 7. NJ INDICES BY AGE

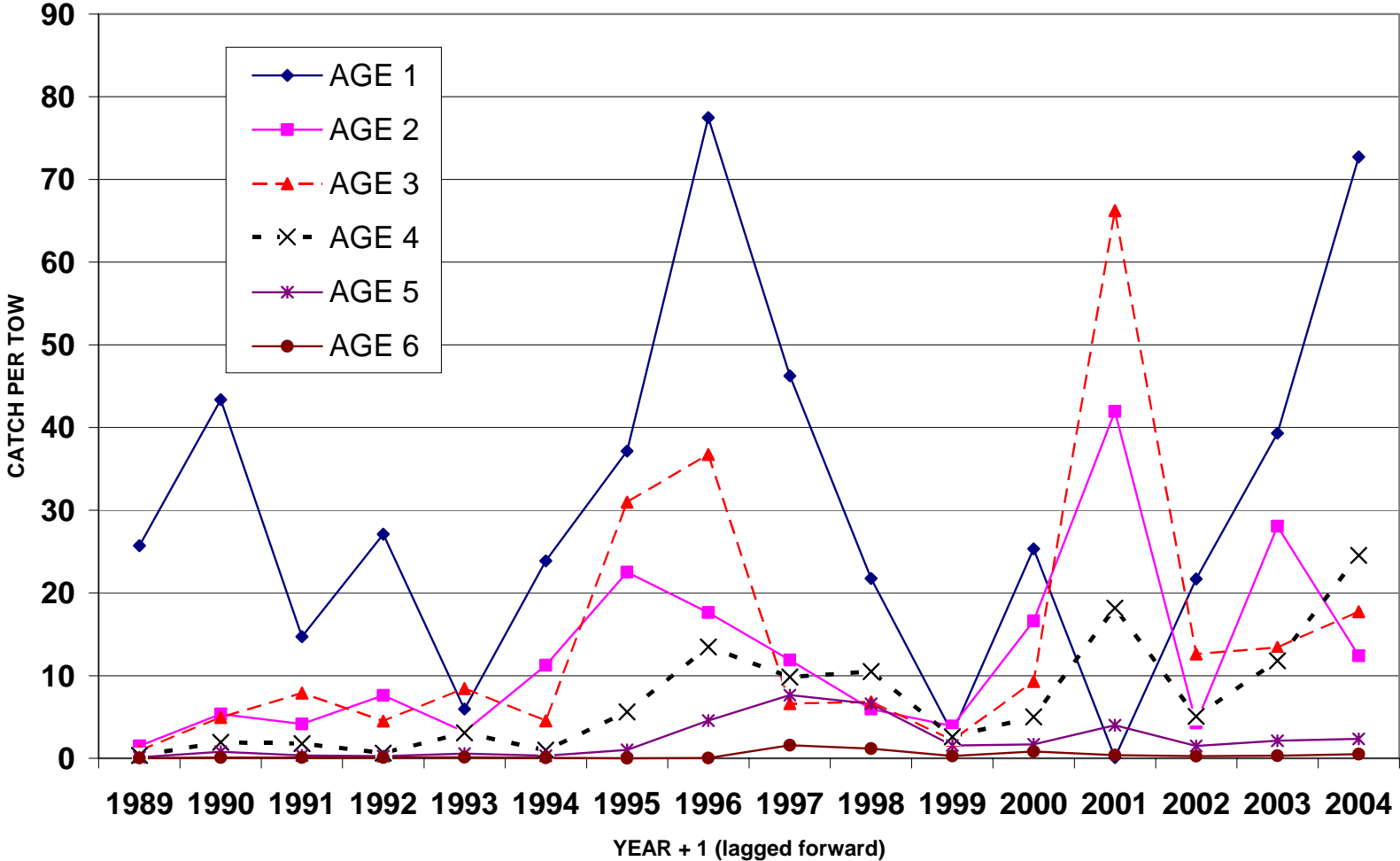


FIGURE 8. DELAWARE SURVEY TOTAL CATCH PER TOW

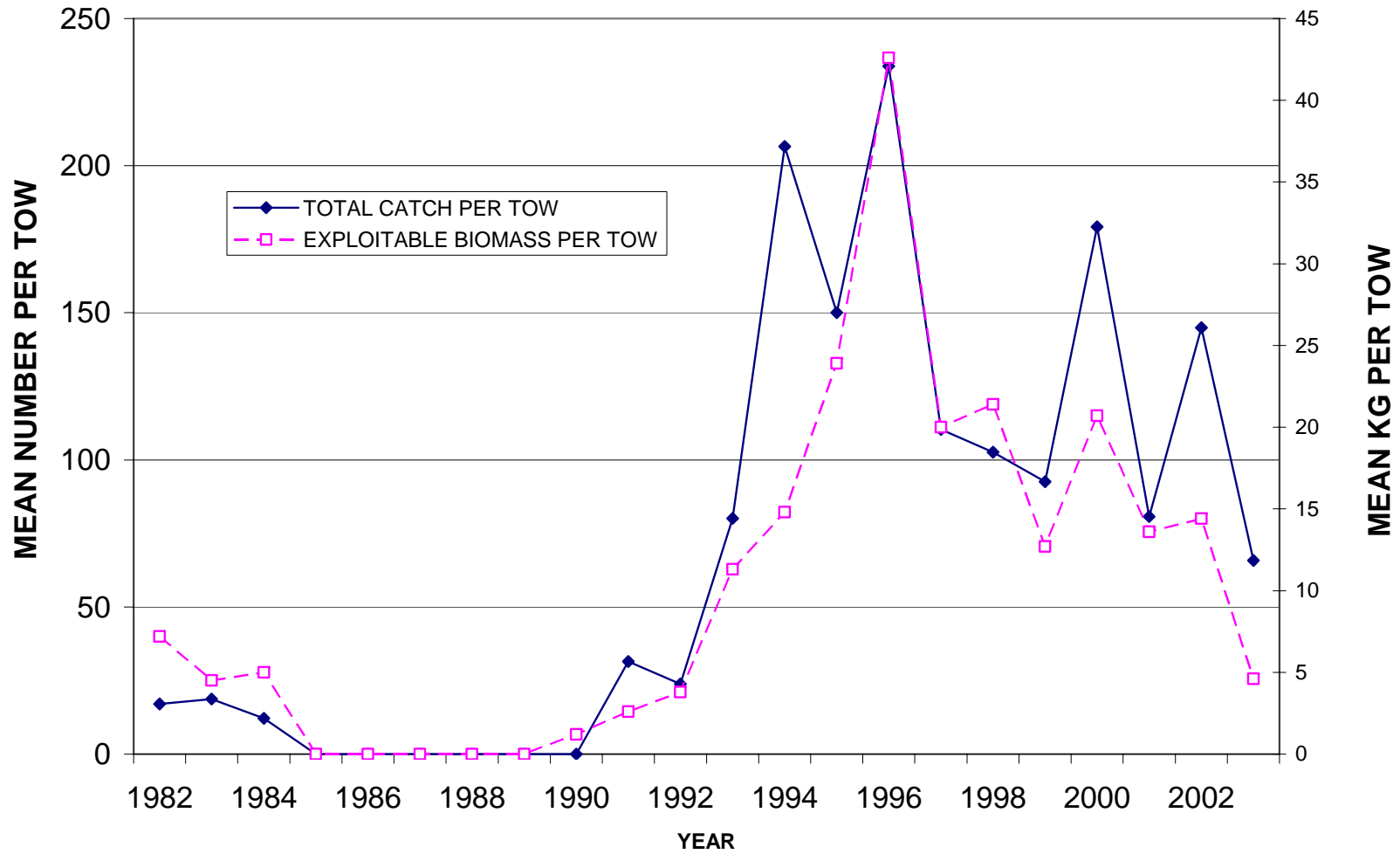


FIGURE 9. DE SURVEY INDICES BY AGE

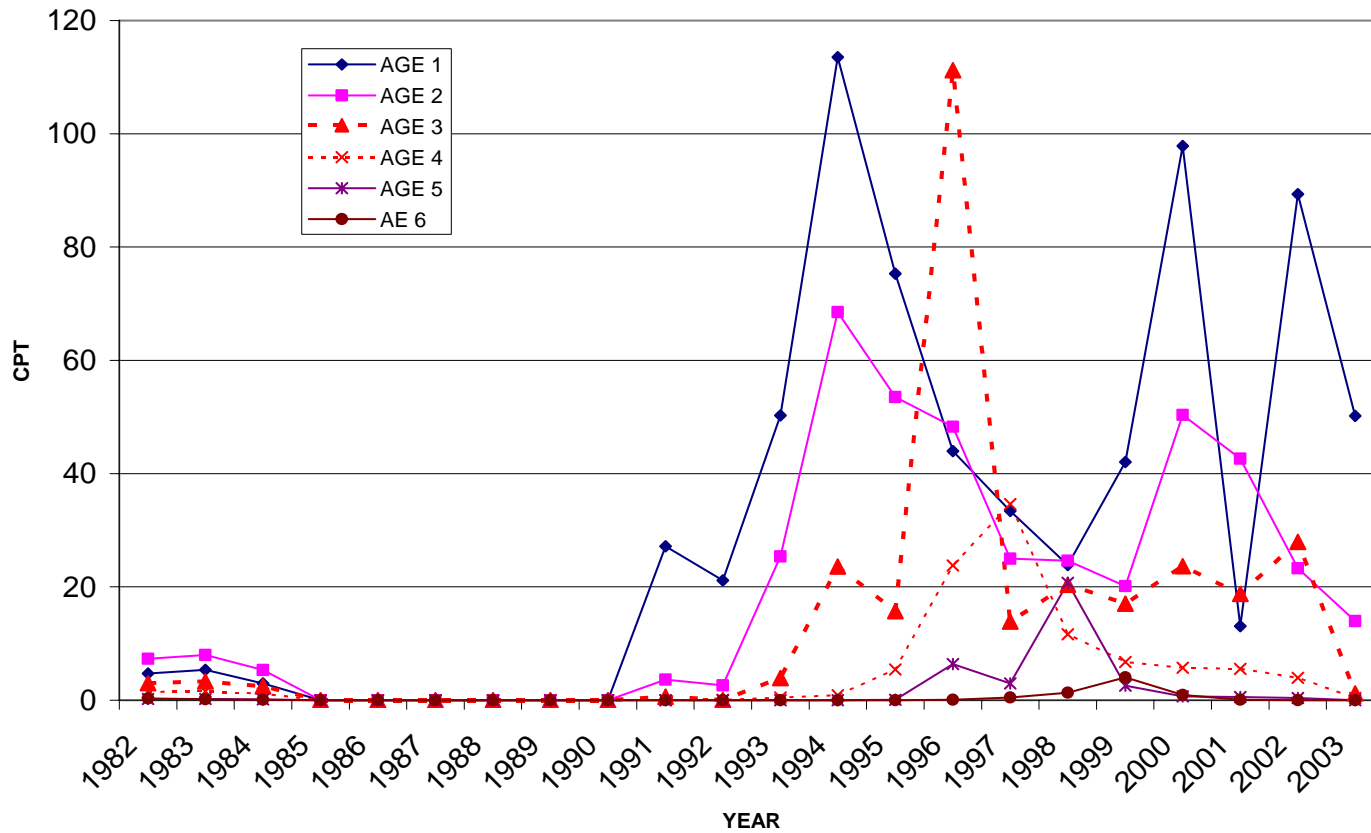


Figure 9a. SEAMAP INDICES BY AGE

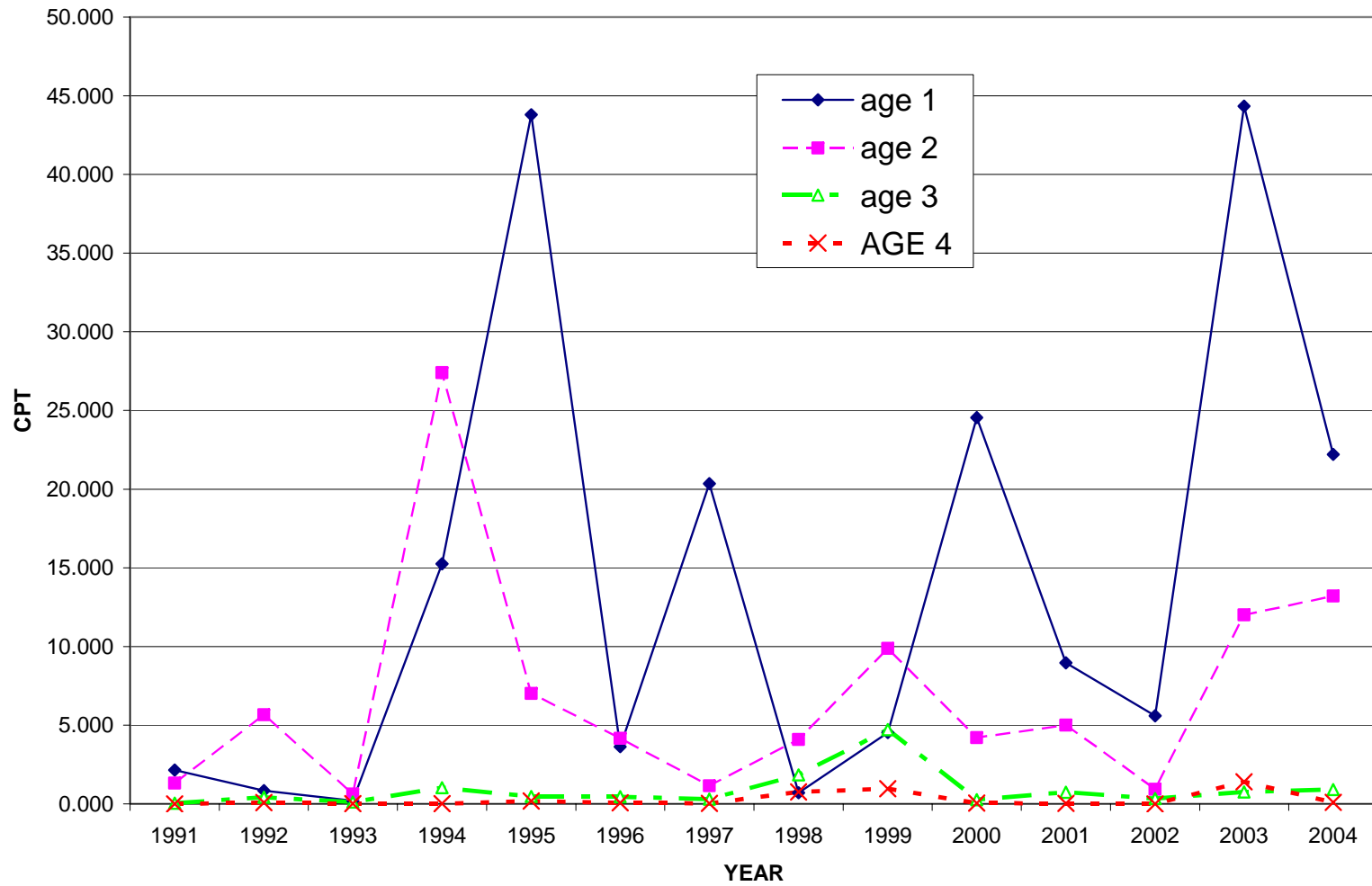


Figure 9b. Relative F based on Recreational Total Catch in Weight and in Number

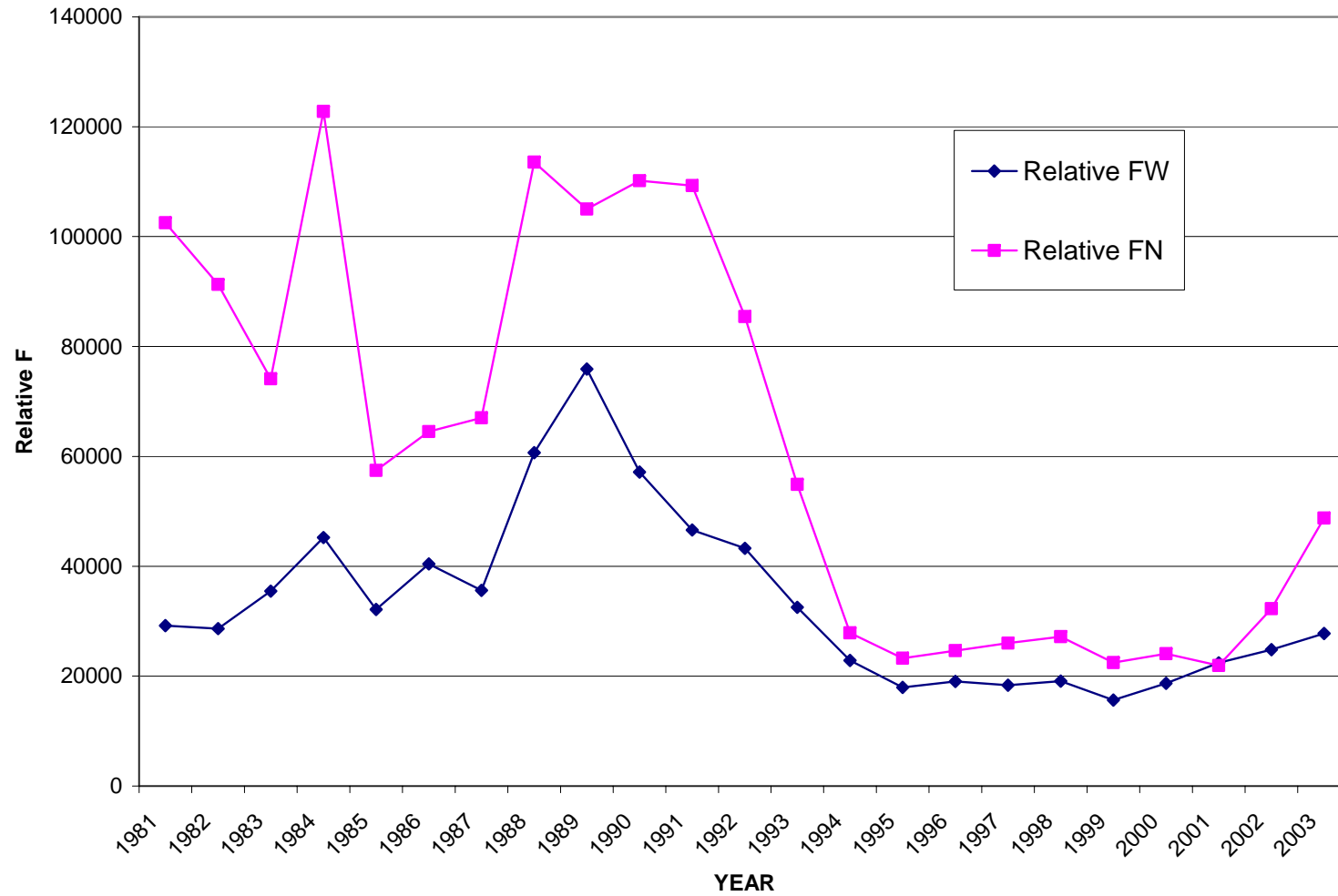


Figure 9c. Relative F derived from Z + 1 transformed trawl survey exploitable biomass indices, divided by 10,000 to rescale.

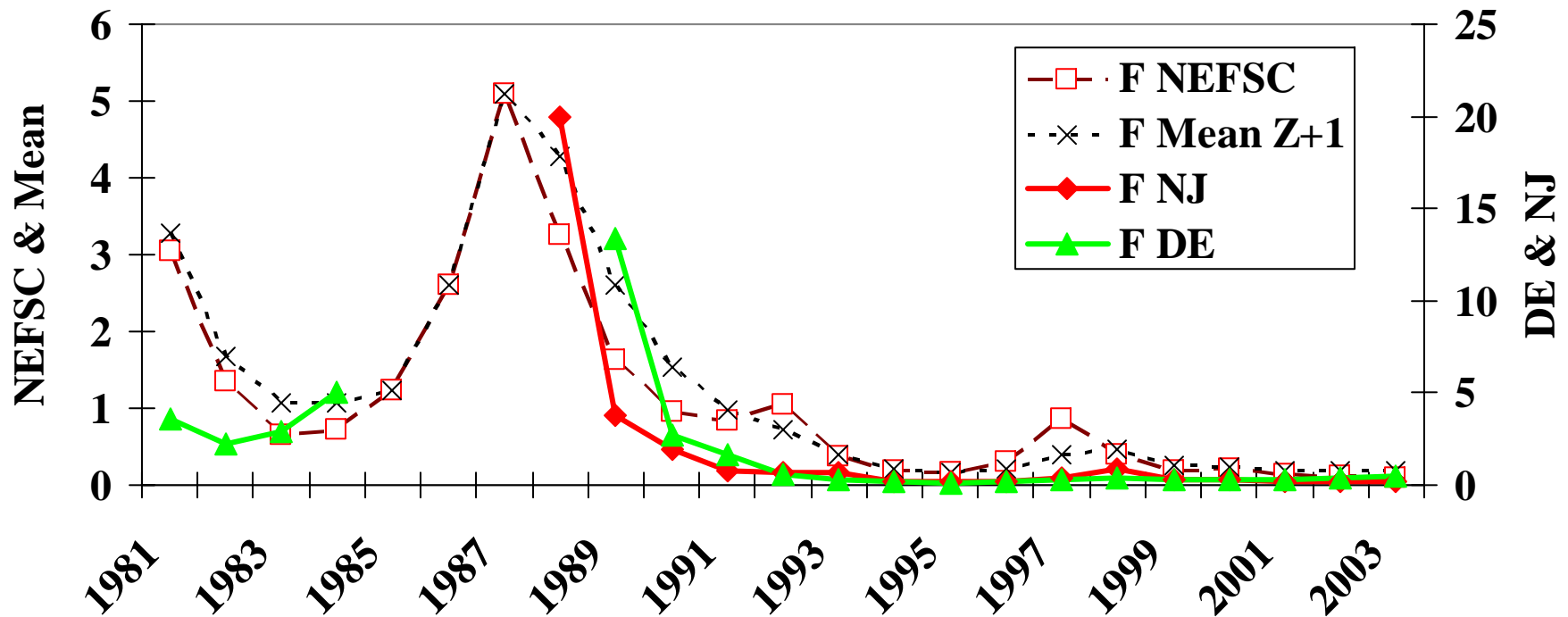


Figure 10.EFFECT OF INCLUDING CORE SPECIES COMMERCIAL DISCARDS ON 2002 CAA

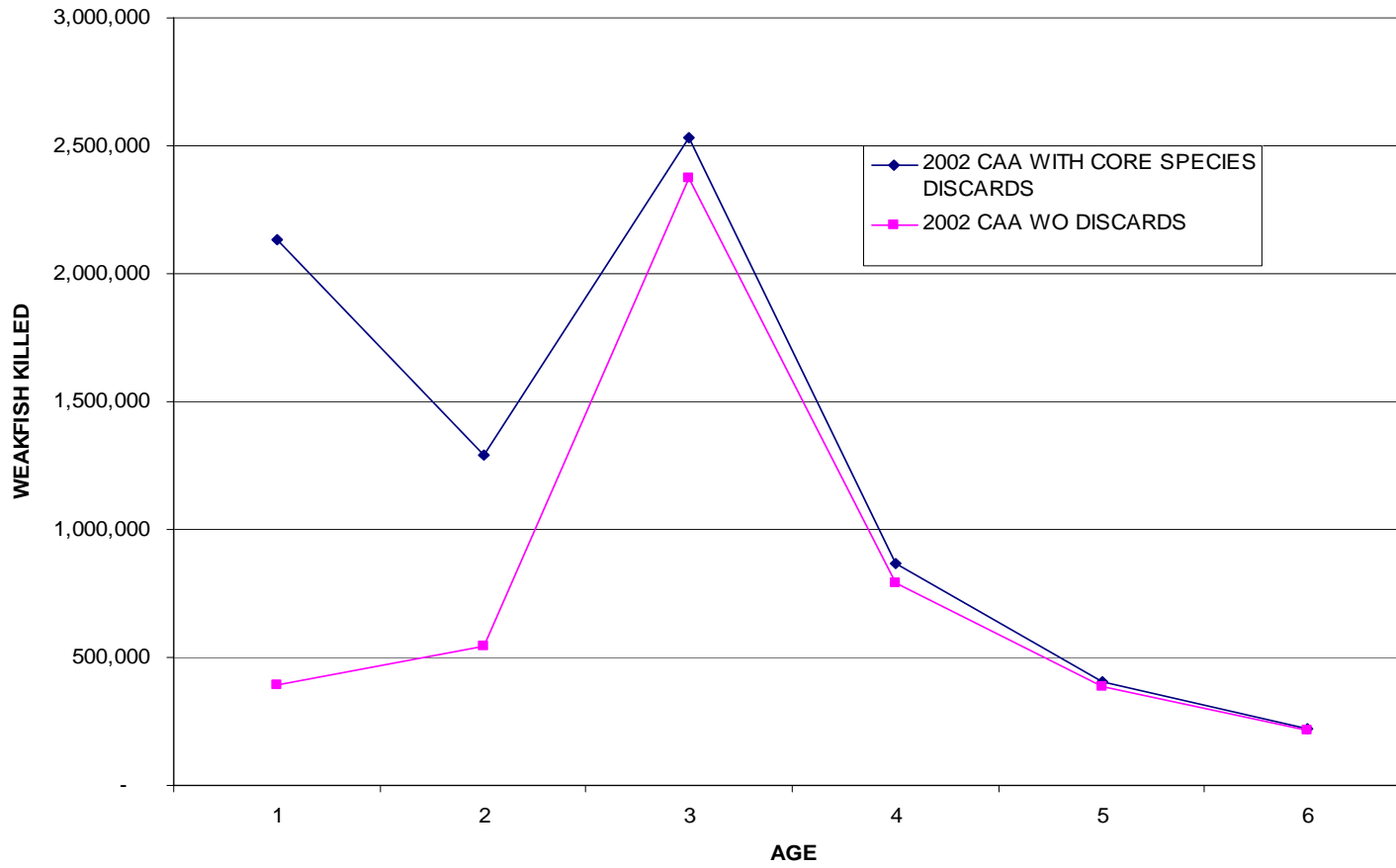


Figure 11. Mean Weight at Age of the Commercial Harvest by State, 2001

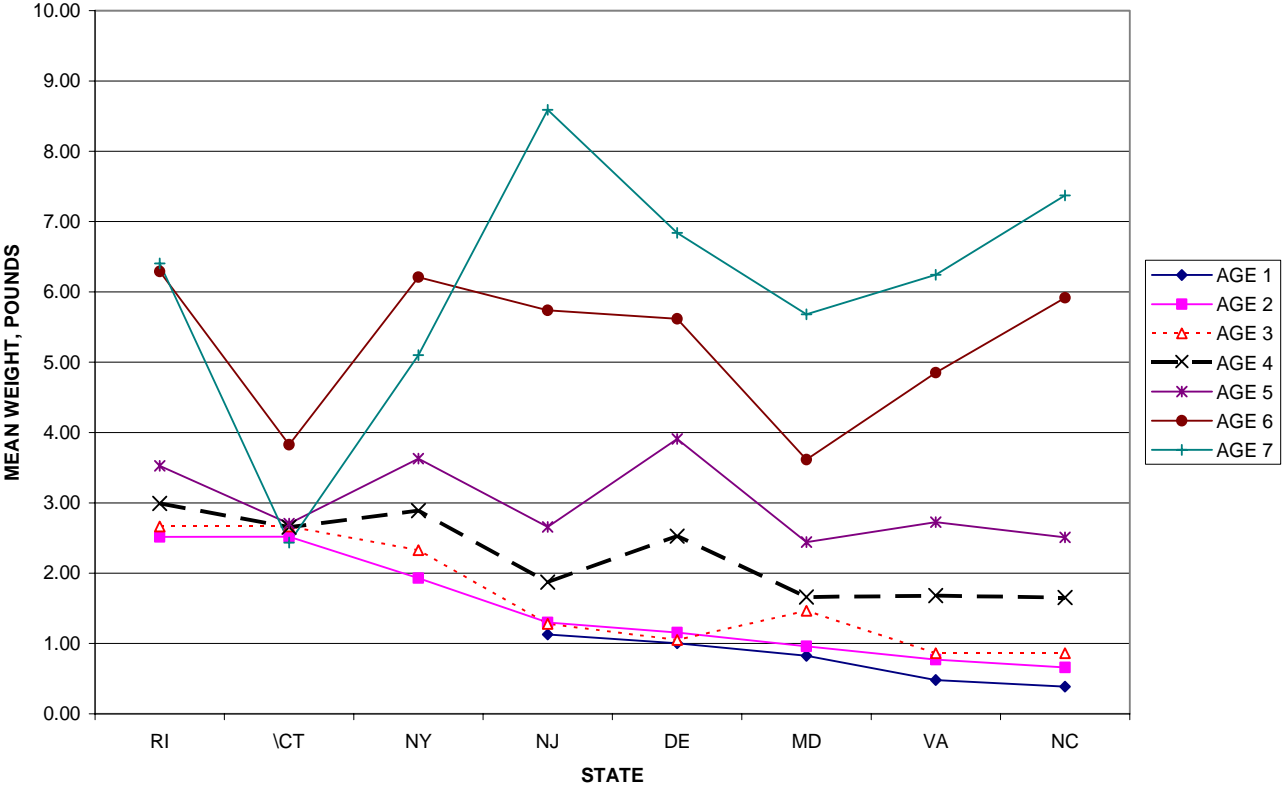


Figure 12. Mean Weight at Age of the Commercial Harvest by State, 2002

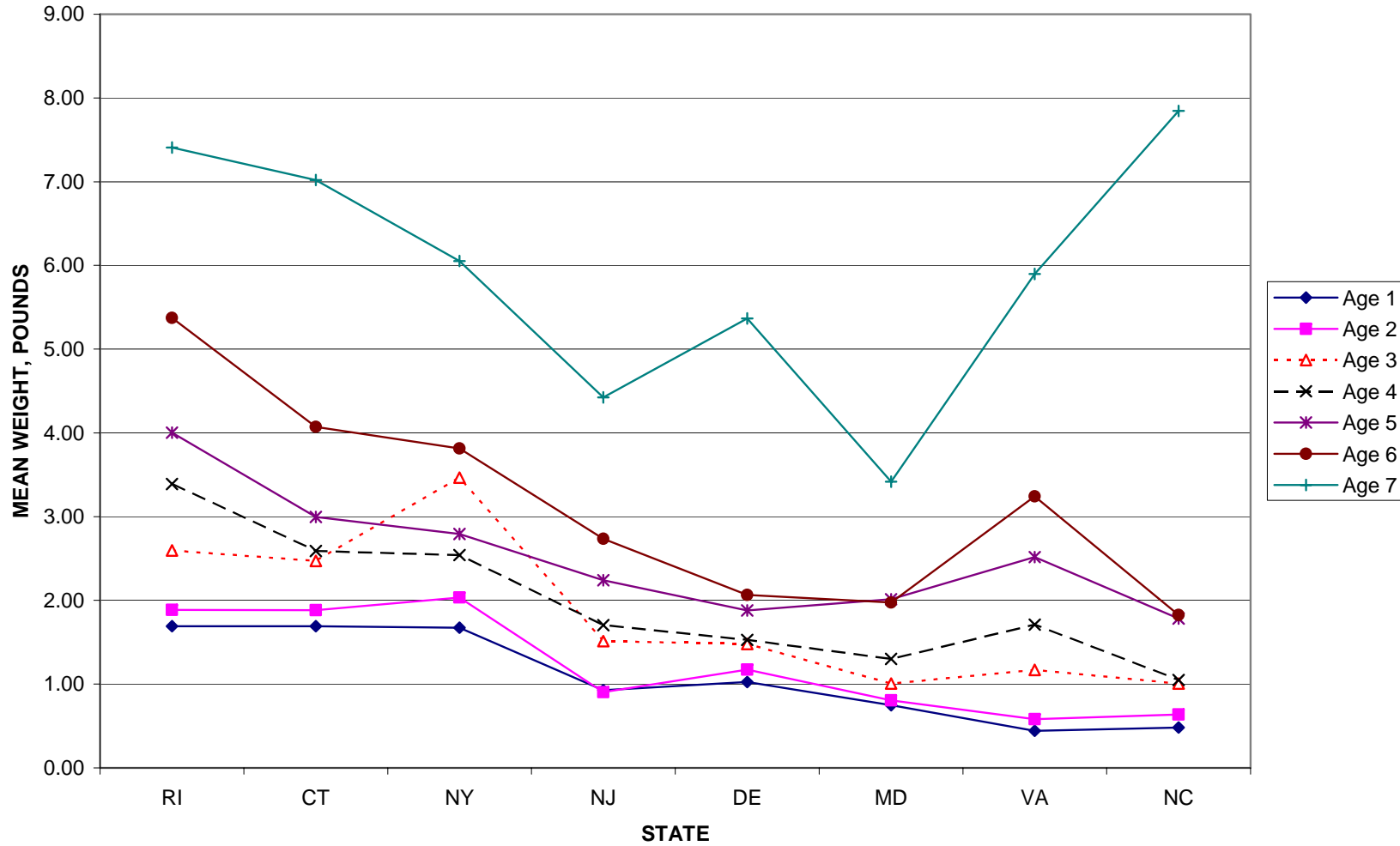


Figure 13. Mean Weight at age of the commercial harvest by state, 2003

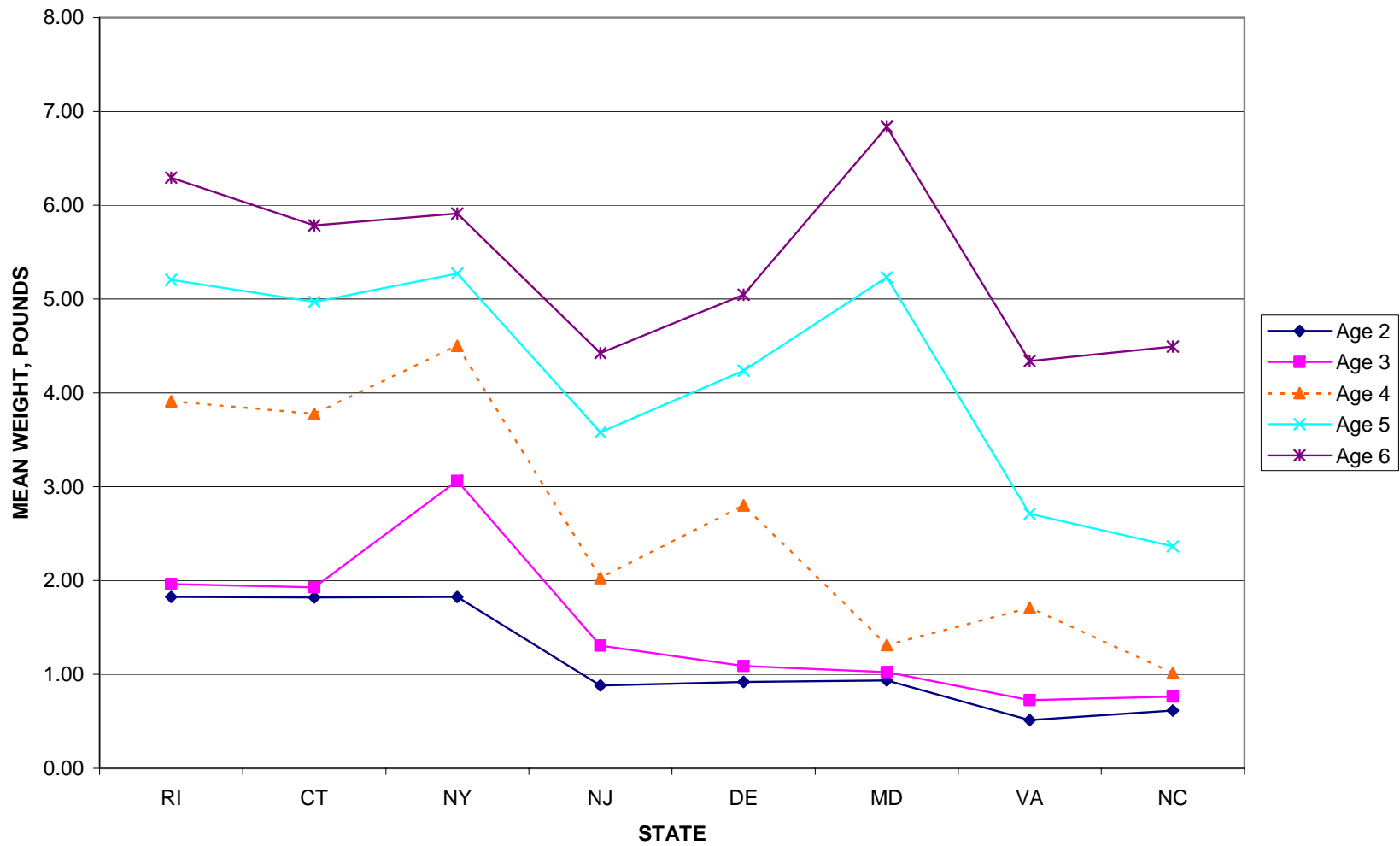


Figure 14. January 1 Stock Mean Weight at Age

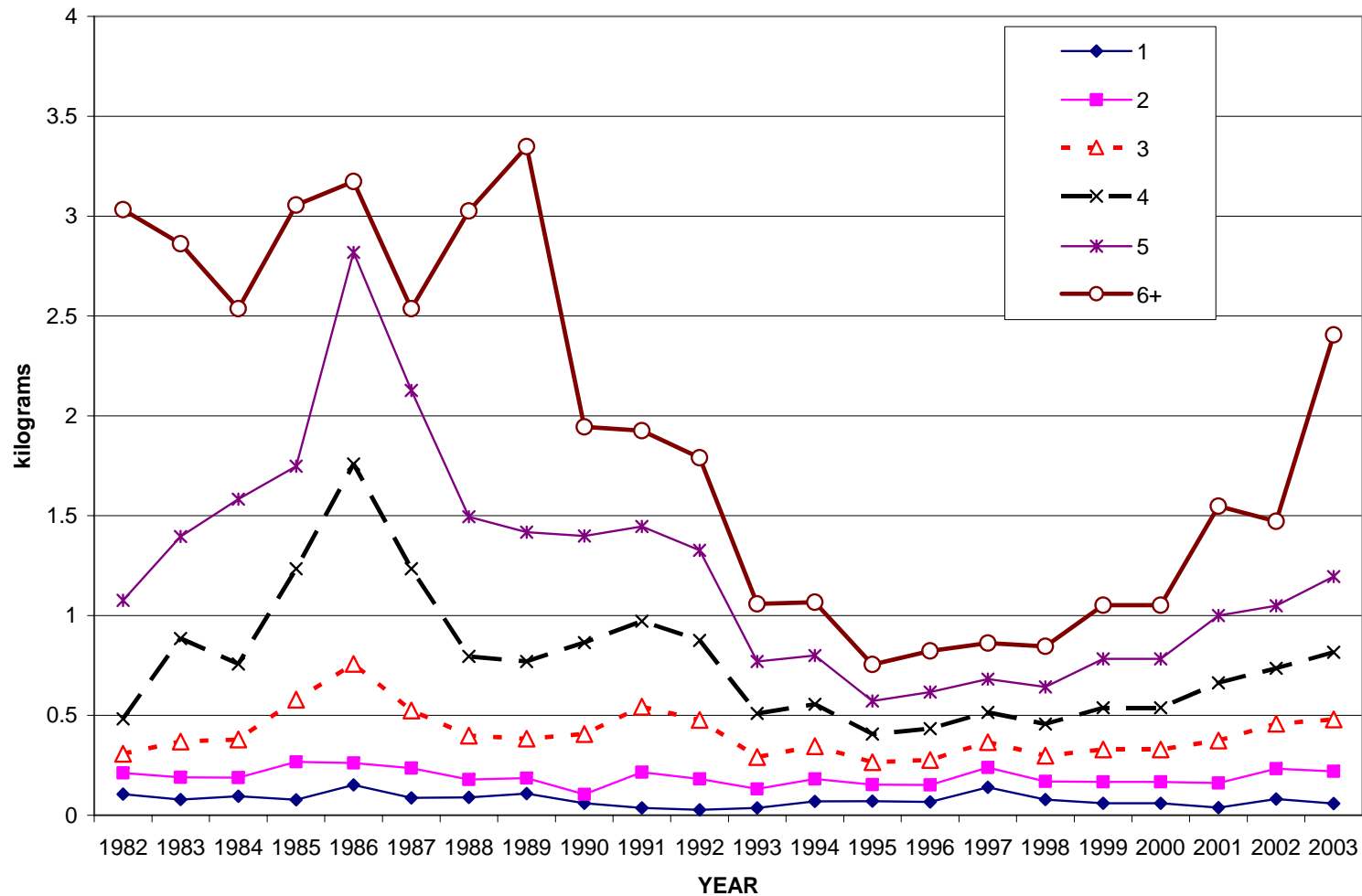


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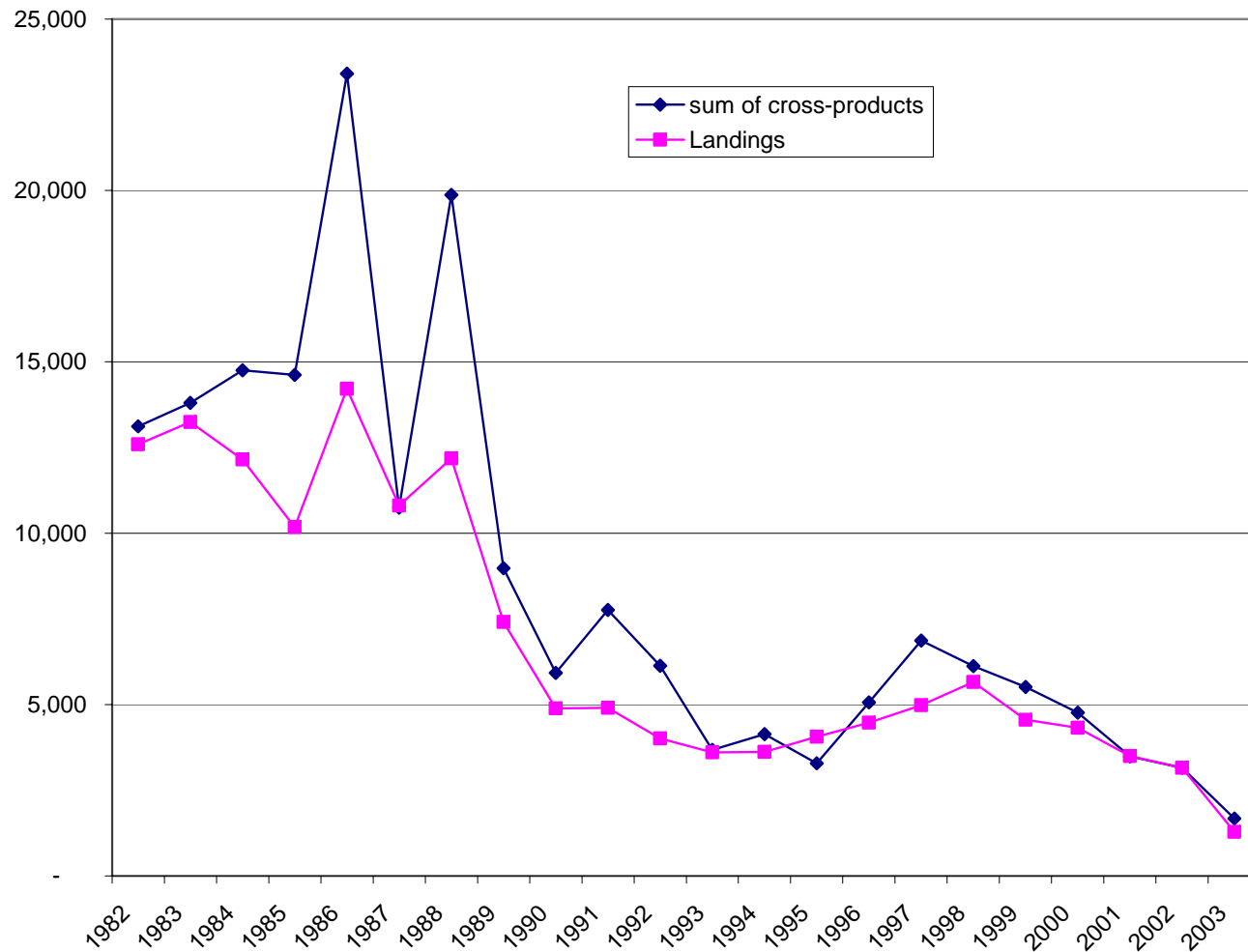


Figure 16. Weakfish Mortality Estimated From Catch Curve Analysis of the Catch at Age Matrix

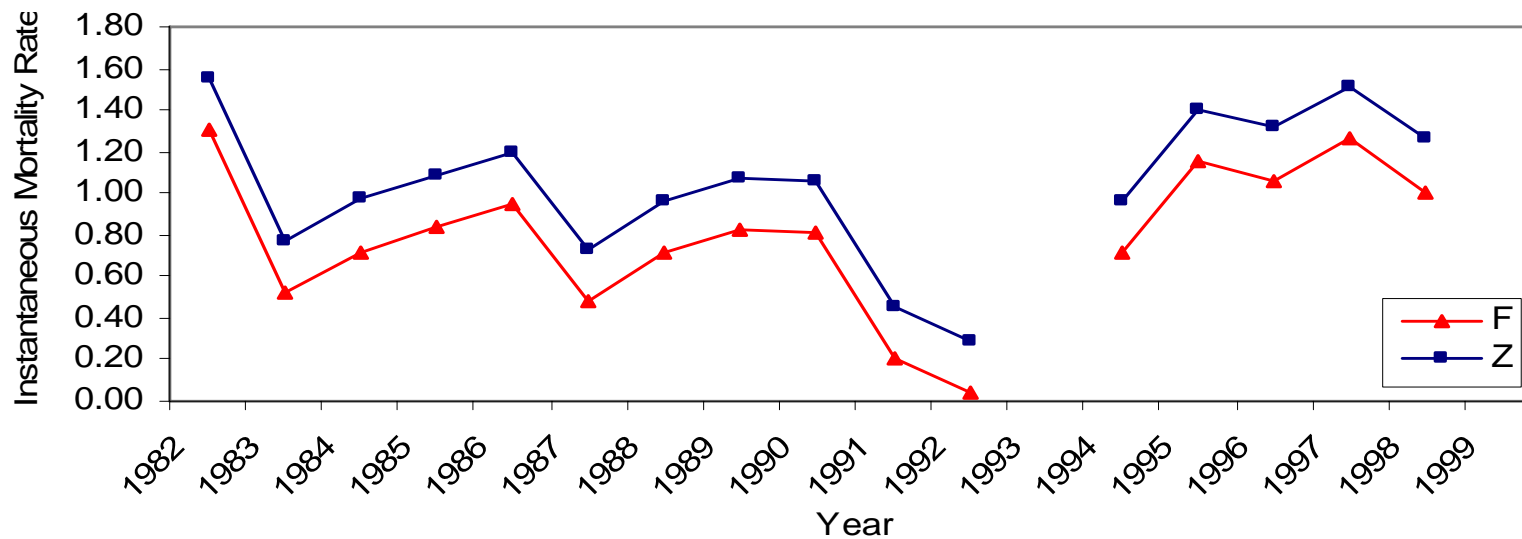


FIGURE 17. CORRELATION BETWEEN YEAR CLASS Z ESTIMATES FROM CATCH CURVE ANALYSIS OF THE CATCH AT AGE MATRIX AND THE DELAWARE SURVEY FOR THE 1990 - 1998 YEAR CLASSES (1993 YEAR CLASS EXCLUDED)

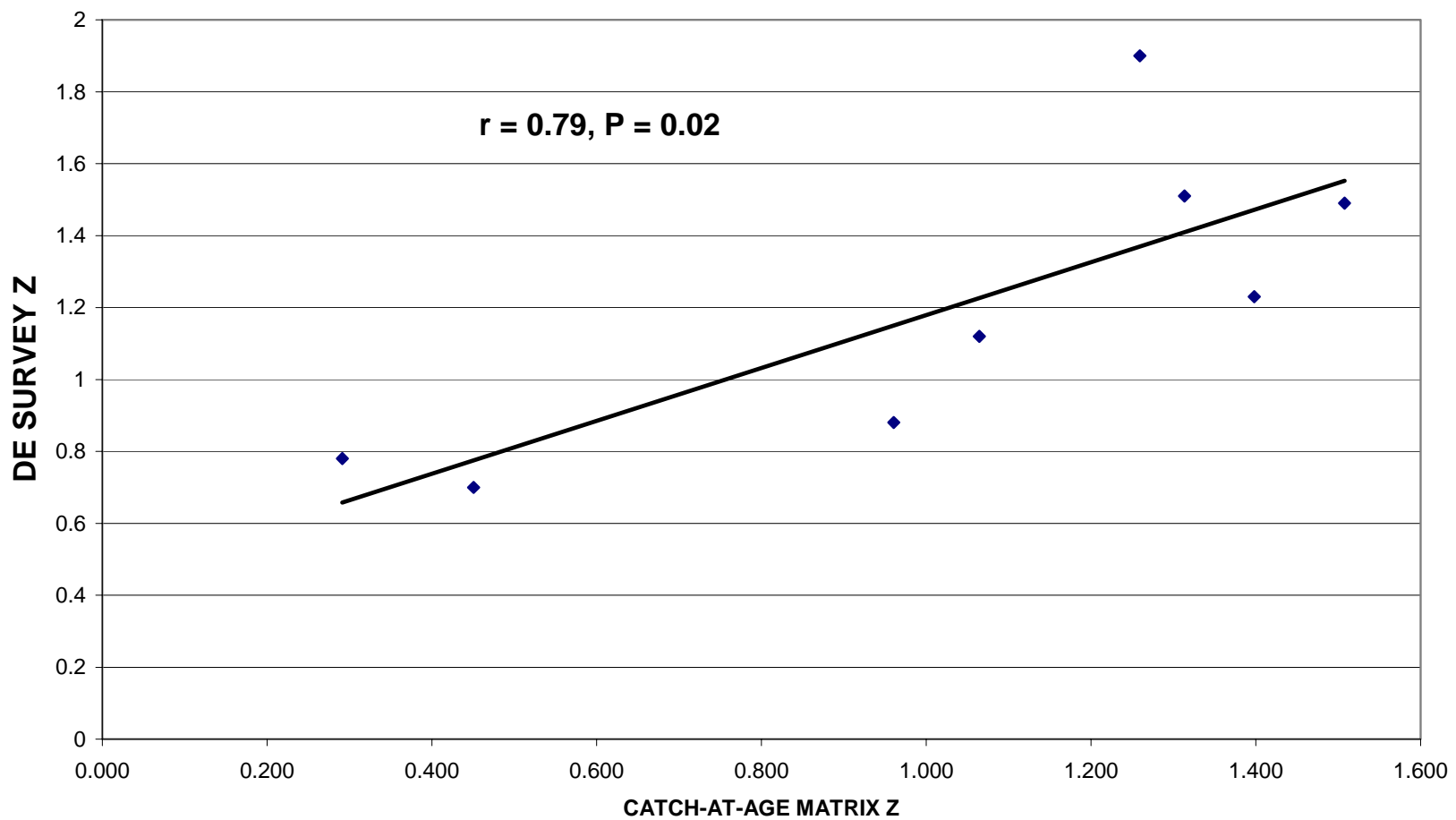
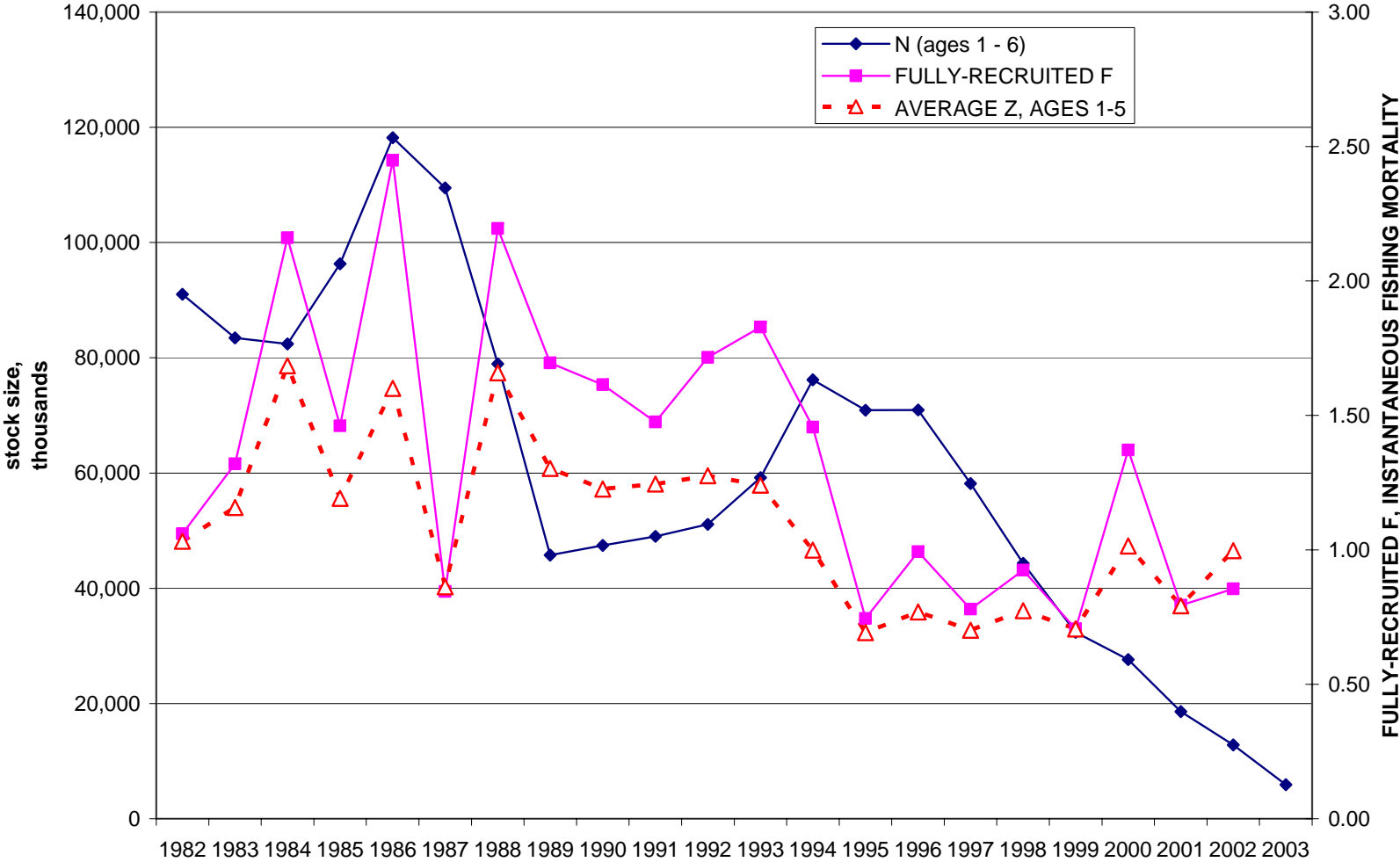


Figure 18. Gulland's Cohort analysis for weakfish



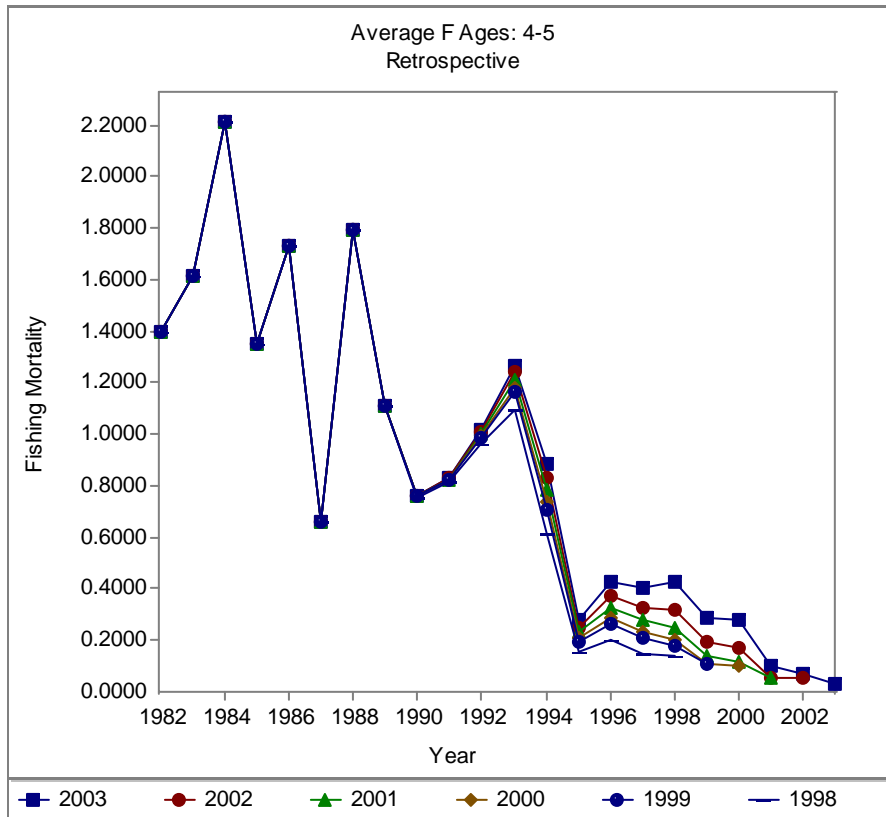


Figure 19. ADAPT run 24 retrospective pattern in fully-recruited F estimates. The 1998 line shows estimated F when only data available through 1998 is used. The 2003 line shows estimates when all data through 2003 is used.

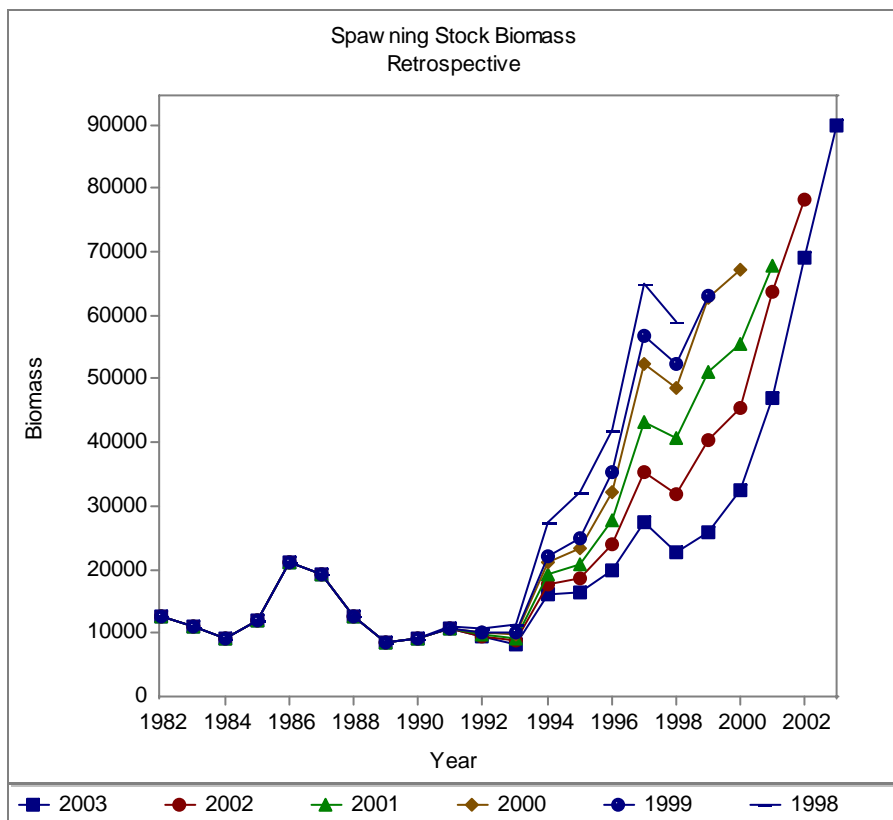


Figure 20. ADAPT run 24 retrospective pattern in estimated spawning stock biomass. Each line shows estimates when the label is the final year of data in the analysis.

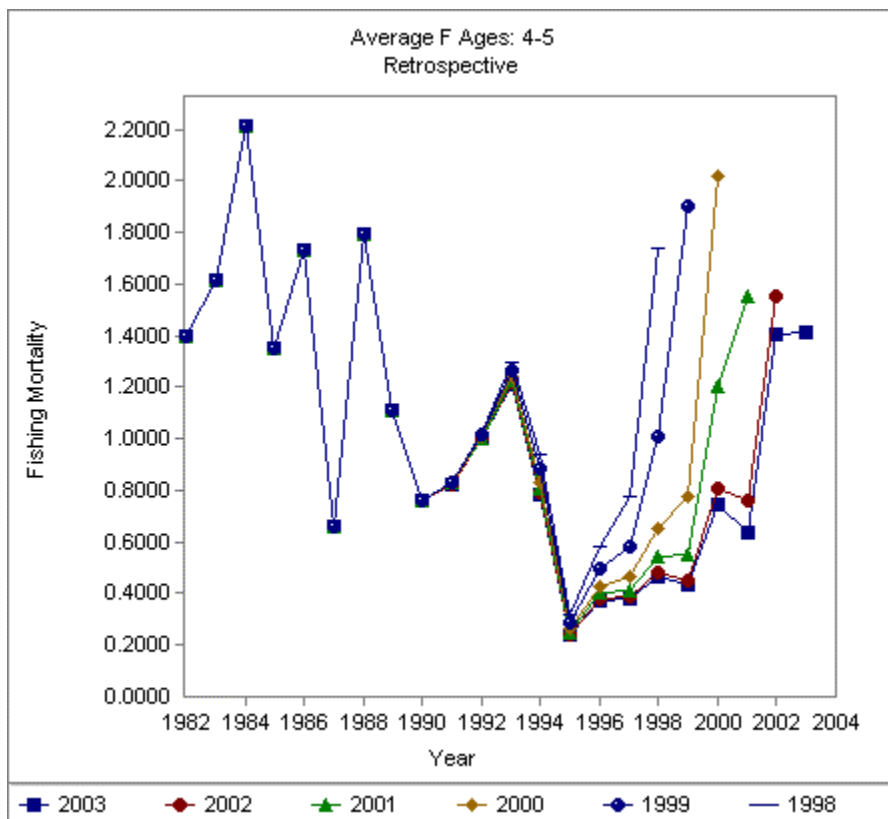


Figure 21. Run 14 retrospective pattern in fully-recruited fishing mortality.

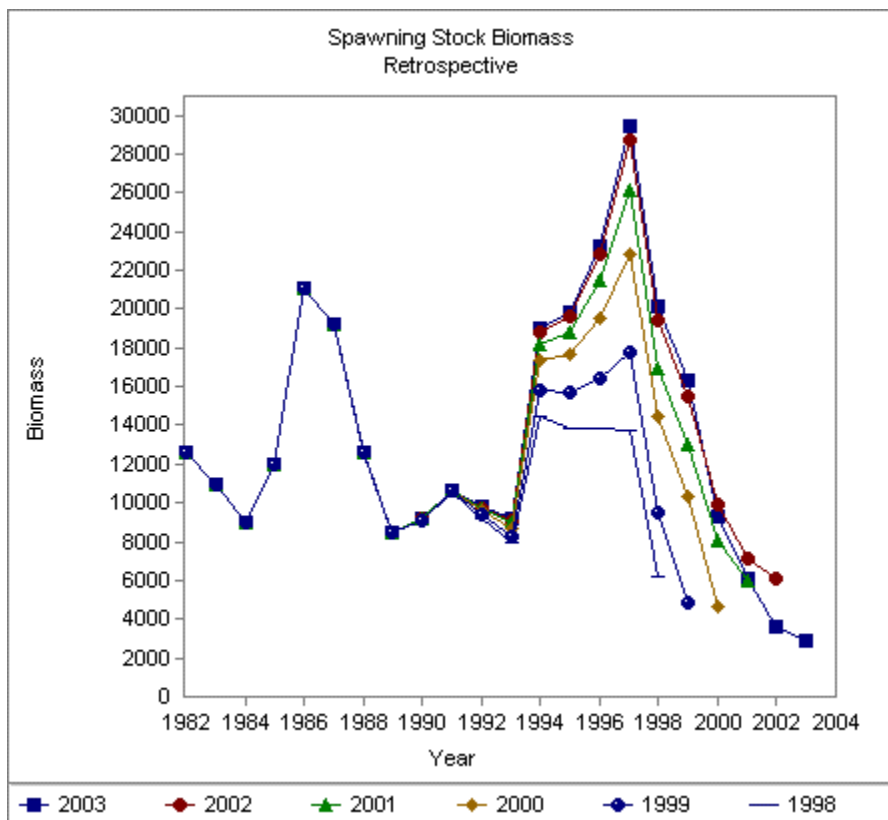


Figure 22. Run 14 retrospective pattern in spawning stock biomass estimates.

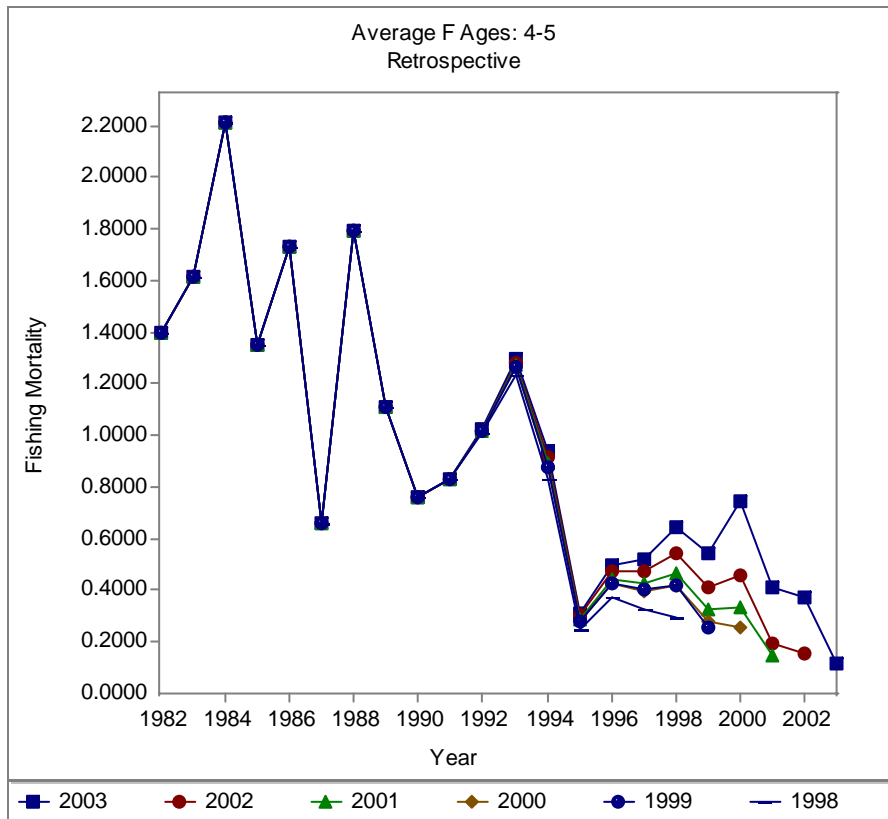
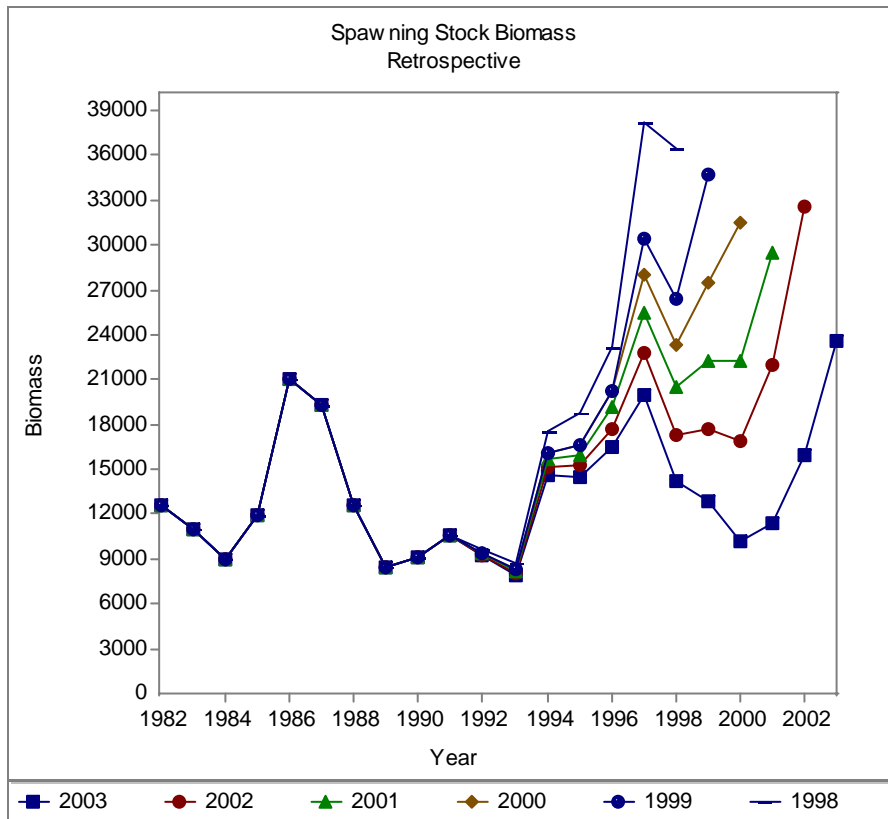


Figure 23A. Run 18 retrospective pattern for fully-recruited F.



Run 23B. Run 18 retrospective pattern in spawning stock biomass.

Figure 24A. DEDFW Survey residuals from Run 18

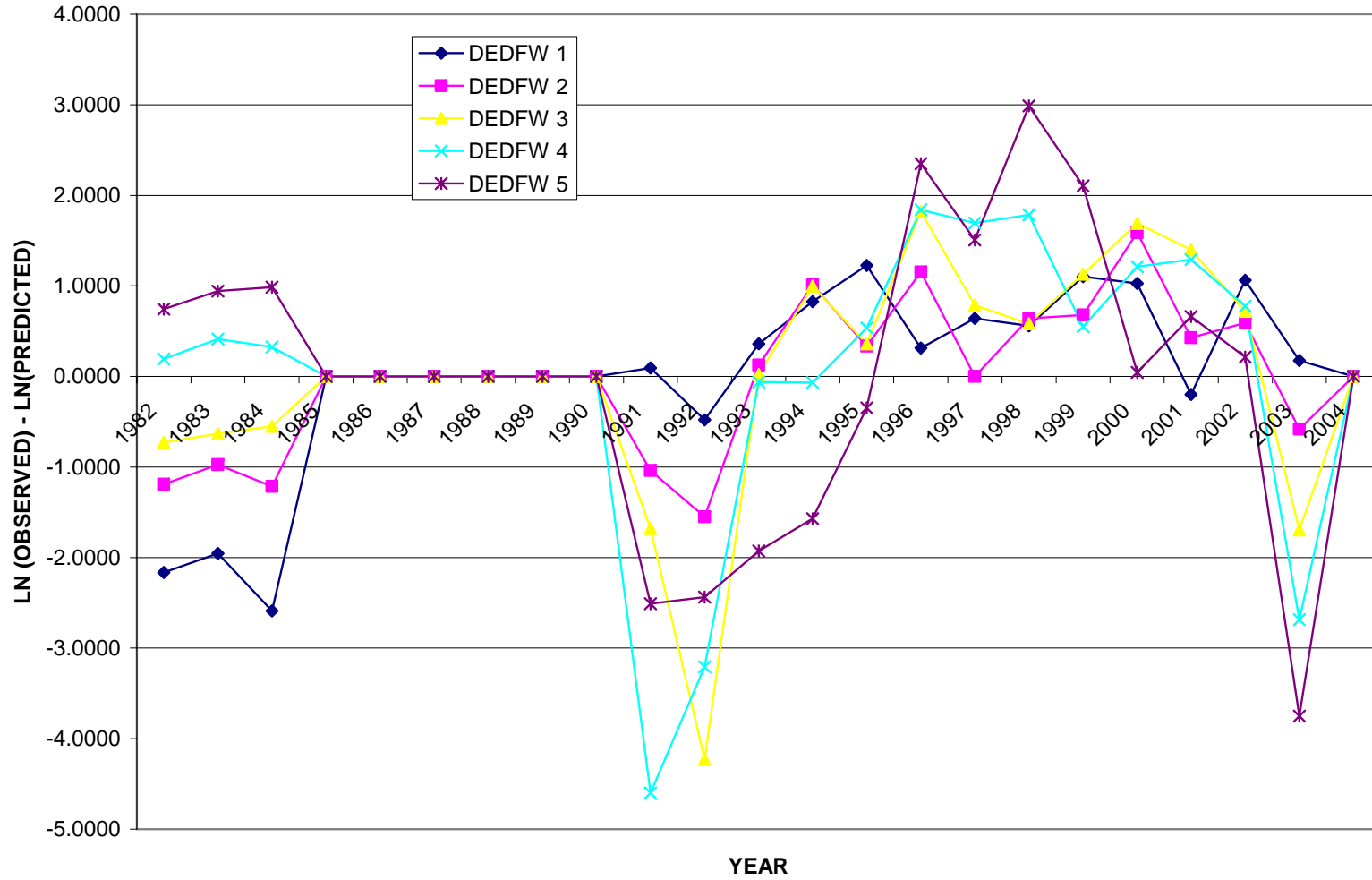


Figure 24B. SEAMAP Residuals from Run 18

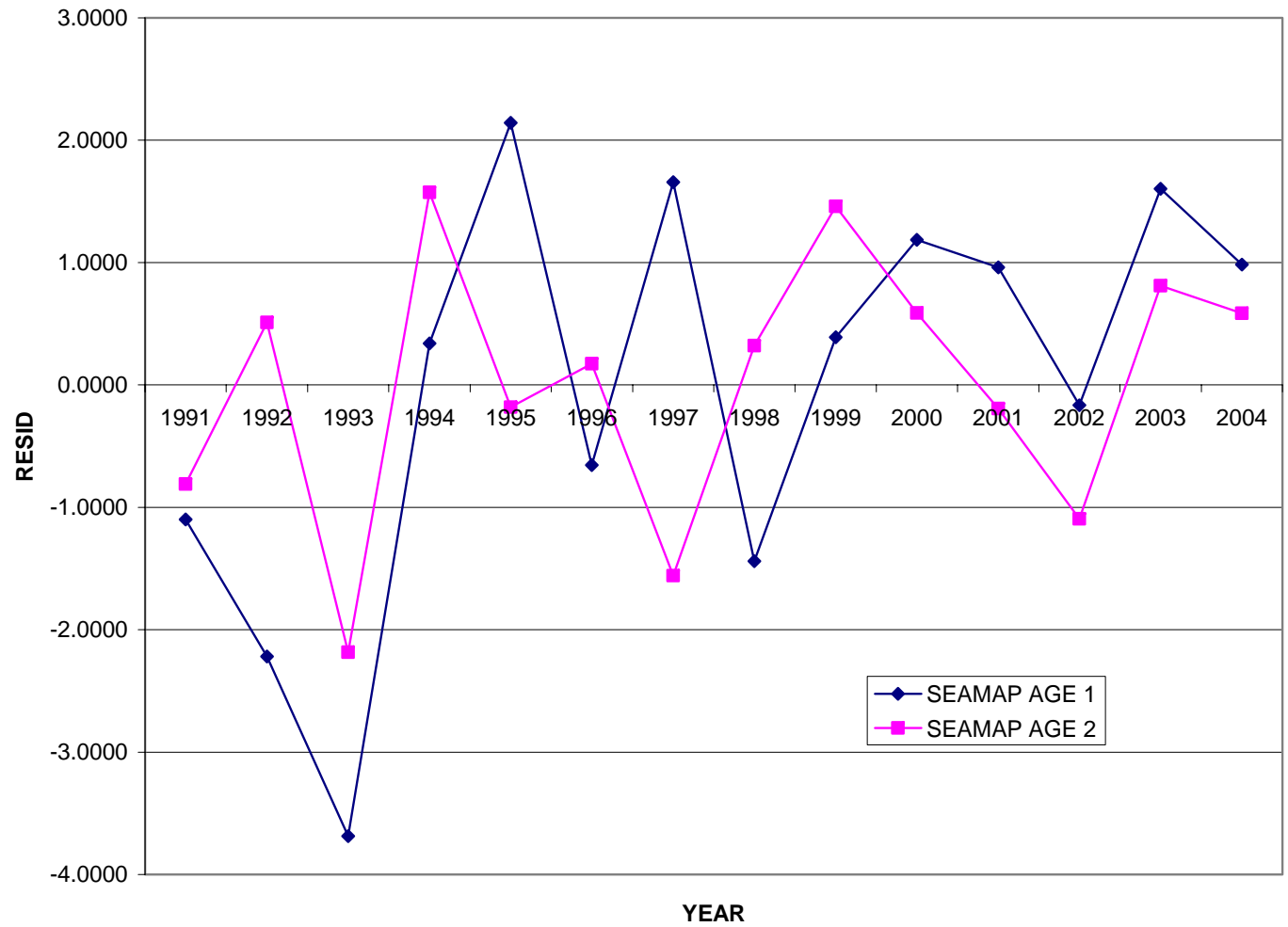


Figure 24C. Residuals from recruitment indices, Run 18

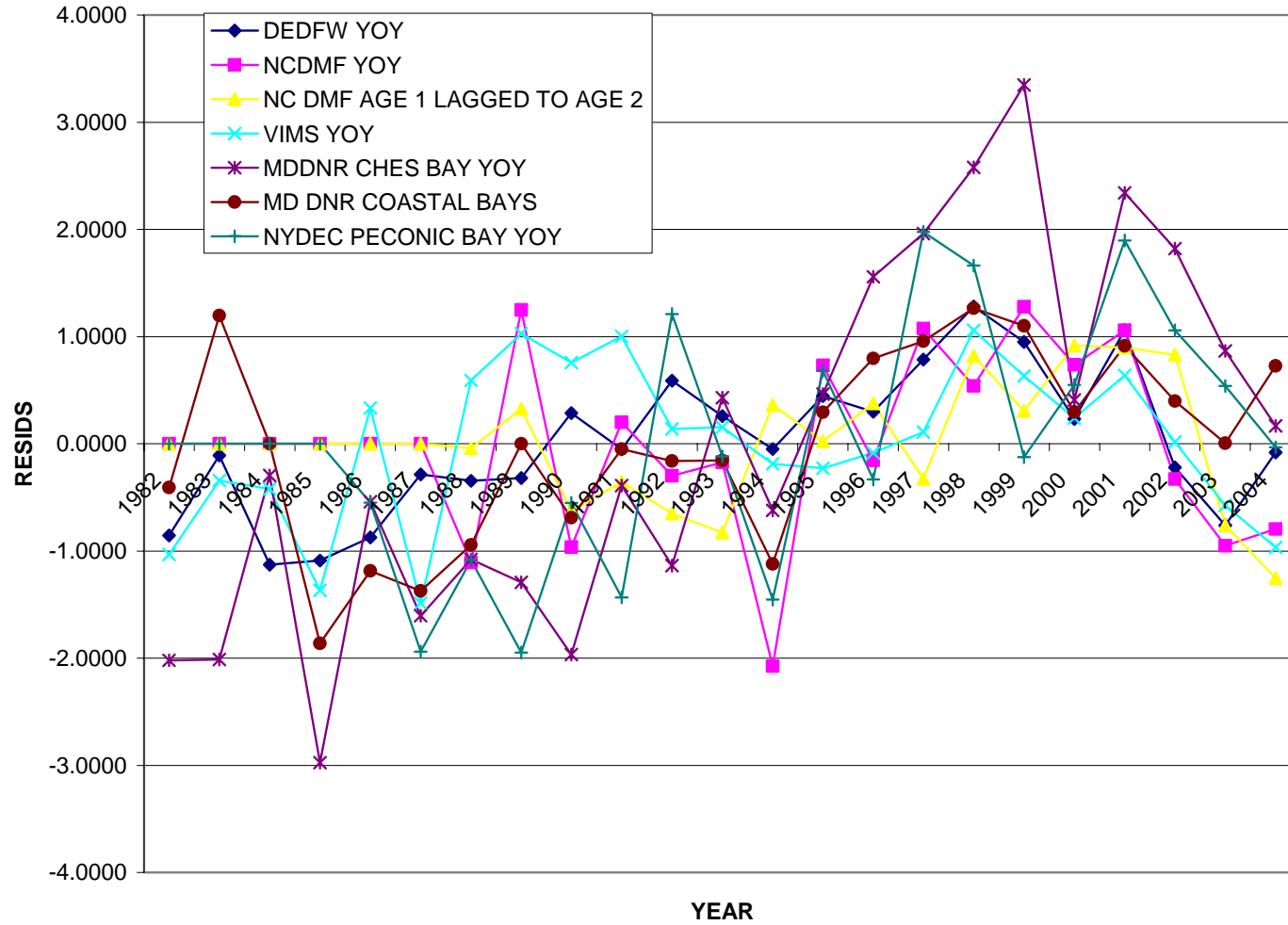


Figure 24D. Recreational CPUE residuals from Run 18

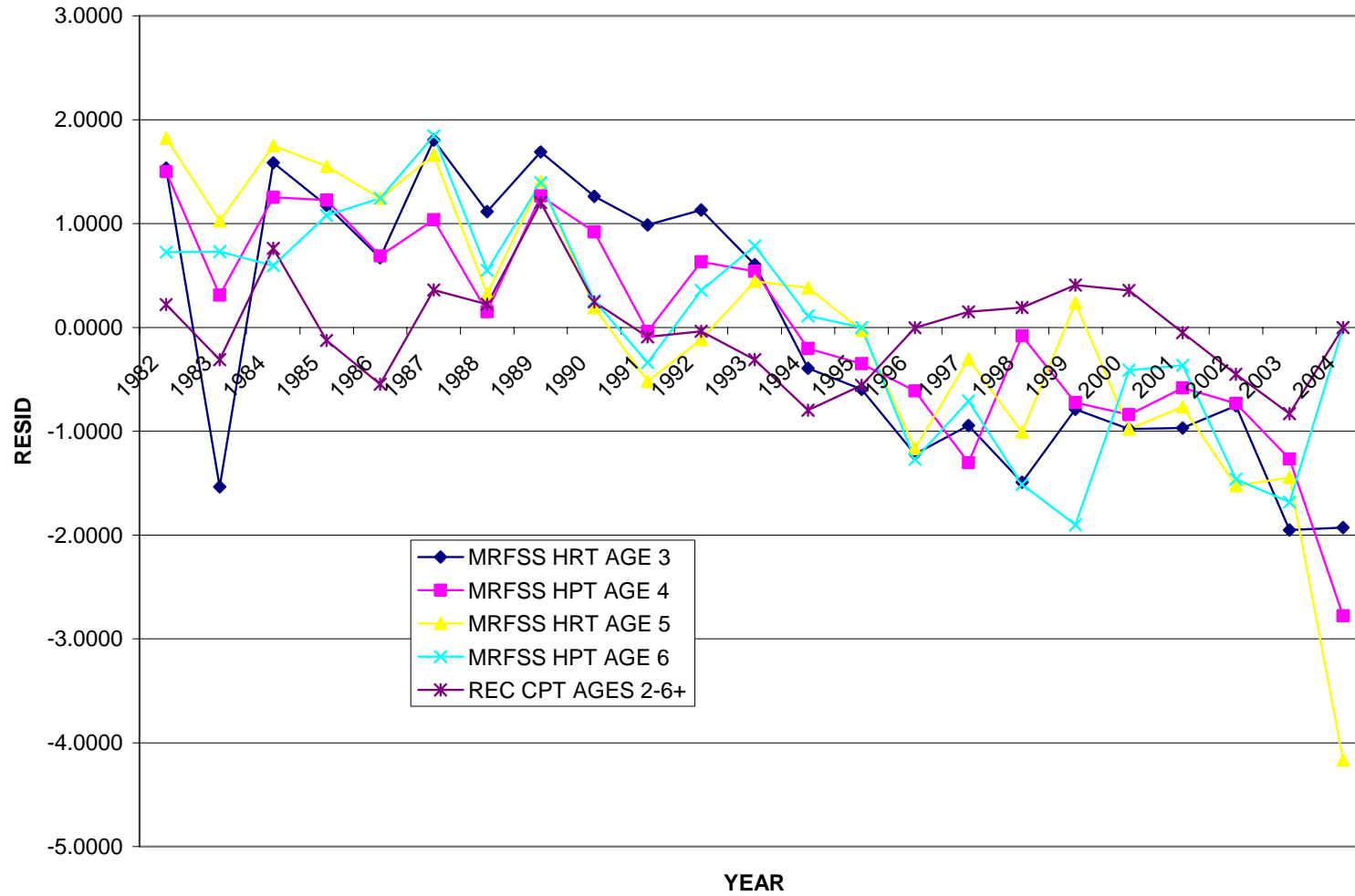


Figure 25. Run 20 Retrospective Pattern in F

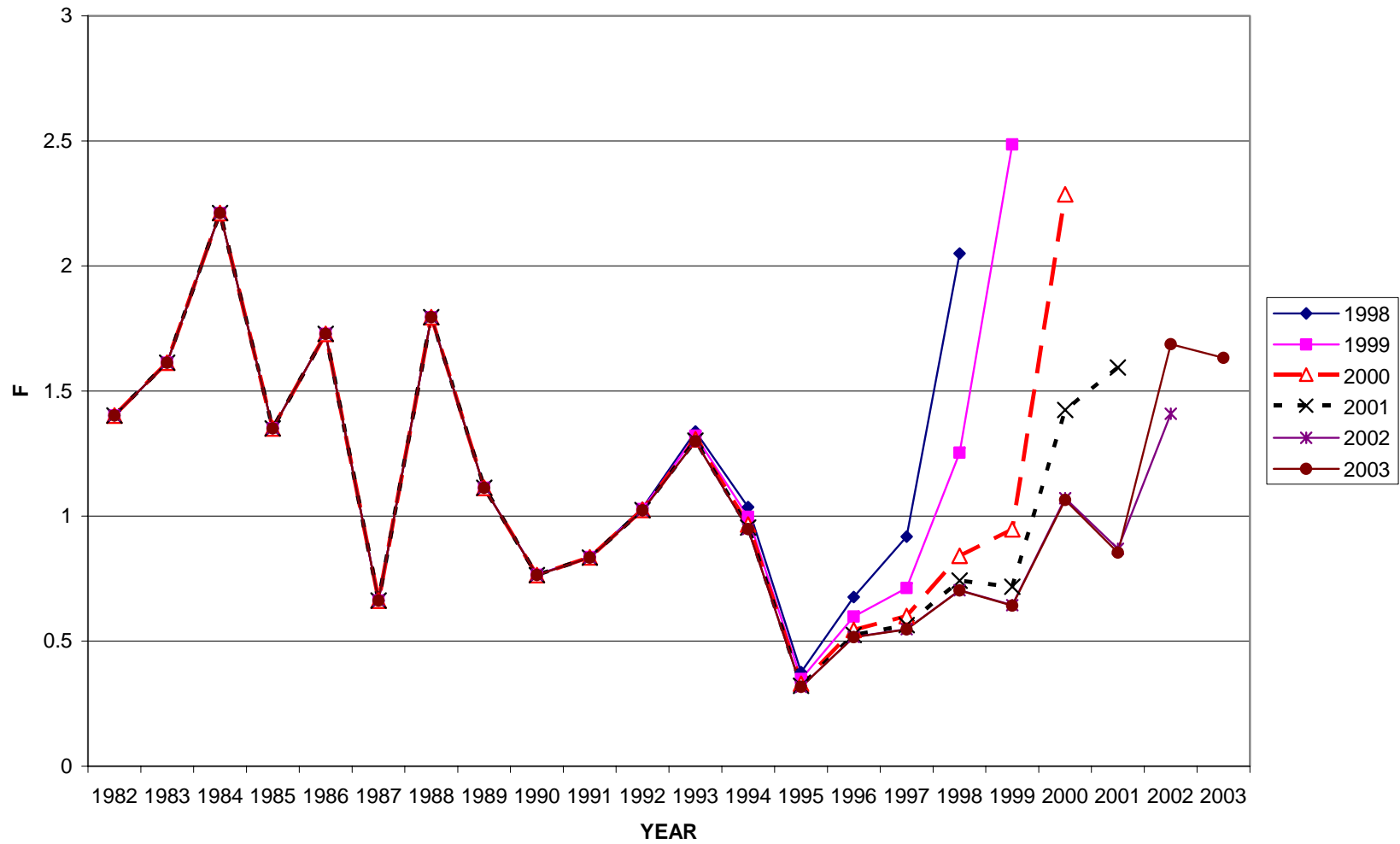


Figure 26. Run 20 Retrospective Pattern in SSB

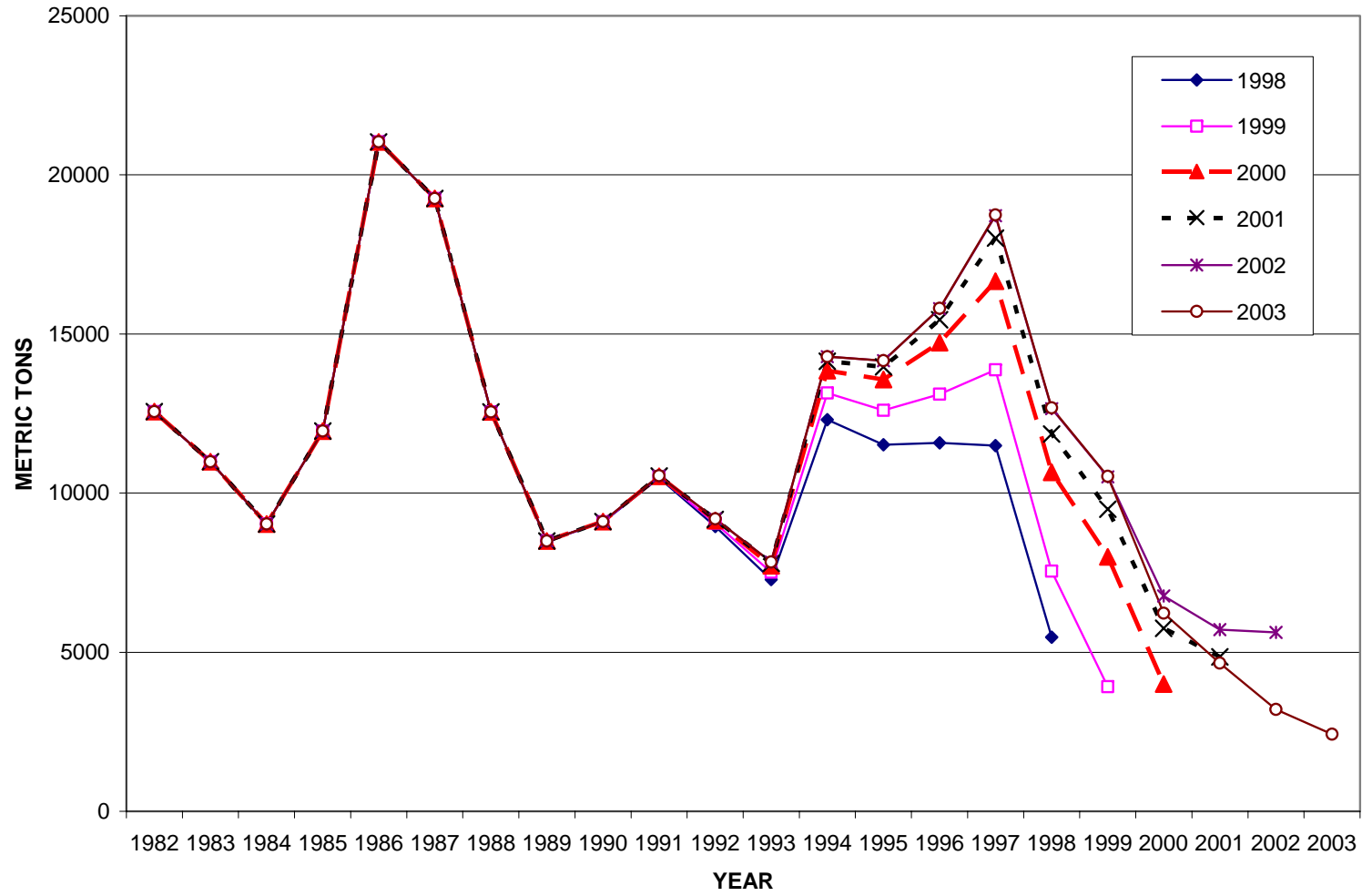


Figure 26B. Run 20 residuals from recreational CPUE indices

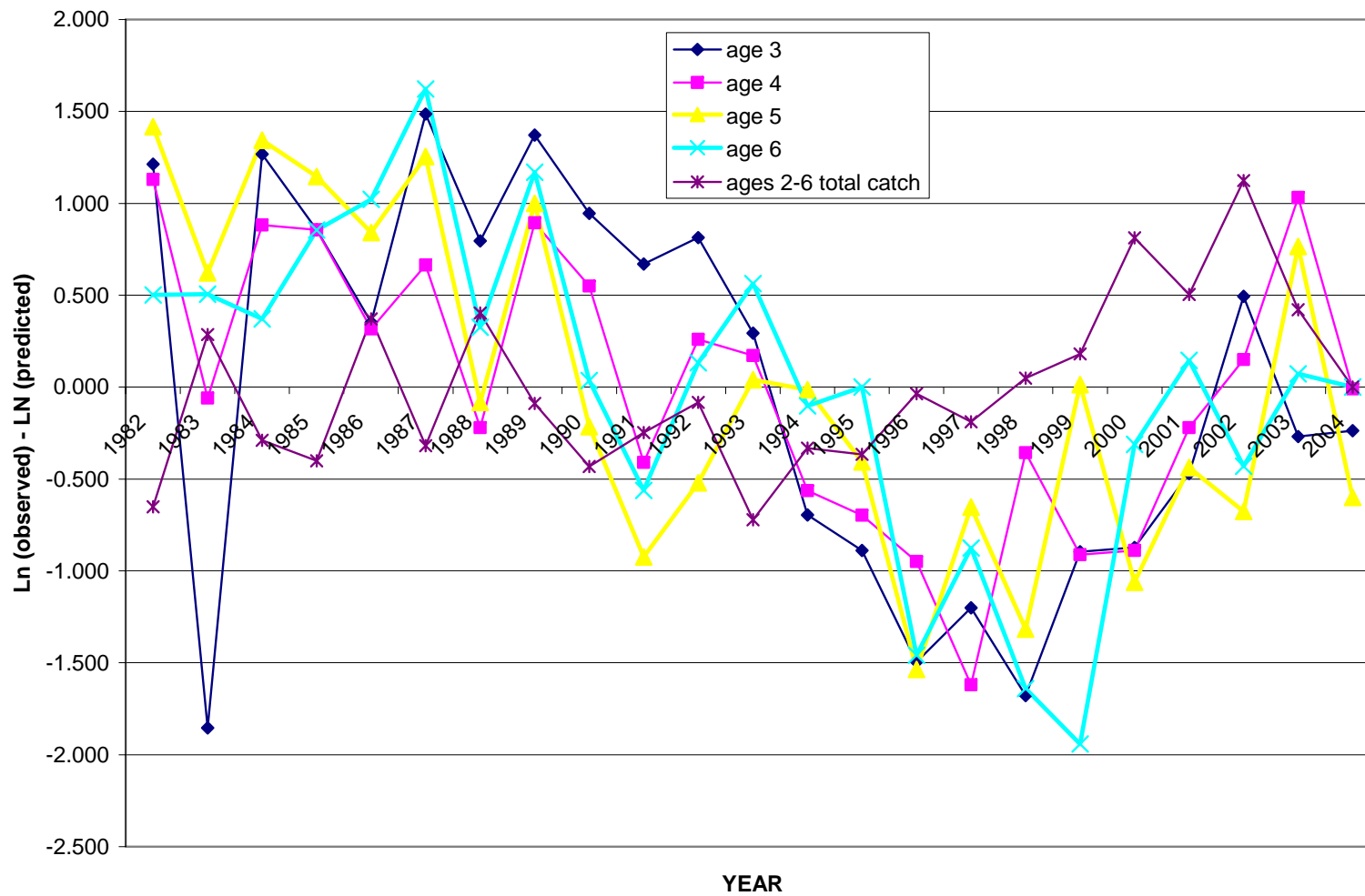


Figure 27. Delaware and New Jersey trawl survey
Quality+ PSD

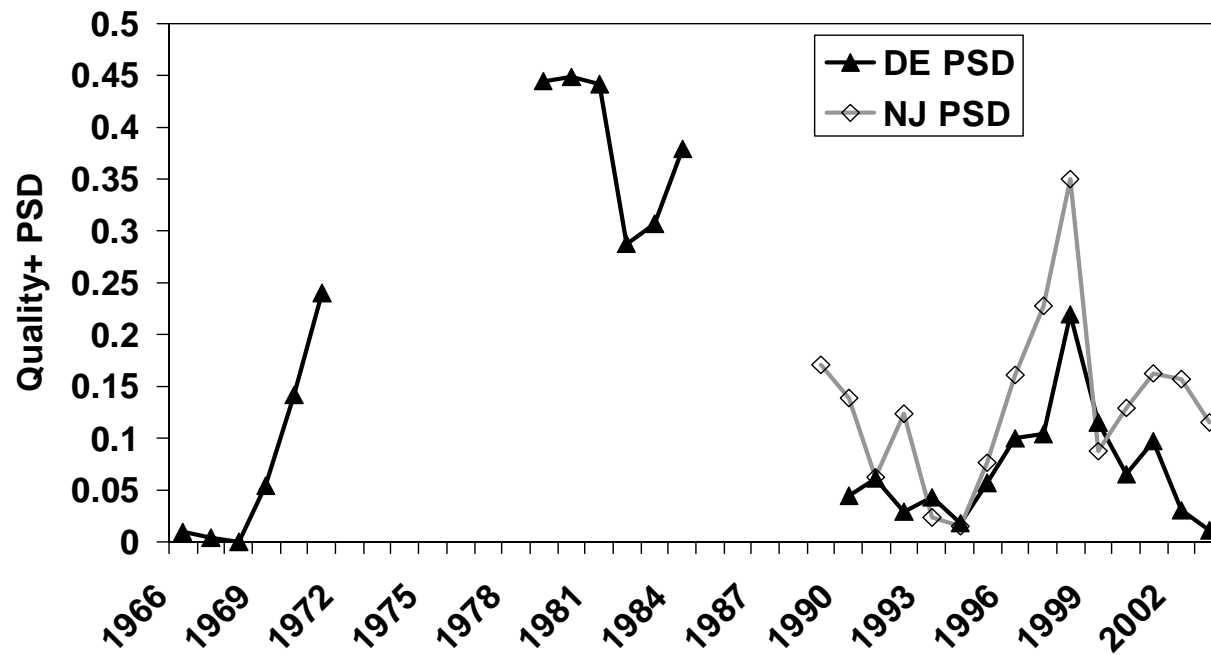


Figure 27a. Percent of population estimate that was ages 6 and older in three ADAPT VPA runs.

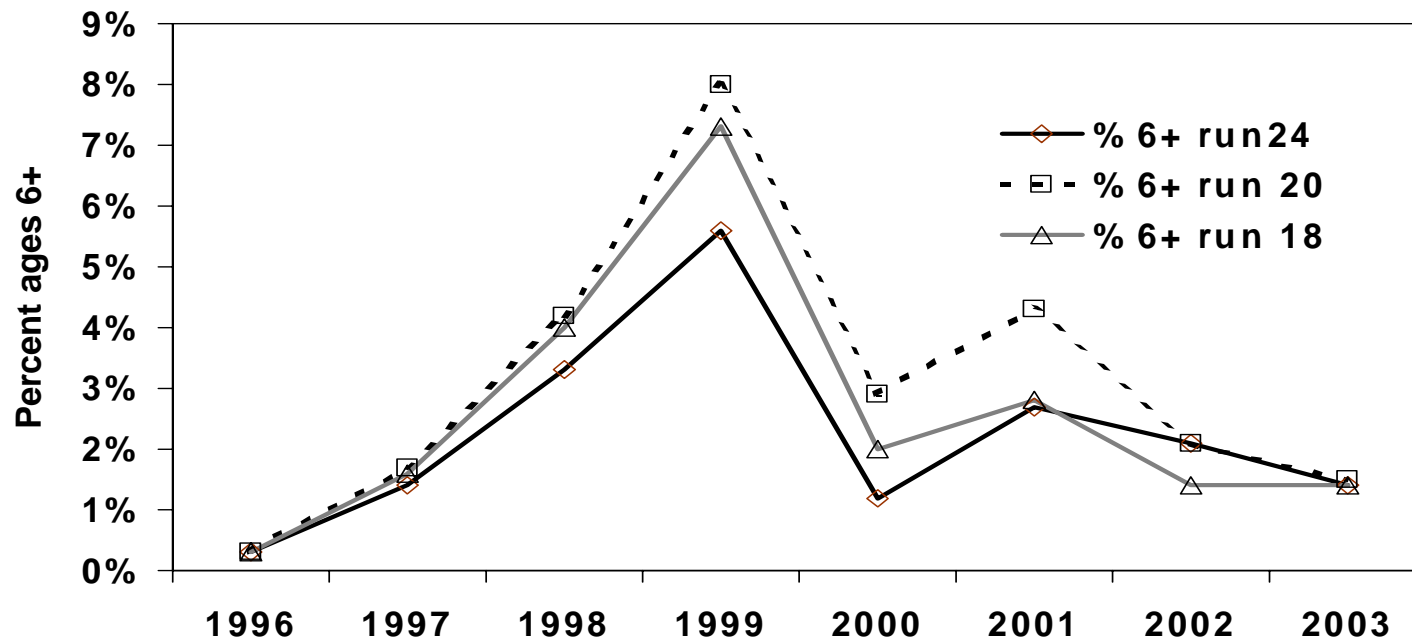
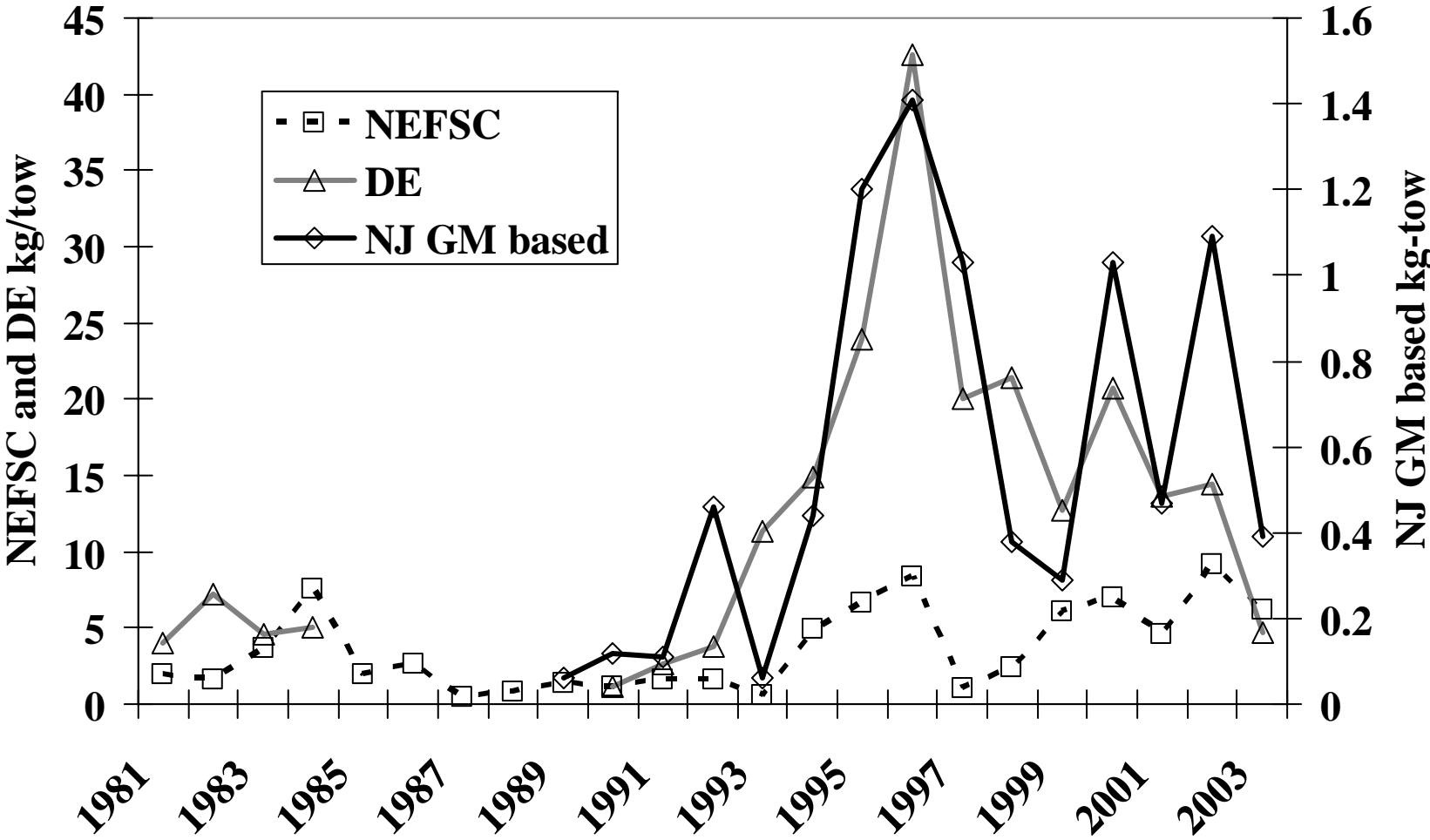


Figure 28. Trawl survey biomass indices (kg/tow of 25 cm + weakfish)



**Figure 29. Model results based on trawl survey indices:
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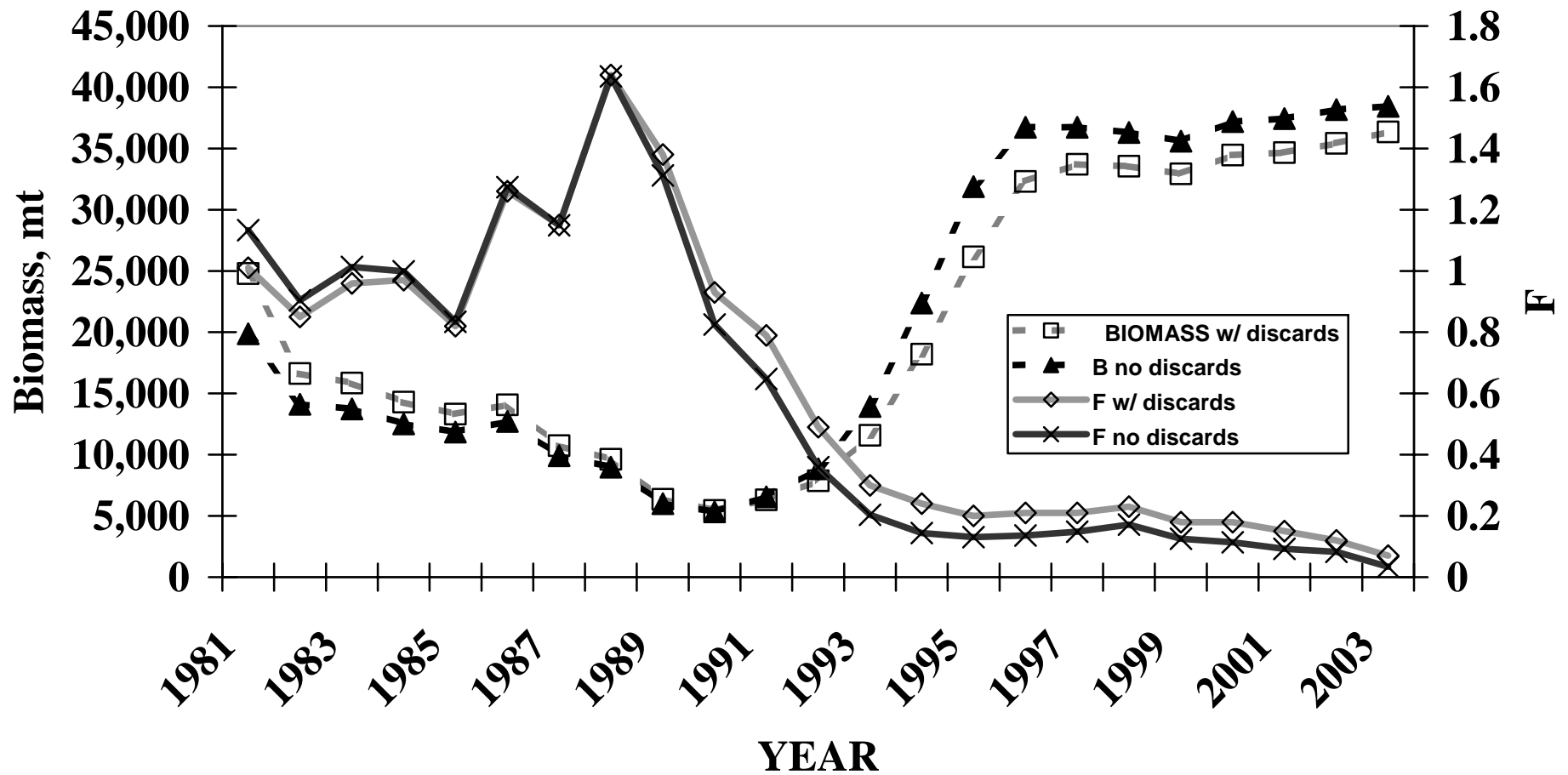


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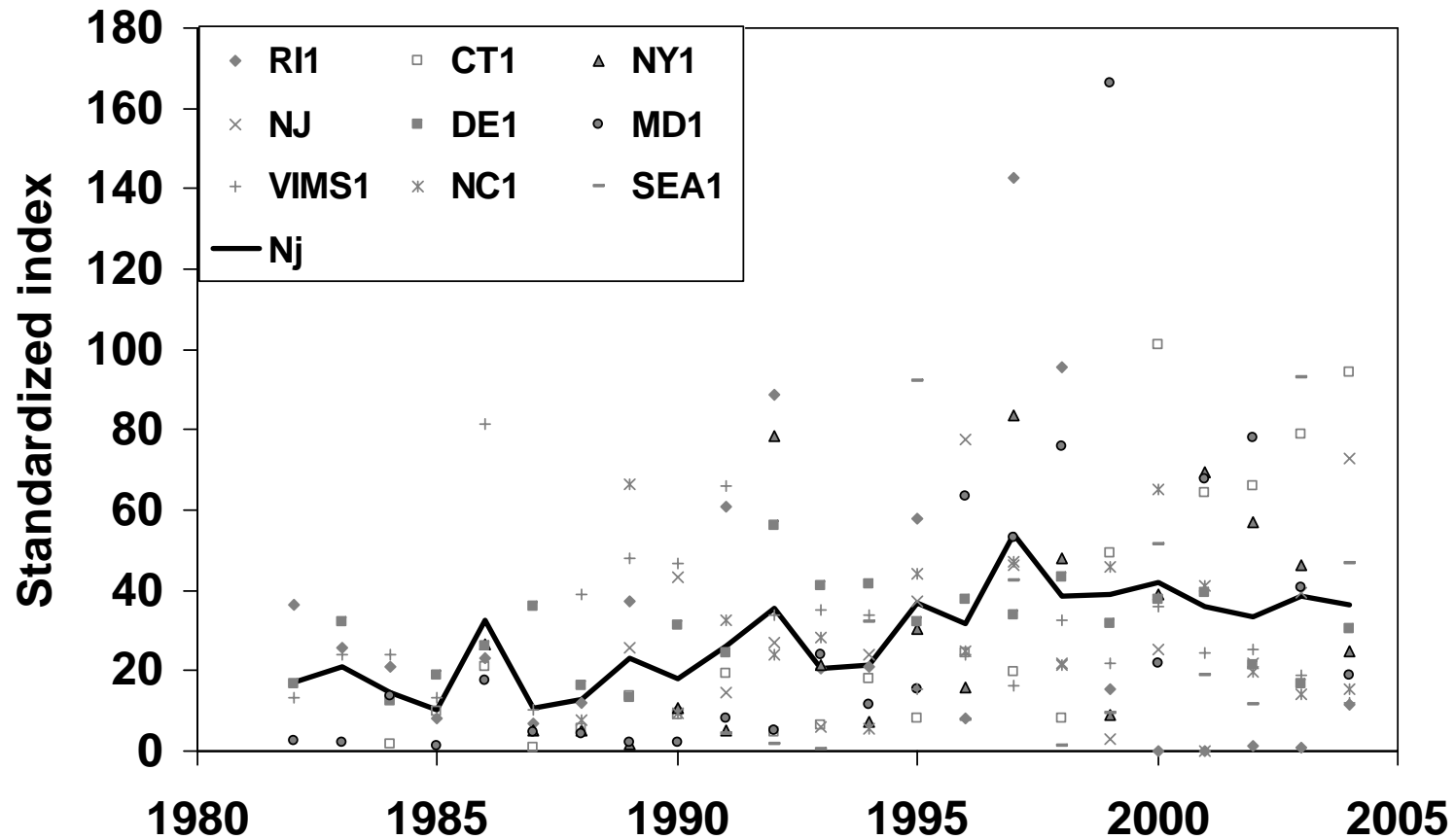


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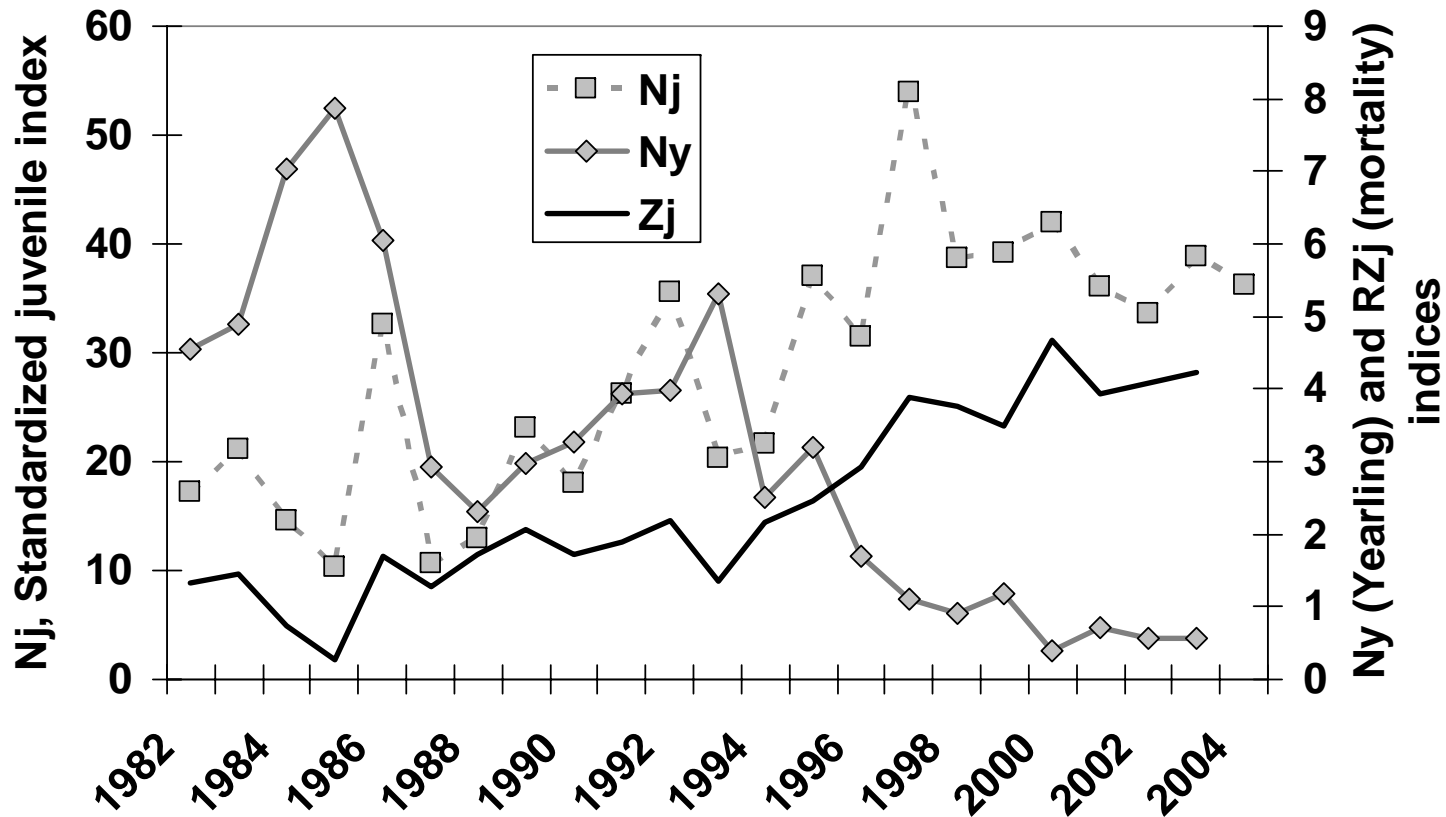


Figure 34. Trends in M_{Wt} .

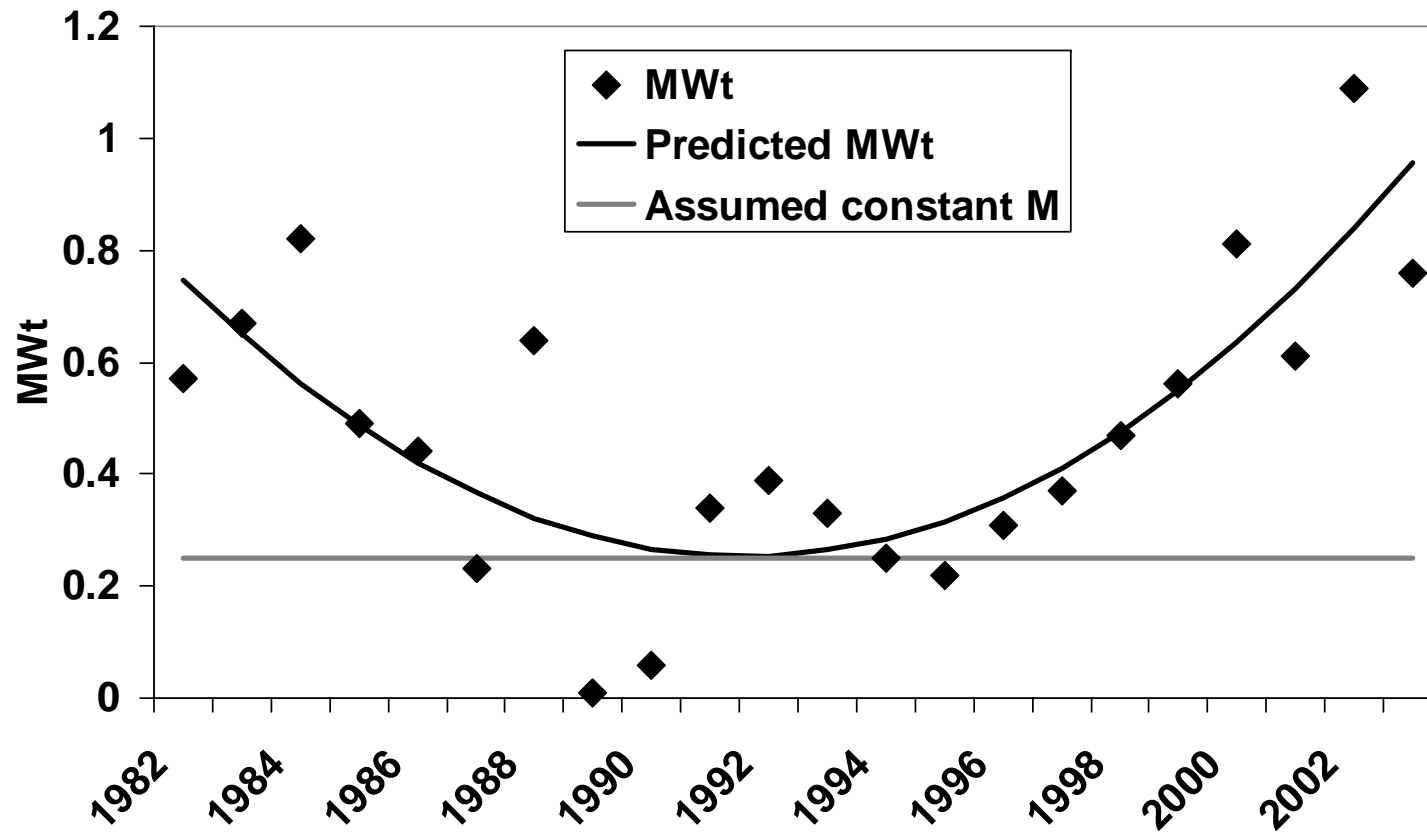


Figure 35. Juvenile weakfish mortality index (units arbitrary) and VPA-based estimates of biomass of ages 8+ striped bass.

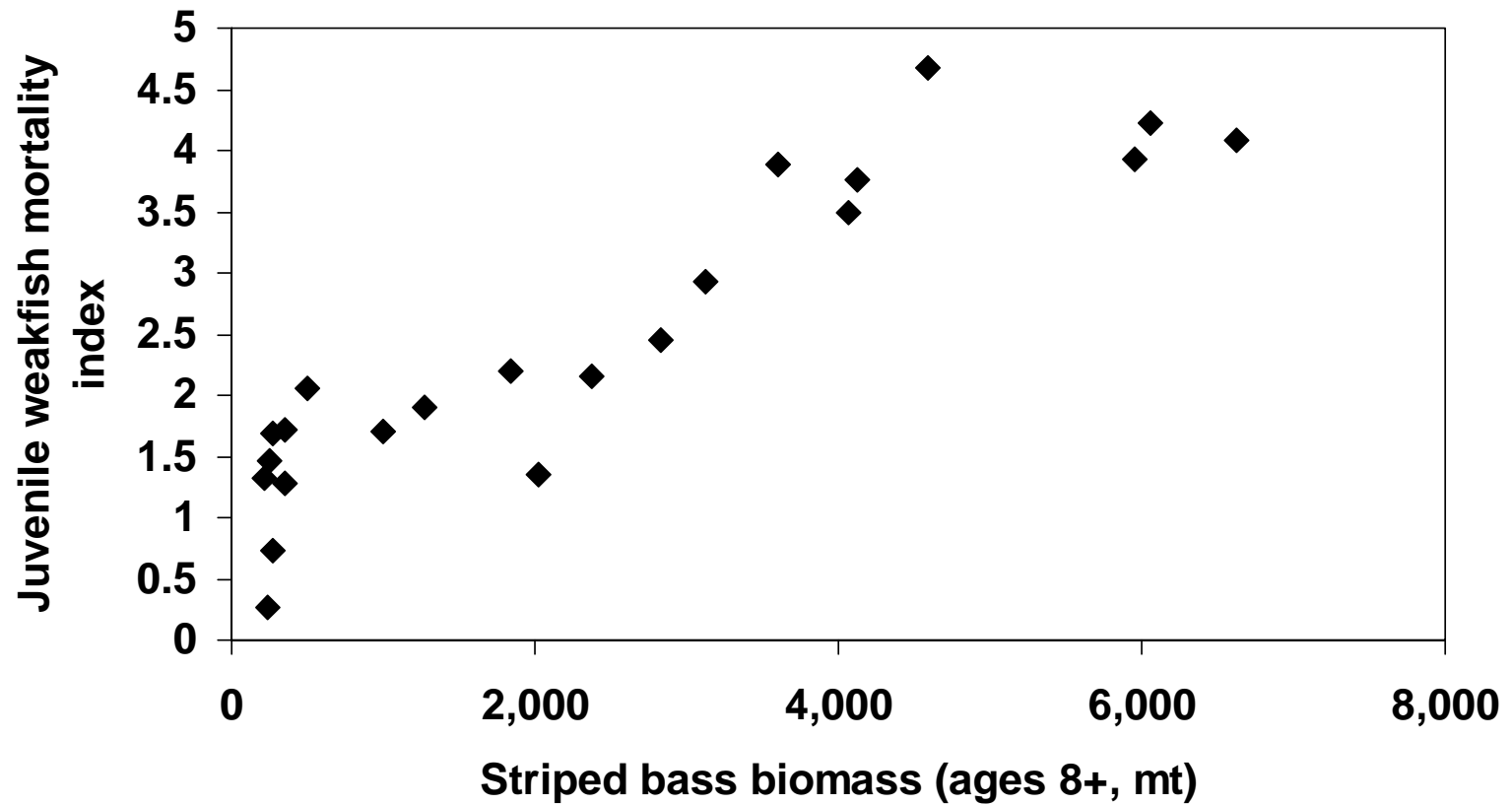


Figure 36. Gompertz striped bass-weakfish predator-prey model estimates of F , F_{msy} , M_p , and M_{vpa} . Model with iterative reweighting was used.

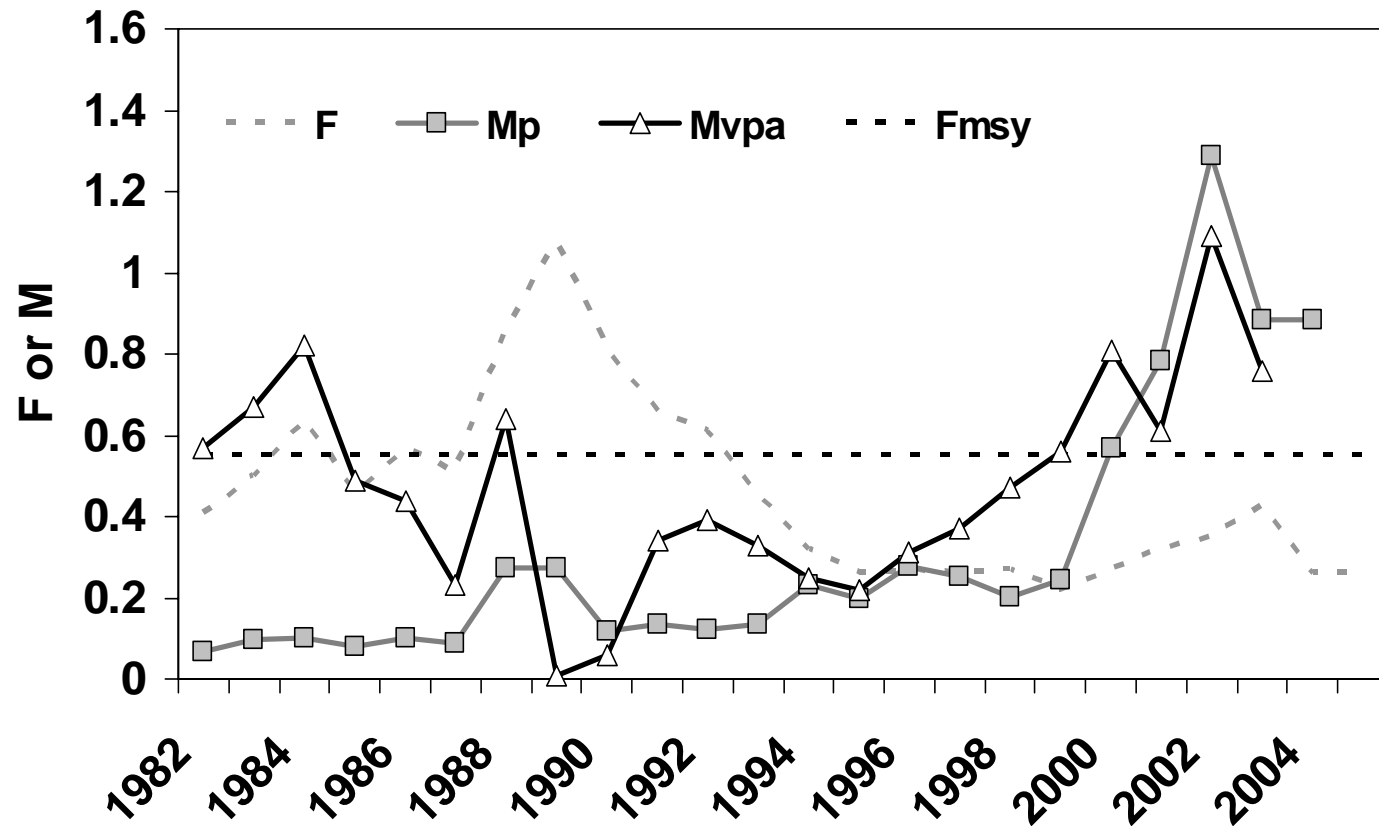


Figure 37. Gompertz surplus production model estimate of weakfish biomass with and without striped bass predation terms. IR indicates iterative reweighting was used in the fitting process.

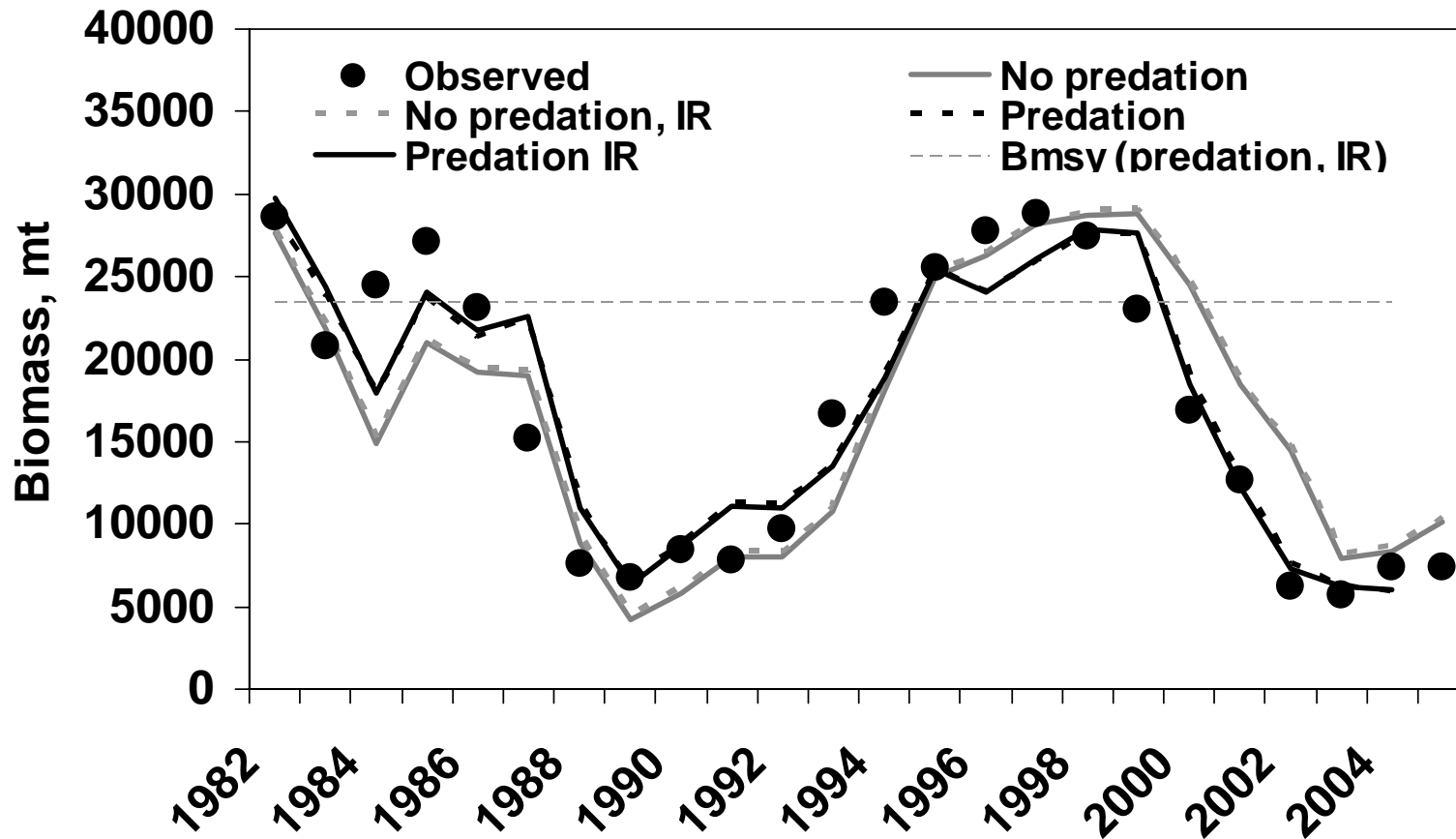


Figure 38. Observed (average biomass; Crecco 2005) and biomass predicted by external production model with varying M and F.

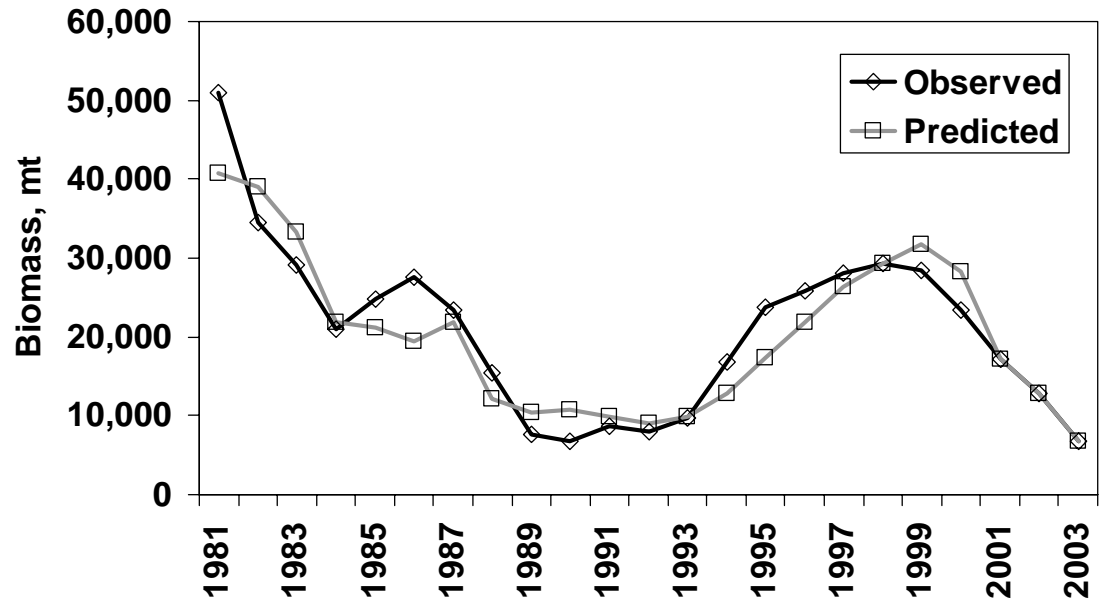


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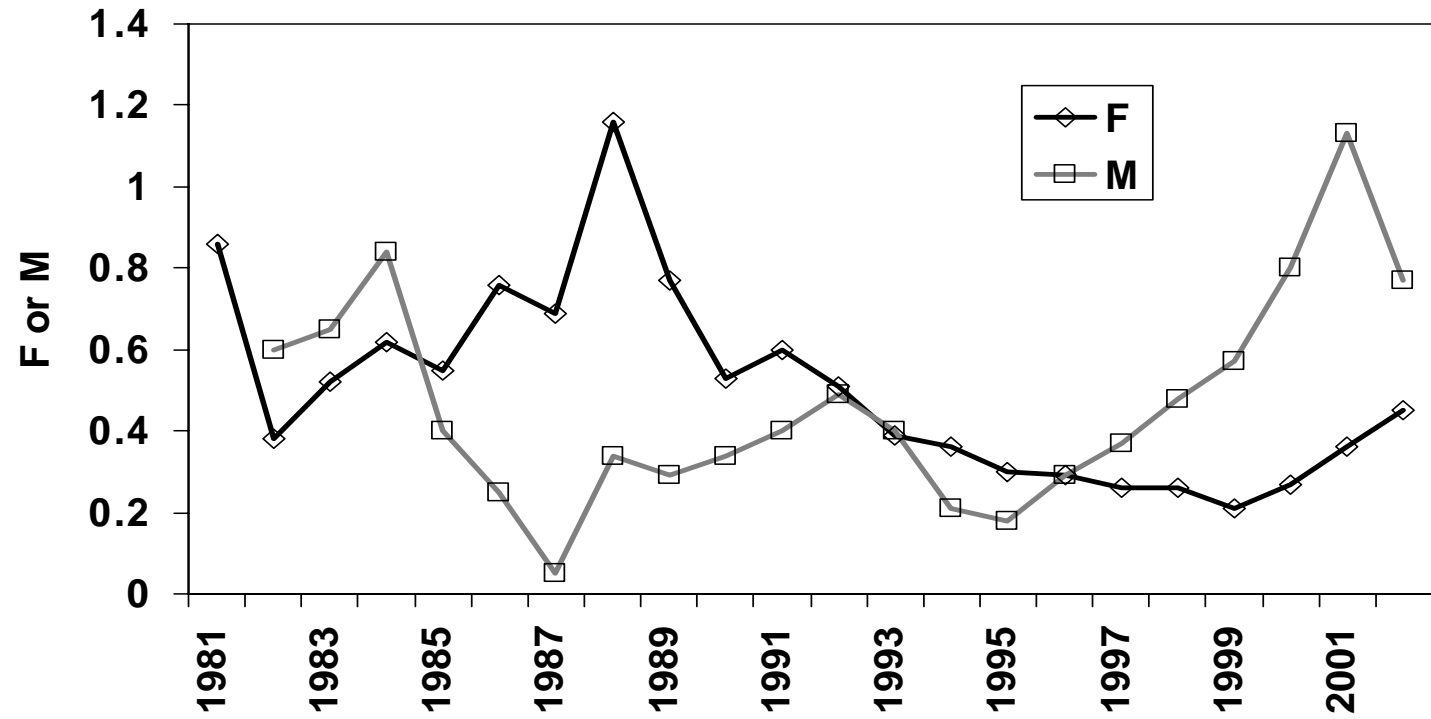


Figure 40. Estimated weakfish biomass during 1981-2003 based on different assumptions. All versions were “forced” to converge with 2001-2003 estimates of Crecco (2005). See Table 1 for model details. The 3 parameter model is the base model.

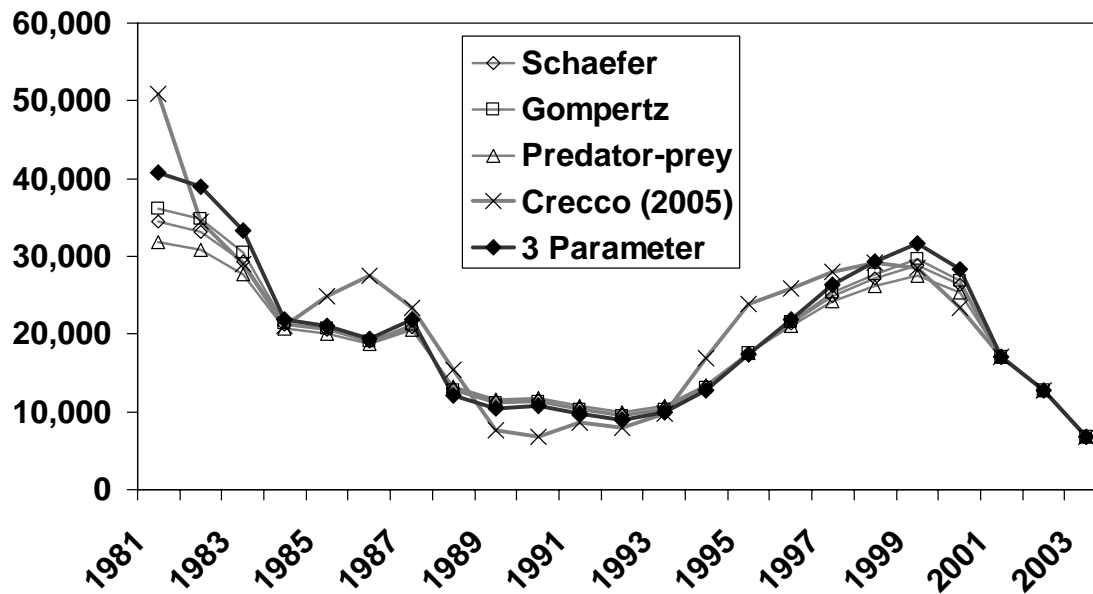


Figure 41. Estimates of F under different model conditions. See Table 1 for model details. All versions were “forced” to converge with 2001-2003 estimates of Crecco (2005). The 3 parameter model is the base model.

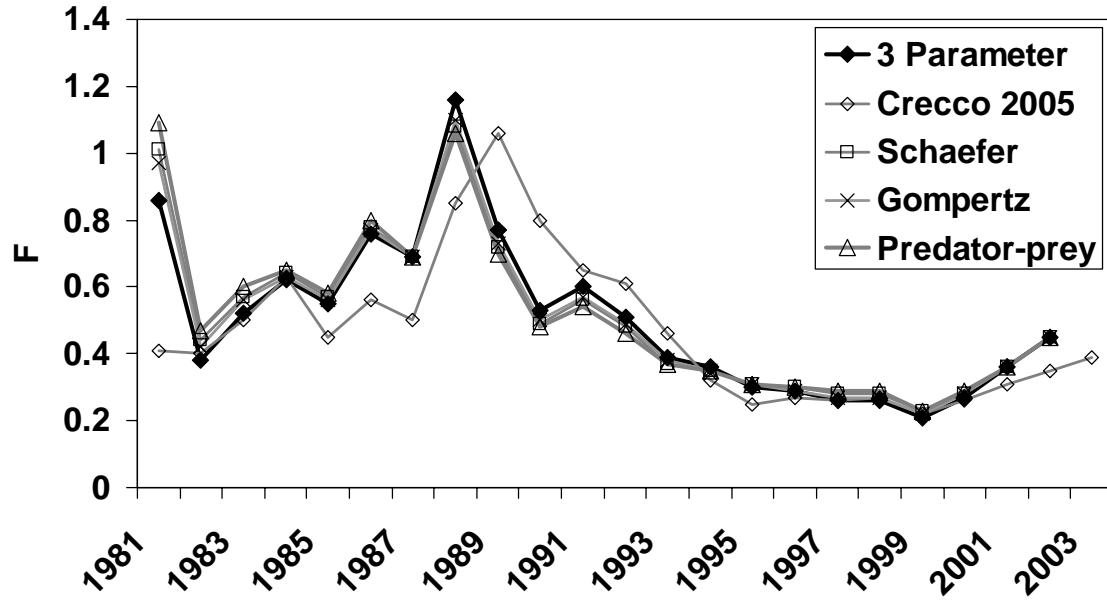


Figure 42. Estimates of M under different model conditions. See Table 1 for model details. All versions were “forced” to converge with 2001-2003 estimates of Crecco (2005). The 3 parameter model is the base model.

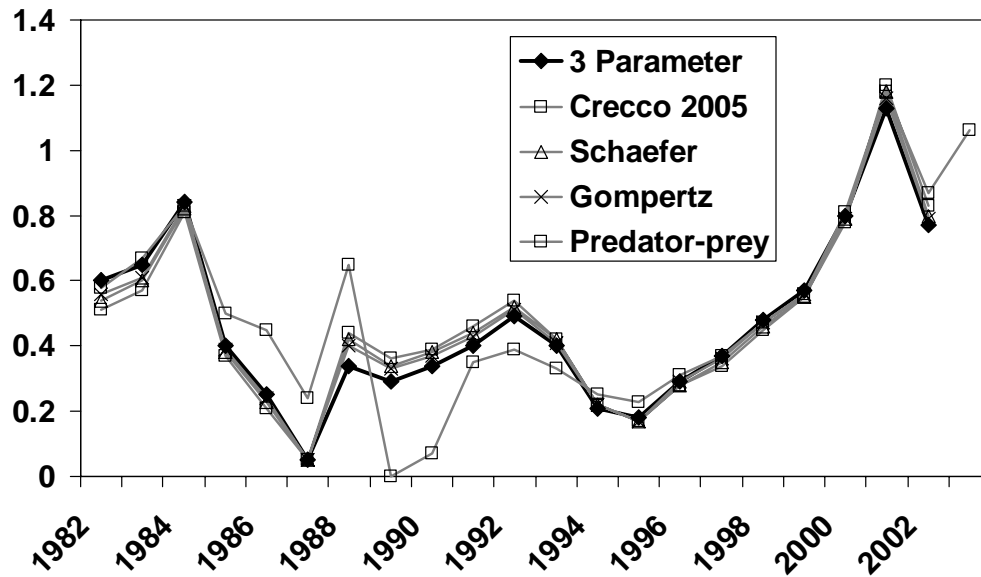


Figure 43. Projected weakfish biomass without a reduction in F (F = current), with F cut by 25%, 50%, and 75%, and a complete moratorium (discard mortality = 0) when M is kept at its current level.

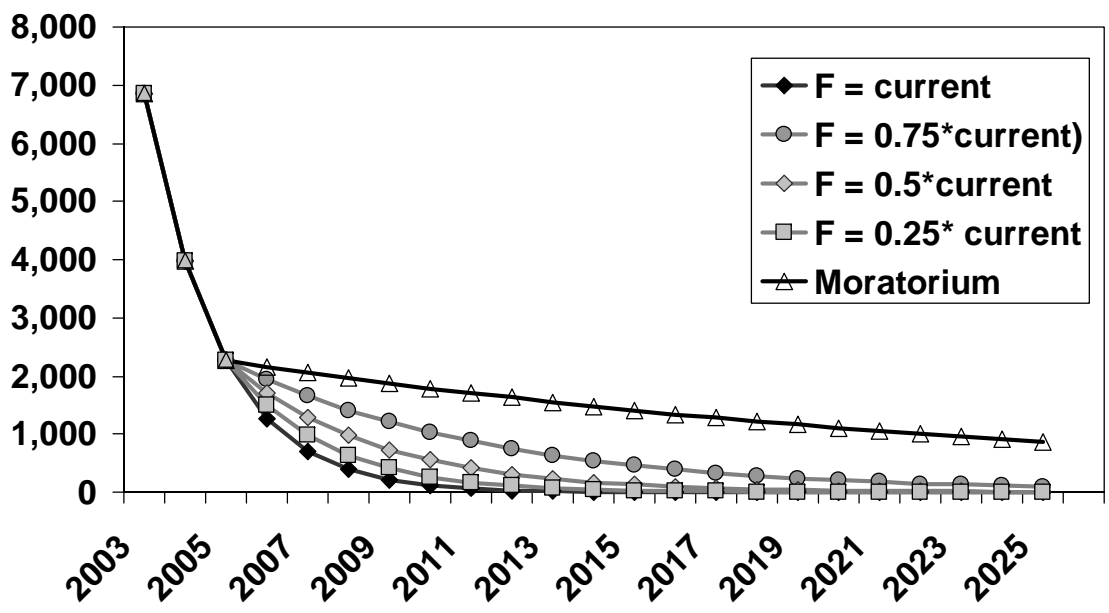


Figure 44. Projected weakfish biomass with reduction in Z equivalent to 1.5 * F or 2.0* F after 2005. See Table 1 for model details. Recovery is indicated at mean 1998-2000 biomass (29,785 mt).

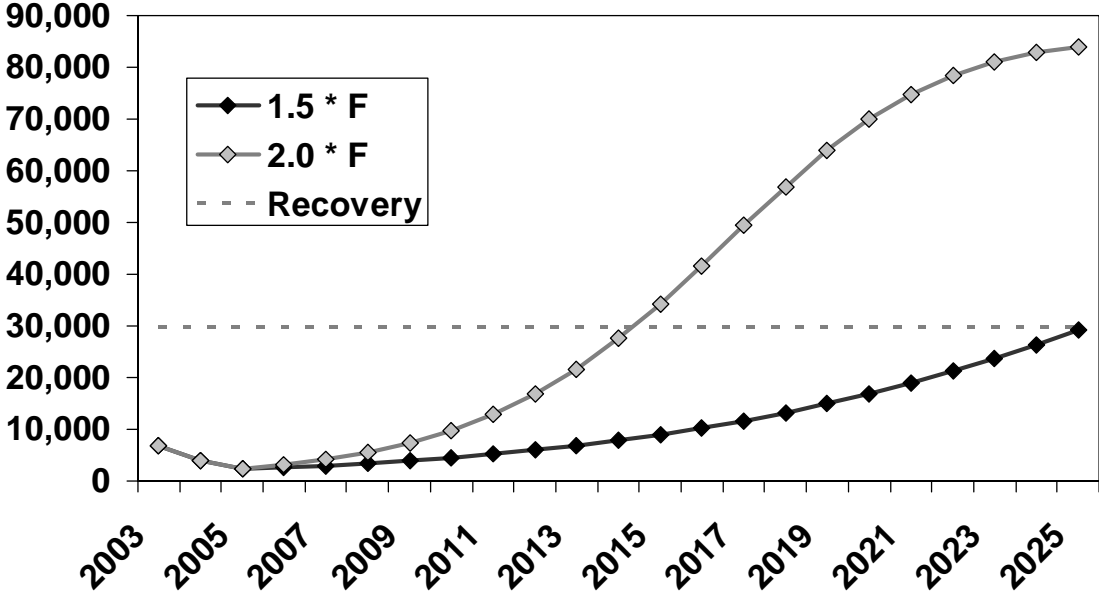


Figure 45. Projected biomass with no reduction in current F or cuts of 25%, 50%, or 75% when M depreciates from 0.95 in 2006 by a constant coefficient (0.8) to a base level ($M_b = 0.25$). Recovery is indicated at mean 1998-2000 biomass (29,785 mt).

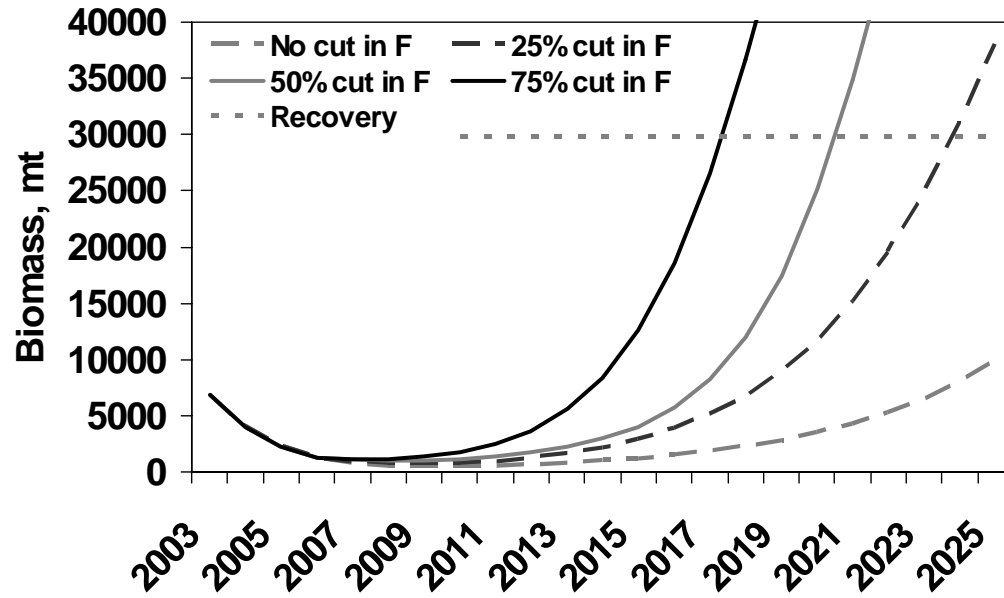
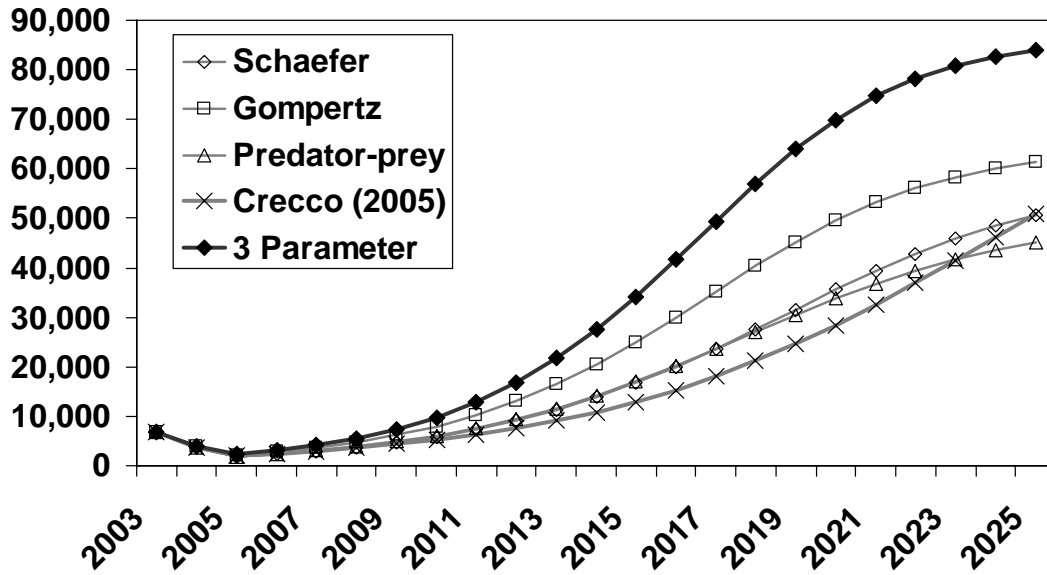


Figure 46. Projected weakfish biomass with reduction in Z equivalent to $2.0 * F$. See Table 1 for model details. Recovery occurs in 2016 for the Schaefer, 2014 for Gompertz, 2015 for predator-prey, 2020 for Crecco (2005), and 2014 for the base model.



WEAKFISH STOCK ASSESSMENT REPORT FOR PEER REVIEW (PART 2)

**An Ecological Assessment of Weakfish: Examination of Fishing and Trophic Effects on the
Recent Stock Decline**

February 2006

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Note: an external peer review panel did not endorse the recommendations within this assessment report. For more information see the Peer Review Report (Section B), and the Weakfish Technical Committee's responses to that report (Section C).

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INTRODUCTION

Weakfish fisheries were subject to increasingly restrictive management during the early and mid-1990s (ASMFC 2002). After imposition of Amendment 3 in 1996, weakfish were thought to be on the path to recovery. Recent recreational and commercial landings of weakfish along the Atlantic Coast approached all-time lows as population estimates derived from age structured (de Silva 2002; Kahn 2002a) or biomass dynamic models (Uphoff 2005a) reached all-time highs. This increasing dichotomy between weakfish stock assessment results and fishery performance has led the Weakfish Technical Committee to re-examine the basic survey data used and to consider external factors that lie outside of the purview of traditional assessments. One of the recommendations of the 40th Stock Assessment Review Committee (or SARC) examination of our preliminary assessment was for the Weakfish Stock Assessment Subcommittee (WSAS) to formulate multiple hypotheses about stock status and evaluate them with empirical data, an approach advocated by Sainsbury (1998) as part of a new paradigm for stock assessment in the 21st century.

A great preponderance of evidence indicates that weakfish abundance and surplus production have declined to low levels (Crecco 2005a or b; Weakfish Stock Assessment Report Part 1 or Part 1). The most reliable estimates of trends or values of F during 1981-2003 have indicated it has been modest since at least 1995 (Part 1). This has led to the formulation of a hypothesis that the most likely cause of declining weakfish fortune is increased natural mortality (Part 1).

We specifically addressed the hypothesis that M has been changing for weakfish and that it is the primary driver behind the recent crash. At this stage, an increase in M in recent years is a default postulate reflecting the difficulty in finding evidence of increasing F other than by adhering dogmatically to the notion that M is constant for weakfish at 0.25. We investigated hypotheses that weakfish biomass dynamics, growth, and survival were negatively affected by striped bass competition and predation exacerbated by depleted forage (hereafter, predation hypothesis). In particular, this is an alternative, nonfishing mortality hypothesis explaining an aborted weakfish recovery. We conducted an analysis on a portion of the food web (weakfish, its primary fish competitor, and three forage fish species) along the mid-Atlantic region (New York-North Carolina) that constitutes the “core area” for weakfish stock assessment (Northeast Fisheries Science Center 2000). We also addressed the potential for other candidate predator-competitors to contribute to the recent decline of weakfish.

We undertook statistical explorations of the inter-relationship of weakfish, striped bass (weakfish’s primary competitor and potential predator), four other potential predator-competitors (summer flounder, bluefish, spiny dogfish, and Atlantic croaker), two main forage species for piscivores along the mid-Atlantic (bay anchovy and Atlantic menhaden) and a forage species of demonstrated importance within Chesapeake Bay but possibly more limited importance elsewhere (spot; Mercer 1985; Taylor 1987; Hartman and Brandt 1995; Hartman 2003; Walter et al. 2003). Weakfish have been specifically identified as a minor food item in some, but not all, striped bass diet studies in the mid-Atlantic region (Manooch 1973; Austin and Walter 1998). In the mid-Atlantic region, bay anchovy represent the prey most consumed by these species as they initiate piscivory, but both predators switch predominately to similar, larger prey (clupeids, primarily Atlantic menhaden) within a year (Mercer 1985; Taylor 1987; Hartman and Brandt

1995). Juvenile menhaden and spot are most relevant to younger, smaller weakfish making the transition through piscivory because piscivorous fishes are size selective and gape limited, and typically select prey that are 20-30% of their length (Stein et al. 1988; Juanes 1994). Stakeholders, Weakfish Management Board members, and other biologists have suggested that Atlantic croaker, summer flounder, bluefish, and spiny dogfish may be potential predator-competitor candidates for explaining the decline in weakfish. Each of these species inhabits mid- and north Atlantic waters where concerns about the depletion of weakfish are greatest.

Weakfish and striped bass make an early switch to a fish diet (age 0 for weakfish and age 1 for striped bass; Mercer 1985; Hartman and Brandt 1995; Hartman 2003; Walter et al. 2003). This early switch to a fish diet indicates that both species are specialized piscivores (Persson and Bronmark 2002). Early switching requires high growth rate, which implies high densities of proper forage and safe foraging opportunities. If unfavorable conditions prevail, growth is reduced, and vulnerability to competitors and predators is increased. Population densities of specialists are generally a result of their success in a feeding stage where they experience intense competition (Persson and Bronmark 2002). Species undergoing ontogenetic diet shifts face a risk of delayed transitions among feeding stages if food resources are limited and competition is intense. Such ontogenetic bottlenecks are common in piscivorous fishes because competition may retard growth and prevent size advantage necessary to feed on the larger prey in the next stage. Individuals not reaching size advantage over prey may become stunted at size where consumption balances metabolic requirements (Bax 1998; Persson and Bronmark 2002).

Empirical models can be used to detect potentially important patterns of fish predation and may provide insight into processes that may be responsible for patterns described in the analysis (Whipple et al. 2000). We explored whether shifts in weakfish landings growth, natural mortality, and distribution (or their proxies) coincided with forage abundance and striped bass biomass using correlation and regression analyses, inferring competition for forage if test results were logical and statistically significant. This approach provided a bridge between the common assumption that predation and competition imposes a constant rate of mortality allowing the stock to be assessed in isolation from its environment (single species assessment; Bax 1998; Sainsbury 1998) and the more complex process-based multispecies models (Whipple et al. 2000).

Abundances of many exploited fish stocks vary widely and these variations may not be explained with single species population models (Spencer 1997). There is increasing evidence that concurrent changes multiple stocks occur at longer time-scales (Steele 1996). Often environmental factors are sought as underlying causes for changes in status, but sustained periods of high and low abundance are often exhibited by fish populations that are much more dramatic than shifts in climate or other environmental factors (Steele and Henderson 1984). These shifts can be regarded as jumps between alternative equilibrium states of ecological systems (Steele and Henderson 1984). The term “regime shift” has been used to suggest these changes are causally connected and can be linked to other changes in an ecosystem (Steele 1996). There is an assumption that environmental factors are the underlying mechanism with fishing as a contributing element (Steele 1996). Continuous changes in intrinsic population growth rate, carrying capacity, F , or rate of predation can lead to “flips” into periods of high or low abundance (Spencer and Collie 1995). A classical logistic population growth formula

combined with an S-shaped predation function reproduces these fluctuations when subjected to simulated directional environmental fluctuations or changes in fishing pressure (Steele and Henderson 1984; Collie and Spencer 1993; Spencer and Collie 1997a).

We applied the predator-prey model of Steele and Henderson (1984) to examine relative effects of fishing and striped bass predation and competition on recent (1981-2003) weakfish dynamics. We interpreted the “effect” of striped bass to potentially include collateral damage to weakfish such as starvation and cannibalism due to competition, as well as direct consumption by striped bass.

When applied generally, this predator-prey model has reproduced the types of rapid shifts in abundance exhibited by marine populations and it was useful in exploring the role of dogfish predation on Georges Bank haddock recovery and management (Spencer and Collie 1995; 1997b). This predator-prey model is essentially a Schaefer biomass dynamic model with a sigmoidal type III predation function added to estimate additional predation losses (Collie and Spencer 1993). In this analysis, it provided estimates of M and F that quantified the extent that striped bass, other candidate predator-competitors species, and fishing mortality could be influencing weakfish.

METHODS

Data Sources for Statistical Analyses for Forage and Striped Bass Influences with Indices and Landings as Stock Size Indicators

Commercial landings (1950-2003) of striped bass and weakfish were obtained from the National Marine Fisheries Service’s website (www.st.nmfs.gov/st1/). These data (Figure 1) represented the longest time-series readily available for both species that could be used to look for changes on decadal time-scales.

We used weakfish recreational catch rates (private/rental harvest and releases per trip in mid-Atlantic state waters based on all trips) as an indicator of stock biomass (Crecco 2006). Recreational catch per effort indices (WRI) were available for 1981-2003. Weakfish 25 cm FL and larger weakfish have constituted the bulk of weakfish measured by the MRFSS. Brust (2004) determine a MRFSS directed trip index of weakfish catch (harvest and releases) per trip when effort was limited to weakfish plus a suite of five species (summer flounder, Atlantic croaker, spot, bluefish, and black sea bass) considered to be commonly associated with weakfish. This index was very closely correlated (Pearson $r = 0.96$, $P < 0.0001$) with the more general WRI (Crecco 2005b), so it was not considered further in analyses.

Delaware and New Jersey trawl survey exploitable biomass indices (Z-transformed with one added for plots; Uphoff 2005a; Part 1) were used as additional indicators of weakfish stock biomass. Larger sizes of weakfish present in WRI were present in these surveys (Uphoff 2004). A third trawl survey, the Northeast Fisheries Science Center fall survey was not considered because exploitable-sized weakfish were not well represented (Uphoff 2004) and because of concerns about high inter-annual variability expressed by reviewers at the 40th SARC.

We examined whether catchability (q) of WRI, DE, and NJ indices was proportional to stock biomass. Each annual biomass index was divided by converged VPA biomass estimates (1982-1993 were considered converged) to estimate q . These estimates of q were plotted against converged biomass. Correlation analysis was used to describe associations of WRI, and DE and NJ trawl indices.

Weakfish weight at age data consisted of estimates for 1982-2003 from the VPA catch-at-age matrix. Trends in ages 1-5 were examined; older ages aggregated into an age 6+ group were not used because of possible bias from different age compositions. Scale ages had been converted to otolith ages for the early half of the matrix (NEFSC 1998).

We used the quality+ proportional stock density index (DE PSD Q+) derived from the Delaware trawl survey to summarize long-term 1966-2003 length-frequency data (Uphoff 2004; Part 1). The Delaware trawl survey time-series was not continuous (1966-1971, 1979-1984, and 1990-2003). The Delaware Quality+ PSD (DE PSD Q+) equaled the proportion of weakfish greater than or equal to 210 mm that were 340 mm or larger. The DE PSD Q+ index was significantly and positively associated with recreational fishing quality (trophy citations) over a broad area (DE, MD, and VA), commercial and recreational yield along the Atlantic Coast, WRI, and the proportion of recreational harvest outside of 3 miles (Uphoff 2004; Part 1).

The annual sum of ADAPT VPA biomass estimates for 2 year-old and older striped bass were used as competitor biomass during 1982-2003 (A. Sharov, MDDNR, personal communication; Uphoff 2003). The average of age 2+ striped bass biomass estimates for 1982-1986 was used as an estimate for 1981 (6,236 mt). Atlantic menhaden dominated diets of striped bass age 2 and greater in the early 1990s in upper Chesapeake Bay (Hartman and Brandt 1995) and were one of the dominant food items along the Atlantic coast (Walter et al. 2003; Nelson et al. 2003). Striped bass of this size would also be important predators of spot (Hartman and Brandt 1995) and would be capable of consuming weakfish.

Striped bass egg-presence absence in Chesapeake Bay spawning rivers (proportion of plankton tows with striped bass eggs; hereafter the large bass biomass index or LBI) was used as a long-term index of biomass (1955-2003; Uphoff 1997). Egg presence-absence scaled relative biomass between 0 and 1. This index tracks biomass of mature spawning females, generally bass 6 years-old or older, but was assumed to be a general indicator of stock biomass.

Indices of Atlantic menhaden, spot, and bay anchovy relative abundance were obtained from agencies in NC, VA, MD, DE, and NJ. Survey locations, gears, time-series duration, estimator of central tendency reported, data source, and which of the three forage species was surveyed were summarized in Table 1. In one case (Atlantic menhaden in Ablemarle Sound), two estimators of central tendency were used. All forage indices were z-transformed to place them on the same scale and split into two regions (NC-MD or DE and NJ) for graphic presentation.

Age 0 abundance estimates of Atlantic menhaden along the Atlantic Coast were available for 1955-2002 (ASMFC 2004a). A long-term juvenile menhaden time-series (1959-2003 proportion of seine hauls in Maryland's portion of Chesapeake Bay; Table 1; Uphoff 2003) was substituted for age 0 abundance estimates in some analyses. The menhaden juvenile index and juvenile abundance estimates are significantly related ($r^2 = 0.67$, $P < 0.001$), but the MD index indicated substantially higher relative abundance of juveniles in the 1970s and 1980s than the abundance estimates. Coastal abundance estimates of our other forage species did not exist, so we could not determine whether they might be applied broadly as we did with the MD juvenile menhaden index.

Statistical Analyses for Forage and Striped Bass Influences with Indices and Landings as Stock Size Indicators

Statistical analyses featured either weakfish commercial landings (1950-2003), DE PSD Q+ (1966-2003, discontinuous time-series), DE and NJ trawl survey indices, or WRI (1981-2003) as dependent variables. Estimates of absolute stock size based on single species stock assessment models were not used for statistical analysis because assumptions of constant natural mortality (VPA) and ecological stability (biomass dynamic models) were necessary to derive these estimates (Mertz and Meyers 1997; Spencer and Collie 1997b). Scatter plots of all bivariate analyses were examined for linear or nonlinear associations or relationships. Ecological systems are highly nonlinear and it was likely that the dynamics of interest might not be reasonably approximated as linear functions (Sainsbury 1998). Functional responses of predators are often nonlinear and time lags in the response of predators to prey and vice versa would be likely (Bax 1998). Natural log transformations were used to induce linearity in some cases.

Correlation analysis was used to empirically explore associations. Level of significance was adjusted for multiple comparisons by dividing $P = 0.10$ by the number of comparisons made for each species. Regressions and correlations were considered significant at $P \leq 0.10$. Residuals were examined for normality and serial trends.

Weakfish landings were plotted against striped bass landings, with each decade (starting in 1950) illustrated by a different symbol; landings from 2000-2003 were included with the 1990s. This differentiation of decades allowed examination of whether the 1970s and 1980s had a different or shifted relationship than the remaining decades. This plot allowed for development of an analytical strategy for examining the hypotheses that (1) striped bass and weakfish landings were negatively related and (2) a shift over decadal time scales was present in the landings data.

We used categorical regression to evaluate these hypotheses. The different decadal groups were coded as 0 (low period; 1960-1969 and 1990-2003) and 1 (high period; 1970-1989) and used with landings as independent variables in a multiple regression (Neter and Wasserman 1974). This test assumed that the slopes of the weakfish and striped bass landings were equal, but intercepts (decades) were different (Neter and Wasserman 1974).

Stepwise multiple regression analysis was used to test the relationship of ln-transformed weakfish landings to ln-transformed menhaden abundance or the Maryland juvenile menhaden

index and LBI. Partial correlation coefficients were calculated to judge the contribution of each variable to model fit (Ott 1997).

We used correlation analysis to explore associations of DE PSD Q+ and available forage fish (Atlantic menhaden, spot, and bay anchovy) relative abundance indices along the mid-Atlantic. The long-term association (1966-2003) of weakfish size quality and striped bass biomass was examined by correlating weakfish DE PSD Q+ and LBI. The short-term (1982-2003) association of weakfish DE PSD Q+ and age 2+ striped bass biomass was also explored with correlation analysis. We developed a multiple regression model of DE PSD Q+ against LBI and Atlantic Coast age 0 menhaden abundance to test the hypothesis that both this predator/competitor and important forage species influenced the size dynamics of weakfish during 1966-2002. In general, forage species were expected to have positive associations or relationships with weakfish DE PSD Q+ and striped bass were expected to have been negatively associated or related.

Weakfish abundance along the Atlantic Coast during 1981-2003, indexed by WRI or the standardized (z-transformation) DE and NJ trawl surveys, were plotted against age 2+ striped bass biomass. Statistical descriptions of these data were not attempted after viewing the complex relationships exhibited in the plots.

Time Varying Natural Mortality from VPA

Standard output from the ADAPT VPA includes biomass weighted fishing mortality rates of ages 1+ weakfish (F_{vpa}) from 1982 to 2003 (Part 1; Crecco 2006; 2005a). As a result, biomass weighted total mortality (Z_{vpa}) rates from 1982 to 2003 were easily derived by adding 0.25 to each of the aggregated F_{vpa} values from VPA run 20 (MRFSS recreational CPUE as the sole tuning index). The WSAS chose run 20 as the preferred VPA model run. The degree of retrospective bias in recent F_{vpa} and weakfish stock size estimates from run 20 was much less extensive than the bias for other model runs that were tuned by either trawl survey indices alone, or a combination of trawl survey and MRFSS indices (D. Kahn, DEDFW, personal communication). Note that rescaled F analysis (Crecco 2005a) estimated biomass weighted F of the same ages encompassed in Z_{vpa} , but independently from the VPA (Crecco 2006). As a result, a time-series of biomass weighted natural mortality rates (M_{vpa}) of ages 1+ weakfish were calculated for each year t from 1982 to 2003 by

$$(5) M_{vpat} = Z_{vpat} - F_{RFAt};$$

where F_{RFAt} is the rescaled F estimate and Z_{vpat} total mortality estimate based on VPA run 20.

Striped Bass-Weakfish Predator-Prey Modeling

WRI during 1981-2003 was used as an index of relative abundance for analyses with the Steele and Henderson (1984) model (Table 2). Landings during 1981-2003 included both directed harvest and discards by the recreational and commercial fisheries (Crecco 2005; Part 1). Biomass estimates for 2 year-old and older striped bass were used as predator-competitor biomass during 1981-2003 (Table 2).

We used a spreadsheet version of a Schaefer (logistic) biomass dynamic model formulated by Haddon (2001) and added the type III predation function to it (Steele and Henderson 1984; Collie and Spencer 1993; Spencer and Collie 1995). The predator-prey model used the following discrete time-step equation:

$$(6) B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1} - [(cP_{t-1}(B_{t-1})^2)/(A^2 + (B_{t-1})^2)] + \varepsilon;$$

where B_t was weakfish biomass in year t ; B_{t-1} = biomass the previous year, r = intrinsic rate of population increase; K = maximum population biomass; H_{t-1} = harvest (commercial and recreational, including discard estimates) in the previous year; maximum c is per capita consumption by predator (striped bass) biomass (P_{t-1}); A is weakfish biomass where predator satiation begins; and ε is measurement error (Hilborn and Walters 1992; Collie and Spencer 1993; Spencer and Collie 1995). The Haddon (2001) spreadsheet version of the logistic model was easy to adapt to this predator-prey formulation because it estimated weakfish biomass as a first step and then estimated catchability (q) as B_t/WRI_t (Haddon 2001). Estimating weakfish biomass first allowed striped bass biomass to be used directly and parameter A to be estimated directly rather than converting them to weakfish index equivalents. The mean of the estimates of q_t (q_{mean}) was used to predict WRI_t as $B_t * q_{mean}$. Biomass was estimated directly for 1981 (B_{1981}) and then the model estimated subsequent years (Haddon 2001).

Standard Schaefer model equilibrium biological reference points (BRPs), F_{msy} ($r/2$) and B_{msy} ($K/2$) were estimated (Haddon 2001). In providing guidance on precautionary approach BRPs to the National Marine Fisheries Service, Restrepo et al. (1998) recommended F_{msy} and B_{msy} as limit reference points. F_{msy} constituted the threshold over which overfishing was occurring and B_{msy} determined the stock size below which the stock would be considered overfished (Restrepo et al. 1998). BRPs adopted for weakfish management under Amendment 4 (ASMFC 2002) were $F_{30\%}$ as a target F (0.31), $F_{20\%}$ (0.5) as a limit BRP, and 20% SSB of unfished stock size (14,400 mt) as a threshold. These reference points were dependent on an age-structured stock assessment and could not be used in comparisons with production model-based BRPs, therefore F_{msy} and B_{msy} were substituted.

An observation error model was used that assumed all residual errors were in the index observations and the equation used to describe the time-series was deterministic and without error (Haddon 2001). A genetic algorithm super solver (Evolver, Palisade Corporation) was used to estimate predator-prey model parameters that minimized observation error (observed $\ln WRI_t$ - predicted $\ln WRI_t$)² (Haddon 2001). The spreadsheet version of the model combined with the genetic algorithm provided a great deal of flexibility for trying different model variations. Residuals were examined to see if they were normally distributed with a mean of zero and to see if serial trends were present.

Instantaneous annual fishing mortality rate in year t was estimated as:

$$(7) F_t = H_{t-1} / [(B_t + B_{t-1}) / 2] \text{ (Ricker 1975).}$$

An equivalent instantaneous natural mortality rate associated with striped bass predation and competition was estimated. The type III predator-prey term in equation 1:

$$(8) [(cP_{t-1}(B_{t-1})^2)/(A^2+(B_{t-1})^2)],$$

estimated loss of weakfish biomass, D_t , that was equivalent to catch H_t . Instantaneous annual natural mortality associated with striped bass was estimated as:

$$(9) M_{bt} = D_{t-1} / [(B_t + B_{t-1}) / 2].$$

Total instantaneous annual mortality of weakfish (Z_{bt}) due to due to fishing and striped bass predation equaled $F_t + M_{bt}$. This estimate did not account for other sources of natural mortality not associated with food web effects represented by striped bass.

Consumption of weakfish biomass per unit of striped bass biomass was estimated as

$$(10) D_t / P_t.$$

The functional response of striped bass to weakfish was displayed visually by plotting D_t / P_t against B_t .

Surplus production estimated from the logistic predator-prey analysis (SP_t) was estimated with two equations. The first estimate was from the standard equation.

$$(11) B_t - B_{t-1} + H_{t-1} \text{ (Prager 1994; Jacobson et al. 2002).}$$

The second estimate of SP_t explicitly accounted for losses due to striped bass predation and competition:

$$(12) B_t - B_{t-1} - D_{t-1} + H_{t-1}.$$

Surplus production estimated by both equations was plotted against F and M_b to explore the possible linkage of these two sources of mortality to changes in surplus production.

Estimates of F or Z from the predator-prey model were compared to F_{Rfa} (F from rescaled relative F analysis; Crecco 2005a; 2006), estimates of F from a Schaefer biomass dynamic model without a predator-prey term (based on WRI), or Z_{vpat} . Biomass estimates were compared among predator-prey, Schaefer (described below), and rescaled relative F models. Weakfish exploitable biomass estimated by the predator prey model was plotted with age 2+ striped bass biomass and combined weakfish and striped bass biomass.

Bootstrapping (Efron and Gong 1983) was used to describe variability of parameters estimated in the spreadsheet version of the logistic predator-prey model. The bootstrapping technique recommended by Prager (1994) that preserved the original order of the time-series was used. For each set of bootstrap trials, a synthetic data set was constructed by combining a random sample of the original residuals with the ordered predictions from the original fit; the model was then rerun with each set of synthetic observations (Efron and Gong 1983; Prager 1994; Haddon

2001). Percentile confidence intervals of model parameters (r , K , B_{1981} , c , and A) and the time-series of biomass, F , and M_b were estimated.

Several approaches were used to investigate sensitivity of predator-prey model parameters (r , K , B_{1981} , c , and A) and estimates of F , M_b , and biomass. All sensitivity analyses were deterministic; bootstrapping was not used. We were particularly attuned to changes that created a decline in biomass at the end of the time-series such as that exhibited in the rescaled relative F analysis (Crecco 2005). Results (SSQ, r^2 , residuals, estimates of parameters r and K , and F and biomass time-series) from the S-H striped bass-weakfish model were contrasted with a Schaefer biomass dynamic model using the WRI. Sensitivity of model parameters to the beginning or ending of the time-series was tested by removing up to three years of data and running the model. An additional sensitivity run used a fixed value for B_{1981} , 49,000 mt, predicted from a regression of DE PSD $Q+$ and average biomass estimated from rescaled F analysis.

Alternative Weakfish Predator-Competitors

Three approaches were used to evaluate the potential of Atlantic croaker, bluefish, summer flounder, and spiny dogfish to influence weakfish biomass dynamics. First, long-term trends in Atlantic Coast commercial landings were analyzed for trends to determine if negative associations with weakfish landing were suggested any of the candidate species. We assumed that landings would be largely reflective of stock biomass in this long-term analysis. Second, biomass estimates from single species assessments, generally within 1981-2003, were obtained and used as predator biomass (P_{t-1} in equation 6) in a species-specific predator-weakfish model used above to explore weakfish-striped bass biomass dynamics (see *Predator-Prey Modeling*). Analytical results were contrasted with those found for striped bass. Finally, literature on candidate species' diets was reviewed to see if predation on weakfish and competition for bay anchovy, Atlantic menhaden, and spot were possible.

Commercial landings (1950-2003) of the four candidates were obtained from the National Marine Fisheries Service's website (www.st.nmfs.gov/st1/; Figure 1). Some bias was likely introduced by increasingly restrictive management after the early 1980s for most of these species (Atlantic croaker would possibly be a lightly regulated exception). After inspection of trends, landings of dogfish were dropped from the analysis. Dogfish landings (not separated by species) were generally low throughout the time-series, but were punctuated by several year periods of high landings. These rapid changes did not appear in character with an animal with modest reproductive characteristics (slow growth, delayed maturity, long gestation, and small number of pups produced; NEFSC 2003). Correlation analysis was used to determine associations ($P \leq 0.10$) among candidate species' landings and weakfish landings. A significant, negative correlation with weakfish would suggest that a candidate species was a predator-competitor.

Biomass time-series from single species assessments were obtained for Atlantic croaker (ASMFC 2005), bluefish (NEFSC 2005a), summer flounder (NEFSC 2005b), and spiny dogfish (NEFSC 2003). Species, time-series and estimation technique were as follows: age 1+ Atlantic croaker SSB, 1981-2002, age structured surplus production model; age 1+ bluefish biomass, 1982-2003, VPA; age 1+ summer flounder biomass, 1982-2003, VPA; and total biomass of spiny dogfish, 1981-2003, expanded area swept by survey trawl (Lowess smoothed). Atlantic

croaker SSB was considered a reasonable proxy for biomass since 90% were considered mature at age 1 and all were mature afterward (ASMFC 2005).

We applied the logistic predator-prey model to examine relative effects of fishing and potential predation and competition from each of our four candidate species on recent (1981-2003) weakfish dynamics. The application of this model for weakfish and striped bass was described in detail above. Deterministic runs were made; bootstrapping and sensitivity analyses were not explored. Model fit (r^2 of observed and predicted indices), its ability to produce a decline in weakfish biomass after the late 1990s, and how closely its levels of weakfish biomass were to those estimated by Crecco (2005) were used to evaluate each candidate predator-competitor. Weakfish biomass estimates of Crecco (2005a) were termed “observed” quantities in these comparisons.

A “scorecard” was kept for each candidate species. Ten criteria were considered - four based on statistical or model results, and six on surveyed diet literature (Table 3). Each variable could be scored between 0 and 1. Maximum score summed to ten. Eight variables were dichotomous, while two were continuous. All six literature variables on diet presence were dichotomous: presence of weakfish, bay anchovy, Atlantic menhaden, spot, clupeids, and sciaenid. In order for a score of one to have been considered in the clupeid and sciaenid categories, diet summaries had to either consider these general categories only or species other than weakfish, menhaden or spot had to be identified. The regression coefficient between the biomass estimates made for 1981-2003 by rescaled F analysis and predator-prey analysis was one of the continuous scores and the biomass closeness ratio of mean 1981-2003 biomass made by rescaled F and predator-prey analyses was the other. This latter ratio was expressed to keep it less than one (either rescaled F biomass to predator-prey biomass or vice-versa) depending on species-specific bias (Table 3).

RESULTS

Statistical Analyses for Forage and Striped Bass Influences with Indices and Landings

Recreational catch per trip (WRI) was intermittently high during 1981-1988, fell to a low level through 1993, rose to a peak similar to previous highs by 1996 and then fell steadily to its nadir in 2003 (Figure 2). Delaware biomass indices were consistently low during 1981-1984. Both the NJ and DE biomass indices indicated a substantial rise during 1990-1996 and both fell afterwards, although the decline in DE was more substantial and consistent than NJ. All three indices had declined by 2003, but the magnitude of decline was inconsistent. WRI had approached its lowest level ever, DE was quite low but not at its lowest, and NJ was somewhat low but had been fluctuating annually between low and high for five years (Figure 2).

Estimated q of the WRI varied over the time-series ($CV = 36\%$), but was generally stable across the estimates of converged VPA biomass (Figure 3). The highest estimate was about three-times greater than the lowest. The two highest estimates of WRI q occurred earliest in the time-series (1981 and 1982) at a mid-level of biomass ($\approx 20,000$ mt). Scatter plots of DE and NJ biomass indices suggested inverse relationships of q and biomass at worst and random 10-fold fluctuations at best. Highest estimates of q in both state surveys occurred at the lowest biomass (Figure 3). Coefficients of variation of these two surveys' catchability estimates during 1982-

1993 were near 100%. We considered these trawl surveys qualitative indicators of changes in biomass.

Correlation of WRI and DE biomass index was not significant when 1981-1984 were included. All three sets of indices were positively correlated with each other during 1989-2003 (WRI and DE, Pearson $r = 0.90$; WRI and NJ, Pearson $r = 0.71$; DE and NJ, Pearson $r = 0.79$; $P \leq 0.003$ in all cases).

Mean weight-at-age was higher prior to 1993 and the difference between mean weights-at-age during 1981-1992 and 1993-2003 increased with age (Figure 4). Median average weight-at-age during 1982-1992 was 7% larger than during 1993-2003 for age 1, 22% larger for age 2, 34% for age 3, 80% for age 4, and 85% for age 5.

The DE PSD Q+ index indicated that size quality of weakfish in recent years was quite poor (Figure 5). A recovery in size quality in the 1990s faltered after 1998; the percentage of quality sized and larger weakfish fell from about 20% in 1998 to 1% in 2003. Weakfish DE PSD Q+ has the potential to be a good bit higher than the peak observed in 1998. Values in the early 1980s were twice as high (45%) as the peak observed in 1998 (Figure 5).

Biomass estimates of age 2+ striped bass were low (5,500-7,700 mt) during 1982-1986 (Figure 6). Biomass rose steadily from 12,000 mt to 83,000 mt during 1987-1998 and then rose slowly to 104,000 mt by 2003. *LBI* was near 0.80-0.90 during 1966-1978 with a hiatus near 0.65 during 1971-1974. After 1978, *LBI* steadily fell and reached a low of about 0.35 during 1983-1984. *LBI* rose rapidly after 1985 and reached 0.80 in 1992; *LBI* has fluctuated near 0.80 during 1993-2003. Age 2+ biomass was positively and significantly correlated with *LBI* (Pearson $r = 0.71$, $P < 0.0002$). Striped bass recovery indicated by *LBI* was much more rapid than VPA estimates of age 2+ biomass, but the relative scale of recovery was similar (Figure 6).

Striped bass biomass was negatively associated with weakfish size quality. Weakfish DE PSD Q+ was significantly and negatively correlated with ln-transformed striped bass age 2+ biomass during 1982-2003 (Pearson $r = -0.74$, $P = 0.0007$) and with *LBI* during 1966-2003 (Pearson $r = -0.67$, $P = 0.0002$).

Indices of relative abundance of Atlantic menhaden in MD, VA, and NC exhibited similar trends (Figure 7). Abundance was low during 1966-1970 and after the early 1990s. Abundance was slightly to well above the time-series average during 1974-1986. Relative abundance was slightly below average during 1987-1991 and then decreased to previous lows observed during 1966-1970 (Figure 7). Relative abundance of menhaden in Delaware Bay surveys (DE and NJ) was steady and low prior to 1993 and has fluctuated from slightly below average to well above the time-series average in the past decade (Figure 8). Coastal abundance estimates of age 0 Atlantic menhaden ranged from 128 to $850 \cdot 10^9$ (Figure 9). Age 0 abundance estimates were frequently in the lowest quartile ($< 280 \cdot 10^9$) during 1966-1972 and after 1994 and frequently in the upper quartile ($> 569 \cdot 10^9$) during 1975-1991. These age 0 population estimates exhibited a pattern similar to that observed in Chesapeake Bay and Ablemarle Sound surveys, but dissimilar to Delaware Bay.

Trends in spot relative abundance in MD and VA were similar to those of Atlantic menhaden; abundance was low during 1966-1971 and after 1991, and high in the interim (Figure 10). Relative abundance in MD and VA appeared to have been suppressed since the mid-1990s, while Pamlico Sound exhibited similar fluctuations as those observed in the early half of its time-series (Figure 10). Indices in Delaware Bay (DE and NJ) exhibited large fluctuations of high to low during the 1980s and early 1990s (Figure 11). These fluctuations began to dampen in the mid-1990s and indices have been below the time-series average since 1998 (Figure 11).

Bay anchovy indices in MD and VA fluctuated about their time-series means during 1966-1986 (Figure 12). An index was not available for NC. After 1994, MD Chesapeake Bay and MD coastal bay indices became asymptotic at a below average level, and VA indices were typically, but not always, below average (Figure 12). Bay anchovy indices in DE and NJ fluctuated above and below their means throughout their time-series (Figure 13).

Significant ($P < 0.10$) positive associations of DE PSD Q+ and forage relative abundance occurred consistently (but not among all surveys) in MD and VA for all three species (Table 4). Significant positive correlations (Pearson $r = 0.60-0.79$) with DE PSD Q+ were exhibited for Atlantic menhaden in MD, VA, and NC indices (3 of 4 comparisons) and coastwide abundance estimates. Spot were significantly and positively associated (Pearson $r = 0.54-0.73$) with PSD Q+ in DE Bay (1 of 3 surveys), and both Chesapeake Bay seine surveys. The VA Chesapeake Bay trawl survey index was positively, but not significantly correlated. Bay anchovy were significantly associated with PSD Q+ in MD ocean bay trawl and VA Chesapeake Bay seine surveys (Pearson $r = 0.61$ and 0.63 , respectively). Two other Chesapeake Bay surveys of bay anchovy were not significantly correlated with PSD Q+, nor were surveys from other geographic locations (Table 4).

The multiple regression model of weakfish DE PSD Q+ versus *LBI* and age 0 menhaden abundance during 1966-2003 was significant ($r^2 = 0.61$, $P < 0.0001$). Partial regression coefficients indicated age 0 menhaden abundance ($P < 0.0001$) explained 50% of variation in weakfish PSD Q+ and striped bass biomass accounted for 11% ($P = 0.02$). The model picked up the general trend (rise and fall) of the DE PSD Q+ time-series, but did not always match the details (Figure 14).

The scatter plot of WRI and age 2+ striped biomass (Figure 15) exhibited at least four stanzas that were sequential with both striped bass biomass and time. During 1982-1988, striped bass biomass was below 20,000 mt and WRI was high on average, but also highly variable. During 1989-1993, striped bass biomass rose from 20,000 to 40,000 mt while WRI was steady and low. Both striped bass biomass and WRI rose steadily and then held steady during 1994-2000; striped bass biomass reached about 85,000 mt while WRI returned to high levels exhibited during the early 1980s. In the final stanza (2001-2003), striped bass biomass continued to increase past 100,000 mt, but WRI steadily decreased and reached its nadir (Figure 15).

The relationship of weakfish exploitable biomass, indicated by DE and NJ trawl survey indices and striped bass age 2+ biomass (Figure 16) was similar to that exhibited by WRI. The earliest data, DE trawl survey during 1982-1984 indicated much lower relative biomass than WRI. Trends in remaining data (1989-2003) were very similar to WRI. Exploitable biomass indices

increased from about 0.1 to 3.0-4.0 concurrently with striped bass increasing from 20,000 to 70,000 mt. As striped bass biomass continued to rise to 100,000 mt in 2003, weakfish biomass indices fell to 0.5-1.0. A consistent indication of decline was not evident until 2003 (Figure 16).

Examination of long-term commercial landings of weakfish and striped bass (1950-2003) suggested that periods of high weakfish landings corresponded to low periods of striped bass landings and vice-versa. Decadal differences were evident when weakfish landings were plotted against striped bass landings (Figure 17). Weakfish landings were shifted upwards during 1970-1989 at the same amount of striped bass landings as during remaining years. A negative relationship of striped bass and weakfish landings that reflected two periods of underlying weakfish productivity was described by the multiple regression of weakfish landings against period and striped bass landings (Figure 17). This regression was significant at $P < 0.001$ and explained 76% of variation in weakfish landings. The relationship was described by the equation $W = -0.47*B + 5720*C + 4115$; where W = weakfish landings; B = striped bass landings, and C = decadal category (low = 0 = 1950-1969 and 1990-2003; high = 1 = 1970-1980). The net visual effect of this regression was to depict predicted weakfish landings as abrupt decadal blocks, with slight declines in the low periods and a slight increase during the high period (Figure 18). All coefficients and the intercept were significantly different from zero. Residuals appeared normally distributed with a mean near zero (mean = $3 \cdot 10^{-13}$). When categories were ignored, relationship of striped bass to weakfish commercial landings was negative and significant, but weak ($r^2 = 0.12$); a pronounced serial trend corresponding to the high and low periods was evident in the residuals.

Multiple regression analysis suggested a negative, nonlinear relationship between weakfish landings striped bass biomass (indexed by LBI) and a positive, nonlinear relationship with \log_e -transformed juvenile Atlantic menhaden abundance (MJ) during 1955-2002. There was an overall significant negative correlation between LBI and \log_e MJ (Pearson $r = -0.49$, $P < 0.0004$), but this correlation was not “close” enough to warrant elimination of either variable in the multiple regression (Ricker 1975). The model was significant ($R^2 = 0.52$, $P < 0.0001$) and was described by the equation:

$$\log_e W = -1.75*LBI + 0.53 * \log_e MJ - 4.41;$$

where W = weakfish landings, LBI = striped bass egg presence-absence index, and MJ = menhaden juvenile abundance. Partial r^2 indicated that \log_e MJ explained 41% of variation ($P < 0.0001$) and LBI explained 11% ($P < 0.0023$). The model moderately overestimated weakfish landings during 1955-1960 and usually underestimated them (sometimes badly) during 1972-1982 (Figure 19). Landings were well predicted after 1985. Residuals were nearly always negative during 1955-1970, always positive during 1971-1982, and then alternated with 3-4 year periods of negative and then positive residuals. The mean of the residuals was very close to zero.

Stepwise regression of the Maryland juvenile menhaden index (MI) and LBI indicated both terms were significant ($R^2 = 0.76$, $P < 0.0001$; Figure 19); MI accounted for the vast majority of the fit (partial $r^2 = 0.73$, $P < 0.0001$) while striped bass accounted for a minor, but significant, fraction (partial $r^2 = 0.03$, $P < 0.04$). As above, there was an overall significant negative correlation between LBI and MI (Pearson $r = -0.52$, $P < 0.0002$), but this correlation was not

“close” enough to warrant elimination of either variable in the multiple regression (Ricker 1975). Residuals appeared normally distributed with a mean near 0; a serial trend was not evident. This regression model reasonably reproduced the time blocks of weakfish landings featured in the categorical regression expressed above; both predictions were significantly correlated (Pearson $r = 0.81$, $P < 0.001$).

Time Varying Natural Mortality from VPA

Although natural mortality of age 1+ weakfish was assumed to be constant in previous assessments (NEFSC 1998; 2000), the trend in annual biomass weighted natural mortality (M_{vpa}) from 1982 to 2003 does not support this assumption (Figure 20). M_{vpa} estimates, though variable, changed systematically. A quadratic function (inverse parabola) fit the time trend of M_{vpa} (all terms significant at $P < 0.00013$; $R^2 = 0.63$, $P < 0.0001$; Figure 20) and depicted M_{vpa} falling from 0.75 in 1982 to near 0.25 during 1990-1993, then rising to 0.95 by 2003. The 1982-2003 average M_{vpa} (mean $M_{vpa} = 0.47$, $SE = 0.06$) was significantly higher (t-statistic = 3.67, $P < 0.01$) than the assumed constant M of 0.25 (Crecco 2006).

Predator-Prey Modeling

The Steele and Henderson (1984) predator-prey model fit the data well and explained 78% of the variation in the weakfish index (Figure 21). The model generally overestimated the index somewhat during 1989-1994 and underestimated it during 1996-2000 (Figure 21). Residuals had a mean of -0.052 and were skewed towards negative values. Model parameters were estimated as $r = 1.17$, $K = 49,080$ mt, $c = 0.076$, and $A = 1,959$ mt. Maximum sustained yield reference points were $F_{msy} = 0.58$ and $B_{msy} = 24,540$ mt.

The initial biomass estimate (1981) equaled 65,274 mt and exceeded the estimate of K (Figure 22). B_{msy} was never exceeded after 1981. Biomass was about 19,000 mt in 1982 and fell to 7,900 mt by 1990; biomass rose to 16,000 mt in 1997 and then began to decline again, reaching 8,000 mt in 2003 and 5,700 mt in 2004 (Figure 22). Biomass estimates in 2003 and 2004 represented 40% and 32%, respectively, of B_{msy} (or 20% and 16% of unfished biomass) under this set of ecosystem conditions.

Instantaneous annual fishing mortality rates rose from 0.5 to 1.3 during 1981-1988, fell to 0.4 by 1993, and then remained steady below F_{msy} at about 0.4-0.5 during the following decade (Figure 22). Instantaneous annual natural mortality rates associated with striped bass were very low during 1981-1986 ($M_b < 0.04$). Estimates of M_b rose steadily after 1986, equaled F by 1999-2000 ($M_b \approx 0.5$), exceeded F by 50% by 2002 ($M_b \approx 0.7$), and easily doubled F by 2003 ($M_b \approx 1.1$; Figure 22). Changes in biomass in the of the time-series largely reflected F until the late 1990s, when a decline largely reflected M_b . The large drop in biomass between 1981 and 1982 was not explained by either F or M_b .

Surplus production estimates (SP_t), calculated with or without accounting for striped bass competition, were near 12,000 mt during 1982-1986 (Figure 23). Estimates from both equations then began to fall through 1990. When striped bass losses were not included in estimation, SP_t

rose after 1990, peaked at about 7,400 mt in 1995, and then underwent a steady decline to about 400 mt by 2003. When striped bass associated losses were included in the estimation of SP_t it declined slowly to 3,500 mt in 1992, remained steady there for three years, rapidly declined to near zero by 1998, and became increasingly negative afterwards (Figure 23).

Plots of SP_t (estimated by either equation) against F (Figure 24) or M_b (Figure 25) indicated a strong logical and negative influence of M_b , but not F ; 1981 was omitted from the plots and appeared to represent an outlier. The plot of F versus SP_t estimated by either equation indicated an illogical, positive association.

A low number of bootstrap trials were performed ($N=50$) after determining that precision of all estimated parameters was extremely high. Coefficients of variation (CV) of r , K , s , B_{1981} , c , and A were all below 1%. CV's of annual biomass, F , and M_b were below 1% for 1981-2000 and would rise to 4-5% at the end of the time-series (biomass estimates were made through 2004; F and M_b were estimated through 2003).

A Schaefer production model could not reproduce the recent decline in WRI without addition of a striped bass predation term (Figure 26). Sum of squares fell from 3.49 (Schaefer) to 2.62 (Steele and Henderson) with the addition of striped bass as a predator-competitor, primarily due to improved fit at the end of the time-series (2003 squared residual = 1.33 in the Schaefer model and 0.10 in the Steele and Henderson model). Improvement in overall fit was modest ($r^2 = 0.72$ without and 0.78 with predation term). Mean of the Schaefer model residuals was -0.075. Schaefer model residuals exhibited much of the same serial pattern described above for the Steele and Henderson model, but became sharply negative at the end of the time-series. Intrinsic rate of increase was estimated to have been much lower, 0.47, in the Schaefer model and K was considerably higher, 71,000 mt.

The time-series modeled had great bearing on the nature of the predator-prey relationship (c and A), but little influence on production parameters (r and K ; Table 5) and equilibrium MSY reference points derived from them. Parameters r and K varied by no more than $\pm 10\%$ from the original fit (1981-2003) with the seven treatments. Predator-prey parameter c varied only 10% from the original estimate when years were removed from the end of the time-series, but changes were substantial (105-267%) when years were removed from the beginning. Parameter A was extremely sensitive to the time-series used. The original series produced the lowest estimate and other years' estimates were 370-1300% greater (Table 5).

Only models fit to the 1981-2003 or 1982-2003 time-series could reproduce the recent decline of the observed WRI out of the seven sensitivity treatments (up to three years removed from beginning or end of time-series and 1981 fixed at 49,000 mt; Figure 27). The remaining treatments indicated that biomass leveled-off or was rising slightly at the end of the time-series (Figure 27). The 2003 estimate of WRI influenced trajectory the most at the end of the time-series. Only the 1981-2003 and 1982-2003 time-series will be described further because of the inability of the other treatments to create a decline at the end of the time-series.

If the time-series began in 1981, modeled striped bass functional response was highly depensatory in the Steele-Henderson model; weakfish per bass was nearly constant ($\approx 0.71-0.76$)

across the entire range of estimated biomass (6,700-66,000 mt; Figure 28). The asymptotic value of weakfish biomass where striped bass demand for weakfish was satiated ($A=1,959$ mt) was greatly exceeded throughout the time-series. If 1982 was used to initiate the time-series, weakfish consumption per striped bass was much more density-dependent (changing nearly linearly from about 0.04 to 0.11 as weakfish biomass rose from 7,900 to 19,000 mt (Figure 28). Weakfish biomass at satiation ($A=11,387$) was reached within the time-series.

Trends in estimates of F were generally comparable among the predator-prey, rescaled relative F , and the Schaefer biomass dynamic models except during the more recent years (Figure 29). Fishing mortality was highest and generally increasing during the 1980s; F fell substantially during the early 1990s, and remained steady and low during the remainder of the 1990s. Trajectories of F after 1999 diverged among models; F remained steady in the predator-prey model, increased with the rescaled F analysis (but remained modest), and fell to a very low level with the Schaefer biomass dynamic model. Estimates of F from both time-series (1981-2003 and 1982-2003) used with the predator prey model were similar. Recent estimates of F from the Schaefer model exhibited the same pathology (trending in recent years to near zero; Figure 29) as estimates made with biomass dynamic models based on trawl survey indices (Uphoff 2005a).

Trends of estimates of M_b were similar (increasing) through 2001 for Steele and Henderson models applied to the two time-series (Figure 30). Estimates of M_b were below 0.05 through 1985 and then steadily increased to approximately 0.70 by 2001. Estimates diverged after 2001, M_b based on the 1982-2003 time-series stabilized at about 0.7 while those based on 1981-2003 continued to increase to 1.1 by 2003 (Figure 30).

Trends in Z were similar between predator-prey models and the 1982-2003 ADAPT VPA (Figure 31); estimates of 1982-2003 Z_b based on the 1981-2003 and 1982-2003 time-series were significantly correlated with Z_{vpa} (Pearson $r = 0.57$, $P < 0.005$ and Pearson $r = 0.53$, $P < 0.012$, respectively). Total mortality rates estimated by the predator-prey model using the 1981-2003 time-series were high during 1982-1992 ($Z_b = 0.8-1.5$), dropped significantly by 1995 ($Z_{bt} = 0.5-0.7$), began a steady climb and returned to previous high levels by 2000, and continued to increase through 2003. Estimates of Z_b based on the 1982-2003 time-series were similar to those from the other two approaches over most of the time-series, but were much lower in 2003. The 1982-2003 predator-prey model estimates of Z_b plateaued at about 1.1-1.3 during 1998-2002 and then fell to just below 1.0 in 2003; these levels of Z_b were only sustained previously during 1986-1989. Estimates of Z_{vpa} have been between 0.9 and 1.4 since 2001; this sustained level was not evident in the ADAPT time-series since 1981-1992 (Figure 31).

Excluding 1981, weakfish biomass estimated by the predator-prey models did not exceed 20,000 mt (Figure 32). Biomass estimated from rescaled F analysis peaked at about 30,000 mt, while the Schaefer model estimates were as high as 60,000 mt. Predator-prey and rescaled F estimates of biomass were very similar in trend (declines and increases were concurrent) and magnitude of low biomass, but differed in magnitude of how high biomass could have been (Figure 32).

Striped bass-weakfish model estimates of weakfish biomass began to recover after 1990 and rose from 7,000 mt to 18,000 mt by 1997 (regardless of predator-prey model time-series). Increases in striped bass biomass beyond 70,000-80,000 mt were offset by decreases in weakfish biomass.

After 1997, weakfish biomass decline to about 8,000 mt. When weakfish biomass was added to striped bass biomass, an asymptotic combined biomass level of 100,000 -110,000 mt was suggested. This may represent the combined carrying capacity for these two species under the limited forage conditions in existence since the mid-1990s.

Alternative Predator-Competitors

Only striped bass commercial landings were significantly and negatively correlated (Pearson $r = -0.35$, $P = 0.0095$) with weakfish landings during 1950-2003 (Table 6). Bluefish and summer flounder landings during 1950-2003 had highly significant positive correlations (Pearson $r = 0.73$ and $P < 0.0001$ in both cases) with weakfish landings. Atlantic croaker harvests were marginally correlated with those of weakfish (Pearson $r = 0.24$, $P = 0.08$).

Trends in candidate species biomass during 1981-2003 varied (Figure 33). Striped bass exhibited a steady increase throughout, starting at about 5,000 mt and ending at about 100,000 mt. Summer flounder exhibited a general saddle shape trend – declining from 30,000 mt to 11,000 mt during 1981-1989, steadying at this low level until 1994, and then steadily increased to 50,000 mt. Bluefish declined steadily from 150,000 mt until reaching 50,000 mt in the late 1990s; the stock stabilized for several years, and then increased to 90,000 mt over the last two years. Spiny dogfish steadily increased from 300,000 mt in the early 1980s to 560,000 mt by the early 1990s. Then they steadily decreased to about 400,000 mt (Figure 33).

Atlantic croaker biomass in the predator-prey model resulted in weakfish biomass estimates of about the same magnitude as that observed (biomass closeness ratio = 0.93), but did not produce a decline in recent years (Table 6, Figure 34). The croaker biomass time-series ended in 2002. Absence of an Atlantic croaker biomass estimate for 2003 is of concern because it was a crucial data point for producing a decline in biomass in predator-prey modeling of striped bass and weakfish. Estimates of r and K (1.00 and 53,823 mt, respectively) were similar to those for the striped bass-weakfish model. Parameter A (weakfish biomass where satiation occurs) estimated for the Atlantic croaker-weakfish model was greater than carrying capacity, casting doubt on the applicability of this model or at least the type III functional response to these data.

With bluefish as a predator, the Steele and Henderson model indicated that weakfish biomass was fluctuating about a mean level (Figure 35). Estimated biomass was about 10-times the observed level (closeness ratio = 0.08). There was little support for bluefish as an influential predator-competitor competitor based on the logistic Steele and Henderson model ($r^2 = 0.01$; Table 6).

Summer flounder biomass produced a similar trend in estimated biomass ($r^2 = 0.62$), but the estimates were 20% to 150% higher than the observed values (biomass closeness ratio = 0.56; Table 6, Figure 36). A recent decline in weakfish biomass was produced with summer flounder biomass as a predator. The intrinsic rate was about half that of the striped bass-weakfish model, while K was about 6-times higher (Table 6). Per capita consumption estimates ($c = 1.2$ mt tons consumed per mt of flounder) indicated flounder would need to consume over 15-times the amount weakfish as estimated for striped bass to produce a similar trend in biomass (Table 6).

Using spiny dogfish biomass estimates in the predator-prey model resulted in weakfish biomass estimates similar to those observed (closeness ratio = 0.89), but did not result in a good fit ($r^2 = 0.29$) or a decline in recent years (Table 6; Figure 37). Parameters r and K were similar to those in the striped bass-weakfish model; A was in excess of K , indicating this model was not entirely appropriate for examining the dogfish-weakfish relationship (Table 6).

Striped bass-weakfish landings analysis and predator-prey model results have already been described extensively and will not be described further here. Comparison of striped bass landings regression and predator-prey model results with other candidate species is presented in Table 6. Striped bass consumed weakfish directly and all five of the forage categories were well represented in diet studies reviewed (Table 7).

The ecological role of Atlantic croaker as a potential competitor or predator on weakfish is poorly understood but croaker trophic status is below that of weakfish (Hartmann et al. 1995; Miller et al. 2004). Diet information for Atlantic croaker were limited and confined to recent years in Chesapeake Bay (Miller et al. 2004). Weakfish were not listed as a diet item for Atlantic croaker. The only diet item on the scorecard that was consumed by Atlantic croaker was bay anchovy (Table 7).

Bluefish diets were summarized for both Chesapeake Bay (Hartman and Brandt 1995) and the North Atlantic and Mid-Atlantic coasts (Buckel et al. 1999a; 1999b). Weakfish were present at a low level in bluefish diets along the North and Mid-Atlantic (Buckel et al. 1999a; Table 7). Bay anchovy, Atlantic menhaden, spot, and clupeids were important items in at least one of the three studies (Table 8). Sciaenids other than spot and weakfish were present, but were a minor component of bluefish diets.

Summer flounder diets for the North and Mid-Atlantic were summarized in Bowman et al. (2000), Link and Almeida (2000), and Link et al. (2002a). Weakfish were present at a low level in summer flounder diets in the inshore region North of Cape Hatteras (Bowman et al. 2000; Table 8). Bay anchovy was the only candidate forage species specifically identified, while anchovies in general comprised less than 5% of summer flounder diets. Neither Atlantic menhaden nor spot were specifically mentioned, but clupeids generally comprised less than 10% of diet by weight. Sciaenids were a minor component of summer flounder diets (Table 8).

Rountree (1999), Link and Almeida (2000), and Link et al. (2002b) described spiny dogfish diets along the North and Mid-Atlantic region. Rountree (1999) reported that spiny dogfish prey on weakfish (Table 8). Only general categories containing our candidate species were identified: anchovies, clupeids, and sciaenids (Table 8). Clupeids were a major component of diet by weight, ranking second among all groups summarized in these two studies. Anchovies and sciaenids were minor components.

The weakfish predation-competition scorecard indicated that striped bass was by far the most likely predator-competitor candidate (9.5 out of 10 possible), followed by summer flounder (6.2), bluefish (6.1; 6.0 points due to diet studies), spiny dogfish (5.2), and Atlantic croaker (2.65; Table 8).

Summer flounder was the only candidate species beside striped bass to both have weakfish in its diet and reproduce the recent fall of weakfish biomass in predator-prey modeling. Based on these factors, summer flounder biomass was combined with striped bass in a predator-prey model run (hereafter, bass - flounder model).

The striped bass – flounder model fit the three predator-prey model criteria better than both species alone. The trend in biomass estimated from rescaled F analysis was faithfully reproduced (Figure 38). Fit of predicted biomass to observed improved ($r^2 = 0.83$). The ratio of predicted to observed biomass was close to 1 (0.95) and the decline in recent biomass was reproduced in the model (Figure 38). Model production parameters r and K (1.22 and 53,302 mt) and predator-prey functional response parameters ($c = 0.096$ and $A = 2,976$ mt) were much closer to those of striped bass than summer flounder (see Table 7). The shading of these parameters towards striped bass far exceeded the weighting expected based on biomass alone; average summer flounder biomass during 1982-2003 was 59% of striped bass biomass.

DISCUSSION

At this time, weakfish have changed from a harvestable commodity to a supporting role as forage for the striped bass recovery (and perhaps other species recoveries). Weakfish in the Mid-Atlantic region are currently in a period of low productivity that has low forage abundance underlying it. Since the early to mid-1990s, regional (NC-NJ) indices of three important forage species (Atlantic menhaden, spot, and bay anchovy) have dropped to low levels. Under these conditions, their population dynamics are dominated by striped bass, perhaps with a negative contribution by summer flounder. Changes in weakfish size, abundance, and biomass dynamics were consistent with trends in forage relative abundance, striped bass and summer flounder biomass, and M over a broad geographic area and time-span. These changes supported the hypothesis that the aborted weakfish recovery largely reflected increased natural mortality due to striped bass competition and predation precipitated by depleted forage.

Equilibrium MSY limit reference points, consistent with the precautionary approach and estimated from the S-H model (Steele and Henderson striped bass-weakfish predator-prey model), indicated that F had been below the F_{msy} limit for nearly a decade, but biomass had fallen well below the B_{msy} limit in the last several years. Biomass was below B_{msy} for most of the time-series, regardless of S-H model employed and F was below F_{msy} since the early 1990s. Under equilibrium conditions, this should not be possible; fishing mortality rates in this range should have been associated with stock expansion or stability. S-H models indicated these equilibrium-based biological reference points have limited utility in this situation where M is rising rapidly. Biological reference points can be sensitive with species subject to predation and cannibalism and may represent moving targets (Collie and Gislason 2001). In the case of the striped bass-weakfish model, by 2003 rising M_b ate away at the amount of F that represented a level of “safe” fishing; $r - M_b$ fell from 0.66 in 2001 to 0.08 in 2003. Even though F fell from 0.50 during 2000-2002 to 0.39 in 2003, the amount of residual productivity ($r - M_b - F$) went from positive to negative (0.16 to -0.31) during 2000-2003.

Natural mortality of age 1+ weakfish was originally assumed to be constant across ages and years ($M = 0.25$; NEFSC 1998; 2000). We found that the trend in annual natural mortality

estimates during 1982-2003 did not support the constant M assumption. Striped bass-weakfish predator-prey models corroborated the rise in M over the past decade. In future assessments here and elsewhere, the assumption of constant M for ages 1+ fish needs to be critically examined. The weakfish ADAPT VPA assumes constant natural mortality (Part 1). Misspecification of M in a VPA creates particularly large biases for lightly fished stocks and, when combined with changes in F , leads to a time-varying bias (Mertz and Meyers 1997).

For medium-term advice, it may suffice to consider one-way interactions and categorize each species as predator or prey (Collie and Gislason 2001). This simplification, which we employed in our S-H models, is convenient because it allows predators to be added to a single species assessment (Collie and Gislason 2001). However, Walters and Martell (2004) warned that this approach does not recognize how trophic ontogeny may cause more complex dynamics.

Biomass dynamic models implicitly assume that underlying ecological conditions remain unchanged (variations in stock size reflect random variation and fishing; Spencer and Collie 1997). Without an additional predator-competitor term, a Schaefer model based on WRI indicated that biomass was continuing to increase and F was falling to a very low level. This same unrealistic trend was produced by trawl index-based biomass dynamic models (Uphoff 2005a). After modeling under an ecological stability assumption, we were left with concluding that recreational and commercial fishers were no longer interested in harvesting a high population of weakfish or that the population was so high that they stunted at smaller sizes than are of interest (Uphoff 2005a). Decreasing forage and increasing competition among predators make it appear unlikely that ecological conditions were stable over the time span modeled.

Term c (per capita consumption by striped bass) in the predator-prey model represented combined effects of direct striped bass predation, but it could be that cannibalism and starvation induced by intense competition with striped bass were being described as well. Predators' responses to prey can be divided into a functional response (number of prey consumed per unit area per unit time by an individual) and a numerical response (numbers change with prey density; Yodzis 1994; Bax 1998). Numerical response to prey density may have a direct development component (growth, birth, and death), but interaction with other predators may also influence this response (Bax 1998). Individual predators may interfere with one another in a number of ways that might affect population growth (Yodzis 1994). Among these interactions, the ones most relevant to weakfish and striped bass would be (1) they may hinder one another's feeding activities, leading to starvation; (2) they may eat one another (including cannibalism); (3) they may indulge in territorial behavior; and/or (4) they may emigrate (Yodzis 1994).

Striped bass did not appear tightly linked to weakfish as prey and loss of weakfish at high recent striped bass population size represented collateral damage. Versions of the 1981-2003 or 1982-2003 logistic predator-prey models, where c was a constant multiplied by striped bass biomass to estimate bass-related weakfish losses ($D_t = c \cdot P_t$) instead of a type III function, produced similar estimates of r and c , but more variable estimates of K , as logistic models employing a sigmoid predator-prey function. Variations in c were particularly important; both trials with linear predator-prey functions produced estimates of c (0.073 and 0.069 for 1981 and 1982 as initial years, respectively) that were similar to the 1981-2003 sigmoid function model estimate ($c = 0.076$). In the case of the 1982-2003 time-series logistic model, a constant c improved fit over

the sigmoid version (linear model SSQ = 2.49, sigmoid SSQ = 2.71). Only the sigmoid model of the 1982-2003 time-series supported the concept of striped bass predation dependent on the biomass of weakfish. The three other versions of the logistic predator-prey model indicated striped bass related mortality was depensatory. Trends in biomass, F , and M_b from constant c were quite similar to those presented for sigmoid functions.

In this analysis, there was a good chance that striped bass represented predators that were not tightly linked to weakfish, a secondary prey. Depensatory mortality may exist when a fish population is faced with a predator that spends much of its time feeding on one prey species, but also has secondary prey (Hilborn and Walters 1992). Predator abundance may be independent of the secondary prey and if the predators are efficient at finding and capturing secondary prey, then the number eaten will be more or less constant. As primary prey abundance declines, the mortality rate caused by the predators on the secondary prey increases (Hilborn and Walters 1992). As indicated by Spencer and Collie (1997), fish stocks that are subject to moderate to severe depensatory predatory mortality often undergo a sudden and persistent drop in surplus production over time, even when fishing mortality rates have remained low for several years.

There is little experience to judge the suitability of parameter A (weakfish biomass where predator satiation begins) in the type III predation function. The ratio of A/K in the 1981-2003 logistic weakfish - striped bass model (0.04) was comparable to estimates generated by Spencer and Collie (1995; 1997) for Georges Bank haddock and spiny dogfish (0.05 – 0.07). Dogfish were considered a depensatory predator of haddock (Spencer and Collie 1995; 1997b). The ratio of A/K in the 1982-2003 weakfish logistic predator-prey model (with Type III function) was considerably higher (0.23) and would indicate a more density-dependent relationship.

The “effect” of these different predator-prey relationships estimated from the two time-series (1981-2003 or 1982-2003) may be best illustrated by exploratory projections. In these projections, striped bass biomass was held constant at its 2003 level through 2022. A constant exploitation rate of 0.1 was set after 2004 (a rate higher than this crashes the 1981-2003 based model). With $c = 0.76$ and $A = 1,956$ mt (estimates from 1981-2003), the population stabilized by 2010 at very low biomass (555 mt). With $c = 0.155$ and $A = 11,387$ mt (estimates from 1982-2003 time-series), the population increases to about 23,300 mt by 2022.

Unlike harvest, we are unlikely to ever have a natural mortality “body count” to unequivocally estimate M . Results from regression and production models demonstrate striped bass-weakfish linkage and they are consistent with studies of striped bass predation in the mid-Atlantic. Weakfish have been specifically identified as a food item in some, but not all, striped bass diet studies in the mid-Atlantic region. They have appeared in studies conducted in lower Chesapeake Bay (Austin and Walter 1998) and Ablemarle Sound (Manooch 1973). They were infrequent diet items, comprising about 1-8% of diets by number and 2-3% on a weight or volume basis (Manooch 1973; Austin and Walter 1998). Weakfish were present in striped bass larger 400 mm in lower Chesapeake Bay in fall and winter (Austin and Walter 1998), while they were present in striped bass larger than 300 mm in Ablemarle Sound during “cooler” months (Manooch 1973). During winter 2004-2005, weakfish comprised 5% of striped bass diets by weight along North Carolina’s coast (A. Overton, East Carolina University, personal communication).

Significant ($P < 0.10$) positive associations of weakfish size quality (1966-2003 DE PSD Q+ index) and forage abundance occurred consistently (but not among all surveys) in MD and VA for all three species and sporadically in NJ and NC. Regressions of striped bass biomass and juvenile menhaden abundance against DE PSD Q+ indicated a significant positive influence of forage and negative influence of striped bass on weakfish size quality. The DE PSD Q+ index represented the longest time-series of fishery-independent data on weakfish population dynamics. Length-frequency data, indexed by DE PSD Q+, contains much of the information contained in age-frequency data and even cursory examination of a length-frequency can give useful insights into population dynamics of a stock (Powell 1979; Hoenig et al. 1987). A population's length-frequency distribution results from its recent history of recruitment and mortality, integrated with growth (Barry and Tegner 1989). Two major parameters affecting DE PSD Q+, growth and mortality, have likely been influenced directly by competition for limited forage and predation.

The DE PSD Q+ index could have been influenced by trends in recruitment; size quality could have diminished because of recent recruitment of strong year-classes. Recruitment indices for weakfish in Crecco (2006) have exhibited wide regional variation, but the overall trend of the grand mean has been an increase during the 1990s and stabilization at a high level afterward (Figure 39). However, juvenile mortality has risen (Crecco 2006), while landings and other relative abundance or biomass indices have declined to well below average during 2002-2004. If the DE PSD Q+ index was to decline due to strong recruitment alone, then these abundance indicators and landings should be increasing.

The PSD Q+ index was chosen over growth equations because a long-time series of otolith-based growth estimates was not available and of because uncertainties in growth estimation. Otoliths were not primary aging media until the early to mid-1990s (NEFSC 1998). Fitting von Bertalanffy growth curves to these data has not been straightforward. Kahn (2002b) estimated von Bertalanffy curves by assigning constant asymptotic maximum length (L_{∞}) of 765 mm because of implausible estimates of this parameter; this same step was necessary to update 2001-2003 growth equations. Estimates of remaining von Bertalanffy parameters could have been confounded because they are interrelated (Jensen 1997). If assignment of constant L_{∞} represents a real phenomenon, a 17% loss in maximum size (from 919 mm) has occurred since the otolith-based estimate made by Lowerre-Barbieri et al. (1995) for the early 1990s.

Kahn (2002b) described a decreasing trend in weakfish Brody growth coefficients in von Bertalanffy growth equations estimated for weakfish during 1990-1999. The Brody growth coefficient dropped by over half after 1990-1992 (Kahn 2002b). Kahn (2002b) also noted lower mean weights-at-age estimated for 2000 than those estimated for 1989-1992 by Lowerre-Barbieri et al. (1995). Weakfish weight-at-age has exhibited large drops since 1992 and the magnitude of the decline increased with increasing age. These changes corresponded to decreased forage relative abundance and increased striped bass biomass.

Significant decreases in weight-at-age of weakfish implied a corresponding rise in natural mortality. Natural mortality rates of fishes increase as a nonlinear function of decreasing body mass under equilibrium conditions when mortality is primarily derived by predation (Peterson and Wroblewski 1984; Boudreau and Dickie 1989).

Median weights-at-age of weakfish of 3 year-old weakfish during 1993-2003 were 75% of those during 1981-1992, while those of ages 4 and 5 during 1993-2003 were 55%. These changes in weight suggest that starvation-related mortality was possible for weakfish and indicated that older weakfish would be most likely to be subjected to starvation. Mortality due to starvation is a size-dependent process (Persson and Bronmark 2002). In a simulation of ontogenetic predator-prey dynamics of pikeperch, Persson and Bronmark (2002) assumed starvation occurred if individual weight dropped below 70% of weight predicted by a length-weight equation. In experiments with fasting juvenile rainbow trout, mean wet weight of starved fish that died were 77% of weight at the beginning of the experiment (Simpkins et al. 2003). A weight-based prediction of starvation is likely to be pretty coarse and may not correspond to the point where protein is broken down to meet metabolic demands (J. Jacobs, NOAA, personal communication).

Declines in weakfish mean weight exhibited in the weight-at-age matrix reflected a bottleneck of appropriate sized prey (juvenile menhaden and spot). If weakfish have stunted at these diminished weights because of low forage availability, this stunted size should reflect the size of forage needed. In experiments, Hartman (2000) found that peak profitability for striped bass occurred at a ratio of prey weight to predator weight of 0.12. When this ratio is applied to weakfish median weight-at-age estimates for 2-5 year-olds since 1993, it indicates that prey weighing an average of 29-109 grams (prey size increasing with weakfish age) would have been most profitable. These weights corresponded to menhaden (growth parameters in ASMFC 2004a) or spot (Fishbase parameters; www.fishbase.org) less than one year-old.

Atlantic menhaden is a buffer species that absorbs predatory pressure from other species when they are sufficiently abundant (ASMFC 2004b). Striped bass actively select for Atlantic menhaden, but will feed on other species when menhaden are not sufficiently abundant (Overton 2003; ASMFC 2004b; Ruderhausen et al. 2005) and they appear capable of limiting their prey populations along the Atlantic coast (Hartman 2003). As menhaden have become less abundant, striped bass searching is likely to lead to increasing encounters with vulnerable-sized weakfish (Walters and Martel 2004) reflected in the close correspondence of weakfish mortality and striped bass biomass.

Attack success of striped bass on Atlantic menhaden has likely been falling (Uphoff 2003). As the ratios of biomass of age 2+ striped bass to: biomass of age 0-2 menhaden (or biomass of bass big enough to eat menhaden: biomass of menhaden small enough to be eaten; based on ASMFC assessments; Uphoff 2003) along the Atlantic Coast fell to asymptotic lows in the mid-1990s, natural mortality of weakfish rose (as indicated by M_{vpa} , M_b , and the estimated trend of juvenile mortality; Crecco 2006; Figure 40). Potential attack success of striped bass on menhaden would be indexed by this ratio because a predator's functional response (number of prey consumed per unit area, per unit time by an individual predator) is both a function of attack success and prey handling time (Yodzis 1994). Handling time varies little for a given predator (Yodzis 1994). Predator feeding efficiency will be inversely related to prey density as consumption rate declines and search effort increases (Ney 1990).

Hartman and Brandt (1995) found striped bass in Chesapeake Bay increased their use of the pelagic food web (primarily menhaden) as they aged, while weakfish increased use of benthic

resources (spot). Stable isotope analysis of striped bass scales collected during 1982-1997 from Chesapeake Bay indicated striped bass increased their use of the benthic food web as menhaden abundance decreased (Pruell et al. 2003). This shift would place them in more direct competition with weakfish and increase the chance striped bass would be occupying the same habitat as weakfish while feeding.

The percent of weakfish and striped bass diets represented by menhaden in Chesapeake Bay dropped substantially between the early 1990s and 2000s (Hartman and Brandt 1995; Overton 2003; Bonzek et al. 2004). The menhaden fishery, centered in Chesapeake Bay, harvests larger menhaden than those consumed by most weakfish or smaller striped bass (ASMFC 2004b), but direct competition with large striped bass is possible. Generalizations of functional response suggested that the fishery would outcompete striped bass at low menhaden densities (Uphoff 2003). A prey-size cascade could have been precipitated by competition between large striped bass and the fishery. Large striped bass are now relying more on small pelagic prey (bay anchovy and juvenile clupeids) needed by small striped bass and weakfish, while diets of these smaller piscivores have shifted to benthic invertebrates (see below; Griffin and Margraf 2003; Overton 2003). Slowing of weakfish growth due to a diet bottleneck, in turn, leaves them more vulnerable to striped bass predation, as do increased searching times for juvenile weakfish attempting the transition to piscivory. Increased searching time for small weakfish to obtain anchovy increases their exposure to larger striped bass that are now competing for the same resource. There is a horrific linkage between getting food and being food (Walters and Martell 2004).

Spot harvests in the mid-Atlantic are considerably less than those of the menhaden fishery, but their impact in relation to the population is unknown because a stock assessment is lacking. Bycatch in shrimp fisheries would kill spot at a size needed by weakfish and striped bass; however, shrimp fisheries do not operate in the geographic range (NJ-VA) where depletion of spot was suggested by correlation analysis. Pamlico Sound (NC) spot indices did not suggest depletion where substantial shrimp bycatch might have been expected. Striped bass generally do not range into Pamlico Sound (Boreman and Lewis 1987).

Weakfish biomass estimated for 1981 with the S-H model exceeded the estimate of K by nearly 30%, was over three-times the next highest biomass, and generally appeared as an outlier in most time-series and bivariate plots with other parameters. However, it could also have indicated rapid initiation of a regime shift in weakfish production. This year (1981) was the last year of an extraordinary three-year period of high landings, maximum DE PSD Q+, and trophy catches (Uphoff 2005a). A speculative Schaefer biomass dynamic model of an extended time-series (1975-2003; Uphoff 2005a) indicated that by 1981 weakfish biomass was in rapid decline from a period of very high biomass. This extended time-series analysis was similar in trend over years in common to the Schaefer biomass dynamic model included in comparisons with the predator-prey model. Estimates of stock biomass trends were in general agreement among models featured in this analysis until the late 1990s, when the Schaefer biomass dynamic model estimates began to diverge from S-H model estimates.

A negative relationship of Atlantic Coast striped bass and weakfish landings that reflected one high (1970s - 1980s) and two low periods (1950s -1960s, and 1990s -current) of underlying

weakfish productivity was indicated by categorical and multiple regression analyses of 1950-2003 weakfish and striped bass commercial landings, LBI, and indicators of Atlantic menhaden juvenile abundance. Earlier landings appear to confirm the negative relationship between these species' landings and rapid changes in weakfish status. Commercial records for these two species were intermittent from 1887-1950, but were reasonably frequent by the late 1920s (Joseph 1972; Boreman and Austin 1985). Catches of weakfish were generally high during the late 1920s through the mid-1940s while striped bass landings were low, but rising. Weakfish landings collapsed within four years in the late 1940s, while striped bass landings continued a sustained increase that lasted into the early 1970s (Joseph 1972; Boreman and Austin 1985). Correlations with landings of other four other candidate predator-competitors did not suggest a significant negative interaction.

Generally, autocorrelated variability that mimics long-term environmental patterns has been the forcing function that underlies regime shifts in simulations of populations with the S-H model (Steele and Henderson 1984; Collie and Spencer 1993; Spencer and Collie 1995; Spencer and Collie 1997a). Movement from high to low equilibrium states can be induced by increased fishing and depensatory mortality (Collie and Spencer 1993). Additional stress, usually attributed to fishing, reduces the level of environmental perturbation necessary to flip a system into a low equilibrium state (Collie and Spencer 1993) and causes flips to occur with higher frequency (Steele 1996). In the recent case of weakfish, it is likely depensatory mortality associated with striped bass under low forage conditions is the force leading to low productivity. The regime shift concept implies that different regimes have inherent stability, so that significant forcing is required to flip the system into alternative states (Steele 1996). The simple models for regime shifts have only two states, but nature may more complex and returning to a desired state may not be guaranteed by reduced fishing on weakfish, increased fishing on striped bass, or betting on beneficial environmental change (Steele 1996).

It is not possible to look at all predation processes in a fishery ecosystem (Bax 1998; Sainsbury 1998) and we have concentrated on a limited representation of the food web. Atlantic menhaden, spot, and bay anchovy represented 65-99% of total prey consumed by striped bass and weakfish during their estuarine residency in Chesapeake Bay during the early 1990s (Hartman and Brandt 1995). Atlantic menhaden and bay anchovy accounted for nearly 65% of weakfish diets by weight in Delaware Bay during 1985 (Taylor 1987). Clupeoid fishes (herrings and anchovies) have dominated the diets of striped bass above age 1 (Walter et al. 2003). Summer and fall diets of striped bass were dominated by these three forage species in Chesapeake, Delaware, and North Carolina regions (Walter et al. 2003). Young-of-year weakfish feed on mysid shrimp and bay anchovy, while older weakfish feed on clupeid species that are abundant in a given area (Mercer 1985; Hartman and Brandt 1995). Striped bass evolved as a schooling species to take advantage of the great energy resource of clupeids along the Atlantic Coast (Stevens 1979). Diet studies strongly suggest that these two piscivores compete strongly for the same forage resources and changes in population characteristics such as growth, mortality, and abundance could be linked under conditions of scarce forage.

Weakfish diet studies have been infrequently conducted in the recent past. Age-specific weakfish diets (ages 0-2) were described by Hartman and Brandt (1995) during 1990-1992 in mid-Chesapeake Bay, while Virginia Institute of Marine Science's Chesapeake Bay Multispecies

Monitoring and Assessment Program (ChesMMAP) characterized age-specific diets of weakfish (ages 0-5) in the entire Bay during 2002-2003 (Bonzek et al. 2004). These latter data were presented at the 2004 American Fisheries Society Meeting and were provided for this analysis (R. Latour, VIMS, personal communication). We have limited the following discussion to Atlantic menhaden, spot, bay anchovy, and invertebrates. Diets in Hartman and Brandt (1995) preceded decreases in forage relative abundance and weakfish weight-at-age, while those described in Bonzek et al. (2004) represent recent conditions of depleted forage and reduced weight-at-age.

Weakfish diet studies in Chesapeake Bay have indicated major changes in the past decade. Atlantic menhaden, spot, bay anchovy, and invertebrates comprised over 75% of diets, by weight, during 1990-1992 and 2002-2003 (Figure 41). Bay anchovy and Atlantic menhaden have become much less frequent in weakfish diets between the early 1990s and early 2000s, spot have disappeared, and invertebrates have comprised a greater part of the diet.

Weakfish in Chesapeake Bay during both periods made a rapid transition to piscivory - bay anchovy dominated their diets as young-of-year. However, transition from bay anchovy to dominance of spot and Atlantic menhaden at age 2 that occurred during the early 1990s was not evident in recent years. Older weakfish (ages 2+) are subsisting on the same invertebrate and bay anchovy diet as ages 0-1, with a much smaller supplement of menhaden (Figure 41). Striped bass diets in Chesapeake Bay have undergone similar changes as those described for weakfish; small striped bass are now relying more on invertebrate prey and large striped bass on small pelagic prey such as bay anchovy and age 0 clupeids, and benthic invertebrates (Griffin and Margraf 2003; Overton 2003). This suggests that larger prey (juvenile menhaden and spot) were insufficiently available for both species to complete the transition to piscivory.

Weakfish are cannibalistic; weakfish juveniles were specifically mentioned in diet studies reviewed by Mercer (1985) and comprised about 20% of weakfish diet by weight in Delaware Bay during 1985 (Taylor 1987). Weakfish were not specifically mentioned in Chesapeake Bay diets during 1990-1992 (Hartman and Brandt 1995), but other weakfish comprised 10% of age 2 and 6% of ages 3-5 diets in Chesapeake Bay during 2002-2003 (R. Latour, VIMS, personal communication). Increased cannibalism could be a consequence of weakfish's inability to compete with striped bass for other forage fish.

Discussions between the Weakfish Stock Assessment Subcommittee (WSAS) and the 40th SARC panel spawned the multiple hypothesis approach adopted by the WSAS to deal with conflicting signals in the data and analyses in this assessment. This approach led the WSAS to formulate and test, and reformulate and retest alternative hypotheses about M to explain patterns in the underlying data. This assessment strategy indicated the likelihood that the dynamics of striped bass predation and competition, and regime change took over in the 1990s once overharvesting had been controlled. Models used in this investigation were of intermediate complexity because of the basic limitations of the observational data. Simplified procedures allowed for the testing of overfishing and ecological hypotheses; overfishing was not supported as the sole cause of the recent decline.

Because of the large numbers of inputs into complex age-structured models, other parameters that may be indicative of other underlying dynamics (growth, productivity, recruitment, natural

mortality, spatial dynamics, fishing efficiency, and fishing selectivity) are held constant or ignored. This assessment strategy must become less favored in light of realization that the dynamics of predation, competition, environmental regime shifts, and habitat alteration or deterioration may take over once overharvesting has been controlled (Link 2002). Simply presuming that ceasing exploitation on an overfished stock will result in recovery ignores the uncertainty imposed by ecological systems (Link 2002). Stationarity of ecological conditions or constancy of M cannot be an acceptable default assumption (Sainsbury 1998).

We do not advocate abandoning more complex age-structured models, but are advocating testing the overfishing hypothesis with less time-consuming methods before plunging into their application. If these less intensive methods (trend analyses, relative F , year-class catch curves, and surplus production models, for example) lend some support to F as a major contributor to the dynamics in question, then proceed with more intensive analysis. If the hypothesis that fishing is a major contributor to a decline is not well supported, time should be spent pursuing alternate explanations.

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TABLES

Table 1. Descriptions of surveys used as estimates of forage relative abundance in the mid-Atlantic region. MH = Atlantic menhaden, S = spot, and BA = bay anchovy.

Location	Gear	Time-series	Estimator	Source	Species
NC, Ablemarle Sound	Seine	1972-2002	Geometric mean	L. Paramore, NC DMF	MH
NC, Ablemarle Sound	Seine	1972-2002	Delta log-normal	ASMFC 2004a	MH
NC, Pamlico Sound	16 ft trawl	1988-2002	Arithmetic mean	L. Paramore, NC DMF	S
VA, Chesapeake Bay	Seine	1967-1973, 1980-2003	Geometric mean	www.fisheries.vims.edu	MH, S, BA
MD, Chesapeake Bay	Seine	1959-2003	Presence-absence	E. Durell, MD DNR	MH S, BA
MD, Coastal Bays	16 ft trawl	1972-2003	Geometric mean	Casey and Doctor (2004)	S, BA
DE, Delaware Bay	30 ft trawl	1966-1972, 1980-1984, 1990-2002	Arithmetic mean	Michels and Grecco (2003)	MH, S
DE, Delaware Bay	16 ft trawl	1980-2002	Arithmetic mean	Michels and Grecco (2003)	S, BA
NJ, Delaware Bay	Seine	1980-2003	Arithmetic mean	J. Brust, NJ DEP	MH, S, BA

Table 2. Weakfish landings (mt commercial and recreational, including discards), weakfish biomass indices (WRI, kg catch per trip), and striped bass biomass (mt) estimates used as inputs for the logistic predator-prey model.

Year	Weakfish Harvest	WRI	Striped Bass
1981	20814	0.96	6236
1982	13823	0.46	5495
1983	14426	0.5	4656
1984	13338	0.31	6532
1985	11181	0.28	6794
1986	15590	0.42	7701
1987	11691	0.35	11741
1988	13090	0.3	14739
1989	8136	0.13	20402
1990	5440	0.09	24857
1991	5610	0.1	28655
1992	4793	0.14	35771
1993	4448	0.09	41806
1994	5395	0.19	49611
1995	5977	0.29	60005
1996	6879	0.38	70322
1997	7206	0.34	78778
1998	7809	0.44	83480
1999	6219	0.38	84275
2000	6126	0.42	85650
2001	5369	0.24	95746
2002	4434	0.24	89280
2003	2660	0.12	104282

Table 3. Variables and scoring criteria for the candidate species “scorecard”. RF = rescaled F analysis (Crecco 2005) and PP = predator-prey model.

Variable	Scoring criterion
1950-2003 landings correlation	1 if significant and negative, 0 otherwise.
Biomass correlation	1981-2003 RF and PP estimates r^2 = score
Recent decline	1 if decline and 0 otherwise
Biomass closeness	Mean RF: mean PP or mean PP:mean RF, whichever < 1
Weakfish in diet	Absent or present, 0 or 1
Bay anchovy in diet	Absent or present, 0 or 1
Atlantic menhaden in diet	Absent or present, 0 or 1
Spot in diet	Absent or present, 0 or 1
Clupieds in diet	Absent or present, 0 or 1
Scianeids in diet	Absent or present, 0 or 1

Table 4. Correlations of relative abundance of major weakfish forage species in Mid-Atlantic region surveys and the DE Trawl Survey Quality+ PSD. Pearson correlation coefficient = r. Grey highlights indicate significant at $P < 0.1$ after adjustment for multiple comparisons. Ablemarle Sound correlations depict two estimators of central tendency applied to the same data.

State	Survey	Menhaden (r, P)	Spot (r, P)	Bay Anchovy (r, P)
NJ	DE Bay Seine	-0.35, 0.137	0.68, 0.0015	0.22, 0.368
NJ	Ocean trawl CPUE, number			-0.43, 0.129
NJ	Ocean trawl CPUE, weight			-0.36, 0.209
DE	16 foot DE Bay Trawl		0.13, 0.601	-0.27, 0.277
DE	30 foot DE Bay Trawl	-0.43, 0.040	-0.09, 0.669	
MD	Coast Bays 16 ft trawl		0.49, 0.03	0.61, 0.0063
MD	Chesapeake 100 ft seine	0.79, <0.0001	0.73, <0.0001	-0.163, 0.427
VA	Chesapeake 100 ft seine	0.60, 0.004	0.54, 0.009	0.63, 0.0016
VA	30 foot trawl, Chesapeake		0.39, 0.088	-0.52, 0.025
NC	Ablemarle Sound Seine GM	0.33, 0.159		
NC	Ablemarle Sound Seine Delta log-normal	0.64, 0.003		
NC	Pamlico Sound Trawls		-0.22, 0.439	
Coastwide	Abundance estimate	0.71, <0.0001		

Table 5. Sensitivity of logistic Steele-Henderson predator-prey (striped bass-weakfish) model parameters to different time-series. Original treatment is the 1981-2003 time-series. Parameter r is the intrinsic rate of increase, K = carrying capacity (mt), B_0 = initial biomass of time-series, c = per capita consumption of weakfish by striped bass, A is weakfish biomass where satiation of striped bass occurs, and mean q is average catchability.

Treatment	r	K	B_0	c	A	Mean q
Original	1.17	49080	65275	0.076	1959	0.000020
end 2002	1.14	49080	65262	0.084	9242	0.000020
end 2001	1.11	49081	65146	0.082	12500	0.000019
end 2000	1.08	49080	65298	0.076	20335	0.000018
start 1982	1.26	49139	16744	0.155	11387	0.000020
start 1983	1.23	48738	16738	0.164	14824	0.000020
start 1984	1.22	43989	16744	0.155	15705	0.000020
1981 fixed	1.16	44719	49000	0.279	27293	0.000020

Table 6. Estimates of logistic predator-prey model parameters r , K , and c (per capita effect of weakfish by striped bass) under sigmoidal and linear (constant) striped bass predation functions. SSQ is the sum of squared deviations of observed and predicted WRI indices.

Time-series	Predator function	r	K (mt)	c	SSQ
1981-2003	Sigmoid	1.17	49,080	0.076	2.62
1982-2003	Sigmoid	1.26	49,139	0.155	2.71
1981-2003	Linear	1.09	55,136	0.073	2.89
1982-2003	Linear	1.25	37,883	0.069	2.49

Table 7. Summary of results of analyses of candidate predator-competitors and weakfish. Model refers to a logistic Steele and Henderson (1984) predator-prey model . “Recent decline?” indicates that the predator-prey model with weakfish produced a decline in weakfish biomass in the most recent years. Biomass ratio is the ratio of predicted to observed (or vice-versa, depending on which is < 1) 1981-2003 mean biomass estimates. Parameter r = intrinsic rate of population increase; K = maximum population biomass (mt); maximum c is per capita consumption by candidate predator; A is weakfish biomass (mt)where predator satiation begins.

	Atlantic croaker	Bluefish	Summer flounder	Spiny dogfish	Striped bass
Landings	r = 0.24	r = 0.73	r = 0.73	NA	r = -0.34
Pearson correlation	P = 0.08	P < 0.0001	P < 0.0001		P = 0.01
Predator-prey model r²	0.72	0.01	0.62	0.29	0.78
Recent decline?	No	No	Yes	No	Yes
Biomass ratio	0.93	0.08	0.56	0.89	0.73
r (intrinsic rate)	1.00	2.92	0.62	1.22	1.17
K (mt)	53,823	296,109	293,653	56,672	49,080
c	0.398	1.198	1.190	0.214	0.076
A (mt)	88,414	90,377	43,843	92,115	1,959

Table 8. Scorecard for candidate species interaction with weakfish. Scoring criteria are explained in Table 3. Potential maximum score = 10 and indicates how strong a candidate a species would be for negative interactions with weakfish.

Variable	Score				
	Atlantic croaker	Bluefish	Summer flounder	Spiny dogfish	Striped bass
Landings correlation	0	0	0	0	1
Biomass correlation	0.72	0.01	0.62	0.29	0.78
Recent decline	0	0	1	0	1
Biomass closeness	0.93	0.08	0.56	0.89	0.73
Weakfish in diet	0	1	1	1	1
Anchovy in diet	1	1	1	1	1
Menhaden in diet	0	1	0	0	1
Spot in diet	0	1	0	0	1
Clupieds in diet	0	1	1	1	1
Scianeids in diet	0	1	1	1	1
Score (sum)	2.65	6.09	6.18	5.18	9.51

FIGURES

Figure 1. Atlantic Coast commercial landings of weakfish, striped bass, Atlantic croaker, summer flounder, and bluefish.

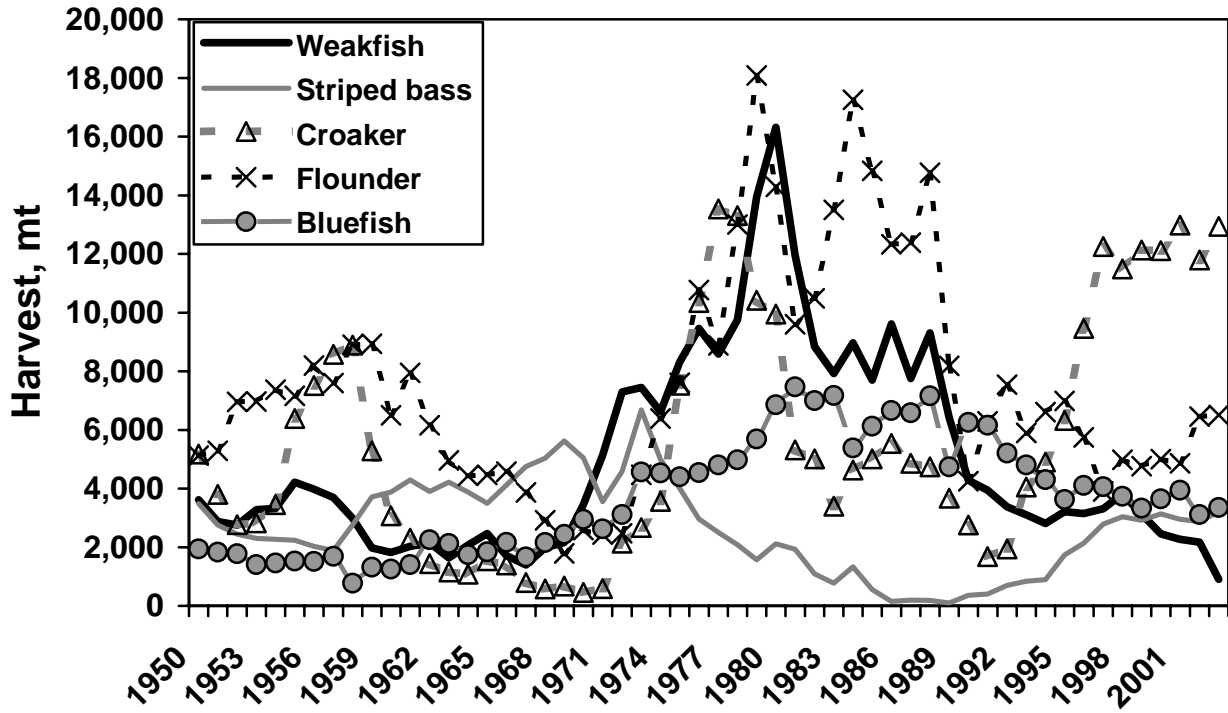


Figure 2. Relative abundance or biomass indices for weakfish. WRI is an MRFSS catch per trip index and equals MRFSS estimates of private/rental catch per trip in mid-Atlantic state waters (harvest and releases). Delaware and New Jersey trawl survey exploitable biomass indices estimated relative biomass of 25 cm and larger weakfish and are z-transformed values plus one.

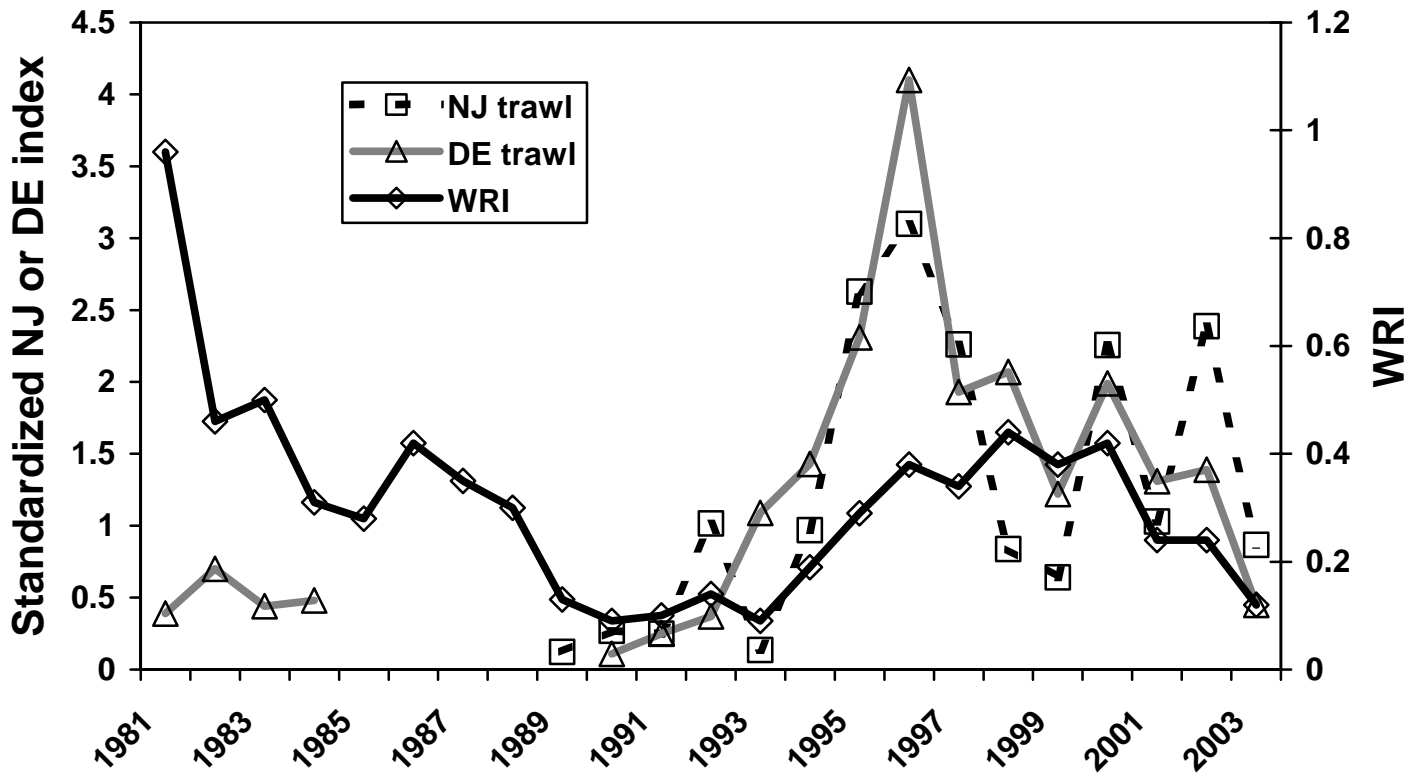


Figure 3. Estimated WRI and New Jersey and Delaware trawl survey catchability versus converged VPA biomass. Delaware estimates have been divided by 10 to place them on the figure.

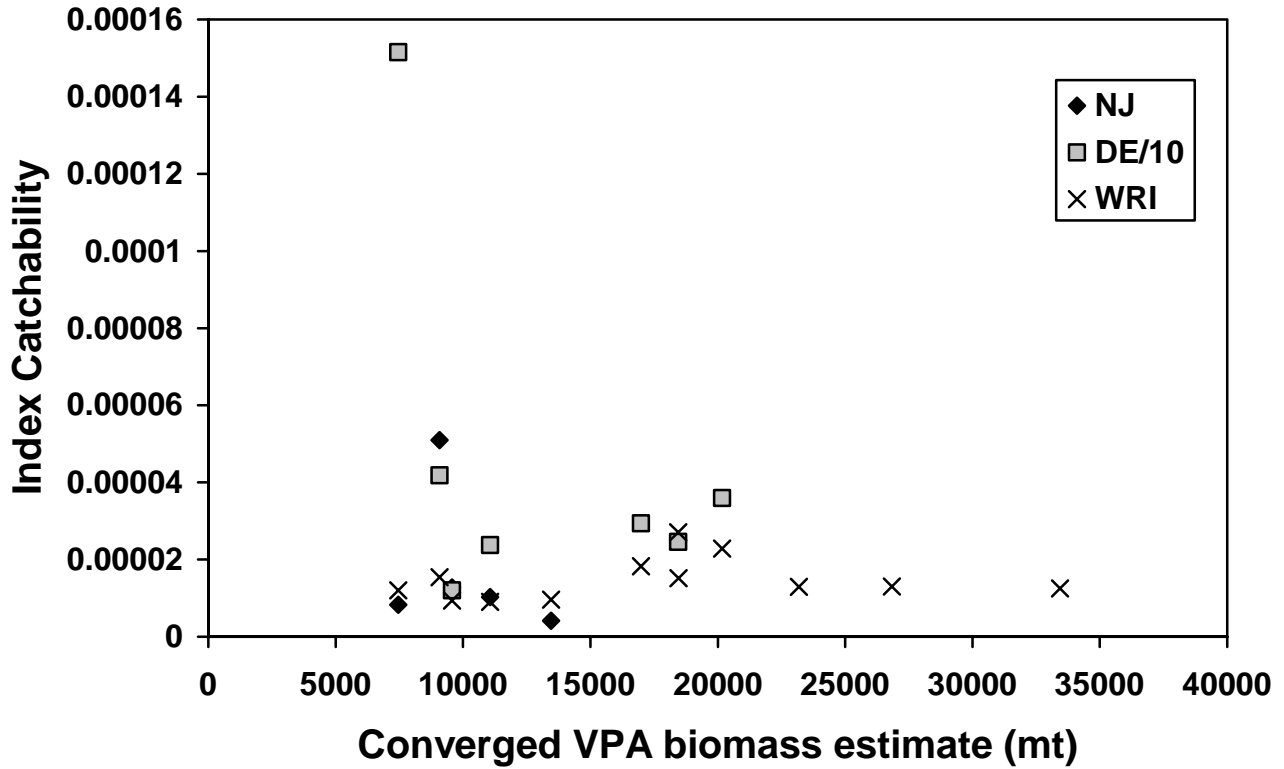


Figure 4. Mean weight at age of 1-5 year-old weakfish in the VPA catch-at-age matrix.

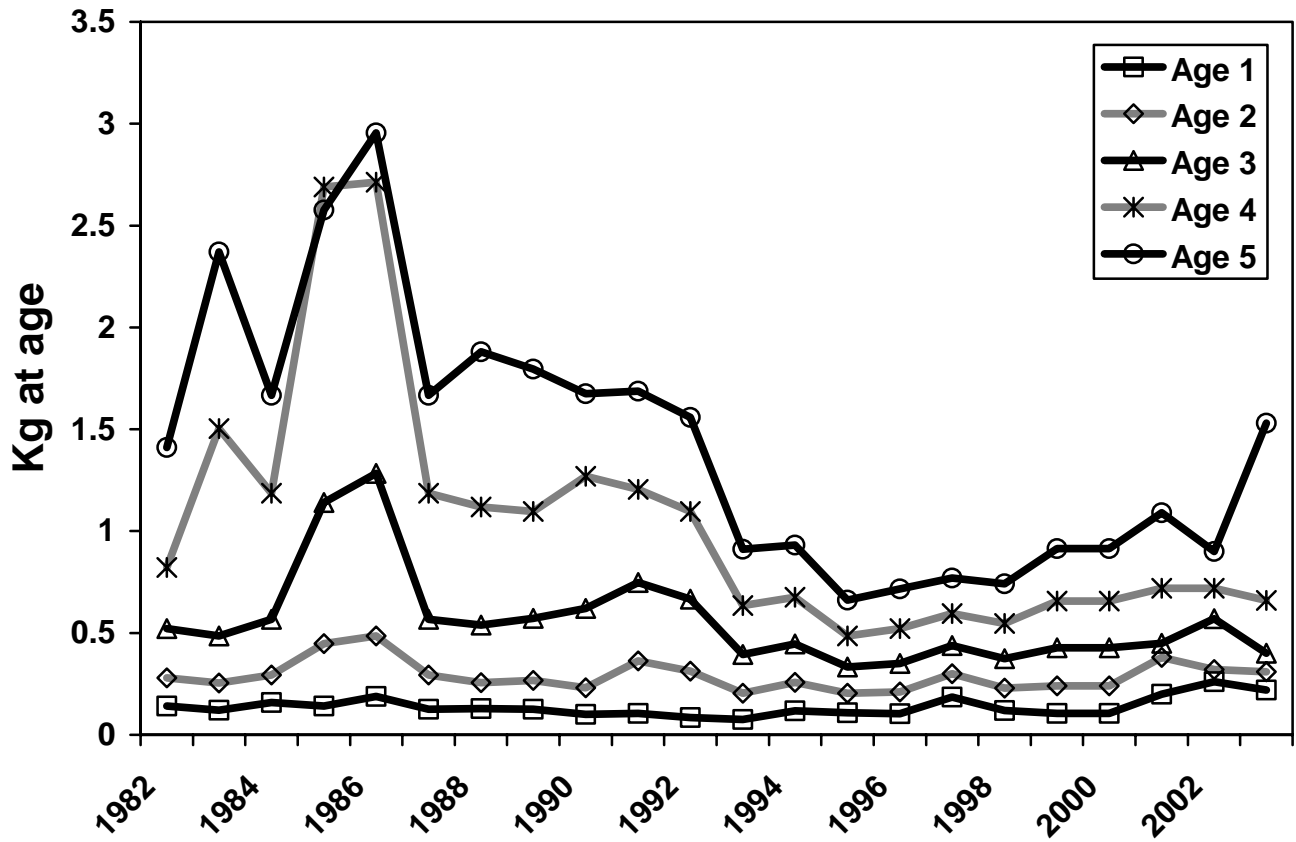


Figure 5. Delaware trawl survey Quality Proportional Stock Density (PSD Quality +). This index equals number of 340 mm+ weakfish divided by number that are 210 mm+ and indicates the proportion of weakfish of interest to anglers.

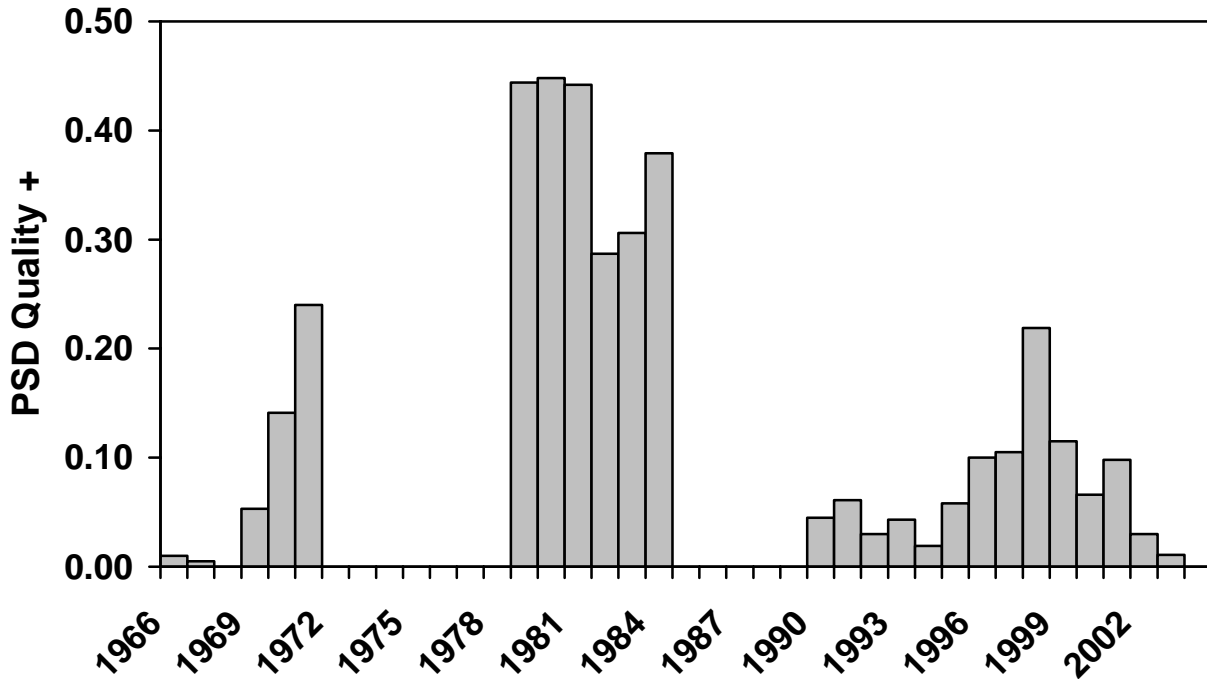


Figure 6. Striped bass biomass indicators along the Atlantic Coast. Age 2+ biomass was estimated by ADAPT VPA (A. Sharov, MDDNR, personal communication). Egg presence-absence is the proportion of plankton tows with eggs in Chesapeake Bay spawning rivers and is a long-term indicator of mature female biomass (Uphoff 1997).

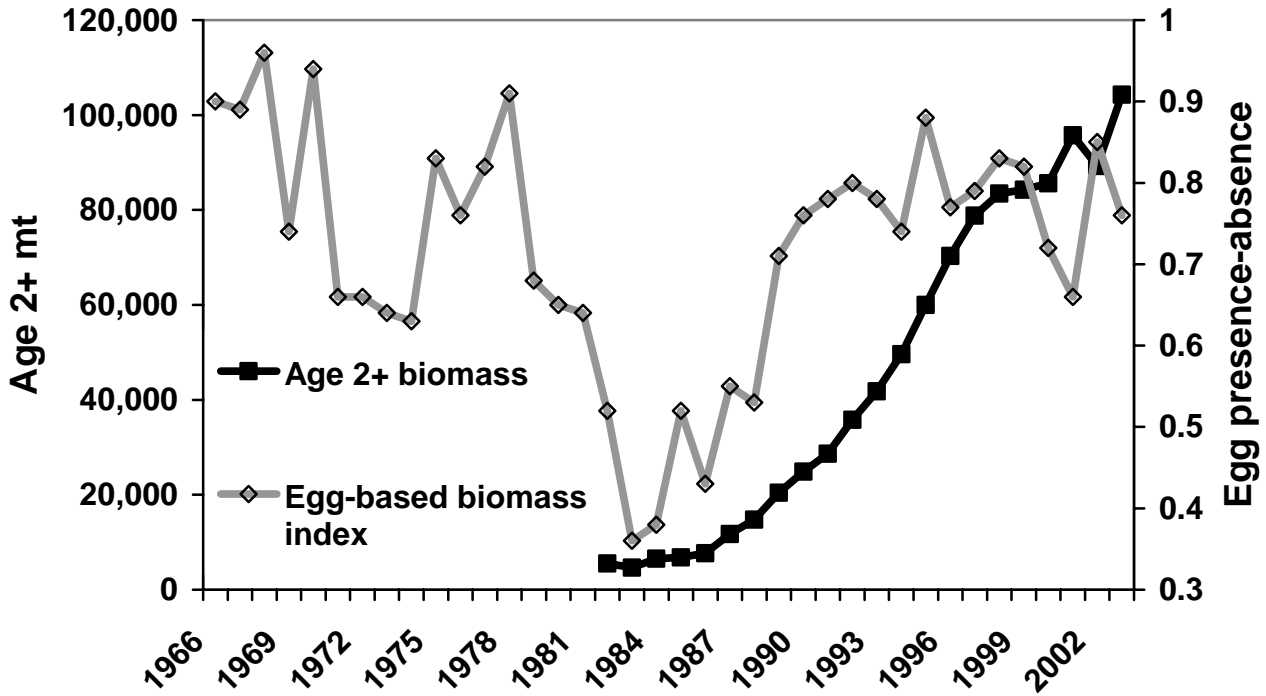


Figure 7. Z-transformed indices of relative abundance of age 0 menhaden in Maryland, Virginia, and North Carolina. Methodologies and locations are summarized in Table 1. Indices for NC are based on the same survey, but different estimators of central tendency; GM = geometric mean and DLN is delta log-normal. A value of zero indicates average abundance for the available time-series.

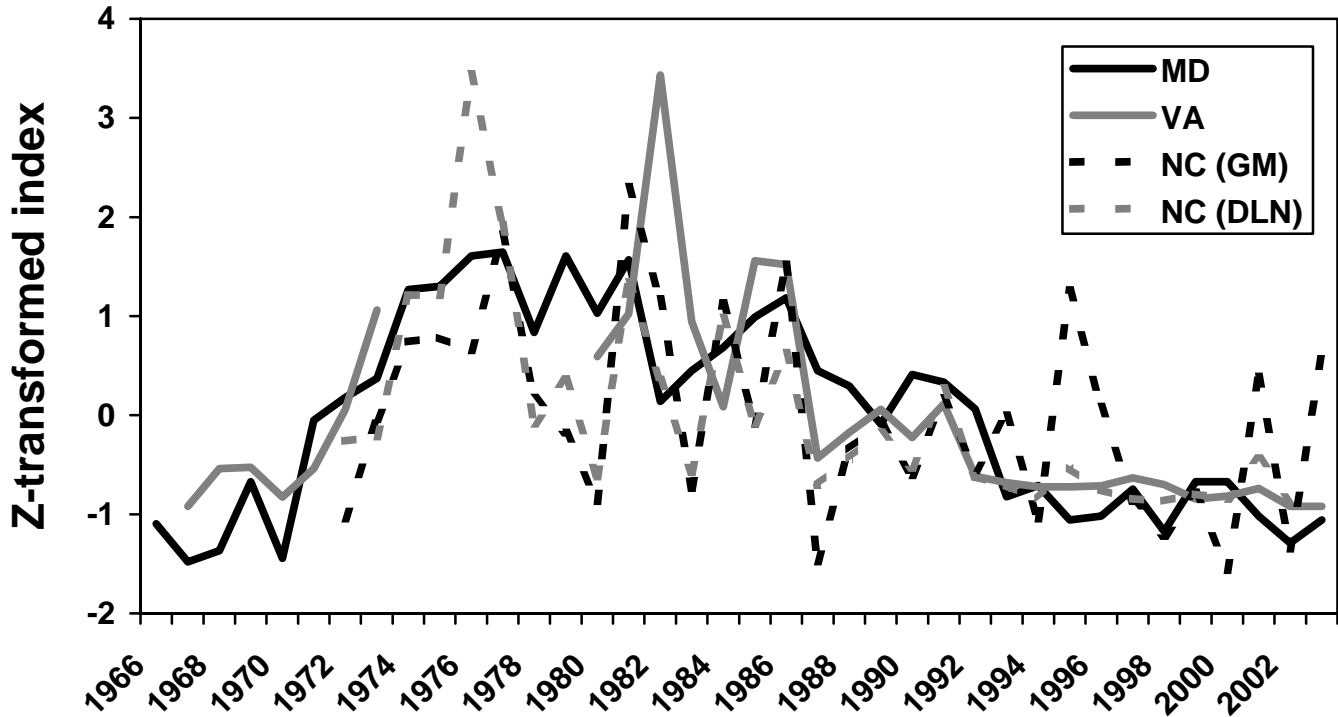


Figure 8. Z-transformed indices of relative abundance of age 0 menhaden in Delaware Bay. Methodologies and locations are summarized in Table 1. A value of zero indicates average abundance for the available time-series.

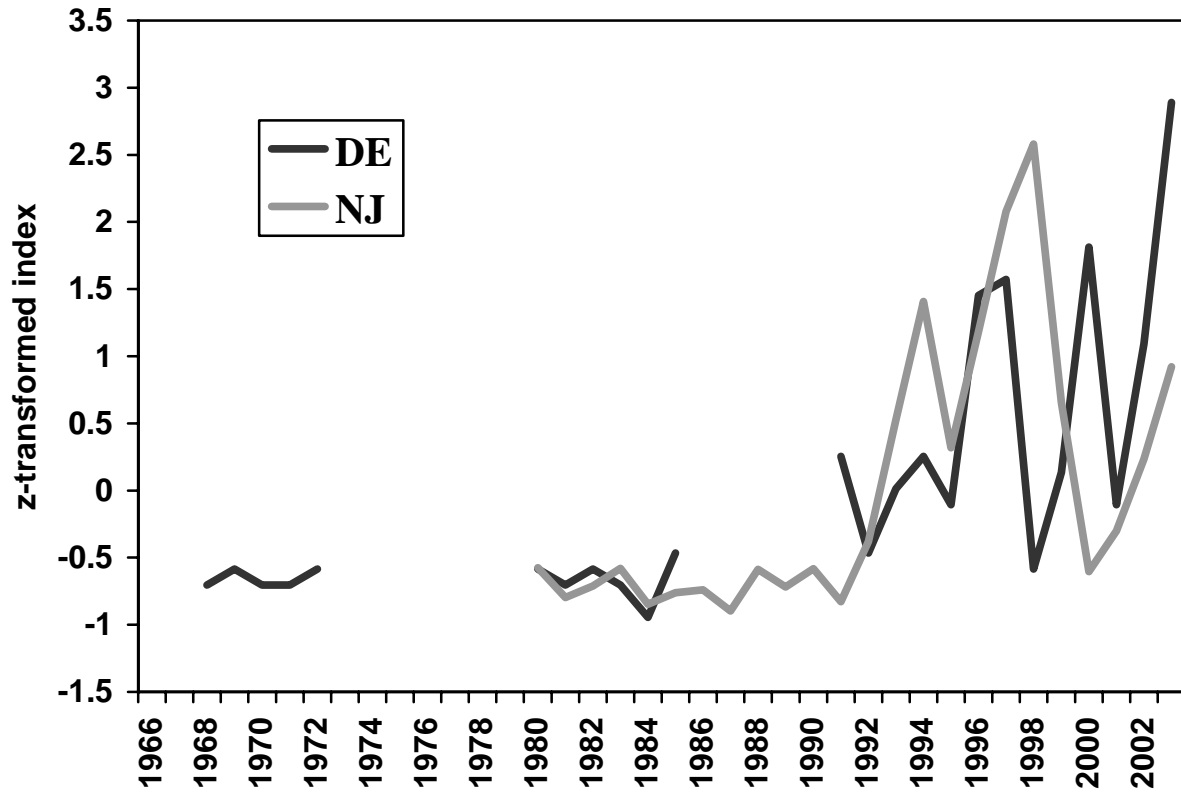


Figure 9. Estimated abundance of age 0 Atlantic menhaden along the Atlantic Coast (ASMFC 2004).

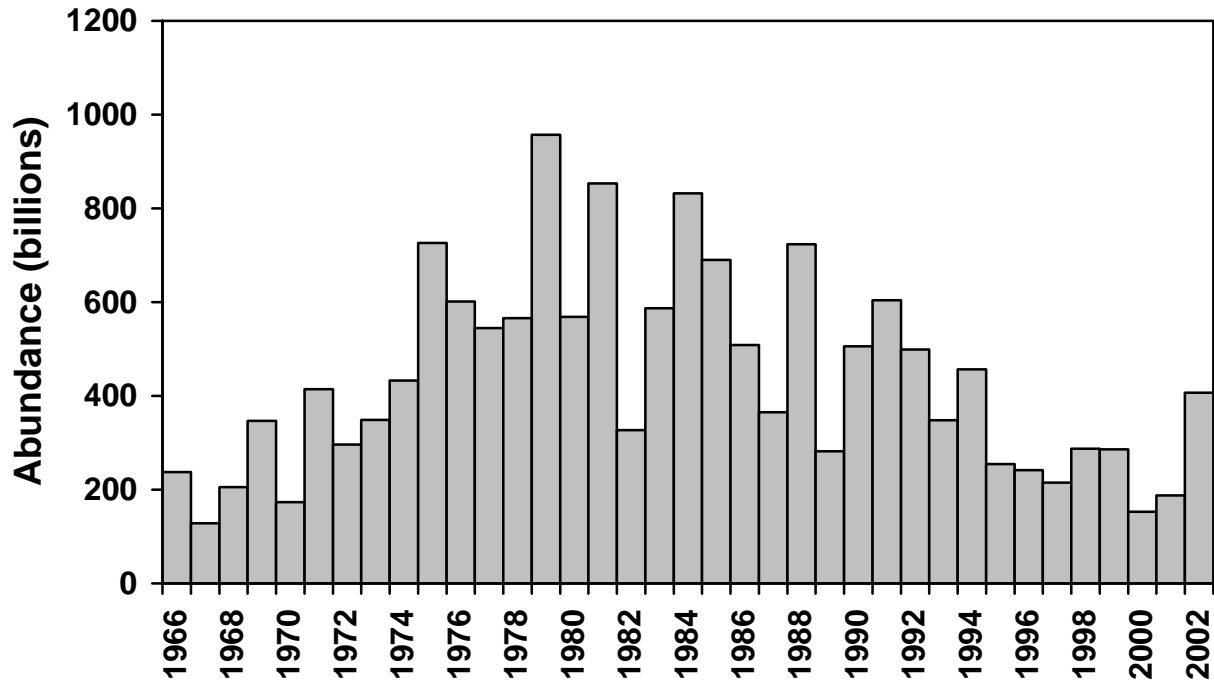


Figure 10. Z-transformed indices of relative abundance of spot in Maryland, Virginia, and North Carolina. Methodologies and locations are summarized in Table 1. Indices T = trawl and S = seine. A value of zero indicates average abundance for the available time-series.

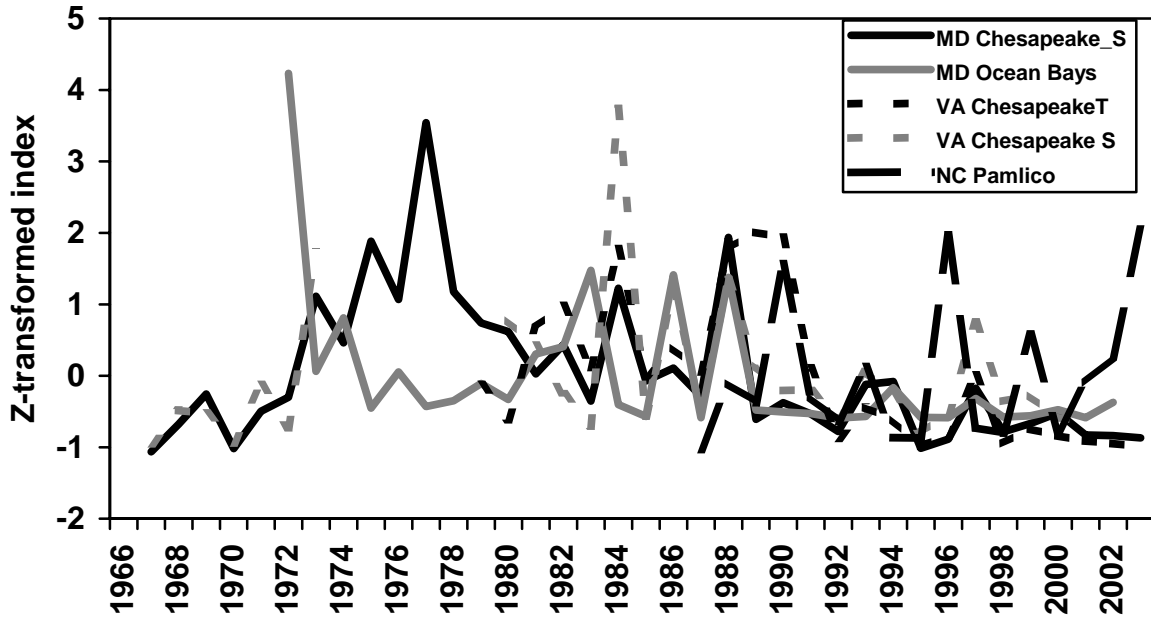


Figure 11. Z-transformed indices of relative abundance of spot in Delaware and New Jersey. Methodologies and locations are summarized in Table 1. Indices T = trawl and S = seine. A value of zero indicates average abundance for the available time-series.

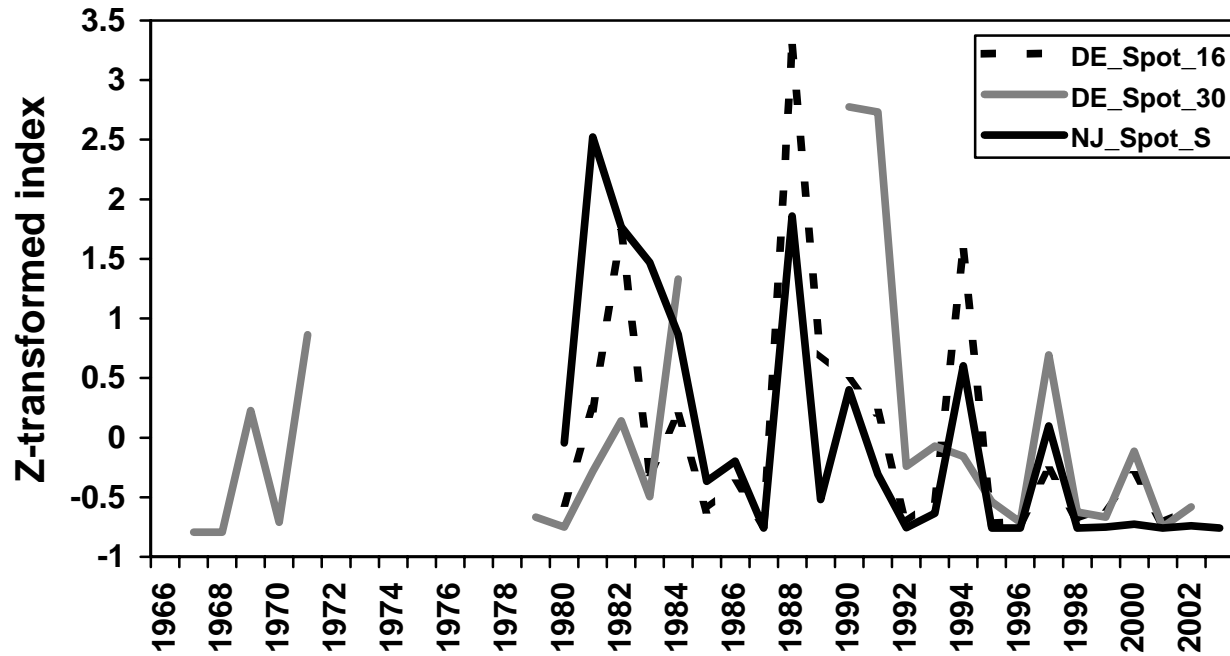


Figure 12. Z-transformed indices of relative abundance of bay anchovy in Maryland, Virginia, and North Carolina. Methodologies and locations are summarized in Table 1. Indices T = trawl and S = seine. A value of zero indicates average abundance for the available time-series.

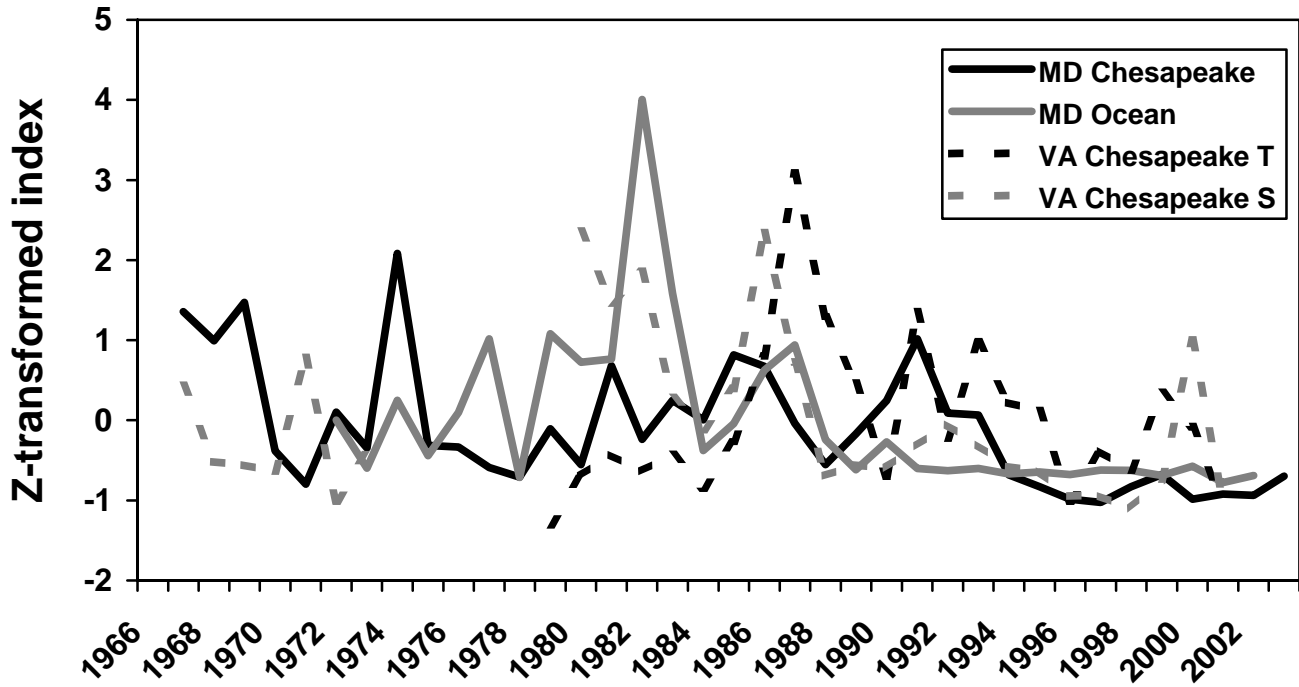


Figure 13. Z-transformed indices of relative abundance of bay anchovy in Delaware and New Jersey. Methodologies and locations are summarized in Table 1. Indices T = trawl and S = seine. A value of zero indicates average abundance for the available time-series.

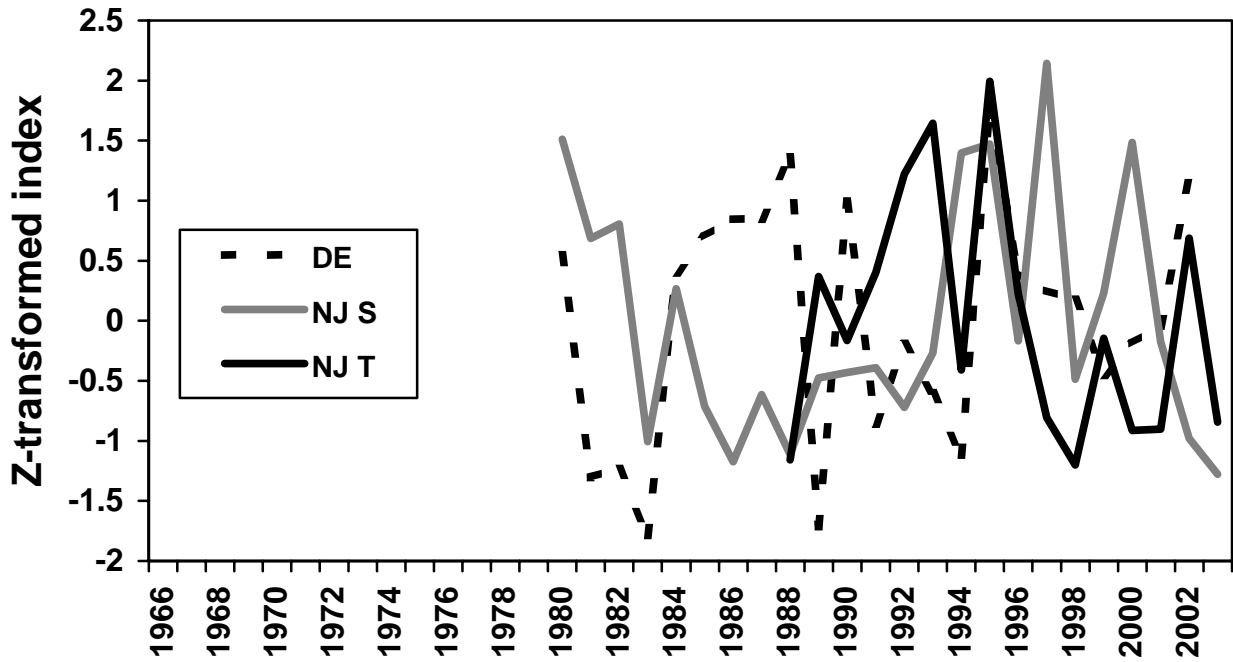


Figure 14. Predicted and observed PSD Q+ weakfish size quality index from multiple regression of age 0 menhaden abundance and a striped bass biomass index.

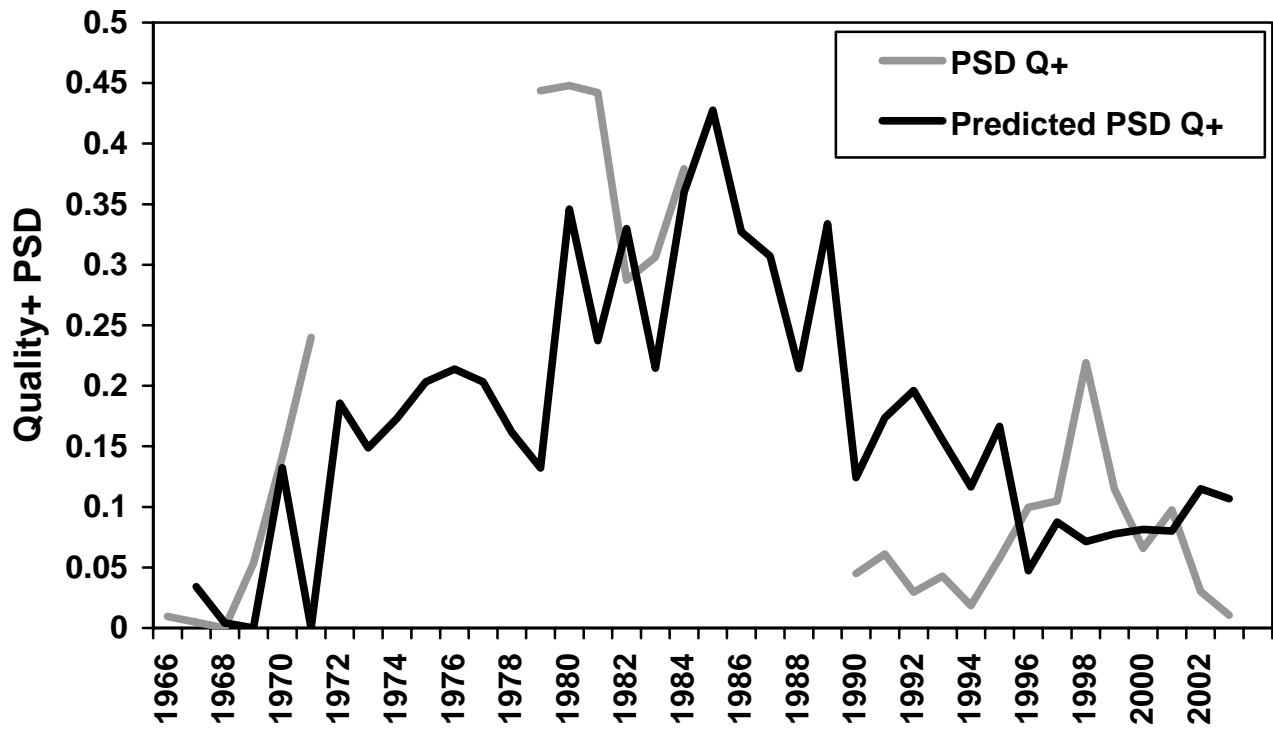


Figure 15. Weakfish catch per trip (includes number harvested and released) index (WRI) versus age 2+ striped bass biomass. Different stanzas are represented by different symbols.

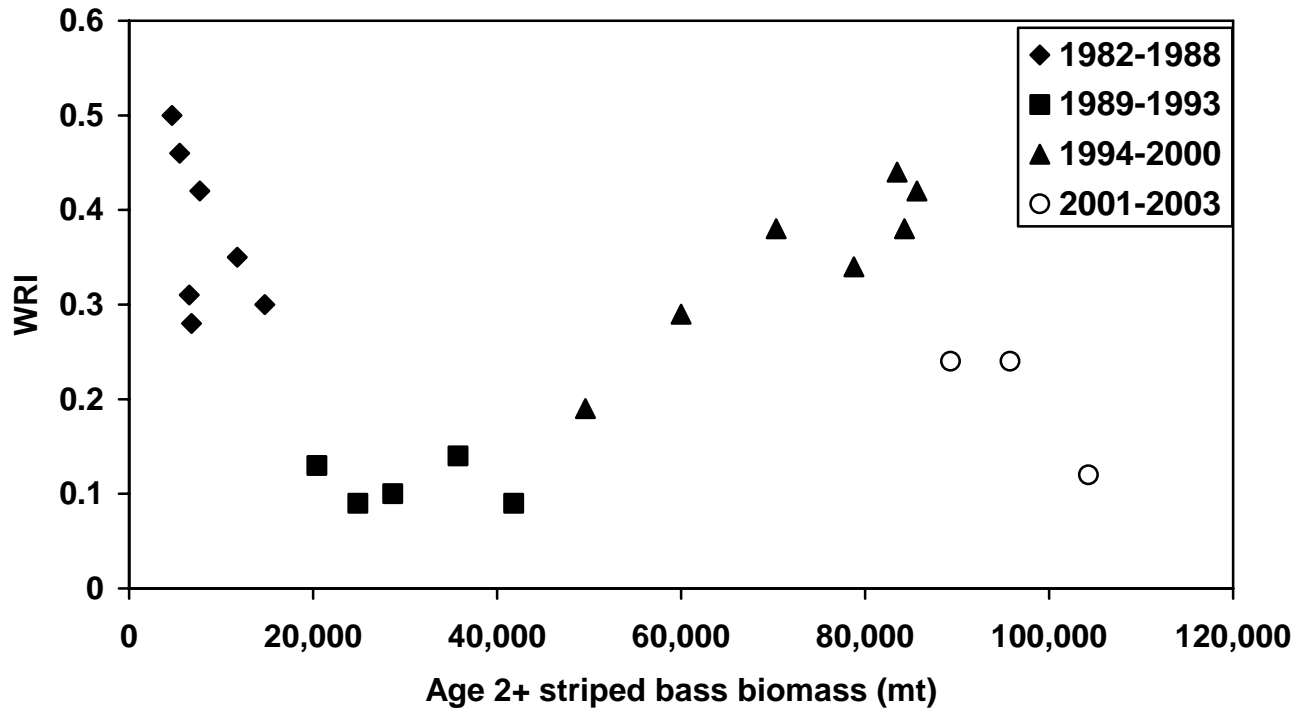


Figure 16. Weakfish biomass indices for DE and NJ trawl surveys versus age 2+ striped bass biomass. Indices are Z-transformed + 1 values. Stanzas in Figure 15 (WRI and striped bass) are maintained and represented by different symbols. Solid symbols represent DE and unfilled symbols indicate NJ.

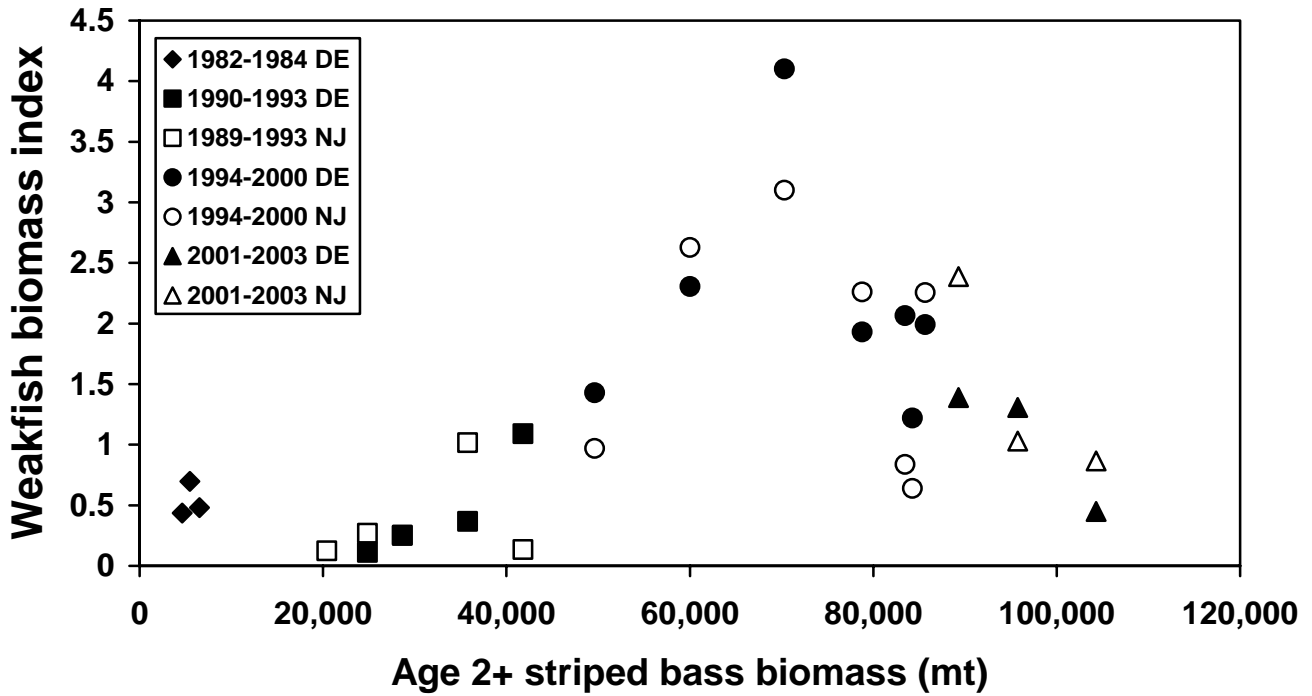


Figure 17. Atlantic Coast commercial landings of weakfish plotted against striped bass landings during 1950-2003. Decades are identified by different symbols; 1990s included 2000-2003. Predicted landings estimated by categorical regression are indicated by solid lines.

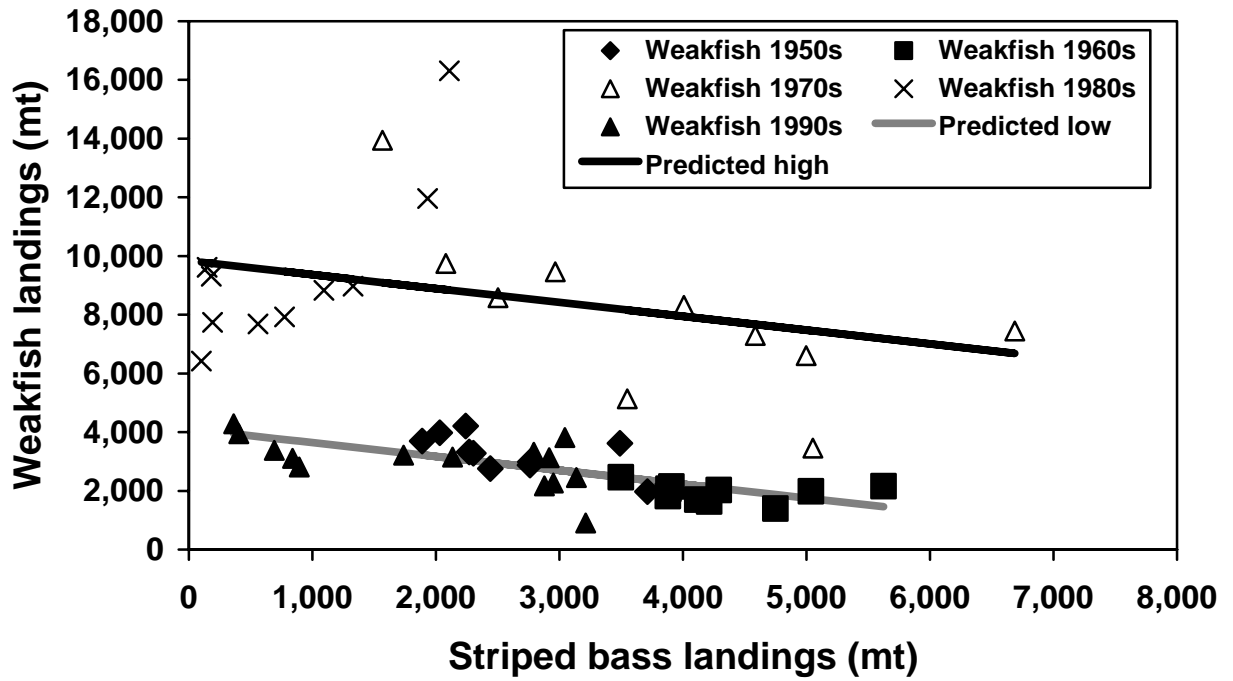


Figure 18. Observed and predicted weakfish landings from categorical regression of bass egg index and time period category.

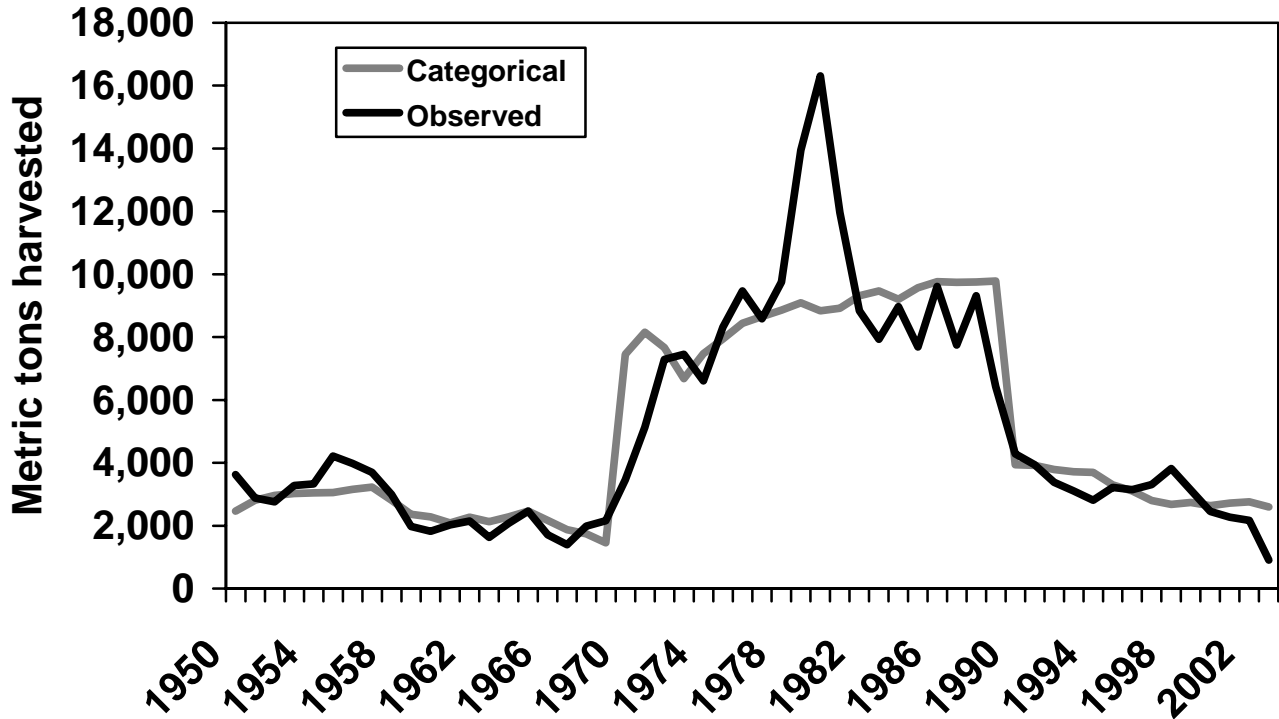


Figure 19. Observed and predicted weakfish landings from multiple regressions of bass egg index and menhaden abundance estimates or menhaden juvenile index.

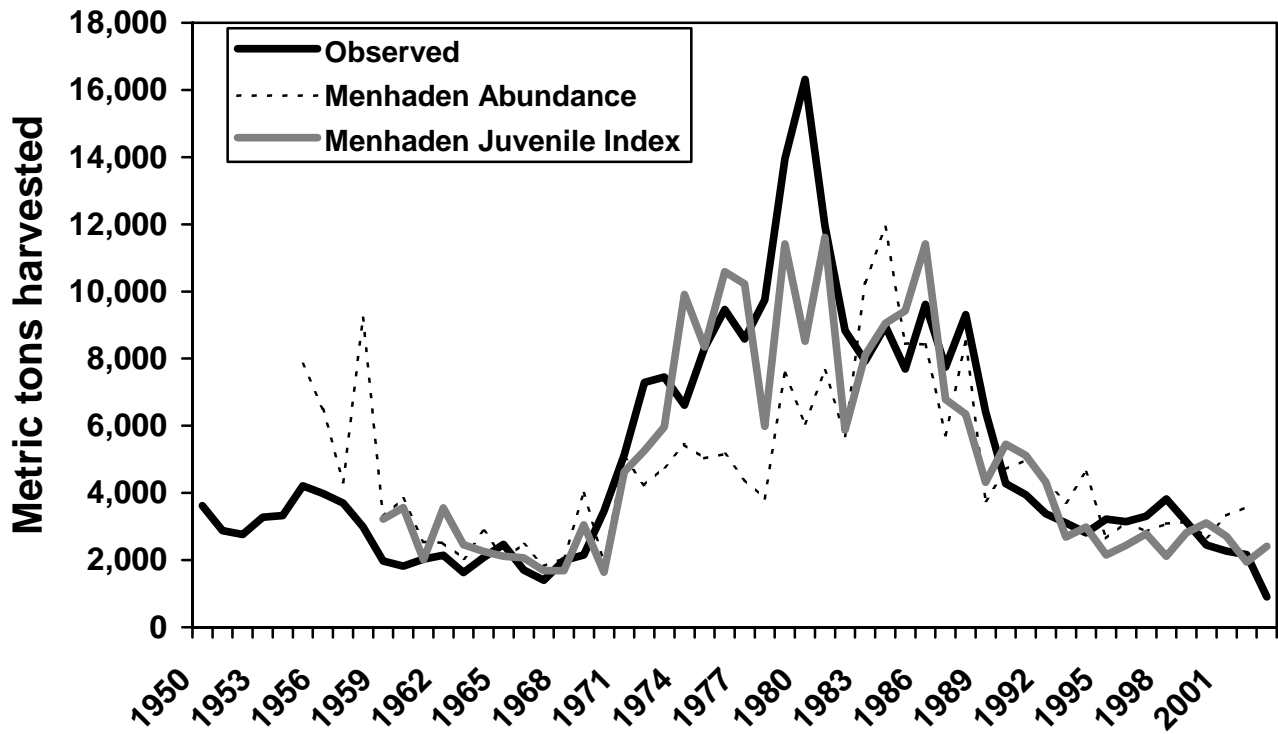


Figure 20. Trends in M_{vpa} .

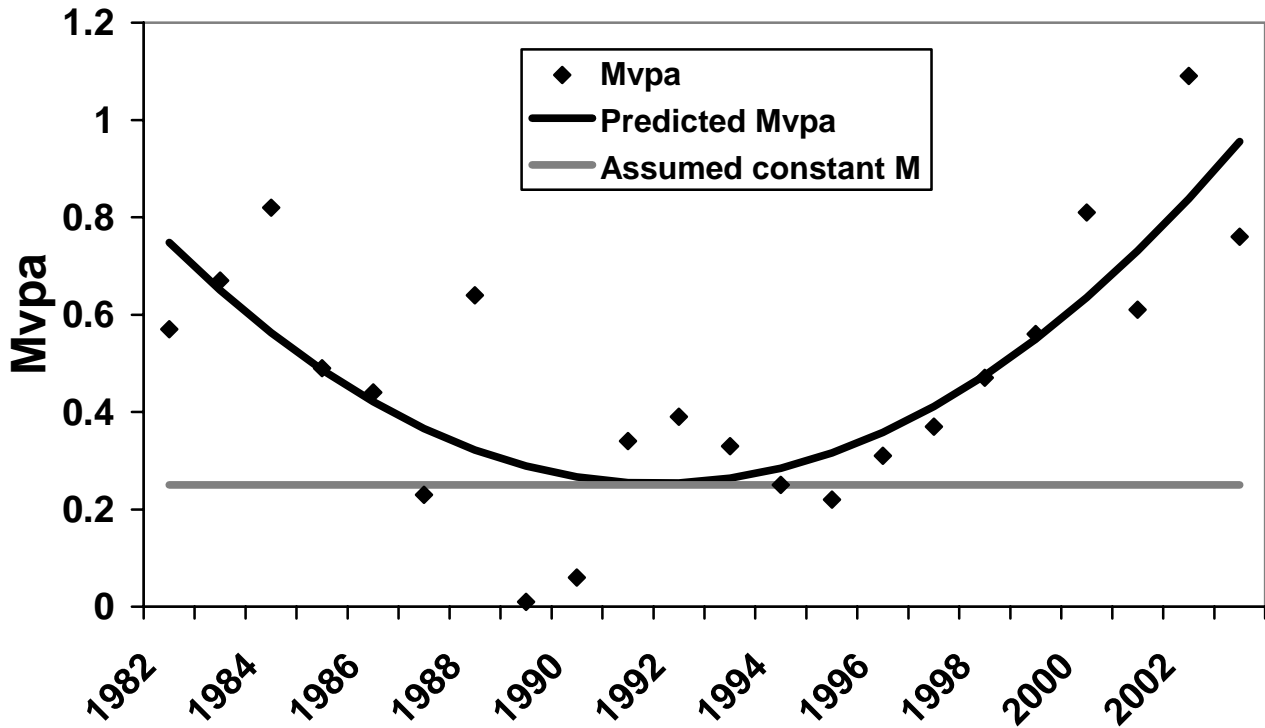


Figure 21. Observed and weakfish biomass indices estimated by logistic striped bass-weakfish predator-prey model.

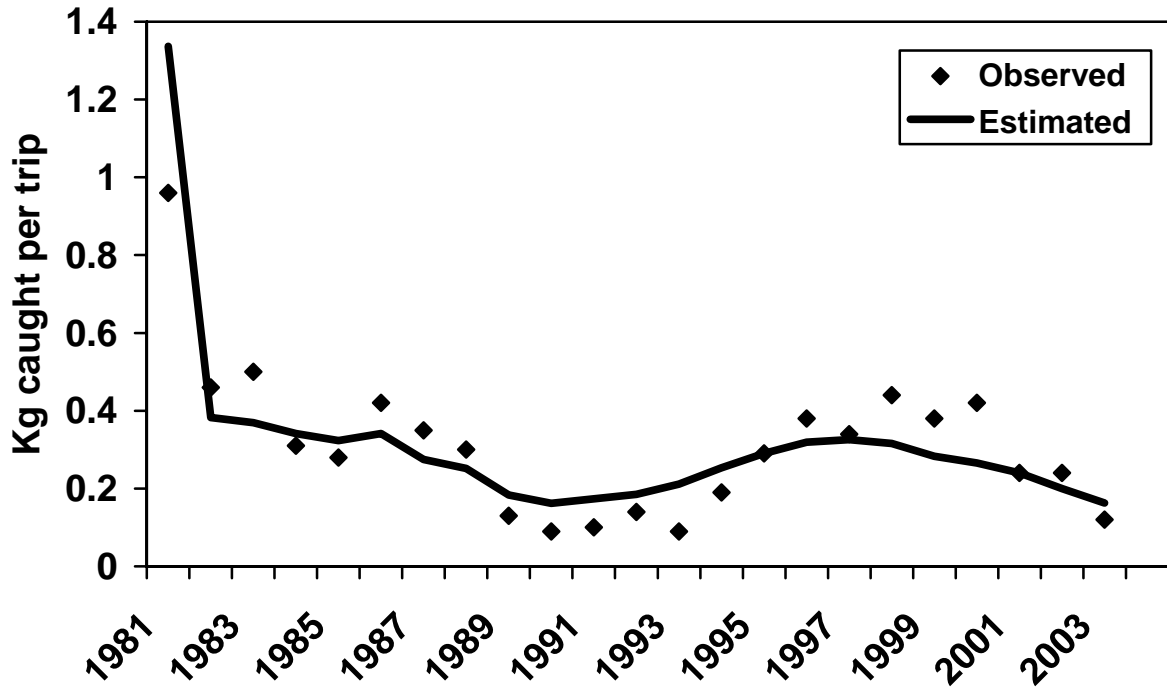


Figure 22. Annual instantaneous mortality rates due to fishing (F), striped bass competition and predation (Mb), F+Mb (Z), and biomass estimated from the logistic predator-prey model.

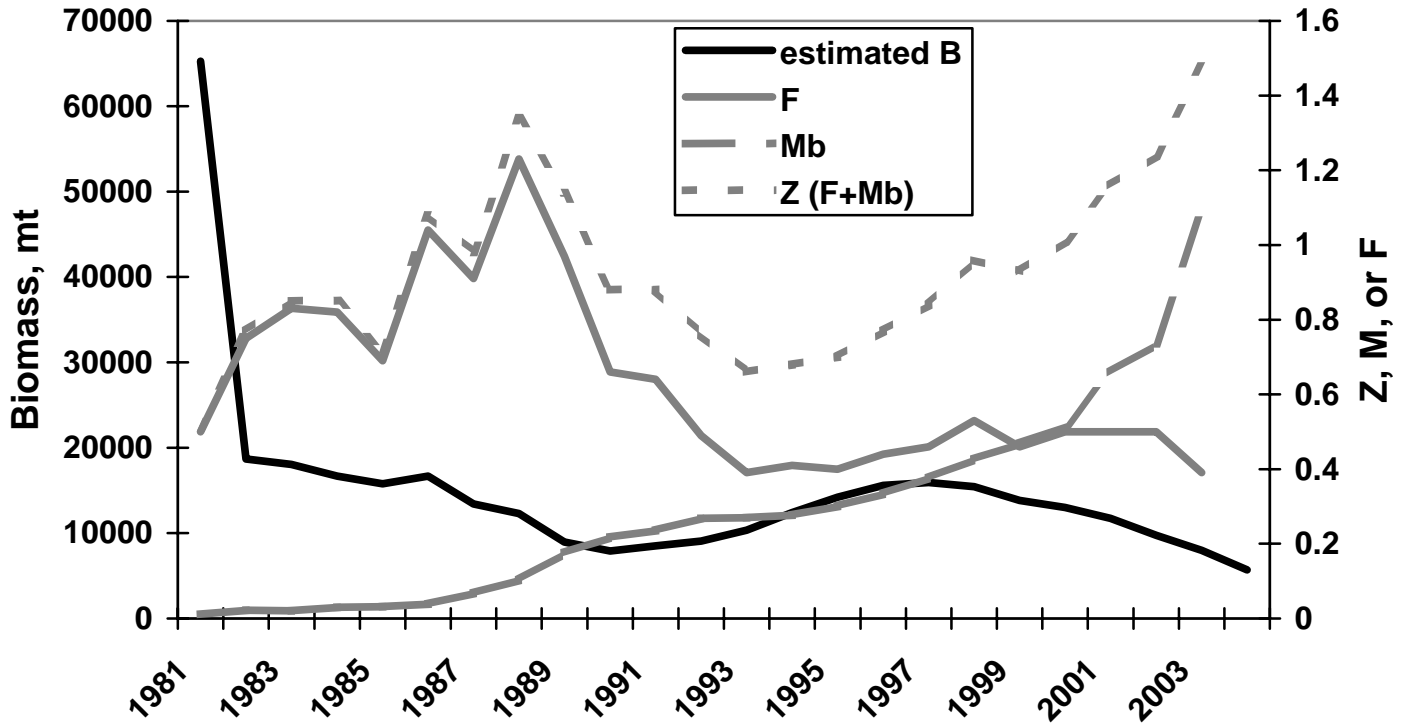


Figure 23. Surplus production estimated from the logistic predator-prey model with and without (traditional) accounting for striped bass predation and competition.

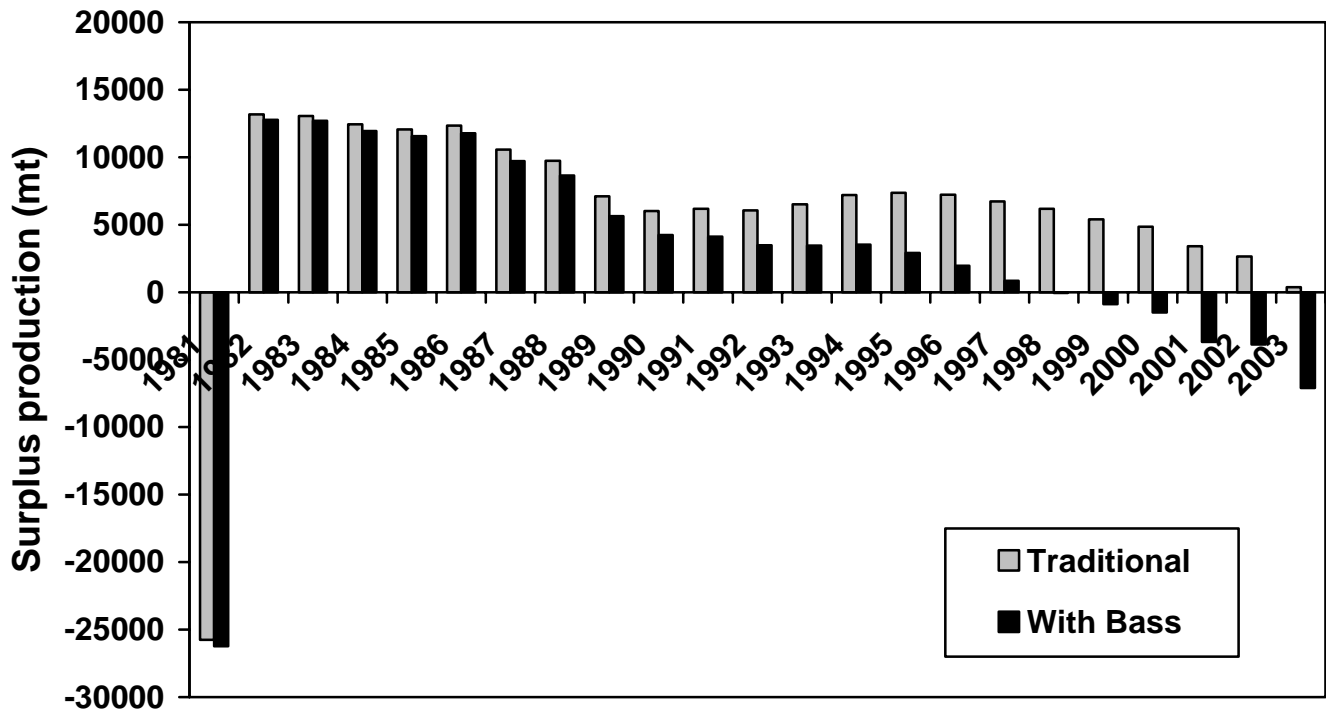


Figure 24. Surplus production calculated with and without (traditional) striped bass associated losses plotted against estimates of F during 1982-2003 from the logistic predator-prey model. Points from 1981 were treated as outliers and omitted from this plot.

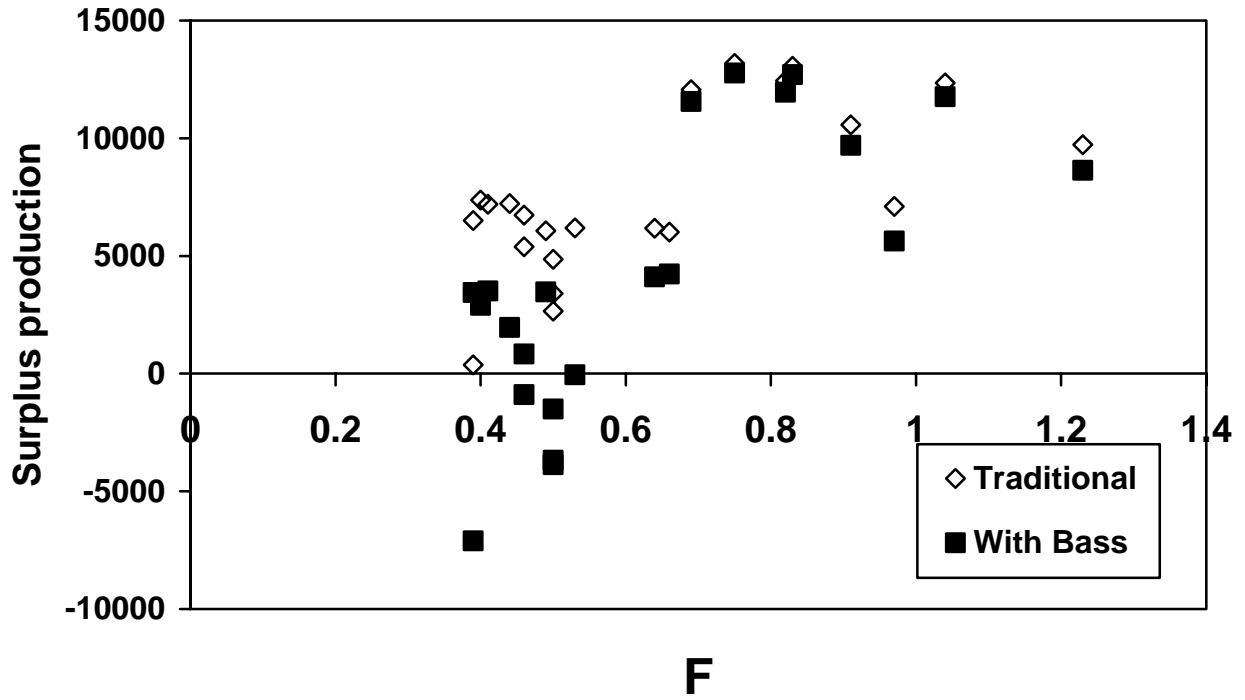


Figure 25. Surplus production calculated with and without (traditional) striped bass associated losses plotted against estimates of Mb during 1982-2003 from the logistic predator-prey model. Points from 1981 were treated as outliers and omitted from this plot.

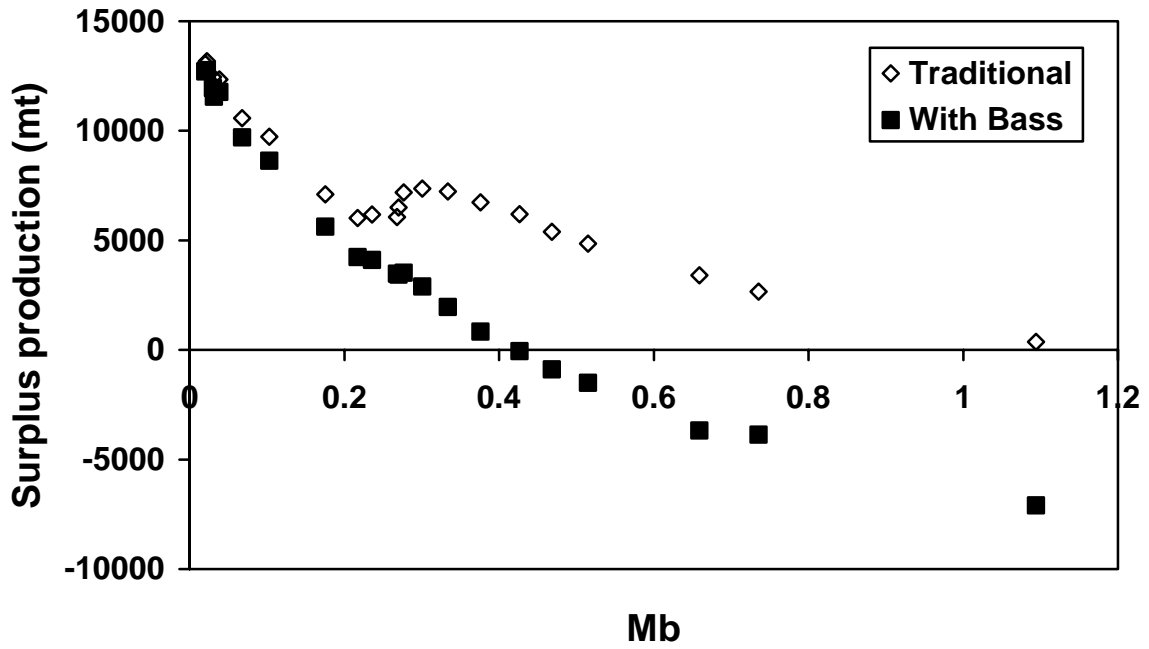


Figure 26. Fit of Schaefer and logistic Steele and Henderson (predator-prey) models to the WRI time-series

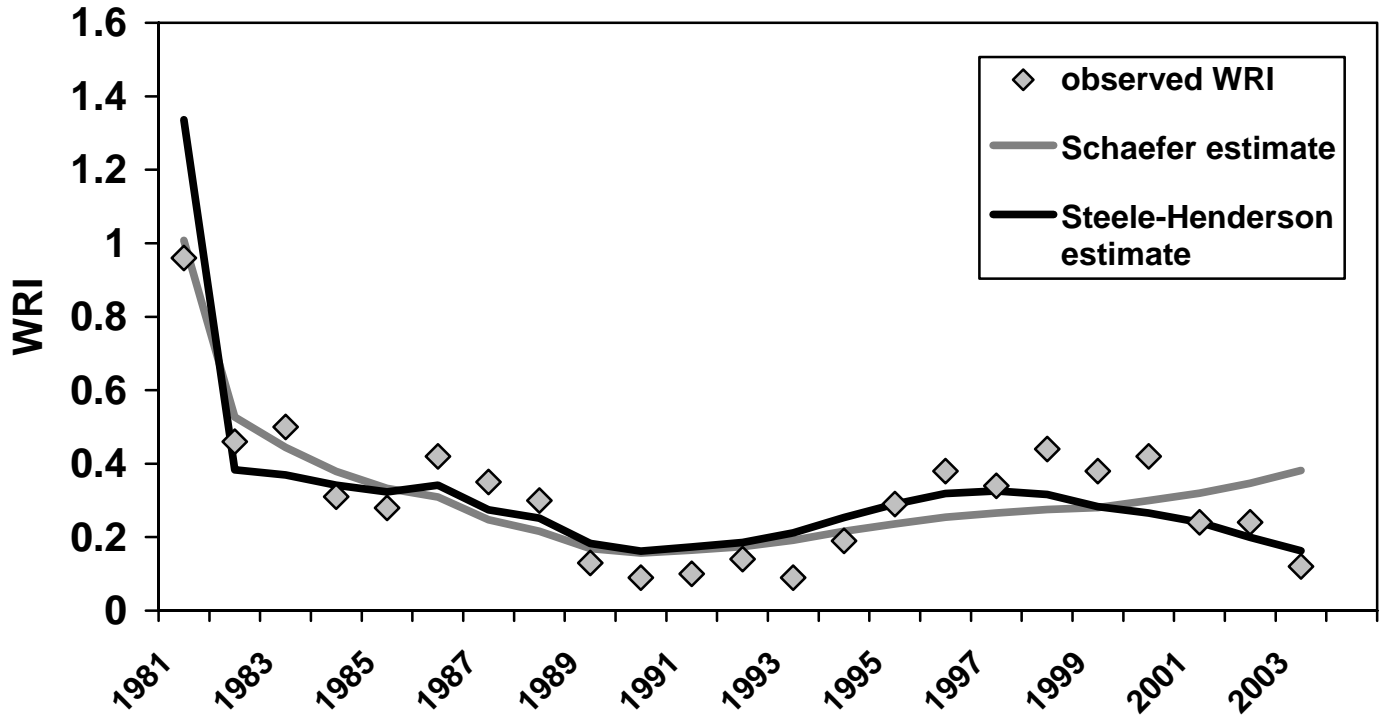


Figure 27. Observed and predicted indices from sensitivity analysis of time-series of the logistic predator-prey model. Original data – 1981-2003. 1-3 years removed from beginning (start) and end; 1981 fixed indicates a value was set to initiate time-series.

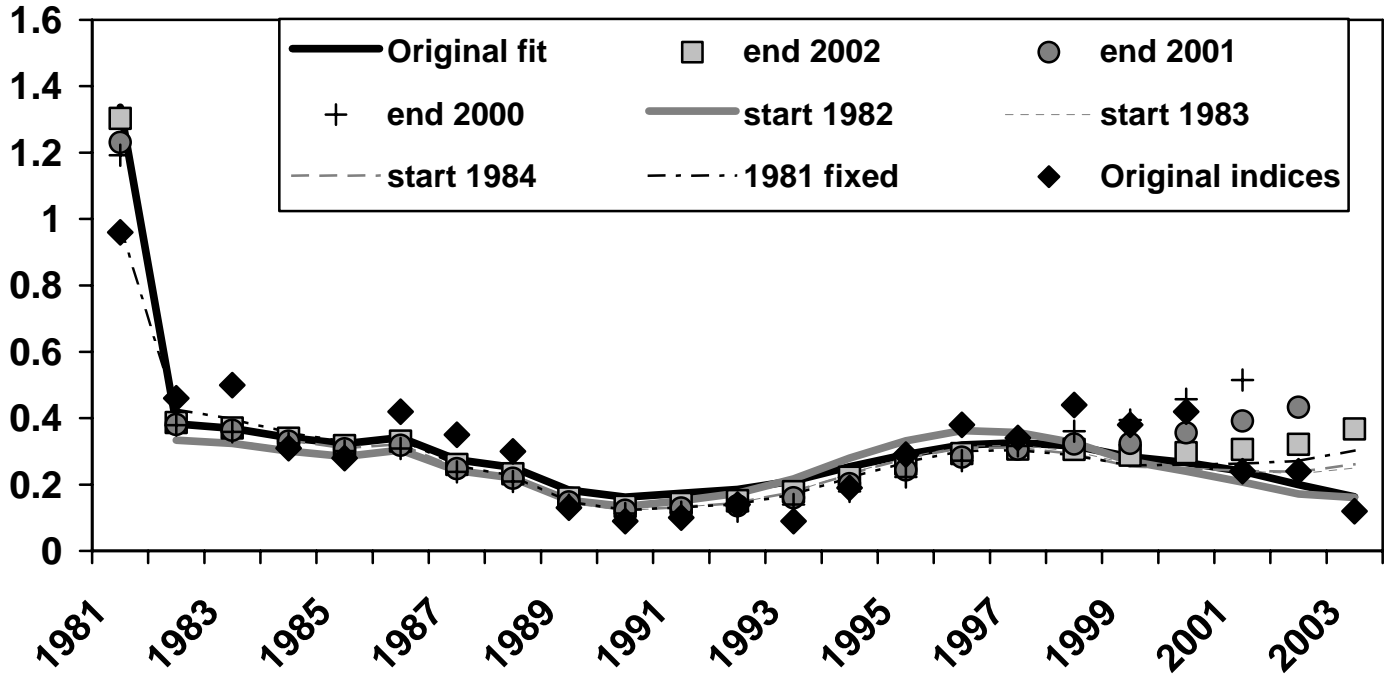


Figure 28. Logistic predator-prey model predicted weakfish biomass consumed per unit of striped bass biomass plotted against weakfish biomass (striped bass functional response) for two time-series. Intersections of predator-prey parameters c and A are indicated for each time-series by open symbols.

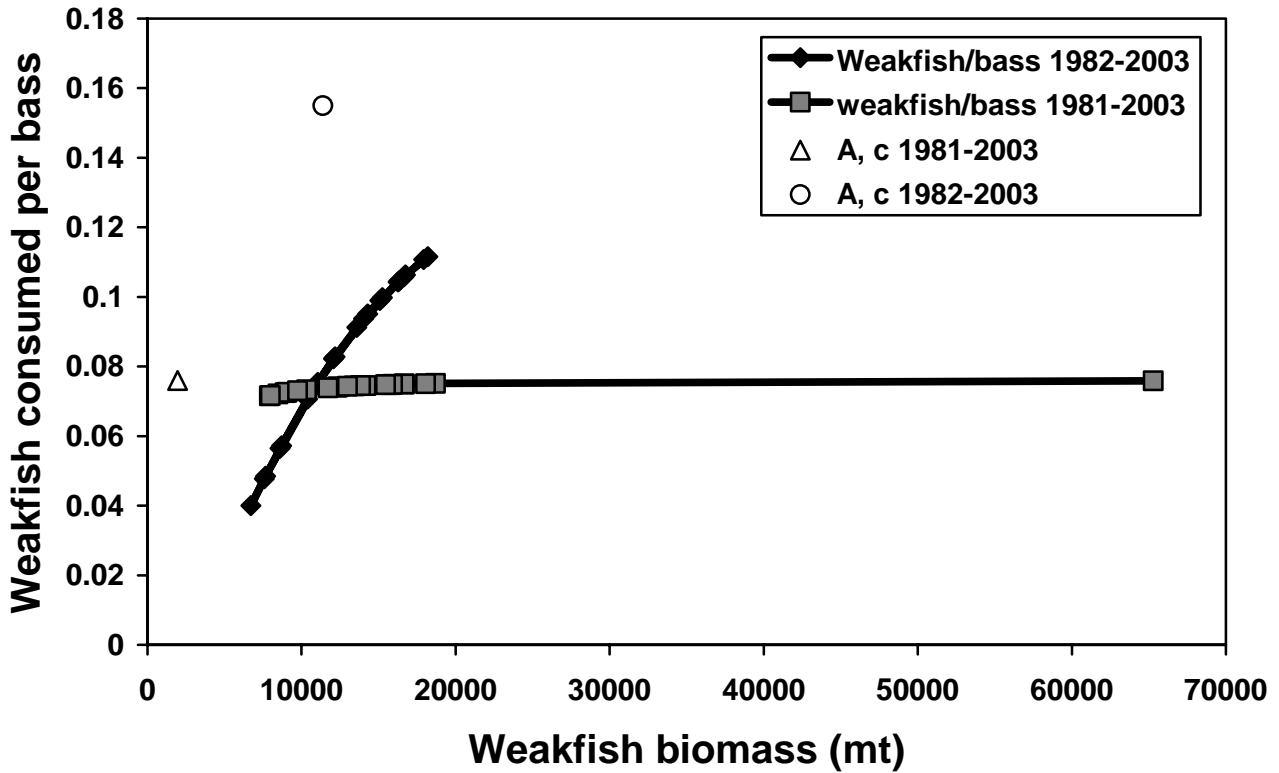


Figure 29. F estimated from rescaled VPA, Schaefer, and logistic Steele and Henderson predator-prey models with the WRI as their basis.

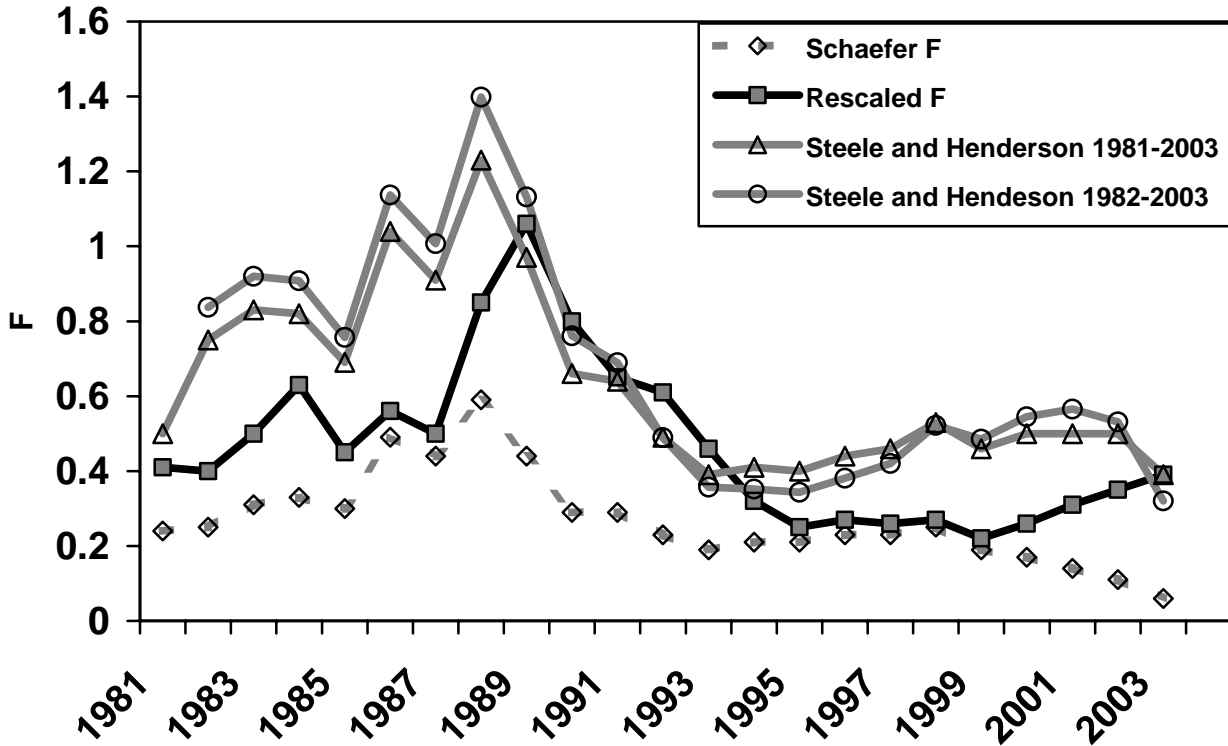


Figure 30. Estimates of M_b from the logistic weakfish-bass predator-prey model applied to two time-series.

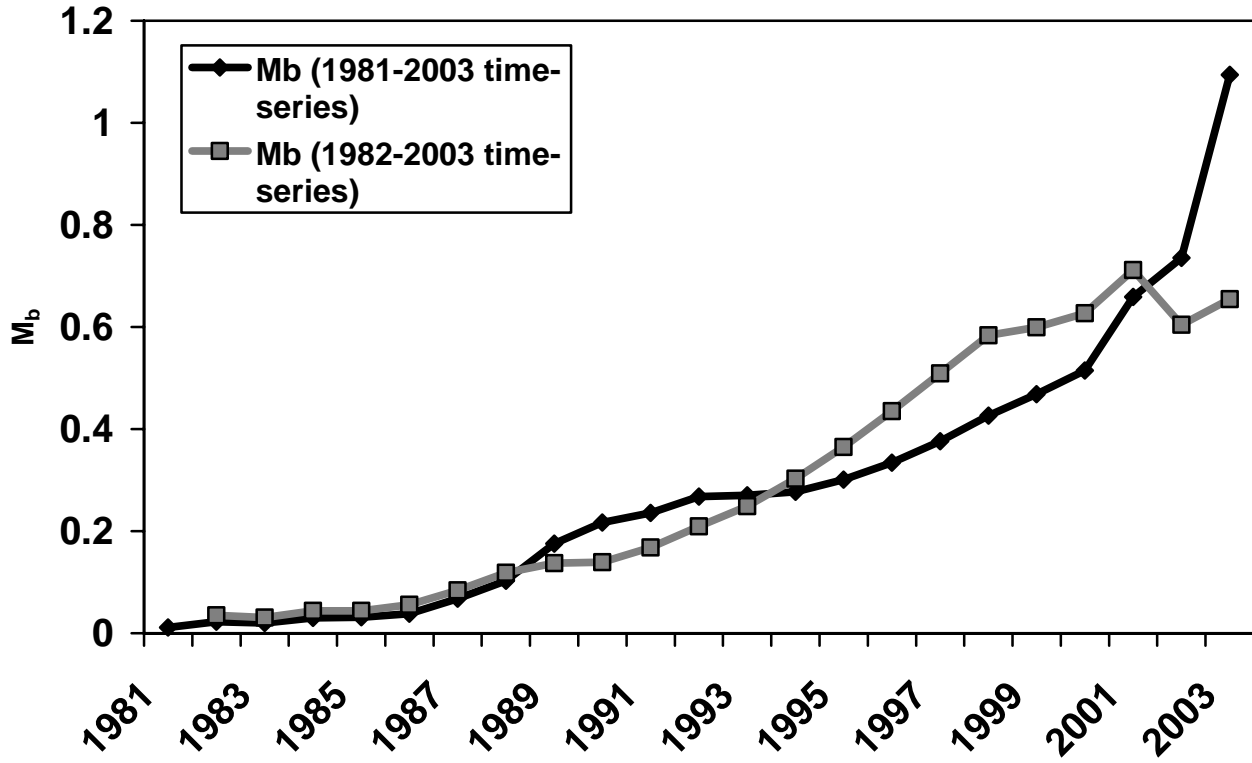


Figure 31. Estimates of Z from ADAPT and from logistic striped bass-weakfish predator-prey models (Steele and Henderson model). Predator-prey model does not include M from sources other than striped bass.

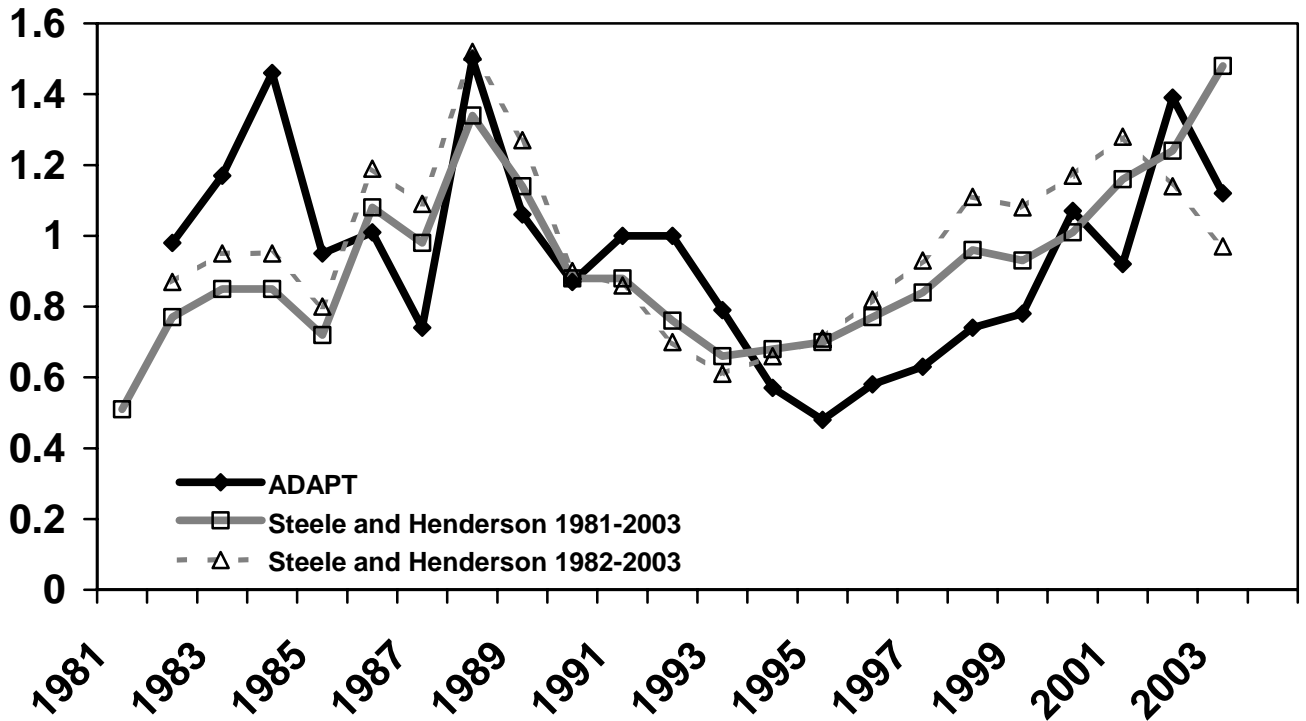


Figure 32. Logistic predator-prey model biomass estimates based on WRI. Figure omits 1981 estimates.

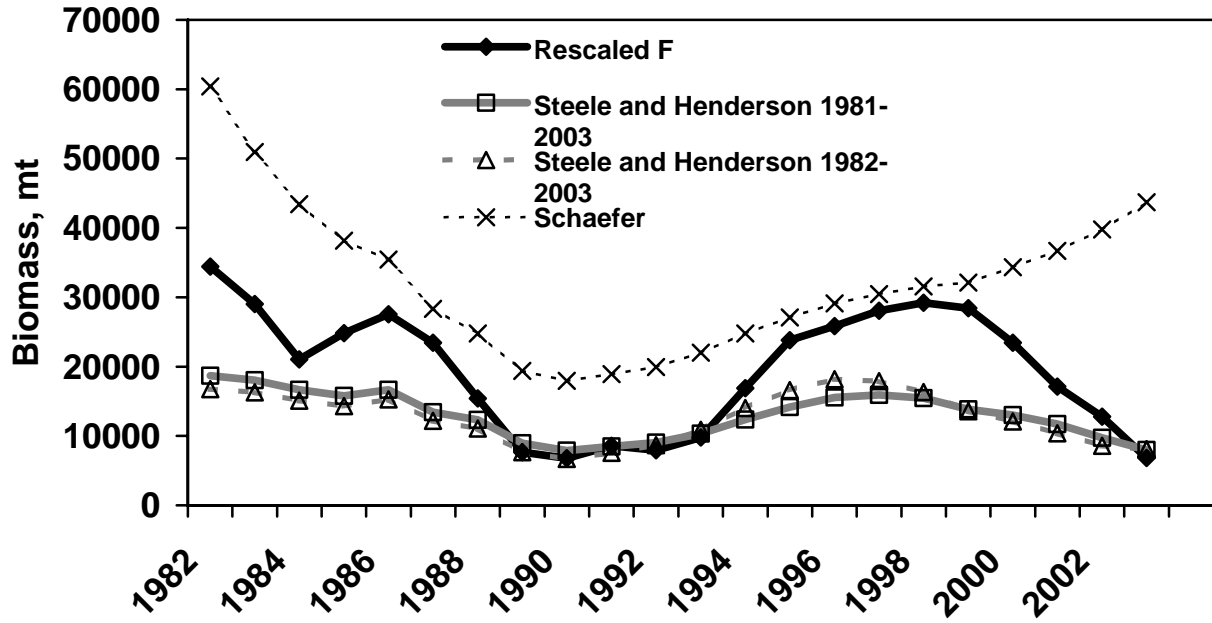


Figure 33. Trends in biomass of weakfish and other candidate alternative predator-competitors for the logistic predator-prey model.

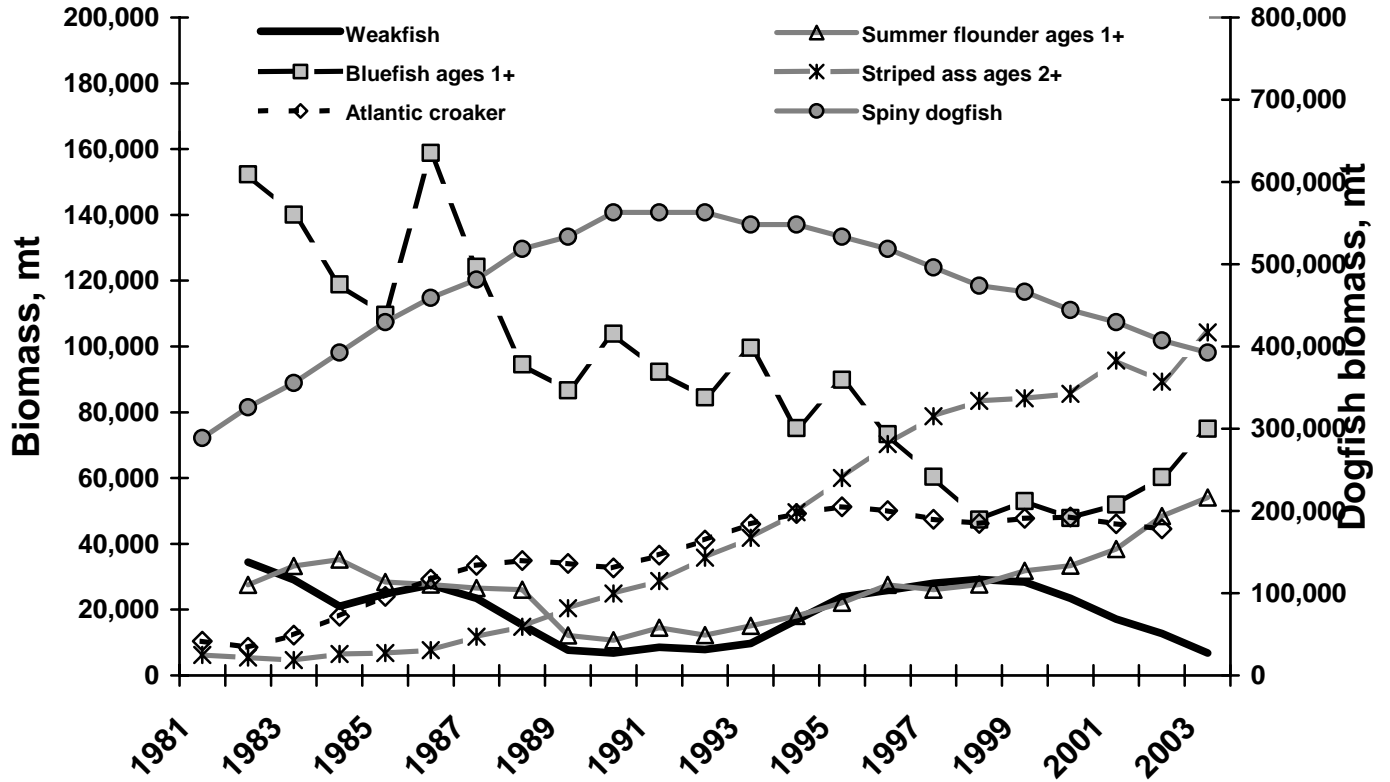


Figure 34. Weakfish biomass predicted from an Atlantic croaker- weakfish logistic predator-prey model and “observed” weakfish biomass from rescaled relative F analysis. Both are based on the same MRFSS catch per effort index for weakfish.

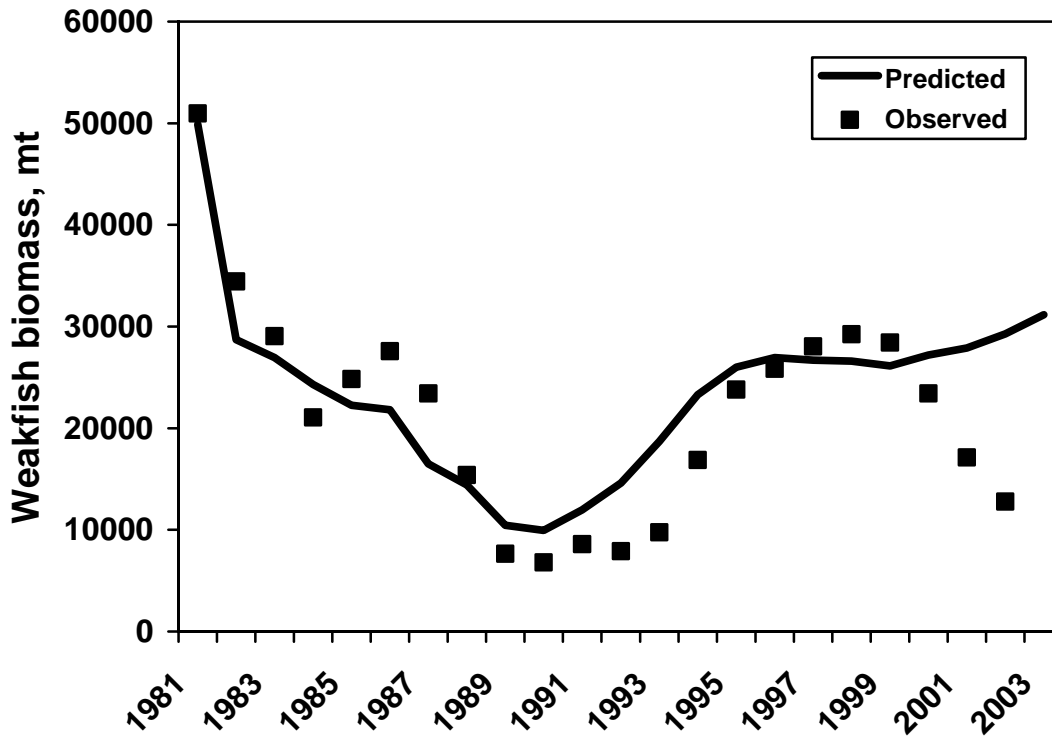


Figure 35. Weakfish biomass predicted from a logistic bluefish - weakfish predator-prey model and “observed” weakfish biomass from rescaled relative F analysis. Both are based on the same MRFSS catch per effort index for weakfish.

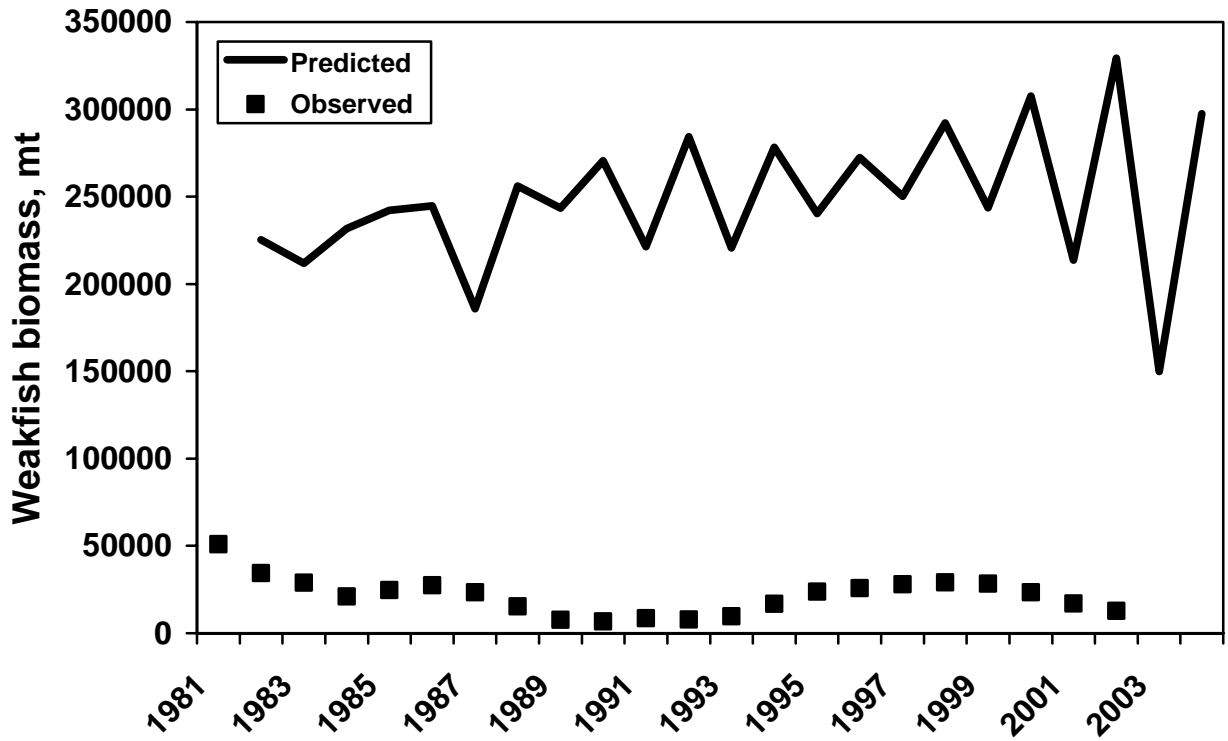


Figure 36. Weakfish biomass predicted from a logistic summer flounder - weakfish predator-prey model and “observed” weakfish biomass from rescaled relative F analysis. Both are based on the same MRFSS catch per effort index for weakfish.

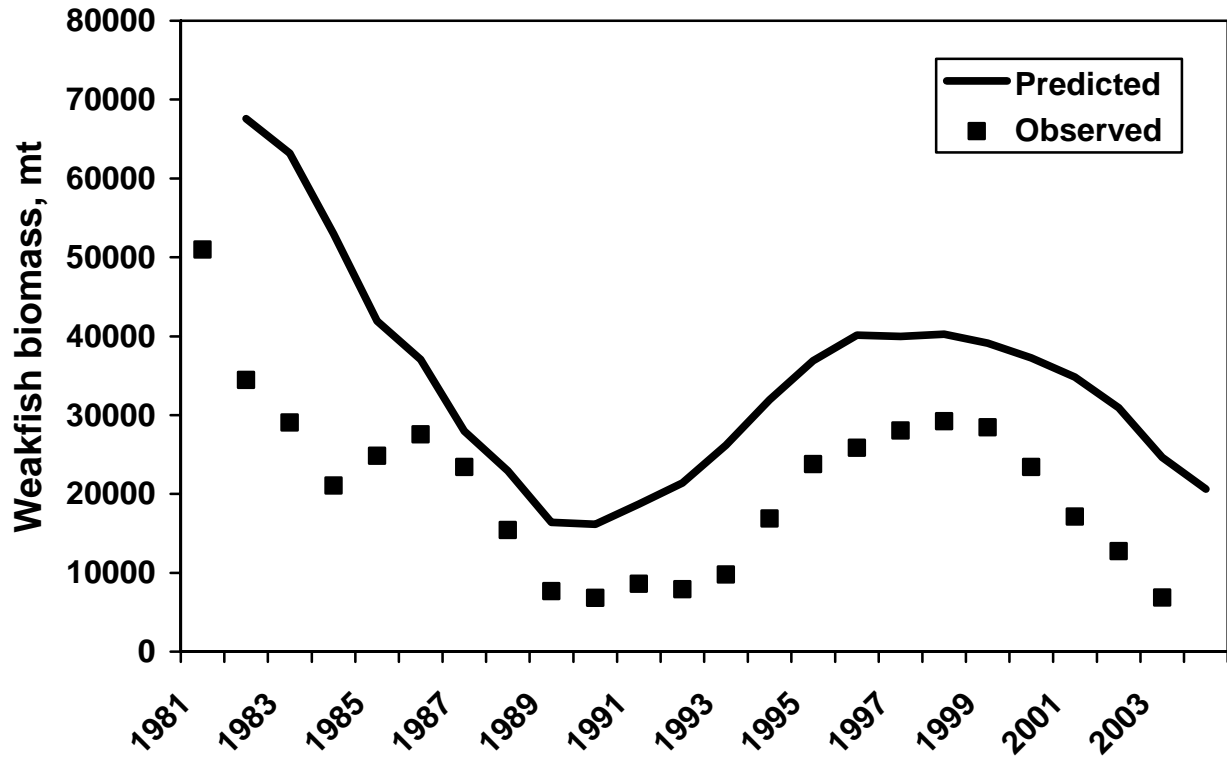


Figure 37. Weakfish biomass predicted from a logistic spiny dogfish - weakfish predator-prey model and “observed” weakfish biomass from rescaled relative F analysis. Both are based on the same MRFSS catch per effort index for weakfish.

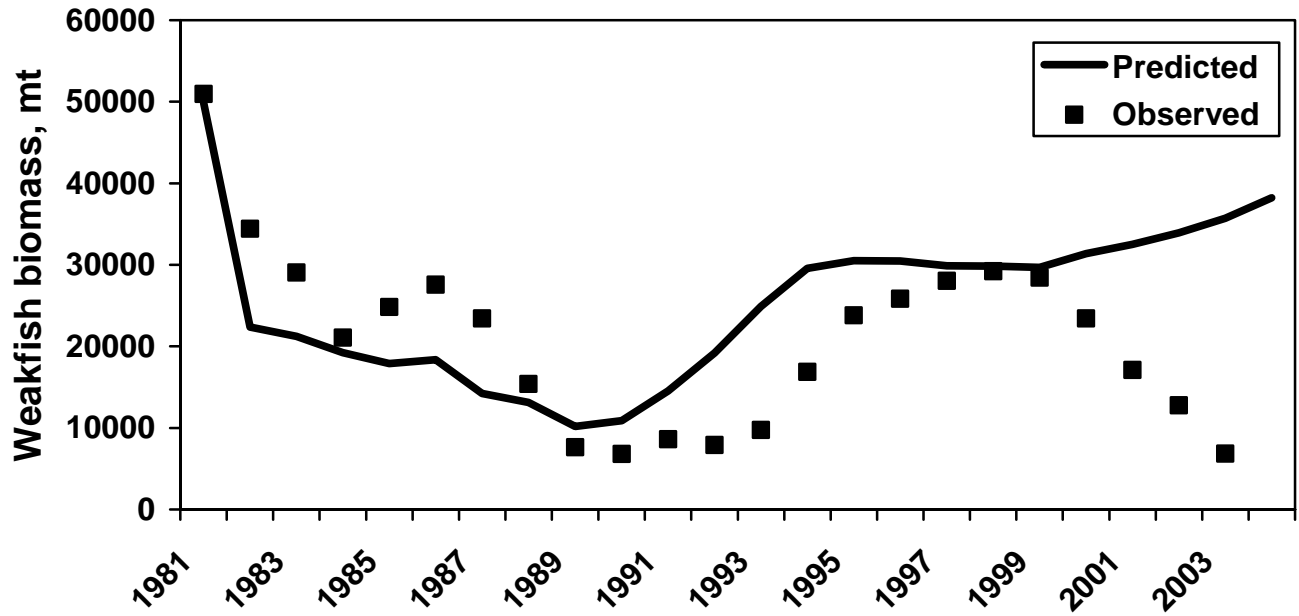


Figure 38. Weakfish biomass predicted from striped bass and summer flounder - weakfish logistic predator -prey model and “observed” weakfish biomass from rescaled relative F analysis. Both are based on the same MRFSS catch per effort index for weakfish.

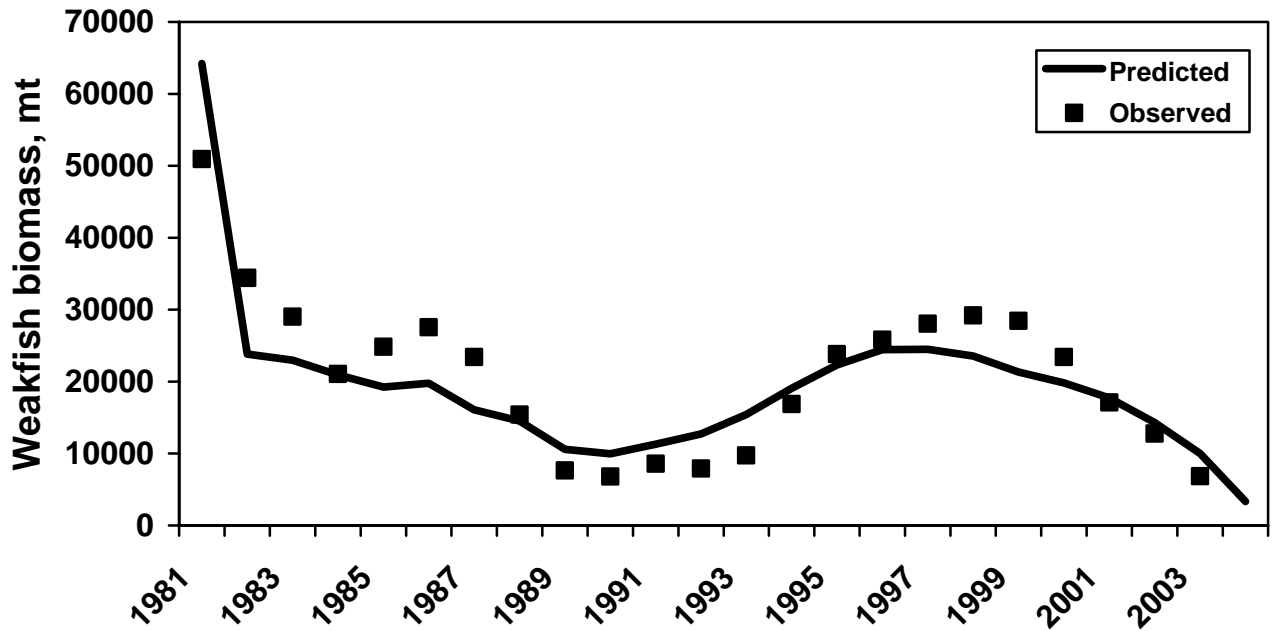


Figure 39. Standardized age 0 weakfish indices from Crecco (2006). Nj (grand mean) indicates general trend.

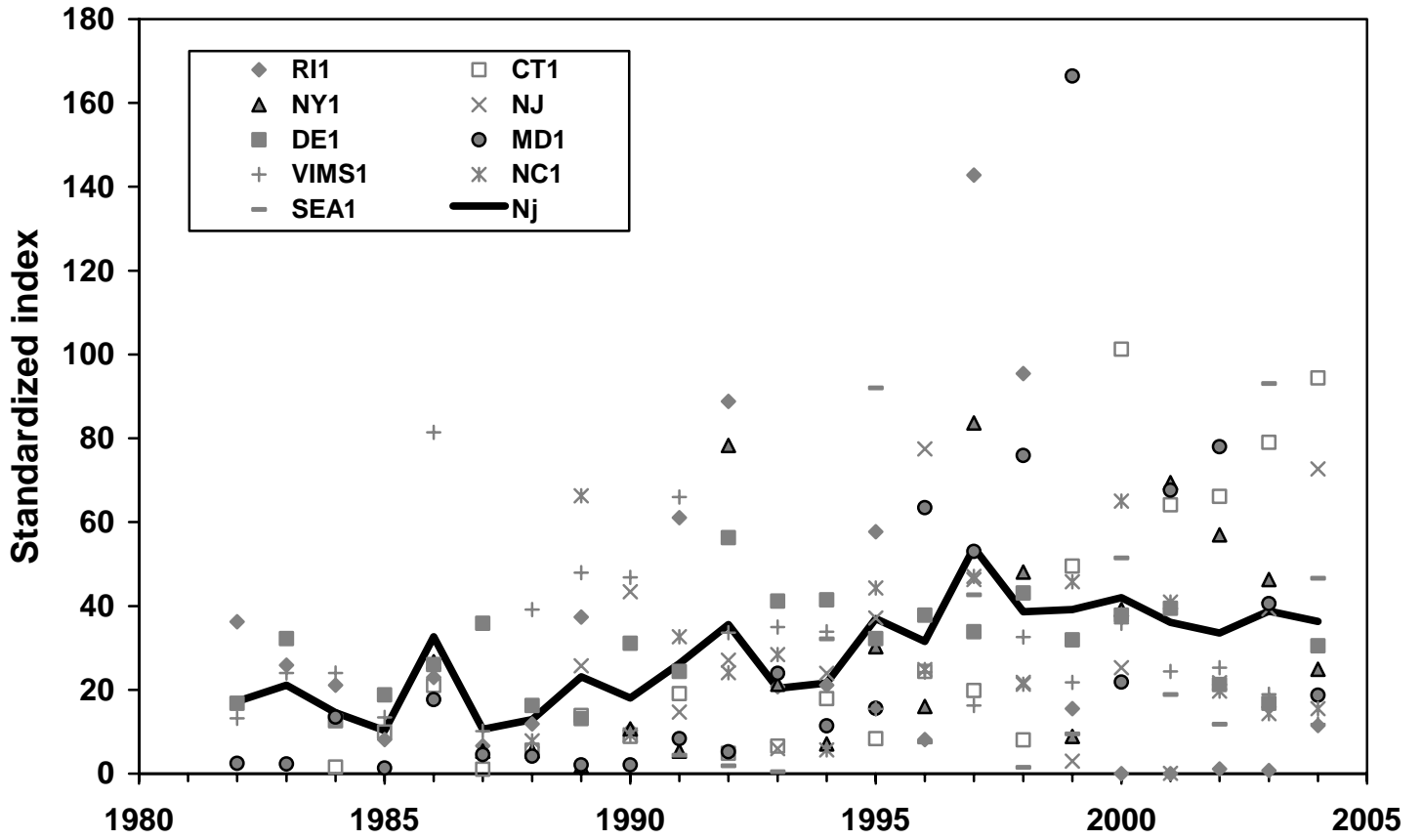


Figure 40. Decreased menhaden : bass ratio corresponded to increased weakfish M attributed to bass competition and predation. Note: rapid rise in M estimates as the ratio becomes asymptotically low in the mid-1990s. Age 0 weakfish mortality index (RZj; Crecco 2006) has been divided by 4 to scale it to the plot.

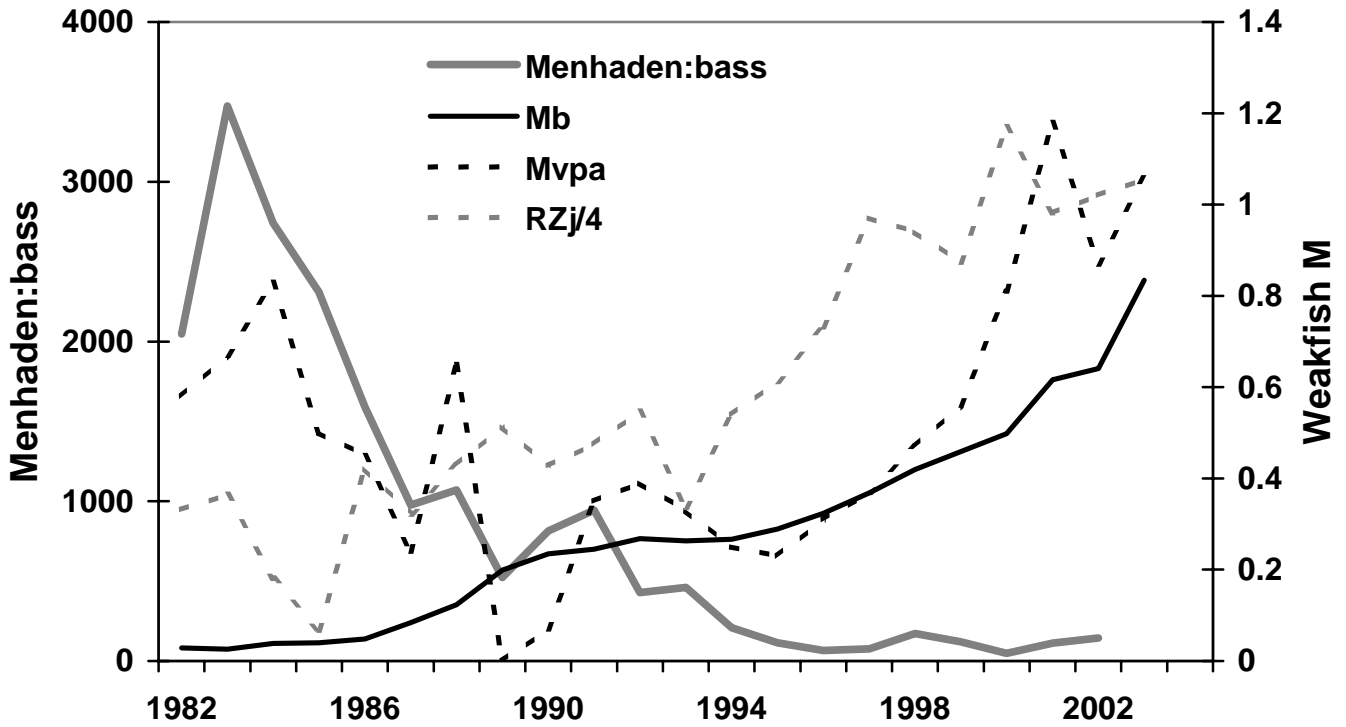
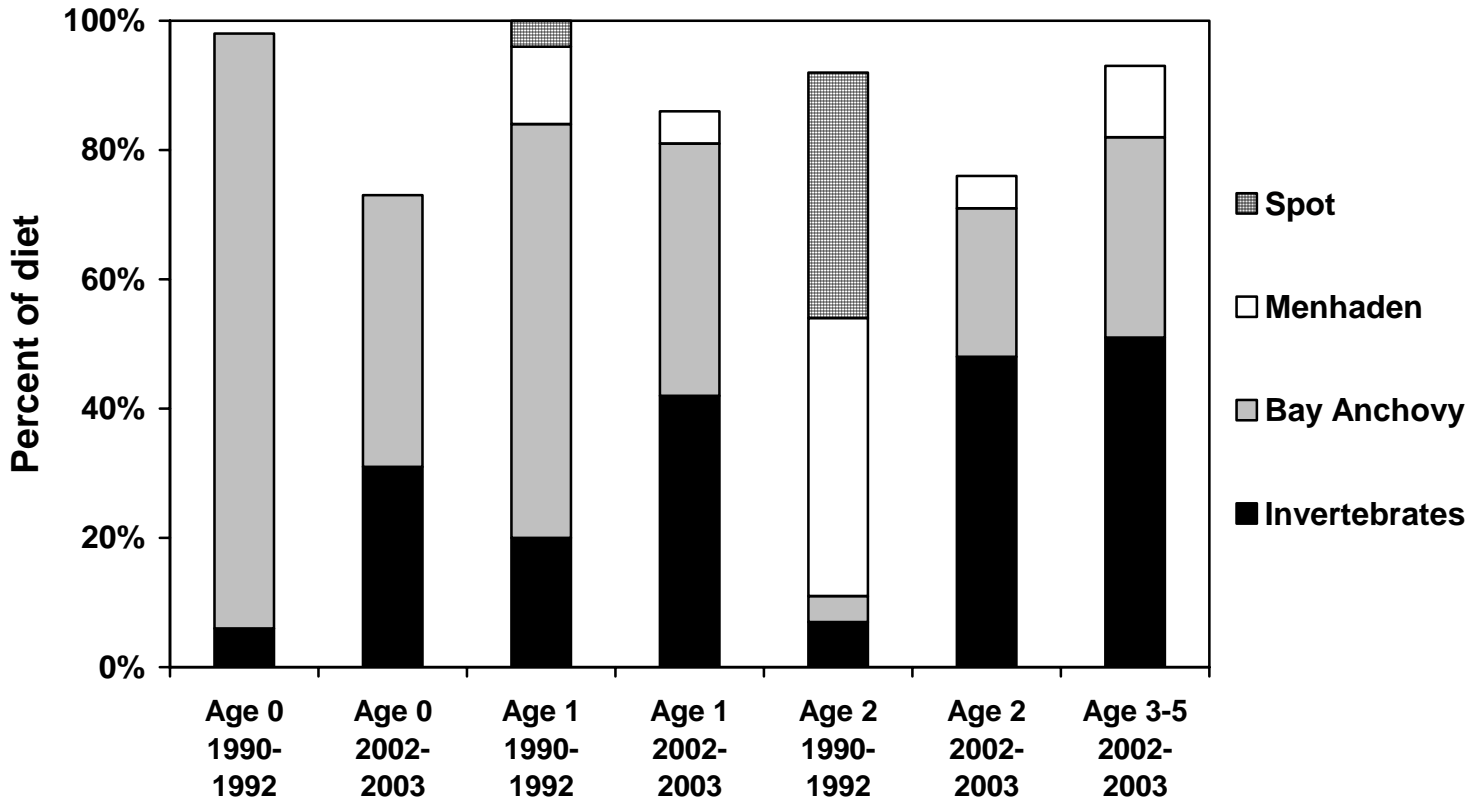
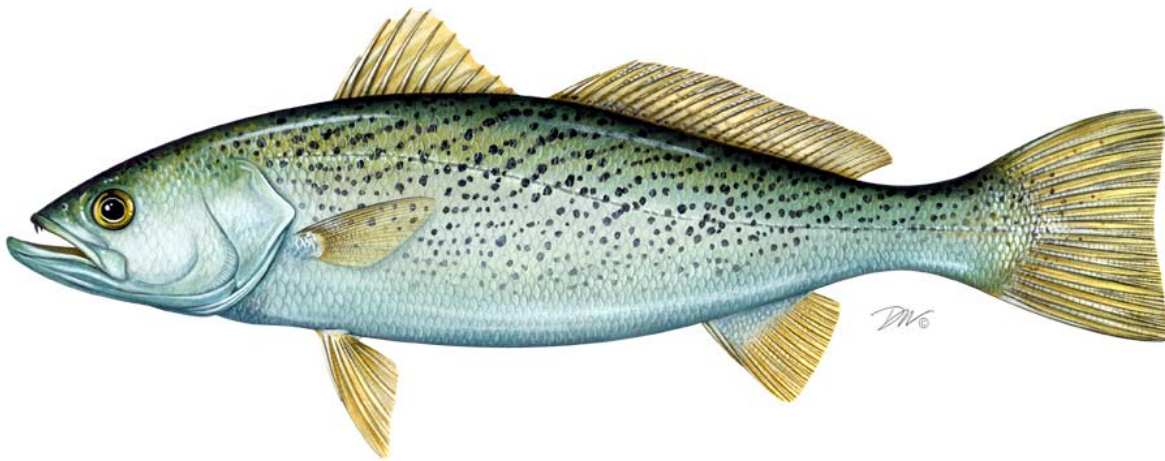


Figure 41. Comparison of age-specific weakfish diets (by weight) in Chesapeake during 1990-1992 and 2002-2003. Data for 1990-1992 were from Hartman and Brandt (1995) and data for 2002-2003 were provided by R. Latour (VIMS).



Atlantic States Marine Fisheries Commission

Terms of Reference & Advisory Report to the Weakfish Stock Assessment Peer Review



April 2006



Healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015

Atlantic States Marine Fisheries Commission

Terms of Reference & Advisory Report to Weakfish Stock Assessment Peer Review

Conducted on
March 20-22, 2006
Providence, RI

Dr. Brian J. Rothschild, Panel Chair, University of Massachusetts – Dartmouth
Dr. Steven X. Cadrin, NOAA/UMass Cooperative Marine Education and
Research Program
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Preface

Summary of the Commission Peer Review Process

The Benchmark Stock Assessment Peer Review Process, adopted in October 1998 and revised in 2002 and 2005 by the Atlantic States Marine Fisheries Commission (ASMFC or Commission), was developed to standardize the process of stock assessment reviews and validate the Commission's stock assessments. The purpose of the peer review process is to: (1) ensure that stock assessments for all species managed by the Commission periodically undergo a formal peer review; (2) improve the quality of Commission stock assessments; (3) improve the credibility of the scientific basis for management; and (4) improve public understanding of fisheries stock assessments. The Commission stock assessment review process includes evaluation of input data, model development, model assumptions, scientific advice, and review of broad scientific issues, where appropriate.

The Commission's Benchmark Stock Assessment Peer Review Process outlines options for conducting an external peer review of Commission managed species. These options are:

1. The Stock Assessment Workshop/Stock Assessment Review Committee (SAW/SARC) conducted by the National Marine Fisheries Service (NMFS), Northeast Fisheries Science Center (NEFSC).
2. The Southeast Data and Assessment Review (SEDAR) conducted by the National Marine Fisheries Service, Southeast Fisheries Science Center (SEFSC).
3. The Transboundary Resources Assessment Committee (TRAC) reviews stock assessments for the shared resources across the USA-Canada boundary and it is conducted jointly through the National Marine Fisheries Service and the Canada Department of Fisheries and Oceans (DFO).
4. An external Commission stock assessment review panel conducted by 3-4 stock assessment biologists (state, federal, university). The Commission review panel will include scientists from outside the range of the species to improve objectivity.
5. A formal review using the structure of existing organizations (i.e. American Fisheries Society, International Council for Exploration of the Sea, or the National Academy of Sciences).

The ASMFC Weakfish Stock Assessment Peer Review Panel (Option 4 from above) met on March 20-22, 2006 in Providence, Rhode Island to review the Weakfish Stock Assessment.

Purpose of the Terms of Reference and Advisory Report

The Terms of Reference and Advisory Report provides summary information concerning the weakfish stock assessment and results of the External Peer Review to evaluate the accuracy of the data and assessment methods for this species. Specific details of the assessment are documented in a supplemental report entitled Weakfish Stock Assessment Report for Peer Review. A copy of the supplemental report can be obtained via the Commission's website at www.asmfc.org under the Weakfish page or by contacting the Commission at (202) 289-6400.

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TERMS OF REFERENCE FOR THE WEAKFISH PEER REVIEW

The ASMFC Weakfish Stock Assessment Peer Review Panel (Panel) does not endorse the recommendations of the ASMFC Weakfish Technical Committee (TC) regarding stock status of weakfish along the Atlantic coast. The TC based its conclusions on a biomass dynamics model that included striped bass predation. The Panel feels that the biomass dynamic model relies too heavily on recreational catch per unit effort (CPUE) data that excludes a portion of the geographic range of weakfish and that the model excludes information on relative stock size from research surveys. The Panel felt that there was insufficient attention to the fact that juvenile survey abundance indices were generally increasing over the time series and too much reliance on the assumption that recent trends in landings directly reflect abundance. The TC did not fully consider the substantial portion of catch being discarded (e.g., commercial and recreational discards increased since 1994 and in 2003 reported discards were greater than total landings). Evidence for predation of exploitable size weakfish by striped bass is weak. Furthermore, if geographic stock structure were considered in analysis of these data, conclusions about stock status may be different than those provided by the TC.

1. Characterize commercial and recreational catch including landings and discards.

There is evidence of stock structure. Nesbitt (1954) indicated at least three stocks from tagging and Thorrold et al. (2001) indicated high fidelity of weakfish to their natal estuary, and there is substantial regional heterogeneity in population dynamics of stock structure. The fact that geographic stock structure has not been included in the analysis may explain the observed annual variation in growth and mortality. Taking account of stock structure would probably reduce uncertainty in the assessment and lead to better stock status determination on a regional basis.

Despite the evidence for stock structure, weakfish along the coast have been assumed to be single stock. All assessments to date have been based on this assumption. In fact, data have been pooled as if there had been a single stock and consequently the appropriate statistical weights among stocks have not been determined.

Commercial landings since 1950 appear to be estimated with moderate precision and landings since 1994 appear to be well estimated; however, given that weakfish are often taken as bycatch and have relatively low value, the overall estimates may be less accurate than the TC believes. The TC felt that the National Marine Fisheries Service (NMFS) Marine Recreational Fisheries Statistics Survey (MRFSS) recreational landings since 1981 are well estimated, but the Panel was concerned that there was substantial uncertainty in the MRFSS estimates. The accuracy of these data may be heterogeneous over regions and time. The panel noted some inconsistencies in the various catch tables, where the total estimated landings from all fisheries are not equal in the various tables that presented total landings.

The discard patterns are not what we would expect given the change in regulations in the mid-1990s. Discards rates varied in recent years (proportion of total commercial catch that was discarded: 1999 – 51%, 2000 – 17%, 2001 – 55%, 2002 – 61%). The high discards rates are a concern for conservation and management of weakfish, as most estimates of discard rates are generally underreported given the change in fishermen behavior when observers are onboard vessels. Commercial discards are estimated with moderate precision to 1994, but there are no direct data in earlier years. Recreational discards are estimated with moderate precision since 1981. Discard mortality is uncertain (assumed variable by commercial gears and assumed 20% for the recreational fishery).

Dedicated age-length keys are unavailable for most portions of the catch (e.g., northern portions of commercial fishery, all recreational catch, some surveys, commercial discards before 1994). Overlap among ages for small fish is worrisome because this probably adds considerable bias in the catch-age information that used in the ADAPT virtual population analysis (VPA; NOAA Fisheries Toolbox Version, 2004. VPA/ADAPT, Version 2.1.9. [Internet address: <http://nft.nefsc.noaa.gov>]). Geographic patterns in growth may confound estimation of catch-at-age. Age-structure is poorly estimated particularly for recreational and commercial discarded portions of the catch.

There are major problems with the calculations of the catch-at-age as indicated by substantial sum of products differences (Catch at age * mean weight at age does not equal the estimates of catch biomass). Abrupt changes in mean weight-at-age data may result from errors in the calculation of the catch-at-age. The changes in mean weight are coincident with a conversion from scale ages to otolith ages. The Panel was not afforded the opportunity to review a statistical analysis of the scale-otolith conversions. In addition, there was no information from fishery independent surveys on changes in growth in the stock assessment report.

There are substantial deficiencies in biological samples (i.e. age and length) in some regions (Table 2A in the stock assessment report). Particularly notable is the absence of sampling in New York or New Jersey; thus, it is impossible to make precise evaluation of this stock without information from this region. It seems likely that larger and older fish migrate further north seasonally and may be less represented in the sampling, given the lack of data from these regions. Estimates of total mortality from sample catch-at-age assume that older fish are fully represented in the samples. Some age-length samples are collected with the intention of obtaining estimates of growth; therefore, they are not representative of age-frequency of the catch (e.g., North Carolina).

In conclusion, deficiencies in stock identification and sampling make this stock assessment very difficult to validate. The age-composition of the catch may not be reliable (e.g., no validation of annuli in otolith based ages, in scale based ages, or in the relationship of otolith to scale based ages); therefore, the results from the ADAPT and subsequent analyses dependent upon these results are suspect, as VPA assumes no errors in catch-at-age.

2. Review adequacy and uncertainty of fishery-independent and dependent indices of relative abundance.

Fishery Independent Surveys

There is a lot of information in the fishery independent surveys, but the information is noisy and difficult to interpret. However, there are consistent long-term trends in the young of year indices among almost all of the surveys. Including all of the survey information into a VPA calibration, in the form assumed by ADAPT, implicitly assumes equal weighting of all surveys regardless of geographic coverage, representativeness, or statistical properties. The TC believed that ADAPT would be able to sort out the different survey variances; however, ADAPT is not capable of doing so. Spatial analysis of fishery and survey data would help to determine what the portions of the resource the data represent. In general, a comprehensive review of the use of state and federal survey fishery independent surveys would benefit ASMFC stock assessments.

The TC evaluated the following four fishery independent surveys: the NMFS fall inshore trawl survey, the New Jersey ocean trawl survey, the Delaware trawl survey, and the SEAMAP trawl survey). Other surveys were included in a graphical analysis of the young of year trends.

Both the TC and the Panel felt the NMFS Northeast Fisheries Science Center's fall inshore survey had the broadest geographic coverage of available surveys, could be used to monitor long-term trends of relative

abundance, and tracked some cohorts well. Concerns with the survey as an indicator of abundance include the interannual variability of the timing of the survey relative to weakfish migration, the fact that the survey only catches young weakfish, catch curve analysis from the survey yields negative estimates of total mortality rates (Z), and confounding temperature effects. The survey has a dedicated age-length key and fish were initially aged by scales before changing to otoliths.

Trends of relative weakfish abundance for the New Jersey ocean trawl survey were consistent with perception of weakfish stock development up to mid-1990s, but were then inconsistent with fishery catch rates. The survey is limited to sampling the coastal ocean of New Jersey. The survey is conducted five times per year in January, April, June, August, and October, but only survey tows from August and October were used to develop the weakfish index and there was a skewed distribution of catches. Age data are available for some years, but in others age data were ‘borrowed’ from adjacent surveys.

Although the Delaware Bay trawl survey sampling area is limited geographically, it is located near the center of weakfish population abundance. The survey is conducted monthly with dedicated age samples, and abundance-at-age patterns are generally consistent (e.g., all Z estimates are positive).

The SEAMAP survey is the southern most survey examined by the Weakfish TC. The survey is conducted in spring, summer, and fall from Cape Hatteras, North Carolina to Cape Canaveral, Florida, but only data from the fall survey from North Carolina were utilized. The abundance-at-age patterns are generally consistent, as evident by positive Z estimates from cohort catch curve analysis, but the index is noisy. Some age data are available from the SEAMAP survey; however, for years when no ages were available, age data were ‘borrowed’ from adjacent surveys.

Fishery dependent indices

Commercial catch per unit effort indices (from North Carolina) were presented to the Panel. Catch per unit effort for both the North Carolina estuarine commercial gears and the North Carolina ocean winter fishery were affected by regulations in 1994 and only cover a small portion of the geographic range of weakfish.

The TC utilized a recreational CPUE based on the NMFS MRFSS using mid-Atlantic private boat CPUE (all trips or trips targeting black sea bass, bluefish, Atlantic croaker, summer flounder, weakfish, and spot). This index includes kept and released catch and, compared to the commercial indices, is less affected by regulations. This fleet historically accounted for a small component of total catch (11% in 1989, 12% in 1990), but accounts for more than 50% in recent years and does not account for trends in abundance outside the fished area (North, South, and offshore). For the recreational fishery, there are no dedicated age data and no lengths are available for the released portion; therefore, ‘borrowed’ age-length keys were applied from commercial and survey data, which could have caused a false consistency with the catch-at-age.

Each fishery dependent index has nuances with catchability (i.e., technological changes), which were not discussed. The mid-Atlantic private boat fleet appears not to reflect abundance. For example, there is a concern that a proportion of the stock migrates out of the mid-Atlantic area and that proportion varies annually, which would affect the catchability of the mid-Atlantic private boat fleet.

The Panel concluded that fishery independent survey data requires a more comprehensive analysis to get a coast wide perception of relative stock size and that fishery dependent indices may not be a reliable indicator of stock size.

3. Review the appropriateness of constant and variable natural mortality (M) estimates in the assessment.

The sampled age-structure indicates extraordinarily high Z estimates during periods of low catch, suggesting that M rates are high. However, there are two additional reasons that Z could be inflated, 1) older ages are truncated by the selectivity of the gear and 2) catches of older fish in areas where fish are not sampled (e.g., New Jersey).

The TC makes the case for increasing Z based on the ADAPT (Run #20 – the preferred run from Stock Assessment Report). All other ADAPT analyses indicate that total mortality is decreasing. The distinguishing feature of Run #20 is that it is tuned solely with the recreational CPUE from the mid-Atlantic private boats. All of the subsequent analyses were dependent upon the results of Run #20. The Panel found the results from the ADAPT and subsequent analyses dependent upon these results are suspect.

Inferences on changes in M were based on nominal fishing effort (in units of private boat day in the mid-Atlantic, $f = \text{catch divided by CPUE}$) scaled to ADAPT estimates of fishing mortality rate from Run #20 over 1987 to 1991 where M was assumed to be 0.25. This short time period may reflect atypical catchability based on migration of weakfish (mid-Atlantic private boat catch was as low as 10% of total catch during this time period, but was much higher in other periods). Changes in M were calculated to be the residual between estimates of fishing mortality rates (F) and Z where the F came from the VPA estimates.

The TC tried to explain the increase in Z as increased predation mortality by striped bass using a production model coupled with a Type III functional response. No statistics were presented that allowed the Panel to evaluate the Steele-Henderson model relative to models that assume constant M. The evidence for striped bass predation on exploitable sized weakfish is weak. The Steele-Henderson model is inappropriate for the interactions among striped bass, weakfish, and menhaden, because it assumes that predation occurs only on fish that are fully vulnerable to the fishing gear and ignores predation on smaller fish.

A more rigorous approach to considering multispecies interactions would be to consider a precise hypothesis and incorporate the appropriate process in the population model. For example, if the precise hypothesis is predation of age-0 and age-1 weakfish by striped bass, then a stock recruitment function with a predation term would be appropriate. Alternatively, if competition with other piscivores were the precise hypothesis, then a competition term in the biomass dynamics model would be appropriate. The Steele-Henderson model applied by the TC would be appropriate for predation of exploitable size weakfish by striped bass. The Panel felt that the evidence for predation of exploitable size weakfish by striped bass is weak. Even if the hypothesis is well founded and the population process equation is appropriate, then accurate data on relative stock size and independent information on total mortality are needed.

If M has increased over time, then the corresponding estimate of F_{msy} would be greater than the estimate of F_{msy} that assumes constant M. If M is truly increasing, but we assume that it is constant, then we would advise for future F estimates that are less than the true F_{msy} . Conversely, if M is truly constant and we assume that it is increasing, we would recommend that future F estimates that are greater than F_{msy} . Therefore, assuming constant M is more risk averse than assuming an increase in M. This is why investigating variability in M is very important.

4. Review the estimates of fishing mortality (F), spawning stock biomass, and total stock biomass for 1981-2004, and characterize the uncertainty of these estimates.

The Panel applauds the effort to examine alternative explanations to the data (e.g., change in M) and appreciates creative solutions to these problems; however, the data are not adequate to test the hypothesis of increasing natural mortality of weakfish. Because this issue is so important along the Atlantic coast, it is worthwhile investigating historical data (i.e., data prior to 1981). The Panel encourages continued exploration of trophic dynamic models, as being conducted for other fishery resources with reduced productivity (e.g., the reduced productivity of cod throughout the north Atlantic). Members of the Panel recommended conducting tagging studies to directly measure changes in natural mortality rates.

The Panel was reluctant to recommend the use of any age-based analysis, given the questions about the reliability of the catch-at-age data. Refer to the discussion under Term of Reference 1 and 2 regarding poorly sampled catch-at-age and noisy survey indices.

There is some optimism for weakfish stock status, as recruitment has not declined in any portion of the range. Despite our uncertainty in stock productivity there is no apparent decrease in recruitment. In fact, evidence suggests that there have been recent increases in recruitment over the last 15 years. These differences in recruitment are correlated with substantial decreases in weight-at-age, which suggests that there is some limitation of weakfish carrying capacity. However, since no estimates of weight-at-age were provided for the survey data, this hypothesis was difficult to evaluate.

Estimated landings decreased since the early 1990s, but estimated discards increased since 1994. All indices of stock size indicate a positive response to the regulations during the early and mid 1990s. Since 1994, abundance indices are equivocal; some show an increase (NEFSC Fall; Connecticut and SEAMAP young-of-year surveys), some show no trend (New Jersey), and others a decrease (mid-Atlantic recreational CPUE) (see Figures 1 and 2).

The Panel is uncertain of the direction of Z, F, and M estimates, and, as a result, it is difficult to choose among alternative explanations of observed changes. Despite the uncertainty of recent indices of stock size, it appears that current regulations should be continued or strengthened to decrease discarded catch until a stronger indication of population response (decline or recovery) is seen. The Panel recommends that the ASMFC engage in a comprehensive monitoring program, including the estuarine and ocean environment that would help address short falls in stock assessments of weakfish and other species.

5. Review the estimated biological reference points, as appropriate.

For the reasons stated in Term of Reference 3, reference points from multispecies biomass dynamics model are not well estimated. Neither stock and recruitment analyses nor growth studies were presented to the Panel. In lieu of an accepted analytical assessment, conventional reference points are irrelevant and fisheries management needs to rely on historical measures of relative abundance (e.g., a period of high stock abundance from research surveys). Management needs to be adaptive to see if there is a response in the stock.

6. Review stock projections.

- **Review the projection of impacts on the stock of recent estimated rise in fishing mortality.**
- **Review the projection of stock response to reductions in fishing mortality given the estimated increase in natural mortality.**

The only projections presented to the Panel were based on the hypothesis of increased natural mortality. For the reasons stated in Term of Reference 3, these projections are not a reliable basis for management. In lieu of an alternate analytical model, projections were not available.

7. Make research recommendations for improving data collection and assessment.

The Panel makes the following research recommendations to improve weakfish data collection and stock assessment efforts:

Catch

- Sample age-composition of commercial catch throughout the range of the fishery.
- Continued observer coverage to estimate discards.
- Partition catch and catch samples by stock.
- Make a substantial investment in monitoring and assessment.
- Develop statistically efficient measurement of catch.
- Develop estimates of discard rates in the commercial fishery.
- Collect information on fishing effort (e.g., number of fishing vessels, number of units, and bycatch by gear).

Aging

- As recommended in SARC 40, further research should be conducted on growth (i.e. validation of scale to otolith conversion).
- Improve regional growth estimates, including studies of density dependent growth.
- Conduct a detailed analysis of the development of the age-length key.

Stock size indices

- Evaluate selectivity of gears (to improve estimates of Z).
- Conduct tagging studies to directly measure changes in natural mortality rates.
- Conduct a detailed analysis of the young of year surveys and the interrelationship among YOY surveys, and the design of a future comprehensive YOY survey.
- Focus on the development of very high resolution catch per unit of effort time series from among the various regions.
- Evaluate the effect of interannual variation on distribution and catchability.
- Conduct a comprehensive review of the use of state and federal survey fishery independent surveys for use in all ASMFC stock assessments.
- Development of analytic methods is needed to determine the status of the resources, which absolutely requires the development of more reliable data sets. Discounting a survey does not imply that it is not necessarily informative about the sub-stock in that region.
- Conduct a comprehensive monitoring program, including the estuarine and ocean environment. Such a program would address short falls in stock assessments of this and other ASMFC managed species.
- Conduct a comprehensive examination of the assemblages of species in a historical construct.

Growth

- A detailed analysis of factors affecting somatic growth should be carried out using data from research surveys if at all possible.

Advisory Report

Status of Stocks

The catch has decreased since the early 1990s. All indices of stock size indicate a positive response to the regulations enacted in Amendment 3 to the Weakfish Fishery Management Plan. Since 1994, survey indices are equivocal; some show an increase (NEFSC Fall, YOY – Connecticut, New Jersey, SEAMAP), some no trend (New Jersey), and others a decrease (Mid-Atlantic recreational CPUE).

Trends in mortality are uncertain. Despite the uncertainty of recent indices of stock size, it appears that current regulations should be continued or strengthened to decrease discarded catch until a stronger indication of population response (either a decline or a recovery) is seen.

Stock Identification and Distribution

Weakfish are part of the Family Sciaenidae or “drums.” Weakfish are found along the Atlantic coast with its primary range from North Carolina through southern New England and have been reported as far south as Florida and as far north as the Gulf of Maine. Weakfish are primarily estuarine and inshore oceanic inhabitants that migrate seasonally. In the autumn, they migrate south to North Carolina or Virginia coastal waters, where they overwinter. In spring weakfish move northward and inshore into estuarine spawning grounds including Pamlico Sound in North Carolina, Chesapeake and Delaware Bays and smaller coastal estuaries up through Gardiners Bay on eastern Long Island, New York and as far north as Long Island Sound and Narragansett Bay in Rhode Island.

The ASMFC Fisheries Management Plan considers weakfish to be a unit stock, based on genetic analysis (Graves et al. 1992). Recent work however, using geochemical composition of otoliths, found weakfish homed to their natal estuaries (Thorrold et al. 2001). Earlier tagging work performed in the 1930s also indicated that weakfish returned to their natal estuary (Nesbit 1954).

Management Unit

The weakfish management unit spans the East Coast of the United States from Massachusetts through Florida.

Fishery Description and Landings

Landings from 1950-2003 peaked in 1981. During the period from 1982-2004, landings were highest at the beginning, totaling about 12,500 mt in 1982. Landings declined in 1989 through 1993. An increase in landings began in 1995. Landings then peaked at above 5,000 mt in 1998. Weakfish landings began to decline again in 1999, which continued through 2004 to record low level. Trends of strong declines in recent years are seen in landings of weakfish by both the commercial and recreational sectors.

In the winter, overwintering aggregations off North Carolina are harvested by gill nets and fly net trawlers. Virginia and New Jersey also comprise an important component of commercial landings. During their annual migration, weakfish are harvested by a variety of estuarine gears, including gill nets, pound nets and floating traps, seines and hand lines. In the fall migration along the coast, weakfish aggregations are targeted with trawls and gill nets. In 1999, for instance, gill nets caught the largest share of the commercial landings, 48%, while trawl landings were second at 27%, followed by pound nets at 20% and haul seine at 5%.

Patterns in recreational total catch and harvest changed after imposition of minimum sizes and creel limits in 1995. After this time, total catch (A + B1 + B2 of the MRFSS) became significantly larger than harvest. This suggests that the regulations had a significant impact on recreational harvest, causing a large increase in live releases (discards).

In 1993, Virginia implemented a commercial fishermen mandatory reporting system, whereby each harvester reports daily activity (harvest, effort) on a monthly basis. This program replaced a voluntary buyer-reporting system. In 1994, North Carolina mandated trip-level reporting by licensed dealers for all commercial landings that requires dealers to complete a trip ticket for each transaction with a fisherman. These reports must be submitted monthly to the North Carolina Division of Marine Fisheries (NCDMF). Previously, the state had a voluntary reporting system from 1978-1993 obtained through a NMFS/NCDMF cooperative statistics program. In addition, NMFS implemented a mandatory electronic reporting system in 2004 for all federally permitted seafood buyers.

There are no explicit commercial discard estimates after 2002 and no information on the operation of the fishery. Only total catch and landings were provided in the Stock Assessment Report. The Panel had to calculate discards from total catch and landings. This information is crucial to understand the dynamics of fisheries impacts on the stock. In addition, the Panel found it important to consider how the “scrap” fishery affects commercial discard rates or biases the results of a production based model that assumes fixed selectivity.

The high discards rates pose an additional concern for conservation and management of weakfish, as most estimates of discard rates are generally underreported given the change in fishermen behavior when observers are onboard vessels.

Data and Assessment

In lieu of an accepted analytical stock assessment, status determination and advice are based on a descriptive summary of catch and survey data.

Biological Reference Points / Fishing mortality

For the reasons stated in Term of Reference 3, reference points from multispecies biomass dynamics model are not well estimated. Neither stock and recruitment analyses nor growth studies were presented to the Panel. In lieu of an analytical assessment, conventional reference points are irrelevant and fisheries management needs to rely on historical measures of relative abundance (e.g., a period of high stock abundance from research surveys). Management needs to be adaptive to see if there is a response in the stock.

Recruitment / Spawning Stock Biomass

There is some optimism for weakfish stock status. Despite uncertainty in stock productivity, recruitment has not declined in any portion of the range. In fact, evidence suggests that there has been a recent increase in recruitment over the last 15 years.

Sources of Information

- Graves, J. E., J. R. McDowell and M. L. Jones. 1992. A genetic analysis of weakfish *Cynoscion regalis* stock structure along the mid-Atlantic Coast. Fishery Bulletin 90:469-475.
- Nesbit, R. A. 1954. Weakfish migration in relation to conservation. Special Scientific Report - Fisheries No. 115. Fish and Wildlife Service, U. S. Dept. of the Interior.
- Thorrold, S. R., C. Latkoczy, P. K. Swart and C. M. Jones. 2001. Natal homing in a marine fish Metapopulation. Science 291: 297-299.

Figures

Figure 1. Relative abundance indices for age-0 weakfish from various fisheries independent surveys. The overall mean shows an increasing trend, but some show declining trends in recent years (e.g., VIMS) and others show increasing trends.

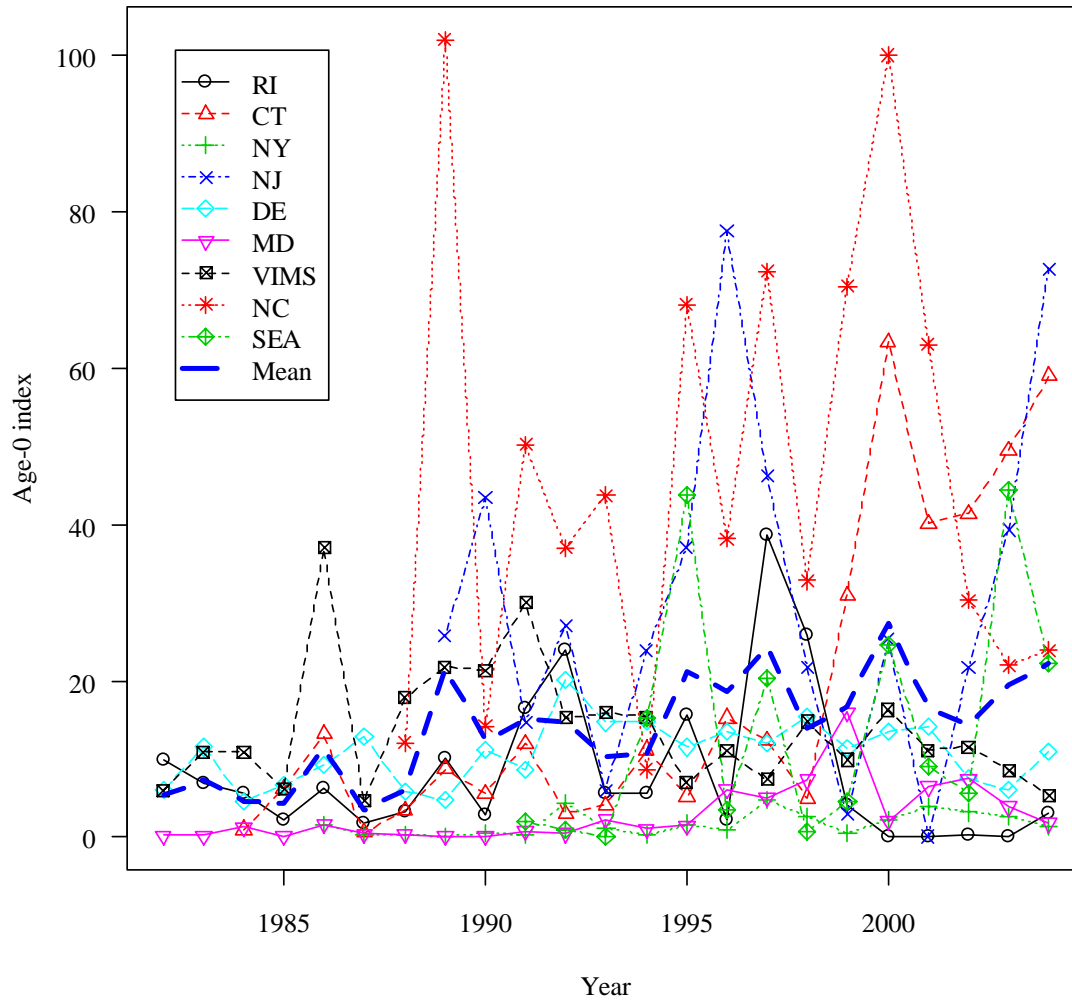
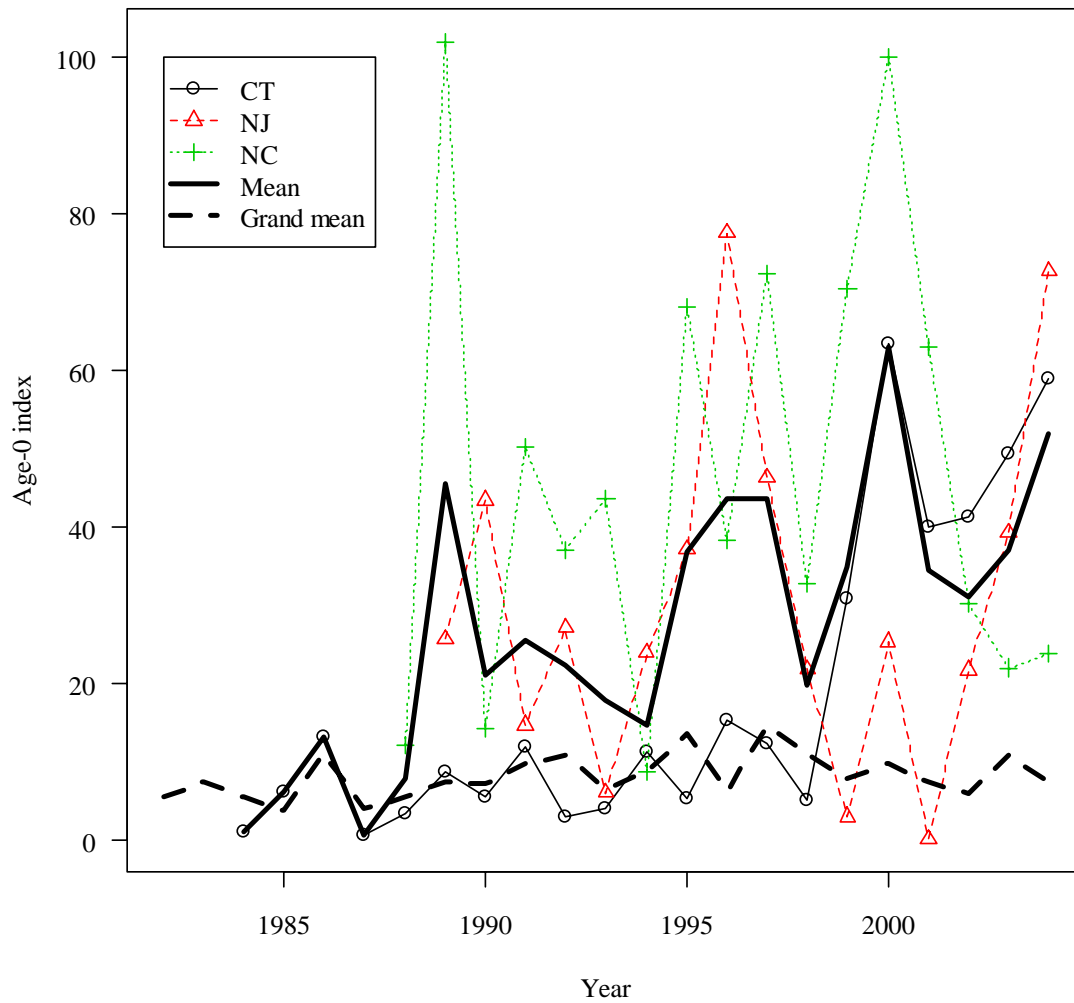


Figure 2. Age-0 abundance indices from the Connecticut (CT), New Jersey (NJ), and North Carolina (NC) fisheries independent surveys. The dark solid line is the mean trend of the CT, NJ, and NC surveys, and the dashed line is the trend of all other surveys shown in Figure 1. These trends suggest an increase in age-0 recruits to the fishery, but are inconsistent with the trends in the recreational CPUE index that was used for tuning the ADAPT VPA.



**Supplemental Material Requested by the Weakfish Management Board
from the Weakfish Technical Committee following Peer Review**

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Implementation of 40th SARC Recommendations

Weakfish Technical Committee
June 2006

Implemented Items: January 2005 – February 2006.

Commercial and recreational data should be examined with regard to its precision and accuracy, both in terms of absolute estimates of catches and its age composition.

Aggregate catches:

1. Commercial catches are a census, so precision cannot be estimated. As with any species' landings, we assume they are accurate.
2. Recreational catch precision is estimated by MRFSS (PSE) and is acceptable. NRC (2006) review indicates serious shortcoming in estimates (any species) that may make accuracy doubtful.
3. Discard estimates and their precision were estimated by De Silva (2004). Precision is sometimes poor. Accuracy cannot be addressed.

Age composition of catches:

It is unclear how precision and accuracy of age structure could be addressed. Sampling of age composition from areas not well covered in the past (NY and NJ) could address a major TC concern about accuracy of CAA if an age-structured assessment is pursued.

Evaluate information content of input data regarding year-class strength.

1. Year-class catch curves of survey data were conducted and indicated only the DE large trawl survey provided suitable consistency (Kahn et al. 2006).
2. Bubble plot of CAA indicates year-classes can be followed over time (D. Kahn DEDFW, personal communication; presented to 2006 peer-review and Board).
3. Gulland-type cohort analysis of CAA provided estimates of N and F (Kahn et al. 2006).

Validate ageing.

Ageing of weakfish by otoliths has been validated in the primary literature (Lowerre-Barbieri et al. 1994). Further work for the TC was summarized by Wenner (1999).

Develop age aggregated indices where CAA inconsistencies are problematic.

1. Age aggregated biomass indices for trawl surveys were developed (Uphoff 2005; summarized in Kahn et al. 2006), evaluated, and accepted by TC.
2. Size structure indices were developed (Proportional Stock Density indices or PSD's (Uphoff 2004; summarized in Kahn et al. 2006), evaluated, and accepted by TC.
3. Age aggregated MRFSS-based recreational index was developed (Crecco 2005a; 2005b; Brust 2005), evaluated, and accepted by TC.

Research into divergence of trawl survey indices and MRFSS index.

1. Aggregated biomass indices evaluated (Uphoff 2006).
2. Correlation analysis indicates that MRFSS, DE, and NJ are significantly correlated. NEFSC significantly correlated with NJ and marginally with DE. MRFSS and NEFSC not correlated. MRFSS index was not "odd man out" (Uphoff 2006).
3. Inclusion of NEFSC with NJ and DE leads to most optimistic status.
4. Substitution of MRFSS for NEFSC indicates declining status.
5. PSD (length-frequency) analysis indicates proportion exploitable in NEFSC too small to estimate reliably (Uphoff 2004). NJ and DE sample sizes provide reliable, precise estimates, and are well correlated. MRFSS CPUE well correlated with PSD.
6. Assessment of index catchability (NEFSC, NJ, DE, MRFSS) and converged VPA biomass indicated MRFSS catchability was most consistent (Uphoff 2006). Survey catchability was much more variable and inverse relationships with biomass were suggested (Uphoff 2006).

Review ecological and other explanations for possible decline of weakfish.

1. Food web dysfunction was explored by Crecco (2005a; 2005c), Kahn et al. (2006), and Uphoff (2006).
2. Selected climatic variables (mean annual water temperatures (C) and annual deviations in the North Atlantic Oscillation index) were tested (Kahn et al. 2006).

Items Not Implemented

1. *Evaluate basic data for suitability of CAA approach.* The 30th SARC considered data adequate for CAA analysis, but they were not aware of the details on how the fisheries were sampled (i.e., NJ and NY missing samples). A detailed re-analysis of basic data was not started due to time constraints. The 2006 review casts doubt on applying an age-structured approach.
2. *Survey catch rates at age should be evaluated in space and time for consistent year-class signals.* Detailed spatial analysis has not been conducted on an age-structured or aggregated basis. TC may not have spatial analysis (GIS) expertise.
3. *Compare survey distributions to observed changes in pattern of landings to try and explain inconsistencies in trends in different series.*

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- Lowerre-Barbieri, S. K., M. E. Chittenden, and C. M. Jones. 1994. A comparison of a validated otolith method to age weakfish, *Cynoscion regalis*, with the traditional scale method.
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Summary of Weakfish Technical Committee (TC) agreements and disagreements with the 2006 peer-review and the record of implementing recommendations of the 40th SARC

July 2006

Overall, the TC agreed with major statements in the review 9 times and disagreed 14; there were 2 agreements and 12 disagreements on main issues (top 4). Eight issues were identified by the 40th SARC and all but 2 were implemented by the TC. “TC Mean Rank” is the average rank of importance each TC member assigned to general issues in the peer-review (1 = most important & 9 = least). Columns labeled “Statements TC agrees” or “Statements TC disagrees” indicate the number of major statements in the peer-review relating to the general issue that the TC agreed or disagreed with; agreements or disagreements were unanimous among TC members. Number of disagreements does not constitute importance. Letters in “Agree statement & details” or “Disagree statement & details” columns refer to specific statements by the peer-review panel that can be found in Section 1 (pages 2-4 of this report). These letters also reference where details of TC agreements or disagreements can be located (Section 2, pages 5-18). “SARC 40 concern” indicates an issue present in 40th SARC panelist documents. NEFSC is the Northeast Fisheries Science Center index and MRFSS is the recreational catch index. CAA = catch at age.

TC Mean Rank	2006 Peer Review General Issue	Statements TC agrees	Statements TC disagrees	Agree statement & details	Disagree statement & details	SARC 40 concern	SARC 40 Implemented
1.4	Indices (NEFSC and JI's useful, MRFSS not)	1	5	P	K, L, M, N, V	Yes	Yes
3.2	Uncertain direction of Z, F, M, biomass		1		W		
4.1	Food web / ecological explanations Landings not accurate	1	5	O	Q, R, S, T, U	Yes	Yes
4.4	(recreational, commercial, & discards)		1		D	Yes	Yes
5.6	CAA matrix problem - missing samples	3	1	C, E, H	F	Yes	No
5.9	Evidence of stock structure	2		A, B			
5.9	Growth - survey estimates not provided	1		J			
5.9	Growth - no validation		1		G	Yes	Yes
6.4	Spatial analysis could help	1		I		Yes	No
Sum for 2006 peer review		9	14				
SARC 40 Issue not in Peer-Review							
	CAA - evaluate indices					Yes	Yes
	Research divergence of trawl and MRFSS indices					Yes	Yes

Section 1: Specific Issues or Statements in the 2006 Peer Review Summarized in Table 1 Agree details or Disagree details columns. List is in order of appearance in the peer review document.

- A. There is evidence of stock structure.*
- B. Taking account of stock structure would probably reduce uncertainty in the assessment and lead to better stock status determination on a regional basis.*
- C. Dedicated age-length keys are unavailable for most portions of the catch (e.g., northern portions of commercial fishery, all recreational catch, some surveys, commercial discards before 1994).*
- D. ... given that weakfish are often taken as bycatch and have relatively low value, overall accuracy of landings may be less than TC believes.*
- E. Missing CAA samples (northern region), overlap of lengths-at-age, and geographic patterns in growth may confound estimation of CAA.*
- F. There are major problems with the calculations of the catch-at-age as indicated by substantial sum of products differences (Catch at age * mean weight at age does not equal the estimates of catch biomass).*
- G. No validation of annuli in otolith based ages, in scale based ages, or in relationship of otolith to scale based ages.*
- H. The age-composition of the catch may not be reliable (e.g., no validation of annuli in otolith based ages, in scale based ages, or in the relationship of otolith to scale based; therefore, the results from the ADAPT and subsequent analyses dependent upon these results are suspect, as VPA assumes no errors in catch-at-age.*
- I. Spatial analysis of fishery and survey data would help to determine what the portions of the resource the data represent.*
- J. No information from fishery independent surveys on changes in growth was presented.*
- K. The TC believed that ADAPT would be able to sort out the different survey variances; however, ADAPT is not capable of doing so.*
- L. Both the TC and Panel felt the NEFSC fall inshore survey had the broadest geographic coverage of available surveys, could be used to monitor long-term trends of relative abundance, and tracked some cohorts well.*
- M. Each fishery dependent index [in this case, the MRFSS aggregated biomass index] has nuances with catchability (i.e., technological changes) which were not discussed. The mid-Atlantic fleet appears not to reflect abundance. For example, there is a concern that a proportion of the stock migrates out of the mid-Atlantic area and that proportion varies annually, which would affect the catchability of the mid-Atlantic private boat fleet.*

N. This [private boat] fleet historically accounted for a small component of total catch (11% in 1989, 12% in 1990), but accounts for more than 50% in recent years and does not account for trends in abundance outside the fished area (North, South, and offshore).

O. The sampled age-structure indicates extraordinarily high Z estimates during periods of low catch, suggesting that M rates are high. However, there are two additional reasons that Z could be inflated, 1) older ages are truncated by the selectivity of the gear and 2) catches of older fish in areas where fish are not sampled (e.g., New Jersey).

P. The TC makes the case for increasing Z based on the ADAPT Run #20 (the preferred run from Stock Assessment Report). All other ADAPT analyses indicate that total mortality is decreasing. The distinguishing feature of Run #20 is that it is tuned solely with the recreational CPUE from the mid-Atlantic private boats. All of the subsequent analyses were dependent upon the results of Run #20.

Q. The TC tried to explain the increase in Z as increased predation mortality by striped bass using a production model coupled with a Type III functional response. No statistics were presented that allowed the Panel to evaluate the Steele-Henderson model relative to models that assume constant M.

R. The evidence for striped bass predation on exploitable sized weakfish is weak. The Steele-Henderson model is inappropriate for the interactions among striped bass, weakfish, and menhaden, because it assumes that predation occurs only on fish that are fully vulnerable to the fishing gear and ignores predation on smaller fish.

S. A more rigorous approach to considering multispecies interactions would be to consider a precise hypothesis and incorporate the appropriate process in the population model.

T. For example, if the precise hypothesis is predation of age-0 and age-1 weakfish by striped bass, then a stock recruitment function with a predation term would be appropriate.

U. If M has increased over time, then the corresponding estimate of F_{msy} would be greater than the estimate of F_{msy} that assumes constant M. If M is truly increasing, but we assume that it is constant, then we would advise for future F estimates that are less than the true F_{msy} . Conversely, if M is truly constant and we assume that it is increasing, we would recommend that future F estimates that are greater than F_{msy} . Therefore, assuming constant M is more risk averse than assuming an increase in M.

V. There is some optimism for weakfish stock status, as recruitment has not declined in any portion of the range. Despite our uncertainty in stock productivity there is no apparent decrease in recruitment. In fact, evidence suggests that there have been recent increases in recruitment over the last 15 years.

W. The Panel is uncertain of the direction of Z, F, and M estimates, and, as a result, it is difficult to choose among alternative explanations of observed changes.

**Section 2: Details of Agreements and Disagreements with the April 2006 Peer Review Panel Outlined
in Table 1**

Format of Section 2

This section provides detailed responses to the Peer review panel comments (in italics) that are represented by letters in Table 1 and listed in order of appearance (and lettered) in Section 1. The TC agreement or disagreement is in bold. An additional response from the TC is in normal type. Part 1 or Part 2 in the text refers to sections of the weakfish assessment. References can be found in the report on the TC response to the 40th SARC.

What was the TC trying to do?

- 1. Answer the question "Is the stock declining?"**
- 2. If so, what are the likely underlying causes for the decline?**

Some important aspects of the assessment were not addressed by the Panel. Development of aggregated biomass indices, estimates of relative F (catch / index), proportional stock densities (PSD, a length structure index), biomass dynamic modeling without predation, and the concept of rescaling relative F to absolute F units were not commented on.

Peer Review Section 1. Characterize commercial and recreational catch including landings and discards.

A. There is evidence of stock structure.

TC Agrees

It has been discussed regularly in the past decade. The TC provided stock structure information to the panel and has summarized it for the Board for this meeting (August 2006). Tagging, meristics, and otolith microchemistry support regional dynamics, but suggest moderate exchange among regions. Genetic analyses indicate largely homogeneous populations. The TC notes that difficulty in defining stocks and implementing stock-specific assessments in a mixed stock fishery would be major obstacles.

Other ASMFC assessments (striped bass, Atlantic menhaden, and red drum) have known or potential stock structure, but are assessed as single stocks.

B. Taking account of stock structure would probably reduce uncertainty in the assessment and lead to better stock status determination on a regional basis.

TC Agrees

- TC agrees that evidence of stock structure exists.
- But there is inadequate data and enough potential mixing that pinpointing where to split is questionable.

C. Dedicated age-length keys are unavailable for most portions of the catch (e.g., northern portions of commercial fishery, all recreational catch, some surveys, commercial discards before 1994).

TC Agrees

The TC notes that this is not a feature restricted to the weakfish assessment alone. In particular, all ASMFC age-structured assessments (striped bass, summer flounder, bluefish) that feature recreational catches rely on borrowed age-length distributions.

D. ... given that weakfish are often taken as bycatch and have relatively low value, overall accuracy of landings may be less than TC believes.

TC Disagrees

The TC did not express an opinion on the accuracy of landings. The assessment used landings as they are used in other assessments.

4. Weakfish are valuable and most major harvesting states have or had directed fisheries.
5. Directed landings and legal bycatch are recorded.
6. Prior to imposition of Amendments 2-4, regulation was minimal and there was little regulatory reason for discarding.
7. Weakfish recreational catches (survey methodology) follow commercial catches (census methodology) closely (correlation $r = 0.86$).
8. Discard estimates and their precision were estimated by De Silva (2004). Precision is sometimes poor. Accuracy cannot be addressed.
9. Including and excluding discards had little effect on assessment trends and estimates.
10. Amount of commercial discards needed to account for decline is much larger than current estimates (3-13 X's in 2003) based on modeling. Evidence does not point toward any Atlantic coast fishery capable of generating 3,000-13,000 mt of additional unreported weakfish discards. The results of these bycatch scenarios taken at face value suggest that weakfish regulations created this massive boost in discards.

E. Missing CAA samples (northern region), overlap of lengths-at-age, and geographic patterns in growth may confound estimation of CAA.

TC Agrees

*F. There are major problems with the calculations of the catch-at-age as indicated by substantial sum of products differences (Catch at age * mean weight at age does not equal the estimates of catch biomass).*

TC Disagrees

Part 1, Figure 15 indicates good agreement during 1981-2003, except for two years in the late 1980s.

G. No validation of annuli in otolith based ages, in scale based ages, or in relationship of otolith to scale based ages.

TC Disagrees

Ageing of weakfish by otoliths has been validated in the primary literature (Lowerre-Barbieri et al. 1994). Further work for the TC was summarized by Wenner (1999). Conversion of scale to otolith ages was covered in the 30th SARC.

H. The age-composition of the catch may not be reliable (e.g., no validation of annuli in otolith based ages, in scale based ages, or in the relationship of otolith to scale based); therefore, the results from the ADAPT and subsequent analyses dependent upon these results are suspect, as VPA assumes no errors in catch-at-age.

TC Agrees

This applies primarily to VPA. Index driven and biomass dynamic model assessments using aggregated landings provided similar trends; the latter were presented to the Panel, but not mentioned in their review.

I. Spatial analysis of fishery and survey data would help to determine what the portions of the resource the data represent.

TC Agrees

Detailed spatial analysis has not been conducted on an age-structured or aggregated basis. TC may not have spatial analysis (GIS) expertise.

J. No information from fishery independent surveys on changes in growth was presented.

TC Agrees

Growth estimates could be made for some years and surveys with extended age structure and otolith collections (DE and NJ). Bias could result if large weakfish avoid trawls or if growth variation reflects stock structure.

Peer Review Section 2. Review adequacy and uncertainty of fishery-independent and dependent indices of relative abundance.

K. The TC believed that ADAPT would be able to sort out the different survey variances; however, ADAPT is not capable of doing so.

TC Disagrees

This was not one of our beliefs. This was the configuration of the SARC 30 approved model. We ran multiple index configurations of VPA, untuned VPA, and biomass dynamic models (Part 1, pages 27-28 & a 15 min presentation to the panel), to try and understand survey contributions.

L. Both the TC and Panel felt the NEFSC fall inshore survey had the broadest geographic coverage of available surveys, could be used to monitor long-term trends of relative abundance, and tracked some cohorts well.

TC Disagrees

While the TC would agree that the NEFSC survey has good geographic coverage and adequate precision, we have found it to be unsuitable as an index of exploitable weakfish abundance or biomass for the following reasons:

1. Cohort catch curves of NEFSC survey indicate spontaneous generation (year-class increases in abundance) in some cases.
2. Too few weakfish of sizes of interest to fishery were present in this survey after minimum size was raised in mid-1990s (fishery selectivity changed and NEFSC no longer sampled relevant sizes). PSD (length-frequency) analysis indicated proportion 13 inches+ in NEFSC was too small to estimate reliably (Uphoff 2004). NJ and DE sample sizes provide reliable, precise PSD estimates, and are well correlated.
3. Use of NEFSC index leads to conclusion of high stock status, contradicts other indices and landings.
4. NEFSC index significantly correlated with NJ and marginally with DE. MRFSS and NEFSC not correlated ($r = 0.03$; J. Uphoff, recent analysis).

M. Each fishery dependent index [in this case, the MRFSS aggregated biomass index] has nuances with catchability (i.e., technological changes) which were not discussed. The mid-Atlantic fleet appears not to reflect abundance. For example, there is a concern that a proportion of the stock migrates out of the mid-Atlantic area and that proportion varies annually, which would affect the catchability of the mid-Atlantic private boat fleet.

N. This [private boat] fleet historically accounted for a small component of total catch (11% in 1989, 12% in 1990), but accounts for more than 50% in recent years and does not account for trends in abundance outside the fished area (North, South, and offshore).

TC Disagrees with M and N.

This is a strong statement that can't be supported without an independent estimate of abundance and movement.

During the Weakfish TC meeting in Providence, V. Crecco was asked to comment on the contention by the Weakfish Peer Review Committee that the private boat recreational catches have comprised an ever increasing proportion of the total coast-wide weakfish harvest from 1989 to 2005. Based on these findings, the Review Committee concluded that the Mid Atlantic private boat CPUE was an uninformative index of coast-wide weakfish abundance.

V. Crecco examined the inputs (total recreational catch and effort) in the Mid Atlantic private boat indices from 1989 to 2005 based on the MRFSS database. There appears to be major confusion among the Reviewers regarding their definition of "landings" and "catches" in their assertion about the private boat index. The Reviewers used column 2 (total coast-wide landings and discard losses) and column 4 (Mid Atlantic private boat catches mt) in Part 1, Table 10, page 70 to generate ratios of private boat catches to total coast-wide landings from 1989 to 2005. Unfortunately for the Reviewers, the data in col 4 in Table 10 are recreational total catches including weakfish released alive (A, B1 and B2) in the Mid Atlantic private boat fishery. These recreational catch data are not directly comparable with the coast-wide landings (commercial and recreational harvest) plus discard losses (20% of recreational releases and all commercial gill net and trawl releases) in column 2. As a result, the apparent rise in the percentage of private boat catches to total landings from 1989 to 2005 is baseless because the units of catch and harvest are incompatible. If you compare the Mid Atlantic private boat harvest (A and B1) to the total coast-wide harvest, the percentage composition of the private boat harvest varies without trend from 1989 to 2005 with a long-term mean of 18%. The imposition of coast-wide minimum size limits and creel limits in 1994 on the recreational fisheries generated a sharp rise in the magnitude of B2 catches (releases) in the Mid Atlantic private boat fishery from 1994 to 2005. Therefore, the argument put forth by the Reviewers regarding the reliability of the Mid Atlantic private boat index is an artifact of a faulty comparison of apples (coast-wide landings) and oranges (private boat catches).

The 40th SARC recommended development of a recreational CPUE index. The TC developed the MRFSS index as a substitute for the NEFSC survey. Substitution of MRFSS for NEFSC indicates a decline to low status instead of an increase to high status indicated by the NEFSC survey. These conflicting trends clearly required a choice between these indices.

Correlation analysis indicates that MRFSS, DE, and NJ biomass indices are significantly correlated. MRFSS and DE indices are significantly correlated with total landings (J Uphoff, recent analysis). MRFSS CPUE well correlated with DE PSD (long-term indicator of status).

Analyses were conducted with multiple indices and this was indicated to the panel. They were available for review upon request. They were not included because of space limitations and information overload.

MRFSS index coverage is broad (NY-VA) and constitutes the core area where weakfish are likely to be a sought gamefish rather than an angler's afterthought. The MRFSS Index includes offshore catches. Minor sportfisheries exist to the north and south of the index region. Northern region (north of New York) MRFSS catch estimates average 0.3% of total catch and southern region (NC-FL) averages 14%. Recreational catches in south have increased, but most have been in NC. Commercial catches in NC have fallen. If weakfish distribution were shifting, wouldn't both sectors in NC reflect an increase? Omitting south catches (NC-FL recreational and portion of NC commercial) from the coastal Steele Henderson model (hence, a mid- to north Atlantic assessment) produces results with minimal differences.

Assessment of index catchability (NEFSC, NJ, DE, MRFSS) and converged VPA biomass during 1982-1993 indicated MRFSS catchability was most consistent (Part 2, page 12). Survey catchabilities were much more variable and inverse relationships with biomass were suggested. NEFSC survey had the worst attributes in catchability. If converged VPA is considered a reasonable estimate of biomass, than MRFSS index q varied up to 3x's during this period, while NEFSC survey q 's varied up to 28x's. Trawl survey q must be quite variable too, given the sawtooth pattern exhibited in NJ and NEFSC indices.

Changes in catchability over the recent decade were not described in Parts 1 or 2, however. Evidence for MRFSS index q to change is supported by estimating relative q (MRFSS index divided by DE or NJ index during 1989-2003 (biomass units for all; this was estimated for the panel at the review). MRFSS catchability rose from and fell to baseline during 1997-2000. Using NJ and DE survey biomass indices as tuning could ameliorate this.

Peer Review Section 3. Review the appropriateness of constant and variable natural mortality (M) estimates in the assessment.

O. The sampled age-structure indicates extraordinarily high Z estimates during periods of low catch, suggesting that M rates are high. However, there are two additional reasons that Z could be inflated, 1) older ages are truncated by the selectivity of the gear and 2) catches of older fish in areas where fish are not sampled (e.g., New Jersey).

TC Agrees

Selectivity and catchability can be confounded with M in age structured assessments.

The Z's have increased in recent years. These phenomena (selectivity and q) would have had to only increased greatly in recent years after imposition of substantial conservation measures (minimum mesh sizes, increased length limits, seasons, NC area closures).

We did not have samples from NJ and NY during early and late in the time-series (when stock was relatively high), so this bias should be the same both periods of the assessment (extended age and size structure were present). Age-structured and age-aggregated assessments detected much higher F's during the early period than recent years.

The mixed nature of the commercial fishery off NC in winter and its migration down the coast through MD and VA in fall does provide an opportunity to sample these "missing" fish.

P. The TC makes the case for increasing Z based on the ADAPT Run #20 (the preferred run from Stock Assessment Report). All other ADAPT analyses indicate that total mortality is decreasing. The distinguishing feature of Run #20 is that it is tuned solely with the recreational CPUE from the mid-Atlantic private boats. All of the subsequent analyses were dependent upon the results of Run #20.

TC Agrees

In ADAPT runs other than #20 (tuned to trawl surveys), F decreases to an unrealistically low level and the stock biomass increases 3x's beyond anything estimated previously and shows no sign of decreasing. It implies decreased catches because fishers are disinterested or weakfish have become more devious. Run 20 retrospective bias was low compared to trawl survey tuned runs. The 40th SARC did not consider runs with trawl surveys credible.

The 40th SARC panel encouraged development of an MRFSS index.

At least three other ASMFC assessments feature MRFSS indices: Atlantic croaker, striped bass and bluefish.

Q. The TC tried to explain the increase in Z as increased predation mortality by striped bass using a production model coupled with a Type III functional response. No statistics were presented that allowed the Panel to evaluate the Steele-Henderson model relative to models that assume constant M.

TC Disagrees

Part 2, page 17: SSQ, r^2 , bootstrapping, and Figure 26 (observed and predicted points for models with and without predation term). Biomass dynamic models without predation based on surveys were presented in Part 1, pages 25-29. Also part of slide presentation. Part 1, pages 35, 41-42, Tables 15-21, included parameter SE, SSQ, R^2 , and residual patterns.

R. The evidence for striped bass predation on exploitable sized weakfish is weak. The Steele-Henderson model is inappropriate for the interactions among striped bass, weakfish, and menhaden, because it assumes that predation occurs only on fish that are fully vulnerable to the fishing gear and ignores predation on smaller fish.

TC Disagrees

The panel was presented with this summary of the reach of statistical analyses and Steele and Henderson model:

- **High-low weakfish biomass domains evident long-term (statistical).**
- **Menhaden has large influence over long-term domains (statistical).**
- **Now in low domain – cutting F may not allow for recovery because bass M high (Steele and Henderson).**
- **Bass-weakfish model (Steele and Henderson) applies to low domain.**

The Steele and Henderson (S-H) model was not developed to fully explain the triangular food web (menhaden, bass, weakfish) presented. It was used to explore depensatory mortality caused by striped bass during the most recent two decades of declining and low forage abundance. The combination of statistical analyses and S-H model explored this food web hypothesis. This combined approach was explicit in the Part 2 introduction of the scope of the analyses. The model is appropriate for the exploration, but it will not supply a full explanation. It does account for 78% of variation in the MRFSS index since 1981.

This is something we may not have explicitly talked about with the Panel, but the MRFSS catch index is composed of some weakfish that are small. MRFSS harvest length-frequencies contain some weakfish as small as 5 inches and 8-10 inch fish are not uncommon. The catch (releases) will be comprised of an even larger fraction of small weakfish vulnerable to predation. The MRFSS index, based on catch, is not as

vulnerable to selection changes as a harvest-based index or a survey index, such as the NEFSC fall survey, where a very large fraction of fish present in the survey went from legal to sublegal after regulations were imposed.

Recent diet analyses of large striped bass collected off Cape Hatteras (2005) and Virginia Beach (2006) in fall and winter by Dr Anthony Overton (East Carolina University) indicate a fairly steady prey-predator length ratio (0.1-0.3) for striped bass, before it tapers off to 0.45. In other words, a 900-1100 mm striped bass can readily eat weakfish that would be legal-sized. Smaller weakfish (8 inches) would be fairly vulnerable to striped bass as small as 500 mm. Weakfish comprised 5% of diets by weight in this study, the highest percentage I know of. The Steele and Henderson model in Part 2 indicates that a decline in weakfish would be induced by striped bass even though weakfish are only a minor part of the diet (1-2% of total when compared to bioenergetics estimates of bass total consumption). This is consistent with the importance of weakfish in bass diet studies; Part 2 page 23.

Consumption of weakfish by striped bass associated with the model (0.09-0.10 kg weakfish per kg striped bass) is very modest compared to total consumption of fish by bioenergetics (4-7 kg of fish per kg of striped bass) and is consistent with mid-Atlantic diet studies.

The Panel's interpretation that the model can only be applied to exploitable weakfish eaten by striped bass is not correct. A production model subsumes recruitment, growth, and natural mortality into a single function. The effects of predation may not be evenly applied across the three sub-functions and the model is just going to calculate a general loss associated with a predator trend. See Part 2, page 22.

Analysis of predation on small weakfish (relative Z) was presented in Part 1, pages 32-34. The rise in relative Z was consistent with M associated with bass in the predator-prey model.

S. A more rigorous approach to considering multispecies interactions would be to consider a precise hypothesis and incorporate the appropriate process in the population model.

TC Disagrees

We operated from very specific hypotheses and used methods suited to modest data sets. The approach suggested by the Panel would likely lead to an overparameterized model. We developed the hypothesis that M has been changing for weakfish and that it is the primary driver behind the recent crash as a specific alternative, nonfishing mortality hypothesis explaining an aborted weakfish recovery. We investigated specific hypotheses that weakfish biomass dynamics, growth, and survival were negatively affected by striped

bass and exacerbated by depleted forage. We postulated and eliminated other candidate predator/competitors and environmental factors (water temperature and North Atlantic Oscillation index) to see how easily “spurious” hypotheses could be confirmed. See the Introduction of Part 2 and Part 1, page 35 for further specifics.

T. For example, if the precise hypothesis is predation of age-0 and age-1 weakfish by striped bass, then a stock recruitment function with a predation term would be appropriate.

TC Disagrees

Stock-recruitment analysis requires a good underlying assessment. Errors in the estimation of SSB can produce a “relationship” that may not exist. If you start with a single species assessment under an assumption of constant M when M isn't constant, you will end up with a misleading relationship. Stock-recruitment relationships are seldom well estimated or can appear to be when they are, in fact, seriously biased.

U. If M has increased over time, then the corresponding estimate of F_{msy} would be greater than the estimate of F_{msy} that assumes constant M . If M is truly increasing, but we assume that it is constant, then we would advise for future F estimates that are less than the true F_{msy} . Conversely, if M is truly constant and we assume that it is increasing, we would recommend that future F estimates that are greater than F_{msy} . Therefore, assuming constant M is more risk averse than assuming an increase in M .

TC Disagrees

Constant M was a very risk prone assumption in most model configurations. Most VPA runs (except run 20 with MRFSS tuning) and standard Schaefer models without predation did not indicate that F was at anything that would have caused concern.

Only VPA run 20 under constant $M=0.25$, rescaled F analysis based on run 20 that allowed for estimation of varying M , or biomass dynamic models with a bass predation component indicated otherwise. VPA run 20 under constant $M=0.25$ would indicate that large cuts in F were necessary and would lead to recovery. Rescaled F analysis and bass predation models indicated that M was much higher than F and recovery was unlikely by cutting F without a drop in M .

Equilibrium MSY limit reference points estimated from the S-H model (Steele and Henderson striped bass-weakfish predator-prey model; Part 2, page 21), indicated that F had been below the equilibrium F_{msy} limit for nearly a decade, but biomass had fallen well below the B_{msy} limit in the last several years. By 2003, rising M_b (bass-related natural mortality) ate away at the amount of F that represented a level of “safe” fishing; residual production after bass predation ($r - M_b$) fell from 0.66 in 2001 to 0.08 in 2003. Even though F fell from 0.50

during 2000-2002 to 0.39 in 2003, the amount of residual productivity after bass predation and fishing ($r - M_b - F$) went from positive to negative (0.16 to -0.31) during 2000-2003. By 2003, completely cutting F to compensate for high M_b would result in slow biomass growth at best.

Peer Review Section 4. Review the estimates of fishing mortality (F), spawning stock biomass, and total stock biomass for 1981-2004, and characterize the uncertainty of these estimates.

V. There is some optimism for weakfish stock status, as recruitment has not declined in any portion of the range. Despite our uncertainty in stock productivity there is no apparent decrease in recruitment. In fact, evidence suggests that there have been recent increases in recruitment over the last 15 years.

TC Disagrees

This is true for the grand mean of multiple surveys. However, in the last 3 years (2000-2003) in the north (RI-DE) and south (MD-NC) there is some evidence of regional declines and a more chaotic picture than indicated by the grand mean. In the north, 2 state indices have declined to below average, 1 is steady at below average relative abundance, and 2 have increased (1 to above average and 1 to average). In the south, 3 have declined below average, and 1 has increased to above average. Age 0 sized weakfish relative abundance in SEAMAP and NEFSC is above average.

W. The Panel is uncertain of the direction of Z, F, and M estimates, and, as a result, it is difficult to choose among alternative explanations of observed changes.

TC Disagrees

The TC unanimously still supports the five points of agreement reached in February 2005 regarding weakfish stock status. The five points were

- **stock is in decline;**
- **total mortality is increasing;**
- **not much evidence of overfishing;**
- **something other than F is going on; and**
- **strong circumstantial evidence of increasing M.**

Stock assessment can provide support for decision making by describing possible alternative states of nature and their consequences to management (See Part 1, Tables 1A and 1B, pages 54-55) . This is what the TC sought to do with the divergent signals it obtained after extensive analysis. There are uncertainties and different interpretations associated with the quality of fisheries data, analysis, and interpretation, as evidenced between the amount of disagreement between the TC and Panel. There clearly are risks associated with

paring away too much information as the Panel has done – there is simply nothing left as a basis for making decisions.

In a decision analysis context, it can be very important to carry a hypothesis through the assessment and examine its management implications even if it has a low probability of being correct. A hypothesis that is rejected as being improbable based on usual scientific criteria should remain in a decision analysis if it has particularly important management implications. The implications of increasing M have been demonstrated by the TC. Simply presuming that ceasing exploitation on an overfished stock has to result in recovery ignores the uncertainty imposed by ecological systems. Weakfish may or may not recover with more regulation in a world where food webs exist.

The Panel's alternative to a decline because of rising M is massive discarding arising from Amendments 2-4. In this case, weakfish regulations created a huge discarding problem (manifold greater than landings) that cannot be solved by only regulating the weakfish fishery. This hypothetical problem, which cannot be confirmed with our current discard data and methodology, likely requires keeping major mobile finfish fleets of the mid-Atlantic from fishing for any species inshore. It is interesting to note that the Panel did not indicate or suggest that directed overfishing was occurring.

Finally, there is a body of models in this assessment, based on accepted conventions, that indicates things are going swimmingly and nothing further needs to be done. They are not necessarily reflective of indices, landings, or fishers' experience over much of the mid-Atlantic. These models require application of pretzel logic to be acceptable - no one really wants to catch weakfish anymore.

Executive Summary: How Much Additional Discarding and Under-Reporting is Needed to Cause the Decline in Weakfish?

Jim Uphoff

June 20, 2006

The Weakfish Peer Review panel suggested that a steep rise in unreported commercial discards of weakfish after 1995 could have created the recent weakfish stock collapse in lieu of our hypothesis of a rise in natural mortality. As a response, I fit a biomass dynamic model to three biomass indices, harvest, estimated discards, and an additional loss term (L_F) mimicking various commercial bycatch scenarios (constant function of harvest plus discards, two increasing functions of time, or an inverse function of biomass). Additional losses were imposed after 1995 to reflect regulatory discards. Results were contrasted with those of a striped bass–weakfish predator-prey model and a biomass dynamic model without L_F . Additional losses from simulated bycatch scenarios or striped bass predation improved fit over the unaltered model (Figure 1). Median L_F estimates during 1996-2003 (6,100-6,800 mt) were greater than median bycatch (1,700 mt) and harvest (4,400 mt). By 2003, L_F estimates were 3-13 times discards used in the assessment depending on how trends in L_F were simulated (Figure 2). There is no evidence available thus far that points toward any Atlantic coast fishery capable of generating 3,000-13,000 mt of additional unreported weakfish discards. The results of these bycatch scenarios taken at face value suggest that weakfish regulations created this massive boost in discards. Thus, implementation of further conservation measures short of a coast-wide moratorium on all Atlantic coast fisheries will not minimize this discard problem.

Rising natural mortality provided a reasonable alternative to enormous undetected discards. Estimated losses of weakfish to modeled striped bass predation fell just below upper range of commercial bycatch scenarios. Striped bass biomass has increased approximately 20-fold since 1981 and modeling indicated little change in weakfish consumed per unit area per unit time (depensatory mortality). This situation is analogous to a fishery able to apply more effort while maintaining catch efficiency. Consumption of weakfish by striped bass associated with the model (0.09-0.10 kg weakfish per kg striped bass) is very modest compared to total consumption of fish by bioenergetics (4-7 kg of fish per kg of striped bass) and is consistent with mid-Atlantic diet studies.

Figure 1. Fit of biomass dynamic model using original harvest + discards with and without additional loss functions. WRI = MRFSS private boat catch per trip index; NJ and DE are trawl survey indices. Loss functions depict striped bass predation, losses as an inverse function of weakfish biomass, quadratic or linear functions of time, and as a constant function of the sum of harvest and estimated discards.

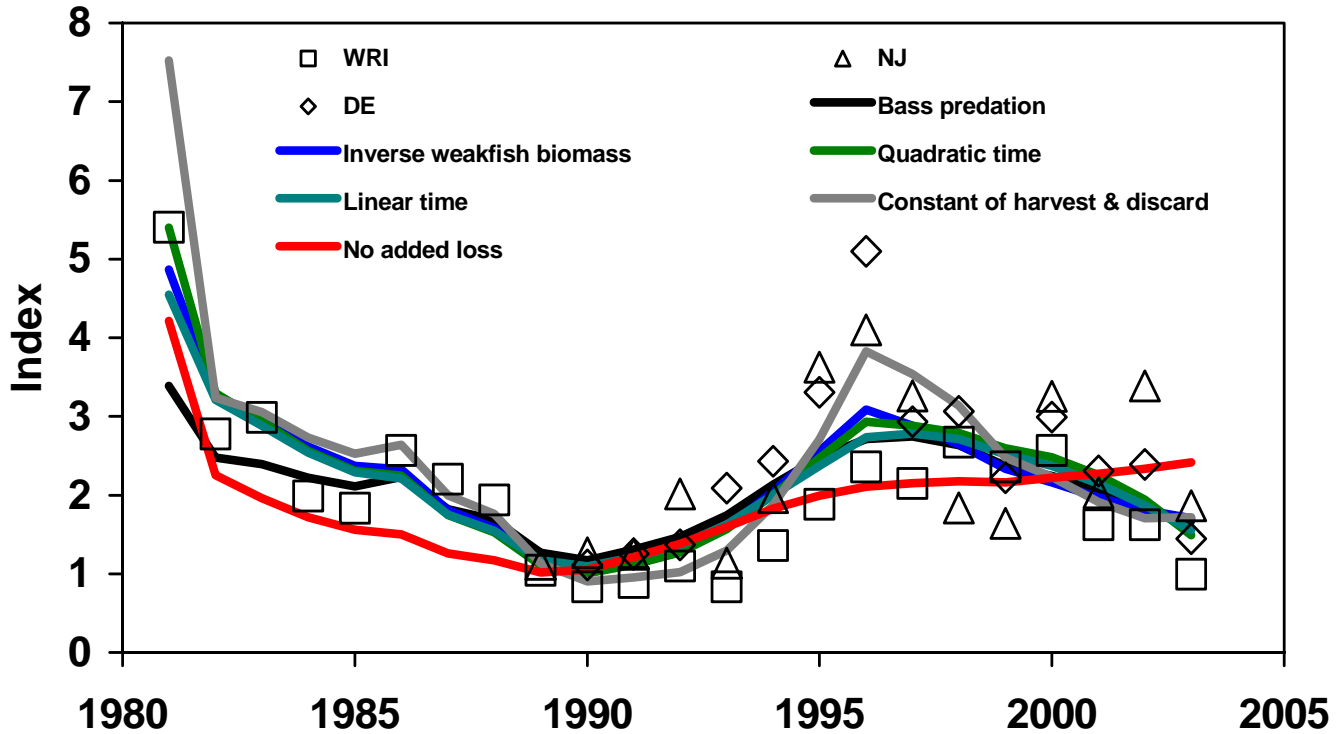
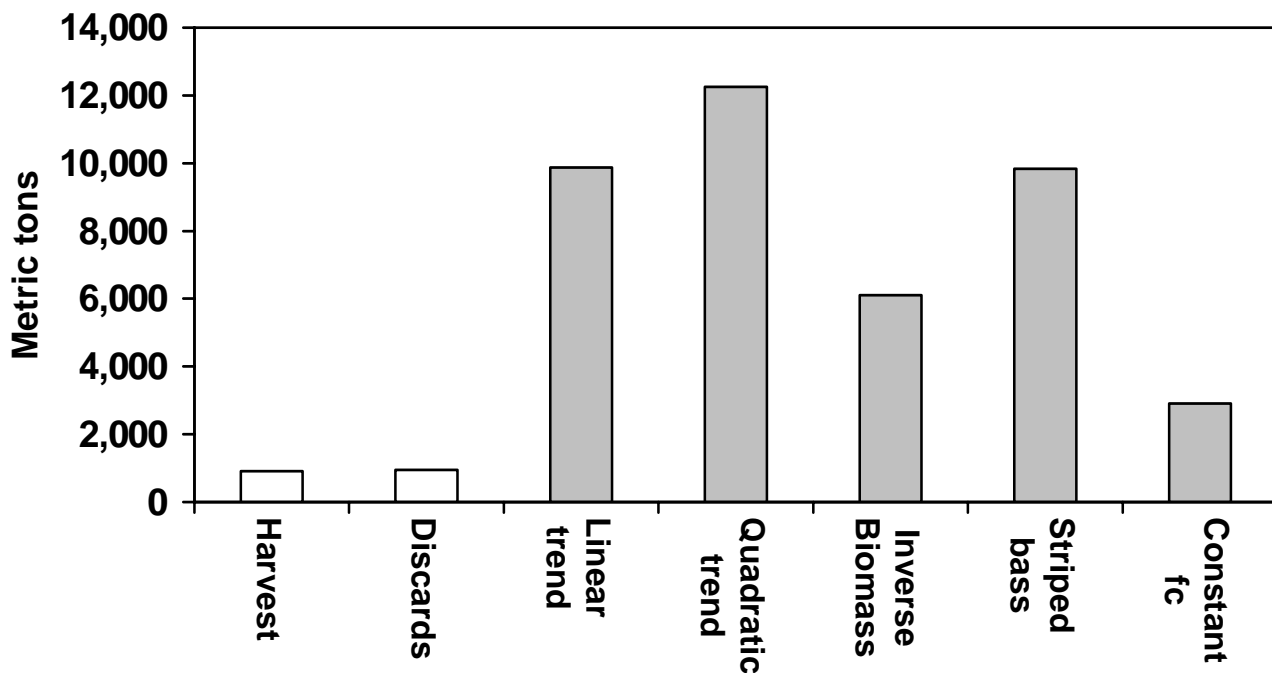


Figure 2. Metric tons of additional losses in 2003 from different loss scenarios contrasted with commercial harvest and discard estimates



Executive Summary: Estimating At-Sea discards of weakfish using NEFSC Observer database

Weakfish Technical Committee

July 2006

This is a brief summary of *Estimating At-Sea discards of weakfish using NEFSC Observer database*, a report prepared by Janaka de Silva (FL) in 2004 for the Weakfish Technical Committee. This report provided the technical basis for discard estimates used in the recent stock assessment.

Estimates of commercial discards were developed from 1994-2003 NMFS at sea observer data. The database contained information on weakfish discards, target species catches, estimates of landings of species kept on the haul, and length data of both discards and fish kept. We only evaluated hauls that were completely sampled for discards. During 1994-2003, discard data were primarily available for gillnets (287 trips and 615 hauls; 89% landed in Virginia, North Carolina or New Jersey) and otter trawls (196 trips and 519 hauls; 41% landed in Maryland and 49% in New Jersey, New York, or Rhode Island.). The annual number of gillnet trips where weakfish discards were observed ranged from 5 to 68. The numbers of otter trawl trips where weakfish discards were observed ranged from 4 to 45.

We developed estimates of weakfish discards using annual (1994-2003) discard ratios (weakfish : target species) and NMFS commercial landings of a suite of target species. The ratio of at sea observations of weight or number of discarded weakfish to pounds of harvested target species by gear (trawl and gill net) was scaled up to total discard estimates based on total landings of the target species. Discards were subdivided into regulatory and market discards. Identification of target species was only based on complete trips and hauls where weakfish discards were observed. Target species were weakfish, longfin squid, Atlantic croaker, butterfish, summer flounder, Atlantic menhaden, spiny dogfish, spot, and bluefish. We assumed that gear-specific relationships among the target species in the data set reflected the entire fishery.

Slightly more than half of ratio estimates were significantly different from zero. Ratio estimators work best when there is evidence of a positive linear relationship between the response and explanatory variable. Total discard estimates presented in this report included estimates from target species where there was not evidence of a strong linear relationship between weakfish discards and the target species we examined.

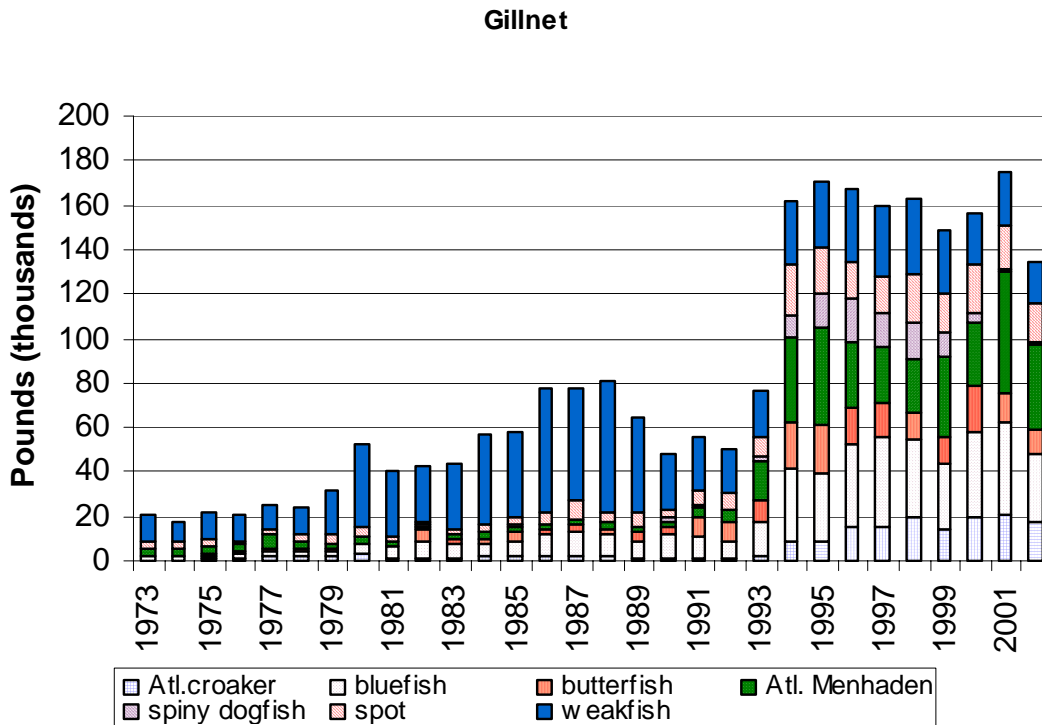
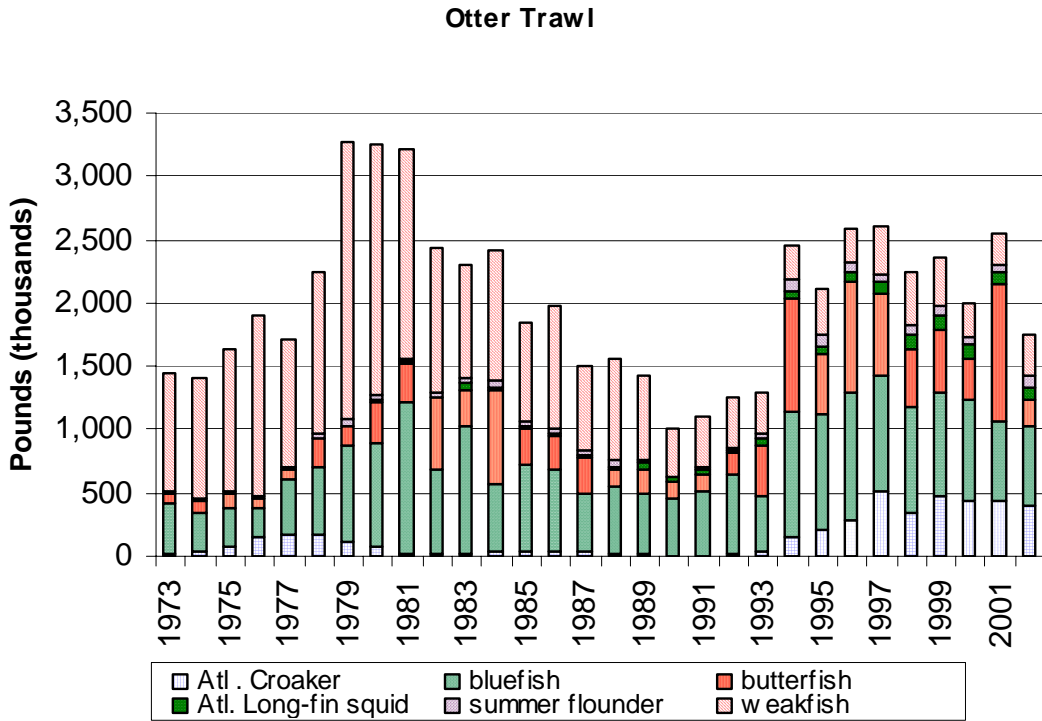
During 1973-1992, coast-wide commercial fishing regulations on weakfish were not in place. Market-related discard ratios were used to estimate weakfish discards during 1973-1992. In 1993, New Jersey and Delaware enacted weakfish regulations that may have lead to non-market discarding and the total discard ratio was used for these states. For all other states, discard estimates for 1993 were based on the non-regulatory discard ratios. During 1994-2002, total discard ratios were used to estimate weakfish discards for all states. Commercial discard data was not available for 2003 when these estimates were formulated for the assessment and we averaged the commercial discard estimates for 2001 and 2002.

We considered all estimated commercial discards to have died in the stock assessment. Weakfish discards for the mid-Atlantic were predominantly associated with otter trawls (Figure 1). Trends in otter trawl weakfish discards for the mid-Atlantic show a cyclical trend, with peaks around the late 1970's (~ 3 million pounds) declining to a low of around 1 million pounds in 1990, and stabilizing around 2.5 million pounds in recent years. Trawl fisheries for Atlantic croaker, summer flounder, bluefish, butterfish, and weakfish account for most discards once regulations were imposed (since mid-1990s). For mid-Atlantic gillnets, discards rose from 50,000-80,000 pounds prior to 1994 to 150,000-160,000 pounds during 1994-2001. Estimated discards fell to around 120,000 pounds in 2002. Gill net discards are now mostly from the Atlantic croaker, Atlantic menhaden, spot, butterfish, and weakfish fisheries; discards from the spiny dogfish fishery no longer appear in our estimates (Figure 1).

We also developed discard estimators using a regression approach, but found the models to be of poor quality. While effort (trip) based estimators could be developed, the corresponding effort information on the target species in the fishery are only available for some states and in certain years.

Most commercial fisheries were eliminated from the South Atlantic by the mid-1990s. Weakfish discard estimates for the South Atlantic were low and of poor quality because they were based on estimates from the mid-Atlantic. They were not included in the assessment.

Figure 1. Estimated weight of weakfish discarded by mid-Atlantic otter trawl and gill net fisheries



The weakfish population structure along the Atlantic Coast: a literature review

A report to the Atlantic States Marine Fisheries Commission's Weakfish Technical Committee

By

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July 2006

Weakfish population structure

This note summarizes available information on weakfish population structure, part of which (especially the tagging outputs, meristic, morphological and life history characters of weakfish inhabiting the Mid-Atlantic) was reviewed by Mercer (1985, 1989). Weakfish, *Cynoscion regalis*, occur along the Atlantic coast of the United States of America from eastern Florida to Massachusetts, but are most abundant between New York and North Carolina (Mercer 1989). Weakfish stock assessments and related fishery management plans have been configured and implemented on the basis of a single, panmictic weakfish population (e.g., Mercer 1985, Anon. 2002).

Investigations of weakfish population structure along the US Atlantic coast have been undertaken through tagging, meristic, morphological, life history, genetic and otolith chemistry studies (Table 1). The conclusions reached are conflicting. While Crawford et al (1988), Graves et al. (1992) and Cordes & Graves (2003) did not detect genetic differentiation within the weakfish population, Chapman et al. (unpublished report) found that weakfish are made up of a series of overlapping stocks, without complete panmixia. Non-genetic studies found evidence of existence of multiple weakfish sub-populations (e.g., Nesbit 1954; Shepherd & Grimes 1983, 1984; Scoles 1990) or important spatial structure of the weakfish population (Thorrold et al. 1998, 2001). Mark-recapture, meristic, morphological and life-history studies (e.g., review by Crawford et al. 1988) indicated that weakfish could be partitioned into sub-stocks, and assessed and managed on a regional basis.

Whether studies did or did not detect discrete weakfish populations, it is worth mentioning views expressed and observations made by authors which may have assessment and management implications:

- Although the weakfish population is genetically homogeneous throughout its area of distribution, it exhibits variations in the population parameters (i.e. growth, mortality and fecundity), upon which the ability of a population to sustain a harvest is largely dependent (...);

therefore, as a practical matter, it is probably best to manage weakfish as discrete northern and southern units, even though these units are not reproductively isolated (Crawford et al. 1988).

- The genetic homogeneity found within the weakfish population contrasts with the geographical (and likely eco-phenotypic) variations of morphological and life history characters reported in other studies (...). There is clearly some interdependence among areas as also supported by tagging data (...). To obtain a meaningful estimate of the magnitude of the interdependence between these areas would require an extensive, time-consuming and expensive tagging study. Until such information is available, it would be best to manage the weakfish resource as a single interdependent stock (Graves et al. 1992).
- Weakfish tagging data indicate that low levels of exchange occur between geographically distant populations of weakfish (e.g., Schwartz et al. 1979; Music & Pafford 1984; Bain et al. 1998).
- Our data show much more spatial structure than is currently assumed by fisheries managers, and it may be useful to consider weakfish population dynamics from a metapopulation perspective. However, these results do not contradict the genetic findings, because there are sufficient exchanges even among those estuaries with the highest levels of natal homing, to prevent genetic divergence (Thorrold et al. 2001).
- The results suggest that a cline of morphometric characters or substantial mixing among the morphological types occur intermediate in the range for weakfish. Moreover, weakfish samples collected from the winter fishery off NC suggest that 79.1% belong to the Long Island (med) reference sample and 20.9% to the Pamlico Sound reference sample. This finding suggests that a significant proportion of weakfish that overwinter off NC are similar in morphology to those found in Long Island Sound in spring... It appears likely that a majority of these fish are found in Delaware Bay and north in spring, and Chesapeake Bay and north in fall (Scoles 1990).
- Overall, there are two groups of weakfish (north and south) along the Atlantic coast. Still, due to the coastal nature and migratory behavior of the species, these groups are not genetically isolated. However, there is sufficient evidence that gene flow (and subsequent recruitment) is so limited that the southern aggregations make minimal contributions to northern populations (Chapman et al, unpublished report).

Conclusion

Two main contrasting findings emerged from the literature on weakfish population structure along the Atlantic Coast. On the one hand, most genetic studies indicate that the weakfish population is genetically homogeneous. On the other, this population appears to be a mosaic of reproductively interdependent regional (even natal estuaries) sub-populations, as evidenced by meristic, morphological and life-history traits and, more importantly, by one genetic investigation and otolith chemistry studies.

On the basis of this information, pragmatism may dictate whether assessment and management strategies have to deal with a single unit-stock or separate unit-stocks (i.e., isolated, panmictic sub-populations, characterized by uniform growth, mortality, fecundity parameters, etc.). Gulland (1988) considered definition of a unit-stock an essentially operational matter, being tied to the models used, the questions asked, and information available. Stock structure is not considered in the striped bass assessment, even though stock structure is well defined.

It is worth keeping in mind that subdividing a stock (initially treated as a whole) into sub-stocks may meet homogeneity requirements, but it presents potential for not meeting the condition of isolation. One may expect that multi-stock assessment models will be complex and data-consuming, without any warranty of improvement in comparison with single-stock assessment models. In fact, the complications here potentially stem from (i) weakfish migration patterns (e.g., Mercer 1985, 1989), resulting in unknown mixing rates between various fishing locations, and (ii) the sequential nature of the exploitation along the weakfish migratory route.

Technical Committee Position

On June 20, 2006, the Weakfish Technical Committee (TC) reviewed this information on stock structure. Full agreement was reached on the following:

- Evidence of stock structure exists.
- Data is inadequate to define stock structure and there is enough potential mixing that pinpointing north/south (N/S) split is not possible at this time.
- If a north to mid-Atlantic subpopulation is in serious decline, this does not warrant a N/S split based on conservation concerns.
- The Board needs to clarify its reasoning for a N/S split.
- Does the Board want research recommendations on this from the TC?

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Overview of the Weakfish Resource

By Jim Uphoff, ASMFC Weakfish Technical Committee Chair

Weakfish fisheries were subject to increasingly restrictive management during the early and mid-1990s and were thought to be on the path to recovery. Between 1999 and 2003, recreational and commercial landings of weakfish along the Atlantic coast plummeted to all-time lows while population estimates from traditional single species assessments reached all-time highs. This dichotomy of assessment results and fishery performance lead the Weakfish Technical Committee to consider less traditional assessment techniques.

A preponderance of evidence from these alternative approaches indicated that a large rise in natural mortality that started in the mid-1990s largely caused weakfish biomass and size structure to decline greatly by 2003. These declines could not be attributed to a slight rise in fishing mortality. When concerns about weakfish first arose during the late 1980s and early 1990s, estimates of fishing mortality were high. Conservative management measures were imposed beginning in the early 1990s and fishing mortality rates fell to modest levels after 1994. Biomass and size structure rebuilt for a time, but underwent a rapid decline after the late 1990s.

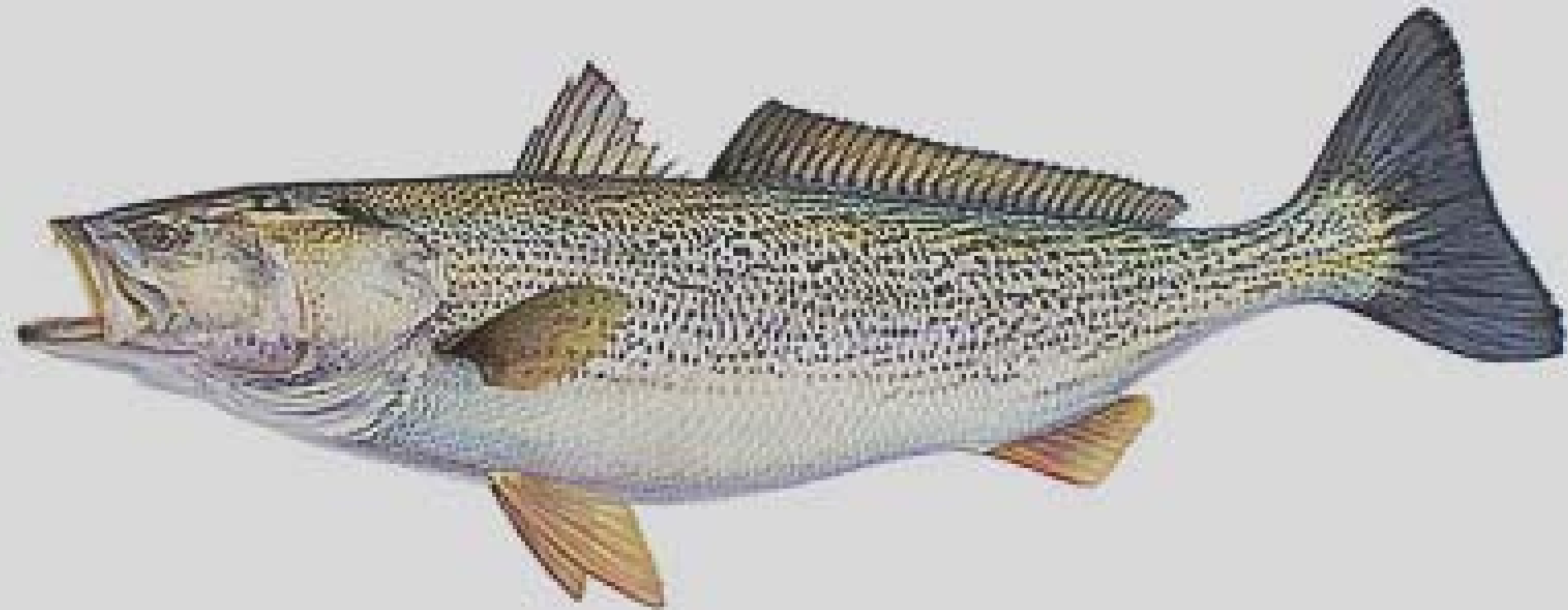
High and rising natural mortality does not provide managers much leverage for recovery by managing the weakfish fishery alone. However, projections indicated that cuts in fishing mortality will be needed for timely recovery if natural mortality declines.

In the past, we have presumed that ceasing exploitation on an overfished stock has to result in recovery. This presumption has ignored uncertainty imposed by ecological systems. The rapid decline in recent weakfish biomass and landings was reminiscent of rapid transitions between extended periods of high or low commercial landings dating back to the late 1920s. In theory, these rapid changes could reflect an underlying environmental driver whose effect has been accelerated by high fishing or predation rates.

The Technical Committee developed and tested specific hypotheses to evaluate candidate predator/competitors (striped bass, summer flounder, bluefish, spiny dogfish and Atlantic croaker), forage species (Atlantic menhaden, bay anchovy, and spot), climatic factors (water temperature and North Atlantic Oscillation index), high discard losses, and overfishing. Insufficient forage, especially Atlantic menhaden, and increased predation by striped bass have emerged from these analyses as leading candidates behind rising natural mortality. Contributions by remaining candidates or others may exist, but have not been detected or tested.

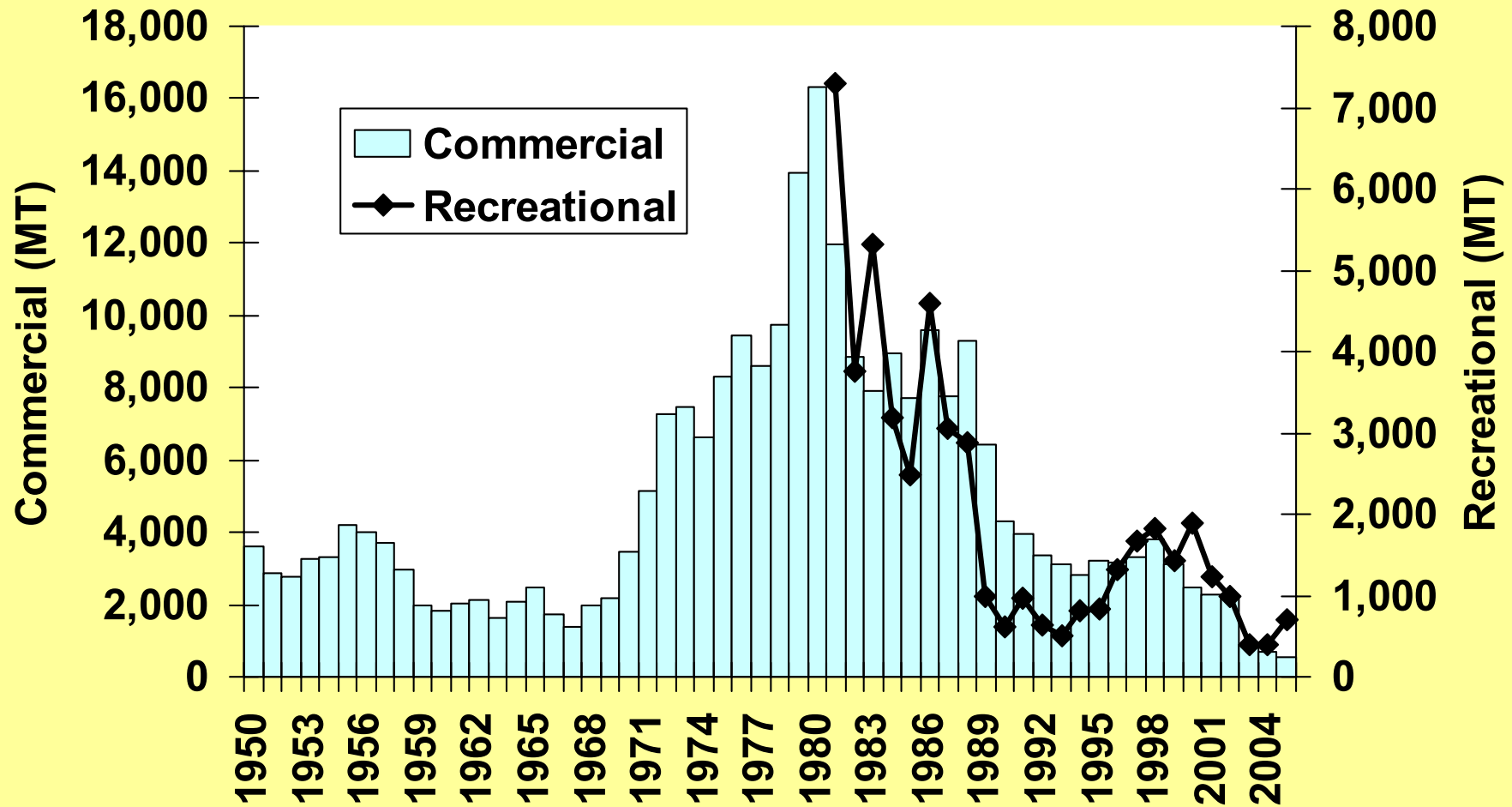
The following presentation was provided to the Commission's Weakfish Management Board at its August 2006 meeting. It was prepared to address questions raised by the Management Board and External Peer Review Panel regarding fishery status and possible next steps in the management program.

What's Next with Weakfish?

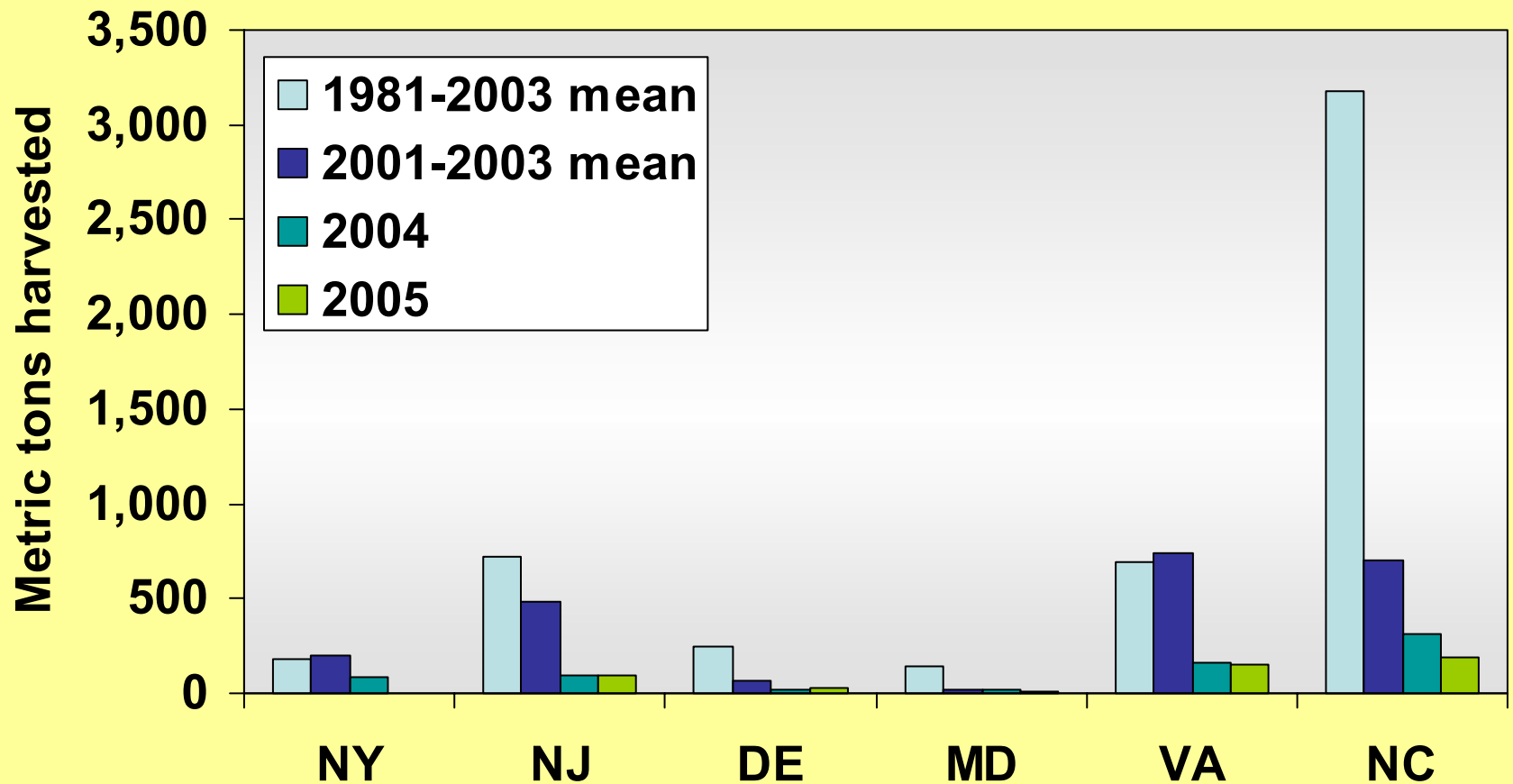


**A Report of the Weakfish Technical Committee
- Fishery Status & Board Requests
August, 2006**

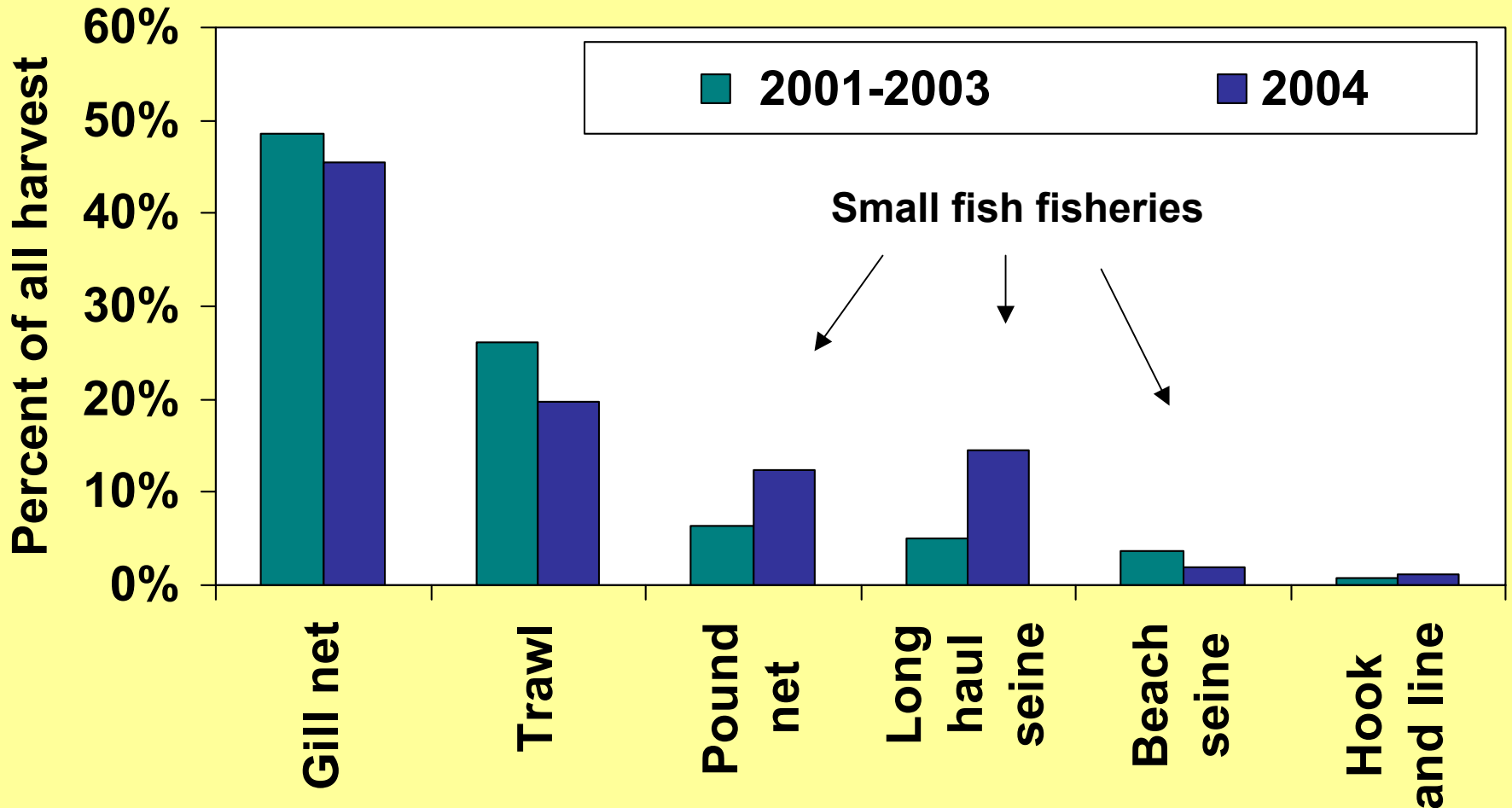
Atlantic coast harvest of weakfish, 1950-2005



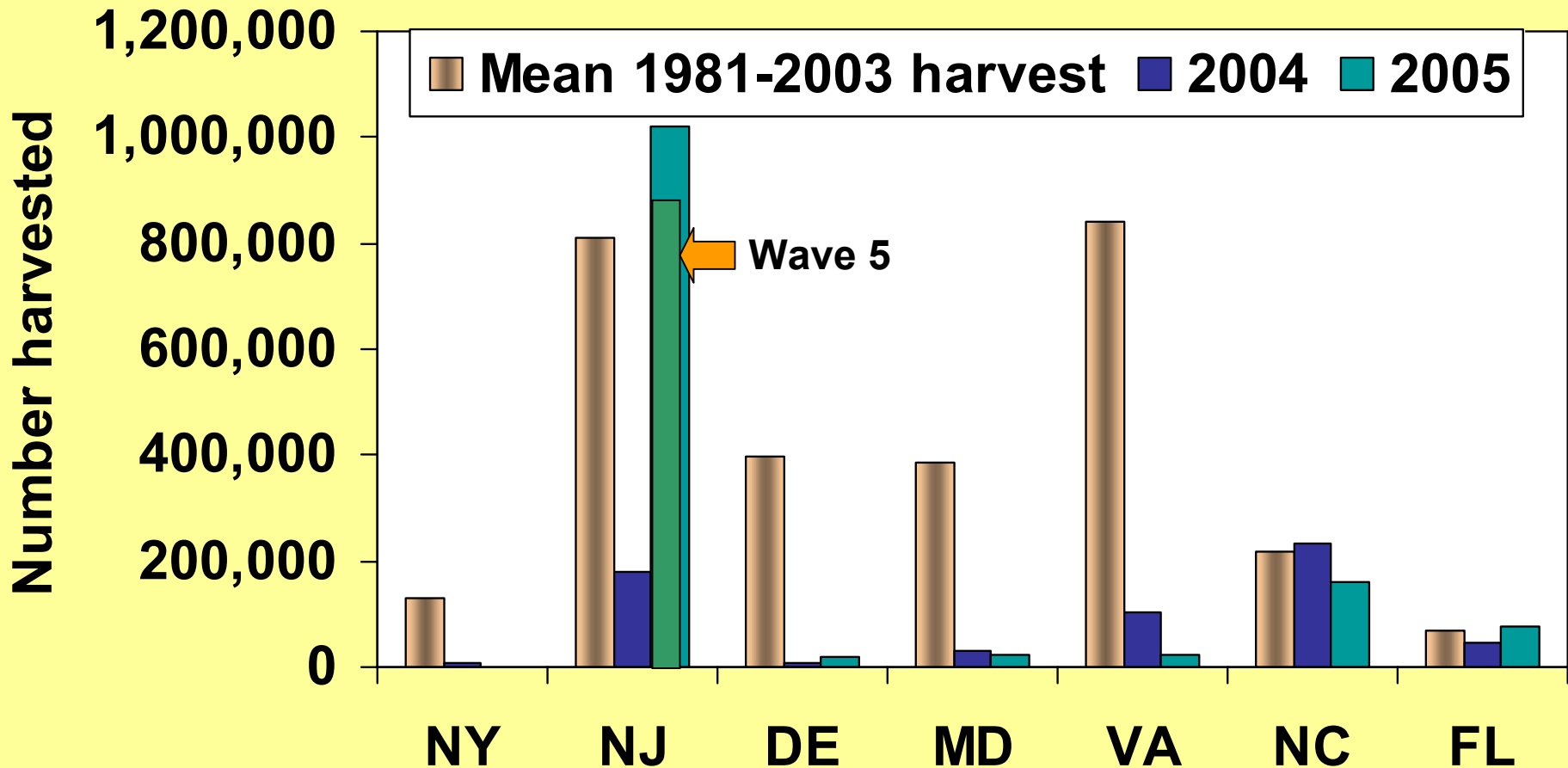
Average commercial harvest by state during 1981-2003 and 2001-2003 vs 2004 and 2005 (incomplete)



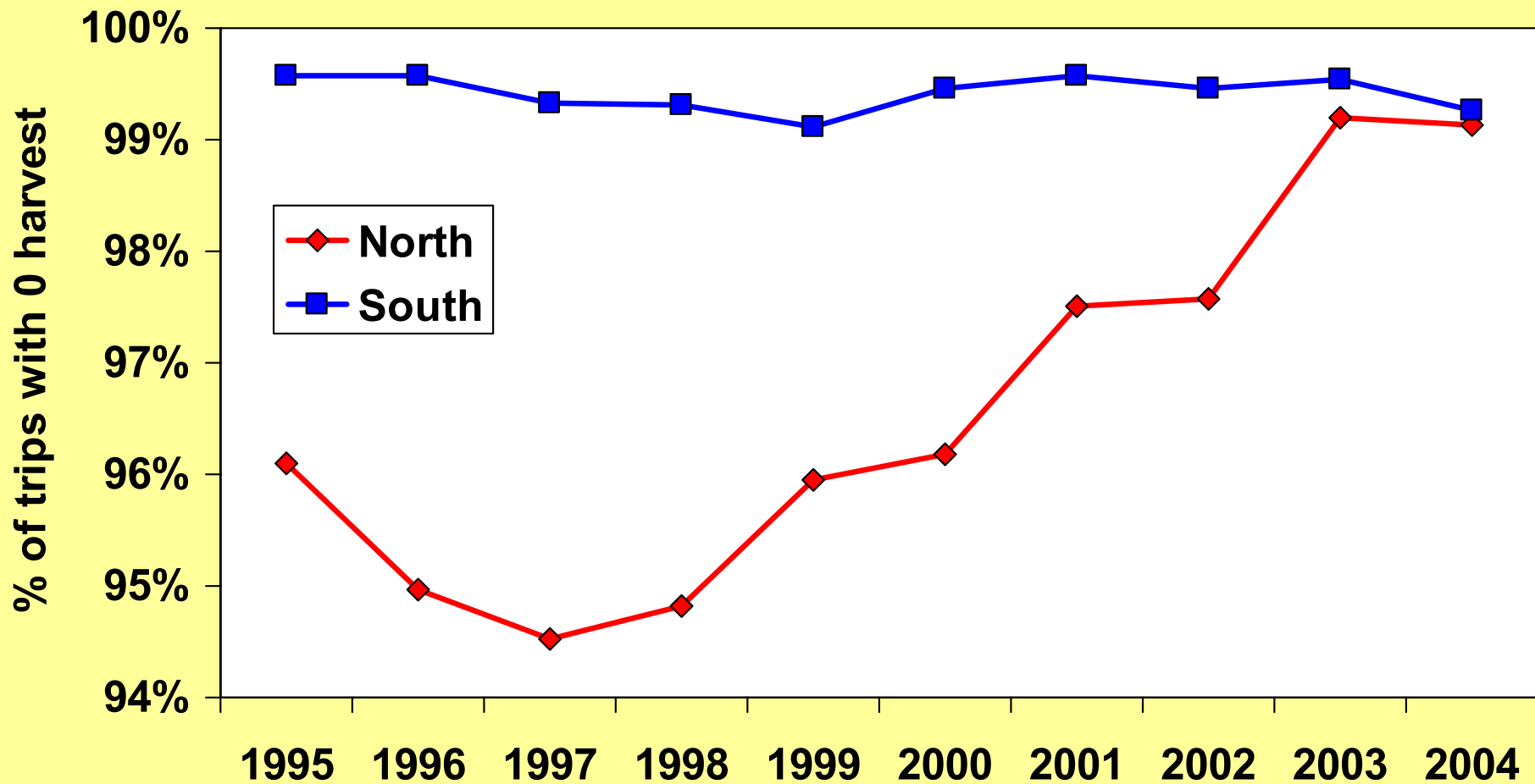
Distribution of mean 2001-2003 commercial catch and 2004 by gear



Average 1981-2003 and 2004, 2005 recreational harvest, in numbers, by state



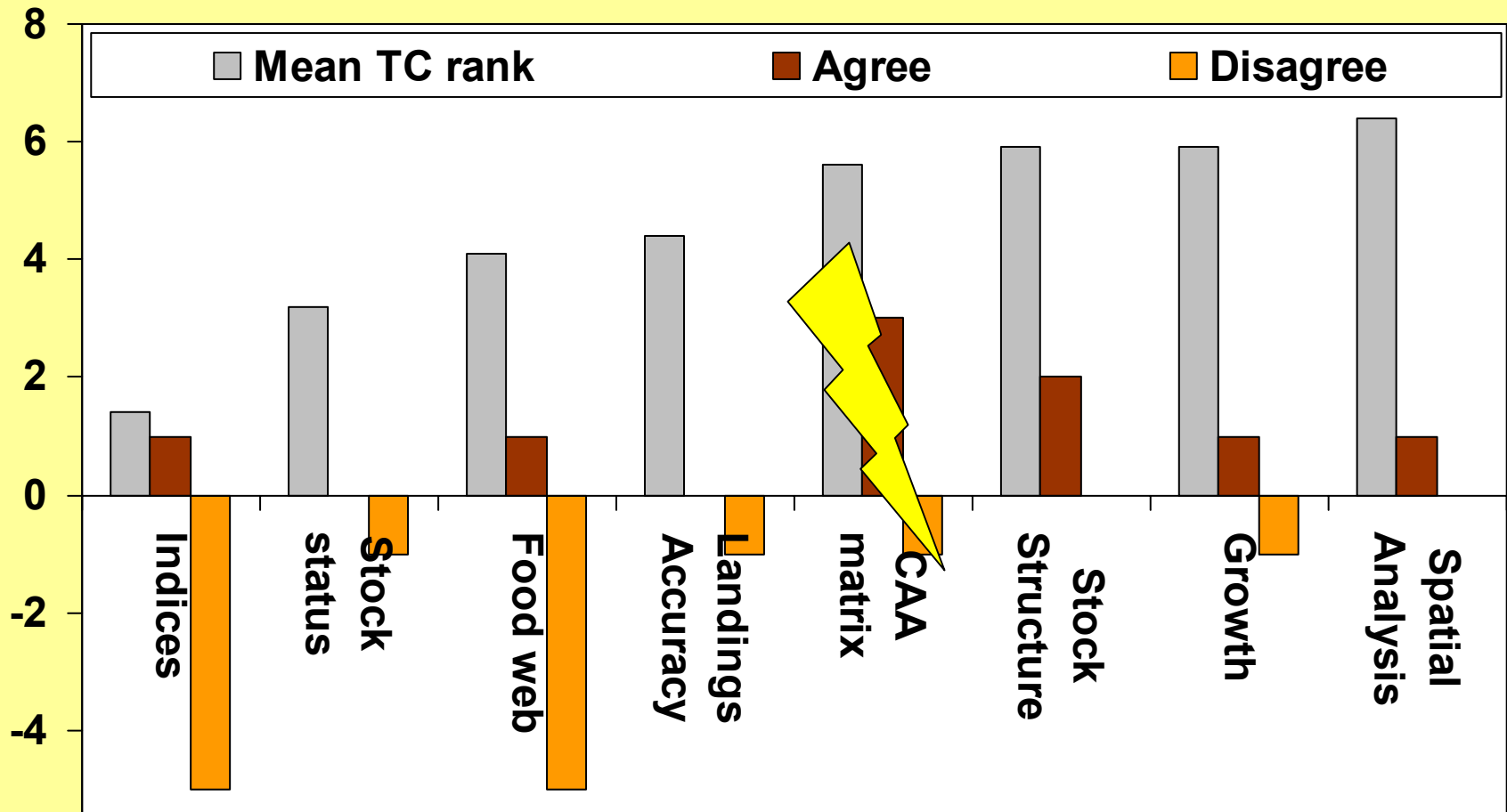
Percent of all recreational trips in the north (NY-VA) & south (NC-FL) that did not harvest weakfish



40th SARC Recommendations

- The TC did not disagree with recommendations
- 6 of 8 recommendations were implemented in the year between reviews
- 2 not implemented required much more time (CAA review) or additional capability (spatial analysis may need GIS to examine anything but gross patterns).

TC agreements and disagreements with 2006 peer-review statements. Lowest mean rank = most important. Agree or disagree displays number of specific statements.



Agreements or disagreements usually unanimous

Indices – main issue is NEFSC trawl survey vs MRFSS index

- TC chose MRFSS catch index
- MRFSS index suggested at 40th SARC
- Panel rejected MRFSS – catchability
- TC rejected NEFSC – catchability, spontaneous generation, poor size representation
- Striped bass, bluefish, croaker assessments have MRFSS indices

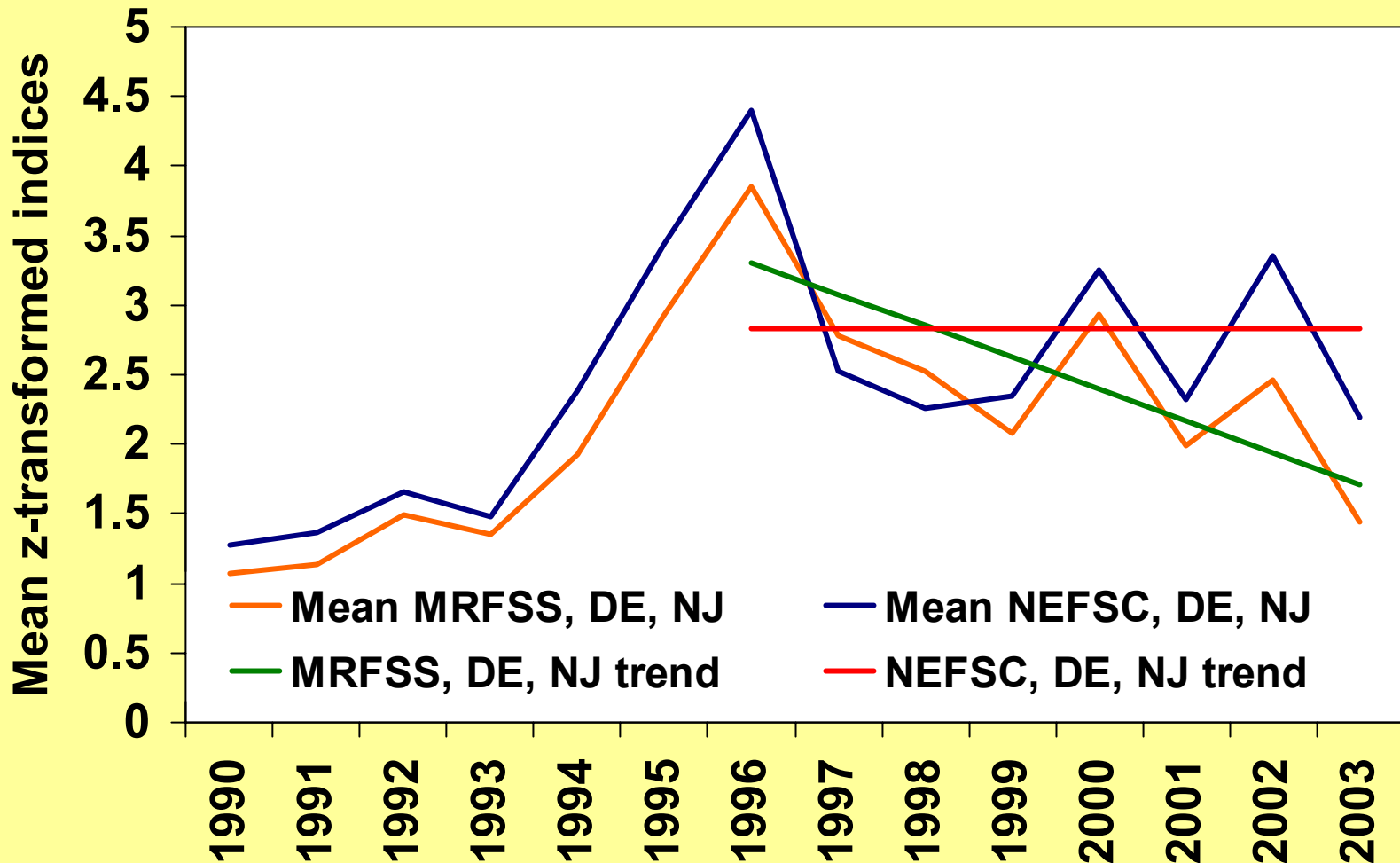
Indices – No correlation of MRFSS vs NEFSC

Disagreement requires choice

Green = $P < 0.05$, Yellow = $P < 0.10$, Red = NS

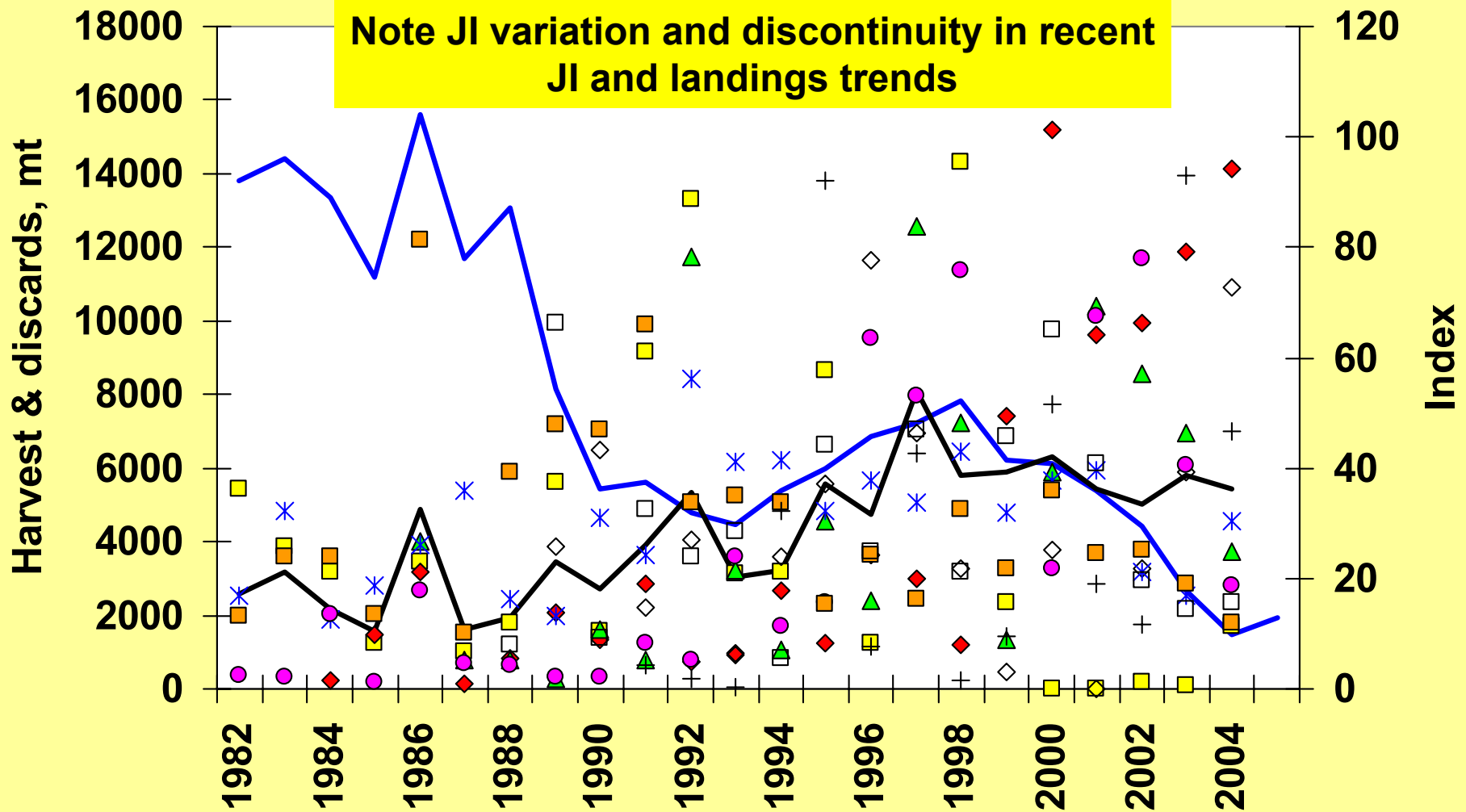
Index	NJ	DE	NEFSC
MRFSS	0.59	0.75	-0.05
NJ		0.78	0.66
DE			0.52

Consequences – very different view of recent trend depending on whether MRFSS or NEFSC is averaged with DE & NJ surveys (all standardized)



Increasing trend in JI a cause for optimism?

Standardized age 0 weakfish indices versus harvest + discards (blue line). JI grand mean (black line) indicates general JI trend; symbols = different surveys.

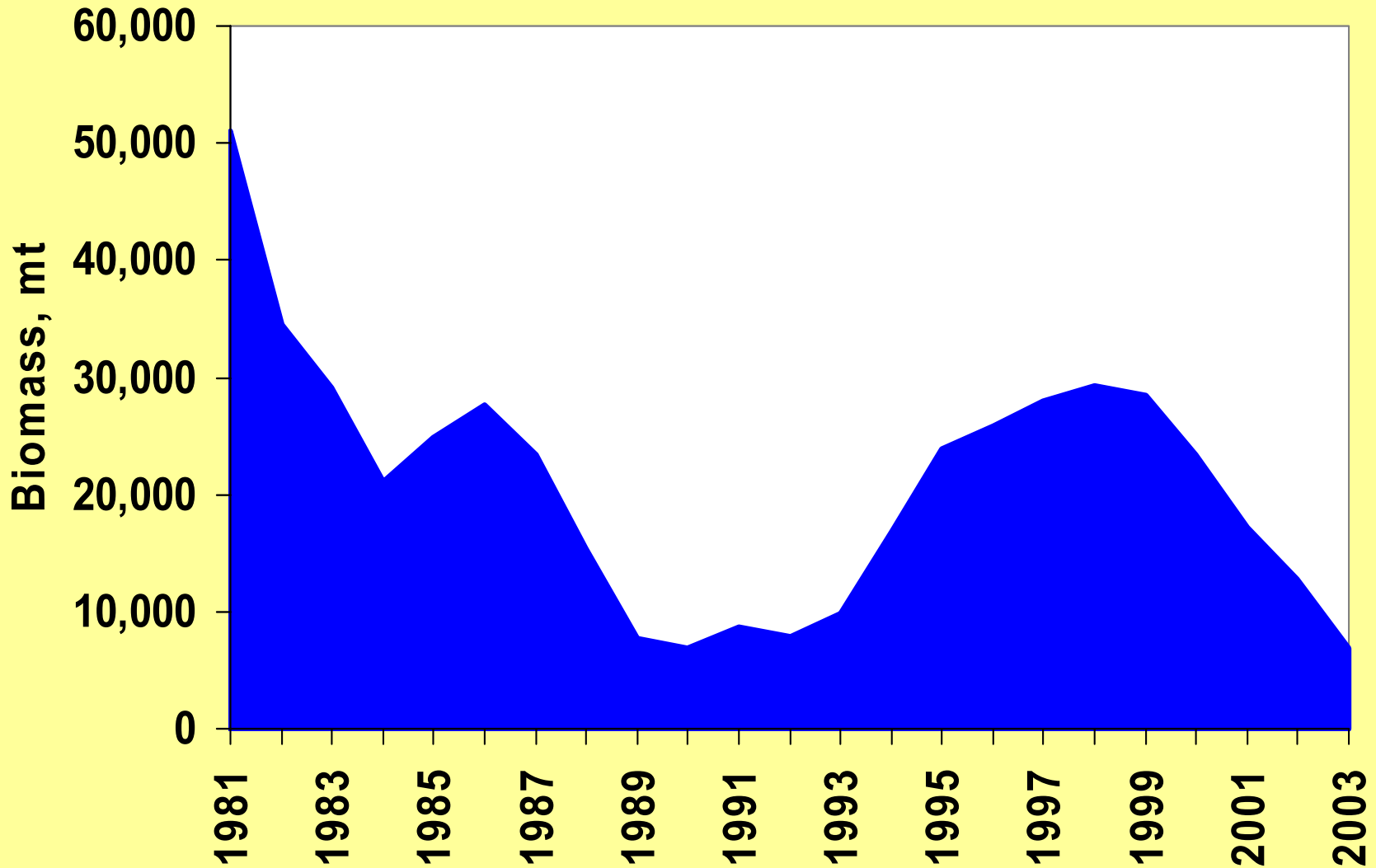


Stock Status: Review didn't offer view

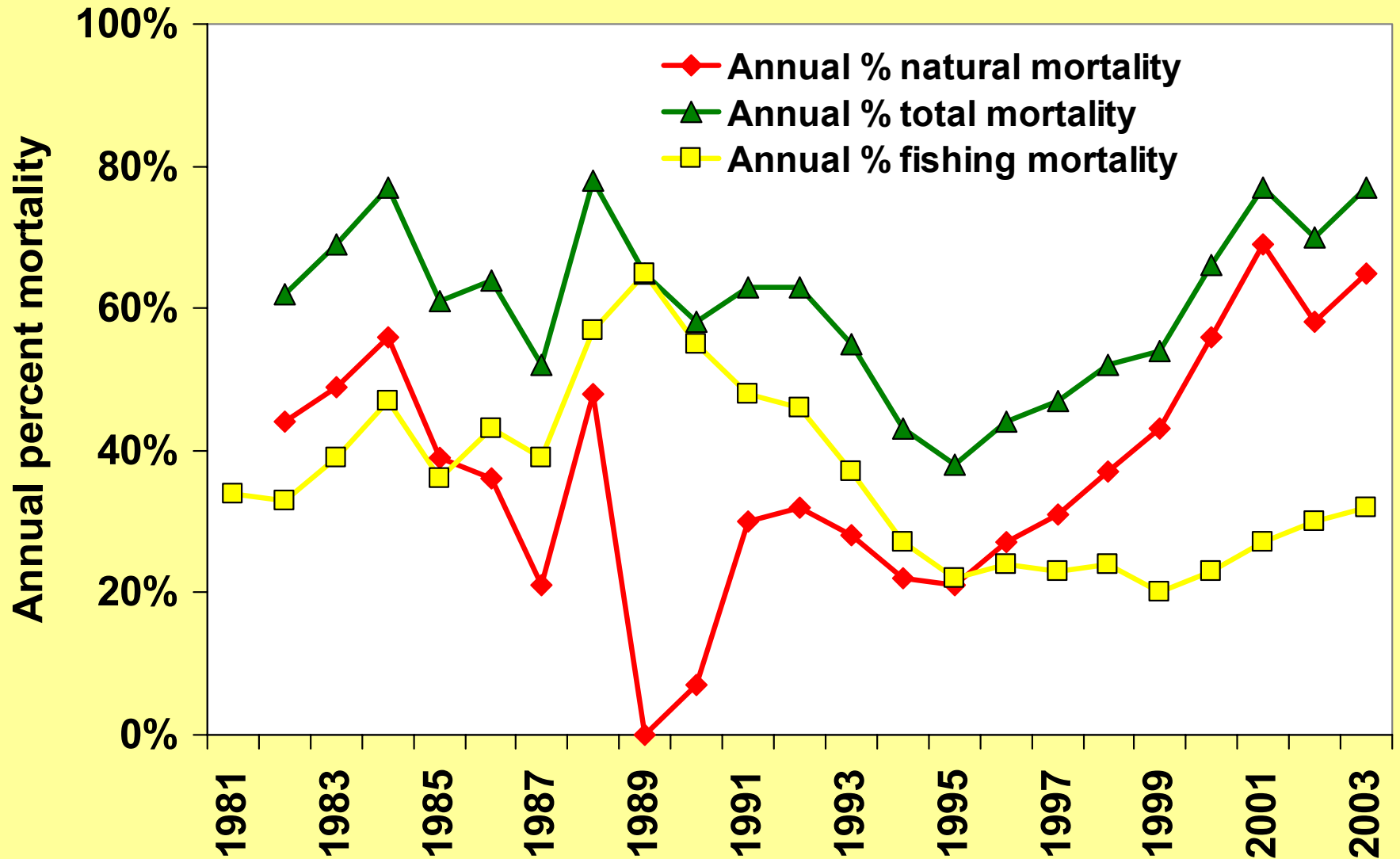
TC position:

- Stock declining
- Total mortality increasing
- Not much evidence of overfishing
- Something other than F is going on (increasing M)
- Strong chance regulating fishery won't help.

Estimated weakfish biomass during 1981-2003.



Annual total, fishing and natural mortality. These rates have been translated into annual percentages, so fishing and natural mortality are not additive.



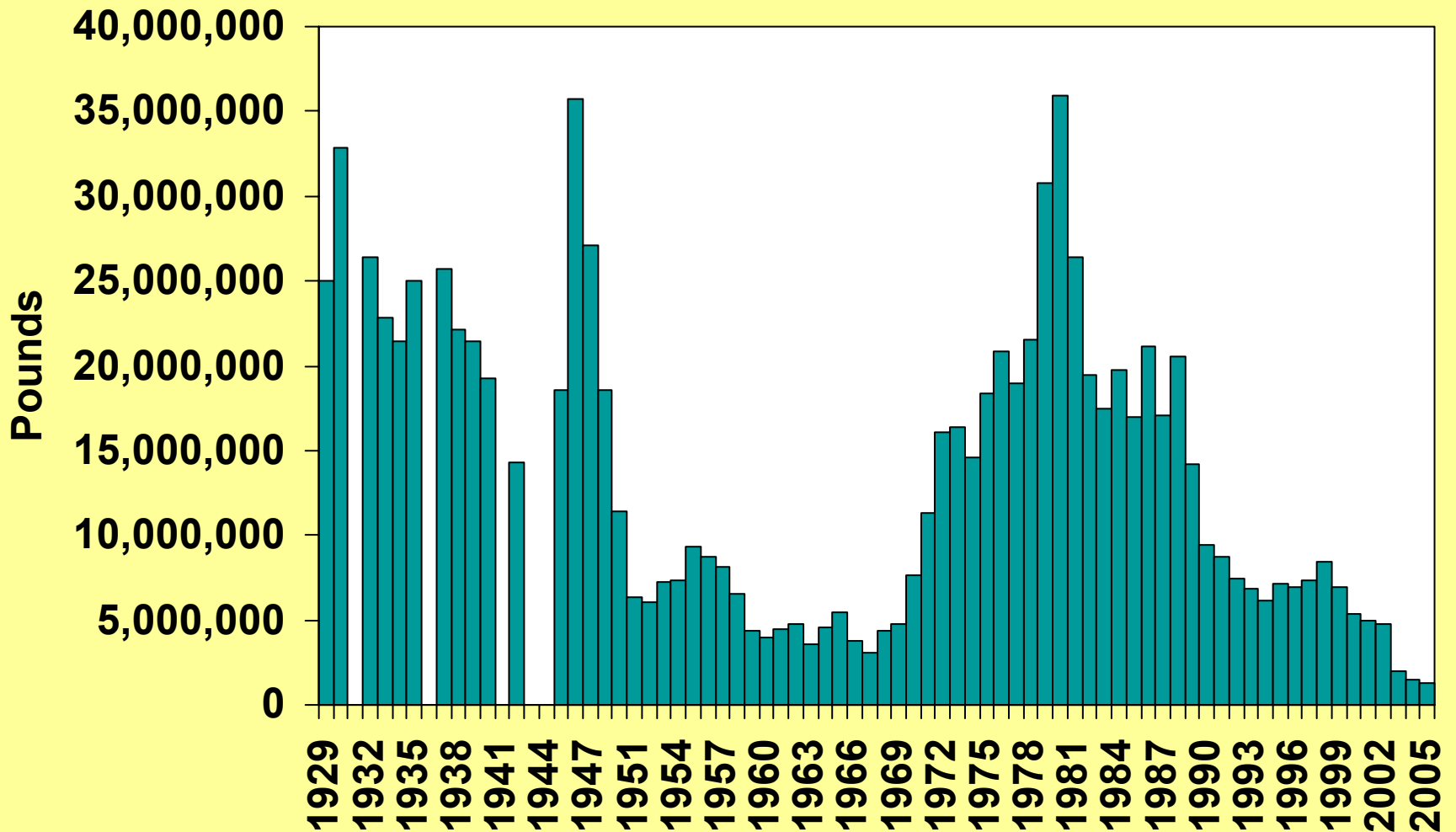
Major Reason for Decline?

- Neither TC, 40th SARC, nor peer-review implicated directed F as lead cause
- TC considers food-web dysfunction as most plausible contributor to high M
- Panel suggested undetected commercial discards lead to high F
- Food-web or discards indicate little direct leverage on stock and do not present a clear-cut solution

Food-Web Conclusions

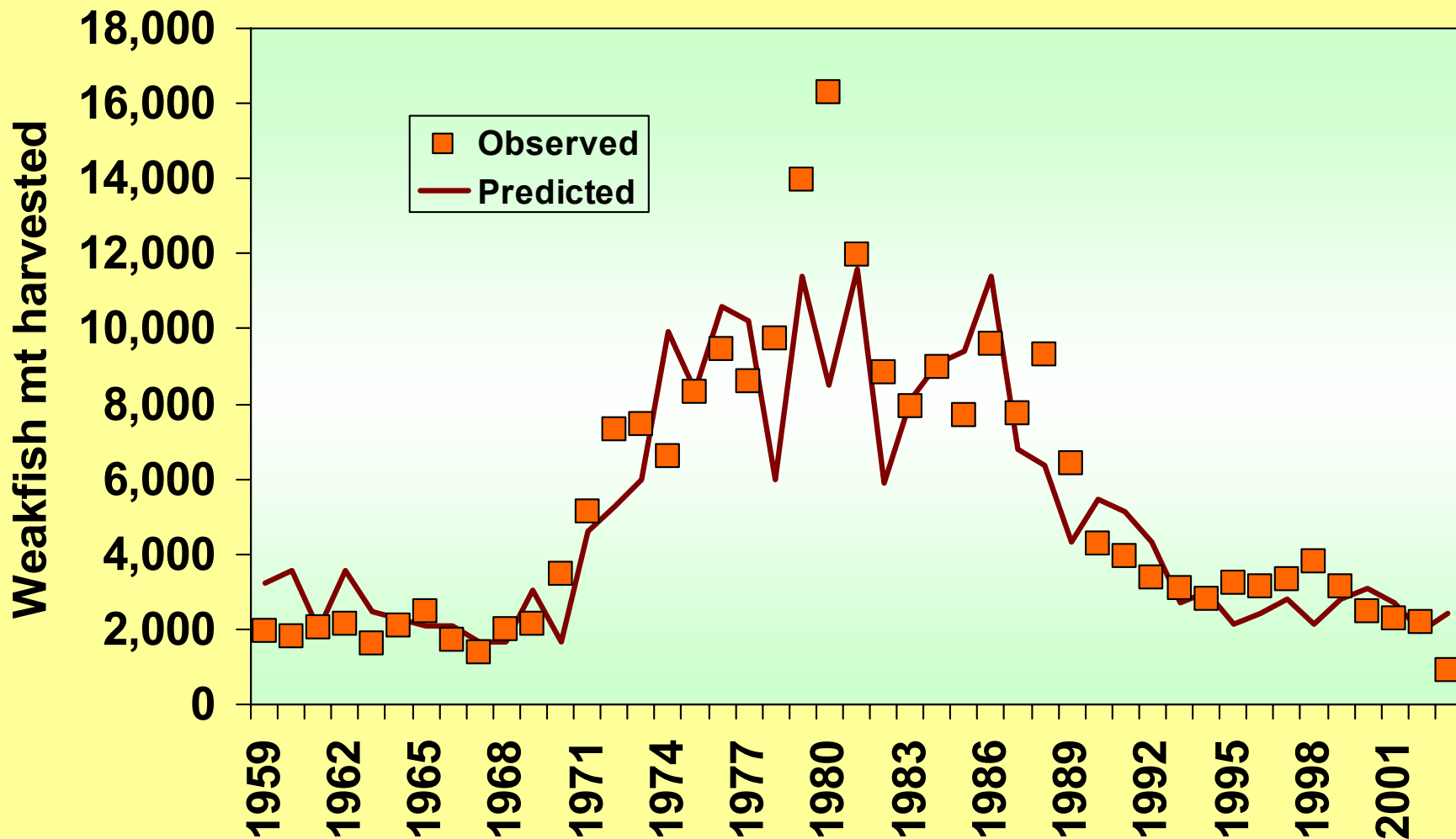
- Diet bottleneck plausible
- High-low weakfish biomass domains evident in long-term landings.
- Now in low domain – cutting F may not allow for recovery because M high.
- Menhaden has large influence over long-term domains.
- Adding bass to single-species model necessary to fit recent decline.
- Bass-weakfish model applies to low domain.

High-low domains appear in 1929-2005 commercial landings

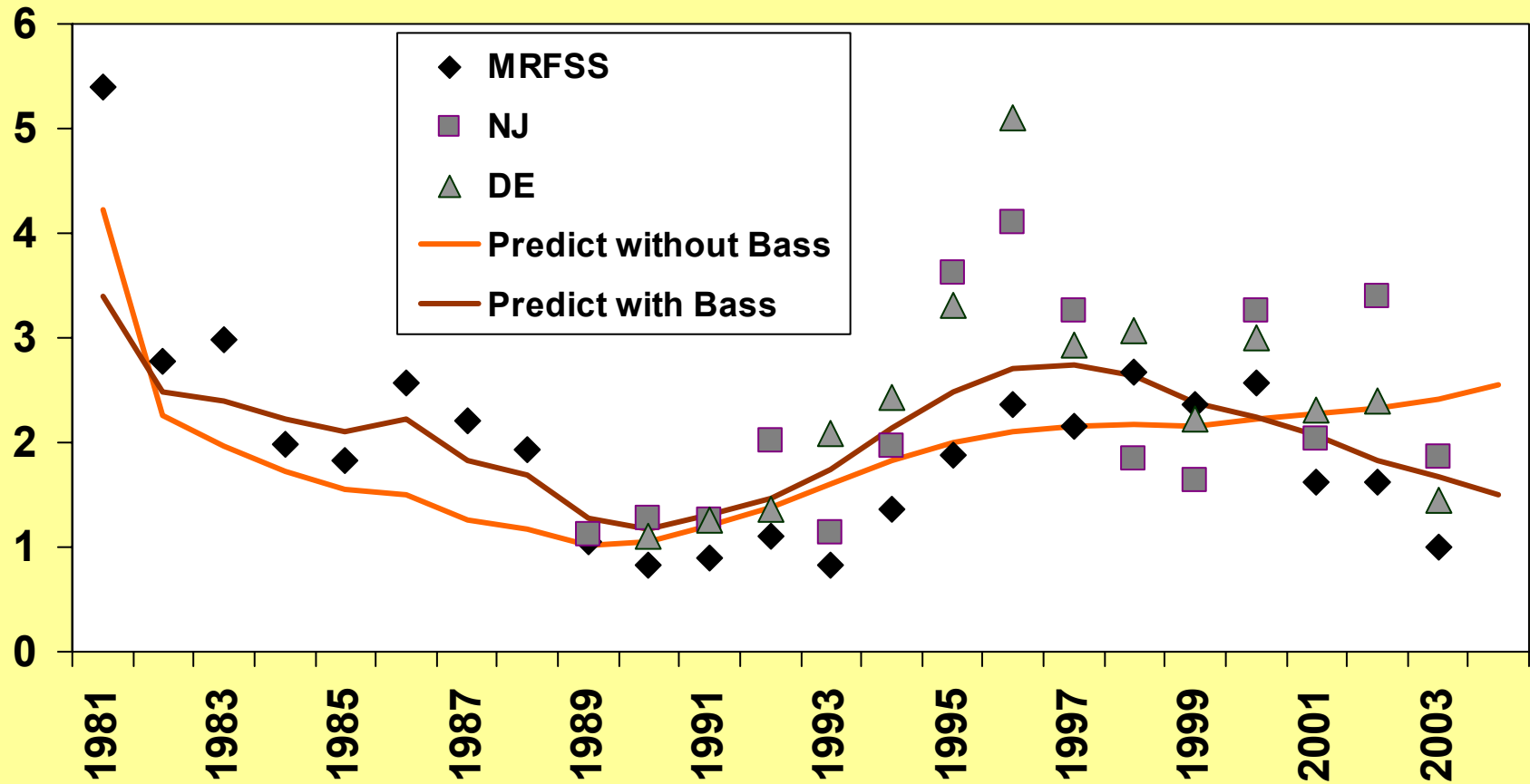


Food-web. Fit to weakfish commercial landings from multiple regression of large bass index and menhaden juvenile index (both terms significant).

Menhaden partial $r^2 = +0.73$; bass partial $r^2 = -0.03$.

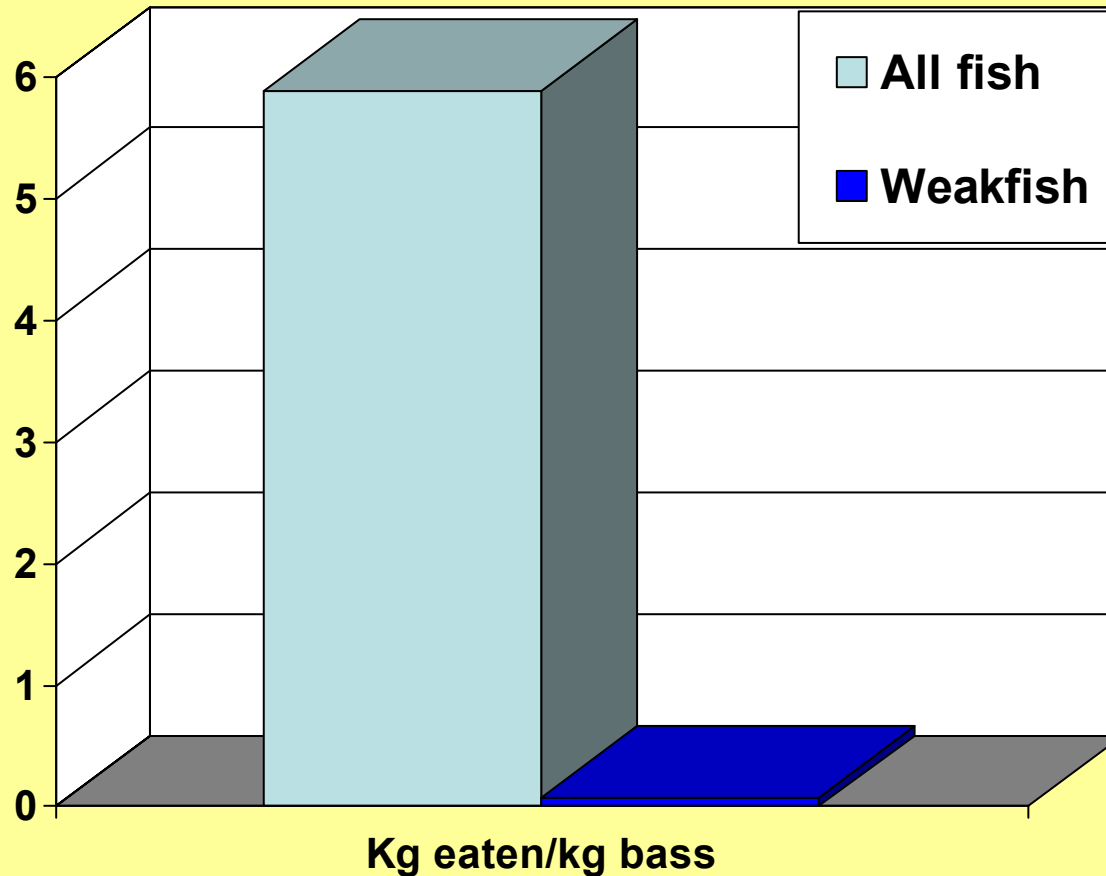


Food-web. If bass losses aren't included, you don't get a declining trend in recent years. DE and NJ surveys included with MRFSS.



Food Web –weakfish only need to be a very minor constituent of bass diet.

Comparison of bass bioenergetics total fish consumption and bass-weakfish model consumption



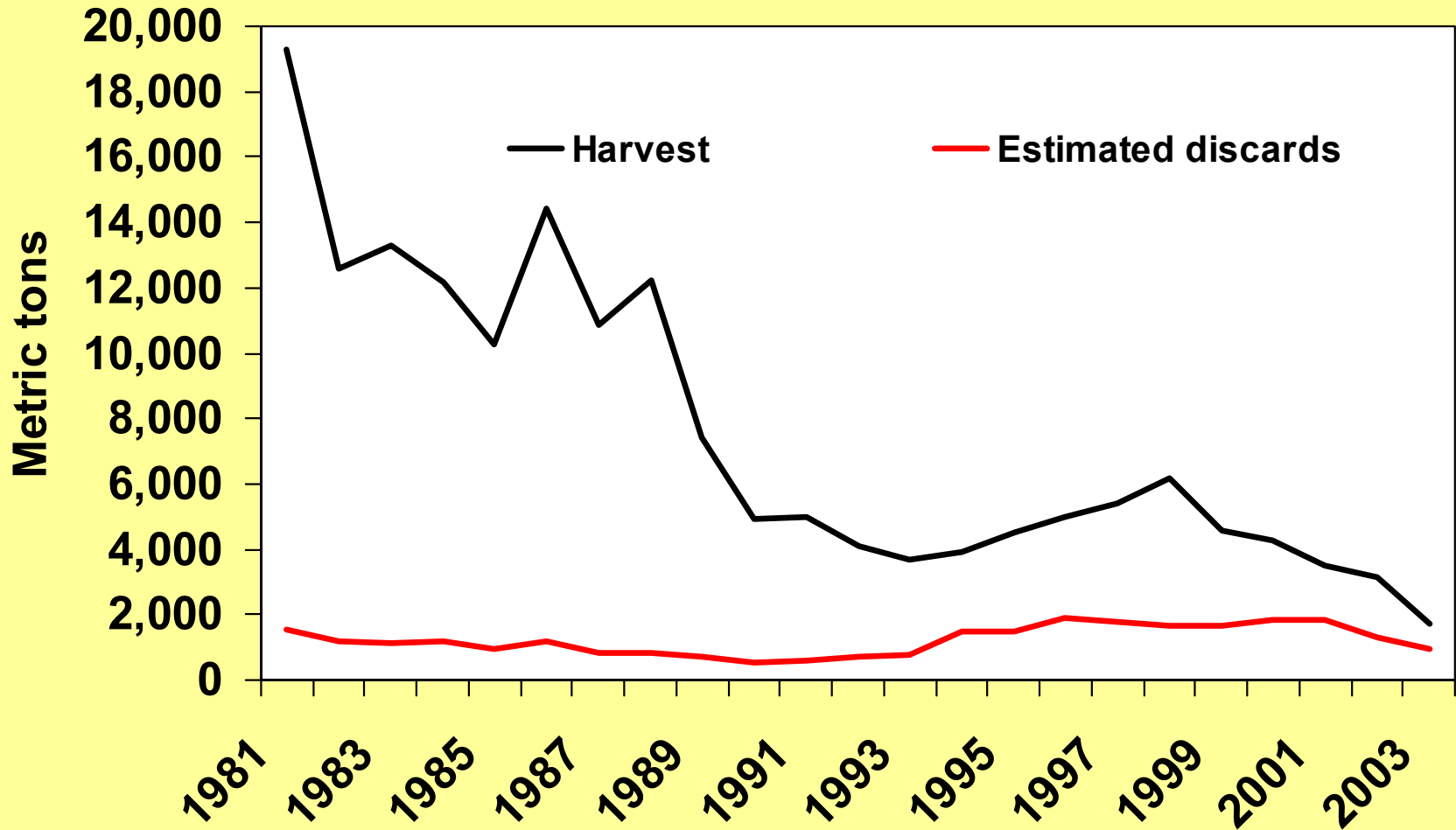
Landings Inaccurate?

- We use the same commercial and recreational landings as other ASMFC assessments
- Commercial discards estimated by technique used for Atlantic croaker (passed peer-review)
- Recreational ages have to be “borrowed” in any assessment

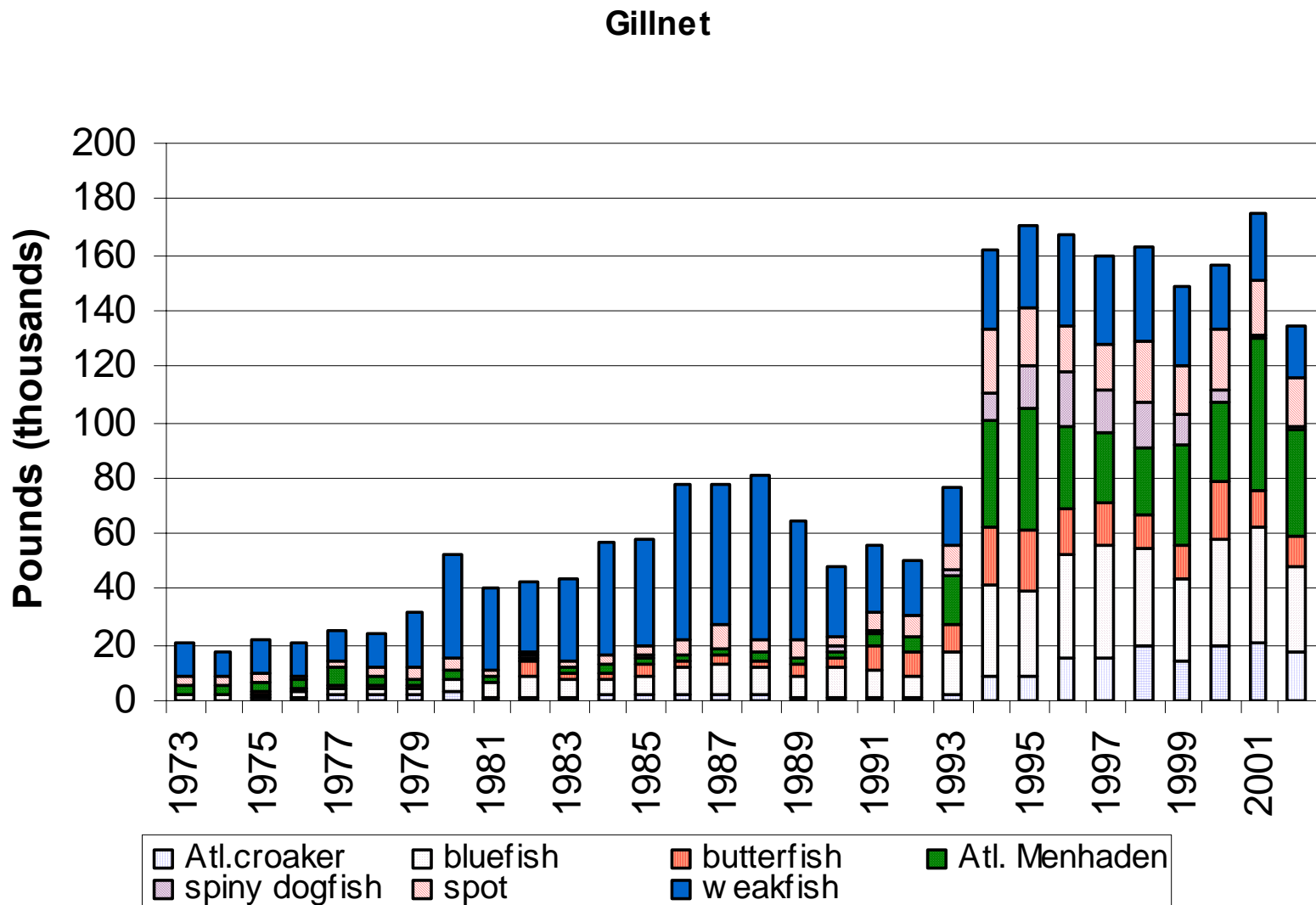
Commercial Discards

- De Silva (2004) thoroughly estimated gill net and trawl discards during 1994-2003 from ratios of harvest to discards in observer samples
- Precision poor to adequate (95% CI overlaps 0 about 50% of estimates)
- Estimates prior to 1994 estimated from 1994-2002 market discard ratios
- Estimates after 1994 based on market & regulatory discards
- Loss estimates on high side - all commercial assumed dead (+ 20% recreational)

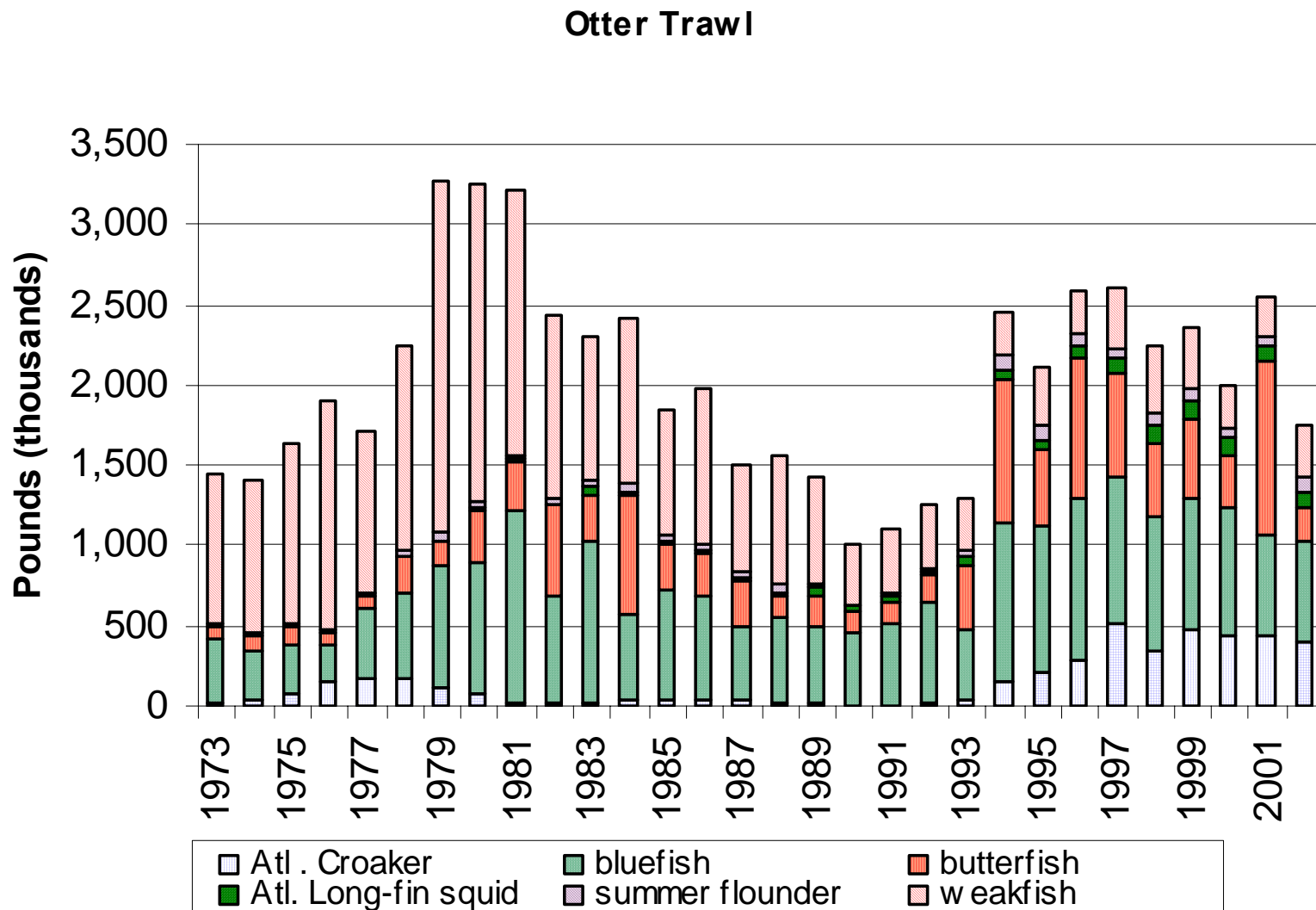
Reported harvest and estimated discards (recreational and commercial combined)



Weakfish discards from gillnets in the mid-Atlantic by target species. Note discard estimates prior to 1994 were based on non-regulatory discard ratios and include regulatory discards after 1993. From de Silva (2004)



Weakfish discards from otter trawl in the mid-Atlantic by target species. Note discard estimates prior to 1994 were based on non-regulatory discard ratios and include regulatory discards after 1993. From de Silva (2004)

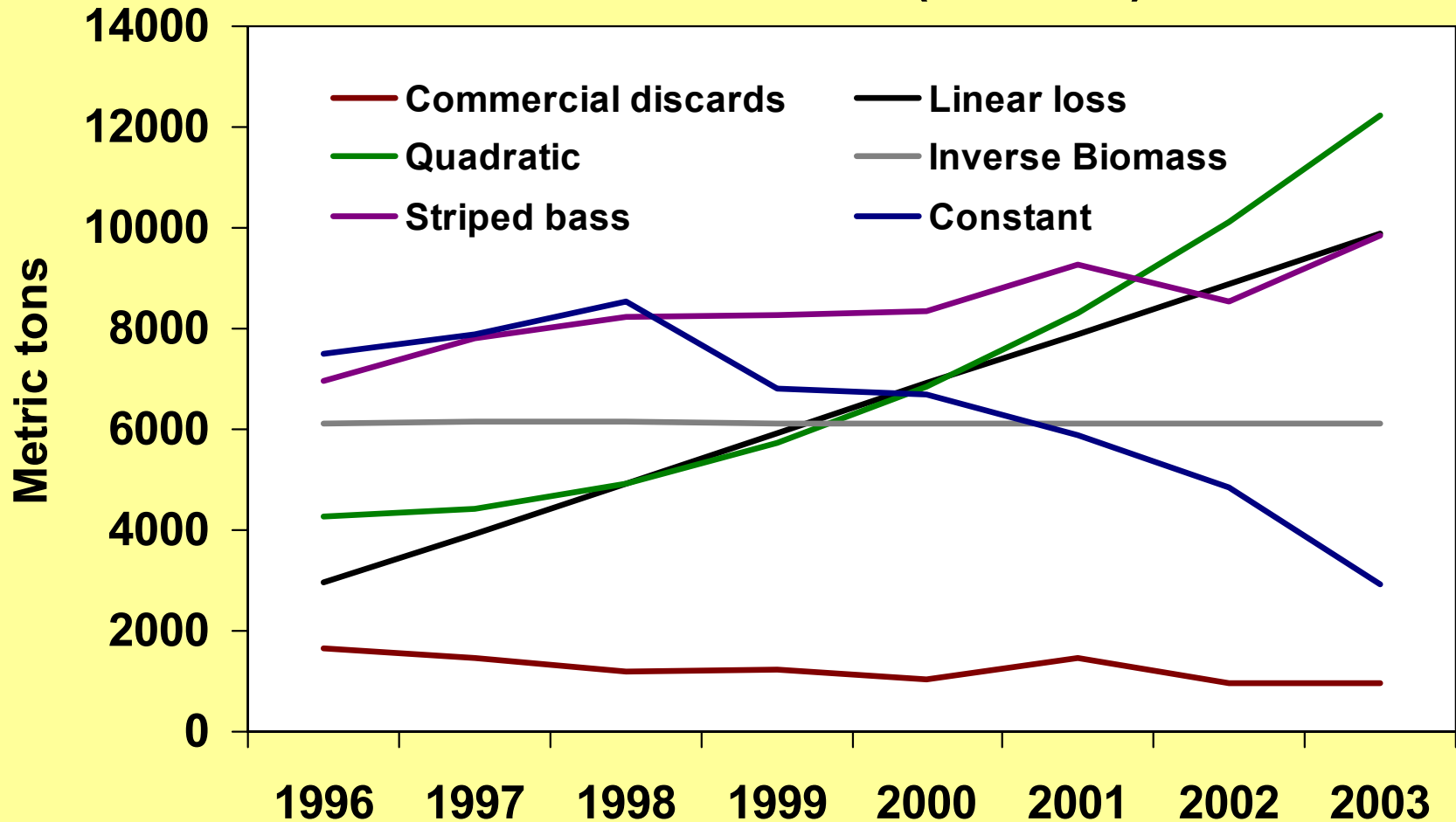


How much commercial “ghost” discard loss do you need to produce a decline?

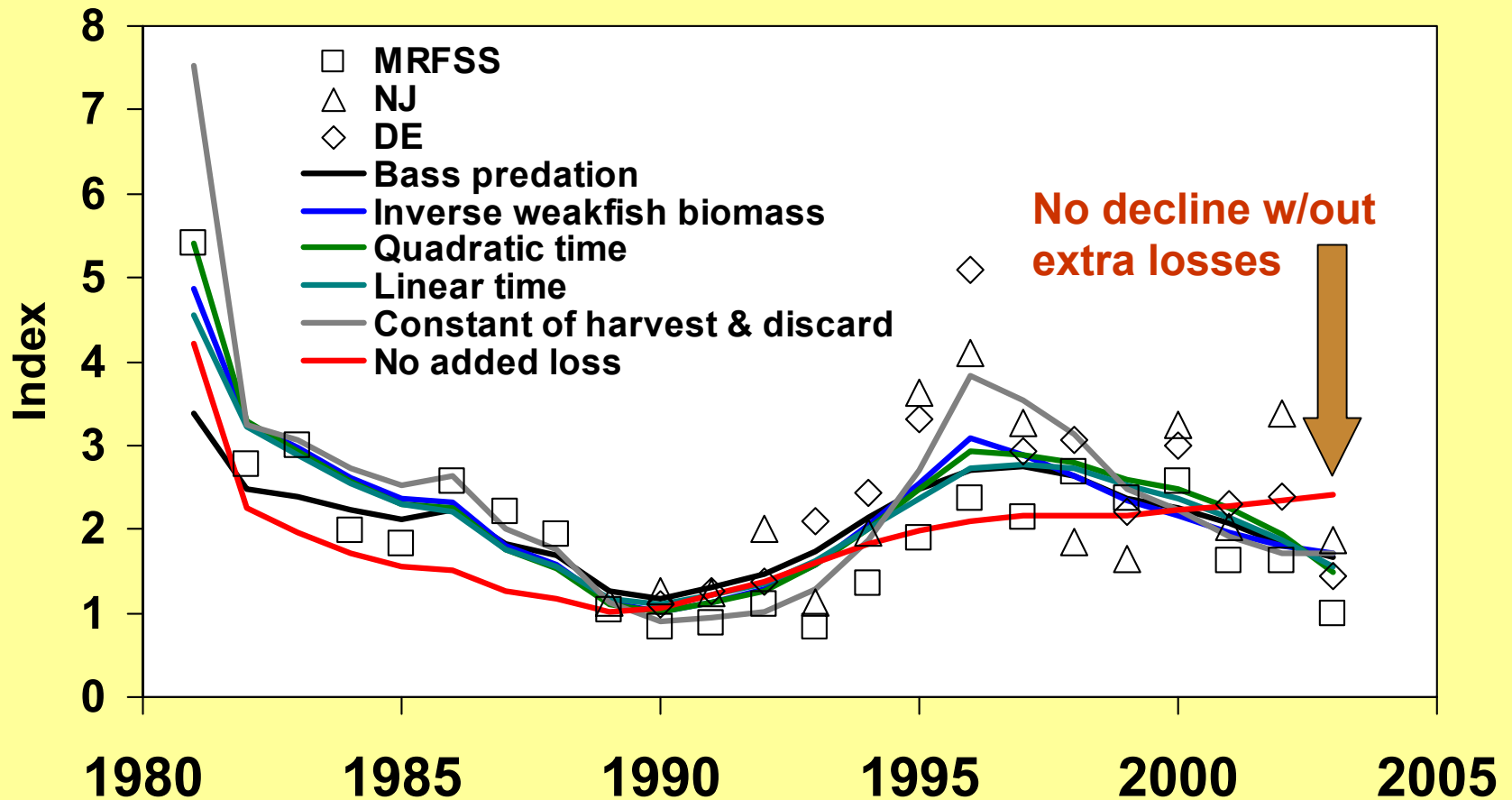
- Peer-review postulated unmeasured commercial (“ghost”) discards as explanation for decline.
- Biomass dynamic model with generic additional losses (time trends, fc of current losses, inverse biomass) tested hypothesis.
- Contrast with no extra loss and striped bass predation
- Used MRFSS, NJ & DE biomass indices.
- “Ghost” losses modeled to start after Amendment 2 (begin in 1996).
- Produce range of possible estimates.
- Search for fisheries capable of producing them.

“Ghost” losses needed to create decline compared to estimated commercial discards.

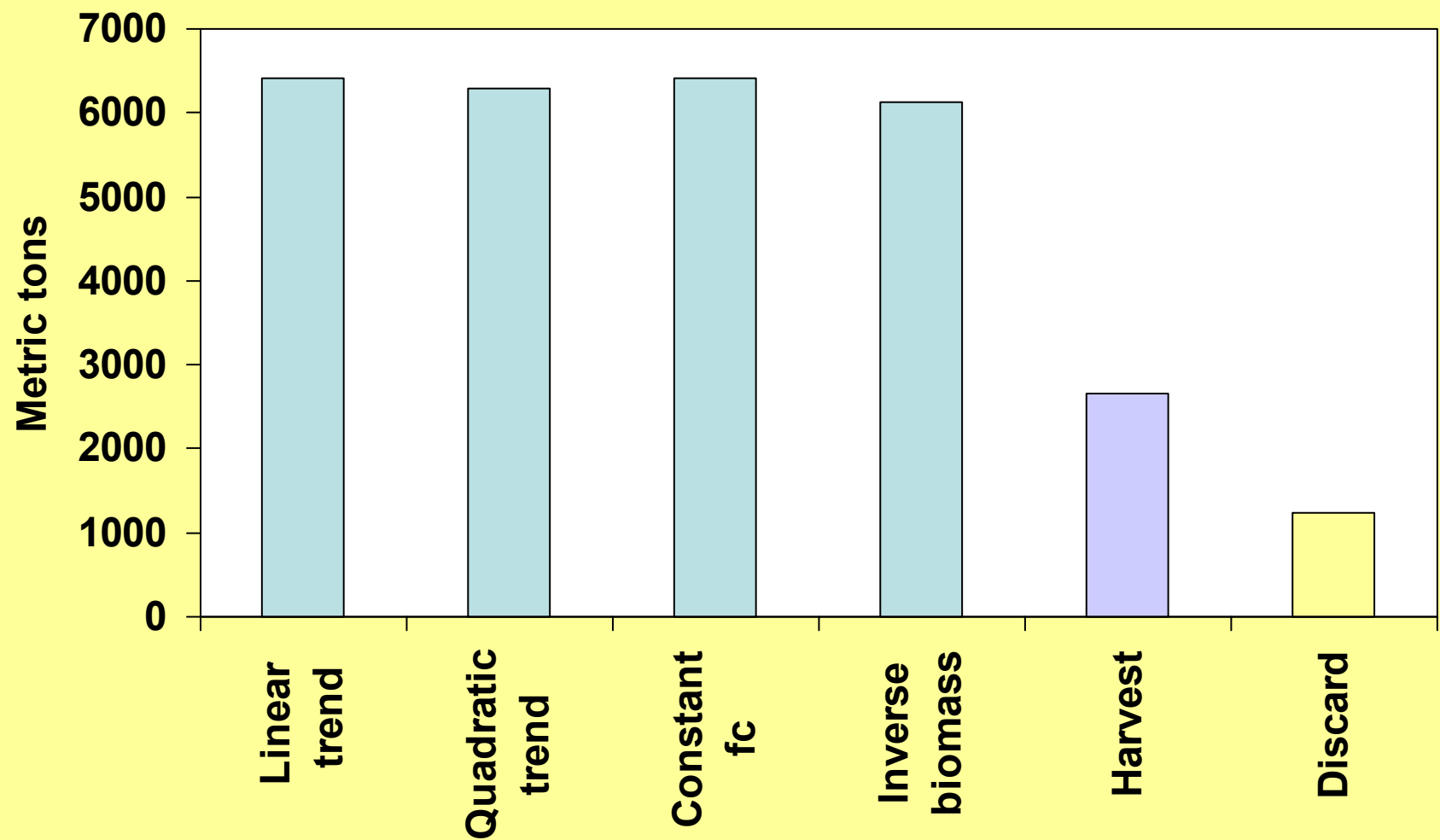
All fit about the same ($R^2 \approx 0.6$).



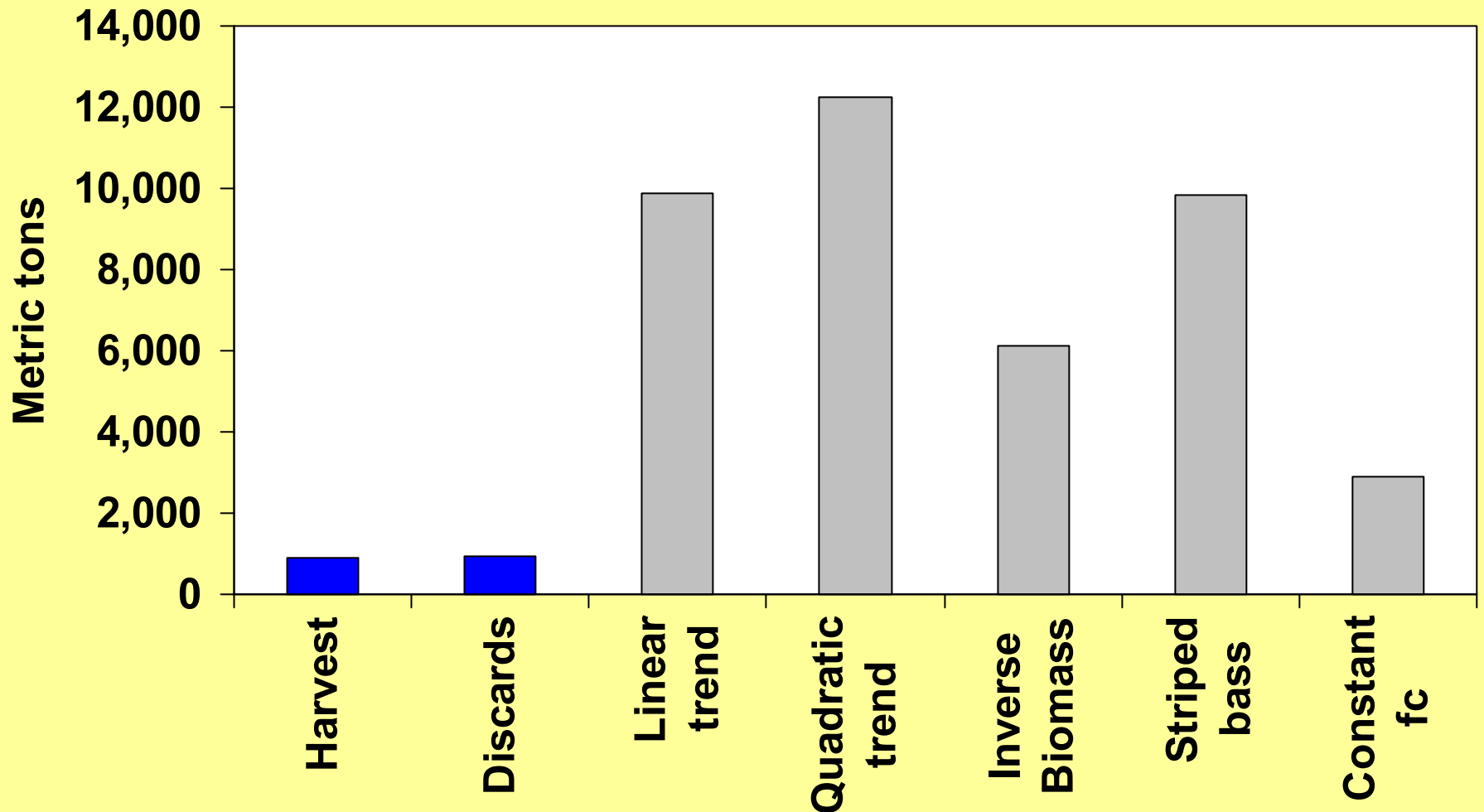
Fit of biomass dynamic model using original harvest + discards with and without additional loss functions. MRFSS = private boat catch per trip index; NJ and DE are trawl survey indices. NJ and DE are trawl survey indices.



Averages of 1996-2003 commercial "ghost" losses from models, observed commercial harvest, and estimated commercial discard losses



Metric tons of additional losses in 2003 from different loss scenarios contrasted with commercial harvest and discard estimates



Discard models - conclusions

- Discard losses needed to create decline manifold higher than estimated discards
- Little \$ incentive for harder fishing that would lead to this magnitude of discards
- TC could not offer candidate fisheries
- Other species should be showing same symptoms



Ghost Discard Management Conundrum

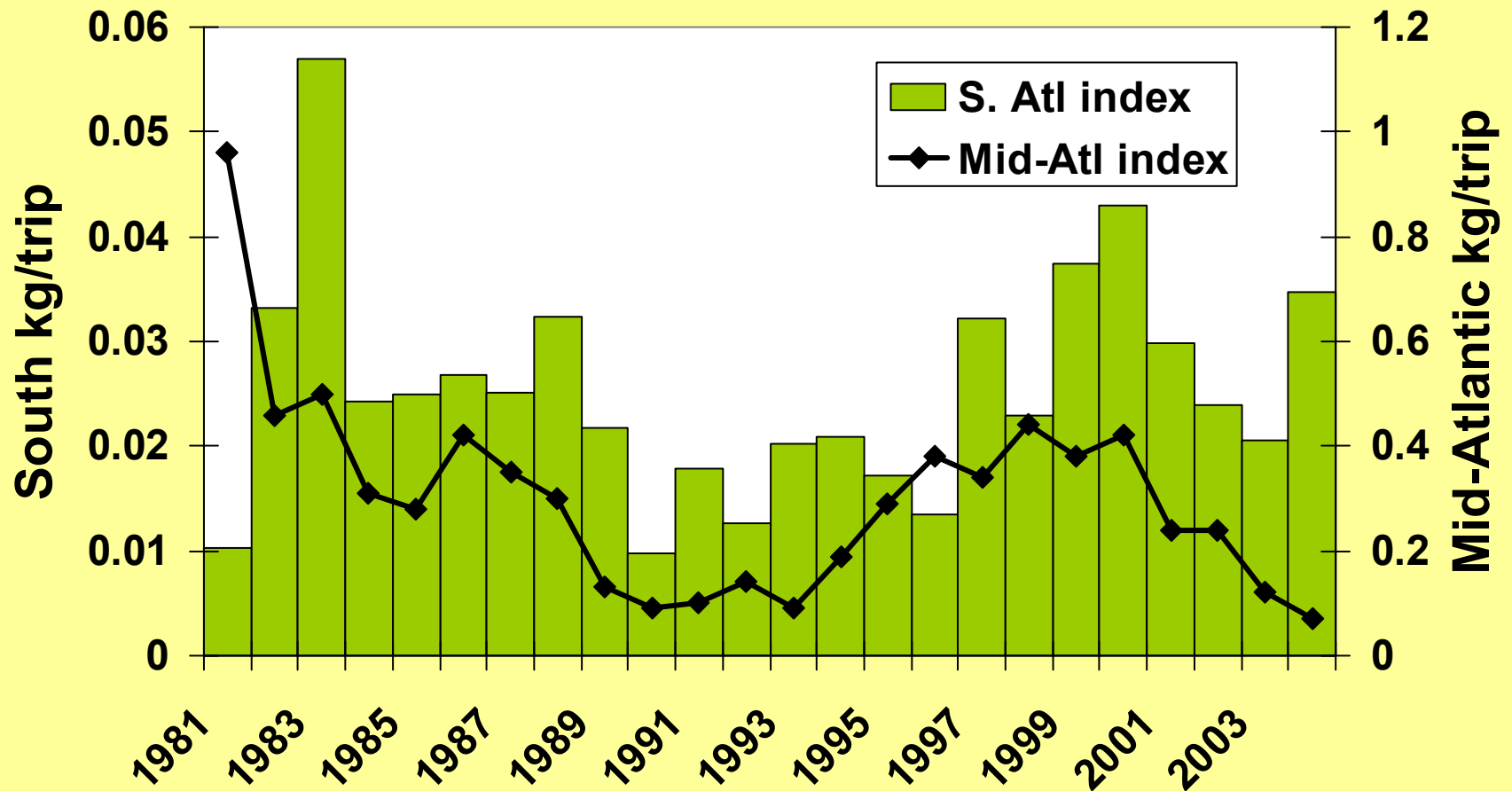
- If “ghost” commercial discards are responsible for decline, then they likely resulted from increased regulation starting with Amendment 2.
- How will more regulation, short of stopping fishing on all species, stem a decline created by regulation?

Stock Structure

- Is there enough evidence to recommend that the Board consider a split?
- If so, what difference would be suggested?
- Where should the split occur?

South versus mid-Atlantic relative biomass indices suggest different trends

(MRFSS; mean weight harvested * catch per private boat trip; includes releases)



Genetic analyses indicate homogeneous stock

**Tagging, otolith microchemistry, meristics,
and morphology indicate stock structure**

This information was reviewed by the Technical Committee in June 2005 and 2006

The Technical Committee declined recommending a north / south split on both occasions.

June 2006 TC Meeting

- Evidence of stock structure exists.
- Data inadequate to define structure
- Enough potential mixing, can't pinpoint N/S split.
- If north to mid-Atlantic stock is in serious decline, this does not warrant split based on conservation concerns.
- Board needs to clarify reasoning for split.
- Does the Board want stock structure research recommendations from the TC?