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A Report of the 48th Northeast Regional Stock Assessment Workshop

48th Northeast Regional Stock Assessment Workshop (48th SAW)

Assessment Report

by Northeast Fisheries Science Center

NOAA's National Marine Fisheries Serv., 166 Water St., Woods Hole MA 02543

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National Oceanic and Atmospheric Administration
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Foreword

The Northeast Regional Stock Assessment Workshop (SAW) process has three parts: preparation of stock assessments by the SAW Working Groups and/or by ASMFC Technical Committees / Assessment Committees; peer review of the assessments by a panel of outside experts who judge the adequacy of the assessment as a basis for providing scientific advice to managers; and a presentation of the results and reports to the Region's fishery management bodies.

Starting with SAW-39 (June 2004), the process was revised in two fundamental ways. First, the Stock Assessment Review Committee (SARC) became smaller panel with panelists provided by the Independent System for Peer Review (Center of Independent Experts, CIE). Second, the SARC provides little management advice. Instead, Council and Commission teams (e.g., Plan Development Teams, Monitoring and Technical Committees, Science and Statistical Committee) formulate management advice, after an assessment has been accepted by the SARC. Starting with SAW-45 (June 2007) the SARC chairs were from external agencies, but not from the CIE. Starting with SAW-48 (June 2009), SARC chairs are from the Fishery Management Council's Science and Statistics Committee (SSC), and not from the CIE. Also at this time, some assessment Terms of Reference were revised to provide additional science support to the SSCs, as the SSC's are required to make annual ABC recommendations to the fishery management councils.

Reports that are produced following SAW/SARC meetings include: An *Assessment Summary Report* - a brief summary of the assessment results in a format useful to managers; this *Assessment Report* – a detailed account of the assessments for each stock; and the SARC panelist report – a summary of the reviewer's opinions and recommendations as well as individual reports from each panelist. SAW/SARC assessment reports are available online at http://www.nefsc.noaa.gov/nefsc/publications/series/crdlist.htm. The CIE review reports and assessment reports can be found at *http://www.nefsc.noaa.gov/nefsc/saw/*.

The 48th SARC was convened in Woods Hole at the Northeast Fisheries Science Center, June 1-4, 2009 to review three assessments (golden tilefish *Lopholatilus chamaeleonticeps*, ocean quahog *Arctica islandica*, and weakfish *Cynoscion regalis*). CIE reviews for SARC48 were based on detailed reports produced by NEFSC Assessment Working Groups for tilefish and ocean quahogs, and by the ASMFC Assessment Working Group for weakfish. This Introduction contains a brief summary of the SARC comments, a list of SARC panelists, the meeting agenda, and a list of attendees (Tables 1 – 3). Maps of the Atlantic coast of the USA and Canada are also provided (Figures 1 - 5).

Outcome of Stock Assessment Review Meeting:

Based on the Review Panel reports (available at http://www.nefsc.noaa.gov/nefsc/saw/ under the heading "SARC 48 Panelist Reports"), the SARC review committee concluded that the assessment terms of reference were satisfied for each of the three stocks.

For tilefish, neither of the two assessment models presented (an ASPIC surplus production model and a statistical, age-and-length-structured model fit to the CPUE and length-frequency data) fit the data well. However, because both models and their uncertainty were adequately investigated, the SARC was able to conclude that stock is not overfished and that overfishing is not occurring. The ASPIC model results suggested a recent increase in abundance; however, the commercial CPUE index has been declining in recent years in a manner consistent with the passage of a strong cohort through the stock, lack of age-structure in the population, and nonequilibrium stock conditions. Based on these considerations and some additional factors, the SARC review committee was not convinced that the stock had rebuilt to B_{TARGET}. They concluded that the tilefish projections are useful for displaying the extent of uncertainty in future stock size, but not for predicting future stock size. They also concluded that for the most recent years the biomass estimates from the ASPIC model are likely overestimates.

For ocean quahogs, the SARC felt commercial landings and fishing effort were well characterized, and the analyses were very thorough. As a whole, the stock is slowly being fished down to its B_{MSY} proxy reference point (1/2 of the virgin biomass), the stock is not overfished and overfishing is not occurring. The unique biology of ocean quahogs (slow growth, low levels of recruitment and very long-lived) creates time lags that are outside the planning horizons for most managed activities and presents unique challenges for the assessment of this stock.

For weakfish, multiple analyses were presented to provide estimates of abundance, total mortality and fishing mortality, including an ADAPT VPA, an analysis of survey data as abundance indices, and a Steele-Henderson production model including predation effects. There are technical issues with some of the modeling, but overall the analyses indicate that abundance has declined markedly, total mortality is high, non-fishing mortality has recently increased and that the stock is currently in a depleted state.

Table 1. 48th Stock Assessment Review Committee Panel.

48th Northeast Regional Stock Assessment Workshop (SAW 48)
Stock Assessment Review Committee (SARC) Meeting

June 1-4, 2009
Woods Hole MA

SARC Chairman:

Dr. Patrick Sullivan
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SARC Panelists (CIE):

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Table 2. Agenda, 48th Stock Assessment Review Committee Meeting.

48th Northeast Regional Stock Assessment Workshop (SAW 48)
Stock Assessment Review Committee (SARC) Meeting

June 1-4, 2009

Stephen H. Clark Conference Room – Northeast Fisheries Science Center
Woods Hole, Massachusetts

AGENDA* (version: 5-27-09)

TOPIC	PRESENTER(S)	SARC LEADER	RAPPORTEUR
Monday, 1 June			
10:00 – 10:30 AM			
Opening			
Welcome	**James Weinberg**, SAW Chairman		
Introduction	**Patrick Sullivan**, SARC Chairman		
Agenda			
Conduct of Meeting			
10:30 - Noon	Tilefish Assessment Presentation (A)		
	Paul Nitschke/ Mike Palmer/ Tiffany Vidal	**Jamie Gibson**	**Palmer/Vidal**
Noon – 1:00 PM	Lunch		
1:00 – 2:30 PM	SARC Discussion of Tilefish (A)		
	Patrick Sullivan, SARC Chairman		
2:30 – 3:00 PM	Break		
3:00 - 5:00 PM	Ocean quahog Assessment Presentation (B)		
	Larry Jacobson/ Toni Chute	**Mike Bell**	**Ralph Mayo**
5:00 – 6:00 PM	SARC Discussion of Ocean quahog (B)		
	Patrick Sullivan, SARC Chairman		

Tuesday, 2 June

9:00 – 10:15 AM	Revisit Tilefish Assessment with Presenters (A)
10:15 – 10:30 AM	Break
10:30 - Noon	Revisit Ocean Quahog Assessment with Presenters (B)
Noon – 1:00 PM	Lunch
1:00 – 3:45 PM	Weakfish Assessment Presentation (C) **Jeff Brust/ Vic Crecco/ Jim Uphoff** **Sven Kupschus** **Russ Allen**
3:45 – 4:00 PM	Break
4:00 – 5:30 PM	SARC Discussion of Weakfish (C) **Patrick Sullivan**, SARC Chairman

Wednesday, 3 June

9:00 – 10:15 AM	Revisit Weakfish Assessment with Presenters (C)
10:15 – 10:30 AM	Break
10:30 - Noon	Tilefish follow up + review Assessment Summary Report (A)
Noon – 1:00 PM	Lunch
1:00 – 3:00 PM	Ocean qua. follow up + review Assessment Summary Report (B)
3:00 – 3:15 PM	Break
3:15 – 5:15 PM	Weakfish follow up + review Assessment Summary Report (C)

Thursday, 4 June

9:00 – 10:15 AM	Final Revisits with presenters, if needed (A, B, C)
10:15 – 10:30 AM	Break
10:30 AM – 5 PM	SARC Report writing. (closed meeting)

*Times are approximate, and may be changed at the discretion of the SARC chair. The meeting is open to the public, except where noted.

Table 3. 48th SAW/SARC, List of Attendees

Name	Affiliation	email
Tom Hoff	MAFMC	thoff@mafmc.org
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Vic Crecco	CT Marine Fisheries	victor.crecco@ct.us
Nichola Meserve	ASMFC	nmeserve@asfmc.org

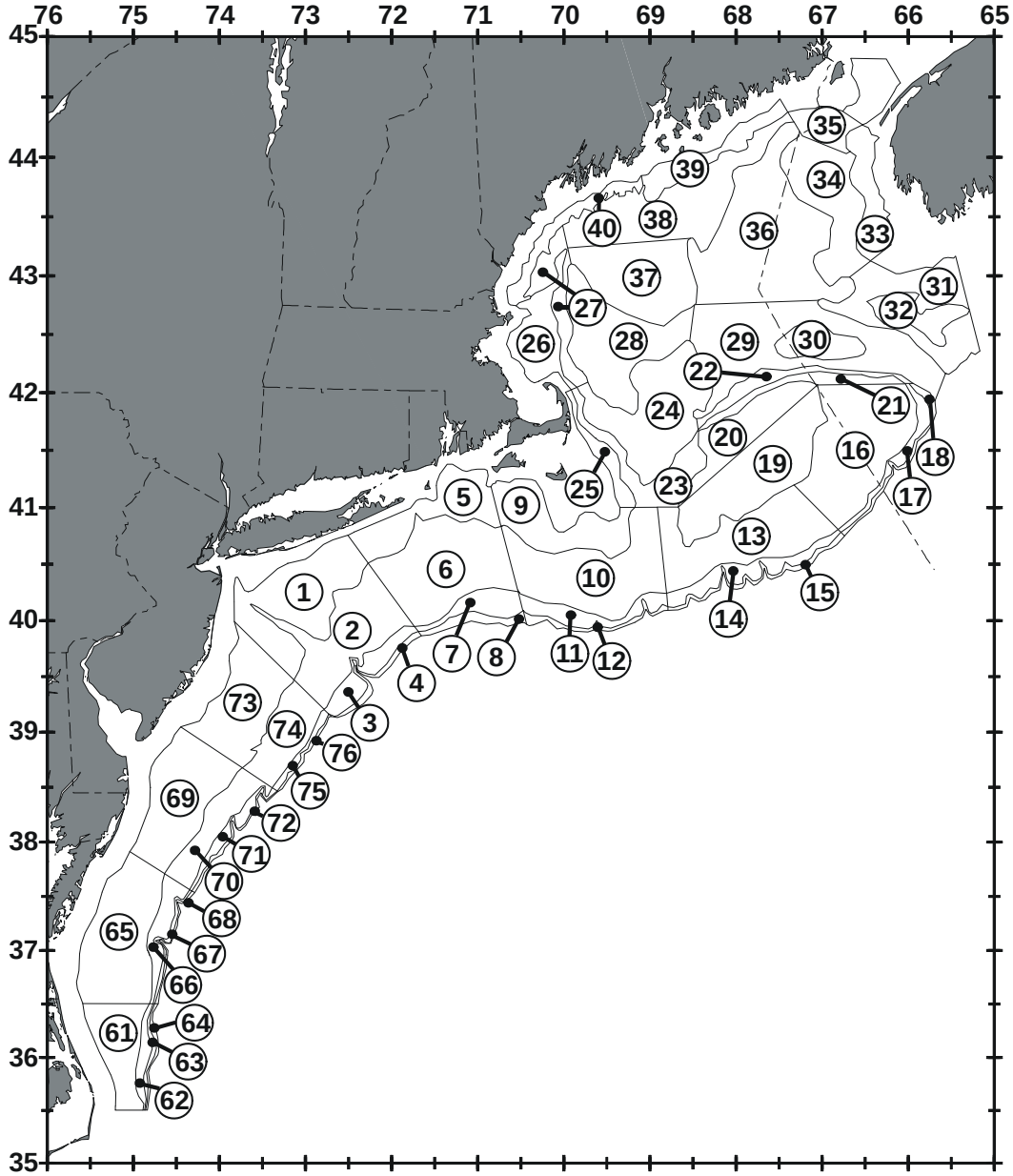


Figure 1. Offshore depth strata sampled during Northeast Fisheries Science Center bottom trawl research surveys.

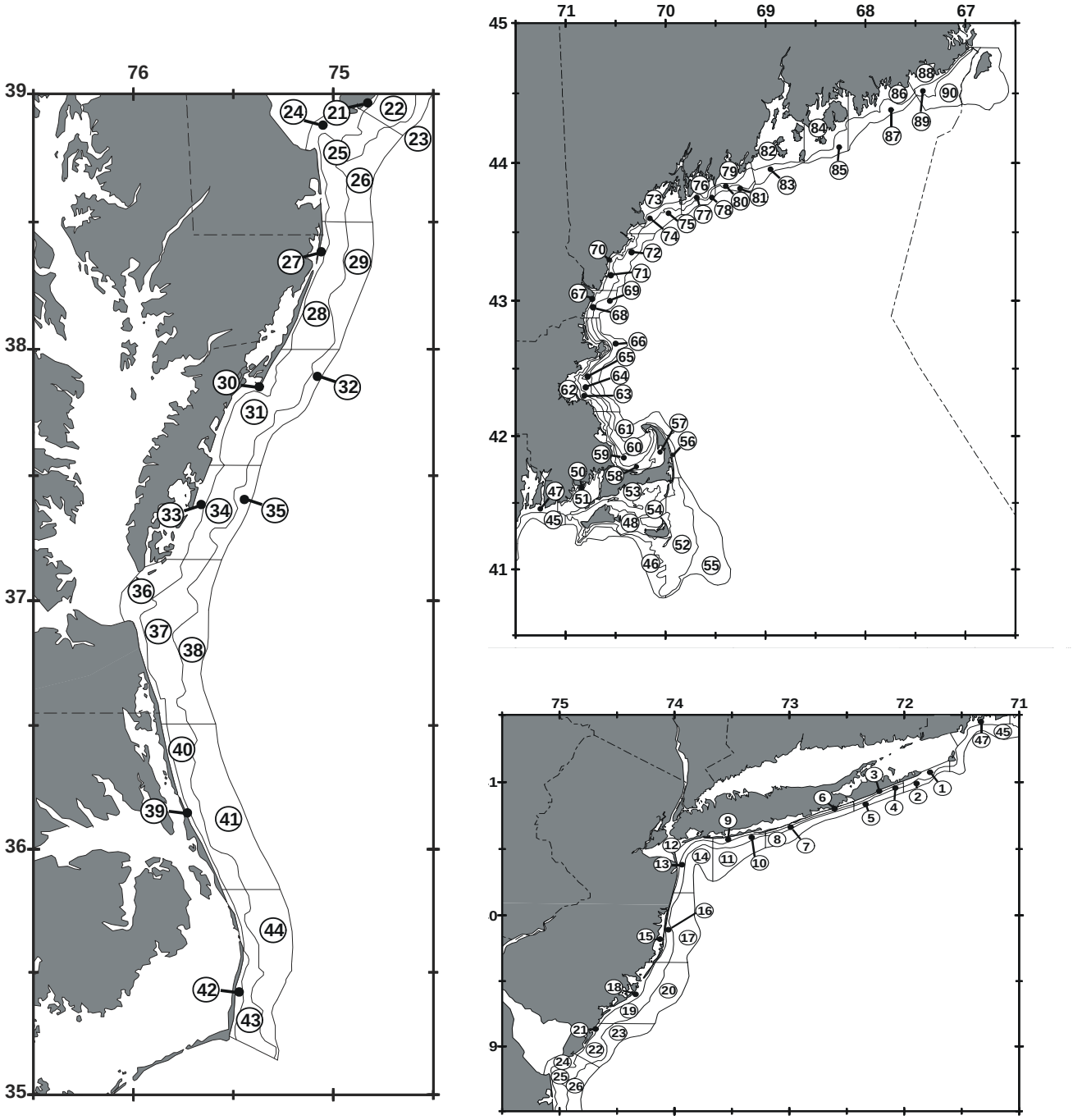


Figure 2. Inshore depth strata sampled during Northeast Fisheries Science Center bottom trawl research surveys.

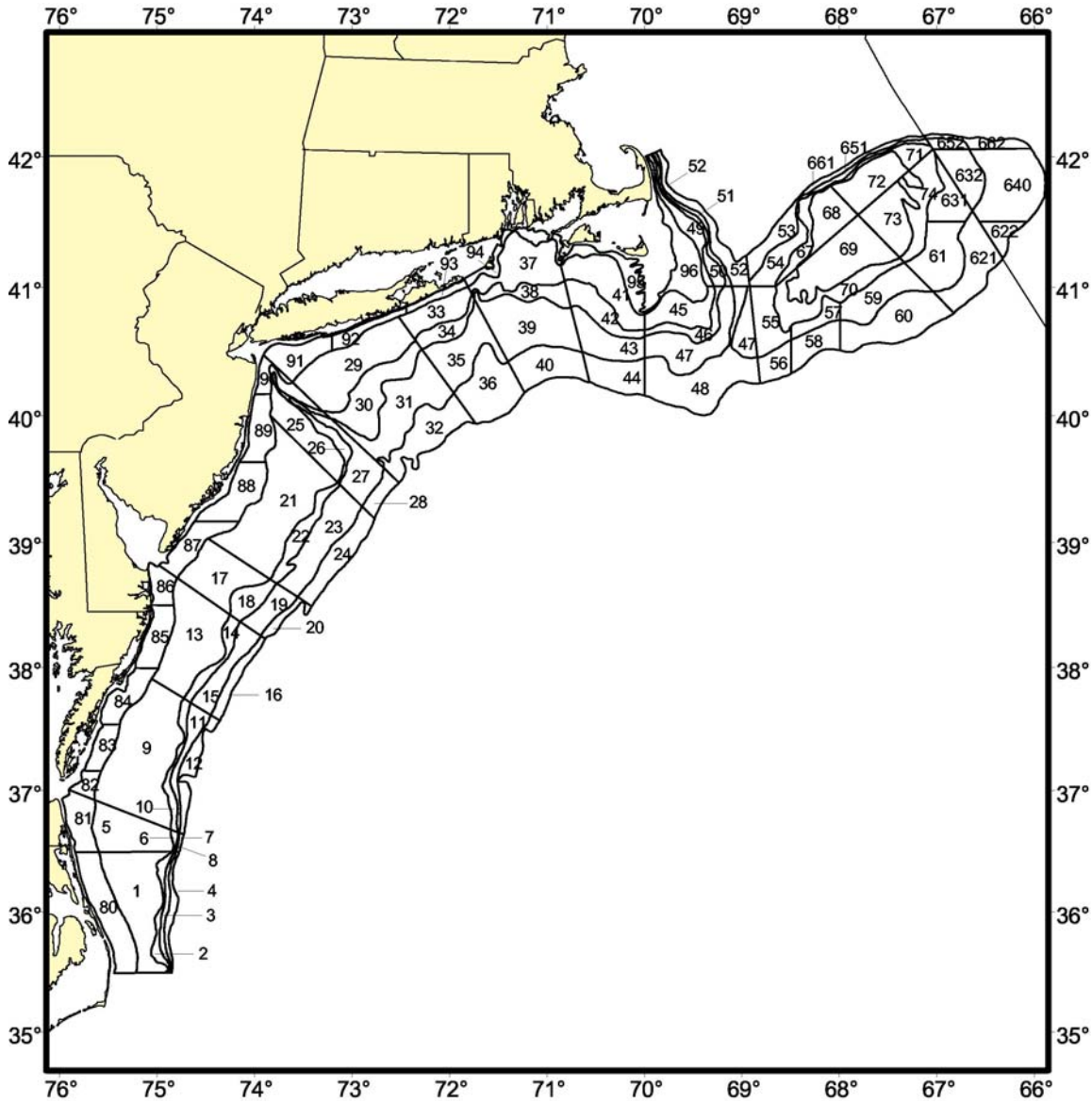


Figure 3. Depth strata sampled during Northeast Fisheries Science Center clam dredge research surveys.

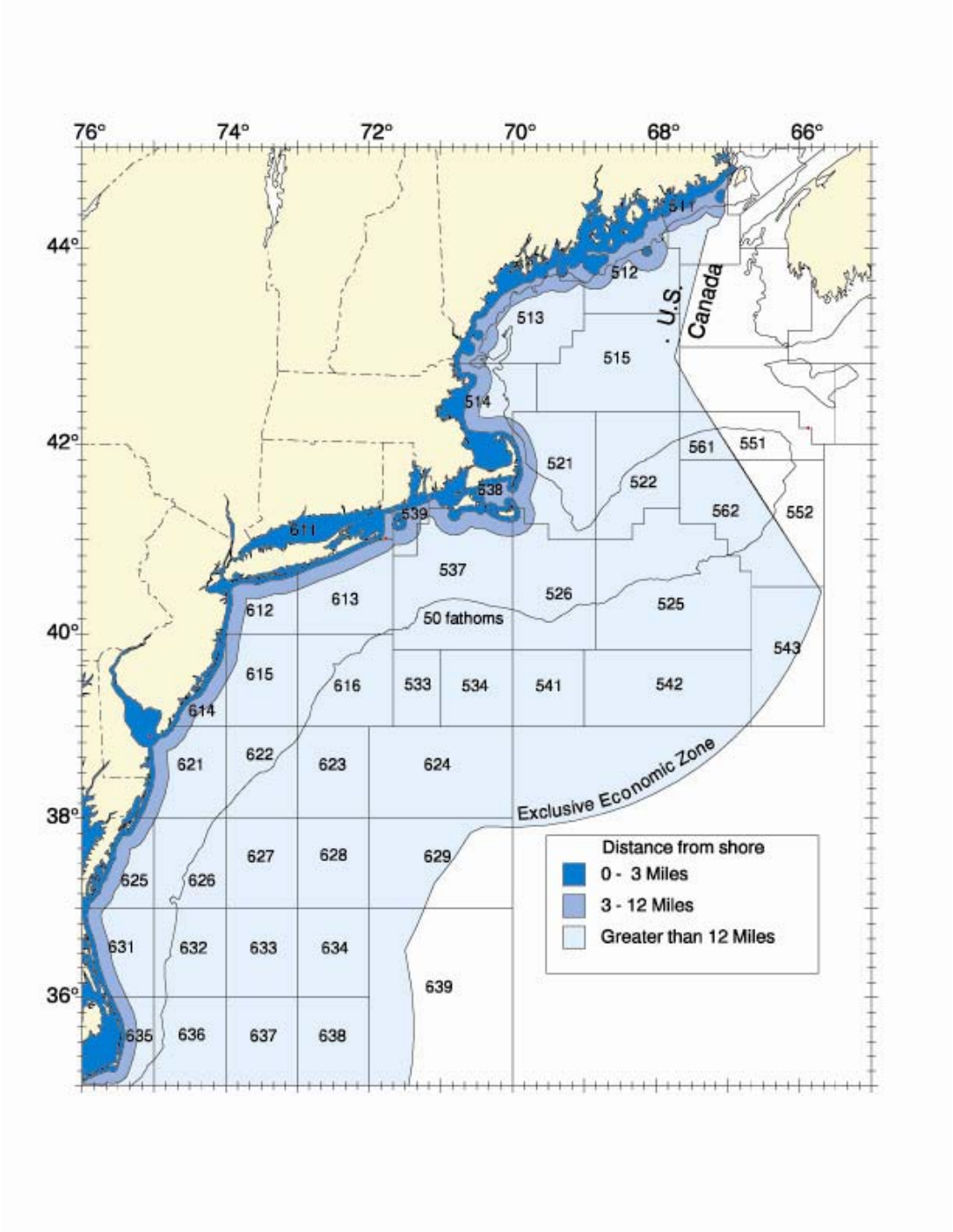


Figure 4. Statistical areas used for reporting commercial catches.

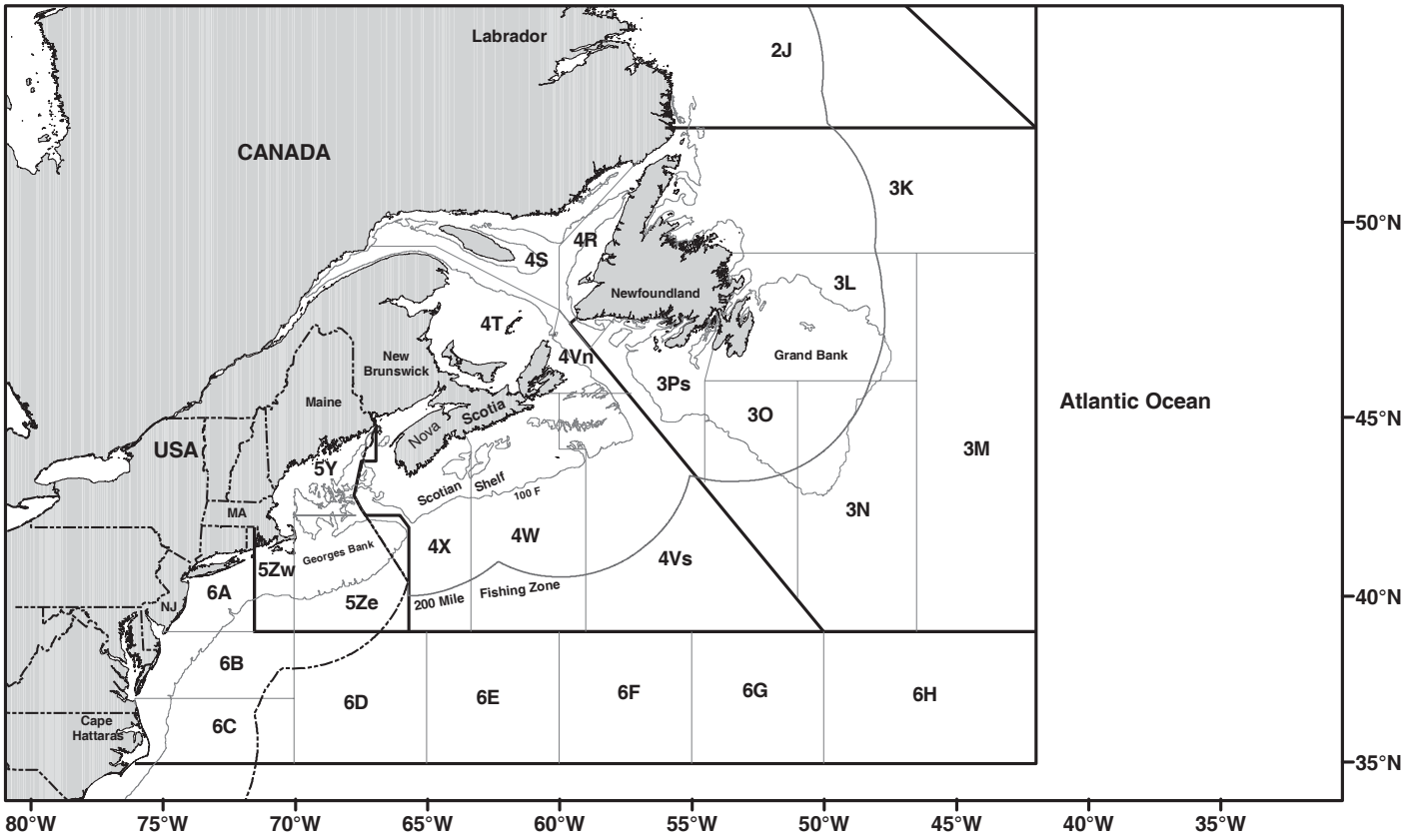
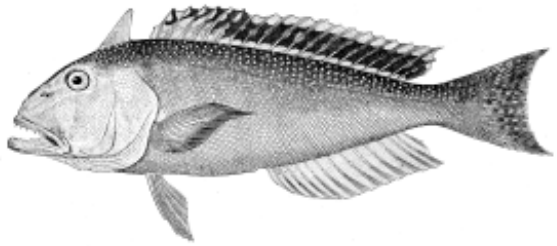


Figure 5. Catch reporting areas of the Northwest Atlantic Fisheries Organization (NAFO) for Subareas 3-6.

A. Assessment of Golden Tilefish, *Lopholatilus chamaeleonticeps,* in the Middle Atlantic-Southern New England Region



A Report of the
Southern Demersal Working Group
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, MA 02543

Executive Summary

The Southern Demersal Working Group met from 27-28 April, 2009 at the Northeast Fisheries Science Center, Woods Hole, Massachusetts to address the terms of reference agreed by the NRCC for tilefish. The following members were in attendance:

Dan Farnhan	F/V Kimberly
Chris Legault	NEFSC
Richard McBride	NEFSC
Jose´ Montañez	MAFMC
Josh Moser	NEFSC
Paul Nitschke	NEFSC (Assessment Lead)
John Nolan	F/V Seacapture
Laurie Nolan	F/V Seacapture
Michael Palmer	NEFSC
Barbara Rountree	NEFSC
Gary Shepherd	NEFSC
Martin Smith	Duke University (SSC lead, phone)
Katherine Sosebee	NEFSC
Mark Terceiro	NEFSC (Chair)
Tiffany Vidal	NEFSC
Susan Wigley	NEFSC

The current status for this stock is based on the ASPIC surplus production model which was the basis of the stock assessment for the last three assessments. The model is calibrated with CPUE series, as there are no fishery-independent sources of information on trends in population abundance. While the Working Group expressed concern about the lack of fit of the model to the VTR CPUE index at the end of the time series, we agreed to accept the estimates of current fishing mortality and biomass and associated reference points. The instability of model results in the scenario projections was also a source of concern. It was noted that the bootstrap uncertainty estimates do not capture the true uncertainty in the assessment. The ASPIC model indicates that the stock is rebuilt. However, the working group acknowledges that there is high uncertainty on whether the stock is truly rebuilt.

Terms of Reference

1. Characterize the commercial catch including landings, effort and discards. Characterize recreational landings. Evaluate utility of study fleet results as improved measures of CPUE.

Total commercial landings (live weight) increased from less than 125 metric tons (mt) during 1967-1972 to more than 3,900 mt in 1979 and 1980. Annual landings have ranged between 666 and 1,838 mt from 1988 to 1998. Landings from 1999 to 2002 were below 900 mt (ranging from 506 to 874 mt). An annual quota of 905 mt was implemented in November of 2001. Landings in 2003 and 2004 were slightly above the quota at 1,130 mt and 1,215 mt respectively. Landing from 2005 to 2008 have been at or below the quota. Landings in 2007 and 2008 were 751 mt and 736 mt respectively. During the late 1970s and early 1980s Barnegat, NJ was the principal tilefish port;

more recently Montauk, NY has accounted for most of the landings. Most of the commercial landings are taken by the directed longline fishery. Discards in the trawl and longine fishery are a minor component of the catch. Recreational catches have also been low for the last 25 years (i.e., less than 1 mt caught annually).

A fishery independent index of abundance does not exist for tilefish. Three different series of longline effort data were analyzed. The first series was developed by Turner (1986) who used a general linear modeling approach to standardize tilefish effort during 1973-1982 measured in kg per tub (0.9 km of groundline with a hook every 3.7 m) of longline fished obtained from logbooks of tilefish fishermen. Two additional CPUE series were calculated from the NEFSC weighout (1979-1993) and the VTR (1995-2008) systems. The number of vessels targeting tilefish has declined over the time series; during 1994-2003, five vessels accounted for more than 70 percent of the total tilefish landings. The length of a targeted tilefish trip had been generally increasing until the mid 1990s. At the time of the last assessment (2005) trip lengths have shorten to about 5 days. Since then trip length has been increasing.

Six market categories exist in the database. From smallest to largest they are: small, kitten, medium, large and extra large as well as an unclassified category. The proportion of landings in the kittens and small market categories increased in 1995 and 1996. Evidence of two strong recruitment events can be seen tracking through these market categories. At the time of the last tilefish assessment (2005) the proportion of large market category has declined since the early 1980s. However more recently most of the landings come from the large market category as the last strong year class (1999) has grown. Commercial length sampling has been inadequate over most of the time series. However some commercial length sampling occurred in the mid to late 1990s. More recently there has been a substantial increase in the commercial length sampling from 2003 to 2008.

Study fleet analysis is addressed in Appendix A1.

2. Estimate fishing mortality and total stock biomass for the current year, and for previous years if possible, and characterize the uncertainty of those estimates. Incorporate results of new age and growth studies.

As in SARC 41 the 2009 Working Group accepted the formulation that began the analysis in 1973, separated the Turner, weighout and VTR CPUE into three series and fixed the $B1/B_{MSY}$ ratio at 1 as the final run (base run). The working group expressed some concern over whether the CPUE in this fishery is more a reflection of changes in fishing practices and changes in spatial distribution of the fish rather than fluctuations in population size. Commercial length data indicate that increases in total biomass are predominantly due to a strong 1999 year class. It appears that most of the commercial catch over the 2002-2007 period were derived from this year class. Process error in the ASPIC model associated with the recent large year class has increased at the end of the time series due to an assumed constant recruitment/growth parameter.

The Working Group examined results obtained from an alternative forward projecting age/size structured model (SCALE) due to the difficulties with ASPIC model fitting the CPUE index at the end of the time series. An earlier version of this model was call catch-length model in SARC 41. The SCALE model incorporates population growth and length information into the model framework. This allows for the estimation of strong recruitment events which can be seen in the commercial length frequency distributions over time. However the overall lack of data and issues with independence of the data sources is a source of concern with the SCALE model results. The

lack of a recruitment index, inability to estimate uncertainty using mcmc, and questions with the estimated flat top selectivity curve are also sources of uncertainty. However SCALE model results suggests that the surplus production model may have overestimated the productivity of the stock.

New age and growth study is addressed in Appendix A2.

3. Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and F_{MSY}). Comment on the scientific adequacy of existing and redefined BRPs.

Biological reference points estimated by the 2009 ASPIC BASE run are moderately different from the 2005 SAW 41 assessment. B_{MSY} is estimated to be 11,400 mt (a 22% increase), F_{MSY} is estimated to be 0.16 (a 24% decrease), and MSY is estimated to be 1,868 mt (a 6% decrease), compared to B_{MSY} = 9,384 mt, F_{MSY} = 0.21, and MSY = 1,988 mt from the 2005 SAW 41 assessment.

SCALE yield per recruit biological reference points suggest that SSB_{MSY} is between 9,878 mt and 15,108 mt for the combine sex run using F_{40} or F_{MAX} as the F_{MSY} proxy. The separate sex run suggests female SSB_{MSY} is between 5,335 mt and 7,100 mt. For both the single sex and separate sex run the F_{MSY} is between 0.079 and 0.128 and MSY ranging from 1,072 mt to 1,200 mt using either F_{40} or F_{MAX} as the F_{MSY} proxy.

4. Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 3).

The biomass-based surplus production model (ASPIC) indicates that the tilefish stock biomass in 2008 has improved since the last assessment in 2005. Total biomass in 2008 is estimated to be 104% of B_{MSY} and fishing mortality in 2008 is estimated to be 38% of F_{MSY}. The tilefish stock is not overfished and overfishing is not occurring. The SARC 48 review panel accepted the ASPIC model but concluded that the ASPIC model is likely over optimistic and that the stock has not rebuilt above B_{MSY}.

SCALE model result suggests a different status determination. The 2009 BASE SCALE model run (separate sex run) and the combined sex run results indicate that the 2009 Golden tilefish stock is at a low biomass (29% to 47% of SSB_{MSY}) and is overfished with respect to the update SSB reference points. Both SCALE runs also suggest recruitment and growth overfishing (147% to 260% of F_{MSY}) is occurring with respect to the F_{40} or F_{MAX} updated biological reference points. However fishing mortality has been decreasing and biomass has been increasing since the beginning of the FMP in 2001.

5. Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch).

a. Provide numerical short-term projections (2-3 years). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out

projections, consider a range of assumptions about the most important uncertainties in the assessment (alternate states of nature).

b. If possible, comment on the relative probability of the alternate states of nature and on which projections seem most realistic.

c. For a range of candidate ABCs, compute the probabilities of rebuilding the stock by November 1, 2011.

d. Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.

The Working Group examined several ASPIC projections, including the current TAC of 905 mt. The ASPIC model indicates the stock is rebuilt and F in 2008 is low. Therefore the projections suggest the stock will continue to build if catches remain below MSY (1,854 mt). Projection scenarios that incorporated a possible future CPUE index illustrate the concern with the model stability due to the year class effects in the CPUE index. The scenario projections suggest that uncertainty with the stock status determination is much higher than what is suggested from the bootstrap uncertainty distributions and the standard projections.

Several options (age-based AGRPRO, deterministic SCALE projection) are available for 63SCALE model projections depending on whether growth is model as a single sex or with the sexes separated. Continued stock rebuilding is projected in the SCALE model with status quo conditions. Uncertainty estimates were not possible likely due to the overall lack of data in the model. Results of the SCALE model should be considered as a possible alternative state of nature for judging the extent of the overall uncertainty in the assessment when setting an ABC.

6. Review, evaluate and report on the status of the research recommendations offered in recent SARC reviewed assessments. Identify new research recommendations, including recruitment estimation.

Most of the research recommendations were addressed through the new study fleet project and updated growth study. Several new research recommendations were also suggested at the working group meeting, including continuation of the tilefish study fleet program or possibly modifying the study fleet program into an industry based survey that could obtain a recruitment index as part of the sampling design. Research recommendations TOR 6 are summarized on pages 32-33.

Introduction

Golden tilefish, *Lopholatilus chamaeleonticeps*, inhabit the outer continental shelf from Nova Scotia to South America, and are relatively abundant in the Southern New England to Mid-Atlantic region at depths of 80 to 440 m. Tilefish have a narrow temperature preference of 9 to 14 C. Their temperature preference limits their range to a narrow band along the upper slope of the continental shelf where temperatures vary by only a few degrees over the year. They are generally found in and around submarine canyons where they occupy burrows in the sedimentary substrate. Tilefish are relatively slow growing and long-lived, with a maximum observed age of 46 years and a

maximum length of 110 cm for females and 39 years and 112 cm for males (Turner 1986). At lengths exceeding 70 cm, the predorsal adipose flap, characteristic of this species, is larger in males and can be used to distinguish the sexes. Tilefish of both sexes are mature at ages between 5 and 7 years (Grimes et. al. 1988).

Golden Tilefish was first assessed at SARC 16 in 1992 (NEFSC 1993). The Stock Assessment Review Committee (SARC) accepted a non-equilibrium surplus production model (ASPIC). The ASPIC model estimated biomass-based fishing mortality (F) in 1992 to be 3-times higher than F_{MSY}, and the 1992 total stock biomass to be about 40% of B_{MSY}. The intrinsic rate of increase (r) was estimated at 0.22.

The Science and Statistical (S&S) Committee reviewed an updated tilefish assessment in 1999. Total biomass in 1998 was estimated to be 2,936 mt, which was 35% of B_{MSY} = 8,448 mt. Fishing mortality was estimated to be 0.45 in 1998, which was about 2-times higher than F_{MSY} = 0.22. The intrinsic rate of increase (r) was estimated to be 0.45. These results were used in the development of the Tilefish Fishery Management Plan (Mid-Atlantic Fishery Management Council 2000). The Mid-Atlantic Fishery Management Council implemented the Tilefish Fishery Management Plan (FMP) in November of 2001. Rebuilding of the tilefish stock to B_{MSY} was based on a ten-year constant harvest quota of 905 mt.

SARC 41 reviewed a benchmark tilefish assessment in 2005. The surplus production model indicated that the tilefish stock biomass in 2005 has improved since the assessment in 1999. Total biomass in 2005 is estimated to be 72% of B_{MSY} and fishing mortality in 2004 is estimated to be 87% of F_{MSY}. Biological reference points did not change greatly from the 1999 assessment. B_{MSY} is estimated to be 9,384 mt and F_{MSY} is estimated to be 0.21. The SARC concluded that the projections are too uncertain to form the basis for evaluating likely biomass recovery schedules relative to B_{MSY}. The TAC and reference points were not changed based on the SARC 41 assessment.

Term of Reference 1: Commercial Fishery

TOR 1: Characterize the commercial catch including landings, effort and discards. Characterize recreational landings. Evaluate utility of study fleet results as improved measures of CPUE.

See Appendix A1 for details on the utility of study fleet results as an improved measures of CPUE.

Data Sources

Commercial catch data

Total commercial landings (live weight) increased from less than 125 mt during 1967-1972 to more than 3,900 mt in 1979 and 1980 (Table A1, Figure A1). Landings stabilized at about 2,000 mt during 1982-1986. An increase in landings occurred in 1987 to 3,200 mt but subsequently declined to 450 mt in 1989. Annual landings have ranged between 454 and 1,838 mt from 1988 to 1998. Landings from 1999 to 2002 were below 900 mt (ranging from 506 to 874 mt). An annual quota of 905 mt was implemented in November of 2001. Landings in 2003 and 2004 were above the quota at 1,130 mt and 1,215 mt respectively. Landing from 2005 to 2008 have been at or below the quota. Landings in 2007 and 2008 were 751 mt ant 736 mt respectively. Over 75% of the landings came from Statistical Areas 537 and 616 since 1991 (Table A2). Since the 1980s, over 85% of the commercial landings of tilefish in the MA-SNE region have been taken in the longline fishery (Table

A3, Figure A2). During the late 1970s and early 1980s Barnegat, NJ was the principal tilefish port; more recently Montauk, NY has accounted for most of the landings. The shift in landings can be seen in the proportion of the landings by state in Table A4 and Figure A3. In the late 1970s and earlier 1980s a greater proportion of the landings were taken in quarters 1 and 2 (Table A5, Figure A4). Recent landings have been relatively constant over the year.

Commercial discard data

Very little discarding (< 1%) of tilefish was reported in the vessel trip report (VTR) from longline vessels that target tilefish and there is little reported discarding of tilefish in the trawl fishery in the VTR data (SARC 41). Recent observer directed tilefish longline trips also suggest that discards of tilefish is minimal. Observer trawl data produce more variable discard estimates across years for tilefish. Discard to kept ratios for trawl trips that either kept or discarded tilefish in the observer data varied from 0 in 1993 to 1.4 in 2001 (Table A6). Twelve of the sixteen years had less than 15 trips sampled that caught tilefish from 1989 to 2003. The number of observer trips that caught tilefish has increase from 2004 to 2008 (average 47). Trawl discards were not expanded to derive total discards due to the relativity minor component of the trawl landings to the total and due to the high uncertainty associated with the hindcast estimates.

Commercial CPUE data

Analyses of catch (landings) and effort data were confined to the longline fishery since directed tilefish effort occurs in this fishery (e.g. the remainder of tilefish landings are taken as bycatch in the trawl fishery). Most longline trips that catch tilefish fall into two categories: (a) trips in which tilefish comprise greater than 90% of the trip catch by weight and (b) trips in which tilefish accounted for less than 10% of the catch. Effort was considered directed for tilefish when at least 75% of the catch from a trip consisted of tilefish (NEFSC 1993).

Three different series of longline effort data were analyzed. The first series was developed by Turner (1986) who used a general linear modeling approach to standardize tilefish effort during 1973-1982 measured in kg per tub (0.9 km of groundline with a hook every 3.7 m) of longline obtained from logbooks of tilefish fishermen. Two additional CPUE series were calculated from the NEFSC weighout (1979-1993) and the VTR (1995-2008) systems as well as a combined 1979-2008 series. Effort from the weighout data was derived by port agents' interviews with vessel captains whereas effort from the VTR systems comes directly from mandatory logbook data. In this assessment and in the 1998 and 2005 tilefish assessments we used Days absent as the best available effort metric. In the 1998 assessment an effort metric based on Days fished (average hours fished per set / 24 * number of sets in trip) was not used because effort data were missing in many of the logbooks and the effort data were collected on a trip basis as opposed to a haul by haul basis. For this assessment effort was calculated as:

Effort = days absent (time & date landed - time & date sailed) - number of trips.

For some trips, the reported days absent were calculated to be a single day. This was considered unlikely, as a directed tilefish trip requires time for a vessel to steam to near the edge of the continental shelf, time for fishing, and return trip time (Grimes et al. 1980). Thus, to produce a realistic effort metric based on days absent, a one day steam time for each trip (or the number of trips) was subtracted from days absents and therefore only trips with days absent greater than one day were used.

The NEFSC Weighout and VTR CPUE series were standardized using a general linear model (GLM) incorporating year and individual vessel effects (Mayo et al. 1994). The CPUE was standardized to an individual longline vessel and the year 1984; the same year used in the last assessment. For the VTR series the year 2000 was used as the standard. Model coefficients were back-transformed to a linear scale after correcting for transformation bias (Granger and Newbold 1977). The full GLM output for the Weighout and the VTR CPUE series is included as Appendix A3.

The number of vessels targeting tilefish has declined over the time series (Table A7, Figure A5); during 1994-2003, five vessels accounted for more than 70 percent of the total tilefish landings (Table A8, Figure A6). The number of vessels targeting tilefish has remained fairly constant since the last assessment in 2005. The length of a targeted tilefish trip had been generally increasing until the mid 1990s. At the time of the last assessment (2005) trip lengths have shorten to about 5 days. Since then trip length has been increasing (Figure A5). In the weighout data the small number of interview is a source of concern; very little interview data exists at the beginning of the time series (Table A7, Figure A7). The 5 dominant tilefish vessels make up almost all of the VTR data with the exception of 2004 when there appears to be more vessels targeting tilefish (Figure A6). In some years there were higher total landings reported in the VTR data than the Dealer data for the 5 dominant tilefish vessels. After the FMP was implemented the IVR (Interactive Voice recorder) database was developed to monitor the quota. In 2005 the IVR database had the highest landings level despite that this system only applies to the limited access tilefish fishery. The IVR 2005 total was assumed to be a better estimate of the total landings in that year then the other data sources. The IVR total landing in 2005 was used as the total removals in all tilefish modeling.

The number of targeted tilefish trips declined in the early 1980s while trip length increased at the time the FMP was being developed in 2000 (Figures A5 and A8). During the last assessment in 2005 the number of trips became relatively stable as trip length decreased. Since the last assessment trip length has increased. The interaction between the number of vessels, the length of a trip and the number of trips can be seen in the total days absent trend in Figure A8. Total days absent remained relatively stable in the early 1980s, but then declined at the end of the weighout series (1979-1994). In the beginning of the VTR series (1994-2004) days absent increased through 1998 but declined to 2005. Since 2005 total days absent has increase somewhat. Figure A8 also shows that a smaller fraction of the total landings were included in the calculation of CPUE compared to the VTR series.

Figure A9 illustrates difference between the nominal CPUE and vessel standardized (GLM) CPUE with the weighout and VTR data combined. CPUE trends are very similar for most vessels that targeted tilefish (Figure A10). A sensitivity test of the GLM using different vessel combinations was done in SARC 41. The SARC 41 GLM was found not to be sensitivity to different vessels entering the CPUE series.

Very little CPUE data exist for New York vessels in the 1979-1994 weighout series despite the shift in landing from New Jersey to New York before the start of the VTR series in 1994. The small amount of overlap between the weighout and VTR series is illustrated in Figures A11 and A12. Splitting the weighout and VTR CPUE series can be justified by the differences in the way effort was measured and difference in the tilefish fleet between the series. In breaking up the series we omitted 1994 because there were very little CPUE data. The sparse 1994 data that existed came mostly from the weighout system in the first quarter of the year. Very similar trends exist in the four years of overlap between Turner (1986) CPUE and the weighout series (Figure A13).

Since 1979, the tilefish industry has changed from using cotton twine to steel cables for the backbone and from J hooks to circle hooks. The gear change to steel cable and snaps started on New

York vessels in 1983. In light of possible changes in catchability associated with these changes in fishing gear, the working group considered that it would be best to use the three available indices separately rather than combined into one or two series. The earliest series (Turner 1986) covered 1973-1982 when gear construction and configuration was thought to be relatively consistent. The Weightout series (1979-1993) overlapped the earlier series for four years and showed similar patterns (Figure A13) and is based primarily on catch rates from New Jersey vessels. The VTR (1995-2004) series is based primarily on information from New York vessels using steel cable and snaps.

In SARC 41 a month vessel interaction was significant but explained only a small amount of the total sum of squares (6%). Adding a month - vessel interaction term to the GLM model had very little influence on the results at SARC 41 and was not updated for this assessment. The GLM output for the Weighout and VTR CPUE series standardized for individual vessel effects can be seen in Appendix A3.

In this assessment the sensitivity of the assumed error structure used in VTR GLM CPUE index was explored. The nominal VTR CPUE data distribution does appear over-dispersed relative to normal or lognormal distribution, suggesting that a model with poisson or negative binomial distribution may be more appropriate (Figure A14). However the GLM CPUE indices using different error assumptions showed very little differences in the CPUE trends (Figure A15). Therefore the lognormal error distribution was retained.

Commercial market category and size composition data

Six market categories exist in the database. From smallest to largest they are: small, kitten, medium, large and extra large as well as an unclassified category. In 1996 and 1997, the reporting of tilefish by market categories increased, with the proportion of unclassified catch declining to less than 20% (Table A9, Figure A16). The proportion of landings in the small and kitten market categories increased in 1995 and 1996. Small and kitten market categories had similar length distributions and samples from 1995 to 1999 were combined. Evidence of several strong recruitment events can be seen tracking through the market category proportions (Figures A16 and A17). At SARC 41 the proportion of the large market category has declined since the early 1980s (Figure A16). Landings data obtained directly from the New York tilefish industry shows a similar decline in the proportion of the large market category between 1980 and 1990 (Figure A18). Landings by market category has shifted from smalls and kittens in 2004 to larges in 2007 and 2008 which is likely the result of a strong year class effect (Figure A17).

Extensive size sampling was conducted in 1976-1982 (Grimes *et al.* 1980, Turner 1986) however that data are not available by market category (Figure A19). Since then commercial length sampling has been inadequate in most years (Table A4). However some commercial length sampling occurred in the mid to late 1990s. More recently there has been a substantial increase in the commercial length sampling in 2003 and 2004. Commercial length sampling in New York has also increased since the last assessment in 2005 (Table A4). Expanded length frequency distributions from 1995 to 1999 from SARC 41 are shown in Figure A20. In this assessment expanded length frequency distributions were estimated form 2002 to 2008 (Figure A21 and A22). The stratification used in the expansion can be seen in table A10. The large market category length frequencies appear to have been relatively stable for years when more than 100 fish were measured. However the small market category exhibits shifts in the size distribution in certain years as strong year classes move through the fishery (Figure A23). The tracking of a year class can be seen as the cohort grows over the year in 2003 and 2004 (Figure A23). The strong 1998/1999 year class seen in

the kept length frequency distributions from tilefish longline observer trips matches well with the expanded commercial length frequency distributions (Figures A24). In addition, the 2008 study fleet length distribution looks similar to the 2008 commercial landings distribution (Figure A25).

Smaller fish sizes are seen in the trawl gear length distributions for the small and kitten market category compared to longline gear (Figure A26). Therefore trawl length frequency distribution where not used to characterize the catch (Table A10). Longline tilefish fishermen often receive forecasts from the draggers of when a strong year class will be entering the fishery. There is some anecdotal information from draggers for the existence of a stronger year class in 2009.

Commercial length frequencies were expanded for years where sufficient length data exist (1995-1999 and 2002-2008) (Table AC10). The large length frequency samples from 1996 to 1998 were used to calculate the 1995 to 1999 expanded numbers at length while the large length samples from 2001 and 2003 were used to calculate the 2002 expanded numbers at length. Evidence of strong 1992/1993 and 1998/1999 year classes can be seen in the expanded numbers at length in the years when length data existed (1995-1999 and 2002-2008) (Figure A20). The matching of modes in the length frequency with ages was done using Turner's (1986) and Vidal's (2009) aging studies. In 2004 and 2005 the 1998/1999 year class can be seen growing into the medium market category and in 2006 and 2007 the year class has entered the large market category (Figure A20). From 2002 to 2007 it appears that most of the landings were comprised of this year class. The catch appears to be comprised of multiple year classes in 2008 after catch rates have declined in the VTR series. An increase in the landings and CPUE can be seen when the 1992/1993 and 1998/1999 year classes recruit to the longline fishery. As the year classes gets older the catch rates decline (Figure A13 and A21).

Recreational data

A small recreational fishery occurred briefly in the mid 1970s (< 100 mt annually, Turner 1986) but subsequent recreational catches have been quite low for the last 30 years (i.e., less than 1 mt caught annually) (Table A11). Party and charter boat vessel trip reports also show low numbers of tilefish being caught since 1994 (Table A12).

NEFSC Trawl survey data

Only a few fish per survey are caught during NEFSC bottom trawl surveys. This survey time series is not useful as an index of abundance for tilefish.

Term of Reference 2: Mortality and stock size estimates

TOR2: Estimate fishing mortality and total stock biomass for the current year and for previous years if possible, and characterize the uncertainty of those estimates. Incorporate results of new age and growth studies.

See Appendix A2 for details on the new age and growth study.

ASPIC Surplus production model

The ASPIC surplus production model (Prager 1994; 1995) was used to determine fishing mortality, stock biomass and biological reference points (F_{MSY}, and B_{MSY}) for the development of the tilefish FMP in 2001. SARC 41 in 2005 accepted the ASPIC model as a basis for determining whether the stock was on schedule for rebuilding by 2011.

As a first step in the surplus production modeling, the landings and index data from the 2005

SAW41 assessment were used as input in the latest version (5.33) of the ASPIC software and compared with the results from the 2005 SAW 41 assessment, which was run in ASPIC version 3.93. There were no significant differences in the results due to the ASPIC version update (Table A13). The three commercial fishery CPUE index series (Turner 1973-1982; NEFSC Weighout 1982-1993; and VTR 1995-2004) as configured in the 2005 SAW 41 assessment were retained in constructing the 2009 ASPIC model configurations. The VTR CPUE index of abundance and commercial fishery landings were updated through 2008 to create the 2009 BASE run. A bootstrap with 1000 iterations was used to estimate confidence intervals for annual F and stock biomass estimates and biological reference points. Several sensitivity runs were made to further evaluate the impact on results of the assumption for the B1/K ratio starting condition (equivalent to the $B1/B_{MSY}$ ratio in the 2005 SAW 41 assessment ASPIC v3.93). A retrospective analysis of the BASE run was made to evaluate model performance.

The trends in fishing mortality (F; in the ASPIC model, this is the ratio of annual catch to average annual stock biomass) were very similar in the 2005 SAW 41 and in the 2009 BASE results through 2004. The 2005 SAW 41 F estimates generally followed the 75%ile of the 2009 BASE estimates of F (i.e., were generally somewhat higher), while the 2005 SAW 41 biomass estimates followed the 25%ile of the 2009 BASE estimates of biomass (i.e., were generally somewhat lower; Figures A27 and A28). The early period (Turner 1973-1982) indices fit better (higher r2 value) in the 2009 BASE run than in the 2005 SAW 41 assessment; conversely, the two later series (NEFSC Weighout 1982-1993 and VTR 1995-2008) fit worse (lower r2 values) (Figure A29). Catchability coefficients (q) decreased for all three index series (Turner by 34%; NEFSC Weighout by 22%; VTR by 34%). The biomass reference points (B_{MSY} and K) increased by 22% from the 2005 SAW 41 run to the 2009 BASE run, while FMSY decreased by 22% and MSY decreased by 6%. The 2009 BASE run estimates provide a more optimistic evaluation of stock status in 2004 than did the 2005 SAW 41 model estimates (e.g., the B_{2004}/B_{MSY} ratio; Table A13).

As in the last assessment, sensitivity runs were made to explore the effect of the value of the B1/K ratio on results (B1 is the stock biomass in the first year of the analysis time series; K is the carrying capacity of the stock, equivalent to the biomass when fishing mortality is zero over the long-term). In the 2009 BASE run configuration the B1/K ratio was fixed at 0.50 (equivalent to the $B1/B_{MSY}$ ratio = 1.00 in the 2005 SAW 41 ASPIC v3.93). The BASE results were compared with runs fixing B1/K at 0.10, 1.00, and a run in which B1/K was estimated at 1.19. The run with B1/K fixed at 0.10 provides a value for the Root Mean Squared Error (RMSE) value over 50% higher than the BASE run and negative r2 values for all 3 CPUE index series. The estimates of K (carrying capacity), MSY (Maximum Sustainable Yield), and FMSY (fishing mortality rate providing MSY) for this run are infeasible given the historical pattern and magnitude of fishery landings and the life history characteristics of tilefish (Table A13, dashed lines in Figures A30 and A31).

The runs fixing B1/K = 1.00 and estimating B1/K = 1.19 provided results and diagnostics comparable to the BASE run with B1/K = 0.50. Estimates of F and biomass for 1979 and later years are nearly identical to the BASE run. The major differences are for 1973-1978, when the B1/K = 1.00 and B1/K = 1.19 runs obviously indicate that the stock declined from a high biomass level near K. Estimates of MSY and K for these sensitivity runs are about 10% (B_{MSY}) and 16% (K) lower than the BASE run, while estimates of F_{MSY} are 10-15% higher (Table A13, Figures A30 and A31). The runs fixing/estimating B1/K ratio near 1.00 in 1973 imply that the stock was near carrying capacity in the early 1970s, which is unlikely given the historical pattern and magnitude of fishery landings. The 2005 SAW 41 review concluded that the most likely assumption for the B1/K ratio was 0.50 (equivalent to $B1/B_{MSY}$ = 1.00). That assumption is again supported by the current

sensitivity analysis results, and so has been retained for the 2009 BASE run configuration.

A retrospective analysis (sequential removal of the last year of data) was conducted for the 2009 BASE run configuration with ten "peels" (ten years sequentially removed from the end of the analysis). The BASE run results are fairly stable for the 1999, 2002-2008 terminal years, both in terms of time series trends (Figures A32 and A33) and in the estimated catchability coefficients and reference points (left side of Table A14). For the 1998, 2000-2001 terminal years, however, the 2009 BASE run converged at a different solution but with a comparable value of the RMSE. For the 1998, 2000-2001 runs, the estimated catchability coefficients were about 25-50% of the 1999, 2002-2008 runs, and the estimated reference points were infeasible given the historic trend and magnitude of the fishery landings (right side of Table A14). These results indicate that the current 2009 BASE model solution is stable for the last several terminal years, but also indicates that future runs should continue to be examined in a similar manner (multiple retrospectives and sensitivity analyses) to evaluate performance.

The 2009 BASE run indicates that the tilefish stock biomass has continued to increase since the 2005 SAW 41 assessment (Figures A28 and A29). Fishing mortality (F = 0.06) is estimated to be 38% of F_{MSY} and stock biomass in 2008 (B = 11,910 mt) is estimated to be 4% above B_{MSY} (Table A13). Bootstrap (1000 iterations) estimates of the 2008 F were 0.05 (25%ile) to 0.07 (75%ile), with a median of 0.06 (50%ile; Figure A34). Bootstrap estimates of the 2008 stock biomass were 9,550 mt (25%ile) to 13,538 mt (75%ile), with a median of 11,767 mt (50%ile; Figure A35). The complete ASPIC model output with bootstrap results is included as Appendix A3.

Expanded landing length frequency distributions and trends in the VTR CPUE suggest recent strong year class effects in the fishery. The recent strong 1998/1999 year class results in increase process error with the fit to the VTR series in the ASPIC model since the surplus production model assumes constant growth/recruitment (Figure A30). The increase in error is reflected in the comparison of the r^2 from the SARC 41 ASPIC assessment (0.54) with the updated assessment (0.20).

SCALE Model

The working group investigated the use of an age and size structured forward projection model (SCALE) for assessing the tilefish stock due to the inability of the ASPIC surplus production model in fitting the observed year class effects. Incomplete or lack of age-specific catch and survey indices often limits the application of a full age-structured assessment (e.g. Virtual Population Analysis and many forward projecting age-structured models). Stock assessments will often rely on the simpler size/age aggregated models (e.g. surplus production models) when age-specific information is lacking. However the simpler size/age aggregated models may not utilize all of the available information for a stock assessment. Knowledge of a species growth and lifespan, along with total catch data, size composition of the removals, recruitment indices and indices on numbers and size composition of the large fish in a survey can provide insights on population status using a simple model framework.

The Statistical Catch At LEngth (SCALE) model, is a forward projecting age-structured model tuned with total catch (mt), catch at length or proportional catch at length, recruitment at a specified age (usually estimated from first length mode in the survey), survey indices of abundance of the larger/older fish (usually adult fish) and the survey length frequency distributions (NOAA Fisheries Toolbox 2008a). The SCALE model was developed in the AD model builder framework. The model parameter estimates are fishing mortality and recruitment in each year, fishing mortality to produce the initial population (Fstart), logistic selectivity parameters for each year or blocks of

years and Qs for each survey index.

The SCALE model was developed as an age-structured model that does NOT rely on age-specific information on a yearly basis. The model is designed to fit length information, abundance indices, and recruitment at age which can be estimated by using survey length slicing. However the model does require an accurate representation of the average overall growth of the population which is input to the model as mean lengths at age. Growth can be modeled as sex-specific growth and natural mortality or growth and natural mortality can be model with the sexes combined. The SCALE model will allow for missing data.

Model Configuration

The SCALE model assumes growth follows the mean input length at age with predetermined input error in length at age. Therefore a growth model or estimates of mean length at age are essential for reliable results. The model assumes static growth and therefore population mean length/weight at age are assumed constant over time. A depiction of model assumed population growth at age using the input mean lengths at age and variation can be seen in Table A15).

The SCALE model estimates logistic parameters for a flattop selectivity curve at length in each time block specified by the user for the calculation of population and catch age-length matrices or the user can input fixed logistic selectivity parameters. Presently the SCALE model can not account for the dome shaped selectivity pattern

The SCALE model computes an initial age-length population matrix in year one of the model as follows. First the estimated populations numbers at age starting with age-1 recruitment get normally distributed at one cm length intervals using the mean length at age with the assumed standard deviation. Next the initial population numbers at age are calculated from the previous age at length abundance using the survival equation. An estimated fishing mortality (Fstart) is also used to produce the initial population. This F can be thought of as the average fishing mortality that occurred before the first year in the model. Now the process repeats itself with the total of the estimated abundance at age getting redistributed according to the mean length at age and standard deviation in the next age (age+1).

This two step process is used to incorporate the effects of length specific selectivities and fishing mortality. The initial population length and age distribution is constructed by assuming population equilibrium with an initial value of F, called F_{start}. Length specific mortality is estimated as a two step process in which the population is first decremented for the length specific effects of mortality as follows:

$$N^*_{a,len,y_1} = N_{a-1,len,y_1} e^{-\left(PR_{len}F_{start}+M\right)}$$

In the second step, the total population of survivors is then redistributed over the lengths at age *a* by assuming that the proportions of numbers at length at age *a* follow a normal distribution with a mean length derived from the input growth curve (mean lengths at age).

$$N_{a,len,y_1} = \pi_{len,a} \sum_{len=0}^{L_\infty} N^*_{a,len,y_1}$$

where

$$\pi_{len,a} = \Phi\left(len+1 \mid \mu_a, \sigma_a^2\right) - \Phi\left(len \mid \mu_a, \sigma_a^2\right)$$

where

$$\mu_a = L_\infty\left(1 - e^{-K(a-t_0)}\right)$$

Mean lengths at age can be calculated from a von Bertalanffy model from a prior study as shown in the equation above or mean lengths at age can be calculated directly from an age-length key. Variation in length at age a = σ_s^2 can often be approximated empirically from the growth study used for the estimation of mean lengths at age. If large differences in growth exist between the sexes then growth can be input as sex-specific growth with sex-specific natural mortality. However catch and survey data are still fitted with sexes combined.

This SCALE model formulation does not explicitly track the dynamics of length groups across age because the consequences of differential survival at length at age a do not alter the mean length of fish at age a+1. However, it does more realistically account for the variations in age-specific partial recruitment patterns by incorporating the expected distribution of lengths at age.

In the next step the population numbers at age and length for years after the calculation of the initial population use the previous age and year for the estimate of abundance. Here the calculations are done on a cohort basis. Like in the previous initial population survival equation the partial recruitment is estimated on a length vector.

$$N^*_{a,len,y} = N_{a-1,len,y-1} e^{-\left(PR_{len} F_{y-1} + M\right)}$$

second stage

$$N_{a,len,y} = \pi_{len,a} \sum_{len=0}^{L_\infty} N^*_{a,len,y}$$

Constant M is assumed along with an estimated length-weight relationship to convert estimated catch in numbers to catch in weight. The standard Baranov's catch equation is used to remove the catch from the population in estimating fishing mortality.

$$C_{y,a,len} = \frac{N_{y,a,len}\, F_y\, PR_{len}\left(1 - e^{-\left(F_y PR_{len} + M\right)}\right)}{(F_y \;\; PR_{len}) + M}$$

Catch is converted to yield by assuming a time invariant average weight at length.

$$Y_{y,a,len} = C_{y,a,len} W_{len}$$

The SCALE model results in the calculation of population and catch age-length matrices for the starting population and then for each year thereafter. The model is programmed to estimate recruitment in year 1 and estimate variation in recruitment relative to recruitment in year 1 for each year thereafter. Estimated recruitment in year one can be thought of as the estimated average long term recruitment in the population since it produces the initial population. The residual sum of squares of the variation in recruitment $\sum(\text{Vrec})^2$ is than used as a component of the total objective function. The weight on the recruitment variation component of the objective function (Vrec) can be used to penalize the model for estimating large changes in recruitment relative to estimated recruitment in year one.

The model requires an age-1 recruitment index for tuning or the user can assume relatively constant recruitment over time by using a high weight on Vrec. Usually there is little overlap in ages at length for fish that are one and/or two years of age in a survey of abundance. The first mode in a survey can generally index age-1 recruitment using length slicing. In addition numbers and the length frequency of the larger fish (adult fish) in a survey where overlap in ages at a particular length occurs can be used for tuning population abundance. The model tunes to the catch and survey length frequency data using a multinomial distribution. The user specifies the minimum size (cm) for the model to fit. Different minimum sizes can be fit for the catch and survey data length frequencies.

The number of parameters estimated is equal to the number of years in estimating F and recruitment plus one for the F to produce the initial population (Fstart), logistic selectivity parameters for each year or blocks of years, and for each survey Q. The total likelihood function to be minimized is made up of likelihood components comprised of fits to the catch, catch length frequencies, the recruitment variation penalty, each recruitment index, each adult index, and adult survey length frequencies:

$$\mathrm{L}_{\text{catch}} = \sum_{years}\left(l\,\mathrm{n}(Y_{\text{obs,y}} + 1) - \ln\left(\sum_{a}\sum_{len} \mathrm{Y}_{\text{pred,len,a,y}} + 1 \right)\right)^2$$

$$L_{catch_lf} = -N_{eff} \sum_{y}\left(\sum_{inlen}^{L_\infty}\left(\left(C_{y,len} + 1\right)\ln\left(1 + \sum_{a} C_{pred,y,a,len} \right) - \ln\left(C_{y,len} + 1\right)\right) \right)$$

$$\mathrm{L}_{\text{vrec}} = \sum_{y=2}^{Nyears} \left(Vrec_y\right)^2 = \sum_{y=2}^{Nyears} \left(R_1 - R_y\right)^2$$

$$\sum L_{rec} = \sum_{i=1}^{Nrec} \left[\sum_{y}^{Nyears} \left(\ln\left(I_{rec_i, inage_i, y}\right) - \ln\left(\sum_{len}^{L_\infty} N_{y, inage_i, len} * q_{rec\,i} \right) \right)^2 \right]$$

$$\sum L_{adult} = \sum_{i=1}^{Nadult} \left[\sum_{y}^{Nyears} \left(\ln\left(I_{adult_i, inlen+_i, y}\right) - \left(\sum_{a} \sum_{inlen_i}^{L_\infty} \ln\left(N_{pred, y, a, len} * q_{adult\,i}\right) \right) \right)^2 \right]$$

$$\sum L_{lf} = \sum_{i=1}^{Nlf} \left[-N_{eff} \sum_{y} \left(\sum_{inlen_i}^{L_\infty} \left(\left(I_{lf_i, y, len} + 1\right) \ln\left(1 + \sum_{a} N_{pred, y, a, len} \right) - \ln\left(I_{lf_i, y, len} + 1\right) \right) \right) \right]$$

In equation L_{catch_lf} calculations of the sum of length are made from the user input specified catch length to the maximum length for fitting the catch. Input user specified fits are indicated with the prefix "in" in the equations. LF indicates fits to length frequencies. In equation L_{rec} the input specified recruitment age and in L_{adult} and L_{lf} the input survey specified lengths up to the maximum length are used in the calculation.

$$Obj\ fcn = \sum_{i=1}^{N} \lambda_i L_i$$

Lambdas represent the weights to be set by the user for each likelihood component in the total objective function.

Tilefish SCALE Model Configuration and results

Two growth studies are available for Golden tilefish (Figure A36 and A37). Turner's aging study was done during the development of the longline fishery (1978-1982). Vidal updated growth from fish collected recently after three decades of fishing in 2008 (Appendix A2). Inferences on the assumed natural mortality were made using Turner's aging work since landings were relativity low before this period. Tilefish have sexual dimorphic growth with the males growing larger than the females. There is some indication from the study fleet length distributions by sex that a greater proportion of the larger fish are males (Figures A38 and A39). Natural mortality may be higher on male than females judging from the number of older fish seen by sex in Turner's sample (Table A16 and A17). In general Turner saw fewer older males than females during his study. Vidal's study was done after a long period of fishing in which the directed longline fishery was active. Large fish were present in Vidal's sampling but very few older fish (>20) were aged. The lack of older fish in Vidal study made the estimation of L infinity more difficult. The sensitivity of the SCALE model results to the assumed growth model (Turner's and Vidal's) was examined (Table A18). The modeling of growth as a combined sex model or with sex specific growth was also investigated. A natural mortality rate of 0.15 on males and 0.1 on females was assumed in runs when sex specific

growth was used. In the combined sex model a natural mortality rate of 0.1 was used. The assumed variation around the mean lengths at age can be seen in Table A15 and Figure A40. The sensitivity of the assumed variation (run 5) around the mean lengths at age was also examined with a run were the variation in the mean lengths at age was increased (Table A18). The length weight relationship was updated using the data collected from the study fleet and growth study (Figure A41). The update relationship was used in the SCALE model. However the update relationship did not differ greatly from Turner's estimate.

A model which used Vidal's growth by sex and estimated selectivity in two time blocks (1971-1981, 1982-2008) was used as the base run (Table A18 and Figure A42 through A46). The SCALE model was dimensioned from ages 1-35, lengths 1-120 cm from years 1971-2008 as either a combined sex or separate sex model. A recruitment index does not exist for tilefish so a straight line index (constant recruitment index) was used as a proxy for an index with the model allowed to loosely fit the recruitment index (Figure A42). A low penalty weight (0.05) on recruitment variation was use in fitting the recruitment. The SCALE model appears to be able to pick up a recruitment signal from the commercial expanded length frequency distributions. The same general recruitment trend is estimated by the model even when yearly selectivity blocks were used. However this model run was not used since large changes in selectivity on a yearly basis seem unrealistic. A proxy for a recruitment index was developed as a sensitivity run (Table A18; run 6). This was done by through the redistribution of the VTR CPUE index according to the proportion of the expanded landing length frequency distribution and then slicing out the 40-50 cm fish as an age 5 index of recruitment (Figure A47). The CPUE indices were fit to fish sizes that were approximate according to the landing length frequency distributions. Turner's CPUE series was fit to 47+ cm fish and the Weighout and VTR series were fit to 37+ cm fish.

The catch length frequency distributions are an important component of the SCALE model. Turner collected landing length frequency information in 1974 and from 1976 to 1982. Note that Turner's length frequency data is only available in 5 cm blocks. NEFSC expanded landing size information exist from 1995 to 1999 and from 2002 to 2008. There appears to be a shift to smaller fish sizes between 1981 and 1982 in Turner's size distributions. Two selectivity blocks were assumed in the SCALE model (1971-1981, 1982-2008). The sensitivity of assuming a single selectivity block (run 3) over the time series was also tested. However in some years this run has trouble fitting the left side of the catch length frequency distribution due to the apparent change in selectivity over the time series.

The SCALE model time series starts in 1971 at the beginning of the tilefish directed longline fishery. However the SCALE model estimates an Fstart close to 0.2. This estimated equilibrium F that is assumed to occur before the beginning the time series appears to be on the high end since there was only a small limited fishery before 1971. A strong retrospective pattern did not exist in the base run (Figure A48). Little differences in the results are seen among the different model configurations (Table A18). There is a general concern with the lack of data and with the data independence used in the SCALE model. The lack of tuning information may result in little difference between the sensitivity runs. The lack of data, in particular the lack of recruitment index, could be preventing the mcmc from producing realistic results so uncertainty estimates around a particular model run could not be estimated. The estimated selectivity curve is also a source of concern given the tilefish longline fleet has some ability to target certain fish sizes by fishing different areas and depths. The SCALE model estimates of F during the late 1990s appear to be unrealistically high (over ten times F_{MSY}), while estimates of biomass in that period were correspondingly unrealistically low.

Term of Reference 3: Biological Reference Points

TOR3: Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and F_{MSY}). Comment on the scientific adequacy of existing and redefined BRPs.

ASPIC Surplus Production Model

Biological reference points estimated by the 2009 BASE run are moderately different from the 2005 SAW 41 assessment (Table A19). B_{MSY} is estimated to be 11,400 mt (a 22% increase), F_{MSY} is estimated to be 0.16 (a 24% decrease), and MSY is estimated to be 1,868 mt (a 6% decrease), compared to B_{MSY} = 9,384 mt, F_{MSY} = 0.21, and MSY = 1,988 mt from the 2005 SAW 41 assessment. The bootstrap (1000 iterations) median estimate (50%ile) of B_{MSY} was 10,135 mt; quartiles were 8,974 mt (25%ile) and 11,436 mt (75%ile). The bootstrap mean estimate of B_{MSY} was 10,336 mt, with a standard deviation (sd) of 2,089 mt and coefficient of variation (cv; sd/mean) of 20%. The bootstrap median (50%ile) estimate of F_{MSY} was 0.19; quartiles were 0.16 (25%ile) and 0.23 (75%ile). The bootstrap mean estimate of F_{MSY} was 0.20, with a standard deviation (sd) of 0.06 and coefficient of variation (cv; sd/mean) of 30%. The bootstrap results indicated that deterministic point estimates of the reference points are likely to be more precise than those accepted for the 2005 SAW 41 assessment, and are negatively biased by about 9% for B_{MSY} and positively biased by about 21% for F_{MSY} (Table A19).

SCALE model

Non-parametric yield per recruit (F_{MAX}) and spawners per recruit (F_{40}) biological reference points (BRP) were developed for SCALE base run 1 (separate sex model, two selectivity blocks) and run 2 (combined sex model, two selectivity blocks) (Table A20). BRPs were estimated both within the SCALE model and by converting the YPR inputs (selectivity, maturity schedule, stock and catch weights) to age based equivalents for use in an age based yield per recruit model (Table A21). The update maturity schedule from Vidal was used in the SPR analysis (Figure A49). MSY and SSB_{MSY} BRPS were estimated from the product of the model estimated initial recruitment (long term average recruitment) and the YPR or SSB per recruit estimates. The conversion to an age based YPR recruit model and an age based projection using AGEPRO is only possible in SCALE runs which modeled growth with the sexes combined (Figure A50). Similar BRPs are seen between the two methods (age based and SCALE). Uncertainty in recruitment can be incorporated into the AGEPRO projection by resampling from the CDF of the recruitment estimates. Reference points can also be estimated from long term projections with the CDF of recruitment and a F_{MSY} proxy. An example for run 2 using the CDF for the entire time series of recruitment and F_{MAX} produced a higher estimate of SSB_{MSY} at 14,000 mt relative to the simple product calculation of around 10,000 mt in Table A20 (Figure A51). The SSB_{MSY} estimate for the separate sex run is based on female fish (run 1). Note that a female estimate of SSB_{MSY} is not possible using the age based YPR model. In addition the age based projections in AGRPRO can not account for the sex specific effects that exist in the separate sex model. However for the separate sex model a simple deterministic projection can be done within the SCALE model.

The estimates of F_{MAX} and F_{40} were similar to the estimates from SARC 41 (F_{MAX} = 0.138 and F_{40}=0.08). F_{MAX} is estimated from a well defined yield curve (Figure A52). The predicted terminal year age and length distributions were slightly truncated in comparison to the equilibrium distribution at F_{MAX} for both run1 and run 2 (Figure A53). Run 2 has a greater proportion of larger

fish in the F_{MAX} equilibrium distribution relative to run 1 because run 1 assumes a higher natural mortality rate on males (Figure A52). SCALE YPR BRPs suggest that SSB_{MSY} is between 9,878 mt and 15,108 mt for the combine sex run using F_{40} or F_{MAX} as the F_{MSY} proxy (Table A20). The separate sex run suggests female SSB_{MSY} is between 5,335 mt and 7,100 mt. For both the single sex and separate sex run the F_{MSY} is between 0.079 and 0.128 and MSY ranging from 1,072 mt to 1,200 mt using either F_{40} or F_{MAX} as the F_{MSY} proxy.

Term of Reference 4: Stock Status

TOR4: Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 3).

ASPIC Surplus Production Model

The 2009 BASE model run results indicate that the Golden tilefish stock is not overfished and that overfishing is not occurring. With respect to the reference points from the 2005 SAW 41 assessment, fishing mortality in 2008 was estimated to be 0.06, 29% of F_{MSY} = 0.21, and total biomass in 2008 was estimated to be 11,910 mt, 127% of B_{MSY} = 9,384 mt. For this TOR note that for the ASPIC surplus production model it may not be appropriate to compare stock status relative to biological reference points from a different model run.

With respect to the updated reference points from the 2009 BASE run, fishing mortality in 2008 was estimated to be 0.06, 38% of F_{MSY} = 0.16. Total biomass in 2008 was estimated to be 11,910 mt, 104% of B_{MSY} = 11,400 mt (Table A13, Figure A54 and A55). The 50% confidence interval (range between the 25%ile and 75%ile) for the 2008 F/F_{MSY} ratio was between 0.25 and 0.42 and for the 2008 B/B_{MSY} ratio was between 0.87 and 1.46. The SARC 48 review panel accepted the ASPIC model but concluded that the ASPIC model is likely over optimistic and that the stock has not rebuilt above B_{MSY}.

SCALE Model

With respect to the existing reference points from the 2005 SAW 41 assessment, SCALE base run 1 fishing mortality in 2008 was estimated to be 0.188, 90% of F_{MSY} = 0.21, and total biomass in 2008 was estimated to be 4,950 mt, 53% of B_{MSY} = 9,384 mt. For this TOR note that this is a comparison of terminal year F (fully selected) and biomass from an age/size structured model relative to biological reference points from the SARC 41 surplus production model. This comparison results in a different status determination (no overfishing and not overfished) than if the update biological reference points were used.

With respect to the updated reference points from the SCALE BASE run (separate sex run), fishing mortality in 2008 was estimated to be 0.188, 147% of F_{MSY} = 0.128 using F_{MAX} as the proxy for F_{MSY}. Total female SSB in 2009 was estimated to be 2,520 mt, 47% of SSB_{MSY} = 5,335 mt using F_{MAX} as the proxy for F_{MSY}. With respect to the updated reference points from the SCALE (run2) combined sex run, fishing mortality in 2008 was estimated to be 0.205, 169% of F_{MSY} = 0.121 using F_{MAX} as the proxy for F_{MSY}. Total SSB in 2009 was estimated to be 4,399 mt, 41% of SSB_{MSY} = 10,794 mt using F_{MAX} as the proxy for F_{MSY}.

The 2009 BASE SCALE model run (separate sex run) and the combined sex run results indicate that the 2009 Golden tilefish stock is at a low biomass (29% to 47% of SSB_{MSY}) and is overfished with respect to the update SSB reference points. Both SCALE runs also suggest recruitment and growth overfishing (147% to 260% of F_{MSY}) is occurring with respect to the F_{40} or

F_{MAX} updated biological reference points. However fishing mortality has been decreasing and biomass has been increasing since the beginning of the FMP in 2001. Comparison of F to F_{MSY} and Biomass to B_{MSY} ratios over time between the ASPIC and SCALE model can be seen in figures A56 and A57.

Term of Reference 5: Projections

TOR 5: Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch).

a. **Provide numerical short-term projections (2-3 years). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions about the most important uncertainties in the assessment (alternate states of nature).**
b. **If possible, comment on the relative probability of the alternate states of nature and on which projections seem most realistic.**
c. **For a range of candidate ABCs, compute the probabilities of rebuilding the stock by November 1, 2011.**
d. **Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.**

ASPIC Surplus Production Model

Standard ASPIC model projections can either project fishery yield (i.e., total catch) for a given trajectory of F or project F for a given trajectory of yield. In neither case are any assumptions made about the future trajectory of the calibration indices - for tilefish, the commercial fishery VTR CPUE index series. For this assessment, two types of projections have been made. The first type is the standard ASPIC projection just described. The second type of projection makes assumptions about the future trajectory and magnitude of the VTR CPUE series in addition to projected F, catch, and biomass, and is intended to further respond to TOR5. The projections with the CPUE assumptions, however, result in changes in the overall model fit, re-scaling of the historical development of the stock, and different reference points. These results are therefore not directly comparable to the 2009 BASE run results, but should be useful in demonstrating how stock status might change in the future given some possible trends in fishery CPUE.

The standard projections were made for 2009-2011 assuming A) constant *status quo* catch = 905 mt, B) constant MSY catch = 1,868 mt, and C) constant F_{MSY} = 0.16. The *status quo* catch = 905 mt (1.995 million lb) has been the TAC since the FMP was implemented in 2001. Status determination was evaluated with respect to the updated reference points from the 2009 BASE run (threshold F_{MSY} = 0.16, target B_{MSY} = 11,400 mt, threshold B_{MSY} = 5,700 mt). Projection results for these three scenarios indicate 15%, 39%, and 45% chances that the stock will decline below the biomass target of B_{MSY} by 2011, and <1% chance that the stock will decline below the biomass threshold of ½ B_{MSY} by 2011. The projections indicate 0%, 40%, and 50% chances that F will exceed the fishing mortality threshold of F_{MSY} by 2011 (Table A22, Figures A58 and A59).

For the projections incorporating the CPUE index, runs were made with constant *status quo* catch = 905 mt, and 2009-2011 index assumptions of A) constant at the 1995-2008 average VTR CPUE = 2.095 (mt/da), B) constant at the 2001-2008 average VTR CPUE = 2.6475 C) increasing an

average rate of +25% per year, D) decreasing at an average rate of 25% per year, constant at the 2008 value of 1.434 (mt/da), and F) constant at the 2008 value rounded up to 1.4 (mt/da). Options C and D were specified to loosely mimic the ~25% average annual rate of increase in VTR CPUE during 2000-2005 that was followed by a ~33% decrease during 2005-2008. Status determination was evaluated with respect to the different reference points calculated in each run. For runs A, B and E (different mean levels of CPUE), the estimates of F_{MSY} increase and B_{MSY} and MSY decrease, relative to the 2009 BASE run estimates. These scenarios indicate about a 10% or less chance that biomass will decline below the target biomass B_{MSY} by 2011, and <1% chance that biomass will decline below the biomass threshold ½ B_{MSY} by 2011. For scenario C (increasing CPUE), F_{MSY}, B_{MSY}, and MSY all decrease, but like scenarios A and B, the projection indicates about a 10% chance that biomass will decline below B_{MSY} by 2011, and <1% chance that biomass will decline below ½ B_{MSY} by 2011 (Table A23, Figures A60 and A61). CPUE projection scenario E is *status quo* for both the fishery TAC and CPUE index, and so is considered the most likely in the short-term. Scenario E provides estimates of fishing mortality, stock biomass, and reference points in line with those from scenarios A, B and C. Scenario F is similar to the status quo CPUE of scenario E with the exception that the CPUE was rounded up to1 decimal place (CPUE was 1.4 instead of 1.434). This minor difference resulted in a large change in the results of the ASPIC model (Figure A62).

Projection scenario D (decreasing CPUE) re-scales the stock size and changes the reference points by a larger amount than the other four CPUE projection scenarios, and is particularly relevant to TOR5d. F_{MSY} decreases by about 60%, while B_{MSY} increases by 32% and MSY decreases by about 50%. These changes indicate a stock with lower resilience and productivity when compared to the other scenarios, in that the recent *status quo* TAC = 905 mt is above the estimated MSY. For scenario D, the time series estimates of F and B indicate that the stock has been below B_{MSY} since the late 1980s and F has consistently been above F_{MSY} since about 2000. The scenario D projection indicates a greater than 75% chance that fishing mortality will be above F_{MSY} and biomass will be below the target B_{MSY} by 2011, and a greater then 50% chance that biomass will be below the threshold ½ B_{MSY} by 2011 (Table A23, Figures A58 and A59). This projection scenario illustrates that the stock is vulnerable to being classified as "overfished" (below the threshold ½ B_{MSY}) if the VTR CPUE continues to decrease during 2009-2011 even as the catch remains near the recent *status quo*.

SCALE Model

As noted under TOR 3 age based projections can not be done in AGEPRO for SCALE separate sex model runs (base run 1). However, a deterministic projection can be done within the SCALE model by fixing the parameters in the model at the model solution and projecting into future years. Figure A63 and Figure A64 are examples of deterministic projections from run 1 at F_{MSY} = F_{MAX} = 0.13 and F_{2008} = 0.19, respectively. Combined sex model runs can be converted to an age based equivalent and projected using the AGEPRO projection program. Some uncertainty in recruitment can be accounted for in AGEPRO through resampling of the CDF of recruitment estimated from the SCALE model. Constant catch projections for run 2 (combined sex run) using agepro are shown in Figure A65. Note that using constant catches over 500 mt allows overfishing (F_{MSY} = F_{MAX}) in the first year of the projection.

Conclusions

The possibility of unknown refuge effects due to conflicts with lobster and trawl gear, effects

of targeting incoming year classes, and the unknown effects on tilefish CPUE due to competition/interference from increased dogfish abundance introduce uncertainty in interpreting CPUE from this fishery as a measure of stock abundance. CPUE index of abundance and catch length frequency distributions are likely a reflection of both the population abundance and the unaccounted changes in fishing practice.

The Working Group accepted the ASPIC model solution but noted that there is very high uncertainty regarding whether the stock is rebuilt. The SARC 48 review panel concluded that the ASPIC model is likely over optimistic and that the stock has not rebuilt above B_{MSY}. The surplus production model inability to fit the decline in CPUE due to at year class effect at the end of the time series is a source of concern. The bootstrap uncertainty estimates from the ASPIC model likely do not capture the true uncertainty in this assessment. Results from the SCALE model which incorporates the species lifespan, growth, and recruitment dynamics evident in the commercial length distributions provide reason to be concerned that the stock is not rebuilt. However the overall lack of data within the scale model and questions on the estimated selectivity may result in a pessimistic stock status determination. The uncertainty in this assessment is encompassed by the results from two very different models which resulted in different status determinations. However increases in biomass and lower fishing mortality rates since the beginning of the FMP are evident in the results from both models. Consideration should be given to the possibility that the SCALE model results may be a reflection of the true state of nature when setting ABCs rather then using the results of the ASPIC surplus production model which states that the stock is rebuilt.

Term of Reference 6: Research Recommendations

TOR 6: Review, evaluate and report on the status of the research recommendations offered in recent SARC reviewed assessments. Identify new research recommendations, including recruitment estimation.

New research recommendations from 2009 SARC 48

1) Continue the development of an improved haul based fishery dependent cpue index (i.e., continue the current study fleet project) or design a tilefish longline survey as a semi fishery independent index of abundance that could be conducted by an existing longline vessel and the study fleet platform. If a tilefish longline survey is developed then size information should be incorporated into the survey design for the estimation of a recruitment and size specific index of abundance which could improve the tilefish assessment.

2). For the study fleet project and any potential semi fishery independent survey, include additional information on conflicts with lobster and trawl gear, the possibility of unknown effects on tilefish CPUE due to competition/interference from an increased abundance of dogfish, the unknown effects of bait type on tilefish CPUE (e.g., substitutes for the preferred squid).

3). Develop protocols to ensure consistency between dealer, VTR, and IVR reports of the tilefish landings.

4). Develop protocols to ensure consistency in market category designation among fishing ports.

5). Explore the influence of water temperature and other environmental factors on trend in the commercial fishery CPUE index of stock abundance.

Research recommendations from the 2005 SARC 41 review

1) Conduct a hook selectivity study to determine partial recruitment changes with hook size.

Determine catch rates by hook size. Update data on growth, maturity, size structure, and sex ratios at length.
Hook selectivity study was not done. Funding was initially available, but subsequently rescinded. Updated growth, maturity, and size structure studies were completed.
2) Collect data on spatial distribution and population size structure. This can help answer the question of the existence of a possible dome shaped partial recruitment pattern where larger fish are less vulnerable to the fishery due to spatial segregation by size.
This research recommendation was examined in the study fleet data.
3) Continue to develop the forward projecting catch-length model as additional length data becomes available. Investigate the influence of adding a tuning index of abundance and model estimated partial recruitment (logistic) to the catch-length model.
This research recommendation was completed. The improved catch-length model was renamed as the SCALE model.
4) Collect appropriate effort metrics (number and size of hooks, length of main line, soak time, time of day, area fished) on a haul basis to estimate commercial CPUE.
This research recommendation was examined with the study fleet analysis.
5) Initiate a study to examine the effects of density dependence on life history parameters between the 1978-82 period and present.
This research recommendation was examined with the update growth and maturity study.
6) Increased observer coverage in the tilefish fishery to obtain additional length data.
Observer coverage has improved in the tilefish fishery.
7) Develop a bioeconomic model to calculate maximum economic yield per recruit.
This research recommendation has not been initiated.

Research recommendations from 1999 Science and Statistical Committee review
1) Ensure that market category distributions accurately reflect the landings. Sampling of the commercial lengths has improved over the last six years. Small, kitten, and medium market category distributions can shift from one year to the next due to the growth of a strong yearclass. Intensive length sampling of the landings by market categories is needed to account for possible shifts in the distribution within a market category over time. Similar landings distributions were seen among the observer, study fleet, and commercial port sampling data sources.
2) Ensure that length frequency sampling is proportional to landings by market category.
Commercial length sampling has been sporadic during the beginning of the time series. In particular length samples from the large market category have been lacking. However commercial length sampling has greatly improved over the last six years with a higher proportion of the sampling coming from Montauk where most of the fish are landed.
3) Increase and ensure adequate length sampling coverage of the fishery.
See comments for research recommendations 1 and 2.
4) Update age- and length- weight relationships.
This TOR has been addressed.
5) Update the maturity-at-age, weight-at-age, and partial recruitment patterns.
This TOR has been addressed.
6) Develop fork length to total length conversion factors for the estimation of total length to weight relationships.
This work was addressed in SARC 41.
7) Incorporate auxiliary data to estimate r independent of the ASPIC model.
This TOR has not been addressed. SARC 41 questioned if this can be done or should be done.

However SARC 48 SCALE results suggest that r is overestimated in the ASPIC model.

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Tables

Table A1. Landings of tilefish in live metric tons from 1915-2008. Landings in 1915-1972 are from Freeman and Turner (1977), 1973-1989 are from the general canvas data, 1990-1993 are from the weighout system, 1994-2003 are from the dealer reported data, and 2004-2008 is from Dealer electronic reporting. - indicates missing data.

year	mt	year	mt	year	mt
1915	148	1960	1,064	2005	676
1916	4,501	1961	388	2006	907
1917	1,338	1962	291	2007	751
1918	157	1963	121	2008	736
1919	92	1964	596		
1920	5	1965	614		
1921	523	1966	438		
1922	525	1967	50		
1923	623	1968	32		
1924	682	1969	33		
1925	461	1970	61		
1926	904	1971	66		
1927	1,264	1972	122		
1928	1,076	1973	394		
1929	2,096	1974	586		
1930	1,858	1975	710		
1931	1,206	1976	1,010		
1932	961	1977	2,082		
1933	688	1978	3,257		
1934	-	1979	3,968		
1935	1,204	1980	3,889		
1936	-	1981	3,499		
1937	1,101	1982	1,990		
1938	533	1983	1,876		
1939	402	1984	2,009		
1940	269	1985	1,961		
1941	-	1986	1,950		
1942	62	1987	3,210		
1943	8	1988	1,361		
1944	22	1989	454		
1945	40	1990	874		
1946	129	1991	1,189		
1947	191	1992	1,653		
1948	465	1993	1,838		
1949	582	1994	786		
1950	1,089	1995	666		
1951	1,031	1996	1,121		
1952	964	1997	1,802		
1953	1,439	1998	1,334		
1954	1,582	1999	508		
1955	1,629	2000	504		
1956	707	2001	871		
1957	252	2002	843		
1958	672	2003	1,130		
1959	380	2004	1,215		

Table A2. Percent landings by statistical area. Landings before 1990 are taken from the general canvas data. Percent landings after 1993 are estimated from the AA tables.

year	unknown	626	622	616	537	526	525	other
1962	100%	0%	0%	0%	0%	0%	0%	0%
1963	65%	0%	0%	0%	4%	28%	0%	3%
1964	83%	0%	0%	0%	4%	14%	0%	0%
1965	83%	0%	0%	0%	1%	16%	0%	0%
1966	97%	0%	0%	0%	0%	1%	1%	0%
1967	96%	0%	0%	0%	0%	4%	0%	0%
1968	96%	0%	0%	0%	1%	0%	0%	3%
1969	93%	0%	0%	0%	2%	4%	0%	1%
1970	87%	0%	0%	0%	8%	5%	0%	0%
1971	99%	0%	0%	0%	0%	0%	0%	0%
1972	92%	0%	0%	1%	1%	0%	0%	6%
1973	0%	0%	0%	62%	16%	0%	0%	21%
1974	0%	0%	0%	51%	27%	0%	0%	22%
1975	0%	0%	0%	48%	34%	8%	0%	10%
1976	0%	0%	0%	58%	28%	13%	0%	1%
1977	1%	0%	0%	44%	32%	22%	0%	1%
1978	0%	0%	0%	29%	40%	31%	0%	0%
1979	0%	0%	0%	18%	37%	45%	0%	0%
1980	0%	0%	0%	22%	34%	44%	0%	0%
1981	0%	0%	0%	28%	37%	35%	0%	0%
1982	0%	0%	0%	19%	52%	27%	0%	2%
1983	0%	1%	0%	22%	54%	23%	0%	0%
1984	0%	1%	3%	9%	53%	34%	0%	1%
1985	0%	0%	2%	25%	33%	38%	2%	1%
1986	0%	0%	1%	28%	44%	25%	3%	1%
1987	0%	0%	0%	12%	53%	32%	1%	2%
1988	0%	1%	2%	21%	41%	32%	0%	2%
1989	0%	0%	1%	63%	9%	26%	1%	1%
1990	0%	2%	0%	15%	14%	36%	0%	33%
1991	0%	0%	1%	64%	25%	1%	0%	10%
1992	0%	0%	1%	22%	70%	5%	1%	1%
1993	0%	0%	2%	14%	72%	7%	3%	2%
1994	0%	1%	1%	11%	78%	1%	2%	6%
1995	0%	0%	2%	26%	53%	0%	1%	19%
1996	0%	0%	0%	29%	61%	5%	0%	4%
1997	0%	0%	0%	18%	67%	0%	0%	15%
1998	0%	0%	0%	11%	68%	3%	1%	18%
1999	0%	0%	0%	32%	48%	0%	1%	18%
2000	0%	0%	0%	41%	38%	1%	0%	20%
2001	0%	0%	0%	61%	26%	4%	0%	9%
2002	0%	0%	0%	36%	40%	7%	1%	17%
2003	0%	0%	0%	42%	34%	2%	1%	21%
2004	0%	0%	0%	25%	53%	5%	1%	16%
2005	0%	12%	0%	25%	47%	0%	0%	16%
2006	0%	8%	0%	28%	46%	1%	0%	16%
2007	0%	0%	2%	31%	47%	0%	0%	20%

Table A3. Landings of tilefish (mt, live) by gear. Landing before 1990 are from the general canvas data. Percent by gear per year are also given.

	Gear				Percent by Gear		
Year	longli	traw	othei	Total	longline	trawl	other
1962		167	2	169	0%	99%	1%
1963		121		121	0%	100%	0%
1964		596		596	0%	100%	0%
1965		614		614	0%	100%	0%
1966		437		437	0%	100%	0%
1967		51		51	0%	100%	0%
1968		30		30	0%	100%	0%
1969		30		30	0%	100%	0%
1970		57	1	58	0%	99%	1%
1971		62	1	62	0%	99%	1%
1972	93	26	2	121	77%	21%	2%
1973	370	24	1	394	94%	6%	0%
1974	531	33	22	586	91%	6%	4%
1975	588	111	11	710	83%	16%	2%
1976	950	58	1	1,010	94%	6%	0%
1977	1,772	309	1	2,082	85%	15%	0%
1978	2,938	309	10	3,257	90%	9%	0%
1979	3,362	449	156	3,968	85%	11%	4%
1980	3,794	94	0	3,889	98%	2%	0%
1981	3,366	128	5	3,499	96%	4%	0%
1982	1,935	49	6	1,990	97%	2%	0%
1983	1,857	8	11	1,876	99%	0%	1%
1984	2,003	6	1	2,009	100%	0%	0%
1985	1,929	31	0	1,961	98%	2%	0%
1986	1,874	76	0	1,950	96%	4%	0%
1987	3,029	180	0	3,210	94%	6%	0%
1988	1,319	42		1,361	97%	3%	0%
1989	421	33	0	454	93%	7%	0%
1990	850	22	0	871	98%	2%	0%
1991	1,164	25	0	1,189	98%	2%	0%
1992	1,497	155	0	1,653	91%	9%	0%
1993	1,597	241	0	1,838	87%	13%	0%
1994	764	22	0	786	97%	3%	0%
1995	618	47	1	666	93%	7%	0%
1996	1,005	111	4	1,121	90%	10%	0%
1997	1,716	79	7	1,802	95%	4%	0%
1998	1,193	134	7	1,334	89%	10%	1%
1999	470	28	10	508	93%	6%	2%
2000	460	38	7	504	91%	7%	1%
2001	819	52	0	871	94%	6%	0%
2002	759	83	1	843	90%	10%	0%
2003	1,004	124	2	1,130	89%	11%	0%
2004	905	211	99	1,215	75%	17%	8%
2005	495	20	161	676	73%	3%	24%
2006	717	32	158	907	79%	3%	17%
2007	711	8	32	751	95%	1%	4%
2008	557	11	167	736	76%	2%	23%

Table A4. Landings of tilefish (mt, live) by state. Number of length measurements are in parentheses. Landings before 1990 are from general canvas data. Percent by state per year are also given.

Year	ME	MA	RI	NY	NJ	other	Total	Percent by State					
								ME	MA	RI	NY	NJ	other
1962	0	28	31	57	42	12	169	0%	16%	18%	34%	25%	7%
1963	0	42	46	13	14	6	121	0%	35%	38%	10%	12%	5%
1964	0	102	424	37	30	2	596	0%	17%	71%	6%	5%	0%
1965	0	106	478	20	9	2	614	0%	17%	78%	3%	1%	0%
1966	0	13	366	55	3	2	437	0%	3%	84%	13%	1%	0%
1967	0	2	27	8	8	5	51	0%	4%	54%	16%	17%	9%
1968	0	1	23	3	3	0	30	0%	4%	76%	9%	11%	0%
1969	0	2	13	4	10	0	30	0%	7%	44%	15%	35%	0%
1970	0	8	36	3	10	1	58	0%	13%	62%	5%	17%	2%
1971	0	0	21	25	15	1	62	0%	1%	34%	40%	24%	2%
1972	0	2	3	6	111	0	121	0%	1%	2%	5%	92%	0%
1973	0	51	17	3	323	0	394	0%	13%	4%	1%	82%	0%
1974	0	163	21	22	380	0	586	0%	28%	4%	4%	65%	0%
1975	0	174	101	2	434	0	710	0%	24%	14%	0%	61%	0%
1976	0	212	56	23	718	0	1,010	0%	21%	6%	2%	71%	0%
1977	0	84	354	314	1,331	0	2,082	0%	4%	17%	15%	64%	0%
1978	0	95	292	969	1,900	0	3,257	0%	3%	9%	30%	58%	0%
1979	0	22	432	1,365	2,148	0	3,968	0%	1%	11%	34%	54%	0%
1980	0	1	87 (37)	1,451	2,348	2	3,889 (37)	0%	0%	2%	37%	60%	0%
1981	0	6	126	1,284 (25)	2,083	1	3,499	0%	0%	4%	37%	60%	0%
1982	6	5	42 (87)	643	1,288	6	1,990 (87)	0%	0%	2%	32%	65%	0%
1983	0	12	7	844 (158)	1,001	12	1,876	0%	1%	0%	45%	53%	1%
1984	0	1	5	1,094	898 (116)	11	2,009 (116)	0%	0%	0%	54%	45%	1%
1985	2	10	207 (247)	958	777 (163)	6	1,961 (410)	0%	0%	11%	49%	40%	0%
1986	3	1	183 (70)	1,076 (107)	687	1	1,950 (177)	0%	0%	9%	55%	35%	0%
1987	0	7	269 (380)	1,996	924 (203)	13	3,210 (583)	0%	0%	8%	62%	29%	0%
1988	0	33	100 (98)	868	353	6	1,361 (98)	0%	2%	7%	64%	26%	0%
1989	0	1	28	249	174	1	454	0%	0%	6%	55%	38%	0%
1990	7	7	19	606	232	3	874	1%	1%	2%	69%	27%	0%
1991	4	1	19	720	444	1	1,189	0%	0%	2%	61%	37%	0%
1992	8	3	146	963 (36)	530	3	1,653 (36)	0%	0%	9%	58%	32%	0%
1993	59	14	276 (100)	1,003	485	1	1,838 (100)	3%	1%	15%	55%	26%	0%
1994	25	3	51	580	127	0	786	3%	0%	6%	74%	16%	0%
1995	8	1	20	560 (432)	76	1	666 (432)	1%	0%	3%	84%	11%	0%
1996	6 (108)	0	88 (219)	924	98 (328)	5	1,121 (655)	1%	0%	8%	82%	9%	0%
1997	13 (244)	0	54 (422)	1,577 (159)	82 (1,154)	74	1,802 (1,979)	1%	0%	3%	88%	5%	4%
1998	15	4	82 (320)	1,073 (74)	123 (606)	38	1,334 (1,000)	1%	0%	6%	80%	9%	3%
1999	3	2	75 (212)	377	40 (161)	12	508 (373)	1%	0%	15%	74%	8%	2%
2000	7	0	57	423 (143)	14	3	504 (143)	1%	0%	11%	84%	3%	1%
2001	0	0	33 (103)	833 (217)	4	1	871 (320)	0%	0%	4%	96%	0%	0%
2002	4	9	59 (482)	740 (850)	23	8	843 (1,332)	0%	1%	7%	88%	3%	1%
2003	2 (343)	12	104 (168)	848 (1,862)	157 (1,205)	6	1,130 (3,578)	0%	1%	9%	75%	14%	1%
2004	0 (31)	117 (19)	142 (388)	596 (789)	323 (2,159)	37	1,215 (3,386)	0%	10%	12%	49%	27%	3%
2005	0 (9)	3	12 (27)	454 (1,123)	122 (2,307)	85	676 (3,466)	0%	0%	2%	67%	18%	13%
2006	0 (14)	52 (446)	8 (55)	524 (2,176)	226 (3,076)	96	907 (5,767)	0%	6%	1%	58%	25%	11%
2007	1 (6)	0 (5)	7 (133)	632 (5,257)	108 (2,018)	2	751 (7,419)	0%	0%	1%	84%	14%	0%
2008	2	0	32 (607)	544 (3,316)	154 (1,271)	4	736 (5,194)	0%	0%	4%	74%	21%	1%

Table A5. Landings of tilefish (mt, live) by quarter. General canvas data are not included. Percent by quarter per year are also given.

	Quarter								
Year	1	2	3	4	Total	1	2	3	4
1977	1,017	961	93	12	2,082	49%	46%	4%	1%
1978	905	1,128	432	793	3,257	28%	35%	13%	24%
1979	1,351	1,055	538	1,024	3,968	34%	27%	14%	26%
1980	1,524	1,263	505	596	3,889	39%	32%	13%	15%
1981	1,352	1,091	474	581	3,499	39%	31%	14%	17%
1982	1,028	433	239	289	1,990	52%	22%	12%	15%
1983	577	726	289	284	1,876	31%	39%	15%	15%
1984	1,032	491	293	193	2,009	51%	24%	15%	10%
1985	551	632	496	281	1,961	28%	32%	25%	14%
1986	542	597	437	374	1,950	28%	31%	22%	19%
1987	1,048	873	723	565	3,210	33%	27%	23%	18%
1988	737	292	160	172	1,361	54%	21%	12%	13%
1989	147	61	78	167	454	32%	13%	17%	37%
1990	258	240	184	189	871	30%	28%	21%	22%
1991	326	437	182	244	1,189	27%	37%	15%	21%
1992	426	433	401	393	1,653	26%	26%	24%	24%
1993	634	664	267	273	1,838	34%	36%	15%	15%
1994	301	275	72	138	786	38%	35%	9%	18%
1995	214	148	108	195	666	32%	22%	16%	29%
1996	366	215	231	308	1,121	33%	19%	21%	28%
1997	442	571	370	419	1,802	25%	32%	21%	23%
1998	537	361	228	209	1,334	40%	27%	17%	16%
1999	162	135	116	96	508	32%	27%	23%	19%
2000	143	141	76	144	504	28%	28%	15%	29%
2001	190	235	222	223	871	22%	27%	26%	26%
2002	287	197	172	188	843	34%	23%	20%	22%
2003	314	314	242	260	1,130	28%	28%	21%	23%
2004	530	272	187	226	1,215	44%	22%	15%	19%
2005	178	119	170	209	676	26%	18%	25%	31%
2006	281	200	188	238	907	31%	22%	21%	26%
2007	196	175	177	203	751	26%	23%	24%	27%
2008	292	191	116	137	736	40%	26%	16%	19%

Table A6. Observer trawl trips which either kept and/or discarded tilefish in kgs. Discard to kept ratio, the number of trips and observed hauls are also shown.

year	discard kgs	kept kgs	d/k ratio	No. trips	No. hauls
1989	114	131	0.88	8	43
1990	9	85	0.11	4	11
1991	252	449	0.56	19	69
1992	182	856	0.21	22	84
1993	21	4,625	0.00	13	77
1994	14	119	0.11	7	23
1995	20	23	0.90	6	13
1996	57	1,515	0.04	11	53
1997	196	1,082	0.18	13	71
1998	45	522	0.09	11	92
1999	31	153	0.20	14	47
2000	116	112	1.04	8	25
2001	654	456	1.44	10	54
2002	5	58	0.08	3	6
2003	278	1,276	0.22	16	69
2004	420	1,777	0.24	50	205
2005	1,099	1,367	0.80	98	237
2006	439	472	0.93	44	143
2007	84	145	0.58	21	49
2008	275	451	0.61	24	57

Table A7. Total commercial and vessel trip report (VTR) landings in live mt and the commercial catch-per-unit effort (CPUE) data used for tilefish. Dealer landings before 1990 are from the general canvas data. CPUE data from 1979 to the first half of 1994 are from the NEFSC weighout database, while data in the secound half of 1994 to 2004 are from the vtr system (below the dotted line). Effort data are limited to longline trips which targeted tilefish (= or >75% of the landings were tilefish) and where data existed for the days absent. Nominal CPUE series are calculated using landed weight per days absent minus one day steam time per trip. Da represents days absent.

	Weighout		Commerical CPUE data subset								
year	& Dealer landings	vtr landings	interview landings	No. interviews	% interview trips	No. vessels	subset landings	days absent	No. trips	da per trip	nominal cpue
1979	3,968		0.0	0	0.0%	20	1,807	1,187	330	3.6	1.93
1980	3,889		0.8	1	0.3%	18	2,153	1,390	396	3.5	1.99
1981	3,499		35.0	4	1.2%	21	1,971	1,262	333	3.8	1.95
1982	1,990		90.7	13	5.7%	18	1,267	1,282	229	5.6	1.10
1983	1,876		85.8	16	8.9%	21	1,013	1,451	179	8.1	0.73
1984	2,009		140.1	25	18.2%	20	878	1,252	138	9.1	0.72
1985	1,961		297.1	64	30.6%	25	933	1,671	209	8.0	0.59
1986	1,950		120.7	31	16.5%	23	767	1,186	188	6.3	0.71
1987	3,210		198.5	38	18.5%	30	1,014	1,343	206	6.5	0.82
1988	1,361		148.2	30	19.4%	23	422	846	154	5.5	0.56
1989	454		92.8	11	15.7%	11	165	399	70	5.7	0.46
1990	874		32.4	8	11.9%	11	241	556	68	8.2	0.45
1991	1,189		0.8	3	2.8%	7	444	961	107	9.0	0.48
1992	1,653		58.0	9	8.6%	13	587	969	105	9.2	0.62
1993	1,838		71.9	11	10.5%	10	571	959	105	9.1	0.61
1994	-		0	0	0.0%	7	127	385	42	9.2	0.34
1994	786	30				4	26	76	9	8.4	0.36
1995	666	547				5	470	964	100	9.6	0.50
1996	1,121	865				8	822	1,318	134	9.8	0.64
1997	1,810	1,439				6	1,427	1,332	133	10.0	1.09
1998	1,342	1,068				9	1,034	1,517	158	9.6	0.70
1999	525	527				10	516	1,185	133	8.9	0.45
2000	506	446				11	427	942	110	8.6	0.47
2001	874	705				8	691	1,046	116	9.0	0.68
2002	851	724				8	712	951	114	8.3	0.78
2003	1,130	790				7	788	691	101	6.8	1.22
2004	1,215	1,153				12	1,136	811	134	6.1	1.54
2005	676	808				11	802	470	93	5.1	1.95
2006	907	870				12	852	682	105	6.5	1.35
2007	751	710				12	691	727	101	7.2	1.01
2008	736	622				12	620	1,034	113	9.2	0.62

Table A8. Dealer, VTR, and IVR tilefish total landings (live metric tons) compared to the total landings from the five dominant tilefish vessels. Percent of five dominant vessels to the total are also shown.

year	Dealer total (live mt)	Dealer top 5 vessels	Dealer % landing of top 5 vessels to total	VTR total (live mt)	VTR top 5 vessels	VTR % landing of top 5 vessels to total	IVR total (live mt)	IVR top 5 vessels	IVR % landing of top 5 vessels to total
1994	786	485	62%	31	17	57%	-	-	-
1995	666	522	78%	549	538	98%	-	-	-
1996	1,121	803	72%	865	799	92%	-	-	-
1997	1,810	1,292	71%	1,439	1,416	98%	-	-	-
1998	1,342	948	71%	1,068	1,003	94%	-	-	-
1999	508	399	79%	527	486	92%	-	-	-
2000	504	459	91%	446	428	96%	-	-	-
2001	871	817	94%	705	684	97%	-	-	-
2002	843	733	87%	724	687	95%	766	727	95%
2003	1,130	784	69%	790	732	93%	894	779	87%
2004	1,215	561	46%	1,153	688	60%	944	687	73%
2005	676	473	70%	808	596	74%	868	670	77%
2006	907	555	61%	870	569	65%	901	595	66%
2007	751	609	81%	710	601	85%	762	651	85%
2008	736	535	73%	622	466	75%	709	542	76%

Table A9. Landing (metric tons) by market category. Small kitten market category was added to kittens.

year	small	kittens	medium	large	xl	unclassified	total
1990	24	14	103	45	0	687	871
1991	43	16	154	85	0	891	1,189
1992	193	136	88	86	0	1,149	1,653
1993	237	131	206	66	4	1,193	1,838
1994	8	11	89	54	7	617	786
1995	26	73	88	91	2	386	666
1996	169	423	149	156	2	221	1,121
1997	249	878	257	110	2	306	1,802
1998	97	375	699	103	6	54	1,334
1999	37	143	197	106	8	17	508
2000	17	193	153	114	8	19	504
2001	11	553	160	124	6	18	871
2002	26	341	311	128	3	34	843
2003	132	644	170	144	5	34	1,130
2004	169	248	523	129	9	137	1,215
2005	6	12	335	149	1	173	676
2006	8	9	233	369	1	287	907
2007	17	81	148	397	4	105	751
2008	68	99	194	297	18	60	736

Table A10. Number of lengths (1995-2008), samples (2002-2008), and metric tons landed per sample (2002-2008) for Golden tilefish. Number of lengths includes borrowing across years in bold. Trawl lengths were not used in the expansion. Large lengths used from 1995 to 1999 were taken from years 1996, 1997, and 1998. Large lengths in 2002 also used large lengths from 2003. Unclassified were redistributed according to mkt and qtr proportions.

Number of lengths.

year	half	sm	ki	med	lg	xl	total
1995	1		244	208	**332**		
	2						784
1996	1		312	100	**332**		
	2						744
1997	1		958	688	**332**		
	2						1978
1998	1		202	407	**332**		
	2						941
1999	1		211	155	**332**		
	2						698

Number of lengths.

year	half	sm	ki	med	lg	xl	total
2002	1		353	206	**492**		
	2						1051
2003	1	735	385	396	467	32	
	2		522	958			3495
2004	1	788	115	882	432		
	2	106	197	427			2947
2005	1		393	1378	825		
	2			763			3359
2006	1	112	346	1856	1284		
	2	218		1079	752		5647
2007	1	396	379	1128	898	25	
	2	220	1152	1871	1316		7385
2008	1	93	719	1356	1506	20	
	2		369	339			4402

Number of samples

half	sm	ki	med	lg	xl	total
1		6	2	**8**		
2						16
				↑		
1	5	4	3	7	2	
2		6	5			32
1	4	1	6	7		
2	1	2	4			25
1		6	10	12		
2			8			36
1	3	6	14	11		
2	2		11	8		55
1	4	4	12	12	1	
2	1	5	9	8		56
1	1	9	16	28	3	
2		4	6			67

mt/samples

half	sm	ki	med	lg	xl	total
1		61	156	**19**		
2						54
1	26	98	22	21	3	
2		42	21			34
1	37	209	50	20		
2	23	20	55			43
1		3	19	12		
2			18			14
1	2	1	9	19		
2	2		9	21		11
1	1	6	6	18	4	
2	12	11	8	23		12
1	49	8	7	11	6	
2		12	13			10

Table A11. Recreational Golden tilefish data from the Marine Recreational Fishery Statistics Suvey (MRFSS).

year	number fish measured	landed no. A and B1	Released B2	A and B1 kg
1982	0	984	0	98
1983	0	0	0	0
1984	0	0	0	0
1985	0	0	0	0
1986	0	0	0	0
1987	0	0	0	0
1988	0	0	0	0
1989	0	0	0	0
1990	0	0	0	0
1991	0	0	0	0
1992	0	0	0	0
1993	0	0	0	0
1994	0	608	0	0
1995	0	0	0	0
1996	0	10,167	0	0
1997	0	0	0	0
1998	0	0	0	0
1999	0	0	0	0
2000	0	0	0	0
2001	0	148	0	0
2002	0	20,068	1,338	0
2003	18	722	0	2,126
2004	3	112	0	317
2005	0	0	0	0
2006	0	1,208	0	0
2007	2	1,515	0	6,720
2008	0	0	0	0

Table A12. Number of tilefish reported in the Party/charater vessel trip reports.

year	ME	MD	NH	NJ	NY	NC	RI	VA	other	total
1994	275	0	636	0	0	0	0	0	0	911
1995	0	0	0	0	176	0	541	0	0	717
1996	0	0	0	0	81	0	0	0	0	81
1997	0	0	0	0	380	0	0	0	20	400
1998	0	0	0	0	121	52	102	0	20	295
1999	0	6	0	0	88	34	1	0	0	129
2000	0	0	0	39	108	139	0	0	0	286
2001	0	0	0	100	122	1,164	0	0	0	1,386
2002	0	0	0	383	425	0	0	0	0	808
2003	0	0	0	905	71	0	3	0	15	994
2004	0	0	0	624	12	0	0	254	0	898
2005	0	0	0	364	82	25	72	16	14	573
2006	0	133	0	66	265	30	0	12	2	508
2007	0	5	0	457	447	313	0	138	88	1,448
2008	0	30	0	140	383	60	2	10	22	647

Table A13. ASPIC surplus production model run comparison and sensitivity.

Run ID	2005 SAW 41 ASPIC v3.93	2005 SAW 41 ASPIC v5.33	**2009 SAW 48 ASPIC v5.33 BASE; B1/K = 0.5**	2009 SAW 48 ASPIC v5.33 B1/K = 0.1	2009 SAW 48 ASPIC v5.33 B1/K = 1.0	2009 SAW 48 ASPIC v5.33 EST B1/K = 1.19
Diagnostics						
RMSE	0.3069	0.3069	**0.3496**	0.5362	0.3357	0.3401
turner r2	0.180	0.180	**0.224**	-0.715	0.545	0.593
Weighout r2	0.703	0.703	**0.652**	-0.129	0.680	0.684
vtr r2	0.538	0.538	**0.201**	-0.058	0.230	0.232
Turner q	0.0133	0.0133	**0.0088**	0.0108	0.0076	0.0074
Weighout q	0.2246	0.2246	**0.1754**	0.1046	0.1771	0.1762
VTR q	0.3921	0.3921	**0.2604**	0.1684	0.2622	0.2632
Results						
B1:K ratio	0.50	0.50	**0.50**	0.10	1.00	1.19
MSY (mt)	1,988	1,988	**1,868**	11,220	1,706	1,680
r	0.4236	0.4238	**0.3278**	4.0000	0.3502	0.3514
FMSY	0.2118	0.2119	**0.1639**	2.0000	0.1751	0.1757
K (mt)	18,770	18,766	**22,790**	11,220	19,490	19,130
BMSY (mt)	9,384	9,383	**11,400**	5,608	9,745	9,565
B2004/BMSY	0.65	0.65	**0.78**	1.95	0.86	0.87
F2004/FMSY	0.87	0.87	**0.56**	0.05	0.81	0.81
B2008/BMSY	n/a	n/a	**1.04**	1.97	1.17	1.18
F2008/FMSY	n/a	n/a	**0.38**	0.03	0.35	0.36

Table A14. 2009 BASE run retrospective estimated parameters.

	Turner	Qs Weighout	VTR		Turner	Qs Weighout	VTR
1999	0.0079	0.1584	0.3333	1998	0.0025	0.0478	0.1479
2002	0.0085	0.1721	0.3408	2000	0.0025	0.0480	0.1503
2003	0.0094	0.1983	0.3572	2001	0.0024	0.0438	0.1319
2004	0.0104	0.2254	0.3925				
2005	0.0111	0.2487	0.4427				
2006	0.0192	0.2430	0.4272				
2007	0.0101	0.2134	0.3484				
2008	0.0088	0.1754	0.2604				
Mean	0.0107	0.2043	0.3628	Mean	0.0024	0.0465	0.1434
Max	0.0192	0.2487	0.4427	Max	0.0025	0.0480	0.1503
Min	0.0079	0.1584	0.2604	Min	0.0024	0.0438	0.1319

	MSY	K	RMSE		MSY	K	RMSE
1999	1,780	26,030	0.3022	1998	38	103,900	0.3086
2002	1,831	23,980	0.2915	2000	38	103,700	0.2968
2003	1,916	20,940	0.2990	2001	38	107,100	0.3023
2004	1,990	18,710	0.3073				
2005	2,048	17,230	0.3111				
2006	2,034	17,560	0.3067				
2007	1,963	19,510	0.3173				
2008	1,868	22,790	0.3496				
Mean	1,929	20,844	0.3106	Mean	38	104,900	0.3026
Max	2,048	26,030	0.3496	Max	38	107,100	0.3086
Min	1,780	17,230	0.2915	Min	38	103,700	0.2968

Table A15. Numbers at age and length from SCALE base run 1 which used sex specific growth curves.

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35
133																																			1
132																															1	1	1	1	1
131																												1	1	1	2	2	2	3	3
130																										1	1	2	2	3	4	5	6	6	7
129																									1	1	3	4	6	8	9	11	12	13	14
128																								1	2	4	6	9	13	16	20	23	25	26	27
127																							1	3	5	9	15	21	27	33	39	43	46	47	48
126																						1	3	7	13	21	31	42	53	63	70	76	79	80	79
125																					1	3	8	16	29	44	62	80	97	110	119	125	126	124	120
124																				1	3	8	19	36	59	87	115	142	163	179	187	190	187	180	171
123																				2	7	19	41	73	114	157	198	232	256	270	274	269	258	243	225
122																			1	6	18	43	84	140	203	264	315	352	373	378	371	354	330	303	275
121																		1	4	14	40	89	161	248	336	412	467	497	505	493	467	432	393	352	312
120																		2	10	33	86	172	286	408	517	597	642	652	634	596	546	490	434	379	329
119																	1	5	24	73	169	309	471	624	739	804	819	793	740	669	593	516	446	380	322
118																	2	14	54	149	309	514	721	885	981	1006	971	897	801	698	598	505	424	353	293
117																1	6	33	114	280	534	794	1024	1167	1210	1168	1070	942	806	677	560	459	375	305	248
116																2	16	74	225	491	826	1140	1352	1429	1385	1260	1094	919	754	609	487	388	308	244	195
115																6	38	155	413	799	1210	1519	1656	1624	1473	1263	1039	832	654	509	393	304	235	182	142
114															1	15	86	300	701	1207	1646	1881	1885	1715	1455	1175	917	700	528	395	296	222	167	126	97
113															4	36	180	540	1107	1693	2078	2162	1993	1682	1335	1015	752	547	396	286	207	151	111	83	62
112														1	10	83	349	904	1623	2207	2437	2308	1956	1533	1138	816	573	399	277	194	137	98	71	53	40
111														2	27	176	628	1405	2211	2670	2654	2289	1784	1298	902	611	409	274	185	127	90	65	49	38	30
110														5	64	348	1051	2028	2795	3001	2686	2109	1513	1023	669	431	279	184	126	90	68	55	46	40	36
109													1	15	140	637	1633	2718	3283	3134	2525	1807	1196	757	472	297	195	137	104	86	76	70	66	63	60
108													2	37	288	1086	2356	3383	3581	3040	2208	1446	891	539	335	224	167	139	127	122	120	119	117	115	111
107													5	86	549	1718	3158	3911	3630	2744	1804	1092	646	399	276	223	205	203	206	211	214	215	212	208	201
106													15	188	971	2523	3931	4201	3423	2312	1393	808	498	359	313	310	326	344	360	371	375	373	367	356	342
105												1	37	380	1594	3443	4546	4196	3010	1839	1053	643	478	442	465	508	551	585	608	618	618	608	591	568	541
104												4	89	714	2432	4364	4886	3904	2490	1423	853	638	615	675	760	840	903	943	963	964	949	921	885	842	795
103												10	196	1246	3446	5139	4887	3404	1984	1157	846	828	939	1084	1217	1321	1388	1419	1421	1397	1354	1297	1231	1160	1085
102												27	400	2018	4537	5626	4564	2823	1612	1115	1076	1246	1472	1680	1839	1941	1987	1986	1947	1881	1795	1697	1592	1484	1376
101											1	66	760	3036	5549	5727	4014	2315	1481	1354	1578	1907	2215	2448	2592	2653	2644	2581	2479	2352	2210	2061	1911	1763	1620
100											4	151	1339	4244	6312	5471	3398	2026	1671	1914	2360	2795	3125	3322	3396	3370	3269	3116	2932	2732	2527	2325	2130	1945	1772
99											12	322	2194	5513	6689	4933	2904	2078	2234	2795	3387	3841	4105	4191	4134	3975	3752	3493	3220	2947	2684	2436	2205	1993	1799
98											33	635	3339	6659	6634	4306	2712	2551	3172	3947	4562	4914	5013	4911	4673	4356	4000	3637	3284	2953	2647	2370	2120	1897	1697
97											80	1165	4720	7492	6220	3819	2958	3466	4424	5248	5724	5845	5687	5345	4907	4432	3961	3517	3111	2747	2425	2141	1893	1676	1487
96										2	183	1985	6205	7881	5635	3691	3711	4764	5848	6508	6678	6458	5991	5403	4785	4188	3642	3158	2737	2373	2063	1797	1570	1376	1209
95										5	388	3141	7593	7813	5138	4081	4946	6296	7229	7507	7237	6626	5861	5073	4333	3676	3110	2634	2236	1905	1629	1400	1209	1049	914
94										16	763	4621	8671	7418	4998	5048	6527	7827	8320	8045	7284	6315	5326	4423	3644	2996	2467	2040	1696	1419	1195	1013	865	742	641
93										43	1394	6323	9289	6953	5416	6518	8214	9079	8899	8008	6810	5589	4494	3581	2846	2268	1817	1468	1195	982	815	681	574	488	418
92										109	2368	8059	9429	6727	6465	8284	9699	9796	8843	7404	5912	4594	3522	2693	2064	1594	1243	980	782	631	515	425	354	298	253
91									1	253	3738	9596	9242	7014	8051	10028	10678	9822	8161	6357	4767	3507	2564	1881	1391	1041	790	608	475	377	303	247	203	169	142
90									3	544	5491	10733	9014	7952	9915	11392	10933	9148	6995	5069	3569	2486	1733	1220	870	631	466	350	268	209	165	133	108	89	74
89									11	1082	7514	11385	9078	9488	11680	12068	10402	7913	5568	3754	2482	1637	1088	735	506	355	255	188	141	108	84	66	53	44	36
88									33	1985	9603	11647	9698	11361	12947	11892	9193	6357	4117	2582	1603	1001	634	411	273	186	130	93	68	51	39	31	24	20	16
87									90	3368	11512	11771	10951	13163	13403	10890	7546	4743	2826	1649	962	568	343	213	137	90	61	43	31	23	17	13	10	8	7
86									226	5284	13043	12089	12674	14446	12915	9264	5752	3286	1802	978	536	300	173	103	64	41	27	18	13	9	7	5	4	3	3
85								1	517	7681	14133	12865	14493	14860	11570	7320	4072	2115	1067	539	277	147	81	46	27	17	11	7	5	4	3	2	2	1	1
84								4	1085	10372	14896	14171	15927	14255	9629	5371	2678	1264	587	276	133	67	35	19	11	7	4	3	2	1	1	1	1		
83								12	2091	13072	15584	15816	16543	12723	7444	3661	1635	701	300	131	59	28	14	7	4	2	1	1	1						
82								38	3701	15487	16463	17393	16096	10555	5345	2317	927	362	142	58	25	11	5	3	1	1									
81								109	6029	17445	17667	18405	14606	8135	3564	1362	488	173	63	24	9	4	2	1											
80								282	9052	18971	19091	18449	12333	5824	2207	743	239	77	26	9	3	1	1												
79								667	12578	20267	20391	17358	9681	3872	1270	377	108	32	10	3	1														
78							1	1438	16265	21572	21106	15251	7060	2391	678	177	46	12	3	1															
77							6	2825	19741	22980	20833	12480	4783	1371	336	78	18	4	1																
76							20	5070	22756	24316	19399	9498	3010	730	155	32	7	1																	
75							64	8330	25261	25158	16927	6718	1759	361	66	12	2																		
74							185	12575	27359	25019	13786	4415	955	166	26	4	1																		
73							486	17532	29128	23574	10456	2695	481	71	10	1																			
72							1153	22741	30440	20834	7376	1528	225	28	3																				
71						1	2477	27702	30924	17158	4837	804	98	10	1																				
70						3	4823	32028	30123	13115	2947	393	40	4																					
69						13	8539	35480	27758	9284	1668	179	15	1																					
68						46	13795	37870	23938	6077	877	75	5																						
67						147	20432	38922	19173	3676	428	30	2																						
66						424	27919	38273	14191	2054	194	11	1																						
65						1095	35442	35651	9676	1059	82	4																							
64						2543	42090	31132	6066	505	32	1																							
63						5311	47014	25255	3493	222	12																								
62					2	10002	49512	18901	1846	90	4																								
61					9	17044	49091	12987	895	34	1																								
60					38	26380	45611	8166	398	12																									
59					137	37237	39447	4688	162	4																									
58					432	48130	31530	2454	61	1																									
57					1214	57154	23143	1171	21																										
56					3029	62478	15518	508	7																										
55					6727	62885	9468	201	2																										
54					13320	58182	5241	72																											
53				1	23561	49332	2627	24																											
52				8	37005	38180	1191	7																											
51				35	52960	26861	488	2																											
50				141	67512	17112	180																												
49				491	77349	9838	60																												
48				1499	79669	5090	18																												
47				4007	73743	2365	5																												
46				9384	61278	985	1																												
45				19267	45645	368																													
44				34698	30423	123																													
43			1	54838	18109	37																													
42			4	76089	9608	10																													
41			20	92708	4536	2																													
40			101	99199	1903																														
39			422	93213	708																														
38			1506	76902	233																														
37			4574	55687	68																														
36			11830	35378	18																														
35			26053	19708	4																														
34			48859	9621	1																														
33			78035	4114																															
32			106141	1540																															
31			122953	504																															
30			121298	144																															
29		2	101911	36																															
28		23	72918	9																															
27		184	44429	2																															
26		1125	23051																																
25		5222	10183																																
24		18426	3830																																
23		49423	1226																																
22		100792	334																																
21		156238	78																																
20		184138	15																																
19		164986	3																																
18		112384																																	
17		58199																																	
16		22912																																	
15	1	6858																																	
14	18	1560																																	
13	306	270																																	
12	3239	35																																	
11	20808	4																																	
10	81232																																		
9	192696																																		
8	277760																																		
7	243288																																		
6	129488																																		
5	41878																																		
4	8230																																		
3	983																																		
2	71																																		
1	3																																		

Table A16. Empirical mean lengths at age and sample size from Turner et. al. (1983).

	Age	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35
female	empirical mean length	-	-	38	47	52	58	64	65	66	68		90	-	-	84	77	-	84	82	-	-	-	-	-	-	-	-	-	92	89	91	89	95	-	88
	n	-	-	14	47	61	40	65	52	11	1		1	-	-	1	1	-	1	1	-	-	-	-	-	-	-	-	-	1	1	1	3	1	-	2
male	empirical mean length	-	-	40	50	53	60	71	74	79	86	89	93	-	-	99	102	104	-	96	109	-	108	-	-	108	96	-	-	-	-	-	-	-	-	-
	n	-	-	4	51	55	17	44	41	23	5	1	1	-	-	5	1	1	-	2	2	-	1	-	-	1	1	-	-	-	-	-	-	-	-	-

Table A17. Oldest fish aged from Turner's PHD dissertation (1986) and Vidal's MS (2009).

Dissertation 1986 S Turner		Number of females younger than 31	Number of females older than 31
oldest male: 39	1978	234	7
oldest female: 46	1979	87	4
	1980	177	3
	1982	194	21

	Number of males younger than 31	Number of males older than 31
1978	216	0
1979	148	1
1980	91	0
1982	187	1

T. Vidal (2008)
oldest male: 23
oldest female: 21

Table A18. Six SCALE sensitivity runs. Natural mortality was assumed to be 0.1 in combined sex runs and for females in the separate sex runs. The assumed natural mortality rate for males was 0.15 in the separate sex runs. TV = T. Vidal, ST = S. Turner, vb = von Bertalanffy, sel bl = selectivity blocks, var = variation, resid = residuals, par = parameters.

Run	1 (Base run)			2			3			4			5			6		
Description	(TV vb, 2 sex, 2 Sel bl)			(TV vb, 1 sex, 2 Sel bl)			(TV vb, 2 sex, 1 Sel bl)			(ST vb, 2 sex, 2 Sel bl)			(Base + high mean len@age var)			(Base + rec index)		
	weight	qs	resid or par	weight	qs	resid or par	weight	qs	resid or par	weight	qs	resid or par	weight	qs	resid or par	weight	qs	resid or par
Total Objective function			68.23			70.96			76.70			68.34			69.77			63.27
total catch	4		0.23	4		0.23	4		0.23	4		0.25	4		0.24	4		0.15
catch len freq 1+	400		45.31	400		48.21	400		52.36	400		45.22	400		46.84	400		44.05
Variation in recruit penalty (Vrec	0.05		7.79	0.05		8.75	0.05		5.92	0.05		8.41	0.05		8.29	0.05		12.58
Age 5	1	3.0E-06	6.01	1	3.3E-06	5.72	1	3.1E-06	5.92	1	3.0E-06	6.67	1	3.0E-06	6.36	1	4.1E-06	2.31
Turner 47+ (1973-1982)	2	4.1E-07	0.21	2	4.5E-07	0.24	2	3.8E-07	0.26	2	3.5E-07	0.18	2	4.2E-07	0.21	2	4.1E-07	0.31
Weighout 37+ (1979-1993)	2	8.9E-07	0.22	2	9.7E-07	0.22	2	9.3E-07	0.23	2	8.2E-07	0.24	2	9.2E-07	0.22	2	8.8E-07	0.28
VTR 37+ (1995-2008)	4	1.7E-06	0.79	4	1.8E-06	0.72	4	1.7E-06	0.79	4	1.6E-06	0.68	4	1.7E-06	0.72	4	1.8E-06	0.88
survey/catch len freq 65+	100		11.56	100		11.83	100		13.03	100		11.46	100		11.44	100		11.00
Fstart			0.20			0.26			0.13			0.10			0.20			0.18
Recruitment year 1 (1971, 000s)			783			624			946			787			765			721
Selectivity Alpha (L50) 71-81			53.97			53.74			41.80			53.70			53.94			54.27
Selectivity Beta (slope) 71-81			0.35			0.35			0.69			0.35			0.36			0.33
Selectivity Alpha (L50) 82-08			41.38			41.49			-			41.35			41.11			41.40
Selectivity Beta (slope) 82-08			0.81			0.80			-			0.58			0.75			0.81
2008 F			0.19			0.20			0.20			0.18			0.19			0.21
2008 Biomass (000s mt)			4950			4518			4784			5200			4867			4422

Table A19. Biological reference point estimates from the 2000 SSC committee review, 2005 SARC 41 assessment, and the 2009 BASE run.

	SSC 2000 1999	SARC 41 2004	SARC 48 2008
BMSY			
Point	8,448	9,384	11,400
Boot mean	-	9,764	10,336
Boot sd	-	5,152	2,089
Boot median	-	9,193	10,135
Boot 25%ile	-	8,379	8,974
Boot 75%ile	-	10,263	11,436
Boot bias	-	4%	-9%
FMSY			
Point	0.22	0.21	0.16
Boot mean	-	0.24	0.2
Boot sd	-	0.21	0.06
Boot median	-	0.22	0.19
Boot 25%ile	-	0.19	0.16
Boot 75%ile	-	0.25	0.23
Boot bias	-	15%	21%
MSY	1,858	1,988	1,868
r	0.45	0.42	0.33
Turner Q	0.009	0.010	0.009
Weighout	0.222	0.225	0.175
VTR Q	-	0.392	0.260

Table A20. Stock status and biological reference points using F40% and Fmax from both the SCALE model and the age based YPR model. A female only BRP can not be done with run 1 using the age based YPR model.

SCALE run	1 Base		2			
Description	(Vidal growth, 2 sex, 2 Sel block)		(Vidal growth, 1 sex, 2 Sel block)			
YPR model	SCALE	SCALE	AGE based	AGE based	SCALE	SCALE
F_{MSY} proxy	$F_{40\%}$	Fmax	$F_{40\%}$	Fmax	$F_{40\%}$	Fmax
F_{MSY}	0.085	0.128	0.079	0.121	0.082	0.121
YPR	1.37	1.45	1.83	1.92	1.85	1.92
SSB per Recruit	9.07	6.82	23.20	15.83	24.22	17.30
Initial Recruits (000s)	783	783	624	624	624	624
MSY (mt)	1,072	1,137	1,142	1,200	1,153	1,200
SSB_{MSY} (mt)	7,100	5,335	14,473	9,878	15,108	10,794
SSB_{09} (mt)	2,520	2,520	4,399	4,399	4,399	4,399
F_{08}	0.188	0.188	0.205	0.205	0.205	0.205
SSB_{09}/SSB_{MSY}	35%	47%	30%	45%	29%	41%
F_{08}/F_{MSY}	221%	147%	260%	170%	250%	169%

Table A21. Converted input (selectivity, maturity from Vidal, population and catch mean weights) to the age based YPR model from the SCALE run 2. Terminal year + 1 stock size at age is also shown.

age	Stock Size on 1 Jan 2009	Selectivity	Proportion Mature	Mean Weights Spawning Stock	Mean Weights Catch
1	623,830	0.000	0.000	0.002	0.003
2	564,465	0.000	0.000	0.053	0.085
3	510,749	0.001	0.005	0.253	0.417
4	442,060	0.376	0.129	0.662	0.783
5	425,544	0.978	0.672	1.295	1.303
6	421,569	1.000	0.956	2.130	2.130
7	169,889	1.000	0.995	3.131	3.131
8	102,072	1.000	0.999	4.251	4.251
9	100,136	1.000	1.000	5.446	5.446
10	138,090	1.000	1.000	6.675	6.675
11	71,028	1.000	1.000	7.904	7.904
12	6,162	1.000	1.000	9.100	9.100
13	2,870	1.000	1.000	10.249	10.249
14	1,144	1.000	1.000	11.336	11.336
15	267	1.000	1.000	12.354	12.354
16	190	1.000	1.000	13.296	13.296
17	43	1.000	1.000	14.161	14.161
18	7	1.000	1.000	14.951	14.951
19	2	1.000	1.000	15.668	15.668
20	1	1.000	1.000	16.314	16.314
21	1	1.000	1.000	16.896	16.896
22	0	1.000	1.000	17.417	17.417
23	0	1.000	1.000	17.881	17.881
24	0	1.000	1.000	18.295	18.295
25	0	1.000	1.000	18.663	18.663
26	0	1.000	1.000	18.988	18.988
27	0	1.000	1.000	19.277	19.277
28	0	1.000	1.000	19.532	19.532
29	0	1.000	1.000	19.757	19.757
30	0	1.000	1.000	19.955	19.955
31	0	1.000	1.000	20.130	20.130
32	0	1.000	1.000	20.284	20.284
33	0	1.000	1.000	20.418	20.418
34	0	1.000	1.000	20.537	20.537
35	0	1.000	1.000	20.642	20.642

Table A22. Projection results using the standard ASPIC projection model (conditioned on yield or F). Catch and biomass in metric tons (mt).

A) C = 2008 TAC = 905 mt

Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	905	0.07	0.06	0.08	0%	13,030	10,480	14,210	35%	<1%
2010	905	0.06	0.06	0.08	0%	13,930	11,420	14,720	25%	0%
2011	905	0.06	0.06	0.07	0%	14,760	12,200	15,260	15%	0%

B) C = MSY = 1,868 mt

Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	1,868	0.14	0.13	0.18	36%	13,030	10,480	14,210	35%	<1%
2010	1,868	0.14	0.14	0.18	38%	12,990	10,480	13,810	37%	<1%
2011	1,868	0.14	0.14	0.18	40%	12,950	10,470	13,590	39%	<1%

C) F = FMSY = 0.16

Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	2,112	0.16	0.15	0.21	50%	13,030	10,480	14,210	35%	<1%
2010	2,071	0.16	0.15	0.21	50%	12,750	10,230	13,660	39%	<1%
2011	2,038	0.16	0.15	0.21	50%	12,530	9,995	13,290	45%	<1%

Table A23. Projection results incorporating assumptions about future values of the VTR CPUE index of abundance. Catch in metric tons and biomass in 000s metric tons. Scenario F was CPUE was rounded to one decimal place.

A) CPUE = 1995-2008		FMSY = 0.165		BMSY = 9,853 mt			MSY = 1,627 mt			
Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	905	0.070	0.065	0.079	0%	12,836	11,259	13,844	16%	<1%
2010	905	0.069	0.064	0.077	0%	13,082	11,595	14,134	13%	<1%
2011	905	0.067	0.062	0.075	0%	13,322	11,896	14,349	10%	0%

B) CPUE = 2001-2008		FMSY = 0.168		BMSY = 9,759 mt			MSY = 1,643 mt			
Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	905	0.071	0.066	0.082	0%	12,496	10,768	13,502	17%	<1%
2010	905	0.069	0.065	0.077	0%	12,874	11,412	13,843	13%	<1%
2011	905	0.068	0.063	0.075	0%	13,210	11,913	14,142	9%	0%

C) CPUE = +25%		FMSY = 0.158		BMSY = 10,070 mt			MSY = 1,590 mt			
Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	905	0.071	0.065	0.082	0%	12,598	10,751	13,820	20%	0%
2010	905	0.069	0.064	0.078	0%	12,936	11,348	14,087	15%	0%
2011	905	0.067	0.063	0.075	0%	13,255	11,780	14,342	12%	0%

D) CPUE = -25%		FMSY = 0.060		BMSY = 15,000 mt			MSY = 897 mt			
Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	905	0.139	0.084	0.213	84%	6,620	4,357	10,981	84%	57%
2010	905	0.143	0.085	0.223	85%	6,440	4,157	10,741	84%	59%
2011	905	0.148	0.087	0.238	86%	6,211	3,924	10,523	85%	60%

E) CPUE = 2008		FMSY = 0.197		BMSY = 8,989 mt			MSY = 1,774 mt			
Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	905	0.069	0.064	0.075	0%	12,980	12,022	14,038	<1%	6%
2010	905	0.068	0.063	0.074	0%	13,081	12,074	14,233	<1%	0%
2011	905	0.068	0.063	0.074	0%	13,174	12,124	14,398	<1%	0%

F) CPUE = 2008 round		FMSY = 0.104		BMSY = 12,060 mt			MSY = 1,254 mt			
Year	C (mt)	F	F25%ile	F75%ile	P > FMSY	B (mt)	B25%ile	B75%ile	P < BMSY	P < 1/2 BMSY
2009	905	0.088	0.066	0.130	38%	10,125	6,789	13,436	64%	18%
2010	905	0.084	0.065	0.125	36%	10,505	7,115	13,840	63%	15%
2011	905	0.083	0.063	0.119	34%	10,844	7,454	14,156	61%	12%

Figures

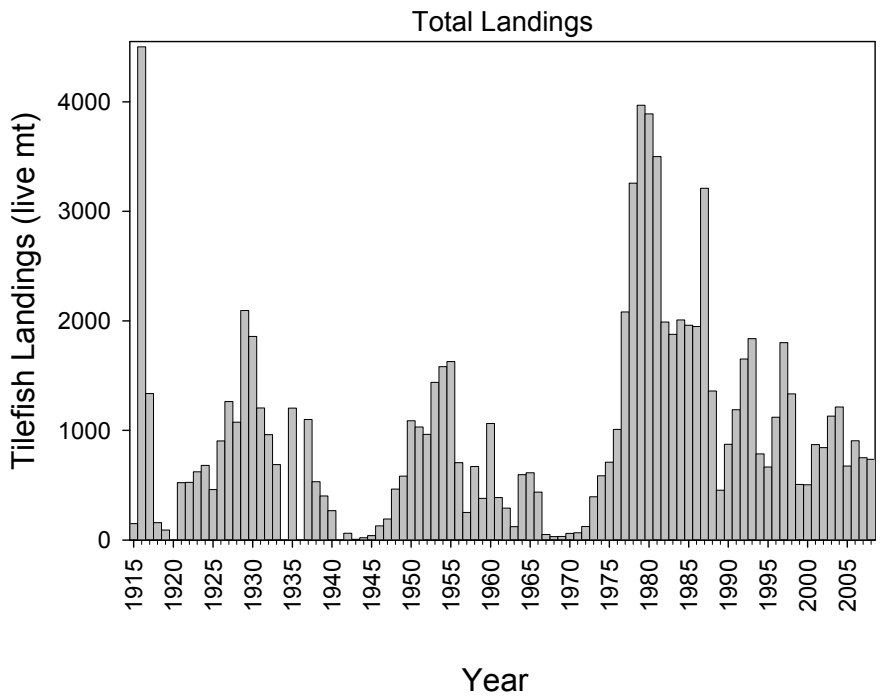


Figure A1. Landings of tilefish in metric tons from 1915-2004. Landings in 1915-1972 are from Freeman and Turner (1977), 1973-1989 are from the general canvas data, 1990-1993 are from the weighout system, 1994-2003 are from the dealer reported data, and 2004-2008 is from dealer electronic reporting.

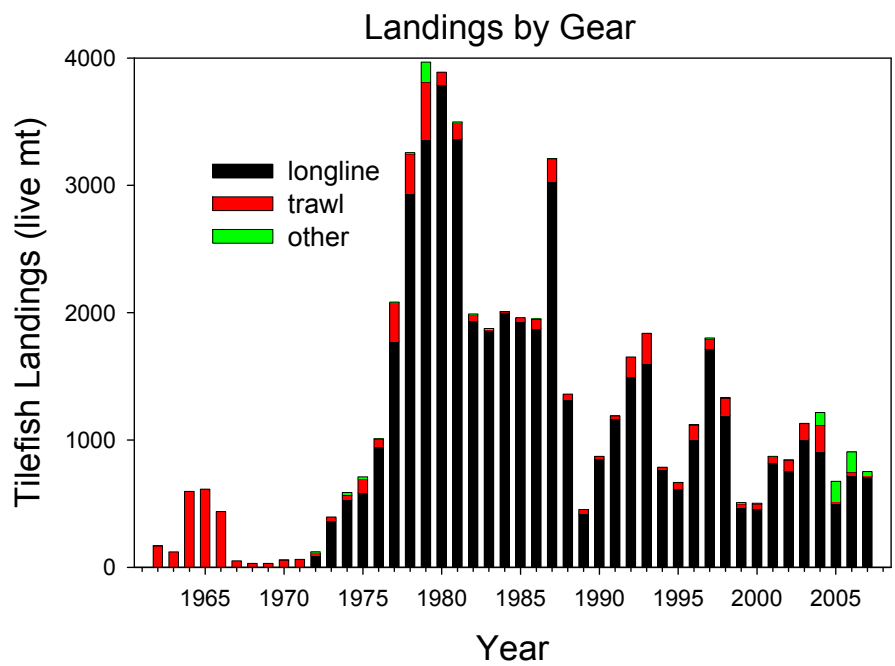


Figure A2. Landings of tilefish (mt, live) by gear. Landing before 1990 are from the general canvas data.

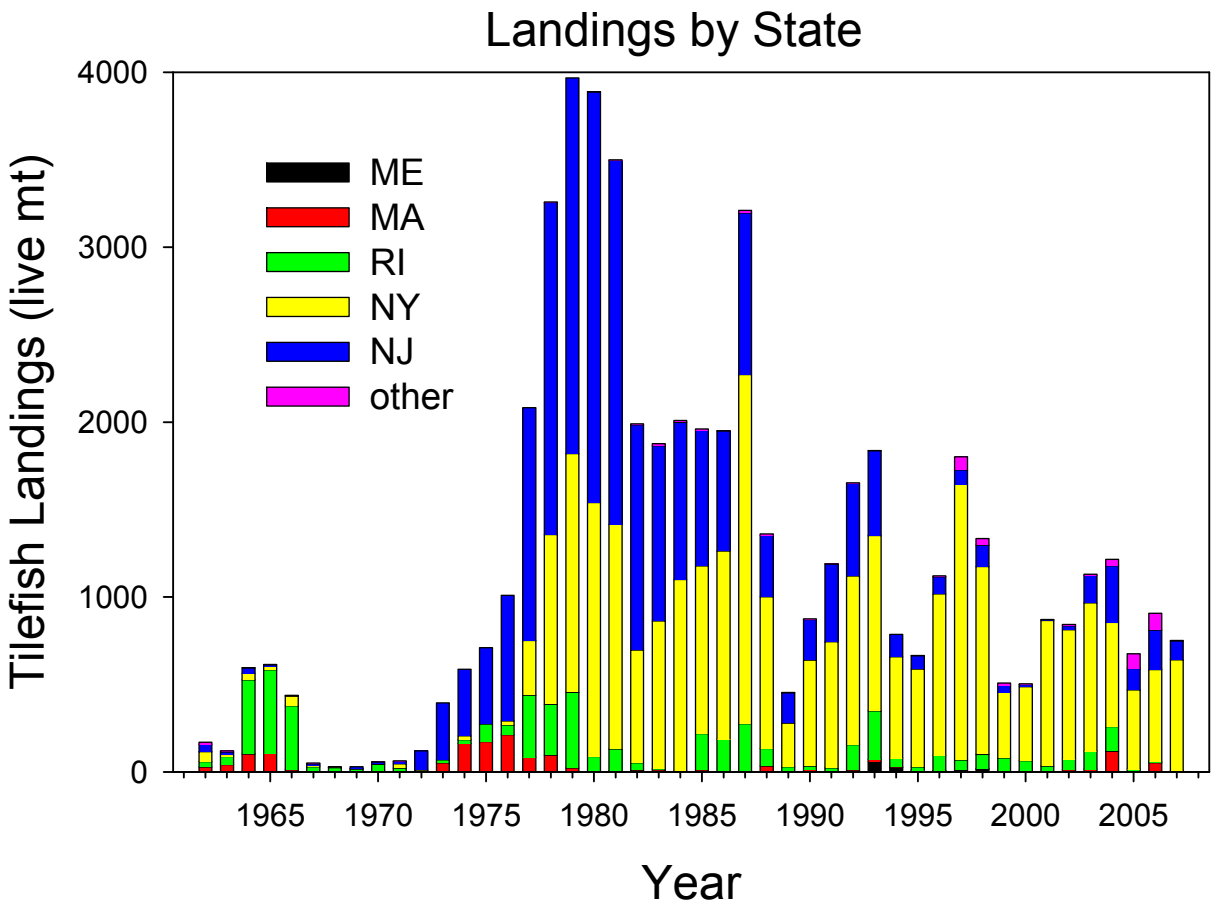


Figure A3. Landings of tilefish (mt, live) by State. Landings before 1990 are from the general canvas data.

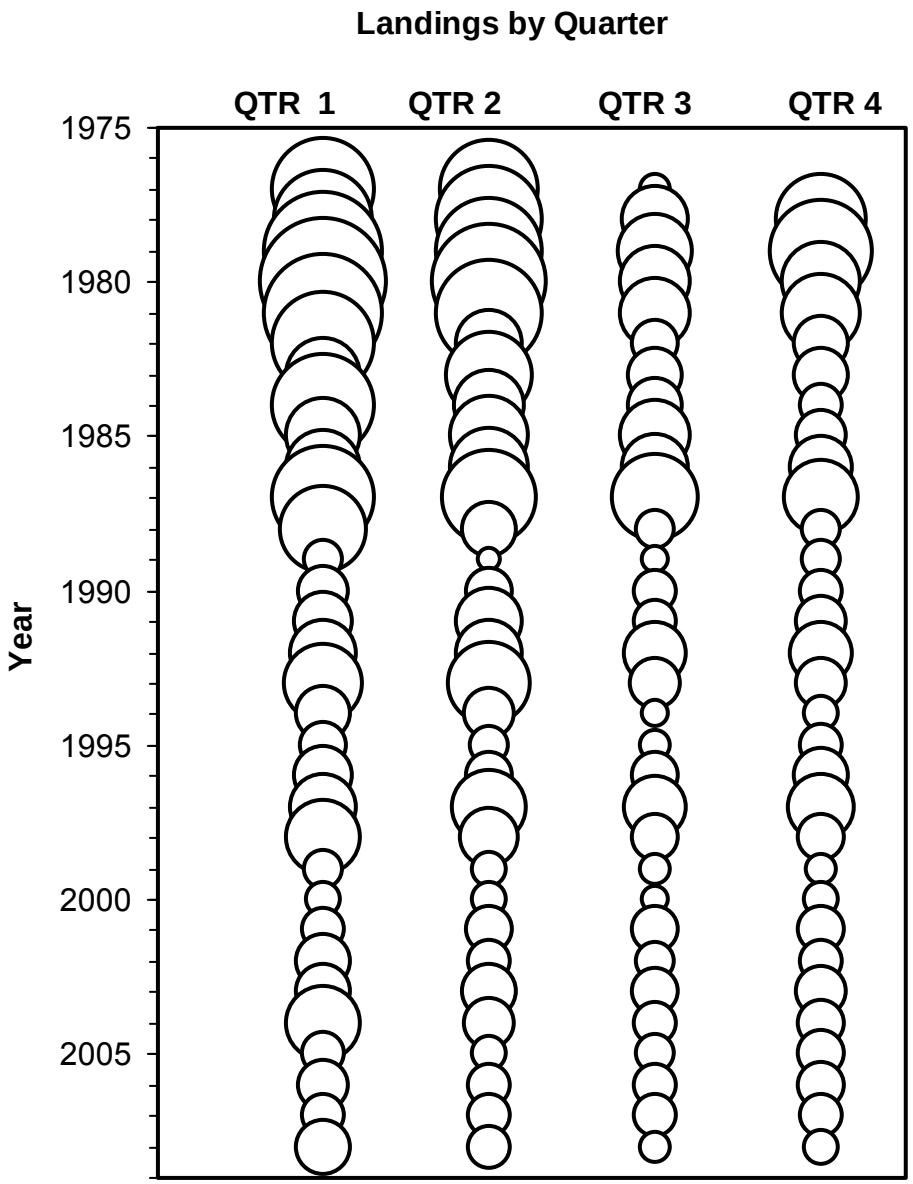


Figure A4. Bubble plot of Golden tilefish landings by quarter.

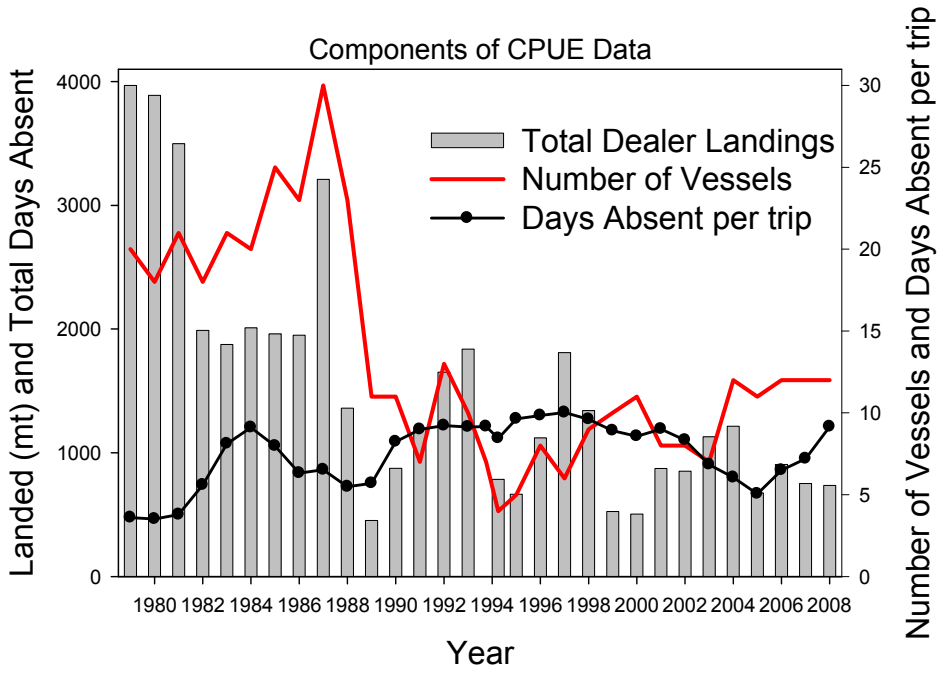


Figure A5. Number of vessels and length of trip (days absent per trip) for trips targeting tilefish (= or >75% tilefish) from 1979-2008. Total Dealer landings are also shown.

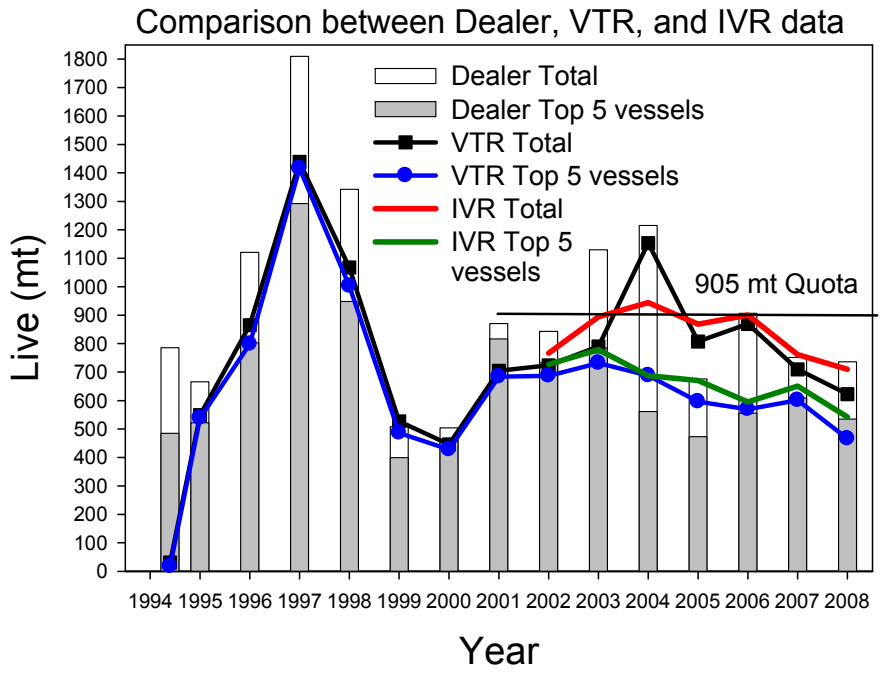


Figure A6. Comparison of dealer, VTR, and IVR total landings in live metric tons. Total landings limited to the top five dominant tilefish vessels are also shown.

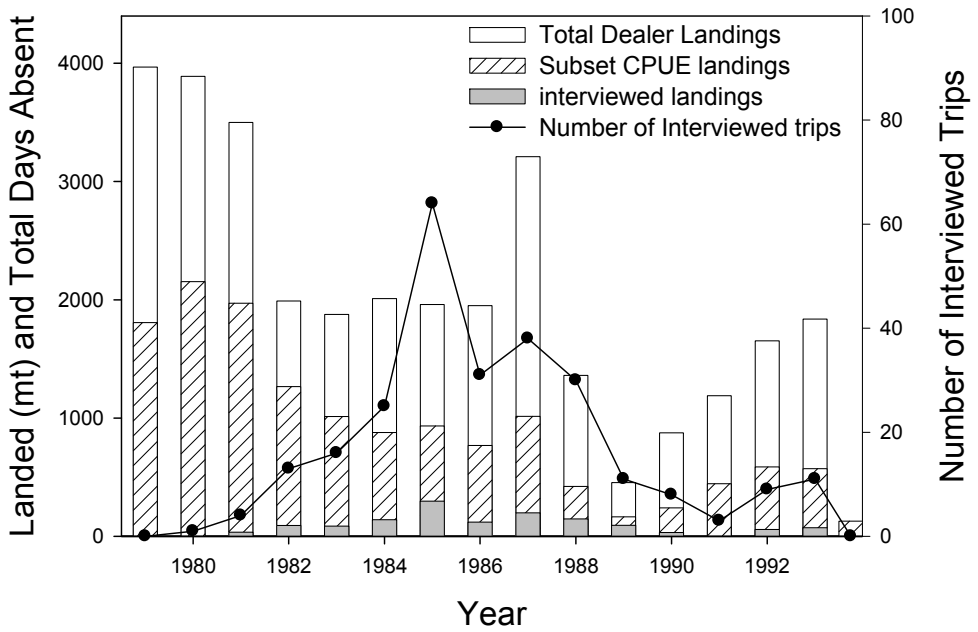


Figure A7. Number of interviewed trips and interviewed landings for trips targeting tilefish (= or >75% tilefish) for the Weighout data from 1979-1994. Total Weighout landings and the subset landings used in CPUE estimate are also shown.

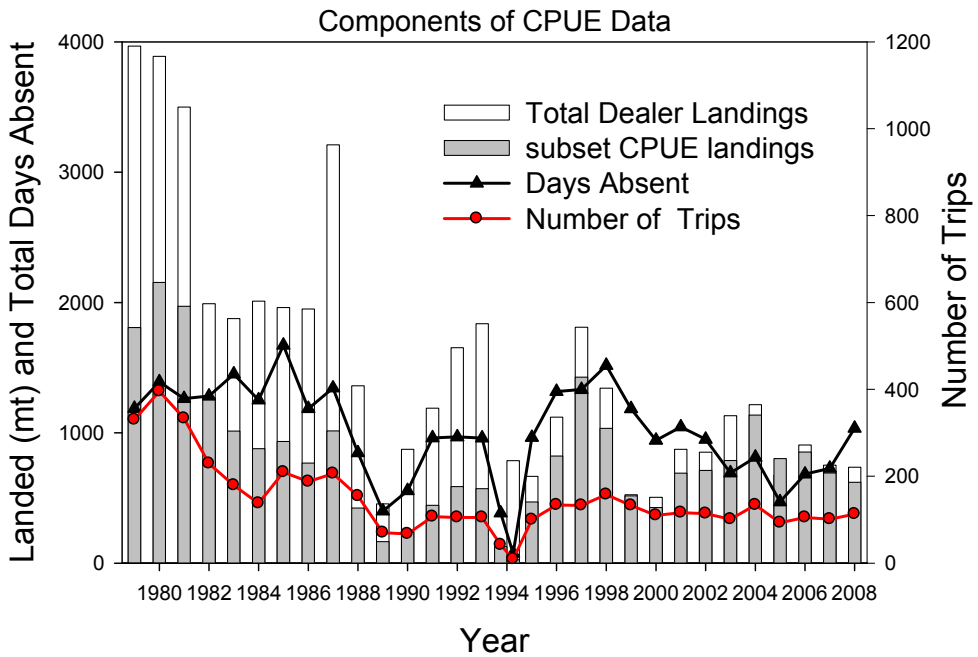


Figure A8. Total number of trips and days absent for trips targeting tilefish (= or >75% tilefish) from 1979-2008. Total Dealer and CPUE subset landings are also shown

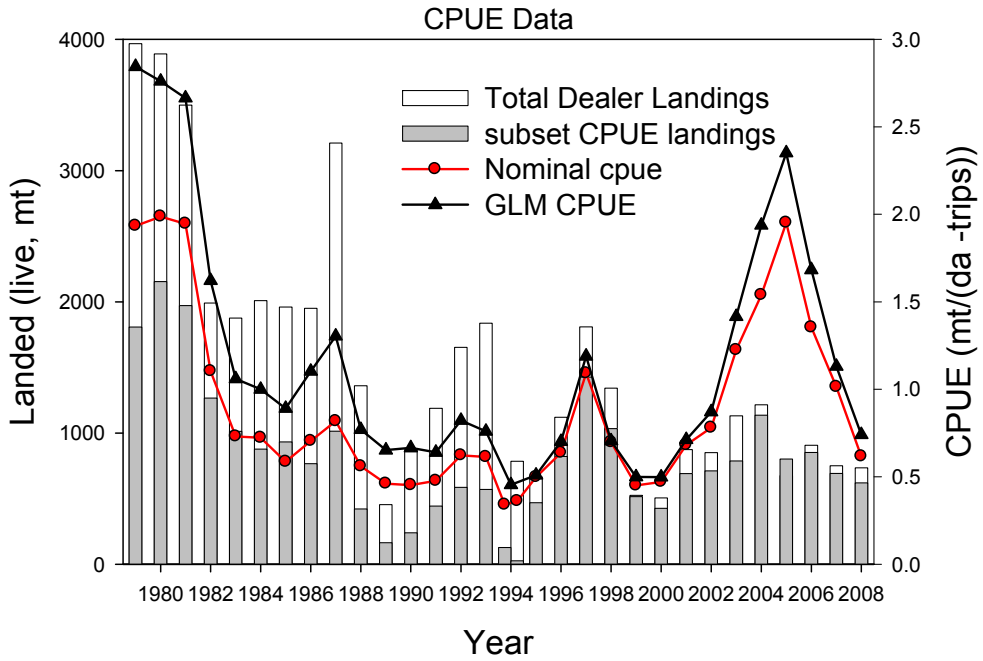


Figure A9. Nominal CPUE (1994 split by Weighout and VTR series) and vessel standard CPUE (GLM) for trips targeting tilefish (= or >75% tilefish) from 1979-2008. Total Dealer and CPUE subset landings are also shown.

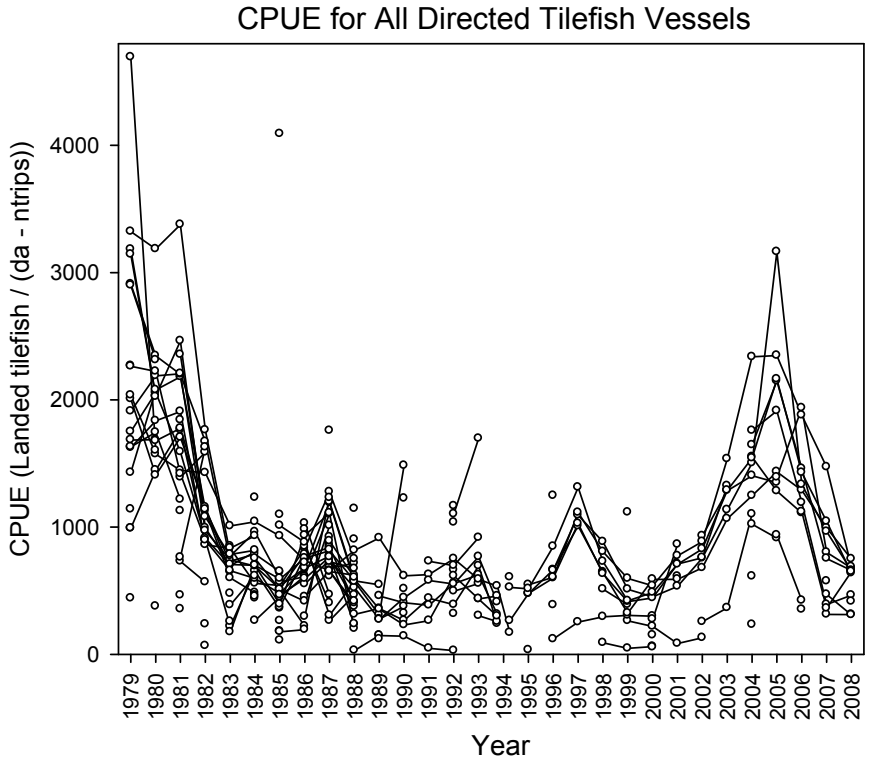


Figure A10. All individual tilefish vessel CPUE data for trips targeting tilefish (= or >75% tilefish) from 1979-200

Figure A11. Depiction of individual vessels (rows) targeting tilefish over the weighout and VTR series. Year 1994 is split by the two series. Below the horizontal line are vessels which are predominantly found in the VTR series.

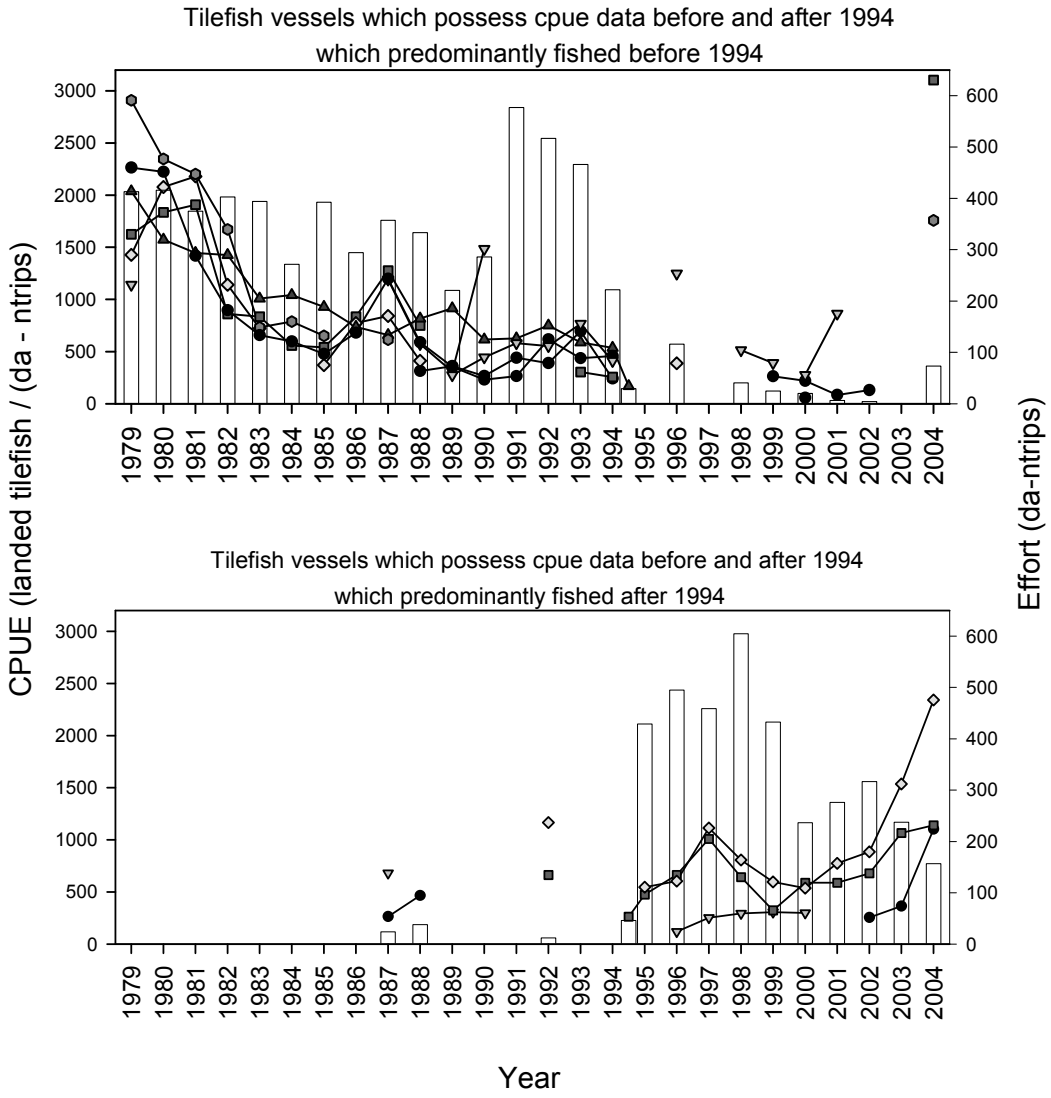


Figure A12. Individual tilefish vessel CPUE and effort data (Bars) for trips targeting tilefish (= or >75% tilefish) from 1979-2004 which are found in both the weighout and VTR series. Top graph are vessels found predominantly in the weighout series. Bottom graph are vessels found predominantly in the VTR series.

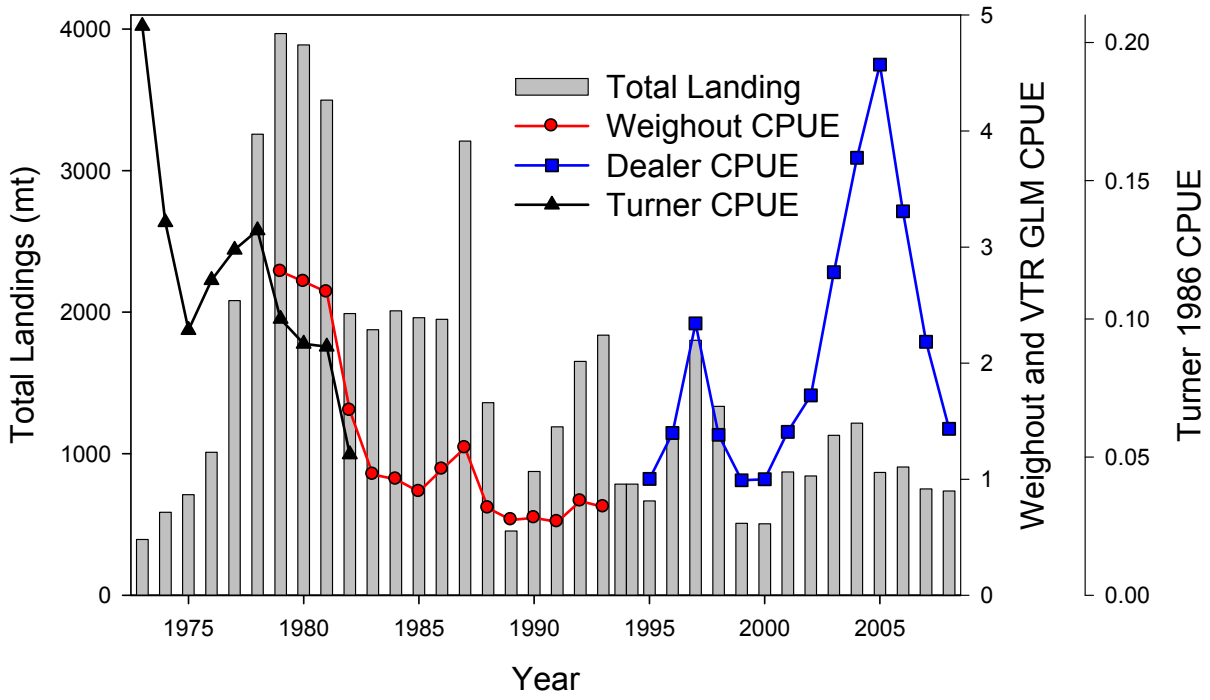


Figure A13. GLM CPUE for the Weighout and VTR data split into two series. Four years of overlap betweenTurner's and the Weighout CPUE series can be seen. Assumed total landings are also shown. Landing in 2005 was taken form the IVR system.

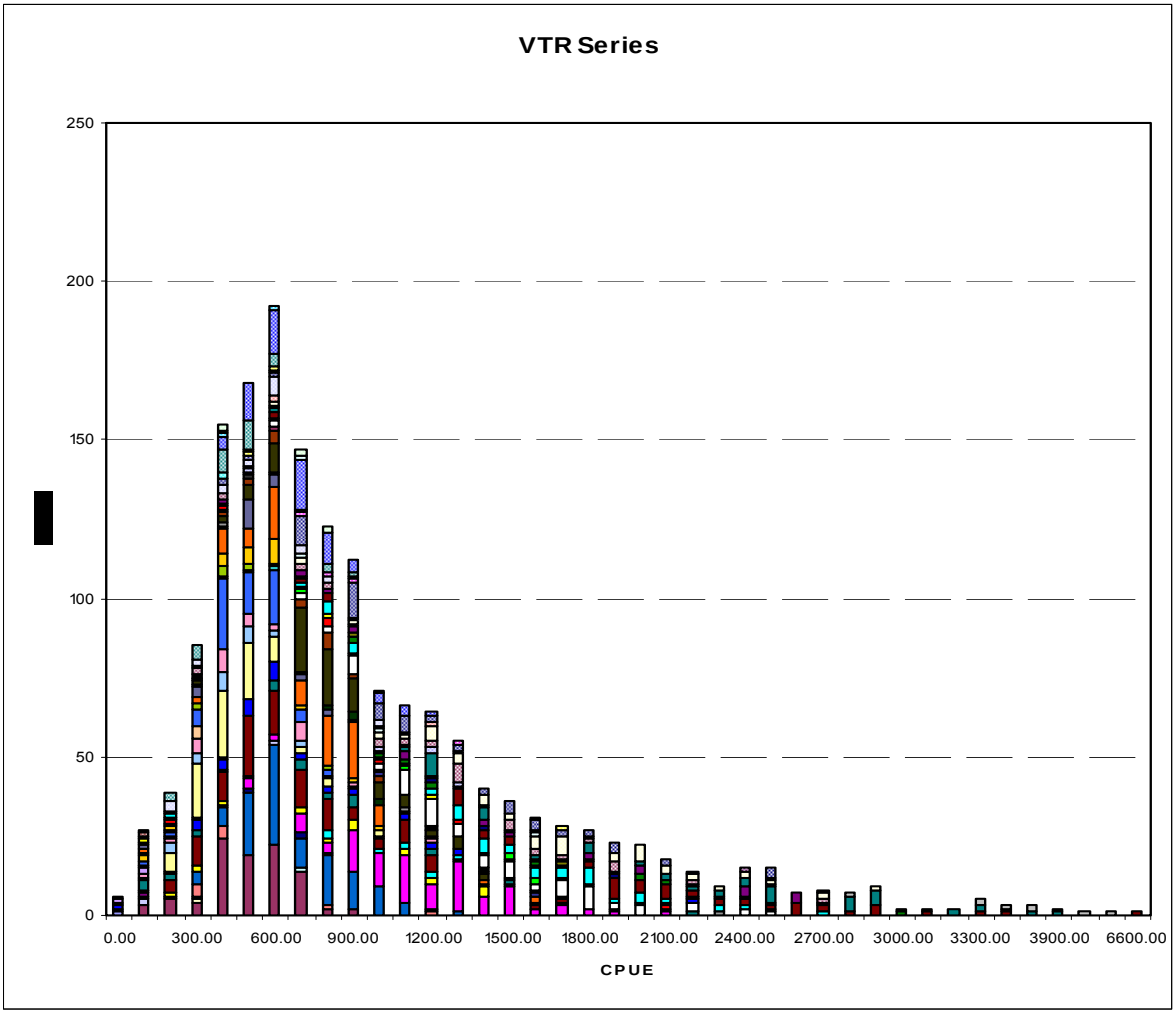


Figure A14. Frequency distribution of the nominal VTR CPUE.

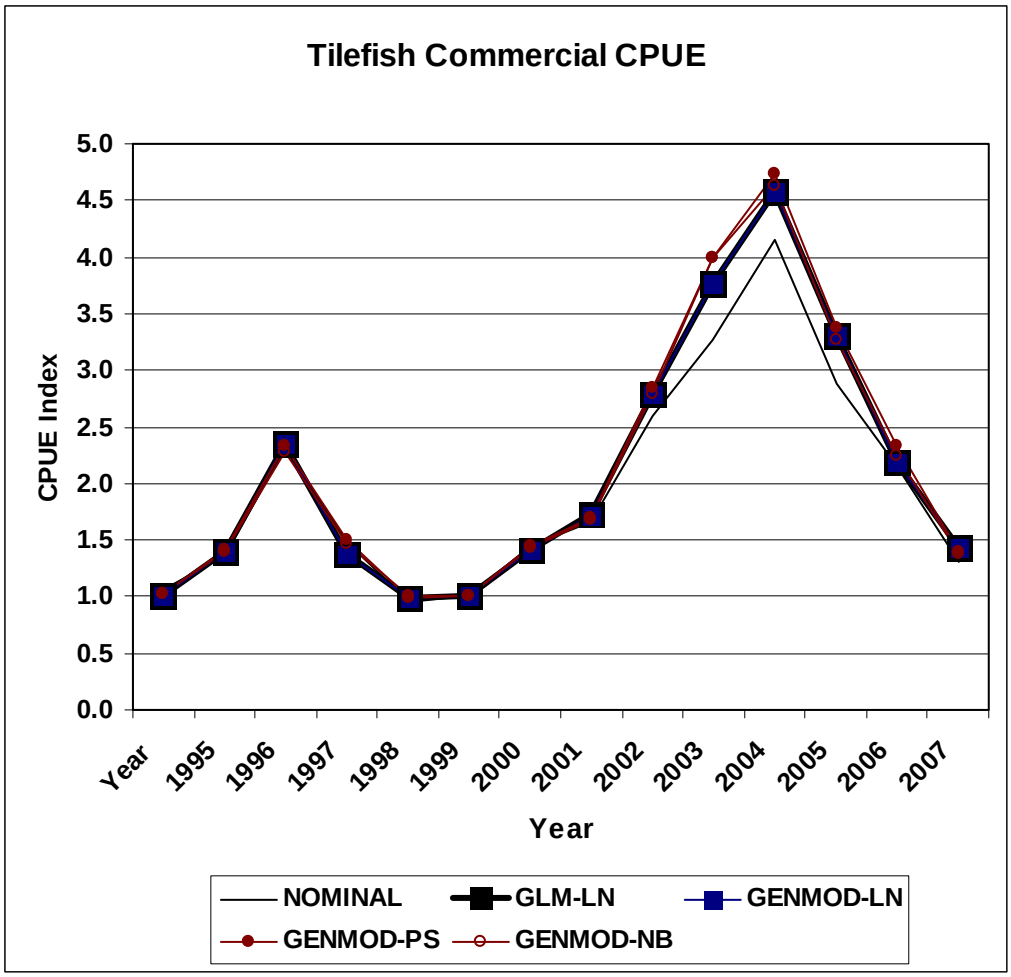


Figure A15. Effect of the assumed error distribution on the vessel standardized GLM CPUE indices.

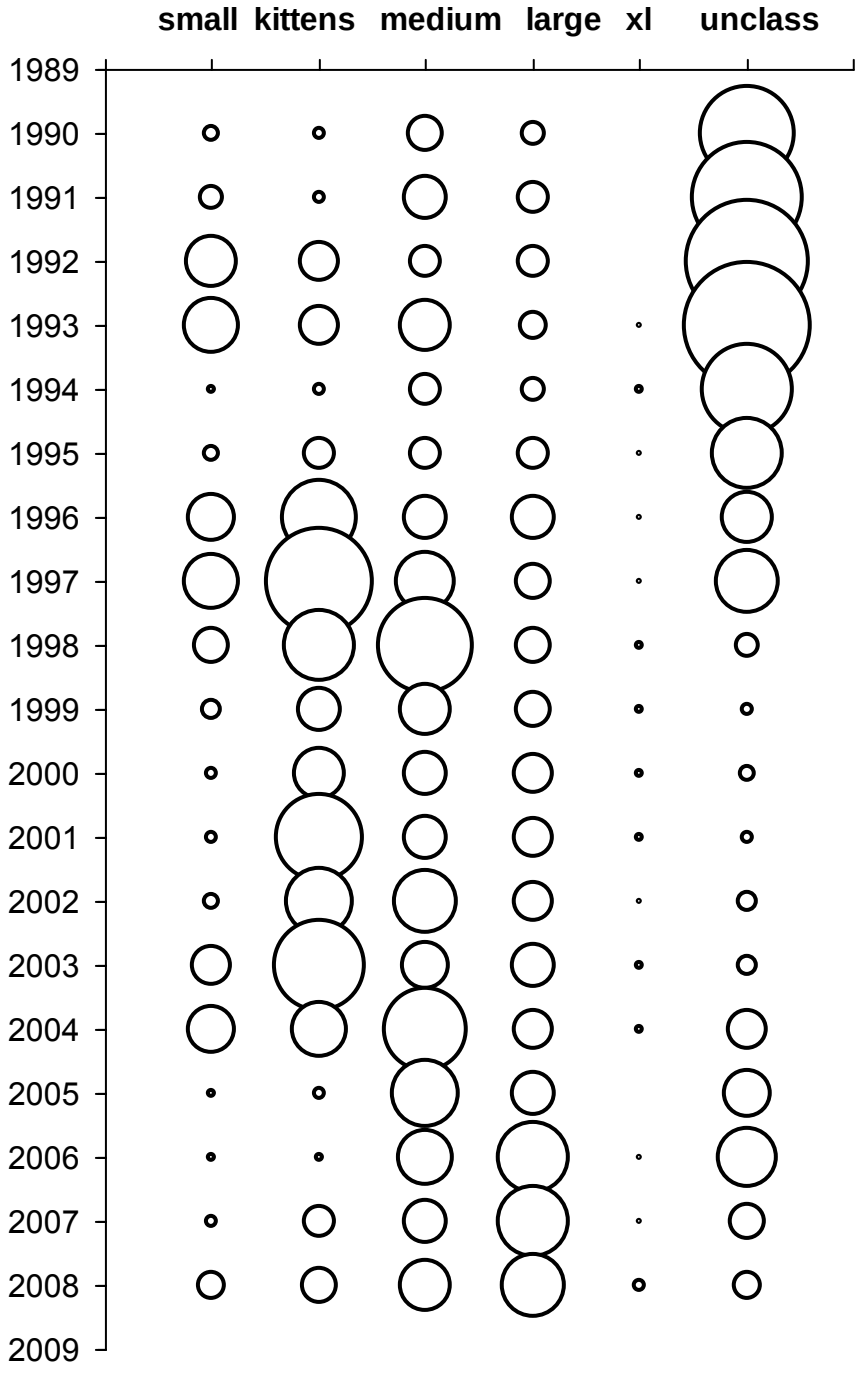


Figure A16. Bubble plot of Golden tilefish landings by market category.

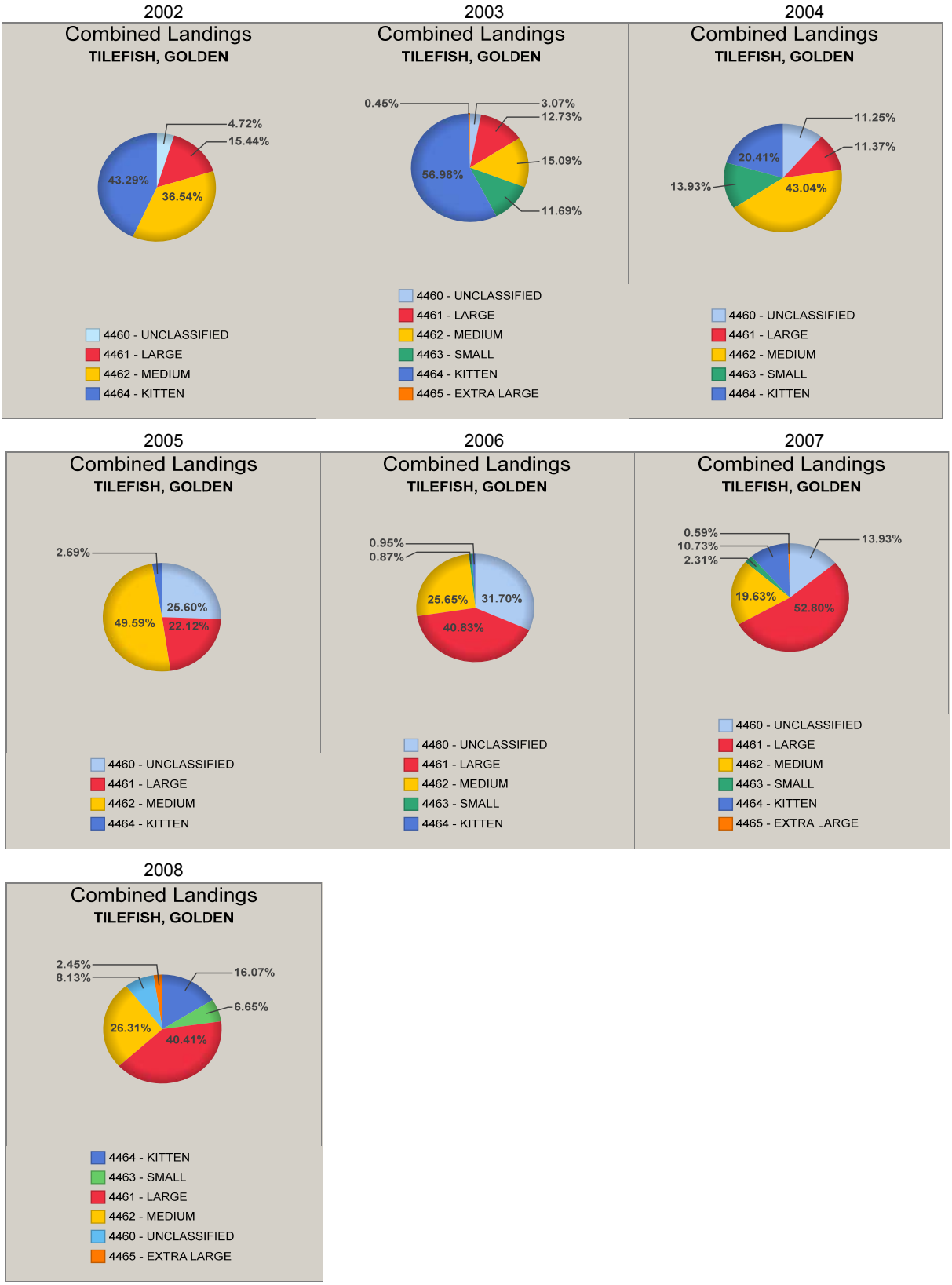


Figure A17. Proportion of landings by market category from 2002-2008.

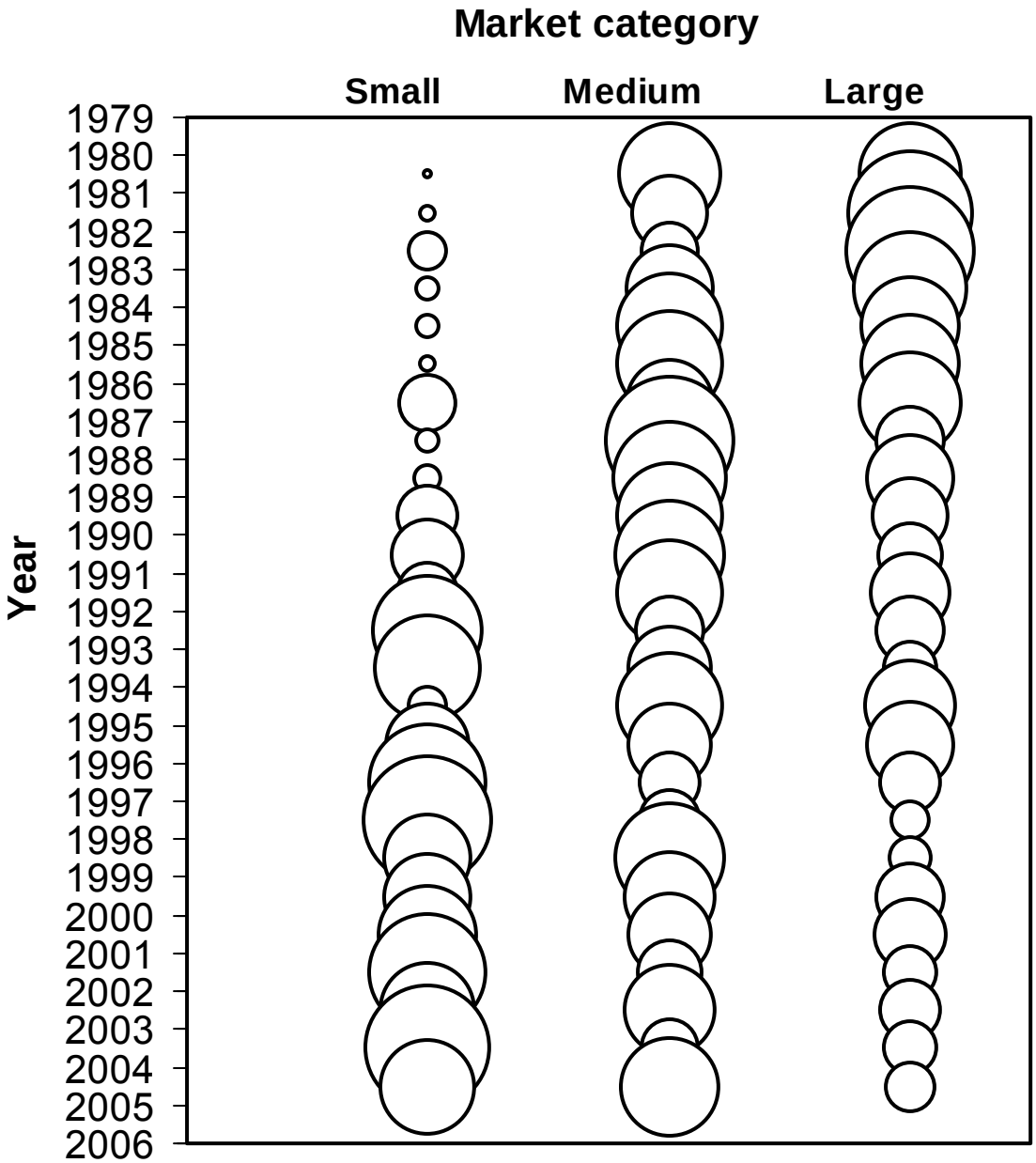


Figure A18. Bubble plot of percent Golden tilefish longline landings by market category. Data from 1980 to 1990 comes from New York tilefish fishermen. Data form 1991-2003 was taken from the dealer data. Data form 2004 are from dealer electronic reporting. Unclassified landings were redistributed according to the other market categories.

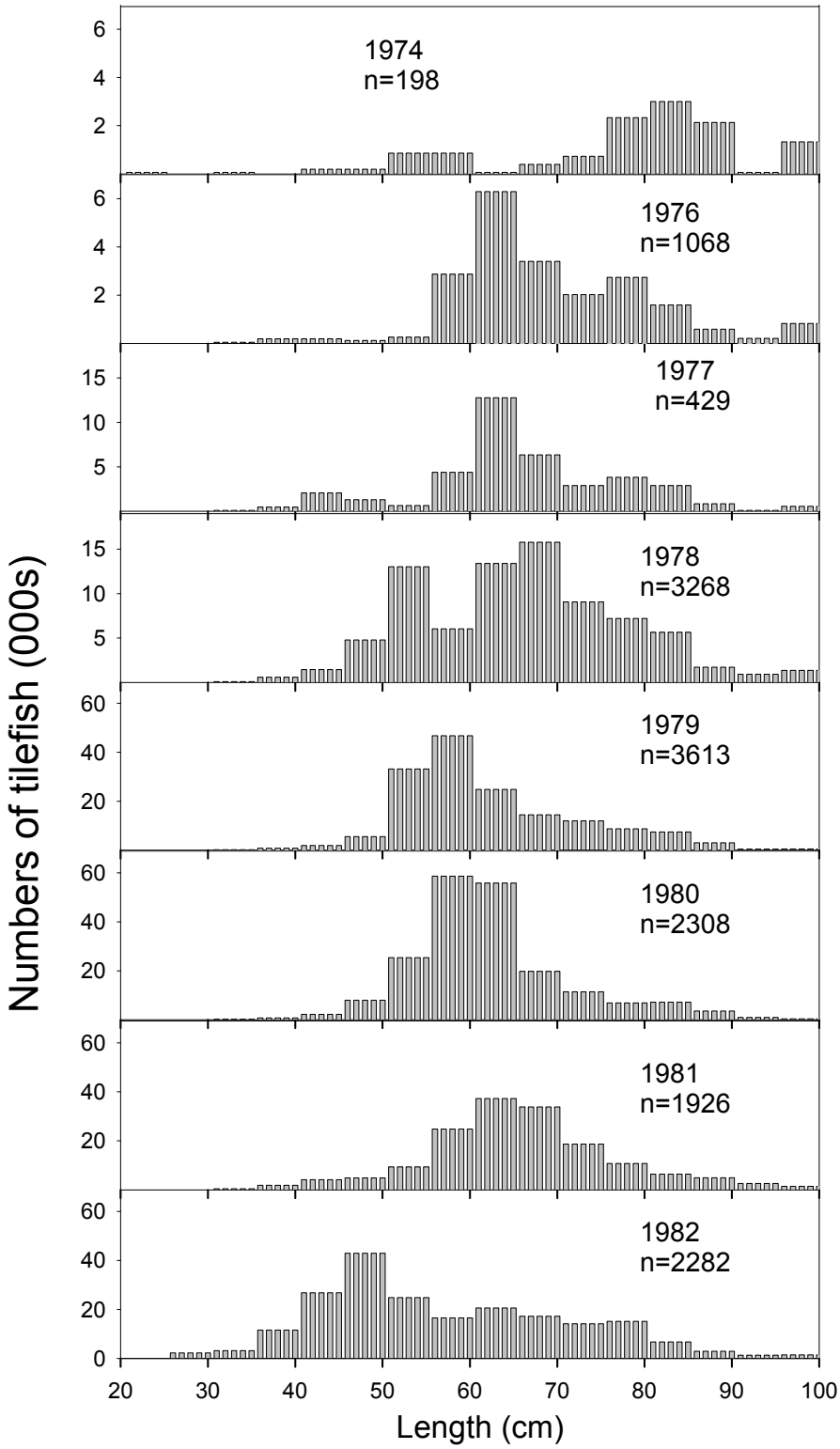


Figure A19. Expanded length frequency distributions using Turner (1986) length samples by 5 cm intervals. Hudson Canyon and Southern New England samples were combined.

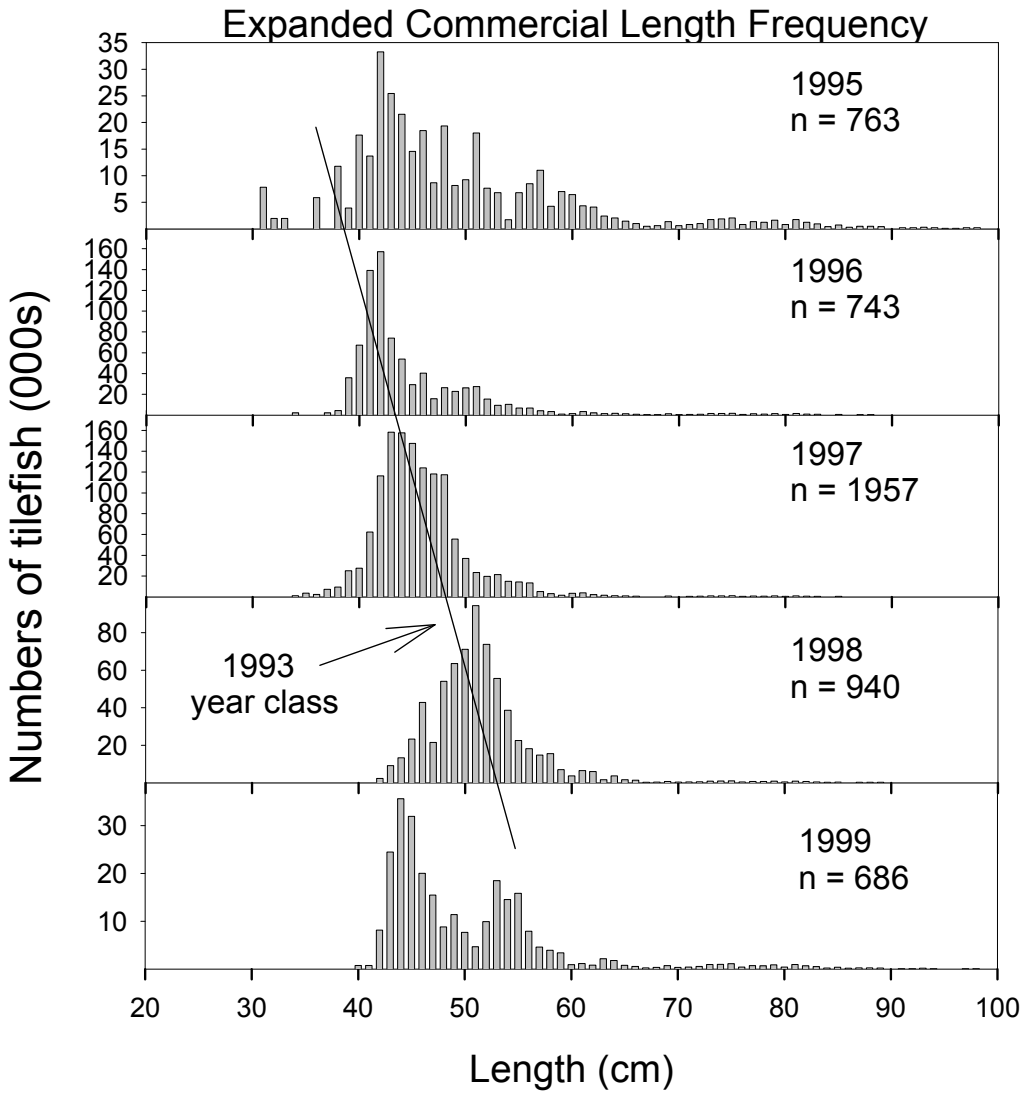


Figure A20. Expanded length frequency distributions by year. Large market category length used from 1995 to 1999 were taken from years 1996, 1998, and 1998. Smalls and kittens were combined and large and extra large were also combined.

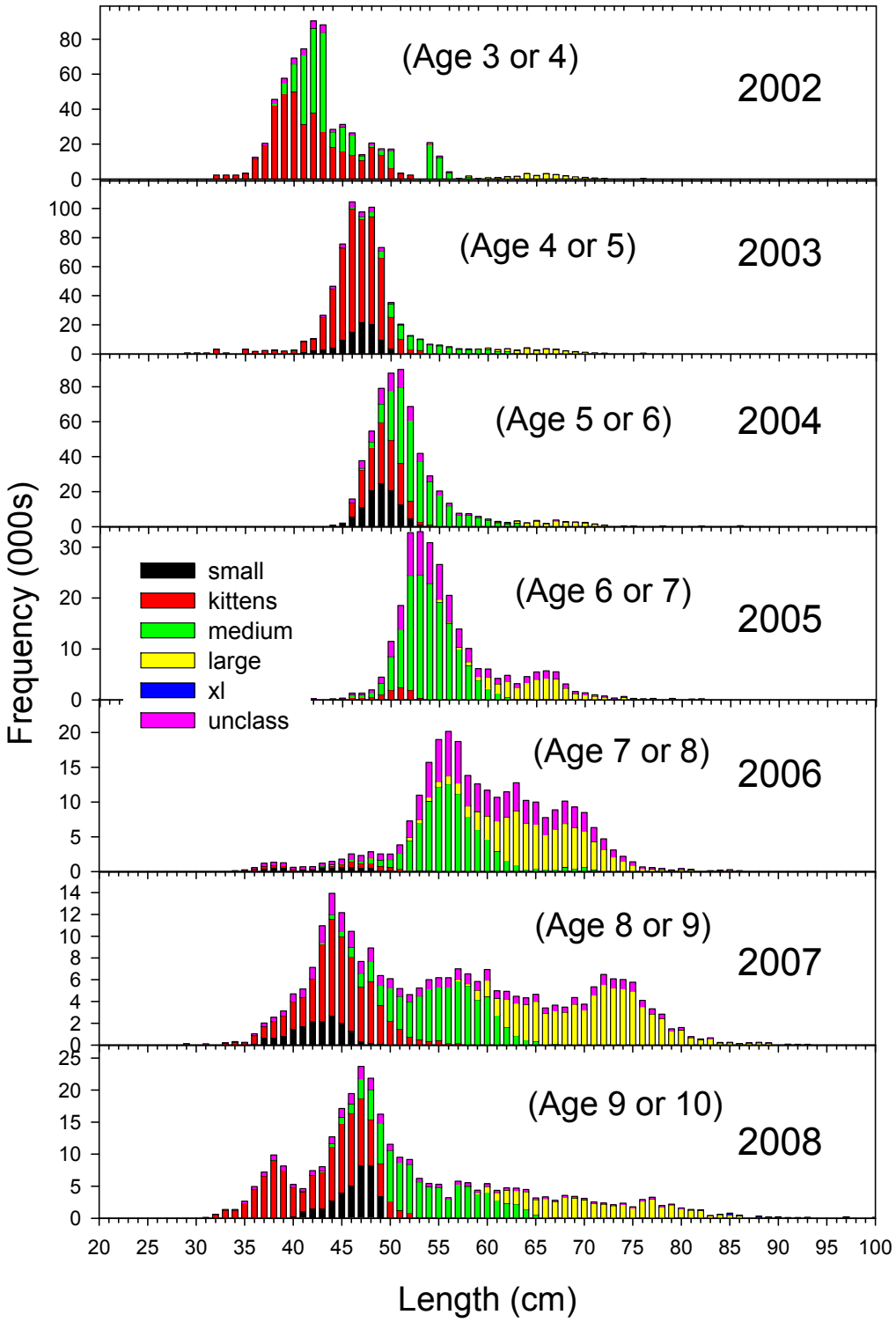


Figure A21. Expanded length frequency distributions by year. Y-axis is allowed to rescale.

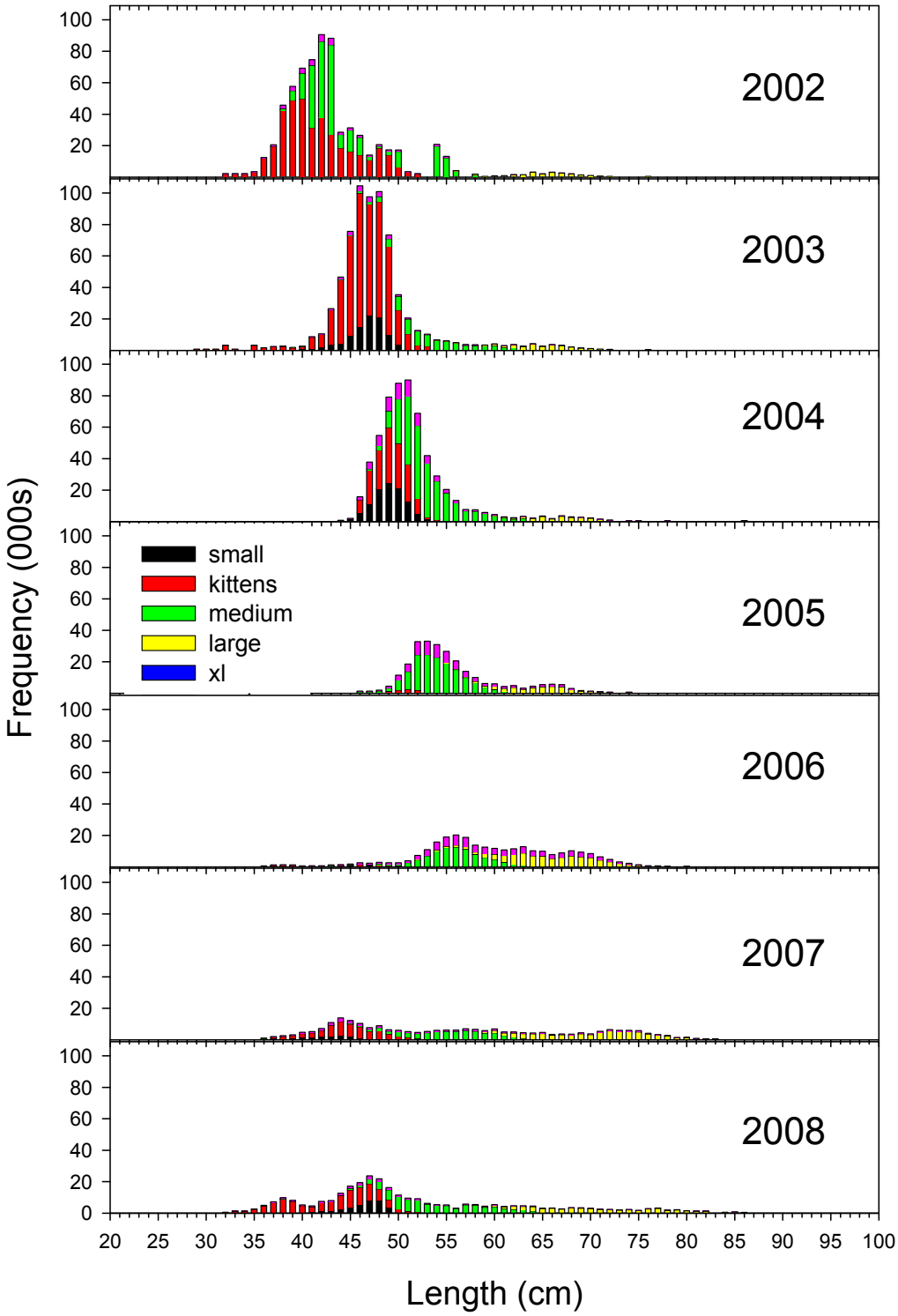


Figure A22. Expanded length frequency distributions by year. Y-axis scale is fixed.

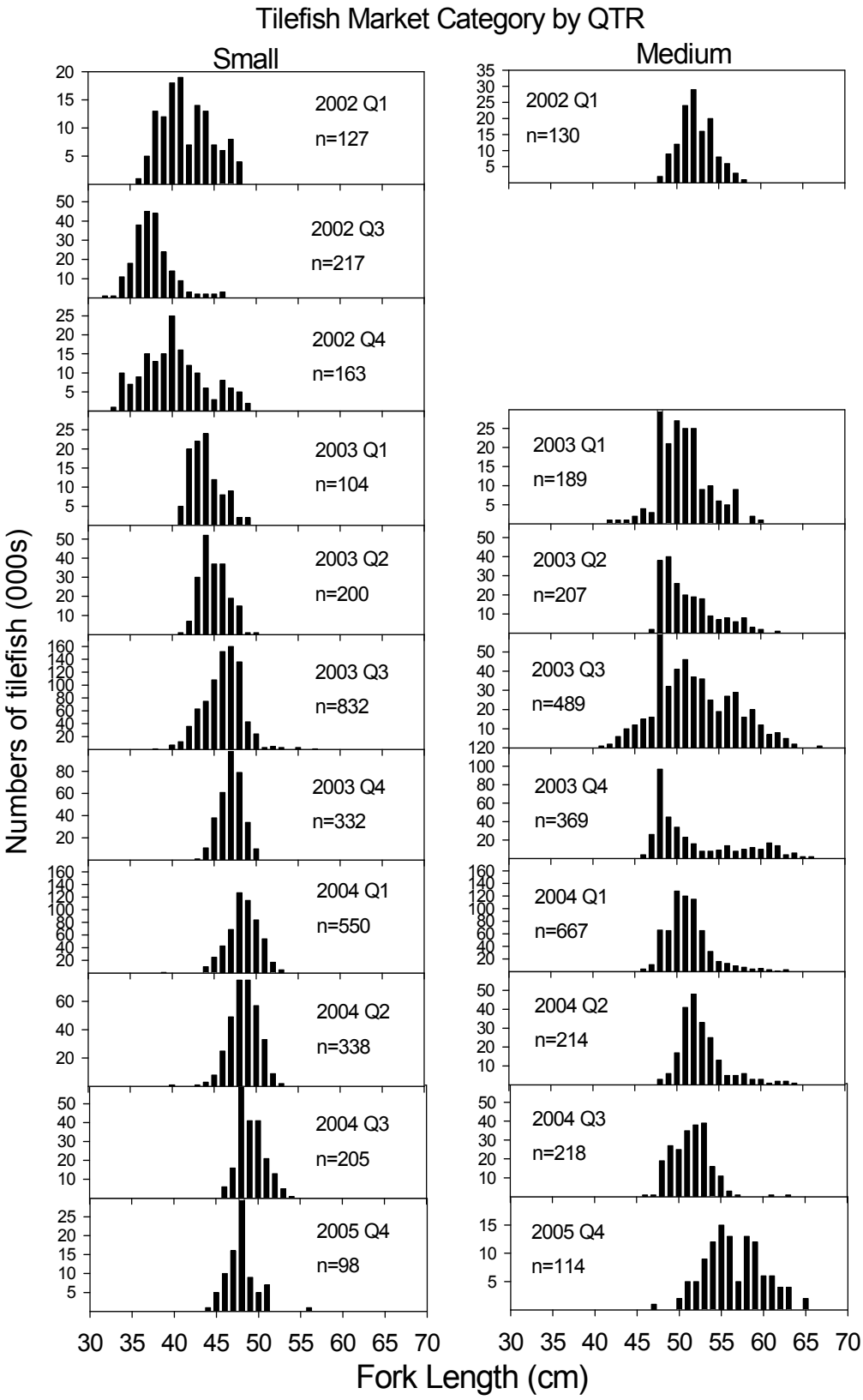


Figure A23. Small and medium tilefish market category length frequency distributions by quarter. Lengths from New York from 2000 to 2004 were converted to fork length.

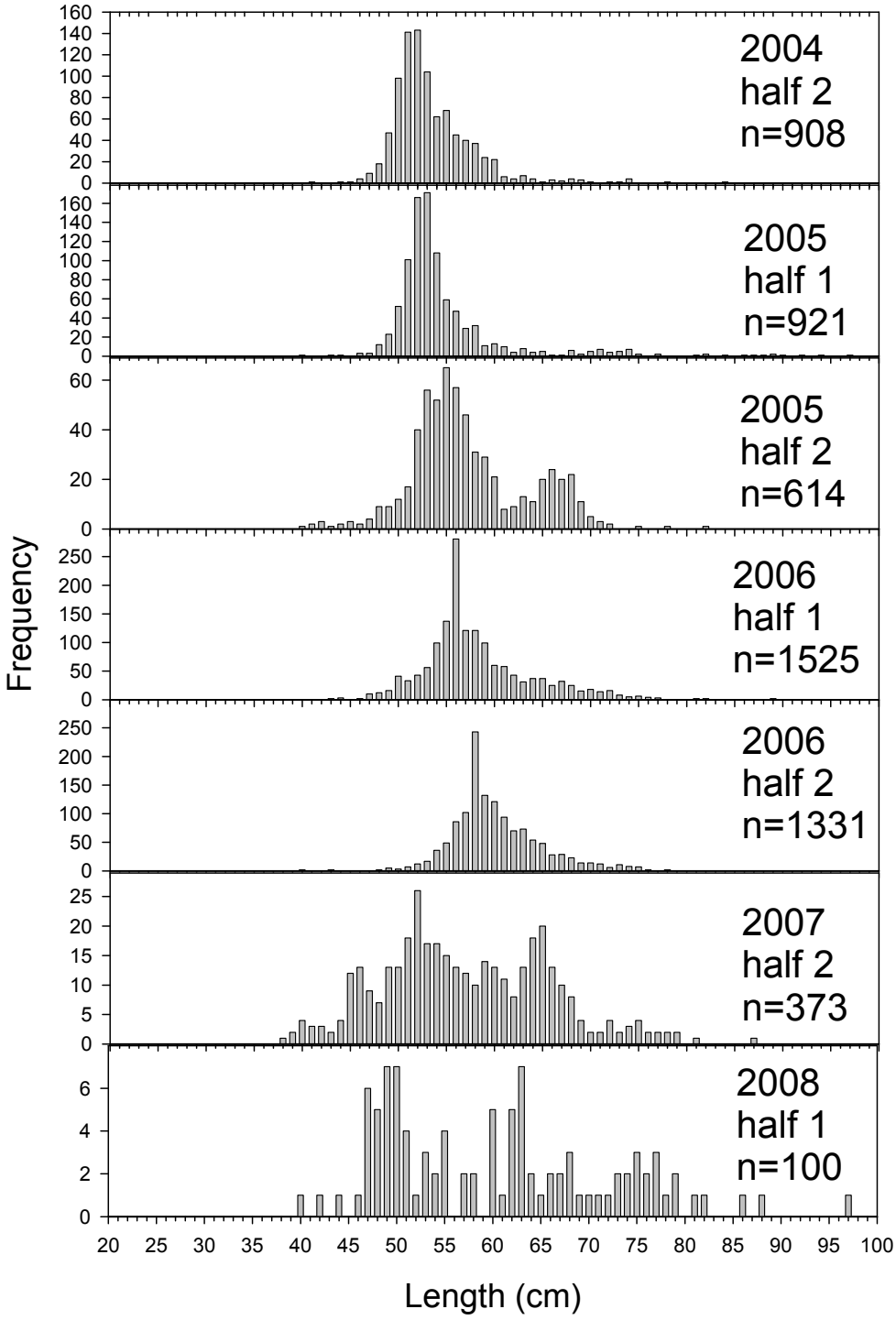


Figure A24. Observer kept length frequency distributions.

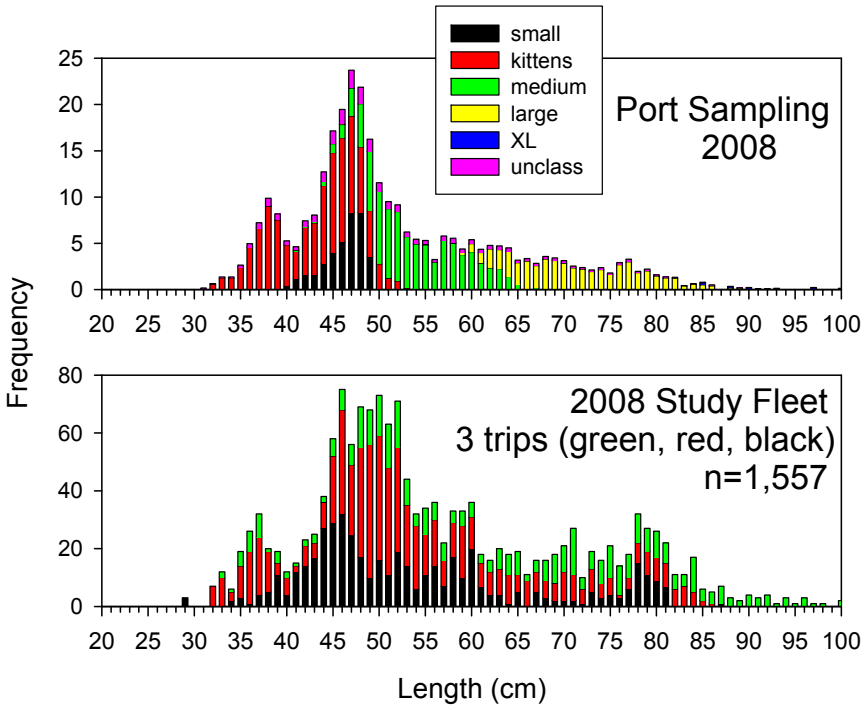


Figure A25. Comparison of study fleet length frequency with expanded landings distribution for 2008.

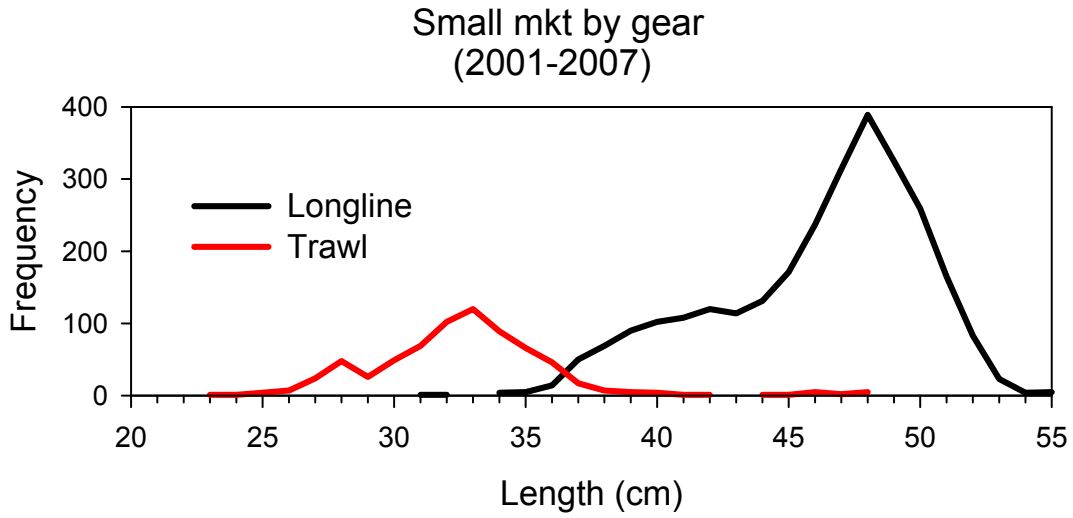


Figure A26. Length frequency distribution of trawl and longline landed fish from the small market category from 2001 to 2007.

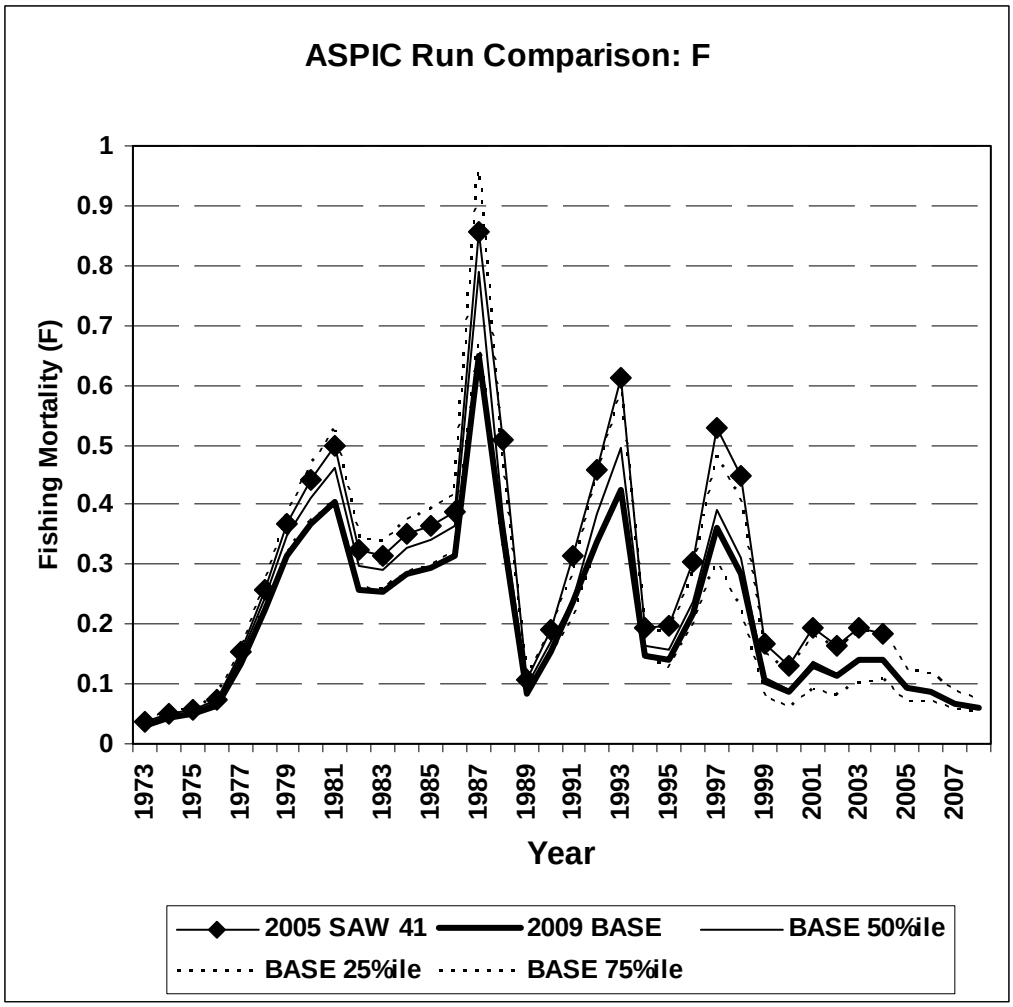


Figure A27. Comparison of the 2005 SAW 41 estimates of fishing mortality (F) with 2009 BASE run estimates.

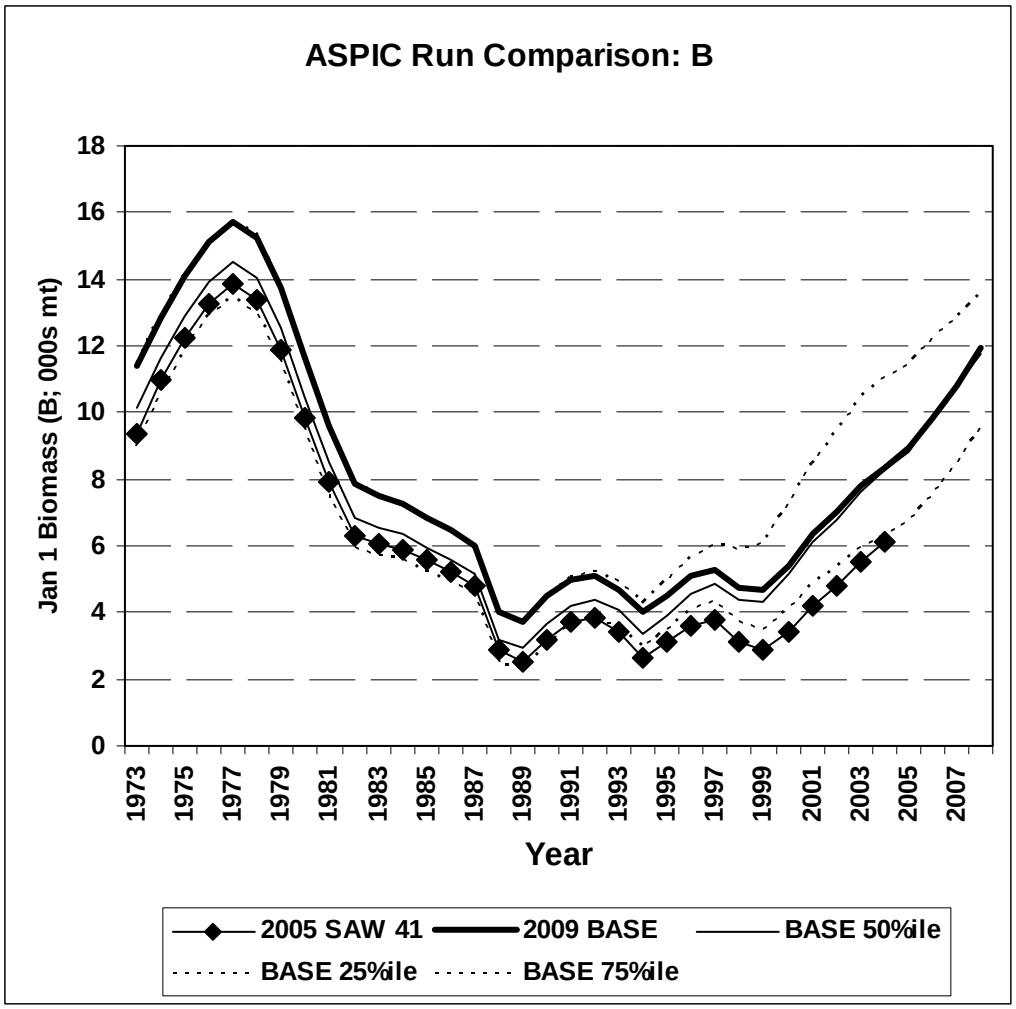


Figure A28. Comparison of the 2005 SAW 41 estimates of stock biomass (B) with 2009 BASE run estimates.

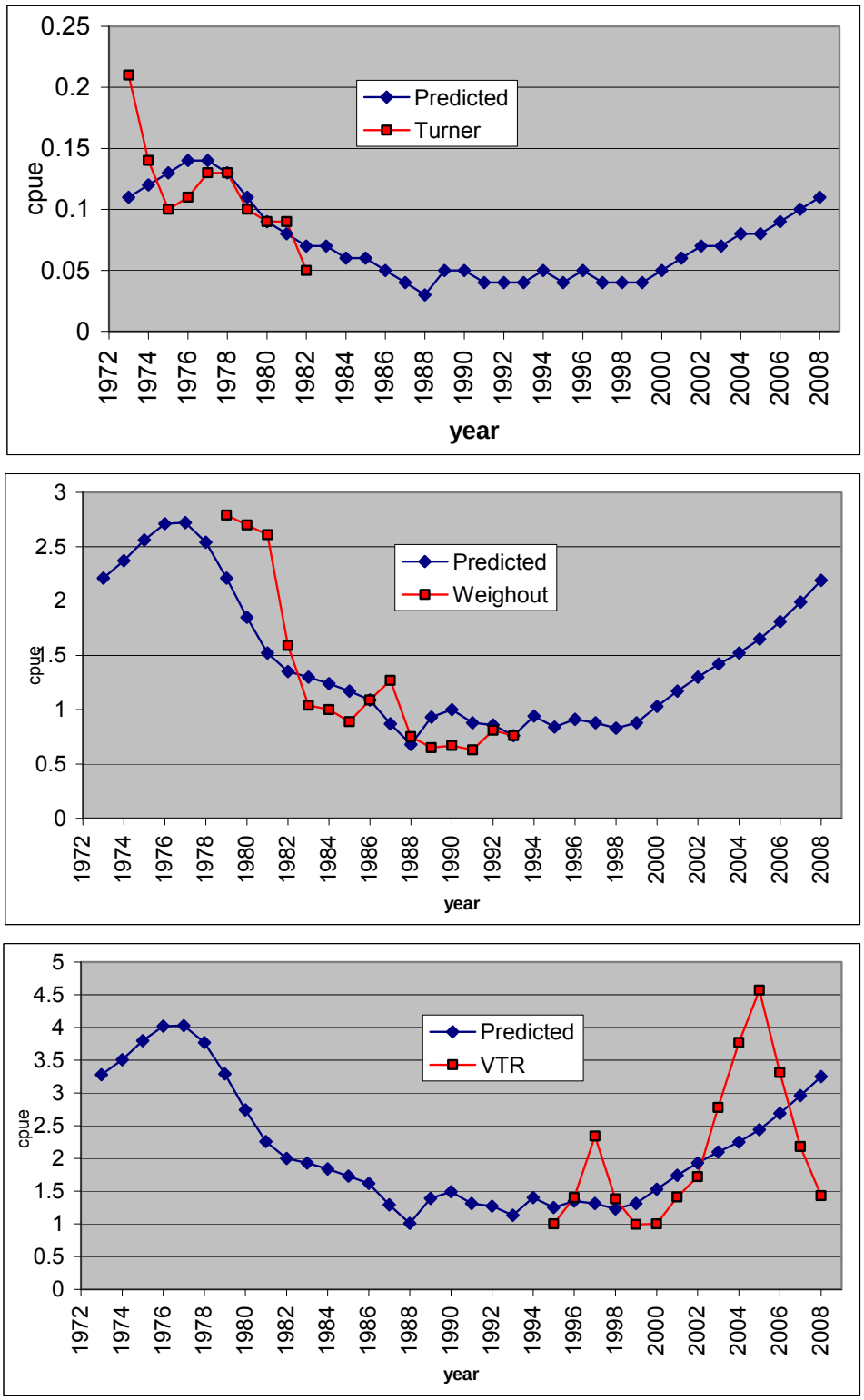


Figure A29. Fit of the ASPIC base run 1 with the three (Turner's, Weighout, and VTR) cpue series.

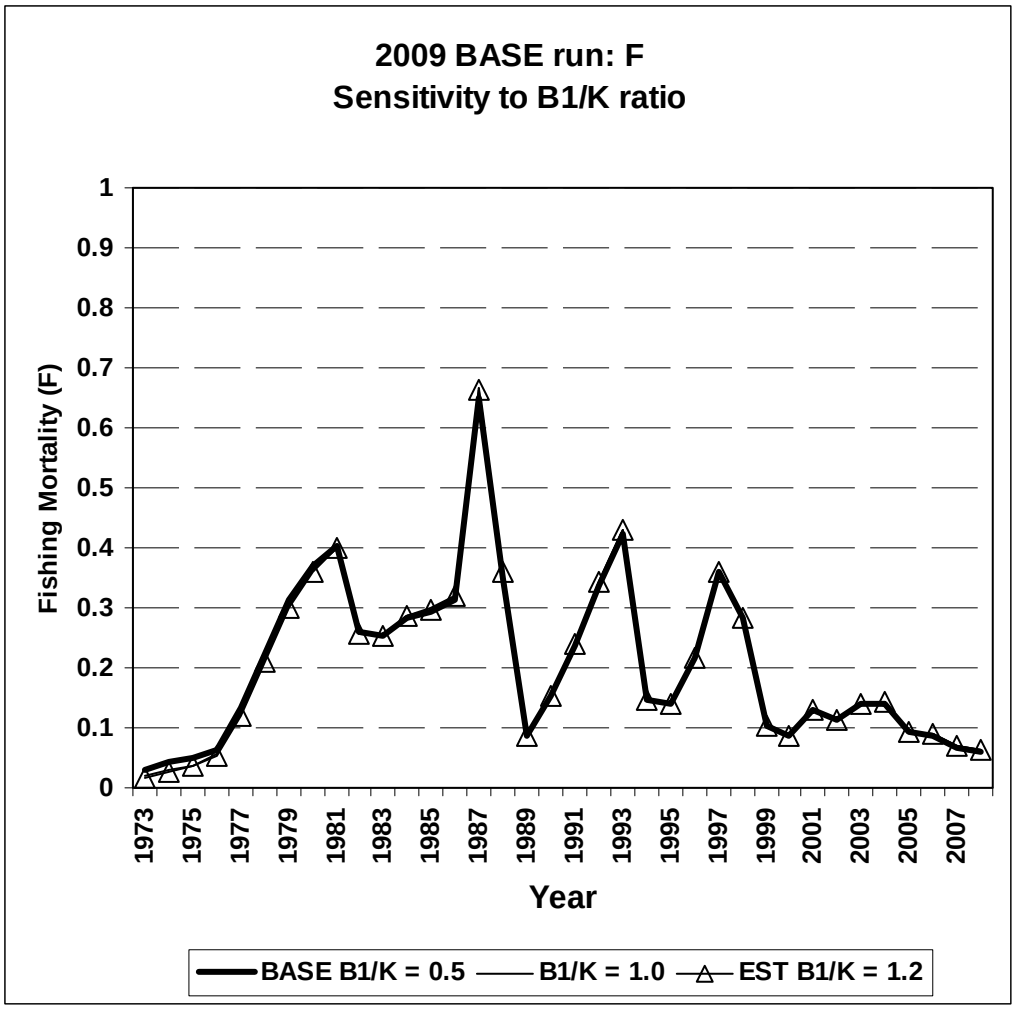


Figure A30. Sensitivity of 2009 BASE run estimated fishing mortality (F) using different values of the time series starting biomass (B1) to carrying capacity (K) ratio. The B1/K = 0.1 run is not shown since this run produced infeasible results by hitting a model bound.

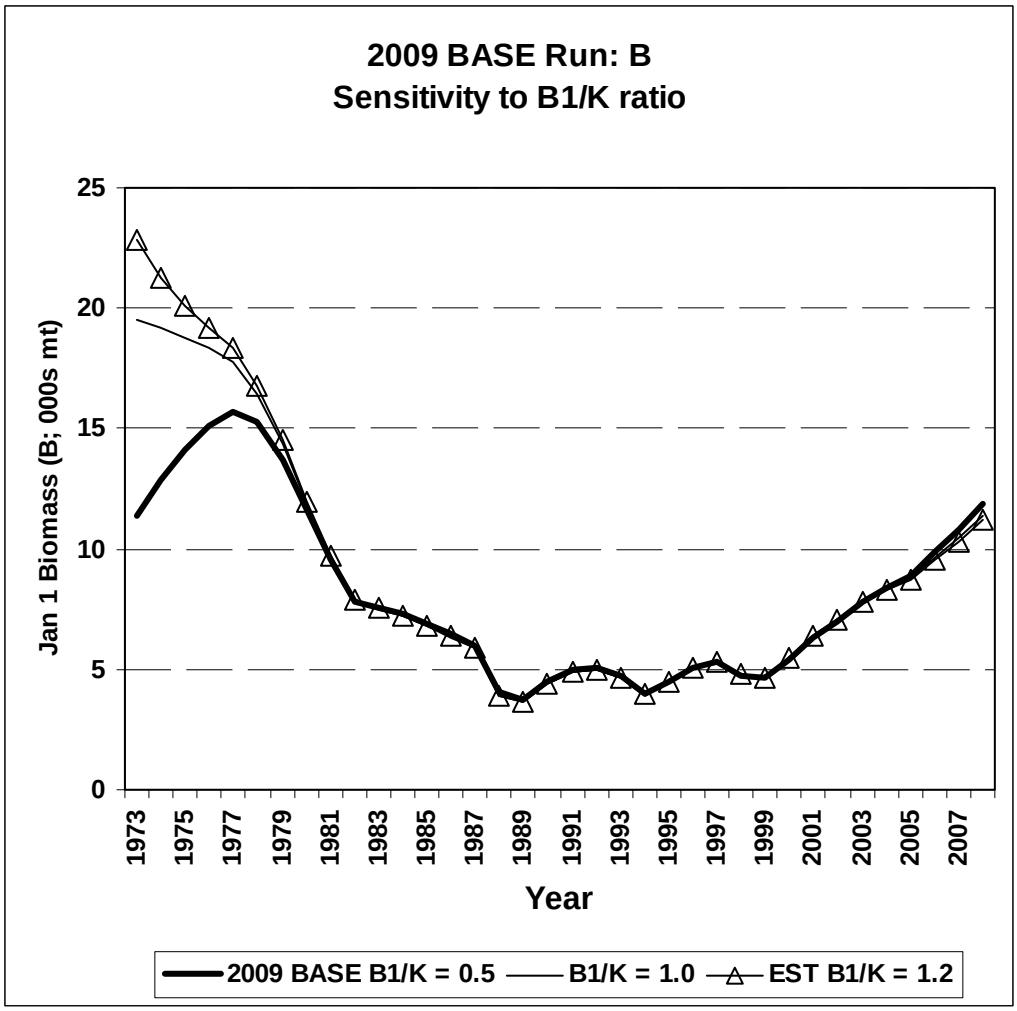


Figure A31. Sensitivity of 2009 BASE run estimated stock biomass (B) for different values of the time series starting biomass (B1) to carrying capacity (K) ratio. The B1/K = 0.1 run is not shown since this run produced infeasible results by hitting a model bound.

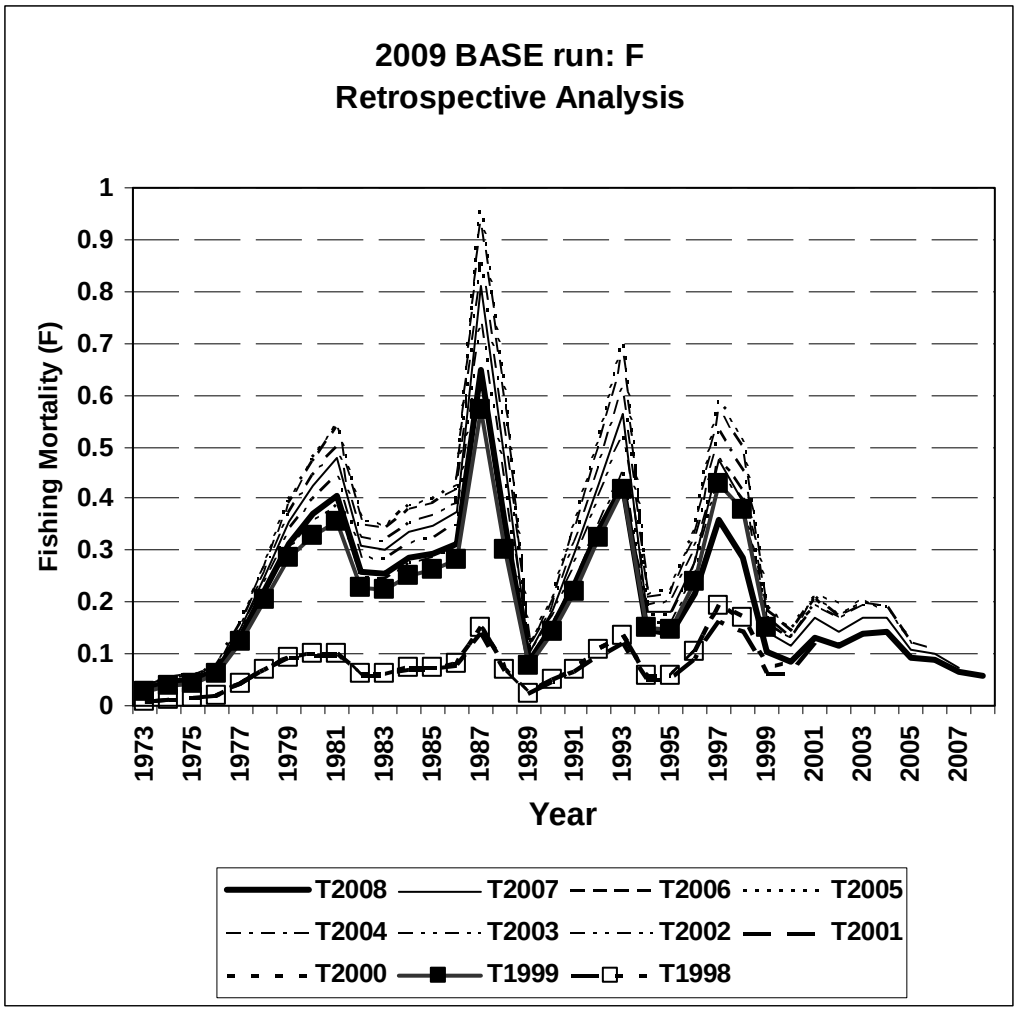


Figure A32. Retrospective analysis results for the 2009 BASE run: fishing mortality (F).

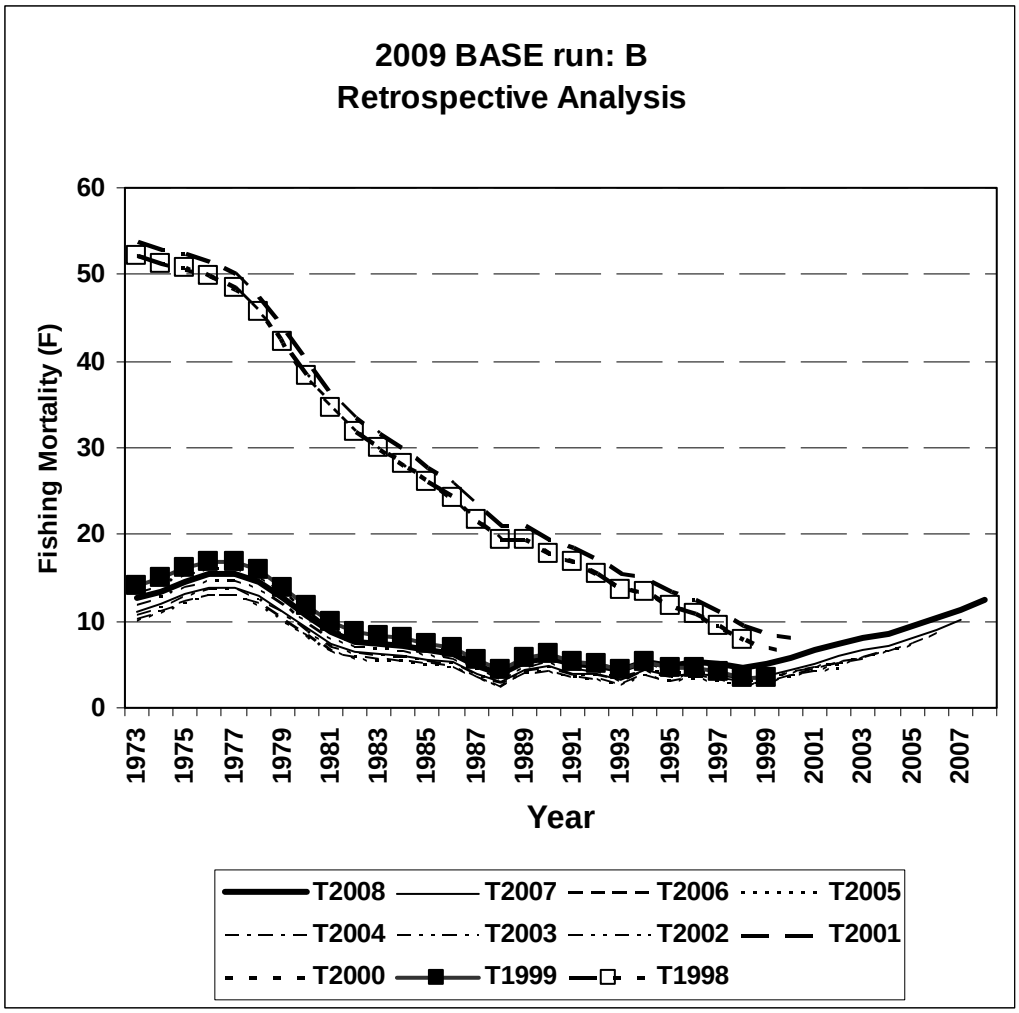


Figure A33. Retrospective analysis results for the 2009 BASE run: stock biomass (B).

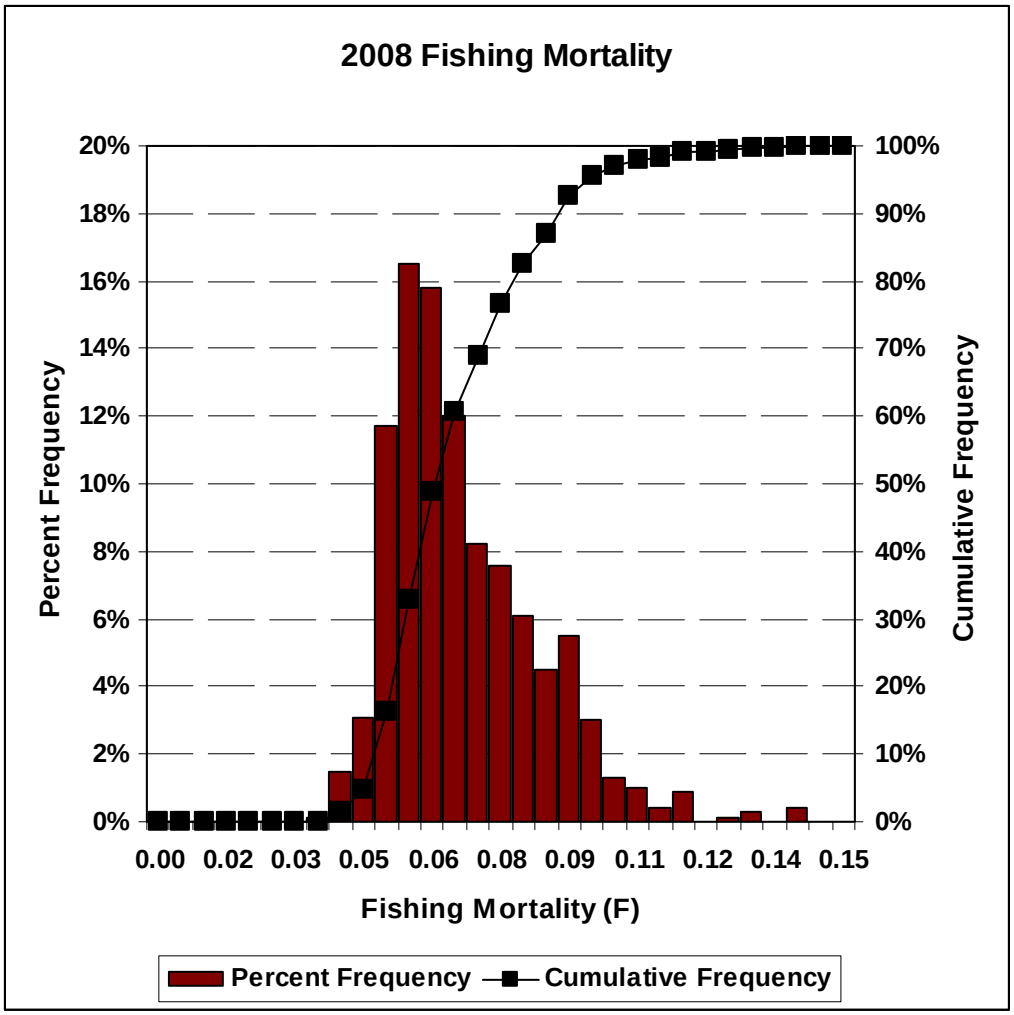


Figure A34. Bootstrap estimates (1000 iterations) of the precision of 2008 fishing mortality from the 2009 BASE run. Vertical bars display the range of the bootstrap estimates; the percent confidence intervals can be taken from the cumulative frequency. The 2008 point estimate of fishing mortality = 0.059.

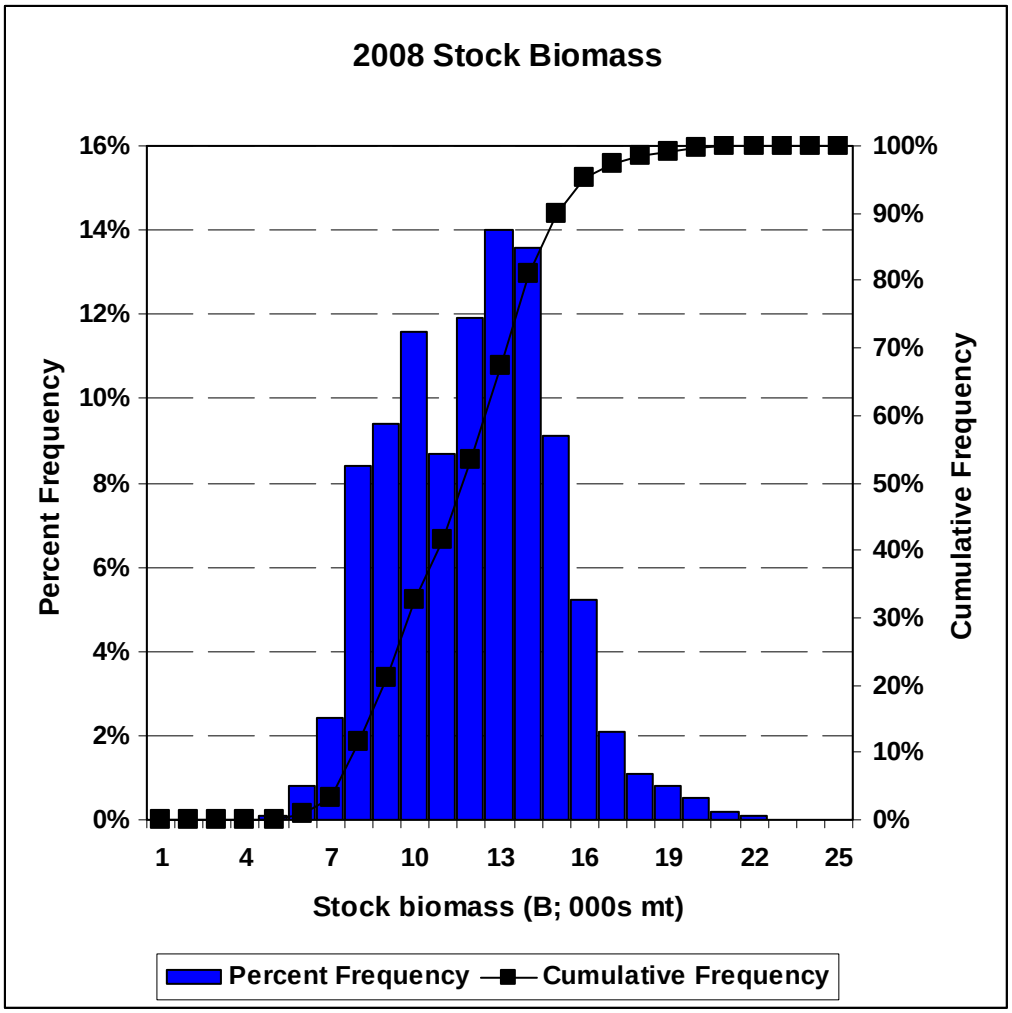


Figure A35. Bootstrap estimates (1000 iterations) of the precision of 2008 stock biomass from the 2009 BASE run. Vertical bars display the range of the bootstrap estimates; the percent confidence intervals can be taken from the cumulative frequency. The 2008 point estimate of stock biomass = 11.910 thousand mt.

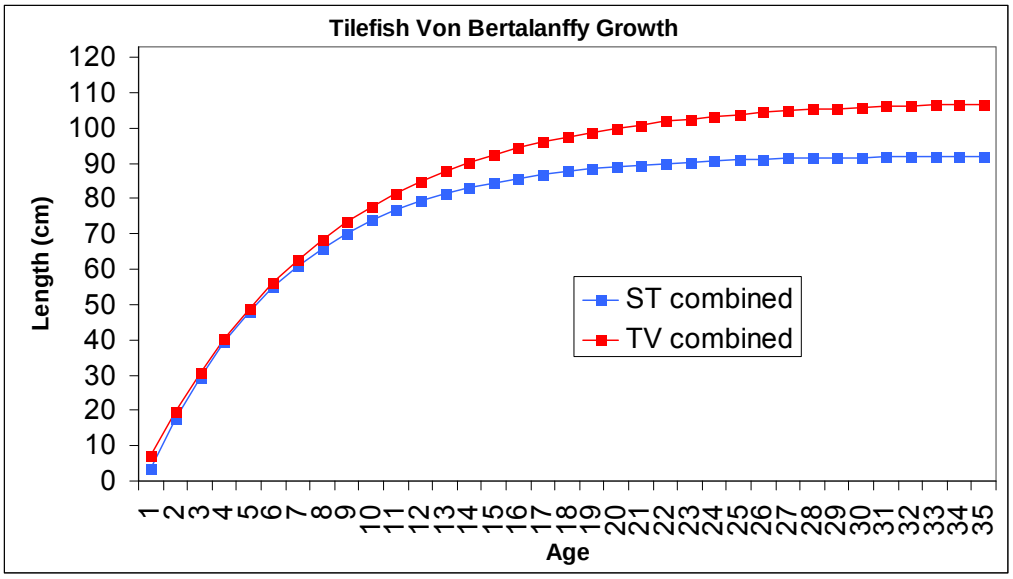


Figure A36. Comparison of Vidal's (2008) and Turner's (1986) von Bertalanffy growth curve with the sexes combined.

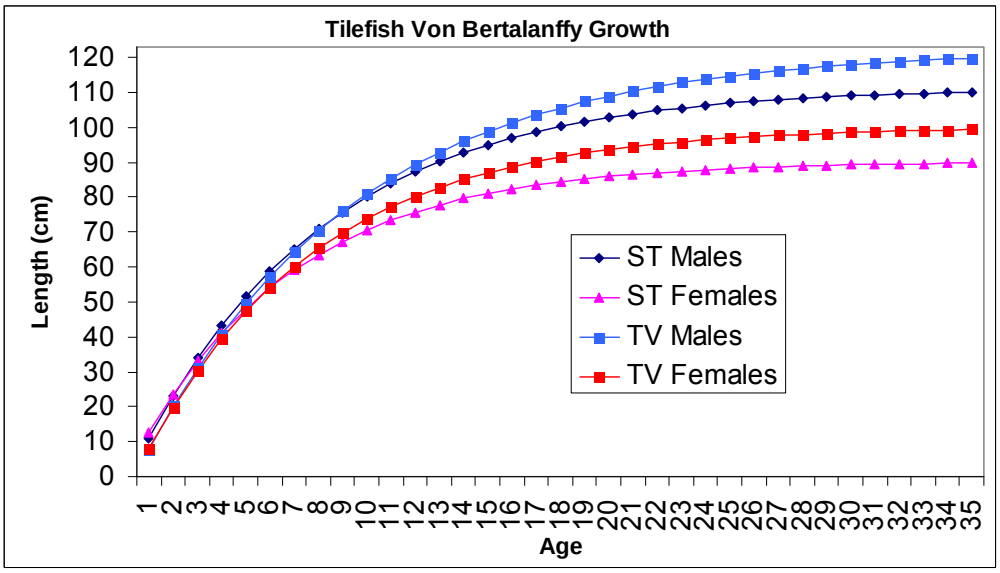


Figure A37. Comparison of Vidal's (2008) and Turner's (1986) von Bertalanffy growth curve with the sexes separated.

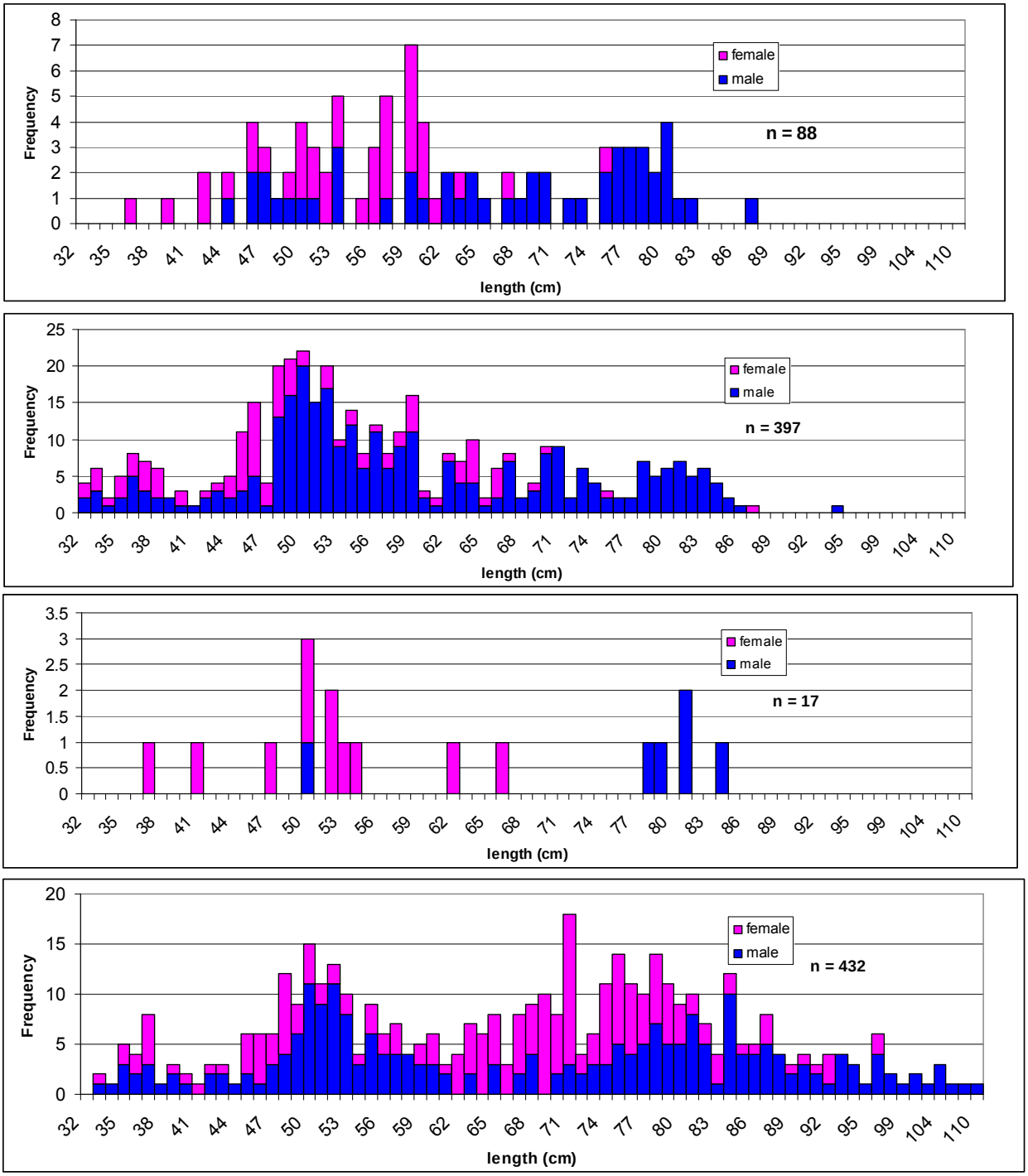


Figure A38. Study fleet length distributions by sex and trip.

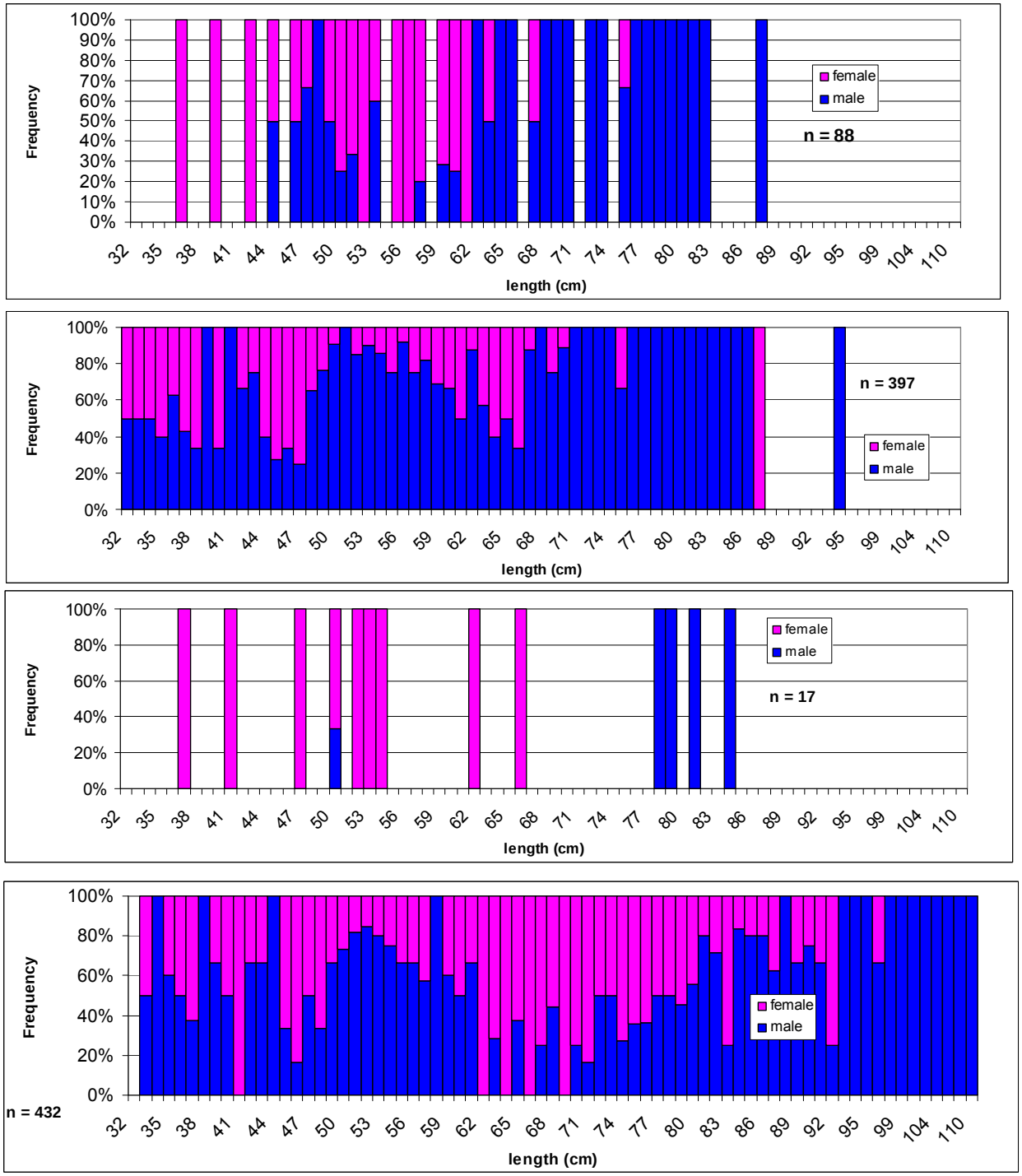
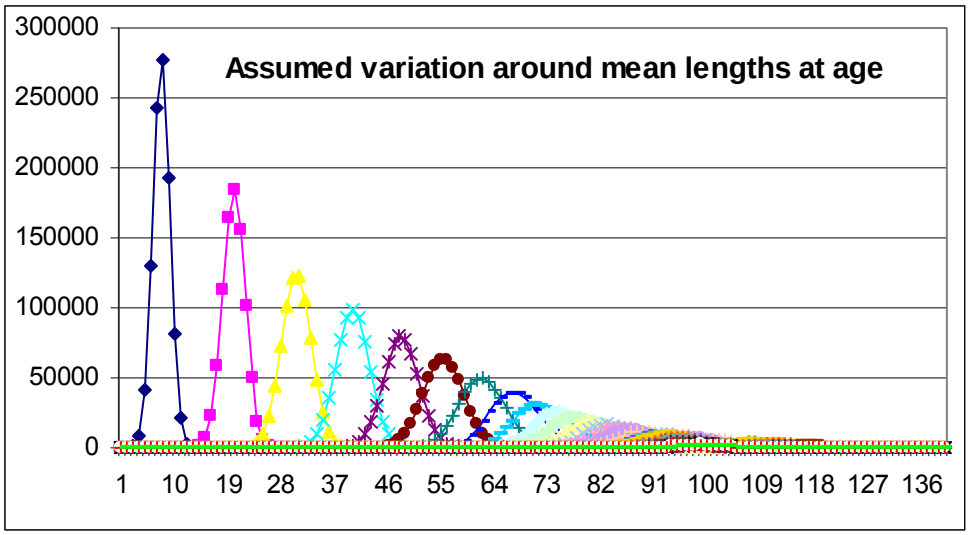


Figure A39. Study fleet sex ratio at length by trip.



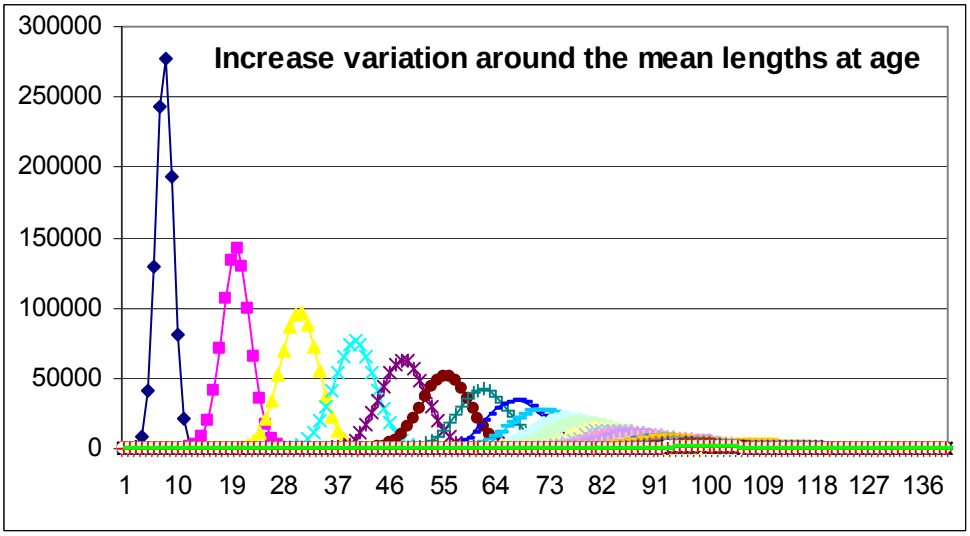


Figure A40. SCALE base run 1 assumed variation around the mean lengths at age (top) and run 5 which increased the assumed variation around the mean lengths at age (bottom).

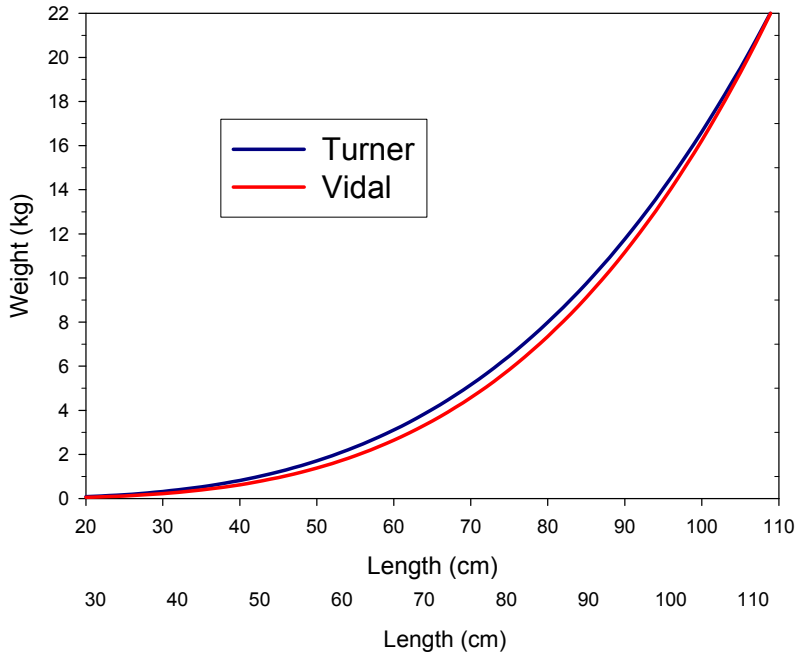


Figure A41. Top graph shows the length weight relationship calculated from the study fleet data (T Vidal 2008). Bottom graph shows the comparison between Turner's (1986) and Vidal length weight relationships.

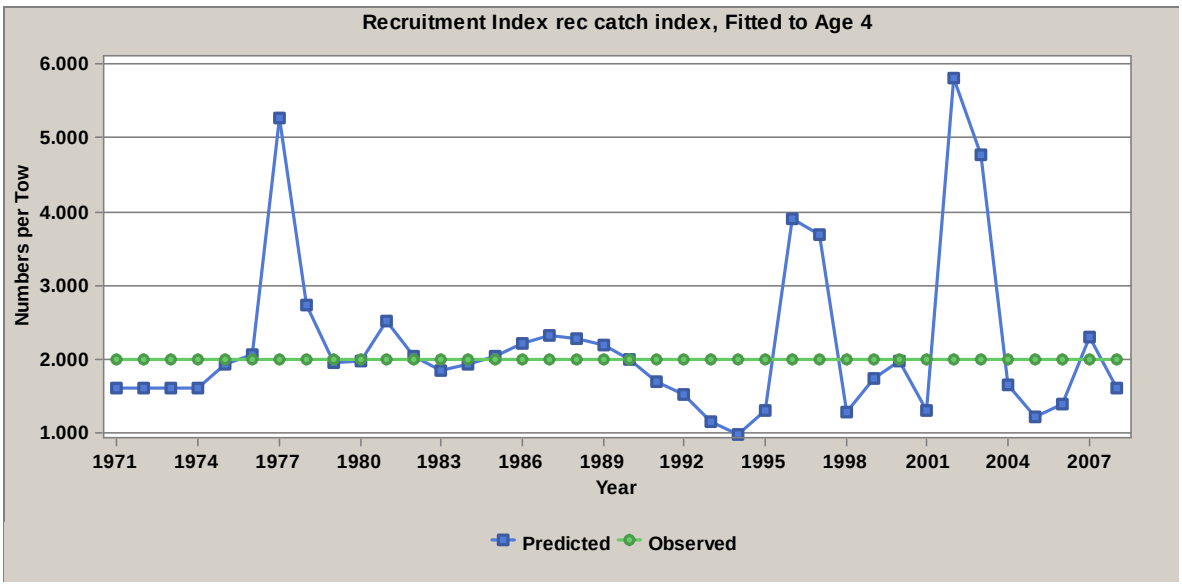


Figure A42. SCALE base run 1 Straight line recruitment index.

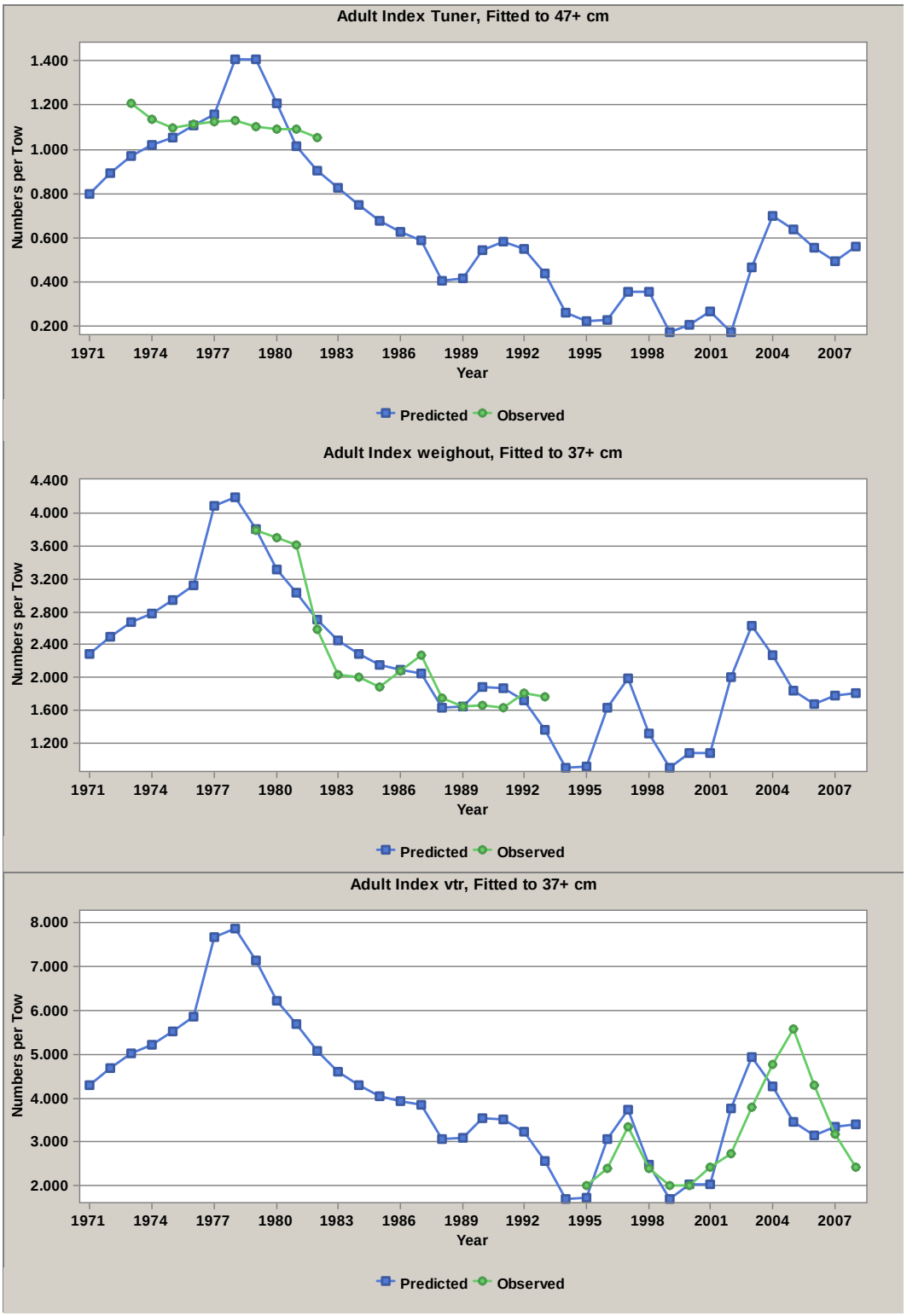


Figure A43. SCALE base run 1 fit to the three cpue indices.

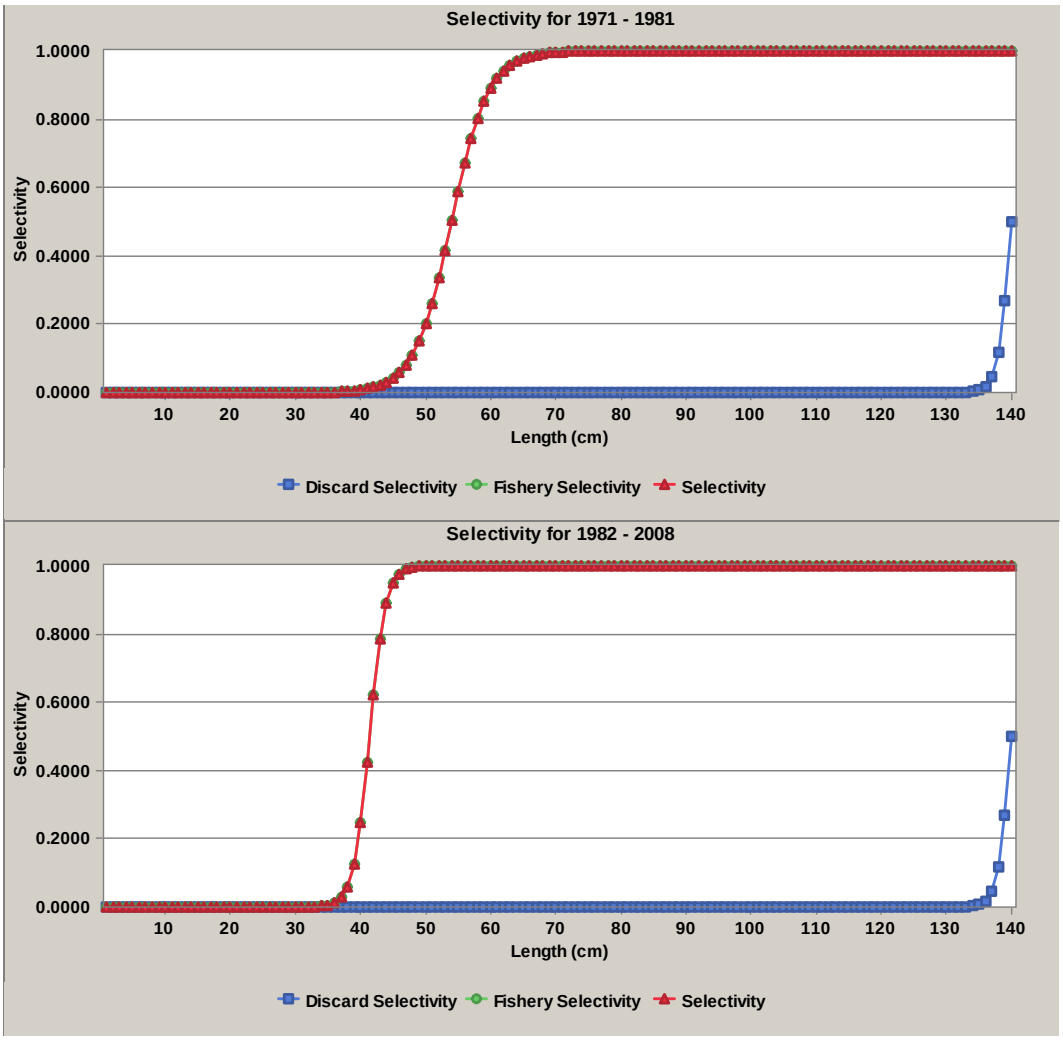


Figure A44. SCALE base run 1 estimated selectivity (block 1 is from1971-1981, block 2 is from 1984-2008).

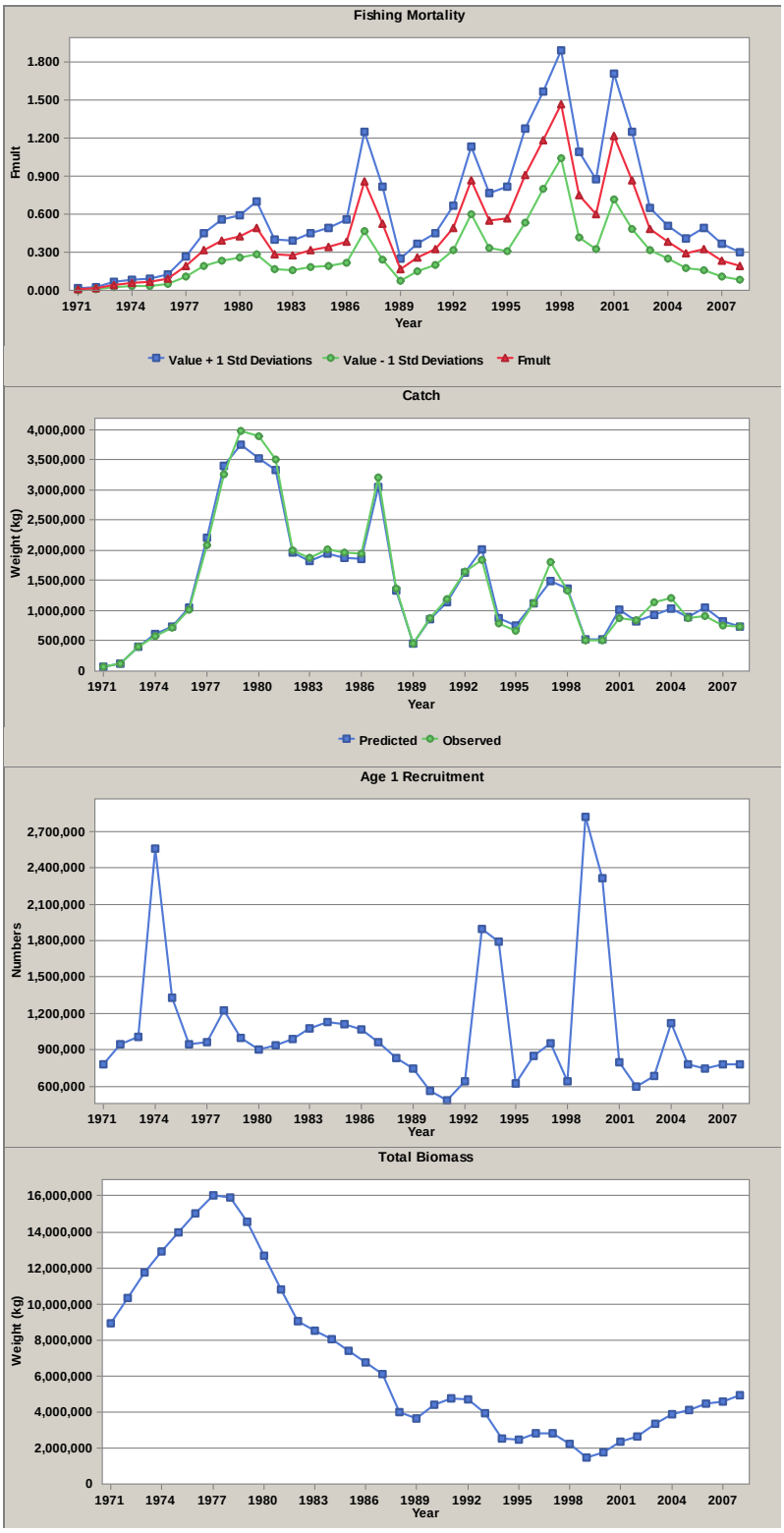
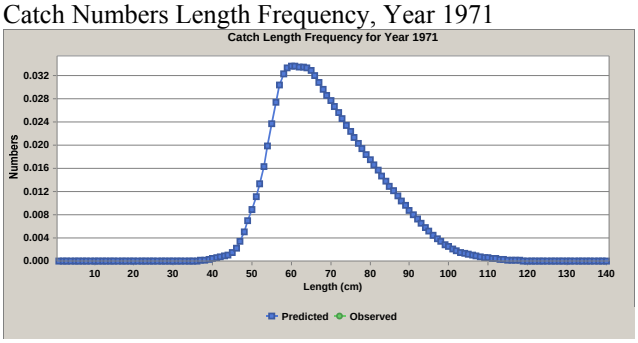
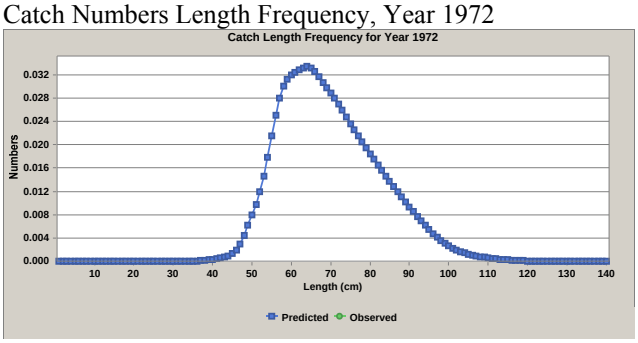


Figure A45. SCALE base run 1 estimated F, fit to the catch, estimated recruitment, and total biomass.

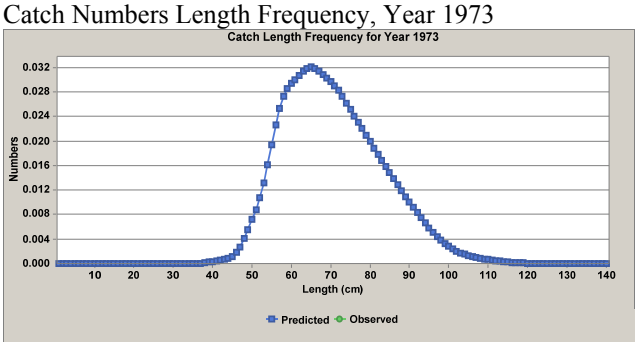
Catch Numbers Length Frequency, Year 1971



Catch Numbers Length Frequency, Year 1972



Catch Numbers Length Frequency, Year 1973



Catch Numbers Length Frequency, Year 1974

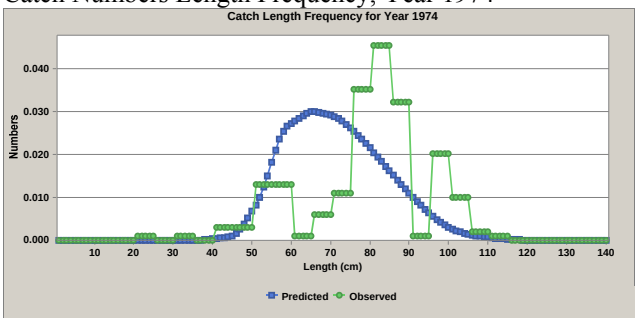
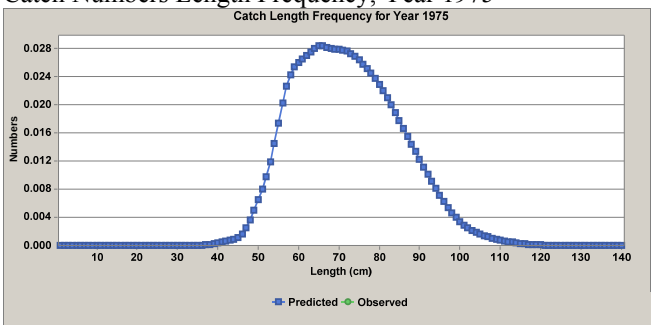
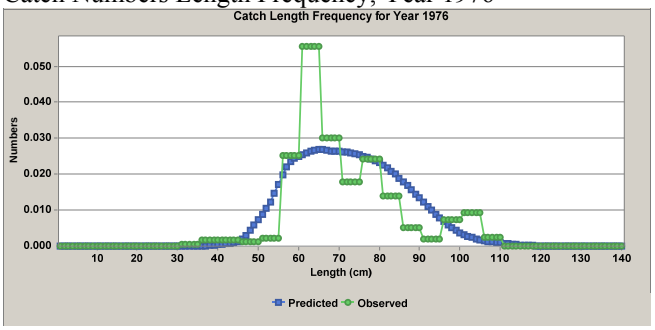


Figure A46. SCALE base run 1 predicted (blue) and observed (green) catch distributions by year. Years which do not have data are also shown.

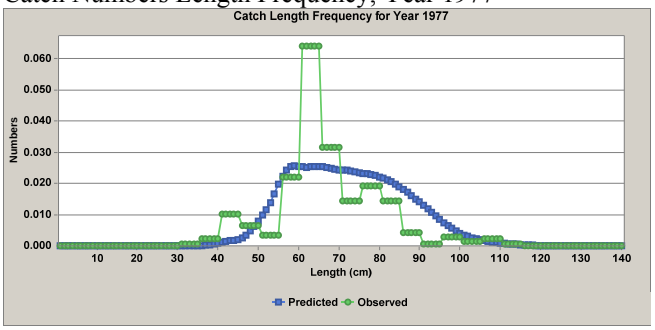
Catch Numbers Length Frequency, Year 1975



Catch Numbers Length Frequency, Year 1976



Catch Numbers Length Frequency, Year 1977



Catch Numbers Length Frequency, Year 1978

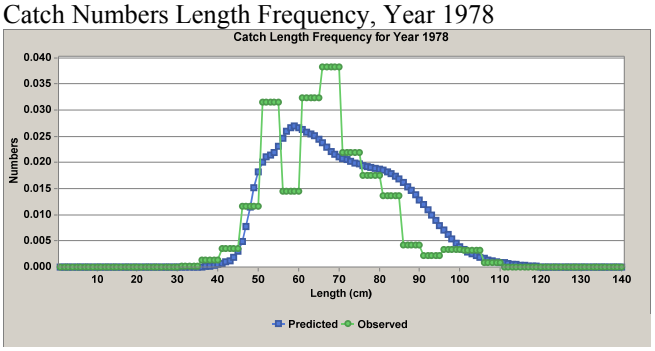
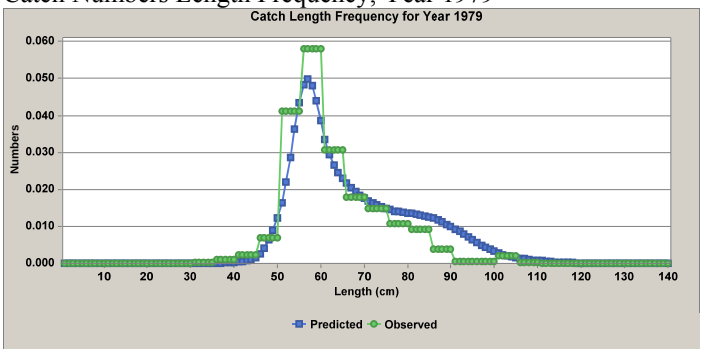
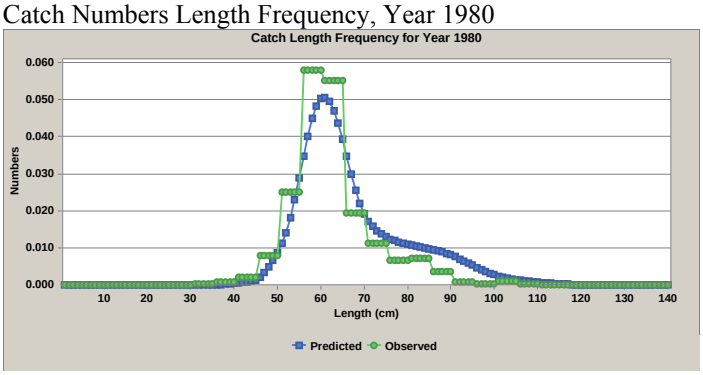


Figure A46. cont.

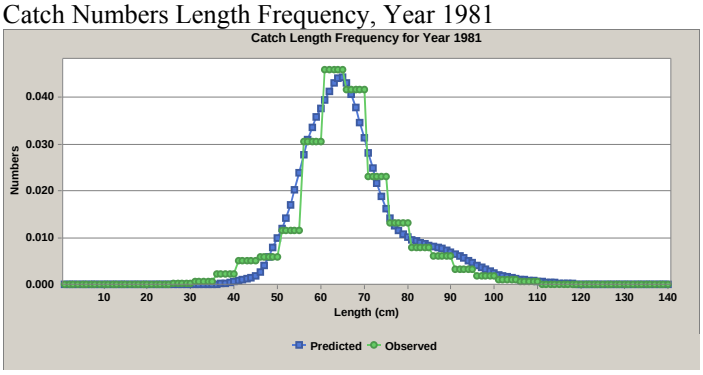
Catch Numbers Length Frequency, Year 1979



Catch Numbers Length Frequency, Year 1980



Catch Numbers Length Frequency, Year 1981



Catch Numbers Length Frequency, Year 1982

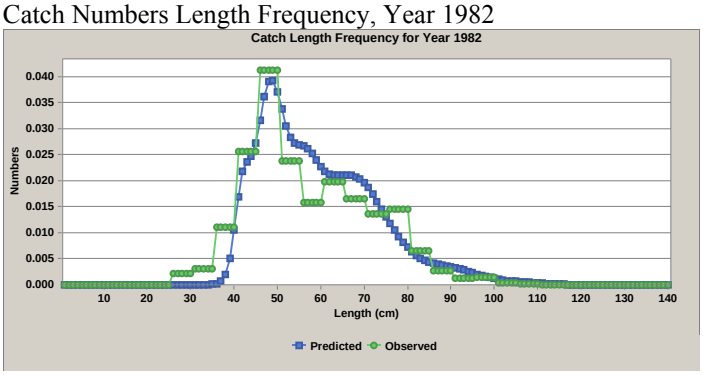
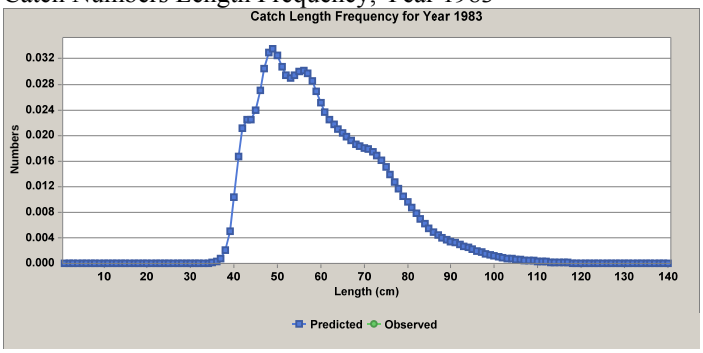
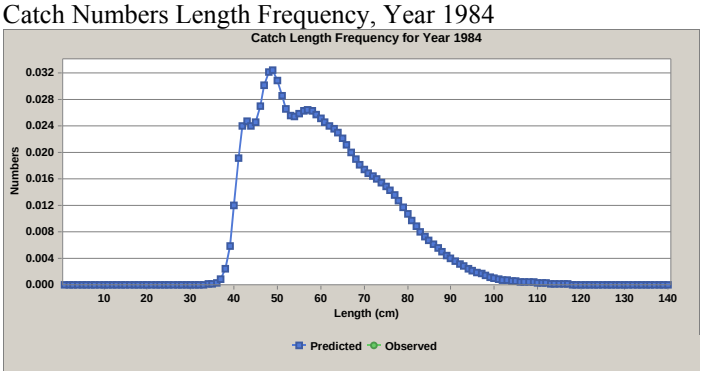


Figure A46. cont.

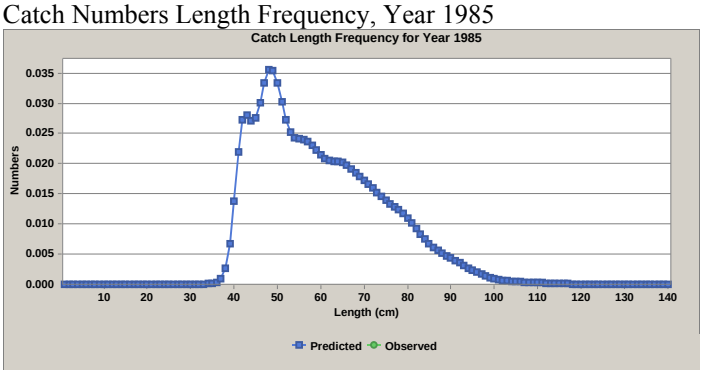
Catch Numbers Length Frequency, Year 1983



Catch Numbers Length Frequency, Year 1984



Catch Numbers Length Frequency, Year 1985



Catch Numbers Length Frequency, Year 1986

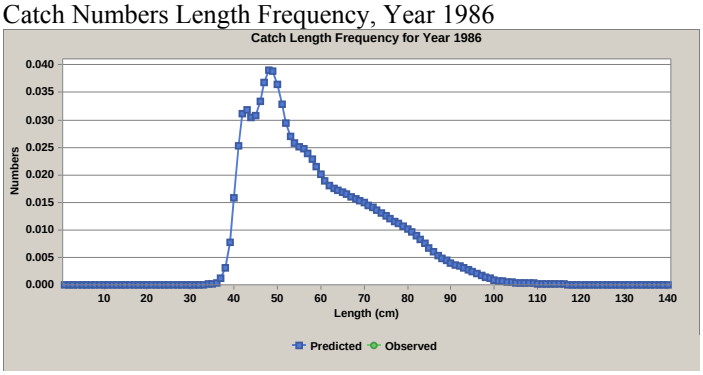
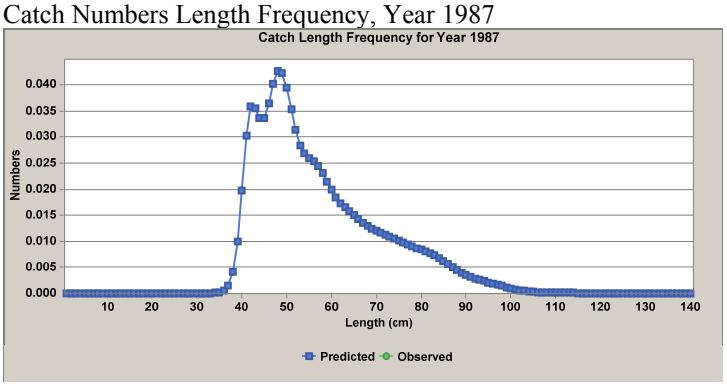
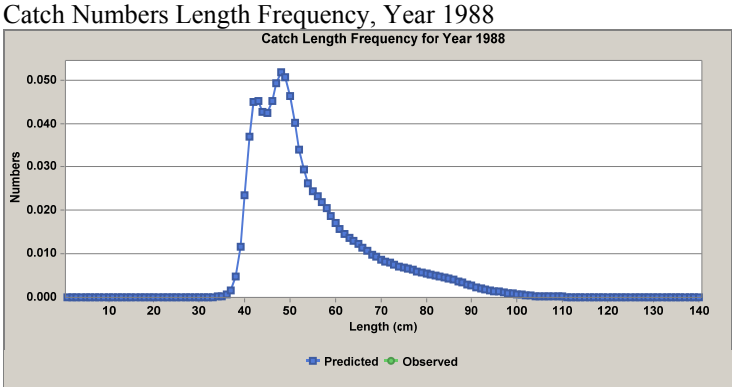


Figure A46. cont.

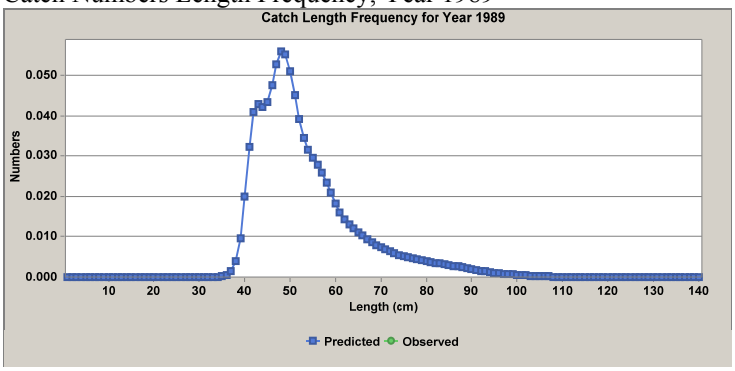
Catch Numbers Length Frequency, Year 1987



Catch Numbers Length Frequency, Year 1988



Catch Numbers Length Frequency, Year 1989



Catch Numbers Length Frequency, Year 1990

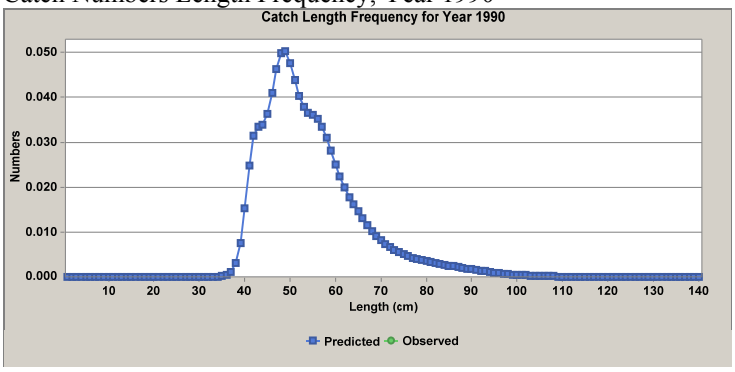
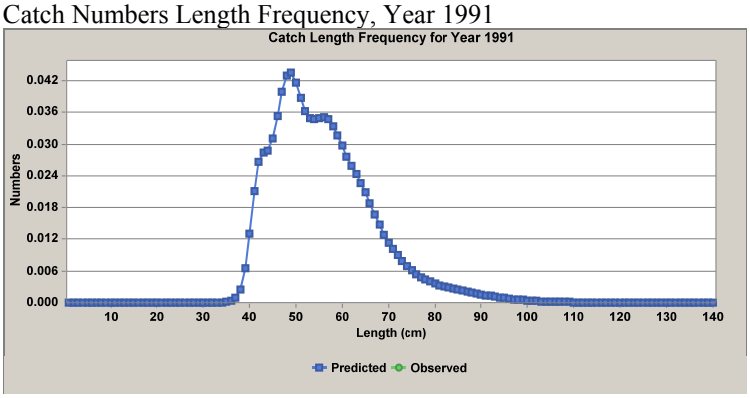
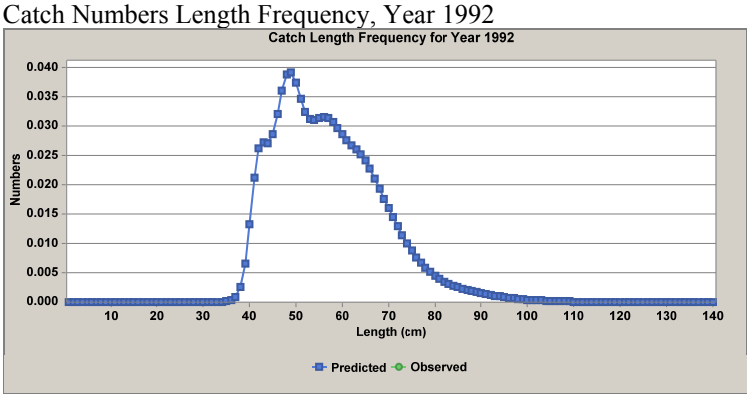


Figure A46. cont.

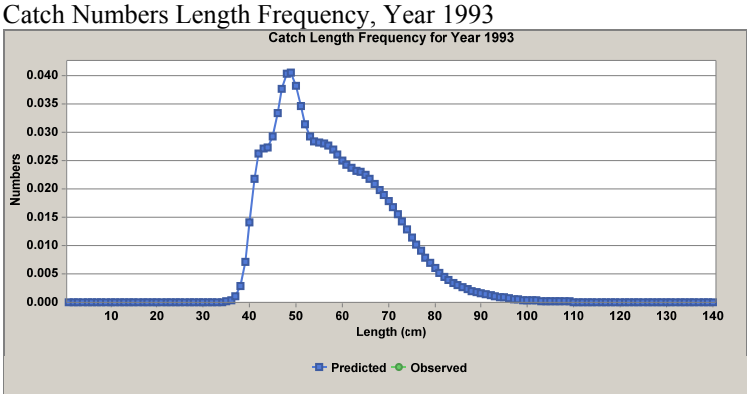
Catch Numbers Length Frequency, Year 1991



Catch Numbers Length Frequency, Year 1992



Catch Numbers Length Frequency, Year 1993



Catch Numbers Length Frequency, Year 1994

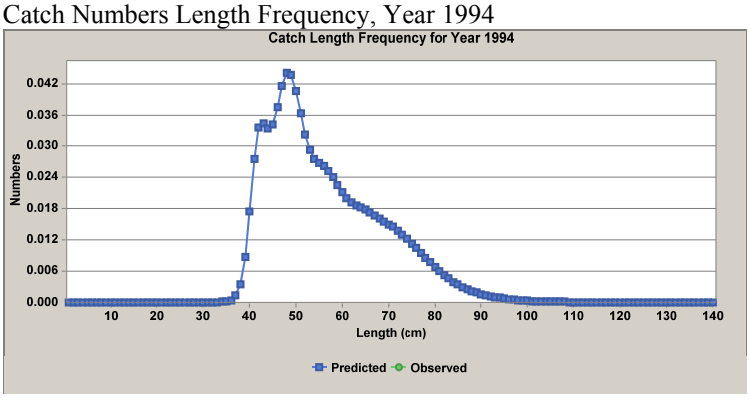
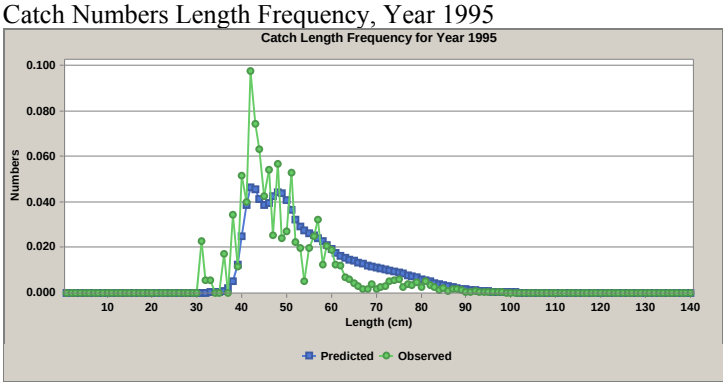
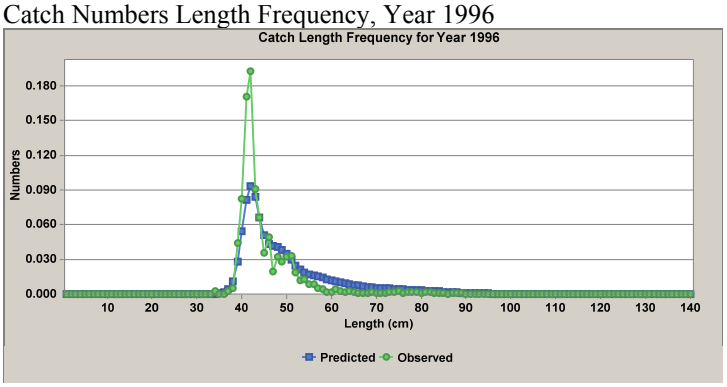


Figure A46. cont.

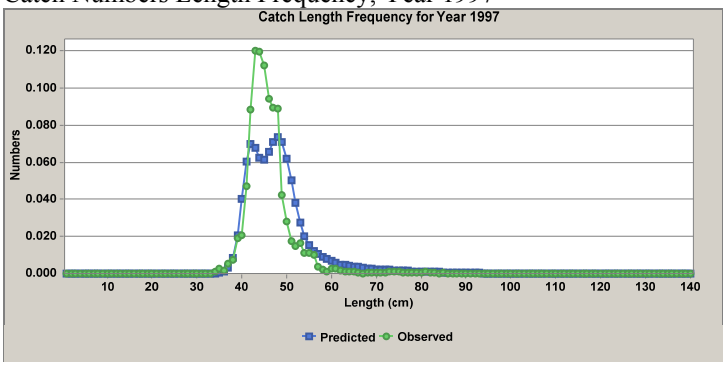
Catch Numbers Length Frequency, Year 1995



Catch Numbers Length Frequency, Year 1996



Catch Numbers Length Frequency, Year 1997



Catch Numbers Length Frequency, Year 1998

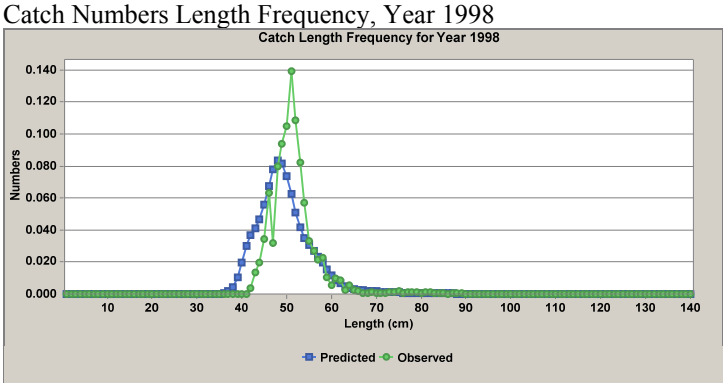
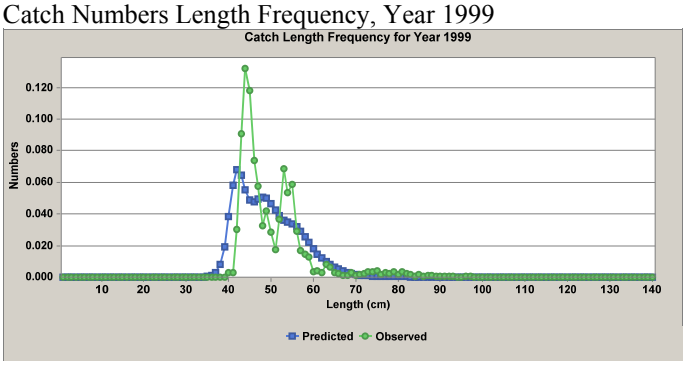
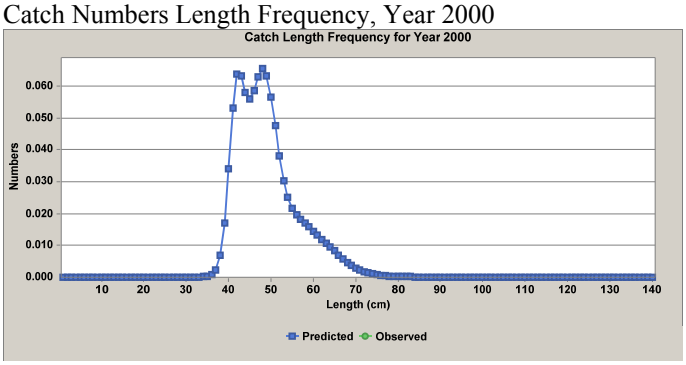


Figure A46. cont.

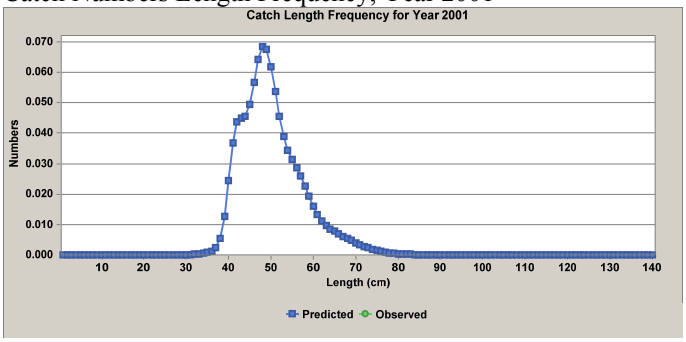
Catch Numbers Length Frequency, Year 1999



Catch Numbers Length Frequency, Year 2000



Catch Numbers Length Frequency, Year 2001



Catch Numbers Length Frequency, Year 2002

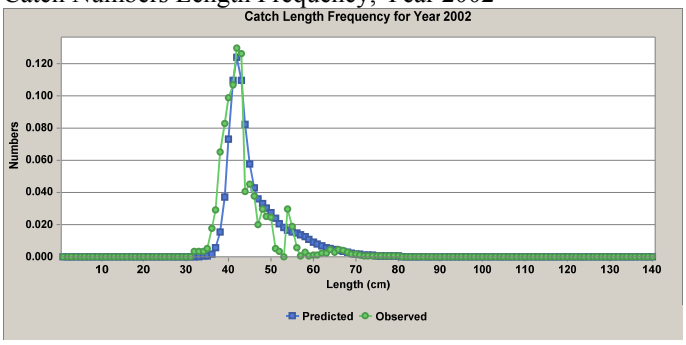
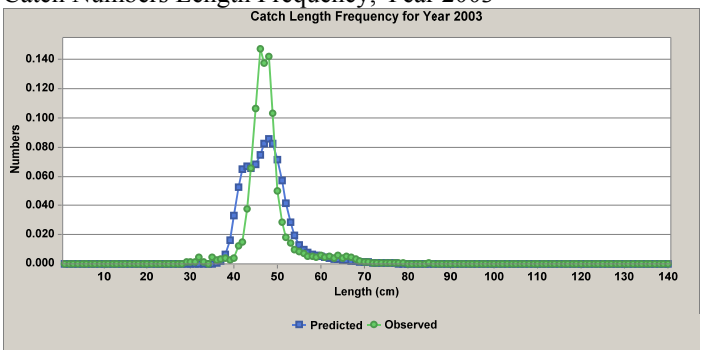
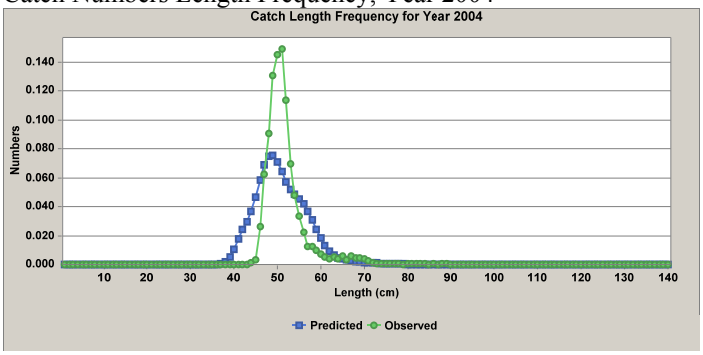


Figure A46. cont.

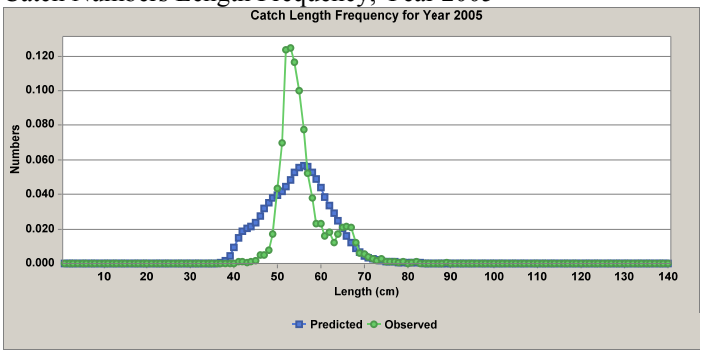
Catch Numbers Length Frequency, Year 2003



Catch Numbers Length Frequency, Year 2004



Catch Numbers Length Frequency, Year 2005



Catch Numbers Length Frequency, Year 2006

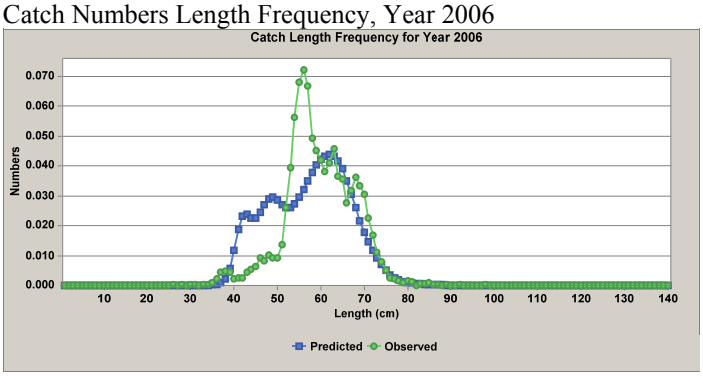
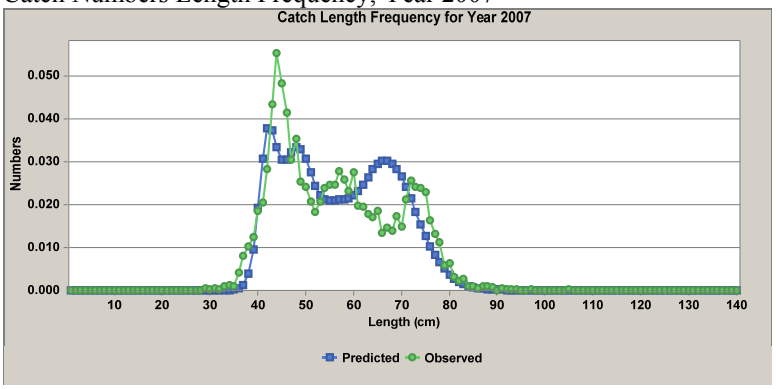


Figure A46. cont.

Catch Numbers Length Frequency, Year 2007



Catch Numbers Length Frequency, Year 2008

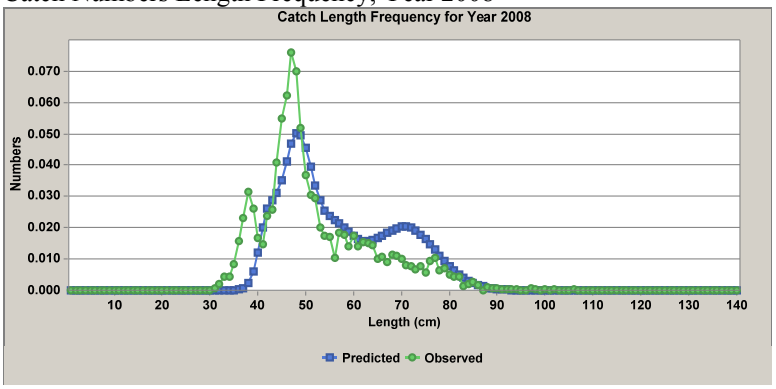


Figure A46. cont.

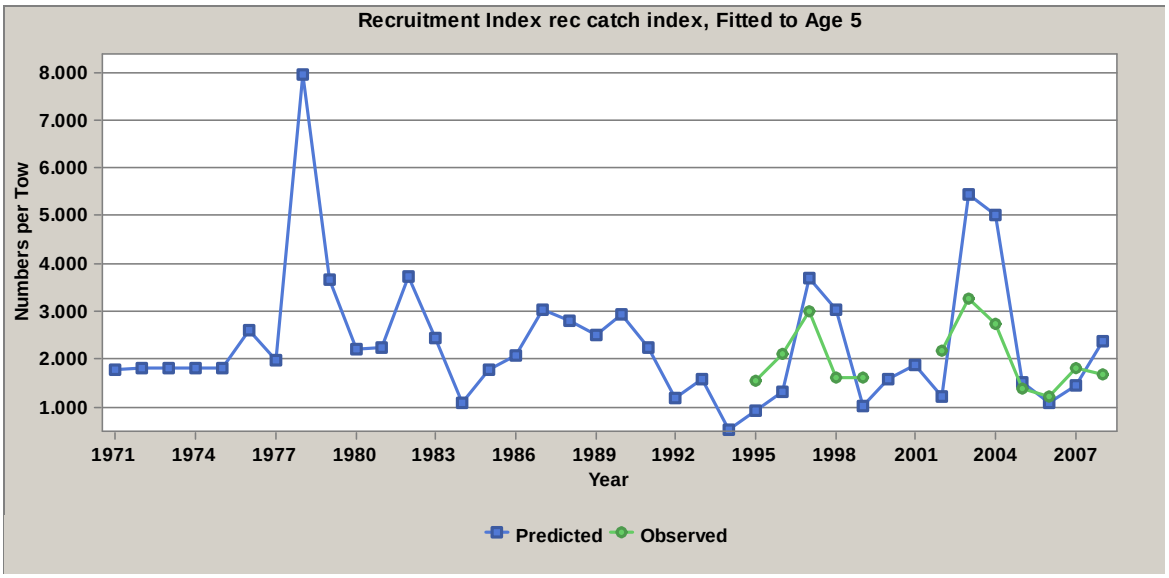


Figure A47. SCALE run 6 was fit to the recruitment index at age 5. The VTR cpue index was applied to the landings proportion at length and 40-50 cm fish were sliced from the index as age 5.

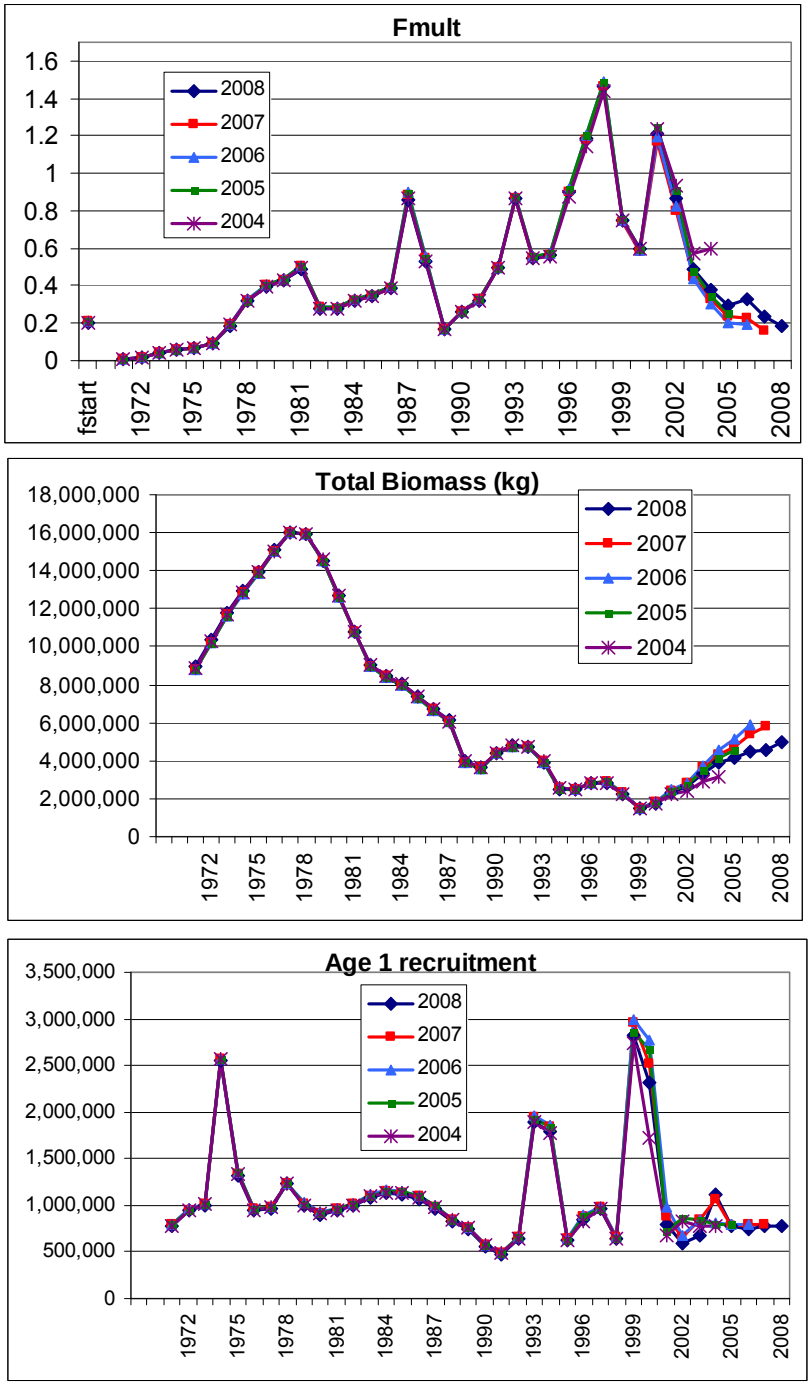


Figure A48. SCALE base run 1 retrospective pattern.

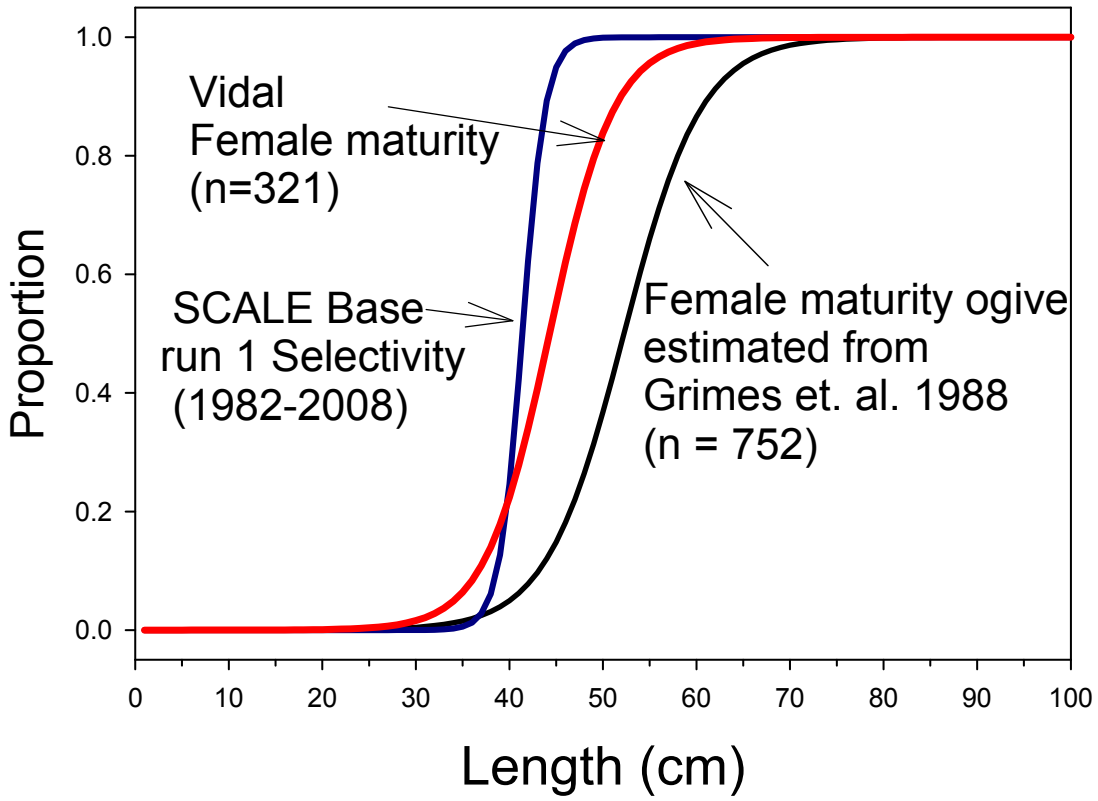


Figure A49. Comparison of SCALE base run 1 selectivity from block 2 (1984-2008), Vidal updated female maturity at length, and Grimes et al (1988) female maturity at length curves.

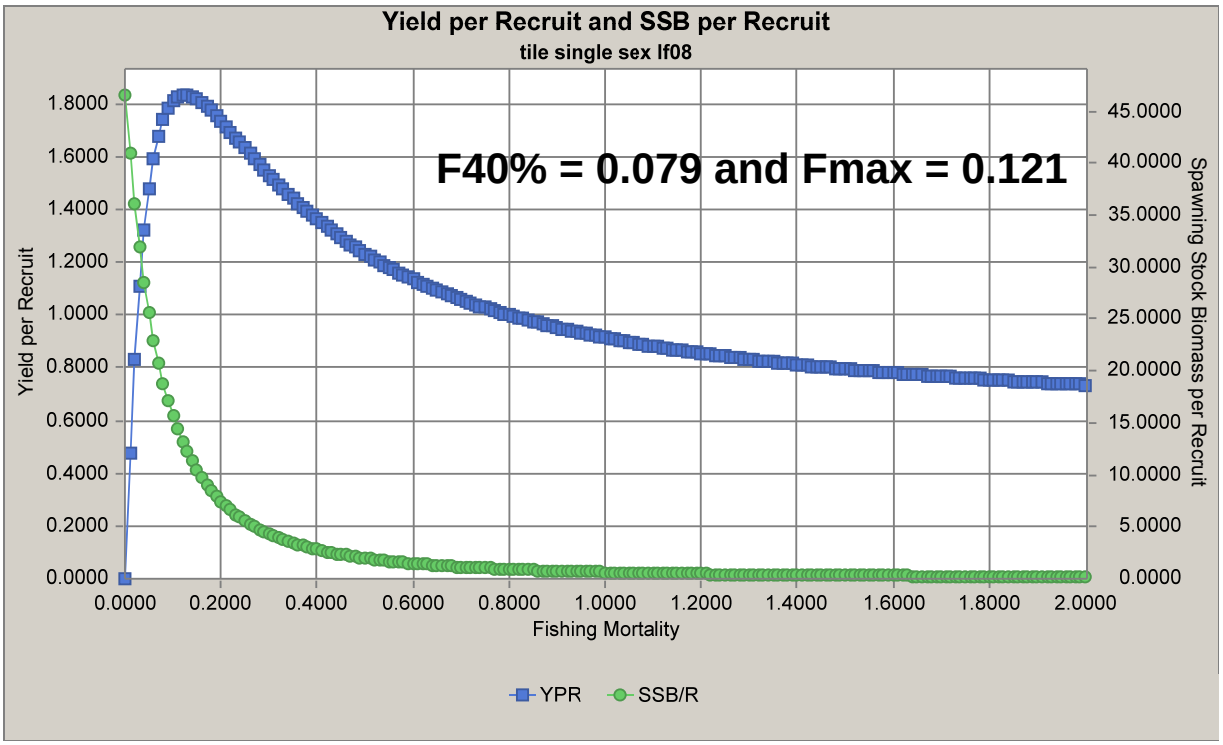


Figure A50. SCALE base run 2 age based YPR and spawners per recruit curves.

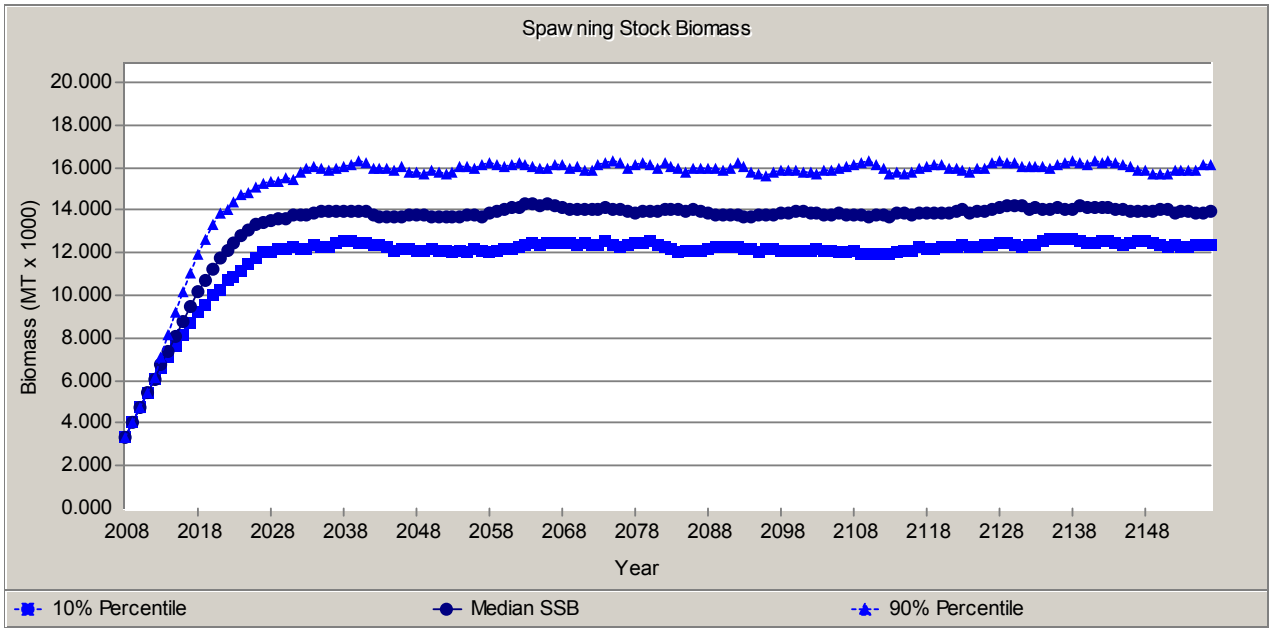


Figure A51. Long term AGEPRO projection at Fmax = 0.121 for run 2 using CDF of recruitment from 1971-2008.

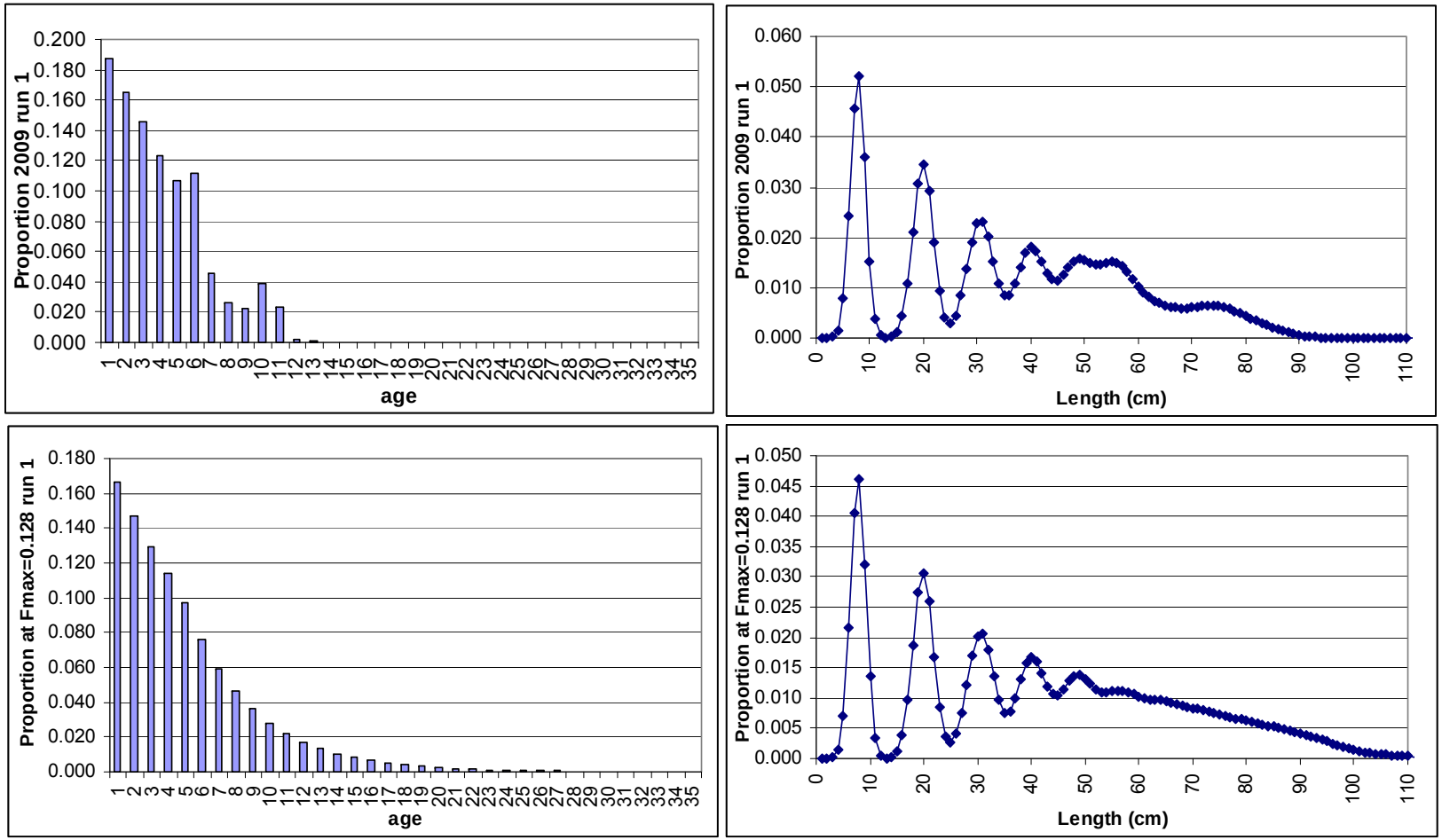


Figure A52. SCALE base run 1 comparison of proportion at length and age in 2009 to Fmax predicted length and age distributions.

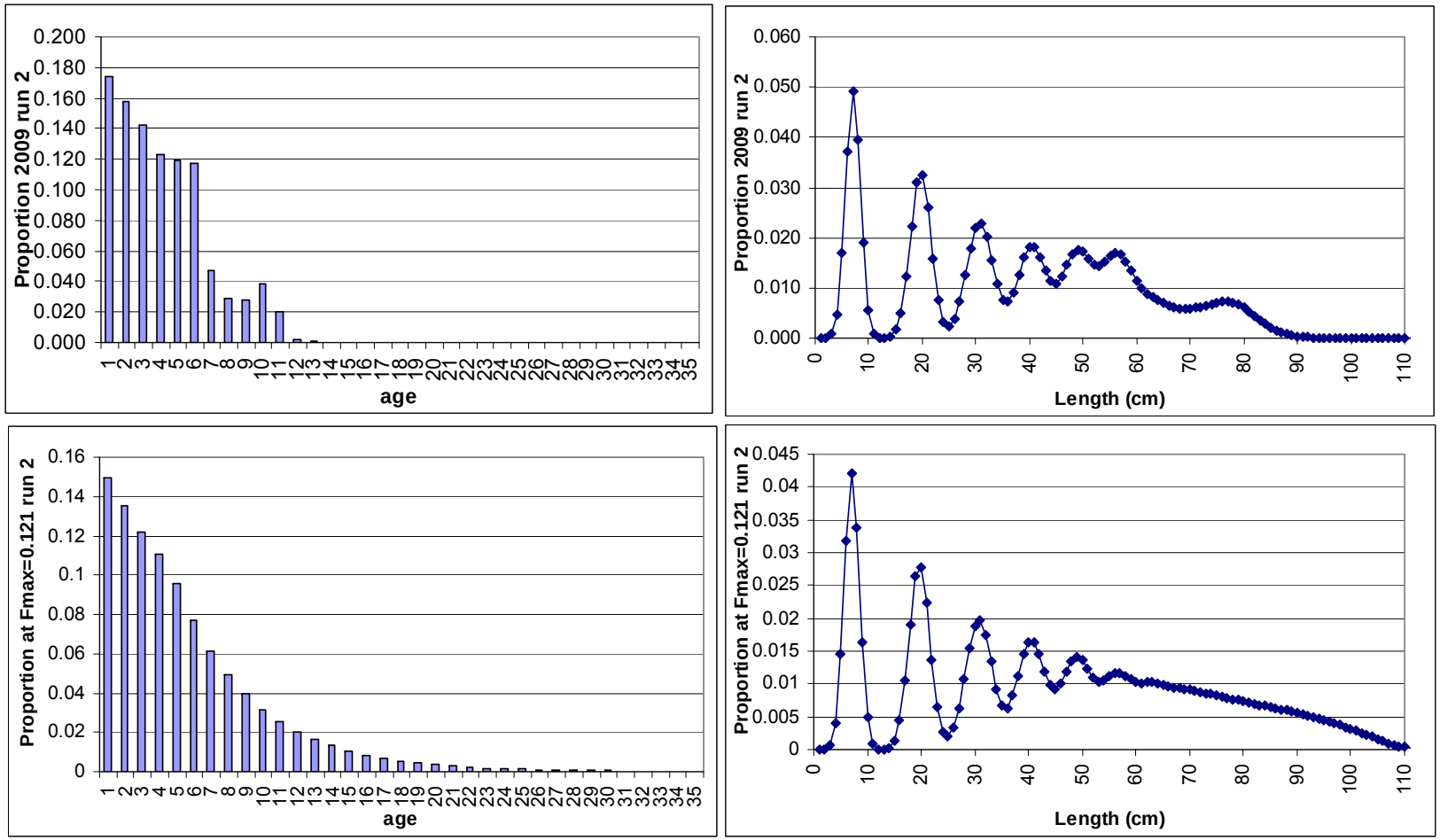


Figure A53. SCALE base run 2 comparison of proportion at length and age in 2009 to Fmax predicted length and age distributions.

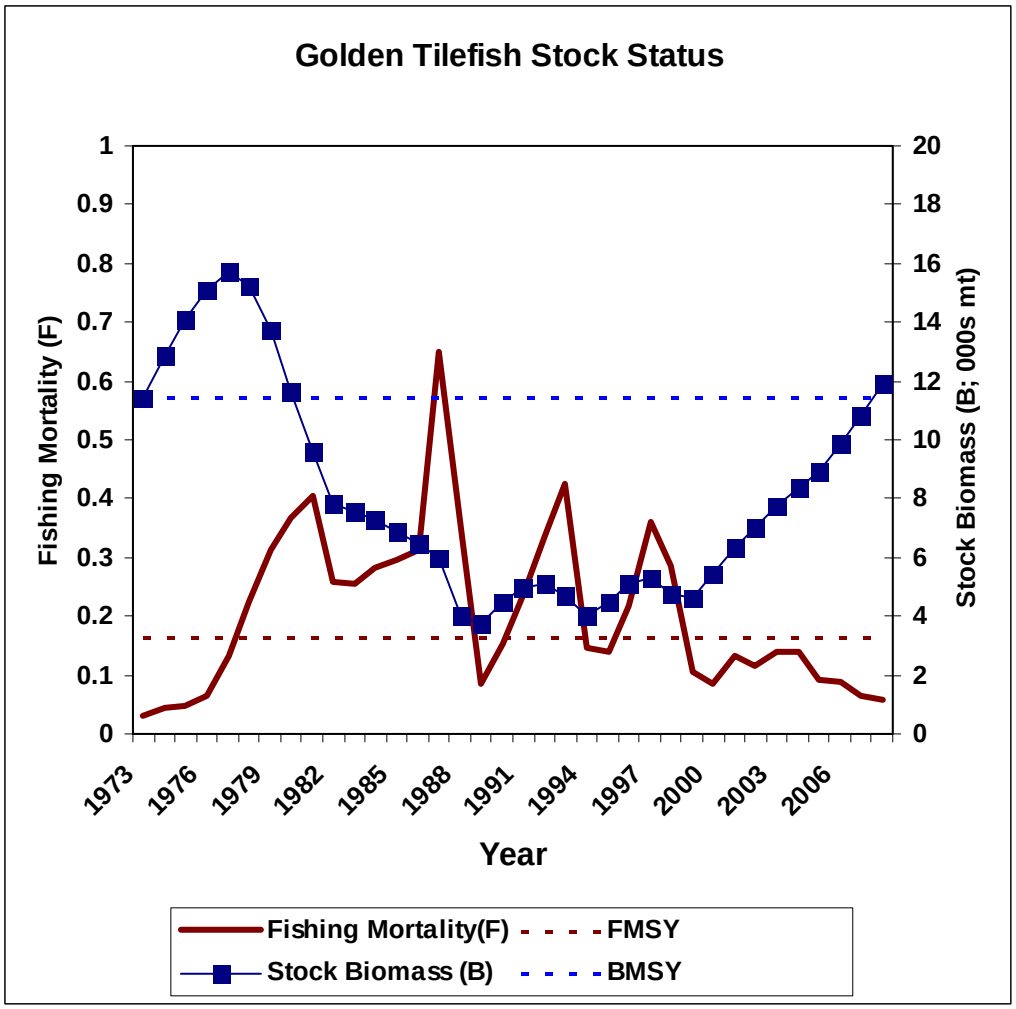


Figure A54. Stock status evaluation for Golden tilefish: 2009 BASE model run.

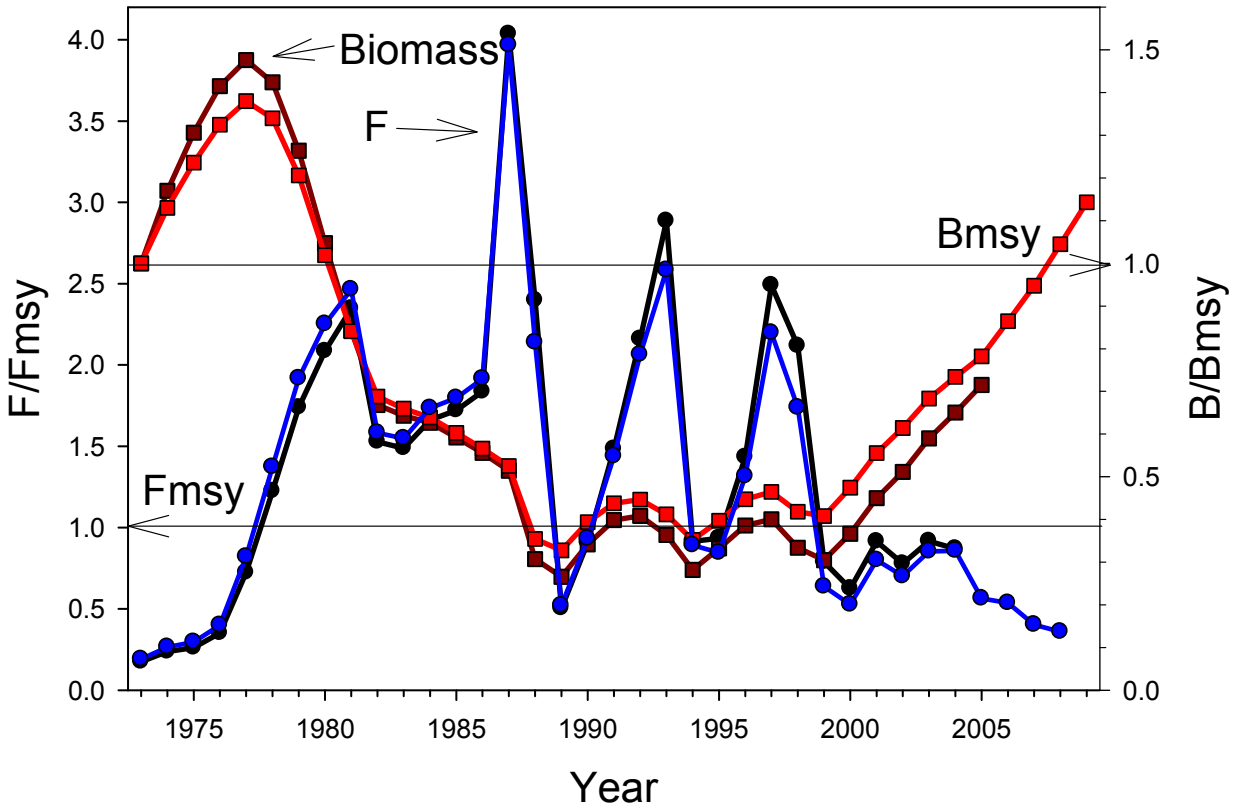


Figure A55. SARC 41 and SARC 48 trends in F/Fmsy and B/ Bmsy ratios for the base ASPIC run which fixed the B1/Bmsy ratio at 1 and used three CPUE series (Turner, Weighout, and VTR).

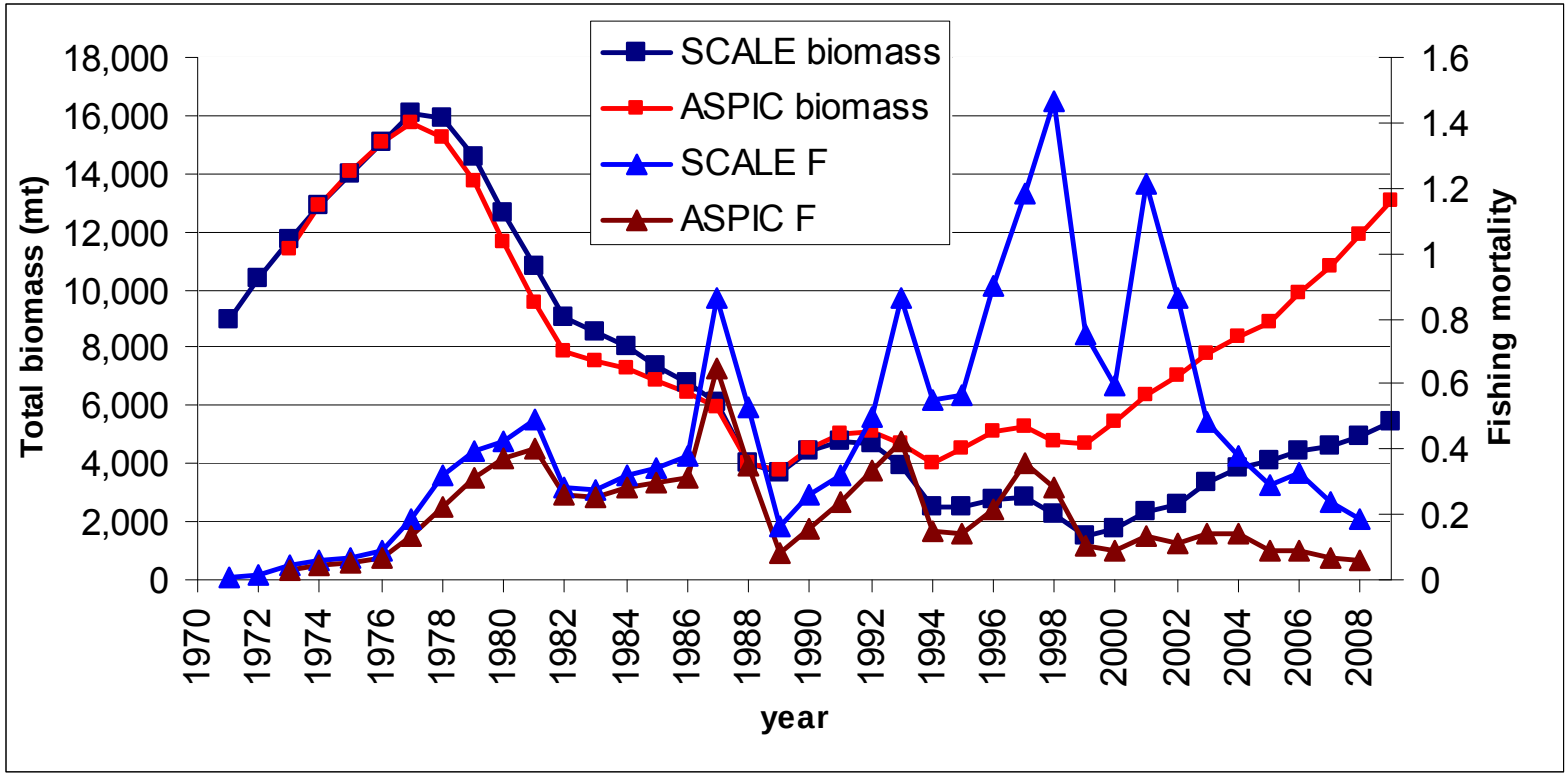


Figure A56. Comparison of F (triangles) and total biomass (squares) between the ASPIC base run 1 with the SCALE base run 1. Note ASPIC base run fixed the biomass in 1973 at Bmsy and SCALE base run estimated Fstart at 0.20.

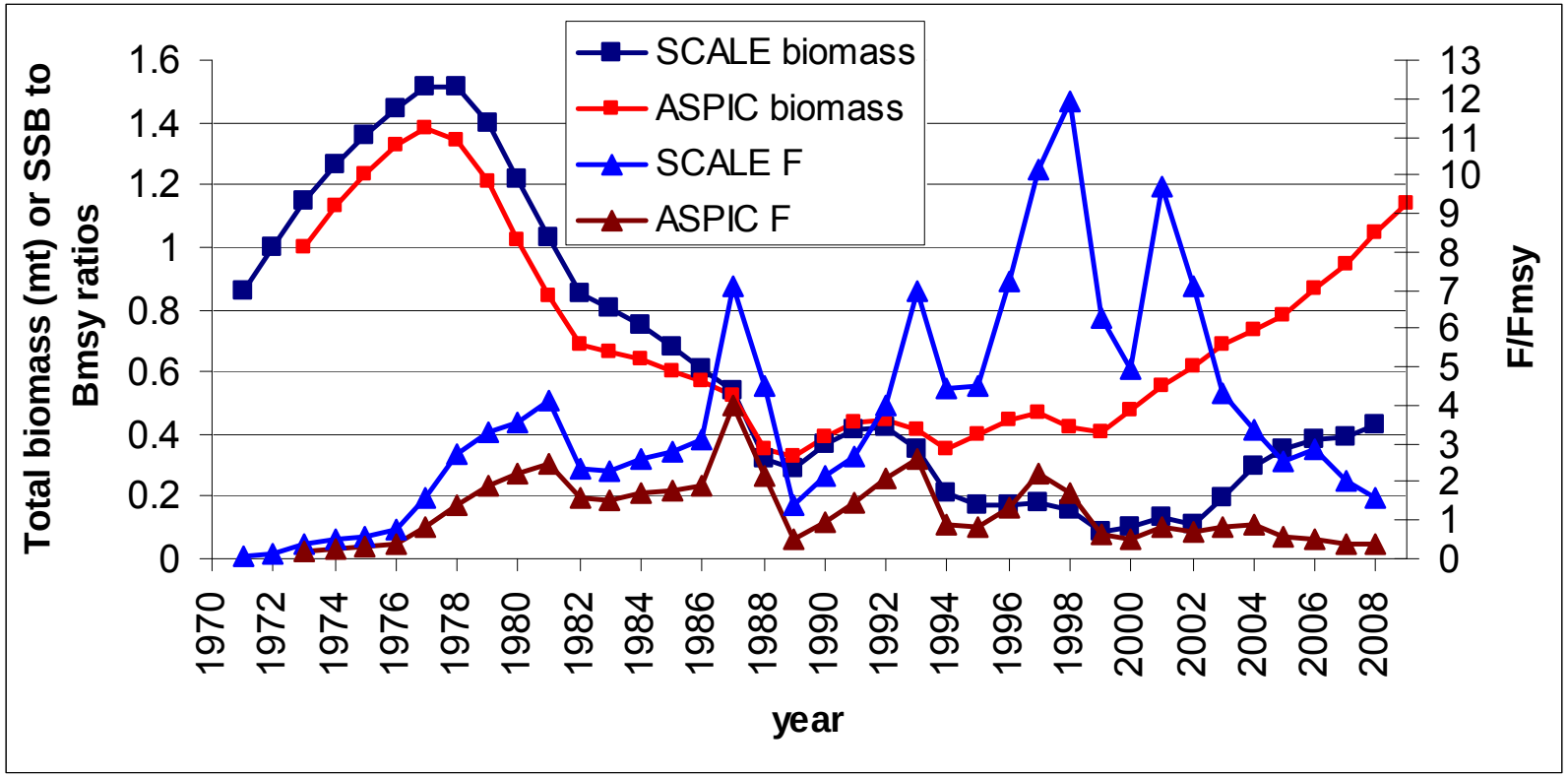


Figure A57. Comparison of F to Fmsy ratio (triangles) and total biomass or SSB to Bmsy ratios (squares) between the ASPIC base run 1 with the SCALE base run 1. Note ASPIC base run fixed the biomass in 1973 at Bmsy and SCALE base run estimated Fstart at 0.20. Fmax (0.128) is used as a proxy for Fmsy and SSBmsy (5,335 mt) is for females only in the SCALE base run 1.

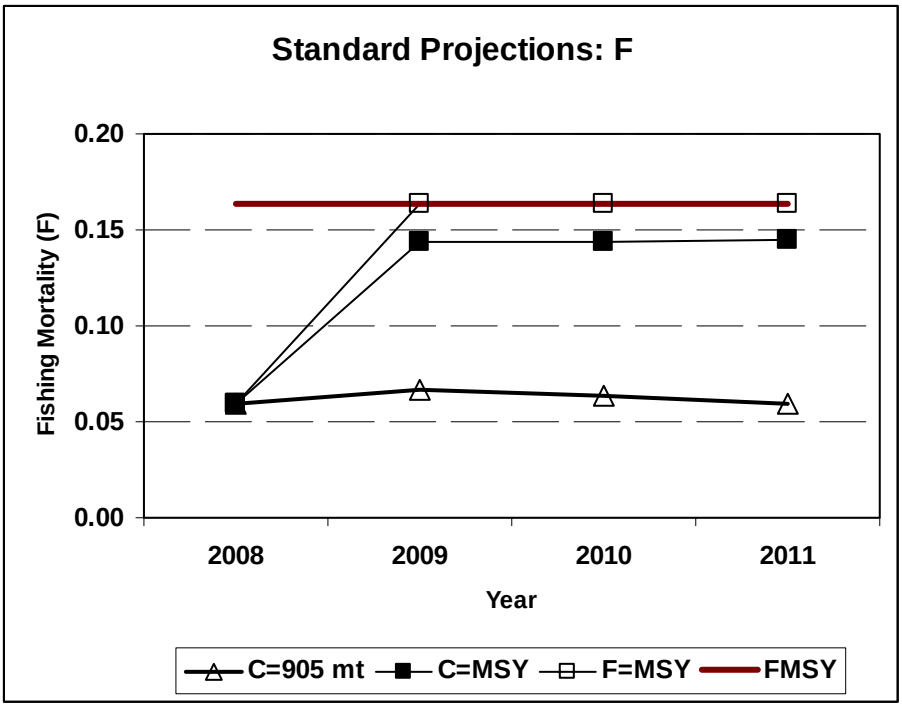


Figure A58. Standard ASPIC projections of fishing mortality (F) for 2009-2011 under alternative assumption for catch (C) or F.

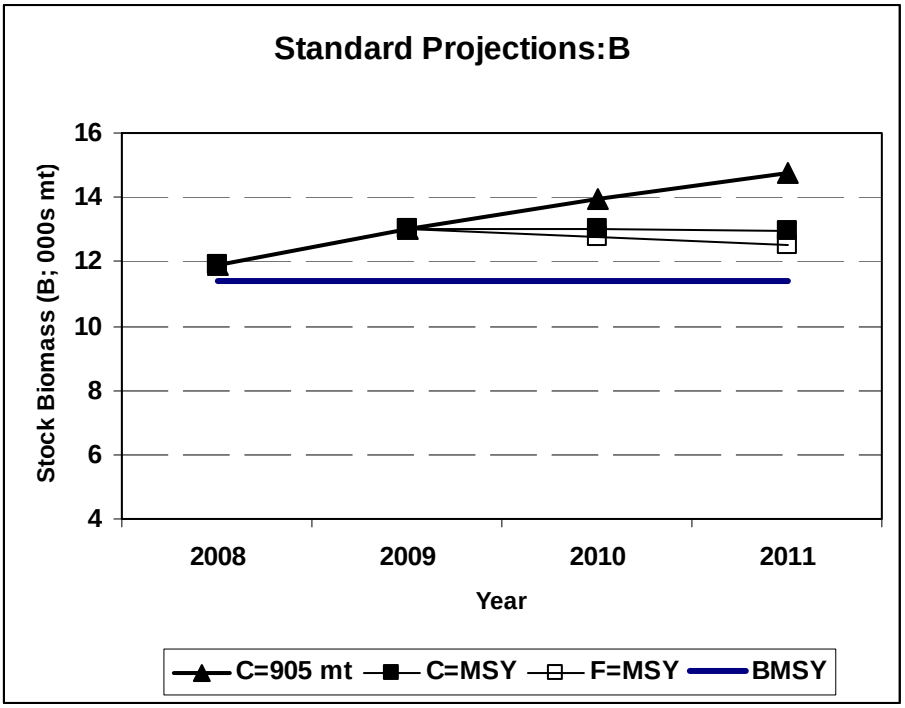


Figure A59. Standard ASPIC projections of stock biomass (B) for 2009-2011 under alternative assumption for catch or F.

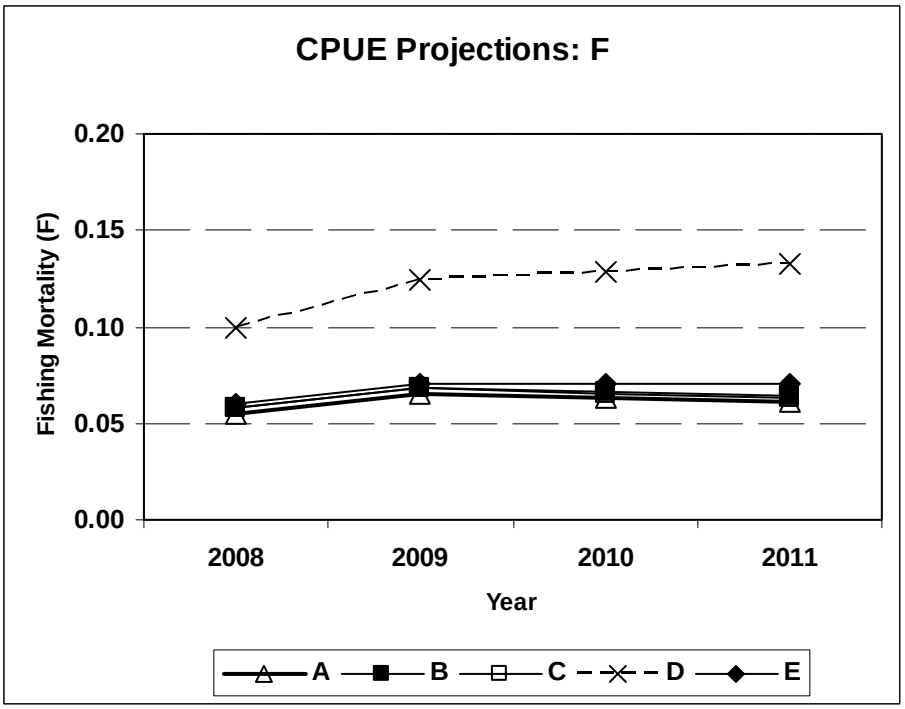


Figure A60. CPUE projections of fishing mortality (F) for 2009-2011 under alternative assumptions for the future trend in fishery VTR CPUE indices (see text).

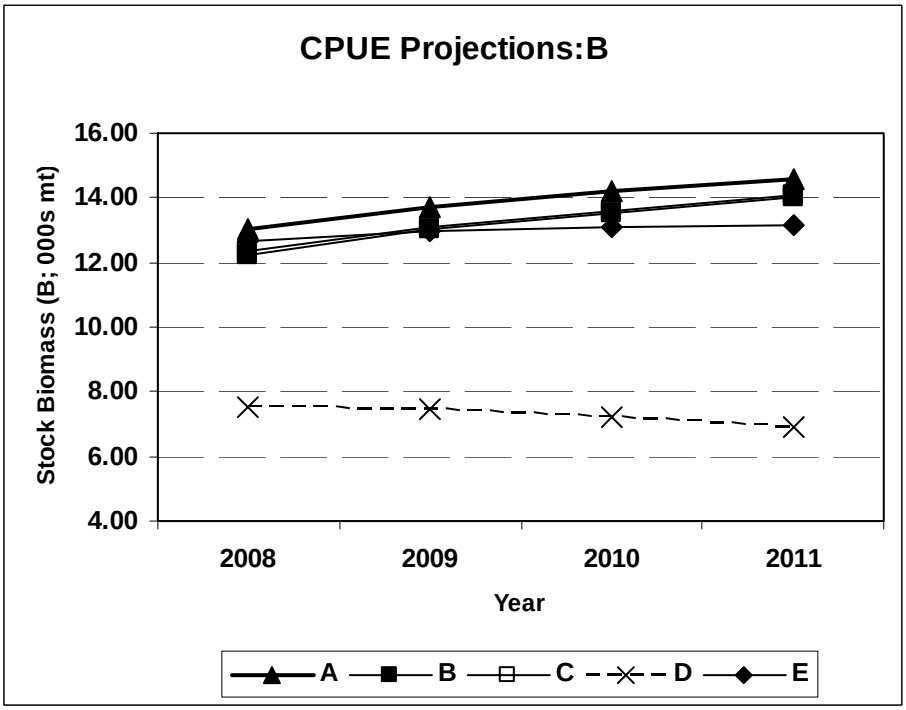


Figure A61. CPUE projections of stock biomass (B) for 2009-2011 under alternative assumptions for the future trend in fishery VTR CPUE indices (see text).

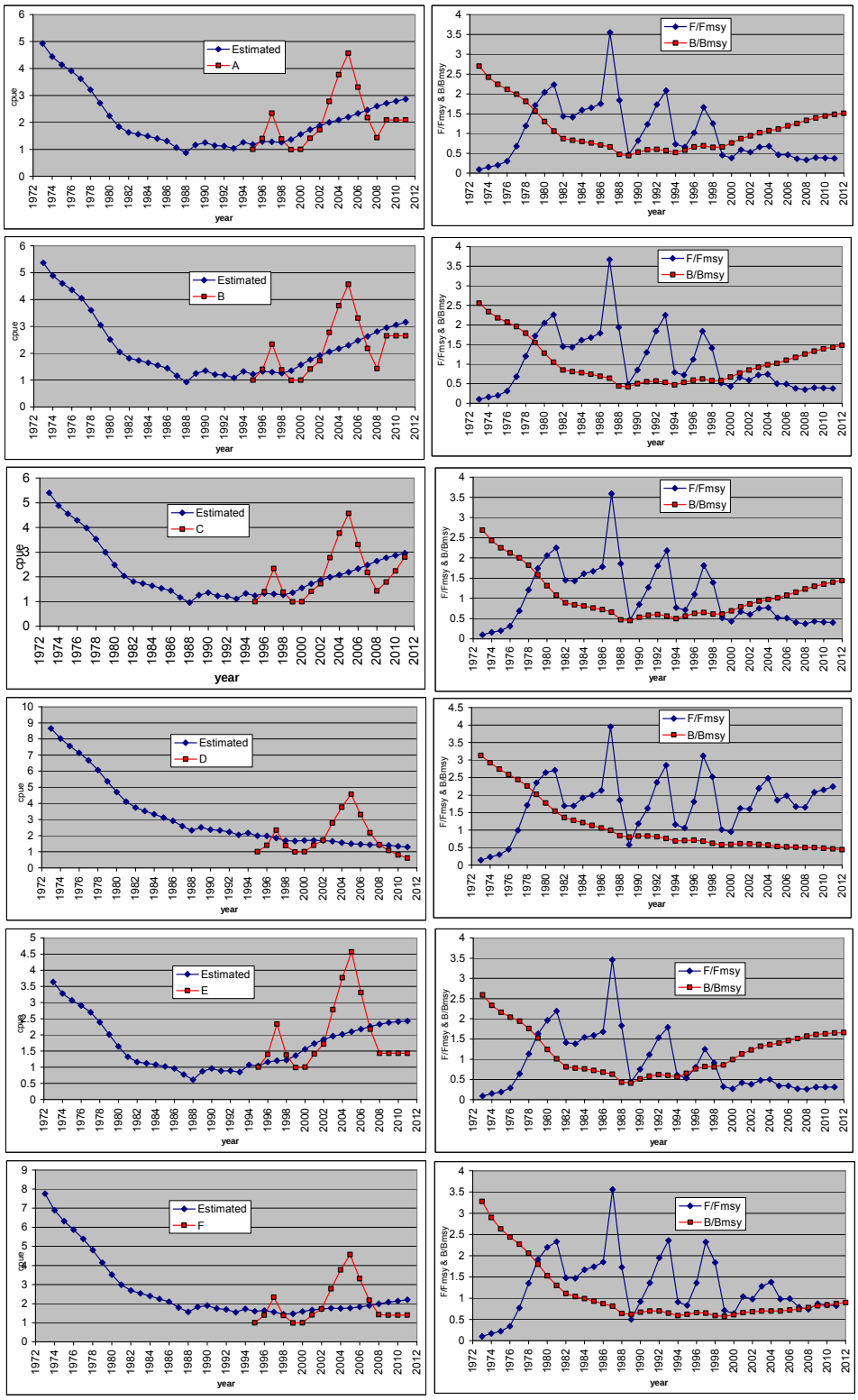


Figure A62. Sensitivity scenarios assuming a constant quota (905 mt) and different cpue estimates from 2009-2011.

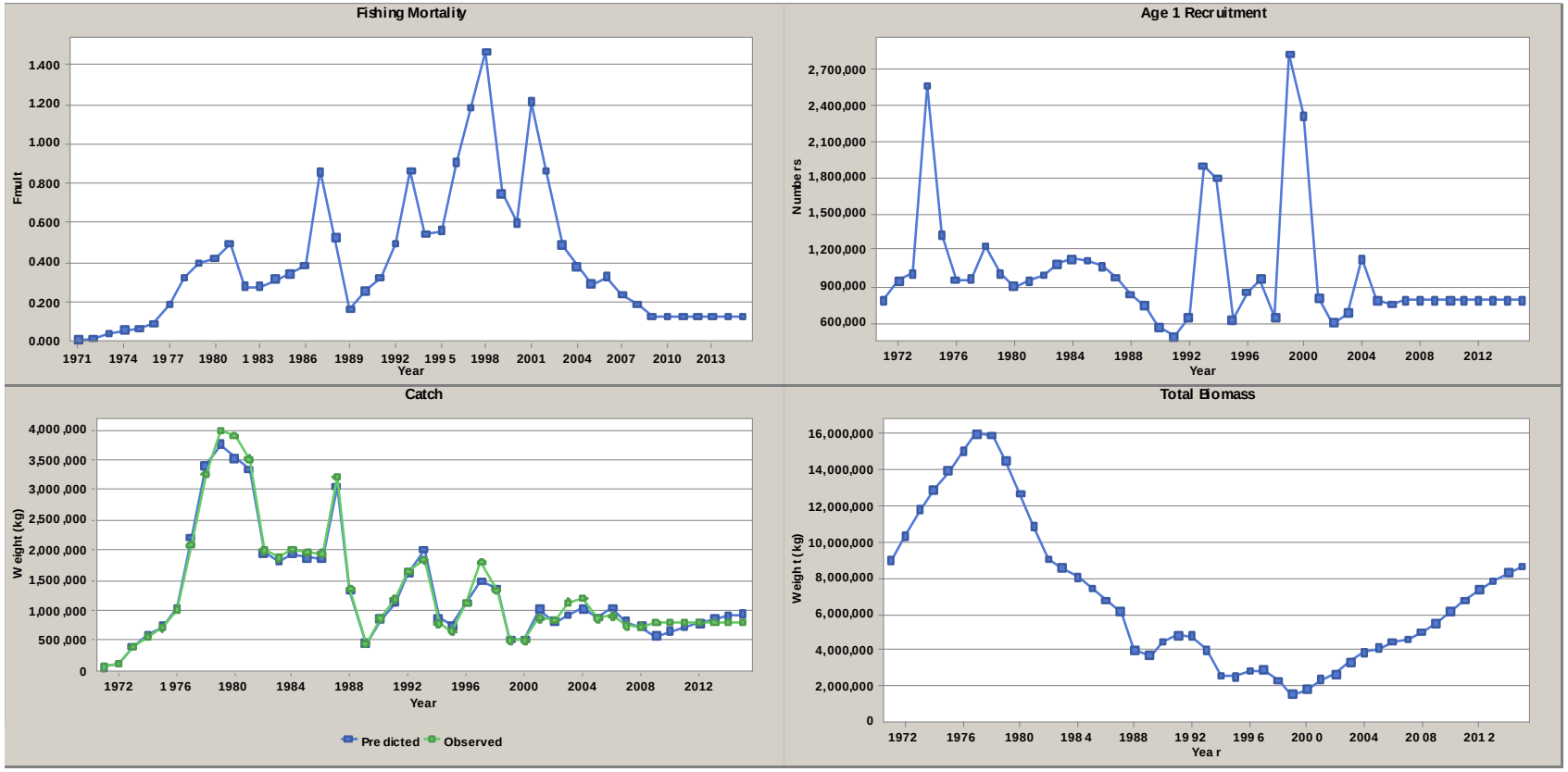


Figure A63. Example of a deterministic SCALE Projection Base run 1 assuming Fmsy=fmax=0.13 from 2009-2015.

Grouped Fmult, Age 1 Recruitment, Observed vs. Predicted Catch Weight, and Total Biomass

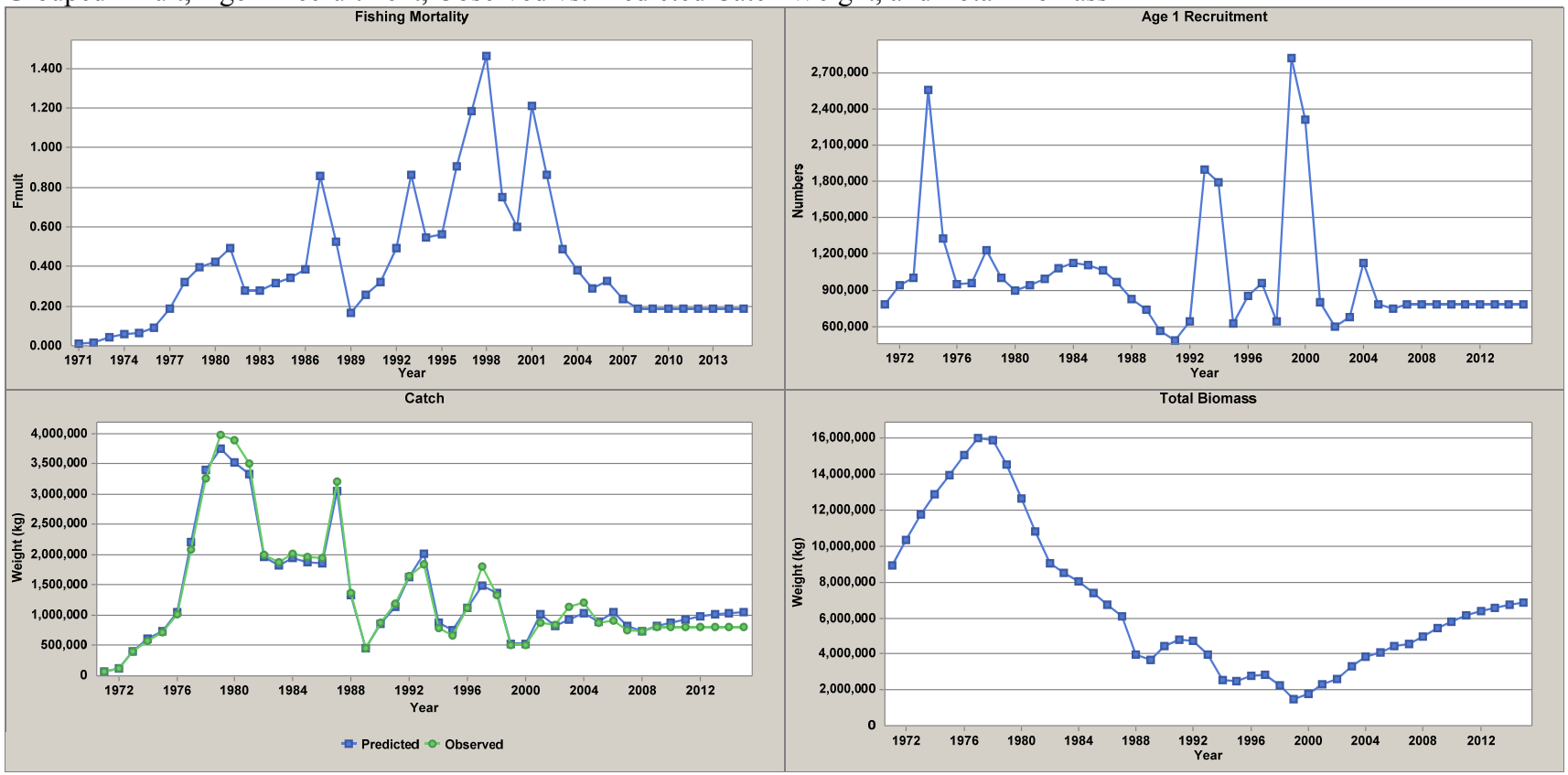


Figure A64. Example of a deterministic SCALE Projection Base run 1 assuming F2008 = 0.19 from 2009-2015.

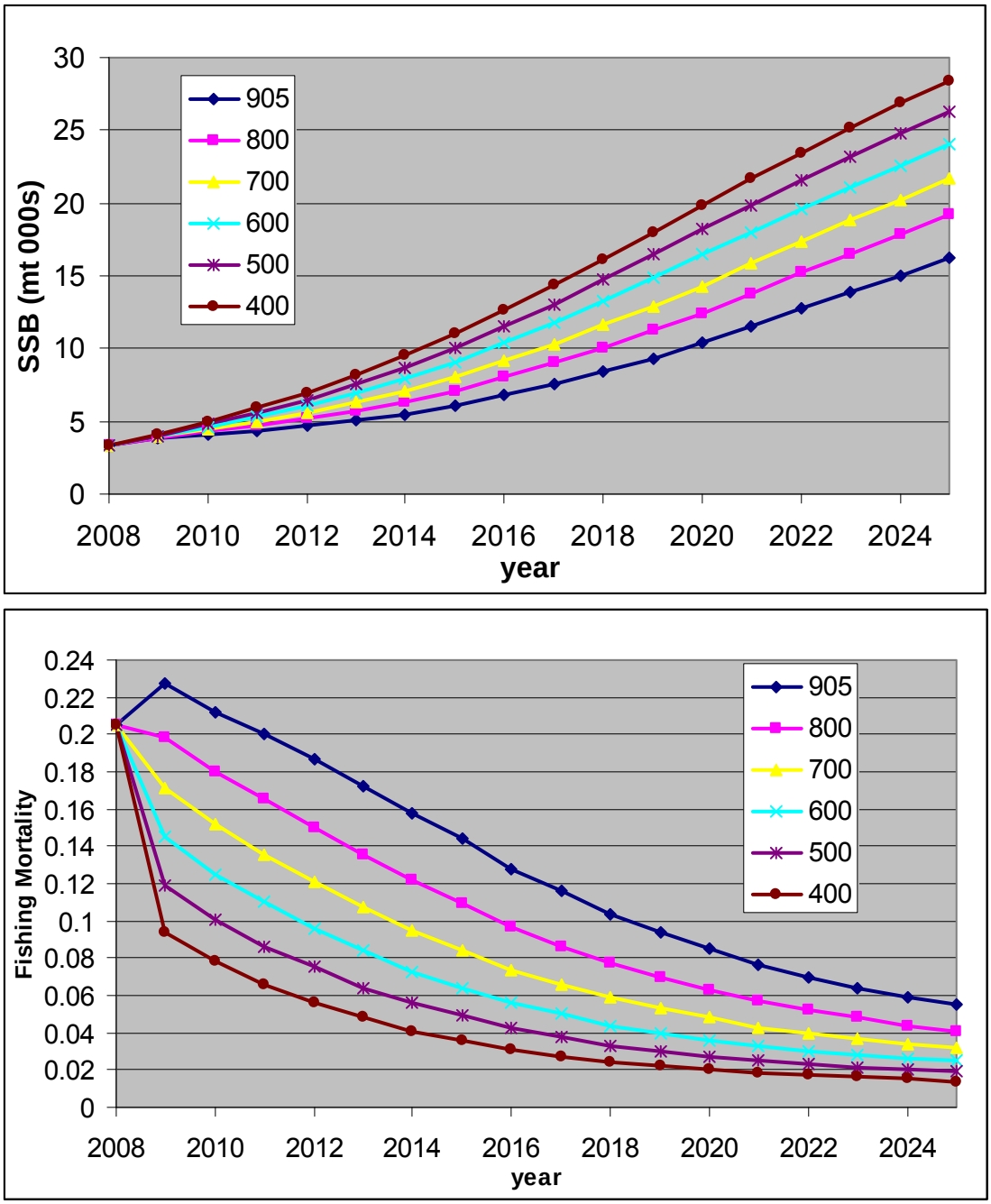


Figure A65. Comparison of SSB and F from Agepro projections for run 2 assuming different constant quotas using the CDF of recruitment from 1971-2008. Note a constant quota no higher than 500 mt is needed to reduce F to Fmax (0.12) in 2009.

Working Paper Appendix A.1
June 1, 2009

An overview of the tilefish data collected through the Northeast Fisheries Science Center's Study Fleet project

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A Working Paper in Support of SAW 48 Terms of Reference A.1

Abstract

The last assessment of golden tilefish, *Lopholatilus chamaeleonticeps*, was based on a surplus production model which utilized a commercial catch per unit effort (CPUE) derived from fishing vessel trip reports (VTRs) as an index of abundance. The 2005 Stock Assessment Review Committee (41st SAW, 2005) concluded that "the effort metric (days absent) in the Weighout and VTR CPUE is a crude measure of effort and could be improved by collecting information (number and size of hooks, length of main line, soak time, time of day, depth fished and area fished) on a haul by haul basis and not by a trip basis." In 2007, the Northeast Fisheries Science Center began a cooperative Study Fleet project with the tilefish industry specifically to address the concerns of the 41st SAW. A brief overview of the program and the data collection protocols is presented along with a general overview of the quality of the data collected by the project to date and a cursory examination of the relationships between haul-based effort metrics and catch. The information is intended to inform the 48th Stock Assessment Review Committee on the types of data available from self-reported haul-by-haul data collection programs. Because of the short time series of these data and data quality concerns, their utility to the current assessment is largely limited to informing the assessment (e.g., accuracy of the days absent effort metric and codification of fishing practices). However, this review serves an important first step in determining whether these types of data can be used in future assessments and whether this, or similar studies, should be extended.

Introduction

The golden tilefish, *Lopholatilus chamaeleonticeps* (hereafter referred to as tilefish), fishery in the Mid-Atlantic region is primarily targeted by a small (< 10 vessels) demersal longline fleet with virtually no observer coverage (Appendix Table A.1.1). Furthermore, this stock lacks a fishery independent index of abundance such that the surplus production model used to assess this stock relies entirely on commercial catch per unit effort (CPUE) derived from fishing vessel trip reports (VTRs) as an index of abundance. The 2005 Stock Assessment Review Committee (41st SAW, 2005) concluded that "…the effort metric (days absent) in the Weighout and VTR CPUE is a crude measure of effort and could be improved by collecting information (number and size of hooks, length of main line, soak time, time of day, depth fished and area fished) on a haul by haul basis and not by a trip basis." Beginning in 2007, the NEFSC began a cooperative Study Fleet project with the tilefish industry specifically to address the concerns of the 41st SAW.

The Northeast Fisheries Science Center (NEFSC) has been operating a Study Fleet Program since 2002. The overall objective of the Study Fleet Program is to assemble a fleet of vessels that are "…capable of providing high resolution (haul-by-haul) self-reported data on catch, effort and environmental conditions while conducting "normal" fishing operations" (Palmer et al. 2007). The Program has been involved in numerous fisheries since 2002 including the groundfish, scallop, hagfish, squid and fluke fisheries. In 2007, four longline vessels which target tilefish for all or part of the year were contracted by the NEFSC to collect fine-scale information on fishing effort and catch. Of the four vessels, two held category A permits (full time) and two held category B permits (part time). The small size of the contracted fleet does restrict how much information can be publically released due to the NEFSC's responsibility to protect vessel confidentiality. The first trip recorded by a tilefish vessel occurred in December 2007 and data collections are currently ongoing. In 2008, the first year of full coverage, 42 trips and 642 hauls were recorded. The trips recorded in 2008 accounted for 237.6 mt of landings, representing 32% of the total annual tilefish landings (736 mt; SAW 48 Working Paper A.1.1). Overall, 52 trips and 702 hauls have been recorded through the Study Fleet Program (*through March 1, 2009*).

Data collection protocols

Electronic logbook

Participating tilefish vessels were equipped with the electronic logbook (ELB) software, Fisheries Logbook Data Recording Software (FLDRS). FLDRS collects all of the information currently collected on paper VTRs, but allows fishermen to record effort and catch information for each haul, rather than aggregated to the subtrip level (i.e., one summary report per gear and area fished). FLDRS can be connected to the vessel's global positioning system (GPS) and depth sounder so vessel captains can capture the date, time, position, statistical area and bottom depth of each haul with the click of the mouse button rather than having to enter this information manually. In addition to basic trip information (vessel, captain, date of sailing, port, etc.) captains were asked to estimate the total length of line and number of hooks hauled (Appendix Figure A.1.1). Because of the complexity associated with the setting behavior of tilefish gear (Appendix Figure A.1.2), captains were asked only to record the hauling activity. For each haul recorded, captains had to provide catch estimates (both retained and discarded). During planning meetings with the industry they had commented that hook competition with other species can negatively impact tilefish catch. In an effort to capture this information captains were also asked to estimate the total number of hooks occupied by non-tilefish species (Appendix Figure A.1.3). On review, the hook competition information appeared incomplete, and was therefore not included in this analysis (in 2008 the number of non-tilefish occupied hooks was only recorded for 331 of 642 hauls). On completion of a trip, captains entered the landings information (date landed port landed, species, amount offloaded, dealer, date sold). Captains were allowed to adjust the landings to reflect the true amount of offloaded catch, such that landings were not affected by hailing errors at the haul-level or by missed hauls during the trip.

GPS polling observations

In addition to the self-reported information, FLDRS was configured to poll the vessel's GPS and depth sounder once every 20 seconds to record fine scale information on vessel cruise paths and bottom topography. These data were stored in a file separate from the trip file and were manually collected by Study Fleet field scientists approximately once per month. By using the ELB entered haul times, it was determined that > 90% of the hauling activity occurs between 3.1 km/hr and 10.2 km/hr, whereas only 12% of non-hauling activity occurs in this speed window (Appendix Figure A.1.4). Plotting fishing tracks in a Geographic Information System (GIS), the hauling vs. non-hauling activity could be differentiated with manual post-processing and used to validate the ELB recorded information (Appendix Figure A.1.5). Of the 42 trips recorded in the ELB in 2008, 36 had GPS polling coverage. Failure of the ELB to communicate with the GPS was the primary reason why GPS polling data were unavailable for a particular trip.

Field scientist observations

NEFSC field scientists were present on four of the ELB-recorded trips (total of 51 hauls). The objectives of the field scientists were to: a) provide independent estimates of tilefish catch; and, b) collect biological samples (e.g., length, weight and age) from the tilefish catch. Field scientists did not observe all hauls during a trip nor did they record observations on the amount of fishing effort (e.g., mainline length, number of hooks, bottom depth). Field scientist information can only be used to assess the accuracy of catch estimates and provide biological information on the resulting catch.

Data quality

Overview

The ELB data collected by the tilefish vessels have not previously been analyzed. This analysis represents the first assessment of the quality and utility of these data. It is a critical first step to determine the overall quality of these data and understand how the quality of both the self-reported and electronically recorded (i.e., by GPS and depth sounder) impact their utility for future tilefish stock assessments. Because of the short time series of these data, their utility to the current assessment is largely limited to informing the assessment (e.g., accuracy of the days absent effort metric and codification of fishing practices). However, this review serves an important first step in determining whether these types of data can be used in future assessments and whether this, or similar studies, should be extended. Data quality analyses focused on the quality of the self reported effort metrics (number of hauls, mainline length, number of hooks, soak duration, and fishing depth) and catch estimates.

Effort metrics were primarily validated by comparing the self-reported estimates to estimates obtained from post-processing of the GPS polling information. The post-processing step is an extremely time consuming process taking approximately 4-8 staff hours per trip file depending on the length of the trip and spatial density of the fishing patterns. Due to the time intensive nature of this activity, only 23 of the 36 trips with GPS polling information were post-processed. Unfortunately, all of these trips were from a single vessel so the results of the data quality analysis should not be overly interpreted as indicative of all of the self-reported data. Because of the limited applicability of these data, no statistical tests were performed.

Number of hauls per trip

During preliminary review of the tilefish data it was observed that the sum of individual catches was often much less than the total landings (Appendix Figure A.1.6). This could indicate that either the individual haul hail estimates were consistently low, or not all hauls were recorded in the ELB. Follow-up conversations with vessel captains suggested that the greatest contributor to these discrepancies was missing hauls. Comparison of the number of self-reported hauls per trip to the number estimated from the GPS indicated that hauls do occasionally go unreported in the logbook (Appendix Figure A.1.7). Of the 23 trips examined there was complete agreement in the haul counts on eight trips and no instances of the ELB recording more hauls compared to the GPS analysis. The degree of underestimation in the ELB was variably, but generally less than 5 hauls per trip.

Mainline length hauled

Mainline length was determined from the GPS polling data by calculating the cumulative haversine distance (Sinnott 1984) of all points between the start and ending points of a haul. In general, the ELB estimated mainline length hauled agreed reasonably well with the GPS calculated mainline length, though there was considerable variability and the numerous outliers (Appendix Figure A.1.8).

Number of hooks hauled

There was no way to directly validate the number of hooks self-reported on the ELB,

however by comparing these estimates to the GPS calculated mainline length a general understanding of the accuracy of these estimates can be obtained. However, the variability observed in the relationship will be contingent on the accuracy of the self-reported data and the setting hook density (number of hooks per km of line set). There is general agreement between ELB hooks hauled and the GPS calculated mainline length (Appendix Figure A.1.9); however, there is greater spread in the relationship compared to the ELB mainline to GPS mainline comparisons.

Soak duration

GPS soak duration was calculated as the average of the soak durations (time difference between when a particular section of gear was set and when the same section was hauled) from five observations taken along the length of the haul. The soak duration associated with the start haul and end haul was always taken and the intent was that the remaining three observations would be equally spaced out across the haul. The average soak duration and standard deviation were calculated for each haul. The ELB estimates of soak duration were generally higher than those calculated from the GPS polling files (Appendix Figure A.1.10). In conversations with the vessel owners, it could be that this difference is partly attributable to the fact that vessel captains calculate soak duration differently (difference between when the last piece of gear was set and when the last piece of gear was hauled).

There was an interesting trend in the relationship of the standard deviation to the average soak duration (Appendix Figure A.1.11). Two different trends are present, one representing efforts where the gear was hauled in the same direction it was set in (lower ratio of variability to average soak duration), and the other when gear were hauled in the opposite direction from which they were set (higher ratio of variability to average soak duration).

Fishing depth

Because tilefish are caught with bottom tending gear, the fishing depth is the bottom depth. Average fishing depth was calculated from the GPS polling file by calculating the average bottom depth between the start of the haul and the end of the haul. The ELB estimates of bottom depth agreed well with the GPS calculated values, though several outliers exist (Appendix Figure A.1.12).

Catch estimates

ELB-reported catch estimates were compared to the catch estimates recorded by the Study Fleet field scientists. The haul-by-haul difference in reported tilefish catch was generally similar with the median centered near 0 and the spread uniform about the median (Appendix Figure A.1.13). There were three hauls where the ELB estimates were considerably higher than the estimates of the field scientists.

Data quality conclusions

Overall, the self-reported ELB data examined did track the general trends derived from alternate sources (GPS/depth sounder or field scientists). While these conclusions are based on a small subset that was generally limited to a single vessel, they do suggest that the overall quality of the self-reported data are sufficient for use in making general inferences about catch relationships and trends.

Use of VTR days absent as a proxy for fishing effort

The 41st SAW (2005) characterized days absent as calculated from the VTR as a "…a crude measure of effort". The availability of more precise and more accurate (particularly when derived

from GPS observations) allows the inaccuracy of VTR days absent to be assessed. There are two fundamental questions: 1) does VTR days absent minus one accurately reflect the amount of time spent on the fishing grounds?; and, 2) does this metric track well with alternate effort metrics such as the amount of mainline length fished?

To evaluate the first question, the GPS data were used to determine the total amount of days the vessel spent on the fishing grounds and compare this to the VTR days absent minus one metric. The agreement between the two was highly significant (Appendix Figure A.1.14; $n = 23$, $r = 0.937$, $p < 0.0001$) indicating that the VTR days absent minus one metric accurately reflects the true time spent fishing. When comparing these two metrics to the GPS estimated mainline length fished, the GPS days fished explains a greater degree of the variability in the mainline length hauled ($r^2 = 0.73$) compared to the VTR days absent minus one metric ($r^2 = 0.52$). These results suggests that while the VTR days absent metric accurately reflects the time spent on the fishing grounds and explains some of the variability in mainline length hauled, more precise metrics may offer improvements over the current metric used in the surplus production model.

Catch relationships as a basis for alternate CPUE estimates

SAW 41 (2005) stated that "…the effort metric [used in calculating CPUE]…could be improved by collecting information (number and size of hooks, length of main line, soak time, time of day, depth fished and area fished) on a haul by haul basis." We've taken an exploratory look at the relationship between these alternate haul-based determinants of tilefish catch. Based on the relative accuracy of the self-reported ELB data all recorded haul records (702 hauls recorded between December 1, 2007 and March 1, 2009) were used in these comparisons. The effort metrics examined here are: mainline length, number of hooks, hook density (hooks/km), soak duration, depth and latitude fished. There is a high degree of multicollinearity among these variables which is expected, particularly among those effort metrics that are closely related such as mainline length and number of hooks (Appendix Table A.1.2).

Catch appears most closely related to the number of hooks fished (Appendix Figure A.1.16), with a weaker relationship to the mainline length (Appendix Figure A.1.17), though because of the collinearity between number of hooks and mainline length, it is unclear if this is direct relationship. Interestingly, there is no linear relationship between catch and hook density (Appendix Figure A.1.18); the highest catch rates occur between 200 and 300 hooks/km, but catch rates are lower at densities outside this range. There a weak linear relationship of catch to soak duration (Appendix Figure A.1.19), but again, because of the collinearity of soak duration to both number of hooks and mainline length it is impossible to determine if soak duration is a determinant of catch. There is no linear relationship between catch and depth (Appendix Figure A.1. 20) or latitude (Appendix Figure A.1.21), however catches do appear to be lower at greater depths and lower latitudes. The interpretation of these results is difficult because vessel tended to fish in shallower depths at higher latitudes (Appendix Figure A.1.22).

The length frequency information collected by the field scientists was cursorily examined for trends with respect to depth (Appendix Figure A.1.23) and latitude (Appendix Figure A.1.24). There were significant relationships of size to both of these variables, with latitude explaining a greater degree of the variability in tilefish fork length.

Catch trends over time

Based on the relative strength of the relationship between catch and the number of hooks fished, a CPUE metric was constructed as the catch (live wt. kg) per hook hauled. CPUEs observed in this time series ranged from 0.0 to 1.0 kg/hook. Three different CPUEs trends were examined; 1)

using all data across the time series fit with a loess smoother (Appendix Figure A.1.25); 2) using only hauls occurring within a 40 minute square region in the vicinity of Hudson Canyon (Appendix Figure A.1.26); and, 3) using only hauls occurring within a 40 minute square region in the vicinity of Block Canyon (Appendix Figure A.1.26). The area in the vicinity of Hudson Canyon was the most heavily fished area for the duration of the time series, with the Block Canyon region being the second most heavily exploited area. While there is some evidence of declining CPUE in each of the time series, the data are insufficient to draw any conclusions, as the trends are driven by high catches early in the time series and may associated with seasonal effects or some other unknown effect.

Conclusions

The information presented in this working paper is intended to inform the 48th Stock Assessment Review Committee on the types of data available from Study Fleet-like projects focusing on the collection of self-reported haul-by-haul information. The data quality is sufficient to detect relationships and perhaps general trends, but the overall quality of the data can be improved. It should be noted that many of the vessels in the tilefish fleet utilize multiple captains, which increases the time period necessary to familiarize one self with the electronic logbook and data collection protocols. Through closer collaboration with the tilefish industry the quality of these data are likely to improve. Because of the quality of these data, more in depth analyses were not performed, however the results do indicate that the current VTR days absent effort metric does provide a reasonable measure of fishing effort, but that it could be improved on by collecting information at a finer scale.

Acknowledgements

Funding for this project was provided by the NEFSC's Cooperative Research Partners Program. Administrative support was provided by Dr. John Hoey and Carolyn Woodhead. Lauren Marcinkiewicz and Tiffany Vidal provided valuable assistance in the field. This project relied heavily on the NEFSC's information technologies capabilities; a debt of gratitude is owed to Heidi Maraotta, Holly McBride, David Hiltz, Bobbi North and Joan Palmer. Most importantly the success of the project was predicated on the participation and cooperative of the owners, captains and crews of the participating tilefish vessels.

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Tables

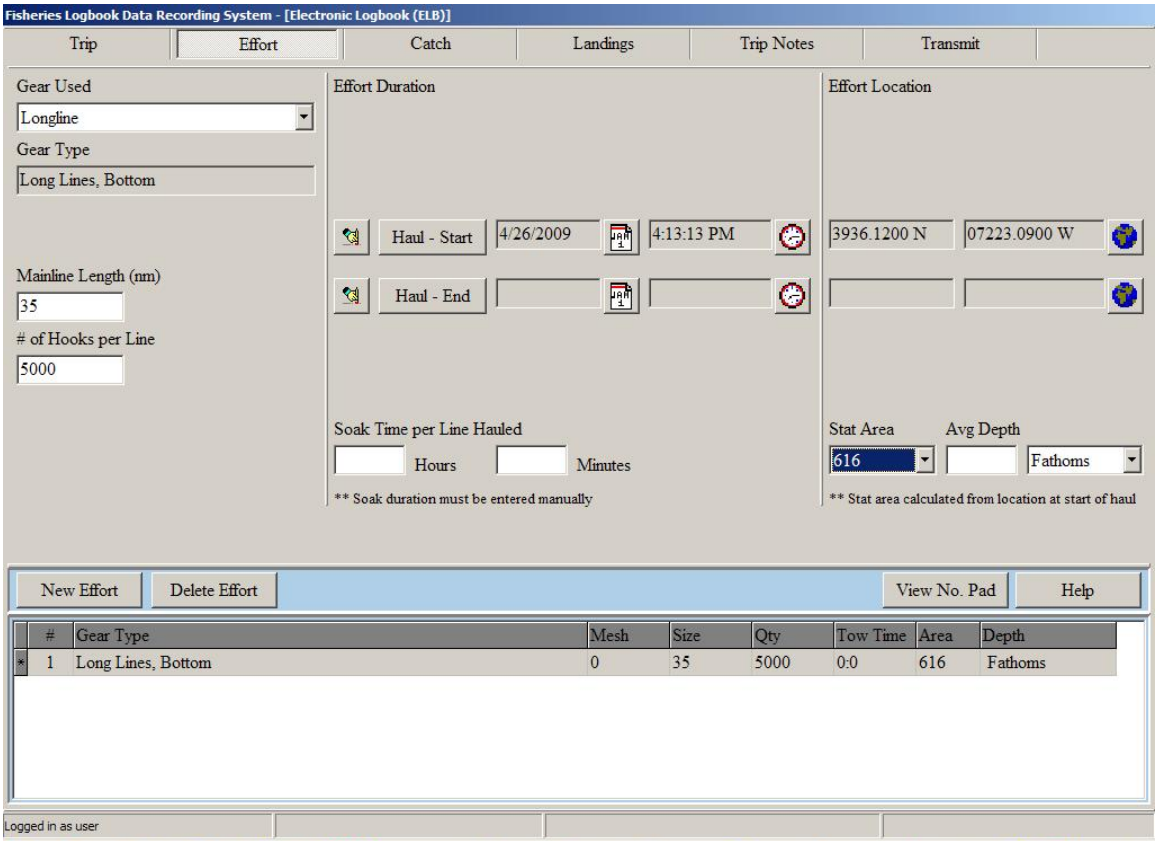
Appendix Table A.1.1. Number of directed tilefish trips (longline gear only) observed by the Northeast Fisheries Observer Program by year.

Year	Number of directed tilefish trips observed (*longline gear only*)
1992	1
2004	1
2005	4
2006	4
2007	2
2008	1

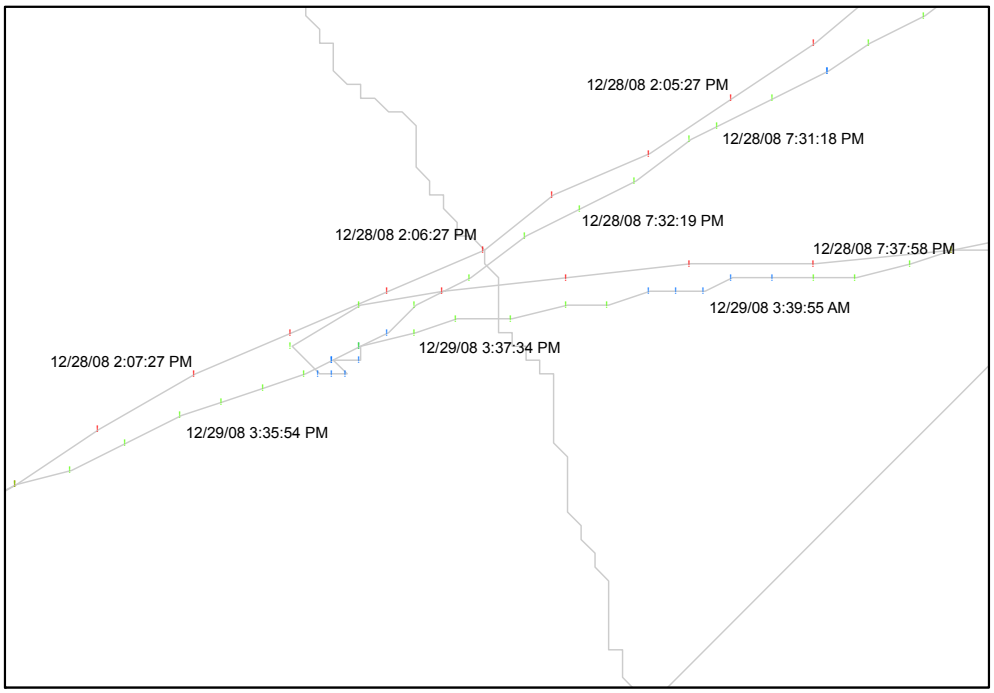
Appendix Table A.1.2. Correlation matrix of tilefish catch and effort metrics from data reported by captains using the electronic logbook. Relationships significant at the $p < 0.05$ are shown in bold.

	Tilefish catch (live wt. kg)	Mainline length (km)	Number of hooks	Hook density (hooks/km)	Soak duration (hours)	Bottom depth (m)
Mainline length (km)	**0.589** ***(<0.0001)***					
Number of hooks	**0.607** ***(<0.0001)***	**0.819** ***(<0.0001)***				
Hook density (hooks/km)	-0.053 *(0.158)*	**-0.308** ***(<0.0001)***	**0.208** ***(<0.0001)***			
Soak duration (hours)	**0.447** ***(<0.0001)***	**0.638** ***(<0.0001)***	**0.604** ***(<0.0001)***	0.017 (0.654)		
Bottom depth (m)	-0.060 *(0.115)*	-0.061 *(0.107)*	-0.066 *(0.083)*	0.008 *(0.832)*	-0.011 *(0.772)*	
Latitude (dd)	-0.049 *(0.1972)*	0.008 *(0.8229)*	**0.094** ***(0.0123)***	**0.123** ***(0.0011)***	**-0.189** ***(<0.0001)***	**-0.361** ***(<0.0001)***

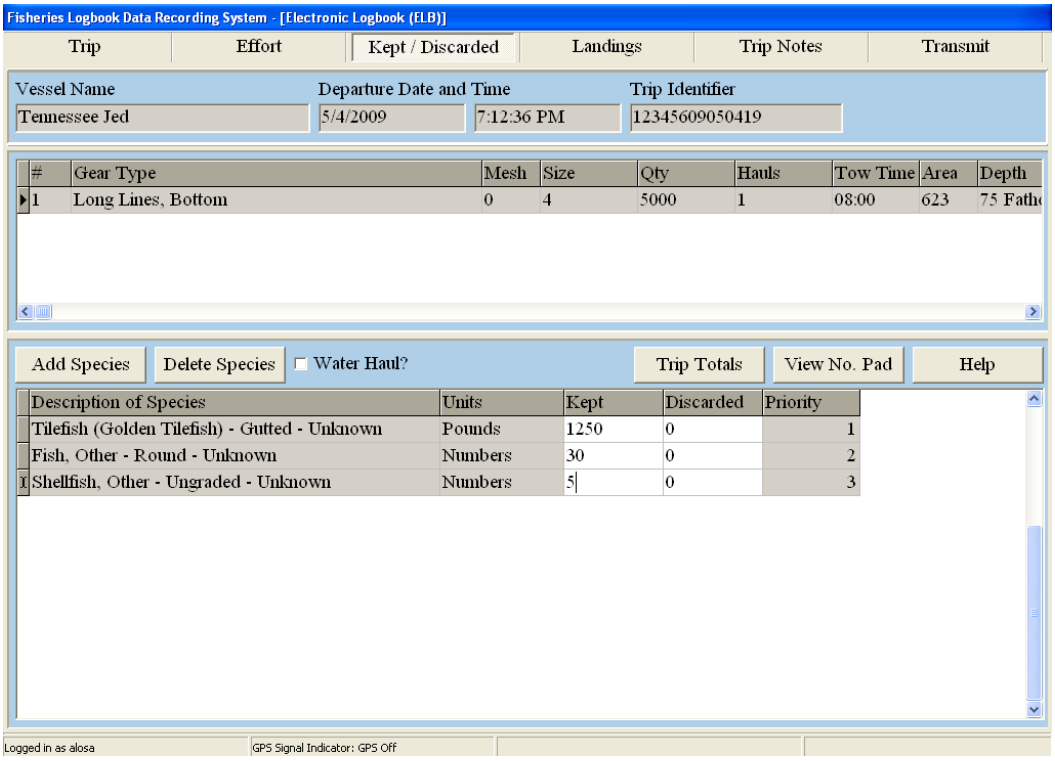
Figures



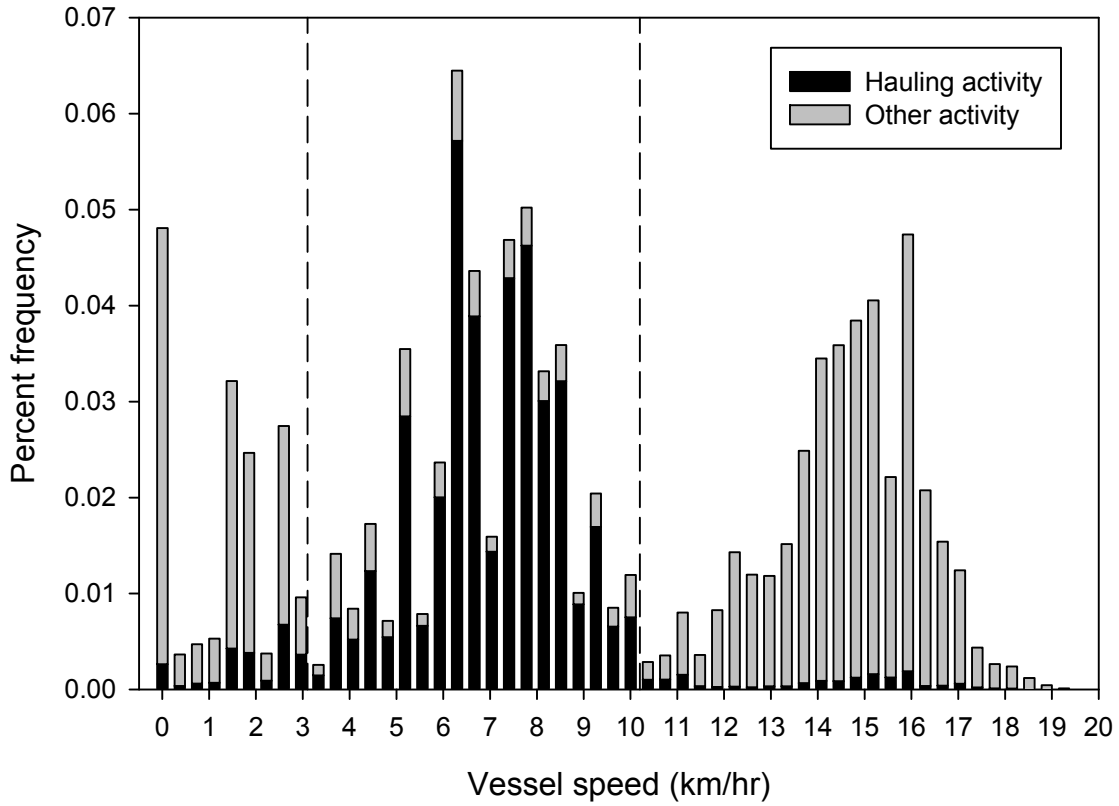
Appendix Figure A.1.1. A screen shot of the Fisheries Logbook Data Recording Software (FLDRS) effort data entry screen. This screen shot is similar to that used by tilefish vessel captains to record information on the gear hauled.



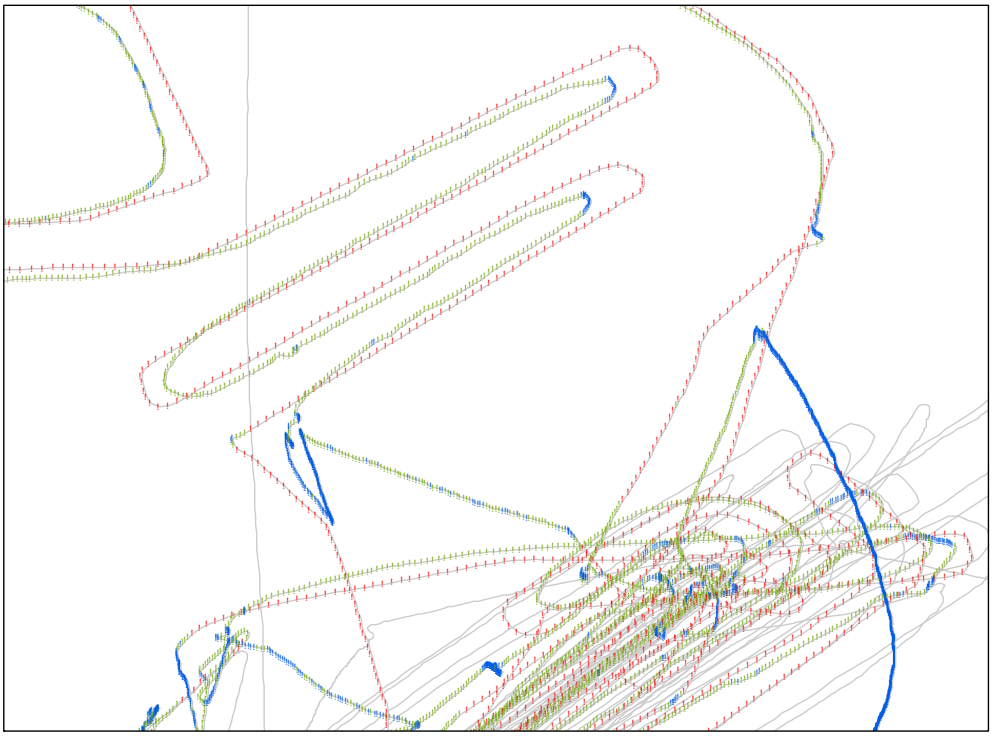
Appendix Figure A.1.2. Example of a tilefish haul where line is hauled from two separate setting events. The 12/29 haul includes gear set on 12/28 around 2:00 PM and also gear set around 7:30 PM. Spatial reference information is intentionally not shown to protect the confidentiality of the vessel data.



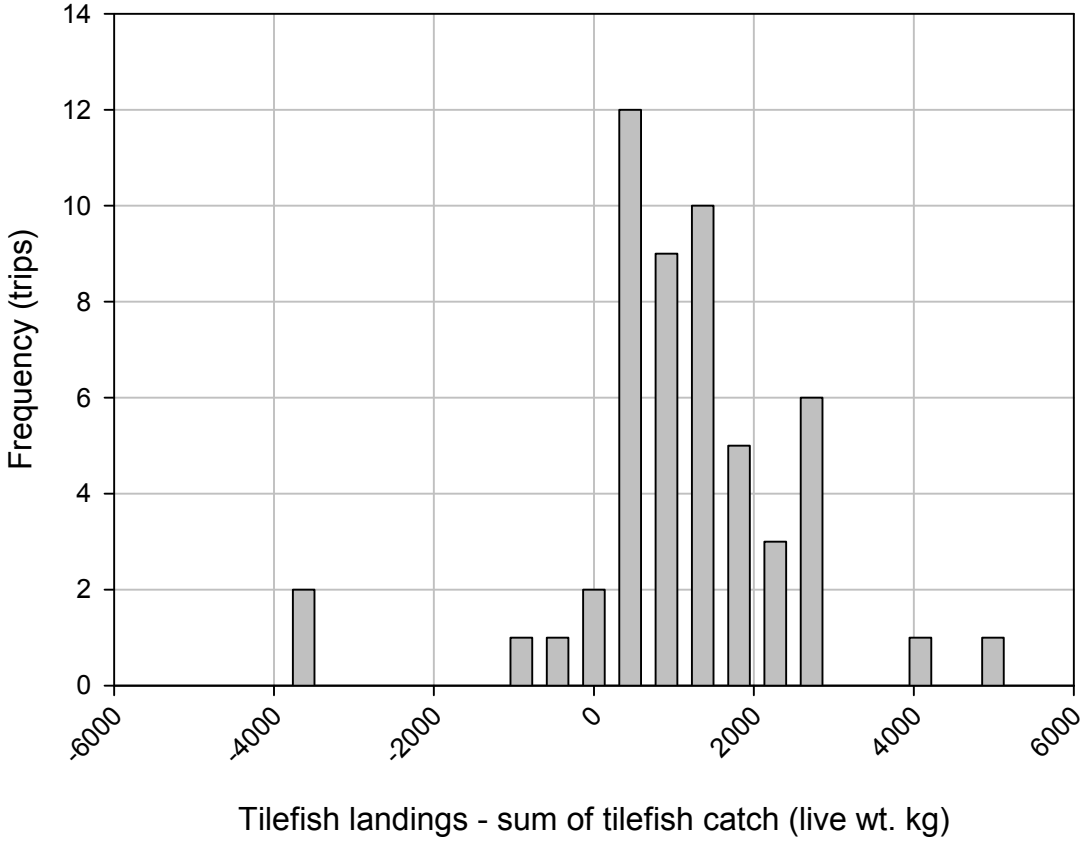
Appendix Figure A.1.3. A screen shot of the Fisheries Logbook Data Recording Software (FLDRS) catch data entry screen. This screen shot is similar to that used by tilefish vessel captains to record information on the fish caught for each haul.



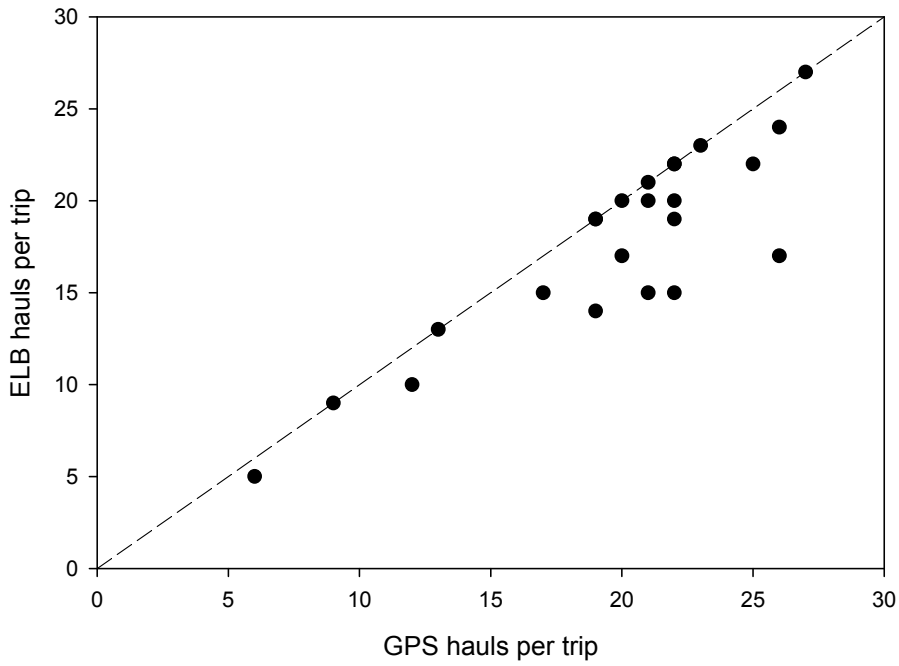
Appendix Figure A.1.4. Percent frequency distribution of recorded tilefish vessel speeds divided into hauling and other activity. The dashed lines (3.1 km/hr and 10.2 km/hr) indicate the speed window where >90% of the hauling activity occurs.



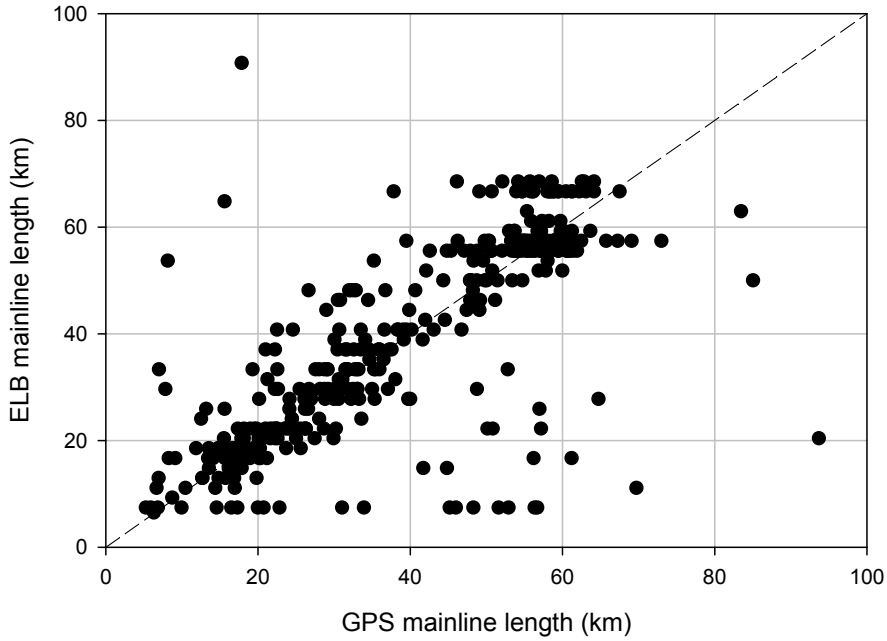
Appendix Figure A.1.5. Example of a global positioning system (GPS) polling file collected from a tilefish vessel. The cruise track is color coded based on vessel speed (blue < 1.7 knots, 1.7 ≥ green ≤ 5.5 knots, red > 5.5 knots). Spatial reference information is intentionally not shown to protect the confidentiality of the vessel data.



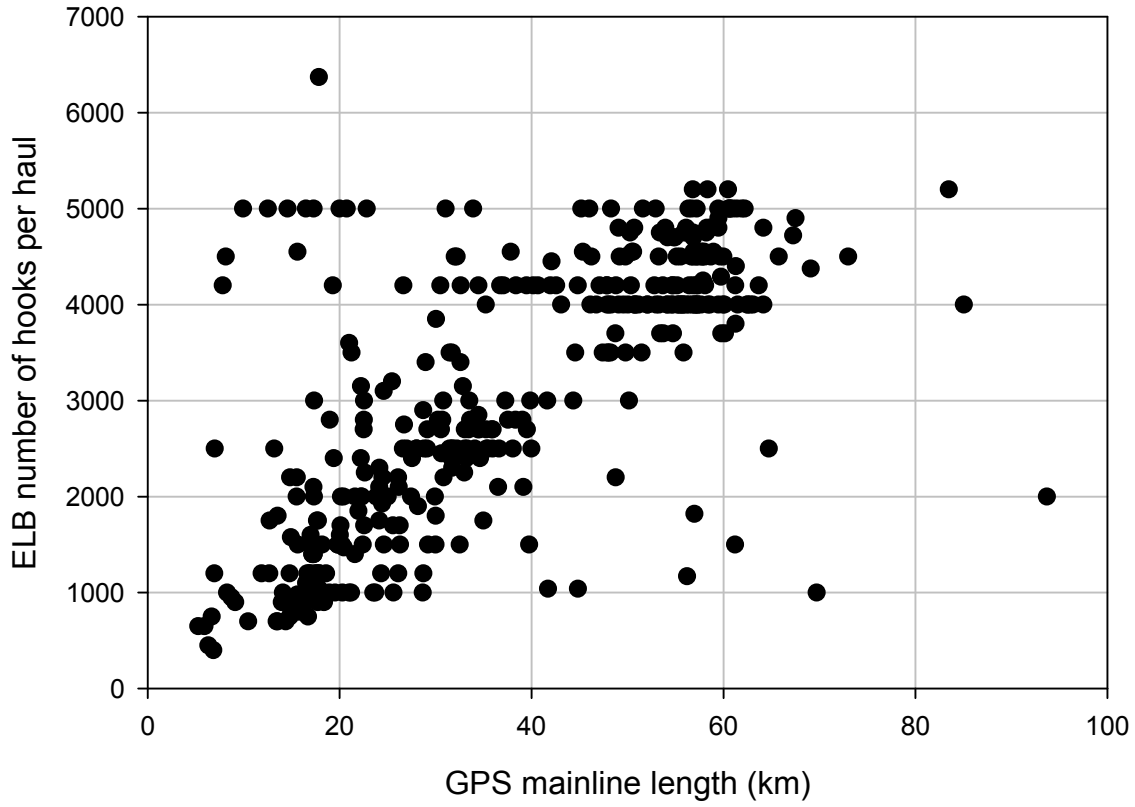
Appendix Figure A.1.6. Frequency distribution of the difference between the amount of landed tilefish and the sum of the individual haul hail weights for a trip. Positive values indicate more landed catch than recorded for the individual hauls, negative values indicates that there was more catch hailed than actually landed.



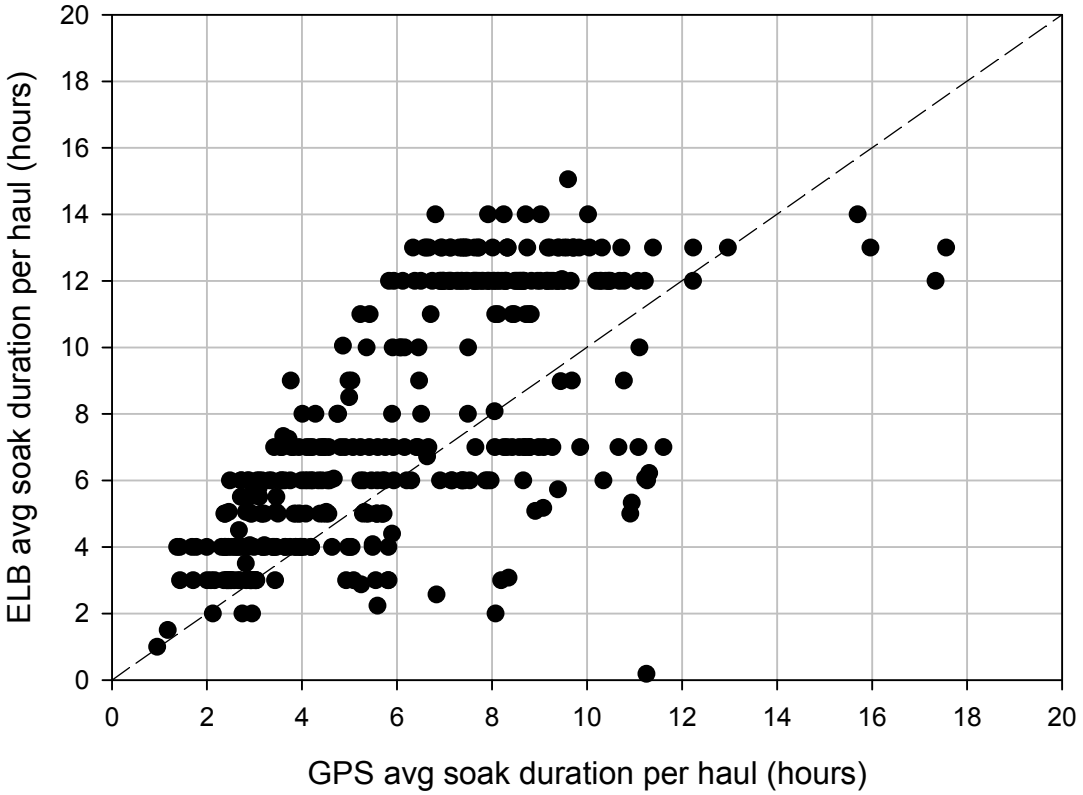
Appendix Figure A.1.7. The number of hauls recorded by the captain in the electronic logbook (ELB) compared to the number of hauls estimated from analysis of the global positioning system (GPS) polling file. The dashed line indicates the 1:1 identity line.



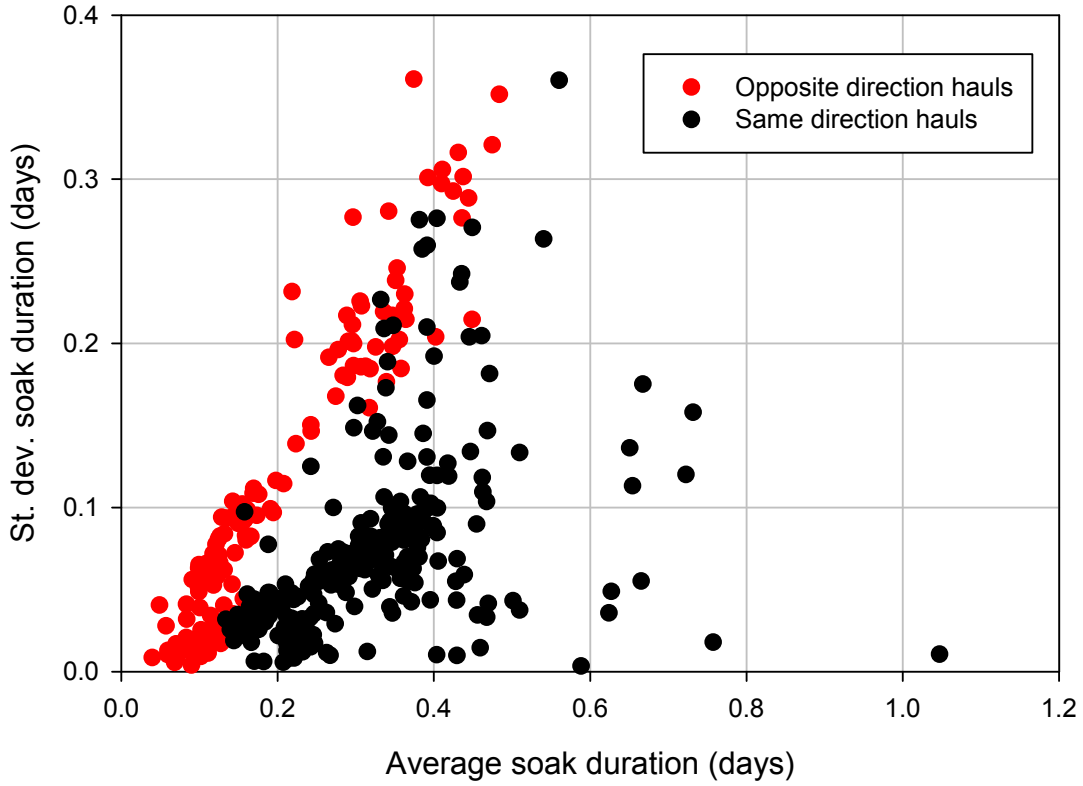
Appendix Figure A.1.8. The captain's estimate of mainline length hauled recorded in the electronic logbook (ELB) compared to the mainline length estimated from analysis of the global positioning system (GPS) polling file. The dashed line indicates the 1:1 identity line.



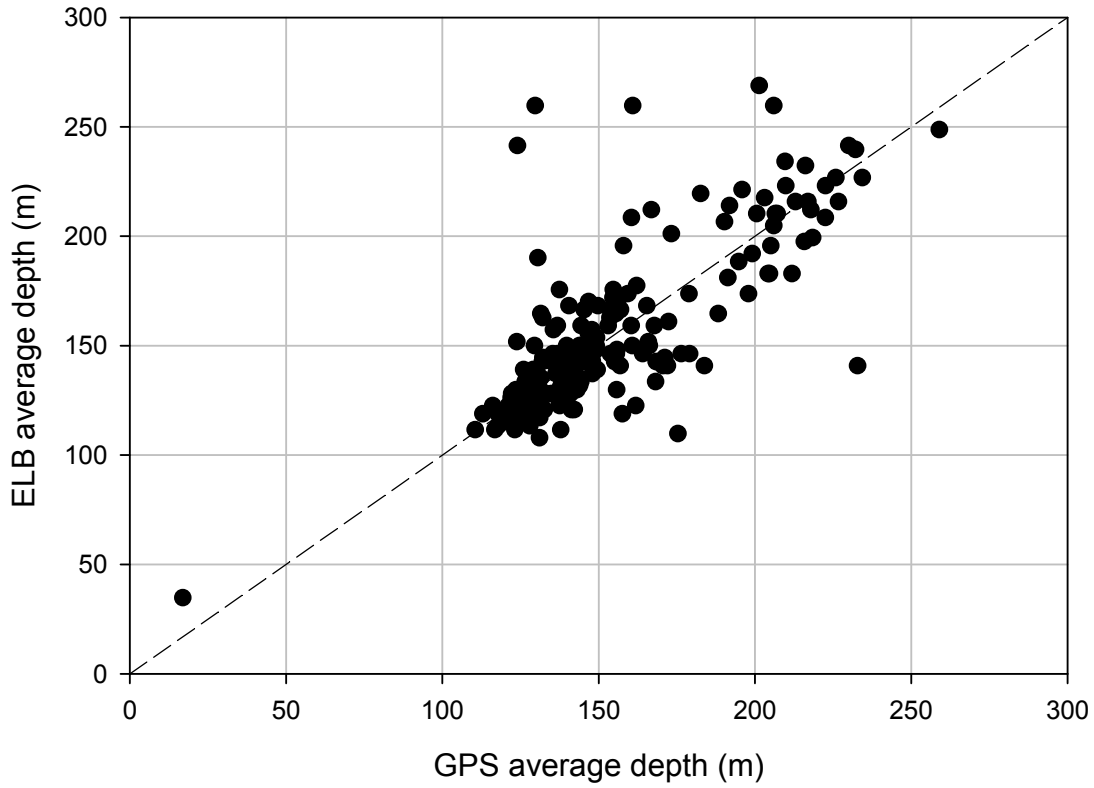
Appendix Figure A.1.9. The captain's estimate of the number of hooks hauled as recorded in the electronic logbook (ELB) compared to the mainline length estimated from analysis of the global positioning system (GPS) polling file.



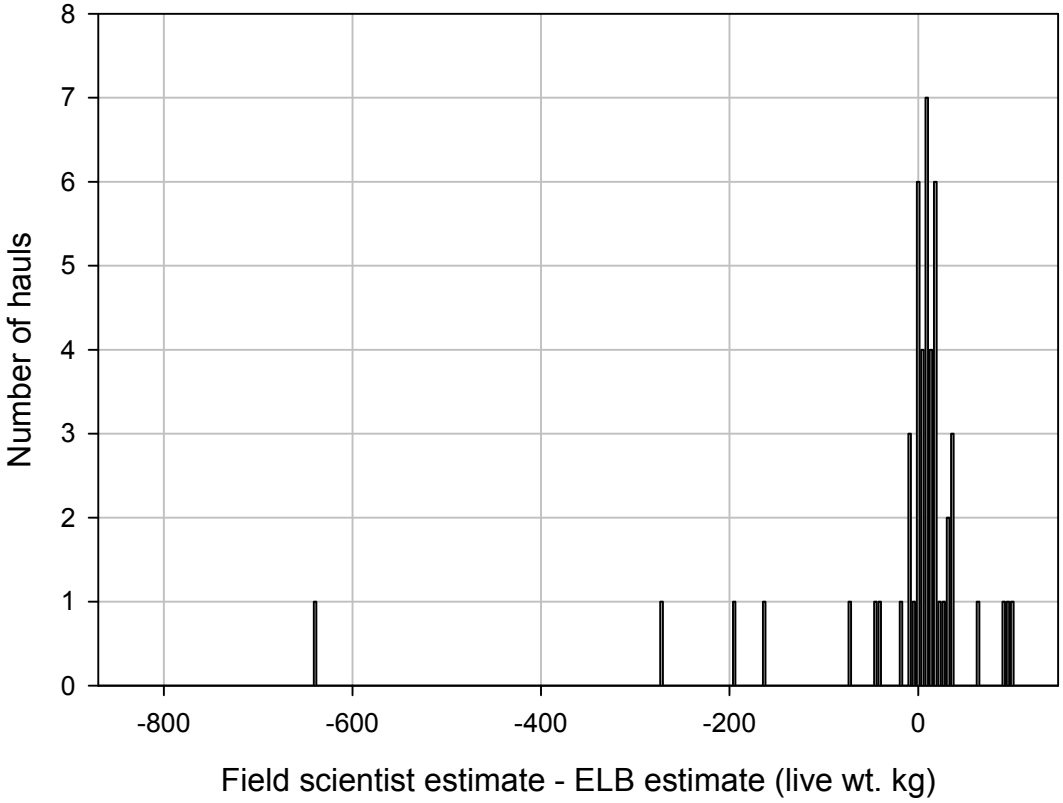
Appendix Figure A.1.10. The captain's estimate of the average soak duration of each haul recorded in the electronic logbook (ELB) compared to the average soak duration estimated from analysis of the global positioning system (GPS) polling file. The dashed line indicates the 1:1 identity line.



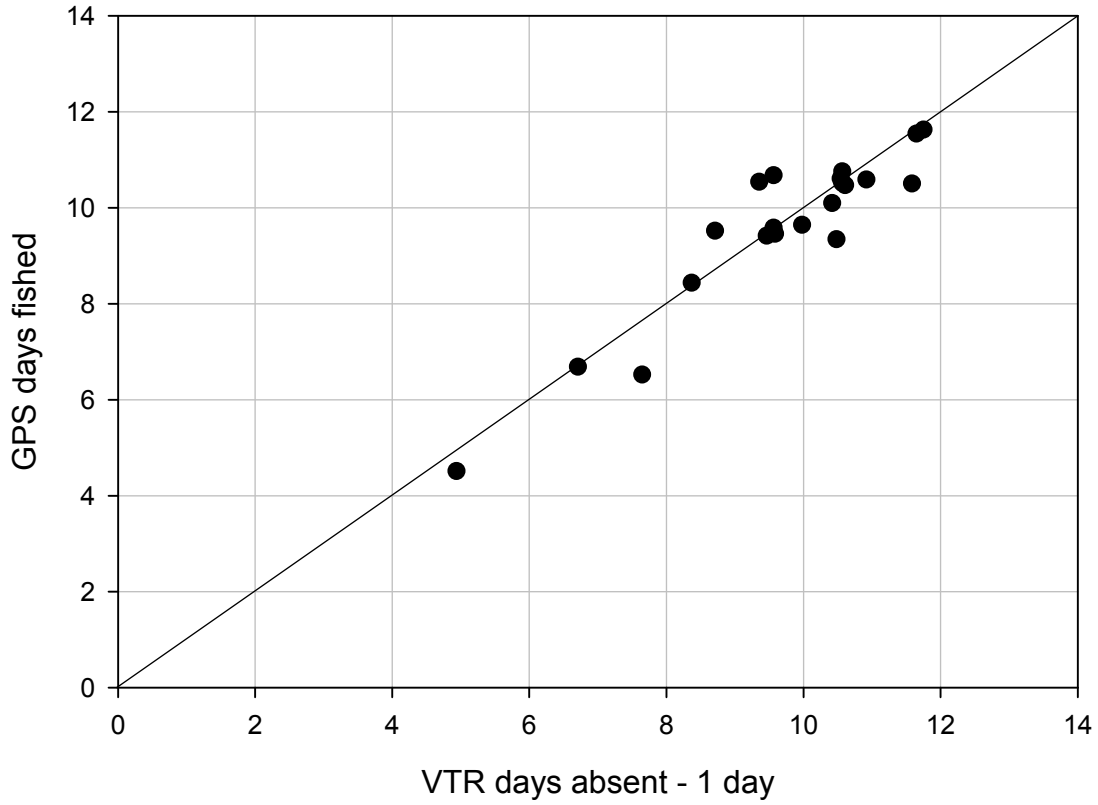
Appendix Figure A.1.11. Comparison of the amount of variability in haul soak times to the overall average soak time for the individual haul. Data points in red represent hauls that were hauled in the opposite direction from which they were set and the points in black represent hauls that were hauled in the same direction they were set.



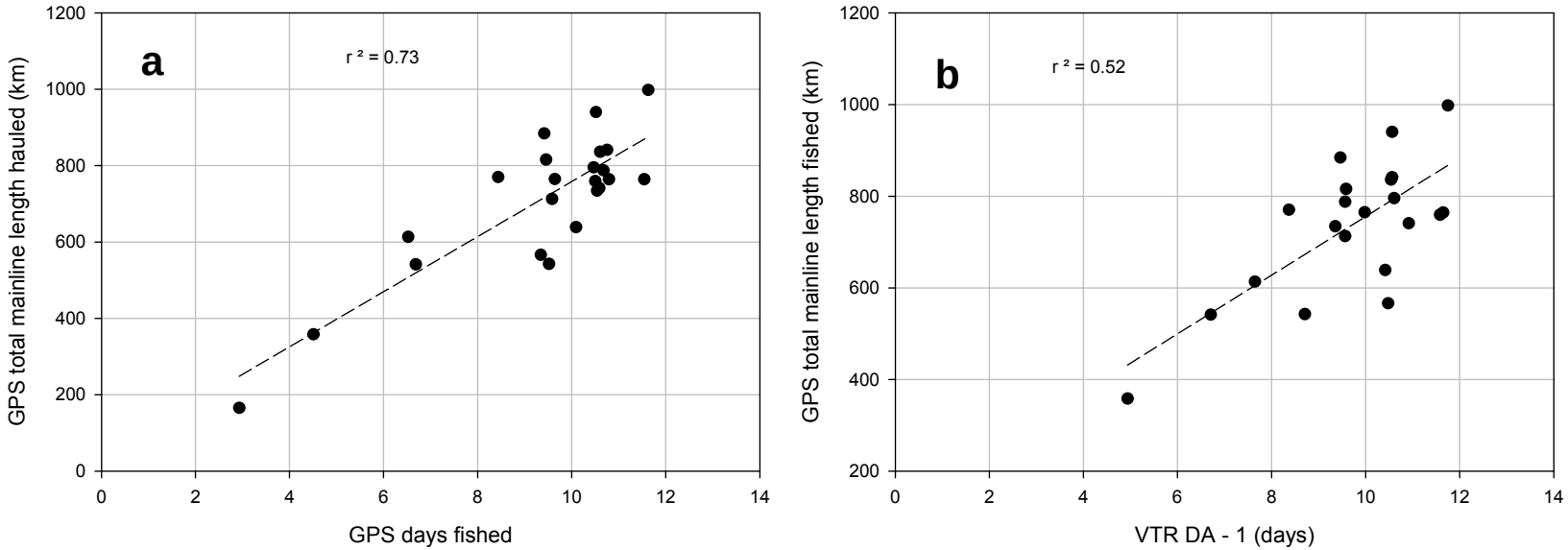
Appendix Figure A.1.12. The captain's estimate of the fishing depth of each haul recorded in the electronic logbook (ELB) compared to the average haul depth (m) estimated from analysis of the global positioning system (GPS) polling file.



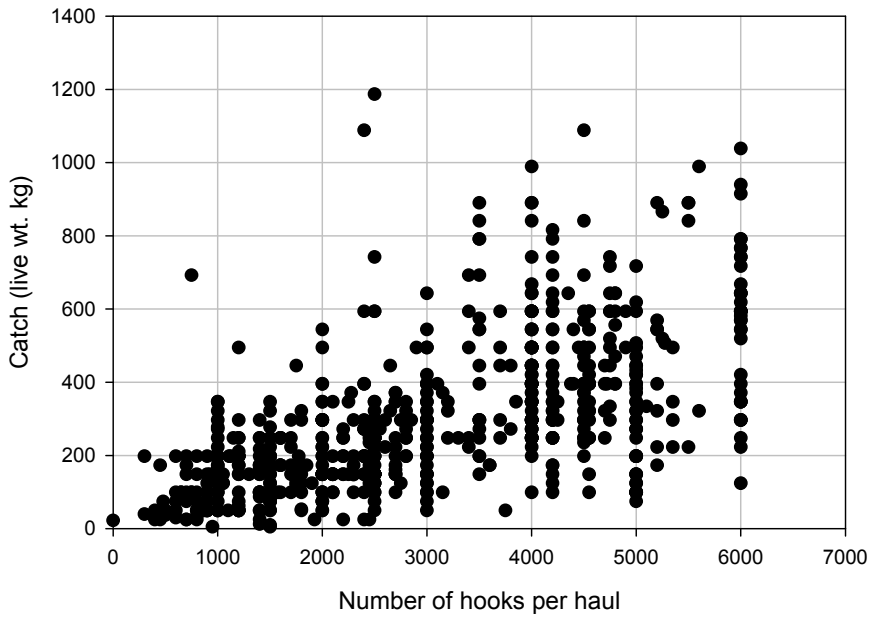
Appendix Figure A.1.13. Frequency distribution of the difference between the captain's haul-level hail weights and those estimated by Study Fleet field scientists. The compared weights span three different trips on three different vessels.



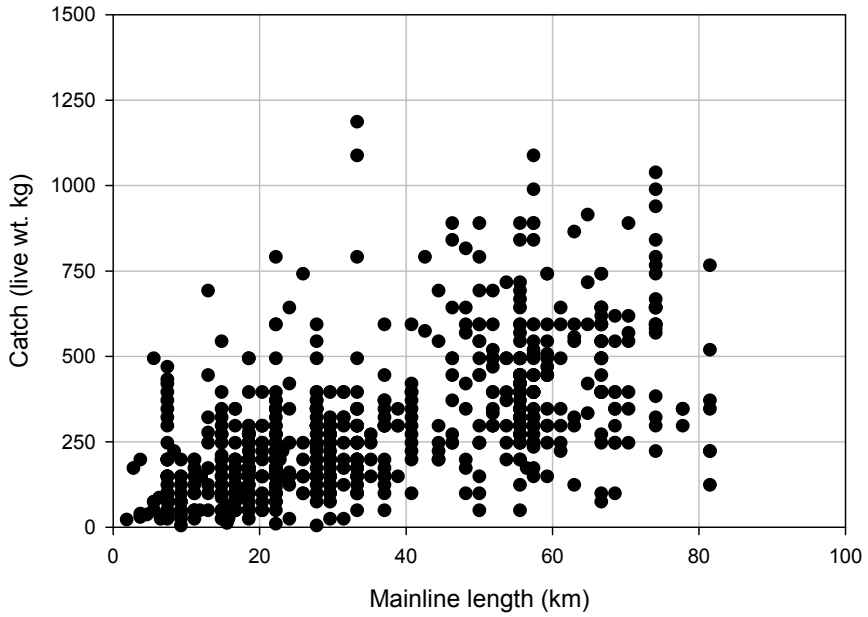
Appendix Figure A.1.14. Relationship between the total number of days fished as determined from analysis of global positioning system (GPS) data and the effort metric used is the surplus production model, the total days absent minus one calculated from the vessel trip reports (VTR).



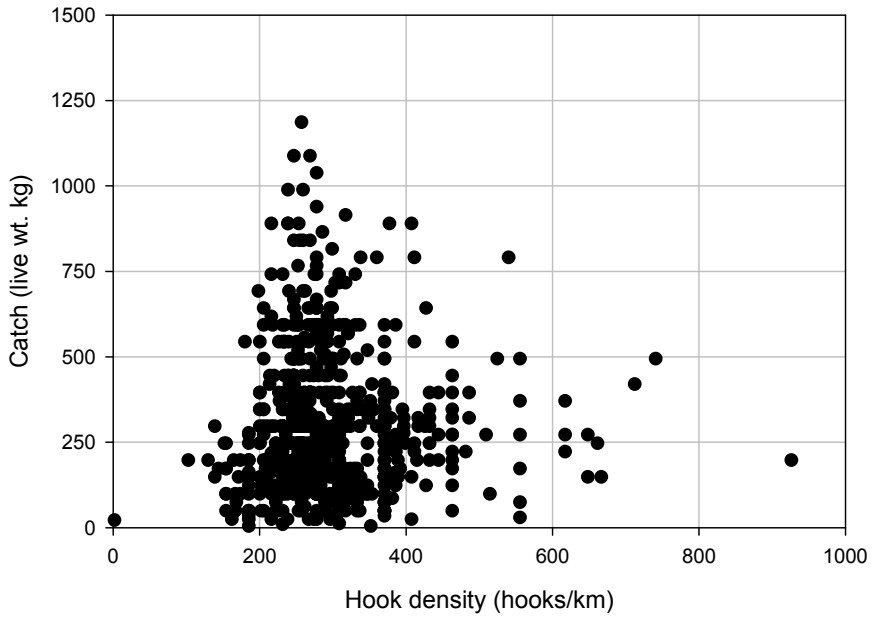
Appendix Figure A.1.15. Relationship between the total mainline length fished per trip as calculated from analysis of global positioning system (GPS) data and the total number of days fished (a) and the total days absent minus one calculated from the vessel trip reports (VTR; b).



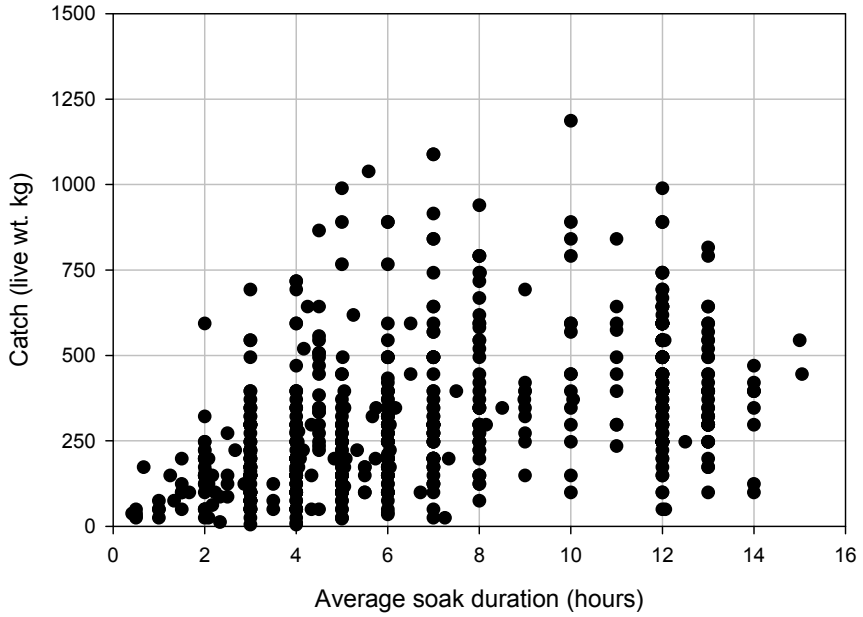
Appendix Figure A.1.16. Tilefish catch (kg live wt.) as a function of the number of hooks fished per haul. Tilefish catches are reported at the haul level by the vessel captains in the electronic logbook.



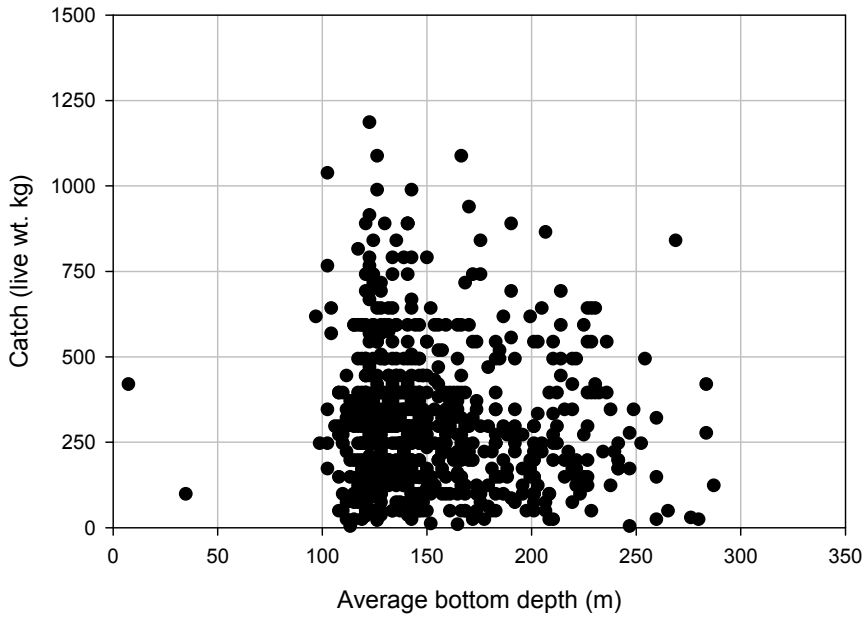
Appendix Figure A.1.17. Tilefish catch (kg live wt.) as a function of mainline length (km). Tilefish catches are reported at the haul level by the vessel captains in the electronic logbook.



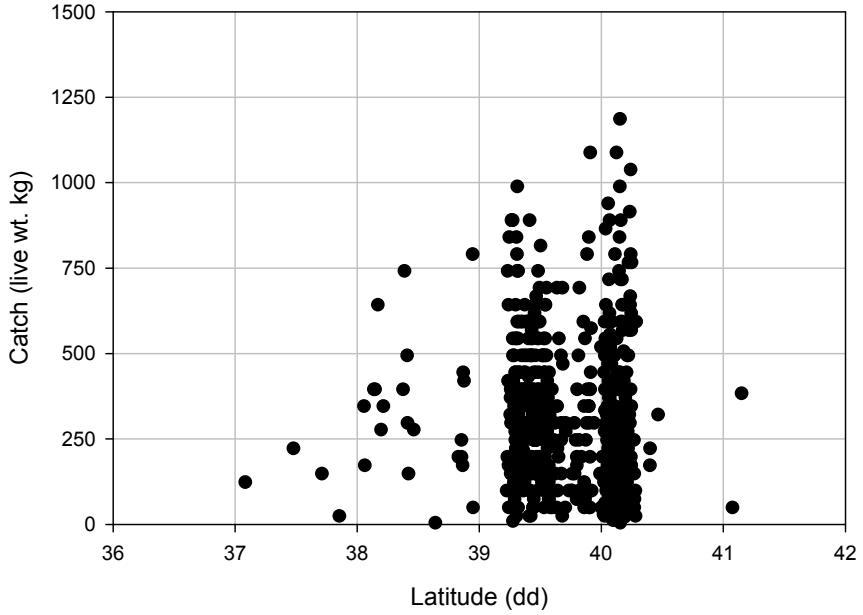
Appendix Figure A.1.18. Tilefish catch (kg live wt.) as a function of hook density (hooks/km). Tilefish catches are reported at the haul level by the vessel captains in the electronic logbook.



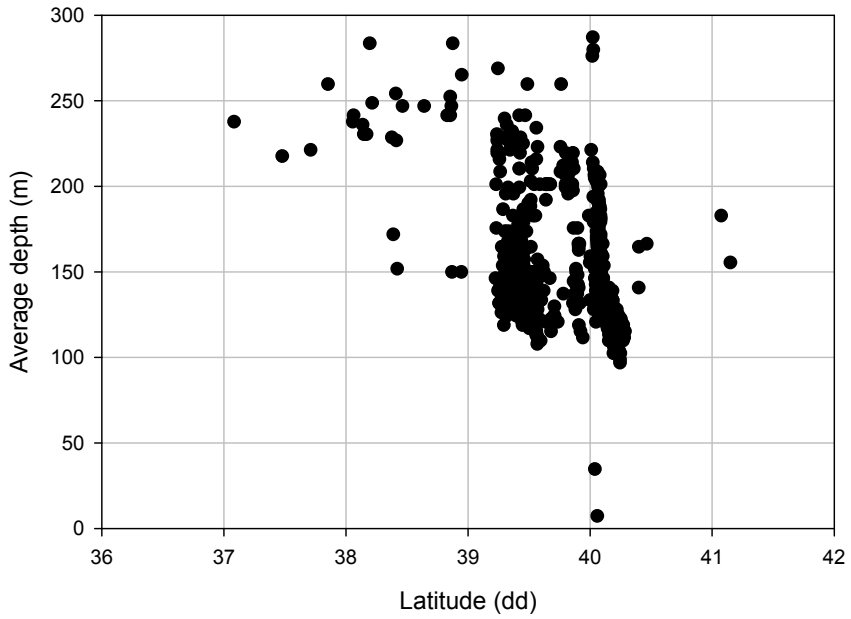
Appendix Figure A.1.19. Tilefish catch (kg live wt.) as a function of average soak duration (hours). Tilefish catches are reported at the haul level by the vessel captains in the electronic logbook.



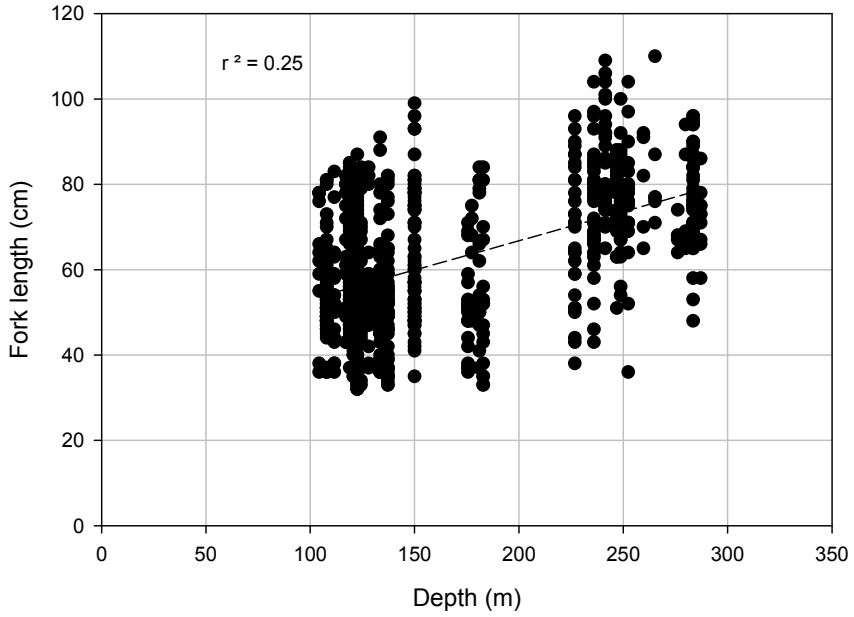
Appendix Figure A.1.20. Tilefish catch (kg live wt.) as a function of bottom depth (m). Tilefish catches are reported at the haul level by the vessel captains in the electronic logbook.



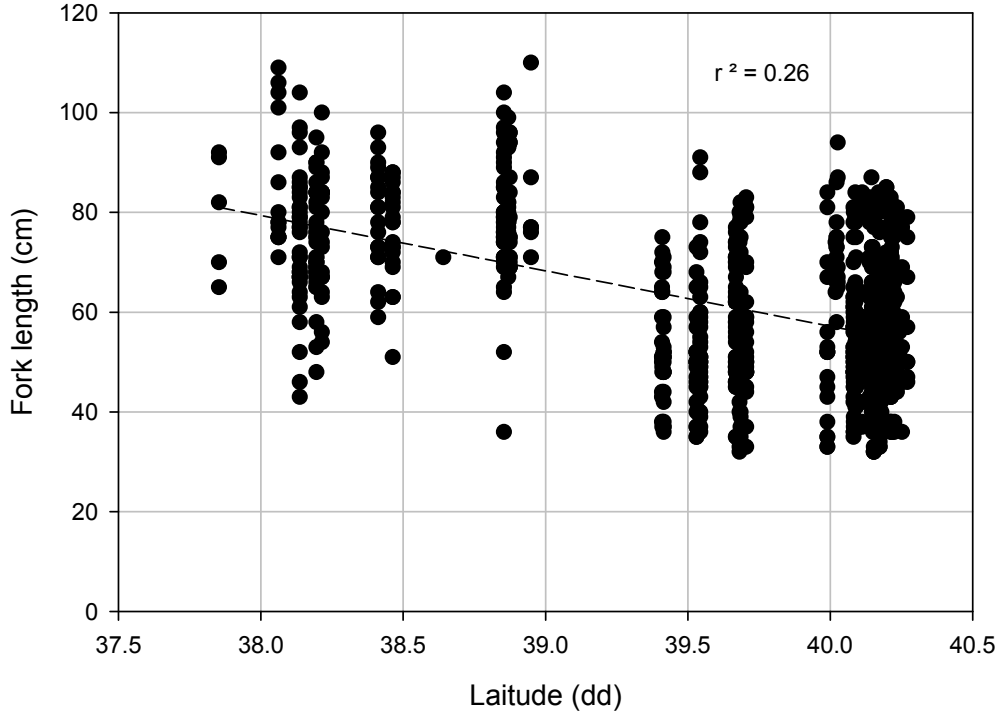
Appendix Figure A.1.21. Tilefish catch (kg live wt.) as a function of latitude (decimal degrees, dd). Tilefish catches are reported at the haul level by the vessel captains in the electronic logbook.



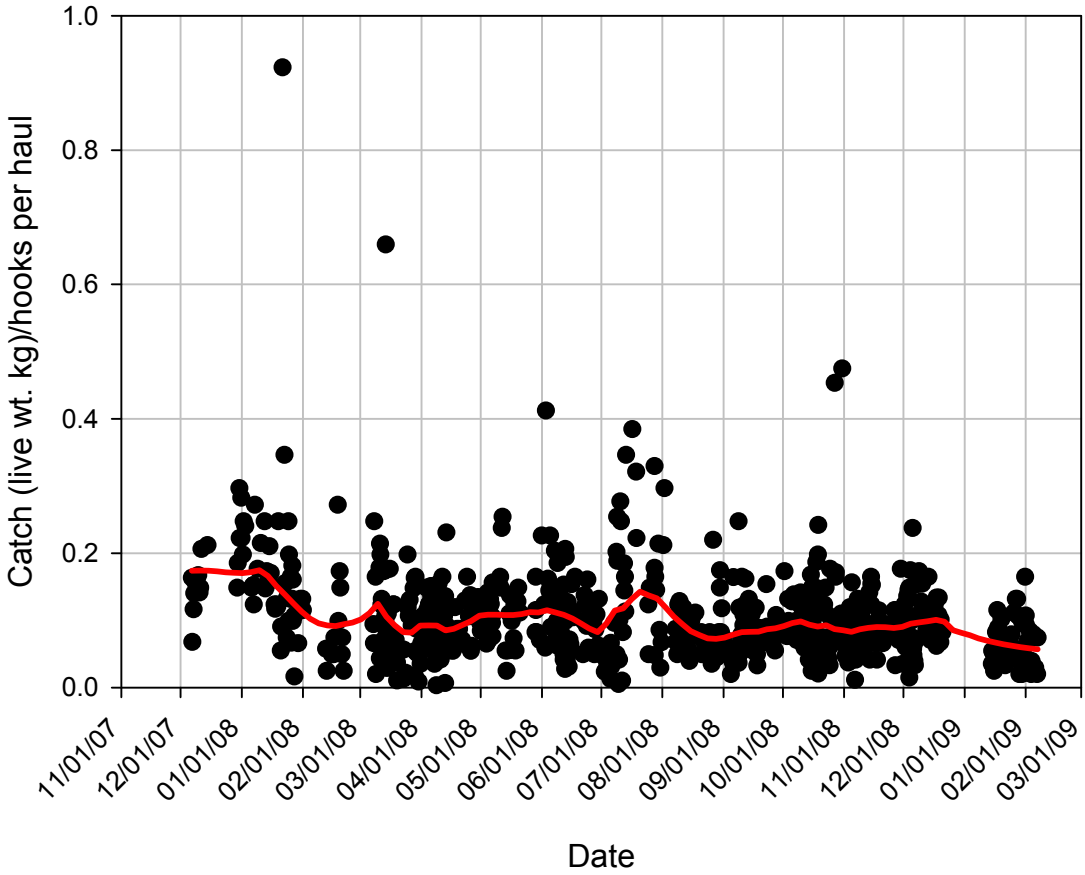
Appendix Figure A.1.22. Bottom depth fished (m) as a function of latitude (decimal degrees, dd).



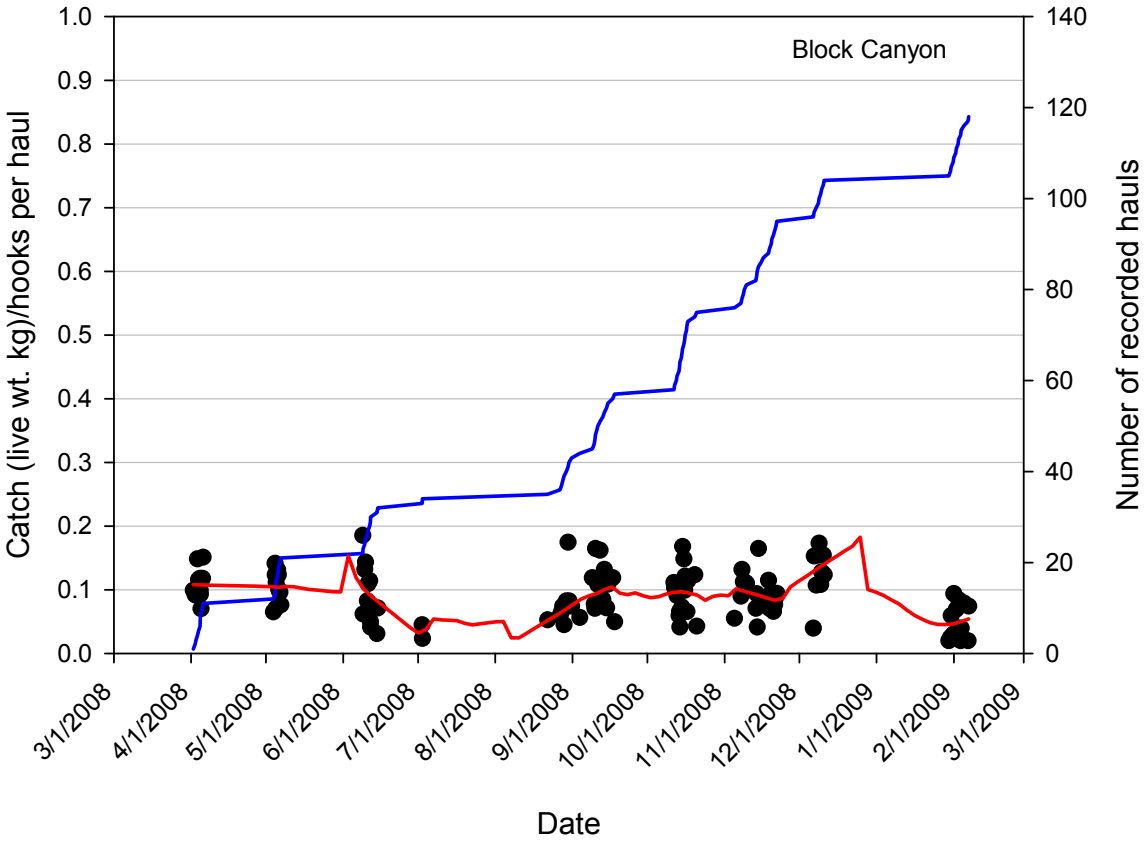
Appendix Figure A.1.23. Tilefish fork length (cm) as a function of bottom depth fished (m).



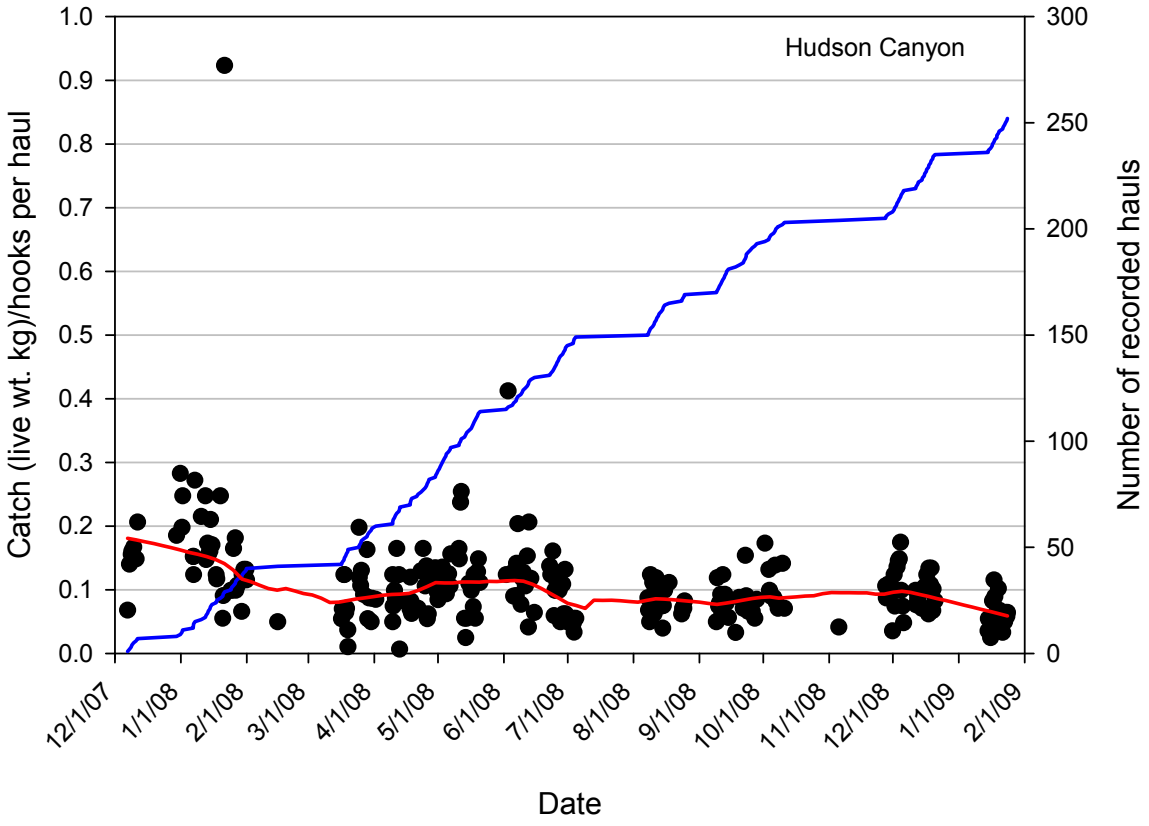
Appendix Figure A.1.24. Tilefish fork length (cm) as a function of the latitude fished (decimal degrees, dd).



Appendix Figure A.1.25. Tilefish haul-level catch (kg live wt.) over time (all data). The red line represents a loess smoothed trend of the time series.



Appendix Figure A.1.26. Tilefish haul-level catch (kg live wt.) over time in the vicinity of Block Canyon. The red line represents a loess smoothed trend of the time series.

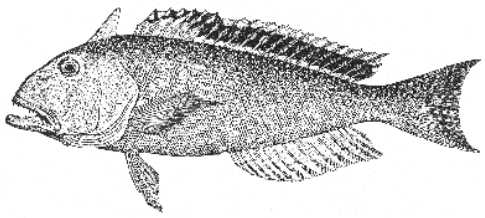


Appendix Figure A.1.27. Tilefish haul-level catch (kg live wt.) over time in the vicinity of Hudson Canyon. The red line represents a loess smoothed trend of the time series.

SAW48 Appendix A2
April 9 2009

Evaluating shifts in size and age at maturity of Golden tilefish from the Mid-Atlantic Bight

Tiffany Vidal
NOAA/UMass Cooperative Marine Education & Research Program
Northeast Fisheries Science Center



Summary
Macroscopic and histological analysis of golden tilefish sampled from the 2008 fishery indicates smaller size at maturity and younger age at maturity than similar analysis of samples from the 1982 fishery. Histology results from analysis of 2008 data indicate that size at 50% maturity was 46cm for females and 48cm for males. Size at age observations also suggest changes in growth rates since the 1980s.

Introduction

The objective of this research was to evaluate size and age at maturation for male and female tilefish, *Lopholatilus chamaeleonticeps*, from the Mid-Atlantic stock. This analysis used macroscopic maturity class data from at-sea sampling on commercial longline vessels combined with histological analysis. The size at maturation for the 2008 stock was then compared to the 1982 stock, to determine if the proportion mature, as a function of size, has shifted towards maturation at smaller sizes. A shift towards maturation at smaller sizes could be an indication that the population size has decreased (Grift et al. 2003; Ernande et al. 2004; Anderson et al. 2007). An ageing study was performed to evaluate changes in the growth curves since 1982 and to determine age at length and maturation and to assess whether or not size at maturity has shifted from 1982, the last time the reproductive biology was evaluated (Grimes et al. 1988). Understanding and evaluating changes in size and age at maturation are important in understanding the broader population dynamics of this stock.

Methods

Sampling Design

Tilefish were sampled from commercial longline catches using a systematic sampling design stratified by fish length and gender; sampling one fish per cm interval per sex. The systematic sampling design was to ensure that the entire size distribution of the fish encountered was sampled, and that the sizes more and less frequently encountered, were not over or under-sampled, respectively. Two commercial trips, for sample collection, were made during the spawning season; June and July. Additional samples, approximately 10 fish bimonthly, were collected portside from commercial trips to obtain samples throughout the year. These fish were selected randomly from market categories: kitten, medium, and large, from the last haul of the trip.

Macroscopic staging

Tilefish are gonochoristic (i.e., they have separate sexes) and are indeterminate serial spawners (i.e., they spawn in multiple batches). Tilefish gonads are paired organs located posteriorly in the body cavity below the swim bladder, with the ovaries suspended by thin mesovaria; testis by mesorchia (Idelberger 1985). Gonads were classified to six macroscopic classes: immature, developing, ripe, ripe and running, spent, and resting; the criteria to classify individuals to a given class were based on Idelberger's (1985) classification criteria. All classes, except immature (and fish of unknown sex and/or class) were considered to be mature. Fish developing to spawn for the first time were not differentiated from repeat spawners.

One ovarian lobe or testis was removed and preserved in 10% buffered formalin; alternatively a transverse section of the medial portion of one ovary or testis was preserved for histology. In the laboratory, the gonad tissue samples were dehydrated through a series of increasing ethanol concentrations, cleared with Clear Rite™, and embedded into paraffin. The paraffin blocks were allowed to harden, trimmed around the edges using a razor blade to remove

excess paraffin, sectioned at a thickness of 4µm using a microtome, mounted on glass slides, stained with hematoxylin, counterstained with eosin and coverslipped. The hematoxylin and eosin (H&E) staining method used was based on H&E procedures detailed by Luna (1968).

Microscopic staging

Microscopic criteria for staging gonadal cells were based on maturity classifications described for the following species: tilefish (Grimes et al. 1988, Erickson et al. 1985), round scad (McBride et al. 2002), tilapia (Hyder 1969), and common snook (Grier et al. 1998). Females were considered immature if the perinucleolar stage was the most advanced stage of oocyte development observed. An individual was considered to be mature if cortical alveolar, vitellogenic, or hydrated oocytes were observed. The presence of postovulatory follicles was also an indication of prior spawning. For males, the presence of spermatozoa in the spermatogenic crypts and/or lobules was the criterion for maturity.

Ageing

The fish sampled for histology were also aged. The sagittal otoliths were extracted at sea, mounted on a wax pillow atop a paper tab with crosshairs for alignment with a low-speed diamond blade Isomet® saw, completely embedded in wax, and thin sectioned through the core. The right sagittae was used unless it was broken or unavailable. Annular rings were counted to determine fish age. Each annulus, or ring, represents one year of growth; with the annuli typically laid down by June of each year (Turner 1986). Confirmation of this aging method has been done through marginal increment analysis. Otoliths from Turner's (1986) aging study were used as a reference collection to maintain consistency in the aging method.

Statistical Analysis

Logistic regression was used predict the maturity ogives for males and females from the 2008 population using the GLM function with a logit link, in the R statistical software program.

$$P_i = \frac{e^{\beta_0+\beta_1 X_i}}{1+e^{\beta_0+\beta_1 X_i}} \qquad (1)$$

P_i: proportion mature at size or age i
B_0: intercept of logistic model
B_1: logistic regression coefficient for explanatory variable X1
X_i: the ith observation of the explanatory variable (size or age)

The 95% confidence bands were calculated as +/- 1.96 times the standard error of the estimate of proportion mature at a given size.

The maturity ogives, for males and females, based on macroscopic and histological data were compared, and precision estimates between the two methods were determined. The macroscopic results were compared to the Grimes et al. (1988) data. The raw data were not available from the Grimes et al. (1988) study, so the binned data were expanded out and treated as raw data. This is not an ideal method for comparison, but should provide a general idea as to whether or not there have been shifts in the ogives.

To quantitatively determine whether the proportion mature as a function of length was significantly different between the macroscopic and histological methods logistic regression models

were used. Logistic regression was also used to test difference in length and age at maturation between 1982 and 2008. The p-values associate with the z-statistics from the model output, in addition to the Bayesian information Criterion (BIC)

$$BIC = z - statistic^2 - \ln(n) \quad (2)$$

were used to test the significance of the regression parameters (Pampel 2000).

Growth curves were computed for the sampled 2008 population using a von Bertalanffy (1938) growth model,

$$L_t = L_\infty [1 - e^{-k(t-t_0)}] \quad (3)$$

Lt: length at age *t*
L∞: asymptotic length
k: Brody growth coefficient
t_0: age at length=0

and a von Bertalanffy growth model with equally weighted mean length at age values. Growth model parameters were estimated using the SAS nlin procedure using Turner's (1986) parameter estimates as the initial values for L_∞, k, and t_0. Age at length was calculated and used to asses shifts in age at maturation, ignoring growth variation and overlapping length distributions, but associating each length with an age using the estimated von Bertalanffy parameter estimates (Hilborn and Walters 1992).

$$\hat{t} = t_0 - \left(\frac{1}{k}\right)\log\left[1 - \left(\frac{L_t}{L_\infty}\right)\right] \quad (4)$$

Growth curves were estimated for both sexes combined as well as males and females separately.

Results

Females – macroscopic

The logistic regression model predicted the proportion of fish mature at length with 95% confidence bands around the estimates. The macroscopic data analyzed were for fish sampled for histology as well; the results indicate that female tilefish begin maturing around 40 cm and are almost 100% mature by 50 cm (Figure 1). The regression cannot fully predict to the lower tails due to a lack of small fish. There is some size selectivity based on the hook size, which selects against the smallest fish in the population. As a result there is limited data for the small sizes, however the ogive fits the data fairly well. Fifty percent maturity (M_{50}) is achieved at approximately 45 cm (n=66; Table 1) and 5 years (Table 2).

Females – histological

Histological evaluation indicated that M_{50} is 46 cm (n=70; Table 3; Figure 2) and 5 years (Table 2). There was strong agreement between the two staging methods for females, with 92% precision. Eighty percent of the disagreement was due to immature fish between 42 and 50 cm being classified as developing macroscopically.

Males – macroscopic

The macroscopic maturity ogive for the 2008 males (Figure 3) shows that they begin maturing around 48 cm and are almost 100% mature at about 73 cm. The length range over which maturation occurs is much wider for the males than for the females. M_{50} is approximately 56 cm (n=149; Table 4; Figure 4) and 6 years (Table 2).

Males – histological

Agreement between the two staging methods for males was less than for the females with 85% precision. Ninety one percent of the disagreement was due to developing fish classified as immature in the field. Fifty percent maturity based on histological evaluation was predicted to be 48 cm (n=151; Table 5) and 5 years (Table 2).

All macroscopic staging

Additional macroscopic observations were made beyond those that were paired with histology. Figures 5 and 6 show all macroscopic staging data for females and males respectively from 2008. Length at 50% maturity (L_{50}) for females is predicted at 44 cm (n=321) and L_{50} for males predicted at 57 cm (n=479; Tables 6 and 7); ages 5 and 6 respectively.

Comparison to 1982 stock

The 1982 data were macroscopic observations expanded out based on the sample sizes noted on the logistic regression plots in the Grimes et al. (1988) study. The data represented proportion mature at each 5 cm length bin; the raw data were not available. Both the macroscopic and histological results were compared to the 1982 macroscopic data. Figures 7 and 8 are qualitative ways to visualize the shifts in maturity ogives from 1982 to the present. The blue line represents the 2008 data and the green line is the 1982 data from Grimes et al. (1988). Each of these plots indicates a shift toward maturation at smaller sizes in 2008 as compared to observations in 1982.

The full regression models, sexes combined, indicated that maturity schedules were significantly different between sexes; sexes were therefore analyzed separately. For all models, year was significant ($p<<0.05$; BIC>10; Tables 8-13), indicating a significant shift in size and age at maturation between 1982 and 2008. M_{50} in 1982 for females was approximately 52 cm (Table 14) and 6 years; 8 cm larger than the combined macroscopic results in 2008 and 6 cm larger than the histology results. M_{50} for males in 1982 was approximately 63 cm (Table 15) and 8 years; 6 cm larger than the combined macroscopic results in 2008 and 16 cm larger than the histology results.

Age at Length

The age-length keys developed from the two growth models: von Bertalanffy using raw data and the von Bertalanffy growth model using equally weighted mean length-at-age values are shown in Tables 16 and 2.

Growth models

Von Bertalanffy growth model results based on individual observations are displayed in Tables 17-19; Figures 9-11. Asymptotic length was substantially larger than previous estimates, due to few old fish in the sample and relatively high frequency of fish ages 5-10. To address this uneven sample distribution, alternative von Bertalanffy growth models were fit to mean length-at-age, which weights each age equally (Tables 20-22; Figures 12-14).

Discussion

These results show a significant decrease in size and age at maturation since the last evaluation of this stock in the early 1980's (Grimes et al. 1986). An environment in which survival rates are low for potentially reproducing individuals, often favors selection of individuals that are able to reproduce at smaller sizes and younger ages (Hutchings 1993; Reznick et al. 1990). In a

hook fishery, it is assumed that the smallest fish in the population are less vulnerable to the gear depending on the hook size. In this fishery, hook size has been intentionally increased to avoid catch of the smallest fish in the population. The fact that such dramatic changes have manifested in this stock may suggest a density-dependent effect of decreased population size. It is uncertain at this point in time, whether these changes are consequences of phenotypic plasticity or selection towards genotypes with lower size and age at maturation.

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Table 1. Proportion mature at length for 2008 females (macroscopic)

Proportion	Length	SE
p = 0.025	36.35355	3.005000
p = 0.250	42.22186	1.487536
p = 0.500	44.73536	1.115889
p = 0.750	47.24885	1.203578
p = 0.975	53.11716	2.545929

Table 2. Age-length keys from von Bertalanffy growth model using mean length at age (sexes combined)

Age at Length		Length at Age	
Length (cm)	Age (years)	Length (cm)	Age (years)
10	1	7	1
11	1	20	2
12	1	31	3
13	1	40	4
14	2	49	5
15	2	56	6
16	2	63	7
17	2	68	8
18	2	73	9
19	2	78	10
20	2	81	11
21	2	85	12
22	2	88	13
23	2	90	14
24	2	92	15
25	2	94	16
26	3	96	17
27	3	98	18
28	3	99	19
29	3	100	20
30	3	101	21
31	3	102	22
32	3	103	23
33	3	103	24
34	3	104	25
35	3	104	26
36	4	105	27
37	4	105	28
38	4	106	29
39	4	106	30
40	4	106	31
41	4	106	32
42	4	107	33
43	4	107	34

44	4	107	35
45	5	107	36
46	5	107	37
47	5	107	38
48	5	107	39
49	5	107	40
50	5	107	41
51	5	107	42
52	5	108	43
53	6	108	44
54	6	108	45
55	6	108	46
56	6	108	47
57	6	108	48
58	6	108	49
59	6	108	50
60	7		
61	7		
62	7		
63	7		
64	7		
65	7		
66	8		
67	8		
68	8		
69	8		
70	8		
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86	12		
87	13		
88	13		
89	14		
90	14		
91	14		
92	15		
93	15		
94	16		

95	16
96	17
97	18
98	18
99	19
100	20
101	21
102	22
103	24
104	25
105	28
106	31
107	36

Table 3. Proportion mature at length for 2008 females (histological)

Proportion	Length	SE
p = 0.025	36.62657	3.160495
p = 0.250	43.10680	1.433769
p = 0.500	45.88239	1.043394
p = 0.750	48.65799	1.256798
p = 0.975	55.13821	2.898430

Table 4. Proportion mature at length for 2008 males (macroscopic)

Proportion	Length	SE
p = 0.025	39.32151	3.381805
p = 0.250	51.07196	1.644096
p = 0.500	56.10488	1.289149
p = 0.750	61.13780	1.496608
p = 0.975	72.88825	3.145142

Table 5. Proportion mature at length for 2008 males (histological)

Proportion	Length	SE
p = 0.025	38.14695	2.954953
p = 0.250	45.13220	1.528347
p = 0.500	48.12411	1.142997
p = 0.750	51.11601	1.141340
p = 0.975	58.10127	2.299208

Table 6. Proportion mature at length for 2008 females (all macroscopic observations)

Proportion	Length	SE
p = 0.025	31.60688	2.2969273
p = 0.250	40.49261	1.1497262
p = 0.500	44.29852	0.8305603
p = 0.750	48.10443	0.8333328
p = 0.975	56.99016	1.7842602

Table 7. Proportion mature at length for 2008 males (all macroscopic observations)

Proportion	Length	SE
p = 0.025	38.11876	1.8763305
p = 0.250	51.60568	0.8664657
p = 0.500	57.38236	0.7582732
p = 0.750	63.15904	1.0026450
p = 0.975	76.64596	2.0903147

Table 8. Logistic regression model output for length at maturation (females - macro)

Coefficients:

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-12.91363	0.74598	-17.311	< 2e-16 ***
length	0.24692	0.01372	17.994	< 2e-16 ***
year2008	2.05630	0.25472	8.073	6.87e-16 ***

Table 9. Logistic regression model output for length at maturation (males - macro)

Coefficients:

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-8.787480	0.443466	-19.815	< 2e-16 ***
year2008	0.741363	0.159973	4.634	3.58e-06 ***
length	0.139662	0.007022	19.889	< 2e-16 ***

Table 10. Logistic regression model output for length at maturation (females – histo 2008; macro 1982)

Coefficients:

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-12.8166	0.7826	-16.376	< 2e-16 ***
length	0.2451	0.0144	17.017	< 2e-16 ***
year2008	1.5979	0.3856	4.144	3.41e-05 ***

Table 11. Logistic regression model output for length at maturation (males – histo 2008; macro 1982)

Coefficients:

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-8.310188	0.485691	-17.110	< 2e-16 ***
year2008	2.445288	0.298275	8.198	2.44e-16 ***
length	0.131946	0.007707	17.120	< 2e-16 ***

Table 12. Logistic regression model output for age at maturation (females)

Coefficients:

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-8.88270	0.58353	-15.22	<2e-16 ***
age	1.49627	0.09428	15.87	<2e-16 ***
year2008	2.26650	0.24190	9.37	<2e-16 ***

Table 13. Logistic regression model output for age at maturation (males)

Coefficients:

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	-5.23012	0.27635	-18.926	< 2e-16 ***
age	0.62969	0.03419	18.415	< 2e-16 ***
year2008	1.20293	0.15711	7.657	1.91e-14 ***

Table 14. Proportion mature at length for 1982 females (Grimes et al. 1988)

Proportion	Length	SE
p = 0.025	37.05423	1.0855842
p = 0.250	47.69894	0.5337725
p = 0.500	52.25825	0.3908343
p = 0.750	56.81757	0.4133665
p = 0.975	67.46228	0.8934191

Table 15. Proportion mature at length for 1982 males (Grimes et al. 1988)

Proportion	Length	SE
p = 0.025	33.76355	1.8815446
p = 0.250	54.25703	0.8505181
p = 0.500	63.03475	0.7033085
p = 0.750	71.81246	0.9232099
p = 0.975	92.30595	1.9925294

Table 16. Age-length keys from von Bertalanffy growth model (sexes combined)

Age at Length		Length at Age	
Length (cm)	Age (years)	Length (cm)	Age (years)
10	1	12	1
11	1	23	2
12	1	32	3
13	1	40	4
14	1	48	5
15	1	55	6
16	1	61	7
17	1	67	8
18	2	72	9
19	2	77	10
20	2	81	11
21	2	85	12
22	2	89	13
23	2	92	14
24	2	95	15
25	2	98	16
26	2	100	17
27	2	102	18
28	3	104	19
29	3	106	20
30	3	108	21
31	3	109	22
32	3	111	23
33	3	112	24
34	3	113	25
35	3	114	26
36	3	115	27
37	4	116	28
38	4	116	29
39	4	117	30
40	4	118	31
41	4	118	32
42	4	119	33
43	4	119	34
44	4	120	35
45	5	120	36
46	5	120	37
47	5	121	38
48	5	121	39

49	5
50	5
51	5
52	6
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54	6
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112	24
113	25
114	26
115	27
116	28
117	30
118	32
119	33
120	36
121	39
122	44
123	52

Table 17. von Bertalanffy growth model parameter estimates (sexes combined)

Parameter	Estimate	Std Error	Approximate 95% Confidence Limits	
li	123.8	7.7452	108.5	139.1
k	0.0969	0.0127	0.0719	0.1219
t0	-0.0778	0.2908	-0.6519	0.4962

Table 18. von Bertalanffy growth model parameter estimates (females)

Parameter	Estimate	Std Error	Approximate 95% Confidence Limits	
li	112.0	9.1182	93.8035	130.2
k	0.0964	0.0175	0.0614	0.1313
t0	-0.5450	0.4590	-1.4618	0.3717

Table 19. von Bertalanffy growth model parameter estimates (males)

Parameter	Estimate	Std Error	Approximate 95% Confidence Limits	
li	141.5	12.1959	117.3	165.7
k	0.0833	0.0136	0.0564	0.1102
t0	-0.0920	0.3331	-0.7527	0.5687

Table 20. von Bertalanffy growth model parameter estimates using mean length at age (sexes combined)

Parameter	Estimate	Std Error	Approximate 95% Confidence Limits	
li	107.9	5.7375	95.9875	119.8
k	0.1338	0.0226	0.0869	0.1807
t0	0.4944	0.5182	-0.5802	1.5690

Table 21. von Bertalanffy growth model parameter estimates using mean length at age (females)

Parameter	Estimate	Std Error	Approximate 95% Confidence Limits	
li	100.1	7.1457	84.1627	116.0
k	0.1393	0.0337	0.0643	0.2142
t0	0.4136	0.7551	-1.2688	2.0961

Table 22. von Bertalanffy growth model parameter estimates using mean length at age (males)

Parameter	Estimate	Std Error	Approximate 95% Confidence Limits	
li	122.2	7.6163	105.0	139.5
k	0.1134	0.0196	0.0691	0.1577
t0	0.4276	0.5271	-0.7649	1.6200

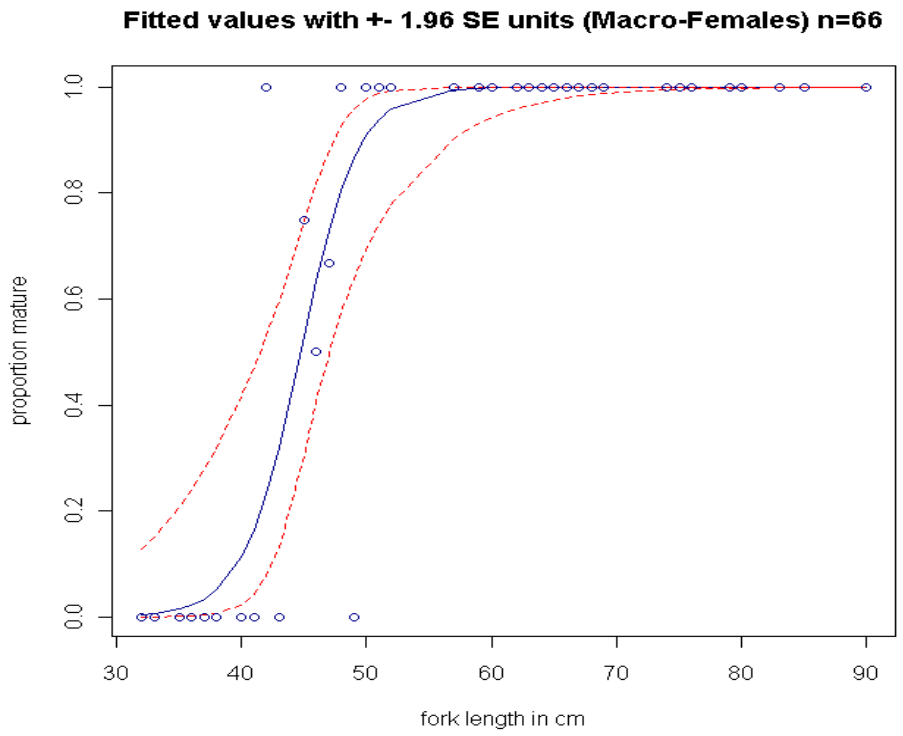

Figure 1. Maturity ogive for females based on macroscopic data (2008)

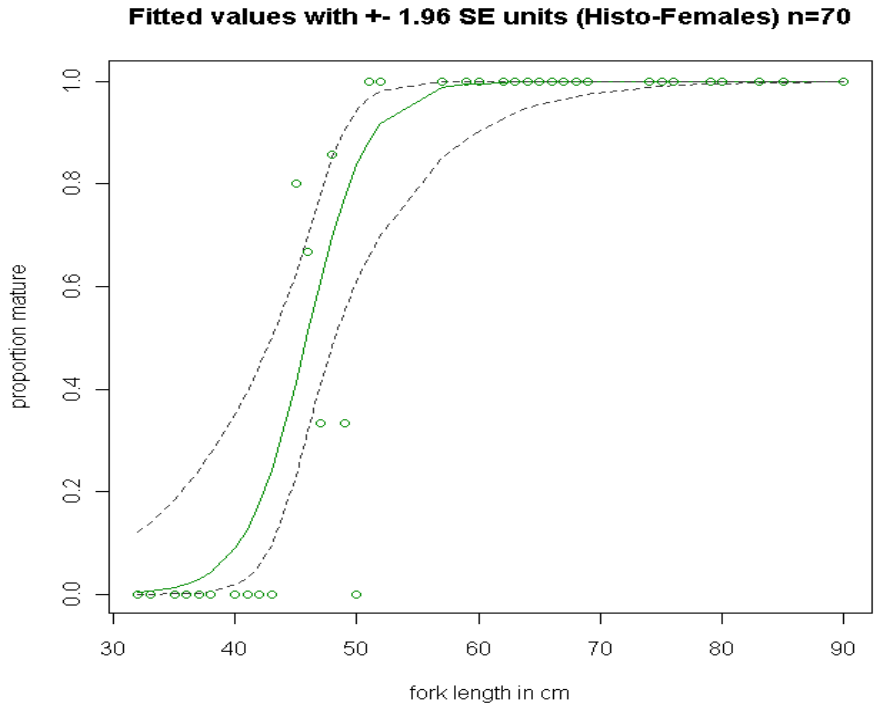

Figure 2. Maturity ogive for females based on histological data (2008)

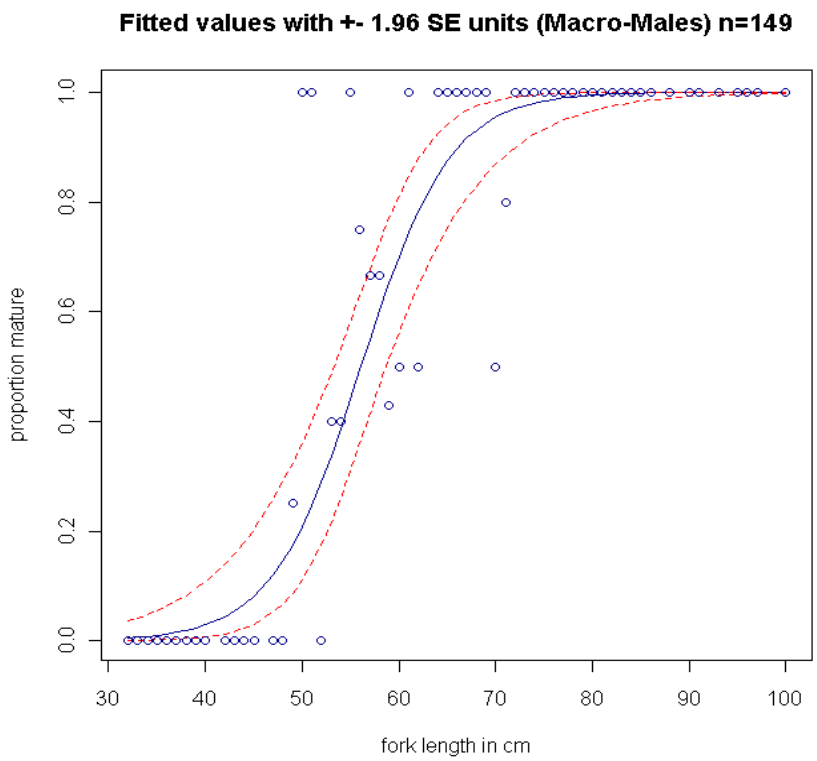

Figure 3. Maturity ogive for males based on macroscopic data (2008)

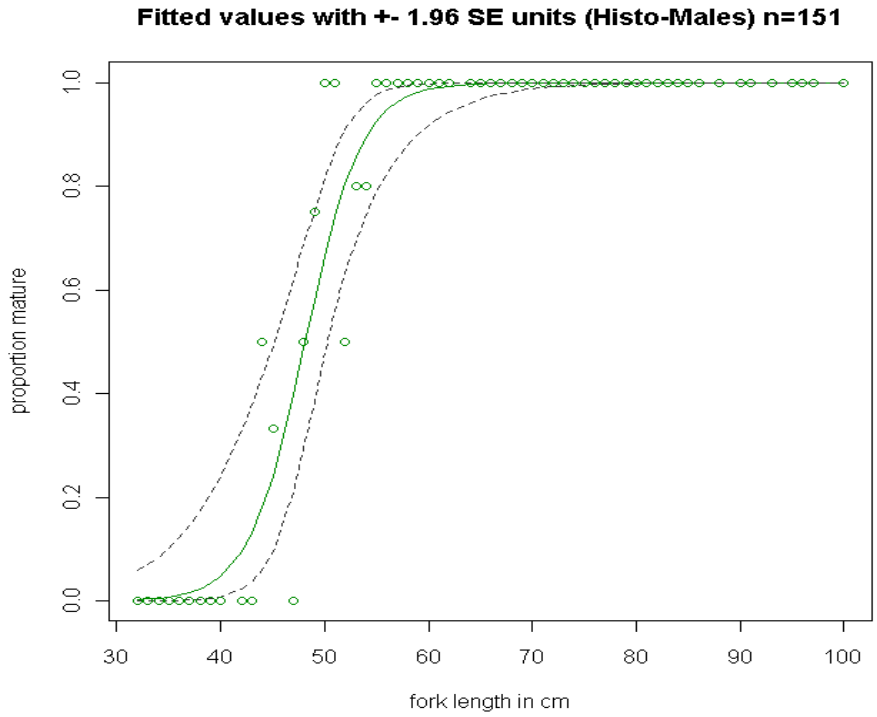

Figure 4. Maturity ogive for males based on histological data (2008)

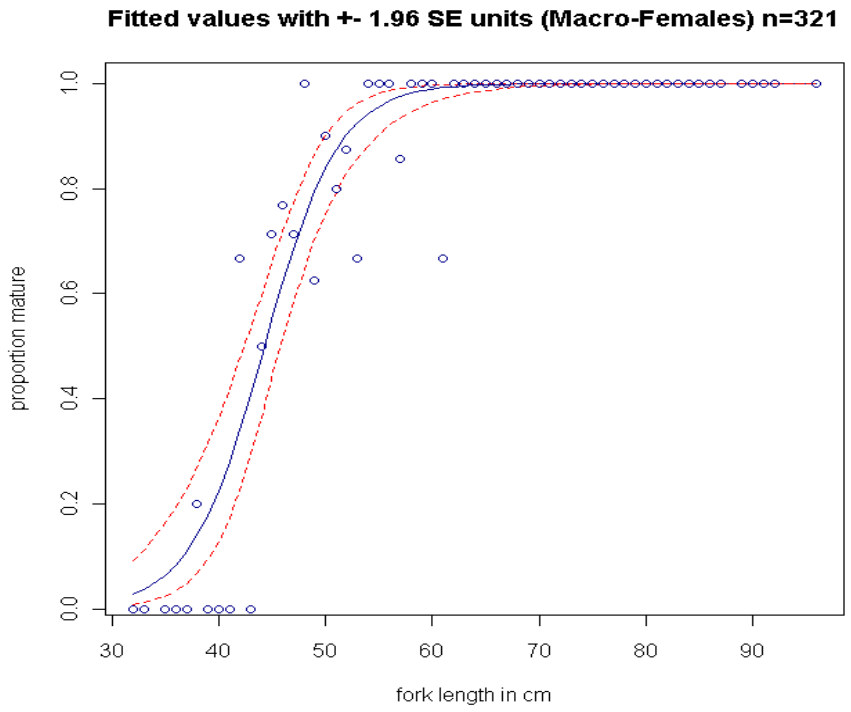

Figure 5. All macroscopic observations for females (2008)

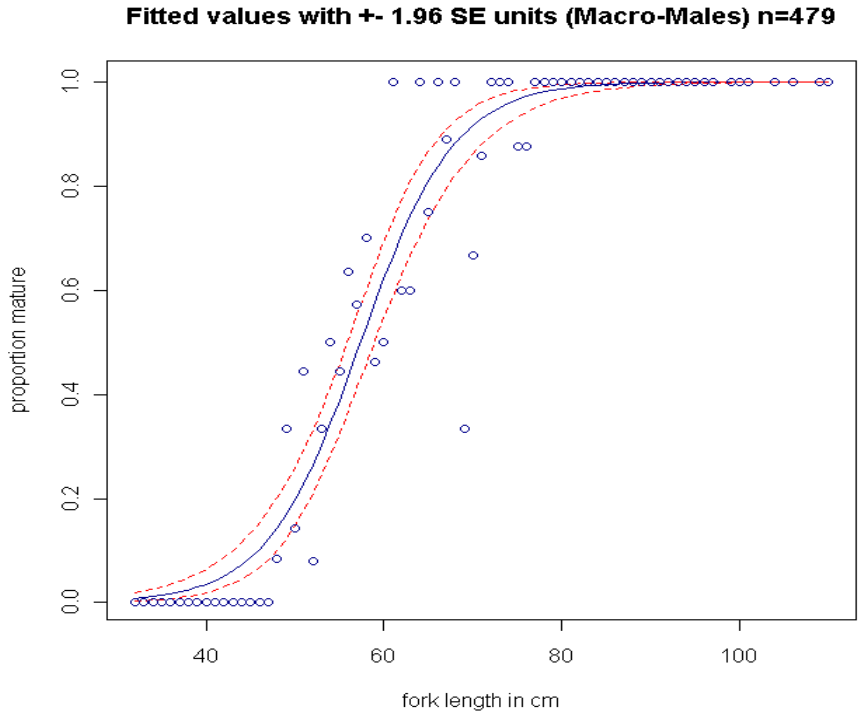

Figure 6. All macroscopic observations for males (2008)

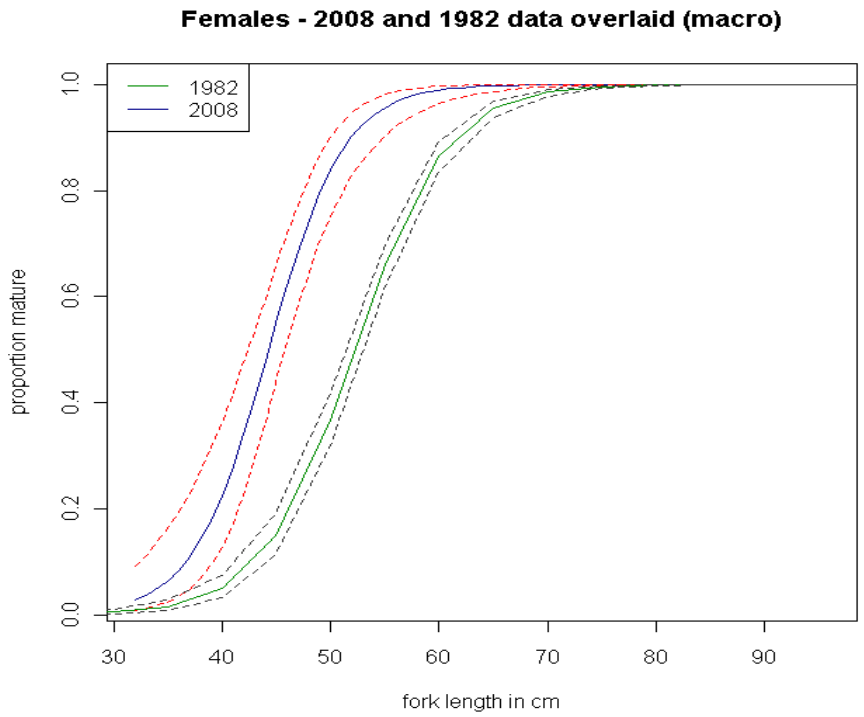

Figure 7. Maturity ogives, with 95% confidence limits, for the 1982 and 2008 females: green line=1982; blue line=2008. The 2008 data is based on all macroscopic observations..

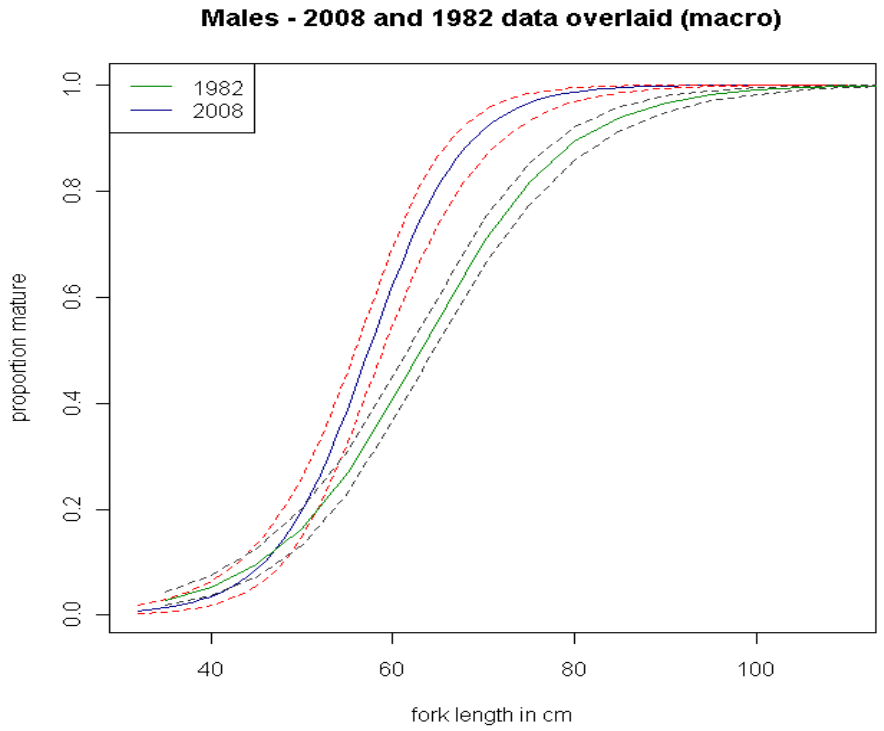

Figure 8. Maturity ogives, with 95% confidence limits, for the 1982 and 2008 males: green line=1982; blue line=2008. The 2008 data is based on all macroscopic observations

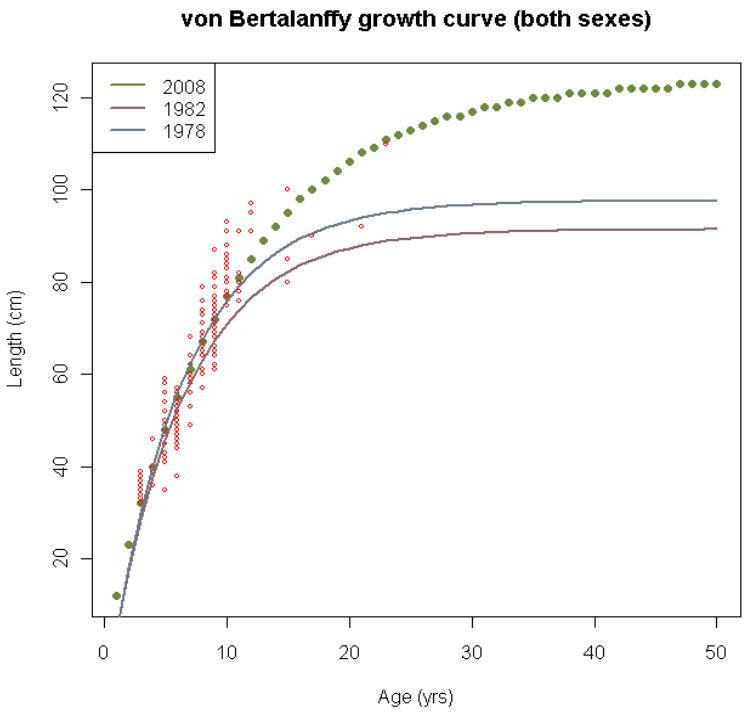

Figure 9. von Bertalanffy growth curve fit to observations of length at age (sexes combined)

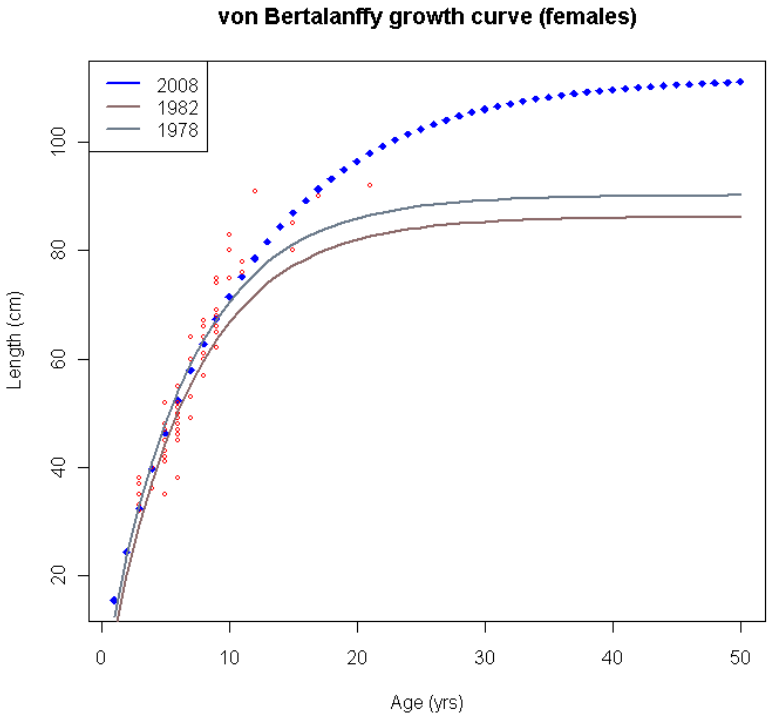

Figure 10. von Bertalanffy growth curve fit to observations of length at age (females)

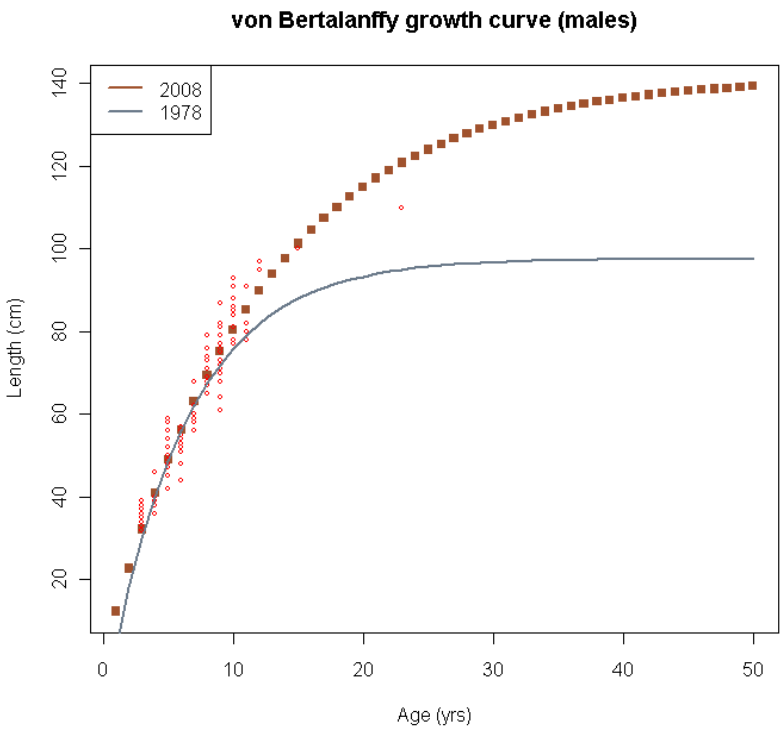

Figure 11. von Bertalanffy growth curve fit to observations of length at age (males)

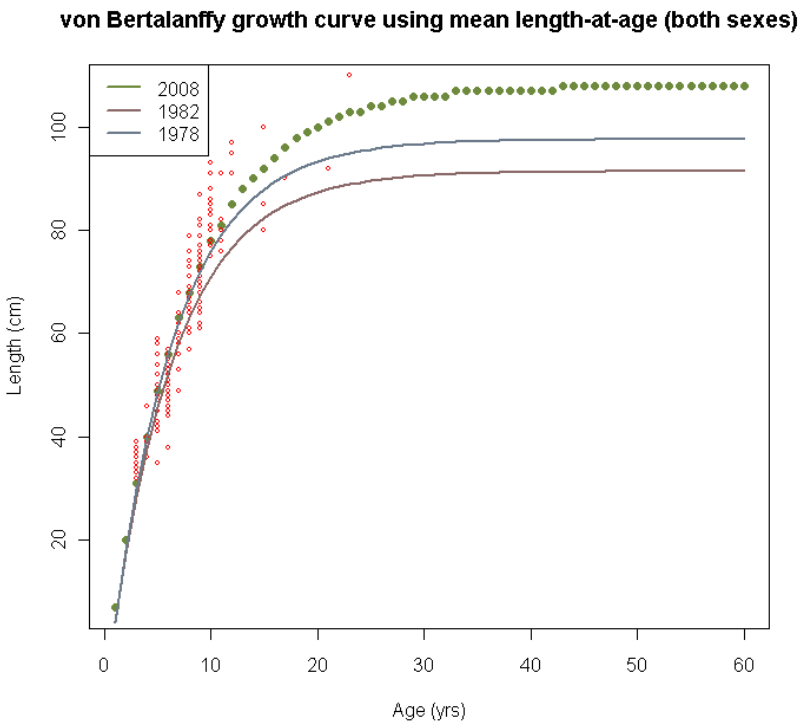

Figure 12. von Bertalanffy growth curve fit to mean length at age (sexes combined)

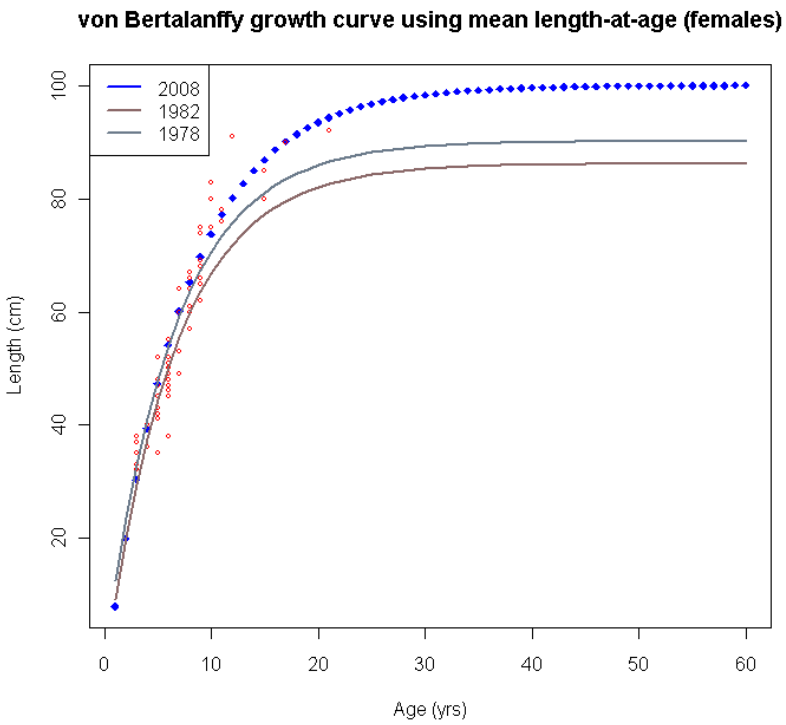

Figure 13. von Bertalanffy growth curve fit to mean length at age (females)

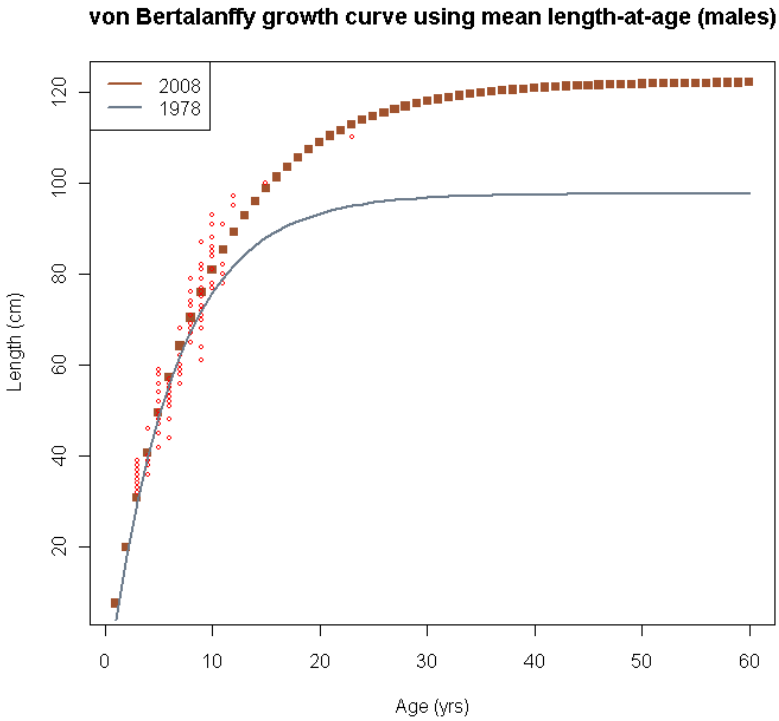

Figure 14. von Bertalanffy growth curve fit to mean length at age (males)

NEFSC Weighout CPUE GLM model

```
The SAS System
14:00 Thursday, March 31, 2005   1

The GLM Procedure
                                                        Class Level Information
Class       Levels  Values

lndyear         15  1979 1980 1981 1982 1983 1985 1986 1987 1988 1989 1990 1991 1992 1993 9999


permit          92        delete permit numbers
Number of observations    1897
The SAS System
 14:00 Thursday, March 31, 2005   2

The GLM Procedure
 Dependent Variable: LNCPUE

                                          Sum of
Source                      DF          Squares     Mean Square    F Value    Pr > F
Model                      105       743.569869        7.081618      23.67    <.0001
Error                     1791       535.787323        0.299155

Corrected Total           1896      1279.357192

R-Square     Coeff Var      Root MSE    LNCPUE Mean
0.581206      8.116663      0.546951       6.738619

Source                      DF        Type I SS     Mean Square    F Value    Pr > F
lndyear                     14      566.9637531      40.4974109     135.37    <.0001
permit                      91      176.6061156       1.9407265       6.49    <.0001

Source                      DF      Type III SS     Mean Square    F Value    Pr > F
lndyear                     14      281.1521083      20.0822934      67.13    <.0001
permit                      91      176.6061156       1.9407265       6.49    <.0001

                                            Standard
Parameter              Estimate              Error    t Value    Pr > |t|

Intercept           6.232567267 B       0.11429828      54.53      <.0001
lndyear   1979      1.022878443 B       0.07430951      13.77      <.0001
lndyear   1980      0.991305758 B       0.07181247      13.80      <.0001
lndyear   1981      0.957632235 B       0.07168379      13.36      <.0001
lndyear   1982      0.461931590 B       0.07359297       6.28      <.0001
lndyear   1983      0.036989477 B       0.07511938       0.49      0.6225
lndyear   1985     -0.116577906 B       0.07301030      -1.60      0.1105
lndyear   1986      0.078237855 B       0.07992860       0.98      0.3278
lndyear   1987      0.235247667 B       0.07689409       3.06      0.0023
lndyear   1988     -0.290869711 B       0.08580020      -3.39      0.0007
lndyear   1989     -0.437414680 B       0.11355219      -3.85      0.0001
lndyear   1990     -0.412418009 B       0.10524248      -3.92      <.0001
lndyear   1991     -0.462210977 B       0.09637704      -4.80      <.0001
lndyear   1992     -0.213720208 B       0.09349023      -2.29      0.0224
lndyear   1993     -0.277906028 B       0.09113548      -3.05      0.0023
lndyear   9999      0.000000000 B        .                .          .
permit    -         0.053877941 B       0.39953947       0.13      0.8927
permit    -         0.290799259 B       0.40217631       0.72      0.4697
permit    -         2.200653904 B       0.55660933       3.95      <.0001
permit    -        -0.720065816 B       0.33062733      -2.18      0.0295
permit    -         1.204048080 B       0.23673422       5.09      <.0001
permit    -        -0.918838210 B       0.55660933      -1.65      0.0990
permit    -         0.884977111 B       0.55660933       1.59      0.1120
permit    -         0.089186369 B       0.13030426       0.68      0.4938
permit    -         0.351073875 B       0.55660933       0.63      0.5283
permit    -        -0.474685588 B       0.40127024      -1.18      0.2370
permit    -        -1.051239079 B       0.55796370      -1.88      0.0597
permit    -         0.883791874 B       0.55876605       1.58      0.1139
permit    -         0.042036558 B       0.15197217       0.28      0.7821
```

permit	-	-2.501448583	B	0.55827964	-4.48	<.0001
permit	-	0.450272193	B	0.12822212	3.51	0.0005
permit	-	0.471191134	B	0.55809344	0.84	0.3986
permit	-	-0.050060896	B	0.14723604	-0.34	0.7339
permit	-	-0.138317903	B	0.24734699	-0.56	0.5761
permit	-	0.288864363	B	0.40301160	0.72	0.4736
permit	-	-0.719753788	B	0.55856606	-1.29	0.1977
permit	-	0.539895149	B	0.20257954	2.67	0.0078
permit	-	0.200325406	B	0.14810284	1.35	0.1764
permit	-	0.166798650	B	0.13012707	1.28	0.2001
permit	-	0.171959971	B	0.11302093	1.52	0.1283
permit	-	0.231976547	B	0.12244851	1.89	0.0583
permit	-	0.024125664	B	0.13432034	0.18	0.8575
permit	-	0.094051267	B	0.16446785	0.57	0.5675
permit	-	0.371090946	B	0.17507191	2.12	0.0342
permit	-	0.068525060	B	0.15621988	0.44	0.6610
permit	-	0.291237884	B	0.55606608	0.52	0.6005
permit	-	0.250774748	B	0.19444954	1.29	0.1973
permit	-	-1.365464039	B	0.19254217	-7.09	<.0001
permit	-	0.202892095	B	0.11692497	1.74	0.0829
permit	-	-0.150565146	B	0.55660933	-0.27	0.7868
permit	-	-1.227887492	B	0.55827964	-2.20	0.0280
permit	-	-1.316984788	B	0.55796370	-2.36	0.0184
permit	-	0.055682092	B	0.55606608	0.10	0.9202
permit	-	0.476788308	B	0.56089822	0.85	0.3954
permit	-	-1.513147475	B	0.22407363	-6.75	<.0001
permit	-	0.925030445	B	0.56089822	1.65	0.0993
permit	-	-0.260880622	B	0.40623775	-0.64	0.5208
permit	-	0.277147040	B	0.11033921	2.51	0.0121
permit	-	-0.894403775	B	0.26894018	-3.33	0.0009
permit	-	-0.087797738	B	0.21953680	-0.40	0.6893
permit	-	0.002668324	B	0.19877790	0.01	0.9893
permit	-	0.496364007	B	0.10872728	4.57	<.0001
permit	-	-0.163600190	B	0.55796370	-0.29	0.7694
permit	-	0.467983305	B	0.12033347	3.89	0.0001
permit	-	0.024708856	B	0.13276574	0.19	0.8524
permit	-	-1.665756882	B	0.40275435	-4.14	<.0001
permit	-	-0.008289609	B	0.21203679	-0.04	0.9688
permit	-	0.422212817	B	0.56253472	0.75	0.4530
permit	-	-0.994541917	B	0.41068120	-2.42	0.0155
permit	-	0.640814312	B	0.17122800	3.74	0.0002
permit	-	0.289229697	B	0.11245469	2.57	0.0102
permit	-	0.232020794	B	0.11406216	2.03	0.0421
permit	-	0.435287696	B	0.23285239	1.87	0.0617
permit	-	-0.093362255	B	0.55876605	-0.17	0.8673
permit	-	0.565119319	B	0.29382393	1.92	0.0546
permit	-	0.185883996	B	0.10864670	1.71	0.0873
permit	-	0.383628924	B	0.26777330	1.43	0.1521
permit	-	-0.429338431	B	0.15476255	-2.77	0.0056
permit	-	0.941153790	B	0.26751142	3.52	0.0004
permit	-	-0.144900138	B	0.55876605	-0.26	0.7954
permit	-	-0.018365360	B	0.39831869	-0.05	0.9632
permit	-	0.233109656	B	0.24325318	0.96	0.3380
permit	-	0.579583698	B	0.55656992	1.04	0.2979
permit	-	0.280357477	B	0.14815327	1.89	0.0586
permit	-	-0.220190021	B	0.33549831	-0.66	0.5117
permit	-	0.477244382	B	0.17126647	2.79	0.0054
permit	-	0.586558492	B	0.29544304	1.99	0.0473
permit	-	1.003951166	B	0.55606608	1.81	0.0712
permit	-	0.882877530	B	0.33498687	2.64	0.0085
permit	-	0.191509700	B	0.24286878	0.79	0.4305
permit	-	0.297364159	B	0.29099874	1.02	0.3070
permit	-	0.283495433	B	0.12957609	2.19	0.0288
permit	-	1.042813481	B	0.56089822	1.86	0.0632
permit	-	-0.065468315	B	0.19188028	-0.34	0.7330
permit	-	-0.153684912	B	0.40328873	-0.38	0.7032
permit	-	0.036432483	B	0.15621610	0.23	0.8156
permit	-	0.099929826	B	0.29223882	0.34	0.7324
permit	-	0.224377910	B	0.11753056	1.91	0.0564
permit	-	0.334472400	B	0.29263852	1.14	0.2532
permit	-	0.346528767	B	0.39933585	0.87	0.3856
permit	-	0.131354900	B	0.17613902	0.75	0.4559
permit	-	0.056859718	B	0.15272950	0.37	0.7097
permit	-	-1.420176111	B	0.55660933	-2.55	0.0108
permit	-	-1.054505031	B	0.33062733	-3.19	0.0015
permit	-	1.290671749	B	0.56253472	2.29	0.0219

permit	-	-0.545675103	B	0.55660933	-0.98	0.3270
permit	-	0.722755358	B	0.12789264	5.65	<.0001
permit	-	0.000000000	B	.	.	.

NEFSC VTR CPUE GLM model

```
The SAS System
14:33 Monday, March 9, 2009

The GLM Procedure
Class Level Information

Class       Levels  Values

lndyear         14  1995 1996 1997 1998 1999 2001 2002 2003 2004 2005 2006 2007
                    2008 9999

permit          28  delete permit numbers

Number of observations    1644

The SAS System
14:33 Monday, March 9, 2009

                              The GLM Procedure

Dependent Variable: LNCPUE
```

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	40	486.8316755	12.1707919	56.67	<.0001
Error	1603	344.2626234	0.2147615		
Corrected Total	1643	831.0942989			

R-Square	Coeff Var	Root MSE	LNCPUE Mean
0.585772	6.982976	0.463424	6.636478

Source	DF	Type I SS	Mean Square	F Value	Pr > F
lndyear	13	349.5367440	26.8874418	125.20	<.0001
permit	27	137.2949315	5.0849975	23.68	<.0001

Source	DF	Type III SS	Mean Square	F Value	Pr > F
lndyear	13	300.7498197	23.1346015	107.72	<.0001
permit	27	137.2949315	5.0849975	23.68	<.0001

Parameter		Estimate		Standard Error	t Value	Pr > |t|
Intercept		5.105961941	B	0.27514746	18.56	<.0001
lndyear	1995	-0.000311337	B	0.06567651	-0.00	0.9962
lndyear	1996	0.333314839	B	0.06159706	5.41	<.0001
lndyear	1997	0.849015959	B	0.06047455	14.04	<.0001
lndyear	1998	0.322043216	B	0.05885335	5.47	<.0001
lndyear	1999	-0.010958858	B	0.06068052	-0.18	0.8567
lndyear	2001	0.340009452	B	0.06244886	5.44	<.0001
lndyear	2002	0.541877218	B	0.06287945	8.62	<.0001
lndyear	2003	1.021480120	B	0.06520389	15.67	<.0001
lndyear	2004	1.324952771	B	0.06417921	20.64	<.0001
lndyear	2005	1.517578755	B	0.06802508	22.31	<.0001
lndyear	2006	1.193859874	B	0.06813050	17.52	<.0001
lndyear	2007	0.778697695	B	0.06658842	11.69	<.0001
lndyear	2008	0.358006552	B	0.06567768	5.45	<.0001
lndyear	9999	0.000000000	B	.	.	.
permit	-	0.971373595	B	0.53879108	1.80	0.0716
permit	-	-1.049233248	B	0.34106397	-3.08	0.0021

```
   permit    -     -0.211985376 B      0.42788650      -0.50      0.6204
   permit    -      0.637114469 B      0.29088986       2.19      0.0287
   permit    -      1.043620837 B      0.53836635       1.94      0.0527
   permit    -     -0.207701079 B      0.32349487      -0.64      0.5209
   permit    -      0.199074689 B      0.29734291       0.67      0.5033
   permit    -      0.795214347 B      0.33240705       2.39      0.0169
   permit    -      0.631300722 B      0.29044120       2.17      0.0299
   permit    -      0.056104033 B      0.28182625       0.20      0.8422
   permit    -      0.900218135 B      0.27302248       3.30      0.0010
   permit    -     -0.029499084 B      0.29005518      -0.10      0.9190
   permit    -      0.710693173 B      0.28013526       2.54      0.0113
   permit    -      0.490335540 B      0.31508786       1.56      0.1199
   permit    -      0.841245620 B      0.28298212       2.97      0.0030
   permit    -      1.922829272 B      0.53861803       3.57      0.0004
   permit    -      0.967713437 B      0.27304640       3.54      0.0004
   permit    -      0.370539541 B      0.30374715       1.22      0.2227
   permit    -     -1.091964427 B      0.53895045      -2.03      0.0429
   permit    -     -0.084261747 B      0.35851162      -0.24      0.8142
   permit    -      0.953641916 B      0.27327679       3.49      0.0005
   permit    -      0.929799416 B      0.28667927       3.24      0.0012
   permit    -      1.158830352 B      0.27203468       4.26      <.0001
   permit    -      0.552623254 B      0.35951185       1.54      0.1245
   permit    -     -1.584154615 B      0.53917468      -2.94      0.0033
   permit    -      0.944499945 B      0.28519020       3.31      0.0009
   permit    -      1.066086228 B      0.27210354       3.92      <.0001
   permit    -      0.000000000 B       .                .         .

NOTE: The X'X matrix has been found to be singular, and a generalized inverse
      was used to solve the normal equations.  Terms whose estimates are
      followed by the letter 'B' are not uniquely estimable.
```

ASPIC Base Boostrap run 1

```
TILEFISH 2009 BASE Bootstrap

    Wednesday, 11 Mar 2009 at 11:09:35
ASPIC -- A Surplus-Production Model Including Covariates (Ver. 5.33)

      BOT program mode
Author:         Michael H. Prager; NOAA Center for Coastal Fisheries and Habitat Research
                           LOGISTIC model mode   101 Pivers Island Road; Beaufort, North
Carolina  28516  USA                               YLD conditioning
Mike.Prager@noaa.gov
SSE optimization

Reference:  Prager, M. H. 1994. A suite of extensions to a nonequilibrium              ASPIC
User's Manual is available surplus-production model.  Fishery Bulletin 92: 374-389.
              gratis from the author.

CONTROL  PARAMETERS  (FROM  INPUT  FILE)                                          Input  file:
c:\tile2009\aspic\f73fix1_v5_2008_base_boot.inp
-------------------------------------------------------------------------------------------------
-------------------
Operation of ASPIC:  Fit logistic (Schaefer) model by direct optimization with bootstrap.
Number of years analyzed:        36         Number of bootstrap trials:     1000
Number of data series:            3         Bounds on MSY (min, max): 3.750E-02 3.000E+02
Objective function:      Least squares      Bounds on K (min, max):  8.000E-01  2.000E+03
Relative conv. criterion (simplex):       1.000E-08
Monte Carlo search mode, trials:          1         50000
Relative     conv.     criterion     (restart):                                     3.000E-08
Random number seed:                       973142085
Relative conv. criterion (effort):        1.000E-04
Identical convergences required in fitting:       6
Maximum F allowed in fitting:                 5.000

PROGRAM STATUS INFORMATION (NON-BOOTSTRAPPED ANALYSIS)
        error code   0
-----------------------------------------------------------------------------------------
Normal convergence

CORRELATION AMONG INPUT SERIES EXPRESSED AS CPUE (NUMBER OF PAIRWISE OBSERVATIONS BELOW)
-----------------------------------------------------------------------------------------
                                        |
 1  weighout cpue                       |   1.000
```

```
                                    |      15
                                    |
 2  turner                          |   0.994   1.000
                                    |       4      10
                                    |
 3  vtr                              |   0.000   0.000   1.000
                                    |       0       0      14
                                    ------------------------------------------------
                                            1       2       3


GOODNESS-OF-FIT AND WEIGHTING (NON-BOOTSTRAPPED ANALYSIS)
-----------------------------------------------------------------------------------------

                                  Weighted      Weighted     Current    Inv. var.    R-squared
Loss component number and title     SSE     N      MSE        weight      weight      in CPUE

Loss(-1)  SSE in yield           0.000E+00
Loss(0)   Penalty for B1 > K     0.000E+00   1      N/A         0.000E+00       N/A
Loss(1)   weighout cpue          1.255E+00   15    9.653E-02    1.000E+00    1.141E+00   0.652
Loss(2)   turner                 6.429E-01   10    8.037E-02    1.000E+00    1.370E+00   0.224
Loss(3)   vtr                    2.258E+00   14    1.881E-01    1.000E+00    5.852E-01   0.201
.......................................................................................
TOTAL OBJECTIVE FUNCTION, MSE, RMSE:          4.15554979E+00          1.222E-01    3.496E-01
Estimated contrast index (ideal = 1.0):               0.5261          C* = (Bmax-Bmin)/K
Estimated nearness index (ideal = 1.0):               1.0000          N* = 1 - |min(B-Bmsy)|/K
TILEFISH                        2009                       BASE                          Bootstrap

MODEL PARAMETER ESTIMATES (NON-BOOTSTRAPPED)
-----------------------------------------------------------------------------------------------
Parameter                                 Estimate User/pgm guess 2nd guess  Estimated User guess

B1/K Starting relative biomass (in 1973)  5.000E-01  5.000E-01    9.000E-01     0            1
MSY  Maximum sustainable yield            1.868E+00  3.000E+00    1.271E+00     1            1
K    Maximum population size              2.279E+01  4.000E+01    7.629E+00     1            1
phi  Shape of production curve (Bmsy/K)   0.5000     0.5000            ----     0            1

--------- Catchability Coefficients by Data Series ---------------
q(1)      weighout cpue                 1.754E-01   3.000E-02    4.750E-01      1            1
q(2)      turner                        8.791E-03   3.000E-02    4.750E-01      1            1
q(3)      vtr                           2.604E-01   3.000E-02    4.750E-01      1            1

MANAGEMENT and DERIVED PARAMETER ESTIMATES (NON-BOOTSTRAPPED)
-----------------------------------------------------------------------------------------------
Parameter                                Estimate          Logistic formula           General formula

MSY   Maximum sustainable yield          1.868E+00            ----                        ----
Bmsy  Stock biomass giving MSY           1.140E+01            K/2            K*n**(1/(1-n))
Fmsy  Fishing mortality rate at MSY      1.639E-01          MSY/Bmsy             MSY/Bmsy

n     Exponent in production function      2.0000             ----                        ----
g     Fletcher's gamma                   4.000E+00            ----      [n**(n/(n-1))]/[n-1]

B./Bmsy Ratio: B(2009)/Bmsy              1.143E+00            ----                        ----
F./Fmsy Ratio: F(2008)/Fmsy              3.598E-01            ----                        ----
Fmsy/F. Ratio: Fmsy/F(2008)              2.779E+00            ----                        ----

Y.(Fmsy) Approx. yield available at Fmsy in 2009   2.136E+00   MSY*B./Bmsy         MSY*B./Bmsy
          ...as proportion of MSY         1.143E+00            ----                        ----
Ye.  Equilibrium yield available in 2009  1.830E+00     4*MSY*(B/K-(B/K)**2) g*MSY*(B/K-(B/K)**n)
          ...as proportion of MSY         9.795E-01            ----                        ----

--------- Fishing effort rate at MSY in units of each CE or CC series ---------
fmsy(1)   weighout cpue                  9.349E-01           Fmsy/q( 1)                Fmsy/q( 1)
```

TILEFISH 2009 BASE Bootstrap

ESTIMATED POPULATION TRAJECTORY (NON-BOOTSTRAPPED)

Obs	Year or ID	Estimated total F mort	Estimated starting biomass	Estimated average biomass	Observed total yield	Model total yield	Estimated surplus production	Ratio of F mort to Fmsy	Ratio of biomass to Bmsy
1	1973	0.031	1.140E+01	1.259E+01	3.940E-01	3.940E-01	1.872E+00	1.908E-01	9.999E-01
2	1974	0.043	1.287E+01	1.349E+01	5.860E-01	5.860E-01	1.803E+00	2.649E-01	1.130E+00
3	1975	0.049	1.409E+01	1.461E+01	7.100E-01	7.100E-01	1.719E+00	2.965E-01	1.236E+00
4	1976	0.065	1.510E+01	1.542E+01	1.010E+00	1.010E+00	1.635E+00	3.995E-01	1.325E+00
5	1977	0.134	1.573E+01	1.549E+01	2.082E+00	2.082E+00	1.627E+00	8.200E-01	1.380E+00
6	1978	0.225	1.527E+01	1.447E+01	3.257E+00	3.257E+00	1.730E+00	1.373E+00	1.340E+00
7	1979	0.314	1.374E+01	1.262E+01	3.968E+00	3.968E+00	1.842E+00	1.918E+00	1.206E+00
8	1980	0.369	1.162E+01	1.054E+01	3.889E+00	3.889E+00	1.853E+00	2.251E+00	1.019E+00
9	1981	0.404	9.581E+00	8.663E+00	3.499E+00	3.499E+00	1.757E+00	2.464E+00	8.407E-01
10	1982	0.259	7.839E+00	7.675E+00	1.990E+00	1.990E+00	1.669E+00	1.582E+00	6.878E-01
11	1983	0.254	7.518E+00	7.396E+00	1.877E+00	1.877E+00	1.638E+00	1.548E+00	6.597E-01
12	1984	0.284	7.279E+00	7.069E+00	2.009E+00	2.009E+00	1.599E+00	1.734E+00	6.387E-01
13	1985	0.295	6.869E+00	6.656E+00	1.961E+00	1.961E+00	1.545E+00	1.797E+00	6.027E-01
14	1986	0.314	6.453E+00	6.212E+00	1.950E+00	1.950E+00	1.482E+00	1.915E+00	5.662E-01
15	1987	0.650	5.984E+00	4.936E+00	3.210E+00	3.210E+00	1.264E+00	3.967E+00	5.251E-01
16	1988	0.351	4.038E+00	3.882E+00	1.361E+00	1.361E+00	1.056E+00	2.139E+00	3.543E-01
17	1989	0.085	3.733E+00	5.323E+00	4.540E-01	4.540E-01	1.206E+00	5.203E-01	3.275E-01
18	1990	0.153	4.485E+00	5.717E+00	8.740E-01	8.740E-01	1.380E+00	9.325E-01	3.936E-01
19	1991	0.236	4.991E+00	5.041E+00	1.189E+00	1.189E+00	1.287E+00	1.439E+00	4.380E-01
20	1992	0.338	5.090E+00	4.888E+00	1.653E+00	1.653E+00	1.259E+00	2.063E+00	4.466E-01
21	1993	0.424	4.696E+00	4.340E+00	1.838E+00	1.838E+00	1.152E+00	2.583E+00	4.120E-01
22	1994	0.146	4.009E+00	5.374E+00	7.860E-01	7.860E-01	1.299E+00	8.922E-01	3.518E-01
23	1995	0.139	4.522E+00	4.808E+00	6.660E-01	6.660E-01	1.244E+00	8.450E-01	3.967E-01
24	1996	0.216	5.099E+00	5.197E+00	1.121E+00	1.121E+00	1.315E+00	1.316E+00	4.474E-01
25	1997	0.360	5.294E+00	5.023E+00	1.810E+00	1.810E+00	1.284E+00	2.198E+00	4.645E-01
26	1998	0.285	4.767E+00	4.708E+00	1.342E+00	1.342E+00	1.225E+00	1.739E+00	4.183E-01
27	1999	0.104	4.650E+00	5.024E+00	5.250E-01	5.250E-01	1.284E+00	6.374E-01	4.080E-01
28	2000	0.086	5.409E+00	5.864E+00	5.060E-01	5.060E-01	1.427E+00	5.264E-01	4.746E-01
29	2001	0.131	6.330E+00	6.665E+00	8.740E-01	8.740E-01	1.546E+00	7.998E-01	5.554E-01
30	2002	0.115	7.002E+00	7.395E+00	8.510E-01	8.510E-01	1.637E+00	7.020E-01	6.144E-01
31	2003	0.140	7.788E+00	8.080E+00	1.130E+00	1.130E+00	1.710E+00	8.531E-01	6.834E-01
32	2004	0.141	8.368E+00	8.643E+00	1.215E+00	1.215E+00	1.759E+00	8.575E-01	7.342E-01
33	2005	0.092	8.912E+00	9.385E+00	8.680E-01	8.680E-01	1.809E+00	5.642E-01	7.820E-01
34	2006	0.088	9.853E+00	1.033E+01	9.070E-01	9.070E-01	1.851E+00	5.356E-01	8.645E-01
35	2007	0.066	1.080E+01	1.136E+01	7.510E-01	7.510E-01	1.867E+00	4.032E-01	9.474E-01
36	2008	0.059	1.191E+01	1.248E+01	7.360E-01	7.360E-01	1.850E+00	3.598E-01	1.045E+00
37	2009		1.303E+01						1.143E+00

TILEFISH 2009 BASE Bootstrap

RESULTS FOR DATA SERIES # 1 (NON-BOOTSTRAPPED)
weighout cpue

Data type CC: CPUE-catch series
Series weight: 1.000

Obs	Year	Observed CPUE	Estimated CPUE	Estim F	Observed yield	Model yield	Resid in log scale	Statist weight
1	1973	*	2.209E+00	0.0313	3.940E-01	3.940E-01	0.00000	1.000E+00
2	1974	*	2.366E+00	0.0434	5.860E-01	5.860E-01	0.00000	1.000E+00
3	1975	*	2.562E+00	0.0486	7.100E-01	7.100E-01	0.00000	1.000E+00
4	1976	*	2.705E+00	0.0655	1.010E+00	1.010E+00	0.00000	1.000E+00
5	1977	*	2.716E+00	0.1344	2.082E+00	2.082E+00	0.00000	1.000E+00
6	1978	*	2.537E+00	0.2251	3.257E+00	3.257E+00	0.00000	1.000E+00
7	1979	2.789E+00	2.213E+00	0.3145	3.968E+00	3.968E+00	-0.23140	1.000E+00
8	1980	2.702E+00	1.848E+00	0.3690	3.889E+00	3.889E+00	-0.37962	1.000E+00
9	1981	2.612E+00	1.519E+00	0.4039	3.499E+00	3.499E+00	-0.54194	1.000E+00
10	1982	1.591E+00	1.346E+00	0.2593	1.990E+00	1.990E+00	-0.16731	1.000E+00
11	1983	1.041E+00	1.297E+00	0.2538	1.877E+00	1.877E+00	0.21989	1.000E+00
12	1984	1.000E+00	1.240E+00	0.2842	2.009E+00	2.009E+00	0.21482	1.000E+00
13	1985	8.920E-01	1.167E+00	0.2946	1.961E+00	1.961E+00	0.26888	1.000E+00
14	1986	1.085E+00	1.089E+00	0.3139	1.950E+00	1.950E+00	0.00409	1.000E+00
15	1987	1.269E+00	8.656E-01	0.6503	3.210E+00	3.210E+00	-0.38250	1.000E+00
16	1988	7.500E-01	6.808E-01	0.3506	1.361E+00	1.361E+00	-0.09680	1.000E+00
17	1989	6.500E-01	9.335E-01	0.0853	4.540E-01	4.540E-01	0.36198	1.000E+00
18	1990	6.660E-01	1.003E+00	0.1529	8.740E-01	8.740E-01	0.40913	1.000E+00
19	1991	6.330E-01	8.840E-01	0.2359	1.189E+00	1.189E+00	0.33403	1.000E+00
20	1992	8.110E-01	8.572E-01	0.3382	1.653E+00	1.653E+00	0.05536	1.000E+00
21	1993	7.610E-01	7.611E-01	0.4235	1.838E+00	1.838E+00	0.00008	1.000E+00
22	1994	*	9.424E-01	0.1463	7.860E-01	7.860E-01	0.00000	1.000E+00
23	1995	*	8.432E-01	0.1385	6.660E-01	6.660E-01	0.00000	1.000E+00
24	1996	*	9.114E-01	0.2157	1.121E+00	1.121E+00	0.00000	1.000E+00
25	1997	*	8.808E-01	0.3604	1.810E+00	1.810E+00	0.00000	1.000E+00
26	1998	*	8.256E-01	0.2851	1.342E+00	1.342E+00	0.00000	1.000E+00
27	1999	*	8.811E-01	0.1045	5.250E-01	5.250E-01	0.00000	1.000E+00
28	2000	*	1.028E+00	0.0863	5.060E-01	5.060E-01	0.00000	1.000E+00
29	2001	*	1.169E+00	0.1311	8.740E-01	8.740E-01	0.00000	1.000E+00
30	2002	*	1.297E+00	0.1151	8.510E-01	8.510E-01	0.00000	1.000E+00
31	2003	*	1.417E+00	0.1398	1.130E+00	1.130E+00	0.00000	1.000E+00
32	2004	*	1.516E+00	0.1406	1.215E+00	1.215E+00	0.00000	1.000E+00
33	2005	*	1.646E+00	0.0925	8.680E-01	8.680E-01	0.00000	1.000E+00
34	2006	*	1.811E+00	0.0878	9.070E-01	9.070E-01	0.00000	1.000E+00
35	2007	*	1.992E+00	0.0661	7.510E-01	7.510E-01	0.00000	1.000E+00
36	2008	*	2.188E+00	0.0590	7.360E-01	7.360E-01	0.00000	1.000E+00

* Asterisk indicates missing value(s).

TILEFISH 2009 BASE Bootstrap

RESULTS FOR DATA SERIES # 2 (NON-BOOTSTRAPPED)
turner

Data type I1: Abundance index (annual average)
Series weight: 1.000

Obs	Year	Observed effort	Estimated effort	Estim F	Observed index	Model index	Resid in log index	Statist weight
1	1973	1.000E+00	1.000E+00	--	2.060E-01	1.107E-01	0.62086	1.000E+00
2	1974	1.000E+00	1.000E+00	--	1.350E-01	1.186E-01	0.12930	1.000E+00
3	1975	1.000E+00	1.000E+00	--	9.600E-02	1.284E-01	-0.29099	1.000E+00
4	1976	1.000E+00	1.000E+00	--	1.140E-01	1.356E-01	-0.17339	1.000E+00
5	1977	1.000E+00	1.000E+00	--	1.250E-01	1.362E-01	-0.08555	1.000E+00
6	1978	1.000E+00	1.000E+00	--	1.320E-01	1.272E-01	0.03715	1.000E+00
7	1979	1.000E+00	1.000E+00	--	1.000E-01	1.109E-01	-0.10375	1.000E+00
8	1980	1.000E+00	1.000E+00	--	9.100E-02	9.267E-02	-0.01815	1.000E+00
9	1981	1.000E+00	1.000E+00	--	9.000E-02	7.616E-02	0.16699	1.000E+00
10	1982	1.000E+00	1.000E+00	--	5.100E-02	6.747E-02	-0.27987	1.000E+00
11	1983	0.000E+00	0.000E+00	--	*	6.502E-02	0.00000	1.000E+00
12	1984	0.000E+00	0.000E+00	--	*	6.214E-02	0.00000	1.000E+00
13	1985	0.000E+00	0.000E+00	--	*	5.851E-02	0.00000	1.000E+00
14	1986	0.000E+00	0.000E+00	--	*	5.462E-02	0.00000	1.000E+00
15	1987	0.000E+00	0.000E+00	--	*	4.340E-02	0.00000	1.000E+00
16	1988	0.000E+00	0.000E+00	--	*	3.413E-02	0.00000	1.000E+00
17	1989	0.000E+00	0.000E+00	--	*	4.680E-02	0.00000	1.000E+00
18	1990	0.000E+00	0.000E+00	--	*	5.027E-02	0.00000	1.000E+00
19	1991	0.000E+00	0.000E+00	--	*	4.432E-02	0.00000	1.000E+00
20	1992	0.000E+00	0.000E+00	--	*	4.297E-02	0.00000	1.000E+00
21	1993	0.000E+00	0.000E+00	--	*	3.815E-02	0.00000	1.000E+00
22	1994	0.000E+00	0.000E+00	--	*	4.724E-02	0.00000	1.000E+00
23	1995	0.000E+00	0.000E+00	--	*	4.227E-02	0.00000	1.000E+00
24	1996	0.000E+00	0.000E+00	--	*	4.569E-02	0.00000	1.000E+00
25	1997	0.000E+00	0.000E+00	--	*	4.416E-02	0.00000	1.000E+00
26	1998	0.000E+00	0.000E+00	--	*	4.139E-02	0.00000	1.000E+00
27	1999	0.000E+00	0.000E+00	--	*	4.417E-02	0.00000	1.000E+00
28	2000	0.000E+00	0.000E+00	--	*	5.155E-02	0.00000	1.000E+00
29	2001	0.000E+00	0.000E+00	--	*	5.860E-02	0.00000	1.000E+00
30	2002	0.000E+00	0.000E+00	--	*	6.501E-02	0.00000	1.000E+00
31	2003	0.000E+00	0.000E+00	--	*	7.104E-02	0.00000	1.000E+00
32	2004	0.000E+00	0.000E+00	--	*	7.598E-02	0.00000	1.000E+00
33	2005	0.000E+00	0.000E+00	--	*	8.251E-02	0.00000	1.000E+00
34	2006	0.000E+00	0.000E+00	--	*	9.081E-02	0.00000	1.000E+00
35	2007	0.000E+00	0.000E+00	--	*	9.988E-02	0.00000	1.000E+00
36	2008	0.000E+00	0.000E+00	--	*	1.097E-01	0.00000	1.000E+00

* Asterisk indicates missing value(s).

TILEFISH 2009 BASE Bootstrap

RESULTS FOR DATA SERIES # 3 (NON-BOOTSTRAPPED)
vtr

type I1: Abundance index (annual average) Data Series
weight: 1.000

Obs	Year	Observed effort	Estimated effort	Estim F	Observed index	Model index	Resid in log index	Statist weight
1	1973	0.000E+00	0.000E+00	--	*	3.279E+00	0.00000	1.000E+00
2	1974	0.000E+00	0.000E+00	--	*	3.513E+00	0.00000	1.000E+00
3	1975	0.000E+00	0.000E+00	--	*	3.803E+00	0.00000	1.000E+00
4	1976	0.000E+00	0.000E+00	--	*	4.015E+00	0.00000	1.000E+00
5	1977	0.000E+00	0.000E+00	--	*	4.032E+00	0.00000	1.000E+00
6	1978	0.000E+00	0.000E+00	--	*	3.767E+00	0.00000	1.000E+00
7	1979	0.000E+00	0.000E+00	--	*	3.285E+00	0.00000	1.000E+00
8	1980	0.000E+00	0.000E+00	--	*	2.744E+00	0.00000	1.000E+00
9	1981	0.000E+00	0.000E+00	--	*	2.255E+00	0.00000	1.000E+00
10	1982	0.000E+00	0.000E+00	--	*	1.998E+00	0.00000	1.000E+00
11	1983	0.000E+00	0.000E+00	--	*	1.926E+00	0.00000	1.000E+00
12	1984	0.000E+00	0.000E+00	--	*	1.840E+00	0.00000	1.000E+00
13	1985	0.000E+00	0.000E+00	--	*	1.733E+00	0.00000	1.000E+00
14	1986	0.000E+00	0.000E+00	--	*	1.617E+00	0.00000	1.000E+00
15	1987	0.000E+00	0.000E+00	--	*	1.285E+00	0.00000	1.000E+00
16	1988	0.000E+00	0.000E+00	--	*	1.011E+00	0.00000	1.000E+00
17	1989	0.000E+00	0.000E+00	--	*	1.386E+00	0.00000	1.000E+00
18	1990	0.000E+00	0.000E+00	--	*	1.489E+00	0.00000	1.000E+00
19	1991	0.000E+00	0.000E+00	--	*	1.312E+00	0.00000	1.000E+00
20	1992	0.000E+00	0.000E+00	--	*	1.273E+00	0.00000	1.000E+00
21	1993	0.000E+00	0.000E+00	--	*	1.130E+00	0.00000	1.000E+00
22	1994	0.000E+00	0.000E+00	--	*	1.399E+00	0.00000	1.000E+00
23	1995	1.000E+00	1.000E+00	--	1.002E+00	1.252E+00	-0.22256	1.000E+00
24	1996	1.000E+00	1.000E+00	--	1.398E+00	1.353E+00	0.03267	1.000E+00
25	1997	1.000E+00	1.000E+00	--	2.342E+00	1.308E+00	0.58275	1.000E+00
26	1998	1.000E+00	1.000E+00	--	1.382E+00	1.226E+00	0.12002	1.000E+00
27	1999	1.000E+00	1.000E+00	--	9.910E-01	1.308E+00	-0.27765	1.000E+00
28	2000	1.000E+00	1.000E+00	--	1.000E+00	1.527E+00	-0.42307	1.000E+00
29	2001	1.000E+00	1.000E+00	--	1.408E+00	1.735E+00	-0.20906	1.000E+00
30	2002	1.000E+00	1.000E+00	--	1.723E+00	1.925E+00	-0.11101	1.000E+00
31	2003	1.000E+00	1.000E+00	--	2.783E+00	2.104E+00	0.27984	1.000E+00
32	2004	1.000E+00	1.000E+00	--	3.770E+00	2.250E+00	0.51608	1.000E+00
33	2005	1.000E+00	1.000E+00	--	4.572E+00	2.443E+00	0.62654	1.000E+00
34	2006	1.000E+00	1.000E+00	--	3.307E+00	2.689E+00	0.20676	1.000E+00
35	2007	1.000E+00	1.000E+00	--	2.183E+00	2.958E+00	-0.30376	1.000E+00
36	2008	1.000E+00	1.000E+00	--	1.434E+00	3.249E+00	-0.81780	1.000E+00

* Asterisk indicates missing value(s)

TILEFISH 2009 BASE Bootstrap
ESTIMATES FROM BOOTSTRAPPED ANALYSIS

Param name	Point estimate	Estimated bias in pt estimate	Estimated relative bias	Bias-corrected approximate confidence limits 80% lower	80% upper	50% lower	50% upper	Inter-quartile range	Relative IQ range
B1/K	5.000E-01	1.223E-09	0.00%	5.000E-01	5.000E-01	5.000E-01	5.000E-01	6.215E-11	0.000
K	2.279E+01	-2.125E+00	-9.32%	2.037E+01	3.705E+01	2.275E+01	3.026E+01	7.509E+00	0.329
q(1)	1.754E-01	2.847E-02	16.23%	9.546E-02	2.078E-01	1.194E-01	1.797E-01	6.030E-02	0.344
q(2)	8.791E-03	1.139E-03	12.95%	6.002E-03	1.012E-02	6.967E-03	9.109E-03	2.142E-03	0.244
q(3)	2.604E-01	2.603E-02	10.00%	1.629E-01	3.851E-01	1.931E-01	3.145E-01	1.214E-01	0.466
MSY	1.868E+00	8.060E-02	4.31%	1.577E+00	1.927E+00	1.699E+00	1.869E+00	1.699E-01	0.091
Ye(2009)	1.830E+00	-2.391E-01	-13.06%	1.640E+00	2.010E+00	1.806E+00	1.973E+00	1.670E-01	0.091
Y.@Fmsy	2.136E+00	3.323E-01	15.56%	1.190E+00	3.115E+00	1.514E+00	2.518E+00	1.004E+00	0.470
Bmsy	1.140E+01	-1.062E+00	-9.32%	1.019E+01	1.853E+01	1.138E+01	1.513E+01	3.755E+00	0.329
Fmsy	1.639E-01	3.430E-02	20.92%	8.329E-02	1.899E-01	1.130E-01	1.653E-01	5.230E-02	0.319
fmsy(1)	9.349E-01	5.018E-02	5.37%	8.046E-01	1.060E+00	8.588E-01	9.864E-01	1.276E-01	0.136
fmsy(2)	1.865E+01	1.288E+00	6.90%	1.459E+01	2.134E+01	1.609E+01	1.954E+01	3.450E+00	0.185
fmsy(3)	6.296E-01	1.366E-01	21.69%	4.375E-01	1.150E+00	5.026E-01	8.316E-01	3.291E-01	0.523
B./Bmsy	1.143E+00	1.063E-01	9.30%	6.972E-01	1.597E+00	8.432E-01	1.317E+00	4.743E-01	0.415
F./Fmsy	3.598E-01	-1.037E-02	-2.88%	2.404E-01	6.314E-01	3.019E-01	5.095E-01	2.076E-01	0.577
Ye./MSY	9.795E-01	-1.528E-01	-15.60%	9.078E-01	1.000E+00	9.768E-01	9.999E-01	2.303E-02	0.024
q2/q1	5.013E-02	-3.456E-04	-0.69%	4.188E-02	6.291E-02	4.625E-02	5.674E-02	1.050E-02	0.209
q3/q1	1.485E+00	-3.899E-02	-2.63%	8.846E-01	2.096E+00	1.145E+00	1.827E+00	6.825E-01	0.460

INFORMATION FOR REPAST (Prager, Porch, Shertzer, & Caddy. 2003. NAJFM 23: 349-361)

Unitless limit reference point in F (Fmsy/F.): 2.779
CV of above (from bootstrap distribution): 0.4376

NOTES ON BOOTSTRAPPED ESTIMATES:

- Bootstrap results were computed from 1000 trials.
- Results are conditional on bounds set on MSY and K in the input file.
- All bootstrapped intervals are approximate. The statistical literature recommends using at least 1000 trials for accurate 95% intervals. The default 80% intervals used by ASPIC should require fewer trials for equivalent accuracy. Using at least 500 trials is recommended.
- Bias estimates are typically of high variance and therefore may be misleading.

Trials replaced for lack of convergence: 0
Trials replaced for MSY out of bounds: 0
Trials replaced for q out-of-bounds: 139
Trials replaced for K out-of-bounds: 0
Residual-adjustment factor: 1.0710
Elapsed time: 0 hours, 8 minutes, 23 seconds.

Ocean quahog

B. Stock assessment for ocean quahogs (*Arctica islandica*)

Invertebrate Subcommittee[1]
SAW/SARC 48

1 See Appendix B1 for committee members. The lead authors were Larry Jacobson and Toni Chute, Northeast Fisheries Science Center, Woods Hole, MA.

Terms of Reference

1. Characterize commercial catch including landings, effort, and discards.
2. Estimate fishing mortality, spawning stock biomass, and stock biomass for the current and previous years. Characterize uncertainty of the estimates.
3. Update or redefine biological reference points (BRPs; estimates or proxies for *BMSY*, *BTHRESHOLD*, and *FMSY*). Comment on the scientific adequacy of existing and redefined BRPs.
4. Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 3).
5. Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch).
    a. Provide numerical short-term projections (3-4 years). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for *F*, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions about the most important uncertainties in the assessment (alternate states of nature).
    b. If possible, comment on the relative probability of the alternate states of nature and on which projections seem most realistic.
    c. Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.
6. Review, evaluate and report on the status of SARC/Working Group research recommendations listed in recent SARC reviewed assessments. Identify new research recommendations.

Clarification of terms used in the terms of reference:

(The text below is from DOC National Standard Guidelines, Federal Register, vol. 74, no. 11, January 16, 2009)

Acceptable biological catch (ABC) is a level of a stock or stock complex's annual catch that accounts for the scientific uncertainty in the estimate of (overfishing limit) OFL and any other scientific uncertainty…" (In other words, OFL ≥ ABC.

ABC for overfished stocks. For overfished stocks and stock complexes, a rebuilding ABC must be set to reflect the annual catch that is consistent with the schedule of fishing mortality rates in the rebuilding plan.

NMFS expects that in most cases ABC will be reduced from OFL to reduce the probability that overfishing might occur in a year.

ABC refers to a level of ''catch'' that is ''acceptable'' given the ''biological'' characteristics of the stock or stock complex. As such, (optimal yield) OY does not equate with ABC. The specification of OY is required to consider a variety of factors, including social and economic factors, and the protection of marine ecosystems, which are not part of the ABC concept.

Executive Summary

A) This assessment for ocean quahog in the US EEZ is based on biological information, fishery-dependent data for 1978-2008 and NEFSC clam survey data for 1982-2008. Based on assessment data, the ocean quahog population is an unproductive stock with infrequent and limited recruitment. After three decades of fishing at a relatively low *F*, the stock as a whole it is being fished down towards its target biomass reference point, which is defined as 50% of biomass during 1978 (pre-fishery) based on assessment recommendations.

B) Ocean quahogs in the US EEZ are not overfished and overfishing is not occurring. Total fishable stock biomass (all regions) during 2008 was 2.905 million mt, which is above the current and recommended management target of 1.790 million mt. The fishing mortality rate during 2008 for the exploited region (all areas but GBK) was $F = 0.01\ y^{-1}$, which is below the current $F_{25\%} = 0.0517\ y^{-1}$ and recommended $F_{45\%}$=0.0219 threshold reference points. The recommended $F_{45\%}$ mortality threshold is based on harvest policies for long lived West Coast groundfish, which are probably more productive than ocean quahogs. The $F_{45\%}$ recommendation should be revisited in the next assessment.

C) Fishing effort declined in the EEZ fishery from about 40 thousand hours per year during 1990-1995 to about 25 thousand hours per year recently. The number of active vessels in the EEZ in 2008 was the lowest level on record. LPUE for the EEZ stock as a whole has been stable since 1982 but is currently higher in northern areas (LI and SNE) than in the south (NJ and DMV). Landings have declined since the peak of 22,000 mt during 1992 to 15,000 mt during 2009.

D) The ocean quahog fishery has shifted north over the last two decades as catch rates declined in the original fishing grounds off Delmarva and New Jersey. In the 1980s, the bulk of the fishing effort was off Delmarva and southern New Jersey, with some fishing off southern New England. In the early 1990s effort fell by half in the Delmarva region while effort increased south of Long Island until about 40% of total effort was concentrated there. By the late 1990s, most of the fishing effort had moved to the Southern New England region. In the early 2000s, the majority of fishing effort was in the Long Island region. By the late 2000s only 22% of total effort was in the Delmarva and New Jersey regions.

E) Cooperative ocean quahog depletion experiments conducted in connection with the 1997-2008 NEFSC clam surveys were used to estimate the efficiency of the NEFSC survey dredge. Results of depletion experiments are important in estimating biomass and fishing mortality. Three more successful depletion experiments were carried out this year for a total of 15. Based on all experiments to date, the median NEFSC survey dredge efficiency is 0.169.

F) During the 2008 NEFSC clam survey, which consisted of 453 stations, the electrical cable powering the dredge pump was replaced at station 241 with a longer one, and the dredge pump was replaced at station 170. As a result, special analyses were conducted to determine the effects of these changes on survey catch rates. Based on the results, effects of the

replacement electrical cables and pumps on catches during the 2008 survey could not be distinguished statistically from zero.

G) Dredge tows completed during the 2008 survey tended to be shorter than tows from the 1997, 1999, 2002 and 2005 surveys although differences between 2008 and 2002 were small. Considerable effort was devoted to examining sensor data to determine why survey tows during 2008 were shorter than in previous surveys. The evidence was inconclusive.

H) The estimates of biomass and fishing mortality for the EEZ stock in this assessment do not include the Maine "mahogany" quahog fishery. Maine stock biomass is small (~1% relative to the rest of the EEZ) with fishing effort concentrated in a small area. A stock assessment for ocean quahogs in Maine waters is presented as Appendix B2.

I) Current BRPs were reviewed. The current threshold reference point for fishing mortality $F_{25\%}$=0.0517 y^{-1} is a poor proxy for F_{MSY} in a long-lived species like ocean quahog with natural mortality rate M=0.02 y^{-1}. In absence of simulations for ocean quahog, the best available information is Clark's (2002) simulation analyses of F_{MSY} proxies applicable to long lived West Coast groundfish and a follow-up workshop report (PFMC 2000, reproduced here as Appendix B7). The workshop report recommends an F_{MSY} proxy of $F_{40\%}$ for relatively productive Pacific whiting and flatfish, $F_{45\%}$ for other groundfish, and $F_{50\%}$ for *Sebastes* spp. (rockfish) and *Sebastolobus* spp. (thornyheads). The Invertebrate Subcommittee could not choose between $F_{40\%}$ and $F_{50\%}$ as F_{MSY} proxies. After discussion, $F_{45\%}$ was recommended as the F_{MSY} proxy for ocean quahogs. New recommended reference points are not referred to as MSY reference points because the productivity of the ocean quahog stock is currently unknown.

J) The new recommended biomass target of 1.837 million mt is one-half of the 1978 pre-fishery biomass (virgin biomass probably fluctuated due to infrequent recruitment). The new recommended $B_{Threshold}$ which is 40% of the 1978 biomass (1.432 million mt), which can be compared to the current $B_{Threshold}$ which is 25% of virgin biomass. The recommended $B_{Threshold}$ is ad hoc, but probably better than the current value.

K) Managers will have to decide whether the new fishing mortality threshold should be compared to estimated fishing mortality for the exploited portion of the stock (excluding GBK where no fishing takes place) or to the whole stock. Fishing does not occur on GBK (which current contains about 45% of stock biomass) because of the risk of PSP (paralytic shellfish poisoning).

   a. The current FMP requires comparison of the threshold reference point to fishing mortality in the exploited portion of the stock only. Most other FMPs compare reference points to mortality rates for the whole stock.

   b. This current approach should help maintain higher productivity for a sessile spatially non-homogenous stock like ocean quahogs. MSY theory is difficult to apply to stocks like ocean quahogs because MSY mortality levels for the stock as a whole result in under-exploitation of the unfished portion (with foregone yield) while the fished portion of the stock is over exploited (resulting in foregone yield).

c. Industry sources expect ocean quahog fishing to begin on Georges Bank soon. This assessment contains no direct advice on harvest of ocean quahogs across the entire stock. Almost all fishery calculations use growth curves and other data for the currently exploited portion of the stock. Harvest policies for ocean quahog should be reconsidered when and if a fishery develops on Georges Bank

L) KLAMZ model projections were run with varying "states of nature", a range of possible values for natural mortality (M=0.015, 0.02 and 0.025) and biomass levels. The projections were also run with four landings policies (status quo, FMP minimum quota, FMP maximum quota, and FMP current quota) and five target fishing mortality policies ($F_{0.1}$, $F_{25\%}$, $F_{40\%}$, $F_{45\%}$ and $F_{50\%}$). Both stochastic and deterministic (which approximate median values from stochastic projections) results indicate that overfished (low biomass) stock conditions and overfishing are not likely to occur by 2015 <u>*at current catch levels*</u> under any of the states of nature.

M) In 2008, fishable stock biomass in SVA, DMV and NJ was less than half of pre-fishing (1978) levels. In contrast, stock biomass in the more northern regions of LI and SNE increased after 1978 to due to a recruitment event and growth, and then began to decrease in the early 1990s when recruitment declined and the fishery gradually began to move north into these areas. The LI, SNE and GBK regions contained about 67% of total fishable biomass during 1978 and contained about 84% of the total fishable biomass during 2008. The GBK region, which is currently not fished due to risk of PSP contamination, contained about 32% of total fishable biomass during 1978 and about 45% during 2008.

N) Recruitment events appear to be localized and episodic (i.e. often separated by decades) although survey length composition data show that a very low level of recruitment occurs on a continuous basis. Based on survey length composition data and published studies, some recruitment has been evident in LI, SNE and GBK during recent years. The potential contribution of recent recruitment to stock biomass and productivity is unknown.

O) Fishing mortality rates are relatively low for the ocean quahog stock as a whole and stock biomass is relatively high. However, ocean quahogs are an unproductive stock that is likely vulnerable to overfishing. If overfished (depleted biomass) conditions occur, one or more decades will be required to rebuild the stock.

Introduction

Ocean quahogs (*Arctica islandica*) in the US Exclusive Economic Zone (EEZ, federal waters only) and a small component in Maine (MNE) state waters are regarded as a single stock. However, the EEZ and MNE components have different biological characteristics and support different fisheries that are managed separately. The EEZ fishery (with landings of about 15,000 mt meats during 2008) is managed by under a single individual transferable quota (ITQ) system that was established for ocean quahog and Atlantic surfclam (*Spisula solidissima*) in 1990. Murawski and Serchuk (1989) and Serchuk and Murawski (1997) provide detailed information about the history and operation of the EEZ fishery. The smaller MNE fishery (with landings of about 200 mt meats during 2008) is managed under a separate quota system. This report focuses primarily on the ITQ fishery but includes a brief summary of key results for ocean quahogs in Maine waters. Appendix B2 gives detailed stock assessment information about ocean quahogs in Maine waters.

The ocean quahog stock is often broken down into smaller regions (listed below) based on biology, fishery characteristics, and history. These designated regions are important in understanding the fishery but have no legal importance beyond the distinctions between Maine, Georges Bank (GBK, see below) and the EEZ as a whole.

Region	Abbreviation
US exclusive economic zone	EEZ
Georges Bank	GBK
Southern New England	SNE
Long Island	LI
New Jersey	NJ
Delmarva	DMV
Southern Virginia and North Carolina	SVA
Mid-Atlantic Bight (Delmarva to Long Island)	MAB
Maine	MNE

Entire stock vs. the exploited region

Data and analysis for ocean quahogs in the EEZ are presented in this assessment for the "entire" or "whole" stock and for the "exploited region" only (Figure B1). "Entire" and "whole" stock refers to ocean quahogs in the entire EEZ. The "exploited region", in contrast, excludes Georges Bank (GBK) because the GBK region has been closed to ocean quahog harvesting since 1990 when paralytic shellfish poison (PSP) was detected. The Mid-Atlantic Bight (DMV to LI) includes most of the exploited region where the fishery originally operated.

Interest in reopening GBK for ocean quahog fishing has increased recently because catch rates on southern fishing grounds are relatively low and a large fraction (nearly 50%) of the fishable biomass is found there. Sampling was carried out during 2008 to determine if PSP is still a problem. Industry sources expect the fishery on GBK to reopen in the near future.

Fishable stock vs. exploited region

The "fishable stock" and "exploited region" are not synonymous for ocean quahogs in this report. "Fishable" ocean quahogs are quahogs large enough to be taken in the commercial fishery based on the size selectivity curve for commercial fishing gear (Figure B2).

Units of measurement

Body size in ocean quahogs is measured in terms of shell length (SL), which is the longest anterior-posterior distance along the axis of an intact specimen.

Vessel size categories and units of measure for ocean quahogs used in this assessment are described below. Commercial data are reported in units of "industry bushels" in logbooks and often converted to saleable meat weights (which include all soft tissues within the shell) for use in this assessment.

Unit	Equivalent
Industry or Mid-Atlantic bushel (Industry bu)	1.88 ft^3
Maine (US standard) bushel (Maine bu)	1.2448 ft^3
Industry bushels x 10	Pounds meat wt
Industry bushels x 4.5359	Kilograms meat wt
Maine bushel	0.662 industry bushels
Cage	32 Industry bushels
Vessel ton class 1	1-4 gross registered tons (GRT)
Vessel ton class 2	2-50 GRT
Vessel ton class 3	51-150 GRT

Previous assessments

Stock assessments for ocean quahog in the EEZ were completed by the NEFSC (1995; 1998; 2000; 2004; 2007a). The last assessment (NEFSC 2007a) concluded that the EEZ ocean quahog resource was not overfished and that overfishing was not occurring.

Fishing mortality rates during 2005 for the MNE stock component was near the $F_{0.1}$ level (NEFSC 2007a).

Biological characteristics[2]

Ocean quahogs are common in the eastern Atlantic as far south as Spain, around Iceland, and in the western Atlantic as far south as Cape Hatteras (Theroux and Wigley 1983; Thorarinsdottir and Einarsson 1996; Lewis et al. 2001). They can be found at depths of 10-400 m, depending on latitude (deeper water habitats are utilized in the south, Theroux and Wigley 1983; Thompson et al. 1980).

The US stock is almost completely within the EEZ at depths of 25-95 m. Dahlgren et al. (2000) found no genetic differences between samples taken along the US coast from Maine to Virginia based on mitochondrial cytochrome *b* gene frequencies.

The natural mortality rate and longevity of ocean quahogs are uncertain. Ocean quahogs are certainly long-lived. Individual specimens are commonly aged at over 200 yrs (Jones 1980; Steingrimsson and Thorarinsdottir, 1995; Kilada et al., 2007; Strahl et al. 2007). Early studies of populations off New Jersey and Long Island (Thompson et al. 1980; Murawski et al. 1982) demonstrate that clams ranging in age from 50-100 years are common. Wanamaker et al., (2008) aged two ocean quahogs at 287 and 405 y, making the latter specimen possibly the oldest non-colonial animal ever documented. Based on longevity estimates of around 200 y, adult ocean quahogs in the EEZ and off Iceland are assumed to die from natural causes at the rate of about 2% annually (instantaneous rate of natural mortality M=0.02 per year). In particular, about 1% of a cohort is expected to survive after 230 y when M=0.02. Kilada et al estimated M to be 0.03 and 0.10 for the Sable Bank and St Mary's Bay populations in Canadian waters based on age–frequency data for unexploited populations.

Ocean quahogs grow slowly after the first years of life (Lewis et al. 2001; Kilada et al. 2007). Maximum size is typically about 110 mm in shell length (SL) although larger specimens are

2 See Cargnelli et al. (1999) for additional information.

found. Individuals large enough to recruit to the fishery grow only 0.51-0.77% per year in meat weight and < 1 mm per year in shell length (Figure B3). Growth is faster in GBK than further south in the MAB (Figure B3).

Maturity and recruitment information for ocean quahogs in the US EEZ is scant (see review in Cargnelli et al. 1999) but size and age at maturity appear to be variable. Off Long Island, the smallest mature quahog found was a male 36 mm long and 6 years old; the smallest and youngest mature female was 41 mm long and 6 yr old (Ropes et al. 1984). Some clams in this region are still sexually immature at ages of 8-14 years (Thompson et al. 1980; Ropes et al. 1984). Females are more common than males among the oldest and largest individuals in the population (Ropes et al. 1984; Fritz 1991).

The shell length maturity relationship used in this assessment (Figure B2) is from data for Icelandic ocean quahogs (Thorarinsdottir and Jacobson, 2005). The curve indicates that 10%, 50% and 90% of female ocean quahog mature at 40, 64, and 88 mm SL (2, 19, and 61 y, based on the growth curve in Lewis et al., 2001 for MAB). Based on the size range of samples (G. Thorarinsdottir, pers. comm.), the maturity curve is probably valid for ocean quahog in the size range used to estimate fishing mortality. Maturity occurs at roughly 10 mm before, and about 10 years before, recruitment to the fishery (Figure B2).

Shell length-meat weight (SLMW) relationships are important for ocean quahogs because survey catches in number are converted to meat weights based on shell length for many analyses. SLMW relationships in this assessment are region-specific (Table B9) and the same as in the last assessment (NEFSC 2007a). They were estimated using a mixture of frozen and fresh samples. Relationships were re-estimated based on large number of fresh samples taken during the 1997-2008 surveys (Appendix B8). The updated relationships will be used in the next ocean quahog assessment but were not ready in time for use here.

Recruitment patterns

Recruitment events are regional and infrequent in ocean quahog (Powell and Mann 2005, Harding et al. 2008). Small ocean quahogs in survey length composition data indicate that recruitment occurs at a very low level during most years, particularly in northern areas (Figures B24 through B29). However, survey data collected during 1982-2008 show only three noteworthy recruitment events in LI, SNE and GBK (Figures B25 through B27) over regional spatial scales. Because growth is so slow, there are delays of one to three decades between larval settlement and production of recruits to the fishery. Ocean quahogs reach 64 mm SL (50% maturity) at age 12 y in GBK and 19 y in MAB (Figure B2). In contrast, ocean quahogs reach 73 mm (50% commercial selectivity) at age 13 y in GBK and 28 y in MAB (Figure B3). Each of the three recruitment events observed since 1980 were produced while spawning biomass in the same region was unfished or nearly unfished. Recruitment patterns in ocean quahog at reduced biomass levels after fishing are a major uncertainty (NEFSC 2007a).

Commercial and Recreational Catch
(TOR-1)

Mandatory logbooks have been the principle source of fishery data (landings, fishing locations and fishing effort) for the ITQ fishery since 1980. Landings and quotas for the ITQ fishery are reported in different units than landings and quotas for the fishery off Maine. In particular, "industry" bushels (1.88 ft^3) are used for the ITQ component and "Maine" bushels (1.2448 ft^3) are used for the Maine component. Biomass and landings from both fishery components are reported in

this assessment as meat weights, unless otherwise noted.

Total EEZ landings (including both ITQ and Maine fishery components) were relatively high during 1987-1996 with a peak of 22,500 mt meats (Tables B1 and B2; Figure B4) or 4.9 million ITQ bushels (see Table B3 for all landings in bushels) during 1992. After 1996, landings declined to a low of about 15,000 mt during 2000 and then increased again to a high of 19,000 mt during 2003. Landings declined after 2003 to about 14,000 mt during 2005, the lowest level since 1981. After 2005, landings increased slightly to about 15,500 mt. Industry sources report that low landings during the most recent years were due to low market demand. Landings by the Maine component of the fishery were only 1.2% of total EEZ landings during 1990-2008.

Landings from Maine waters increased steadily from 75 mt in 1992 to relatively high levels (≥ 326 mt annually) during 2000-2003 (Tables B2 and B3). Maine landings decreased after 2003, but remained over 300 mt through 2007. Only 201 mt were landed in 2008, the lowest level since 1997.

Landings by the ITQ component averaged 83% of the EEZ quota during 1990-2008 (Table B1). In contrast, the 100,000 Maine bushel quota allocated for ocean quahog in Maine waters was usually exhausted during 1999-2008 with vessels leasing ITQ shares in some years to harvest more than 100,000 mt meats from Maine waters (Tables B2 and B3).

Landings of quahogs from state waters south of Maine are effectively zero because ocean quahogs are found offshore in relatively deep water. There are no recreational landings of ocean quahogs because commercial clam dredges are required to harvest them, and because they provide an industrial product with no recreational value.

Prices

Nominal ex-vessel prices for ITQ ocean quahog landings (expressed as dollars per ITQ bushel) increased by about 66% after 1990 (Table B4 and Figure B5). In real terms, prices stayed fairly stable except for a 30% jump from 2000 to 2001, followed by a steady decline. Prices during 2006-2008 stabilized at about $3.20 a bushel.

Prices for ocean quahog harvested in Maine waters (expressed as dollars per ITQ bushel for the sake of comparison) were roughly ten times higher than prices for ocean quahogs harvested in the rest of the EEZ (Table B4 and Figure B5). In real dollars, Maine prices have fallen about 50% since their peak in the early nineties.

Fishing effort

Total hours fished annually in the ITQ fishery component decreased from a peak of about 40,000 hr per year during 1991-1994 to about 30,000 hr per year during 1996 to 2004, and then to about 20,000 hr per year during 2005-2008 (Table B5 and Figure B6). The total number of trips in the ITQ fishery decreased steadily from about 3000 trips per year during 1991 to about 1200 trips per year during 2008 (Figure B7). In contrast, hours fished and trips increased in the Maine fishery component during 1991-2005, but declined afterward. The number of active permits (vessels with landings during the year in question) in the ITQ fishery remained relatively constant during 1996-2003 but declined by 50% from 2004 to 2006 and has remained stable at around 30 permits ever since (Figure B8). The number of active permits and fishing effort (hours fished and numbers of trips) is high in Maine waters relative to other regions in the EEZ.

Landings per unit effort (LPUE)

LPUE (expressed in bushels landed per hour fished) in the ocean quahog fishery is a better

measure of fishing success than a measure of stock abundance because changes in abundance or biomass may be masked by movement of fishing effort to areas where ocean quahog density and catch rates remain high. In spite of these potential problems, LPUE and NEFSC clam survey data are highly correlated for southern areas (DMV and NJ) where significant levels of fishing occurred over long periods of time (NEFSC 2007a).

LPUE declined by about 60% in the DMV and NJ regions after the mid-1980s to about 60-80 bushels per hour in recent years (Table B6 and Figure B9). LI and SNE show relatively high LPUE levels of about 160 and 180 bushels per hour that have been relatively stable since 2000. The LPUE for the ITQ fishery as a whole has been remarkably constant since the early 1980s (Table B6) at between 100 and 150 bushels per hour because the fishery moves to new grounds when LPUE declines.

The break-even LPUE (where variable costs and revenues are the same) reported in NEFSC (2004) for the EEZ fishery was 80 bushels h^{-1}. This estimate was higher than previously reported (NEFSC 2001) because of inflation, increased steaming time to relatively distant fishing grounds, operation of new larger vessels, and increased costs for food, fuel, insurance, etc. It was not possible to update the estimate of break-even LPUE because of extreme variability in the price of fuel.

In the Maine fishery (Figure B10), standardized LPUE increased to over 6 bushels an hour during 1991-2000, and decreased afterwards, and has fluctuated between 4 and 5.5 bushels per hour for the last 8 years.

NEFSC (2007a) standardized LPUE data by adjusting for vessel, month and vessel size effects. Estimated trends were very similar to trends in nominal LPUE. Standardized LPUE data are not presented in this assessment.

Spatial patterns in fishery data

Spatial patterns are important in interpreting fishery data and in managing fisheries for sessile and unproductive organisms like ocean quahogs. The ocean quahog stock is a complicated spatial mosaic with scattered productive and profitable fishing grounds where abundance is high and where fishing mortality tends to be concentrated. The size of a productive ocean quahog fishing ground appears to be less than the size of a ten-minute square (TMS, 10' x 10' ≅ 100 nm^2), which is the smallest spatial strata reported on logbooks and used in this stock assessment. As described in NEFSC (2004), spatial patterns in cumulative landings, cumulative effort and LPUE reflect a shift in the distribution of the fishery to offshore and northern grounds. During the 1980s, nearly all of the landings and fishing effort were from the southern DMV and NJ regions. As LPUE declined there, fishing effort and landings shifted offshore and north to the LI and SNE regions. During 2008, the southern DMV and NJ regions accounted for only about 15% of landings and fishing effort while the bulk of landings and effort (outside of Maine waters) were from LI.

Fishery data by ten-minute square (TMS)

Vessels that fish for ocean quahogs in the EEZ are required to report landings and fishing effort by TMS for each trip in mandatory logbooks. TMS are identified by six digit numbers. For example, TMS 436523 is a ten-minute square that lies within the one-degree square with southeast corner at 43° N and 65° E. TMS are formed by dividing one-degree squares further into six columns and six rows that are 10' wide. Columns are numbered 1-6 counting from west to east and the column number is given in the TMS name before the row number. Rows are numbered 1-6 counting from north to south. Thus, TMS 436523 is the ten-minute square whose southeast corner is at 43° 30' N and 65° 40' E.

Landings (Figure B11) during 1981-1990 were concentrated in relatively few TMS that were primarily in the south and relatively inshore. Over time, TMS with highest landings shifted offshore and north. Landings during 2001-2008 were concentrated in the LI region.

Fishing effort (Figure B12) was concentrated in a few southern TMS during 1980-1990 with three adjacent TMS having effort levels higher than 1,000 h per year and appreciable fishing effort south of 38° N. Fishing effort spread into additional offshore and northern TMS during 1991-1995 and 1996-2000. After 1995, there were few or no TMS with effort levels above 1000 h per year. During 2001-2008, there was no fishing effort south of 38° N.

LPUE (Figure B13) was relatively high inshore and south during 1980-1990 with ten TMS that had LPUE ≥ 161 ITQ bushels h^{-1}. LPUE in the area below 40° S was generally high. LPUE declined in the south and fishing effort spread northward during 1991-1995 where LPUE was relatively high. During 1996-2000, the fishery continued to move northward into the SNE region where catches were profitable. By the 2001-2005 time period, LPUE was often ≤ 80 ITQ bushels h^{-1} below 40° S.

Trends for important TNMS

Trends in landings and LPUE during 1980-2005 were plotted for individual TMS that were important to the fishery (Figures B14 through B16). Important TMS were selected by sorting TMS according to total cumulative landings during 1980-1990, 1991-1995, 1996-2000, 2001-2005 and 2006-2008 and then selecting the top 20 TMS during each time period. All of the TMS selected in this manner were combined to form a single a single set of TMS that were important to the fishery at some time during 1980-2008.

Trends in LPUE for individual TMS tend to be relatively high during the first years of exploitation and then tend to decline as effort, annual landings and cumulative landings increase over time (Figures B14 through B16). Decreasing trends in LPUE appear strongest in southern areas such as TMS 377422 to 397326 with the longest history of exploitation. LPUE does not appear to increase in a TMS once fishing effort decreases.

Unlike LPUE which is highest in the first years of exploitation, landings and fishing effort tend to peak after 5-10 years of exploitation while LPUE is still relatively high and then to decrease over a 5-10 y period as grounds are fished down (Figures B14 through B16). In some TMS with low recent LPUE levels (e.g. TMS 387443-397316), fishing effort has increased recently with some increase in landings.

Bycatch and discard

Landings and catch are almost equal in the ocean quahog fishery because discards are nil. Discard of ocean quahogs in the ocean quahog fishery does not occur because undersize animals are automatically released by automatic sorting equipment. However, some incidental mortality occurs. Based on Murawski and Serchuk (1989), NEFSC (2004) assumed incidental mortality rates of ≤ 5% for ocean quahog damaged during fishing but not handled on deck. As in previous assessments, fishing mortality and other stock assessment calculations in this report assume 5% incidental mortality rates (i.e. landings x 1.05 = assumed catch).

Bycatch of ocean quahog probably occurs in fishing for Atlantic surfclam. Discard quantities have not been quantified but are probably minor. Off DMV and SVA in the southern end of the ocean quahog's range, survey catches including both surfclam and ocean quahog have become more common in recent years as surfclams have shifted towards deeper water in response to warm water conditions (Weinberg 2005). However, mixed loads of surfclams and ocean quahogs are not

acceptable to processors and it is not practical to sort catches at sea, so vessels tend to avoid areas where both species might be caught together.

Bycatch and discard of ocean quahogs in other fisheries is nil. Ocean quahogs are not vulnerable to bottom trawls, scallop dredges (because they are too deep in sediments), and hook and line gear.

Commercial size selectivity

The commercial fishery selectivity curve used in this assessment is from Thorarinsdottir and Jacobson (2005) who estimated selectivity of commercial dredges that harvest ocean quahogs off Iceland. Based on this commercial selectivity curve ($s_L = 1/\left(1 + e^{7.63-0.105L}\right)$ where L is shell length in mm) about 10%, 50% and 90% of ocean quahogs are available to the fishery at 51, 72, and 93 mm SL (9, 28 and 86 y, based on the growth curve for MAB in Figure B3).

Dredges and towing speeds used in the US fishery are very similar to those used in the selectivity experiments. The dredge used for selectivity experiments was 24 ft (7.35 m) in length, 5 ft (1.5 m) high and 12 ft (3.65 m) wide. The cutting blade was 10 ft (3.05 m) wide and set to penetrate sediments to a depth of 3 in (8 cm). The dredge was made of steel bars with intervening spaces of 1 ¼ in (3.5 cm) and was towed at about 2.1 knots (3.9 km h^{-1}). Water pressure supplied to jets on the dredge from a pump on the ship was about 109 psi (7.5 bars). Water pressure levels in the US fishery are usually lower (~80 psi) but water pressure probably has relatively little effect on size selectivity. Fishery selectivity curves are used in tracking trends in fishable biomass, estimating fishing mortality and in calculating biological reference points.

Commercial size-composition data

Commercial length composition data collected by port agents from landings samples (Table B7) indicate that the size composition of ocean quahogs captured in the DMV region differed during 1987-1994, 1995-2000 (when they were smaller) and 2001-2008 (Figure B17). Lengths for DMV during 1987-1994 and 2001-2008 were similar. The only exception is 2007, when port samples from the DMV region showed slightly larger harvested quahogs.

Commercial length composition data for NJ were stable during 1982-2002 with smaller ocean quahogs landed during 2003-2008 (Figure B18). Length data for LI include relatively high proportions of large individuals (11-12 cm SL) during 1997-1999 (Figure B19). Length data for SNE during 1998-2005 were generally stable but with smaller ocean quahogs landed during 1997-2000 (Figure B20). According to NEFSC (2004), smaller sizes landed from SNE during 1997-2000 were due to vessels targeting specific beds with relatively small ocean quahogs that had relatively high meat yield.

Port sampling levels were increased in the SNE and LI regions during recent years due to increased landings and fishing effort levels (Table B7). Increased port sample frequencies reflect movement of the fishery onto northern grounds in SNE and LI.

Mortality and Stock Biomass
(TOR-2)

Mortality and stock biomass estimates for ocean quahog in the US EEZ are based on triennial NEFSC clam surveys, cooperative survey studies that include depletion experiments used to measure survey dredge efficiency, fishery, and other data.

NEFSC clam surveys

Survey data used in this assessment were from surveys conducted during 1982-2008 by the *R/V Delaware II* during the summer (June-July), using the standard NEFSC survey hydraulic dredge with a submersible pump. The current survey dredge which has been used since 1982 has a 152 cm (60 in) blade and 5.08 cm (2 in) mesh liner to retain relatively small ocean quahogs and Atlantic surfclams. The survey dredge differs from commercial dredges in that it is smaller (5 ft blade instead of 8-12.5 ft), has a small mesh liner, and the pump is mounted on the dredge instead of the deck of the vessel. The survey dredge is useful for ocean quahogs as small as 50 mm SL (size selectivity described below). Changes in ship construction, winch design, winch speed and pump voltage that may have affected survey dredge efficiency are summarized in Table A7 of NEFSC (2004). Each of these factors has been constant since the 2002 survey.

Surveys prior to 1982 were not used in this assessment because they were carried out during different seasons, used other sampling equipment or, in the case of 1981, have not been integrated into the clam survey database (Table A7 in NEFSC 2004).

NEFSC clam surveys are organized around NEFSC shellfish strata and stock assessment regions (Figure B1). Most ocean quahog landings originate from areas covered by the survey. The survey did not cover GBK during 1982, 1983, 1984 or 2005. Individual strata in other areas were sometimes missed (Table B8). Strata not sampled during a particular survey are "filled" for assessment purposes by borrowing data from the same stratum in the previous and/or next survey, if data are available (NEFSC 2004). Survey data are never borrowed from surveys further back than the previous survey or beyond the next survey. Despite research recommendations, a model based approach to filling survey holes has not yet been developed, although the approach appears practical based on results for Atlantic surfclam (NEFSC 2007a).

Surveys follow a stratified random sampling design, allocating a pre-determined number of tows to each stratum. Stations used to measure trends in ocean quahog abundance are either random or nearly random. The few "nearly" random tows were added in previous surveys in a quasi-random fashion to ensure that important areas were sampled. Other non-random stations are occupied for a variety of purposes but not used to estimate relative trends in ocean quahog abundance.

A standard tow is nominally 0.125 nm (232 m) in length (i.e. 5 minutes long at a speed of 1.5 knots). However, sensor data indicate that the actual tow lengths depend on depth and are generally longer than 0.125 nm (Weinberg et al. 2002 and see below).

Occasionally, randomly selected stations are found too rocky or rough to tow. Beginning in 1999, these cases trigger a search for fishable ground in the vicinity (0.5 nm) of the original station (NEFSC 2004). If no fishable ground is located, the station is given a special code (SHG=151) and the research vessel moves on to the next station. The proportion of random stations that cannot be fished is an estimate of the proportion of habitat in a stratum or region that is not suitable habitat for ocean quahog. These estimates are used for calculating ocean quahog swept-area biomass (see below).

Following all successful survey tows, all ocean quahogs and Atlantic surfclams in the survey dredge are counted and shell length is measured to the nearest mm. A few very large catches are subsampled. Mean meat weight (kg) per tow is computed with shell length-meat weight (SLMW) equations from NEFSC (2004).

Survey tow distance and gear performance in trend analysis

For trend analysis, tow distances are based on start and stop locations recorded for each tow. The catch at each station is standardized to a "nominal" tow distance of 1.5 nm for trend analysis.

"Successful" tows suitable for trend analysis are identified using "HG" (haul and gear) database codes ≤ 36, which are recorded at sea by the watch chief following each tow based on criteria used consistently since the late 1970's. Sensor data are not used to calculate tow distance for trend analyses because sensor data are not available prior to 1997. Sensor data are used, however, to calculate tow distance and monitor gear performance during tows for depletion, repeat station and other types of experimental studies conducted since 1997 (see below).

Survey tow distance and gear performance based on sensor data

After the 1994 survey, sensors were used to monitor depth (ambient pressure), differential pressure, voltage, frequency (hertz) and amperage of power supplied to the dredge, *x*-tilt (port-starboard angle), *y*-tilt (fore-aft angle, effectively the "angle of attack" of the dredge) and ambient temperature during survey fishing operations. At the same time, sensors on board the ship monitor electrical frequency, GPS position, vessel bearing and vessel speed. Most of the sensor data are averaged and recorded at 1 second intervals.

Good tows have characteristic sensor data patterns that are easy to interpret (Figure B31). Anomalous patterns indicate potential problems with the tow or sensors. Differential pressure, amperage and *y*-tilt can be particularly important. Differential pressure is the pressure of water pumped through jets in front of the dredge blade to loosen the sediments. Amperage measures the work done by the pump in moving water through the jets. If water is blocked at the entrance to the pump, then both amperage and differential pressure will be low. If water is blocked downstream of the pump, then amperage will be low and differential pressure will be high. As described below, *y*-tilt can be used to determine if the dredge is on the bottom with the blade in the sediment.

NEFSC (2007a) developed a quantitative system for identifying tows with poor performance based on *y*-tilt and differential pressure sensor data that was applied to the 2005 NEFSC clam survey (see Appendix A3 in NEFSC 2007a). The *y*-tilt criterion which was part of this quantitative system was dropped after reconsideration in this assessment (Appendix B3) for 3 reasons: i) the *y*-tilt sensors appear to be strongly affected by vibration, ii) the existing procedure for calculating tow distances (see below) already indentifies periods when the dredge is not fishing, and iii) because the standard database "SHG" code eliminates many of the problematic tows before sensor data are examined. The revised criteria based on differential pressure only was applied to the 2008 and retroactively to 2005 surveys (but not to the 1997-2002 surveys due to lack of time).[3,4] Affects on the 2005 survey were modest with only one additional tow shifted from the poor to good performance categories.

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3 The criterion for differential pressure is a time-weighted approach that penalizes problematic high and low pressures. The weights depend on the extent of the deviation from normal operating range of 35-40 psi. The weighting system for differential pressure data P_t is:

$W_t = 2 *(P_t-40)/40$ when the differential pressure $P_t > 40$ psi

$W_t = 2 * ((35-P_t)/35 \text{ x } 0.83)$ when $P_t < 35$ psi

$W_t = 1$ otherwise

A tow is judged to have poor performance when the weighted time outside the normal range > 25%. See Appendix B3 for more information.

4 Stations with poor performance based on sensor data in the 2005 survey: 1, 2 , 4,17,20,22,23,24,25,26,28,29,30,31,32, 33,34,45,48,56,58,67,75,76,108,218,225,262,282,405,411,413,414,417, 422,423,424.

Stations in the 2008 survey: 15,
,29,35,43,45,48,52,65,95,99,119,137,138,141,150,164,165,169,175,197,198,206,209,226,227,229,241,242,245, 246,248,249,250,252,254,257,258,262,263,288,290,291,293,305,306,307,308,309,310,317,326,358,366,394,402, 403,424,430,433,434,435,436,437,438,448,452,453.

Survey gear selectivity

NEFSC (2004) estimated selectivity curves for ocean quahogs in the NEFSC clam dredge based on catches by a commercial dredge with a chicken-wire mesh liner during 2003 and survey catches in the same area during 2002. The selectivity curve $s_L = 1/\left(1 + e^{8.122-0.119L}\right)$ indicates that 50% of ocean quahogs are fully available to the NEFSC clam dredge at about 68 mm SL, which can be compared to about 73 mm for commercial dredges (Figure B21). The survey dredge tends to take smaller ocean quahogs than commercial dredges because of the relatively small 50 mm (2 in) liner in the survey dredge. Based on sizes retained by the survey dredge (NEFSC 2004), the survey dredge selectivity curve is reliable for ocean quahogs ≥ 50 mm SL.

Survey, stock and fishable abundance and biomass

The survey size selectivity curve with survey catch and size composition data for ocean quahogs ≥ 50 mm SL was used to estimate relative abundance and size composition for the stock as a whole. In particular, $N_L = n_L / s_L$ where N_L is mean stock numbers or biomass per tow at length L in the stock as a whole, n_L is survey catch and s_L is survey selectivity.

Abundance and length composition for the fishable stock (i.e. of a size available to the fishery) were estimated by adjusting stock estimates for fishery selectivity. In particular, $\eta_L = \phi_L N_L$ where η_L is fishable abundance and ϕ_L is fishery selectivity. Fishable abundance can be estimated directly from survey data for ocean quahogs ≥ 50 mm SL using $\eta_L = n_L \phi_L / s_L$ (Figure B21).

Calculations of stock abundance and biomass occasionally produce very large estimates for small sizes where selectivity is small (near zero) when ratios n_L / s_L become very large. Calculation of fishable abundance and biomass from ocean quahog survey data does not suffer from this problem because the adjustment for small sizes is relatively modest (Figure B21).

Survey Trend Results

Based on survey data, abundance and biomass of relatively large quahogs (70+ mm SL) declined during 1997-2008 in all areas but GBK (Table B10 and Figures B22 and B23). The declines in southern areas where the bulk of fishing has occurred (DMV and NJ) appear clear. The apparent trends in SNE and LI since 1997 are not as clear and may be due to sampling error or changes in survey catchability.

Based on survey data for small ocean quahogs (< 70 mm SL, Table B11 and Figure B24), recruitment during 1997-2008 was about average in DMV, higher than average in NJ, SNE and GBK, and below average in LI.

Survey length composition data (Figures B25 through B29) and the distribution of catches in the 2008 survey (Figure B30, lower panel) provide additional information about recruitment. In particular, survey length composition data for LI for 1982 are bimodal with a lower mode at 65-70 mm SL in 1982 due to a strong recruitment event. Based on the growth curve for the MAB (Figure B3), ocean quahogs 65-70 mm SL are about 21-26 y old. The mode gradually shifted to the right over time as the year class grew. By 2005 (23 y later), the strong year class had grown to be indistinguishable from other ocean quahogs in the region. This historical recruitment event is evident in recruit trends for LI, which increased during the 1960-1970's and generally decreased afterwards (Figure B24).

Survey size composition data for SNE during 2005 and 2008 (Figure B26) show a recent recruitment event that is also apparent in the survey trend data for the same years (Figure B24). The

lower mode during 2005 and 2008 was at approximately 50-60 mm SL. Based on the MAB growth curve, ocean quahogs 50-60 mm SL are about 9-15 y old. This strong year class is located southeast of Cape Cod based on catch locations in the 2008 survey (Figure B30, bottom panel). LPUE data show relatively high catch rates in the corresponding TMS southeast of Cape Cod at approximately 40° 30' N 69° 40'E (Figure B13).

Size composition data from the 2008 survey show an apparent recent recruitment event in the GBK region as there is a strong mode at about 60-65 mm SL (Figure B25). Based on a growth curve for GBK from Lewis et al. (2001), ocean quahogs 60-65 mm SL on GBK are 7-10 y old. Small ocean quahogs appear sporadically in survey length composition data for GBK during 1982-2002.

The geographic distribution of survey catches for small ocean quahogs (<70 mm SL, Figure B30) and trends for the same sizes (Figure B24) show that small ocean quahogs are most common in the north (LI, SNE and GBK). Large ocean quahogs (70+ mm SL, Figure B30) have the highest densities in the SNE and GBK regions although appreciable densities are also found in LI and offshore in the NJ region.

2008 clam survey

The 2008 clam survey consisted of 453 stations. The total number of useful random stations (with database HG codes ≤ 36) was 337. There were 97 useful nonrandom stations of which three were to identify areas of high recruitment, seven were test tows, and 87 were repeat tows to test for gear effects or setup tows for commercial depletion experiments.

As described below, sensor data (Figure B31) provide additional useful information about gear performance. GPS position information, speed- and course over ground, and amperage data are available for all stations in the 2008 survey. Survey sensor package (SSP) data from the 2008 survey are available for stations 1-405 and backup sensor data are available for tows 406-453. The backup sensor data include ambient pressure but not y-tilt, manifold pressure or voltage.

There were at least three potentially important events during the 2008 clam survey that might affect dredge gear performance and capture efficiency (Figures B32 and B33): a new pump was installed on the dredge and used starting at station 170 due to failure of the original equipment, a new electrical cable to send power to the pump was installed and first used at station 241 so that the dredge could be deployed in relatively deep water, and a new SSP sensor data package was installed and first used at station 270. Mean differential pressure, voltage and amperage calculated for each tow during periods when the dredge was fishing effectively (smoothed y-tilt ≤ 5.16°, see below) reflect each of these events (Figure B33). Based on these data, and in comparison to previous surveys (Figure A29 in NEFSC 2007a), sensor data indicate no major gear performance issues during the 2008 clam survey.

Tow distance

The NEFSC survey dredge is assumed to be effectively fishing when the angle of attack (y-tilt, after smoothing with a 7-second moving average) is less than 5.16°. The 5.16° figure is a standard criterion which corresponds to the dredge blade extending 1 inch into the sediments based on the geometry of the dredge (NEFSC 2003). The criterion was selected based on sensitivity analysis; tow distance estimates were not sensitive to small changes in the critical angle around 5.16° (NEFSC 2003). Tow distances from sensor data are not used in trend analysis but are very important in depletion studies and other types of studies where absolute estimates of quahog density are required.

The procedures used to calculate 2008 survey tow distances were the same as in NEFSC

(2007a). The first step was to replace missing speed over ground and inclinometer data (which occur infrequently) for each station with interpolated values from a cubic spline. The second step was to smooth the original plus interpolated speed over ground and inclinometer data using a centered seven second moving average (e.g. the smoothed value for $t = 3$ seconds was the average for $t = 1$ to 7 seconds).[5] The final step was to compute the effective tow distance for each tow d_j using:

$$d = \frac{\sum_t \delta_t s_t}{3600}$$

where t is for a one-second time interval, δ_t was a dummy variable equal to one when the dredge was fishing effectively (smooth y-tilt $\leq 5.16^o$), zero otherwise, s_t was smoothed speed over ground (knots) and 3600 is the number of seconds per hour.

Tows during the 2008 survey tended to be shorter than tows during the 1997, 1999, 2002 or 2005 surveys although differences between 2008 and 2002 were relatively small (Figure B34 and see below). Median tow distances for 1999 to 2005 are similar and longer (0.19-0.22 nm). As pointed out in NEFSC (2003), the median tow distance for 1997 (0.26 nm) was larger than median tow distances from other surveys because a slower winch was used to retrieve the survey dredge (Table C7 in NEFSC 2003).

Year	Median Tow Distance (NM)
1997	0.26
1999	0.22
2002	0.19
2005	0.21
2008	0.16

The relatively short tow distance during 2008 triggered a detailed analysis of all available data to determine the possible causes.

Tow distance and depth

Relationships between tow distance and depth differed among surveys (Figure B35). As expected based on medians, tow distance was relatively low during 2008 at all depths (Figure B35). Regression relationships for depth and tow distance were statistically significant and the best model for the entire set includes separate regression lines for each survey (NEFSC 2007a). However, a single regression model (see below) fit to all of the available data (surveys combined) might be useful in future for predicting tow distance based on depth (Figure B36). The combined model indicates that tow distance increases by 0.0014 nm (2.6 meters) for each additional meter of depth.

Parameter	Estimate	SE
Intercept	0.1635	0.003
Depth	0.0014	0.0001
Residual standard error		0.0479

5 Steps 1-2 were done in SAS (note that interpolation precedes smoothing). proc expand data=sdata1 out=sdata2 to=second; by station; ID TowTime; convert TiltY=SmoothAngle / transform=(cmovave 7); convert GPS1_SOG=SmoothSOG / transform=(cmovave 7); run;

Residual degrees of freedom	1497
Multiple R^2	22%

Short tow distance in 2008 survey

Considerable effort was devoted to examining sensor data to determine why survey tows during 2008 were shorter than in previous surveys. A number of possible explanations were considered and four principal hypotheses were examined: 1) the dredge during 2008 may have been towed at relatively high angle of attack (high *y*-tilt) possibly due to minor differences in gear; 2) *y*-tilt sensors were not calibrated during 2008 in the same manner as during 2005; 3) survey protocols differed slightly in the two surveys; or 4) tow distance estimates from SSP sensor data are sensitive to assumptions about the critical angle for effective fishing. Unfortunately, it was not possible to completely eliminate any of these possible explanations.

If *y*-tilt sensors were calibrated so that the apparent *y*-tilt based on sensors was greater than the actual *y*-tilt, then distance estimates based on sensor data may be too low during 2008 but survey data trends would be unaffected. On the other hand, if the angle of attack was actually higher during 2008 or survey protocols differed, then the distance estimates for 2008 should be unbiased but trend estimates may be affected to the extent that the efficiency of the dredge changed.

Station records for successful random tows (survey SHG codes ≤136) indicate that the average duration (based on start and stop times recorded on the bridge), average nominal tow distance (based on ships GPS start and stop locations) and average depth were similar for the 2005 and 2008 surveys. Survey personnel were interviewed but could not recall any changes in protocol. The captain of the R/V Delaware was involved in both the 2005 and 2008 surveys. The chief scientist and watch chiefs were very familiar with clam survey operations. The crewman who operated the winch during 2005 was present in 2008 and on duty 12 h each day, and trained the new operator. The winch and hawser were the same as during the 2005 survey.

Incorrect calibration or mechanical errors affecting y-tilt sensor were considered as a potential cause for the apparently shorter tow distances. To test this hypothesis, tow distance was plotted against depth in the 2008 survey for successful random tows using different symbols for tows with the original and replacement SSP equipment (Figure B37). The relationships between depth and tow distance were very similar indicating that the units were calibrated and working in the same manner. It is still possible, however, that both of the *y*-tilt sensors used during 2008 were calibrated incorrectly.

Tests show that tow distance estimates are not sensitive to the critical angle (5.16^o) assumed in tow distance calculations. A sensitivity analysis in NEFSC (2003) was repeated using data from the 2005 and 2008 surveys (Figure B38). Results indicate that median tow distances for all of the surveys since 1997 are robust to assumptions about critical angle in the range of $4\text{-}6^o$, which includes the current 5.16^o criterion.

Additional analyses used sensor data from successful random tows during the 2005 and 2008 surveys (Figure B39). All of these analyses used sensor data that were collected between the first and last seconds of each tow during which the smoothed *y*-tilt was less than or equal to 5.16^o (while the dredge was potentially fishing). In particular, the proportion of time on bottom that the dredge was effectively fishing (i.e. proportion of time between the first and last seconds of the tow with smoothed *y*-tilt $\leq 5.16^o$), depth, speed over ground, and the mean and standard deviation of unsmoothed *y*-tilt and *x*-tilt were calculated for each tow. The statistical distribution of each variable in each survey was described graphically using box plots with notches that approximate 95% confidence intervals for each median (Figure B39). In addition, linear correlation coefficients

were calculated between each pair of variables in each survey (Tables B12 and B13).

Based on box plots (Figure B39) distributions of speed over ground while dredges were potentially fishing were similar for 2005 and 2008 although median speed over ground was slightly lower during 2008. Median time on bottom (difference between the first and last second when the dredge was effectively fishing) was lower in 2008 by about 0.01 hr (36 seconds, which amounts to about 12% of a five minute tow). The proportion of time that the dredge was effectively fishing was lower during 2008. In particular, the median proportions differed by only about 0.01 but the distribution of the proportions was skewed towards smaller values in 2008.

The median *y*-tilt was about 2.5^{o} during 2005 and 3.7^{o} during 2008 (Figure B39). As expected, these values were less than the 5.16 ^{o} criterion used to estimate tow distance. The standard deviations for *y*-tilt measurements were similar during both surveys.

The biggest and most surprising (though possibly least important) difference between the 2005 and 2008 surveys was between *x*-tilt measurements (Figure B39). In particular, *x*-tilt values were almost always negative during 2005 and almost always positive during 2008. The standard deviations for *x*-tilt measurements were similar in both surveys. It is possible that the reversal of sign was due to changes in the orientation of the *x*-tilt sensors within the SSP package during 2005 and 2008.

There were 19 out of 36 "substantial" correlations among sensor variables from the 2008 survey compared to 5 out of 29 for the 2005 survey (Tables B12 and B13). In this analysis, "substantial" correlations had an absolute value ≥ 0.5. Many of the substantial correlations were expected (i.e. correlations involving tow time, proportion of time effectively fishing, y-tilt, SD y-tilt and depth). However, several of the substantial correlations were surprising and may help explain the short tow distances during 2008.

Tow time and proportion of time effectively fishing were positively correlated during 2008 but not during 2005. This result suggests the dredge performed better during longer tows during 2008.

The negative correlation between tow time and speed over ground during 2008 (but not 2005) was surprising because survey protocols are designed to achieve both a constant time (5 minutes) at specified speed (1.5 kt). In the experience of survey personnel, start and stop times used for this purpose are clear and easy to determine. In principle, speed over ground could have been determined very accurately on the bridge based on GPS. The correlation in 2008 suggests, however, that tow time and speed may have been adjusted to obtain the desired distance.

The negative correlation between *x*-tilt and *y*-tilt and between *x*-tilt and depth during 2008 (but not 2005) indicates that dredge performance during 2008 was more sensitive to depth. The positive correlations between *y*-tilt and speed over ground as well as between the SD of *y*-tilt and speed over ground indicate that dredge performance was more sensitive to speed during 2008 than during 2005.

Repeat tow analysis for cable and pump effects

Repeat tow analyses were conducted to estimate effects of different electrical cables and pumps on catch rates during the NEFSC survey. As described above, the original ("old") electrical cable used to send power to the dredge pump at the beginning of the survey was replaced at station 241 because it was too short to accommodate deep stations. The original ("old") pump was replaced and station 170 due to a malfunction.

Two types of repeat tows were carried out in connection with the 2008 NEFSC clam survey to quantify the potential effects of changes in the pump and electrical cables used on the survey

dredge. "DE2DE2" repeat stations were occupied twice by the *R/V Delaware II* (e.g. with the old and then the new cable or pump). "DE2FV" stations were occupied first by the *R/V Delaware II* (with either cable or plump) and afterwards by the *F/V Endeavor*.

Ratio estimators and a linear model analysis (see below) indicate potential cable and pump effects for ocean quahog tows during the 2008 survey were not significantly different from zero. The two ratio estimator and linear model analyses were not completely independent because they used almost the same survey data.

Background

Both electrical cables used during the 2008 survey were the same type and model. Both were purchased from the same vendor in one order prior to the 2005 clam survey. The old cable used during the 2008 survey was used during the 2005 survey also. It was shortened between surveys by removing a section near the end between the end of the 2005 survey and beginning of the 2008 survey, however, because the steel cable used to retrieve the dredge during the 2005 survey had shed wire splinters that penetrated the covering of the electric cable on the end near the dredge.

DE2DE2 repeat stations

Ocean quahog catches (50+ mm SL) were standardized using sensor tow distance to a standard area swept (5 ft x 0.15 nm = 4557 ft^2 = 423 m^2) for use in all analyses. If the sensor based tow distance was missing for a station, then the median tow distance for successful random tows during 2008 was used instead. Pairs of stations were omitted if either tow was "unsuccessful" based on sensor data (NEFSC 2007a) or had a database HG code > 36. DE2DE2 repeats with zero quahog catch in both tows would not affect estimates and were also omitted. Based on these criteria, repeat station data were available for 17 DE2DE2 repeat stations (Table B14).

The DE2DE2 repeat station data were more useful for detecting potential cable effects than pump effects. All of the original tows were made with the old cable and all of the repeat tows were made with the new cable. Five of the original tows were made with the old pump and all of the repeat tows were made with the new pump (Table B14). Fortunately, differential pressure data indicate that pump effects were likely minor because differential pressure was within the normal operating range before and after the new pump was installed (Figure B33).

The null hypothesis of no cable effect was not rejected because the ratio estimator (sum of catches with new cable / sum of catches with old cable) for DE2DE2 repeat stations was 0.8 (SE 0.22) and the 95% confidence interval (0.36, 1.23) included one (Figure B40).

DE2FV repeat stations

The repeat stations used in this analysis included random and nonrandom stations occupied by the Delaware originally during the survey and later by the commercial vessel (Table B15). Some of the survey stations were setup tows for depletion experiments that could be treated as if they were repeated by the first one or two tows in the ensuing commercial depletion experiment (see below). Length composition data were used to calculate numbers of quahogs 90 mm SL or larger, which were adjusted to the same area swept (423 m^2).

Only quahogs over 89 mm SL were used because commercial and survey selectivity curves indicate that ocean quahogs are at least 85% selected at 90 mm SL and the 90 mm cutoff is used in commercial depletion studies that involve the *R/V Delaware II* and a wide range of commercial vessels.

Forty-five stations had survey or commercial catches larger than zero (Table B15 and Figure

B41). Ratio estimators (sum of survey catches / sum of commercial vessel catches) are given below. The difference between the ratio estimators for the new pump with the old and new cables is 0.3520-0.2849=0.0671, the variance is 0.008+0.0006=0.0086, and the 95% confidence interval is (-0.11, 0.24). Thus, DE2FV ratio estimators indicate that the new cable reduced capture efficiency by about (0.3520 - 0.2849) / 0.3520=20% but the difference is not statistically significant (see below). The ratio estimate 0.31 for all of the data indicates that the capture efficiency for the survey dredge was about 31% of the capture efficiency for the commercial dredge.

DE2 configuration	Ratio	N	Var	SE	CV	Low 95% CI	Hi 95% CI	Bias
New pump-Old cable	0.3520	14	0.0080	0.0893	0.2538	0.1769	0.5271	0.0040
New pump-New cable	0.2849	28	0.0006	0.0238	0.0835	0.2383	0.3316	0.0016
Old pump-Old cable	0.4798	3	0.0708	0.2661	0.5546	-0.0418	1.0013	-0.0200
All	0.3183	45	0.0015	0.0386	0.1211	0.2427	0.3939	0.0013

Linear model analysis

Step-wise linear models were fit to the DE2FV data to refine estimates and produce variances that characterize uncertainty in estimated pump and cable effects. The dependent variable was the log of survey catch / commercial catches. Records with zero survey or commercial catches were omitted from linear models because the log of the catch ratio was undefined. A total of 41 observations were available for linear model analysis. Sample size was N=24 for pairs that had survey tows with the new pump and new cable, N=14 for survey tows with the new pump and old cable, and N=3 for survey tows with the old pump and old cable (Table B15).
Models considered in the analysis ranged from:

logRatio ~ 1

that hypothesizes a constant log ratio with no pump or cable effects to

logRatio ~Pump*ElecCable

that hypothesizes pump and electrical cable effects plus their interaction (i.e. different cable effects for each type of pump). The "best" model with the lowest AIC score was identified and estimated by the stepwise search.

The best linear model was the simplest case with logR = -1.277 (se 0.116, $p<e^{-13}$) indicating a constant log ratio with no pump or cable effects. The ratio of survey/commercial catches implied by this model is $e^{-1.277}$ = 0.28 (CV=0.12) with an approximate 95% CI (0.22-0.35).

Depletion studies and survey dredge efficiency

Survey dredge efficiency estimates are important in this assessment because they help scale relative trends to actual biomass levels in modeling and because they can be used to estimate swept-area biomass directly. By definition, dredge efficiency estimated in depletion experiments is the probability of capture (i.e. of being handled on deck) for an ocean quahog that is in the path of the dredge and large enough to fully selected by the gear. Effects of shell length and size selectivity on catches and efficiency estimates are accommodated in depletion study analyses by restricting analysis to ocean quahogs 90 mm SL or larger, which have high size selectivity (≥ 0.85) in both survey and commercial clam dredges (Figure B21).

In brief, depletion experiments usually begin with "setup" tows by the *R/V Delaware II* during the NEFSC clam survey. "Survey density" is calculated for each tow by dividing the catch by area swept, which is the dredge width times the distance traveled while the dredge was effectively

fishing based on sensor data (i.e. where while y-tilt $\leq 5.16^{o}$).

Mean survey density for each depletion experiment site is calculated by averaging the survey density from each setup tow. After the setup tows are completed, additional overlapping tows are made repeatedly by the same or different vessel over the area immediately adjacent to the setup tows until a significant decline in catch per tow is noted. Care is taken to ensure that setup tows are close to each other with little or no overlap and close to the corresponding depletion tows.

Vessel position is used as a proxy for dredge position during depletion experiments. Experiments during 1997-1998 used Loran-C to track the position of the depletion vessel with positions recorded by hand on datasheets at 30 second intervals. GPS with stored data has been used since 2002 to record position data at 6-30 second intervals. Setup tows have always been tracked by GPS at 1 second intervals. In other words, the frequency and type of information has been consistent for setup tows by the *R/V Delaware II* but has varied for depletion tows by commercial vessels.

One "Delaware II" depletion experiment has been completed for ocean quahog (experiment OQ1999-01 DE2 in Tables B16 and B17). In Delaware II depletion experiments, the research vessel carries out both the setup and depletion tows.

A relatively large number of commercial depletion experiments have been carried out (Tables B16 and B17). Commercial depletion experiments use a commercial vessel to make depletion tows after setup tows are made by the *R/V Delaware II*. Commercial depletion experiments are the preferred approach to estimating survey dredge efficiency because commercial dredges perform consistently with high efficiency and deplete the experimental site faster. Commercial dredges are inherently more efficient than the NEFSC survey dredge because water jets run at higher pressure on commercial boats and commercial dredges are heavier and less prone to vibration. Moreover, they are larger so that there is less uncertainty about their location. Bar spacing and sorting equipment on deck are usually adjusted to enhance retention of relatively small ocean quahogs before a depletion study. However, even with these adjustments to gear, commercial dredges catch relatively lower proportion of small quahogs than survey dredges, which have a small mesh liner.

In Delaware II depletion experiments, the survey dredge efficiency is estimated directly. In commercial depletion studies with setup tows, the estimated survey dredge efficiency (e) is:

$$e = \frac{d}{D}$$

where D is the estimated density from the Patch model and d is the mean survey density for the site. One disadvantage of commercial depletion experiments is the extra variance in estimated dredge efficiency due to the variance in mean survey density d. Variance of mean survey density tends to be high because the number of setup tows is typically 3-5 (Table B16).

Survey dredge efficiency estimates are available in NEFSC (2007a) for 12 depletion experiments with setup tows (11 commercial and 1 Delaware II), out of 16 total depletion experiments conducted during 1997 and 2005 (Tables B16 and B17). Three additional new commercial depletion experiments with setup tows were carried out (OQ2008-3 in SNE and OQ2008-1 and OQ2008-2 in LI, Figure B32) following the 2008 NEFSC clam survey by the *F/V Endeavor* with scientific staff from Haskin Shellfish Research Laboratory and NEFSC (Tables B16 and B17; Figures B42 through B44).

As described above, the electrical cable and pump were replaced during the 2008 survey (Figure B32). The original electrical cable and new pump were used for setup tows during the first experiment (OQ2008-01), while the new electrical cable and new pump were used during the second and third experiments (OQ2008-02 and OQ2008-03).

2008 depletion experiment methods

The *F/V Endeavor* used a 12.5 ft clam dredge that operated at a differential pressure of about 60 psi (measured at depth in the manifold of the dredge). At each depletion site, the number of bushels of clams was counted for every tow and fractional bushels were estimated by eye. In addition, one full bushel was counted and measured and an additional full bushel was counted on every fifth tow, beginning with tow two.

The survey sensor package (including GPS) was mounted on the dredge used by the F/V Endeavor during 2008, but was operational at only 106 out of a total of 232 stations due to lack of time between tows to charge batteries (particularly during depletion tows) and lack of staff to operate the unit on leg 3 of the survey. The total number of stations (232) includes stations used for ocean quahog and Atlantic surfclam depletion experiments, repeat tows, and surfclam size selectivity studies.

The start and end of fishing (when the dredge was on the bottom) was easy to determine by visual examination of SSP y-tilt and pressure sensor data. Based on SSP data, the angle of attack for the commercial dredge used by the F/V Endeavor was not prone to excess variability in y-tilt (Figure B45).

To determine tow distance at stations without SSP data, a backup pressure (depth) sensor and a backup GPS were used. The resolution of the backup pressure sensor is 4-5 meters. Backup pressure and GPS data were recorded every five seconds, in contrast to every second on the SSP. For these reasons, backup sensor data are more difficult to use in estimating dredge paths.

To develop a means to estimate tow start and stop time using backup sensor data, times on and off bottom from SSP data for OQ2008- 1 and OQ2008-02 (34 stations total) were compared to visually determined times on and off bottom from backup pressure sensor and backup GPS data. Visually determined time off bottom estimates were similar to time-off-bottom estimates based on SSP data. However, subjectively determined time-on-bottom values were greater than the SSP time on bottom values about 15-20 seconds. Time on bottom was difficult to judge because the commercial dredge was deployed using winches that do not spool freely as the dredge is deployed.

After some experimentation, 15 seconds were subtracted from the subjective time on bottom estimates from backup sensor data. The adjusted time on bottom estimates for the 34 test stations differed from SSP time on bottom values by only 4 seconds on average, with positive differences as likely as negative differences. This alternate approach was used to identify time on and off bottom based on backup sensors for all commercial tows.

See Appendix B4 for a detailed description of the cooperative survey work by the *F/V Endeavor* during 2008 and calculation of tow distances.

Patch model

The Patch model (Rago et al. 2006) was used to analyze all of the depletion experiment data used in this assessment (Table B16). Estimates for the 1997-2005 surveys are from NEFSC (2007a). Estimates for the 2008 survey (Tables B16 and B17) are described below. The Patch model is a standard approach used in NEFSC stock assessment work for a variety of shellfish and sedentary demersal finfish including Atlantic sea scallops NEFSC (2004b), ocean quahog (NEFSC 2004; 2007a), Atlantic surfclam (NEFSC 2003; 2007a) and goosefish (NEFSC 2005). The most important characteristics of the Patch model are that it is spatially explicit and it is not necessary to assume that ocean quahogs mix randomly across the entire site after each depletion tow.

The Patch model estimates three parameters for each depletion experiment: initial ocean

quahog density D; depletion dredge efficiency e, and a measure of variance k in catch data. Cell width in the Patch model was assumed to be twice the dredge width. The "gamma" parameter in the Patch model, used to measure indirect effects on catches (e.g. ocean quahogs lost from the study site without being counted on deck), was fixed at the ratio of the dredge width and cell width (γ=0.5) so that no indirect effects were assumed to occur.

Parameters are estimated by maximizing the likelihood of the observed catch data under the assumptions that the dredge path is known and that the catches are sampled from a negative binomial distribution. In computing the likelihood for the catch in each tow, the model considers the number of times each grid sampled during the tow had been sample during previous tows and adjusts the predicted catch for each tow accordingly. Likelihood profiles are used to compute confidence intervals for all model estimates and residual plots (observed – predicted catches) are used to judge model fit.

Modeling procedures

Revised procedures described in the last ocean quahog assessment (NEFSC 2007a) were used without modification for ocean quahog in this assessment. In particular, latitude and longitude data generated during the tows by GPS were smoothed with cubic splines (Figures B46 through B48). The smoothed latitude and longitude position data were interpolated along straight lines between the smoothed points to a distance of 5 ft. The grid size for 2008 commercial depletion experiments was 25 ft because the dredge was 12.5 ft wide.

As described above, SSP data were available for the OQ2008-1 and OQ2008-3 experiments, but not for the OQ2008-2 experiment. Patch model analyses in this assessment used the adjusted tow paths based on backup sensor data described above, instead of tow paths based on SSP data, to enhance interpretation and comparability of results. Otherwise, differences in start time calculations would have been confounded with effects of different electrical cables.

Survey dredge efficiency and other Patch model estimates

There were 2-4 setup and 17 depletion tows for depletion experiments completed during 2008 (Tables B16 and B17). All of the setup tows used in the analysis were located within approximately 300 m of the depletion tows. All setup tows for the same site used the same combination of electrical cable and pump. All setup tows used in the analysis were successful based on HG codes and analysis of sensor data (Appendix B3). Sensor tow distances were available for all setup tows with the exception of station 355 in OQ2008-03, which used the median tow distance for all successful tows during 2008.

Patch model fit to commercial depletion catch data was poor for OQ2008-1 but reasonably good for OQ2008-2 and for OQ2008-3 (Figures B49 through B51). Commercial dredge efficiency estimates for the OQ2008-1 and OQ2008-3 experiments were on their upper feasible bound (1.0). The area in Long Island where the OQ2008-1 and OQ2008-3 experiments was conducted has a relatively thin layer of sand on top of peat. The thin layer of sand tends to concentrate ocean quahogs near the surface where they are easy to catch (Pers. comm. E. Powell, Rutgers Shellfish Research Laboratory, Port Norris NJ).

The average survey dredge efficiency estimate for 2008 was 0.320 and estimates ranged from 0.207 to 0.467 (Table B17). The mean estimate for 2008 is relatively high compared to the "best" median estimate of 0.165 and mean estimate of 0.248 from the twelve depletion studies completed during 1997-2005 (Table B17). However, the individual and mean estimates for 2008 fall well within the range and distribution of estimates from depletion studies during 1997-2005 (Table B17).

The mean Patch model density estimate for 2008 was 0.091 quahogs per ft^2, which is similar to the estimate 0.097 quahogs per ft^2 in NEFSC 2007a from earlier studies (Table B17).

With the new data (N=15), the new median best estimate of survey dredge efficiency is 0.169 (mean 0.264, Table B18). A 90% confidence interval calculated by bootstrapping the fifteen estimates (15,000 iterations) had bounds of 0.154-0.285.

Based on Patch model estimates (Table B18), The *F/V Endeavor* appears to have consistently high efficiency for ocean quahogs. The estimates of commercial efficiency for 2008 experiments ranged 0.78 to 1.0.

Uncertainty and sensitivity

A vessel towing at 3 knots (a typical commercial tow speed) will travel 25 ft (the width of the grids used in analysis of 2008 depletion study data, see below) in 4.9 seconds (NEFSC 2007a). Thus, variability in start time estimates adds uncertainty to position data that may affect Patch model estimates to some (probably minor) degree.

As described above, the electrical cable and pump were replaced during the 2008 survey. The original electrical cable and new pump were used for setup tows during the OQ2008-01 experiment, while the new cable and new pump were used during the OQ2008-02 and OQ2008-03 experiments. Different cables (and any other gear differences in general) may cause changes in actual dredge efficiency if pump voltage and pressure change. The variance of survey dredge efficiency estimates has not been fully characterized, but is probably substantial based on the variability of estimates within and between years (Table B16). For these reasons, it is probably better to view the full set of depletion experiment dredge efficiency estimates as a distribution with an underlying mean and variance (Table B18). Individual estimates and estimates for a single survey are too imprecise to be used directly in making survey-specific estimates of survey dredge efficiency.

The accuracy of position information, smoothing, choice of grid size and assumptions about indirect effects are important considerations and uncertainties. The accuracy of position data for the ship as a proxy for position of the dredge probably depends on many factors and has probably varied among depletion experiments (NEFSC 2007a). Sensitivity analyses in NEFSC (2007b) showed that smoothed position data produce higher estimates of initial density and lower estimates of dredge efficiency than unsmoothed position data.

Dredge efficiency is harder to estimate for ocean quahogs than Atlantic surfclams (NEFSC 2007b) because ocean quahogs are found in deeper water (which makes dredge position data less reliable) and because they burrow deeper into sediments depending on environmental conditions (and are probably sampled less efficiently).

Results indicate that uncertainty in Patch model estimates is greater than depicted in likelihood profile confidence intervals (Figures B49 through B51). Preliminary results seem to indicate that the statistical properties of estimates vary among experiments in a complicated manner that depends on the spatial distribution of depletion tows, number of tows, accuracy of position data and on the density, variance in density and spatial distribution of ocean quahogs.

The gamma parameter is theoretically estimable but estimation has proven difficult in practice because the estimate for gamma is correlated with other estimates in the model and dependent on assumptions about cell size (Rago et al. 2006). Efficiency and density estimates from the Patch model tend to decrease as the assumed level of γ and indirect effects increases (Rago et al. 2006).

Assumptions about grid size reflect a compromise between the accuracy of position data and

the tenability of the assumption that animals mix within cells after each tow. Patch model estimates for ocean quahog are moderately sensitive to the changes in the assumed grid size. In particular, efficiency estimates tend to increase and density estimates tend to decrease as the grid size increases (NEFSC 2007a).

Efficiency corrected swept area biomass

Efficiency corrected swept area biomass (ESB) was estimated for years when NEFSC clam surveys collected sensor data (1997, 1999, 2002, 2005 and 2008) (Table B19). ESB results are used primarily as prior information for use in fitting other stock assessment models.

ESB for ocean quahog (Table B19) was calculated:

$$B = \frac{B'}{e}$$

where:

$$B' = \frac{\overline{\chi} A'}{a}(1+\phi)u$$

In ESB calculations, e is the best estimate of survey dredge efficiency for ocean quahogs, $\overline{\chi}$ is mean catch of fishable ocean quahogs per standard tow based on sensor data (kg tow^{-1}, see below), A' is habitat area (nm^2), a= 0.00012405 nm^2 tow^{-1} is the area that would be covered by the 5 ft wide survey dredge during a standard tow of 0.15 nm, and u=10^{-6} converts kilograms to thousand metric tons. Tow length thing again. B' is the minimum swept-area biomass prior to correction for survey dredge efficiency.

The term ϕ used in ESB calculations is the fraction of total biomass in deep water strata off LI (strata 32 and 36), SNE (strata 40, 44, 48) and GBK (strata 56, 58, 60 and 62) that were sampled only during 1999. According to NEFSC (2000), deep water strata accounted for 0%, 2% and 13% of total biomass in the LI, SNE and GBK regions during 2005. Data for deep water strata sampled only during 1999 are otherwise omitted in calculations and, in particular, calculation of mean catch per tow $\overline{\chi}$.

Habitat area for ocean quahogs in each region was estimated:

$$A' = Au$$

where $\underline{u}$ is the proportion of random tows in the region not precluded by rocky or rough ground (ocean quahogs occupy smooth sandy habitats), and A is the total area computed by summing GIS area estimates for each survey stratum in the region. Estimates for u in this assessment are the same as in NEFSC (2007a).

Mean catch per standard tow ($\overline{\chi}$) is the stratified mean catch of fishable ocean quahog for individual tows after adjustment to standard tow distance based on tow distance measurements from sensor data (d_s):

$$\chi_i = \frac{C_i d}{d_s}$$

Only random tows were used in calculations of ESB. Tows without sensor data, with gear damage or poor pump performance were excluded from ESB calculations. Following NEFSC (2004a), and as described above, tow distance was measured for each station assuming that the dredge was fishing when the blade penetrated the substrate to a depth of at least one inch. Thus, the tow distance at each station was the sum of the distance covered while the dredge angle was ≤ 5.16^{o}.

ESB estimates for the entire ocean quahog stock during 1997-2005 (Table A15, NEFSC 2004a) were computed using a formula that facilitated variance calculations (see below):

$$B_{total} = \frac{B'_{total}}{e} = \frac{\sum_r B'_r}{e}$$

Catch-ESB Mortality estimates

Fishing mortality rates were estimated directly from the ratio of catch (landings plus an assumed 5% incidental mortality allowance) and ESB data for each region and year (Table B20). The primary purpose for these calculations was as a check on model based fishing mortality estimates. Ocean quahog biomass levels may change slowly, fishing and natural mortality rates are low for ocean quahogs, and the survey during June provides a good approximation to average biomass. It was advantageous to use the ratio estimator because the surveys occur in June and because it was easy to include a wide range of uncertainties in variance calculations (see below).

Uncertainty in ESB and mortality estimates

Variance estimates for ESB and related mortality estimates are important in using and interpreting results (Tables B19 and B20; Figures B52 and B53). Formulas for estimating ESB and mortality for a single region are products and ratios of constants and random variables. Random variables in calculations are typically non-zero (or at least non-negative) and can be assumed to be approximately lognormal. Therefore, we estimated uncertainty in ESB and related mortality estimates using a formula for independent lognormal variables in products and ratios (Deming 1960):

$$CV\left(\frac{ab}{c}\right) = \sqrt{CV^2(a) + CV^2(b) + CV^2(c)}$$

where *ln(ab/c)*, *ln(a)*, *ln(b)* and *ln(c)* are normally distributed. The accuracy of Deming's formula for ESB estimates was checked by comparison to simulated estimates (NEFSC 2002). CVs by the two methods were similar as long as variables in the calculation were log normally distributed. In addition, distributions of the simulated products and ratios were skewed to the right and appeared lognormal.

CV estimates for terms used in ESB and related estimates (Tables B19 and B20) were from a variety of sources and were sometimes just educated guesses. The CV for best estimate of survey dredge efficiency (*e*) was 0.21, calculated by bootstrapping the median (15,000 bootstrap iterations) (Table B18). For lack of better information, CVs for sensor tow distances (*d*), area swept per standard tow (*a*), total area of region (*A*), percent suitable habitat (*u*), and catch were all assumed to be 10%. The CV for area swept (*a*) is understood to include variance due to Doppler distance measurements and variability in fishing power during the tow due, for example, to rocky or muddy ground.

ESB for combined stock assessment areas was estimated as described above. Variance calculations accommodated covariance among regional estimates due to using a single estimate of survey dredge efficiency:

$$CV^2(B_{total}) = CV^2(e) + CV^2(B'_{total})$$

"VPA" estimates

VPA estimates of biomass and fishing mortality (Figure B54) for ocean quahogs are useful as

a way to verify estimates from the KLAMZ model and for regions where the KLAMZ model is not applicable (see below). Surprisingly, for such a crude approach, VPA biomass estimates for the stock in the exploited region are similar to survey trends (not used in calculating VPA) and estimates from other more sophisticated modeling approaches (Figure B55).

Assuming no recruitment and that growth exactly balances natural mortality, ocean quahog biomass on January 1st and annual fishing mortality rates can be estimated for each region using a simple virtual population analysis or "VPA" approach (NEFSC 2004a). Efficiency corrected swept-area biomass estimates for 2002, 2005 and 2008 are averaged and used as the estimated biomass in 2005 which "anchors" the calculations as they work forward and backward in time. Averages for 2002-2008 are used in place of the 2005 ESB because the estimates for individual years are not precise (Table B19).

The VPA biomass estimate for January 1, 2005 is:

$$b_{2005} = \frac{B_{2002} + B_{2005} + B_{2008}}{3} - \frac{C_{2005}}{2}$$

where b_y is the VPA biomass estimate for January 1 in year y, B_y is the efficiency corrected swept area biomass for June in year y, C_{2005} is total catch weight (landings plus a 5% allowance for incidental mortality). The first ratio on the right-hand side is average efficiency corrected swept-area biomass during 2002-2008 and used as an estimate of biomass in June of 2005. Catch for 2005 is divided by two prior to subtraction because NEFSC clam surveys occur during June, when the year is half over.

Biomass estimates for years before 2005 (up to the beginning of 2009) were calculated:

$$b_{y<2005} = b_{2005} + \sum_{i=y}^{2004} C_i$$

Biomass estimates for years after 2005 were calculated:

$$b_{y>2005} = b_{2005} - \sum_{i=2005}^{y-1} C_i$$

Fishing mortality rates from VPA estimates were calculated by solving the catch equation with instantaneous rates for natural mortality and somatic growth both zero. Based on these equations, the VPA biomass estimate for GBK ocean quahogs is the mean of ESB estimates for 2002, 2005 and 2008 (1,651 thousand mt meats) because no catch occurs there.

KLAMZ model

KLAMZ (technical description in Appendix B6) is a forward projecting stock assessment model based on the Deriso-Schnute delay-difference equation (Deriso 1980; Schnute 1985; Quinn and Deriso 1999). The delay-difference equation is an implicitly age structured population dynamics model that is mathematically identical to common age-structured models if fishery selectivity is "knife-edged", somatic growth follows the von Bertalanffy equation, and natural mortality is the same for all individuals in the modeled population. Knife-edge selectivity means that all individuals alive in the model during the same year experience the same fishing mortality rate. Natural mortality rates and growth parameters can change from year to year in the KLAMZ model but are assumed to be the same for all individuals alive during each year. The model is implemented in AD Model Builder and Excel but only the AD Model Builder version was used in this assessment.

The main assumptions in the KLAMZ model for ocean quahog are: recruitment is the same

in all years (and possibly zero) or follows a "step" pattern with one constant level during early years and a different constant level during later years (see below); fishery selectivity is knife-edged; the natural mortality rate is low or constant, and growth in weight can be described by a von Bertalanffy growth curve. Recruitment is assumed to follow a simple function (and inevitably estimated to be very low for ocean quahogs) because no reliable recruitment index current exists, recruitment levels appear to be very low based on survey data, and trends in stock dynamics appear primarily due to fishing mortality.

Recruitment to the ocean quahog fishery is not knife-edged and actually occurs at sizes of about 51-86 mm SL (Figure B21). Under these circumstances, KLAMZ can be used to track trends in fishable (instead of total) biomass. Fishable biomass is dominated by relatively large individual ocean quahogs that are readily captured. Survey data used in the KLAMZ model are in units of mean kg per standard tow for the "fishable" portion of survey catches (Table B10).

Despite simplifying assumptions, KLAMZ has proven to be a relatively robust model with little or no retrospective bias which has been used successfully in for a relatively large number of stocks. It provides useful estimates of long-term biomass and fishing mortality, performs relatively well with very limited information about age and growth and when explicitly age-structured models are difficult to apply. One of the chief reasons for the utility of the KLAMZ model is statistical simplicity. The model used for ocean quahog, for example, estimates only 2-4 parameters.

Model configurations

KLAMZ model estimates were for ocean quahogs in the DMV, NJ, LI and SNE regions or for the stock in the exploited region (entire stock less GBK) during 1977-2008. The model was not used for SVA because survey data for SVA are noisy and incomplete. Configurations of the KLAMZ model for ocean quahog in each region were similar to the "best" configurations identified in the last assessment (NEFSC 2007a) following a thorough analysis of a wide range of alternate configurations. Changes are highlighted in the descriptions below. The most important changes are use of the step function recruitment pattern for LI, SNE and the exploited region. A KLAMZ model was applied to the stock in the exploited region for the first time in this assessment.

Data used in KLAMZ models for ocean quahog in this assessment were: NEFSC clam survey biomass trends and associated CV's for 1982-2008 (mean kg per tow of fishable biomass by region and year, Table B10); efficiency corrected swept-area biomass estimates for 1997-2008; and catch during 1977-2008 (landings in Table B2 with amounts for region unknown prorated by region with landings, plus a 5% allowance for incidental mortality). LPUE data are included in the model (Table B6) but only for comparative purposes (i.e. they had nil effect on model estimates). Catch data for ocean quahogs were assumed accurate and not estimated in the model. Efficiency corrected swept-area biomass (ESB) estimates for 1997-2008 are used as "prior" information that helps scale of model estimates, but were not used to measure trends because the survey data provides trend information (see below).

NEFSC clam survey and swept-area biomass data for 1994 were omitted for all stock areas because electrical voltage supplied to the pump on the survey dredge was set to 480 v, rather than 460 v, artificially increasing dredge efficiency during the 1994 survey (NEFSC 2004). In addition, survey and swept area biomass data for GBK during 1982-1984, 1989, 2002 and 2005 were also omitted because of poor survey coverage during those years.

Assumptions about growth are the same as in the last assessment. In particular, the growth parameters $\rho=e^K$ (where K=0.0176 is the von Bertalanffy growth parameter for weight), $J_t= w_{k-1}/w_k$ = 0.9693 (where w_j is predicted weight at age j) are constant and the same for all regions (NEFSC

2004). These growth parameters mean that quahogs in the model are slow growing, and that quahogs recruit to the fishery (reach 70 mm SL) at age k=26 (Figure A62, NEFSC 2004). Growth patterns differ among regions (Lewis et al. 2001) but ocean quahogs are difficult to age and there is too little information available to use region-specific growth curves (NEFSC 2000). The MAB growth curve was used for all regions where fishing occurs and the growth curve for GBK was used in the model for GBK (Lewis et al., 2001; Figure B3). The assumed natural mortality rate was M=0.02 y^{-1}, except in sensitivity analyses.

An assumed level of variance in instantaneous somatic growth rates (IGR) for old recruits is used to help estimate the initial age structure of ocean quahogs in the initial years of the model (Appendix B6). However, as described in NEFSC (2007a), this constraint is unimportant because estimated age structures were stable due to assumptions about recruitment and low mortality rates.

ESB data are important in KLAMZ models for ocean quahogs as a source of information about biomass scale. To use ESB data as a measure of scale while ignoring trend (see Appendix B6), the likelihood component for trends in ESB data were set to 10^{-6} so that the survey scaling parameter Q was calculated but the trend was ignored. Information in ESB data about biomass scale is contained in the estimated survey scaling parameter Q.

As described in Appendix B6, the likelihood of the survey scaling factor is calculated assuming that estimates of Q are from a lognormal distribution:

$$L = 0.5\left[\frac{\ln(Q) - \tau}{\varphi}\right]^2$$

where L is the negative log likelihood, $\varphi = \sqrt{\ln(1 + CV)}$ and $\tau = \ln(\overline{q}) - \frac{\varphi^2}{2}$ is the mean of the log normal distribution. For ocean quahog ESB data, the mean of the prior $\overline{q}$ = $ln(1)$ = 0 if ESB data measure stock biomass accurately and CV=0.21 is the bootstrap coefficient of variation (standard deviation / mean) for the median survey dredge efficiency used in calculating ESB (Table B18).

Parameters estimated

KLAMZ models for ocean quahog in this assessment estimate two to four parameters by maximum likelihood and numerical optimization. The parameters potentially estimated are logarithms of: 1) biomass at the beginning of 1977, 2) escapement biomass (total biomass less biomass of new recruits) at the beginning of 1978, and 3) annual recruitment biomass (which is assumed constant over time for each region with one parameter or constant during two time periods with two parameters). In models where recruitment was too low to estimate, recruitment was fixed at an assumed value near zero (1 kg y^{-1}) which reduced the number of parameters estimated.

Fishing mortality rates are calculated solving the catch equation numerically. Survey scaling parameters were calculated using a closed form maximum likelihood estimator.

Variance estimates

Variances for biomass and fishing mortality estimates and for model parameters can be estimated by the delta method using exact derivatives calculated by AD Model Builder libraries, by bootstrapping, or by MCMC (Appendix B6). Estimates in this assessment were from the delta method or bootstrapping.

KLAMZ Results-DMV

As in previous assessments (NEFSC 2004; 2007a), estimated recruitment was near zero and hard to estimate in preliminary runs for DMV. The annual recruitment level was therefore fixed at very low value (1 kg y^{-1}) in final runs. Survey data generally indicate that recruitment has been low in DMV since 1978 (Figure B24) although some small ocean quahogs are present (Figure B30).

The KLAMZ model for ocean quahog in the DMV area (Figure B56) fit NEFSC survey and LPUE data well (LPUE data did not affect model estimates). The CV of arithmetic scale residuals (26%) for NEFSC survey data was smaller than the mean CV (35%) for mean kg/tow survey data but within the range of observed values (21%-53%). The estimated survey scaling parameter for ESB data was Q=0.96 indicating that the model was able to match the observed ESB biomass levels on average during 1995-2008 using the catch data and trends in NEFSC survey data.

Based on KLAMZ model results, biomass of ocean quahogs in DMV declined steadily after 1978 (Figure B56). Estimated fishable biomass during 2008 was 30% of the estimate for 1978 (Figure B56).

KLAMZ Results-NJ

The KLAMZ model for ocean quahog in the NJ area (Figure B57) fit NEFSC survey and LPUE data well (LPUE data did not affect model estimates). The CV of arithmetic scale residuals (43%) for NEFSC survey data was larger than the mean (19%) and range (14%-24%) of CV values for mean kg/tow survey data. The estimated survey scaling parameter for ESB data was Q=0.96 indicating that the model was able to match the observed ESB biomass levels on average during 1995-2008 using the catch data and trends in NEFSC survey data.

Based on KLAMZ model results, biomass of ocean quahogs in NJ declined steadily after 1978 (Figure B57). Estimated fishable biomass in NJ during 2008 was 40% of the estimate for 1978.

KLAMZ Results-LI

Preliminary KLAMZ model fits for ocean quahog in the LI area indicated that the model with constant recruitment was not able to match the apparently increasing abundance trends before 1994 and decreasing abundance trend afterwards without estimating an implausible survey scaling parameters Q=0.48 (Figure B58). A step function recruitment model with different levels of constant recruitment before and after a specified point in time was therefore used instead. A series of runs with the change in recruitment occurring at 1990 to 1999 indicated 1994 was the best change year for recruitment (Figure B59). The step function for LI allows for a higher level of recruitment prior during 1977-1993 (Figure B60) while a strong year class was recruiting to the fishery (Figures B24 and B28) and a lower level afterward.

The model (Figure B61) with step function recruitment fit the survey and LPUE data for ocean quahogs better than the model with constant recruitment (LPUE data did not affect model estimates) and the change in total likelihood indicated that the additional parameter was statistically significant. The CV of arithmetic scale residuals (25%) for NEFSC survey data was larger than the mean (18%) but within the range (14%-28%) of CV values for mean kg/tow survey data. The estimated survey scaling parameter for ESB data was Q=1.04 indicating that the model was able to match the observed ESB biomass levels on average during 1995-2008 using the catch data and trends in NEFSC survey data.

Based on KLAMZ model results (Figure B61), biomass of ocean quahogs in LI increased steadily after 1978 until 1993 when recruitment decreases and fishing mortality increased to

maximum levels. Estimated fishable biomass in LI during 2008 was 89% of the estimate for 1978 and 70% of the maximum estimated biomass during 1992 (Figure B61).

KLAMZ Results-SNE

The KLAMZ model for ocean quahog in the SNE area (Figure B62) with a single recruitment parameter did not fit the apparently increasing trend in survey data prior to 1994 and decreasing trend afterwards. A step function recruitment model was therefore used instead. A series of runs with the change in recruitment occurring at 1990 to 1996 indicated 1993 was the best change year for recruitment (Figure B63). The step function for LI allows for a higher level of recruitment prior during 1977-1992 (Figure B64) while a strong year class was recruiting to the fishery (Figures B24 and B28) and a lower level afterward.

The model with step function recruitment (Figure B65) fit NEFSC survey and LPUE data better (LPUE data did not affect model estimates) and the change in total likelihood indicated that the additional parameter was statistically significant. The CV of arithmetic scale residuals (27%) for NEFSC survey data was smaller than the mean CV (35%) for mean kg/tow survey data but was within the range of observed values (18%-47%). The estimated survey scaling parameter for ESB data was Q=1.04 indicating that the model was able to match the observed ESB biomass levels on average during 1995-2008 using the catch data and trends in NEFSC survey data.

Based on KLAMZ model results, biomass of ocean quahogs in SNE increased steadily and then declined after 1992 when recruitment declined and fishing mortality increased dramatically (Figure B65). Estimated fishable biomass in SNE during 2008 was 99% of the estimate for 1978 and 78% of the maximum estimated biomass during 1994.

KLAMZ Results-GBK

The KLAMZ model for ocean quahog in the GBK area fit NEFSC survey data well although only 5 survey observations were available (Figure B66). The CV of arithmetic scale residuals (21%) for NEFSC survey data was smaller than the mean CV (18%) for mean kg/tow survey data but within the range of observed values (18%-27%). Only three ESB observations were available for GBK. The estimated survey scaling parameter for ESB data was Q=1.01 indicating that the model was able to match the observed ESB biomass levels on average during 1995-2008 and trends in NEFSC survey data to some extent. Trends in survey and ESB data were conflicting. The survey data varied without trend during 1986-2008. The shorter (and higher variance) ESB data for 1997, 2000 and 2008 showed a consistent increase.

Based on KLAMZ model results, biomass of ocean quahogs in GBK increased steadily after 1978. Estimated fishable biomass during 2008 was 13% higher than the estimate for 1978 (Figure B66).

KLAMZ Results-exploited region

The KLAMZ model for ocean quahog in the exploited stock area (Figure B67) fit NEFSC survey trends reasonably with a single recruitment pattern. However, the model with step function recruitment was significantly better based on log likelihood. A series of runs with the change in recruitment occurring at 1990 to 1996 indicated 1993 was the best change year for recruitment (Figure B68). The step function allows for a higher level of recruitment prior during 1977-1992 (Figure B69) and a lower level afterward.

The model with step function recruitment (Figure B70) fit NEFSC survey data better but fit LPUE poorly (LPUE data did not affect model estimates). Lack of fit to LPUE data was probably

due to the fishery shifting its distribution across the large area modeled to maintain relatively high catch rates. The CV of arithmetic scale residuals (21%) for NEFSC survey data was larger than the mean (13%) and range (10%-14%) of CV values for mean kg/tow survey data. The estimated survey scaling parameter for ESB data was Q=1.06 indicating that the model was able to match the observed ESB biomass levels on average during 1995-2008 using the catch data and trends in NEFSC survey data.

Based on KLAMZ model results (Figure B70), biomass of ocean quahogs in entire stock area less GBK declined after 1978 and then more steeply after 1994 when recruitment declined and fishing mortality was relatively high. Estimated fishable biomass during 2008 was 62% of the estimate for 1978.

Biomass estimates from the KLAMZ model for the exploited region were similar to the sum of biomass estimates from regional KLAMZ models for DMV, NJ, LI and SNE plus VPA estimates for SVA, and to the sum of regional VPA estimates (Figure B55). Despite this high degree of consistency, 95% confidences intervals from the model for the exploited stock were wide (e.g. 1513 to 3981 thousand mt in 1978 and 1056-2195 thousand mt in 2008) indicating considerable uncertainty in estimated biomass (Figure B55).

Retrospective patterns

A retrospective analysis was carried out using the KLAMZ model for the exploited region by using 2000-2008 as the terminal year in the model (Figure B71). Estimates did not tend to change between runs unless a year with a survey (2002, 2005 or 2008) was dropped. There was no evidence of the typical retrospective pathology. Terminal years tended to be similar in all runs. Historical pre-1983 estimates changed in a random manner between runs, suggesting that recruitment during the first time period (1978-1992) was difficult to estimate.

"Best" biomass estimates

Biomass and fishing mortality estimates from regional KLAMZ models were used as the best estimates of biomass and fishing mortality for ocean quahogs in DMV, NJ, LI, SNE and GBK during 1977-2008 (Tables B21 and B22; Figures B72 through B74). VPA biomass estimates were used for SVA because a KLAMZ model was not available. Biomass estimates for the exploited stock and total stock are the sums of regional estimates. Fishing mortality rates for SVA, the exploited stock and total stock were calculated by solving the catch equation for F using observed landings, biomass and instantaneous rates of recruitment and growth for the appropriate region during the year.

. CVs for best biomass and fishing mortality estimates in DMV, NJ, LI, SNE and GBK are asymptotic estimates from KLAMZ model runs. The CVs for biomass and fishing mortality in the exploited region are from the KLAMZ model for the exploited region (regional variances were not used to avoid assumptions about independence in errors among regions during the same year). CVs for fishing mortality in the entire stock were assumed the same as for the exploited region. CVs for biomass and fishing mortality in SVA were assumed to be the same as the average CV for ESB (0.96, Table B19) in SVA.

As noted before, biomass estimates for ocean quahogs are not sensitive to choice of modeling approach (Figure B55). In addition, updated estimates for recent biomass and fishing mortality in this assessment are similar to estimates and projections in the last assessment (NEFSC 2007a, Figure B73), even for the LI and SNE models which assumed constant recruitment patterns in NEFSC (2007a) and two-step recruitment patterns in this assessment.

Biological Reference Points
(TOR-3)

Managers use biological reference points (BRPs) for fishing mortality and stock biomass in dealing with ocean quahogs and other species in the US EEZ. BRPs for management targets and management thresholds are required. Targets are BRPs that represent desirable stock conditions. Thresholds are BRPs that identify undesirable stock conditions.

BRPs for US fisheries are generally linked in policy to maximum sustained yield (MSY). In particular, the overfishing threshold is often F_{MSY}, MSY, or a proxy for either F_{MSY} or MSY. Fishing mortality levels at or higher than the F_{MSY} threshold constitute overfishing. Managers may choose any fishing mortality target level < F_{MSY} as a target for healthy stocks.

Similarly, the target reference point for biomass ("stock size") is B_{MSY}, which is the stock biomass level that produces MSY when the stock is harvested at F_{MSY}. Policy for choosing biomass thresholds is specified in the National Standard Guidelines. To the extent possible, the stock size threshold should equal whichever of the following is greater: 1) one-half the MSY stock size; or 2) the minimum stock size at which rebuilding to the MSY level would be expected to occur within 10 years if the stock or stock complex were exploited at the maximum fishing mortality threshold.

Current BRPs for ocean quahog

The Atlantic Surfclam and Ocean Quahog Fishery Management Plan (FMP, Amendment 12) specifies $B_{Target} = B_{MSY}$, which is assumed be one-half of virgin biomass *for the whole stock*, and $F_{Target} = F_{0.1}$ for the *exploited region only* (whole stock less GBK) The biomass and fishing mortality thresholds are $B_{Threshold}$= ½ B_{MSY} and $F_{Threshold}$=$F_{25\%}$ (the fishing mortality rate that reduces life time egg production for an average female to 25% of the average level with no fishing). The FMP does not specify whether the thresholds apply to the whole stock or exploited region only. Based on the last assessment, current estimates for the fishing mortality BRPs are F_{Target} = $F_{0.1}$=0.0275 y^{-1} and $F_{Threshold}$=$F_{25\%}$=0.0517 y^{-1}.

Previous assessments and reviews concluded that $F_{25\%}$ is a poor threshold reference point because it is a poor proxy for F_{MSY} in a long-lived species like ocean quahog with assumed natural mortality rate M=0.02 y^{-1} (NEFSC 2007a; 2007b). Simulation analyses in Clark (2002) indicate that long-term yield from unproductive fish stocks is maximized at fishing mortality rates of $F_{45\%}$ or lower. The same simulations show that fishing at $F_{25\%}$ would eventually result in spawning stock biomass levels less than 25% of the virgin level, which is below the B_{MSY} estimate of one-half virgin biomass. Thus, the current proxies for F_{MSY} and B_{MSY} are not compatible.

Revised and recommended fishing mortality rate reference points

Per recruit reference points (Table B23) for ocean quahogs are from a length-based per-recruit model in the NEFSC Stock Assessment Toolbox[6] The length-based approach is better for ocean quahogs because fishery selectivity and maturity have been estimated in terms of shell length. Biological and fishery parameters (Table B24) in per recruit models were the same as in the last assessment (NEFSC 2007a).

The problem of choosing an F_{MSY} for ocean quahogs is difficult because we have relatively little experience with unproductive stocks like ocean quahogs. More importantly, MSY theory may not be applicable to ocean quahogs because low productivity may preclude economically viable

6 Contact Alan Seaver (Alan.Seaver@noaa.gov), Northeast Fisheries Science Center, Woods Hole, MA, USA for information and access to the Stock Assessment Toolbox.

levels of sustained catch. Productivity is low for the stock as a whole and particularly in the south because recruitment events have been infrequent and regional, growth is slow, and there is a long lag time between spawning and recruitment to the mature or fishable stock. There is a chance that fishing on Georges Bank could be sustainable, as growth and potential recruitment rates are relatively high. It is probably not possible to maintain a sustainable fishery on the currently exploited region where recruitment and growth rates are very low. For these reasons, recommended reference points in this assessment are described as thresholds and targets but not as proxies for F_{MSY} or B_{MSY} related reference points.

Quahog specific simulation analyses were not performed for this assessment. In absence of simulations for ocean quahog, the best available information is Clark's (2002) simulation analyses of F_{MSY} proxies applicable to long lived west coast groundfish. The west coast ground fishery includes a substantial number of long-lived fishes that are managed based on Clark's (2002) simulation analyses. F_{MSY} proxies for west coast groundfish were considered at a workshop that resulted in specific recommendations for stocks with a range of life history characteristics (Appendix B7). In particular, the workshop recommended $F_{40\%}$ for relatively productive Pacific whiting and flatfish, $F_{45\%}$ for other groundfish, and $F_{50\%}$ for *Sebastes* spp. (rockfish) and *Sebastolobus* spp. (thornyheads).

The Invertebrate Subcommittee considered $F_{40\%}$ and $F_{50\%}$ as fishing mortality thresholds for ocean quahogs (Table B25). $F_{50\%}$ might be better for ocean quahogs because *Sebastes* spp. are shorter lived, grow faster and reproduce on a more regular basis than ocean quahogs. On the other hand, ocean quahogs have some characteristics that might enhance productivity to some extent (e.g. lack of fishing on Georges Bank). High quality landings and low levels of indirect and discard mortality probably enhance stock assessment information for ocean quahogs and reduce the chances for inadvertent overfishing. After discussion, the subcommittee decided to "split the difference" and recommend $F_{45\%}$ as the fishing mortality threshold which the SARC 48 then accepted.

The current $F_{Threshold}$ for ocean quahogs ($F_{25\%}$) is compared to fishing mortality rates for the exploited portion of the quahog stock (i.e. the whole stock less GBK) to determine if overfishing is occurring. This approach is the result of a policy decision taken by the Mid-Atlantic Fishery Management Council and is unique to ocean quahogs. In the absence of clear policy, the Invertebrate Subcommittee makes no recommendation regarding how fishing mortality should be calculated for comparison to the fishing mortality threshold.

MSY theory may not be applicable to ocean quahogs, as described above. However, from a technical point of view mortality rates calculated for the whole stock including Georges Bank do not describe conditions on either the exploited portion or unexploited portions of the stock (Hart 2003). In particular, fishing mortality may be higher than desired on the exploited portion (resulting in foregone yield and relatively low biomass conditions) and zero on the unexploited portion (resulting in foregone yield).

Very little simulation or other information was available for recommending biomass reference points for ocean quahog. The current proxy was therefore retained as a target reference point except that the target was defined as one-half of the fishable (fully selected) biomass during 1978 (under pre-fishery conditions) instead of one-half of virgin biomass. Fishable biomass during 1978 (pre fishery) was used in place of virgin biomass because it is the only available estimate of stock size under unfished conditions. Results in this assessment indicate that virgin biomass likely varied in long slow cycles prior to fishing as infrequent strong year classes slowly grew to fishable size.

The recommended biomass threshold of 1.432 mmt (40% of the pre-fishery biomass during

1978) is an *ad hoc* approach judged to be more realistic than the current threshold (25% of virgin biomass). It is possible that a higher threshold may be required, particularly if the stock on GBK is found to be unproductive.

The growth curve used in calculations was for the ocean quahogs in the Mid-Atlantic Bight that did not include growth data from the GBK area where growth is faster and maximum size is larger (Lewis et al., 2001). Growth and recruitment assumptions should be revisited if managers decide to apply threshold fishing mortality rates to the whole stock (including GBK) or if a fishery develops on GBK.

Uncertainty in biological reference points

Ocean quahogs (including GBK) may or may not have the potential for supporting sustainable catches in the long term. Some recruitment and growth occurs each year but at low levels. Much depends on the response of the stock on Georges Bank to fishing, where growth and potential recruitment rates are relatively high. It is probably not possible to maintain a sustainable fishery on the currently exploited region where recruitment and growth rates are very low.

It is probably constructive and technically valid to view the ocean quahog fishery and fishing on Georges Bank is as an adaptive management experiment. The stock (including Georges Bank) may or may not support a sustainable fishery, the answer should be clear after a decade or two of fishing on Georges Bank, and managers should be prepared to react in either case. Policy and management actions in the event the fishery is not sustainable should be considered carefully beforehand. One obvious option would be to discontinue fishing, for ocean quahogs, potentially for a decade or more, if stock biomass reaches its biomass threshold.

In conducting the adaptive management experiment, it is important that removal rates are low enough to provide one or two decades for increased recruitment following fishing because the lag time between spawning and recruitment to the fishery is relatively long. At high fishing mortality rates, it would be theoretically be possible to eliminate the spawning biomass before recruitment has a chance to occur.

Threshold reference points were sensitive to assumptions about natural mortality. The range of values for $F_{45\%}$ was 0.017, 0.019 and 0.027 y^{-1} at assumed natural mortality levels of M=0.015, 0.02 and 0.025 y^{-1}. Thus, there is considerable uncertainty associated with uncertainty in M. Uncertainty in biomass reference points is probably about the same as relative uncertainties in fishing mortality thresholds.

Stock Status
(TOR-4)

Ocean quahogs in the US EEZ are not overfished and overfishing is not occurring. Total fishable stock biomass (all regions) during 2008 was 2.905 million mt (Table B21), which is above the current and recommended management target of 1.790 million mt. As shown in Figure B74, there is nil probability based on model results that 2008 biomass for the entire stock was below the management target. The fishing mortality rate during 2008 for the stock in the exploited region was F= 0.01 y^{-1} (Table B22) which is below the current $F_{25\%}$ = 0.0517 y^{-1} and recommended $F_{45\%}$=0.0219 threshold reference points. As shown in figure B74, there is nil probability based on model results that fishing mortality during 2008 exceeded the current or recommended threshold values. For comparison, the fishing mortality rate for the entire fishable stock (all areas) during 2008 was 0.0055 y^{-1}.

Biological condition of the EEZ stock

The ocean quahog population is relatively unproductive. Total biomass is gradually declining and approaching the recommended biomass target (½ virgin of the unfished biomass during 1978) after about three decades of relatively low fishing mortality (Figure B74).

Based on survey data (Figure B23), LPUE data (Figure B9) and best estimates for 1977-2008 (Figure B72), declines in stock biomass have occurred in southern regions (SVA, DMV and NJ) where the fishery has been active longest and where little recruitment has occurred. During 2008, fishable stock biomass in SVA, DMV and NJ was less than half of pre-fishing (1978) levels (Figure B72). In contrast, stock biomass in northern regions LI and SNE increased after 1978 to due to recruitment and growth and then began to decrease in the mid-1990s when fishing commenced (Figure 72). Biomass in the unfished GBK region appears to have increased gradually since 1978 (Figure B72).

The LI, SNE and GBK regions in the north contained about 67% of total fishable biomass during 1978 and about 84% of the remaining fishable biomass during 2008 (Figures B75 and B76). The GBK region, which is currently not fished due to risk of PSP contamination, contained about 32% of total fishable biomass during 1978 and about 45% during 2008 (Figures B75 and B76).

Recruitment biomass is remarkably low (< 48 thousand mt during all years, Figure B77) for a stock with biomass levels in excess of 3 million mt during 1978-2008 (Figure B75). Almost all recruitment since 1978 occurred in northern regions (LI, SNE and GBK). Estimated recruitment declined during 1992-2000. Since 2000, recruitment (about 17 thousand mt per year) has occurred almost entirely on GBK (Figure B75).

Fishing effort and mortality

Fishing effort has shifted to offshore and northern grounds over time as catch rates and abundance in the south declined (Figure B6). Analysis of LPUE data for individual 10-minute squares indicates considerable fishing-down on fishing grounds that historically supplied the bulk of landings (Figure B12). There is no indication that LPUE increased on historical grounds after fishing effort was reduced.

Fishing mortality rates during 2008 are relatively low for the entire stock (F=0.0056 y^{-1}) and for the exploited stock (F=0.01 y^{-1}), which excludes GBK (Figure B64). Fishing mortality rates in southern areas declined over the last decade to low levels (F = 0.0, 0.003 and 0.0047 y^{-1} for SVA, DMV and NJ during 2008). Fishing mortality rates for LI increased abruptly during 1992 as effort increased, declined and then increased to F=0.0193 y^{-1} during 2008. Fishing mortality rates for SNE increased after 1995 to levels above 0.01 y^{-1} during 1997-2000 and then decreased to 0.0041 y^{-1} during 2008.

Survey size composition (Figures B26 and B30) and fishery data (Figure B13) indicate a strong year class in a relatively small area within SNE off the southwest coast of Cape Cod. Growth rates in this area (which is intermediate between the MAB and GBK) are uncertain but these recruits are expected to enter the fishery over the next decade. Survey data for GBK (Figures B24, B25 and B30) where growth is faster indicate a recent recruitment event that has already reached fishable sizes (Figure B73). This recruitment was not detected until 2008 because of low coverage during the 2002 and 2005 surveys.

Productivity under fishing

Questions about the potential productivity of ocean quahog are becoming important as the stock is fished down from high virgin levels to B_{MSY}. Uncertainties about productivity are closely

related to choice of accurate F_{MSY} and B_{MSY} proxies and to other decisions that affect sustainability and fishery profitability.

Ocean quahogs in the EEZ do not currently show a clear increase in stock productivity due to higher recruitment and increased growth rates, which would be expected as biomass declines to B_{MSY} levels. Indeed, estimated recruitment in northern regions began to decrease in about 1993 (Figure B77) as the fishery moved into the northern LI and SNE regions. Given the long periods between settlement and recruitment and slow growth once ocean quahogs reach fishable size, any increase in stock productivity may be delayed (Powell and Mann 2005).

Biological condition of ocean quahog in Maine waters
See Appendix B2.

Projections

(TOR-5)

Median stochastic projections were similar to corresponding deterministic projections (Table B26). As with the deterministic results, stochastic projections indicate that overfished (low biomass) stock conditions are not likely to occur by 2015 under any of the states of nature or management actions considered (Table B27). Overfishing relative to the true $F_{45\%}$ mortality threshold is not likely to occur under status-quo landings or at the minimum landings level specified in the FMP (Table B27). However, there is some probability of overfishing at the current quota and maximum landings level specified in the FMP, particularly if natural mortality $M \leq 0.02$ (Table B27).

Based on deterministic and stochastic projections, overfishing relative to the true $F_{45\%}$ would occur by 2015 under most of the states of nature considered. Most of these results are artifacts, however, because $F_{45\%}$ is one of the most conservative harvest policies considered and harvest at the relatively aggressive $F_{40\%}$, $F_{20\%}$, $F_{0.1}$ policies would constitute overfishing relative to $F_{45\%}$ by definition.

Projections indicate that landings levels based on $F_{45\%}$ and $F_{50\%}$ and exploited stock biomass would not result in F values for the entire stock larger than $F_{45\%}$ under any of the states of nature.

Stochastic biomass projections (Figure B79) indicate that changes in biomass are likely to be gradual under all harvest policies and states of nature considered. Projected fishing mortality estimates (Figure B80) show that some of the harvest policies considered are relatively aggressive in comparison to the status-quo catch policy.

Projection methods

Projected fishable biomass, fishing mortality and landings during 2010-2015 were calculated in two ways. The first method is a relatively simple approach used in the last assessment that has proven to be useful and reliable. The simple approach works well for ocean quahogs because stock biomass changes very slowly under current conditions. The principle advantage of the simple approach is that it provides projection information for each separate region based on regional conditions, as well as for the exploited region and total stock area. The principle disadvantage is that the uncertainty calculations for the simple approach are relatively crude.

The second approach provides stochastic projections based on the KLAMZ model for ocean quahogs in the exploited portion of the stock. This more complicated method captures uncertainty in 2008 biomass in addition to uncertainty in estimated recruitment levels. The stochastic approach is similar to the methods used for finfish in the US. Stochastic calculations for quahogs are slightly more complicated, however, because they involve interpreting projections for the stock in the

exploited region (less GBK) in terms of the entire stock area.

All projections were started in 2008, the last year with best estimates from stock assessment models for ocean quahogs. At the time the projections were done, reasonable "anticipated" estimates of landings for 2009 were available. Therefore, all projections used actual landings for 2008 and anticipated landings for 2009 (17,690 mt meats = 3.9 million bu).

The range of harvest polices (management actions) used in projections (Table B28) included four constant landings policies (status quo, FMP minimum, FMP maximum, and FMP current quota) and five target fishing mortality policies ($F_{0.1}$, $F_{25\%}$, $F_{40\%}$, $F_{45\%}$ and $F_{50\%}$). As described below, the constant F policies were simulated by calculating a target landings level corresponding to the intended fishing mortality rate policy and the best estimate of 2008 biomass. Total catch impacting the stock in projections was landings plus 5% for assumed incidental mortality.

States of nature assumed in projections involved a range of possible values for natural mortality (M=0.015, 0.02 and 0.025) and a range of biomass levels. Deterministic projections used a range of possible biomass levels in 2008, while stochastic projections included uncertainty in 2008 biomass automatically based on bootstrap results.

Projections with F assumed known are unrealistic because F cannot be controlled directly by managers and is never truly known. Annual catch limits, in contrast, can be specified by managers and landings may be known. In practice, managers specify a landings level for ocean quahogs that are expected to generate a "target" or expected level of F. Therefore, projections in this assessment for ocean quahogs involving a target level of F (e.g. $F_{45\%}$) were carried out by calculating the catch in approximately the same manner as managers would do in managing the actual fishery based on the best biomass estimate for 2008. For example, projections with target $F= F_{45\%}$ were carried out using catch $C=F_{45\%}$ x B_{2008} for years 2010-2015.

Some of the possible states of nature considered in simulation analyses involve different levels of natural mortality M that imply different underlying biomass levels. However, managers are expected to use only the best estimates of biomass during 2008 (assuming M=0.02) in setting catch limits for 2010-2015. Therefore, management actions (landings and catch levels) are always calculated based on the best biomass estimates with M=0.02. Management decisions considered in projection analyses involve choices among harvest policies (e.g. maintain status quo landings/catch or harvest at the $F_{45\%}$ level), rather than choices among biomass estimates.

Reference points and states of nature

Mortality reference points used in simulations to determine the probabilities of overfishing were based on the true state of nature in the scenario tested**.** For example, scenarios with true M=0.015 used $F_{45\%}$=0.017 in comparisons while scenarios with true M=0.20 used $F_{45\%}$=0.0219 (Table B23). The true value of the $F_{45\%}$ reference point depends on the state of nature because the reference point depends on M (Table B23). Mortality reference points and the state of nature are linked in comparisons because the goal of the analysis is to evaluate the probability that fishing mortality in the ocean quahog stock will exceed the true value of the threshold reference point in 2015.

Biomass reference points were not adjusted for the assumed true value of M in deterministic projections although estimated biomass in 1978 and derived biomass reference points depend on natural mortality. The best method for simultaneously incorporating uncertainty in M, 1978 biomass and 2008 biomass was not clear and probably too complicated for simple deterministic calculations.

For stochastic projections, biomass reference points were adjusted for the assumed true value of M. In particular, the threshold biomass was 40% of the estimated biomass during 1978 based on

original model runs for the exploited area and for GBK with the appropriate level of *M*.

Simple deterministic methods

In deterministic projections, bounds for true biomass in 2008 were B_{low}=1,438 and B_{high}=1,899 thousand mt meats for the exploited portion of the stock. The bounds were taken from an 80% bootstrap confidence interval (2000 iterations) analysis with the KLAMZ model for the exploited area. As described above, biomass in GBK during 2008 was assumed to be in the same proportion as the best estimates for 2008. Adjusting for the proportion of the biomass on GBK during 2008 (45%), the bounds for biomass of the entire stock are 2,633 and 3,475 thousand mt.

Deterministic projections are generally similar to the medians of results from more complicated stochastic projections (Jacobson and Cadrin 2004). Deterministic projection calculations for ocean quahog in this assessment use the following equations to represent biomass dynamics:

$$X = G + r - M - F$$

$$B_{t+1} = B_t e^X$$

$$F = \frac{C}{B} \quad or \quad C = FB$$

where X is the net instantaneous annual rate of change, G is the instantaneous rate for somatic growth in weight, r is the rate for recruitment, $M = 0.02\ y^{-1}$ is the rate for natural mortality rate, F is the rate for fishing, C is catch (e.g. landings + 5%), and B is fishable biomass. When catch is assumed known, the fishing mortality rate F can be calculated iteratively. When F is known, catch can be calculated directly.

Instantaneous rates for recruitment and growth during 2009-2015 were assumed to be the same as in 2008 (Table B29). Proportions of total catch in each region during 2010-2015 were assumed to be the same as in 2008 (Table B27). Proportions of stock biomass in each region during 2008 were assumed to be the same as in best estimates for 2008 (Table B29).

Simple projections are probably best interpreted as medians. Some crude measures of uncertainty are, however, available. Uncertainty in deterministic projections is roughly the same as uncertainty in the best biomass estimates for 2008 because recruitment is very low and projections are short-term. Thus, CVs for best estimates of 2008 biomass (based on the variance of 2008 biomass estimates from KLAMZ models for the exploited region and for GBK) can serve as estimates of uncertainty for projected biomass in 2015. If uncertainty in biomass is lognormal, then bounds for an asymmetric 80% confidence interval can be computed approximately as the median estimate multiplied or divided by $e^{1.28\sigma}$ where $\sigma = \sqrt{\ln\left(CV^2 + 1\right)}$. If uncertainty in biomass is lognormal, and uncertainty in assumed catches is zero, then fishing mortality is also lognormal with the same CV as for biomass (Deming 1960).

CVs and standard deviations for uncertainty in projected biomass and fishing mortality from best estimates, with standard deviations (σ).

Region	Total less GBK	Total
CV	0.101	0.135
σ	0.101	0.135
$1/e^{1.28\sigma}$	0.879	1.138
$e^{1.28\sigma}$	0.841	1.189

Deterministic projections for biomass and fishing mortality levels were compared to a range of reference points. Overfishing was judged "likely" for a scenario if projected median fishing mortality exceeded the threshold reference point. Threshold reference points were compared to median fishing mortality for both the exploited portion of the stock and the entire stock area. Overfished stock status was judged likely if projected median biomass for the entire stock was lower than the biomass threshold.

Stochastic projection methods

Uncertainty in biomass and estimated recruitment from the KLAMZ model for ocean quahogs in the exploited and GBK regions was estimated by bootstrapping survey data and KLAMZ models for the two regions (2000 iterations). Projections were carried out for the exploited region using each bootstrap biomass estimates for 2008 as the starting point and assuming recruitment during 2009-2015 at the estimate from the model. See technical documentation for the KLAMZ model in Appendix B6 for detailed description of bootstrap and projection methods.

For simplicity, biomass on GBK during 2000-2015 in projections was assumed the same as in 2008 and uncertainty in GBK biomass was ignored. Thus, stochastic projection calculations for the entire stock ignore key uncertainties but hopefully provide useful (though understated) estimates of uncertainty for the stock as a whole. This is a topic for future research and projections in the next assessment should include the full range of uncertainty for the entire stock.

Distributions of projected biomass and fishing mortality in 2015 from stochastic projections were compared to a range of reference points. The range of natural mortality values considered in stochastic projections (M=0.015, 0.02 and 0.025) was the same as in deterministic projections. It was not necessary to assume a range in 2008 biomass estimates because the stochastic projection analyses include uncertainty in estimated biomass automatically via the bootstrap step. Projections under an assumed state of nature with M=0.015, for example, started with fitting KLAMZ models for the exploited portion of the stock and for GBK with M=0.015 assumed in the model. The resulting model for the stock in the exploited region was bootstrapped and then projections were carried out for each management action considered.

The separation of the exploited region and GBK necessitates additional steps in making comparisons of reference points to whole stock conditions. Biomass reference points were always calculated for the entire stock area based on KLAMZ estimates for1978 biomass for the exploited region and for Georges Bank at the appropriate level of M. Therefore projected values of 2015 biomass for the exploited stock area plus the estimated biomass in 2008 on GBK were compared to biomass reference points so that biomass comparisons were whole stock biomass to whole stock reference point.

Managers currently compare fishing mortality reference points to fishing mortality for the exploited stock area only. They may choose, however, to compare mortality reference points to fishing mortality for the whole stock. Projected fishing mortality rates for the entire stock were

calculated from estimates for the exploited stock only by solving the catch equation for whole stock F using catch $C = \frac{F^x}{F^x + M} B^x \left(1 - e^{F^x + M}\right)$, whole stock biomass $B = B^x + B^{GBK}$ and the assumed true value of M. In these equations F is the fishing mortality estimate for the whole stock in 2015, F^x and B^x are projected estimates for the exploited stock in 2015, and B^{GBK} is the estimated biomass from the KLAMZ model for GBK during 2008. The estimates F^x, B^x and B_{GBK} were from KLAMZ models that used the value of M assumed true under the state of nature.

Vulnerability to overfishing

Ocean quahogs are an unproductive stock that is vulnerable to overfishing. If overfished (depleted biomass) conditions occur, one or more decades will be required to rebuild the stock. Current fishing mortality rates are roughly 0.01 y^{-1} for the exploited area and roughly 0.005 y^{-1} for the stock as a whole (Figure B73). In contrast, the recommended fishing mortality threshold is $F_{45\%}$=0.0219 y^{-1}. The recommended mortality threshold was based on simulation analyses for west coast groundfish and may not be appropriate for ocean quahogs, which are probably less productive than the longest-lived west coast groundfish. Traditional southern fishing grounds in the DMV and NJ regions declined after 1990 to less than ½ of their unfished biomass (Figure B72) while fishing mortality averaged about 0.01 y^{-1} (Figure B73).

Productivity (due to somatic growth and recruitment) is higher in the north (LI, SNE and GBK) but very low in the south (DMV and NJ). Recruitment to the stock as a whole declined from about 48 thousand mt y y^{-1} before 1993 to about 17 thousand mt y^{-1} after 1993 (Figure B77). Most of the recruitment during 2005 was on GBK where a relatively strong year class is reaching fishable size. A strong but very regional recruitment event in SNE southwest of Cape Cod is expected to reach fishable size over the next decade.

Projection analyses indicate that ocean quahog biomass will decline very slowly during 2010-2015 under most of the harvest rates considered in projections (Figure B79). However, there is appreciable probability of $F_{2015} > F_{45\%}$ in the exploited stock if landings during 2010-2015 are at the current quota or maximum quota levels specified in the FMP (Table B27). Fishing mortality rates for the entire stock in 2015 are unlikely to exceed F45% under any harvest policy (Table B27).

Research Recommendations
(TOR-6)

Recommendations from the previous assessment and recommendations for future research are described below.

Recommendations from last assessment (SAW 44)

1) The *R/V Delaware II* may not be available for use on NEFSC clam surveys after 2008, and it appears likely that the clam survey will become a cooperative effort with sampling done by a commercial vessel. Both the *R/V Delaware II* and a commercial vessel should be used during 2008 so that catch rates, efficiency and selectivity patterns for the two vessels can be compared and calibrated. Planning should commence immediately.
Completed. See cruise report from F/V Endeavor in Appendix B4.

2) Fishing mortality and biomass reference points used as proxies for *FMSY* and *BMSY* should be

reevaluated in the next assessment.
Completed. Several proxy reference points were evaluated in the present assessment.

3) Additional estimates of survey dredge efficiency from cooperative depletion studies are required.
Completed. Three additional depletion studies were conducted in 2008.

4) Develop a length (and possibly age) structured stock assessment model for ocean quahogs that makes better use of survey and fishery length composition data which may provide better estimates of recruitment trends.
Not attempted in the present assessment.

5) Conduct further experimental work to determine the relationship between dredge efficiency, depth, substrate and clam density. A comprehensive study coincident with the next NEFSC clam survey would be most useful. The experimental design should include sufficient contrast in variables that may affect dredge efficiency.
Completed. The relationships were evaluated and no obvious relationship was detected at this time.

6) Cover GBK in the next NEFSC clam survey.
Completed. A full survey was conducted in this region in 2008.

7) Investigate the survey data from GBK during the 1989 survey to determine why it is low relative to survey observations during earlier years. This may be important in determining if biomass is increasing in GBK.
This is no longer an important issue.

8) Survey strata with no tows are a particular problem in the GBK region. The current procedure for filling holes in survey data involves borrowing data from adjacent surveys. This may not be optimal for ocean quahog surveys and GBK in particular. In the next assessment, consider filling holes in the GBK survey data using a model with stratum and year effects.
Not attempted due primarily to limited time. The current approach was considered adequate for ocean quahogs that have slow population dynamics, and was continued in the present assessment. Years when borrowing was substantial (e.g. 1989, 2002 and 2005) were excluded from the KLAMZ model of GBK.

9) Evaluate possible increasing trends in biomass for ocean quahog on GBK.
Completed. This was evaluated directly in the KLAMZ model.

10) Evaluate effects and contribution of recruitment to stock productivity.
Completed. This was evaluated directly in the KLAMZ model.

11) Improve estimates of biological parameters for age, growth (particularly of small individuals), and maturity for ocean quahog in both the EEZ and in Maine waters.
Not attempted. No new estimates of the biological parameters were obtained in the present assessment.

12) Survey dredge and commercial dredge efficiency estimates should be reevaluated by field work

during the next NEFSC clam survey. The next survey may be the last opportunity to estimate survey dredge selectivity. The commercial dredge selectivity curve was used in this assessment was estimated from field studies done off Iceland (Thorarinsdottir and Jacobson, 2005) where conditions may differ. Repeat tow experiments (i.e. survey stations reoccupied by commercial vessels) may be useful for this purpose.
Completed in part. Efficiency comparisons were conducted but there were no selectivity studies for the commercial dredge for ocean quahogs.

13) In the next assessment, projection calculations should be carried out using a model that is basically the same as the primary stock assessment model used to estimate biomass and fishing mortality (e.g. delay-difference population model in KLAMZ).
Completed. The projection model uses the same equations as the KLAMZ model in addition to a simple deterministic approach.

14) Recommendations for future depletion studies:

- It was difficult to find areas with high concentrations of ocean quahog for depletion experiment sites during 2005. However, areas with lower densities of ocean quahog can be used if depletion tow distance is increased.
Completed. The 2008 survey design included areas of lower densities for the depletion studies,

- Revised estimators for survey dredge efficiency based on commercial depletion experiments and setup tows use data for relatively large ocean quahogs (i.e. 90+ mm) only. Future depletion sites should contain reasonably high densities of large individuals.
Completed. The 2008 survey design included areas of high densities of >90mm ocean quahogs.

- In the future, every effort must be made to collect and record precise location data at short time intervals during depletion studies.
Completed. Location data were collected at a time interval of <= 5 seconds in the 2008 depletion studies.

- Collect length and bushel count data from survey and depletion tows more frequently (e.g. every 1-2 tows). It might be advantageous to measure fewer individuals sampled from more tows.
This change was not implemented in the 2008 depletion studies because the existing protocol was considered adequate.

- Analyze results from previous depletion studies to determine if differences between bushel counts and length composition data from different tows in the same depletion experiment are significantly different. Use the results to modify sampling protocols as appropriate.
No detailed analyses were attempted.

- Changes in length composition during a depletion experiment might be incorporated into efficiency estimation by, for example, including selectivity parameters in the Patch model. Efficiency estimates (and commercial selectivity) might be more precise because more size groups would be included in catch data.
This was not attempted in the present assessment but it would be useful to conduct this analysis in

the future.

- It would be useful to analyze efficiency estimates in terms of season because ocean quahogs are believed to change their depth in sediments on a seasonal basis.
This was not attempted in the present assessment but it would be useful to conduct this analysis in the future

15) The next stock assessment should review the M=0.02 y^{-1} assumption for ocean quahog.
Not completed although projection and reference point calculations considered a range of M values.

16) In the next assessment, KLAMZ model runs with two recruitment parameters should be explored for LI and SNE. Survey length composition show more recruitment prior to 1994 than afterwards. Model fit was not as good for SNE as other regions.
Completed. The present assessment incorporated two recruitment parameters for these regions and for the exploited stock as a whole.

17) KLAMZ model runs for GBK should be explored further in the next assessment.
Completed.

New Recommendations (in rough order of priority)
1) The next survey should be conducted by a commercial vessel that is more efficient in sampling ocean quahogs compared to *R/V Delaware II.* The pilot program and analysis of existing cooperative survey data suggest that the data collected by a commercial vessel will be more precise and easier to interpret compared to data collected by the existing clam survey. A considerable amount of planning and preparation for this transition has already occurred. The survey should commence immediately in 2010 on a 15 days at sea per year schedule.

2) The 2011 survey should be of sufficient length, including anticipated down time, to cover all of the regions from Delmarva through Georges Bank.

3) Carry out simulations to determine optimum proxies for F_{MSY} and B_{MSY} in ocean quahogs, given their unusual biological characteristics.

4) The survey sensor package (SSP) should be modified so that y-tilt sensors are situated to better measure y-tilt at shallow angles; it is not important to measure y-tilt accurately at steep angles. Consider using a sensor not prone to vibration and resonance effects.

5) The SSP equipment should be redesigned and battery life extended for greater reliability and use on commercial dredges. Backup sensors should be improved as well and used routinely.

6) Estimate relationships between size and number of eggs produced. Determine spawning frequency if possible.

7) Additional age and growth studies are required to determine if extreme longevity (e.g. 400 y) is typical or unusual and to refine estimates of natural mortality. Similarly, additional age and growth studies over proper geographic scales could be used to investigate temporal and spatial recruitment

patterns.

8) Better information about maturity at length is required.

9) There has been progress in improving port sampling for ocean quahogs since the last assessment and efforts in this direction should continue, particularly as the distribution of the fishery shifts and if a fishery develops on Georges Bank.

10) Commercial dredge selectivity estimates should be obtained for the next assessment.

11) Improve estimates of biological parameters for age, growth (particularly of small individuals), and maturity for ocean quahog in both the EEZ and in Maine waters.

12) Additional estimates of survey dredge efficiency from cooperative depletion studies are required.

13) Develop a length (and possibly age) structured stock assessment model for ocean quahog that makes better use of survey and fishery length composition data which may provide better estimates of recruitment trends.

14) Conduct further analyses to determine the relationship between dredge efficiency, depth, substrate, and clam density.

15) Changes in length composition during a depletion experiment might be incorporated into efficiency estimation by, for example, including selectivity parameters in the Patch model. Efficiency estimates (and commercial selectivity) might be more precise because more size groups would be included in catch data.
16) It would be useful to analyze efficiency estimates in terms of season because ocean quahog are believed to change their depth in sediments on a seasonal basis.

17) Investigate model formulations that accommodate spatial heterogeneity.

18) Examine existing underwater photographs of ocean quahogs to evaluate the potential use of HABCAM or other optical surveys for surveying ocean quahogs and for measuring their habitat.

19) Further analysis of commercial vessel performance in making standardized tows would be advantageous to supplement work already completed.

20) Regions used in a future cooperative surveys should be spatially distinct (non-overlapping) and sensible with respect to fishery patterns, management requirements and the biological distribution of the animals. It is important that the spatial resolution of the catch and port sampling data are adequate for use with the new survey regions. The survey should cover the entire habitat area. It may be advisable to break SNE into two portions, one associated with biological patterns on GBK and the other associated with LI.

21) It may be advantageous to use survey strata that are appropriate for ocean quahogs and surfclams per se, rather than for all shellfish including scallops and other shellfish.

22) Presentation of results for SVA complicates the assessment and this area should be dropped or combined with DMV in the next assessment.

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TABLES

B. Stock assessment for ocean quahogs (*Arctica islandica*)

Invertebrate Subcommittee
SAW/SARC 48

October 5, 2009

Table B1. Annual landings and quotas (1000 metric tons meats) for ocean quahog from state waters (including Maine) and from the Exclusive Economic Zone (EEZ, state waters excluded). EEZ landings are from logbooks. Landings from state waters are not used in this assessment unless stated otherwise.

Year	Dealer Database	EEZ (Logbook)	State Waters (Logbook - Dealer)	Percent Landings in EEZ	EEZ Quota	EEZ Landings / Quota (%)
1967[a]	0.020		0.020			
1968	0.102		0.102			
1969	0.290		0.290			
1970	0.792		0.792			
1971	0.921		0.921			
1972	0.634		0.634			
1973	0.661		0.661			
1974	0.365		0.365			
1975	0.569		0.569			
1976	2.510	1.854	0.656	0.739		
1977	8.411	7.293	1.118	0.867		
1978	10.415	9.197	1.218	0.883		
1979	15.748	14.344	1.404	0.911	13.608	105%
1980[b,c]	11.623	13.407		1.000	15.876	84%
1981	11.202	13.101		1.000	18.144	72%
1982	16.478	14.234	2.244	0.864	18.144	78%
1983	16.200	14.586	1.614	0.900	18.144	80%
1984	17.939	17.975		1.000	18.144	99%
1985	22.035	20.726	1.309	0.941	22.226	93%
1986	20.585	18.902	1.683	0.918	27.215	69%
1987	22.709	21.514	1.195	0.947	27.215	79%
1988	21.007	20.273	0.734	0.965	27.215	74%
1989	23.147	22.359	0.787	0.966	23.587	95%
1990	21.235	20.965	0.270	0.987	24.040	87%
1991	22.119	22.064	0.055	0.998	24.040	92%
1992	22.871	22.477	0.395	0.983	24.040	93%
1993	24.843	21.876	2.967	0.881	24.494	89%
1994	21.159	20.985	0.173	0.992	24.494	86%
1995	23.253	21.108	2.145	0.908	22.226	95%
1996	21.122	20.061	1.061	0.950	20.185	99%
1997	19.930	19.628	0.301	0.985	19.581	100%
1998	18.098	17.897	0.201	0.989	18.144	99%
1999	17.557	17.381	0.175	0.990	20.412	85%

Table B1. (cont.)

Year	Dealer Database	EEZ (Logbook)	State Waters (Logbook - Dealer)	Percent Landings in EEZ	EEZ Quota	EEZ Landings / Quota (%)
2000	14.899	14.723	0.176	0.988	20.412	72%
2001	17.234	17.069	0.165	0.990	20.412	84%
2002	18.144	17.947	0.197	0.989	20.412	88%
2003	18.997	18.815	0.182	0.990	20.412	92%
2004	17.812	17.655	0.157	0.991	22.680	78%
2005	13.793	13.635	0.158	0.989	24.190	56%
2006	14.461	14.273	0.188	0.987	24.190	59%
2007	15.734	15.574	0.161	0.990	24.190	64%
2008	14.442	15.479		1.000	24.190	64%

[a] Figures for 1967-1979 are from NEFSC (1990)

[b] Figures for 1980-1993 from NEFSC (2003).

[c] For 1980-2005, "Dealer Database Total" landings are from commercial landings

[d] Dealer database total for 2008 may not be complete.

Table B2. Ocean quahog landings (mt meats) by region reported in logbooks for the US EEZ. Figures for 1978-1979 are not from logbooks may be less reliable.

YEAR	SVA	DMV	NJ	LI	SNE	GBK	MNE	UNK	Grand Total
1978		1,290	6,350					2,775	10,415
1979		5,450	6,030					4,268	15,748
1980		4,230	7,750	6				1,421	13,407
1981	56	3,637	8,402	3				1,003	13,101
1982	6	4,598	8,538					1,092	14,234
1983		5,396	8,249	21	629			291	14,586
1984	6	7,171	8,851		822			1,125	17,975
1985	160	7,200	10,676	40	693			1,956	20,726
1986		8,237	9,059	396	562			649	18,902
1987		10,540	9,070	1,180	696			27	21,514
1988	42	11,716	7,015	640	841			20	20,273
1989		6,439	14,100	605	1,196			20	22,359
1990	14	3,685	15,590	739	934		3		20,965
1991		4,839	14,575	1,674	865		110		22,064
1992		2,378	6,942	11,940	1,143		75		22,477
1993		1,953	10,205	8,642	1,020		56		21,876
1994		992	6,938	12,015	954		65	22	20,985
1995		699	5,357	9,527	5,412		114		21,108
1996		736	4,864	5,943	8,350		142	26	20,061
1997		1,072	4,229	5,141	8,968		218		19,628
1998		1,365	2,684	6,856	6,736		218	39	17,897
1999		1,090	3,039	6,329	6,618		279	27	17,381
2000		1,048	3,318	4,745	5,083	49	357	123	14,723
2001		894	4,560	5,692	4,694	13	326	889	17,069
2002		1,732	2,781	9,113	3,884		387	51	17,947
2003		896	3,683	11,626	2,177		359	73	18,815
2004		624	2,761	10,690	3,273		307		17,655
2005		910	669	9,714	2,021		301	19	13,635
2006		494	467	11,101	1,847		365		14,273
2007		100	1,566	11,290	2,311		306		15,574
2008		270	1,733	11,123	2,151		201	0	15,479

[c] All data for 1980-1993 fron NEFSC (2003), all other data from logbooks.

Table B3. Ocean quahog landings by region as reported in logbooks for the US EEZ. Landings (except for Maine) are in thousands of ITQ bushels.

YEAR	SVA	DMV	NJ	LI	SNE	GBK	MNE	MNE (Maine bushels)	UNK	Grand Total
1980		933	1,709	1					313	2,956
1981	12	802	1,852	1					221	2,888
1982	1	1,014	1,882						241	3,138
1983		1,190	1,819	5	139	64			64	3,280
1984	1	1,581	1,951		181	248			248	4,211
1985	35	1,587	2,354	9	153	431			431	5,001
1986		1,816	1,997	87	124	143			143	4,310
1987		2,324	2,000	260	153	6			6	4,749
1988	9	2,583	1,546	141	185	4			4	4,474
1989		1,420	3,108	133	264	4			4	4,934
1990	3	812	3,437	163	206		1	1		4,623
1991		1,067	3,213	369	191		24	37		4,901
1992		524	1,530	2,632	252		16	25		4,980
1993		431	2,250	1,905	225		12	19		4,841
1994		219	1,530	2,649	210	5	14	21	5	4,653
1995		154	1,181	2,100	1,193		25	38		4,691
1996		162	1,072	1,310	1,841	6	31	47	6	4,476
1997		236	932	1,133	1,977		48	73		4,400
1998		301	592	1,511	1,485	9	48	72	9	4,026
1999		240	670	1,395	1,459	6	62	93	6	3,931
2000		231	732	1,046	1,121	27	79	119	27	3,381
2001		197	1,005	1,255	1,035	196	72	109	196	4,065
2002		382	613	2,009	856	11	85	129	11	4,097
2003		198	812	2,563	480	16	79	120	16	4,284
2004		138	609	2,357	722		68	102		3,994
2005		201	148	2,142	446	4	66	100	4	3,110
2006		109	103	2,447	407		80	121		3,268
2007		22	345	2,489	510		68	102		3,535
2008		59	382	2,452	474	0	44	67	0	3,479

[c] All data for 1980-1993 are landings in NEFSC (2003) / 220.463.

Table B4. Real and nominal prices for ocean quahog based on dealer data. Average price was computed as total revenues divided by total landed meat weight during each year, rather than as annual averages of prices for individual trips, to reduce bias due to small deliveries at relatively high prices. The consumer price index (CPI) used to convert nominal dollars to 1991 equivalent dollars is for unprocessed and packaged fish, which includes shellfish and finfish (Eric Thunberg, NEFSC, pers. comm.).

Year	CPI	Excluding Maine			Maine only		
		Nominal ($/lb)	Real price (1991 $/lb)	Real price (1991 $/ITQ bu)	Nominal ($/lb)	Real price (1991 $/lb)	Real price (1991 $/Maine bu)
1982	0.67	0.31	0.46	4.58	NA	NA	NA
1983	0.71	0.31	0.43	4.33	NA	NA	NA
1984	0.75	0.31	0.41	4.06	0.78	1.03	6.83
1985	0.77	0.31	0.40	4.00	NA	NA	NA
1986	0.84	0.30	0.36	3.62	1.75	2.10	13.88
1987	0.94	0.29	0.31	3.09	2.30	2.46	16.27
1988	0.99	0.29	0.29	2.90	1.90	1.91	12.64
1989	0.96	0.29	0.31	3.06	2.72	2.85	18.86
1990	0.98	0.32	0.32	3.23	2.70	2.75	18.19
1991	1.00	0.34	0.34	3.39	4.10	4.10	27.15
1992	1.04	0.36	0.34	3.40	4.07	3.90	25.80
1993	1.05	0.40	0.38	3.82	3.58	3.42	22.62
1994	1.08	0.38	0.36	3.57	3.83	3.55	23.49
1995	1.14	0.40	0.35	3.52	3.46	3.02	20.03
1996	1.11	0.41	0.37	3.74	3.10	2.79	18.50
1997	1.19	0.42	0.35	3.49	2.62	2.20	14.58
1998	1.23	0.42	0.34	3.45	2.50	2.04	13.52
1999	1.28	0.42	0.33	3.30	2.75	2.16	14.28
2000	1.33	0.43	0.33	3.26	2.74	2.07	13.69
2001	1.28	0.55	0.43	4.32	3.23	2.53	16.77
2002	1.28	0.54	0.42	4.19	3.69	2.88	19.10
2003	1.31	0.53	0.41	4.05	3.75	2.87	19.03
2004	1.38	0.52	0.38	3.75	3.79	2.75	18.20
2005	1.49	0.51	0.34	3.41	3.60	2.42	16.02
2006	1.59	0.51	0.32	3.18	3.23	2.03	13.47
2007	1.62	0.52	0.32	3.18	3.16	1.95	12.90
2008	1.71	0.54	0.32	3.16	3.29	1.93	12.77

Table B5. Ocean quahog fishing effort (hours fished) by region in the US EEZ based on logbook data. “Sub-trips" (deliveries from the same trip to different dealers) are counted only once.

YEAR	SVA	DMV	NJ	LI	SNE	GBK	MNE	UNK	Grand Total
1983		7,131	13,937	50	1,535			56	22,709
1984	15	11,106	15,477		2,523			1,231	30,352
1985	204	10,058	17,890	87	2,066			2,955	33,260
1986		12,260	14,360	361	1,138			1,012	29,130
1987		15,818	14,698	806	1,340			49	32,711
1988	64	19,100	11,598	615	1,639			64	33,079
1989		12,124	24,262	797	2,327			50	39,560
1990	25	8,166	29,327	1,283	1,838		286		40,924
1991		12,048	30,397	1,844	1,433		17,110		62,832
1992		5,513	15,998	13,148	1,964		13,424		50,047
1993		4,622	25,457	12,883	1,783		5,720		50,465
1994		2,260	20,543	19,165	2,082		5,056	57	49,162
1995		1,621	13,598	16,015	8,561		5,731		45,526
1996		1,521	9,340	10,239	11,866		8,404	54	41,423
1997		2,742	9,382	8,295	13,515		11,734		45,669
1998		3,225	6,983	10,509	10,639		11,631	79	43,066
1999		2,595	7,623	9,132	12,258		10,821	90	42,518
2000		2,517	7,966	7,071	10,542	63	12,215	612	40,986
2001		2,170	10,844	7,813	11,404	22	13,113	1,454	46,820
2002		4,290	6,683	11,605	7,797		16,779	85	47,240
2003		2,617	10,750	16,113	4,596		17,832	108	52,016
2004		2,495	7,905	14,582	6,642		19,014		50,638
2005		3,445	1,972	12,519	4,043		16,905	45	38,928
2006		1,811	1,386	14,542	3,314		14,638		35,691
2007		346	3,719	15,618	4,286		13,821		37,791
2008		956	4,768	14,980	3,965		10,734	11	35,414

Table B6. Ocean quahog landings per unit effort (LPUE, total bushels / total hours fished) based on logbook data for all vessels operating in the US EEZ.

YEAR	DMV	NJ	LI	SNE	MNE	Total ITQ
1983	131	123	26			130.16
1984	72	120				95.16
1985	101	105				94.35
1986	97	127	13	122		112.59
1987	100	133		135		129.86
1988	83	203	14	93		313.14
1989	150	82	109	53		164.92
1990	285	68	203	84		134.90
1991	214	51	77	129		77.43
1992	257	194	10	134		111.33
1993	176	135	13	115		109.89
1994	472	156	19	92		130.29
1995	323	113	164	29		146.44
1996	283	241	186	19	0.08	157.81
1997	80	163	319	16	1.21	138.65
1998	48	169	200	112	2.16	155.79
1999	63	141	143	150	2.89	172.67
2000	94	117	160	188	3.94	187.95
2001	139	55	193	130	3.66	143.08
2002	56	100	120	187	3.67	127.55
2003	88	68	65	244	4.41	88.34
2004	79	127	86	156	3.78	108.45
2005	111	311	160	212	5.04	142.28
2006	109	586	176	145	5.41	117.81
2007	398	164	151	168	4.90	103.91
2008	210	31	143	112	6.18	85.95

Table B7. Number of quahogs measured, trips sampled, percentage of trips sampled, and the number quahogs measured per bushel landed by year and region, from port samples.

Region	Year	Quahogs sampled	Trips sampled	% of trips sampled	Samples per bushel landed
SNE	1996	30	1	0.12	0.00002
	1997	310	10	1.20	0.00016
	1998	796	25	3.88	0.00054
	1999	634	21	2.67	0.00043
	2000	822	27	4.12	0.00073
	2001	761	25	3.84	0.00074
	2002	1353	42	7.18	0.00158
	2003	606	20	6.31	0.00126
	2004	1302	43	10.39	0.00180
	2005	1280	42	14.58	0.00287
	2006	996	32	12.45	0.00245
	2007	1282	42	14.84	0.00252
	2008	2406	80	34.19	0.00507
Region	**Year**	**Quahogs sampled**	**Trips sampled**	**% of trips sampled**	**Samples per bushel landed**
LI	1996	30	1	0.12	0.00002
	1997	1012	32	5.02	0.00089
	1998	480	16	2.28	0.00032
	1999	1440	48	7.12	0.00103
	2000	390	13	2.63	0.00037
	2001	180	6	1.05	0.00014
	2002	150	5	0.63	0.00007
	2003	990	33	3.26	0.00039
	2004	360	12	1.37	0.00015
	2005	1866	62	9.00	0.00087
	2006	2928	98	12.68	0.00120
	2007	2099	68	8.58	0.00084
	2008	2482	81	11.81	0.00101
Region	**Year**	**Quahogs sampled**	**Trips sampled**	**% of trips sampled**	**Samples per bushel landed**
NJ	1996	30	1	0.14	0.00003
	1997	390	13	2.03	0.00042
	1998	420	14	3.47	0.00071
	1999	420	14	3.13	0.00063
	2000	600	20	4.13	0.00082
	2001	780	26	3.99	0.00078
	2002	510	17	4.59	0.00083
	2003	390	13	2.68	0.00048
	2004	1080	36	9.92	0.00177
	2005	90	3	3.23	0.00061
	2006	243	8	11.59	0.00236
	2007	343	11	6.04	0.00099
	2008	330	11	4.74	0.00086
Region	**Year**	**Quahogs sampled**	**Trips sampled**	**% of trips sampled**	**Samples per bushel landed**
DMV	1996	180	6	5.08	0.00111
	1997	570	19	10.86	0.00241
	1998	390	13	6.70	0.00130
	1999	960	32	19.39	0.00399
	2000	690	23	14.65	0.00299
	2001	660	22	18.64	0.00335
	2002	120	4	1.78	0.00031
	2003	390	13	10.66	0.00197
	2004	150	5	4.46	0.00109
	2005	511	17	12.32	0.00255
	2006	743	24	29.63	0.00683
	2007	195	6	42.86	0.00887
	2008	120	4	10.00	0.00202

Table B8. Number of random and nearly random NEFSC survey tows used to estimate trends in abundance of ocean quahog. Figures in each cell are the number of tows in calculations for each combination of stratum and cruise. Figures in plain text are the number of original tows (without borrowing). Bold and outlined figures are for cells that had zero tows originally but were filled by borrowing tows from the same strata during previous and/or subsequent cruises. Black cells are for cells with zero tows that could not be filled by borrowing. Survey/region combinations with relatively poor sampling (a relatively large number or relatively large strata) are shown in grey.

Region	Stratum	Area (nm2)	%Total Stratum Area	Survey Year											
				1982	1983	1984	1986	1989	1992	1994	1997	1999	2002	2005	2008
SVA	5	690	0.97	4	9	13	8	8	8	8	8	**16**	8	8	**8**
	6	22	0.03	1	1	1	1	1	1	1	1	**3**	2	1	**1**
DMV	9	1894	0.47	30	26	35	29	37	37	39	39	38	39	39	31
	10	190	0.05	2	2	3	3	3	3	3	3	3	3	3	2
	11	246	0.06	2	2	**4**	2	2	2	2	2	2	2	2	**2**
	13	1149	0.28	19	18	25	20	20	20	21	22	19	20	20	15
	14	205	0.05	2	2	3	3	3	3	5	3	3	3	3	**3**
	15	387	0.10	4	4	**8**	4	4	4	5	4	5	4	4	**4**
NJ	17	703	0.11	11	11	18	12	12	12	12	14	12	12	12	12
	18	240	0.04	3	3	**6**	3	3	3	3	3	3	3	3	3
	19	266	0.04	3	3	**6**	3	3	3	3	3	3	3	3	3
	21	1693	0.26	18	18	22	19	20	20	23	26	39	29	29	28
	22	305	0.05	3	3	**6**	3	3	3	5	3	3	3	3	3
	23	724	0.11	7	6	**11**	5	4	5	5	5	5	5	5	5
	25	647	0.10	9	9	13	8	9	9	9	12	8	9	9	13
	26	190	0.03	2	2	**5**	3	3	3	3	3	3	3	3	3
	27	442	0.07	4	4	**8**	4	4	4	4	4	4	4	4	4
	87	356	0.05	8	7	10	9	9	9	9	9	9	16	16	9
	88	484	0.07	15	15	24	17	20	20	20	21	22	20	20	19
	89	343	0.05	15	15	21	15	18	17	17	19	18	18	18	18
	90	117	0.02	2	2	3	2	2	2	2	2	2	2	2	1

Table B8. (cont.)

Region	Stratum	Area (nm2)	%Total Stratum Area	Survey Year											
				1982	1983	1984	1986	1989	1992	1994	1997	1999	2002	2005	2008
LI	29	1078	0.24	11	10	20	10	10	10	10	10	11	10	10	16
	30	667	0.15	7	8	14	6	6	6	6	6	7	6	6	12
	31	932	0.21	9	7	12	5	7	8	8	8	9	8	8	8
	33	361	0.08	4	4	8	4	4	4	5	4	4	4	4	10
	34	207	0.05	2	2	4	2	2	2	5	2	2	2	2	8
	35	614	0.14	4	2	4	2	5	6	6	6	6	6	6	6
	91	342	0.08	3	2	4	4	3	3	3	3	3	3	3	5
	92	165	0.04	2	2	3	2	2	2	2	2	2	2	2	5
	93	97	0.02	1	1	2	1	1	1	1	1	1	2	2	4
SNE	37	660	0.13	7	4	7	3	6	3	5	4	4	3	3	
	38	268	0.05	3	2	5	3	3	3	5	3	3	3	3	3
	39	946	0.19	6	4	6	2	5	5	5	5	5	5	5	5
	41	580	0.12	6	5	7	5	6	6	6	6	5	6	6	6
	45	407	0.08	3	7	9	4	4	4	4	4	4	3	3	4
	46	205	0.04	2	5	5	3	2	3	5	3	3	2	2	3
	47	873	0.18	4	3	4	2	2	4	5	4	3	1	1	4
	94	215	0.04	1	2	2		1	1	2	2	4	2	2	
	95	278	0.06	4	14	11	4	4	4	4	4	4	4	8	4
	96	490	0.10	12	12	13	1	1	3	2	4	4		1	1

Table B8. (cont.)

Region	Stratum	Area (nm2)	%Total Stratum Area	Survey Year											
				1982	1983	1984	1986	1989	1992	1994	1997	1999	2002	2005	2008
GBK	54	295	0.04		3	3	3	6	3	3	3	3		2	2
	55	386	0.05	3	3	3	3	1	3	3	3	2	2	4	2
	56	214	0.03								4	4	4		
	57	176	0.02			2	2	1	2	5	2	2	2	4	2
	58	303	0.04								5	5	5		
	59	512	0.07		4	5	1	2	6	5	5	4	5	9	4
	60	801	0.10			2	2	2	4	2	5	5	5	9	4
	61	588	0.08	8	1	6	5	12	7	6	6	6	6	11	5
	62	731	0.09			1	1	1	4	4	4	4	4	7	3
	65	184	0.02			3	3	5	2	2	3	4	1	1	
	67	196	0.03		5	5	5	7	7	7	7	7		2	2
	68	380	0.05	1	8	7	3	6	6	5	5	5		6	6
	69	902	0.12	2	5	11	6	6	6	7	6	7	7	4	4
	70	544	0.07	1	2	6	4	8	4	4	4	3	2	6	4
	71	168	0.02		2	2	3	1	2	3	3	1	2	3	1
	72	472	0.06	2	10	8	1	8	8	8	8	6	6	4	4
	73	526	0.07	1	1	4	3	6	6	6	6	5	6	9	3
	74	443	0.06	3	4	1	3	7	4	4	4	3	3	6	3

Table B9. Parameter estimates for the relationship between shell length (L, mm) and meat weight (W, g) in ocean quahog (same as in NEFSC 2004). The equation for the relationship is $W=e^{\alpha}L^{\beta}$.

Region	Alpha	Beta
SVA	-9.042313	2.787987
DMV	-9.042313	2.787987
NJ	-9.847183	2.94954
LI	-9.233646	2.822474
SNE	-9.124283	2.774989
GBK	-8.969073	2.767282

Table B10. Trends in survey, stock and fishable abundance and biomass for ocean quahog ≥ 50 mm SL during 1982-2008 based on NEFSC clam survey data. Figures include original plus borrowed tows. "Number Strata" for a particular year includes strata sampled by the survey during the same year plus strata sampled by tows borrowed from the previous and subsequent surveys. Survey data for 1994 should be ignored because of gear problems that artificially boosted sampling efficiency. Survey coverage was incomplete on GBK prior to 1986 and 2005.

		survey				stock				fishable						
region	year	N/tow	CV	Kg/tow	CV	N/tow	CV	Kg/tow	CV	N/tow	CV	Kg/tow	CV	tows per region	positive tows	strata surveyed in region
GBK	1986	278.06	0.19	6.99	0.18	430.11	0.23	9.66	0.19	233.54	0.19	5.99	0.18	47	21	16
GBK	1989	92.29	0.26	2.72	0.25	126.71	0.24	3.37	0.25	80.19	0.26	2.41	0.25	78	38	16
GBK	1992	346.25	0.21	10.44	0.21	485.71	0.19	12.86	0.20	302.84	0.21	9.30	0.21	74	41	16
GBK	1994	405.23	0.20	12.34	0.20	578.46	0.19	15.22	0.19	355.56	0.20	11.03	0.20	76	40	16
GBK	1997	269.76	0.19	7.99	0.19	389.38	0.19	10.08	0.18	234.25	0.19	7.11	0.19	83	44	18
GBK	1999	273.40	0.17	8.88	0.19	365.97	0.16	10.63	0.18	241.90	0.17	8.04	0.19	77	47	18
GBK	2002	328.37	0.18	10.29	0.19	478.14	0.15	12.68	0.18	288.96	0.18	9.26	0.19	61	38	15
GBK	2008	323.77	0.30	7.09	0.28	693.48	0.31	12.01	0.29	265.74	0.29	6.03	0.27	49	30	15
SNE	1982	277.61	0.27	9.41	0.25	345.84	0.28	11.07	0.26	245.46	0.27	8.47	0.25	48	30	10
SNE	1983	173.21	0.29	5.61	0.30	237.69	0.31	6.92	0.29	151.40	0.29	5.02	0.30	58	37	10
SNE	1984	188.46	0.27	6.40	0.29	234.35	0.26	7.52	0.28	166.80	0.27	5.77	0.29	69	38	10
SNE	1986	289.15	0.31	9.37	0.31	394.36	0.35	11.51	0.32	253.12	0.31	8.39	0.31	27	23	9
SNE	1989	274.66	0.19	9.03	0.18	353.18	0.21	10.83	0.19	241.36	0.19	8.09	0.18	34	29	10
SNE	1992	333.08	0.19	11.64	0.19	400.10	0.19	13.40	0.19	297.00	0.19	10.53	0.20	36	31	10
SNE	1994	529.09	0.22	18.12	0.20	670.13	0.25	21.44	0.21	467.48	0.22	16.37	0.20	43	32	10
SNE	1997	292.89	0.54	8.23	0.45	447.96	0.61	11.27	0.51	246.94	0.52	7.17	0.43	39	27	10
SNE	1999	252.43	0.54	8.31	0.48	312.91	0.56	9.84	0.51	221.84	0.53	7.42	0.47	39	30	10
SNE	2002	180.67	0.22	6.89	0.22	206.74	0.22	7.64	0.22	164.25	0.22	6.34	0.22	29	28	9
SNE	2005	157.78	0.26	4.81	0.23	333.78	0.42	6.93	0.27	137.54	0.25	4.33	0.22	40	34	10
SNE	2008	201.41	0.25	5.48	0.22	523.90	0.42	9.07	0.27	172.65	0.24	4.88	0.22	37	31	8

Table B10. (cont.)

		Survey				Stock				Fishable						
Region	Year	N/tow	CV	Kg/tow	CV	N/tow	CV	Kg/tow	CV	N/tow	CV	Kg/tow	CV	N tows	N positive tows	N strata surveyed
LI	1982	277.91	0.15	6.98	0.16	433.99	0.16	9.29	0.15	238.75	0.15	6.22	0.16	42	36	9
LI	1983	185.88	0.21	5.23	0.21	253.51	0.22	6.36	0.21	163.62	0.21	4.74	0.21	38	36	9
LI	1984	239.24	0.17	6.67	0.16	323.92	0.17	8.11	0.16	210.02	0.17	6.03	0.16	71	63	9
LI	1986	319.60	0.22	8.89	0.20	426.26	0.22	10.78	0.21	280.44	0.21	8.02	0.20	36	31	9
LI	1989	226.21	0.34	5.06	0.29	367.49	0.38	7.15	0.33	190.10	0.33	4.38	0.28	40	36	9
LI	1992	323.33	0.18	8.31	0.16	465.23	0.20	10.62	0.17	279.03	0.17	7.40	0.16	42	36	9
LI	1994	592.57	0.16	15.35	0.16	827.85	0.17	19.30	0.16	513.28	0.16	13.66	0.16	46	44	9
LI	1997	401.64	0.16	11.16	0.16	518.85	0.17	13.35	0.16	353.15	0.16	10.05	0.16	42	35	9
LI	1999	232.27	0.17	6.28	0.15	310.52	0.19	7.67	0.16	202.72	0.17	5.63	0.14	45	41	9
LI	2002	253.06	0.21	6.97	0.20	330.41	0.21	8.39	0.20	222.21	0.21	6.27	0.20	43	40	9
LI	2005	149.38	0.19	4.07	0.19	215.78	0.19	5.06	0.18	131.16	0.19	3.68	0.20	45	39	9
LI	2008	155.33	0.16	4.55	0.15	206.67	0.19	5.41	0.16	137.71	0.16	4.14	0.15	74	66	9
NJ	1982	112.34	0.20	5.09	0.20	129.33	0.20	5.61	0.20	102.55	0.20	4.73	0.20	99	50	13
NJ	1983	86.09	0.21	4.05	0.21	98.42	0.21	4.42	0.21	79.20	0.21	3.79	0.21	98	55	13
NJ	1984	147.61	0.24	6.69	0.24	170.30	0.24	7.37	0.24	134.86	0.24	6.21	0.24	151	79	13
NJ	1986	144.02	0.23	7.03	0.22	159.78	0.24	7.56	0.22	133.62	0.23	6.61	0.22	103	52	13
NJ	1989	72.24	0.22	3.10	0.21	88.60	0.22	3.51	0.21	65.22	0.22	2.85	0.21	109	52	13
NJ	1992	88.04	0.18	4.33	0.17	97.82	0.18	4.65	0.17	81.73	0.18	4.07	0.17	110	52	13
NJ	1994	235.41	0.22	10.90	0.21	269.04	0.22	11.92	0.21	216.05	0.22	10.16	0.20	115	59	13
NJ	1997	122.26	0.15	6.11	0.15	135.78	0.16	6.55	0.15	113.72	0.15	5.76	0.15	124	59	13
NJ	1999	59.48	0.15	2.89	0.14	72.27	0.15	3.18	0.14	54.89	0.15	2.72	0.14	132	61	13
NJ	2002	89.79	0.23	4.62	0.24	101.12	0.22	4.94	0.23	83.82	0.24	4.38	0.24	127	60	13
NJ	2005	47.08	0.16	2.24	0.15	62.36	0.15	2.53	0.15	43.12	0.15	2.11	0.14	103	54	13
NJ	2008	45.15	0.17	2.14	0.16	60.59	0.17	2.43	0.16	41.27	0.17	2.01	0.16	121	65	13

Table B10. (cont.)

		survey				stock				fishable						
Region	Year	N/tow	CV	Kg/tow	CV	N/tow	CV	Kg/tow	CV	N/tow	CV	Kg/tow	CV	N tows	N positive tows	N strata surveyed
DMV	1982	79.16	0.32	2.96	0.34	86.64	0.31	3.16	0.33	73.84	0.32	2.79	0.34	59	24	6
DMV	1983	86.23	0.49	2.55	0.42	106.61	0.52	2.99	0.45	76.16	0.48	2.30	0.41	54	28	6
DMV	1984	52.01	0.35	1.67	0.30	63.19	0.36	1.90	0.31	46.65	0.34	1.53	0.30	78	34	6
DMV	1986	75.68	0.23	2.53	0.22	86.74	0.24	2.80	0.22	68.94	0.23	2.34	0.22	61	28	6
DMV	1989	64.35	0.58	1.80	0.46	82.47	0.62	2.18	0.51	55.95	0.55	1.61	0.44	69	31	6
DMV	1992	71.98	0.36	2.29	0.31	85.41	0.40	2.59	0.33	64.68	0.35	2.09	0.30	69	25	6
DMV	1994	39.46	0.25	1.33	0.23	47.97	0.27	1.49	0.24	35.89	0.25	1.23	0.23	75	28	6
DMV	1997	47.74	0.21	1.67	0.21	56.44	0.22	1.85	0.21	43.72	0.21	1.56	0.21	73	28	6
DMV	1999	28.36	0.29	0.95	0.27	33.39	0.29	1.06	0.27	25.82	0.29	0.88	0.26	70	23	6
DMV	2002	31.81	0.25	1.11	0.23	38.77	0.26	1.23	0.23	29.14	0.24	1.03	0.22	71	19	6
DMV	2005	19.41	0.49	0.69	0.53	24.84	0.45	0.78	0.50	17.91	0.50	0.65	0.53	66	21	6
DMV	2008	17.76	0.54	0.62	0.59	22.61	0.49	0.70	0.56	16.34	0.55	0.58	0.59	57	16	6
SVA	1982	0.039	0.000	0.002	0.000	0.039	0.000	0.002	0.000	0.038	0.000	0.002	0.000	5	1	2
SVA	1983	1.892	0.578	0.099	0.577	1.916	0.577	0.101	0.577	1.854	0.579	0.097	0.577	10	3	2
SVA	1984	0.189	0.846	0.010	0.870	0.191	0.845	0.010	0.868	0.185	0.848	0.010	0.871	14	2	2
SVA	1986	0.285	0.000	0.013	0.000	0.294	0.000	0.013	0.000	0.275	0.000	0.012	0.000	9	1	2
SVA	1989	0.392	0.000	0.018	0.000	0.401	0.000	0.019	0.000	0.380	0.000	0.018	0.000	9	1	2
SVA	1992	0.000		0.000		0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	9	0	2
SVA	1994	4.467	0.787	0.225	0.807	4.559	0.782	0.229	0.805	4.349	0.790	0.220	0.810	8	2	2
SVA	1997	0.154	0.000	0.004	0.000	0.282	0.000	0.006	0.000	0.132	0.000	0.003	0.000	9	1	2
SVA	1999	0.081	0.551	0.002	0.607	0.182	0.501	0.003	0.541	0.069	0.556	0.002	0.614	19	2	2
SVA	2002	0.045	1.000	0.001	1.000	0.133	1.000	0.002	1.000	0.037	1.000	0.001	1.000	10	1	2
SVA	2005	0.000		0.000		0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	9	0	2
SVA	2008	0.000		0.000		0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	9	0	2

Table B11. Survey abundance trends for small quahogs (1-69 mm SL). Mean numbers per tow (N/Tow) are standardized to a 0.15 nm tow distance based on start and end tow position data. Figures include original plus borrowed tows. "Number Strata" for a particular year includes strata sampled by the survey during the same year plus strata sampled by tows borrowed from the previous and subsequent surveys. Survey data for 1994 should be ignored because of gear problems that artificially boosted sampling efficiency. Survey coverage was incomplete on GBK prior to 1986 and 2005.

Year	SVA		DMV		NJ		LI		SNE		GBK	
	N/tow	CV	N/tow	CV	N/tow	CV	N/tow	CV	N/tow	CV	N/tow	CV
1982	0.00		0.74	0.28	2.01	0.33	68.51	0.23	9.50	0.35	10.83	0.16
1983	0.00		1.77	0.57	2.29	0.52	22.24	0.31	22.67	0.73	12.07	0.39
1984	0.00		1.62	0.47	3.30	0.41	26.50	0.22	7.89	0.35	37.12	0.66
1986	0.00		0.54	0.58	1.99	0.59	30.82	0.28	23.76	0.70	40.73	0.59
1989	0.00		1.07	0.78	3.45	0.36	51.56	0.52	14.17	0.59	7.13	0.31
1992	0.00		0.99	0.63	1.02	0.38	42.30	0.36	5.91	0.35	31.75	0.35
1994	0.03	0.00	1.34	0.55	4.02	0.30	62.43	0.27	30.77	0.61	36.29	0.32
1997	0.04	0.00	1.47	0.53	1.50	0.26	21.81	0.29	58.00	0.80	61.97	0.35
1999	0.03	0.50	0.96	0.49	3.65	0.32	14.11	0.30	6.77	0.75	35.35	0.34
2002	0.02	1.00	1.44	0.48	2.29	0.19	16.08	0.41	2.14	0.42	39.72	0.18
2005	0.00		1.26	0.36	4.05	0.19	19.42	0.36	47.95	0.60	97.92	0.34
2008	0.00		1.10	0.40	4.57	0.20	14.15	0.50	82.74	0.55	150.58	0.37

Table B12. Linear correlations between sensor data summary statistics that dredge performance of individual successful random tows during the 2005 (top, above diagonal) and 2008 (bottom, below diagonal) NEFSC clam surveys. Performance statistics were calculated using data from periods when the dredge was potentially fishing (i.e. between the first and last seconds of each tow when smoothed y-tilt ≤ 5.16°). Sample sizes vary between surveys. However, with the exception of backup y-tilt, samples involved several hundred stations and tens of thousands of sensor measurements at 1 second intervals. Backup y-tilt data for 2008 were from only 8 tows and 2341 sensor measurements. No backup suitable y-tilt data are available for 2005. Correlations with absolute value ≥ 0.5 are shown in bold.

		Tow time	**Proportion time fishing**	**X-tilt**	**SD X-tilt**	**Y-tilt**	**SD Y-tilt**	**Depth**	**Speed over ground**	**Backup y-tilt**	
	Tow time		-0.08	0.04	-0.01	0.07	0.13	***0.65***	-0.25	NA	
	Proportion time fishing	***0.94***		0.18	***-0.64***	***-0.68***	***-0.49***	-0.04	-0.19	NA	
	X-tilt	***0.56***	0.31		0.25	-0.20	0.03	0.12	0.00	NA	
2008 survey	**SD X-tilt**	0.35	0.36	0.31		0.31	***0.54***	0.03	0.08	NA	2005 survey
	Y-tilt	***-0.87***	***-0.79***	***-0.51***	***-0.42***		0.11	0.15	0.34	NA	
	SD Y-tilt	***-0.63***	***-0.76***	0.07	0.13	0.26		0.14	-0.05	NA	
	Depth	-0.08	0.21	***-0.44***	0.12	0.17	-0.35		0.23	NA	
	Speed over ground	***-0.91***	***-0.85***	-0.32	-0.30	***0.82***	***0.59***	0.22		NA	
	Backup y-tilt	***0.87***	***0.81***	***0.54***	***0.55***	***-0.98***	-0.25	-0.05	***-0.77***		

Table B13. Summary of linear correlations for sensor data summary statistics that survey dredge performance in NEFSC clam surveys. Correlations ≥ 0.5 are marked "++". Correlations ≤ -0.5 are marked "- -". No backup y-tilt data were available in 2005.

Variable 1	Variable 2	Survey 2005	2008
Tow time	Proportion time		++
	X-tilt		++
	SD X-tilt		
	Y-tilt		--
	SD Y-tilt		--
	Depth	++	
	Speed over ground		--
	Backup y-tilt	na	++
Proportion time	X-tilt		
	SD X-tilt	--	
	Y-tilt	--	--
	SD Y-tilt	--	--
	Depth		
	Speed over ground		--
	Backup y-tilt	Na	++
X-tilt	SD X-tilt		
	Y-tilt		--
	SD Y-tilt		
	Depth		--
	Speed over ground		
	Backup y-tilt	na	++
SD X-tilt	Y-tilt		--
	SD Y-tilt	++	
	Depth		
	Speed over ground		
	Backup y-tilt	na	--
Y-tilt	SD Y-tilt		
	Depth		
	Speed over ground		++
	Backup y-tilt	na	--
SD Y-tilt	Depth		
	Speed over ground		++
	Backup y-tilt		
Depth	Speed over ground		
	Backup y-tilt		
Speed over ground	Backup y-tilt	na	--

Table B14. DE2DE2 (Delaware II-Delaware II) repeat station tow data (50+ mm SL). Catch are numbers of ocean quahogs caught adjusted to a standard area swept based on sensor tow distance data (4,557 ft^2 = 423 m^2). Stations with useful data are at the top of the table. Stations excluded from the analysis because both tows were zero or because of poor dredge performance (based on differential pressure and amperage sensors) are shown at the bottom. "HG" codes are NEFSC survey database codes that describe results of the haul and damage to the dredge based on observations by the watch chief (without using sensor data). By convention, tows with HG ≤ 36 are used in most analyses.

	Original station					Repeat station				
Stratum	Station	Catch	Cable	Pump	SHG	Station	Catch	Cable	Pump	HG
				Useful repeat stations						
6250	16	5.754	old	old	11	315	4.233	new	new	36
6250	17	1.4855	old	old	11	292	2.100	new	new	11
6250	23	3.1124	old	old	11	294	0.000	new	new	11
6250	25	0.9655	old	old	11	313	0.000	new	new	11
6930	170	2.9155	old	new	23	325	1.485	new	new	11
6930	172	21.295	old	new	34	329	284.070	new	new	35
6250	38	0.8368	old	old	11	296	0.000	new	new	11
6930	172	21.2954	old	new	34	327	7.068	new	new	11
6930	173	611.722	old	new	11	328	341.535	new	new	11
6330	174	105.004	old	new	36	328	341.535	new	new	11
6330	178	280.119	old	new	11	333	260.802	new	new	35
6930	179	19.830	old	new	11	335	13.517	new	new	11
6330	180	288.316	old	new	11	336	102.231	new	new	11
6920	181	10.588	old	new	11	337	7.724	new	new	11
6290	182	453.819	old	new	11	338	230.036	new	new	11
6290	183	359.921	old	new	11	339	121.018	new	new	11
6250	214	1.047	old	new	11	295	24.768	new	new	11
				Both catches zero						
6890	13	0.0000	old	old	11	316	0.000	new	new	11
6890	26	0.0000	old	old	11	314	0.000	new	new	11
6890	30	0.0000	old	old	11	312	0.000	new	new	11
6210	37	0.0000	old	old	11	302	0.000	new	new	36
6210	41	0.0000	old	old	11	303	0.000	new	new	11
6890	42	0.0000	old	old	11	304	0.000	new	new	11
6890	45	0.0000	old	old	35	310	0.000	new	new	34
6890	48	0.0000	old	old	35	317	0.000	new	new	11
6880	51	0.0000	old	old	11	318	0.000	new	new	11
6880	53	0.0000	old	old	11	319	0.000	new	new	48
				Poor dredge performance						
6250	22	26.069	old	old	11	293	27.008	new	new	23
6330	171	31.390	old	new	35	326	6.525	new	new	36
6300	206	327.657	old	new	11	287	420.315	new	new	11

Table B15. DE2FV (Delaware II – F/V Endurance) repeat tow data. Catches are numbers or ocean quahogs per standard area swept (4557 ft^2 = 423 m^2). "HG" codes are NEFSC survey database codes that describe results of the haul and damage to the dredge based on observations by the watch chief (without using sensor data). All of the stations shown in the table are useable based on differential pressure and amperage data from sensors. By convention, tows with HG ≤ 36 are used in most analyses.

Sequential FV tow number	DE2 station number	Pump	Electrical cable	HG code	W code	DE2 catch (N per standard tow area)	FV catch (N per standard tow area)	Summary of DE2 Configuration
76	304	New	New	11	2	0.000	0.000	New pump-New cable
77	303	New	New	11	0	0.000	0.000	New pump-New cable
79	312	New	New	11	0	0.000	0.382	New pump-New cable
80	313	New	New	11	0	0.000	0.597	New pump-New cable
81	314	New	New	11	0	0.000	0.000	New pump-New cable
82	316	New	New	11	0	0.000	0.000	New pump-New cable
84	290	New	New	11	1	93.661	286.865	New pump-New cable
84	289	New	New	11	1	81.602	286.865	New pump-New cable
85	290	New	New	11	1	93.661	305.617	New pump-New cable
85	289	New	New	11	1	81.602	305.617	New pump-New cable
102	272	New	New	11	3	71.985	263.336	New pump-New cable
103	274	New	New	36	3	0.966	30.072	New pump-New cable
104	276	New	New	11	-2	28.000	65.263	New pump-New cable
105	278	New	New	11	2	33.736	383.916	New pump-New cable
106	282	New	New	11	0	145.733	320.499	New pump-New cable
107	280	New	New	11	2	0.702	3.541	New pump-New cable
118	354	New	New	11	1	162.193	674.015	New pump-New cable
118	355	New	New	11	1	161.239	674.015	New pump-New cable
118	353	New	New	11	1	143.319	674.015	New pump-New cable
159	319	New	New	48	1	0.000	0.000	New pump-New cable
160	318	New	New	11	2	0.000	0.000	New pump-New cable
161	296	New	New	11	0	0.000	0.000	New pump-New cable
162	295	New	New	11	2	23.642	45.174	New pump-New cable
167	339	New	New	11	1	35.257	200.715	New pump-New cable
168	336	New	New	11	0	62.378	96.687	New pump-New cable
169	334	New	New	11	4	55.518	168.281	New pump-New cable
170	333	New	New	35	0	93.726	315.868	New pump-New cable
171	324	New	New	11	0	66.136	191.406	New pump-New cable
172	326	New	New	36	5	2.175	0.000	New pump-New cable
174	328	New	New	11	0	148.925	430.130	New pump-New cable
191	338	New	New	11	1	113.000	178.561	New pump-New cable

Table B15. (cont.)

Sequential FV tow number	DE2 station number	Pump	Electrical cable	HG code	W code	DE2 catch (N per standard tow area)	FV catch (N per standard tow area)	Summary of DE2 Configuration
192	293	New	New	23	2	24.847	142.024	New pump-New cable
193	294	New	New	11	0	0	7.608	New pump-New cable
194	292	New	New	11	0	1.05	9.009	New pump-New cable
195	315	New	New	36	1	3.175	5.853	New pump-New cable
196	310	New	New	34	5	0	0	New pump-New cable
101	205	New	Old	11	1	52.228	153.64	New pump-Old cable
163	201	New	Old	11	1	70.373	429.723	New pump-Old cable
164	209	New	Old	11	4	101.89	395.804	New pump-Old cable
165	207	New	Old	23	3	47.045	341.305	New pump-Old cable
166	203	New	Old	11	1	46.442	323.178	New pump-Old cable
167	183	New	Old	11	1	110.22	200.715	New pump-Old cable
168	180	New	Old	11	0	150.835	96.687	New pump-Old cable
170	178	New	Old	35	0	97.339	315.868	New pump-Old cable
174	173	New	Old	11	0	374.091	430.13	New pump-Old cable
174	176	New	Old	11	0	113.529	430.13	New pump-Old cable
174	174	New	Old	36	1	44.657	430.13	New pump-Old cable
174	177	New	Old	11	0	43.126	430.13	New pump-Old cable
191	182	New	Old	11	1	221.989	178.561	New pump-Old cable
200	199	New	Old	11	1	16.213	77.062	New pump-Old cable
78	36	Old	Old	11	1	3.435	13.902	Old pump-Old cable
169	2	Old	Old	11	4	25.028	168.281	Old pump-Old cable
171	1	Old	Old	11	0	150.771	191.406	Old pump-Old cable
197	49	Old	Old	11	0	0	0	Old pump-Old cable
198	60	Old	Old	11	1	0	0	Old pump-Old cable
199	64	Old	Old	11	0	0	0	Old pump-Old cable

Table B16. Summary of 2008 commercial depletion experiments for ocean quahog with comparisons to results of experiments during 1997-2005. Depletion experiments are identified by a sequential and field ID codes. The sequential codes are ordered by date (e.g. OQ2008-3 was the third study for ocean quahog completed during 2008). The field identification codes were used in planning and carrying out the experiments (e.g. field ID OQ08-6 for the experiment with sequential ID OQ2008-03). Sequential ID codes are used in this assessment.

Depletion experiment ID (Field ID)	**Commercial dredge efficiency estimate**	**Population density estimate (N/ft^2)**	**Negative binomial k estimate**	**Setup tow station numbers**	**Setup Configuration**	**Setup Density (N/tow)**	**Setup Density (N/ft^2)**	**Setup Density CV**	**Survey dredge efficiency**	**Comment**
OQ2008-01 (OQ08-1)	1.000	0.068	7.55	173, 174, 176, 177	Old cable; new pump	143.851	0.032	0.546	0.467	Poor Patch model fit, note high CV for stock density from Patch model and setup tow density
OQ2008-02 (OQ08-2)	0.780	0.086	14.55	289	New cable; new pump	81.602	0.018	NA	0.207	Good Patch model fit; only 1setup tow
OQ2008-03 (OQ08-6)	1.000	0.120	5.95	353, 354, 355	New cable; new pump	155.584	0.034	0.039	0.285	Good Patch model fit
Mean OQ-08 (N=3)	0.927	0.091	9.349	NA	NA	127.012	0.028	NA	0.320	2008 commercial efficiency estimates higher than average from previous studies; 2008 population density estimates about the same as average from previous studies; survey dredge efficiencies 25% higher than average of previous estimates
All 1997-2005 (N=17)	0.596 (95% CI 0.469 to 0.723)	0.097 (95% CI 0.032 to 0.162)		NA	NA	NA	NA	NA	0.248	

Table B17. Patch model estimates for ocean quahogs 90+ mm SL in commercial and NEFSC survey clam dredges based on depletion experiments during 1997-2008. "NA" means not available. The sequential codes are ordered by date (e.g. OQ2008-3 was the third study for ocean quahog completed during 2008). The field identification codes were used in planning and carrying out the experiments (e.g. field ID OQ08-6 for the experiment with sequential ID OQ2008-03). Sequential ID codes are used in this assessment. Footnotes are on the page following the table.

Study area						Depletion Tows						Patch Model							Setup Tows (if applicable)				
Experiment	Region	Latitude (decimal degrees)	Longitude (decimal degrees)	Depth (m)	Mean Sediment Size (microns)	Depletion Vessel	Date	Ship Position Data (source / nominal accuracy / time interval)	N tows used	N Bushel Counts / Length samples	Depletion Vessel Blade Width (ft)	Cell Size (ft)	Density (N ft^{-2})	Depletion Vessel Efficiency	Neg. binomial *k*	Gamma γ	Neg. Log likelihood	Fit to Catch Data (R2s)	Setup Date	RV stations	Setup or RV Density (N ft^{-2})	NEFSC Survey Dredge Efficiency	Foot-notes
OQ2008-01 (OQ08-1)	LI	-72.04765	40.93762	27	530	F/V Endurance	2-Sep	GPS / 6 ft / 6 sec	17	4 / 4	12.5	25	0.068	1.000	7.55	0.50	118.5	Poor	16-Jul	173-174, 176, 177	0.032	0.467	19
OQ2008-02 (OQ08-2)	LI	-72.84397	40.27445	49	258	F/V Endurance	16-Sep	GPS / 6 ft / 6 sec	17	4 / 4	12.5	25	0.086	0.781	14.55	0.50	115.0	Ok	22-Jul	289	0.018	0.207	19
OQ2008-03 (OQ08-6)	SNE	-70.85472	41.02307	46	357	F/V Endurance	18-Sep	GPS / 6 ft / 6 sec	17	4 / 4	12.5	25	0.120	1.000	5.95	0.50	127.5	Ok	30-Jul	353-355	0.034	0.285	19
Mean				*41*	*382*								*0.091*	*0.927*	*9.350*						*0.0279*	*0.320*	
CV for Mean				*17%*	*21%*								*17%*	*8%*	*28%*						*18%*	*0.241*	
OQ2005-1	LI	40.51903	72.07617	57	536	F/V Lisa Kim	5-Sep	GPS / 6 ft / 6 sec	20	4 / 4	10	20	0.073	0.183	1.97	0.50	127.0	Ok	Jun-05	165, 231-234	0.0120	0.165	1
OQ2005-2	LI	40.38957	72.38950	53	438	F/V Lisa Kim	5-Sep	GPS / 6 ft / 6 sec	21	4 / 4	10	20	0.047	0.402	8.57	0.50	131.8	Ok	Jun-05	162, 235-238	0.0080	0.169	1
OQ2005-3	LI	40.64220	72.65170	35	267	F/V Lisa Kim	5-Sep	GPS / 6 ft / 6 sec	20	4 / 4	10	20	0.085	0.733	9.57	0.50	125.9	Ok	Jun-05	3, 239-242	0.0101	0.119	1
OQ2005-4	LI	40.68817	72.18147	46	308	F/V Lisa Kim	5-Sep	GPS / 6 ft / 6 sec	17	4 / 4	10	20	0.027	0.815	12.31	0.50	89.4	Ok	Jun-05	168, 243-246	0.0042	0.154	1
OQ2005-6	LI	40.05550	72.41673	65	554	F/V Lisa Kim	5-Sep	GPS / 6 ft / 6 sec	20	4 / 4	10	20	0.137	0.660	2.55	0.50	146.3	Ok	Jun-05	252-256	0.0210	0.153	1
Mean				*51*	*421*								*0.074*	*0.559*	*6.99*						*0.0110*	*0.152*	
CV for Mean				*10%*	*14%*								*25%*	*21%*	*29%*						*25%*	*0.058*	
OQ2002-1 (LK-1)	LI	40.72762	71.73730	60	331	F/V Lisa Kim	5-Mar	GPS / 1 ft / 6 sec	24	5 / 5	10	20	0.295	0.489	6.56	0.50	173.1	Ok	Jun-02	5 - 9	0.0290	0.098	1, 2, 5
OQ2002-2 (LK-2)	LI	40.10312	73.19108	48	277	F/V Lisa Kim	5-Mar	GPS / 1 ft / 6 sec	22	4 / 4	10	20	0.165	0.785	10.57	0.50	149.7	Ok	Jun-02	25- 29	0.0245	0.149	1, 2
OQ2002-3 (LK-3)	NJ	38.81491	73.81335	50	195	F/V Lisa Kim	5-Mar	GPS / 1 ft / 6 sec	20	4 / 4	10	20	0.081	0.777	11.57	0.50	133.4	Ok	Jun-02	213 - 217	0.0239	0.297	1, 2
OQ2002-4 (LK-4)	DMV	37.88755	74.64486	48	135	F/V Lisa Kim	4-Mar	GPS / 1 ft / 6 sec	24	5 / 5	10	20	0.073	0.254	12.46	0.50	136.0	Ok	Jun-02	272 - 276	0.0210	0.287	1, 2,9, 16
Mean		*39.38330*	*73.34665*	*52*	*235*								*0.153*	*0.576*	*10.29*						*0.0246*	*0.208*	
CV for Mean				*6%*	*18%*								*34%*	*22%*	*13%*						*7%*	*0.239*	
OQ2000-1 (JN-1)	LI	40.60217	71.98750	58	N/A	F/V John N	1-Mar	GPS / 1 ft / 30 sec	22	5 / 5	12.5	25	0.100	0.730	5.55	0.50	157.4	Ok	Jun-99	194 - 199	NA	NA	1, 2,6
OQ2000-2 (JN-2)	LI	40.39450	72.54300	48	N/A	F/V John N	1-Mar	GPS / 1 ft / 30 sec	16	4 / 3	12.5	25	0.062	0.554	15.10	0.50	98.1	Ok	Jun-99	178 - 180	0.0145	0.234	1, 2,7,11,12, 17
OQ2000-3 (DM-1)	LI	40.58300	72.79683	40	N/A	F/V Danielle Maria	1-May	GPS / 1 ft / 30 sec	27	6 / 6	10	20	0.089	0.560	4.57	0.50	184.2	Ok	Jun-99	3 - 8	0.0147	0.165	1, 2,8,10,12, 18
Mean		*40.52656*	*72.44244*	*49*									*0.084*	*0.615*	*8.405*						*0.0146*	*0.199*	
CV for Mean				*11%*									*14%*	*9%*	*40%*						*1%*	*0.175*	
OQ1999-01 DE2	LI	40.60227	71.98483	57	N/A	R/V Delaware II	1-Jun	GPS / 36 ft / 1 sec	60	8 / 8	5	10	0.007	0.990	4.05	0.25	253.1	Poor		N/A		0.990	14, 15
OQ1998-1 (SH-3)	LI (Shinnecock)	40.76650	72.17950	41	N/A	F/V Cape Fear	1-Mar	Loran / 40 ft / 30 sec.	14	3 / 3	10	20	0.017	1.000	3.48	0.50	76.5	Poor					1, 13
OQ1998-2 (SH-2)	LI (Shinnecock)	40.72200	72.00750	45	N/A	F/V Cape Fear	1-Mar	Loran / 40 ft / 30 sec.	23	5 / 5	10	20	0.067	0.869	10.57	0.50	140.3	Ok		NA		NA	15
OQ1998-3 (NS-1)	SNE (Nantucket Shoals)	40.46700	69.48300	63	N/A	F/V Cape Fear	1-Apr	Loran / 40 ft / 30 sec.	24	5 / 5	10	20	0.255	0.710	7.56	0.50	195.5	Ok					15
Mean		*40.65183*	*71.22333*	*50*									*0.113*	*0.860*	*7.204*								
CV for Mean				*14%*									*64%*	*10%*	*29%*								
OQ1997-1 (SH-1)	LI (Shinnecock)	40.26950	72.29850	58	N/A	F/V Laura Ann	1-Jul	Loran / 40 ft / 30 sec.	28	7 / 7	7.75	20	0.083	0.458	10.57	0.39	164.2	Ok					1,3
OQ1997-2 (WW-1)	NJ (Wildwood)	38.50950	74.11150	49	N/A	F/V Agitator	1-Aug	Loran / 40 ft / 30 sec.	28	13 / 6	10	20	0.084	0.150	2.37	0.50	176.0	Ok		NA		NA	1,4
Mean		*39.38950*	*73.20500*	*54*									*0.083*	*0.304*	*6.47*								
CV for Mean				*8%*									*0%*	*51%*	*63%*								

Footnotes for Table B17

[1] NA
[2] NA
[3] Depletion tows 1, 2, 12 & 18 omitted per NEFSC 1998, Figure E18
[4] Depletion tows 1, 19, 23 & 27 omitted per NEFSC 1998, Figure E21
[5] Setup station 5 dropped because sensor tow distance < 0.04 nm
[6] Length composition data collected at setup tow 194 only for OQ2000-1 (indicated 6% of catch >= 90 mm SL), setup data not useable.
[7] Length composition data collected at setup tow 178 only for OQ2000-2 (indicated 28% of catch >= 90 mm SL), used for all setup tows.
[8] Length composition data collected at setup tows 3 and 6 only for OQ2000-3 (average 33% and 28% of catch >= 90 mm SL), used for all setup tows.
[9] Length composition data collected at setup tow 272 only for OQ2000-4 (33% of catch >= 90 mm SL), used for all setup tows.
[10] Sensor tow distance missing for setup station 4, average tow distance at stations 3, 5, 6, 7, 8 used instead.
[11] Depletion tow 1 omitted because it was outside the study area.
[12] Adjustments for apparent trends in numbers per bushel during depletion experiment.
[13] Original estimates appear to have used incorrect mean number per bushel in depletion tows
[14] Missing GPS location data at survey stations 198 and 216 (depletion tows 5 and 23) replaced by approximate start/stop locations and interpolation.
[15] Anomalously high bushel count and length data at station 200 were not used.
[16] One setup tow with length data for OQ2002-4.
[17] One setup tow with length data for OQ2000-2.
[18] Two setup tows with length data for OQ2000-3.
[19] Used backup GPS and backup depth sensor data in place of SSP sensor data for depletion tows. Setup tows used SSP data.

Table B18. Summary of density, commercial dredge efficiency, and NEFSC dredge efficiency estimates for ocean quahog 90+ mm SL from the Patch model. The 90% confidence interval calculated by bootstrapping the fifteen survey efficiency estimates (15,000 iterations) ranged from 0.154 to 0.285.

Statistic	Density (N ft^{-2})	Commercial Vessel Efficiency	NEFSC Survey Dredge Efficiency
N experiments	21	20	15
Minimum	0.007	0.150	0.098
Maximum	0.295	1.000	0.990
Median	0.083	0.720	0.169
Mean	0.096	0.646	0.263
Distribution of point estimates[1]			
Standard deviation	0.070	0.259	0.222
CV (sd/mean)	0.728	0.402	0.845
Lo 95%	0.000	0.137	0.000
Hi 95%	0.233	1.000	0.697
Distribution of average estimates[1]			
Standard error	0.015	0.058	0.057
CV (se/mean)	0.159	0.090	0.218
Lo 95%	0.066	0.532	0.150
Hi 95%	0.126	0.759	0.375

Table B19. Efficiency corrected swept-area fishable biomass estimates (1,000 mt meats) and CVs for ocean quahog during 1997, 2000, 2002, 2005 and 2008 (years with NEFSC clam surveys), by region. Figures for SVA and GBK during 2005 are, in effect, averages of figures for 2002 and 2008 because little data were available for 2005.

Area of assessment region (*A* , nm^2) - no correction for stations with unsuitable clam habitat

S. Virginia and N. Carolina (SVA)	712	10%
Delmarva (DMV)	4,071	10%
New Jersey (NJ)	6,510	10%
Long Island (LI)	4,463	10%
Southern New England (SNE)	4,922	10%
Georges Bank (GBK)	7,821	10%
Total	28,499	

INPUT: Fraction suitable habitat (*u*)

S. Virginia and N. Carolina (SVA)	100%	10%
Delmarva (DMV)	100%	10%
New Jersey (NJ)	100%	10%
Long Island (LI)	100%	10%
Southern New England (SNE)	96%	10%
Georges Bank (GBK)	90%	10%

Habitat area in assessment region (*A'* , nm2)

S. Virginia and N. Carolina (SVA)	712	14%
Delmarva (DMV)	4,071	14%
New Jersey (NJ)	6,510	14%
Long Island (LI)	4,463	14%
Southern New England (SNE)	4,714	14%
Georges Bank (GBK)	7,039	14%

INPUT: Biomass fraction in unsurveyd deep water

S. Virginia and N. Carolina (SVA)	0%	10%
Delmarva (DMV)	0%	10%
New Jersey (NJ)	0%	10%
Long Island (LI)	0%	10%
Southern New England (SNE)	2%	10%
Georges Bank (GBK)	13%	10%

INPUT: Original survey mean catch from fishable stock (kg/tow, for tows adjusted to nominal tow distance using sensors)

	Estimates for 1997	CV	Estimates for 1999	CV	Estimates for 2002	CV	Estimates for 2005	CV	Estimates for 2008	CV
S. Virginia and N. Carolina (SVA)	0.0013	100%	0.0007	55%	0.0004	100%	0.0004	100%	0.0004	100%
Delmarva (DMV)	0.6528	23%	0.4449	26%	0.6879	24%	0.4221	48%	0.3908	52%
New Jersey (NJ)	1.7341	15%	0.9728	14%	1.8752	23%	1.0553	14%	1.2071	19%
Long Island (LI)	4.5648	17%	3.0065	14%	3.5561	18%	2.1791	16%	3.4396	15%
Southern New England (SNE)	2.2252	37%	2.6964	45%	3.2654	26%	2.0689	22%	2.8049	22%
Georges Bank (GBK)	2.6710	16%	3.1454	18%	3.8760	17%	4.3336	20%	4.7733	27%

Swept-area biomass without efficiency correction (B', 1000 mt):

S. Virginia and N. Carolina (SVA)	0.008	102%	0.004	59%	0.002	102%	0.002	102%	0.002	102%
Delmarva (DMV)	22	30%	15	33%	23	31%	14	52%	13	56%
New Jersey (NJ)	91	25%	51	24%	99	30%	56	24%	64	28%
Long Island (LI)	165	26%	109	24%	129	27%	79	26%	124	25%
Southern New England (SNE)	87	42%	105	49%	127	33%	81	30%	109	30%
Georges Bank (GBK)	172	26%	203	27%	250	26%	279	28%	308	34%
Total fishable biomass less GBK	365	17%	280	21%	378	17%	229	15%	310	16%
Total fishable biomass	537	14%	483	17%	627	14%	508	17%	618	19%

INPUT: Survey dredge efficiency (e)	0.169	21%	0.169	21%	0.169	21%	0.169	21%	0.169	21%

Efficiency adjusted swept area fishable biomass (B, 1000 mt)

S. Virginia and N. Carolina (SVA)	0.045	104%	0.024	62%	0.013	104%	0.013	104%	0.013	104%
Delmarva (DMV)	127	37%	87	39%	134	38%	82	56%	76	60%
New Jersey (NJ)	541	33%	304	32%	585	37%	329	32%	377	35%
Long Island (LI)	977	34%	644	32%	761	34%	466	33%	736	33%
Southern New England (SNE)	513	47%	622	54%	753	39%	477	36%	647	36%
Georges Bank (GBK)	1,019	33%	1,200	34%	1,479	34%	1,653	35%	1,821	40%
Total fishable biomass less GBK	2,159	27%	1,656	30%	2,234	27%	1,355	26%	1,836	26%
Total fishable biomass	3,178	25%	2,856	27%	3,713	25%	3,009	27%	3,657	28%

Lower bound for 80% confidence intervals on fishable biomass (1000 mt, for lognormal distribution with no bias correction)

	Estimates for 1997	Estimates for 1999	Estimates for 2002	Estimates for 2005	Estimates for 2008
S. Virginia and N. Carolina (SVA)	0.015	0.011	0.004	0.004	0.004
Delmarva (DMV)	81	54	84	42	38
New Jersey (NJ)	360	203	370	220	245
Long Island (LI)	643	430	498	309	490
Southern New England (SNE)	290	327	465	304	412
Georges Bank (GBK)	674	785	973	1,067	1,117
Total fishable biomass less GBK	1,539	1,138	1,596	978	1,320
Total fishable biomass	2,311	2,037	2,693	2,142	2,573

Upperbound for 80% confidence intervals on fishable biomass (1000 mt, for lognormal distribution with no bias correction)

S. Virginia and N. Carolina (SVA)	0.134	0.049	0.039	0.039	0.039
Delmarva (DMV)	202	141	214	161	154
New Jersey (NJ)	814	454	926	493	580
Long Island (LI)	1,485	962	1,164	705	1,106
Southern New England (SNE)	909	1,182	1,218	749	1,016
Georges Bank (GBK)	1,540	1,835	2,248	2,561	2,969
Total fishable biomass less GBK	3,029	2,409	3,127	1,879	2,555
Total fishable biomass	4,371	4,004	5,118	4,226	5,198

Table B20. Ocean quahog fishing mortality estimates based on catch and efficiency corrected swept-area biomass for fishable ocean quahog during 1997, 1999, 2002, 2005 and 2008 with NEFSC clam surveys. CVs are based on analytical variance calculations assuming log normality, and include uncertainty in catch, survey data, swept-area, amount of suitable habitat, and survey dredge efficiency.

INPUT: Upper bound incidental mortality allowance	5%
INPUT: Assumed CV for catch	10%

INPUT: Landings (1000 mt, discard ~ 0)	**Estimates for 1997**	**Estimates for 1999**	**Estimates for 2002**	**Estimates for 2005**	**Estimates for 2008**
S. Virginia and N. Carolina (SVA)	0	0.000	0.000	0.000	0.000
Delmarva (DMV)	1.072	1.092	1.737	0.912	0.270
New Jersey (NJ)	4.229	3.043	2.789	0.670	1.733
Long Island (LI)	5.141	6.339	9.140	9.728	11.123
Southern New England (SNE)	8.968	6.628	3.895	2.024	2.151
Georges Bank (GBK)	0.000	0.000	0.000	0.000	0.000
Total	19.410	17.102	17.561	13.334	15.278

Catch (1000 mt, landings + upper bound incidental mortality allowance)					
S. Virginia and N. Carolina (SVA)	0.000	0.000	0.000	0.000	0.000
Delmarva (DMV)	1.126	1.146	1.824	0.957	0.283
New Jersey (NJ)	4.441	3.195	2.928	0.704	1.820
Long Island (LI)	5.398	6.656	9.597	10.215	11.679
Southern New England (SNE)	9.416	6.960	4.090	2.125	2.259
Georges Bank (GBK)	0.000	0.000	0.000	0.000	0.000
Total	20.380	17.957	18.439	14.001	16.042

INPUT: Efficiency Corrected Swept Area Biomass for Fishable Stock (1000 mt)	**Estimates for 1997**	**CV**	**Estimates for 1999**	**CV**	**Estimates for 2002**	**CV**	**Estimates for 2005**	**CV**	**Estimates for 2008**	**CV**
S. Virginia and N. Carolina (SVA)	0	104%	0	62%	0	104%	0	104%	0	104%
Delmarva (DMV)	127	37%	87	39%	134	38%	82	56%	76	60%
New Jersey (NJ)	541	33%	304	32%	585	37%	329	32%	377	35%
Long Island (LI)	977	34%	644	32%	761	34%	466	33%	736	33%
Southern New England (SNE)	513	47%	622	54%	753	39%	477	36%	647	36%
Georges Bank (GBK)	1,019	33%	1,200	34%	1,479	34%	1,653	35%	1,821	40%
Total fishable biomass less GBK	2,159	27%	1,656	30%	2,234	27%	1,355	26%	1,836	26%
Total fishable biomass	3,178	25%	2,856	27%	3,713	25%	3,009	27%	3,657	28%

Fishing mortality (y^{-1})										
S. Virginia and N. Carolina (SVA)	0.000	105%	0.000	63%	0.000	105%	0.000	105%	0.000	105%
Delmarva (DMV)	0.009	38%	0.013	40%	0.014	39%	0.012	57%	0.004	60%
New Jersey (NJ)	0.008	34%	0.011	34%	0.005	38%	0.002	34%	0.005	36%
Long Island (LI)	0.006	NA	0.010	NA	0.013	36%	0.022	35%	0.016	34%
Southern New England (SNE)	0.018	48%	0.011	54%	0.005	40%	0.004	38%	0.003	38%
Georges Bank (GBK)	0.000	NA	0.000	NA	0.000	NA	0.000	0%	0.000	0%
Total fishable biomass less GBK	0.009	29%	0.011	32%	0.008	29%	0.010	28%	0.009	28%
Total fishable biomass	0.006	27%	0.006	29%	0.005	27%	0.005	29%	0.004	30%

Lower bound for 80% confidence intervals for fishing mortality (y^{-1}, for lognormal distribution with no bias correction)	**Estimates for 1997**	**Estimates for 1999**	**Estimates for 2002**	**Estimates for 2005**	**Estimates for 2008**
S. Virginia and N. Carolina (SVA)	NA	NA	NA	NA	NA
Delmarva (DMV)	0.005	0.008	0.008	0.006	0.002
New Jersey (NJ)	0.005	0.007	0.003	0.001	0.003
Long Island (LI)	NA	NA	0.008	0.014	0.010
Southern New England (SNE)	0.010	0.006	0.003	0.003	0.002
Georges Bank (GBK)	NA	NA	NA	NA	NA
Total fishable biomass less GBK	0.007	0.007	0.006	0.007	0.006
Total fishable biomass	0.005	0.004	0.004	0.003	0.003

Upper bound for 80% confidence intervals for fishing mortality (y^{-1}, for lognormal distribution with no bias correction)					
S. Virginia and N. Carolina (SVA)	NA	NA	NA	NA	NA
Delmarva (DMV)	0.014	0.022	0.022	0.023	0.008
New Jersey (NJ)	0.013	0.016	0.008	0.003	0.008
Long Island (LI)	NA	NA	0.020	0.034	0.024
Southern New England (SNE)	0.033	0.021	0.009	0.007	0.006
Georges Bank (GBK)	NA	NA	NA	NA	NA
Total fishable biomass less GBK	0.014	0.016	0.012	0.015	0.012
Total fishable biomass	0.009	0.009	0.007	0.007	0.006

Table B21. “Best” biomass estimates for ocean quahogs during 1978-2008. SVA estimates are from "VPA" and other regional estimates are from KLAMZ models. Whole stock and exploited stock biomass are sums of regional estimates. "KLAMZ (1R)" means from a KLAMZ model that has constant recruitment in each year. "KLAMZ (2R)" means from a KLAMZ model assuming two periods of constant recruitment. "*Q* for ESB" is the estimated (KLAMZ model) or assumed (VPA) survey scaling parameter for efficiency corrected swept area biomass. *Q* values are a diagnostic for KLAMZ model fits and expected to be near one.

Biomass	VPA		KLAMZ (1R)		KLAMZ (1R)		KLAMZ (2R)		KLAMZ (2R)		KLAMZ (1R)		Sum of best regional estimates			
Q for ESB	1.00		0.96		0.96		1.04		1.04		0.98					
Year	SVA	CV	DMV	CV	NJ	CV	LI	CV	SNE	CV	GBK	CV	Exploitable stock	CV	Whole Stock	CV
1978	0.3344	0.96	298	0.14	897	0.13	663	0.28	553	0.38	1,169	0.41	2,412	0.24	3,580	0.21
1979	0.3344	0.96	290	0.15	872	0.13	676	0.26	564	0.36	1,175	0.39	2,403	0.24	3,577	0.21
1980	0.3344	0.96	277	0.15	848	0.13	689	0.25	575	0.34	1,181	0.37	2,389	0.23	3,570	0.20
1981	0.3344	0.96	267	0.15	824	0.14	702	0.24	586	0.32	1,186	0.36	2,378	0.22	3,564	0.19
1982	0.2708	0.96	257	0.15	800	0.14	714	0.23	596	0.30	1,192	0.34	2,368	0.22	3,560	0.19
1983	0.2639	0.96	247	0.16	776	0.14	727	0.22	607	0.28	1,198	0.32	2,358	0.21	3,555	0.18
1984	0.2639	0.96	237	0.16	754	0.14	740	0.21	616	0.26	1,203	0.31	2,347	0.21	3,550	0.17
1985	0.2571	0.96	225	0.17	731	0.14	752	0.20	626	0.24	1,209	0.29	2,334	0.20	3,542	0.17
1986	0.0712	0.96	212	0.17	706	0.14	764	0.19	635	0.23	1,214	0.28	2,318	0.20	3,532	0.16
1987	0.0712	0.96	200	0.18	684	0.15	776	0.19	645	0.22	1,220	0.27	2,305	0.19	3,524	0.16
1988	0.0712	0.96	185	0.19	662	0.15	787	0.18	654	0.21	1,225	0.25	2,289	0.19	3,514	0.15
1989	0.0272	0.96	170	0.20	643	0.15	798	0.17	663	0.20	1,231	0.24	2,275	0.19	3,506	0.15
1990	0.0272	0.96	160	0.21	618	0.15	810	0.17	672	0.19	1,236	0.23	2,260	0.18	3,496	0.14
1991	0.0130	0.96	154	0.22	591	0.16	821	0.17	681	0.18	1,241	0.22	2,247	0.18	3,488	0.14
1992	0.0130	0.96	146	0.22	566	0.16	831	0.16	690	0.17	1,246	0.21	2,233	0.18	3,479	0.14
1993	0.0130	0.96	140	0.23	549	0.16	813	0.16	684	0.17	1,251	0.20	2,187	0.18	3,438	0.13
1994	0.0130	0.96	136	0.23	529	0.16	799	0.17	678	0.17	1,256	0.20	2,142	0.18	3,398	0.13
1995	0.0130	0.96	132	0.23	513	0.17	781	0.17	672	0.17	1,261	0.19	2,098	0.18	3,359	0.13
1996	0.0130	0.96	129	0.23	499	0.17	765	0.17	661	0.17	1,266	0.19	2,054	0.18	3,320	0.13
1997	0.0130	0.96	125	0.24	485	0.17	753	0.17	647	0.17	1,271	0.18	2,011	0.18	3,282	0.13
1998	0.0130	0.96	122	0.24	472	0.17	742	0.17	633	0.17	1,276	0.18	1,969	0.18	3,245	0.13
1999	0.0130	0.96	118	0.24	461	0.17	728	0.17	621	0.18	1,280	0.18	1,928	0.18	3,209	0.13
2000	0.0130	0.96	115	0.24	450	0.17	715	0.17	608	0.18	1,285	0.18	1,888	0.18	3,173	0.13
2001	0.0130	0.96	111	0.25	439	0.17	704	0.17	597	0.18	1,290	0.18	1,852	0.18	3,141	0.13
2002	0.0130	0.96	108	0.25	426	0.17	691	0.17	587	0.18	1,294	0.18	1,813	0.18	3,107	0.13
2003	0.0130	0.96	104	0.25	416	0.18	675	0.18	577	0.18	1,298	0.18	1,773	0.18	3,071	0.13
2004	0.0130	0.96	101	0.25	405	0.18	657	0.18	569	0.18	1,303	0.18	1,732	0.18	3,035	0.13
2005	0.0130	0.96	99	0.26	396	0.18	639	0.18	559	0.18	1,307	0.18	1,693	0.19	3,000	0.13
2006	0.0130	0.96	96	0.26	388	0.18	623	0.18	551	0.18	1,311	0.19	1,658	0.19	2,969	0.13
2007	0.0130	0.96	94	0.26	381	0.18	605	0.19	544	0.18	1,315	0.19	1,623	0.19	2,938	0.13
2008	0.0130	0.96	92	0.26	373	0.18	587	0.19	535	0.18	1,319	0.20	1,586	0.19	2,905	0.13
Min	0.0130	0.96	92	0.145	373	0.132	587	0.163	535	0.171	1,169	0.176	1,586	0.176	2,905	0.127
Median	0.0130	0.96	140	0.226	549	0.160	728	0.178	616	0.182	1,251	0.209	2,187	0.185	3,438	0.135
Mean	0.0934	0.96	166	0.210	586	0.157	727	0.191	616	0.217	1,249	0.242	2,094	0.193	3,343	0.150
Max	0.3344	0.96	298	0.260	897	0.178	831	0.278	690	0.383	1,319	0.407	2,412	0.244	3,580	0.213

Table B22. Best fishing mortality estimates for ocean quahogs during 1978-2008. . Whole stock, exploited region, and SVA estimates are from solving the catch equation for catch given best biomass estimates and instantaneous rates for growth and recruitment. Other regional estimates are from KLAMZ models that provided the best biomass estimates.

Year	SVA	CV	DMV	CV	NJ	CV	LI	CV	SNE	CV	GBK	CV	Exploitable stock	CV	Whole Stock	CV
1978	0.0000	0.00	0.0060	0.15	0.0098	0.13	0.0000	0.00	0.0000	0.00	0.0000	0.00	0.0045	0.24	0.0031	0.24
1979	0.0000	0.00	0.0264	0.15	0.0096	0.13	0.0000	0.00	0.0000	0.00	0.0000	0.00	0.0069	0.24	0.0046	0.24
1980	0.0000	0.96	0.0174	0.15	0.0104	0.14	0.0000	0.25	0.0000	0.00	0.0000	0.00	0.0059	0.23	0.0039	0.23
1981	0.2135	0.96	0.0150	0.15	0.0112	0.14	0.0000	0.24	0.0000	0.00	0.0000	0.00	0.0058	0.23	0.0039	0.23
1982	0.0258	0.00	0.0197	0.16	0.0117	0.14	0.0000	0.00	0.0000	0.00	0.0000	0.00	0.0063	0.22	0.0042	0.22
1983	0.0000	0.96	0.0227	0.16	0.0110	0.14	0.0000	0.22	0.0011	0.28	0.0000	0.00	0.0065	0.21	0.0043	0.21
1984	0.0264	0.96	0.0331	0.16	0.0127	0.14	0.0000	0.00	0.0014	0.26	0.0000	0.00	0.0081	0.21	0.0053	0.21
1985	1.3050	0.00	0.0364	0.17	0.0164	0.14	0.0001	0.20	0.0012	0.24	0.0000	0.00	0.0094	0.20	0.0062	0.20
1986	0.0000	0.00	0.0414	0.18	0.0135	0.14	0.0005	0.19	0.0009	0.23	0.0000	0.00	0.0086	0.20	0.0056	0.20
1987	0.0000	0.96	0.0548	0.19	0.0135	0.15	0.0015	0.19	0.0011	0.22	0.0000	0.00	0.0098	0.19	0.0064	0.19
1988	0.9770	0.00	0.0660	0.20	0.0108	0.15	0.0008	0.18	0.0013	0.21	0.0000	0.00	0.0093	0.19	0.0061	0.19
1989	0.0000	0.96	0.0390	0.21	0.0224	0.15	0.0008	0.17	0.0018	0.20	0.0000	0.00	0.0104	0.19	0.0067	0.19
1990	0.7487	0.00	0.0235	0.21	0.0258	0.15	0.0009	0.17	0.0014	0.19	0.0000	0.00	0.0098	0.18	0.0063	0.18
1991	0.0000	0.00	0.0324	0.22	0.0252	0.16	0.0020	0.17	0.0013	0.18	0.0000	0.00	0.0103	0.18	0.0066	0.18
1992	0.0000	0.00	0.0166	0.23	0.0125	0.16	0.0145	0.16	0.0017	0.17	0.0000	0.00	0.0106	0.18	0.0068	0.18
1993	0.0000	0.00	0.0141	0.23	0.0189	0.16	0.0107	0.17	0.0015	0.17	0.0000	0.00	0.0106	0.18	0.0067	0.18
1994	0.0000	0.00	0.0074	0.23	0.0133	0.16	0.0152	0.17	0.0014	0.17	0.0000	0.00	0.0103	0.18	0.0065	0.18
1995	0.0000	0.00	0.0054	0.23	0.0106	0.17	0.0123	0.17	0.0081	0.17	0.0000	0.00	0.0106	0.18	0.0066	0.18
1996	0.0000	0.00	0.0058	0.24	0.0099	0.17	0.0078	0.17	0.0128	0.17	0.0000	0.00	0.0103	0.18	0.0063	0.18
1997	0.0000	0.00	0.0087	0.24	0.0088	0.17	0.0069	0.17	0.0140	0.17	0.0000	0.00	0.0102	0.18	0.0062	0.18
1998	0.0000	0.00	0.0114	0.24	0.0058	0.17	0.0093	0.17	0.0108	0.18	0.0000	0.00	0.0095	0.18	0.0058	0.18
1999	0.0000	0.00	0.0094	0.24	0.0067	0.17	0.0088	0.17	0.0108	0.18	0.0000	0.00	0.0094	0.18	0.0056	0.18
2000	0.0000	0.00	0.0093	0.24	0.0075	0.17	0.0067	0.17	0.0085	0.18	0.0000	0.00	0.0081	0.18	0.0048	0.18
2001	0.0000	0.00	0.0086	0.25	0.0111	0.17	0.0082	0.17	0.0084	0.18	0.0000	0.00	0.0096	0.18	0.0056	0.18
2002	0.0000	0.00	0.0163	0.25	0.0066	0.17	0.0133	0.18	0.0067	0.18	0.0000	0.00	0.0103	0.18	0.0060	0.18
2003	0.0000	0.00	0.0087	0.25	0.0090	0.18	0.0175	0.18	0.0038	0.18	0.0000	0.00	0.0110	0.18	0.0063	0.18
2004	0.0000	0.00	0.0062	0.25	0.0069	0.18	0.0165	0.18	0.0058	0.18	0.0000	0.00	0.0106	0.18	0.0060	0.18
2005	0.0000	0.00	0.0094	0.26	0.0017	0.18	0.0154	0.18	0.0036	0.18	0.0000	0.00	0.0083	0.19	0.0047	0.19
2006	0.0000	0.00	0.0052	0.26	0.0012	0.18	0.0181	0.19	0.0034	0.18	0.0000	0.00	0.0089	0.19	0.0049	0.19
2007	0.0000	0.00	0.0011	0.26	0.0042	0.18	0.0190	0.19	0.0043	0.18	0.0000	0.00	0.0100	0.19	0.0055	0.19
2008	0.0000	0.00	0.0030	0.26	0.0047	0.18	0.0193	0.19	0.0041	0.18	0.0000	0.00	0.0102	0.19	0.0056	0.19
Min	0.0000	0.00	0.0011	0.15	0.0012	0.13	0.0000	0.00	0.0000	0.00	0.0000	0.00	0.0045	0.18	0.0031	0.18
Median	0.0000	0.00	0.0146	0.23	0.0107	0.16	0.0068	0.17	0.0016	0.18	0.0000	0.00	0.0095	0.19	0.0059	0.19
Mean	0.1099	0.19	0.0193	0.21	0.0113	0.16	0.0069	0.16	0.0039	0.16	0.0000	0.00	0.0090	0.19	0.0056	0.19
Max	1.3050	0.96	0.0660	0.26	0.0258	0.18	0.0190	0.25	0.0140	0.28	0.0000	0.00	0.0110	0.24	0.0068	0.24

Table B23. Biological reference points from per recruit models for ocean quahogs. Reference points from model runs with natural mortality M=0.02 y^{-1} are for potential use by managers. Results with M=0.015 and 0.025 are for sensitivity analyses.

Policy	Fishing mortality rate (F)	Yield per recruit (g)	Spawning biomass per recruit (g)	Total biomass per recruit (g)
***M*=0.015**				
F=0	0.0000	0.00	1124	1341
F_{MAX}	0.0540	9.54	215	346
$F_{0.1}$	0.0220	8.53	431	592
$F_{25\%}$	0.0390	9.41	282	425
$F_{40\%}$	0.0200	8.31	459	623
$F_{45\%}$	0.0170	7.89	507	676
$F_{50\%}$	0.0140	7.32	566	740
$F_{55\%}$	0.0110	6.56	638	819
$F_{60\%}$	0.0090	5.89	696	882
***M*=0.02**				
F=0	0.0000	0.00	704	877
F_{MAX}	0.0759	7.52	129	234
$F_{0.1}$	0.0277	6.59	275	407
$F_{25\%}$	0.0517	7.39	176	292
$F_{40\%}$	0.0266	6.51	282	415
$F_{45\%}$	0.0219	6.11	317	454
$F_{50\%}$	0.0180	5.67	353	495
$F_{55\%}$	0.014	5.05	399	545
$F_{60\%}$	0.0120	4.66	426	575
***M*=0.025**				
F=0	0.0000	0.00	466	608
F_{MAX}	0.1030	6.11	82	169
$F_{0.1}$	0.0360	5.34	179	289
$F_{25\%}$	0.0660	5.98	117	214
$F_{40\%}$	0.0330	5.21	189	300
$F_{45\%}$	0.0270	4.87	212	327
$F_{50\%}$	0.0220	4.49	237	355
$F_{55\%}$	0.0180	4.09	261	382
$F_{60\%}$	0.015	3.72	282	406

Table B24. Input parameters for length based per recruit models used to estimate biological reference points for ocean quahog. The shell height-meat weight relationship is $W = e^{\alpha+\beta \ln L}$ where W is meat weight in grams and L is shell height (mm). Meat weights are in grams. Logistic functions for maturity and fishery selectivity at length were $p_L = 1\Big/\left[1+e^{-(\alpha+\beta L)}\right]$ where L is shell height in mm and p_L is the corresponding proportion.

Parameter	**Value**
von Bertalanffy growth curve	
L_∞	97.28
K	0.0311
Shell height-meat weight relationship	
ln(α)	-9.258
β	2.825
Natural mortality (M)	0.02
Logistic fishery selectivity at size	
α	-7.63
β	0.105
Logistic maturity at size	
α	-5.92
β	0.0927

Table B25. Factors considered in choosing an F_{MSY} proxy for ocean quahogs between $F_{40\%}$ and $F_{50\%}$.

Factors affecting MSY estimates for fishable quahogs	Groundfish proxy ($F_{40\%}$)	Less resilient than groundfish proxy ($F_{50\%}$)
Temporal recruitment pattern (regularity)		x
Accurate catch data	x	
Low bycatch mortality	x	
Long time lags between spawning and recruitment to the fishery and spawning stock		x
Heterogeneous fishing patterns	x	x
Longevity		x
Mature before entering the fishery	x	
Slow growth		x
Time to fix errors if we are wrong	x	

Table B26. Stochastic projection results for ocean quahogs in 2015 with natural mortality M=0.02 under various constant quotas. Starting biomass levels in 2008 are from a bootstrap analysis (1673 iterations) with the KLAMZ model ocean quahogs in the exploited area. Biomass on GBK was assumed constant at the 2008 estimate. Actual landings were used in simulations for 2008 and expected landings (3.8 million bushels or 17.2 mt meats) were used for 2009. For 2010-2015, simulated managers specified a constant level of annual landings (quota) based on a harvest policy. Quotas are calculated by multiplying the target fishing mortality times the current best estimate of biomass during 2008, where the biomass estimate is for either the exploited or entire stock area. Simulated catches were equal to the quota plus 5% to account for incidental mortality. Probabilities of overfished stock conditions ($B_{2015} \leq B_{Threshold}$) and probabilities of overfishing ($F_{2015} \geq F_{45\%}$) in 2015 are shown in the last three columns. The probability of overfishing is for either the exploited stock (F_{2015} for exploited stock $\geq F_{45\%}$) or the entire stock (F_{2015} for entire stock $\geq F_{45\%}$).

How are the landings calculated? (alternative management actions, under constant annual removal)	Annual landings 2010-2015 (million bushels)	Annual landings 2010-2015 (1000 mt meats)	Probability overfished in 2015 ($B_{2015} \leq B_{Threshold}$)	Probability of overfishing for exploited stock in 2015 (F_{2015} for exploited stock $\geq F_{45\%}$)	Probability of overfishing for entire stock in 2015 (F_{2015} for entire stock $\geq F_{45\%}$)
Status quo landings	3.8	17.2	0	0.00	0.00
Current quota	5.3	24.2	0	0.19	0.00
FMP min landings	4.0	18.1	0	0.00	0.00
FMP max landings	6.0	27.2	0	0.54	0.00
Recommended F threshold ($F_{45\%}$) x 2008 biomass in exploited area	7.7	34.8	0	0.90	0.00
Current F target ($F_{0.1}$) x 2008 biomass in exploited area	9.7	44.0	0	0.99	0.00
Current F threshold ($F_{25\%}$) x 2008 biomass in exploited area	18.1	82.2	0	1.00	1.00
Recommended F threshold ($F_{45\%}$) x biomass in entire area	14.0	63.7	0	1.00	0.97
Current F target ($F_{0.1}$) x biomass in entire area	17.8	80.6	0	1.00	1.00
Current F threshold ($F_{25\%}$) x biomass in entire area	33.1	150.4	0	1.00	1.00

Table B27. Probabilities of overfishing and overfished stock status by 2015 for ocean quahogs under various harvest policies and three states of nature (M=0.015, 0.02 and 0.025) based on stochastic projection analyses for 2008-2015. Actual landings were used for 2008 and expected landings were used for 2009. For 2010-2015, simulated managers specify annual landings in terms of a constant landings policy (e.g. status-quo landings) or by multiplying an F based reference point (e.g. F20%) times the best estimate of stock biomass in 2008, where the biomass estimate may be for either the whole stock or the exploited stock only. The specified level of annual landings (+ 5% for incidental mortality) is then extracted from the simulated population during 2010-2015. Figures on the left side of the figure describe management actions (harvest policies) and calculation of annual landings during 2010-2015. Figures on the right hand side of the figure give the probability of overfishing for the exploited stock and the entire stock relative to the true mortality threshold $F_{45\%}$, as well as the probability of overfished stock conditions for the whole stock relative to the assumed true biomass threshold $B_{threshold} = 0.4B_{1978}$. The mortality and biomass thresholds depend on the state of nature because $F_{45\%}$ and B_{1978} depend on M. Probabilities equal zero are not shown to enhance the readability of the table. Figures above the dash line are for constant landings policies. Figures below the dashed line are for F based harvest policies.

Harvest policies (management actions)							States of nature								
							M=0.015			M=0.02			M=0.025		
Policy	Reference point F	Stock area for target landings	Best estimate 2008 biomass for catch calculations	Landings (million bushels)	Landings (1000 mt meats)	Landings + indicidental mortality (1000 mt meats)	Biomass	F for exploit. stock	F whole stock	Biomass	F for exploit. stock	F whole stock	Biomass	F for exploit. stock	F whole stock
Current quota		NA	NA	5.33	1.175	1.234	0.00	0.68	0.00	0.00	0.19	0.00	0.00	0.01	0.00
FMP max landings	NA	NA	NA	6.00	1.323	1.389	0.00	0.86	0.00	0.00	0.54	0.00	0.00	0.12	0.00
FMP min landings	NA	NA	NA	4.00	0.882	0.926	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Status quo landings	NA	NA	NA	3.80	0.838	0.880	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F0.1	0.0277	Whole	2,908	17.76	80.557	84.584	0.00	1.00	1.00	0.00	1.00	1.00	0.00	1.00	1.00
F25%	0.0517	Whole	2,908	33.15	150.353	157.871	0.00	1.00	1.00	0.00	1.00	1.00	0.00	1.00	1.00
F40%	0.0266	Whole	2,908	17.05	77.358	81.226	0.00	1.00	1.00	0.00	1.00	1.00	0.00	1.00	0.99
F45%	0.0219	Whole	2,908	14.04	63.689	66.874	0.00	1.00	1.00	0.00	1.00	0.97	0.00	1.00	0.70
F50%	0.0180	Whole	2,908	11.54	52.347	54.965	0.00	1.00	0.98	0.00	1.00	0.60	0.00	1.00	0.00
F0.1	0.0277	Exploitable	1,589	9.70	44.015	46.216	0.00	1.00	1.00	0.00	0.99	0.00	0.00	0.96	0.00
F25%	0.0517	Exploitable	1,589	18.11	82.151	86.259	0.00	1.00	1.00	0.00	1.00	1.00	0.00	1.00	1.00
F40%	0.0266	Exploitable	1,589	9.32	42.267	44.381	0.00	1.00	1.00	0.00	0.99	0.00	0.00	0.93	0.00
F45%	0.0219	Exploitable	1,589	7.67	34.799	36.539	0.00	0.99	0.01	0.00	0.90	0.00	0.00	0.73	0.00
F50%	0.0180	Exploitable	1,589	6.31	28.602	30.032	0.00	0.90	0.00	0.00	0.65	0.00	0.00	0.24	0.00

Table B28. Harvest policies (management actions) considered in projection analyses for ocean quahogs. Constant landings policies are shown with the corresponding approximate true F for the whole and exploited stock components. Constant F policies are shown with the corresponding landings level determined by multiplying the target *F* by the biomass for the whole stock in 2008.

Whole stock 2008 biomass (1000 mt meats)		2,908		
Constant landings policies				
Policy (management action)	F (whole stock)	Landings Million bu	Thousand mt meats	F exploited stock (for comparison)
Status quo landings	0.006	3.80	17.24	0.011
FMP maximum landings	0.009	6.00	27.22	0.017
FMP minimum landings	0.006	4.00	18.14	0.012
FMP current landings quota	0.008	5.33	24.18	0.015
Constant F policies				
F0.1 (current target)	0.028	17.76	80.56	0.052
F25% (current threshold)	0.052	33.15	150.35	0.100
F40%	0.027	17.05	77.36	0.050
F45% (recommended target)	0.022	14.04	63.69	0.041
F50%	0.018	11.54	52.35	0.034

Table B29. Input data used in simple projection analyses for ocean during 2009-2015.

Year	SVA	DMV	NJ	LI	SNE	GBK	Total Less GBK	Total
Somatic growth rate (G y^{-1})								
2008	1.05011E-07	1.05011E-07	0.00122	0.00792	0.00841	0.01116	0.00588	0.00837
Recruitment rate (r = Recruitment / Average Biomass in 2005 y^{-1})								
2008	0	1.0686E-08	0.00142	0.00002	0.00002	0.01182	0.00035	0.00548
Natural mortality (M y^{-1})								
2008	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Initial biomass proportions by region								
2008	4.46199E-06	0.03151	0.12819	0.20270	0.18399	0.45361	0.54639	1.00000
Proportions landings and catch by region								
2008	0	0.01766	0.11345	0.72807	0.14081	0.00000	1.00000	1.00000

FIGURES

B. Stock assessment for ocean quahogs (*Arctica islandica*)

Invertebrate Subcommittee
SAW/SARC 48

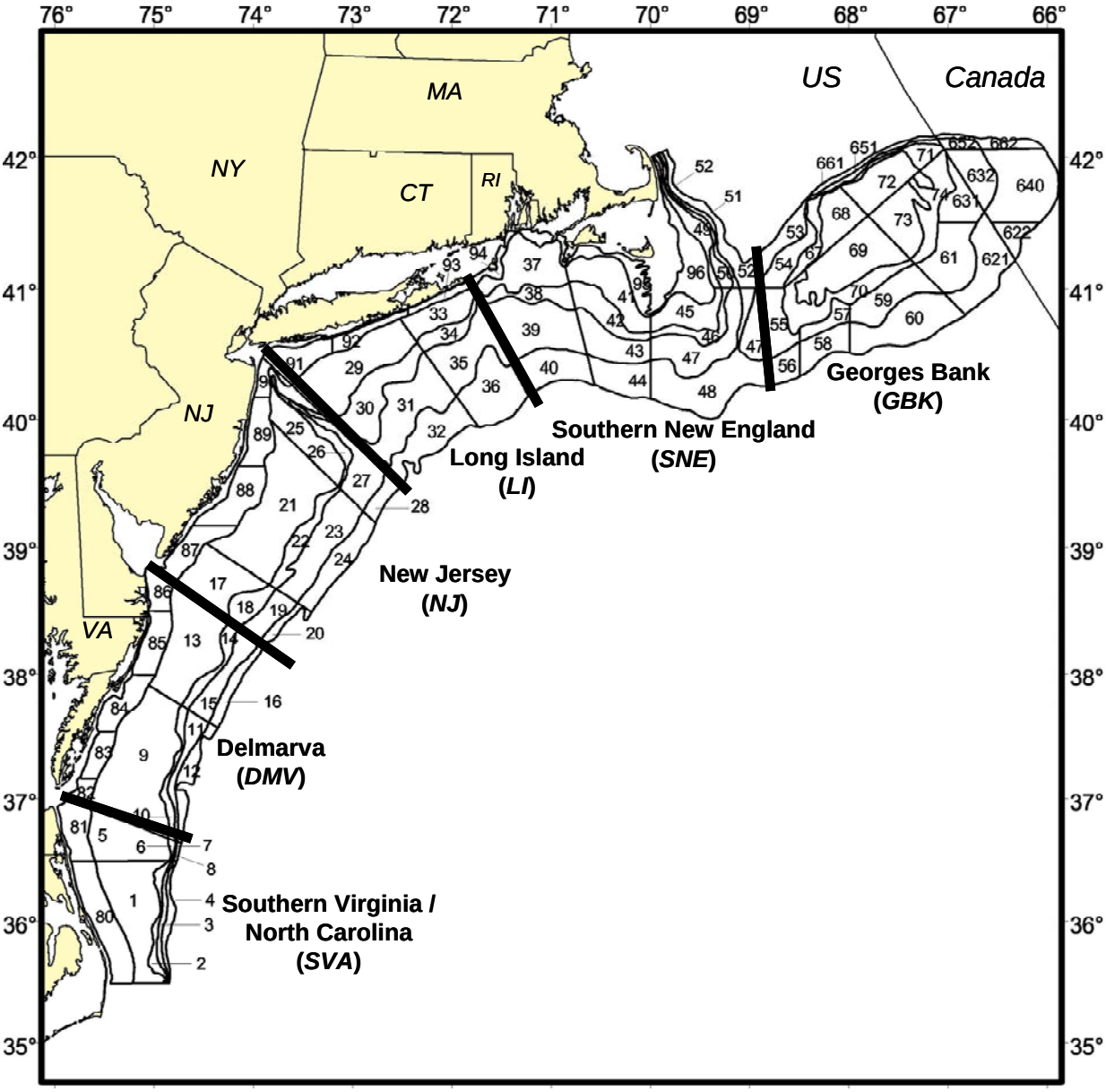


Figure B1. Stock assessment regions for ocean quahog in the US EEZ, with NEFSC shellfish survey strata boundaries.

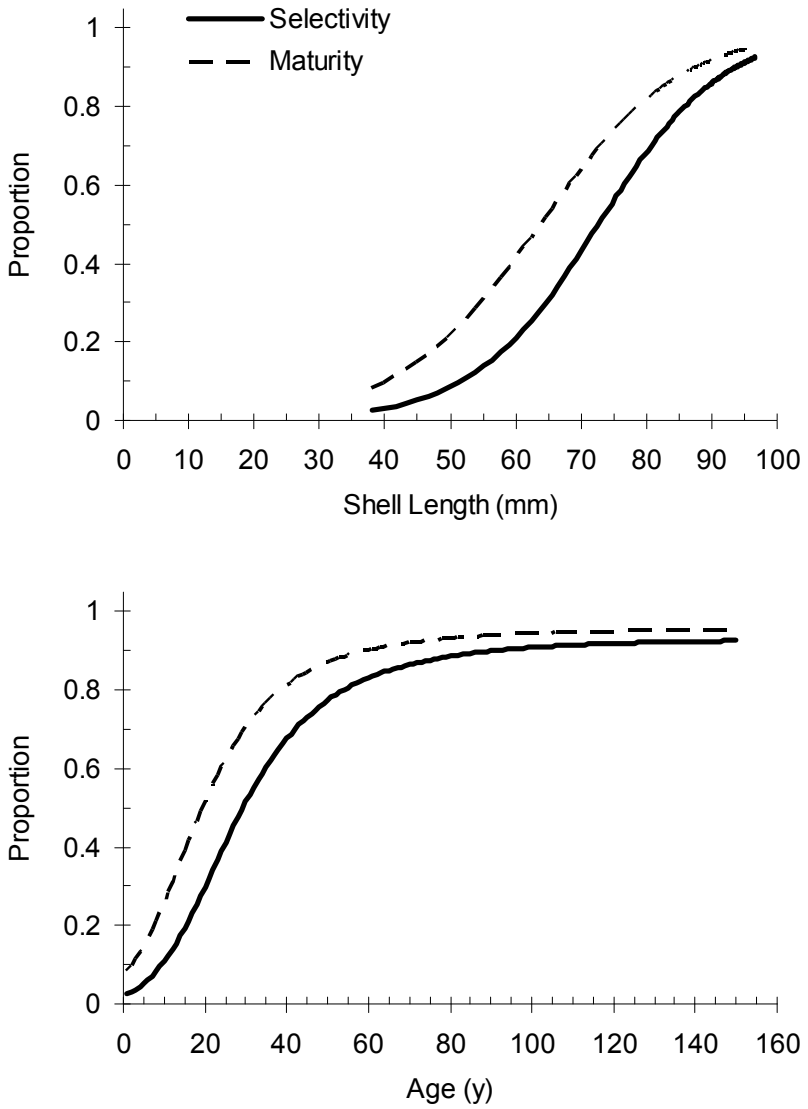


Figure B2. Commercial size-selectivity and maturity by length (*top panel*) and by age (*bottom panel*) assuming the von Bertalanffy growth curve for ocean quahogs in MAB (exploited region). Estimates in upper panel are from Thorarinsdottir and Jacobson, 1995).

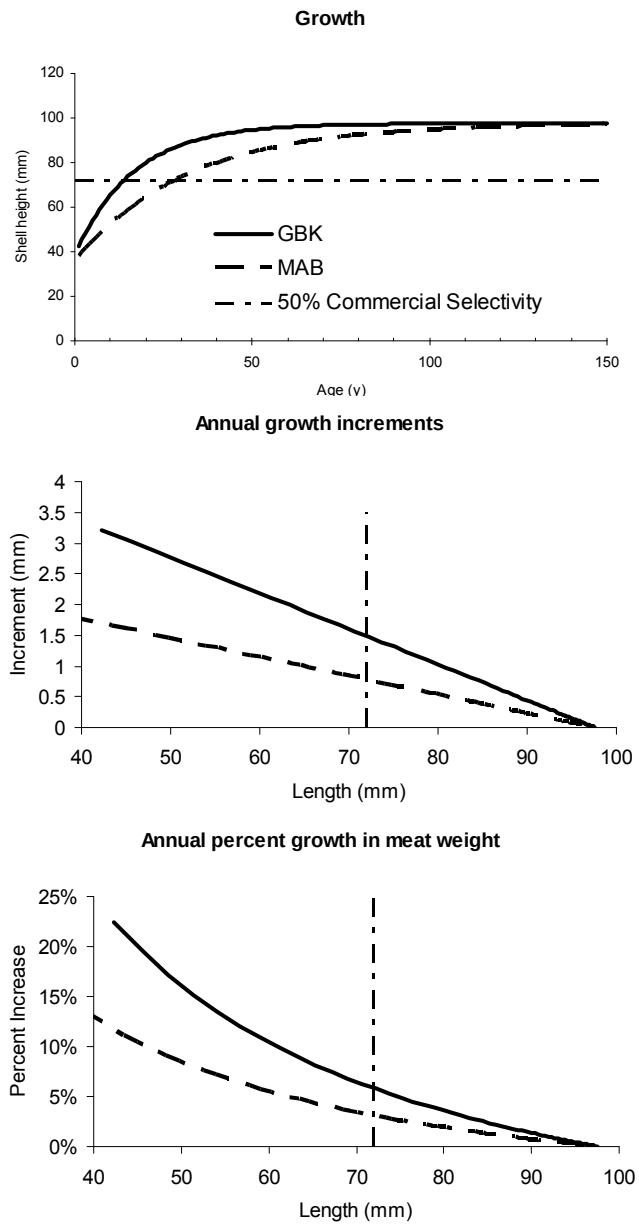


Figure B3. Growth, annual growth increments and percent annual change in meat weights for ocean quahog in GBK and in the Mid-Atlantic Bight (MAB) based on von Bertalanffy growth curves (Lewis et al., 2001) and shell length-meat weight relationships. The growth curve for MAB is used in this assessment for the exploited ocean quahog stock (which excludes GBK).

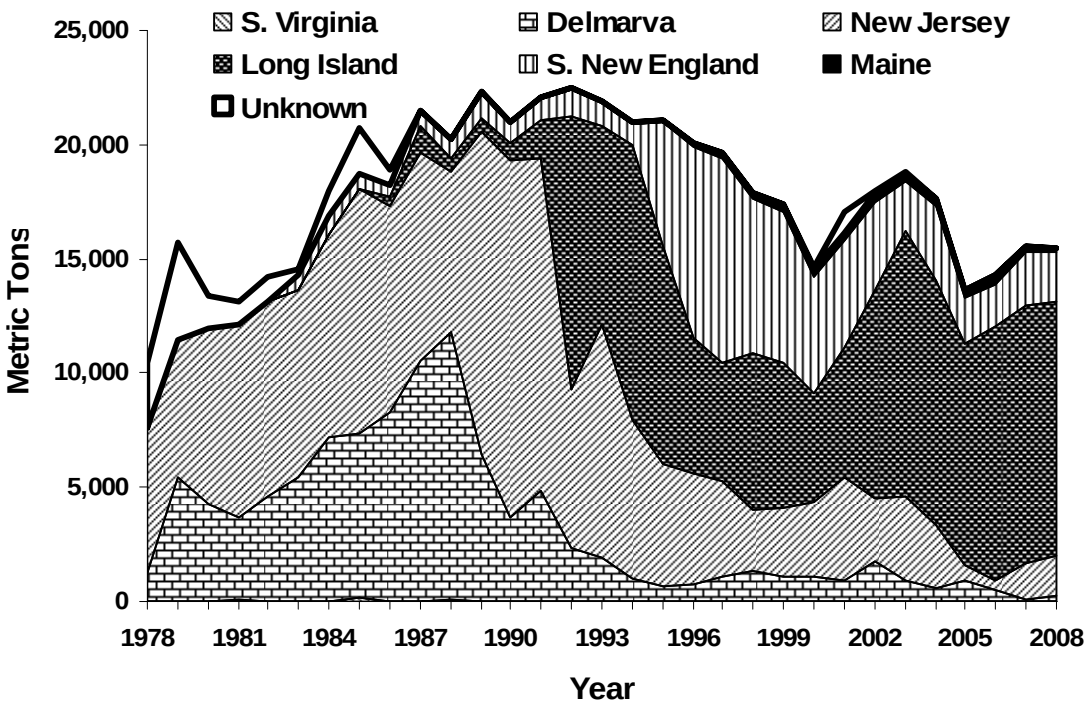
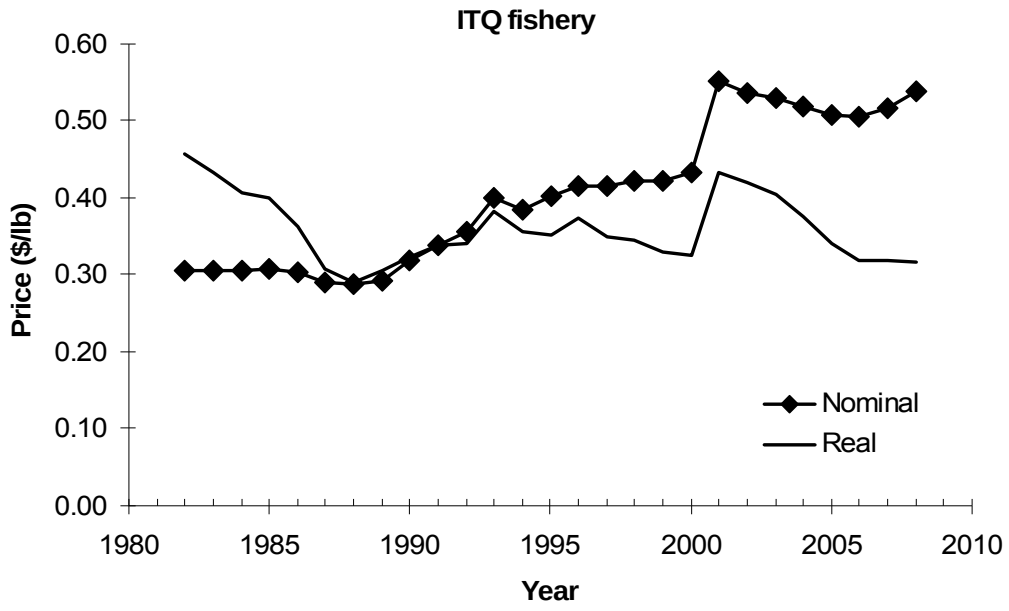


Figure B4. Ocean quahog commercial landings (in metric tons meat weights) from the US EEZ during 1978-2008. Landings in the SVA (S. Virginia) area are too small to be visible in the figure.



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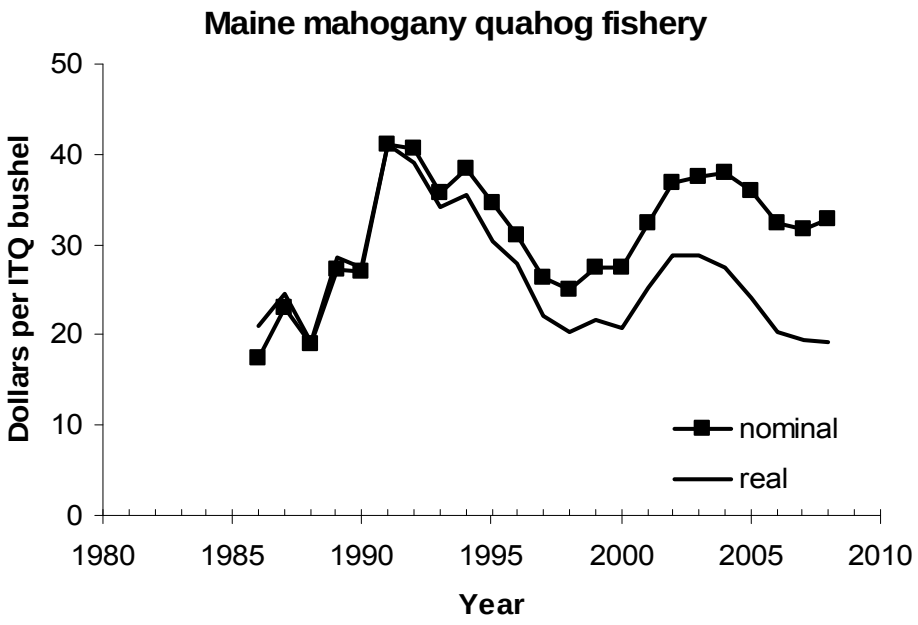


Figure B5. Real and nominal ex-vessel prices (total revenue/total landings) for the ITQ and Maine ocean quahog fisheries. Real prices are 1991 dollars.

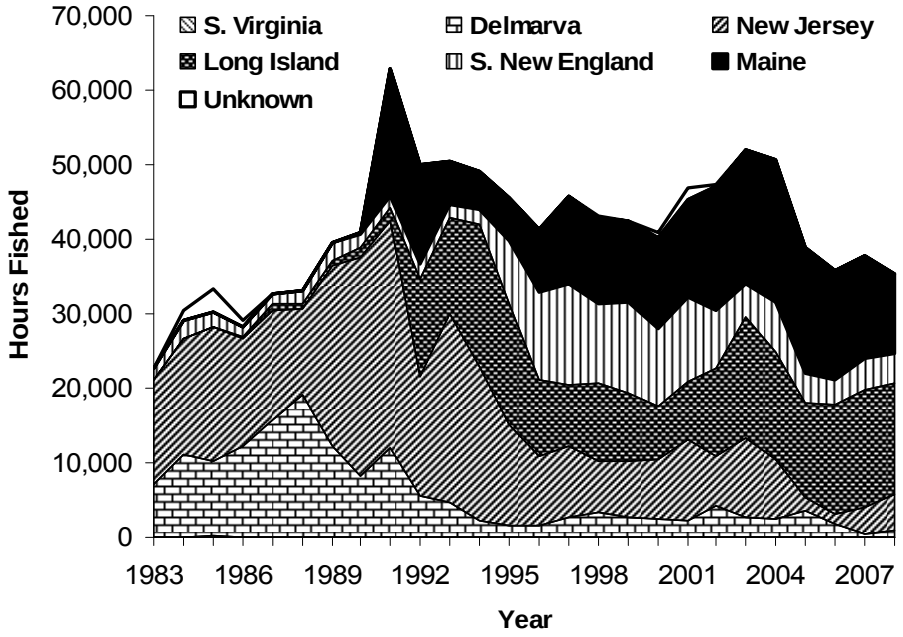


Figure B6. Hours fished for ocean quahog in the US EEZ during 1983-2008 based on logbook records. Hours fished in the SVA (S. Virginia) area are too small to be visible in the figure.

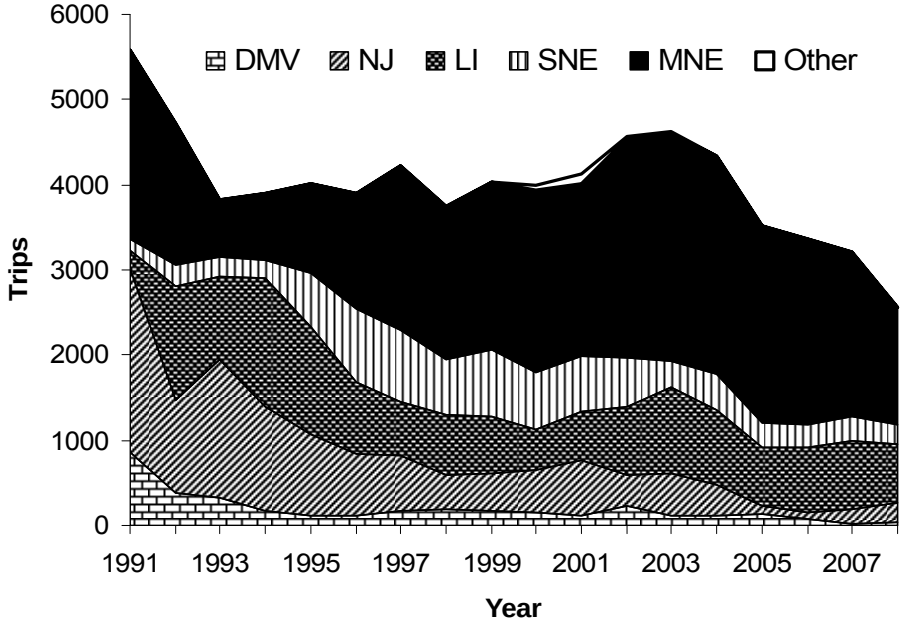


Figure B7. Number of trips for ocean quahog in the US EEZ during 1991-2008 based on logbook records.

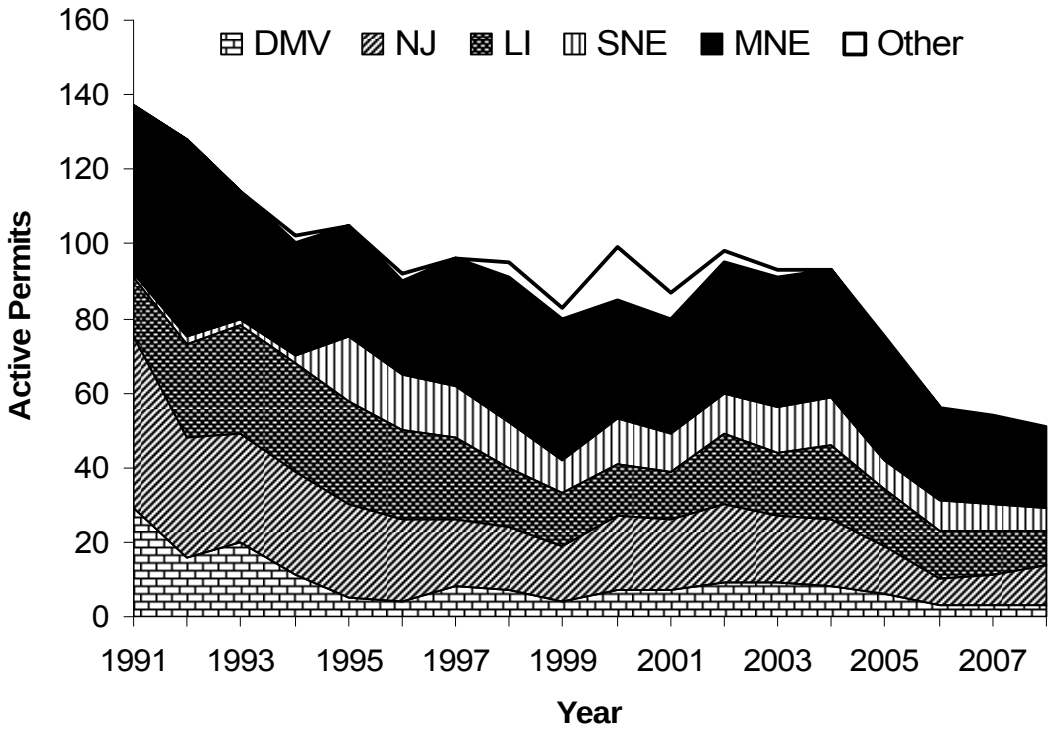


Figure B8. Number of active permits (fishing vessels) for ocean quahog in the US EEZ during 1991-2008 based on logbook records. The total number of permits in the graph for any year may exceed the total number of active permits in the fishery because some vessels fished in more than one area.

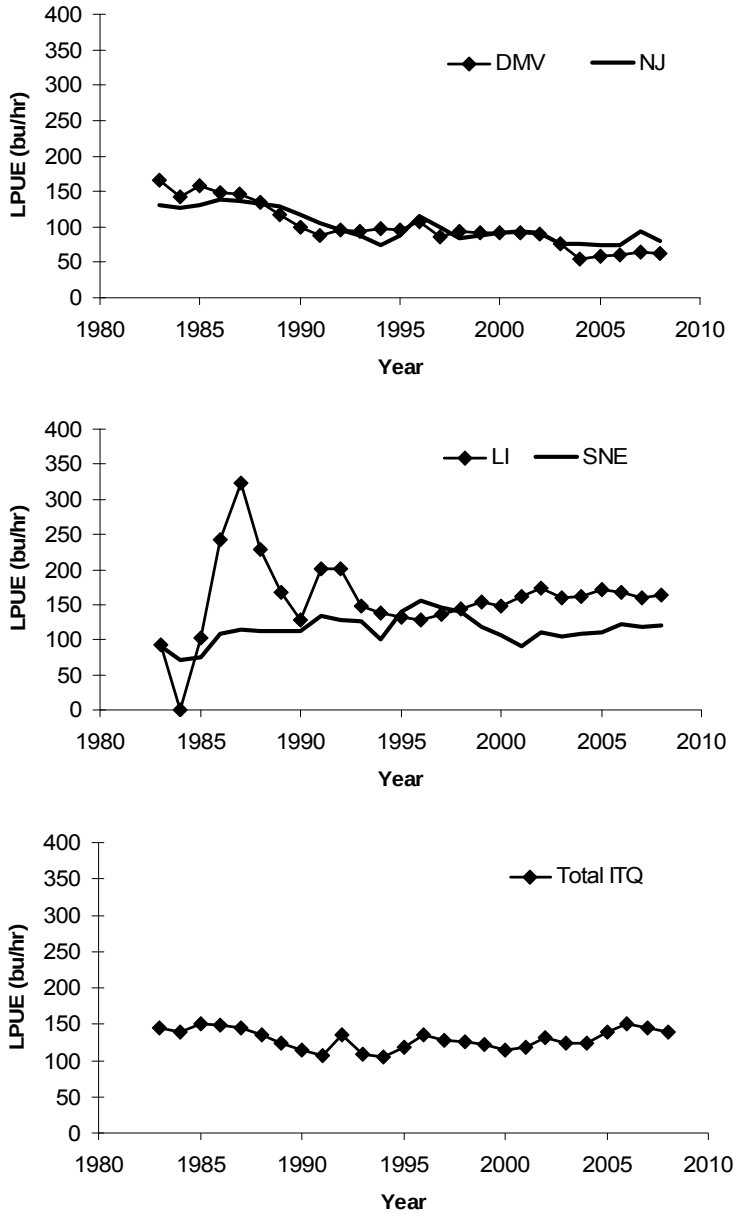


Figure B9. Trends in nominal LPUE for ocean quahog during 1980-2008 by region.

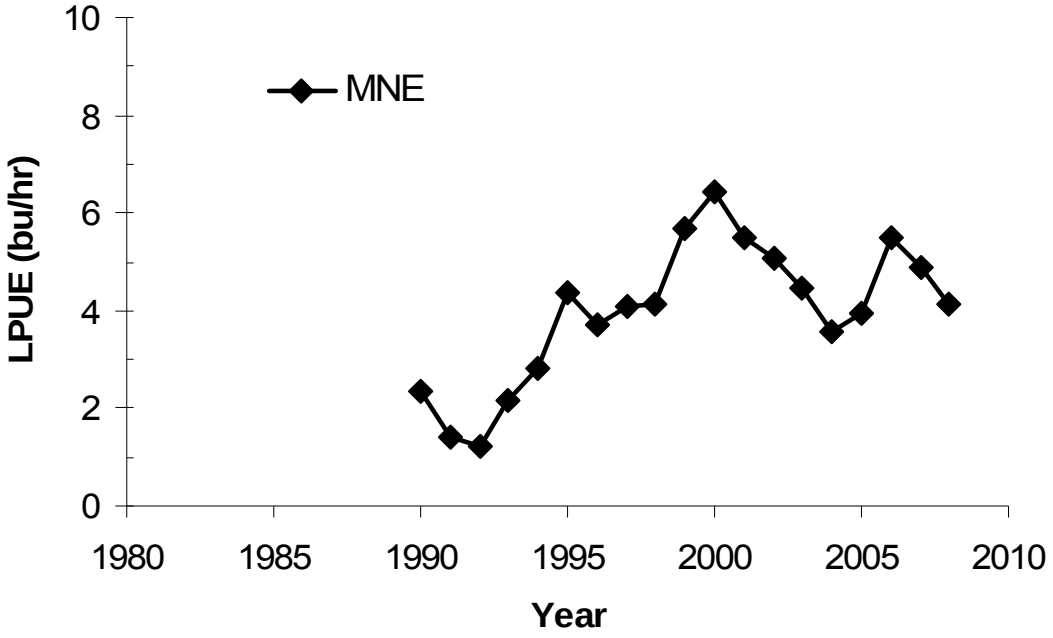


Figure B10. Nominal LPUE (ITQ bushels per hour) in the Maine ocean quahog fishery.

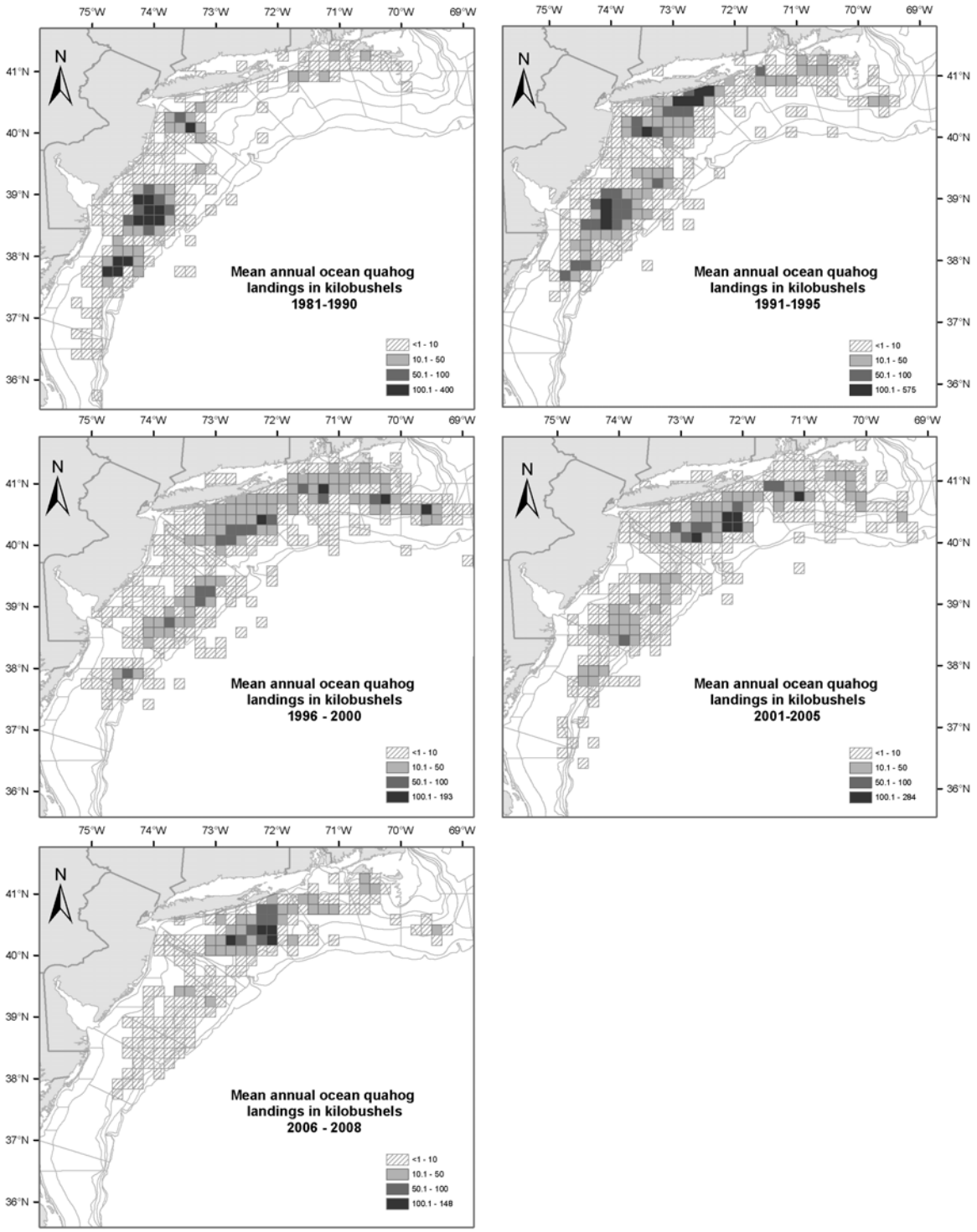


Figure B11. Spatial patterns in average annual landings (1000 ITQ bushels per year) for ocean quahog from logbook records. Data in TNMS far offshore reflect errors in logbook data.

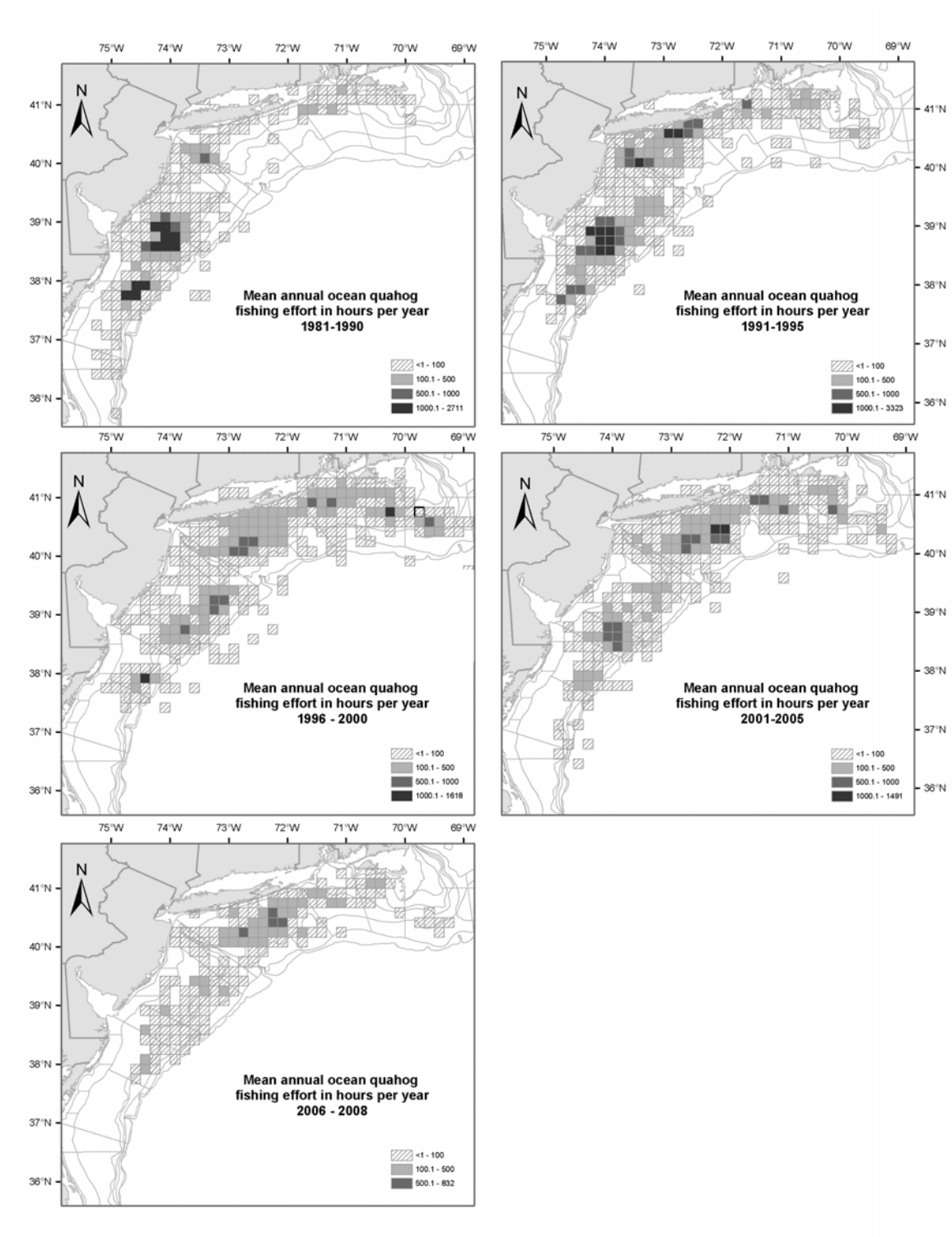


Figure B12. Spatial patterns in average annual fishing effort (hours fished per year) for ocean quahog from logbook records. Data in TNMS far offshore reflect errors in logbook data.

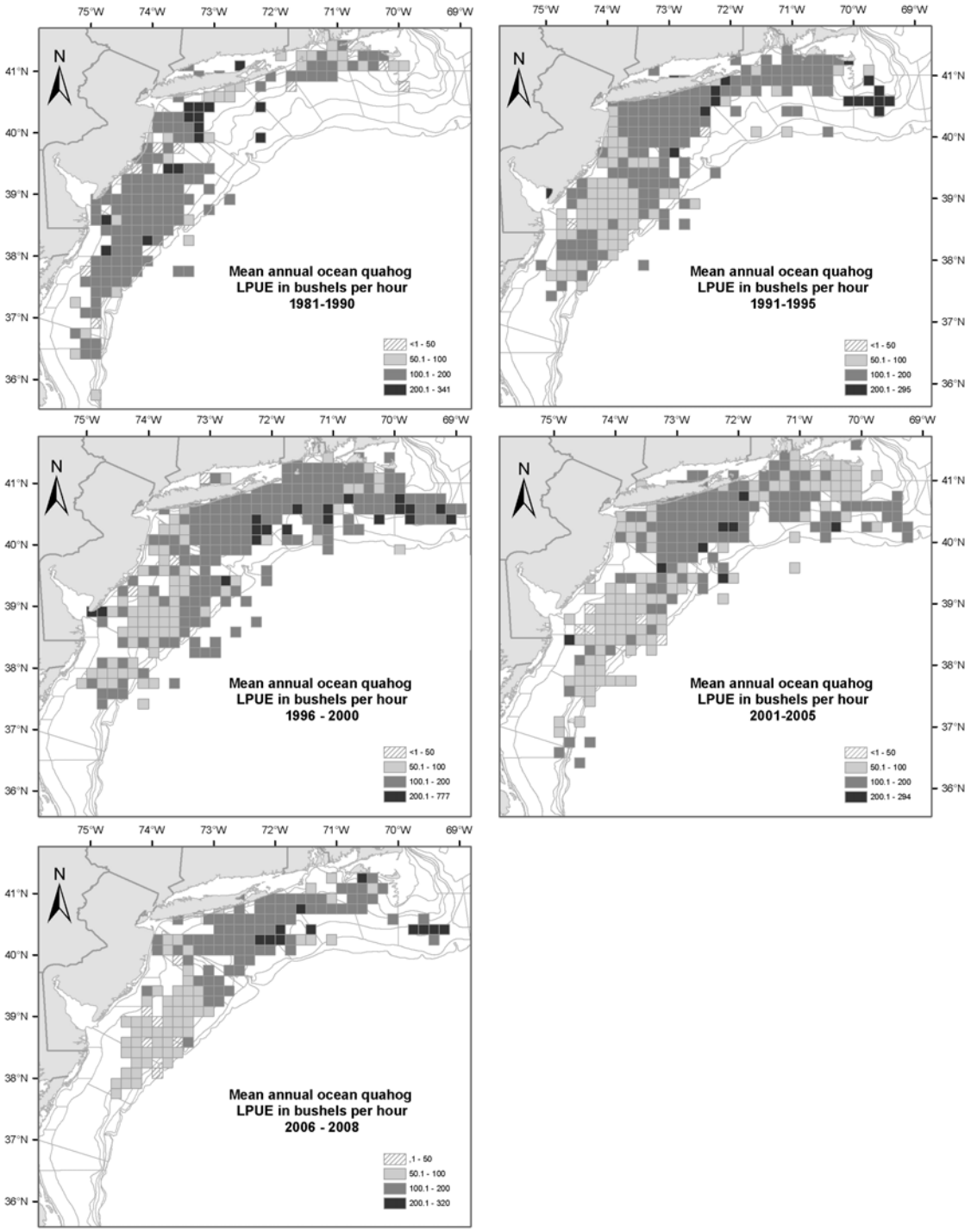


Figure B13. Spatial patterns in average LPUE (ITQ bushels per hours fished) for ocean quahog from logbook records. Data in TNMS far offshore reflect errors in logbook data.

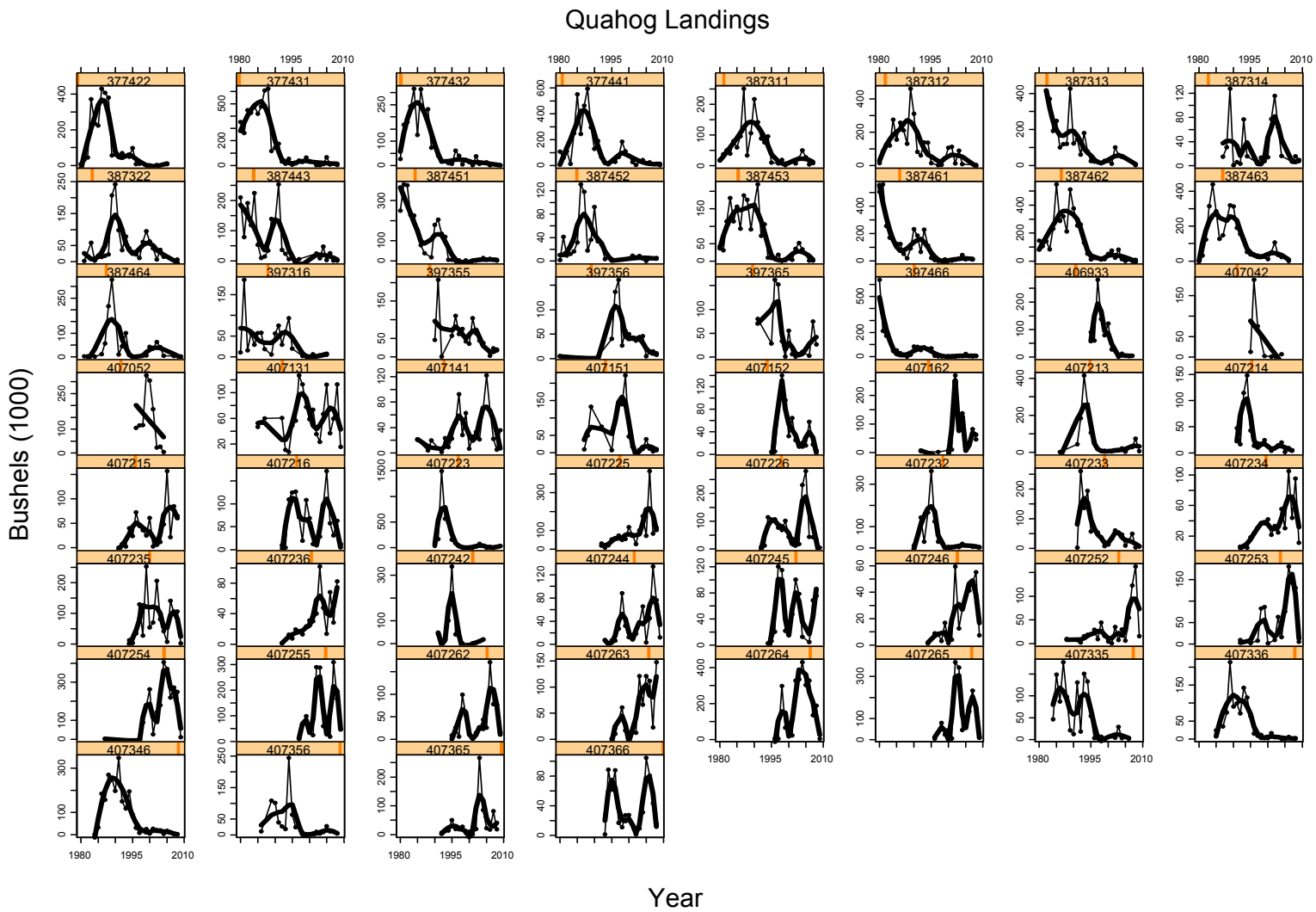


Figure B14. Trends in total annual landings (ITQ bu per year, vessel ton class 3-4) for ocean quahog in important TNMS during 1980-2008.

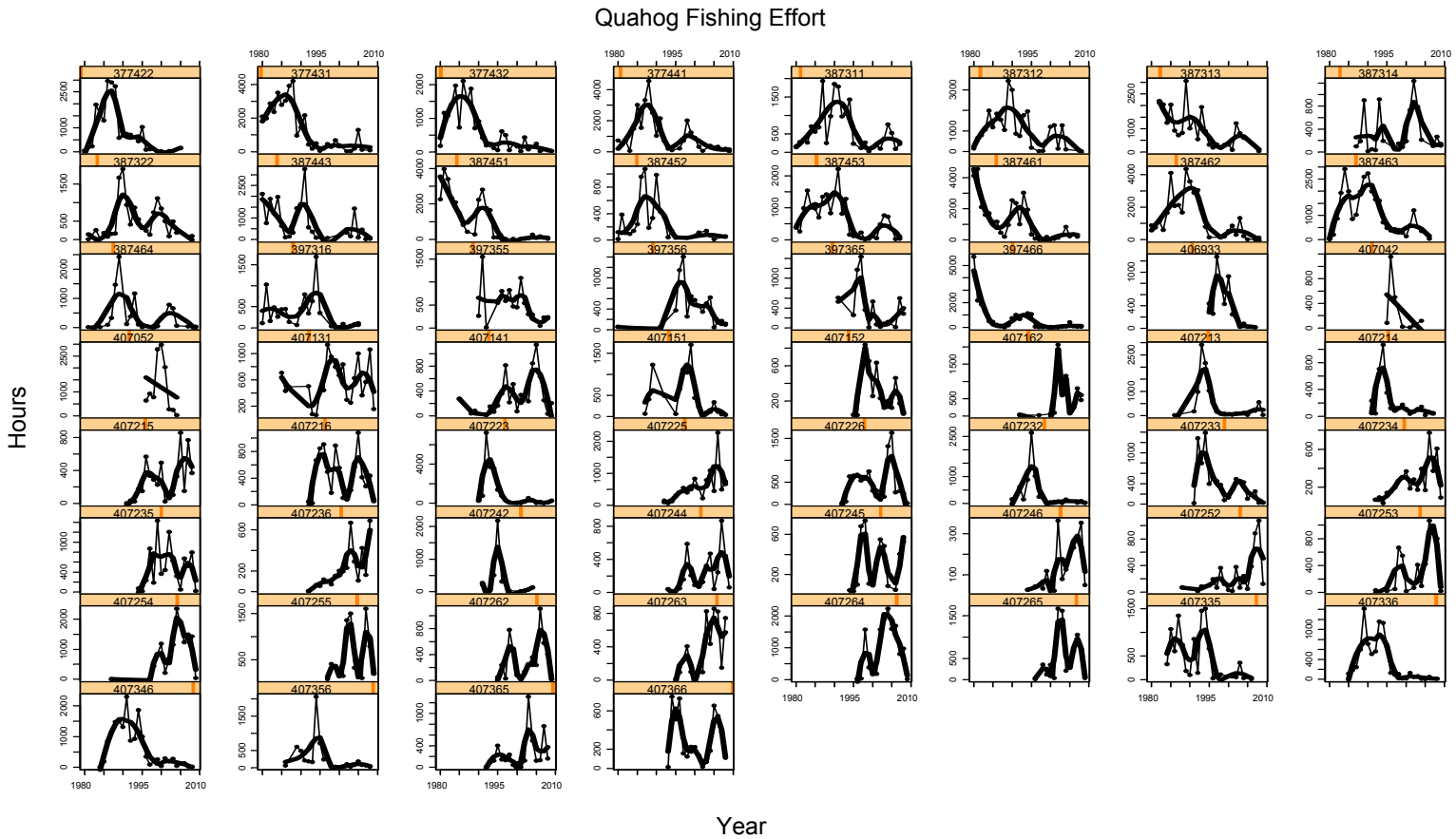


Figure B15. Trends in total annual fishing effort (hours fished per year, vessel ton class 3-4) for ocean quahog in important TNMS during 1980-2008.

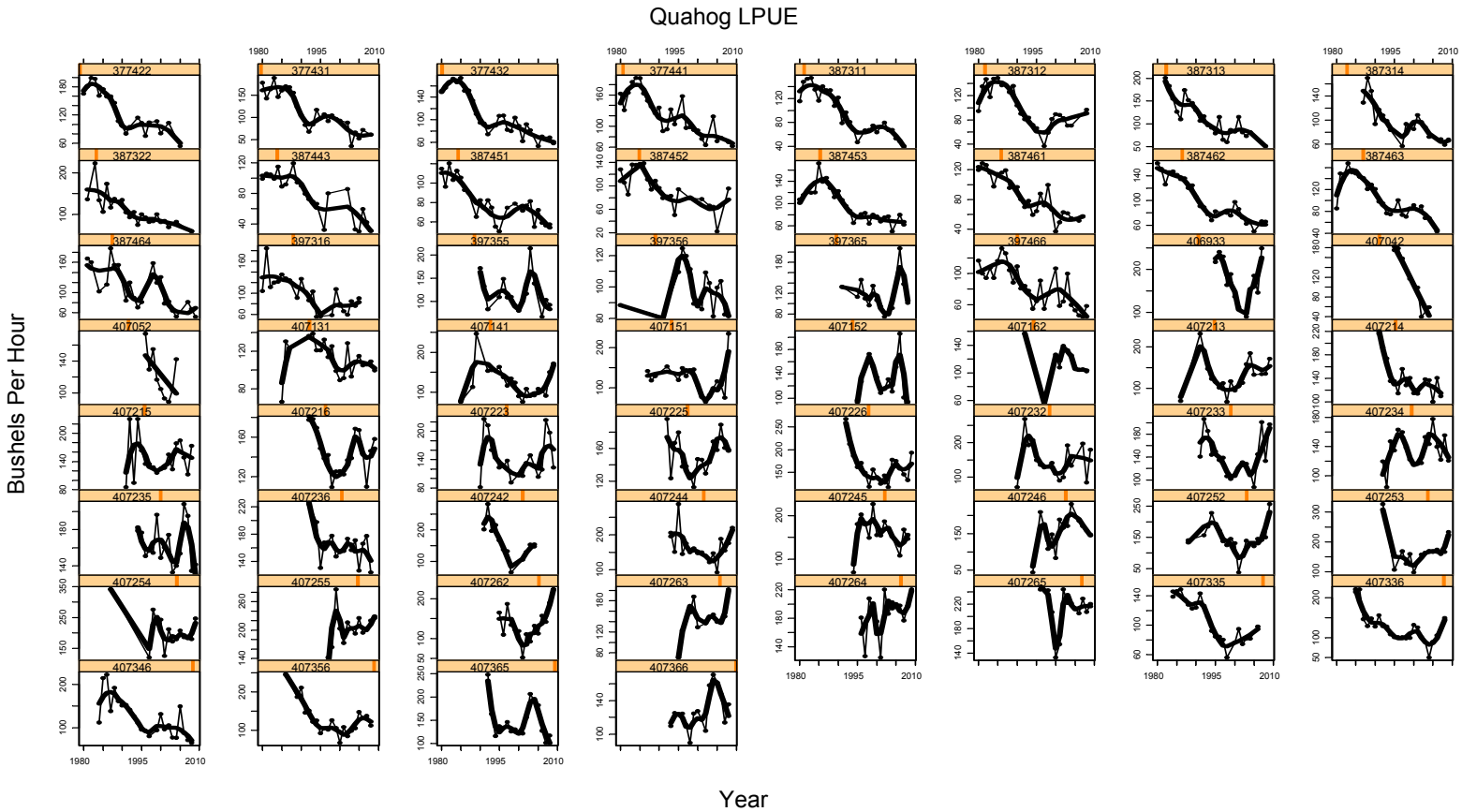


Figure B16. Trends in annual LPUE (ITQ bu h^{-1}, total landings/total hours fished) for ocean quahog in important TNMS during 1980-2008.

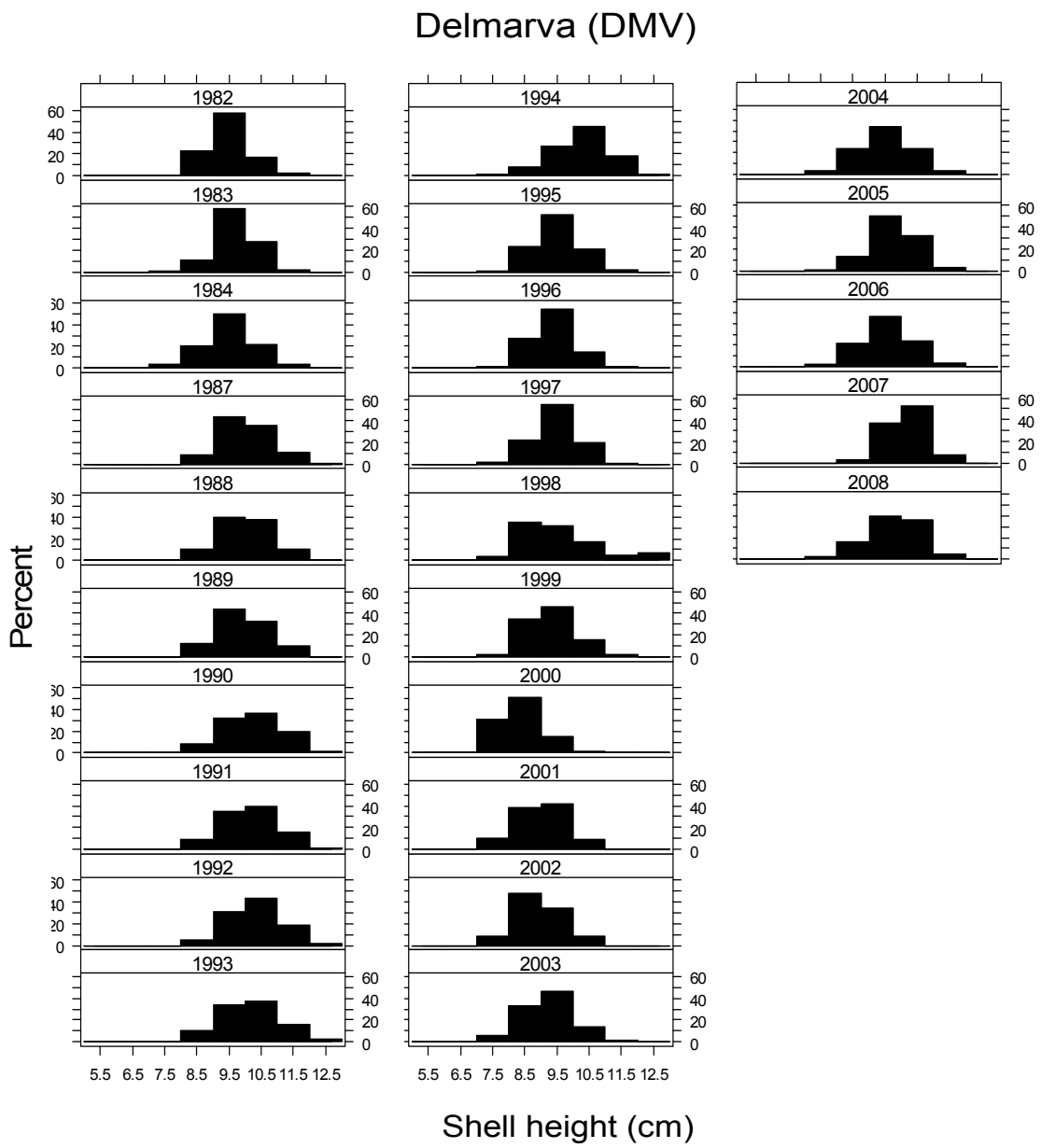


Figure B17. Commercial length composition data for ocean quahogs landed in the DMV region.

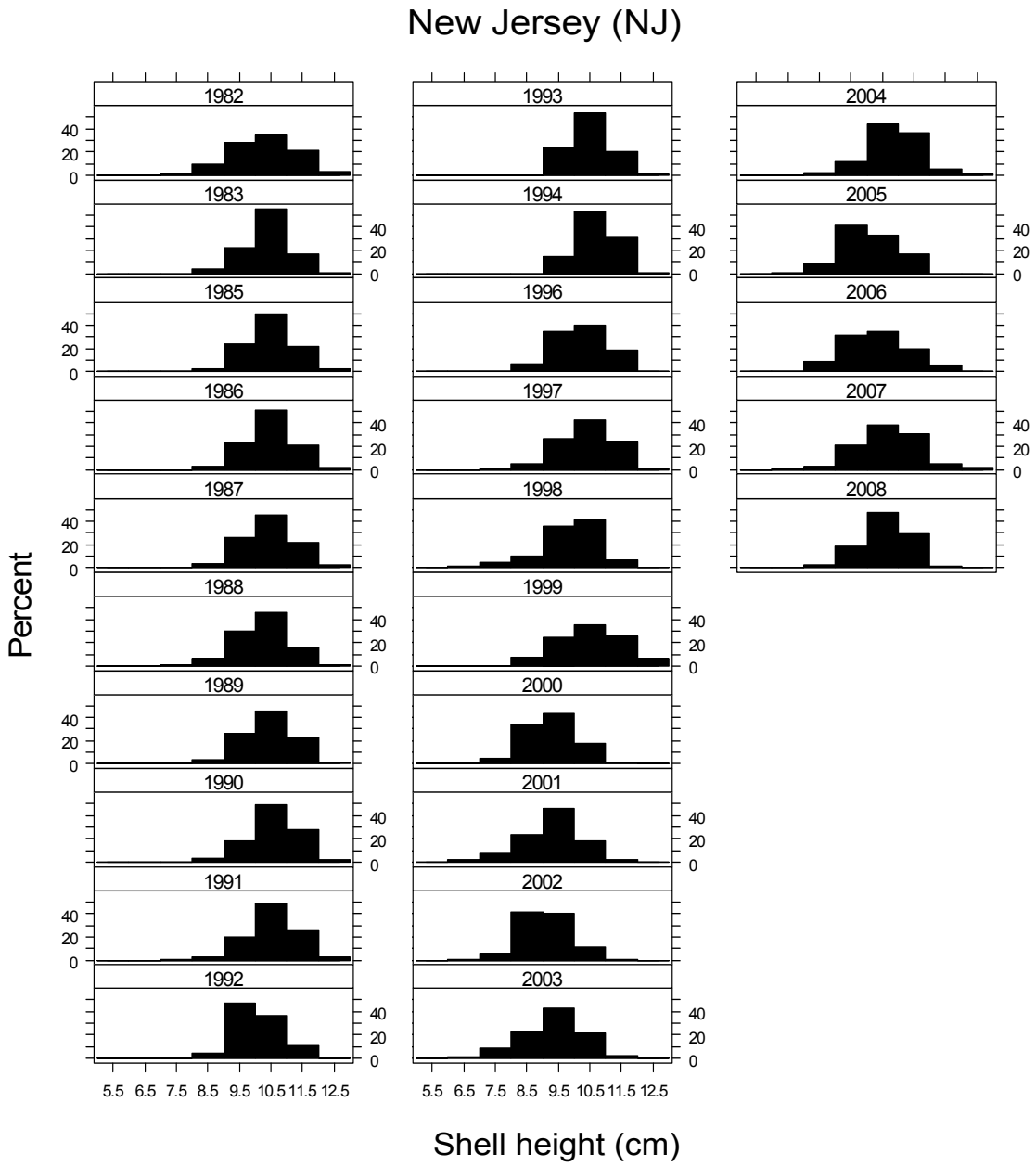


Figure B18. Commercial length composition data for ocean quahogs landed in the NJ region.

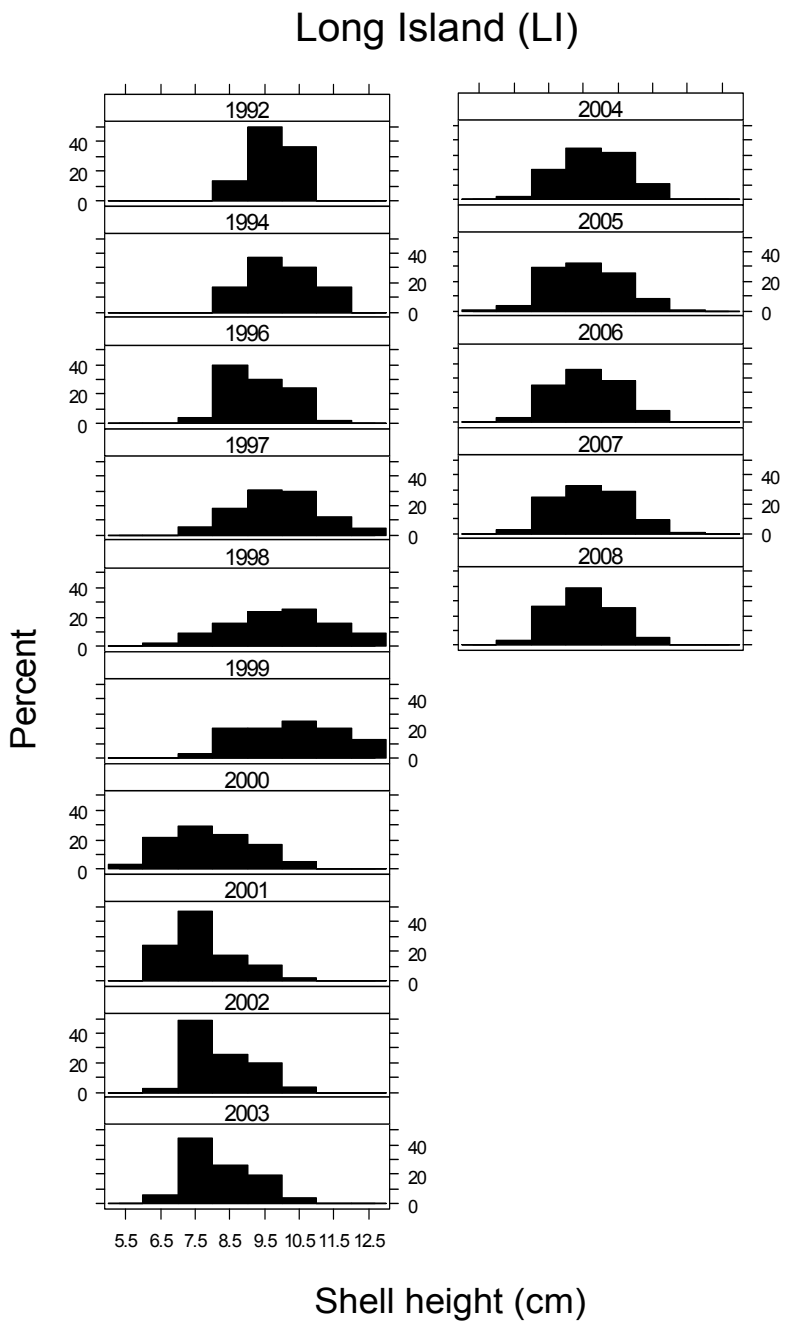


Figure B19. Commercial length composition data for ocean quahog landed in the LI region.

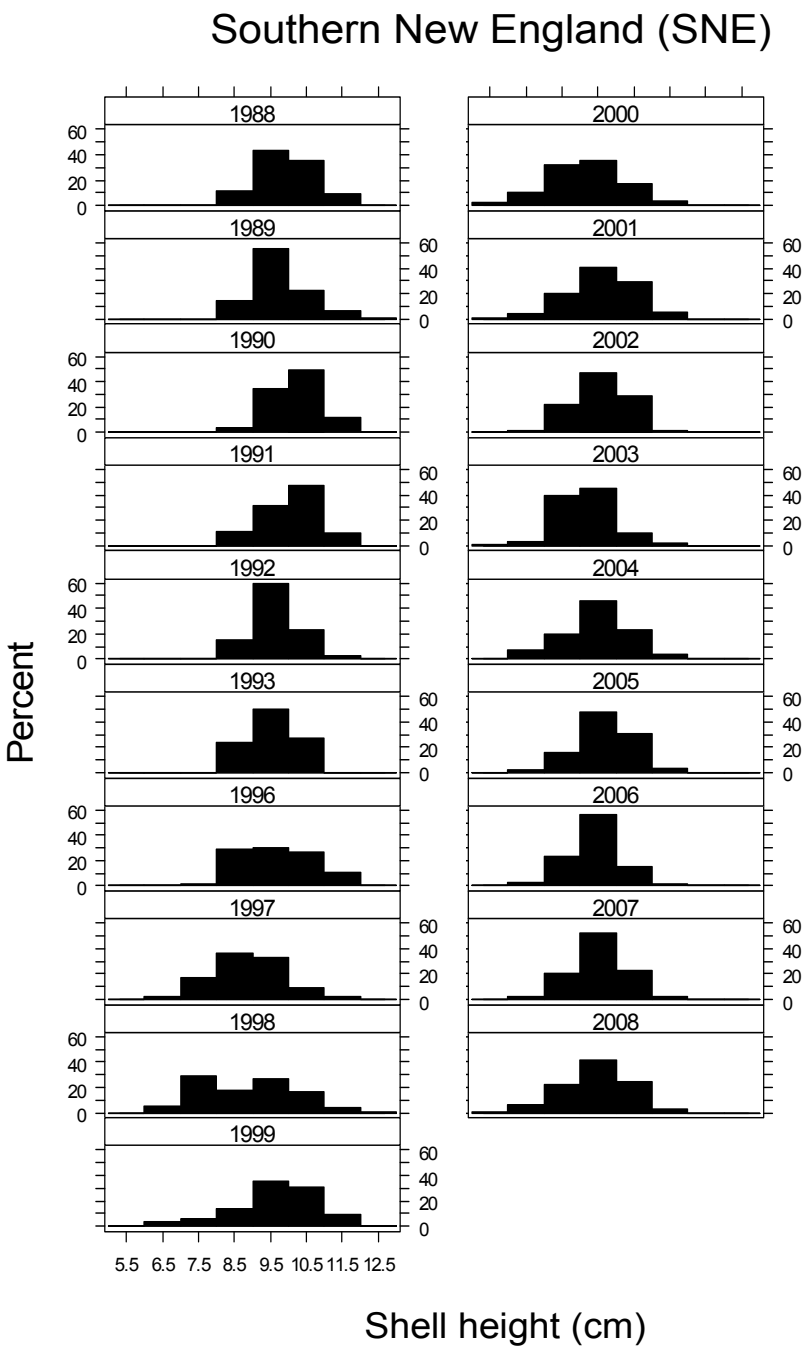


Figure B20. Commercial length composition data for ocean quahog landed in the SNE region.

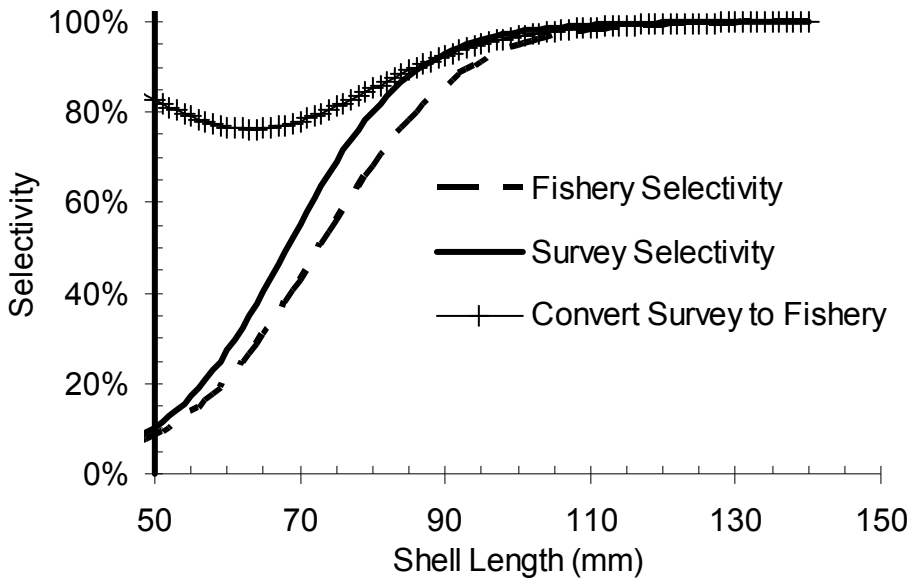


Figure B21. Fishery and survey selectivity curves for ocean quahog from NEFSC (2007a). The ratio of the fishery and survey selectivity curves, which can be used to convert survey abundance at size directly to fishable abundance at size, is also shown.

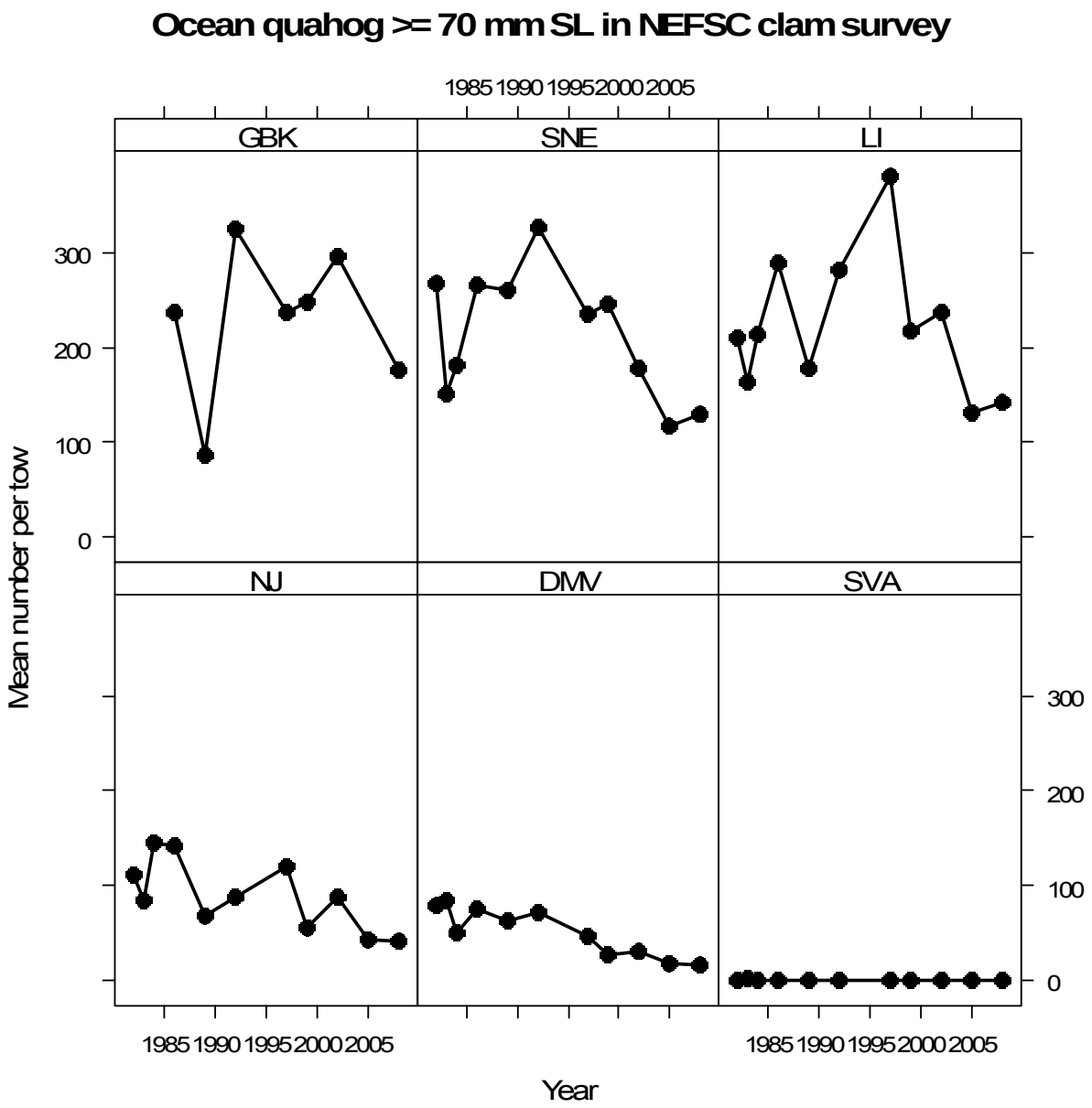


Figure B22. Long-term trends in survey abundance (mean number per tow) for large (≥70 mm SH) ocean quahogs during 1982-2008. Data from the 1994 survey are not shown because of voltage problems that affected catchability of the survey dredge. Sampling was relatively poor and figures are less unreliable for GBK during 1982-1984, 1989, 2002 and 2005; SNE during 1984 and 2005; LI during 1984; NJ during 1984; DMV during 2008; and in SVA during 1999 and 2008 (Table B8).

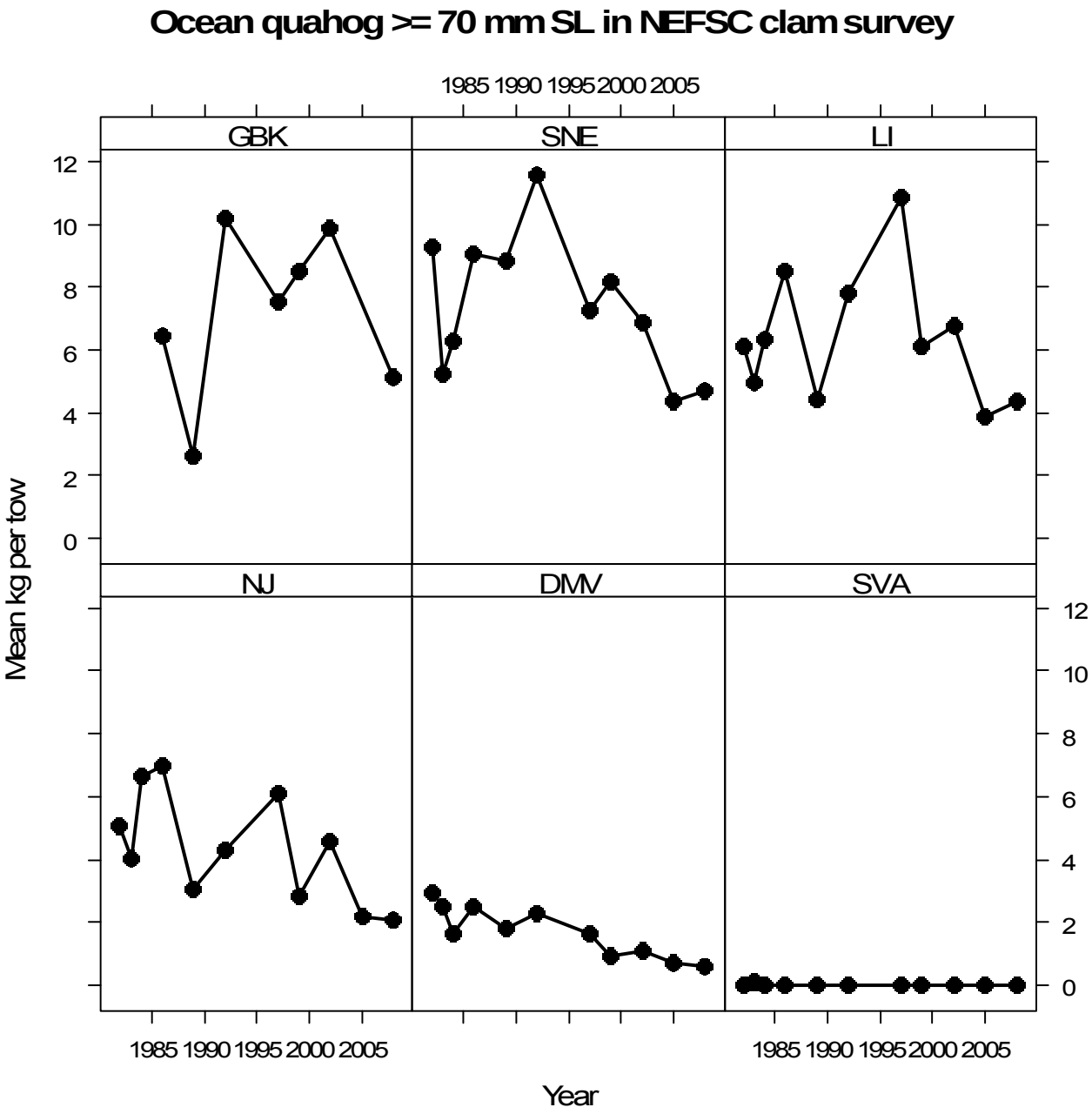


Figure B23. Long-term trends in survey mean biomass per tow for large (≥70 mm SH) ocean quahogs during 1982-2008. Data from the 1994 survey are not shown because of voltage problems that affected catchability of the survey dredge. Data for GBK from the 1982, 1983, 1984 and 2005 surveys are not shown because GBK was poorly sampled during those years (Table B8).

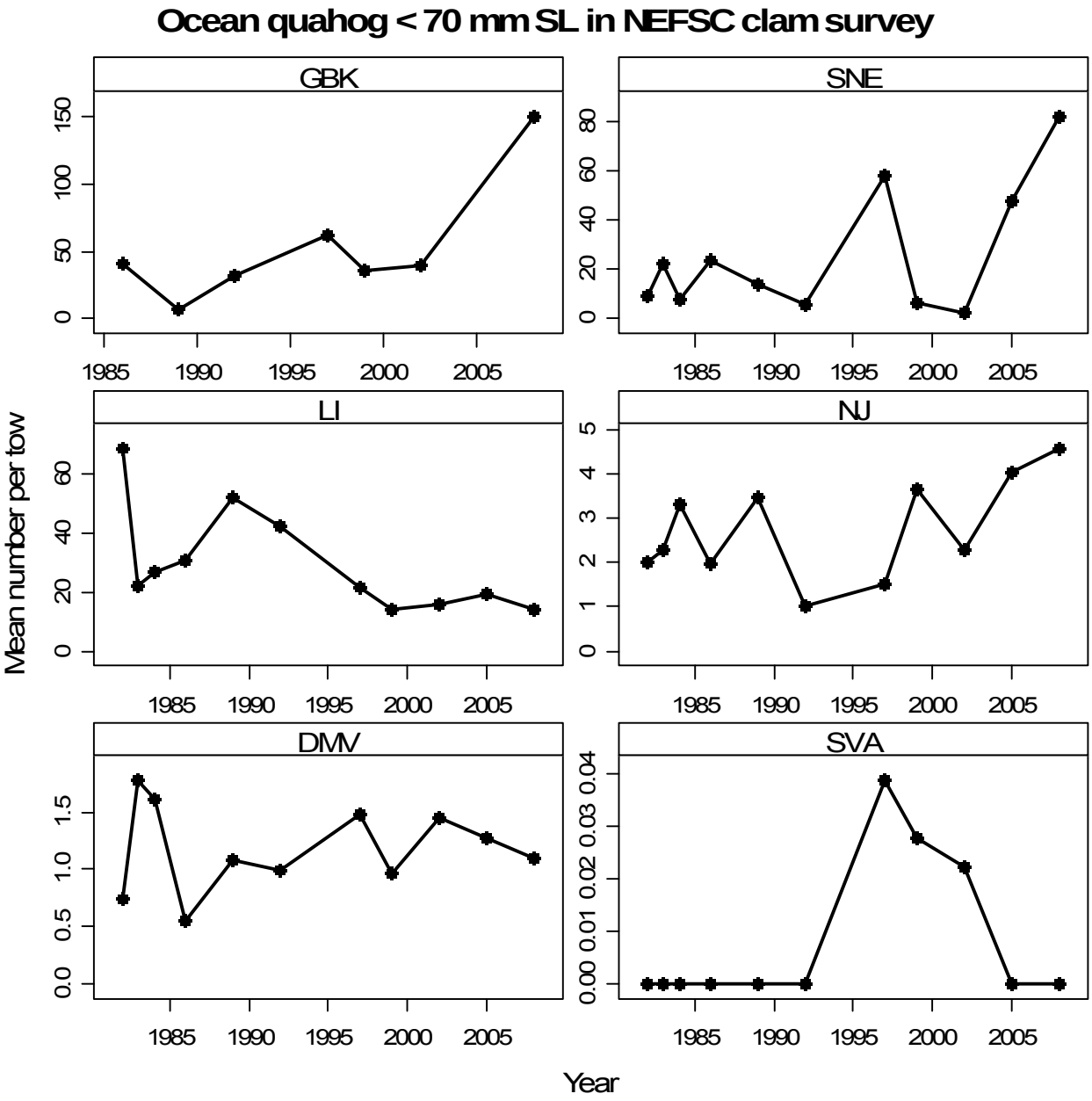


Figure B24. Long-term trends in abundance of small (<70 mm SH) ocean quahogs during 1982-2008. Data from the 1994 survey are not shown because of voltage problems that affected catchability of the survey dredge. Data for GBK from the 1982, 1983, 1984 and 2005 surveys are not shown because GBK was poorly sampled during those years (Table B8).

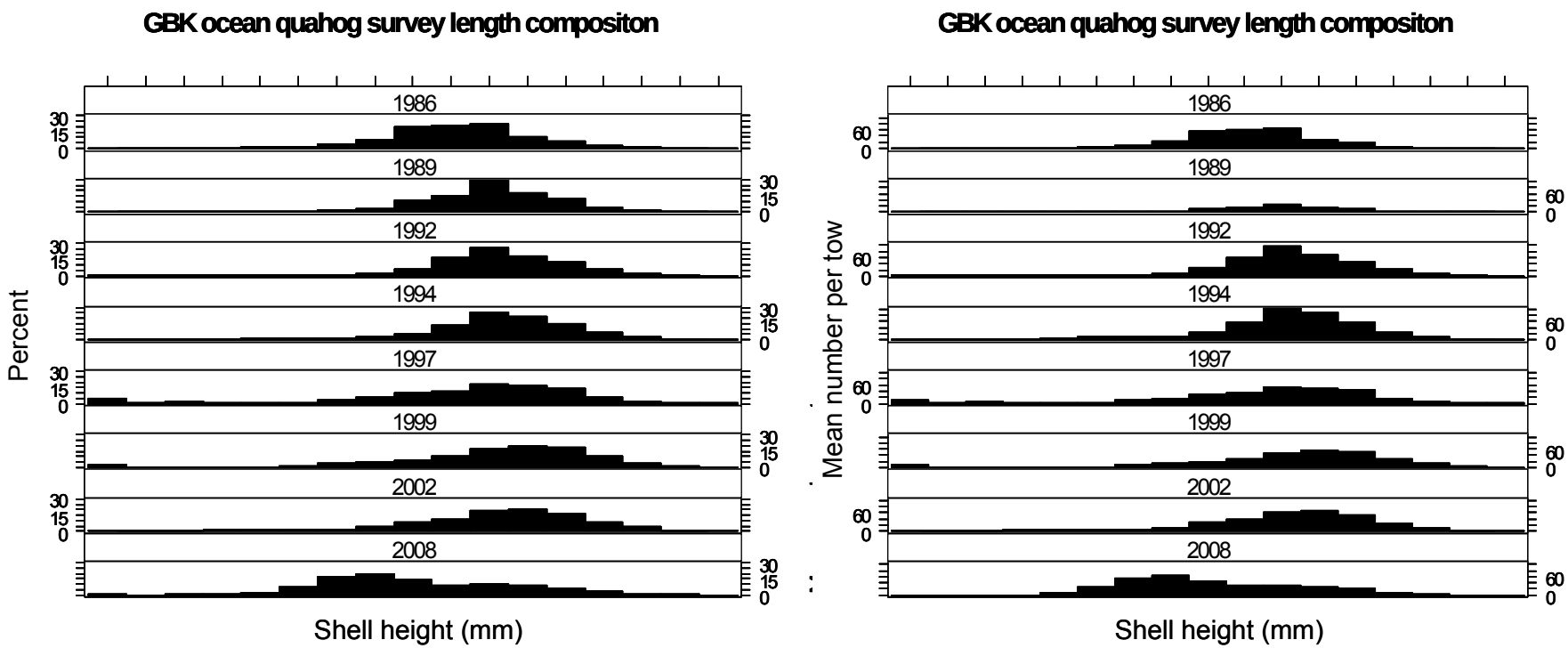


Figure B25. Survey length composition for ocean quahog in NEFSC clam surveys in the GBK region. The plots on the left show proportions of total mean number per tow in each year. The plots on the right show mean numbers per tow. All figures are without adjustment for survey dredge selectivity.

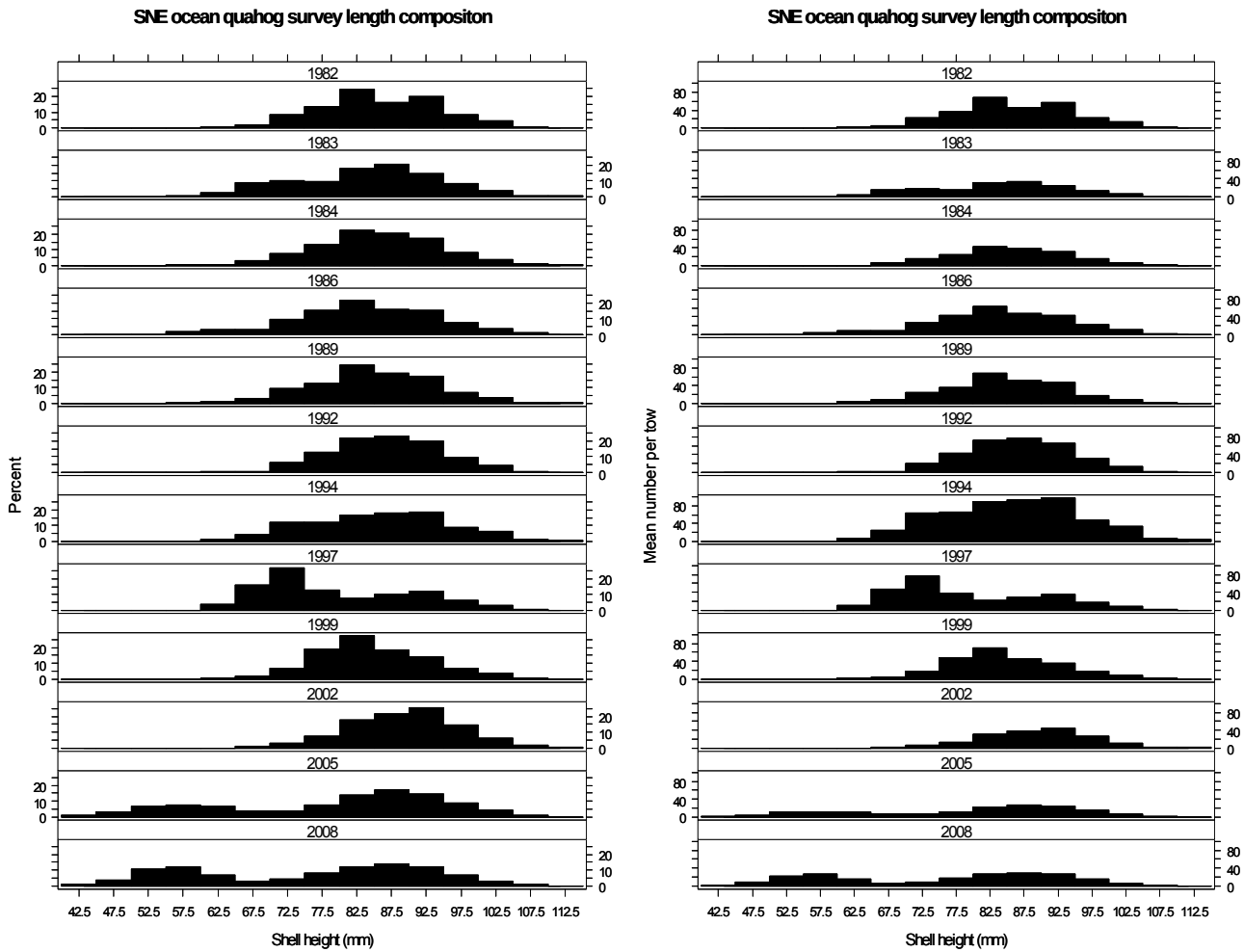


Figure B26. Survey length composition for ocean quahog in NEFSC clam surveys in the SNE region. The plots on the left show proportions of total mean number per tow in each year. The plots on the right show mean numbers per tow. All figures are without adjustment for survey dredge selectivity. Sampling was relatively poor and figures are less unreliable for SNE during 1984 (Table B8).

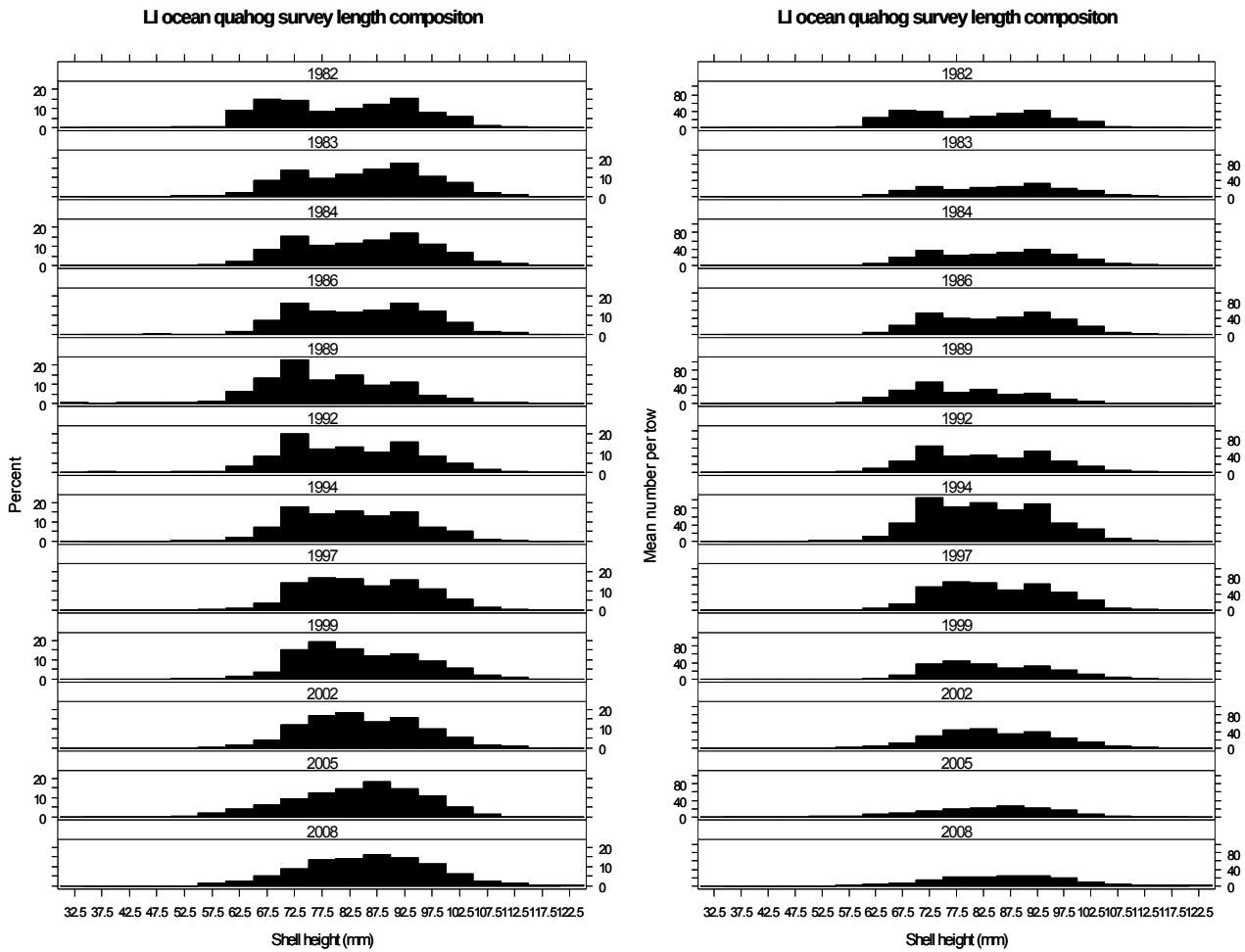


Figure B27. Survey length composition for ocean quahog in NEFSC clam surveys in the LI region. The plots on the left show proportions of total mean number per tow in each year. The plots on the right show mean numbers per tow. All figures are without adjustment for survey dredge selectivity. Sampling was relatively poor and figures are less unreliable for LI during 1984 (Table B8).

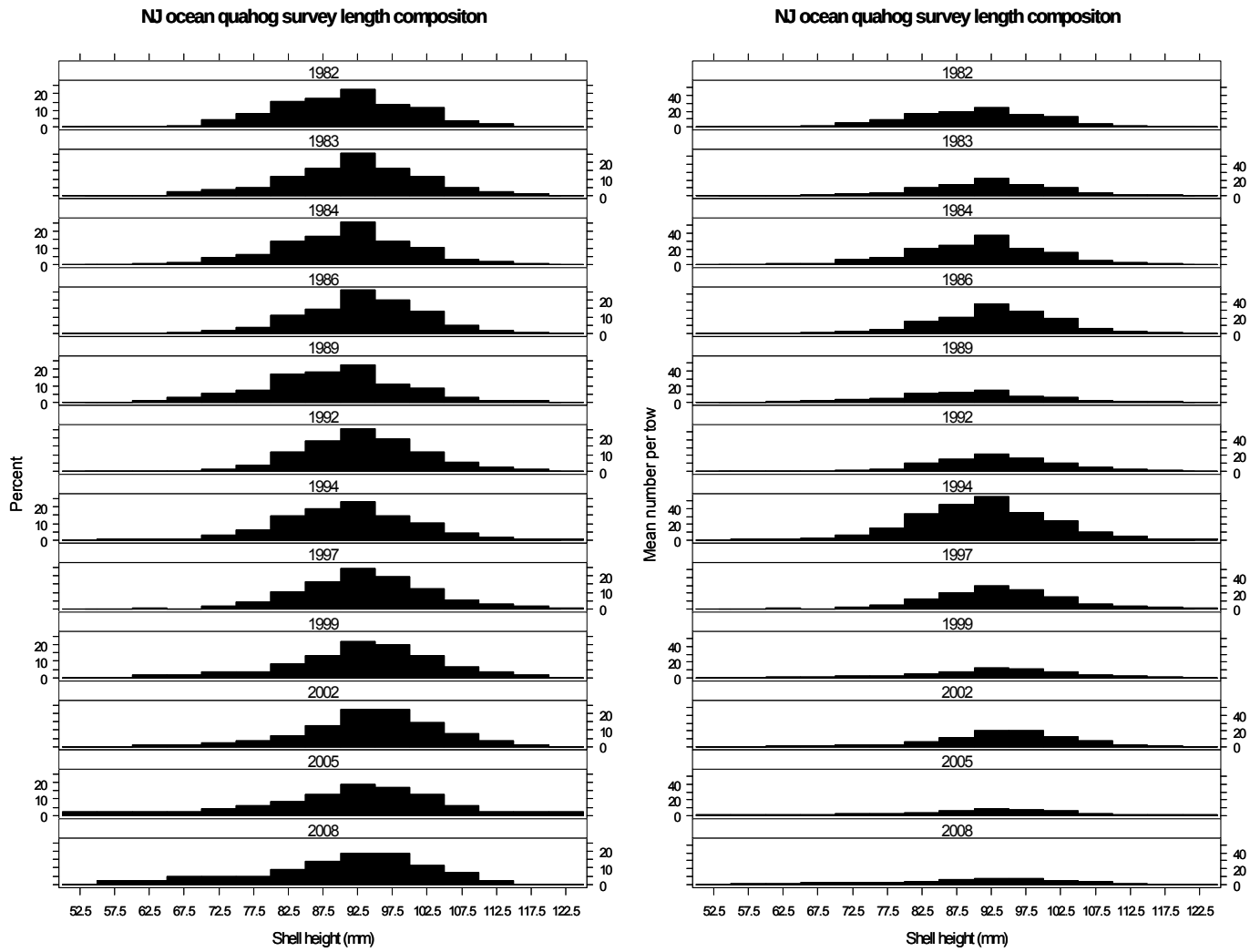


Figure B28. Survey length composition for ocean quahog in NEFSC clam surveys in the NJ region. The plots on the left show proportions of total mean number per tow in each year. The plots on the right show mean numbers per tow. All figures are without adjustment for survey dredge selectivity. Sampling was relatively poor and figures are less unreliable for NJ during 1984 (Table B8).

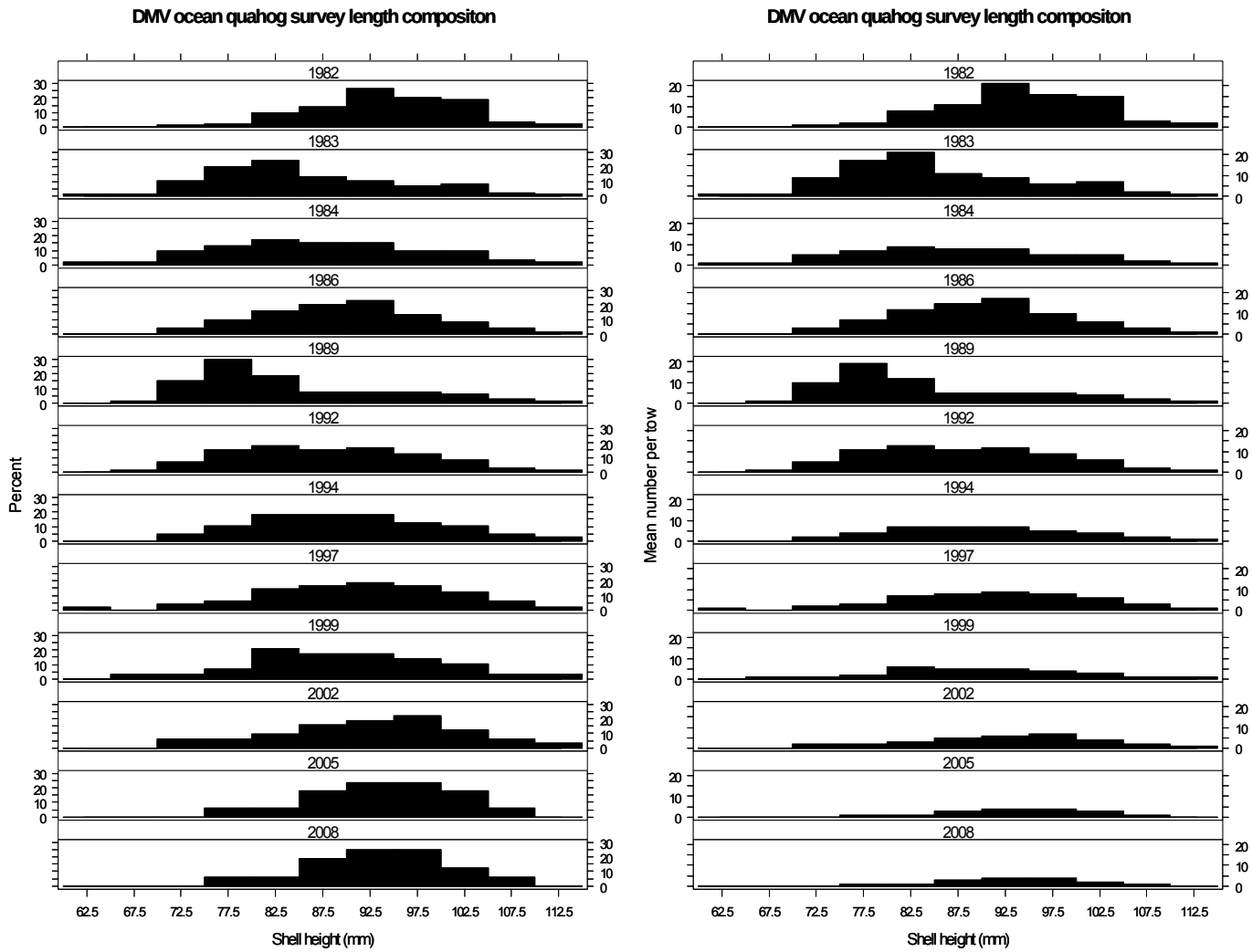


Figure B29. Survey length composition for ocean quahog in NEFSC clam surveys in the DMV region. The plots on the left show proportions of total mean number per tow in each year. The plots on the right show mean numbers per tow. All figures are without adjustment for survey dredge selectivity. Sampling was relatively poor and figures are less unreliable for DMV during 2008 (Table B8).

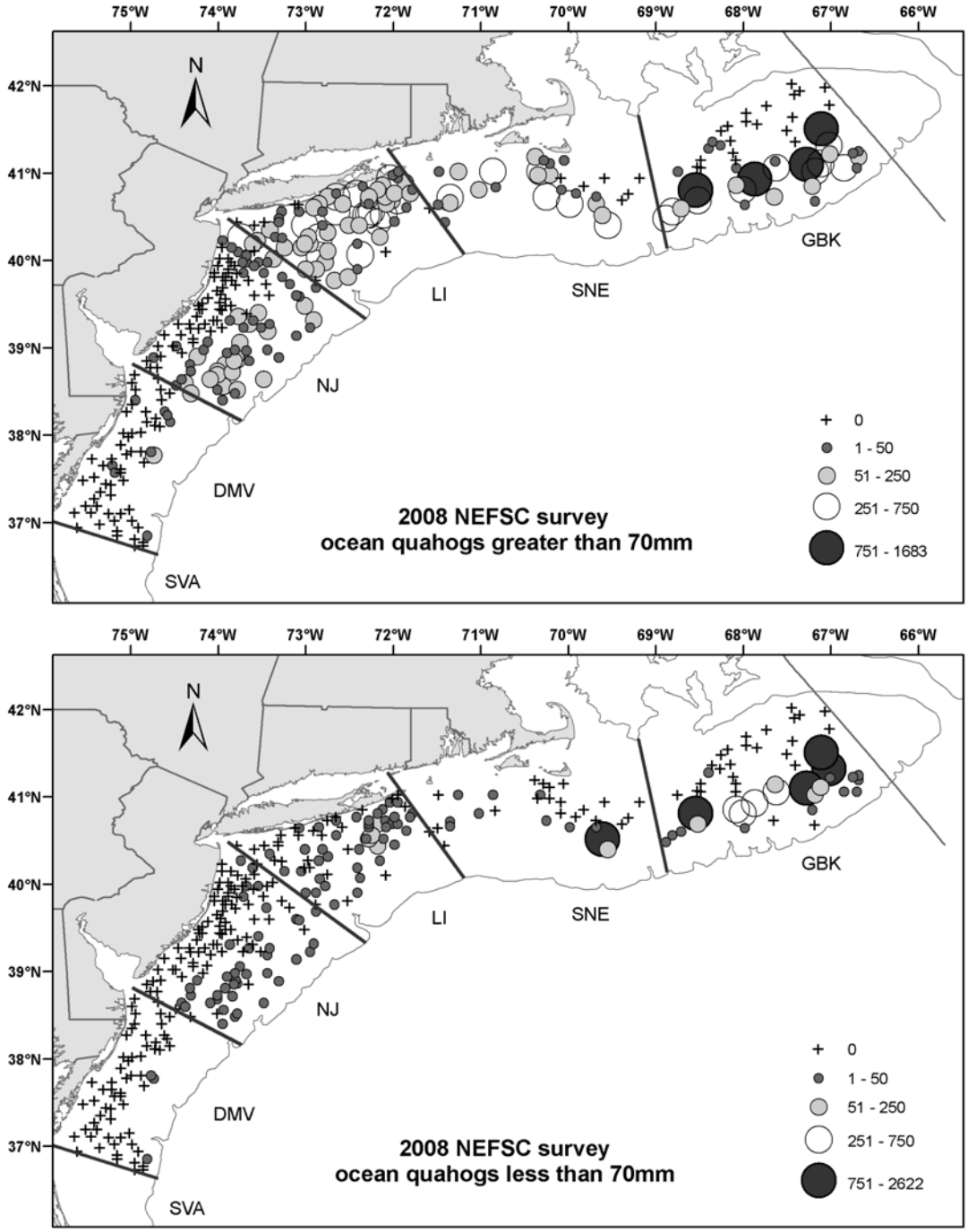


Figure B30. Location of tows and catch of large (≥70 SL) and small (<70 mm) ocean quahogs in 2008 clam survey. See Appendix B5 for other years.

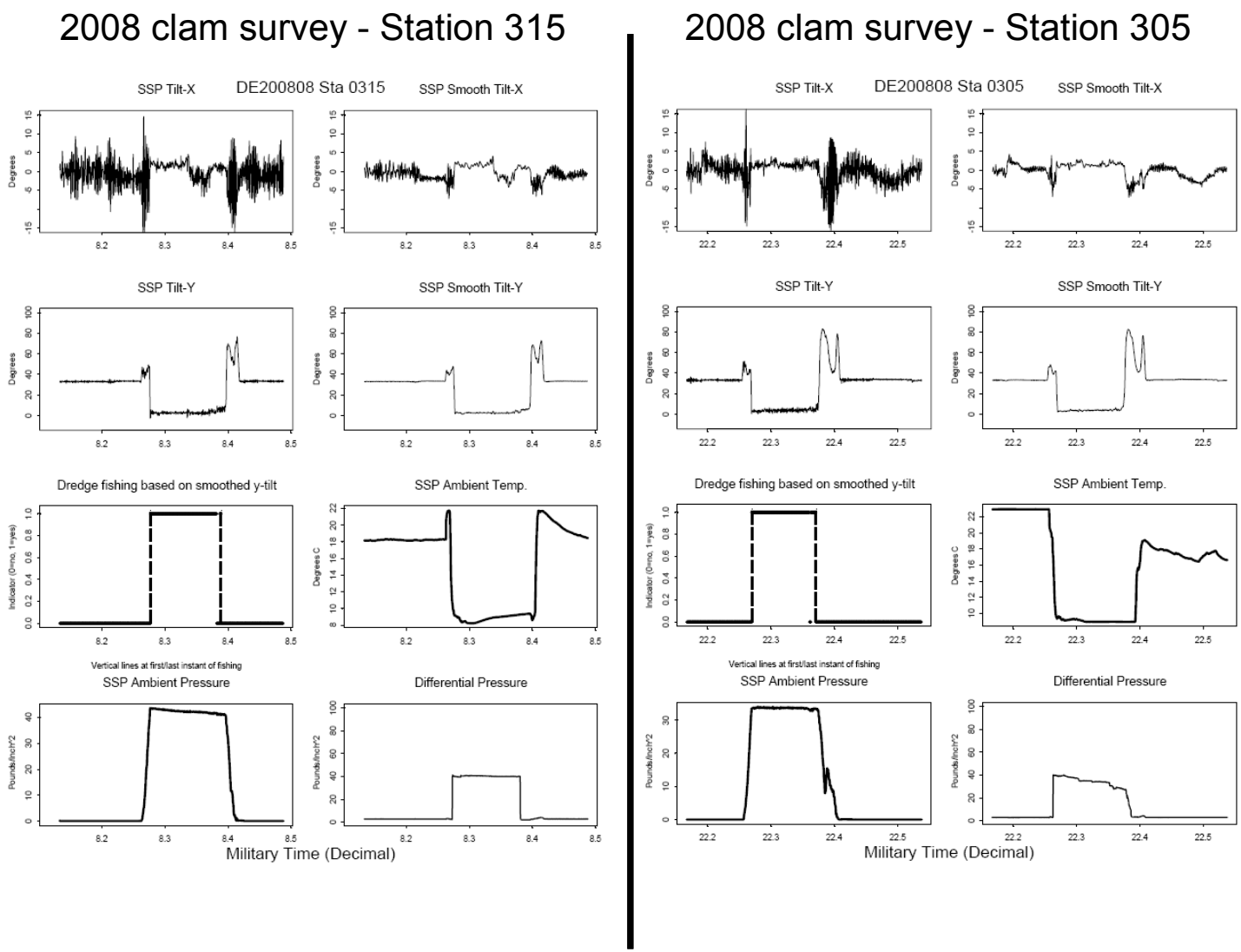


Figure B31. Sensor data from stations 315 (left) and 305 (right) in the 2008 NEFSC clam survey. Based on amperage and differential pressure, dredge performance was better at station 315.

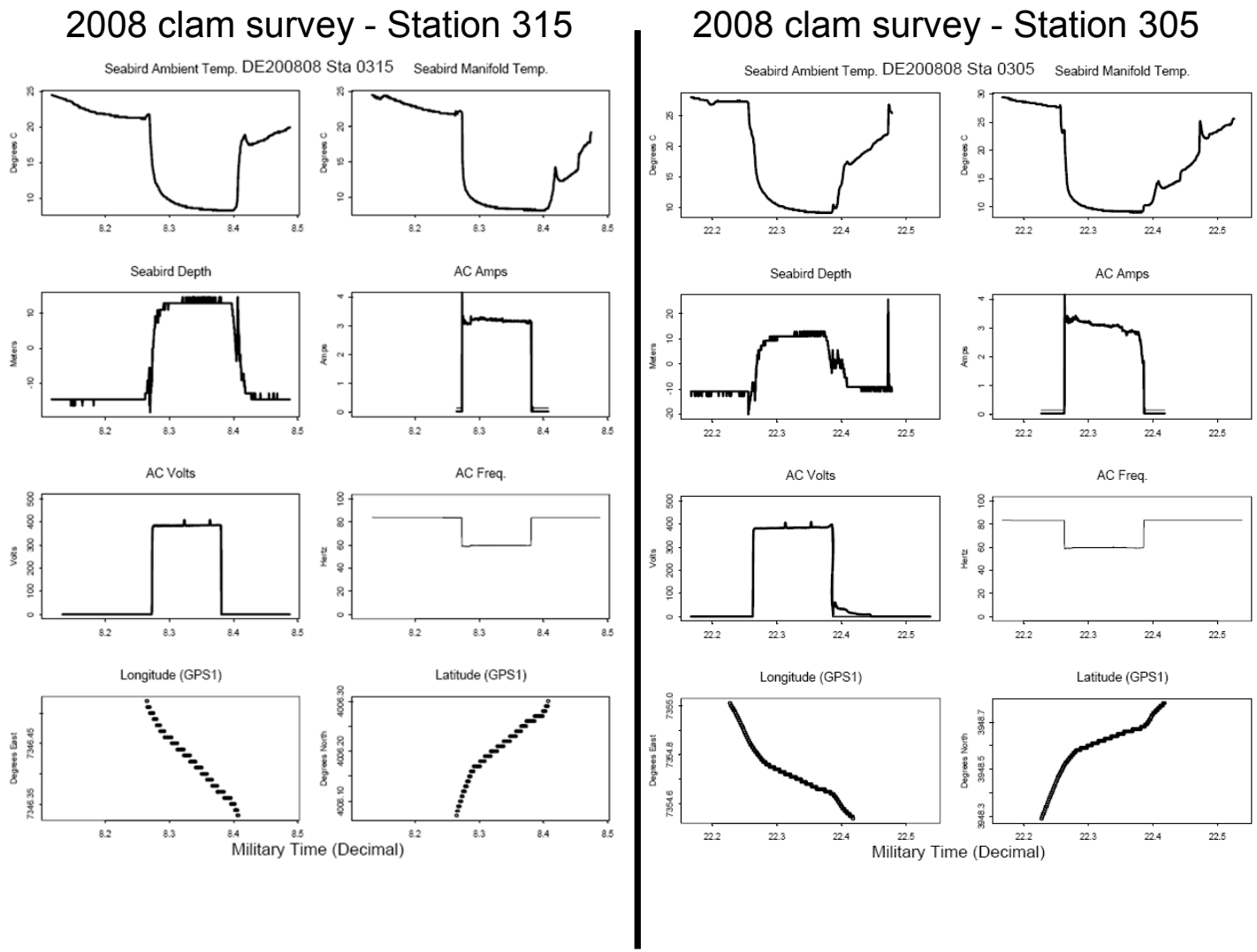
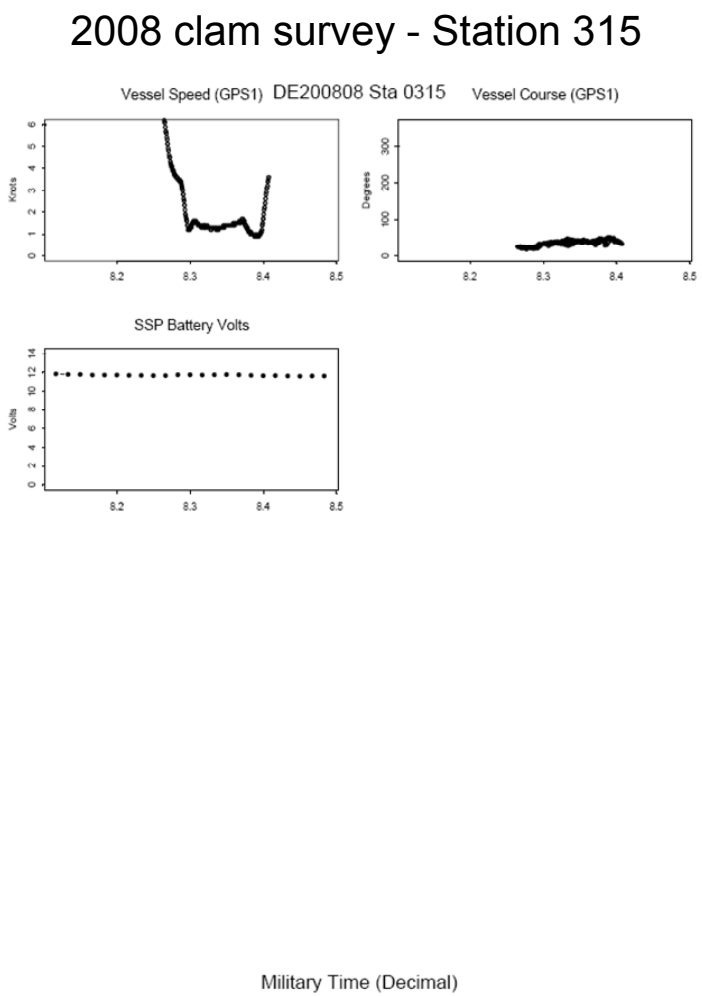


Figure B31 (cont.)



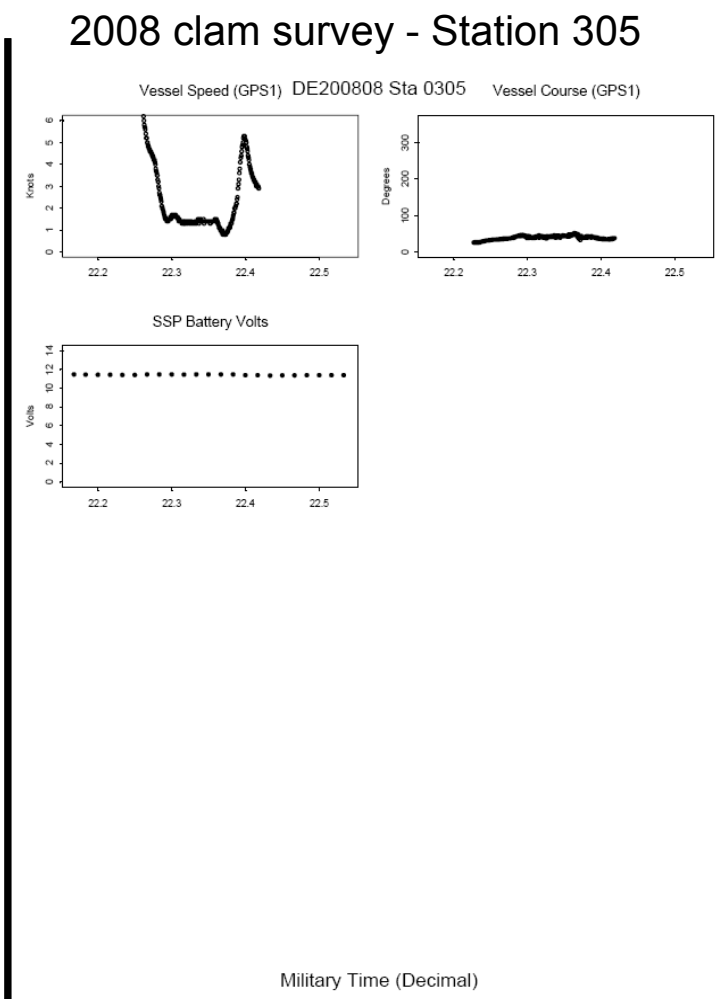


Figure B31 (cont.)

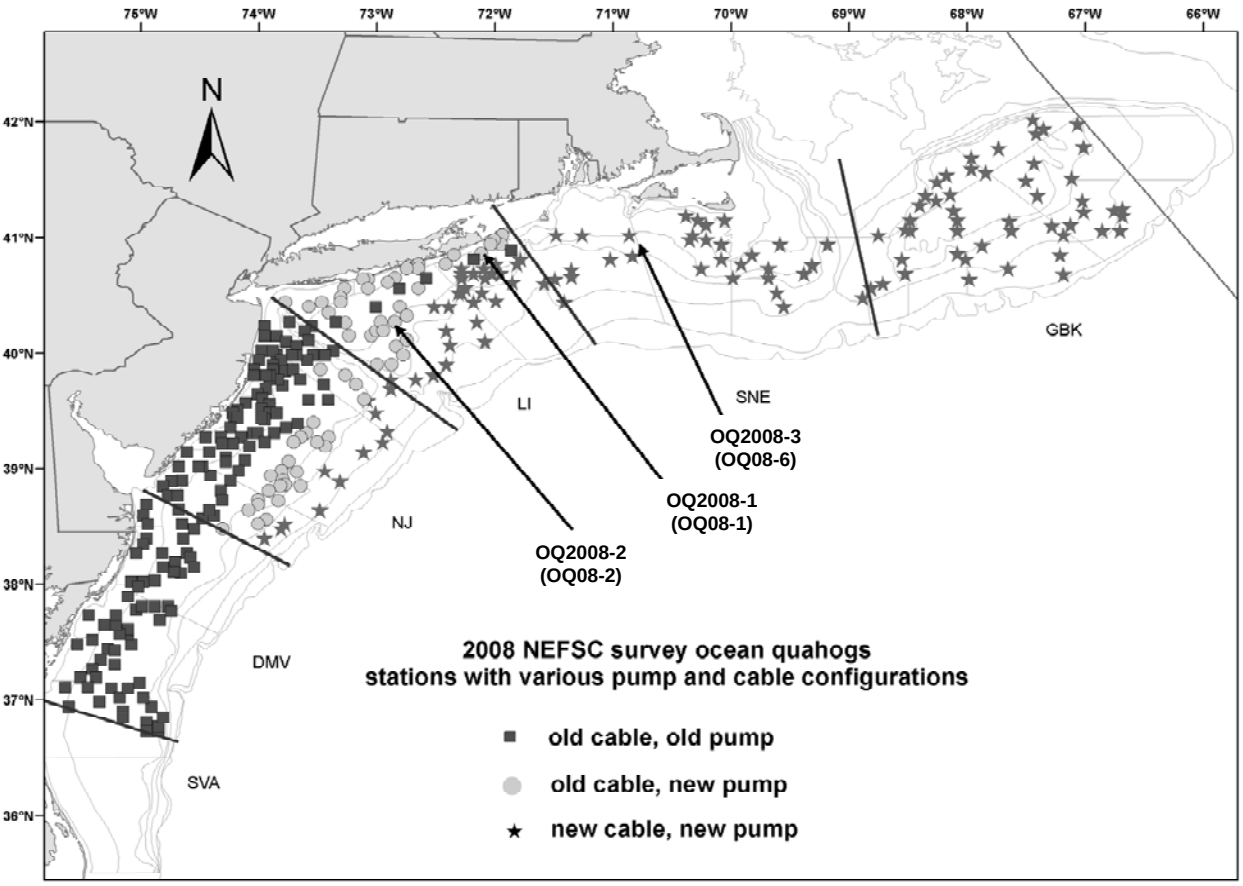


Figure B32. Map showing the locations of random tows done during the 2008 NEFSC clam survey. The different symbols represent different configurations of the electrical cable and dredge pump, which were both replaced during the survey. Arrows point to the areas where the depletion experiments were conducted.

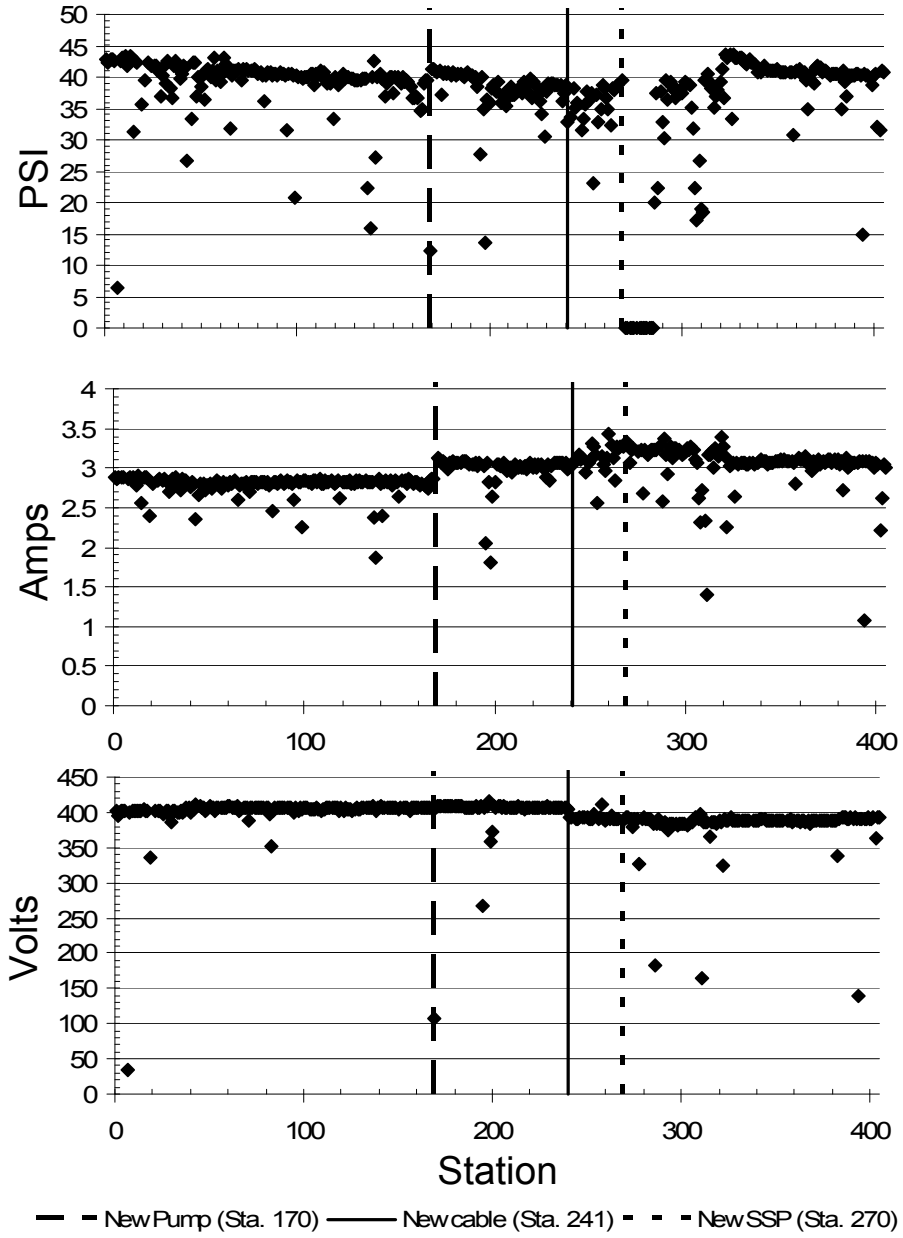


Figure B33. Mean SSP sensor data during periods when the dredge was fishing effectively, for stations 1-405.

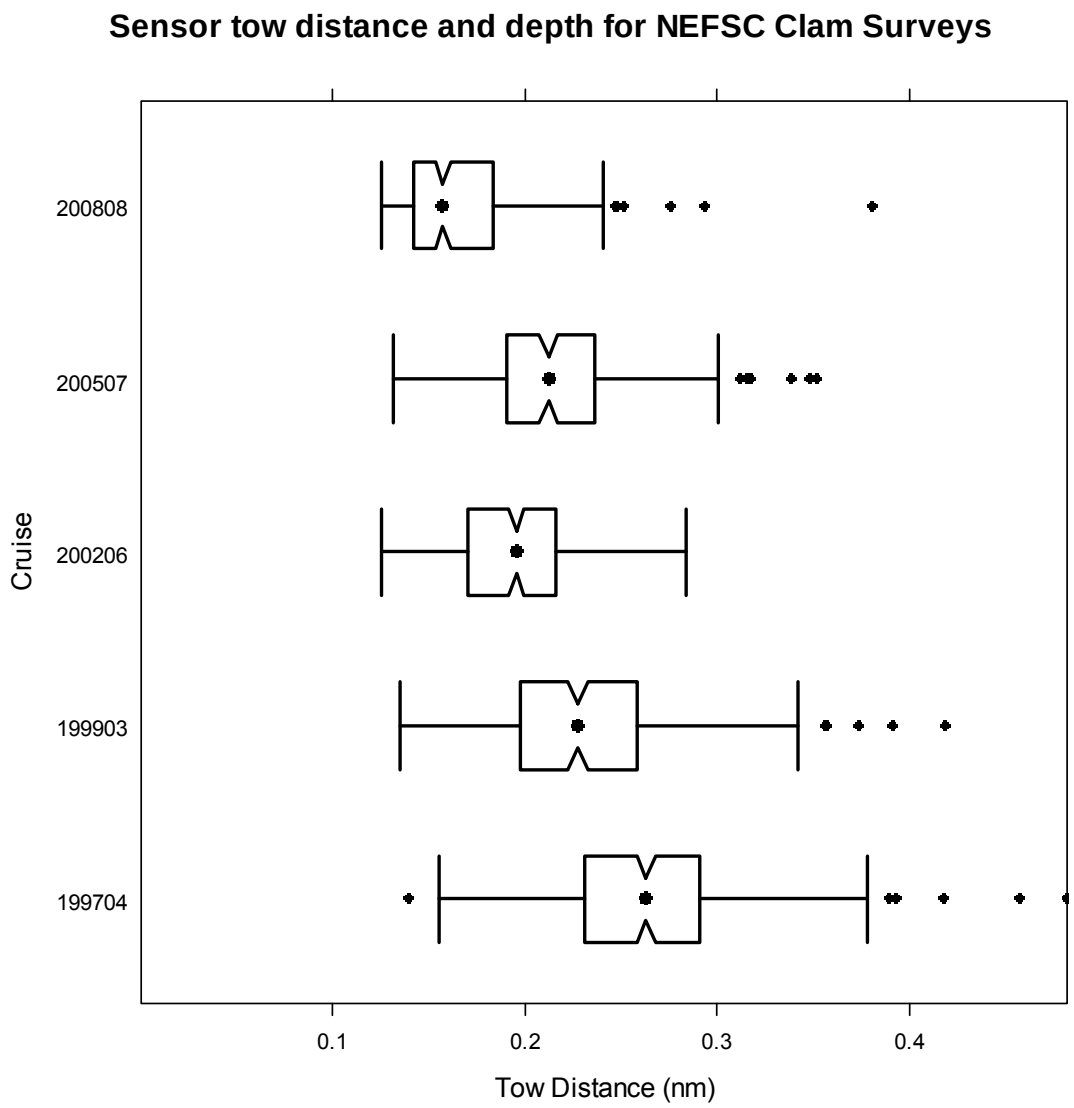


Figure B34. Distribution of sensor based tow distances for all tows in the 1997-2008 surveys with useable y-tilt data.

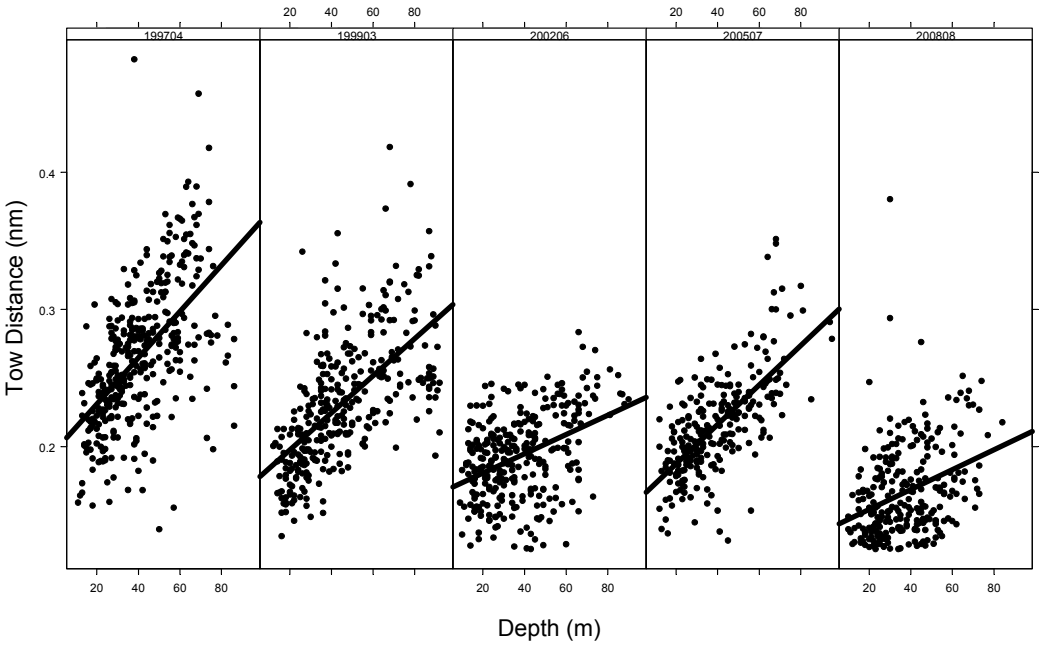


Figure B35. Survey specific linear regression models for relationships between tow distance (based on sensor data) and depth. Data are for successful random tows only.

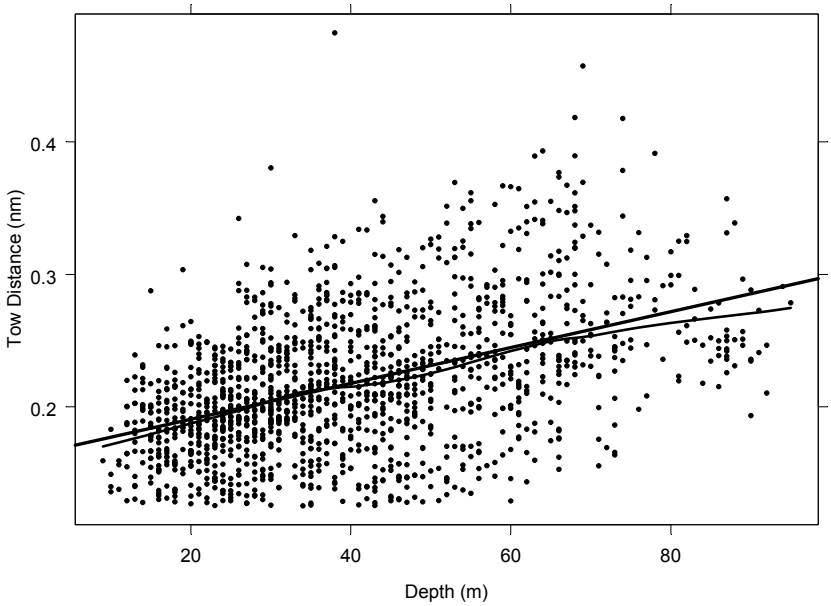


Figure B36. Relationship between tow distance (based on sensor data) and depth for successful random tows in surveys with sensor data conducted between 1997 and 2009. The straight line shows the linear regression model Distance=0.1635+0.0014 x Depth. The nonlinear line is a spline meant to show underlying, potentially nonlinear, trends.

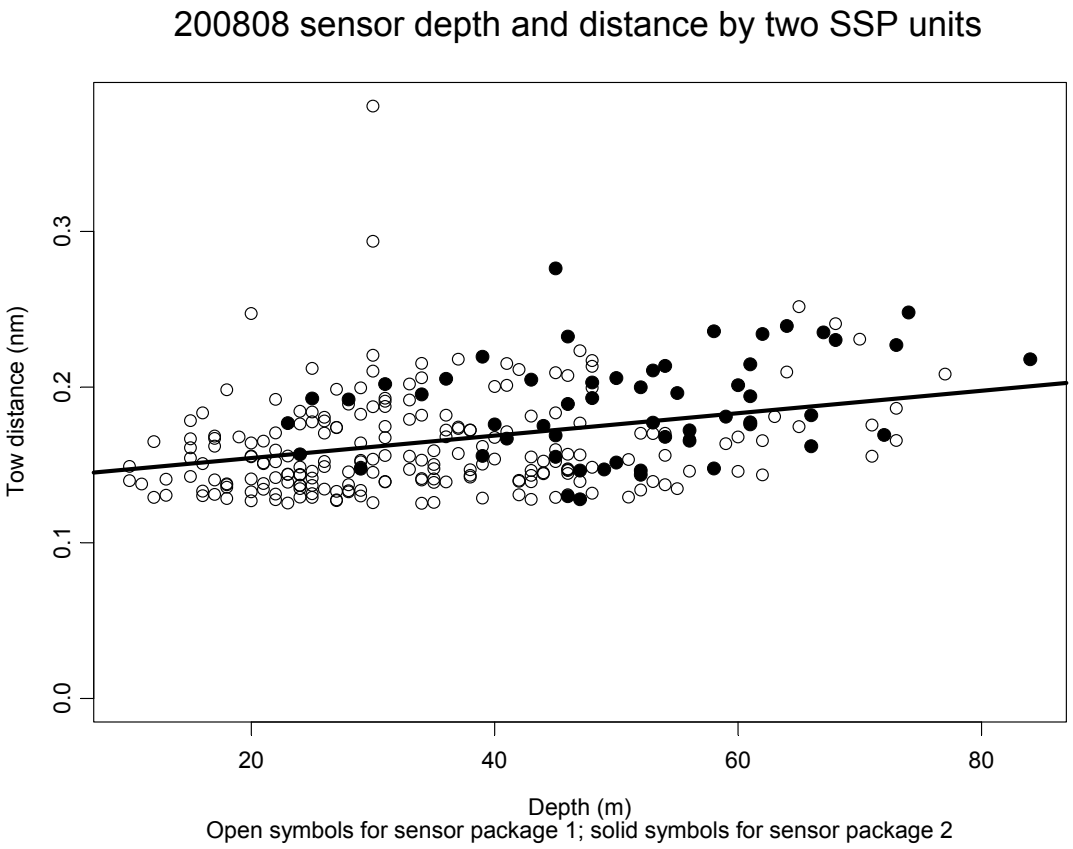


Figure B37. Relationship between tow distance and depth during the 2008 clam survey estimated using y-tilt data from the original (open symbols, stations 1-269) and replacement (dark symbols, stations 270-401) SSP units.

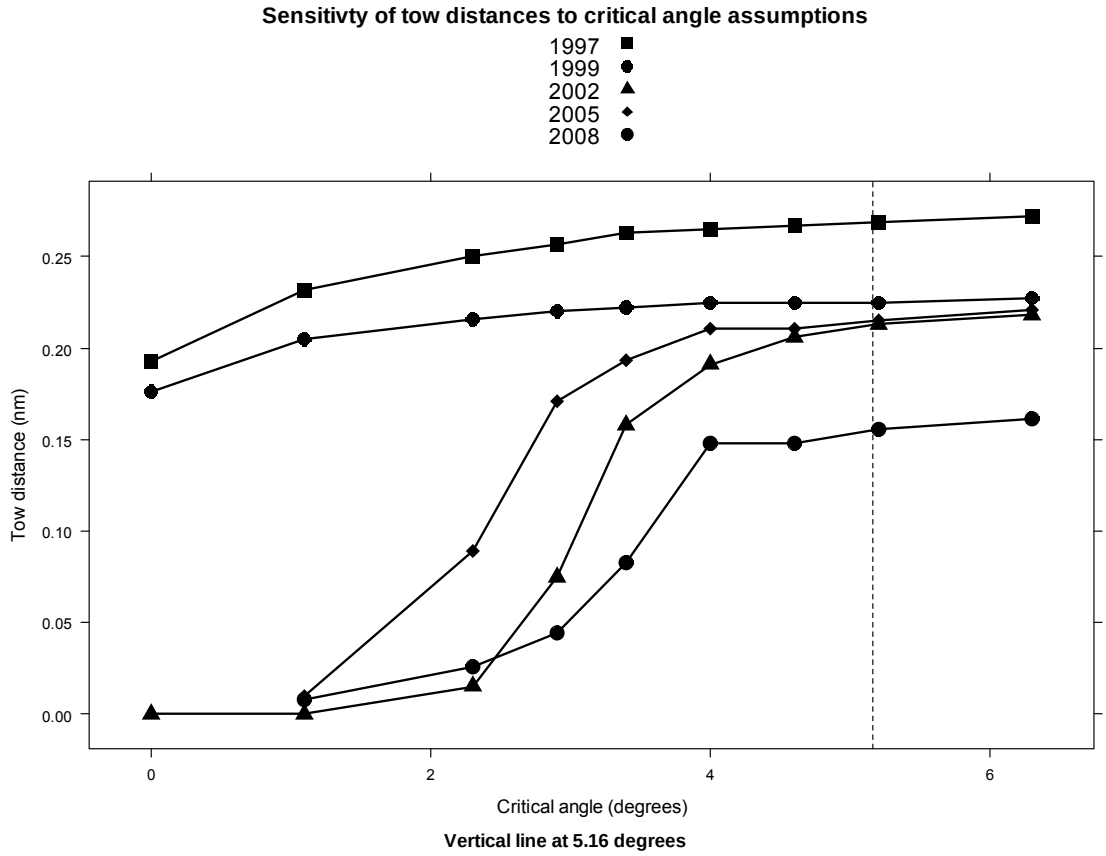


Figure B38. Sensitivity of median survey tow distance to assumptions about the critical angle at which the survey dredge fishes effectively. Median tow distances are for all successful random survey tows with y-tilt data during the 1997-2008 surveys. Surveys during 1997 and 1999 surveys used an inclinometers attached to the dredge. Surveys during 2002, 2005 and 2008 used integrated SSP (survey sensor package) sensors. Over the range of dredge angles shown in the figure, D = 0.731*A -7.947, where D is the blade depth (inches) and A is the critical angle in degrees. This analysis updates Figure C21 in NEFSC (2003).

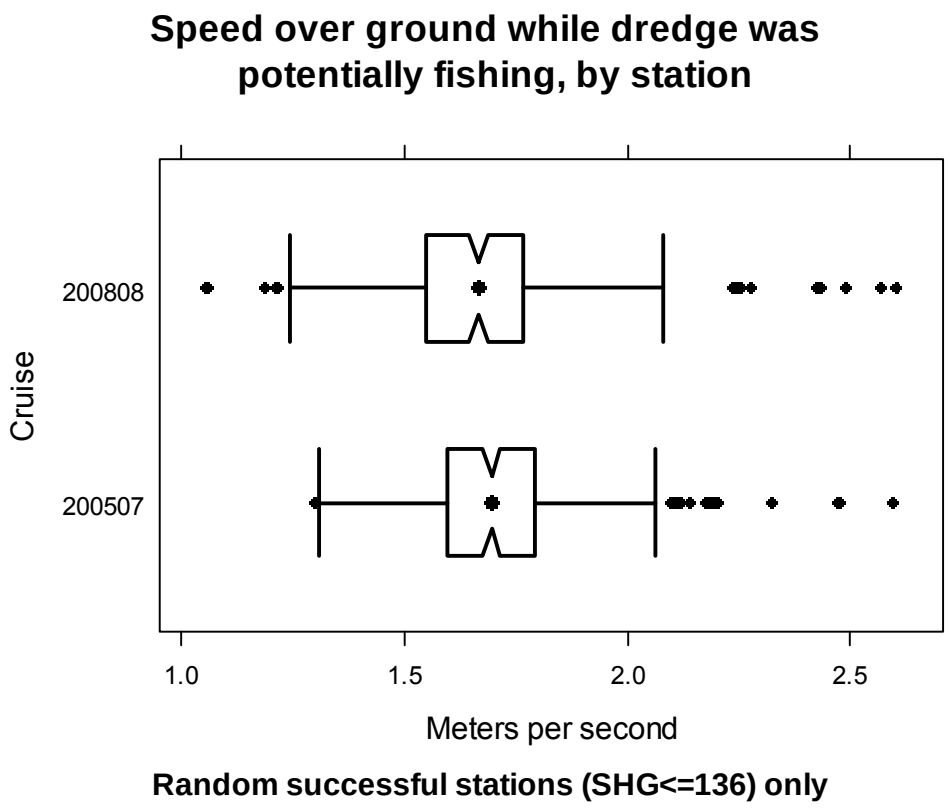
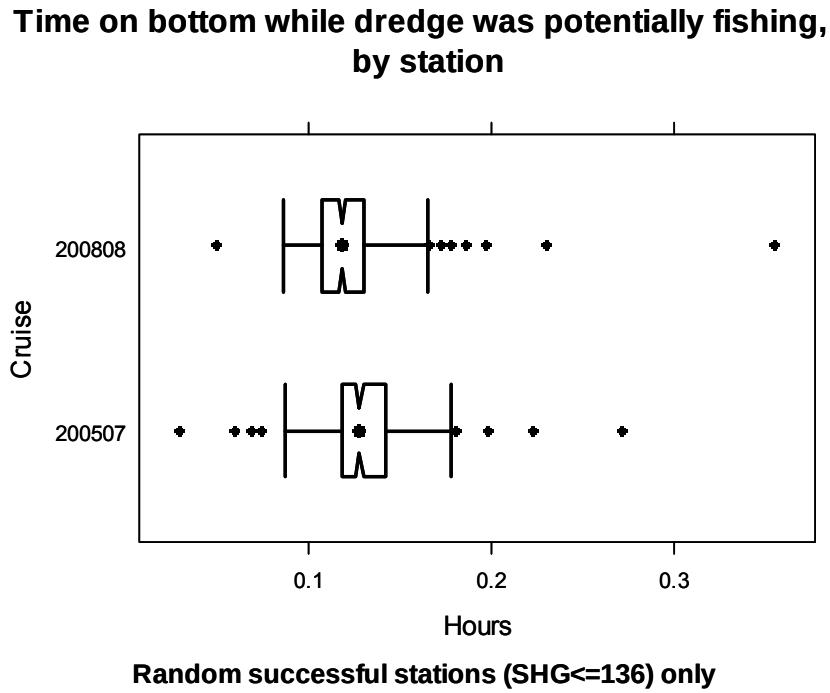


Figure B39. Box plots showing distributions of dredge performance variables from sensor data for successful random tows during the 2005 and 2008 NEFSC clam survey. For some variables that are highly skewed, two boxplots are presented with the plot at the top showing the distribution of all of the data and the plot at the bottom rescaled to exclude outliers and to better depict the relative distributions of most of the data.



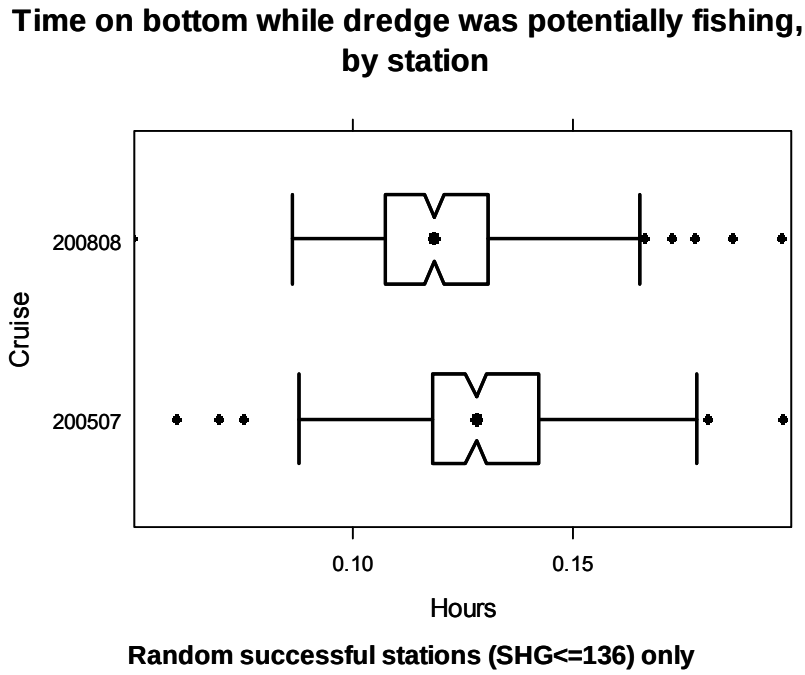
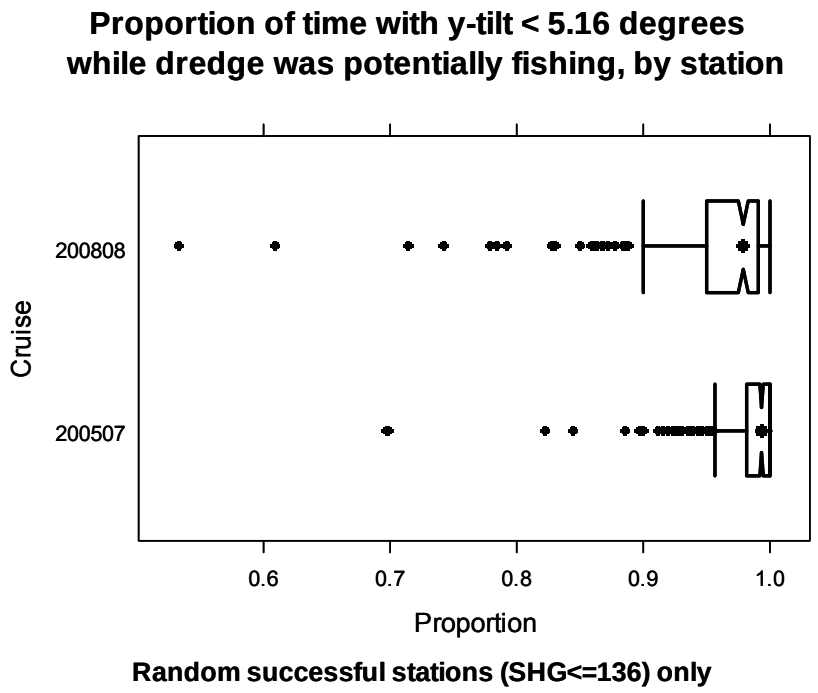


Figure B39. (cont.)



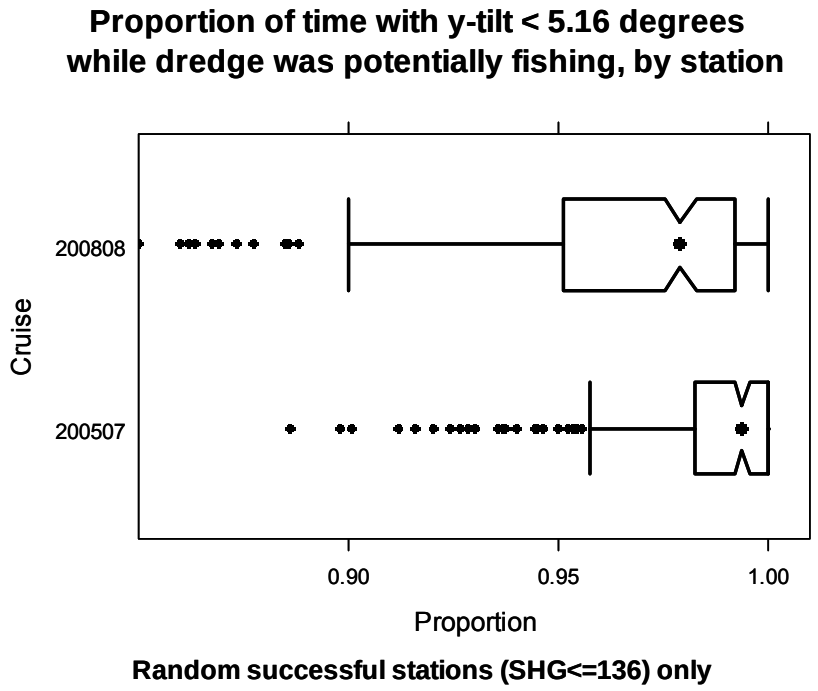
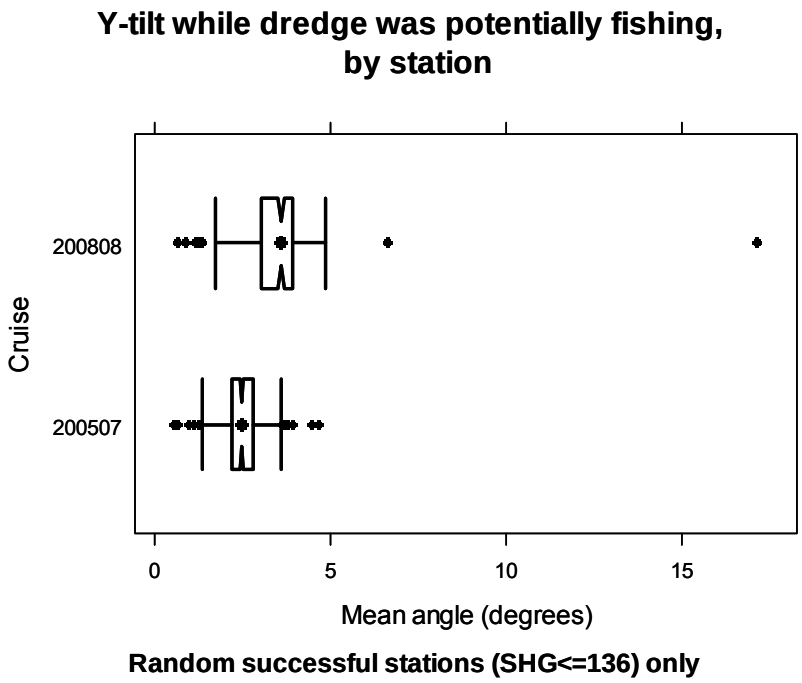


Figure B39. (cont.)



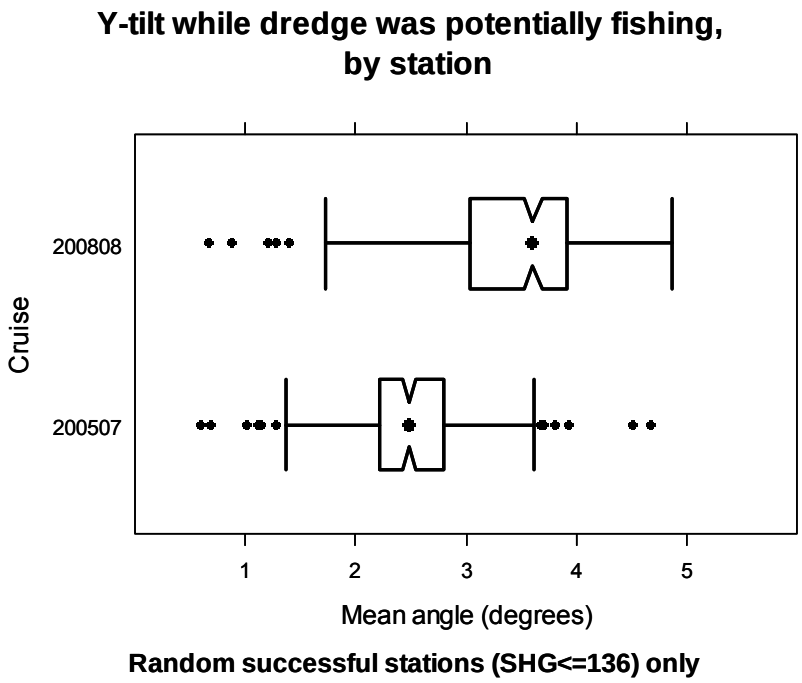
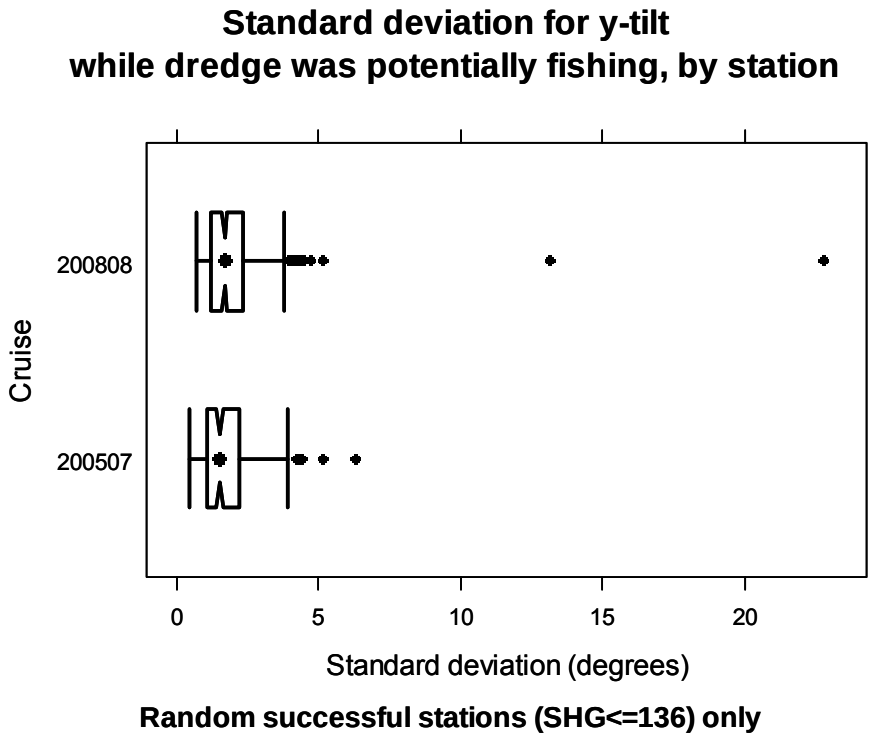


Figure B39. (cont.)



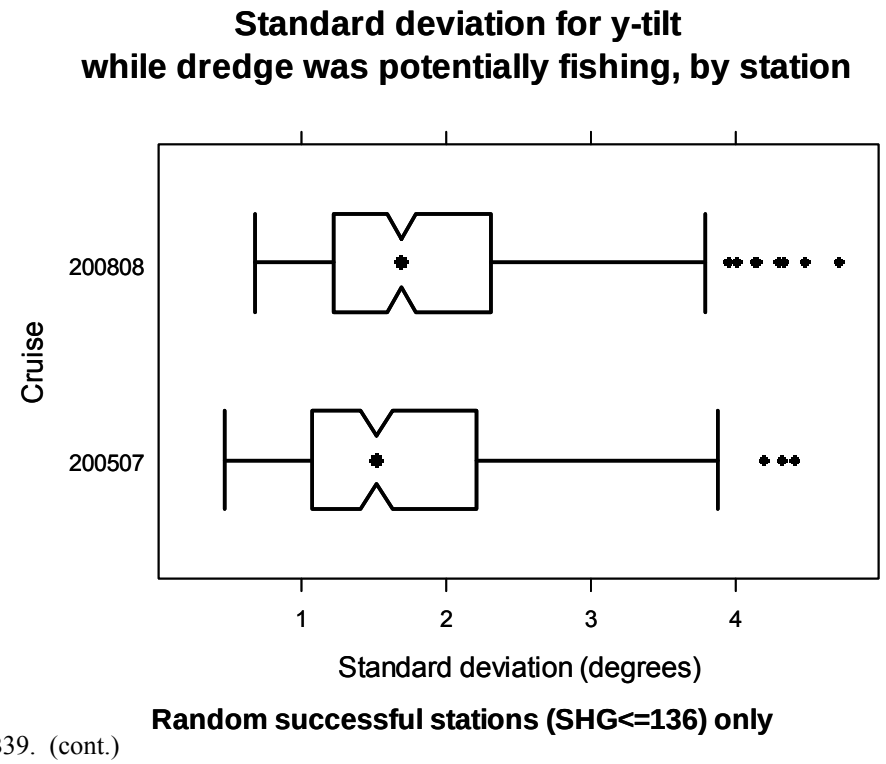
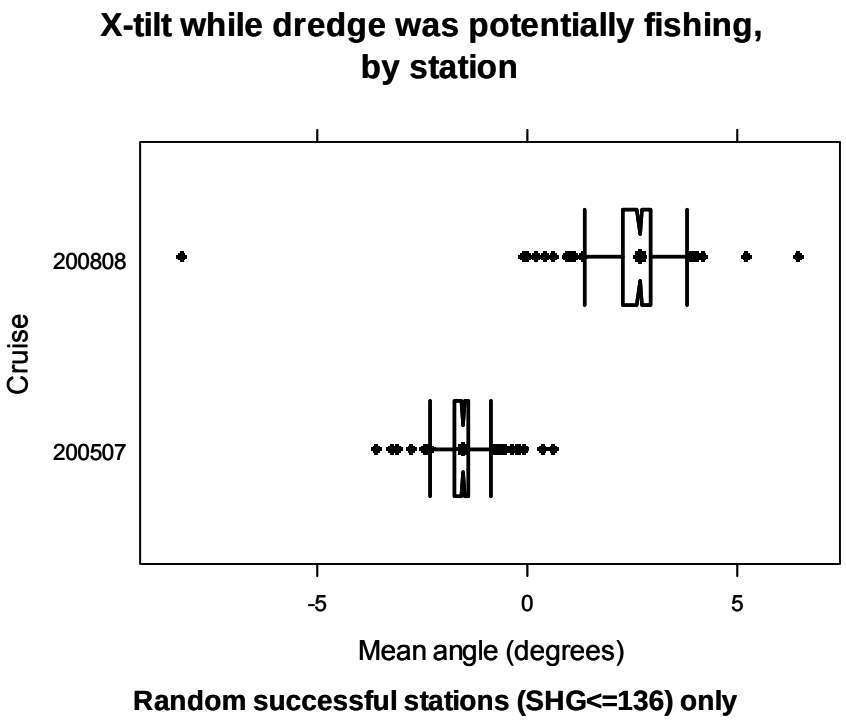


Figure B39. (cont.)



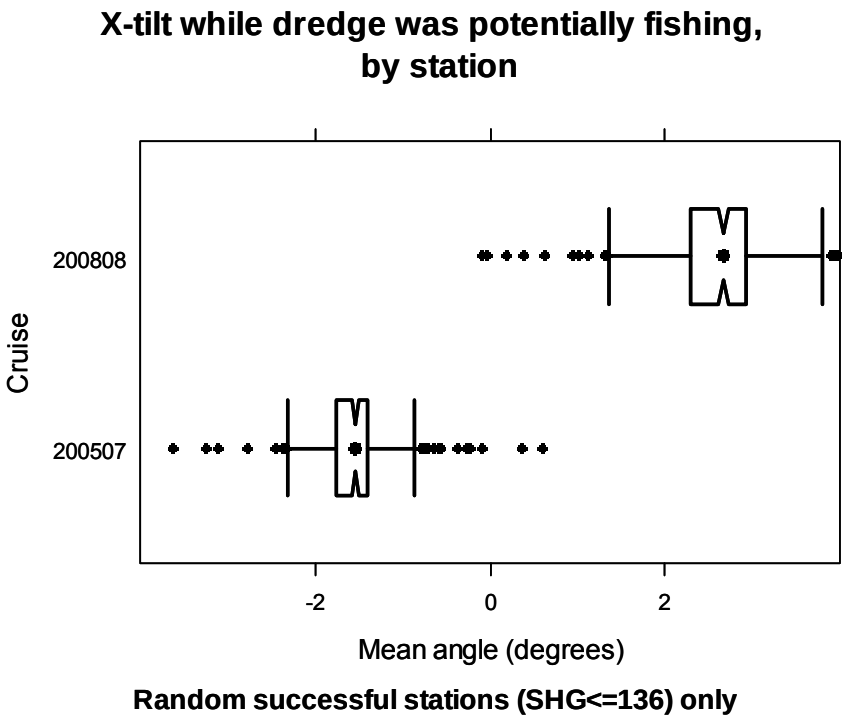


Figure B39. (cont.)

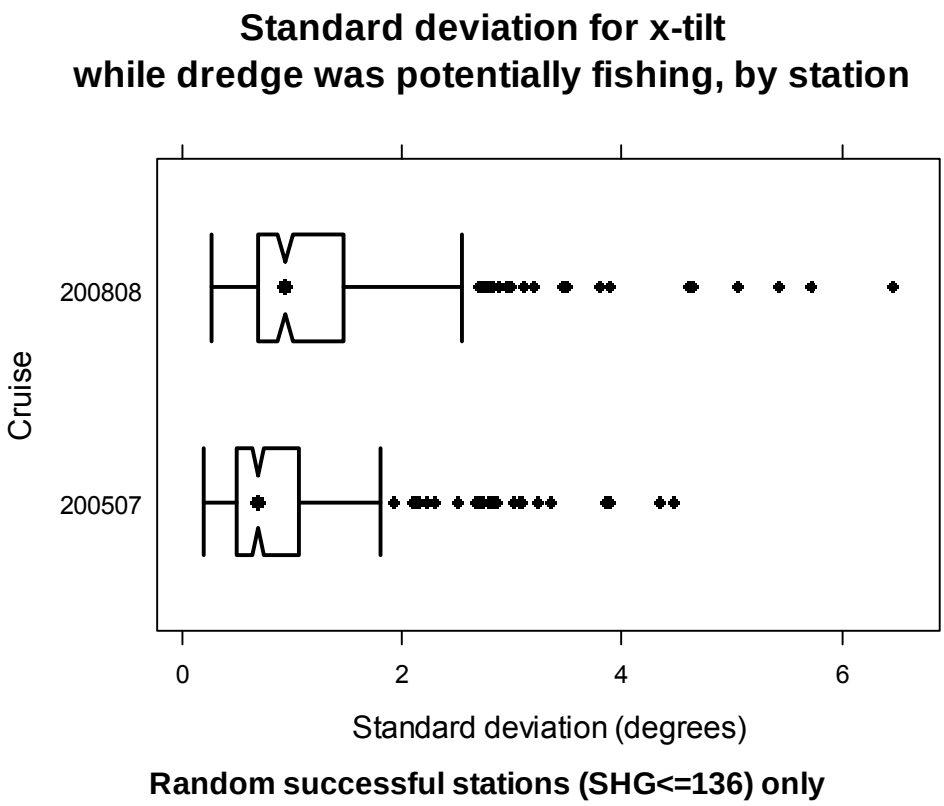


Figure B39. (cont.)

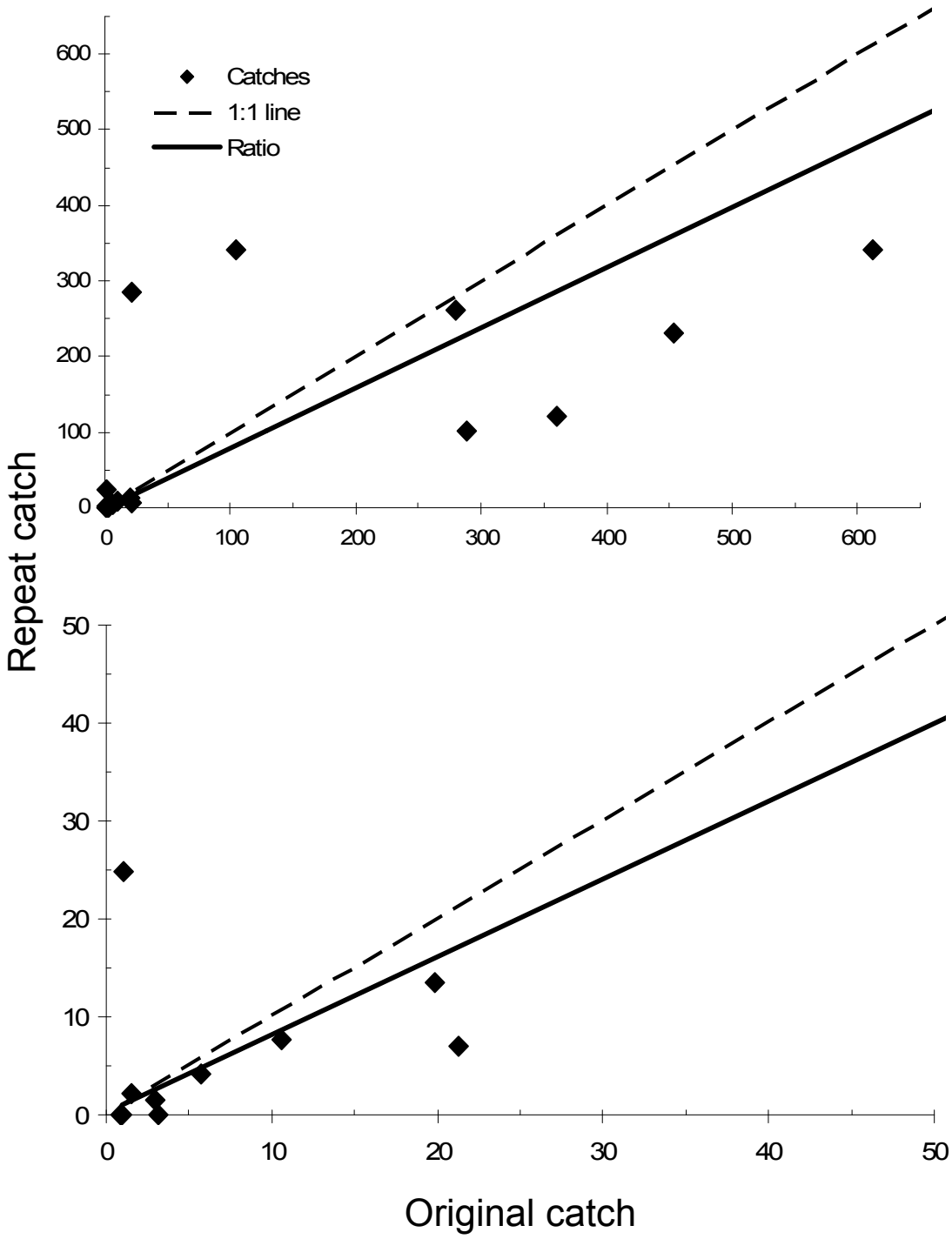


Figure B40. *Delaware II-Delaware II* (De2-De2) repeat station results. *Top*: all data. *Bottom*: showing observations near the origin that are hard to see in the upper panel.

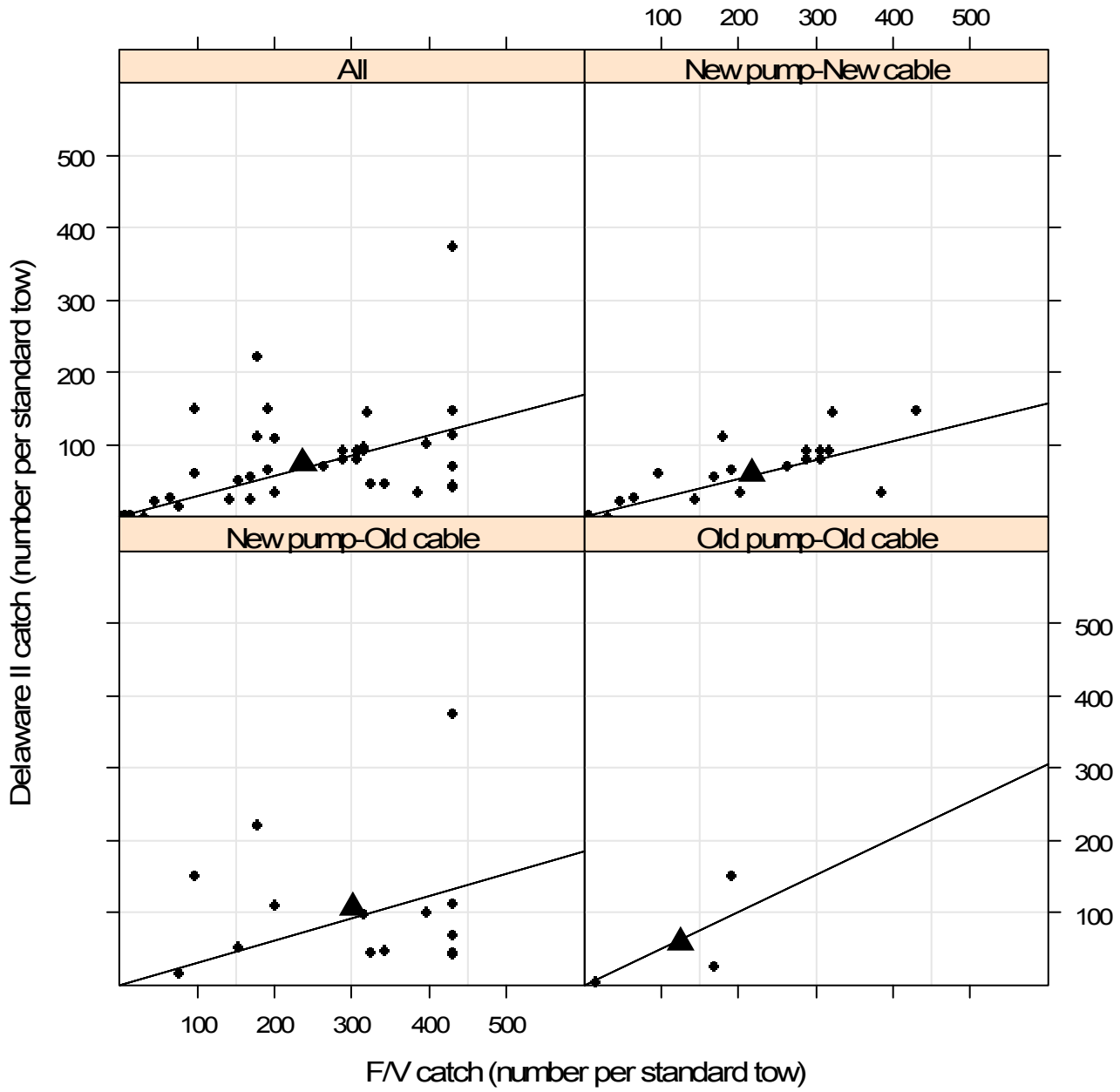


Figure B41. Catch per standard tow in DE2FV (Delaware II – F/V Endurance) repeat tows. The solid line in each panel is a regression line forced through the origin. The dark triangle in each plot shows the mean catch by both vessels.

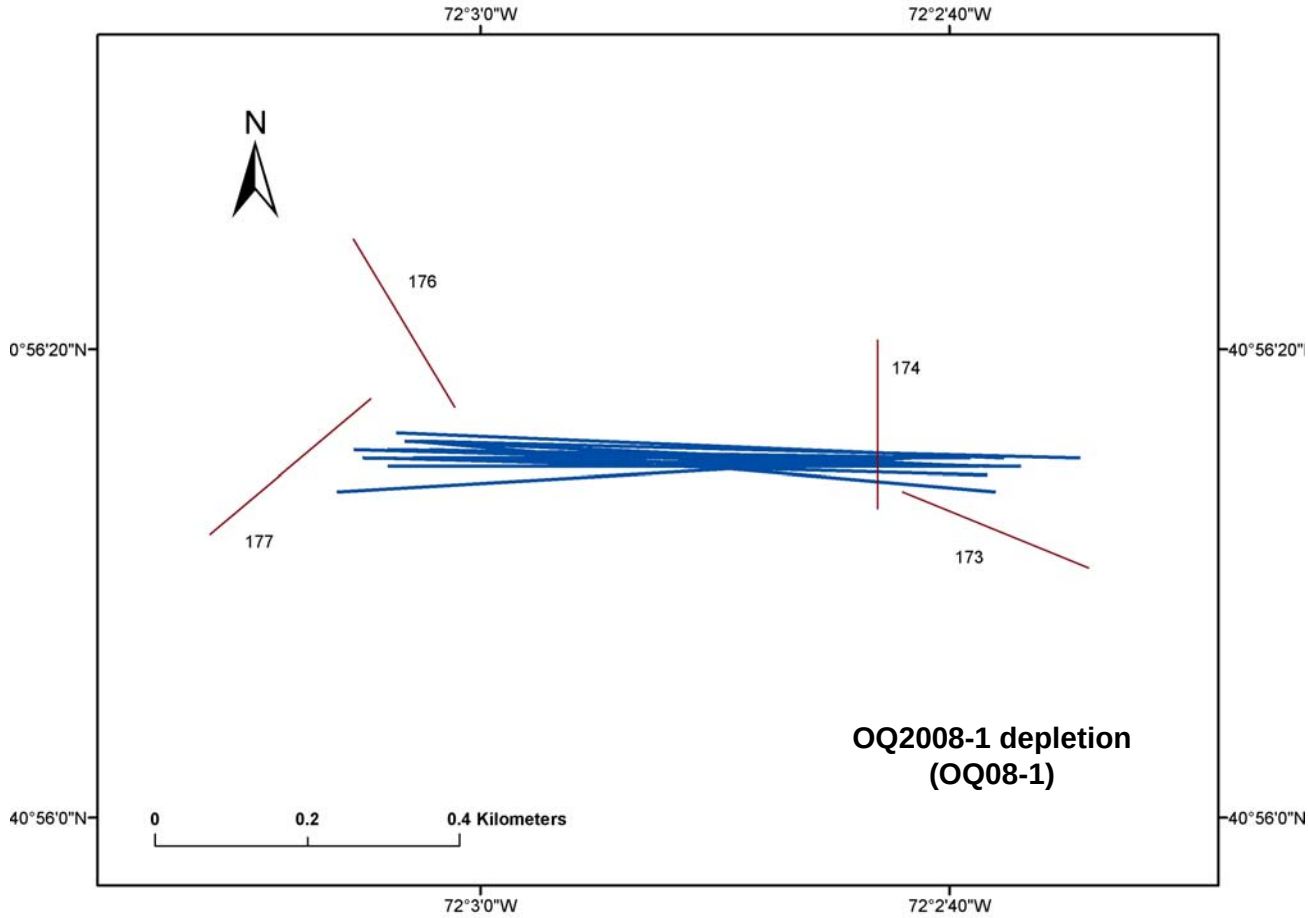


Figure B42. Depletion and setup tows for the OQ2008-1 commercial depletion experiments.

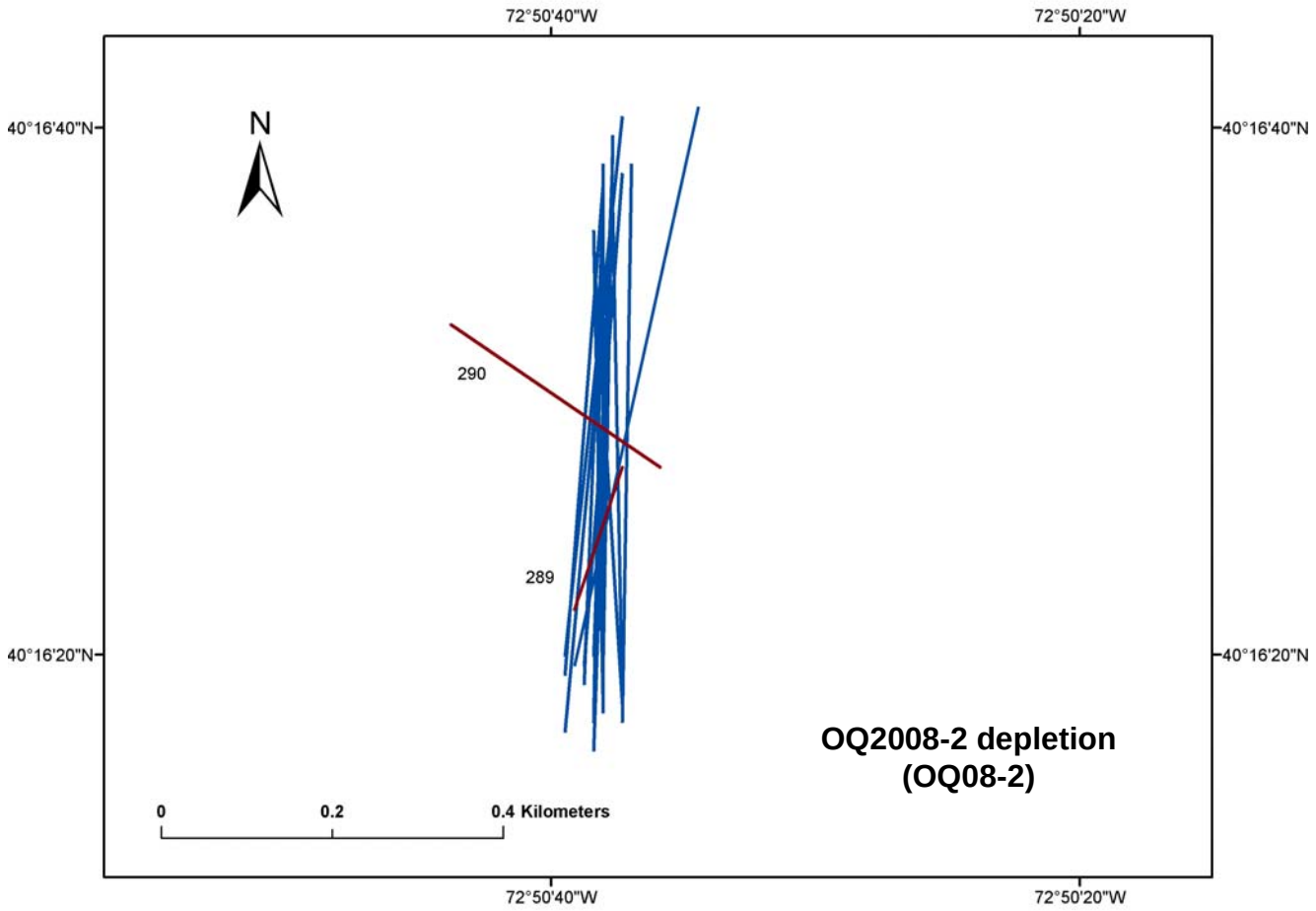


Figure B43. Depletion and setup tows for the OQ2008-2 commercial depletion experiments. The setup tow at station 289 is located under the depletion tows and may not be visible.

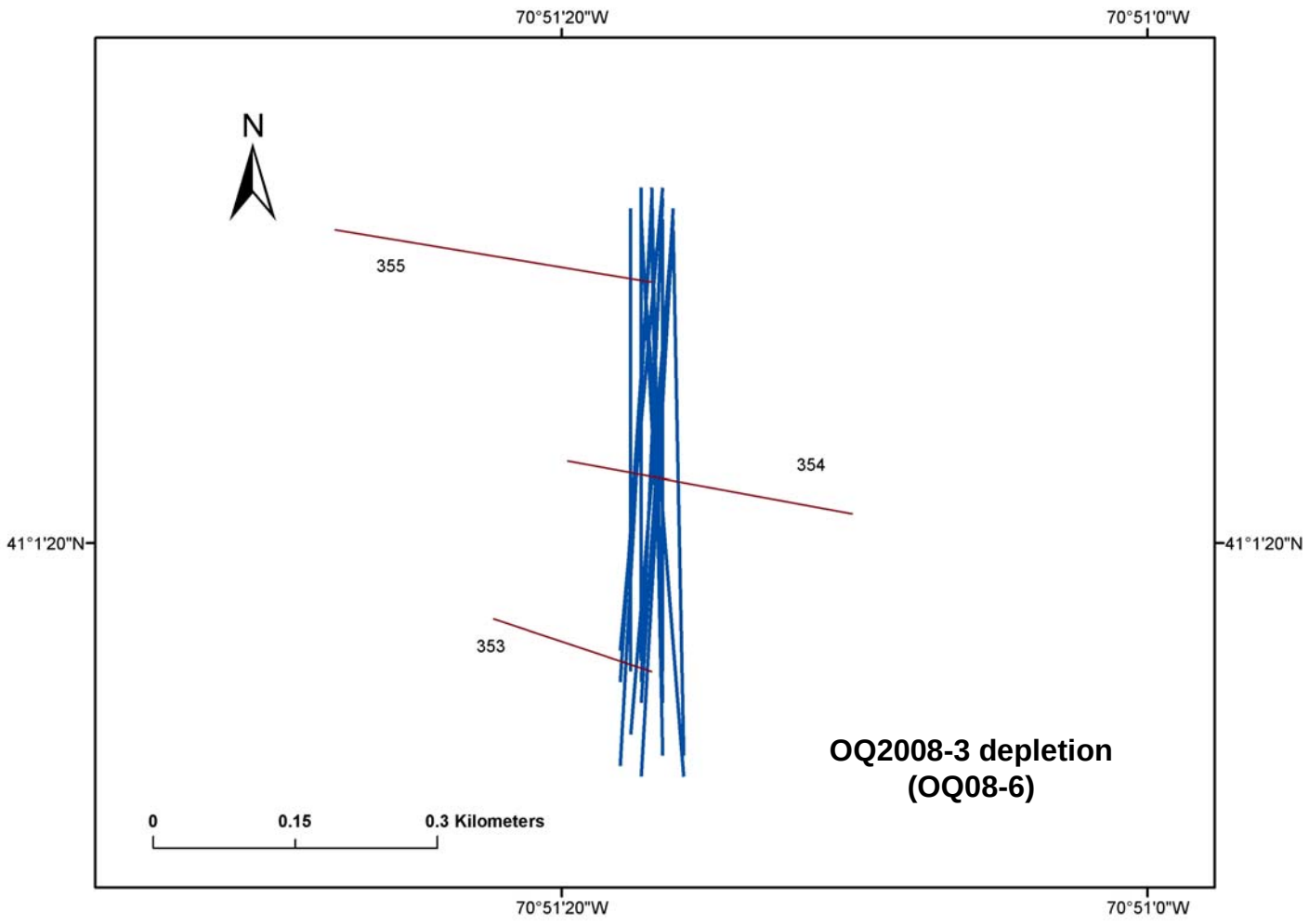


Figure B44. Depletion and setup tows for the OQ2005-3 commercial depletion experiments.

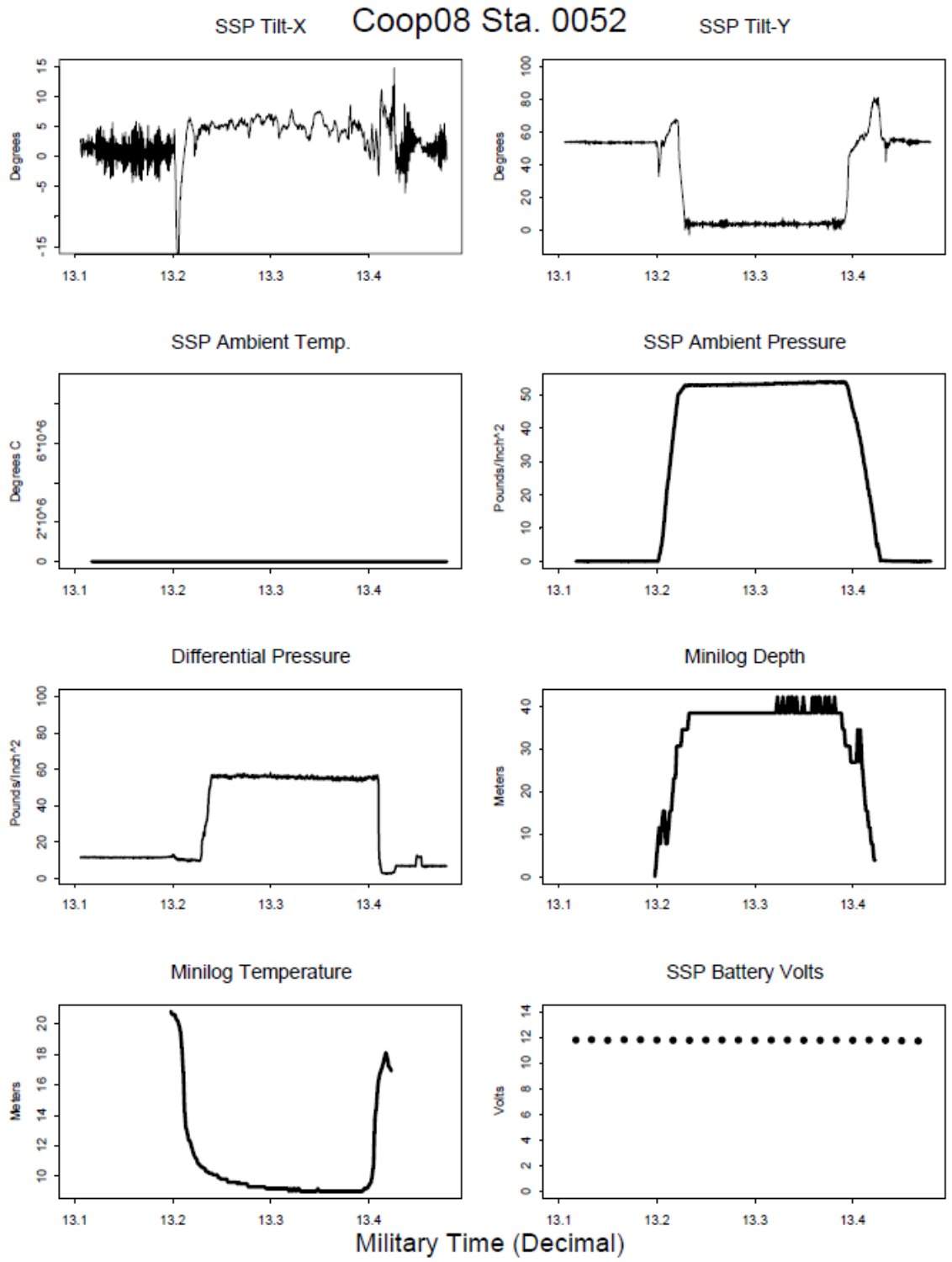


Figure B45. SSP sensor data for a tow by the F/V Endeavor during the 2008 cooperative clam survey.

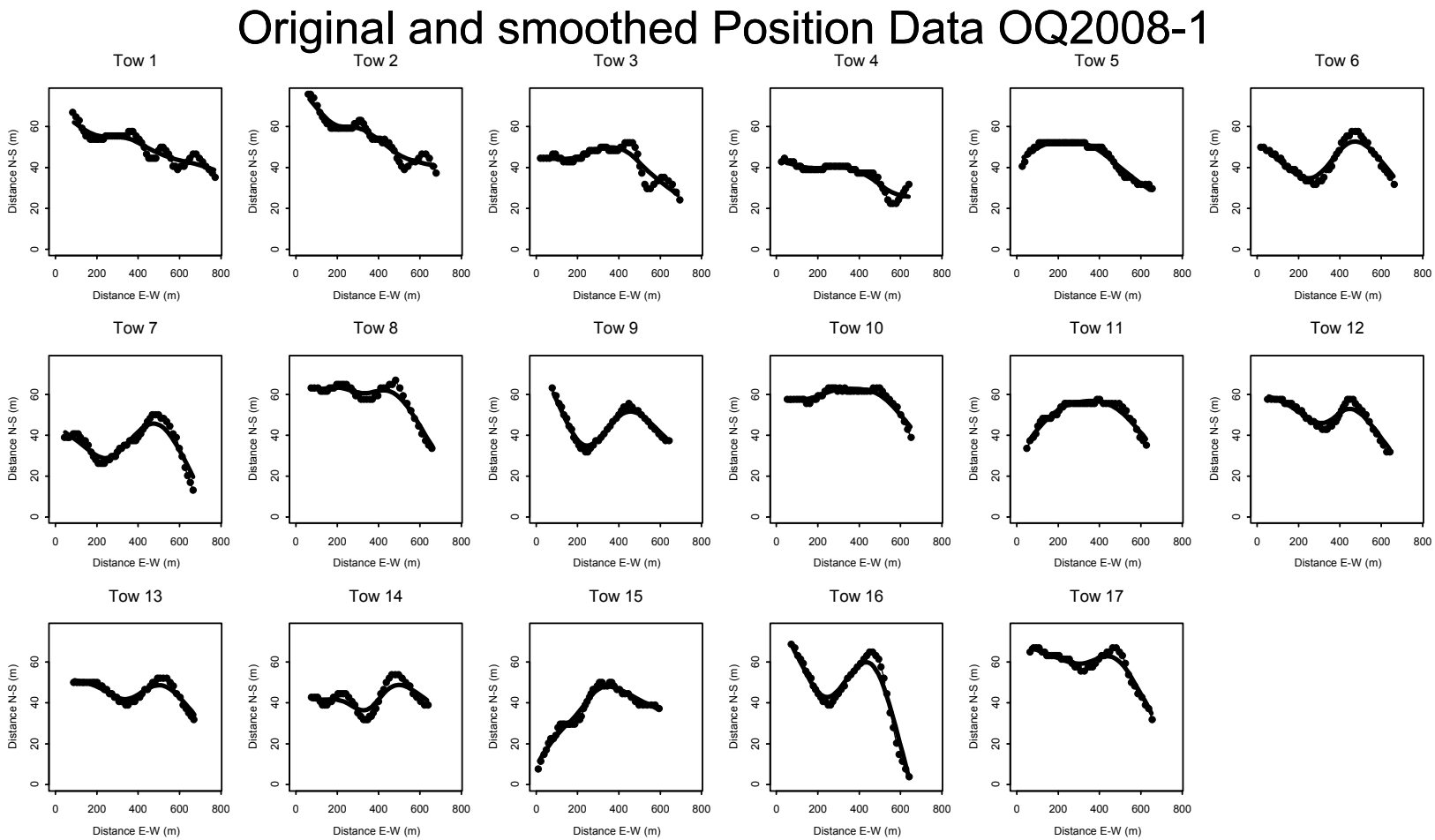


Figure B46. Original and smoothed position data for the OQ2008-1 commercial depletion study.

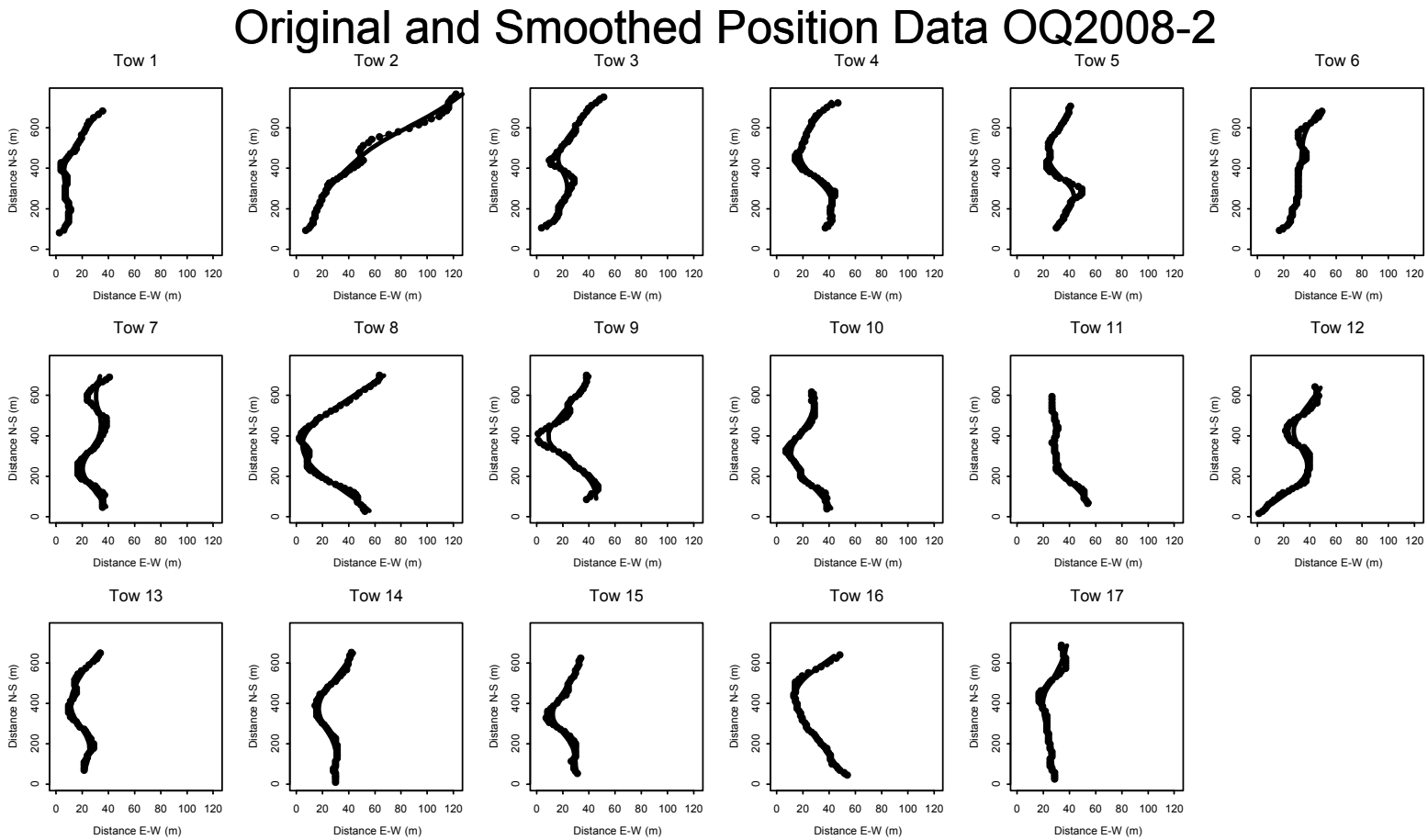


Figure B47. Original and smoothed position data for the OQ2008-2 commercial depletion study.

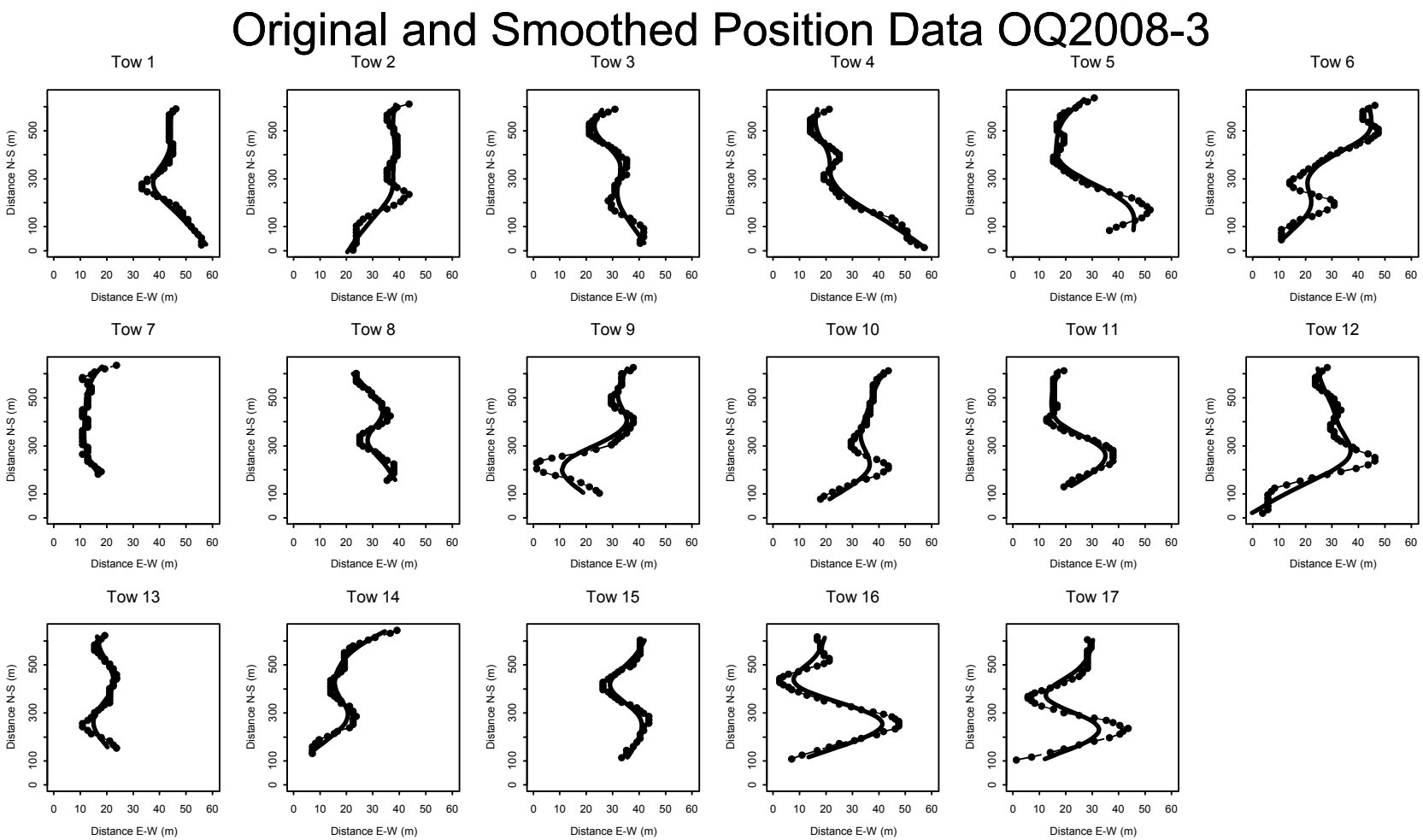


Figure B48. Original and smoothed position data for the OQ2008-3 commercial depletion study.

OQ2008-1 Patch model estimates, goodness of fit and likelihood profiles.

Density	0.0676
Efficiency	0.99994
K	7.54808
Gamma	0.50001
NLL	118.47

Note: estimates for ocean quahogs 90+ mm SL.

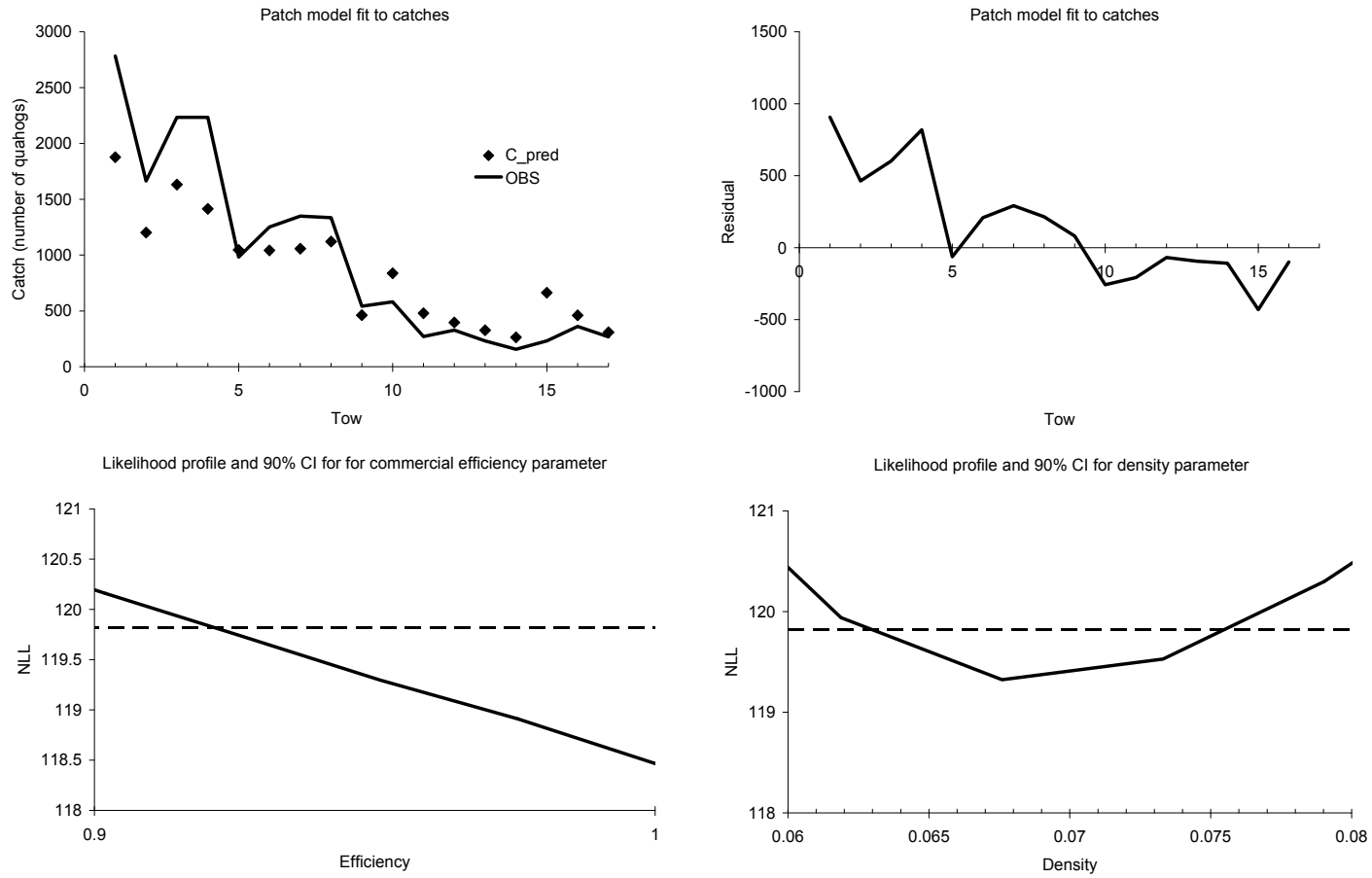
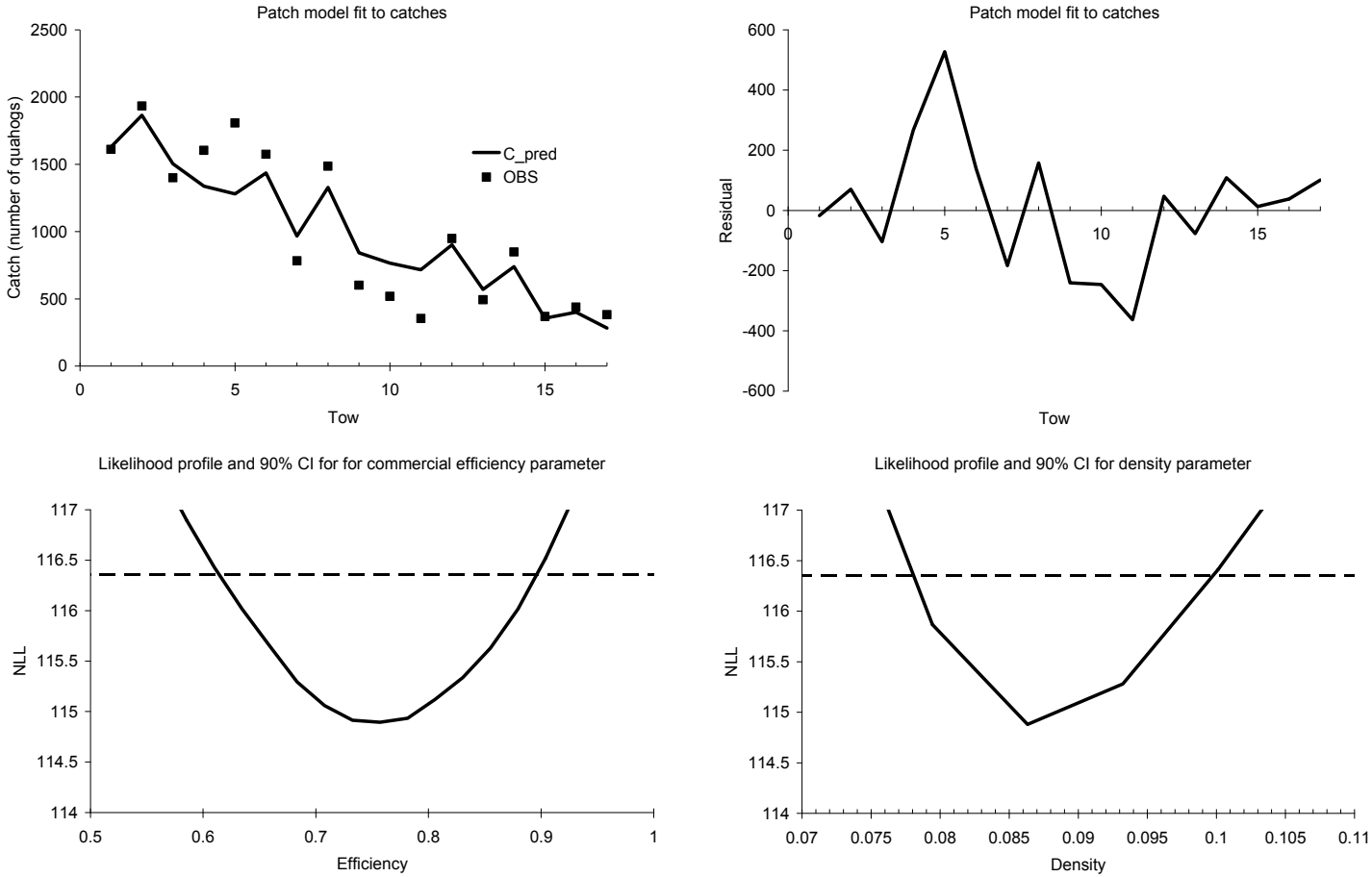


Figure B49. Goodness of fit and likelihood profile confidence intervals for the Patch model estimates for the OQ2008-1 commercial depletion study.

OQ2008-2 Patch model estimates, goodness of fit and likelihood profiles.

Density	0.08633	
Efficiency	0.78149	
K	14.5478	Note: estimates for ocean quahogs 90+ mm SL.
Gamma	0.50001	
NLL	115.002	



FigureB50. Goodness of fit and likelihood profile confidence intervals for the Patch model estimates for the OQ2008-2 commercial depletion study.

OQ2008-3 Patch model estimates, goodness of fit and likelihood profiles.

Density	0.11981	
Efficiency	1	
K	5.95464	Note: estimates for ocean quahogs 90+ mm SL.
Gamma	0.50001	
NLL	127.49	

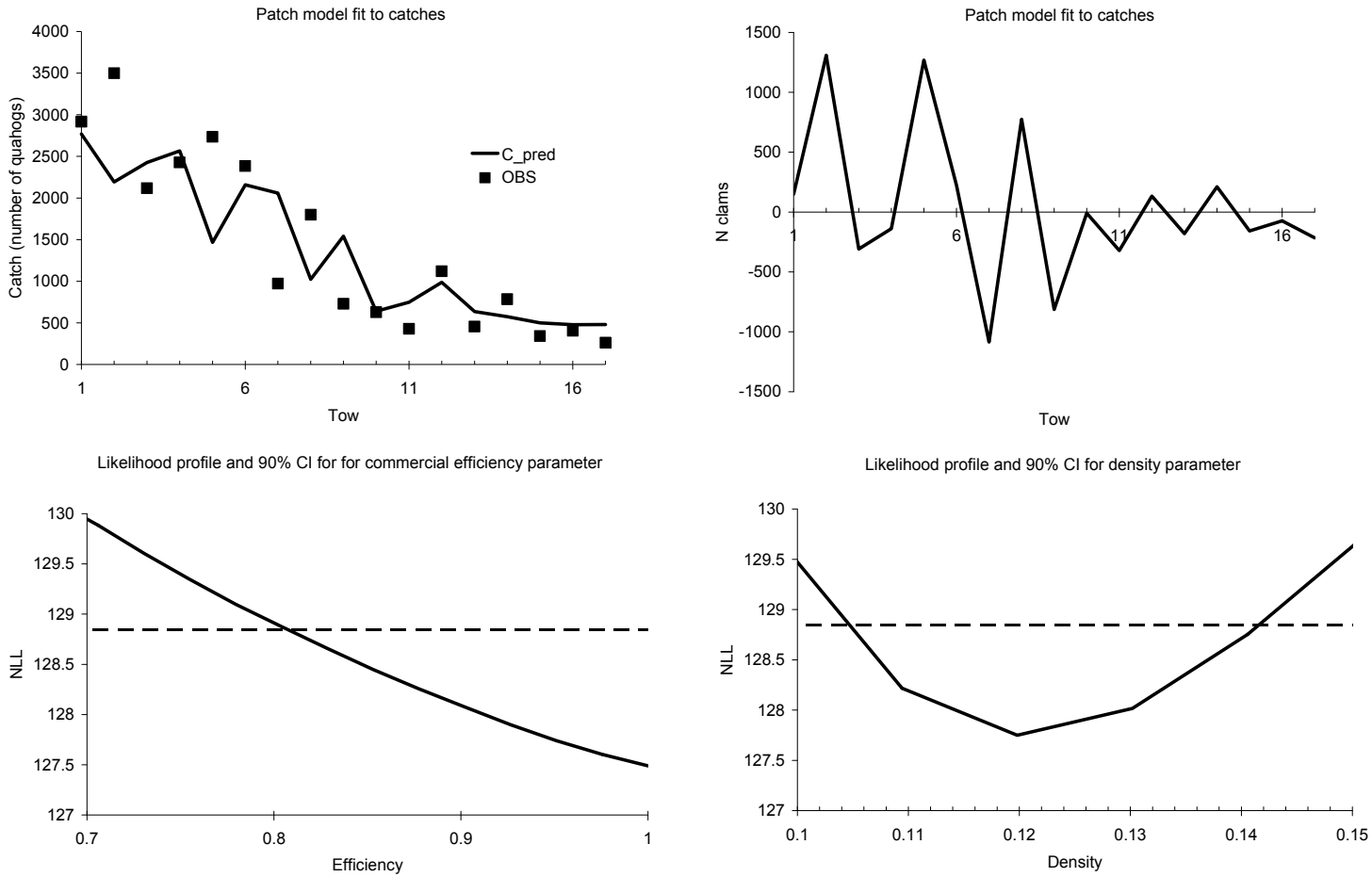


Figure B51. Goodness of fit and likelihood profile confidence intervals for the Patch model estimates for the OQ2008-3 commercial depletion study.

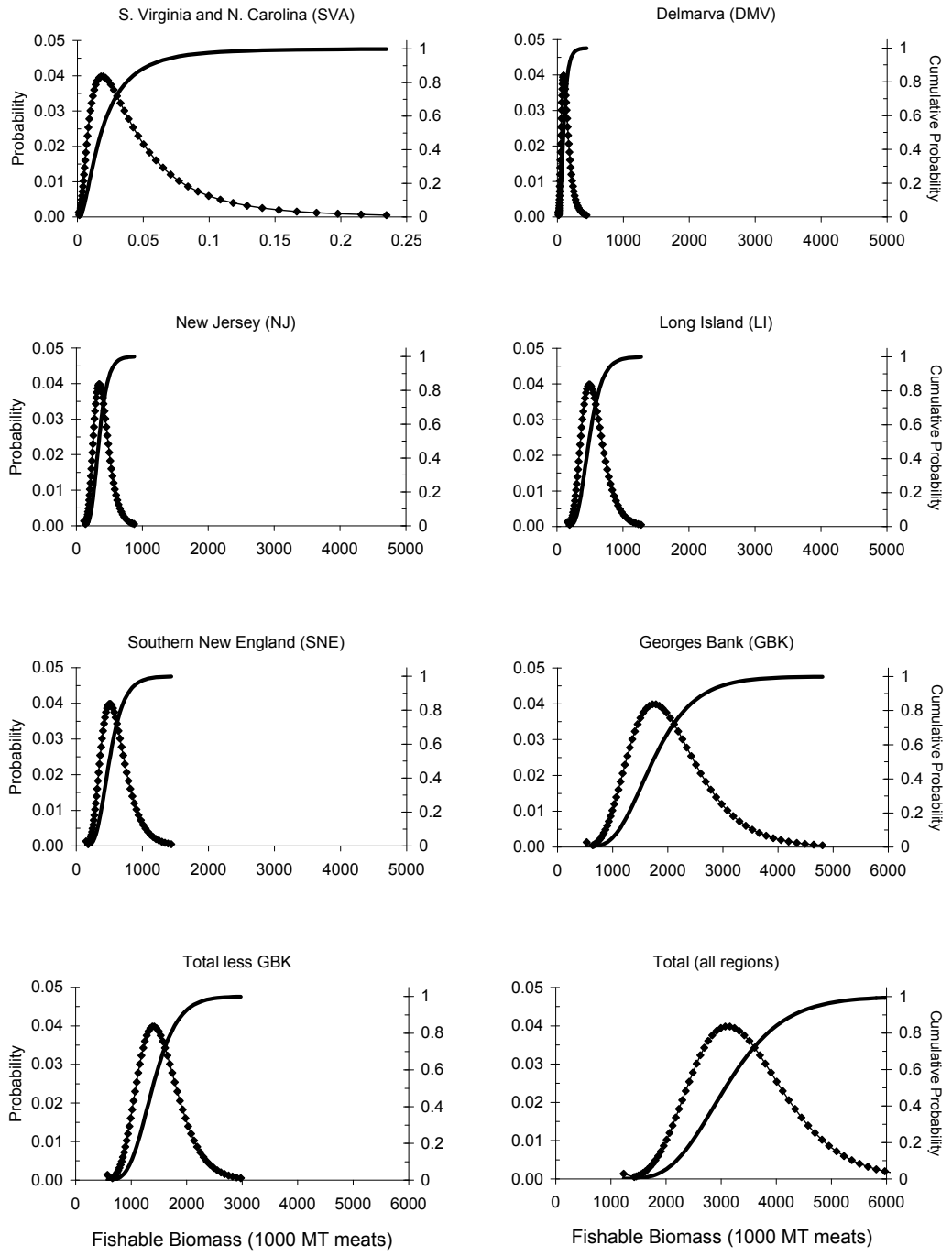


Figure B52. Uncertainty in efficiency corrected swept area biomass ESB) estimates for fishable ocean quahog during 2008. Note that the x-axis differs in the panel for SVA but is the same in all other panels to facilitate comparisons.

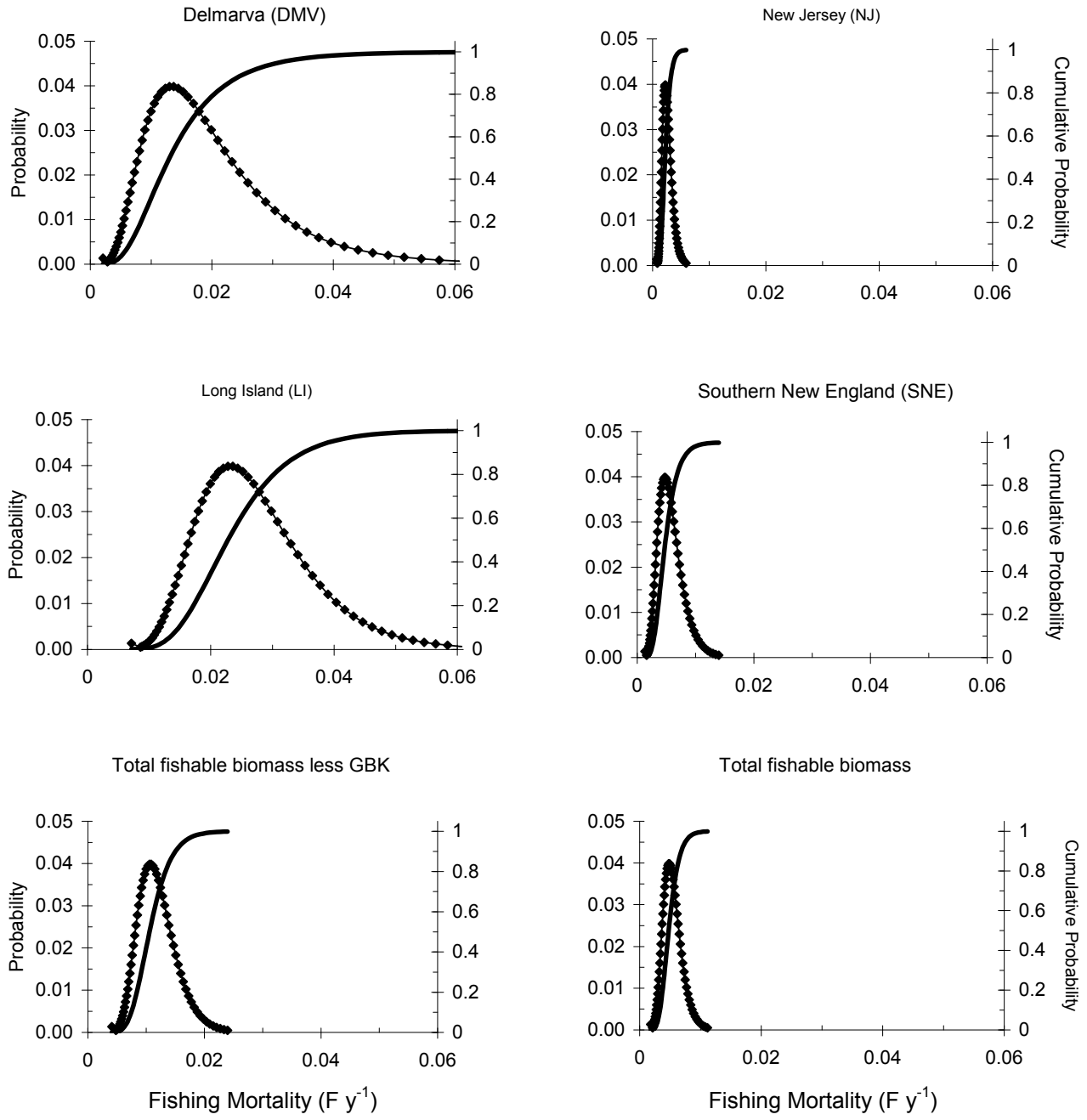


Figure B53. Uncertainty in fishing mortality estimates for ocean quahog during 2008 based on catch data and efficiency corrected swept-area biomass. X-axes are scaled to the same maximum to facilitate comparisons.

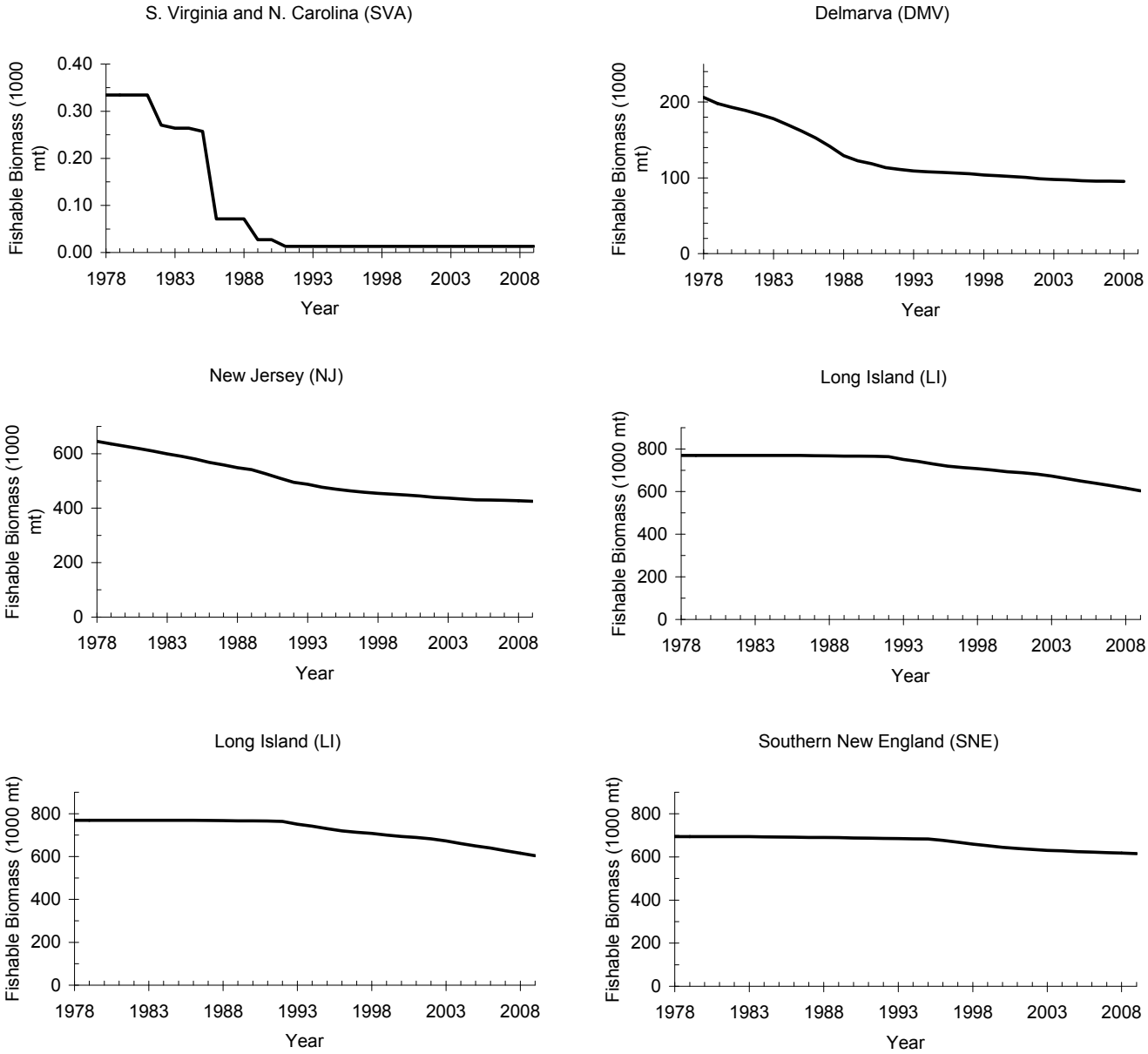


Figure B54. Trends in fishable biomass for ocean quahog from the "VPA" method during 1978-2009, by region. The VPA estimate for GBK is the mean of ESB estimates for 2002, 2005 and 2008 because no catch occurs in GBK.

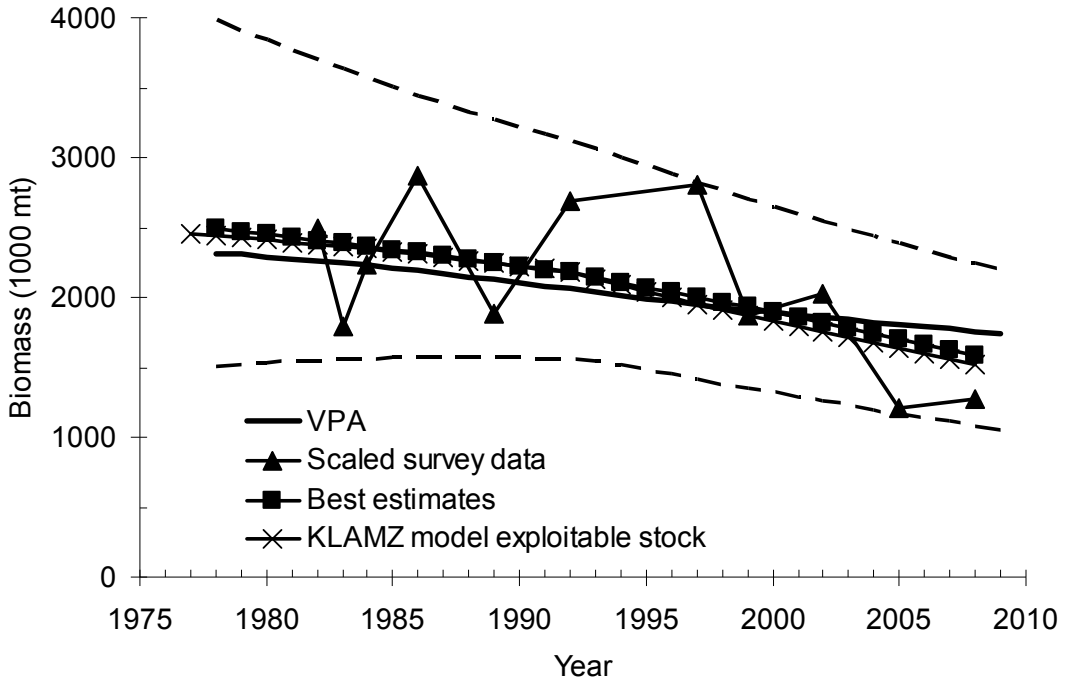


Figure B55. Biomass estimates for ocean quahogs in the exploited region with survey trend data adjusted to the same scale. Estimates are from: i) the sum of best estimates in this assessment (VPA model for SVA and regional KLAMZ models for other areas); ii) VPA (sum of regional VPA estimates); and a KLAMZ model fit to the entire exploited region. The dashed lines show an asymmetric confidence interval for the KLAMZ model fit to the entire exploited region.

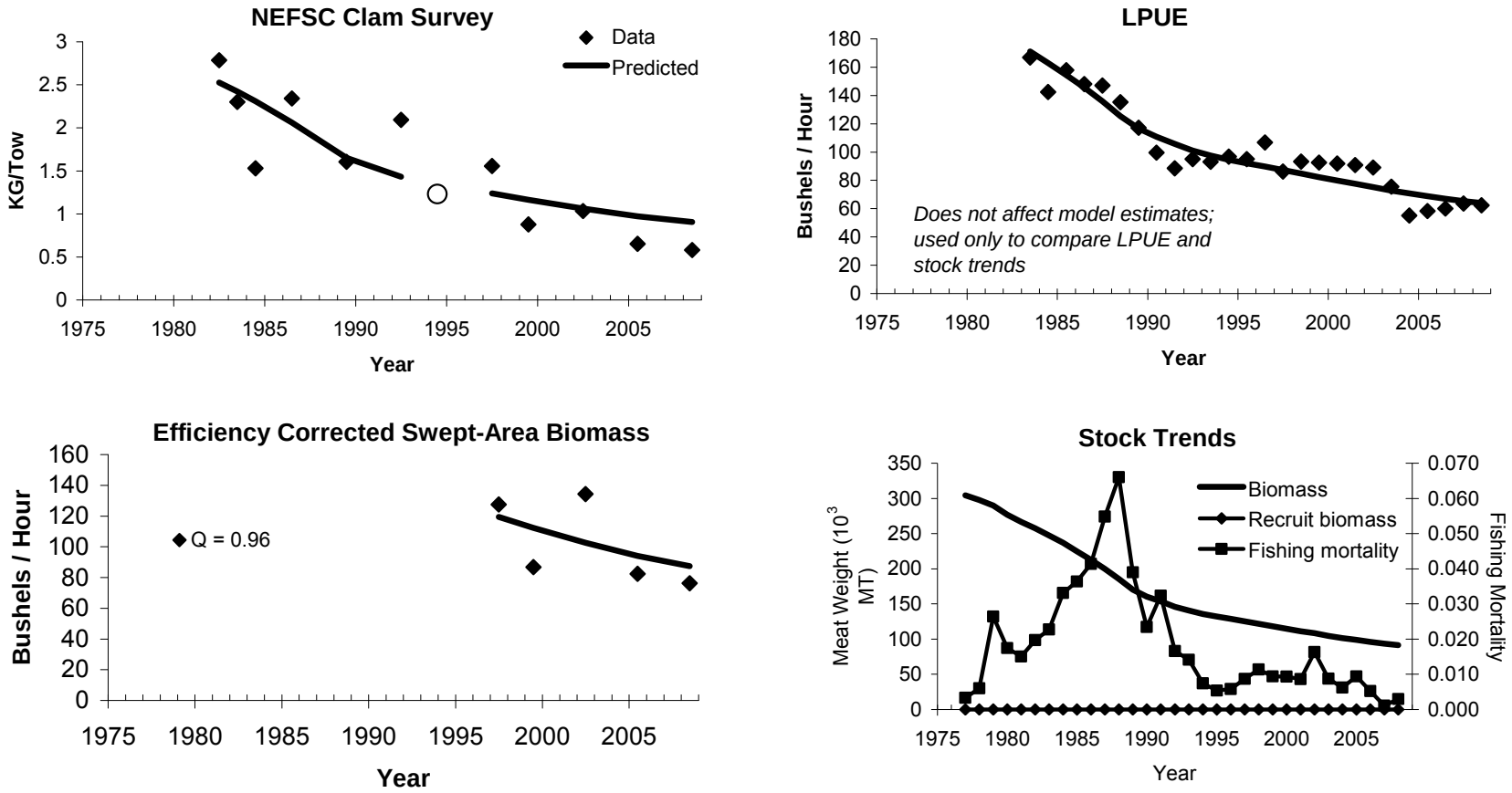


Figure B56. KLAMZ model results for ocean quahog in the DMV stock assessment region during 1977-2008. The bottom right panel shows population estimates. Other panels show goodness of fit to survey, LPUE and swept area biomass trend data. Results are for a KLAMZ model run with M=0.02 y-1 and recruitment biomass fixed near zero. The survey scaling parameter estimate for ESB data is shown in the bottom left panel. The 1994 clam survey observation (open circle) was not used in fitting the model.

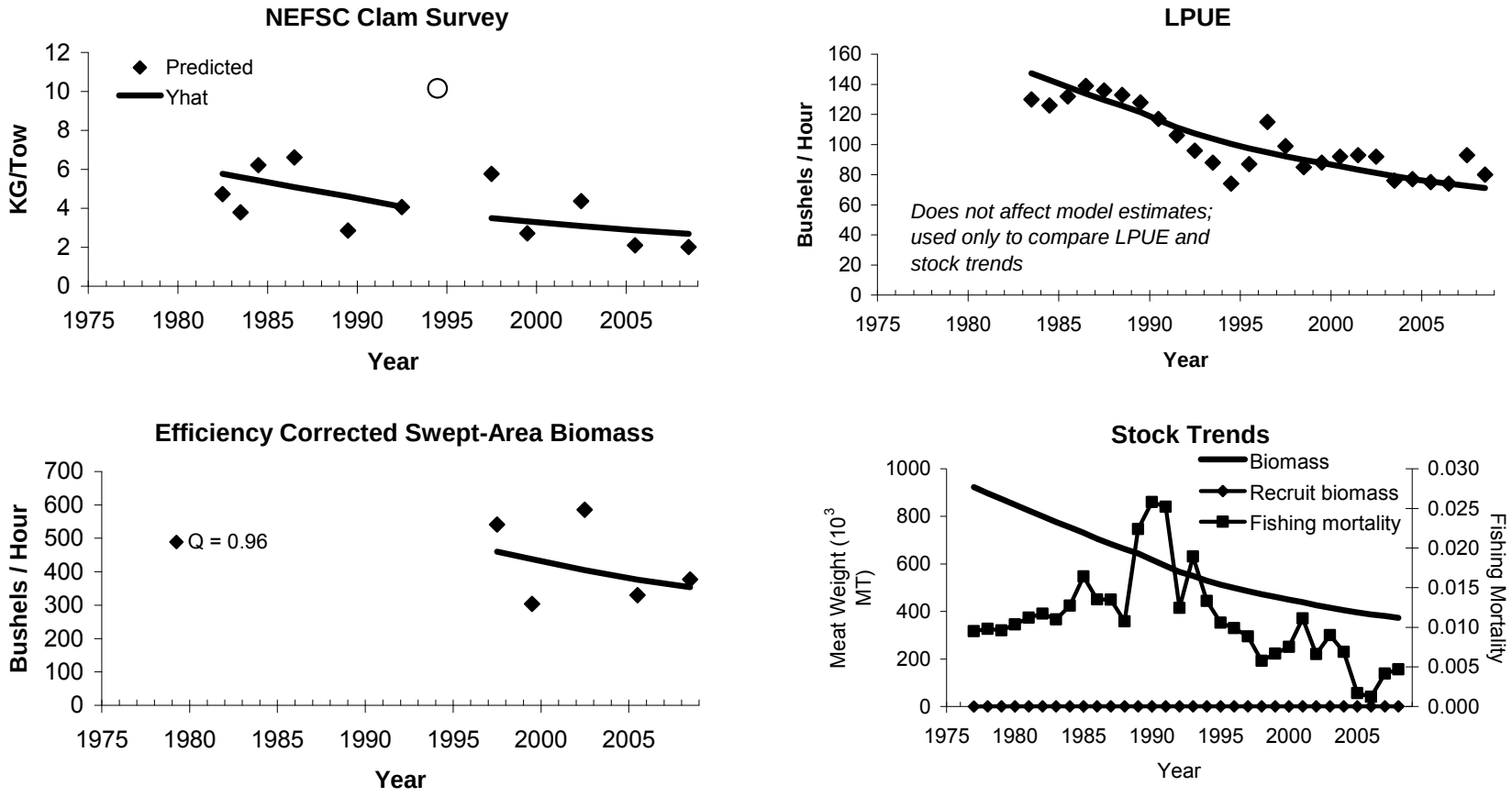


Figure B57. KLAMZ model results for ocean quahog in the NJ stock assessment region during 1977-2008. The bottom right panel shows population estimates. Other panels show goodness of fit to survey, LPUE and swept area biomass trend data. Results are for a KLAMZ model run with M=0.02 y-1 and recruitment biomass estimated at a relatively low level. The survey scaling parameter estimate for ESB data is shown in the bottom left panel. The 1994 clam survey observation (open circle) was not used in fitting the model.

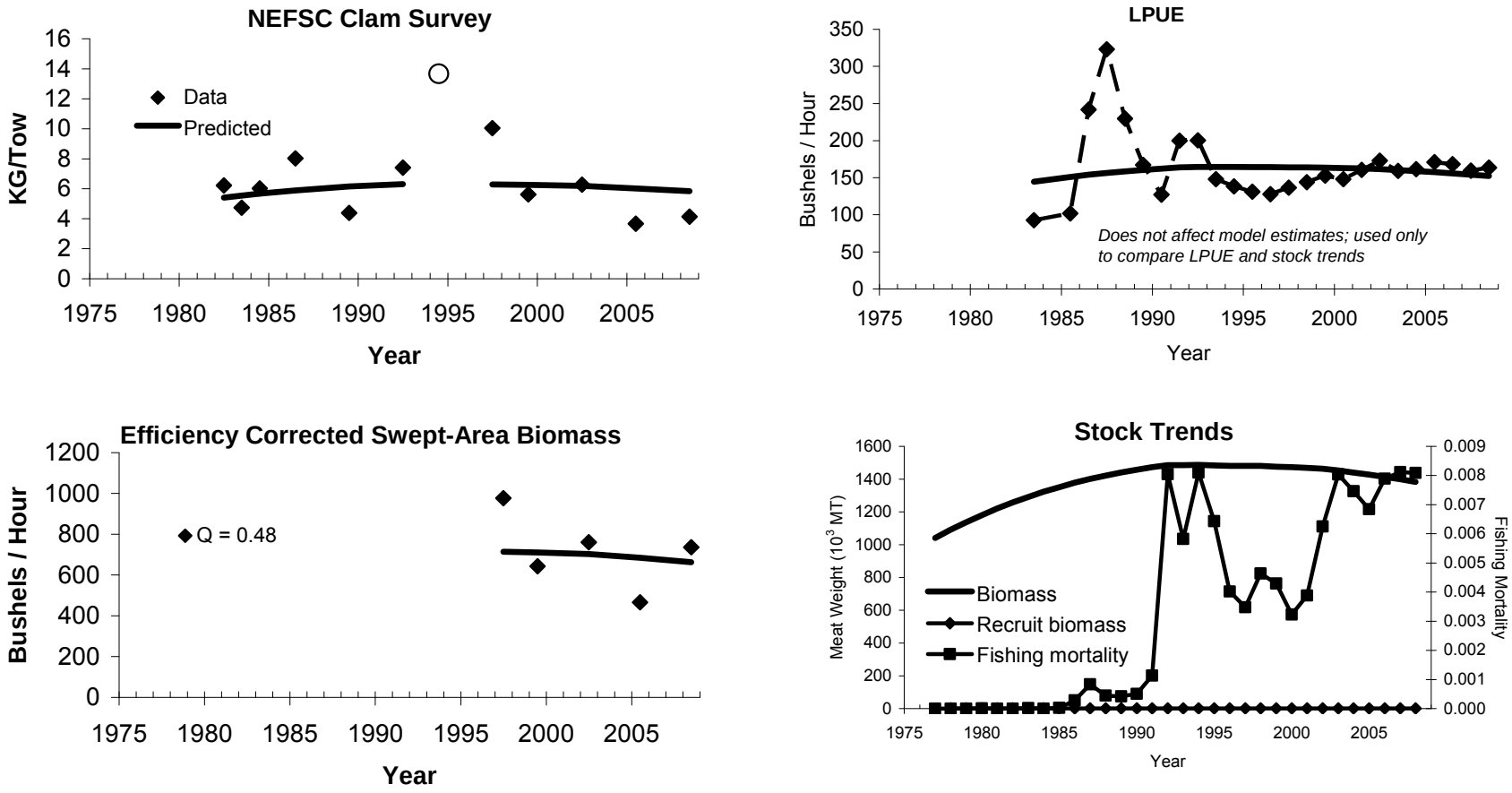


Figure B58. Preliminary results from a KLAMZ model with constant recruitment for ocean quahog in the LI stock assessment region during 1977-2008. Note the slight lack of fit to recent survey data (top left panel) and the anomalous survey scaling coefficient value (Q=0.48) for efficiency corrected swept area biomass (bottom left panel).

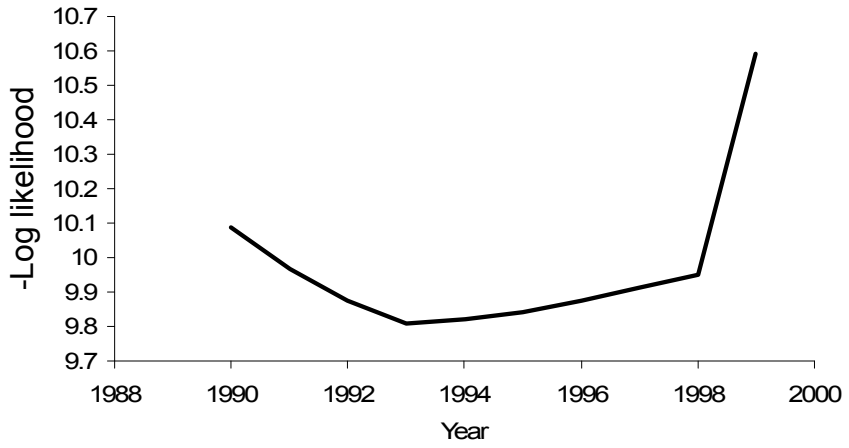


Figure B59. Profile likelihood analysis to determine the change year for the step recruitment function in the KLAMZ model for LI.

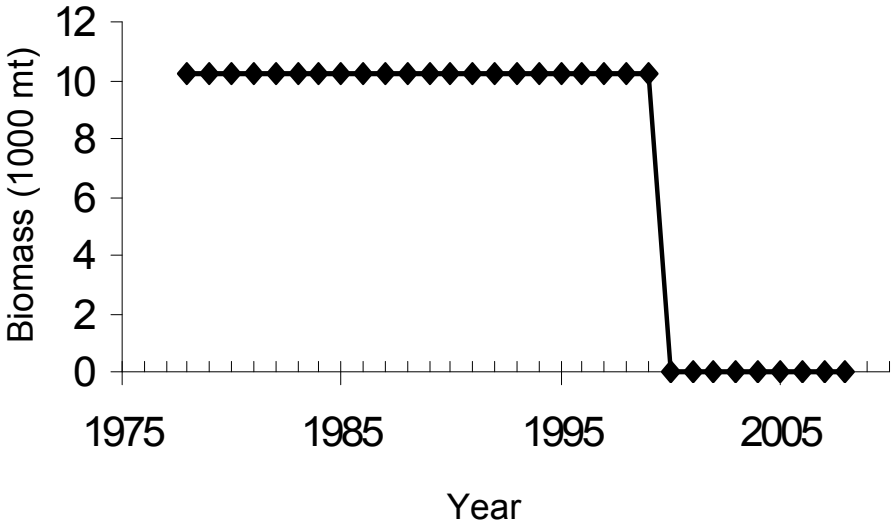


Figure B60. Step function recruitment estimates from the KLAMZ model for LI.

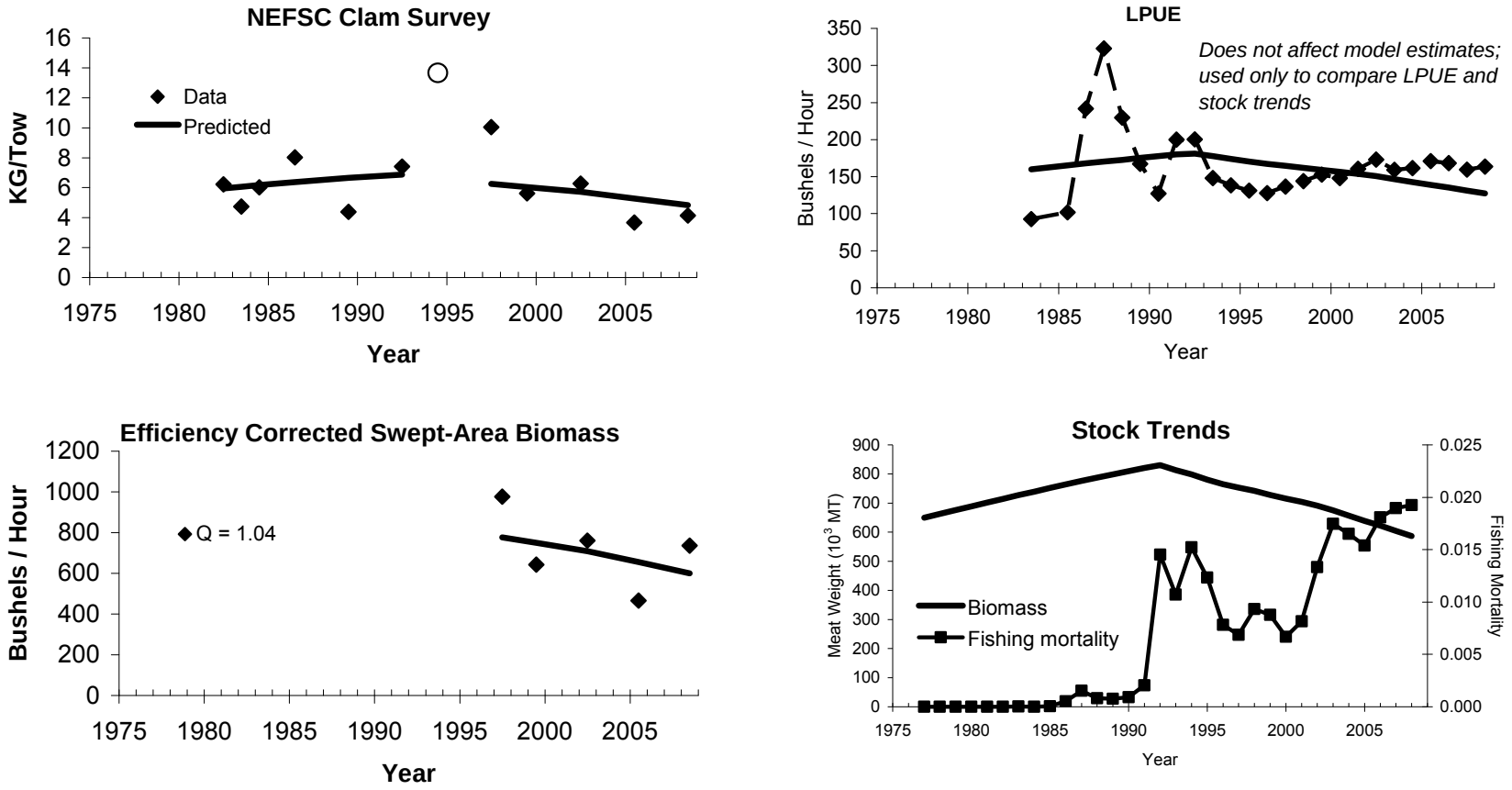


Figure B61. KLAMZ model results for ocean quahog in the LI stock assessment region during 1977-2008. The bottom right panel shows population estimates. Other panels show goodness of fit to survey, LPUE and swept area biomass trend data. Results are for a KLAMZ model run with M=0.02 y-1 and recruitment biomass estimated using a step function with the second period starting in 1994 (Figure K5). The survey scaling parameter estimate for ESB data is shown in the bottom left panel. The 1994 clam survey observation (open circle) was not used in fitting the model.

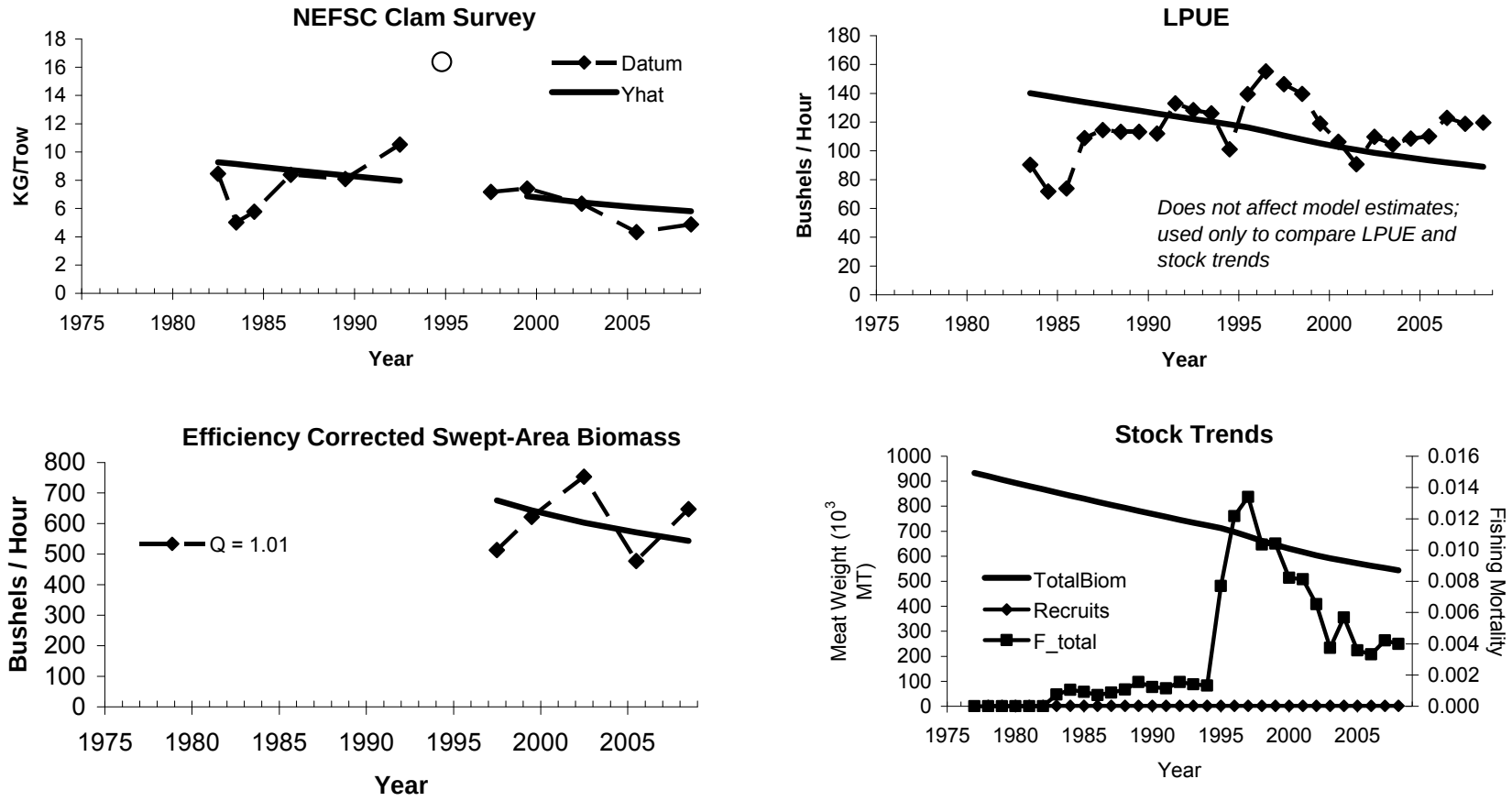


Figure B62. Preliminary results from a KLAMZ model with constant recruitment for ocean quahog in the SNE stock assessment region during 1977-2008. Note lack of fit to survey data (top left panel).

Figure B63. Profile likelihood analysis to determine the change year for the step recruitment function in the KLAMZ model for SNE.

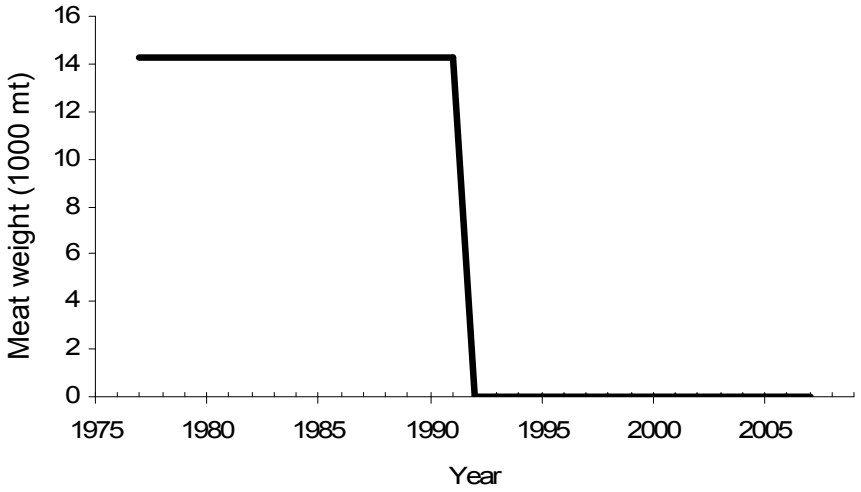


Figure B64. Step function recruitment estimates from the KLAMZ model for SNE.

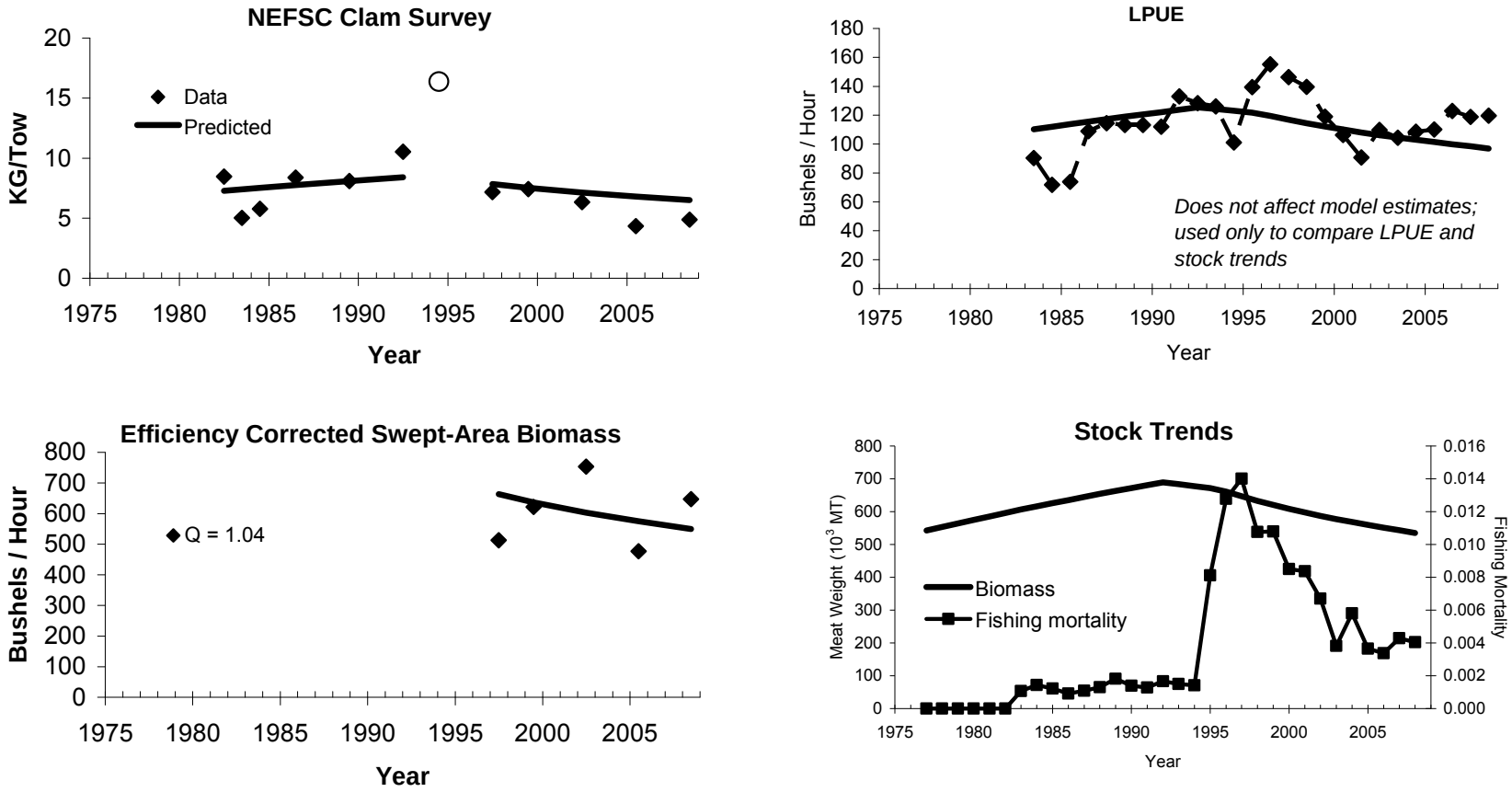


Figure B65. KLAMZ model results for ocean quahog in the SNE stock assessment region during 1977-2008. The bottom right panel shows population estimates. Other panels show goodness of fit to survey, LPUE and swept area biomass trend data. Results are for a KLAMZ model run with M=0.02 y-1 and recruitment biomass estimated using a step function with the second period starting in 1994 (Figure K5). The survey scaling parameter estimate for ESB data is shown in the bottom left panel. The 1994 clam survey observation (open circle) was not used in fitting the model.

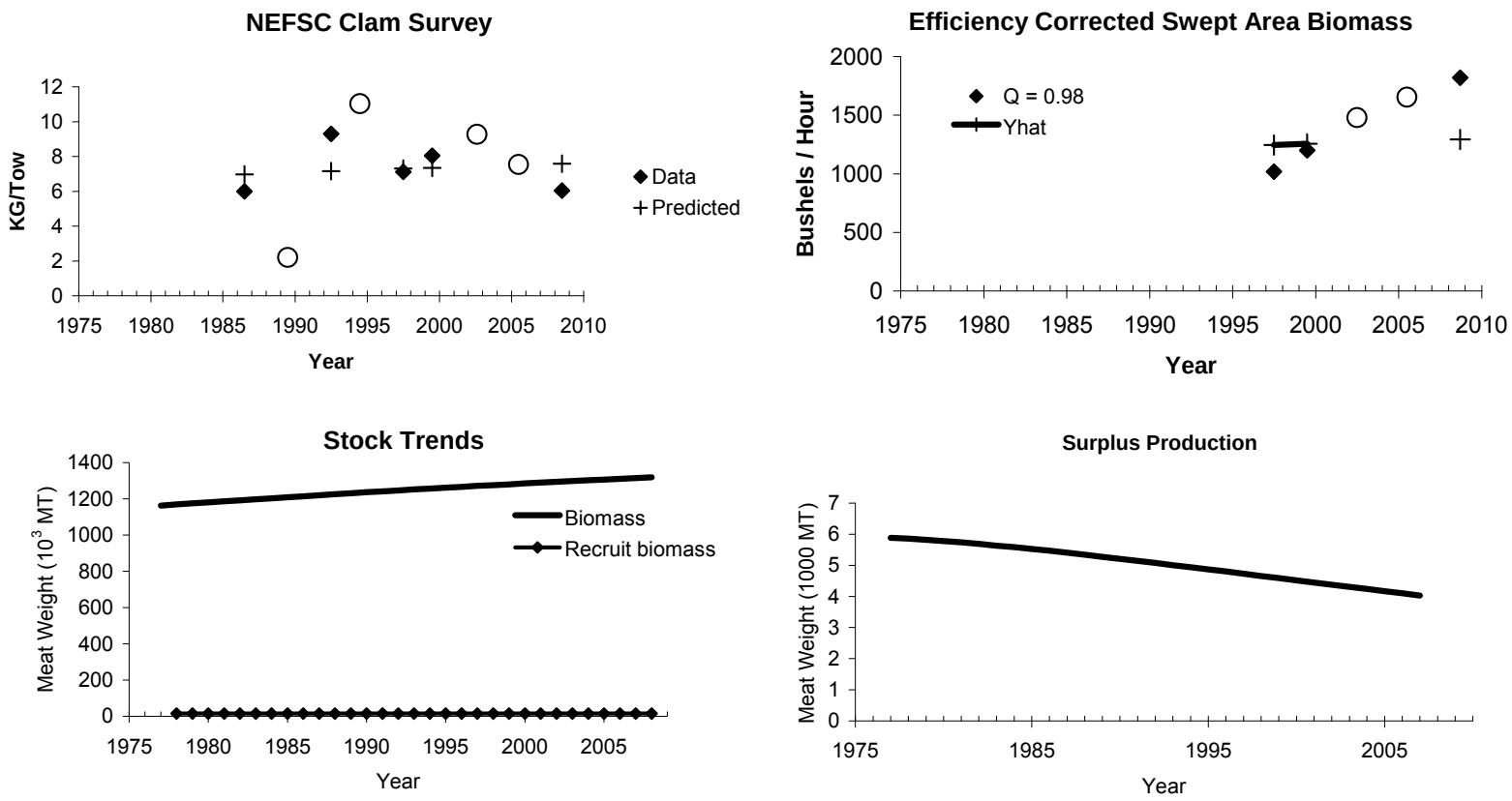


Figure B66. KLAMZ model results for ocean quahog in the GBK stock assessment region during 1977-2008. The bottom two panels show population estimates. Other panels show goodness of fit to survey and swept area biomass trend data. Results are for a KLAMZ model run with M=0.02 y-1 and recruitment biomass estimated at a relatively low level. The survey scaling parameter estimate for ESB data is shown in the bottom left panel. Survey and swept area biomass data for 1989, 1994, 2002 and 2005 (open circles) were not used in fitting the model due to voltage problems in 1994 and poor sampling in other years.

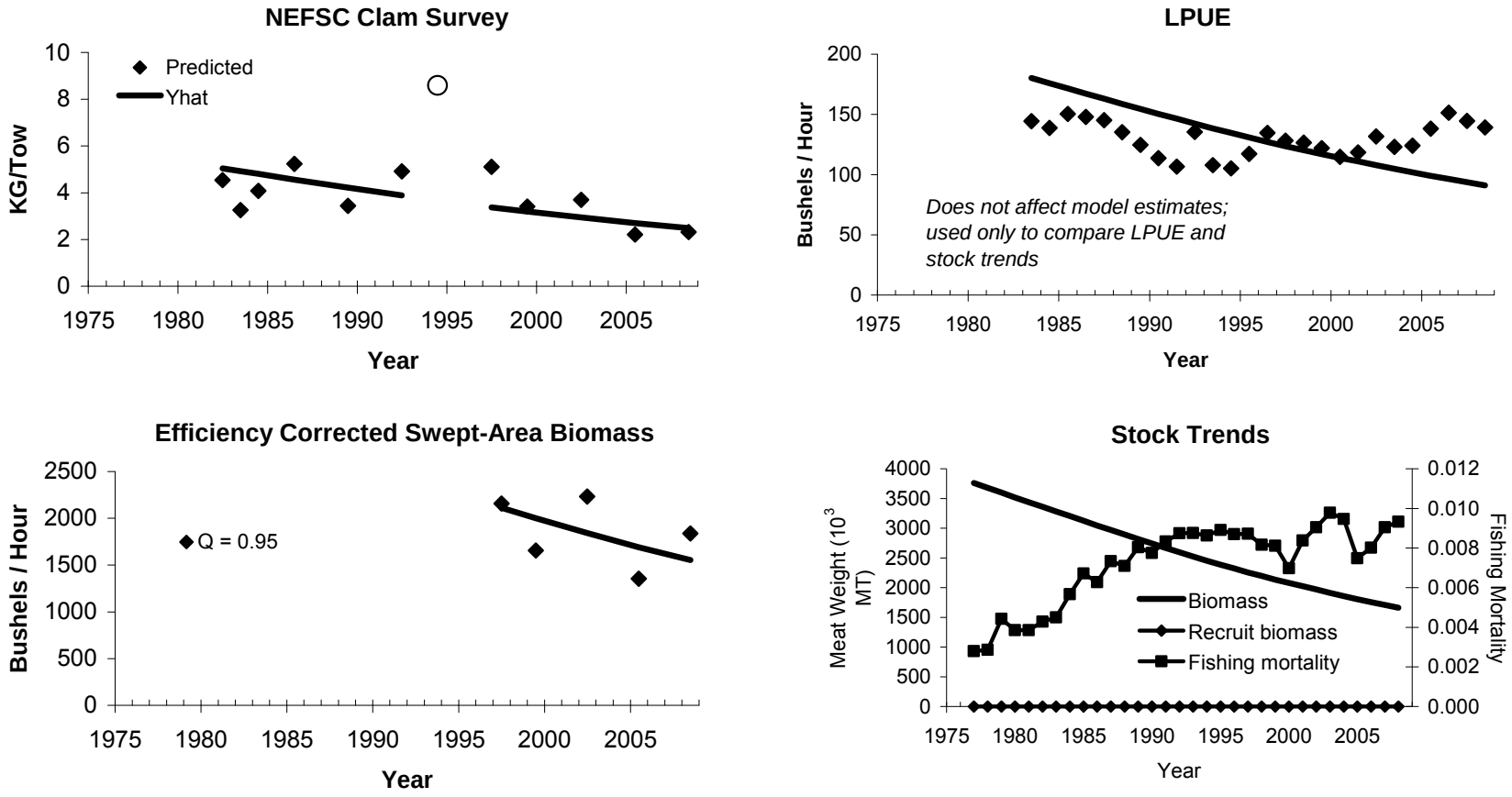


Figure B67. Preliminary results from a KLAMZ model with constant recruitment for ocean quahog in the exploited stock area during 1977-2008. Note lack of fit to survey data (top left panel).

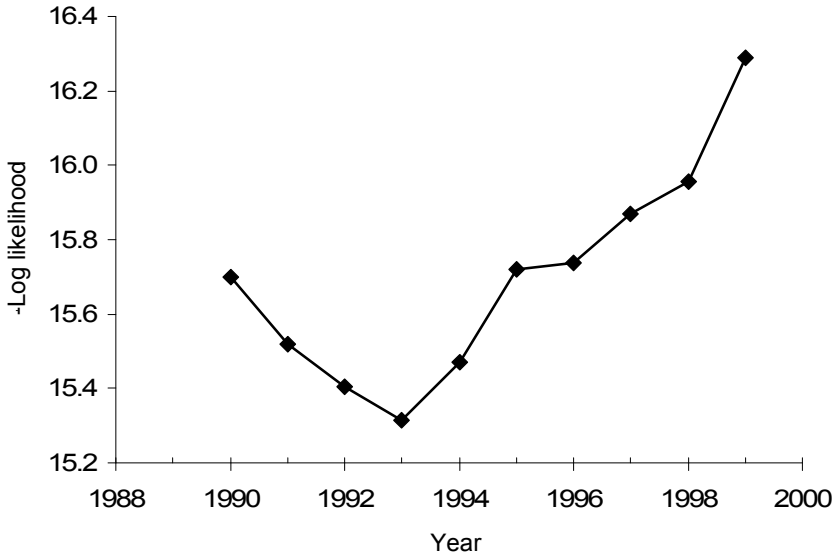


Figure B68. Profile likelihood analysis to determine the change year for the step recruitment function in the KLAMZ model for the exploited stock region.

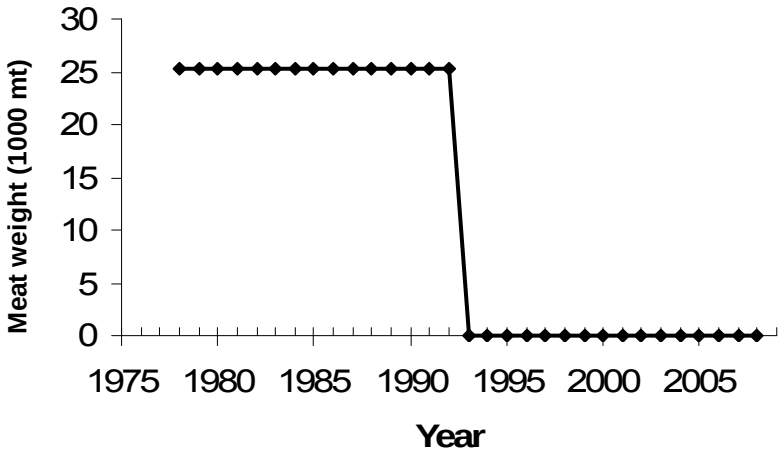


Figure B69. Step function recruitment estimates from the KLAMZ model for ocean quahogs in the exploited stock region.

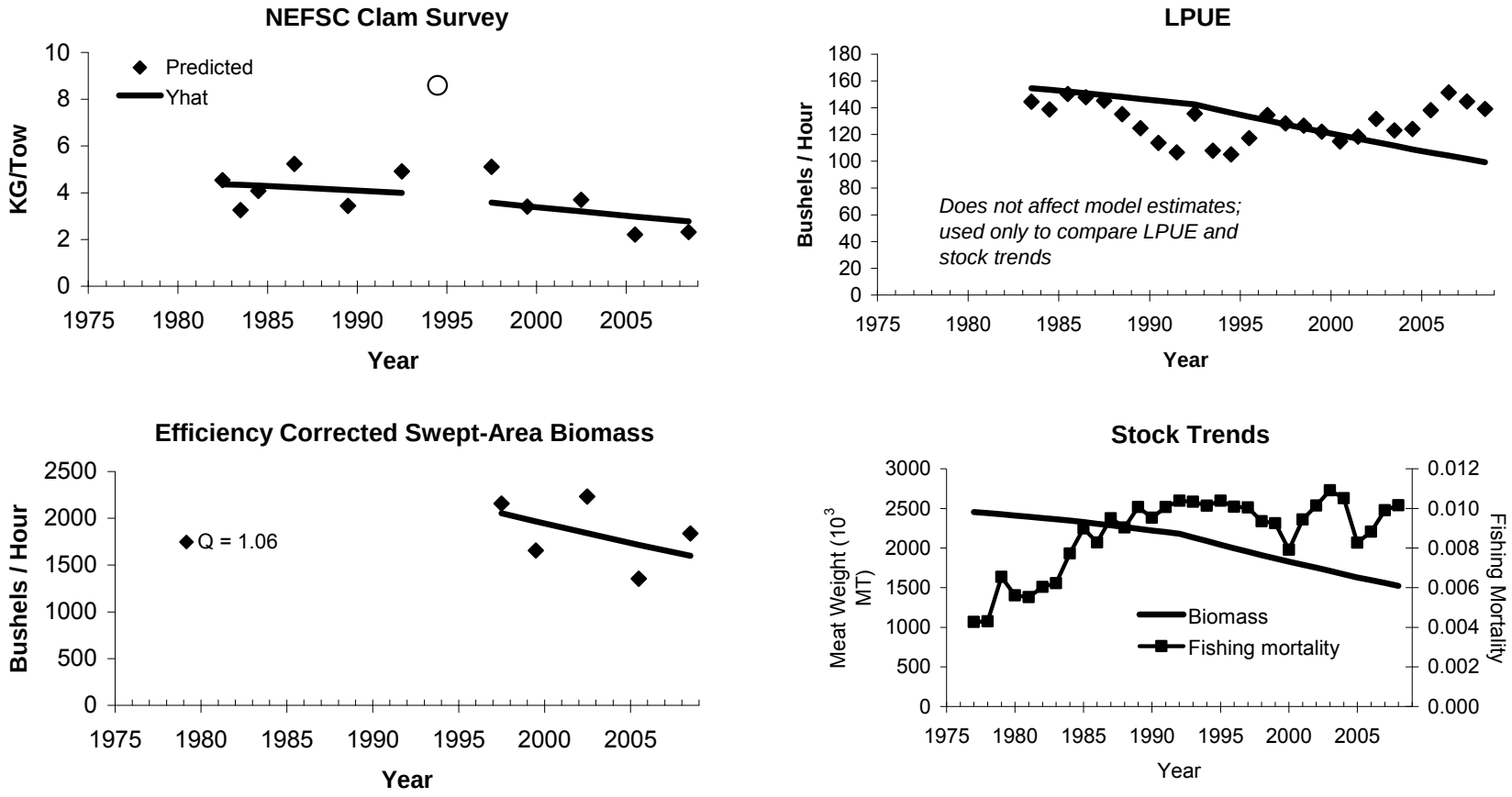


Figure B70. KLAMZ model results for ocean quahog in the LI stock assessment region during 1977-2008. The bottom right panel shows population estimates. Other panels show goodness of fit to survey, LPUE and swept area biomass trend data. Results are for a KLAMZ model run with M=0.02 y-1 and recruitment biomass estimated using a step function with the second period starting in 1994 (Figure B69). The survey scaling parameter estimate for ESB data is shown in the bottom left panel. The 1994 clam survey observation (open circle) was not used in fitting the model.

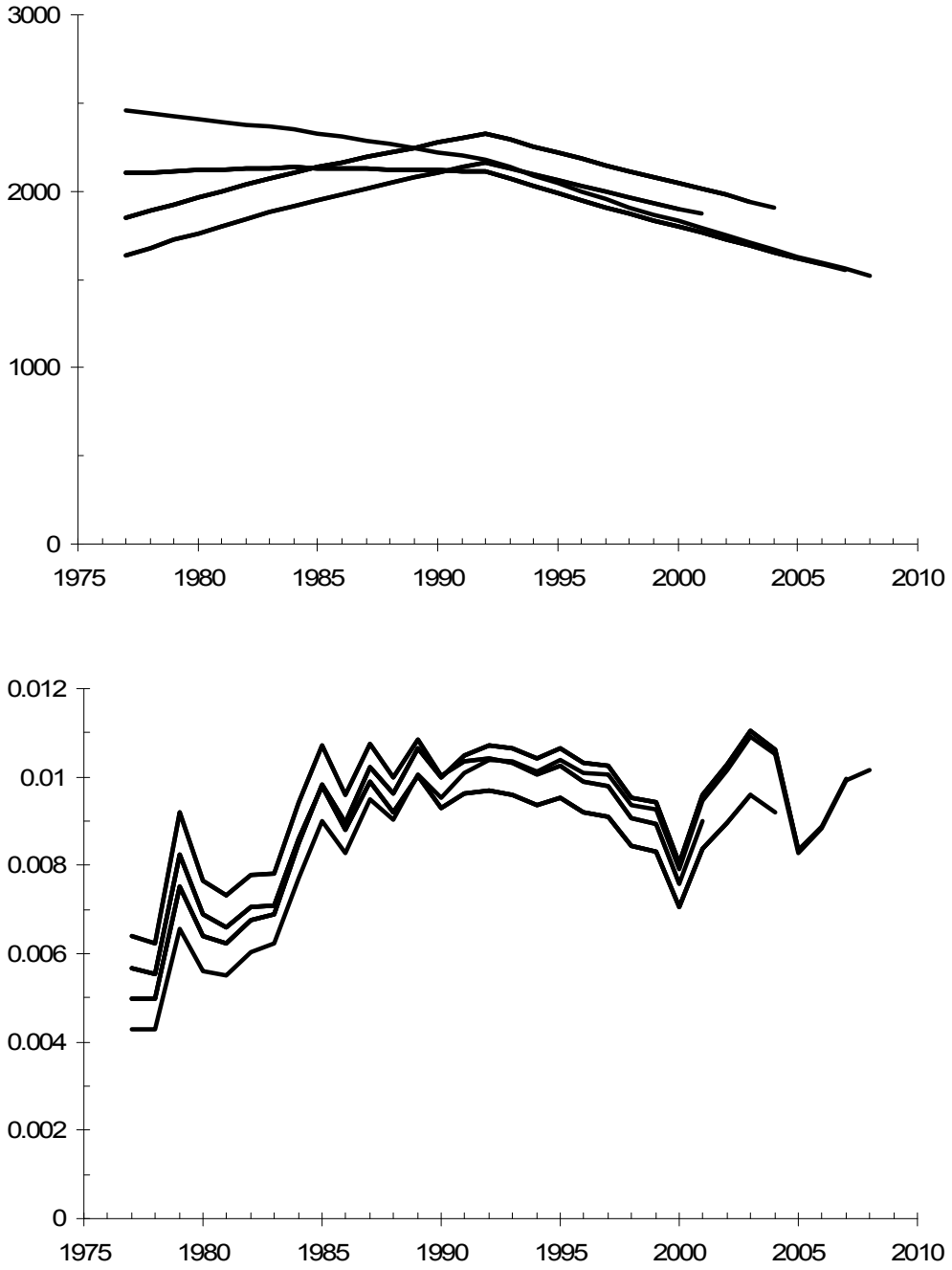


Figure B71. Retrospective analysis with the KLAMZ model for ocean quahogs in the exploited region with 2000-2008 as the terminal year. Results for some terminal years are not visible because the estimates were exactly the same as in an adjacent run (estimates may not change unless a year with survey data is omitted).

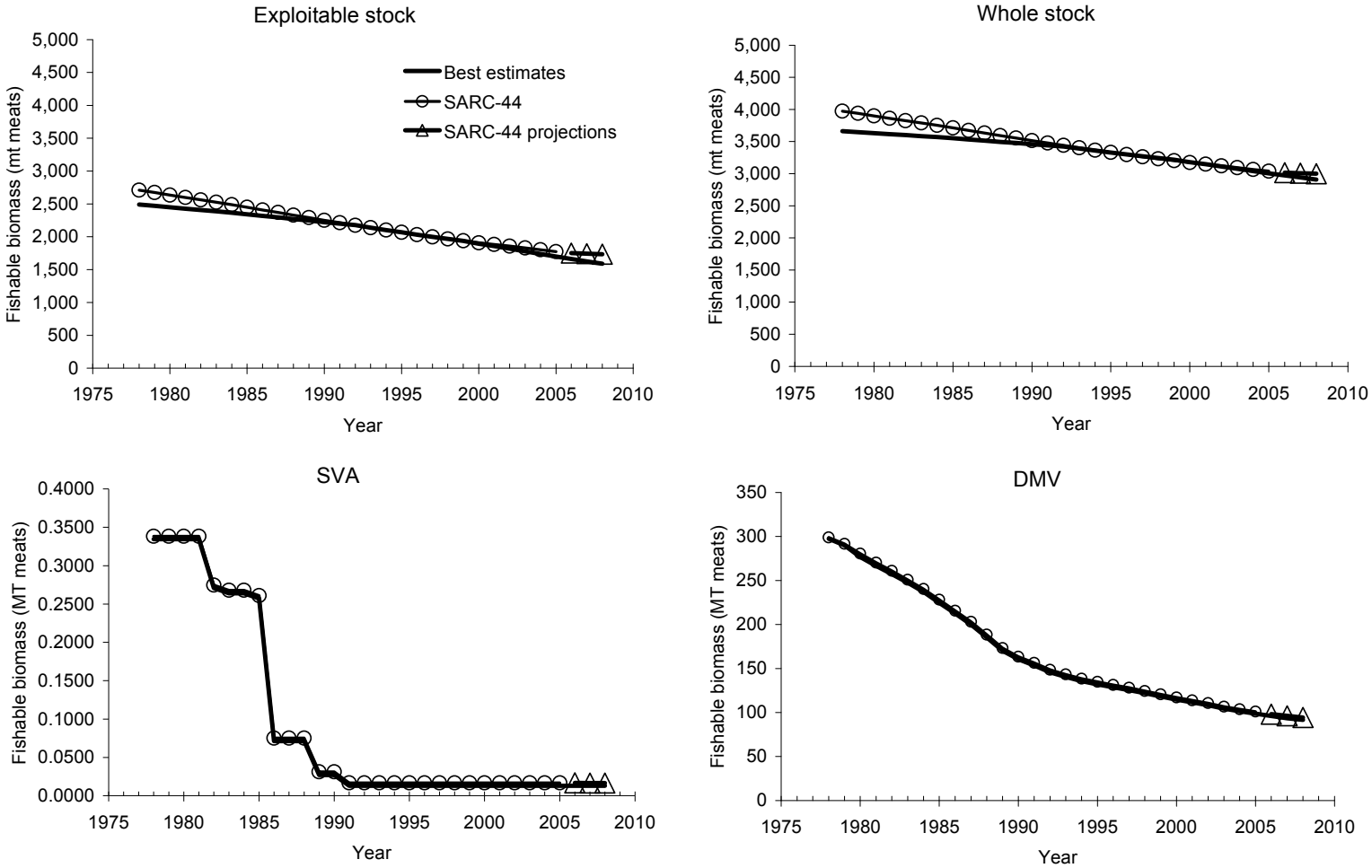


Figure B72. Best biomass estimates for ocean quahogs during 1978-2008, with estimates for 1978-2005 and projections for 2006-2008 from the last assessment (NEFSC 2007a). The report for the previous assessment did not include projections with status-quo catches so the projections for 2006-2008 were rerun starting from the 2005 biomass estimate in the previous assessment and using actual catches during 2006-2008.

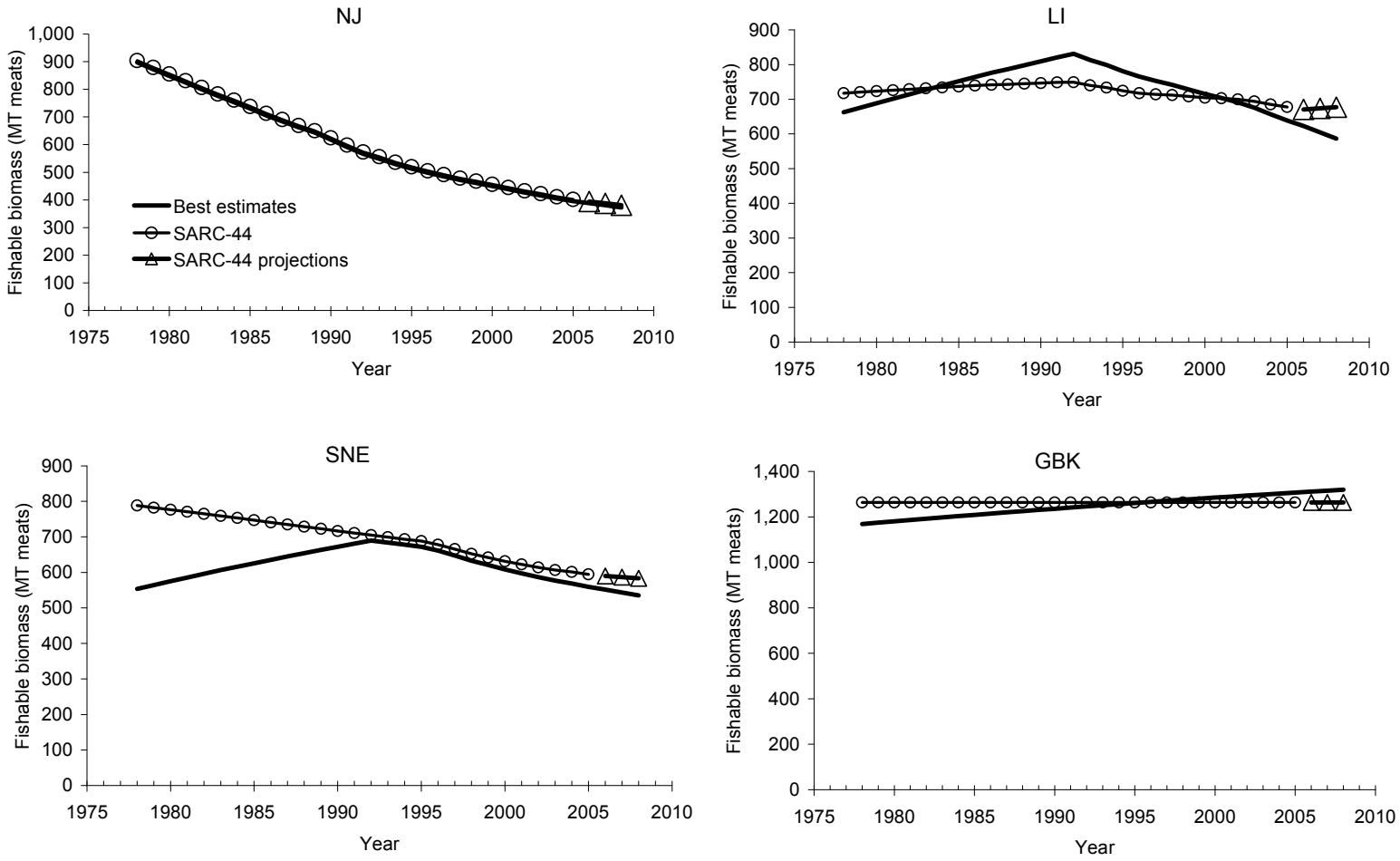


Figure B72. (cont.)

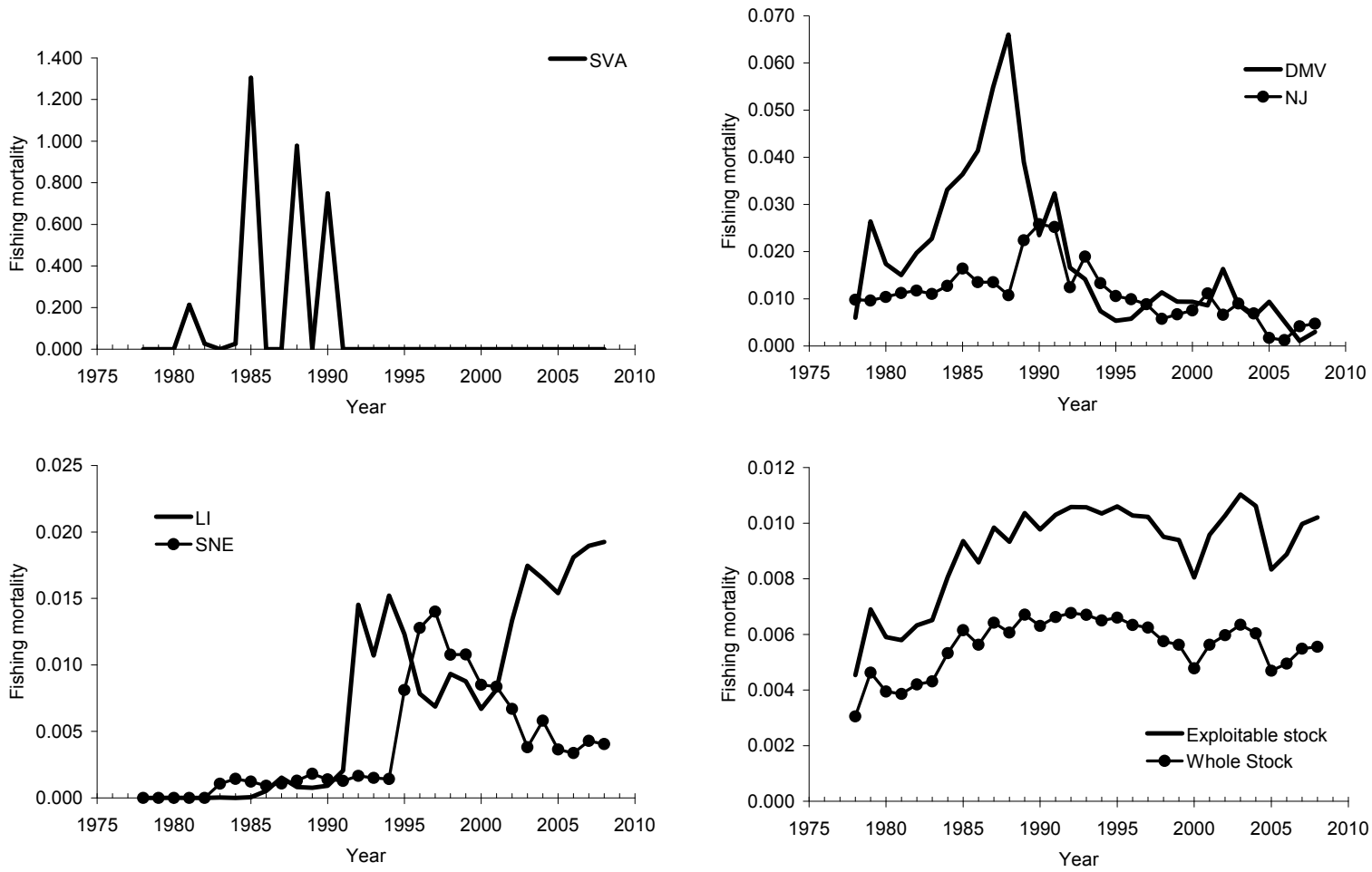


Figure B73. Best estimates of fishing mortality for ocean quahogs during 1978-2008.

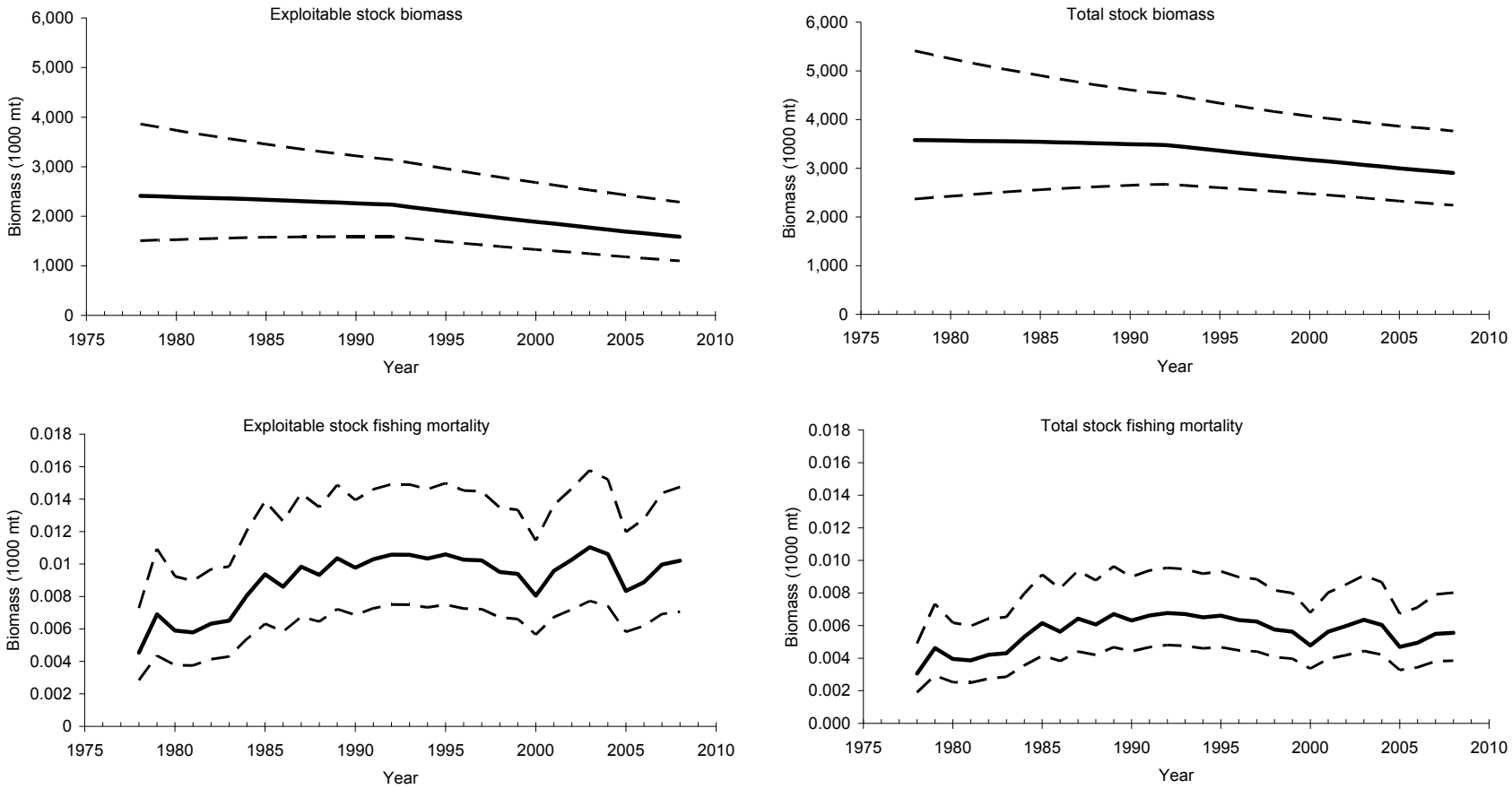


Figure B74. Approximate asymmetric 95% confidence intervals for best biomass and fishing mortality estimates for ocean quahogs in the exploited and total stock regions.

Figure B75. Trends in ocean quahog biomass during 1978-2008, by region based on best estimates. SVA is excluded because biomass is negligible there.

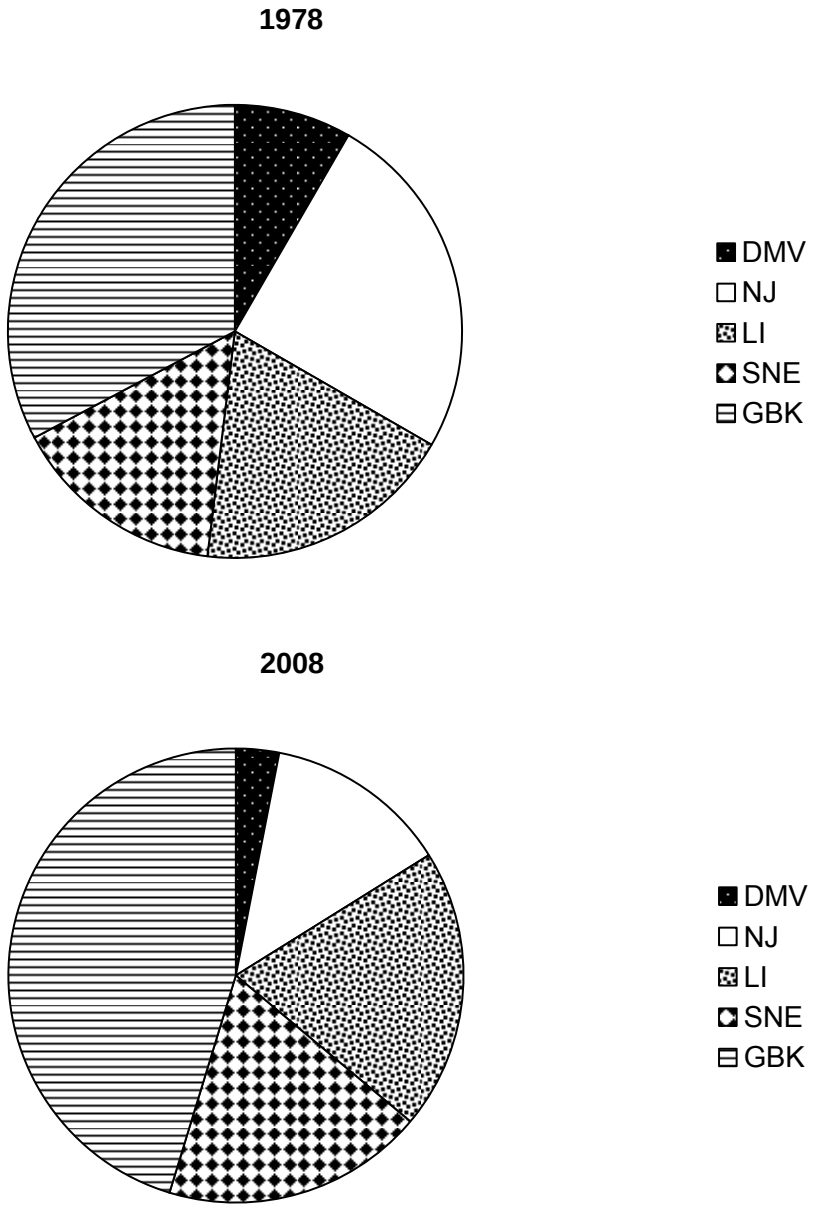


Figure B76. Proportion of ocean quahog biomass by region during 1978 and 2008, based on best estimates. SVA is excluded because it contains negligible biomass.

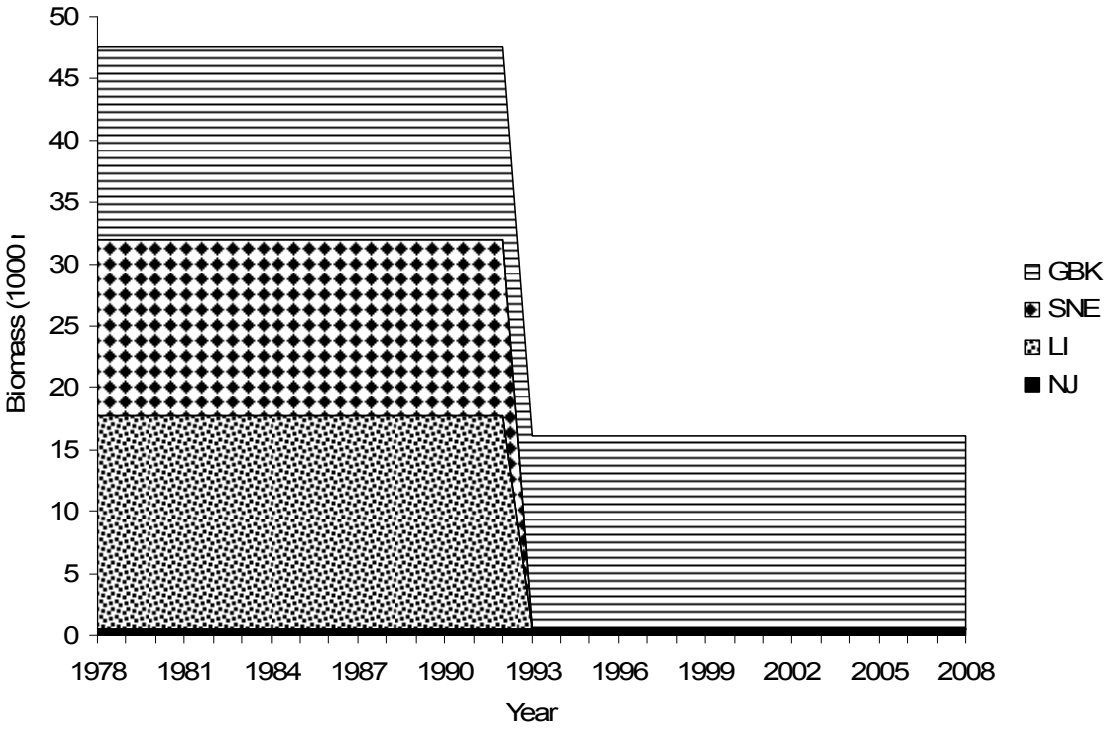


Figure B77. Estimated ocean quahog recruitment during 1978-2008, based on best regional models. Recruitment trends follow a stair step pattern because KLAMZ models for SNE and LI assumed two periods of constant recruitment with changes in level after 1992. SVA and DMV are not shown because recruitment is negligible there.

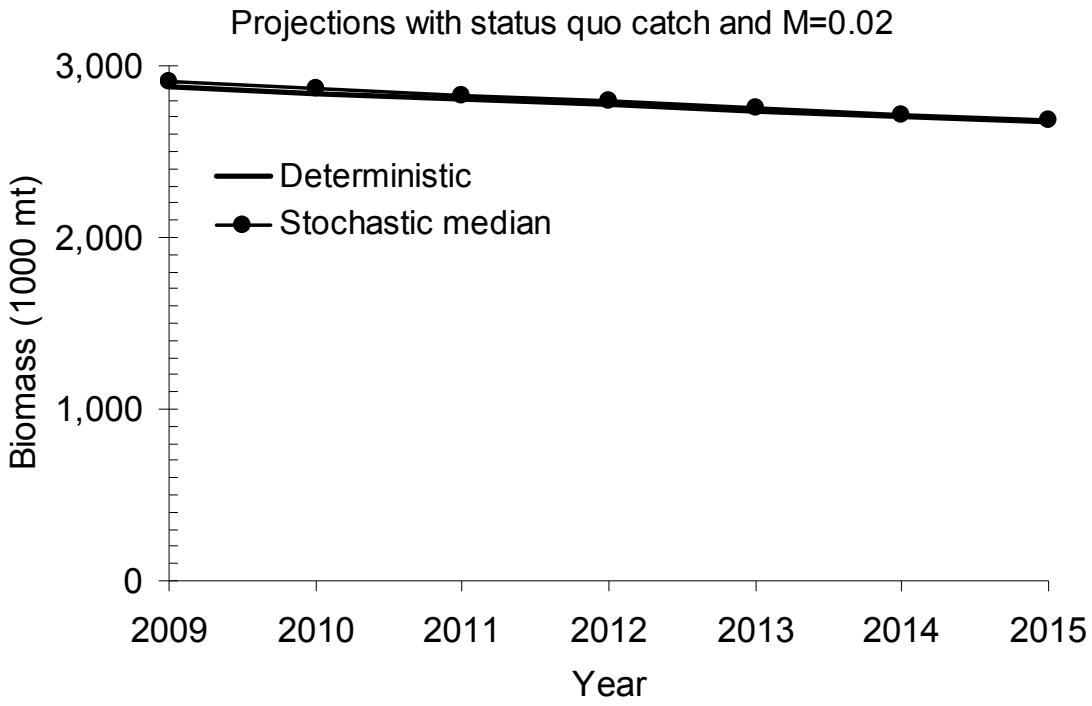


Figure B78. Deterministic and median stochastic projected biomass with M=0.02 and the determinist projection starting at the best estimates for 2008.

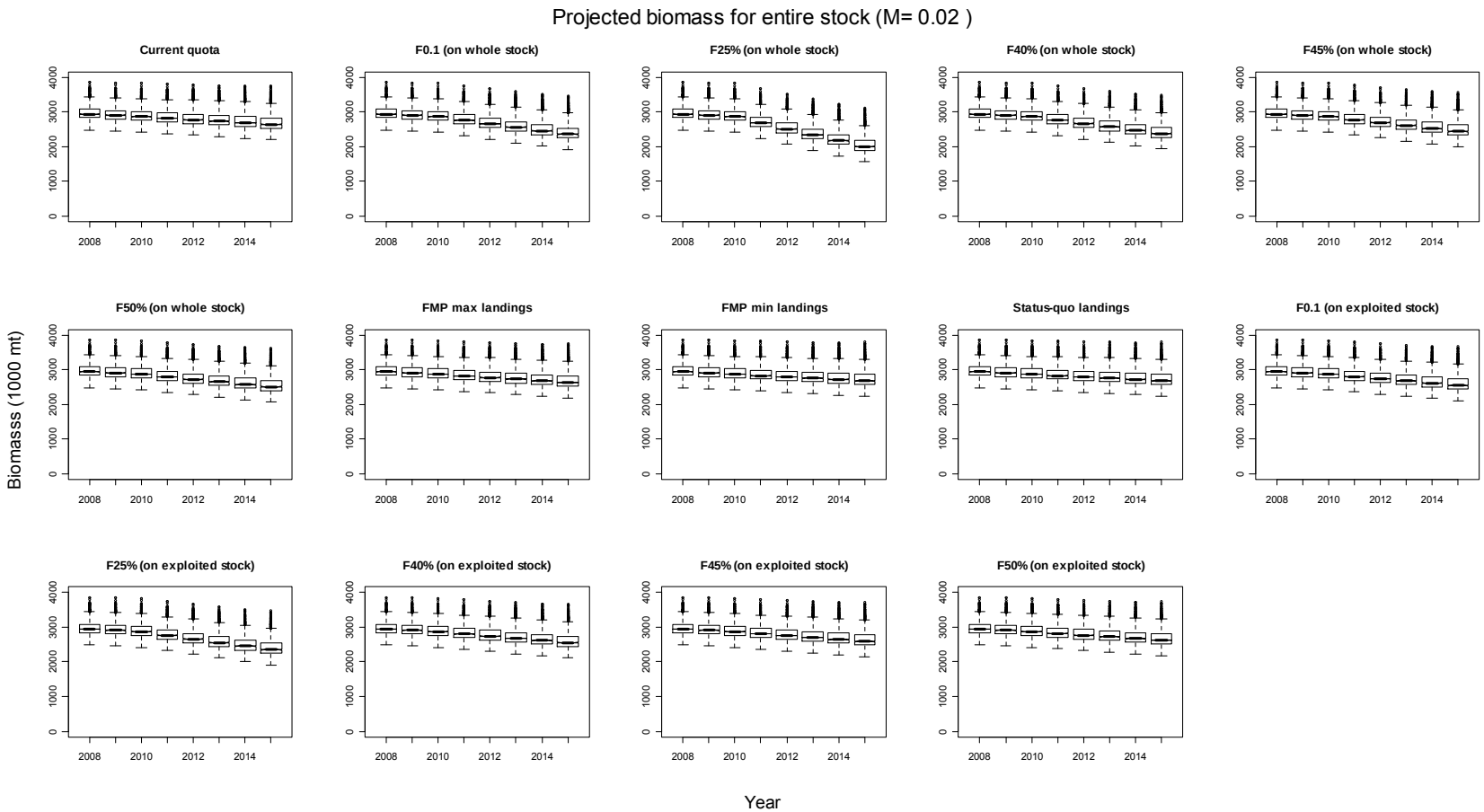


Figure B79. Projected estimates of whole stock biomass for ocean quahogs during 2010-2015 under various harvest policies assuming the true state of nature is *M*=0.02.

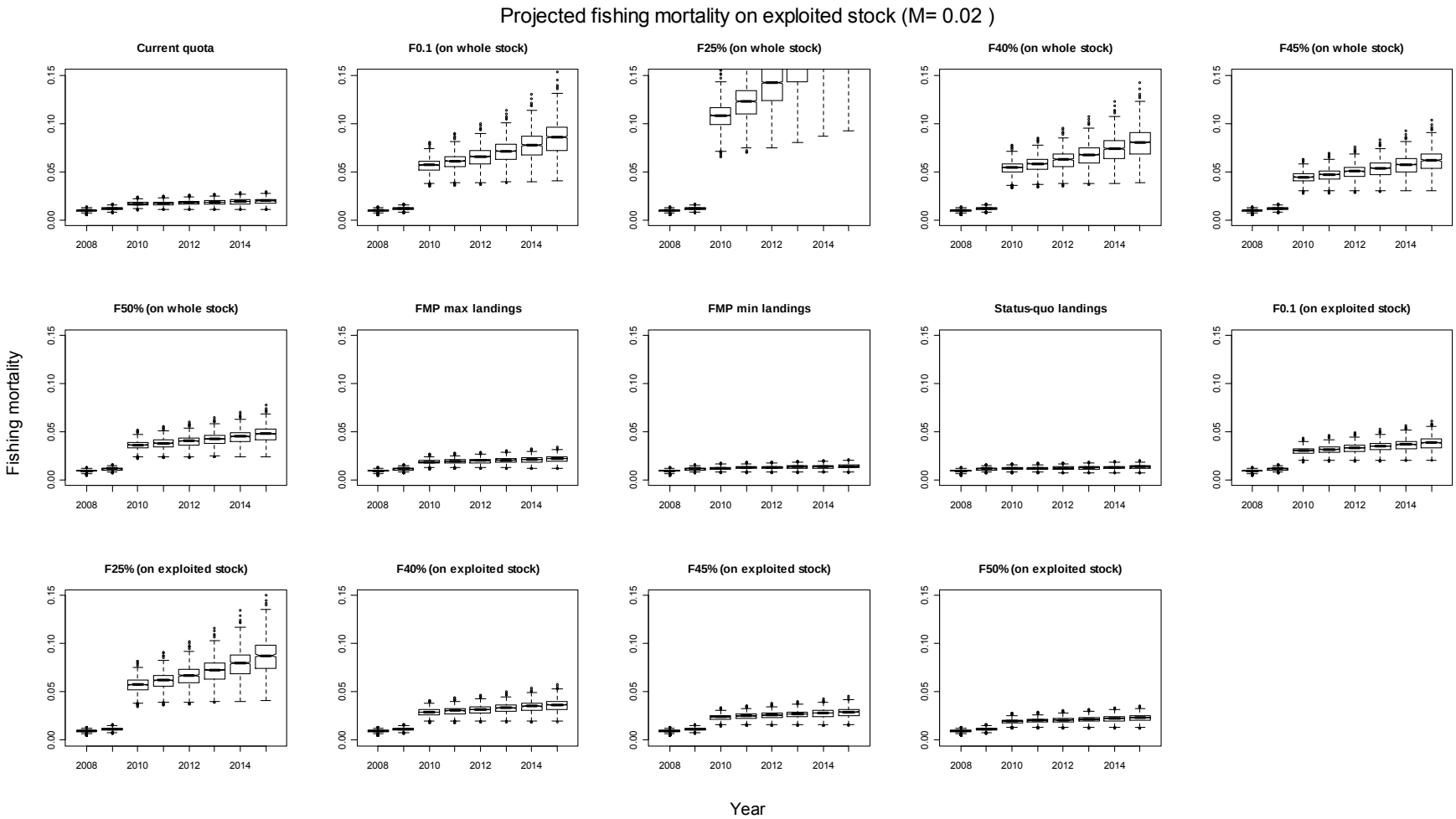


Figure B80. Projected estimates of fishing mortality for ocean quahogs in the exploited region during 2010-2015 under various harvest policies and assuming the true state of nature is *M*=0.02

APPENDIX B1: List of invertebrate working group participants:

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APPENDIX B2: Report on the ocean quahog resource in Maine waters.

2009 Maine Ocean Quahog Assessment

Introduction

The Maine fishery for Ocean quahogs, although harvesting the same species (*Artica islandica*), is persecuted in a different way and fills a different sector of the shellfish market than the rest of the EEZ fishery. The Maine "mahogany" quahog is harvested at a smaller size (38-64 mm or 1.5-2.5 in shell length, SL) than elsewhere in the EEZ fishery where ocean quahogs are harvested at 89-140 mm (3.5-5.5 in) SL.

Ocean quahog from Maine waters are marketed as a less expensive alternative for *Mercenaria mercenaria* (Maine DMR 2003). Harvesting takes place year round with the highest market demand during the summer holidays (Memorial Day through Labor Day). During this peak harvest period 20-30 out of a total of 57 license holders may land some volume of product.

The majority of the vessels in the Maine fleet is between 10.7-13.7 m (35-45 ft) and classified as "under-tonnage" or "small" in issuing permits. All of the vessels use a "dry" dredge (with no hydraulic jets to loosen the sediments) with a cutter bar set by regulation at no more than 0.91 m (36 in). There are no restrictions on any other dimension of the dredge.

Quahog Fishing in Maine takes place in relatively few locations along the coast north of 43 degree 50 minute latitude. Historically the bulk of fishing activity has taken place between Mt. Desert Rock and Cross Island with two significant quahog beds south of Addison and Great Wass Island covering an area of approximately 60 square nautical miles.

The Maine fishery began to expand into Federal waters in the 1980's due in part to PSP closures within state waters. In 1990 it was determined that this fishing activity conflicted with the Magnuson-Stevens Fishery Management Conservation Act which calls for a stock to be managed as a unit throughout its range. The Maine fishery was granted "experimental" status from 1990-1997.

In 1998, the Maine fishery was fully incorporated under Amendment 10 of the FMP and given an initial annual quota of 100,000 bushels based on historical landings data. There was no independent assessment of the resource available at that time. The State of Maine is responsible under Amendment 10 to certify harvest areas free of PSP and to conduct stock assessments.

In 2002 the State of Maine conducted a pilot survey to assess the distribution and abundance of quahogs along the Maine coast. This survey was a critical first step in establishing distribution, size composition and relative abundance information for the Maine fishery and for directing the design of the current survey work. While this initial survey provided valuable information it did not have the resources to estimate dredge efficiency and therefore was not able to estimate total biomass or biological reference points. The survey conducted in 2005 was focused on estimating dredge efficiency and to map quahog density on the commercial fishing grounds.

Estimates of biomass and mortality presented in this report are only for the commercial beds south of Addison and Jonesport/Great Wass Maine. This approach was chosen due to available resources and because it was conservative. Other quahog beds are known to exist along many parts of the Maine coast. If mortality targets could be met using the estimates from the primary fishing grounds then biomass outside the survey area can act as a *de facto* preserve.

Fishery Data

Data through out this report is presented in metric units. In some cases there are specialized terms and conversion factors which are listed below.

"Mid Atlantic" bushels of Ocean Quahogs x 10	=	lbs meat.
"Mid Atlantic" bushels of ocean quahogs x 4.5359	=	kg meat
1 "Mid Atlantic" (= "industry") bushel	=	1.88cubic feet
1 "Maine" (= "US Standard") bushel	=	1.2448 cubic feet
"Under-tonnage" vessel	=	1-4.9 GRT
"Small" vessel	=	5-49.9 GRT
1 "Maine" bushel	=	0.00303 metric tons meat weight

There are 57 ocean quahog licenses in the state of Maine. Since 2004 the number of licenses reporting landings has declined from 36 to 24.

Landings have trended downwards since 2002 (Table 1). The exception to this trend is in 2006 when landings increased to 124,839 bushels. This increase is most likely due to the reopening of a highly productive portion of the fishing grounds that had been closed in previous years from PSP. After the initial boost to landings from additional fishing ground, landings again began to decline. By the end of 2008 only 67,698 bushels out of a 100,000 bushel quota had been landed. LPUE has tracked landings closely over recent years. For 2008 LPUE was at a level 6.21 bushels/hour (Figure 1).

Incidental mortality in the ocean quahog stock off Maine is an important topic for future research. Maine has a very high level of fishing activity relative to the size of the fleet. Approximately 10,776 hours of fishing took place during 2008 representing over 64,000 tows at 10 min per tow. Using standard industry dredge dimensions and tow speeds this level of fishing activity represents 31.42 nautical miles^2 of bottom swept by commercial dredges.

Research Surveys

With the limited funds dedicated for survey work on quahogs, it was decided to focus all of the survey efforts in 2005, 2006 and 2008 on the primary commercial fishing grounds south of Addison and Great Wass Is. This decision is important in the interpretation of all following data as results because estimates pertain only to these two beds and not to the coast of Maine as a whole. Vessel logbooks and the 2002 independent survey abundance indices show that the majority of fishing activity and a sizable portion of the resource was in this region (Figure 2).

The first step in designing the 2005 survey was to establish a 1 km^2 grid overlay using Arcveiw 3.2 over the known commercial beds. Based on number of days at sea, 260 sites (tows) could be completed. The centers of the 260 1 km^2 grids covering the commercial beds were selected as start points for survey tows. These points were transferred to The Cap'n Voyager Software for use on board the survey vessel.

As of 2005 the quahog bed south of Addison, (referred to as "western") had been the only open fishing grounds for 3 years due to PSP issues in other beds. The quahog bed south of Great Wass Island, (referred to as "eastern") had been unfished for 3 years but had previously been one of the most productive fishing grounds. The 2006 survey took place 9 months after the "eastern" bed had been reopened. All areas were open during the 2008 survey.

Survey gear and procedures

The original survey in 2005 was conducted using the commercial vessel F/V Promise Land. It was a 12.8 m (42 ft) Novi Style dragger piloted by Capt. Michael Danforth and was contracted to perform all the survey drag operations in 2005 and 2006. All survey tows during these two years were conducted using the same dredge with dimensions: cutter bar 0.91 m (36 in), 2.44 m (8 ft) long x 1.83 m (6 ft) wide x 1.22 m (4 ft) high, overall weight 1,361 kg (3,000 lbs), bar spacing all grills 19.05 mm (¾ in). The survey dredge was the same dredge used by the F/V Promise Land during normal fishing activity. Prior to the 2008 survey The F/V Promise Land was sold and the captain left the fishery. To conduct the survey we had to contract a new vessel and captain which also meant the drag used was different than the two previous surveys. The new vessel, The F/V Allyson J4, had nearly identical specifications to the F/V Promise Land. Captain of the F/V Allyson J4, Bruce Porter, has been a quahog fisherman for 24 years. The dredge used for the 2008 survey had been built to nearly the same specifications as the original with the difference that the catch box on the original had extensions added to allow it to hold more sediment during longer commercial tows (Figure 3). These extensions meant the original dredge was roughly 400lbs heavier than the current dredge. During tow operations it was noted that the teeth on the cutter bar of the new dredge shined to depth of 3 inches just as they had in the original dredge. From this we assumed that the new dredge was cutting to the same depth as the original. It was also felt that since the survey tows were short (2 min) in order to avoid any overfilling and subsequent material loss that the additional catch box capacity of the original dredge would not give it any advantage over the current dredge.

For the initial survey in 2005 as the vessel approached the center of one of the 260 selected tow grids, bottom type and the feasibility of conducting a tow were assessed. If suitable bottom was not immediately present at the predetermined start point, the vessel would start crossing runs within the grid. If after 5 to 6 crosses no towable bottom or a tow path free of fixed lobster gear could not be found, then the grid location was deemed untowable, a note was made, and the captain continued on to the next site. When a suitable tow path was found within a grid the dredge was lowered to the bottom by free-spooling until the ratio of cable length to depth was 3:1. Once the desired cable length was reached the drum was locked, a two minute timer was started and a GPS point was taken.

Tows were made into the current at approximately 6.48 km/hr (3.5 knots) speed over ground

(average tow 188 m). After two minutes elapsed, a second GPS point was taken and the dredge was brought to the surface.

Tow distances calculated using the start and stop GPS points are good estimates of the distance actually traveled by the dredge. The manner in which the dredge is set and retrieved does not create a situation in which the dredge continues to fish as it is retrieved or before the drum is locked. In particular, the weight of the dredge keeps it in place on the bottom when the drum is unlocked at the end of the tow. In addition, the practice of backing the vessel toward the stopping point at the end of each tow means that the dredge was unlikely to travel very far at the end of the tow as it is lifted into the water column.

After the dredge was retrieved and before it was brought onboard the vessel, excess mud was cleaned from the dredge by steaming in tight circles with the dredge in the vessel's prop wash (Figure 4). Once on board, the dredge was emptied and photographed with a digital camera (Figure 5). The contents were placed on a shaker table (Figure 6), bycatch was noted and then all live quahogs were sorted out from the catch. From each tow a 5 L subsample of quahogs was taken at random (the entire catch was taken if catch was less than 5 L). The subsample was used to estimate tow counts, volume, and size frequency of the catch. The remainder of the catch was placed in calibrated buckets to determine total catch volume.

All data collected on board during operations were entered into a Juniper Systems handheld Allegro field computer running Data Plus Professional Software. All GPS data were collected using a pair of Garmin Etrex handheld units and transmitted in real time to the Allegro and a laptop running Cap'n Voyager Software. Data entry screens on the Allegro for the abundance survey consisted of: 1) trip information (date, time out, weather, sea state, time in, and comments); 2) site information (depth, bottom type, start tow GPS position, speed, end tow GPS position, and comments); 3) catch information (sample portion 5 L or all, volume, weight, count, photo id, size frequency 5 L or all, and comments); and 4) bycatch information (species, abundance).

The lengths (longest dimension) of all subsampled quahogs were measured to the nearest 0.01 mm and entered into the Allegro handheld using a Fowler Ultra-Cal IV digital caliper with an RS232 port. Estimated counts of quahogs were made by counting the number of clams in the 5 L sample and then expanding that value using the total volume of the catch. All data were analyzed using Excel with variances calculated using a bootstrap program (10,000 iterations) written by Dr. Yong Chen at the University of Maine, Orono.

Tow distances were determined by The Cap'n Software and were checked using ESRI ArcInfo software. All data from the tows were standardized to a 200 m tow prior to further analysis. For the 2006 and 2008 surveys only the 183 stations deemed towable during the initial survey were revisited. Due to vessel availability the 2006 survey needed to be conducted in the fall when there is a large amount of fixed lobster gear in the tow area. As a consequence only 130 tows could be completed.

Dredge efficiency

The Maine dry dredge is much less efficient (2-17%, ME DMR 2003) than hydraulic dredges used in the rest of the EEZ which can be up to 95% efficient (Medcolf and Caddy, 1971). A reliable estimate of dredge efficiency is needed to convert survey densities to a biomass estimate (NEFSC 2004).

One method of estimating dredge efficiency is through depletion experiments which are used to measure survey dredge efficiency for NEFSC clam surveys in Federal waters. Depletion studies for ocean quahog involve sensor and data processing equipment that were not readily available in

2005. The dry dredge used in the Maine survey is also relatively small compared to the depth of fishing. We hypothesized that it would be difficult to control the dredge precisely given the depth, size of dredge and strong currents in this region off Maine.

For the conditions off Maine is was determined that the best approach to estimating dredge efficiency would be through the use of box core samples (to directly estimate quahog density) followed by survey tows in the same area. Considering only ocean quahogs available to the fishery, the ratio of density measured by "follow on" dredge tows divided by boxcore density is an estimate of survey dredge efficiency (Thorarinsdottir and Jacobson 2005).

The *F/V Promise Land* with its large A frame and winches was able to deploy the 544 kg (1,200 lb) Ocean Instruments 610 box core with a core capacity of 0.062 m^2 and maximum penetration up to 60 cm (Figure 7). Follow on tows were conducted using the same gear used during all previous portions of the survey.

Box core work was conducted at three locations during three separate trips, one in August of 2005, one in January of 2006 and the last in April 2006. In all three experiments, follow on survey tows were made the day after the cores had been taken. The locations sampled were in the eastern quahog bed in an area of relatively high abundance. This area was also selected because it was a closed fishing ground during the August 2005 trip which would eliminate the possibility of the box core sites being commercially towed before follow on tows could be made. In January and April 2006 the region had been reopened to commercial fishing. However, VHF radio announcements describing the type of work underway were broadcast to local fisherman who were very cooperative and stayed well away from the experimental areas until all follow on tows could be completed the next day. Data entered into the Juniper Systems Allegro field computer included information about: 1) the trip (date, start tow, end tow), core (core #, core length, count, volume, weight, count of newly settled).

Each experiment began by establishing a single long towpath. To do this, the vessel was slowed to the standard tow speed of 3.5 kts and a GPS point was taken and plotted. After 2 min steaming along a fixed heading, a second GPS point was taken and plotted. These waypoints determined the endpoints for the follow on commercial tows and the path for boxcore sampling. Cores were then taken haphazardly along the tow path (60 for the August 2005 trip, 34 on the January 2006 trip and 30 on the April 2006 trip).

Once a core was brought on board it was measured for overall length and sieved through a large screen (1cm^2 mesh size). All quahogs were counted and their total volume and weight were measured.

During coring operations, it was noted that the upper 1-2 cm of very soft sediment contained recently settled quahogs (< 5mm length). The number of quahogs in this size range were recorded separately for all further cores and newly settled quahogs were retained to be preserved. During the January and April 2006 trips the top 5 cm of each core was removed and washed separately through a 300 μ sieve and all quahogs <5mm SL were preserved.

It was noted during boxcore sampling during the August 2005 boxcore trip that there was a change in sediment type beginning around 12-15 cm from the surface of each core. At this transition the sediment turned to a matrix of solid clay and old quahog shell. None of the live quahogs found in the cores in 2005 were below this transition. To assess this, the maximum depth within the core of live quahogs was measured during the 2006 trips.

After the maximum number of cores had been completed for a given trip the commercial dredge was deployed at one of the endpoints of the established tow path. Standard commercial towing was conducted for 2 min along the same path as the cores had been taken allowing the

dredge to tow from one endpoint to the next. After each round of coring, 6 tows were made along the same path, three in one direction and 3 opposing to help mitigate any effect from tide.

Dredge survey results

The original 2005 survey visited 259 potential tow grids. Out of the 259 there were 183 (121 in the western bed and 62 in the eastern bed) or 70.7% that were towable. Only two stations were untowable due to fixed lobster gear or other known obstructions. The remainder of the untowable sites were due to inappropriate substrate.

Tow distance, catch volume and counts were all standardized to a 200m tow. For the 2006 and 2008 surveys only the 183 towable grids were revisited. In 2006 130 of the 183 tows were completed. In 2008 181 of the 183 tows were completed.

For all surveys the highest concentration of biomass was in the eastern bed. The eastern section has had the most variable open and close status due to PSP. Substrate data (Figure 8) from Kelly et al. (1998) show the complexity of the substrate in the eastern section with highest quahog densities found near the boundary of hard rocky substrate with gravels, sands or mud. Substrate data collected independently using sidescan imaging showed that Kelly et al.'s (1998) substrate information was relatively accurate. However, in some cases substrate labeled as "sand" or "gravel-sand mix" near our most productive tows may have been shell hash from old quahog beds that was seen in box cores from the same area.

Size frequencies for all subsampled quahogs (n=20,737 in 2005, n=2,014 in 2006 and n=4,055 in 2008) Show a difference in size structure between the western and eastern beds. The quahogs in the eastern bed were larger (mean SL of 56mm $\pm$ 5for 2008) than the western bed (mean SL 52mm $\pm$ 4.9 for 2008). Cumulative size frequency distributions and a Kolmogorov-Smirnov test were used to test the null hypothesis that the size frequency distributions in the eastern and western areas were the same (Zar 1999). The null hypothesis was rejected (p=0.001). It should also be noted that in the 3 years since the initial survey the mean size for both western and eastern beds has increased by 5.03mm and 4.45mm respectively(Figure 9). Given the growth data available for this stock these size increases should take between 8 and 14 years. This may suggest that harvesting in Maine which targets smaller sizes may be altering the stock towards a larger and older quahog.

Because the two beds have differing size compositions and abundance levels, it was decided to calculate abundance for the two beds separately before estimating combined abundance for the entire survey area. Abundance estimates (see below) include a dredge efficiency that was estimated by applying 10,000 bootstrapped efficiency estimates from the three boxcore trips to 10,000 average abundance estimates from the surveys.

To estimate the total biomass in each year for the commercial fishing grounds the size frequency distributions were converted to proportion of the population in each 1 mm size bin. Shell length (*L*) was converted to meat wet weight (*W*) using W=4.97x10-6 x $L^{3.5696}$ (Maine DMR 2003).

year	bed	Median Abundance Estimate	Median mt Meat Weight	CV
2005	west	1.729E+09	8,653	39%
	east	2.404E+09	17,208	40%
	combined	4.134E+09	25,862	39%

2006	west	1.996E+09	10,166	41%
	east	1.225E+09	8,846	41%
	combined	3.221E+09	19,012	41%
2008	west	7.111E+08	5,471	40%
	east	1.094E+09	11,103	41%
	combined	1.805E+09	16,574	40%

Box core results

Efficiency estimates from box core experiments are presented based on sizes taken in the commercial fishery (35mm SL and greater). The estimated dredge efficiency was 17.91% with a 95% bootstrap confidence interval of 8.0%-34.4%.

Another important result from the boxcore work was that the average depth of live quahogs in the region sampled was no deeper than 9.55 cm (CV 20%). The standard commercial dry dredge has cutting teeth that are set to a depth of 7.62cm. We did not see evidence of anaerobic quahogs located deep in the sediments as has been reported elsewhere (Chenowith and Dennison,1993; Taylor 1976). Based on these results, it would seem that the majority of quahogs in this region would be impacted after one pass of a dredge.

Per recruit modeling

Biological and fishery parameters from a variety of sources were used to carry out a per recruit analysis for ocean quahog in Maine waters. Age at length and growth information was taken from Kraus et al. (1992). Von Bertalanffy growth parameters estimated from a sample of 663 quahogs from Machias Bay were: $L_{inf} = 59.470 \pm 2.089$, $K = 0.055 \pm 0.006$, and $t_o = -0.235 \pm 0.483$. The growth curve from Maine shows relatively fast growth the first few years of life in comparison to curves for other areas (Figure 19). Length-weight parameters were from the 2002 Maine Quahog survey: $W = 4.97 \times 10^{-6} * L^{3.5696}$. Length-weight curves for the Maine ocean quahogs and the rest of the EEZ stock were similar (Figure 10). Size at maturity data estimates were based on Rowell et al. (1990) who found that females became fully mature at an average size of 49.2mm for a quahog stock in Nova Scotia, Canada.

Fishery selectivity was modeled as a linear ramp function that was zero at 37 mm SL and one at 47mm. Following surveys, quahog of various sizes were pushed through the grates on the commercial dredge (19.05 mm, 3/4 in. bar spacing) to see what sizes might be retained. Clams from 34mm to 38mm generally passed through the grate with some getting caught. After 41mm almost all clams were thick enough to be retained. The regression model for shell depth and shell length in Feindel (2003) shows that a 19.05 mm (¾ in) bar spacing is the thickness of an ocean quahog with 38.7 mm SL.

The per recruit model used in this analysis was a length based approach which can be downloaded from the Northeast Fisheries Science Center as part of the NMFS Stock Assessment Toolbox.[7] The length based per recruit model was also used by Thorarinsdottir and Jacobson (2005). The biological reference points estimated in per recruit modeling for ocean quahog were $F_{max} = 0.0561$, $F_{0.1} = 0.0247$ and $F_{50\%} = 0.013\ y^{-1}$ (Figure 11).

Sensitivity analysis shows biological reference points from the per recruit model for ocean quahog are most sensitive to fishery selectivity parameters and, in particular, the length at which

7 Contact Alan.Seaver@noaa.gov for information about the NMFS Stock Assessment Toolbox.

ocean quahog in Maine waters become fully recruited to the fishery. Commercial port sampling conducted in 2009 confirms the size selectivity estimates used in the modeling (Figure 12).

Fishing mortality rate

For this report fishing mortality is estimated as the catch in biomass/average biomass. The surveys each take place over a period of 1 month, but mortality rates are relatively low so that survey biomass is a good proxy for average biomass. Following NEFSC (2004), the catch for each year used in fishing mortality estimation was landings plus a 5% allowance for incidental mortality to account for clams that are killed during fishing activity but not harvested. Catches for 2005, 2006 and 2008 including the 5% for incidental mortality were 528mt , 642mt and 348 mt of meat weight respectively. Biomass estimates for the same years were 25,862mt, 19,012mt and 16,574mt of meat weight respectively(Table 2). F=0.020 y^{-1} for 2005, F=0.033 y^{-1} for 2006 and F=0.021 y^{-1} for 2008. Thus for 2005 and 2008 F is roughly equal to $F_{0.1}$ but higher than $F_{50\%}$.

Stock Status

It is not necessary to evaluate stock status of ocean quahog in Maine waters because the stock component off Maine is a relatively small part of the EEZ stock as a whole. Ocean quahog biomass in Maine waters represented less than 1% of the biomass for the EEZ stock as a whole during 2005. Overfishing definitions apply to the EEZ stock as a whole.

It was not possible to compare or evaluate current biomass levels relative to biological reference points associated with maximum productivity, depleted stock or historical levels because no appropriate biological reference points or historical biomass estimates are available.

The fishing mortality rates during all three surveys has been almost equal to $F_{0.1}$=0.0247 and the assumed natural mortality rate M=0.02 y^{-1} but almost double $F_{50\%}$ =0.013 y^{-1}. $F_{0.1}$ might be a reasonable reference point for managers if the goal is to maximize yield per recruit while preserving some spawning stock. Simulation analysis (Clark 2002) indicates that $F_{50\%}$ (1.3% per year) might be a reasonable reference point for managers if the goal was to preserve enough spawning potential to maintain the resource in the long term. However, preservation of spawning potential may not be necessary if recruitment originates mostly outside of Maine waters.

There is evidence or recent recruitment (newly settled ocean quahog < 5 mm SL) in one of the beds that were surveyed. However, although growth is relatively rapid in Maine waters, it may be 3 decades or longer before these recruits become large enough to enter the fishery.

Stock assessment advice concerning ocean quahog in Maine waters would be easier to provide if management goals were formulated and if biological reference points for biomass and fishing mortality were defined.

Research Recommendations

1. Impact on habitat and substrate should be investigated for the Maine Dredge along with good estimates of area swept by fishing activity,
2. More work needs to be done to determine age, growth rates and size/age at maturity for Maine ocean quahogs. New digitized methods may help in this process.

Acknowledgements

This report was completed with extensive help from the Captains and crews of F/V Promise Land and F/V Allyson J4. Additional assistance was provided by Scott Feindel, Larry Jacobson and

the NEFSC Invertebrate (Clam) working group.

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Table 1. Landings from vessel logbooks.

year	Landings (Maine bushels)all vessel classes combined	Landings (only records with both effort and catch>0)	Effort (hrs fished)	Nominal LPUE (ME bushel/hr)
1990	1018	1018	286	3.56
1991	36679	34360	17163	2.00
1992	24839	24519	13469	1.82
1993	17144	17144	5748	2.98
1994	21672	21672	5106	4.24
1995	37912	37912	5747	6.60
1996	47025	47025	8483	5.54
1997	72706	72706	11829	6.15
1998	72466	72152	11745	6.14
1999	93015	92285	11151	8.28
2000	121274	119103	12739	9.35
2001	110272	110272	13511	8.16
2002	147191	147191	19681	7.48
2003	119675	119675	17853	6.70
2004	102187	102187	19022	5.37
2005	100115	100115	17063	5.87
2006	121373	121373	14902	8.14
2007	102006	102006	14018	7.28
2008	66926	66926	10776	6.21

Table 2. Commercial landings from Dealer Logbooks converted to mt meat weight for estimates of *F*.

year	landings from dealer logs (bushels)	metric tons meat landed w/ 5% incidental mortality	*F*
2005	102,671	528	0.020
2006	124,839	642	0.033
2008	67,698	348	0.021

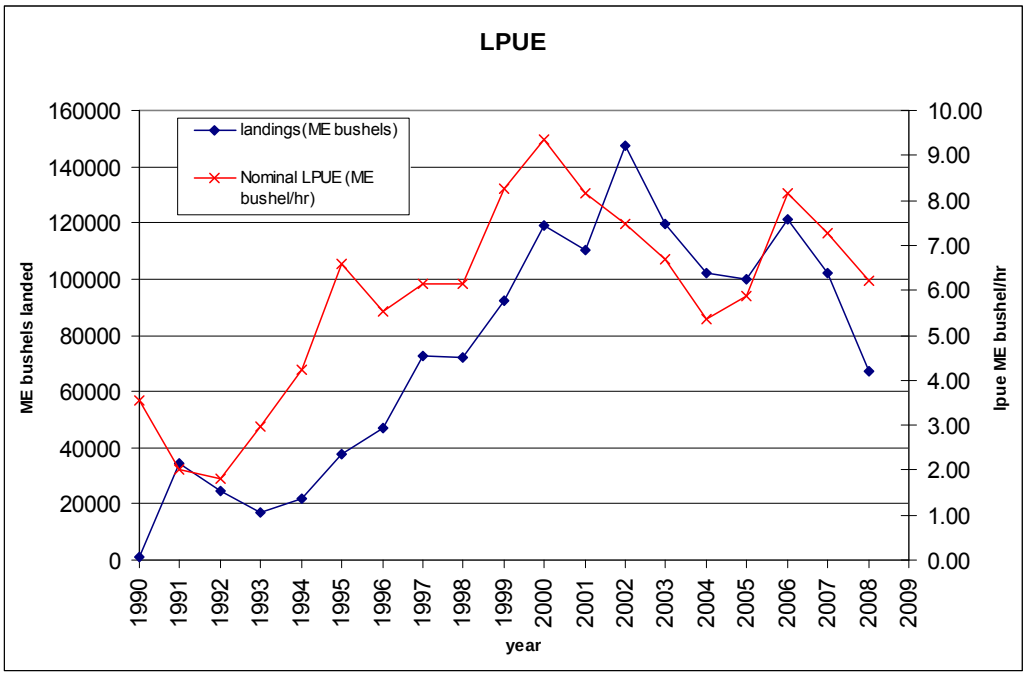

Fig 1. Commercial LPUE and Landings from vessel trip reports.

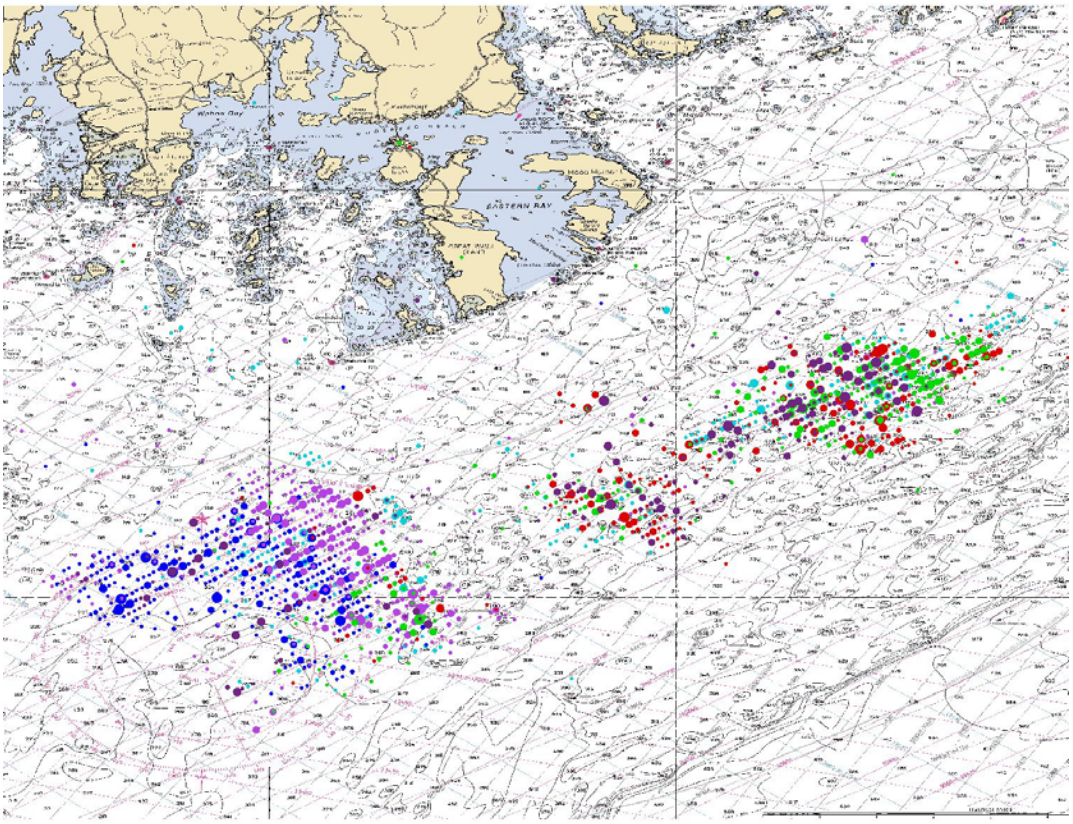

Figure 2. Combined locations of all reported commercial landings 2003-2008.

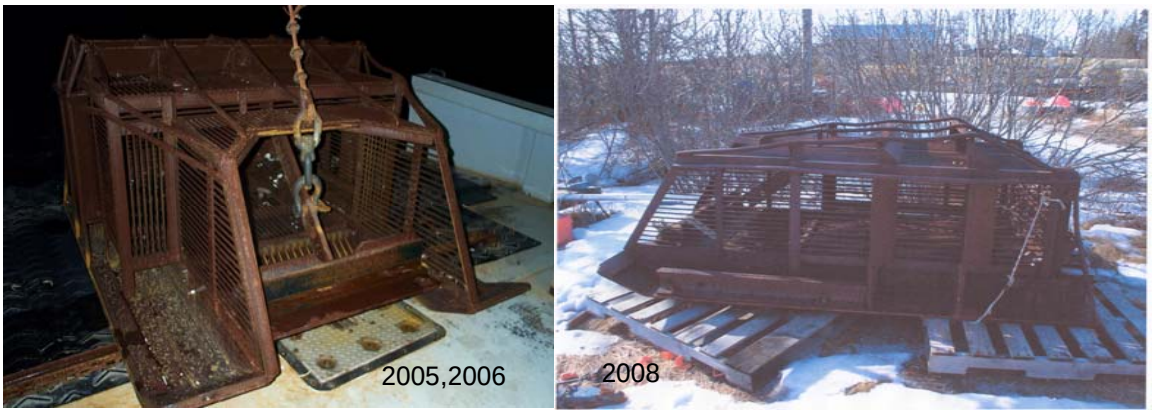

Figure 3. On left, Commercial dredge used in 2005, 2006 operations roughly 3,000lbs. On right commercial dredge used in 2008 roughly 2,600lbs.

Figure 4. Washing the catch in vessel prop wash.

Figure 5. Typical 2 min tow. Note very low bycatch and uniform size of clams.

Figure 6. Processing the catch on shaker table, used to remove shell fragments and mud. This step is performed in commercial operations as well.

Figure 7. Ocean Instruments box corer used during survey.

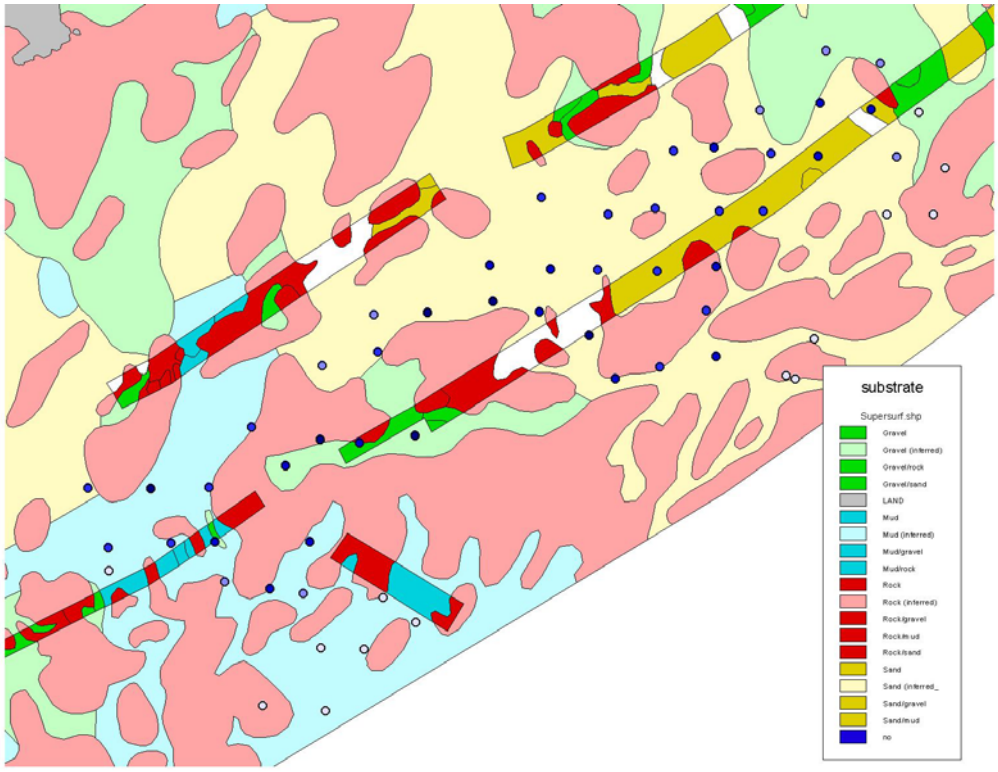

Figure 8. Substrate information from Kelly et al. Showing coincidence of hard bottom edges with high density quahog tows from eastern bed.

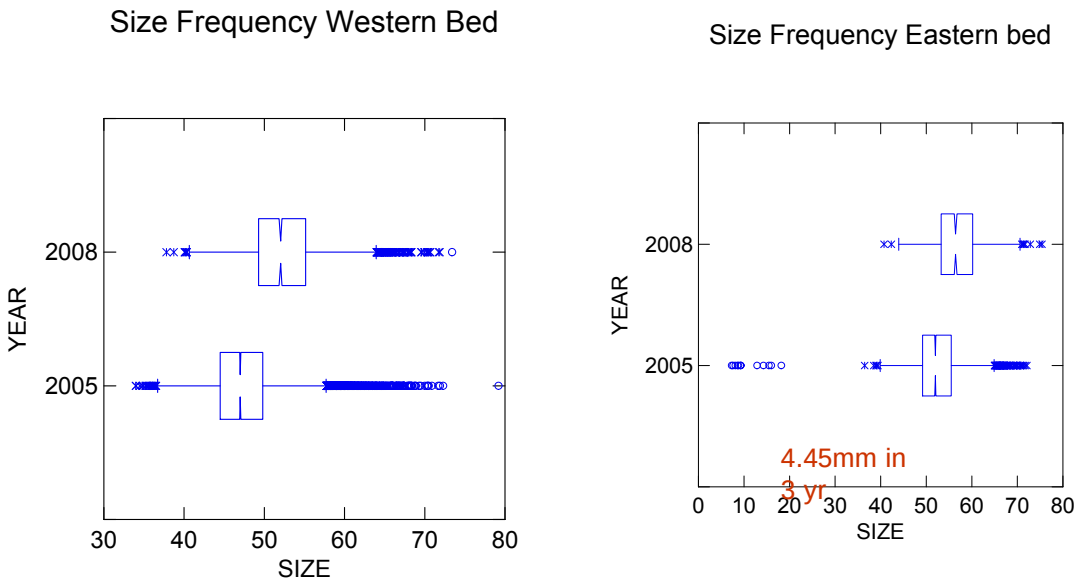

Figure 9. Growth in quahogs between 2005 and 2008 surveys. Based on Maine growth data an increase of 5mm in the western bed should have taken 8 years and the 4.45mm increase in the eastern bed should have taken 14 years.

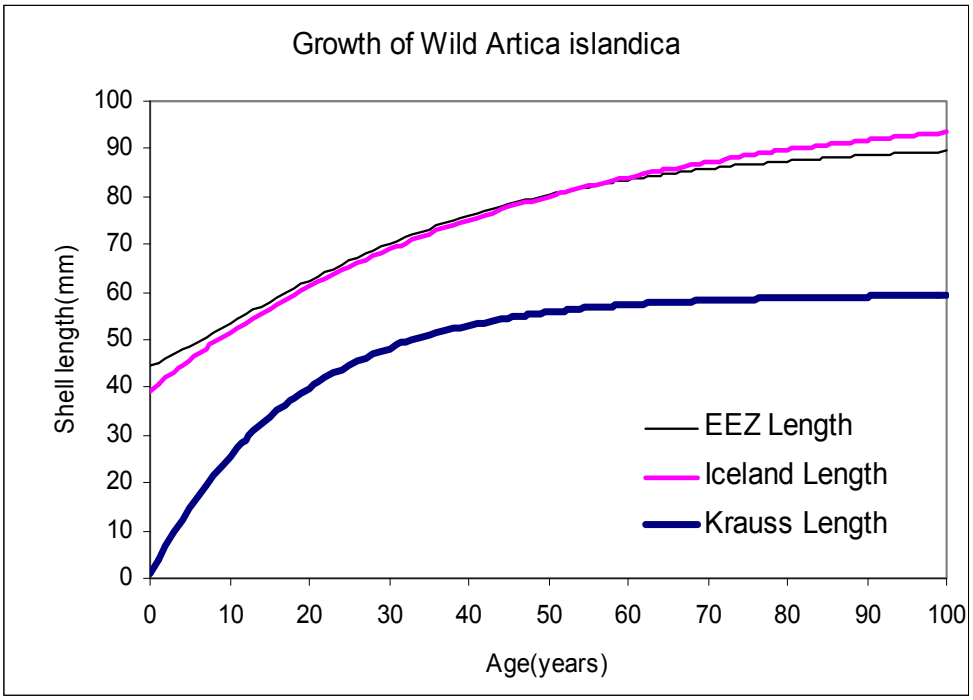

Figure 10. Growth curves for various quahog stocks. Maine (Krauss) shows rapid initial growth with much lower maximum size.

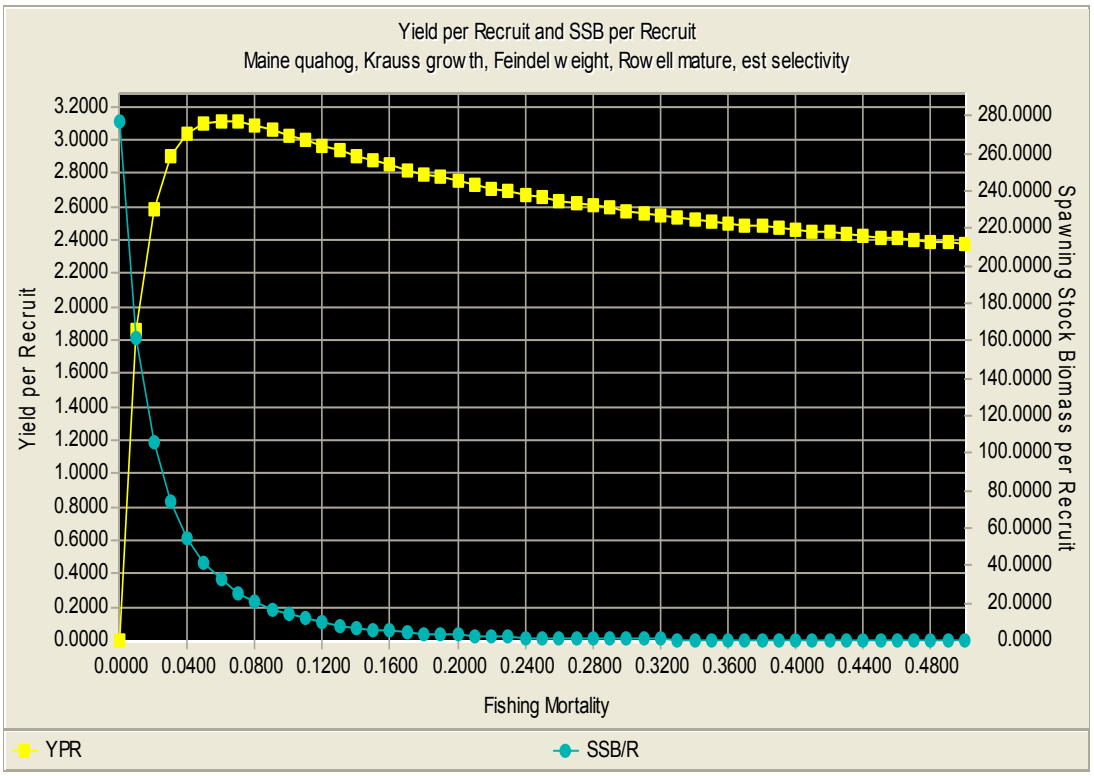

Figure 11. YPR analysis run in 2005. No new information was available to modify these results.

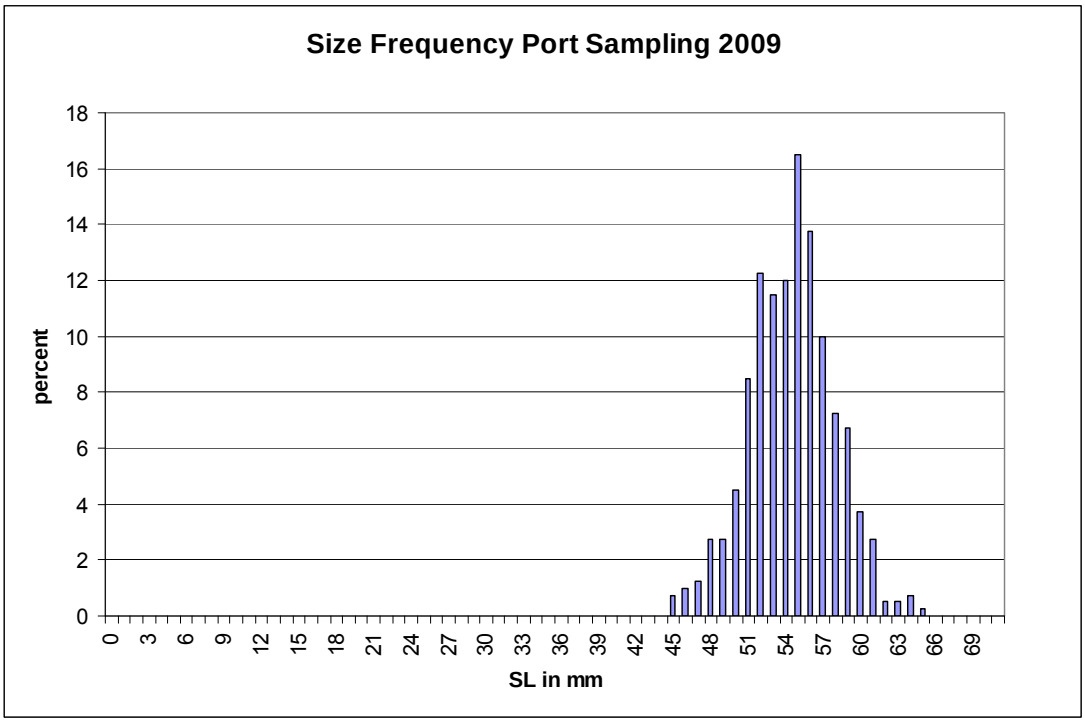

Figure 12. Size frequency for port samples collected in Jan- March 2009 from 6 different vessels. These sizes concur with ramp function used in YPR analysis

APPENDIX B3: Report on dredge performance from SSP (survey sensor package) data.

2008 Survey NOAA Clam Dredge Performance Review

April 23, 2009

Summary

The review of the 2008 NOAA clam survey tows described below accomplished the following tasks.

1) Grade the tows based on the previously developed manifold pressure "good"/"bad" criteria. For the 2008 survey this required development of a manifold pressure proxy based on pump amps due to a SSP failure towards the end of the survey. A total of 67 stations out of 453 were determined to be "bad" by the criteria.
2) Grade the tows based on the previously developed Y Tilt (dredge fore/aft angle) "good"/"bad" criteria. For the 2008 survey it was determined that sensor issues were likely creating false excessive Y tilt motions and the Y Tilt criteria should not be used. Based on this decision, the 2005 NOAA survey tows were re-reviewed for Y tilt issues and a similar determination was made. This resulted in one station, #218, previously labeled bad for Y tilt being included in a re-analysis of the 2005 survey data.
3) Evaluate the effect of changing the dredge pump, pump power cable, and SSP during the survey on the dredge's performance. The end conclusion is there was no noticeable effect on the survey results.
4) Investigate several SSP data anomalies, particularly fluctuations in frequency recorded and minor variations in pump amps and manifold pressure trends that occurred during the survey. It was determined that these anomalies were likely sensor issues or a minor pump problem that had no noticeable effect on the survey results.

Review of Survey Dredge Pump Performance Relationships

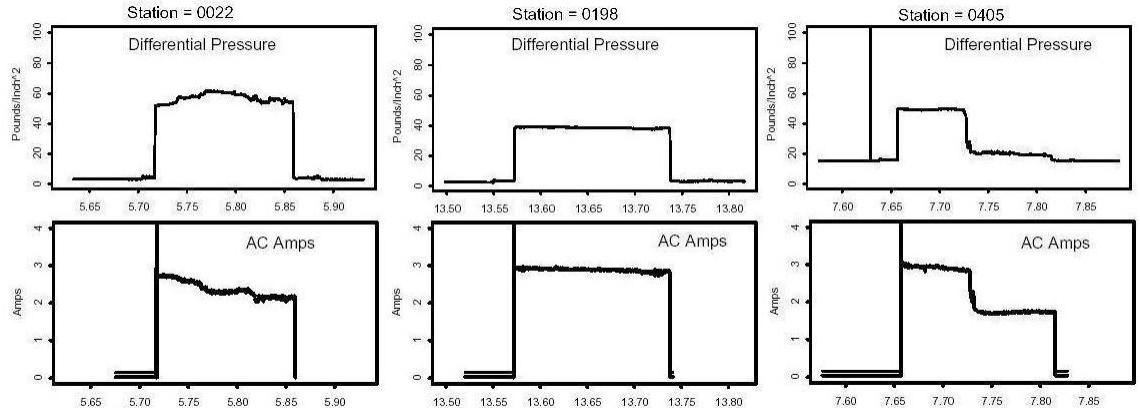

Figure 1 - Blocked Manifold Good Pressure Blocked Intake

In evaluating the performance of the NOAA clam survey, several key pieces of data are used, pump manifold pressure and pump electrical operating parameters. The key data is the manifold pressure with the electrical data serving as a backup to missing manifold pressure data and to verify the pump was seeing a consistent electrical supply.

Figure 1 shows the two dredge pump pressure problems, a blocked pump discharge manifold (pressure increase) or a blocked pump intake (pressure decrease). For a centrifugal pump such as used on the survey dredge, in both blockage cases the pump amps will fall in proportion the increase or decrease in manifold pressure. Thus with a suitable proxy, missing manifold pressures can be recreated using the amps data recorded.

The frequency and voltage data, along with the amps data, is primary used to verify a consistent electrical supply to the dredge pump motor. For the NOAA survey dredge the frequency should be 60 hertz and the voltage should be a relatively consistent value. The frequency is set by the rpm's of the generator which is governed to between 59.5 and 60.5 hertz depending on load. The voltage recorded is the voltage at the dredge pump and typically runs around 400 volts depending on power cable length.

Introduction

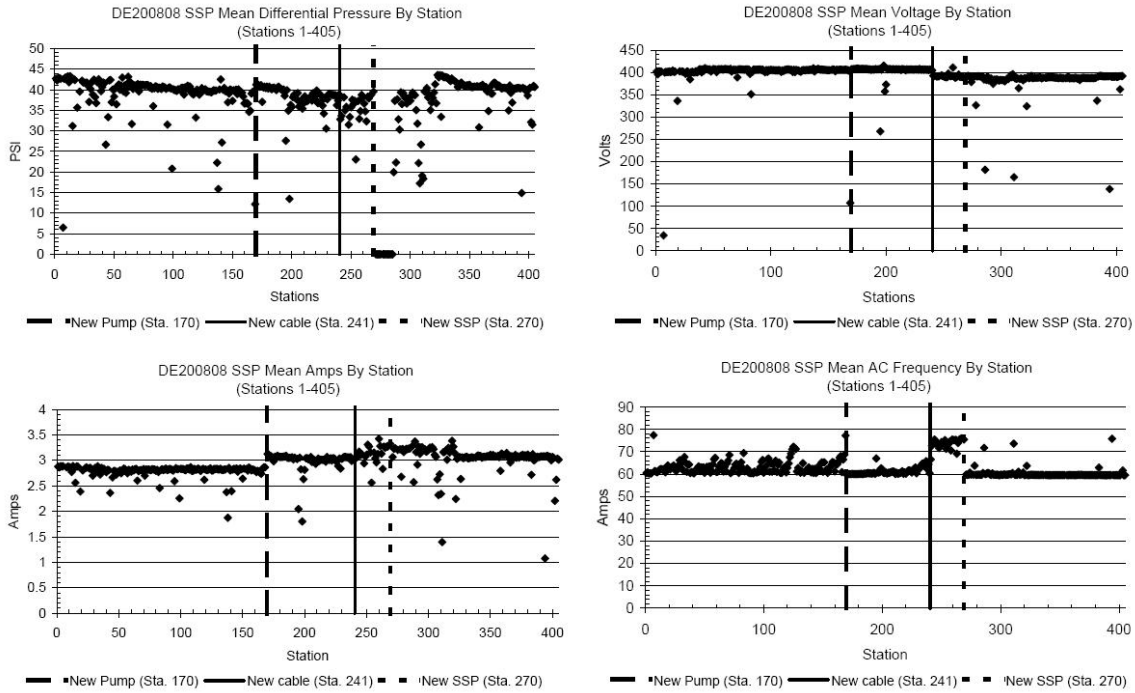

Figure 2 - SSP Mean Values for Differential Pressure, Amps, Volts, Frequency

A review of the Survey Sensor Pack (SSP) data from the 2008 NOAA clam survey was undertaken to evaluate the performance of the dredge for each of the survey tows. The SSP's mean Manifold Differential Pressure, Pump Amps Draw, Pump Voltage, and Frequency for tows 1 to 405 are plotted in Figure 2. Tows 406 to 453 are not plotted due to a failure of the SSP package. For reference survey leg 1 was stations 1 to 169, survey leg 2 was stations 170 to 319, and survey leg 3 was stations 320to 453.

For the 2008 clam survey, (4) onboard events happened.

1) The dredge pump failed during station 169 tow and was replaced with the backup unit for tows 170 till the survey end.
2) The pump power cable was replaced at station 241 with a longer cable to allow tows in deeper waters. The longer cable remained for the rest of the tows.

3) The primary SSP package failed towards station 269 and was replaced with the backup SSP for station 270 on.
4) The backup SSP failed from station 406 till the end of the survey.

A visual review of the SSP data showed the following issues of concern.
1) There were large number of tows with significant drops in the manifold differential pressure and pump amps. (Same as occurred during 2005 survey)
2) There was modest, about 3 to 5 psi, jump in the manifold differential pressure for the last third of the survey.
3) The frequency recorded from station 1 to 169 varied from 60 to 70 hertz. The frequency then stabilized at 60 hertz till about station 220 when it started a slow rise followed by a jump to over 70 hertz at station 241. The frequency then stabilized at 60 hertz till the end of the survey.
4) The dredge Y tilt (fore/aft) and X tilt (side/side) seemed to have greater fluctuations than previous surveys.

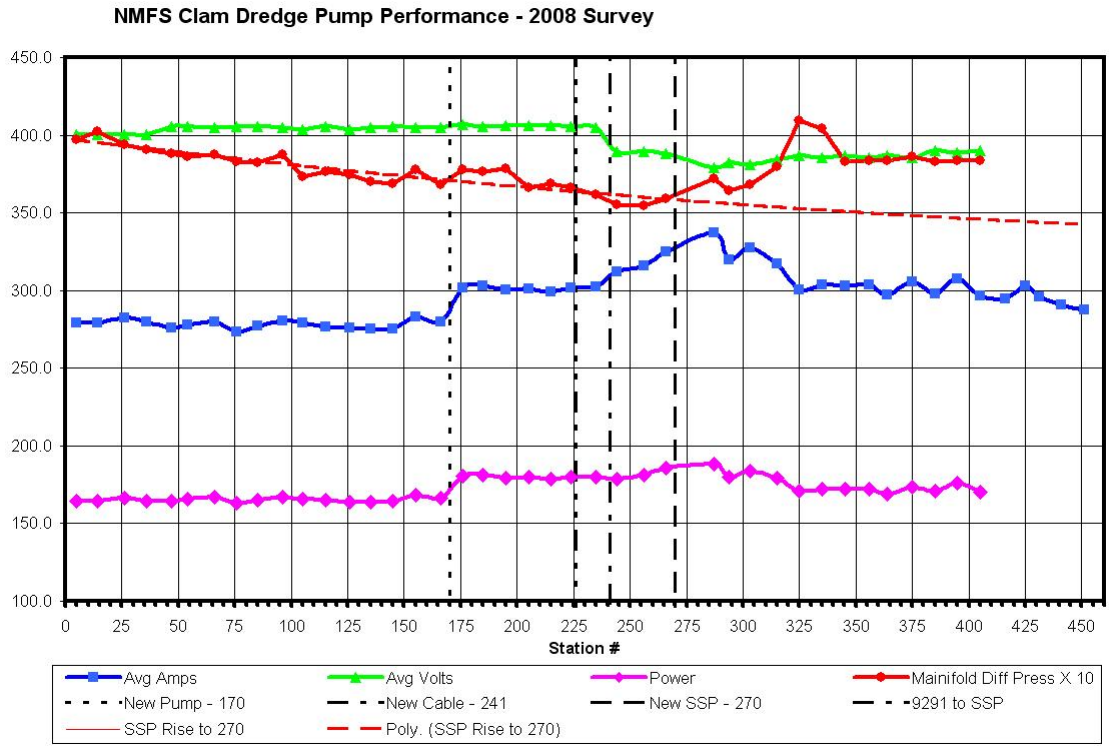

Figure 3 - Clam Dredge Pump Performance - 2008 NOAA Survey

To help evaluate the effect of the onboard events and SSP data concerns a plot of the dredge pump's general operating performance, Figure 3, was done to see trends over the entire survey. This plot was done using stations ending in (5) or if that station had problems, such as a clogged manifold, the next nearest good station was used. Note the manifold pressure, red line, is plotted at a 10 times scale.

Effect of Dredge Pump Replacement at Station 170

The dredge's pump was replaced at station 170 and is shown on figure 3 with the black short dashed line. When the new second pump was installed the manifold pressure jumped up roughly 1

psi to about 38.5 psi (red line figure 3). The pressure increase would be expected over the first pump's now worn condition, but did not increase to the first pump's "new" pressure of about 40 psi.

This is likely due to the fact that the second pump appears to have more internal running resistance than the first pump by the jump in amps draw (blue line figure 3) and power (magenta line figure 3) from about 275 amps, 160 VA, for the first pump to 300 amps, 180 VA, for the second pump. The increase in internal resistance could be from tighter bearings, shaft seals, or running clearances and would cause the second pump to run slightly slower than the first pump which would produce less manifold pressure.

Also interestingly the fluctuation in recorded frequency up to replacement of the pump disappeared and a steady 60 hertz was now being recorded (see figure 2). The variation in frequency from 60 to 70 hertz is not possible as this is a direct function of the ship's generator rpm's which are governed to 59.5 to 60.5 hertz depending on load. Variations of the size recorded would be easily noticed by ship's engineer and at 70 hertz would have likely tripped automatic over-speed safety shutdowns. In addition the higher frequencies, if they did occur, would have caused the dredge pump to run at significantly higher speeds which would have boosted the manifold pressure and raised the amps draw, neither of which occurred. The frequency variations could have been due to problems in the first pump which eventually caused the pump motor failure.

Based on the above the change in dredge pumps would have had no noticeable effect on the performance of the survey dredge as the key manifold pressure remained within the normal operating band of 35 to 40 psi.

Effect of Dredge Pump Power Cable Replacement at Station 241

The dredge pump's power cable was replaced at station 241 with a longer cable to allow sampling in deeper water and is shown on figure 3 with the black long/short dashed line. When the new longer cable was installed there was a drop in voltage (green line figure 3) at the pump from about 405 volts to 390 volts which would be expected from the higher resistance of the longer cable. There was a corresponding increase in the amps draw (blue line figure 3) from 300 amps to 315 which would also be expected as the dredge pump power draw (magenta line figure 3) remained the same.

Most importantly the key manifold pressure (red line figure 3) over the power cable change followed the general small downward typical of a survey pump wearing normally over the course of a survey. Based on this the change in dredge pump's power cable would have had no noticeable effect on the performance of the survey dredge as the key manifold pressure remained within the normal operating band of 35 to 40 psi.

Replacement of Primary SSP at Station 270

The primary SSP was replaced at station 270 due to onboard data review which was indicating a SSP failure. The frequency recorded had started to rise after station 220 and then jumped to a completely impossible 74/75 hertz (see above discussion). In addition station 268 had no SSP differential pressure and station 270 recorded no SSP data at all. These failures had followed a string of stations with low recorded manifold pressures.

The frequency data recorded by the second SSP after station 271 did return to an expected steady value of 60 hertz. In addition the voltages recorded at the pump remained steady at around 390 volts between the first and 2nd SSP's. Both of these indicate a correctly functioning second SSP.

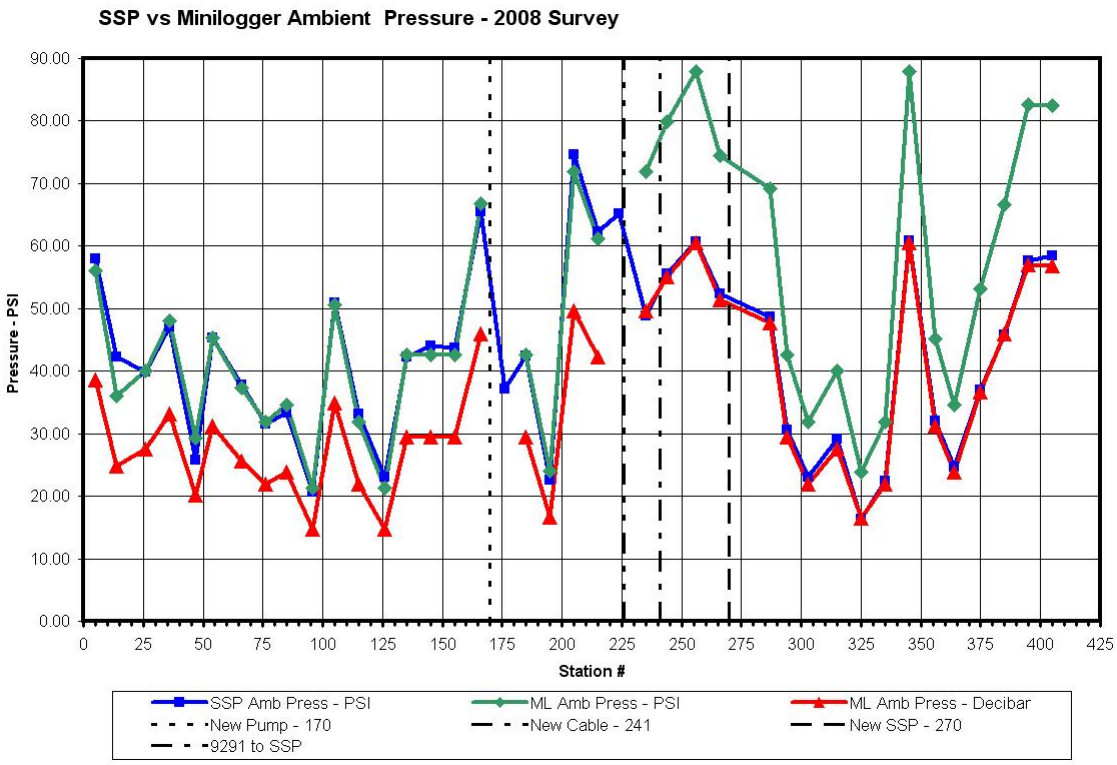

Figure 4 - SSP vs Mini-Logger Ambient Pressure - 2008 NOAA Survey

A further check was done by comparing the SSP recorded ambient pressure to the ambient pressure recorded by the mini-loggers (see figure 4). The SSP ambient pressure (blue line figure 4) tracks the mini-logger pressure (green and red lines figure 4) very closely both before and after the change in SSP's. Note the SSP value changed from psi to decibars at station 226/227 which will be discussed later. This change in units did not affect any of the review work undertaken.

The average dredge running angle recorded by the SSP's inclinometer was also compared between the first and second SSP units. (Note stations used were the good stations used to develop dredge pump performance plot in figure 3.)

	Y Tilt	X Tilt
First SSP Stations 1 to 269	3.39	2.72
Second SSP Stations 270 to 405	2.76	2.63

Both the Y (fore/aft) and X (side/side) tilt angles are within the at sea calibration errors that were done to set up the second SSP.

A review of the pump voltage, recorded by the SSP's, and pump amps, recorded independently of the SSP's, was also done to compare first and second SSP functionality. The amps (blue line figure 3) and voltages (green line figure 3) are steady from station 1 to about station 260 as would be expected. From station 260 to about station 285 though, the amps increased significantly then declined to "normal" values at station 325 and remained steady for the rest of the survey.

This increase in pump amps could only be caused by increased running resistance in the pump such as shell hash binding the pump impeller. An increase in manifold pressure would not

cause this increase in amps. Whatever was causing the binding eventually wore away and the running resistance eventually returned to normal conditions. The corresponding dip and rise in voltage and increase then drop in power demand (magenta line figure 3) supports this theory.

The manifold pressure though should have dipped slightly during this episode as the added running resistance would have slowed the pump rpm's down. This did not occur though as the manifold pressure (red line figure 3) was recorded to be steadily rising and continued do so well past when this anomaly in amps draw was over. From the following discussions it appears the manifold pressure was likely having sensor issues and coupled with the fact that the amps anomaly occurred over the change in SSP's suggests the change in SSP's was not a factor.

The manifold pressure (red line figure 3) on the other hand was not recorded by the second SSP for stations 270 to 285 and then started recording till the complete failure of the second SSP at station 406. When the manifold pressure started recording at station 286 it had jumped slightly about 1 psi above the first SSP's last values, and then showed a sharp rise from about 36 psi to about 41 psi around station 325. The manifold pressure then dropped to a steady value of about 39 psi at station 345 and remained steady there after to the failure of the second SSP at station 405.

The small initial jump in pressure is within calibration errors from the first SSP to the second SSP. However from past surveys the manifold pressure should have followed a steady small downward trend due to pump wear (red dashed line figure 3). The rise in and fall in manifold pressure could be indicating a slightly plugged manifold but the pump amps, recorded independent of the SSPs, (blue line figure 3) did not drop/rise in agreement.

From the analysis of the 2005 NOAA clam survey, an unknown drift in the manifold pressure sensor readings before the pump was started (blue line figure 5b) occurred which created a false rise in the recorded manifold pressure (green line figure 5b). A possible sensor drift was also investigated for the 2008 survey, but as shown in figure 5a the same drift did not occur. Unlike the 2005 survey, the 2008 survey manifold pressure before pump start (blue line figure 5a) staid steady throughout the survey.

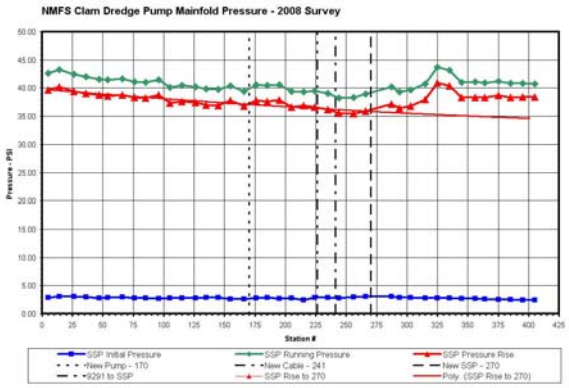

Figure 5a - 2008 Dredge Manifold Pressure

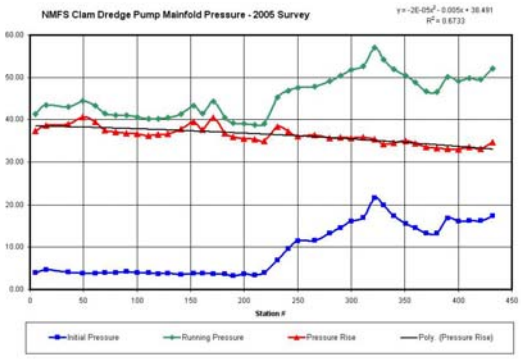

Figure 5b - 2005 Dredge Manifold Pressure

The SSP differential pressure sensor was changed from the 2005 survey's Trans Metric P022 unit to a Stellar Technology DT1900 unit for the 2008 survey which could explain the difference between 2005 and 2008 surveys. Neither manufacture was able to provide any insight into the sensor's performance.

Based on the above, no definitive judgment can be passed on the performance of the second SSP unit or the effect of the data recorded on the survey. However the second SSP's frequency values were steady at 60 hertz, voltage remained the same between the two SSP's, and the SSP

ambient pressure matched the mini-logger values, all indicating consistent SSP operation.

The change in manifold pressure, the key dredge performance measuring criteria, however is a concern about the second SSP unit. The change in manifold pressures though is fairly small and the value stays within the accepted 40 to 35 psi normal range. Further the stations with pump problems shown by the second SSP (station 402 figure 6) data have amp readings, recorded independently of the SSP, that are consistent and follow the patterns as occurred with the first SSP data (station 045 figure 6) and previous surveys (station 262 figure 6). Because of this the good/bad manifold pressure criteria is still valid for stations recorded by the second SSP.

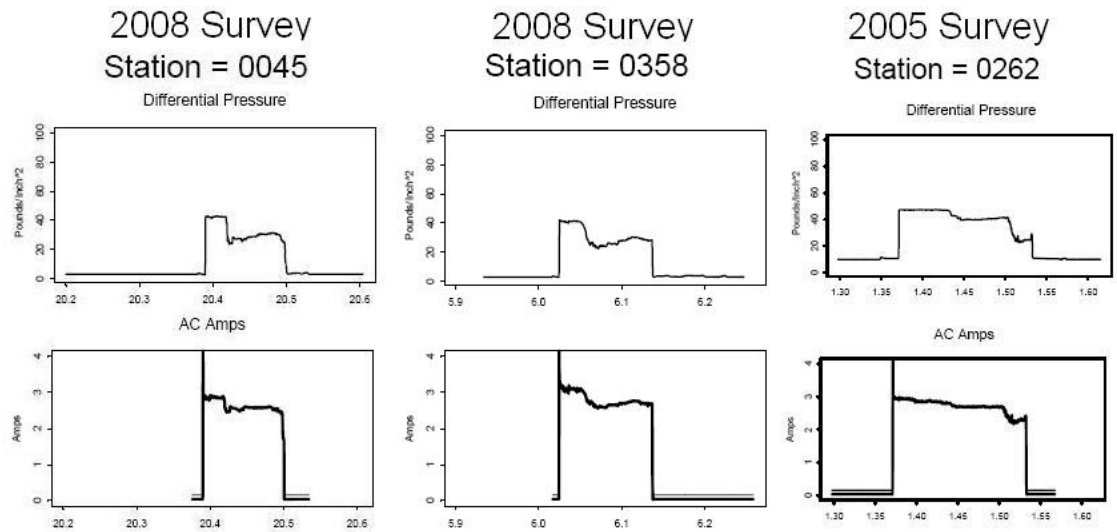

Figure 6 - 2008 & 2005 Survey 1st and 2nd SSP Manifold Pressure vs. Amps

Survey Dredge Y Tilt and X Tilt Fluctuations

From the visual inspection of the survey tow data plots the dredge Y tilt (fore/aft) and X tilt (side/side) seemed to have greater fluctuations than previous surveys. Several examples of tow Y and X tilt are shown in figure 7, with station 187 being typical of a "good" station for Y and X data. (Note different Y and X scales for degrees)

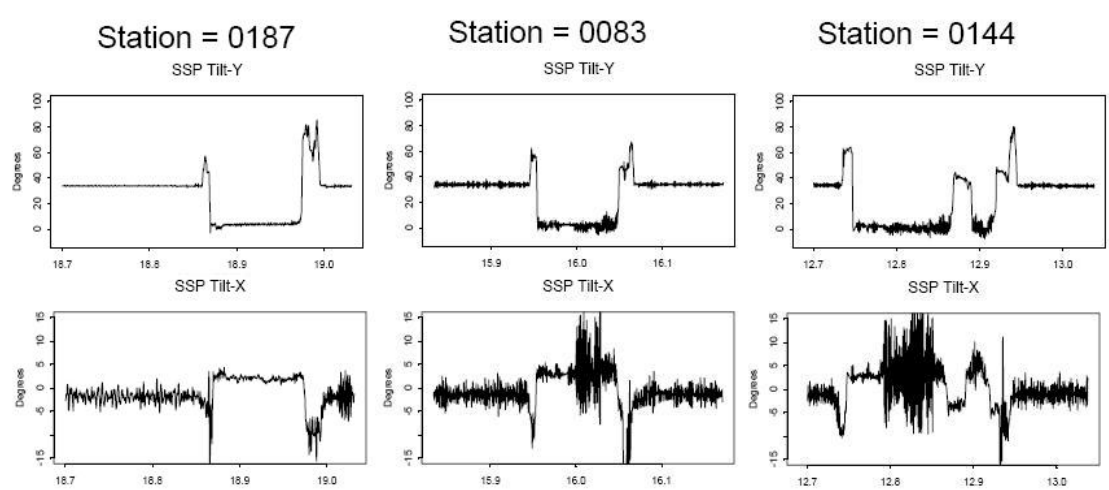

Figure 7 - NOAA Dredge 2008 Survey Y/X Tilt SSP Data Plot Examples

The SSP uses a 2 axis conductive liquid inclinometer to measure the Y and X tilt angles. This type of inclinometer measures the angle by sensing the level of a conductive fluid using (5) probes. Based on discussions with the clinometers' manufacture, the liquid used in the SSP's inclinometer has a viscosity about the same as water. Because of this the clinometers' liquid would be suspect to several error producing situations.

1) The liquid can slosh from sharp impacts or jolts.
2) The liquid can go into harmonic resonance at about 10 hertz (10 times per second).

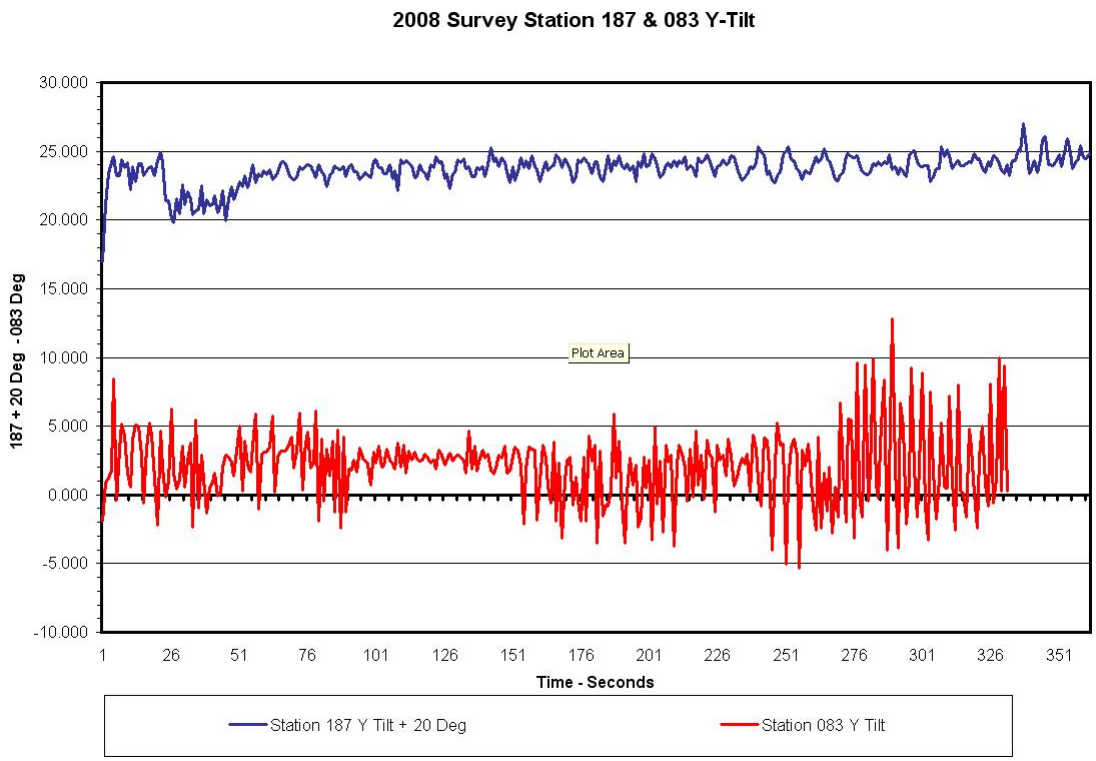

Figure 8 - 2008 Station 187 and 083 Comparison of SSP Y Tilt Plots

The sloshing of the clinometers liquid from sharp fore and aft jolting movements as the dredge jerks horizontally over the bottom can appear as a vertical Y tilting of the dredge. The rapid large vertical swings of station 083 tow (red line figure 8) are most likely from sloshing of the clinometers' conductive liquid due to the dredge jerking fore/aft horizontally through the bottom, not actual dredge vertical movement. The large 10 degree vertical swings at the end the tow are most likely from the clinometers' conductive liquid sloshing in resonance. (Good station 187 Y tilt, blue line, is plotted as a comparison.)

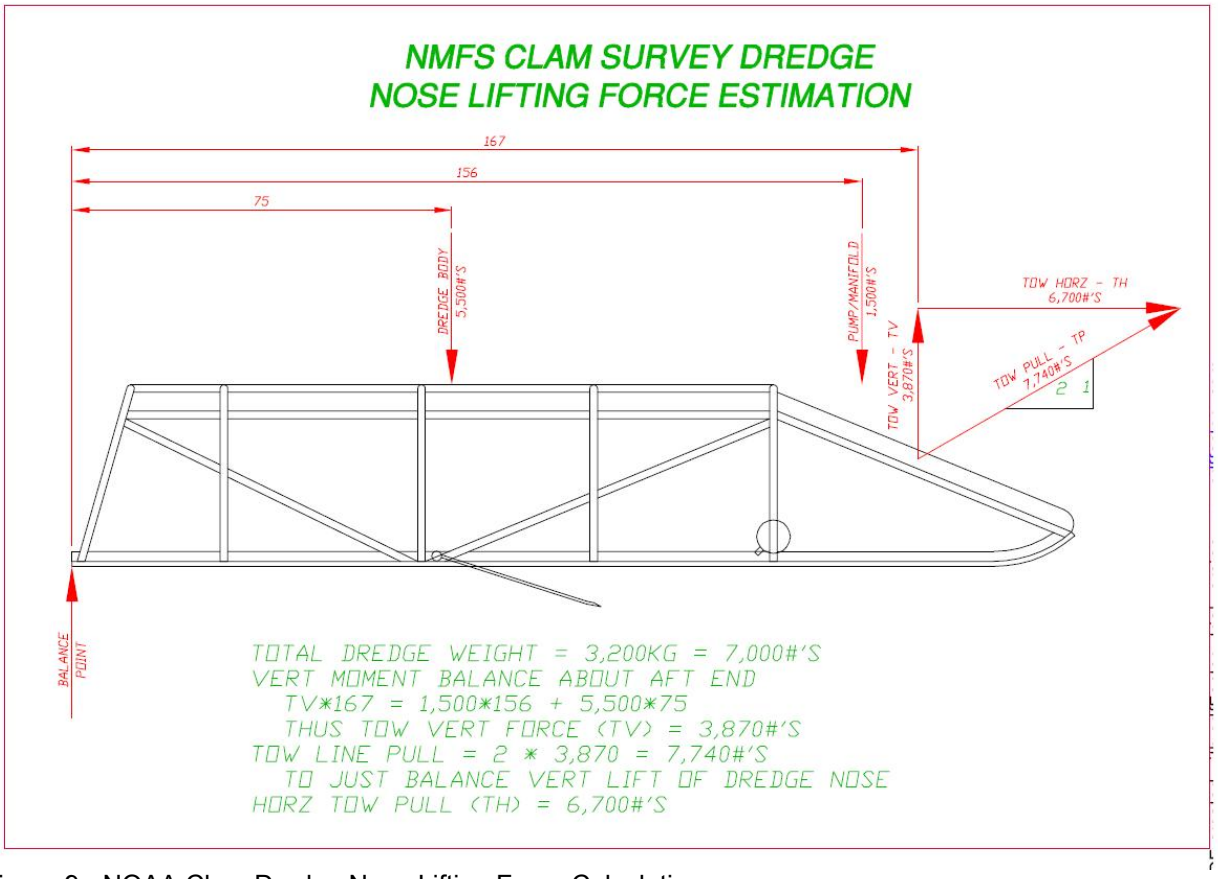

Figure 9 - NOAA Clam Dredge Nose Lifting Force Calculation

Further evidence the large Y tilt swings are from the inclinometer sloshing is the large towline pull that would be required to lift the nose of the dredge off the bottom. Figure 9 is an estimation of the towline pull that would be required to lift the nose of the dredge off the bottom. From a moment balance calculation, approximately 7,700 #'s of towline pull would be required to just balance the dredge on the aft end of the runners. But this 7,700 # towline pull also creates a horizontal pulling force of 6,700 #'s, more than ample to pull the dredge forward, particularly after the dredge's knife is completely above the bottom at a Y angle rise of about 4.4 degrees.

The last evidence the large Y tilt swings are from the inclinometer sloshing is the physical fact that it is not possible for the dredge's large flat runners to bury in the bottom as the plots would suggest. For station 083 shown, its normal running angle appears to be about 3 degrees (time 100 to 150 red line figure 8). Yet from the plots the dredge and its runners are burying 5 to 10 degrees on 1 second intervals in to the bottom, not a realistic situation. The 1 to 2 degree bounces on roughly 5 second intervals for station 187 (blue line figure 8) are realistic.

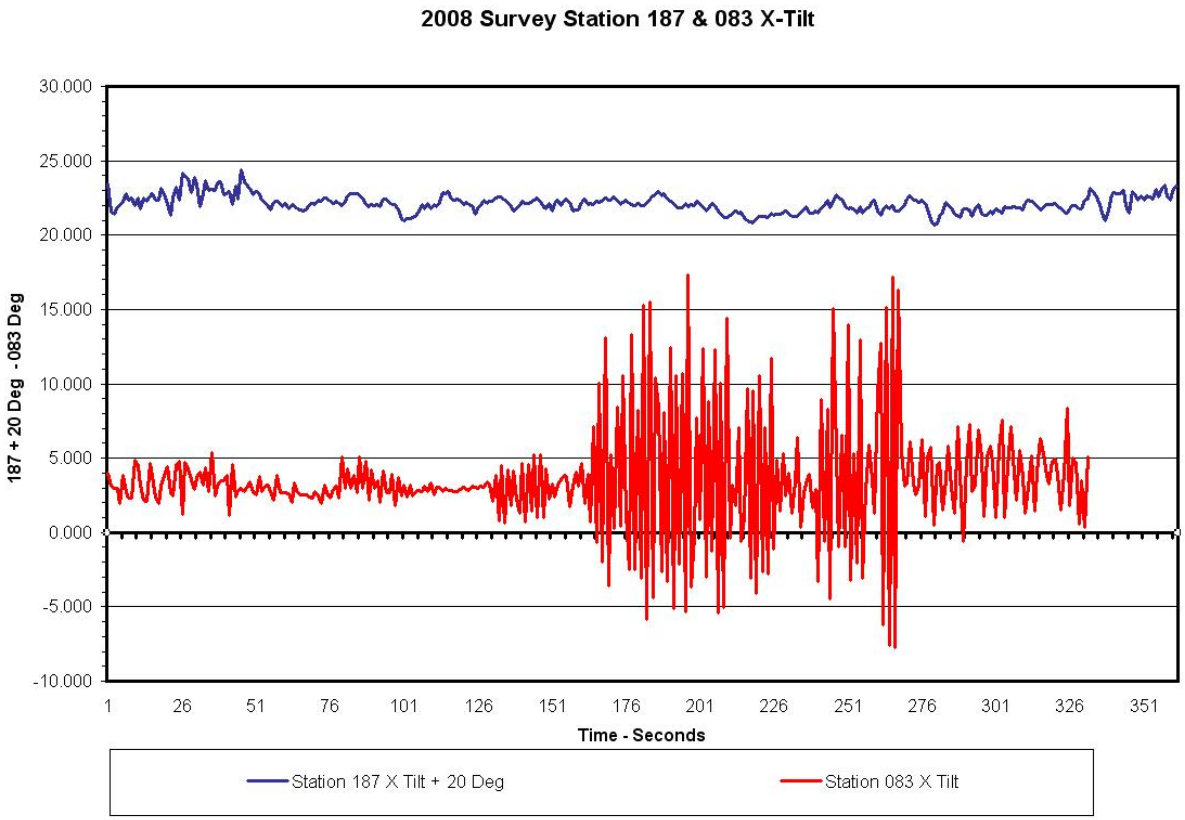

Figure 10 - 2008 Station 187 and 083 Comparison of SSP X Tilt Plots

As with Y tilt, sharp sideways jolting movements of the dredge can appear as an excessive side X tilting of the dredge. The rapid large X swings of station 083 tow (red line figure 10) are again most likely from sloshing of the clinometers' conductive liquid, probably in resonance during the 20 degree plus swings. (Good station 187 Y tilt, blue line, is plotted as a comparison.)

Because these rapid Y and X tilt fluctuations are likely due to a SSP sensor problem, and are not the actual movement of the dredge, these fluctuations can be ignored in evaluating the dredge's performance. Extreme problems in the dredge's running angle such as shown by the station 144 plots in figure 7 will not be ignored by this assumption. In this case the dredge jumped up about 40 degrees for a brief period in the latter part of the tow due to a sudden very large 5 knot increase in vessel speed. This non fishing period though will be compensated for in the tow length calculations and thus be correctly accounted for in the survey results. As such the Y-Tilt Criteria developed for the 2005 survey is no longer applicable and was not applied to the 2008 survey.

Based on the above, the 2005 NOAA survey Y and X tilt plots were re-evaluated and similar Y and X fluctuations were noted, though with a significant lesser number of occurrences than the 2008 survey. Typical examples of stations from the 2005 NOAA survey are shown in figure 11. Station 137 is a typical good station for smooth Y and X tilt plots. Station shows similar Y and X tilt fluctuations to the 2008 survey discussed above. The one 2005 survey station that was flagged as "bad" by the Y-Tilt criteria was station 218 shown in figure 11. As discussed above the Y tilt spike in the middle of the tow will be accounted for in the tow length calculations and thus station 218 can be placed back into the survey calculations.

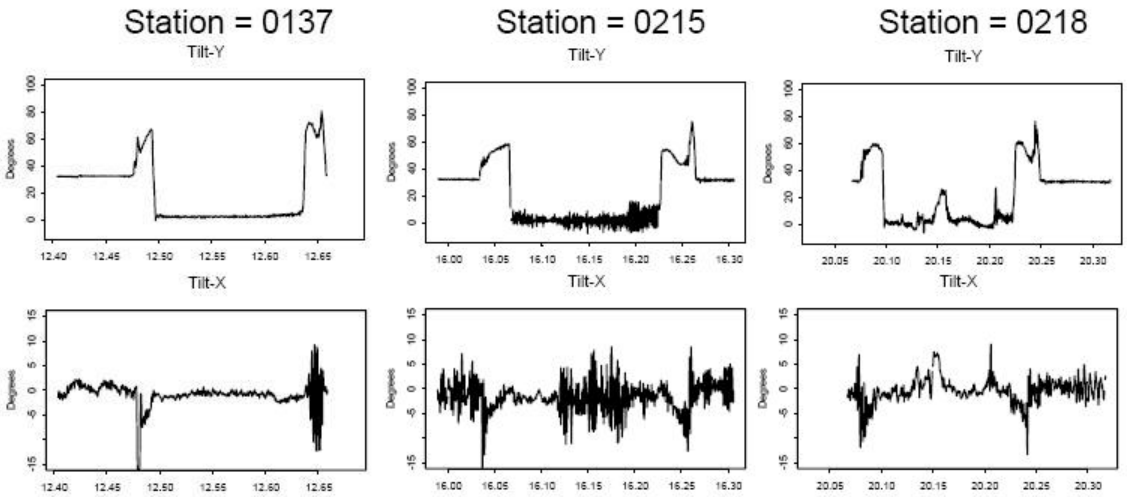

Figure 11 - NOAA Dredge 2005 Survey Y/X Tilt SSP Data Plot Examples

SSP vs Mini-Logger Ambient Pressure Comparison

From figure 4 there was an interesting anomaly in the SSP ambient pressure recorded in the data files. The SSP ambient pressure (blue line figure 4) tracks the mini-logger ambient pressure in psi (green line figure 4) up to station 226. At station 227 the SSP ambient pressure now tracks the mini-logger ambient pressure in decibars (red line figure 4) till the SSP data ends at station 405. In the excel data files the column header for SSP ambient pressure is "PRESS.AM9291" up to station 226, then switches to "PRESS.AM.SSP" for the remainder of the survey tows. This switch in header labels also occurred for SSP ambient temperature, tilt X, and tilt Y.

This unit jump appears to only have occurred in the SSP ambient pressure data. The SSP ambient temperature tracked the mini-logger ambient temperature across the full survey (see figure 12). The average Y tilt and X tilt before and after stations 226/227 was also calculated to see if a problem occurred. The Y and X tilt was stopped at station 269 when the SSP was replaced and there is a minor calibration difference between the two SSP units as discussed previously. Again from the data below it dose not appear if there was any change in the X or Y before to after station 226/227. (Note stations used in these comparisons were the good stations used to develop dredge pump performance plot in figure 3.)

	Y Tilt	X Tilt
First SSP Stations 1 to 226	3.39	2.71
First SSP Stations 227 to 269	3.38	2.75

Based on the above, this unit switch did not affect any of the 2008 survey tow review.

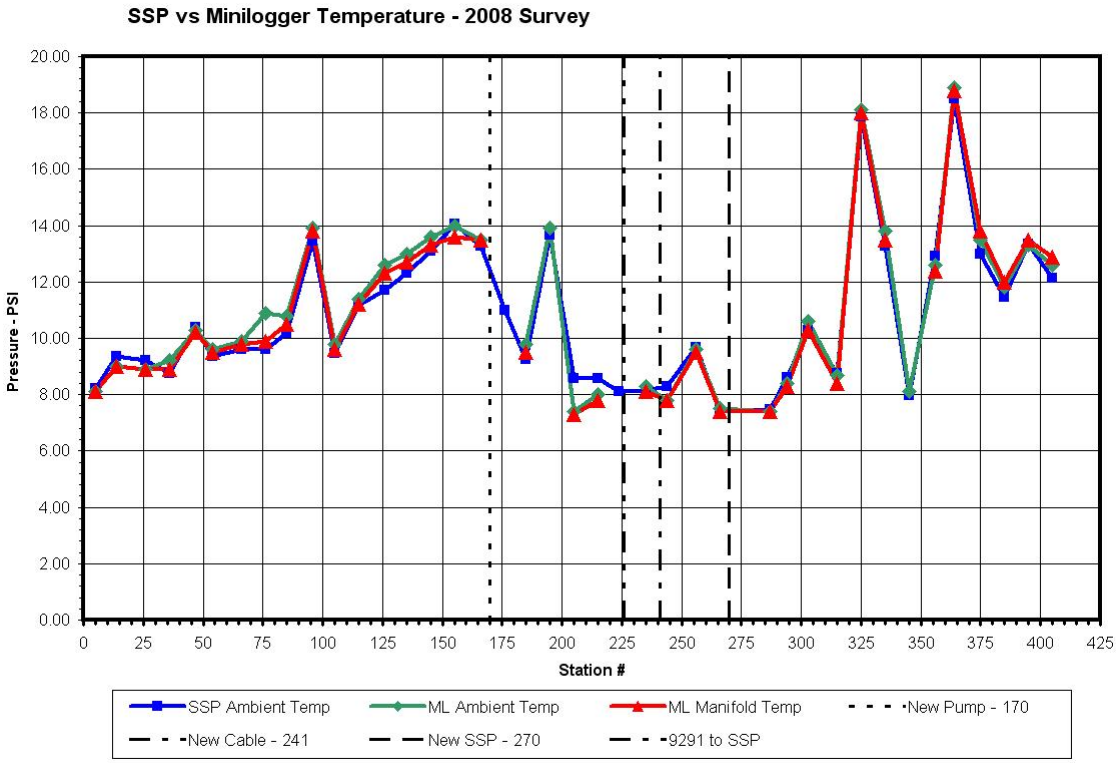

Figure 12 - SSP vs. Mini-Logger Ambient Temperature - 2008 NOAA Survey

Application of Manifold Pressure Good/Bad Tow Criteria to 2008 Survey

As with the 2005 NOAA clam survey, there were numerous stations that experienced manifold pressure problems during the 2008 survey. These suspect stations were evaluated using the good/bad manifold pressure criteria that was developed for the 2005 survey. In summary the criteria compares the time the manifold pressure was in the "normal" operating range of 35 to 40 psi with the time it was outside of that range. If the time outside of the range exceeded the time within the normal range by more than 25%, the tow is labeled a "bad" tow.

The 2008 survey did present one problem in using the good/bad manifold pressure criteria, the lack off SSP manifold pressure data after station 405. Fortunately the dredge pump's amp draw is recorded independent of the SSP's and was available for use in these latter stations. Figure 13 is a plot of several stations that experienced pressure problems were both SSP manifold pressure and amps were available. This plot was used to develop a manifold pressure from amps proxy that would allow use of the good/bad manifold pressure criteria for stations after 405.

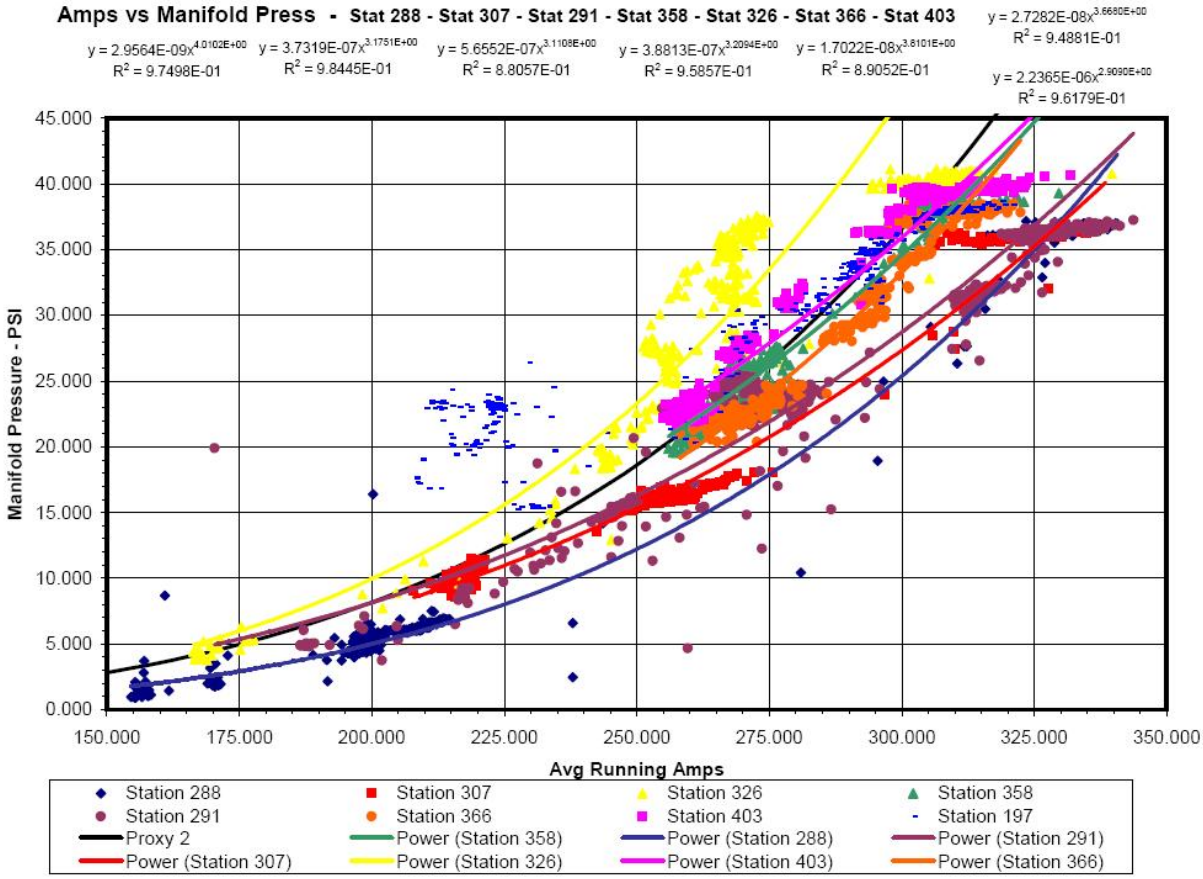

Figure 13 - Manifold Pressure vs. Pump Amps Proxy

The selected proxy is the black line in figure 13. This proxy was set by visual trial and error to best match stations 403, 358, and 366. These stations were selected as they occurred towards the end of the SSP available data and best matched the amps/pressure relationship of a normally operating pump in the latter tows.

The list on the following page are the stations determined to be "bad" by the manifold pressure criteria.

NOAA Clam Survey Station 1 to 405 Manifold Pressure Good/Bad Tow Criteria - All Bad Tows

Station #	Weighted Time < 35 PSI	Weighted Time 35-40 PSI	Weighted Time > 40 PSI	%Time Under	%Time Over	25% Good Bad	Womack #	Other Issues
15	151.95	70.00	0.01	217%	0%	Bad	5	
29	91.34	242.00	0.01	38%	0%	Bad	5	
35	70.37	205.00	0.00	34%	0%	Bad	5	
43	246.60	170.00	0.00	145%	0%	Bad	5	
45	79.92	4.00	11.55	1998%	289%	Bad	5	
48	60.77	160.00	0.00	38%	0%	Bad	4	
52	56.30	137.00	0.00	41%	0%	Bad	5	
65	129.17	119.00	0.00	109%	0%	Bad	5	
95	148.77	39.00	0.00	381%	0%	Bad	5	
99	341.51	47.00	0.00	727%	0%	Bad	5	
119	126.18	87.00	0.00	145%	0%	Bad	5	
137	304.97	54.00	0.00	565%	0%	Bad	5	
138	949.11	232.00	0.00	409%	0%	Bad	5	
141	249.06	217.00	0.00	115%	0%	Bad	5	
150	74.99	278.00	0.12	27%	0%	Bad	3	
164	71.39	88.00	0.00	81%	0%	Bad	3	
165	67.19	165.00	0.00	41%	0%	Bad	3	
169	235.57	39.00	0.00	604%	0%	Bad	5	
175	91.08	338.00	0.00	27%	0%	Bad	4	
197	104.26	183.00	0.00	57%	0%	Bad	4	
198	554.15	63.00	0.00	880%	0%	Bad	5	
206	91.58	269.00	0.00	34%	0%	Bad	4	
209	72.08	195.00	0.00	37%	0%	Bad	4	
226	41.14	112.00	0.00	37%	0%	Bad	3	
227	83.24	169.00	0.00	49%	0%	Bad	5	
229	153.22	137.00	0.00	112%	0%	Bad	5	
241	122.53	144.00	0.00	85%	0%	Bad	5	
242	92.20	96.00	0.00	96%	0%	Bad	5	
245	78.76	13.00	0.00	606%	0%	Bad	5	
246	75.65	0.00	0.00	#######	0%	Bad	3	
248	179.34	51.00	0.00	352%	0%	Bad	5	
249	120.82	80.00	0.00	151%	0%	Bad	5	
250	69.02	15.00	0.00	460%	0%	Bad	4	
252	52.64	135.00	0.00	39%	0%	Bad	3	
254	582.83	166.00	0.00	351%	0%	Bad	5	
257	162.74	71.00	0.00	229%	0%	Bad	5	
258	107.49	127.00	0.00	85%	0%	Bad	5	
262	92.52	229.00	0.00	40%	0%	Bad	5	
263	160.98	196.00	0.03	82%	0%	Bad	5	
288	510.08	160.00	0.00	319%	0%	Bad	5	
290	169.52	272.00	0.00	62%	0%	Bad	5	
291	223.76	141.00	0.00	159%	0%	Bad	5	
293	71.66	230.00	0.00	31%	0%	Bad	2	
305	63.43	88.00	0.00	72%	0%	Bad	4	
306	117.97	97.00	0.00	122%	0%	Bad	5	
307	309.04	76.00	0.00	407%	0%	Bad	5	
308	389.37	54.00	0.00	721%	0%	Bad	5	
309	239.78	141.00	0.00	170%	0%	Bad	5	
310	398.91	117.00	0.00	341%	0%	Bad	5	
317	58.74	89.00	0.00	66%	0%	Bad	3	
326	119.05	134.00	1.59	89%	1%	Bad	5	
358	144.09	92.00	0.00	157%	0%	Bad	5	
366	94.22	109.00	0.00	86%	0%	Bad	5	
394	499.73	453.00	0.00	110%	0%	Bad	5	
402	150.58	175.00	0.01	86%	0%	Bad	5	
403	342.60	209.00	0.00	164%	0%	Bad	5	
424	583.83	35.00	3.96	1668%	11%	Bad	5	Manifold Press Amps Proxy
430	354.42	45.00	11.91	788%	26%	Bad	5	Manifold Press Amps Proxy
433	132.47	251.00	0.14	53%	0%	Bad		Manifold Press Amps Proxy
434	508.41	4.00	0.00	12710%	0%	Bad	5	Manifold Press Amps Proxy
435	428.23	2.00	0.00	21412%	0%	Bad	5	Manifold Press Amps Proxy
436	353.87	1.00	0.00	35387%	0%	Bad		Manifold Press Amps Proxy
437	343.27	17.00	0.00	2019%	0%	Bad	5	Manifold Press Amps Proxy
438	325.45	16.00	0.05	2034%	0%	Bad	5	Manifold Press Amps Proxy
448	236.08	21.00	0.00	1124%	0%	Bad	5	Manifold Press Amps Proxy
452	445.80	20.00	0.01	2229%	0%	Bad	5	Manifold Press Amps Proxy
453	135.09	92.00	0.61	147%	1%	Bad	5	Manifold Press Amps Proxy

APPENDIX B4: Cooperative survey report from F/V Endeavor.

2008 Cooperative Industry Surf Clam/Ocean Quahog Survey Cruise Report
F/V Endeavor

SUMMARY

The 2008 Cooperative Surf Clam/Ocean Quahog Survey took place from September 10-23, 2008 following the 2008 NEFSC clam survey during June. The *F/V Endeavor*, based in Atlantic City, NJ was the commercial vessel used in the cooperative survey while the NEFSC survey used the NOAA Fishing Vessel R/V Delaware II. Leg 1 of the cooperative survey took place during September 11-15, leg 2 during 15-19th; and leg 3 during September 20-23rd, 2009. The cooperative survey was a joint effort by the National Fisheries Institute Clam Committee, Rutgers University, Virginia Institute of Marine Science, and the Northeast Fisheries Science Center.

Principal objectives of the survey were to: (1) further evaluate the feasibility of a cooperative clam survey using commercial vessels; 2) augment the NEFSC clam survey by repeating stations already sampled by the R/V Delaware II using the NEFSC clam survey dredge; (3) estimate efficiency of the NEFSC survey and commercial dredges by conducting depletion experiments; and (4) collect data for use in estimating size-selectivity for surfclams in the commercial and NEFSC survey dredges.

VESSEL, GEAR, and CREW INFORMATION

The *F/V Endeavor* is a 165-foot fishing vessel with a 42-foot beam, a 14,000-gallon fuel tank, and a 12,000-gallon fresh water tank. It has two 12.5-foot wide dredges, deployed by hydraulic power-out winches. The vessel was specifically outfitted with dredges that had bars with spacing reduced to 0.75 inches to retain small ocean quahogs and surfclams. The starboard dredge was lined with 1-inch hexagonal chicken wire for size selectivity studies. The dredge knives were set at 5.25 inches for surf clam sites and at 4.25 inches for quahog sites.

Two small belts ran the catch from the port and starboard hoppers onto a larger, centralized belt that transported the catch across a shaker table and onto a sorting belt. The large belt before the shaker table was about 4 feet wide and 10 feet long. Alongside the belt was a large, metal stand where workers could access the catch before it reached the shaker table, where the catch was mechanically sorted. The average spacing between the rolling bars on the shaker table was 0.73 (+/-0.10) inches.

A NEFSC Survey Sensor Package (SSP) that records latitude, longitude, angle of the dredge (fore/aft and port/starboard), temperature, depth, and internal manifold pressure every second was carried inside the port dredge and was operational for parts of legs 1 and 2. Two Vemco mini-loggers (which record ambient temperature and pressure/depth) were fastened to each dredge on a metal rod welded to the top near the manifold. The mini-logger sensors were operational during all three survey legs.

The crew was split into two, 12-hour shifts so that operations could take place around the clock. Each shift was made up of seven people, including the captain or mate, four scientists, and two crew members. On-deck responsibilities, including sorting and measuring the catch, were shared by all four scientists on shift. In addition, one scientist was responsible for interacting with

the captain to execute the cruise plan and one scientist (from NEFSC) was responsible for operating the SSP software package. Having seven people on each shift worked well and allowed the catch to be processed in a timely fashion while steaming between sites.

SITE DESCRIPTIONS AND METHODS

A. Surf Clam Size Selectivity Sites

Experiments were done at these sites to determine the size-selectivity of the commercial dredge and the NEFSC survey dredge by comparing catches from a lined commercial dredge, an unlined commercial dredge and a NEFSC survey dredge at the same site. Selectivity experiment sites were chosen based on location, and the size and species composition of the NEFSC survey dredge catches in 2008.

Experimental protocol was to first tow 5-minutes with the port (unlined) dredge. The catch was allowed to run over the shaker table and onto the sorting belt in the normal fashion in order to capture effects of both the dredge and shaker table on size selectivity. The shaker table had been pre-configured to increase selectivity of the commercial equipment as a whole for small quahogs. Thus, size selectivity for small ocean quahogs may be higher than during normal commercial operations. The total number of bushels in addition to the number of clams in any partial bushel was counted along with the number of clams in two full bushels to permit conversion of bushel counts to numbers of animals. Clams in two full bushels were also measured to the nearest mm.

The site was then towed for 30 seconds along an adjacent track using the starboard (lined) dredge. This time the catch was sorted before going over the shaker table so that the entire catch was sampled, until at least 6 full bushels of clams had been collected. All clams in the six full bushel samples were measured, regardless of size. The remainder of the catch was discarded. The volume of the catch was too large to sort the entire catch or accurately measure its volume. However, size composition data for surf clams in both tows at the site are directly comparable. Sorting the catches from the lined dredge generally took between one and three hours.

C. Surf Clam and Ocean Quahog Depletion Experiment

Depletion experiments were conducted to estimate capture efficiency of the commercial and NMFS survey dredge. The R/V Delaware II completed five "setup" tows at a predetermined site prior to the arrival of the commercial vessel. The setup tows were generally parallel and oriented either north-south or east-west.

After arriving at the site, the chief scientist aboard the F/V Endeavor selected a rectangular area near as many of the five setup tows as possible. The rectangle was oriented perpendicular to the setup tows to the extent possible with a target width of about 10 times the width of the dredge (125 feet). The length of the site was chosen so that initial catches were at least 10 bushels per tow (typically 1200 to 2400 feet) based on trial tows near the edge and parallel to depletion site.

After the size of the site was defined, depletion tows were carried out repeatedly (typically 17-22 tows per site) by the F/V Endeavor using the port dredge until the site showed substantial depletion and catch per tow declined significantly. Tow paths were adjusted based on GPS data to tow sufficiently over the entire rectangle to see a significant decline in catch per tow in all areas of the rectangle. In most cases, this took place after the entire area of the rectangle was covered at least twice with the dredge –usually between 17 and 22 tows. Each tow was approximately 5-minutes in duration. Ship positions were recorded during maximally every 5 seconds, after which the catch was allowed to run over the shaker table and onto the sorting belt. On every tow the number of clam

bushels was counted and the partial bushel estimated. On every fifth tow, starting with tow two, one full bushel was measured and a second counted. Depletion experiments took anywhere between 9 and 16 hours to complete depending on the conditions at the site and the number of animals in the selected rectangular grid.

D. Surfclam and Ocean Quahog Repeat Stations

About halfway through the 2008 NEFSC clam survey with the Delaware II, an electrical cable used to power the pump on the survey dredge was replaced with a longer cable. Similarly, the pump on the NEFSC survey dredge was replaced after the original pump failed after about a third of the survey. The *F/V Endeavor* reoccupied some stations originally towed by the Delaware II which was using various configurations of old and new equipment to help quantify potential changes in survey dredge efficiency due to changing equipment. In some cases, these repeat station experiments were combined with or carried out at the same location as surfclam size selectivity and depletion experiments.

These sites had already been occupied either once or twice by the Delaware II during 2008 using the NEFSC survey dredge and the old and/or new cable and pump. At these sites the *F/V Endeavor* towed the port dredge for 5-minutes. The catch was run over the shaker table and onto the sorting belt. The total number of bushels was counted. The number of clams in the partial bushel and in two full bushels was counted, and all clams in the two full bushels were measured to the nearest mm.

Results

See Table 1 and Figure 1, which list the location and type of all cooperative stations, along with station numbers from the NEFSC clam survey for repeat stations.

The length frequency of all ocean quahogs measured on the survey can be found in Figure 2. The length frequency of all surf clams measured from 5-minute, unlined tows (size-selectivity experiments and depletion experiments) can be found in Figure 3. The length frequency of all surf clams measured from 30-second, lined tows can be found in Figure 4.

Sensor data and area swept

Sensor data was used to determine when the dredge was on/off bottom. Times on/off bottom were then matched to a GPS record of the ship's position to estimate area swept by the dredge. The NEFSC Survey Sensor Package used during the cooperative survey records latitude, longitude, angle of the dredge (fore/aft and port/starboard), temperature, depth, and internal manifold pressure every second. The frequency and resolution of the output data make it easy to determine when the dredge is on bottom and fishing. SSP data were not collected for some tows during Legs I and II because the battery could not be not fully charged due to lack of time between stations. Also, the SSP was not operational during Leg III due to lack of trained scientific staff. Therefore, SSP sensor data were available for less than half of the sites occupied. Fortunately, backup GPS and sensor data including ambient temperature and pressure (depth) from backup sensors are available for every tow.

The backup GPS and sensors were used to determine time on-bottom and area swept for tows with no SSP data. Backup sensors record depth at a lower resolution (accuracy approximately 5 meters) and at a lower frequency (5 second intervals) than the SSP. It was therefore necessary to use SSP data where available to develop procedures for estimating time on/off bottom and area swept using backup sensor data. The following steps were taken to determine when the dredge was fishing and subsequently estimate the area swept using these sensors for tows where SSP data was not

available:

1. The backup pressure (depth) data for each station was used to estimate times the dredge was on or off bottom. The resolution of the backup pressure data is 5 meters and the apparent trajectory of the dredge during the tow is noisy. In particular, a small change in depth can appear to be a large change. This adds uncertainty to the estimates of time on/off bottom.

2. Initial time on/off bottom estimates based on backup sensor data were compared to estimates from SSP data for 51 surfclam stations with SSP data. In comparing time on/off bottom estimates made using backup sensor and SSP data, it was noted that estimates based on backup sensors lagged SSP estimates by about 15 seconds. Estimates based on backup sensors were therefore corrected by subtraction of 15 seconds. After this adjustment, times on/off bottom differed, on average, by only 1 second (Table 2). Furthermore, after applying this correction, the chance of the backup sensor estimate being ahead of the SSP estimate and the chance of the backup sensor estimate being behind the SSP annotation were equal. The lag method was applied to all tows for which SSP data were lacking.

3. The initial time on/off bottom estimates based on backup sensor data were compared to estimates from SSP data for 34 ocean quahog tows from depletion experiments OQ0801 and OQ0802. Backup sensor estimates of time off bottom matched well with the SSP estimates. However, the backup sensor estimate of time on bottom averaged 15 seconds ahead of the estimates based on SSP data. With the adjustment for a 15 seconds lag described above, the backup sensor estimates differed from the SSP annotations by an average of four seconds. Furthermore, after applying this correction, the chance of the backup sensor estimate being ahead of the SSP estimate and the chance of the backup sensor estimate being behind the SSP annotation were equal. Therefore, the 15 second adjustment was used for all Vemco files across all tows and all experiments for which SSP data were lacking.

4. The SSP and adjusted backup sensor estimates of time on/off bottom were used to determine the area swept.

COMMENTS

Having primary (SSP) and backup GPS and sensor data for each tow is critical. Efforts should be made to increase the reliability of the SSP on commercial vessels and to increase the resolution and the recording frequency of backup sensors.

The ambient pressure sensor on the SSP malfunctioned unexpectedly because the tubing connecting it to the dredge had a tendency to plug up. A different approach to mounting the pressure sensor should be used next time.

Backup sensors should include an inclinometer to measure the fore/aft angle of the dredge, which are useful data in determining time on/off bottom.

Power out winches made it difficult to drop the dredge within a specific rectangular area during depletion experiments, and increased difficulties in interpreting time on/off bottom from backup sensor data. Boat operators were able to adjust towing procedures and to drop the dredge reliably in the rectangular area. However, the number of unsuccessful attempted tows increased over the previous years, adding time to the total time required to conduct the experiments. In the future an effort should be made to use free-fall winches.

The chicken wire liner proved to be sturdy and reliable. No repair was needed except at the leading edge behind the knife. Welding a bar across this leading edge in the future would eliminate this one weak point and permit long-term use of a lined dredge for improved estimates of smaller clams.

SCIENTIFIC CREW

Below is a list of names and email addresses for the scientific crew that participated in the survey. In addition to the science crew, aboard the vessel for all three legs were the captain, first mate, four crew members, and a cook (16 persons in total on each leg).

Legs 1 and 2:

Kathryn Ashton-Alcox, HSRL	kathyryn@hsrl.rutgers.edu
Jenn Gius. HSRL	jengius@hsrl.rutgers.edu
Shad Mahlum, NOAA-NMFS	shad.mahlum@noaa.gov
Roger Mann, VIMS	rmann@vims.edu
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Jason Morson, HSRL	jmorson@hsrl.rutgers.edu
Chris Pickett, NOAA-NMFS	cpickett@mercury.who.whoi.edu
Eric N. Powell, HSRL	eric@hsrl.rutgers.edu
Erin Reilly, VIMS	ereilly@vims.edu

Leg 3:

Kathryn Ashton-Alcox, HSRL	kathyryn@hsrl.rutgers.edu
Roxanne Carter, REMSA Inc.	roxy@remsameso.com
Jenn Gius. HSRL	jengius@hsrl.rutgers.edu
Rebecca Marzec, HSRL	marzec@hsrl.rutgers.edu
Jason Morson, HSRL	jmorson@hsrl.rutgers.edu
Eric N. Powell, HSRL	eric@hsrl.rutgers.edu
Zachariah Sheller, REMSA Inc.	zsheller@yahoo.com

Table 1. 2008 Cooperative Industry Surf Clam/Ocean Quahog Survey station list. "Shape on Map" refers to the map in Figure 1 where all stations are plotted using specific shapes to identify the purpose of the station.

NMFS Site #	NMFS Depletion #	Shape on Map	Site Type	Lat	Long	# of Surf Clam Bushels (Depletion Sites, Tow 1 Only)	# of Quahog Bushels (Depletion Sites, Tow 1 Only)	Comments
36	N/A	STAR	Surf Clam Size Selectivity	39.8597	73.7122	4	1.33	
49	N/A	STAR	Surf Clam Size Selectivity	39.6523	74.0078	6	0	
60	N/A	STAR	Surf Clam Size Selectivity	39.5688	74.1133	5.5	0	
64	N/A	STAR	Surf Clam Size Selectivity	39.4385	74.1782	3	0	
292	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	40.0633	73.6757	22.33	0.67	
293	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.9765	73.5343	22	8.25	
294	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.9427	73.588	22	0.67	
295	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.8575	73.4783	22	3	

296	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.7323	73.4477	29.75	0	
303	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.7213	73.8003	11	0	
304	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.7723	73.844	22.25	0	
310	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.8118	73.9473	17.75	0	
312	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.939	73.814	17	0.01	
313	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.9788	73.7162	19.5	0.25	
314	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.9832	73.8482	9	0	
315	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	40.1027	73.7745	22	0.33	
316	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	40.1465	73.945	28	0	

318	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.5633	73.9113	9.5	0	
319	N/A	CROSS	Repeat Surf Clam / Surf Clam Size Selectivity	39.4768	73.911	11	0	
67	SC08-01	CIRCLE	Surf Clam Depletion	39.3073	74.054	6.5	0	
74	SC08-02	CIRCLE	Surf Clam Depletion	39.188	74.0753	16.67	0	
297	SC08-03	CIRCLE	Surf Clam Depletion	39.6028	73.41	16	0	
305	SC08-04	CIRCLE	Surf Clam Depletion	39.8093	73.9132	11	0	
358	SC08-05	CIRCLE	Surf Clam Depletion	41.1457	70.047	14	0	The running tide, wind, and waves made it impossible to stay inside the rectangle at this location. Therefore, this site was terminated after 6 tows.
N/A	N/A (SC08-09)	CIRCLE	Surf Clam Depletion	39.3117	74.0537	14	0	We picked this site as an additional depletion site because SC08-05 was untowable.
324	N/A	SQUARE	Repeat Quahog	40.8915	71.859	0	14.5	

326	N/A	SQUARE	Repeat Quahog	40.9422	71.9528	0	0	No catch here. This tow was not run through the hopper because the dredge was filled with large rocks.
333	N/A	SQUARE	Repeat Quahog	40.8555	72.12	0	43.33	
334	N/A	SQUARE	Repeat Quahog	40.8138	72.1755	0	20.25	
336	N/A	SQUARE	Repeat Quahog	40.773	72.4152	0	13	
338	N/A	SQUARE	Repeat Quahog	40.726	72.6485	0	14	
339	N/A	SQUARE	Repeat Quahog	40.558	72.6467	0	28	
199	N/A	INV. TRIANGLE	Quahog Old Wire	40.2568	73.2653	0	6	
201	N/A	INV. TRIANGLE	Quahog Old Wire	40.1497	73.0467	0	29.25	
203	N/A	INV. TRIANGLE	Quahog Old Wire	40.2747	72.9737	0	27	
205	N/A	INV. TRIANGLE	Quahog Old Wire	40.3165	72.7473	0	18.5	
207	N/A	INV. TRIANGLE	Quahog Old Wire	40.187	72.9453	0	35.75	
209	N/A	INV. TRIANGLE	Quahog Old Wire	40.0577	72.8393	0	37.5	
272	N/A	TRIANGLE	Quahog New Wire	40.5608	72.2457	0	22.75	
274	N/A	TRIANGLE	Quahog New Wire	40.6503	72.278	0	6.5	
276	N/A	TRIANGLE	Quahog New Wire	40.7298	72.2808	0	5.25	
278	N/A	TRIANGLE	Quahog New Wire	40.7298	72.086	0	64.5	
280	N/A	TRIANGLE	Quahog New Wire	40.8082	71.7798	0	0.67	
282	N/A	TRIANGLE	Quahog New Wire	40.6865	71.948	0	24.67	

173	OQ08-01	DIAMOND	Quahog Depletion	40.9363	72.0428	0	31.33	
287	OQ08-02	DIAMOND	Quahog Depletion	40.2702	72.8483	0	30	
344	OQ08-05	DIAMOND	Quahog Depletion	40.721	71.3465	0	4	This site was untowable.
351	OQ08-06	DIAMOND	Quahog Depletion	41.0172	70.8558	0	34	
N/A	N/A (OQ08-09)	DIAMOND	Quahog Depletion	41.0187	70.8559	0	24	We picked this site as an extra one because OQ08-05 was untowable, however, we needed to leave this site after 6 tows to bring in a sick crew member.

Table 2. (On following pages): 15-second adjustments made to Vemco sensor on-bottom and off-bottom records to more closely match SSP on-bottom and off-bottom records. Columns 1 and 2, Depletion and Tow or Site #, identify the site. Column 3 and 5, On-Bottom-VEMCO and Off-Bottom-VEMCO, are the times the dredge was on the bottom and fishing and then off bottom, respectively, according to VEMCO sensor annotations. Adjusted + 15 seconds in columns 4 and 6 are the same times, but with a 15-second, or three reading adjustment. Columns 7 and 8, On-Bottom SSP and Off- Bottom SSP, are the times the dredge was on the bottom and fishing and then off bottom, respectively, according to SSP sensor annotations. The last four columns calculate the difference in seconds between the SSP data and the Vemco sensor data annotations before and after the 15-second adjustment was made.

Depletion Station	Tow or Site #	On Bottom - VEMCO	Adjusted: + 15 seconds	Off Bottom- VEMCO	Adjusted: + 15 seconds	On Bottom- SSP	Off Bottom- SSP	On Bottom Difference: Un-adjusted	Off Bottom Difference: Un-adjusted	On Bottom Difference: adjusted	Off Bottom Difference: adjusted
SC08-01	2	14:00:22	14:00:38	14:12:42	14:12:57	14:00:50	14:12:56	0:00:28	0:00:14	0:00:12	-0:00:01
SC08-01	3	14:49:15	14:49:30	15:01:15	15:01:30	14:49:25	15:01:27	0:00:10	0:00:12	-0:00:05	-0:00:03
SC08-01	5	16:16:25	16:16:40	16:28:35	16:28:50	16:16:44	16:28:57	0:00:19	0:00:22	0:00:04	0:00:07
SC08-01	6	16:50:35	16:50:50	17:03:25	17:03:40	16:50:27	17:03:50	-0:00:08	0:00:25	-0:00:23	0:00:10
SC08-01	10	18:57:36	18:57:51	19:10:11	19:10:26	18:58:00	19:10:25	0:00:24	0:00:14	0:00:09	-0:00:01
SC08-01	13	20:39:31	20:39:46	20:51:51	20:52:06	20:39:48	20:52:05	0:00:17	0:00:14	0:00:02	-0:00:01
SC08-02	2	2:41:32	2:41:47	2:51:37	2:51:52	2:41:55	2:51:37	0:00:23	0:00:00	0:00:08	-0:00:15
SC08-02	3	3:23:21	3:23:36	3:33:06	3:33:21	3:23:45	3:33:07	0:00:24	0:00:01	0:00:09	-0:00:14
SC08-02	4	3:50:16	3:50:31	3:59:56	4:00:11	3:50:36	4:00:01	0:00:20	0:00:05	0:00:05	-0:00:10
SC08-02	5	4:17:01	4:17:16	4:27:06	4:27:21	4:17:27	4:27:11	0:00:26	0:00:05	0:00:11	-0:00:10
SC08-02	6	4:41:41	4:41:56	4:51:41	4:51:56	4:42:04	4:52:01	0:00:23	0:00:20	0:00:08	0:00:05
SC08-03	1	2:08:29	2:08:44	2:16:59	2:17:14	2:08:55	2:17:22	0:00:26	0:00:23	0:00:11	0:00:08
SC08-03	2	2:37:24	2:37:39	2:46:04	2:46:19	2:37:46	2:46:26	0:00:22	0:00:22	0:00:07	0:00:07
SC08-03	4	3:48:42	3:48:57	3:57:17	3:57:33	3:49:04	3:57:38	0:00:22	0:00:21	0:00:07	0:00:05
SC08-03	5	4:13:22	4:13:38	4:21:22	4:21:37	4:13:30	4:21:38	0:00:08	0:00:16	-0:00:08	0:00:01
SC08-03	7	5:01:52	5:02:07	5:10:32	5:10:47	5:02:14	5:10:55	0:00:22	0:00:23	0:00:07	0:00:08
SC08-03	9	6:00:42	6:00:57	6:08:12	6:08:27	6:01:08	6:08:36	0:00:26	0:00:24	0:00:11	0:00:09
SC08-03	12	7:19:27	7:19:42	7:28:27	7:28:42	7:19:56	7:28:47	0:00:29	0:00:20	0:00:14	0:00:05
SC08-03	13	8:02:05	8:02:20	8:09:45	8:10:00	8:02:29	8:10:00	0:00:24	0:00:15	0:00:09	0:00:00
SC08-03	14	12:00:45	12:01:00	12:10:00	12:10:15	12:00:49	12:10:02	0:00:04	0:00:02	-0:00:11	-0:00:13
SC08-03	15	13:13:33	13:13:48	13:23:28	13:23:43	13:13:42	13:23:34	0:00:09	0:00:06	-0:00:06	-0:00:09
SC08-03	16	13:44:38	13:44:53	13:54:38	13:54:53	13:44:51	13:54:43	0:00:13	0:00:05	-0:00:02	-0:00:10
SC08-03	17	14:18:08	14:18:23	14:27:23	14:27:38	14:18:27	14:27:40	0:00:19	0:00:17	0:00:04	0:00:02
SC08-03	18	15:00:21	15:00:36	15:09:21	15:09:36	15:00:41	15:09:49	0:00:20	0:00:28	0:00:05	0:00:13
SC08-03	19	15:30:06	15:30:21	15:39:26	15:39:41	15:30:16	15:39:53	0:00:10	0:00:27	-0:00:05	0:00:12
SC08-03	21	16:51:16	16:51:31	17:00:11	17:00:26	16:51:36	17:00:32	0:00:20	0:00:21	0:00:05	0:00:06
SC08-03	22	17:17:36	17:17:51	17:27:51	17:28:06	17:17:58	17:28:10	0:00:22	0:00:19	0:00:07	0:00:04
SC08-04	2	22:44:17	22:44:32	22:55:02	22:55:17	22:44:26	22:55:04	0:00:09	0:00:02	-0:00:06	-0:00:13
SC08-04	3	23:23:41	23:23:56	23:34:51	23:35:06	23:23:56	23:35:12	0:00:15	0:00:21	0:00:00	0:00:06

SC08-04	5	0:50:31	0:50:46	1:01:56	1:02:11	0:50:48	1:02:08	0:00:17	0:00:12	0:00:02	-0:00:03
SC08-04	7	2:36:21	2:36:36	2:45:31	2:45:46	2:36:44	2:45:46	0:00:23	0:00:15	0:00:08	0:00:00
SC08-04	8	3:10:44	3:10:59	3:19:49	3:20:04	3:10:55	3:20:10	0:00:11	0:00:21	-0:00:04	0:00:06
SC08-04	9	3:43:39	3:43:54	3:52:44	3:52:59	3:43:51	3:53:07	0:00:12	0:00:23	-0:00:03	0:00:08
SC08-04	10	4:13:49	4:14:04	4:22:59	4:23:14	4:13:55	4:23:19	0:00:06	0:00:20	-0:00:09	0:00:05
SC08-04	11	4:50:09	4:50:24	4:59:19	4:59:34	4:50:16	4:59:42	0:00:07	0:00:23	-0:00:08	0:00:08
SC08-04	12	5:23:24	5:23:39	5:32:29	5:32:44	5:23:40	5:32:49	0:00:16	0:00:20	0:00:01	0:00:05
SC08-04	13	6:28:58	6:29:13	6:38:18	6:38:33	6:29:09	6:38:36	0:00:11	0:00:18	-0:00:04	0:00:03
SC08-04	14	7:00:43	7:00:58	7:10:13	7:10:28	7:00:59	7:10:30	0:00:16	0:00:17	0:00:01	0:00:02
SC08-04	15	7:33:53	7:34:07	7:43:08	7:43:23	7:34:05	7:43:30	0:00:12	0:00:22	-0:00:02	0:00:07
SC08-04	16	8:01:03	8:01:18	8:10:08	8:10:23	8:01:09	8:10:28	0:00:06	0:00:20	-0:00:09	0:00:05
SC08-05	1	16:51:06	16:51:21	16:57:06	16:57:31	16:51:28	16:57:20	0:00:22	0:00:14	0:00:07	-0:00:11
SC08-05	2	17:13:26	17:13:41	17:19:31	17:19:46	17:13:34	17:19:51	0:00:08	0:00:20	-0:00:07	0:00:05
SC08-05	3	19:08:53	19:09:08	19:14:28	19:14:43	19:09:20	19:14:43	0:00:27	0:00:15	0:00:12	0:00:00
SC08-05	6	21:04:18	21:04:33	21:10:48	21:11:03	21:04:43	21:11:01	0:00:25	0:00:13	0:00:10	-0:00:02
	304	9:37:14	9:37:29	9:43:54	9:44:09	9:37:34	9:44:10	0:00:20	0:00:16	0:00:05	0:00:01
	303	14:17:59	14:18:14	14:24:14	14:24:29	14:18:14	14:24:37	0:00:15	0:00:23	0:00:00	0:00:08
	36	17:10:13	17:10:28	17:16:43	17:16:58	17:10:32	17:16:57	0:00:19	0:00:14	0:00:04	-0:00:01
	312	18:43:43	18:43:58	18:51:28	18:51:43	18:44:05	18:51:41	0:00:22	0:00:13	0:00:07	-0:00:02
	313	21:46:33	21:46:48	21:54:28	21:54:43	21:46:46	21:54:39	0:00:13	0:00:11	-0:00:02	-0:00:04
	314	0:22:38	0:22:53	0:30:13	0:30:28	0:22:42	0:30:13	0:00:04	0:00:00	-0:00:11	-0:00:15
	316	2:48:28	2:48:43	2:55:08	2:55:33	2:48:28	2:55:48	0:00:00	0:00:40	-0:00:15	0:00:15
							Average Difference:	**0:00:16**	**0:00:16**	**0:00:01**	**0:00:01**

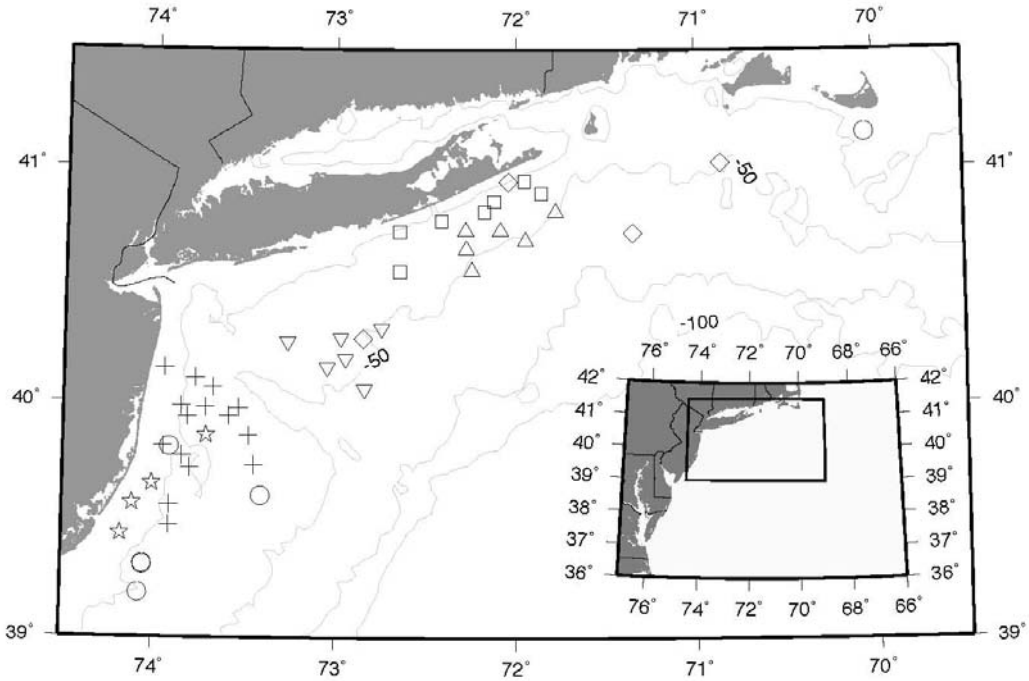


Figure 1. Map of site locations from the 2008 Cooperative Industry Surf Clam/Ocean Quahog Survey. Shapes indicate the type of site. See Table 1 for which tows are represented by which shape.

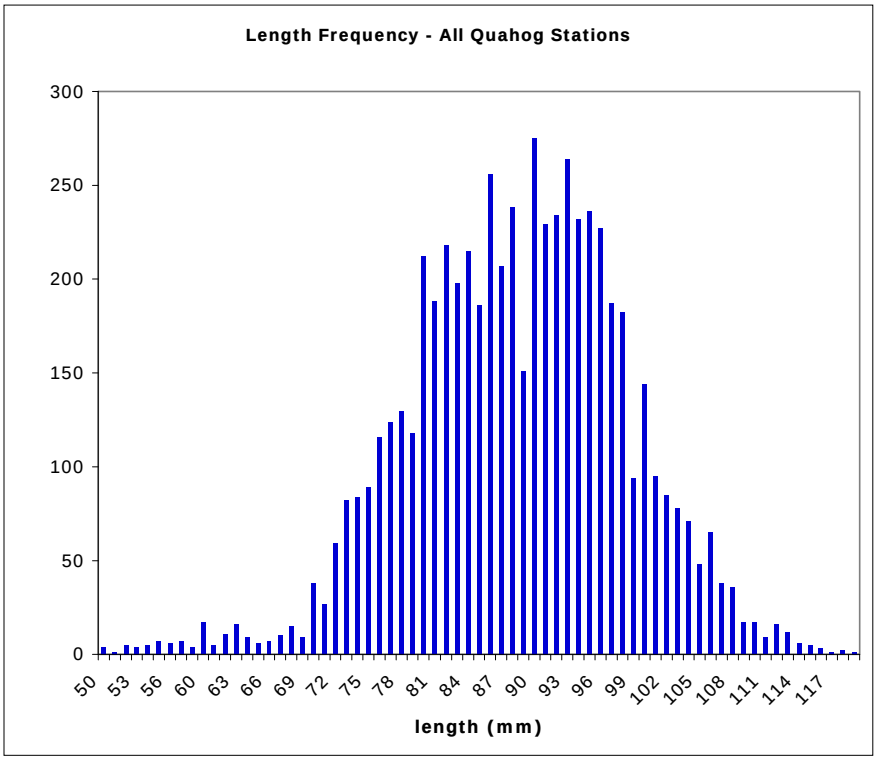


Figure 2. The length frequency of all ocean quahogs measured on 2008 Cooperative Industry Surf Clam/Ocean Quahog Survey

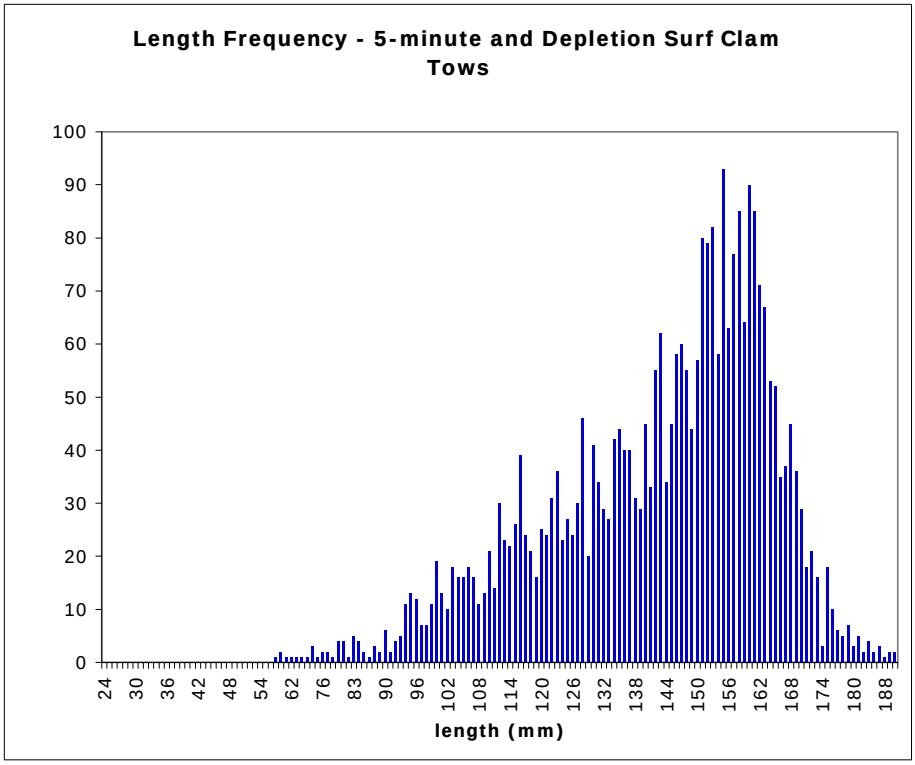


Figure 3. The length frequency of all surf clams measured from 5-minute, unlined tows (size-selectivity experiments and depletion experiments) on 2008 Cooperative Industry Surf Clam/Ocean Quahog Survey

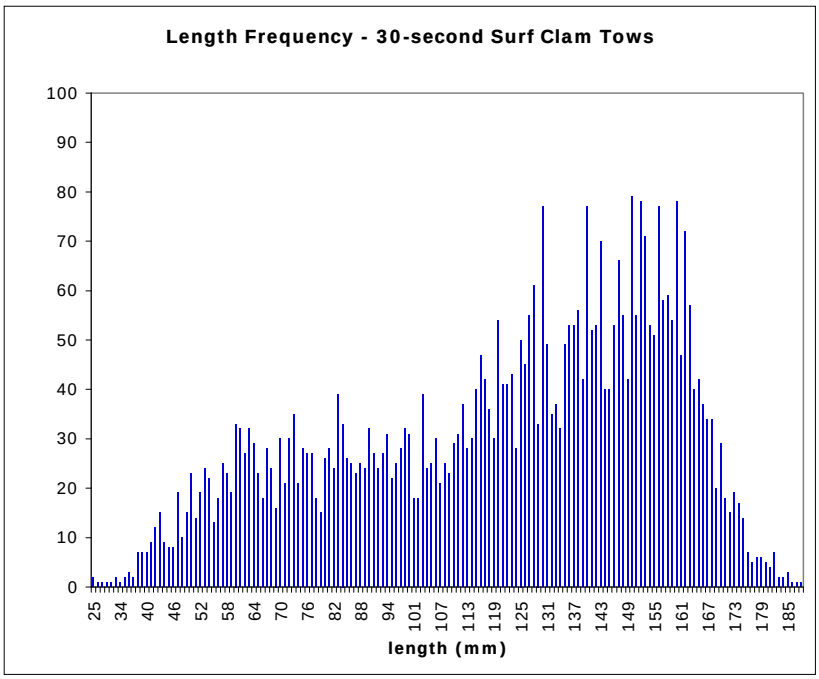


Figure 4. The length frequency of all surf clams measured from 30-second, lined tows on 2008 Cooperative Industry Surf Clam/Ocean Quahog Survey

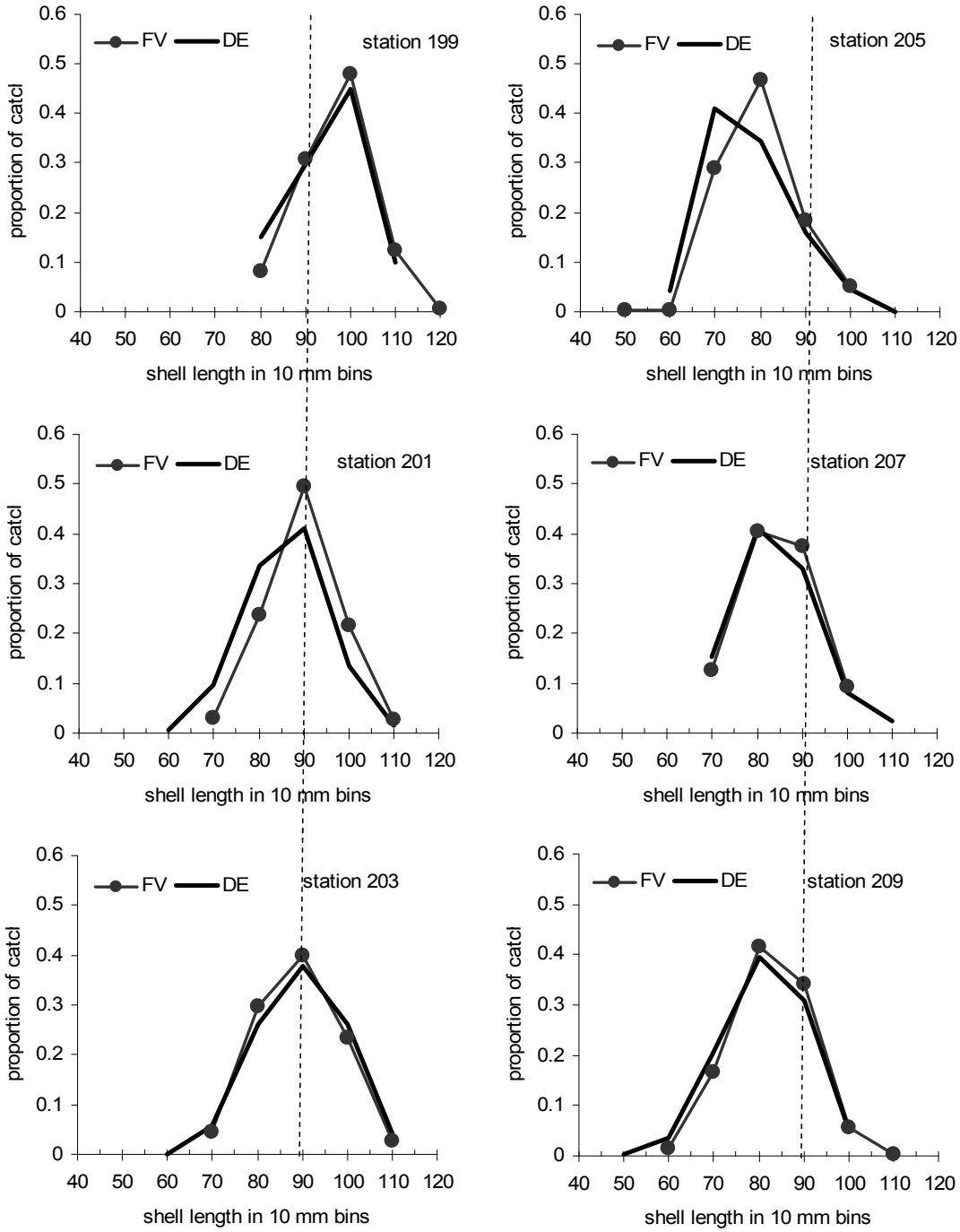


Figure 5. Length composition data for DE2FV repeat tows. For example, 70 mm on the x-axis refers to the 70-79 mm SL bin. Values on the y-axis are proportions of the total.

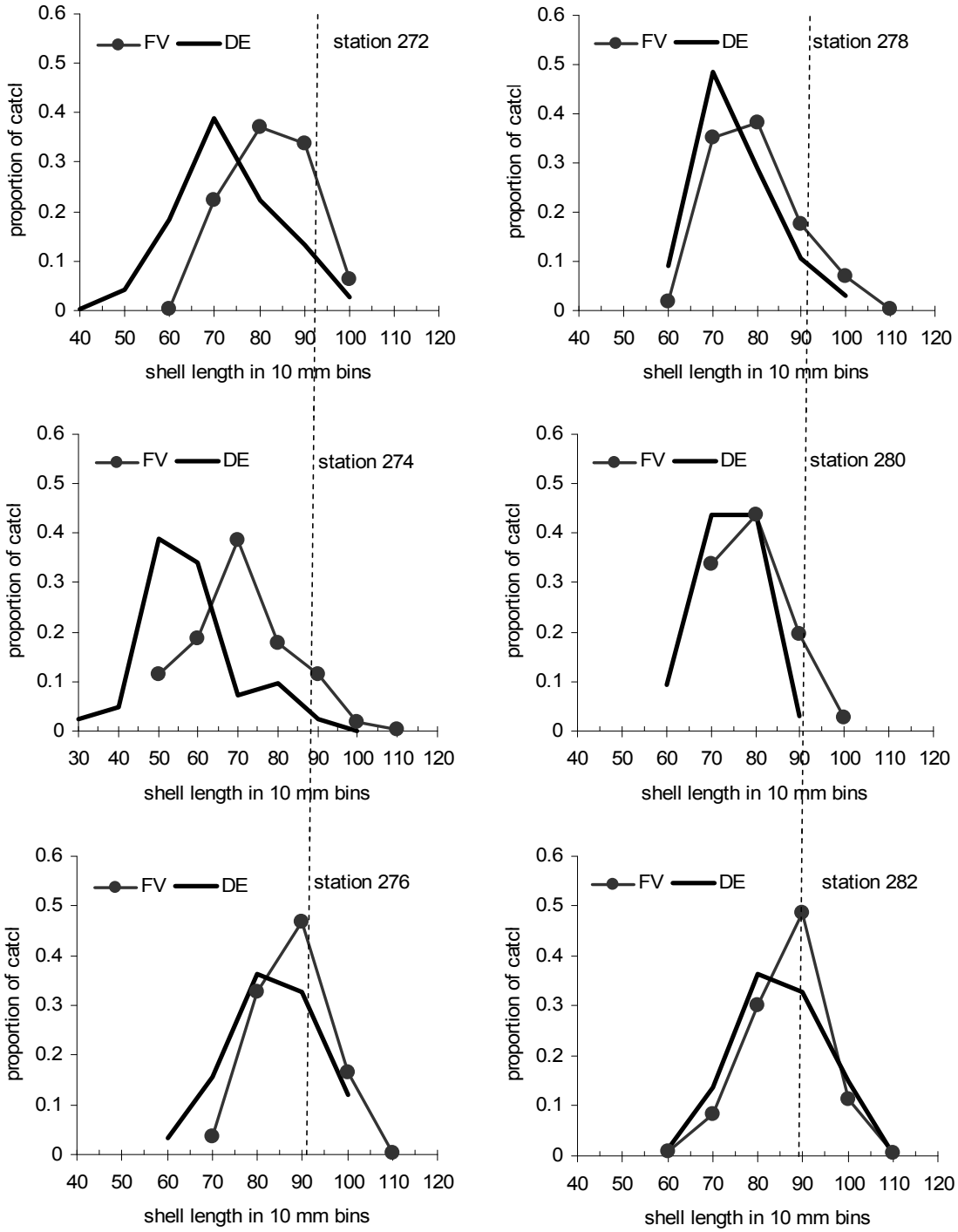
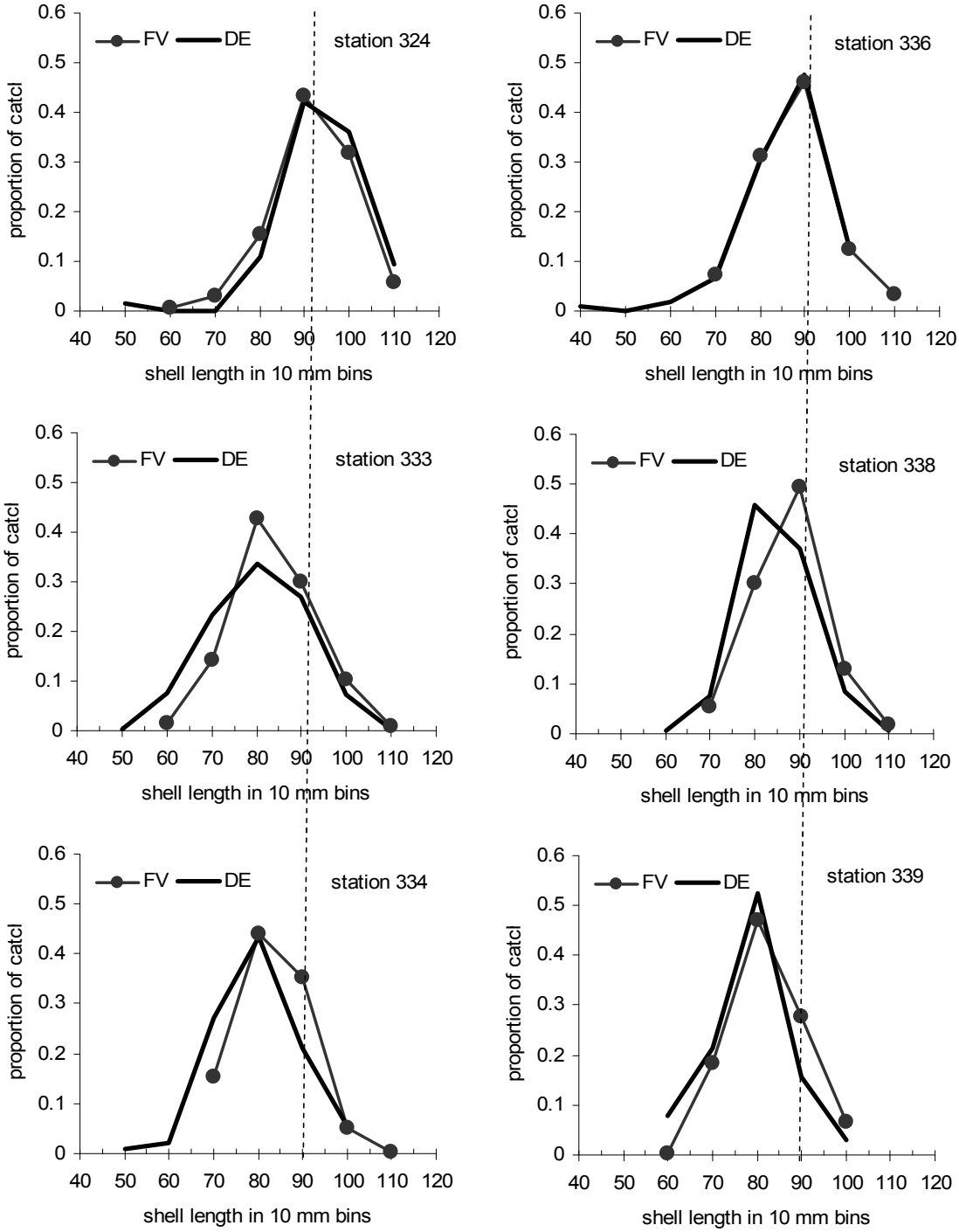
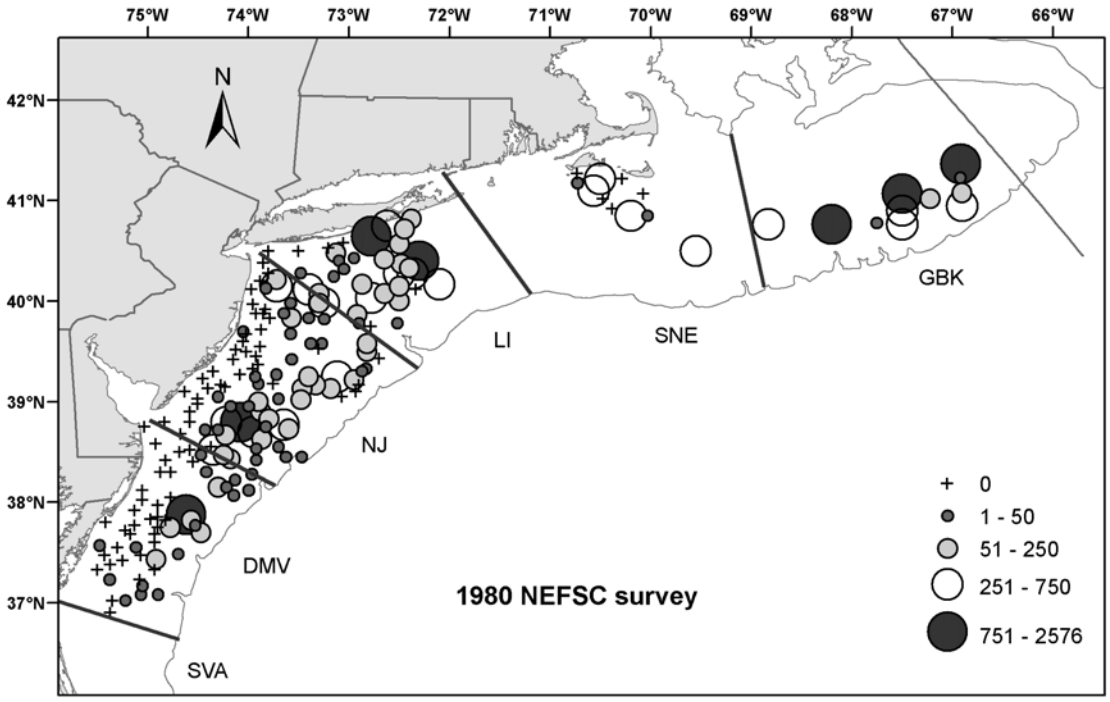
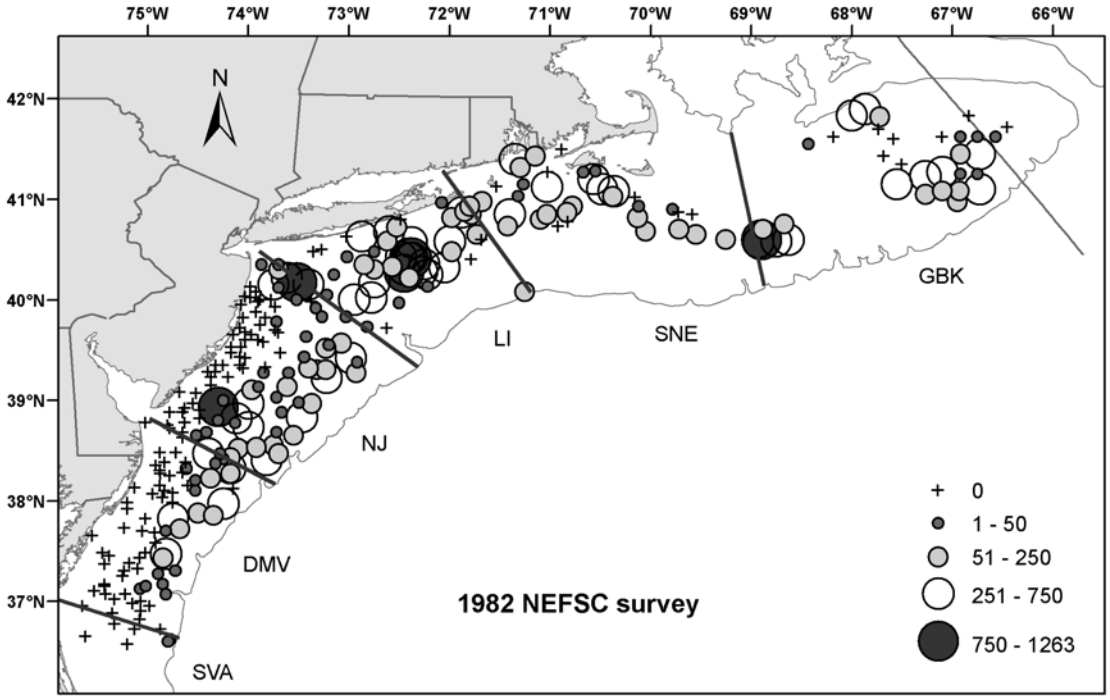


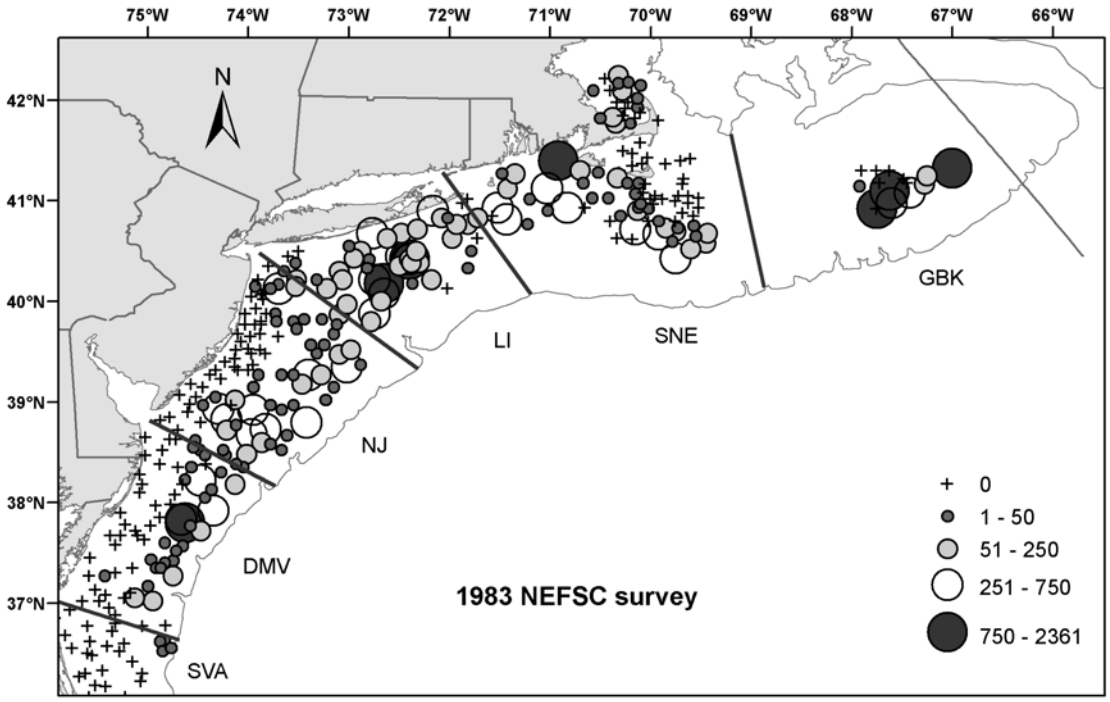
Figure 5. (cont.)

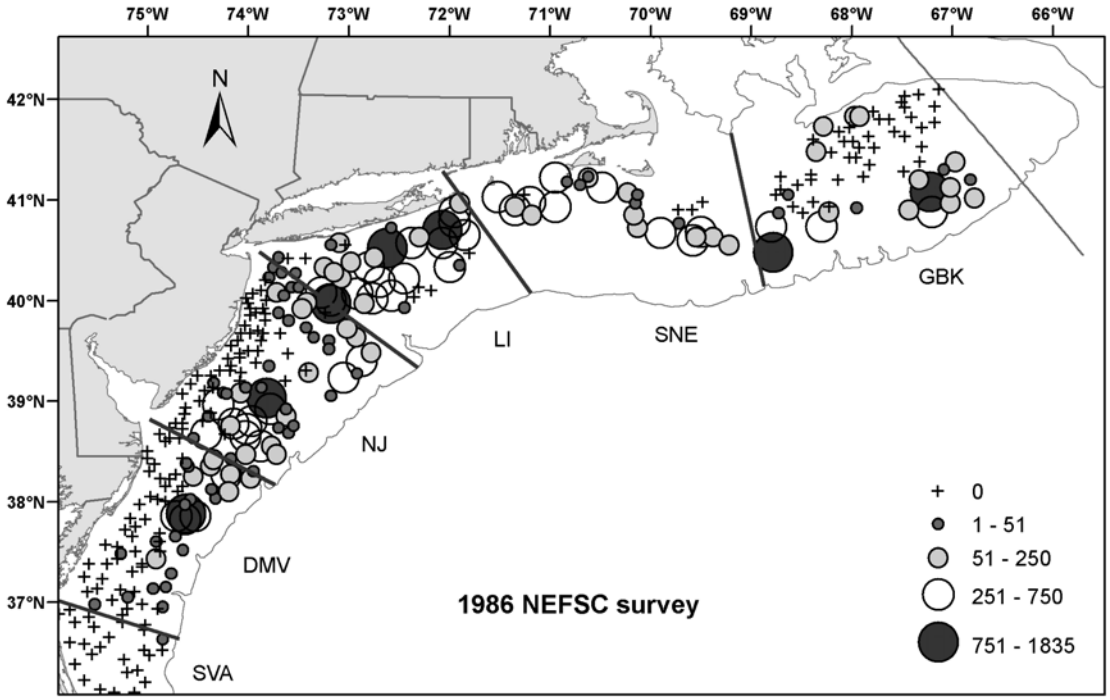

FV
DE
station 324
station 336
station 333
station 338
station 334
station 339
proportion of catcl
shell length in 10 mm bins
0
0.1
0.2
0.3
0.4
0.5
0.6
40
50
60
70
80
90
100
110
120

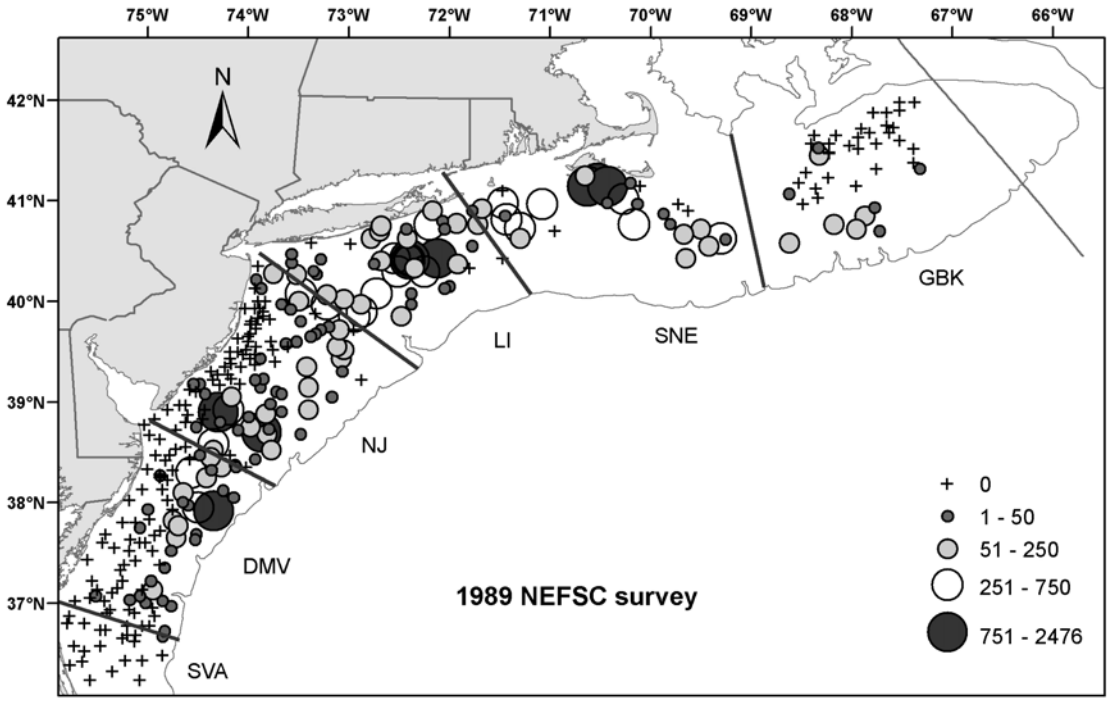
Appendix B5: Maps of NEFSC clam survey catches 1980-2008.

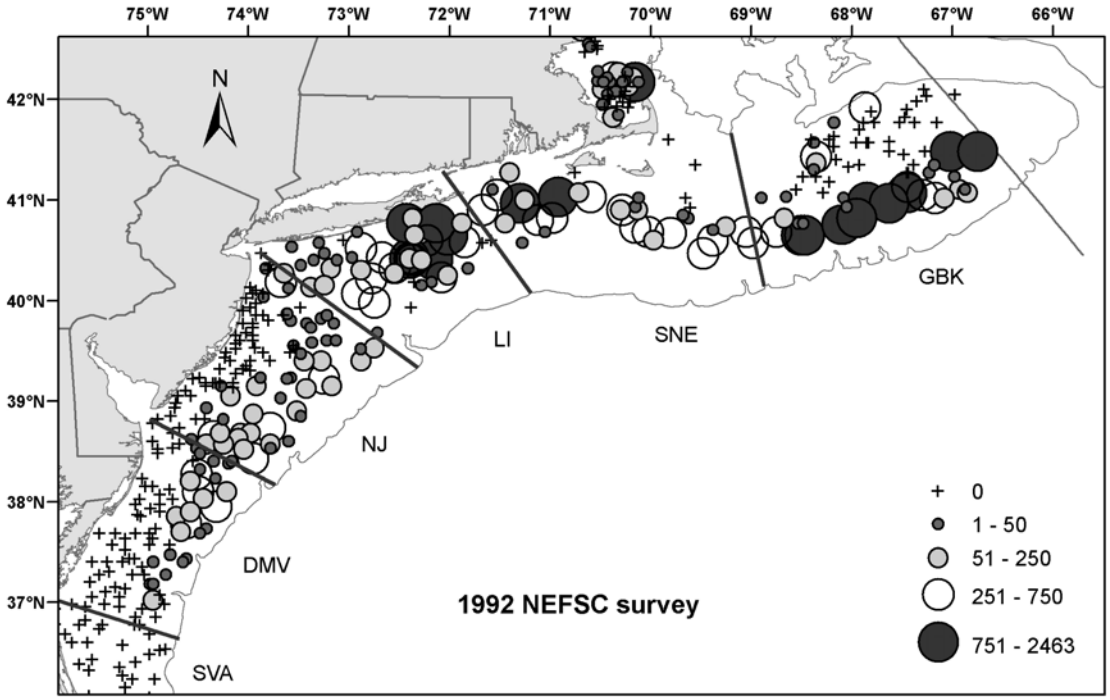


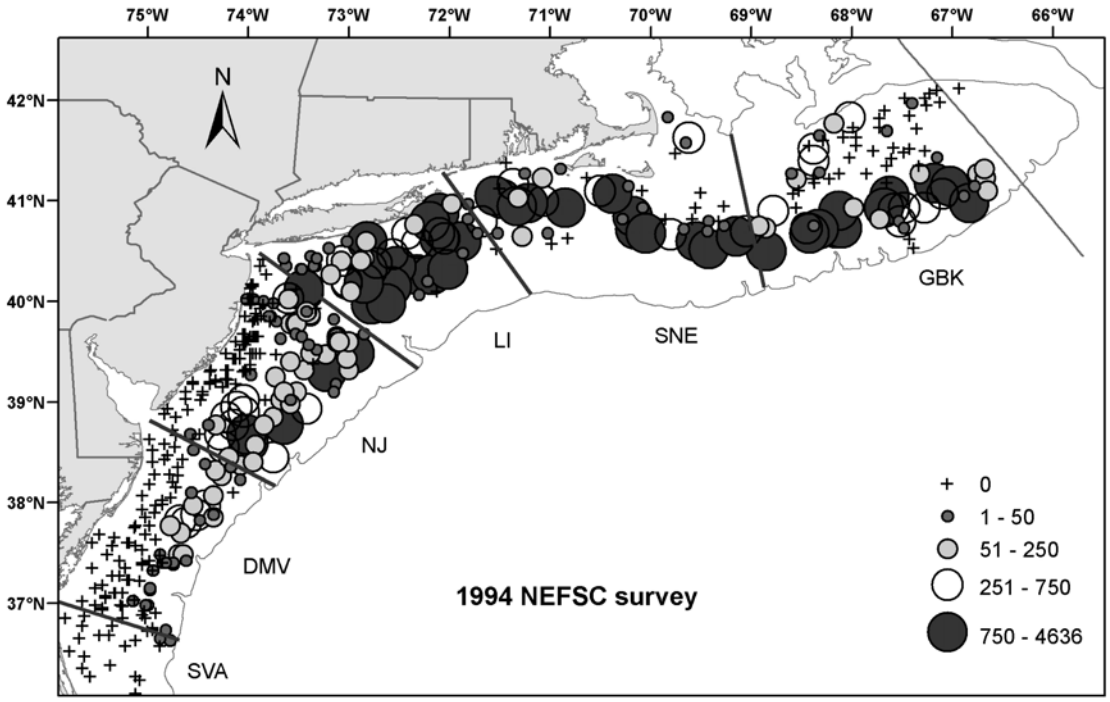


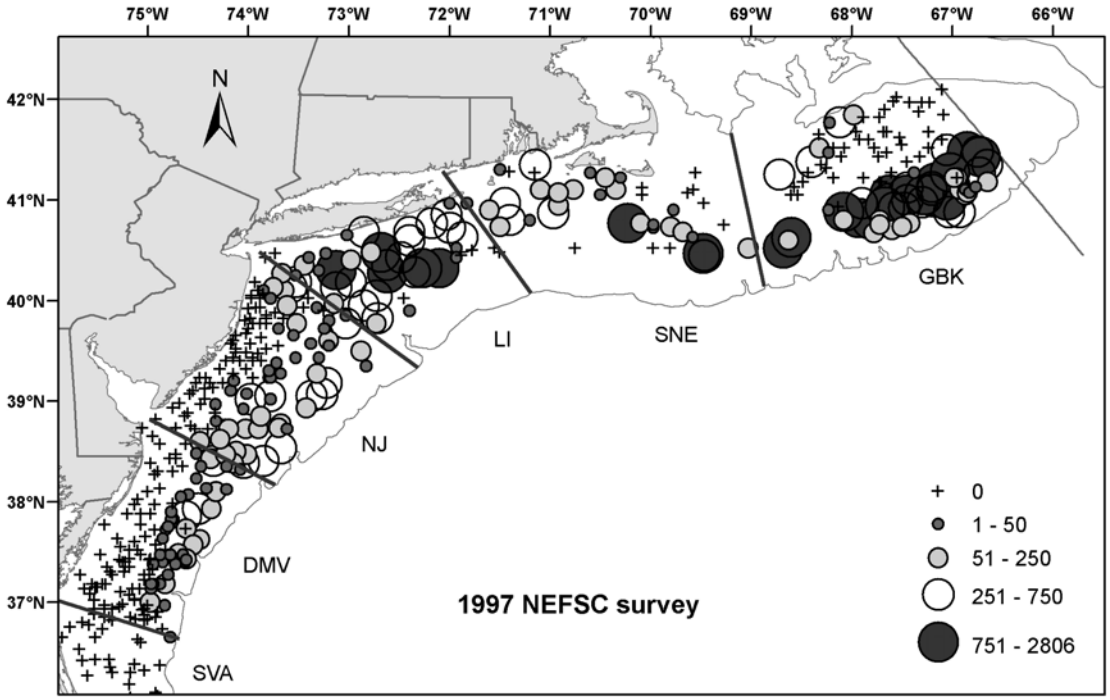

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
1983 NEFSC survey
0
1 - 50
51 - 250
251 - 750
750 - 2361

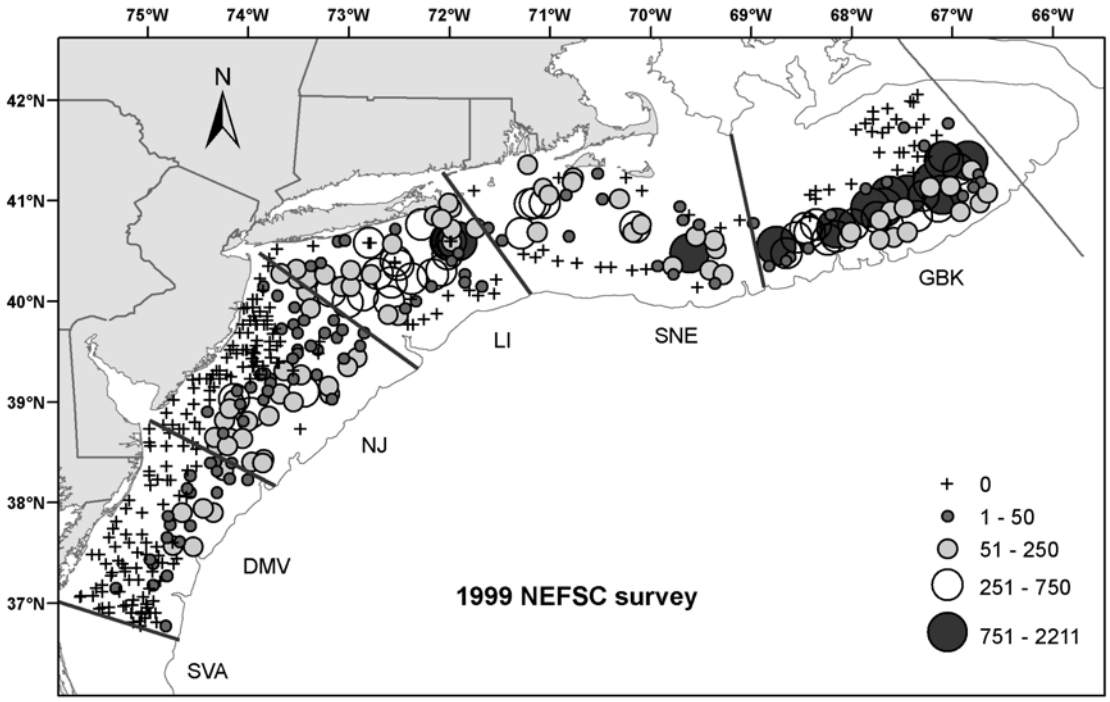

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
1986 NEFSC survey
0
1 - 51
51 - 250
251 - 750
751 - 1835

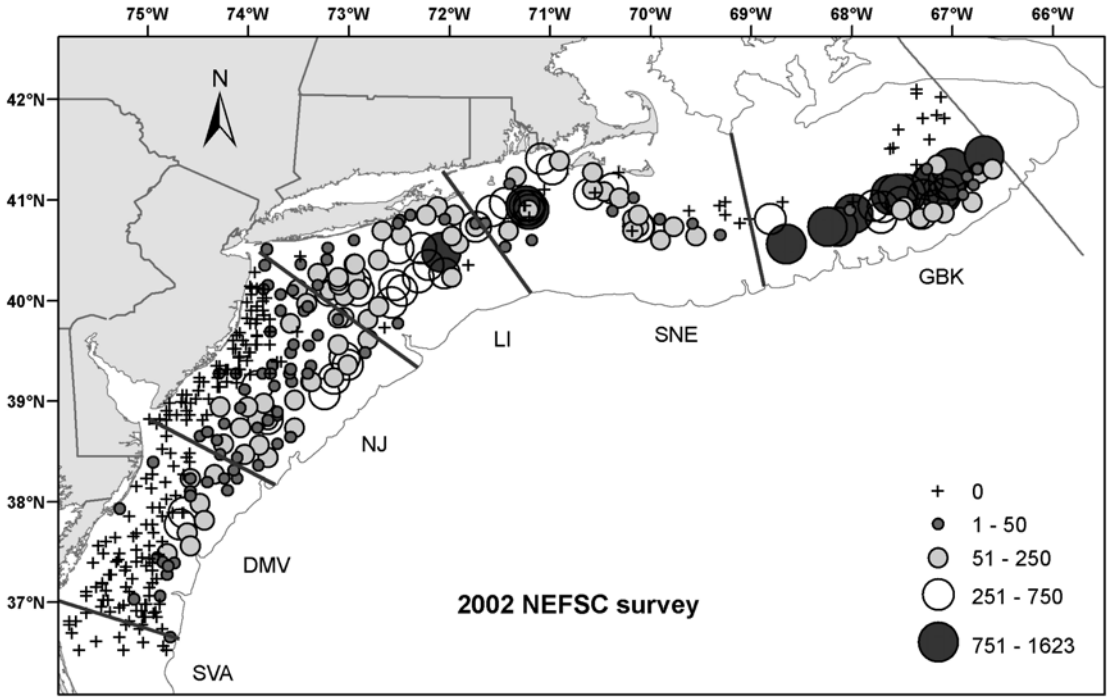

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
1989 NEFSC survey
0
1 - 50
51 - 250
251 - 750
751 - 2476

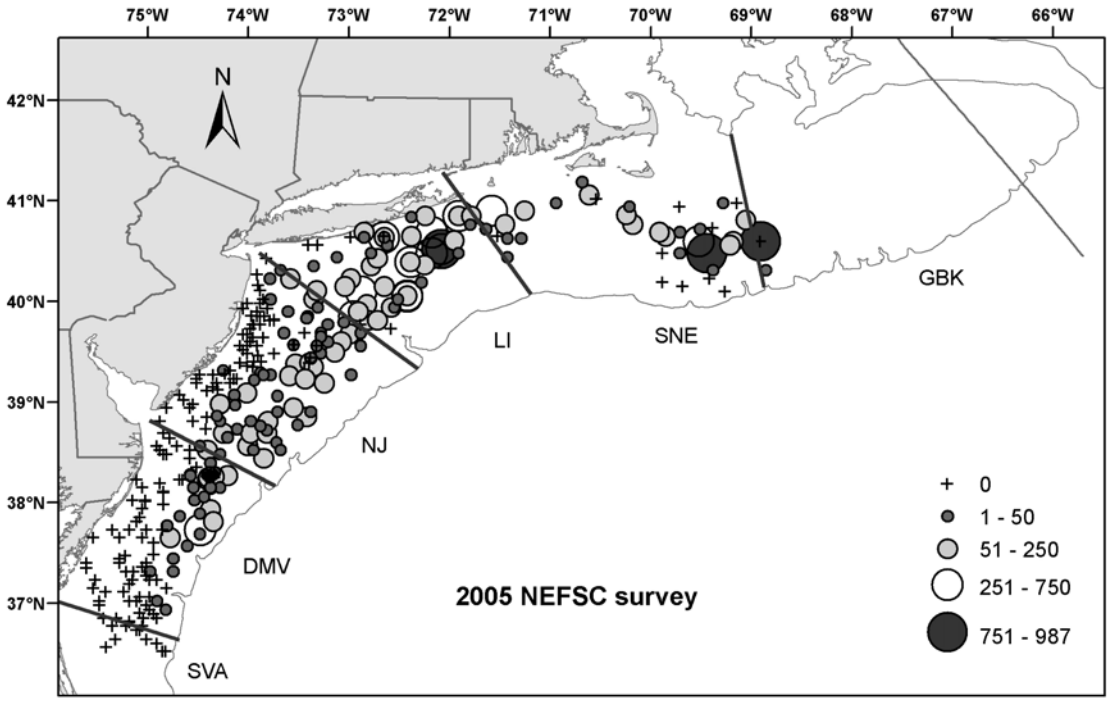

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
1992 NEFSC survey
0
1 - 50
51 - 250
251 - 750
751 - 2463

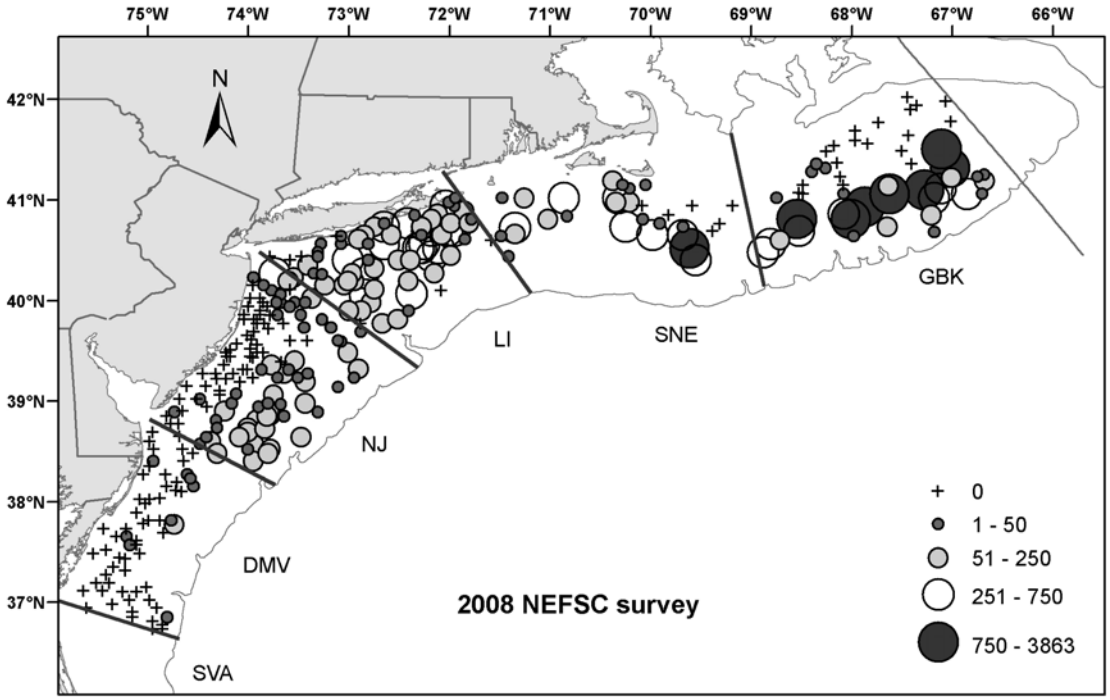

75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
1994 NEFSC survey
0
1 - 50
51 - 250
251 - 750
750 - 4636


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
1997 NEFSC survey
0
1 - 50
51 - 250
251 - 750
751 - 2806


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
1999 NEFSC survey
0
1 - 50
51 - 250
251 - 750
751 - 2211


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
2002 NEFSC survey
0
1 - 50
51 - 250
251 - 750
751 - 1623


75°W
74°W
73°W
72°W
71°W
70°W
69°W
68°W
67°W
66°W
42°N
41°N
40°N
39°N
38°N
37°N
N
GBK
LI
SNE
NJ
DMV
SVA
2005 NEFSC survey
0
1 - 50
51 - 250
251 - 750
751 - 987


75°W
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67°W
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42°N
41°N
40°N
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N
GBK
LI
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NJ
DMV
SVA
2008 NEFSC survey
0
1 - 50
51 - 250
251 - 750
750 - 3863

APPENDIX B6: KLAMZ model details.

KLAMZ Assessment Model – Technical Documentation

The KLAMZ assessment model is based on the Deriso-Schnute delay-difference equation (Deriso 1980; Schnute 1985; Quinn and Deriso 1999). The delay-difference equation is a relatively simple and implicitly age structured approach to counting fish in either numerical or biomass units. It gives the same results as explicitly age-structured models (e.g. Leslie matrix model) if fishery selectivity is "knife-edged", if somatic growth follows the von Bertalanffy equation, and if natural mortality is the same for all age groups in each year. Knife-edge selectivity means that all individuals alive in the model during the same year experience the same fishing mortality rate.[8] Natural and fishing mortality rates, growth parameters and recruitment may change from year to year, but delay-difference calculations assume that all individuals share the same mortality and growth parameters within each year. The KLAMZ model includes simple numerical models (e.g. Conser 1995) as special cases because growth can be turned off so that all calculations are in numerical units (see below).

As in many other simple models, the delay difference equation explicitly distinguishes between two age groups. In KLAMZ, the two age groups are called "new" recruits (R_t in biomass or numerical units at the beginning of year t) and "old" recruits (S_t) that together comprise the whole stock (B_t). New recruits are individuals that recruited at the beginning of the current year (at nominal age k).[9] Old recruits are all older individuals in the stock (nominal ages k+1 and older, survivors from the previous year). As described above, KLAMZ assumes that new and old recruits are fully vulnerable to the fishery. The most important differences between the delay-difference and other simple models (e.g. Prager 1994; Conser 1995; Jacobson et al. 1994) are that von Bertalanffy growth is used to calculate biomass dynamics and that the delay-difference model captures transient age structure effects due to variation in recruitment, growth and mortality exactly. Transient effects on population dynamics are captured exactly because, as described above, the delay-difference equation is algebraically equivalent to an explicitly age-structured model with von Bertalanffy growth.

The KLAMZ model incorporates a few extensions to Schnute's (1985) revision of Deriso's (1980) original delay difference model. Most of the extensions facilitate tuning to a wider variety of data that anticipated in Schnute (1985). The KLAMZ model is programmed in both Excel and in C++ using AD Model Builder[10] libraries. The AD Model Builder version is faster, more reliable

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8 In applications, assumptions about knife-edge selectivity can be relaxed by assuming the model tracks "fishable", rather that total, biomass (NEFSC 2000a; 2000b). An analogous approach assigns pseudo-ages based on recruitment to the fishery so that new recruits in the model are all pseudo-age k. The synthetic cohort of fish pseudo-age k may consist of more than one biological cohort. The first pseudo-age (k) can be the predicted age at first, 50% or full recruitment based a von Bertalanffy curve and size composition data (Butler et al. 2002). The "incomplete recruitment" approach (Deriso 1980) calculates recruitment to the model in each year R_t as the weighted sum of contributions from two or more biological cohorts (year-classes) from spawning during successive years (i.e. $R_t = \sum_{a=1}^{k} r_a \Pi_{t-a}$ where k is the age at full recruitment to the fishery, r_a is the contribution of fish age k-a to the fishable stock, and Π_{t-a} is the number or biomass of fish age k-a during year t).

9 In some applications, and more generally, new recruits might be defined as individuals recruiting at the beginning or at any time during the current time step (e.g. NEFSC 1996).

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and probably better for producing "official" stock assessment results. The Excel version is slower and implements fewer features, but the Excel version remains useful in developing prototype assessment models, teaching and for checking calculations.

The most significant disadvantage in using the KLAMZ model and other delay-difference approaches, beyond the assumption of knife-edge selectivity, is that age and length composition data are not used in tuning. However, one can argue that age composition data are used indirectly to the extent they are used to estimate growth parameters or if survey survival ratios (e.g. based on the Heinke method) are used in tuning (see below).

Population dynamics

The assumed birth date and first day of the year are assumed the same in derivation of the delay-difference equation. It is therefore natural (but not strictly necessary) to tabulate catch and other data using annual accounting periods that start on the assumed biological birthday of cohorts.

Biomass dynamics

As implemented in the KLAMZ model, Schnute's (1985) delay-difference equation is:

$$B_{t+1} = (1+\rho)\,\tau_t\,B_t - \rho\,\tau_t\,\tau_{t-1}\,B_{t-1} + R_{t+1} - \rho\,\tau_t\,J_t\,R_t$$

where B_t is total biomass of individuals at the beginning of year t; ρ is Ford's growth coefficient (see below); $\tau_t=exp(-Z_t)=exp[-(F_t+M_t)]$ is the fraction of the stock that survived in year t, Z_t, F_t, and M_t are instantaneous rates for total, fishing and natural mortality; and R_t is the biomass of new recruits (at age k) at the beginning of the year. The natural mortality rate M_t may vary over time. Instantaneous mortality rates in KLAMZ model calculations are biomass-weighted averages if von Bertalanffy growth is turned on in the model. However, biomass-weighted mortality estimates in KLAMZ are the same as rates for numerical estimates under the assumption of knife-edge selectivity because all individuals are fully recruited. The growth parameter $J_t = w_{t-1,k-1} / w_{t,k}$ is the ratio of mean weight one year before recruitment (age k-1 in year t-1) and mean weight at recruitment (age k in year t).

It is not necessary to specify body weights at and prior to recruitment in the KLAMZ model (parameters v_{t-1} and V_t in Schnute 1985) because the ratio J_t and recruitment biomass contain the same information. Schnute's (1985) original delay difference equation is:

$$B_{t+1} = (1+\rho)\,\tau_t\,B_t - \rho\,\tau_t\tau_{t-1}\,B_{t-1} + w_{t+1,k}N_{t+1} - \rho\tau_t w_{t-1,k-1}\,N_t$$

To derive the equation used in KLAMZ, substitute recruitment biomass R_{t+1} for the product $w_{t+1,k}$ $N_{t+1,k}$ and adjusted recruitment biomass $J_t R_t = (w_{t-1,k-1}/w_{t,k})\, w_{t,k}\, N_{t,k} =$ $w_{t-1,k-1}\ N_t$ in the last term on the right hand side. The advantage in using the alternate parameterization for biomass dynamic calculations in KLAMZ is that recruitment is estimated directly in units of biomass and the number of growth parameters is reduced. The disadvantage is that numbers of recruits are not estimated directly by the model. When required, numerical recruitments must be calculated externally as the ratio of estimated recruitment biomass and the average body weight for new recruits.

Numerical population dynamics

Growth can be turned on off so that abundance, rather than biomass, is tracked in the KLAMZ model. Set J_t=1 and ρ=0 in the delay difference equation, and use N_t (for numbers) in place of B_t to get:

$$N_{t+1} = \tau_t\,N_t + R_{t+1}$$

Mathematically, the assumption J_t=1 means that no growth occurs the assumption ρ=0 means that the von Bertalanffy K parameter is infinitely large (Schnute 1985). All tuning and population dynamics calculations in KLAMZ for biomass dynamics are also valid for numerical dynamics.

Growth

As described in Schnute (1985), biomass calculations in the KLAMZ model are based on Schnute and Fournier's (1980) re-parameterization of the von Bertalanffy growth model:

$$w_a = w_{k-1} + (w_k - w_{k-1})(1+\rho^{1+a-k}) / (1-\rho)$$

where w_k=V and w_{k-1}=v. Schnute and Fournier's (1980) growth model is the same as the traditional von Bertalanffy growth model {$W_a = W_{max}\ [1 - exp(-K(a-t_{zero})]$ where W_{max}, K and t_{zero} are parameters}. The two growth models are the same because $W_{max} = (w_k - \rho\, w_{k-1})/(1-\rho)$, $K = -ln(\rho)$ and $t_{zero} = ln[(w_k - w_{k-1})/(w_k - \rho\, w_{k-1})] / ln(\rho)$.

In the KLAMZ model, the growth parameters J_t can vary with time but ρ is constant. Use of time-variable J_t values with ρ is constant is the same as assuming that the von Bertalanffy parameters W_{max} and t_{zero} change over time. Many growth patterns can be mimicked by changing W_{max} and t_{zero} (Overholtz et al., 2003). K is a parameter in the C++ version and, in principal, estimable. However, in most cases it is necessary to use external estimates of growth parameters as constants in KLAMZ.

Instantaneous growth rates

Instantaneous growth rate (IGR) calculations in the KLAMZ model are an extension to the original Deriso-Schnute delay difference model. IGRs are used extensively in KLAMZ for calculating catch biomass and projecting stock biomass forward to the time at which surveys occur. The IGR for new recruits depends only on growth parameters:

$$G_t^{New} = \ln\left(\frac{w_{k+1,t+1}}{w_{k,t}}\right) = \ln(1+\rho-\rho J_t)$$

IGR for old recruits is a biomass-weighted average that depends on the current age structure and growth parameters. It can be calculated easily by projecting biomass of old recruits S_t=B_t-R_t (escapement) forward one year with no mortality:

$$S_t^* = (1+\rho)S_t - \rho\tau_{t-1}B_{t-1}$$

where the asterisk (*) means just prior to the start of the subsequent year t+1. By definition, the IGR for old recruits in year t is $G_t^{Old} = \ln\left(S_t^*/S_t\right)$. Dividing by S_t gives:

$$G_t^{Old} = \ln\left[(1+\rho) - \rho\tau_{t-1}\frac{B_{t-1}}{S_t}\right]$$

IGR for the entire stock is the biomass weighted average of the IGR values for new and old recruits:

$$G_t = \frac{R_t G_t^{New} + S_t G_t^{Old}}{B_t}$$

All IGR values are zero if growth is turned off.

Recruitment

In the Excel version of the KLAMZ model, annual recruitments are calculated

$R_t = e^{\Omega_t}$ where Ω_t is a log transformed annual recruitment parameter, which is estimated in the model. In the C++ version, recruitments are calculated based on two log geometric mean recruitment parameters (μ, ι_t), and a set of annual log scale deviation parameters (ω_t):

$$\Omega_t = \mu + \iota_t + \omega_t$$

The parameter ι_t is an offset for a step function that may be zero for all years or zero for years up to a user-specified "change year" and any value (usually estimated) afterward. The user must specify the change year, which cannot be estimated. The change year might be chosen based on auxiliary information outside the model, preliminary model fits or by carrying out a set of runs using sequential change year values and to choosing the change year that provides the best fit to the data.

The deviations ω_t are constrained to average zero.[11] With the constraint, for example, estimation of μ and the set of ω_t values (1+ n years parameters) is equivalent to estimation of the smaller set (n years) of Ω_t values.

Natural mortality

Natural mortality rates (M_t) are assumed constant in the Excel version of the KLAMZ model. In the C++ version, natural mortality rates may be estimated as a constant value or as a set of values that vary with time. In the model:

$$M_t = me^{\varpi_t}$$

where $m=exp(\pi)$ is the geometric mean natural mortality rate, π is a model parameter that may be estimated (in principal but not in practical terms), and ϖ_t is the log scale year-specific deviation. Deviations may be zero (turned off) so that M_t is constant, may vary in a random fashion due to autocorrelated or independent process errors, or may based on a covariate.[12] Model scenarios with zero recruitment may be initializing the parameter π to a small value (e.g. 10^{-16}) and not estimating it.

Random natural mortality process errors are effects due to predation, disease, parasitism, ocean conditions or other factors that may vary over time but are not included in the model. Calculations are basically the same as for survey process errors (see below).

Natural mortality rate covariate calculations are similar to survey covariate calculations (see below) except that the <u>*user should standardized covariates to average zero over the time period included in the model*</u>:

$$\kappa_t = K_t - \overline{K}$$

where κ_t is the standardized covariate, K_t is the original value, and $\overline{K}$ is the mean of the original covariate for the years in the model. <u>*Standardization to mean zero is important because otherwise m is not the geometric mean natural mortality rate*</u> (the convention is important in some calculations, see text).

Log scale deviations that represent variability around the geometric mean are calculated:

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11 The constraint is implemented by adding $L = \lambda \overline{\varpi}^2$ (where $\overline{\varpi}$ is the average deviation) to the objective function, generally with a high weighting factor (λ = 1000) so that the constraint is binding.

12 Another approach to using time dependent natural mortality rates is to treat estimates of predator consumption as discarded catch (see "Predator consumption as discard data"). In addition, estimates of predator abundance can be used in fishing effort calculations (see "Predator data as fishing effort").

$$\varpi_t = \sum_{j=1}^{n} p_j \kappa_t$$

where n is the number of covariates and p_j is the parameter for covariate j. These conventions mean that the units for the covariate parameter p_j are 1/units of the original covariate, the parameter p_j measures the log scale effect of changing the covariate by one unit, and the parameter m is the log scale geometric mean.

Fishing mortality and catch

Fishing mortality rates (F_t) are calculated so that predicted and observed catch data (landings plus estimated discards in units of weight) "agree" to the extent specified by the user. It is not necessary, however, to assume that catches are measured accurately (see "Observed and predicted catch").

Fishing mortality rate calculations in Schnute (1985) are exact but relating fishing mortality to catch in weight is complicated by continuous somatic growth throughout the year as fishing occurs. The KLAMZ model uses a generalized catch equation that incorporates continuous growth through the fishing season. By the definition of instantaneous rates, the catch equation expresses catch as the product:

$$\hat{C}_t = F_t \overline{B}_t$$

where $\hat{C}_t$ is predicted catch weight (landings plus discard) and $\overline{B}_t$ is average biomass.

Following Chapman (1971) and Zhang and Sullivan (1988), let $X_t = G_t - F_t - M_t$ be the net instantaneous rate of change for biomass.[13] If the rates for growth and mortality are equal, then $X_t = 0$, $\overline{B}_t = B_t$ and $C_t = F_t B_t$. If the growth rate G_t exceeds the combined rates of natural and fishing mortality ($F_t + M_t$), then $X_t > 0$. If mortality exceeds growth, then $X_t < 0$. In either case, with $X_t \neq 0$, average biomass is computed:

$$\overline{B}_t \approx -\frac{\left(1 - e^{X_t}\right) B_t}{X_t}$$

When $X_t \neq 0$, the expression for $\overline{B}_t$ is an approximation because G_t approximates the rate of change in mean body weight due to von Bertalanffy growth. However, the approximation is reasonably accurate and preferable to calculating catch biomass in the delay-difference model with the traditional catch equation that ignores growth during the fishing season.[14] Average biomass can be calculated for new recruits, old recruits or for the whole stock by using either G_t^{New}, G_t^{Old} or G_t.

In the KLAMZ model, the modified catch equation may be solved analytically for F_t given C_t, B_t, G_t and M_t (see the "Calculating F_t" section below). Alternatively, fishing mortality rates can be calculated using a log geometric mean parameter (Φ) and a set of annual log scale deviation parameters (ψ_t):

$$F_t = e^{\Phi + \psi_t}$$

where the deviations ψ_t are constrained to average zero. When the catch equation is solved

13 By convention, the instantaneous rates G_t, F_t and M_t are always expressed as numbers ≥ 0.

14 The traditional catch equation $C_t = F_t(1 - e^{-Z_t}) B_t / Z_t$ where $Z_t = F_t + M_t$ underestimates catch biomass for a given level of fishing mortality F_t and overestimates F_t for a given level of catch biomass. The errors can be substantial for fast growing fish, particularly if recent recruitments were strong.

analytically, catches must be assumed known without error but the analytical option is useful when catch is zero or very near zero, or the range of fishing mortality rates is so large (e.g. minimum F=0.000001 to maximum F=3) that numerical problems occur with the alternative approach. The analytical approach is also useful if the user wants to reduce the number of parameters estimated by nonlinear optimization. In any case, the two methods should give the same results for catches known without error.

Surplus production

Annual surplus production is calculated "exactly" by projecting biomass at the beginning of each year forward with no fishing mortality:

$$B^*_t = (1+\rho)\, e^{-M}\, B_t - \rho\, e^{-M}\, L_{t-1}\, B_{t-1} - \rho\, e^{-M}\, J_t\, R_t$$

By definition, surplus production $P_t = B^*_t - B_t$ (Jacobson et al. 2002).

Per recruit modeling

Per recruit model calculations in the Excel version of the KLAMZ simulate the life of a hypothetical cohort of arbitrary size (e.g. R=1000) starting at age k with constant M_t, F (survival) and growth (ρ and J) in a population initially at zero biomass. In the first year:

$$B_1 = R$$

In the second year:

$$B_2 = (1+\rho)\,\tau\, B_1 - \rho\,\tau\, J\, R_1$$

In the third and subsequent years:

$$B_{t+1} = (1+\rho)\,\tau\, B_t - \rho\,\tau^2\, B_{t-1}$$

This iterative calculation is carried out until the sum of lifetime cohort biomass from one iteration to the next changes by less than a small amount (0.0001). Total lifetime biomass, spawning biomass and yield in weight are calculated by summing biomass, spawning biomass and yield over the lifetime of the cohort. Lifetime biomass, spawning biomass and yield per recruit are calculated by dividing totals by initial recruitment (R).

Status determination variables

The user may specify a range of years (e.g. the last three years) to use in calculating recent average fishing mortality $\overline{F}_{Recent}$ and biomass $\overline{B}_{Recent}$ levels. These status determination variables are used in calculation of status ratios such as $\overline{F}_{Recent} / F_{MSY}$ and $\overline{B}_{Recent} / B_{MSY}$.

Goodness of Fit and Parameter Estimation

Parameters estimated in the KLAMZ model are chosen to minimize an objective function based on a sum of weighted negative log likelihood (NLL) components:

$$\Xi = \sum_{v=1}^{N_\Xi} \lambda_v L_v$$

where N_Ξ is the number of NLL components (L_v) and the λ_v are emphasis factors used as weights.

The objective function Ξ may be viewed as a NLL or a negative log posterior (NLP) distribution, depending on the nature of the individual L_v components and modeling approach. Except during sensitivity analyses, weighting factors for objective function components (λ_v) are usually set to one. An arbitrarily large weighting factor (e.g. λ_v =1000) is used for "hard" constraints that must be satisfied in the model. Arbitrarily small weighting factors (e.g. λ_v =0.0001) can be used for "soft" model-based constraints. For example, an internally estimated spawner-recruit curve or surplus production curve might be estimated with a small weighting factor to summarize stock-recruit or surplus production results with minimal influence on biomass, fishing mortality and other estimates from the model. Use of a small weighting factor for an internally estimated surplus production or stock-recruit curve is equivalent to fitting a curve to model estimates of biomass and recruitment or surplus production in the output file, after the model is fit (Jacobson et al. 2002).

Likelihood component weights vs. observation-specific weights

Likelihood component weights (λ_v) apply to entire NLL components. Entire components are often computed as the sum of a number of individual NLL terms. The NLL for an entire survey, for example, is composed of NLL terms for each of the annual survey observations. In KLAMZ, observation-specific (for data) or instance-specific (for constraints or prior information) weights (usually w_j for observation or instance j) can be specified as well. Observation-specific weights for a survey, for example, might be use to increase or decrease the importance of one or more observations in calculating goodness of fit.

NLL kernels

NLL components in KLAMZ are generally programmed as "concentrated likelihoods" to avoid calculation of values that do not affect derivatives of the objective function.[15] For $x \sim N(\mu, \sigma^2)$, the complete NLL for one observation is:

$$L = \ln(\sigma) + \ln\left(\sqrt{2\pi}\right) + 0.5\left(\frac{x-u}{\sigma}\right)^2$$

The constant $\ln\left(\sqrt{2\pi}\right)$ can always be omitted because it does not affect derivatives. If the standard deviation is known or assumed known, then $\ln(\sigma)$ can be omitted as well because it is a constant that does not affect derivatives. In such cases, the concentrated negative log likelihood is:

$$L = 0.5\left(\frac{x-\mu}{\sigma}\right)^2$$

If there are N observations with possible different variances (known or assumed known) and possibly different expected values:

$$L = 0.5\sum_{i=1}^{N}\left(\frac{x_i-\mu_i}{\sigma_i}\right)^2$$

If the standard deviation for a normally distributed quantity is not known and is (in effect) estimated by the model, then one of two equivalent calculations is used. Both approaches assume

15 Unfortunately, concentrated likelihood calculations cannot be used with MCMC and other Bayesian approaches to characterizing posterior distributions. Therefore, in the near future, concentrated NLL calculations will be replaced by calculations for the entire NLL. At present, MCMC calculations in KLAMZ are not useful.

that all observations have the same variance and standard deviation. The first approach is used when all observations have the same weight in the likelihood:

$$L = 0.5N \ln\left[\sum_{i=1}^{N}(x_i - u)^2\right]$$

where N is the number of observations. The second approach is equivalent but used when the weights for each observation (w_i) may differ:

$$L = \sum_{i=1}^{N} w_i \left[\ln(\sigma) + 0.5\left(\frac{x_i - u}{\sigma}\right)^2\right]$$

In the latter case, the maximum likelihood estimator:

$$\hat{\sigma} = \sqrt{\frac{\sum_{i=1}^{N}(x_i - \hat{x})^2}{N}}$$

(where $\hat{x}$ is the average or predicted value from the model) is used for σ. The maximum likelihood estimator is biased by $N/(N-d_f)$ where d_f is degrees of freedom for the model. The bias may be significant for small sample sizes but d_f is usually unknown.

Landings, discards, catch

Discards are from external estimates (d_t) supplied by the user. If $d_t \geq 0$, then the data are used as the ratio of discard to landed catch so that:

$$D_t = L_t \Delta_t$$

where $\Delta_t = D_t/L_t$ is the discard ratio. If $d_t < 0$ then the data are treated as discard in units of weight: $D_t = abs(d_t)$.

In either case, total catch is the sum of discards and landed catch ($C_t = L_t + D_t$). It is possible to use discards in weight $d_t < 0$ for some years and discard as proportions $d_t > 0$ for other years in the same model run. If catches are estimated (see below) so that the estimated catch $\hat{C}_t$ does not necessarily equal observed landings plus discard, then estimated landings are computed:

$$\hat{L}_t = \frac{\hat{C}_t}{1 + \Delta_t}$$

and estimated discards are: $\hat{D}_t = \Delta_t \hat{L}_t$.

Calculating F_t

As described above, fishing mortality rates may be estimated based on the parameters Φ and ψ_t to satisfy a NLL for observed and predicted catches:

$$L = 0.5\sum_{t=0}^{N} w_t \left(\frac{\hat{C}_t - C_t}{\kappa_t}\right)^2$$

where the standard error $\kappa_t = CV_{catch}\hat{C}_t$ with CV_{catch} and weights are w_t supplied by the user. The weights can be used, for example, if catch data in some years are less precise than in others. Using observation specific weights, any or every catch in the time series can potentially be estimated.

The other approach to calculating F_t values is by solving the generalized catch equation (see

above) iteratively. Subtracting predicted catch from the generalized catch equation gives:

$$g(F_t) = C_t + \frac{F_t\left(1 - e^{X_t}\right)}{X_t} B_t = 0$$

where X_t=G_t-M_t-F_t. If X_t=0, then $\overline{B}_t = B_t$ and F_t=C_t/B_t.

If $X_t \neq 0$, then the Newton-Raphson algorithm is used to solve for F_t (Kennedy and Gentle 1980). At each iteration of the algorithm, the current estimate F_t^i is updated using:

$$F_t^{i+1} = F_t^i - \frac{g\left(F_t^i\right)}{g'\left(F_t^i\right)}$$

where $g'\left(F_t^i\right)$ is the derivative F_t^i. Omitting subscripts, the derivative is:

$$g'(F) = -\frac{Be^{-F}\left[\left(e^F - e^\gamma\right)\gamma + e^\gamma F\gamma - e^\gamma F^2\right]}{X^2}$$

where γ=G-M_t. Iterations continue until $g\left(F_t^i\right)$ and $abs\left[g\left(F_t^{i+1}\right) - g\left(F_t^{i+1}\right)\right]$ are both less than a small number (e.g. ≤ 0.00001).

Initial values are important in algorithms that solve the catch equation numerically (Sims 1982). If M_t+F_t > G_t so that X_t < 0, then the initial value F_t^0 is calculated according to Sims (1982). If M_t+F_t < G_t so that X_t > 0, then initial values are calculated based on a generalized version of Pope's cohort analysis (Zhang and Sullivan 1988):

$$F_t^0 = \gamma_t - \ln\left[\frac{\left(B_t e^{0.5\gamma_t} - C_t\right)e^{0.5\gamma_t}}{B_t}\right]$$

F for landings versus F for discards

The total fishing mortality rate for each year can be partitioned into a component due to landed catch ${}^L F_t = \frac{D_t}{C_t} F_t$, and a component due to discard ${}^D F_t = \frac{L_t}{C_t} F_t$.

Predator consumption as discard data

In modeling population dynamics of prey species, estimates of predator consumption can be treated like discard in the KLAMZ model as a means for introducing time dependent natural mortality. Consider a hypothetical example with consumption data (mt y^{-1}) for three important predators. If the aggregate consumption data are included in the model as "discards", then the fishing mortality rate for discards ${}^d F_t$ (see above) would be an estimate of the component of natural mortality due to the three predators. In using this approach, the average level of natural mortality m would normally be reduced (e.g. so that $m_{new} + {}^d\overline{F} = m_{old}$) or estimated to account for the portion of natural mortality attributed to bycatch.

Surplus production calculations are harder to interpret if predator consumption is treated as discard data because surplus production calculations assume that F_t=0 (see above) and because surplus production is defined as the change in biomass from one year to the next in the absence of fishing (i.e. no landings or bycatch). However, it may be useful to compare surplus production at a given level of biomass from runs with and without consumption data as a means of estimating maximum changes in potential fishery yield if the selected predators were eliminated (assuming no

change in disease, growth rates, predation by other predators, etc.).

Effort calculations

Fishing mortality rates can be tuned to fishing effort data for the "landed" catch (i.e. excluding discards). Years with non-zero fishing effort used in the model must also have landings greater than zero. Assuming that effort data are lognormally distributed, the NLL for fishing effort is:

$$NLL = 0.5\sum_{y=1}^{n_{eff}} w_y \left[\frac{\ln\left(E_y / \hat{E}_y\right)}{\sigma} \right]^2$$

where w_y is an observation-specific weight, n_{eff} is the number of active effort observations (i.e. with $w_y > 0$), E_y and $\overline{E}_y$ are observed and predicted fishing effort data, and the log scale variance σ is a constant calculated from a user-specified CV.

Predicted fishing effort data are calculated:

$$\hat{E}_y = \zeta F_y^{\vartheta}$$

where $\zeta = e^u$, $\vartheta = e^b$, and u and b are parameters estimated by the model. If the parameter b is not estimated, then $\vartheta=1$ so that the relationship between fishing effort and fishing mortality is linear. If the parameter b is estimated, then $\vartheta \neq 1$ and the relationship is a power function.

Predator data as fishing effort

As described under "Predator consumption as discard data", predator consumption data can be treated as discard. If predator abundance data are available as well, and assuming that mortality due predators is a linear function of the predator-prey ratio, then both types of data may be used together to estimate natural mortality. The trick is to: 1) enter the predator abundance data as fishing effort; 2) enter the actual fishery landings as "discard"; 3) enter predator consumption estimates of the prey species as "landings" so that the fishing effort data in the refer to the predator consumption data; 4) use an option in the model to calculate the predator-prey ratio for use in place of the original predator abundance "fishing effort" data; and 5) tune fishing mortality rates for landings (a.k.a. predator consumption) to fishing effort (a.k.a. predator-prey ratio).

Given the predator abundance data κ_y, the model calculates the predator-prey ratio used in place of fishing effort data (E_y) as:

where B_y is the model's current estimate of total (a.k.a "prey") biomass. Subsequent calculations with E_y and the model's estimates of "fishing mortality" (F_y, really a measure of natural mortality) are exactly as described above for effort data. In using this approach, it is probably advisable to reduce m (the estimate of average mortality in the model) to account for the proportion of natural mortality due to predators included in the calculation. Based on experience to date, natural mortality due to consumption by the suite of predators can be estimated but only if m is assumed known.

Initial population age structure

In the KLAMZ model, old and new recruit biomass during the first year (R_1 and $S_1 = B_1 - R_1$) and biomass prior to the first year (B_0) are estimated as log scale parameters. Survival in the year

prior to the first year ("year 0") is $\tau_0 = e^{-F_0 - M_1}$ with F_0 chosen to obtain catch C_0 (specified as data) from the estimated biomass B_0. IGRs during year 0 and year 1 are assumed equal (G_0=G_1) in catch calculations.

Biomass in the second year of as series of delay-difference calculations depends on biomass (B_0) and survival (τ_0) in year 0:

$$B_2 = (1+\rho)\,\tau_1\,B_1 - \rho\,\tau_1\,\tau_0\,B_0 + R_2 - \rho\,\tau_1\,J_1\,R_1$$

There is, however, there is no direct linkage between B_0 and escapement biomass (S_1=B_1-R_1) at the beginning of the first year.

The missing link between B_0, S_1 and B_1 means that the parameter for B_0 tends to be relatively free and unconstrained by the underlying population dynamics model. In some cases, B_0 can be estimated to give good fit to survey and other data, while implying unreasonable initial age composition and surplus production levels. In other cases, B_0 estimates can be unrealistically high or low implying, for example, unreasonably high or low recruitment in the first year of the model (R_1). Problems arise because many different combinations of values for R_1, S_1 and B_0 give similar results in terms of goodness of fit. This issue is common in stock assessment models that use forward simulation calculations because initial age composition is difficult to estimate. It may be exacerbated in delay-difference models because age composition data are not used.

The KLAMZ model uses two constraints to help estimate initial population biomass and initial age structure.[16] The first constraint links IGRs for escapement (G^{Old}) in the first years to a subsequent value. The purpose of the constraint is to ensure consistency in average growth rates (and implicit age structure) during the first few years. For example, if IGRs for the first n_G years are constrained[17], then the NLL for the penalty is:

$$L_G = 0.5\sum_{t=1}^{n_G}\left[\frac{\ln\left(G_t^{Old}/G_{n_G+1}^{Old}\right)}{\sigma_G}\right]^2$$

where the standard deviation σ_G is supplied by the user. It is usually possible to use the standard deviation of Q_t^{Old} for later years from a preliminary run to estimate σ_G for the first few years. The constraint on initial IGRs should probably be "soft" and non-binding ($\lambda \approx 1$) because there is substantial natural variation in somatic growth rates due to variation in age composition.

The second constraint links B_0 to S_1 and ensures conservation of mass in population dynamics between years 0 and 1. In other words, the parameter for escapement biomass in year 1 is constrained to match an approximate projection of the biomass in year 0, accounting for growth, and natural and fishing mortality. The constraint is intended to be binding and satisfied exactly (e.g. λ=1000) because incompatible values of S_1 and B_0 are biologically impossible. In calculations:

$$S_1^p = B_0 e^{G_1 - F_0 - M_1}$$

where S_1^p is the projected escapement in year 1 and B_0 is the model's estimate of total biomass in year 0. The instantaneous rates for growth and natural mortality from year 1 (G_1 and M_1) are used in place of G_0 and M_0 because the latter are unavailable. The NLL for the constraint:

$$L = \left[\ln\left(\frac{S_1^p}{S_1}\right)^2\right]^2 + \left(S_1^p - S_1\right)^2$$

16 Quinn and Deriso (1999) describe another approach attributed to a manuscript by C. Walters.

17 Normally, $n_G \le 2$.

uses a log scale sum of squares and an arithmetic sum of squares. The former is effective when S_1 is small while the latter is effective when S_1 is large. Constants and details in calculation of NLL for the constraint are not important because the constraint is binding (e.g. λ=1000).

Equilibrium pristine biomass

It may be useful to constrain the biomass estimate for the first year in a model run towards an estimate of equilibrium pristine biomass if, for example, stock dynamics tend to be stable and catch data are available for the first years of the fishery, or as an alternative to the approach described above for initializing the age structure of the simulated population in the model. Equilibrium pristine biomass $\widetilde{B}_0$ is calculated based on the model's estimate of average recruitment and with no fishing mortality (calculations are similar to those described under "Per-recruit modeling" except that average recruitment is assumed in each year).[18] The NLL term for the constraint is:

$$L = \ln\left(\frac{\widetilde{B}_0}{B_0}\right)^2$$

Pristine equilibrium biomass is used as a hard constraint with a high emphasis factor (λ) so that the variance and constants normally used in NLL calculations are not important.

Estimating natural mortality

As described above, natural mortality calculations involve a parameter for the geometric mean value (m) and time dependent deviations (ϖ_t, which may or may not be turned on). Constraints on natural mortality process errors and natural mortality covariates can be used to help estimate the time dependent deviations and overall trend. The geometric mean natural mortality rate is usually difficult to estimate and best treated as a known constant. However, in the C++ version of the KLAMZ model, $m=e^{\pi}$ (where π is an estimable parameter in the model) and estimates of m can be conditioned on the constraint:

$$L = 0.5\left[\frac{\ln\left(w/w_{Target}\right)}{\sigma_{\varpi}}\right]^2$$

where w_{Target} is a user supplied mean or target value and σ_{ϖ} is a log scale standard deviation. The standard deviation is calculated from an arithmetic scale CV supplied by the user. Upper and lower bounds for m may be specified as well.

Goodness of fit for trend data

Assuming lognormal errors[19], the NLL used to measure goodness-of-fit to "survey" data that measure trends in abundance or biomass (or survival, see below) is:

18 Future versions of the KLAMZ model will allow equilibrium initial biomass to be calculated based on other recruitment values and for a user-specified level of F (Butler et al. 2003).

19 Abundance indices with statistical distributions other than log normal may be used as well, but are not currently programmed in the KLAMZ model. For example, Butler et al. (2003) used abundance indices with binomial distributions in a delay-difference model for cowcod rockfish. The next version of KLAMZ will accommodate presence-absence data with binomial distributions.

$$L = 0.5 \sum_{j=1}^{N_v} \left[\frac{\ln\left(I_{v,j} \Big/ \hat{I}_{v,j} \right)}{\sigma_{v,j}} \right]^2$$

where $I_{v,t}$ is an index datum from survey v, hats "^" denote model estimates, $\sigma_{v,j}$ is a log scale standard error (see below), and N_v is the number of observations. There are two approaches to calculating standard errors for log normal abundance index data in KLAMZ and it is possible to use different approaches for different types of abundance index data in the same model (see below).

Standard errors for goodness of fit

In the first approach, all observations for one type of abundance index share the same standard error, which is calculated based on overall goodness of fit. This approach implicitly estimates the standard error based on goodness of fit, along with the rest of the parameters in the model (see "NLL kernels" above).

In the second approach, each observation has a potentially unique standard error that is calculated based on its CV. The second approach calculates log scale standard errors from arithmetic CVs supplied as data by the user (Jacobson et al. 1994):

$$\sigma_{v,t} = \sqrt{\ln\left(1 + CV_{v,t}^2\right)}$$

Arithmetic CV's are usually available for abundance data. It may be convenient to use $CV_{v,t}$=1.31 to get $\sigma_{v,t}$=1.

There are advantages and disadvantages to both approaches. CV's carry information about the relative precision of abundance index observations. However, CV's usually overstate the precision of data as a measure of fish abundance[20] and may be misleading in comparing the precision of one sort of data to another as a measure of trends in abundance (e.g. in contrasting standardized LPUE that measure fishing success, but not abundance, precisely with survey data that measure trends in fish abundance directly, but not precisely). Standard errors estimated implicitly are often larger and more realistic, but assume that all observations in the same survey are equally reliable.

Predicted values for abundance indices

Predicted values for abundance indices are calculated:

$$\hat{I}_{v,t} = Q_v A_{v,t}$$

where Q_v is a survey scaling parameter (constant here but see below) that converts units of biomass to units of the abundance index. $A_{v,t}$ is available biomass at the time of the survey.

In the simplest case, available biomass is:

$$A_{v,t} = s_{v,New} R_t e^{-X_t^{New} \Delta_{v,t}} + s_{v,Old} S_t e^{-X_t^{Old} \Delta_{v,t}}$$

where $s_{v,New}$ and $s_{v,Old}$ are survey selectivity parameters for new recruits (R_t) and old recruits (S_t);

20 The relationship between data and fish populations is affected by factors (process errors) that are not accounted for in CV calculations.

$X_t^{New} = G_t^{New} - F_t - M_t$ and $X_t^{Old} = G_t^{Old} - F_t - M_t$; $j_{v,t}$ is the Julian date at the time of the survey, and $\Delta_{v,t}=j_{v,t}/365$ is the fraction of the year elapsed at the time of the survey.

Survey selectivity parameter values ($s_{v,New}$ and $s_{v,Old}$) are specified by the user and must be set between zero and one. For example, a survey for new recruits would have $s_{v,New}$=1 and $s_{v,Old}$=0. A survey that measured abundance of the entire stock would have $s_{v,New}$=1 and $s_{v,Old}$=1.

Terms involving $\Delta_{v,t}$ are used to project beginning of year biomass forward to the time of the survey, making adjustments for mortality and somatic growth.[21] As described below, available biomass $A_{v,t}$ is adjusted further for nonlinear surveys, surveys with covariates and surveys with time variable $Q_{v,t}$.

Scaling parameters (Q) for log normal abundance data

Scaling parameters for surveys with lognormal statistical errors were computed using the maximum likelihood estimator:

$$Q_v = e^{\frac{\sum_{i=1}^{N_v}\left[\ln\left(\frac{I_{v,i}}{A_{v,i}}\right)\Big/\sigma_{v,j}^2\right]}{\sum_{j=1}^{N_j}\left(1/\sigma_{v,j}^2\right)}}$$

where N_v is the number of observations with individual weights greater than zero. The closed form maximum likelihood estimator gives the same answer as if scaling parameters are estimated as free parameters in the assessment model assuming lognormal survey measurement errors.

Survey covariates

Survey scaling parameters may vary over time based on covariates in the KLAMZ model. The survey scaling parameter that measures the relationship between available biomass and survey data becomes time dependent:

$$\hat{I}_{v,t} = Q_{v,t} A_{v,t}$$

and

$$Q_{v,t} = Q_v e^{\sum_{r=1}^{n_v} d_{r,t}\theta_r}$$

with n_v covariates for the survey and parameters θ_r estimated in the model. Covariate effects and available biomass are multiplied to compute an adjusted available biomass:

$$A'_{v,t} = A_{v,t} e^{\sum_{r=1}^{n_v} d_{r,t}\theta_r}$$

The adjusted available biomass $A'_{v,t}$ is used instead of the original value $A_{v,t}$ in the closed form maximum likelihood estimator described above.

21 It may be important to project biomass forward if an absolute estimate of biomass is available (e.g. from a hydroacoustic or daily egg production survey), if fishing mortality rates or high or if the timing of the survey varies considerably from year to year.

Covariates might include, for example, a dummy variable that represents changes in survey bottom trawl doors or a continuous variable like average temperature data if environmental factors affect distribution and catchability of fish schools. Dummy variables are usually either 0 or 1, depending on whether the effect is present in a particular year. With dummy variables, Q_v is the value of the survey scaling parameter with no intervention ($d_{r,t}$=0).

For ease in interpretation of parameter estimates for continuous covariates (e.g. temperature data), it is useful to center covariate data around the mean:

$$d_{r,t} = d'_{r,t} - \overline{d'_r}$$

where $d'_{r,t}$ is the original covariate. When covariates are continuous and mean-centered, Q_v is the value of the survey scaling parameter under average conditions ($d_{r,t}$=0) and units for the covariate parameter are easy to interpret (for example, units for the parameter are $1/^{o}C$ if the covariate is mean centered temperature in ^{o}C).

It is possible to use a survey covariate to adjust for differences in relative stock size from year to year due to changes in the timing of a survey. However, this adjustment may be made more precisely by letting the model calculate $\Delta_{v,t}$ as described above, based on the actual timing data for the survey during each year.

Nonlinear abundance indices

With nonlinear abundance indices, and following Methot (1990), the survey scaling parameter is a function of available biomass:

$$Q_{v,t} = Q_v A_{v,t}^{\Gamma}$$

so that:

$$\hat{I}_{v,t} = \left(Q_v A_{v,t}^{\Gamma}\right) A_{v,t}$$

Substituting $e^{\gamma}=\Gamma+1$ gives the equivalent expression:

$$\hat{I}_{v,t} = Q_v A_{v,t}^{e^{\gamma}}$$

where γ is a parameter estimated by the model and the survey scaling parameter is no longer time dependent. In calculations with nonlinear abundance indices, the adjusted available biomass:

$$A'_{v,t} = A_{v,t}^{e^{\gamma}}$$

is computed first and used in the closed form maximum likelihood estimator described above to calculate the survey scaling parameter. In cases where survey covariates are also applied to a nonlinear index, the adjustment for nonlinearity is carried out first.

Survey Q process errors

The C++ version of the KLAMZ model can be used to allow survey scaling parameters to change in a controlled fashion from year to year (NEFSC 2002):

$$Q_{v,t} = Q_v e^{\varepsilon_{v,t}}$$

where the deviations $\varepsilon_{v,t}$ are constrained to average zero. Variation in survey Q values is controlled by the NLL penalty:

$$L = 0.5 \sum_{j=1}^{N_v} \left[\frac{\varepsilon_{v,j}}{\sigma_v} \right]^2$$

where the log scale standard deviation σ_v based on an arithmetic CV supplied by the user (e.g. see NEFSC 2002). In practice, the user increases or decreases the amount of variability in Q by decreasing or increasing the assumed CV.

Survival ratios as surveys

In the C++ version of KLAMZ, it is possible to use time series of survival data as "surveys". For example, an index of survival might be calculated using survey data and the Heinke method (Ricker 1975) as:

$$A_t = \frac{I_{k+1,t+1}}{I_{k,t}}$$

so that the time series of A_t estimates are data that may potentially contain information about scale or trends in survival. Predicted values for an a survival index are calculated:

$$\hat{A}_t = e^{-Z_t}$$

After predicted values are calculated, survival ratio data are treated in the same way as abundance data (in particular, measurement errors are assumed to be lognormal). Selectivity parameters are ignored for survival data but all other features (e.g. covariates, nonlinear scaling relationships and constraints on Q) are available.

Recruitment models

Recruitment parameters in KLAMZ may be freely estimated or estimated around an internal recruitment model, possibly involving spawning biomass. An internally estimated recruitment model can be used to reduce variability in recruitment estimates (often necessary if data are limited), to summarize stock-recruit relationships, or to make use of information about recruitment in similar stocks. There are four types of internally estimated recruitment models in KLAMZ: 1) random (white noise) variation around a constant or time dependent mean modeled as a step function; 2) random walk (autocorrelated) variation around a constant or time dependent mean modeled as a step function; 3) random variation around a Beverton-Holt recruitment model; and 4) random variation around a Ricker recruitment model. The user must specify a type of recruitment model but the model is not active unless the likelihood component for the recruitment model is turned on ($\lambda > 0$).

The first step in recruit modeling is to calculate the expected log recruitment level $E[ln(R_t)]$ given the recruitment model. For random variation around a constant mean, the expected log recruitment level is the log geometric mean recruitment:

$$E[\ln(R_t)] = \sum_{j=1}^{N} \ln(R_j) \Big/ N$$

For a random walk around a constant mean recruitment, the expected log recruitment level is the logarithm of recruitment during the previous year:

$$E[\ln(R_t)] = \ln(R_{t-1})$$

with no constraint on recruitment during the first year R_1.

For the Beverton-Holt recruitment model, the expected log recruitment level is:

$$E[\ln(R_t)] = \ln\left[e^a T_{t-\ell} / \left(e^b + T_{t-\ell}\right)\right]$$

where $a=e^{\alpha}$ and $b=e^{\beta}$, the parameters α and β are estimated in the model, T_t is spawning biomass, and ℓ is the lag between spawning and recruitment. Spawner-recruit parameters are estimated as log transformed values (e^{α} and e^{β}) to enhance model stability and ensure the correct sign of values used in calculations. Spawning biomass is:

$$T_t = m_{new} R_t + m_{old} S_t$$

where m_{new} and m_{old} are maturity parameters for new and old recruits specified by the user. For the Ricker recruitment model, the expected log recruitment level is:

$$E[\ln(R_t)] = \ln\left(S_{t-\ell} e^{a-bS_{t-\ell}}\right)$$

where $a=e^{\alpha}$ and $b=e^{\beta}$, and the parameters α and β are estimated in the model.

Given the expected log recruitment level, log scale residuals for the recruitment model are calculated:

$$r_t = \ln(R_t) - E[\ln(R_t)]$$

Assuming that residuals are log normal, the NLL for recruitment residuals is:

$$L = \sum_{t=t_{first}}^{N} w_t \left[\ln(\sigma_r) + 0.5\left(r_t / \sigma_r \right)^2 \right]$$

where λ_t is an instance-specific weight usually set equal one. The additional term in the NLL [$ln(\sigma_r)$] is necessary because the variance σ_r^2 is estimated internally, rather than specified by the user.

The log scale variance for residuals is calculated using the maximum likelihood estimator:

$$\sigma_r^2 = \frac{\sum_{j=t_{first}}^{N} r_j}{N}$$

where N is the number of residuals. For the recruitment model with constant variation around a mean value, t_{first}=1. For the random walk recruitment model, t_{first}=2. For the Beverton-Holt and Ricker models, $t_{first}=\ell+1$ and the recruit model imposes no constraint on variability of recruitment during years 1 to ℓ (see below). The biased maximum likelihood estimate for σ^2 (with N in the divisor instead of the degrees of freedom) is used because actual degrees of freedom are unknown. The variance term σ^2 is calculated explicitly and stored because it is used below.

Constraining the first few recruitments

It may be useful to constrain the first ℓ years of recruitments when using either the Beverton-Holt or Ricker models if the unconstrained estimates for early years are erratic. In the KLAMZ model, this constraint is calculated:

$$NLL = \sum_{t=1}^{t_{first}-1} w_t \left\{ \ln(\sigma_r + 0.5\left[\frac{\ln\left(R_t / E\left(R_{t_{first}}\right)\right)}{\sigma_r} \right]^2 \right\}$$

where t_{first} is the first year for which expected recruitment $E(R_l)$ can be calculated with the spawner-recruit model. In effect, recruitments that not included in spawner-recruit calculations are constrained towards the first spawner-recruit prediction. The standard deviation is the same as used

in calculating the NLL for the recruitment model.

Prior information about the absolute value abundance index scaling parameters (Q)

A constraint on the absolute value one or more scaling parameters (Q_v) for abundance or survival indices may be useful if prior information is available (e.g. NEFSC 2000; NEFSC 2001; NEFSC 2002). In the Excel version, it is easy to program these (and other) constraints in an *ad-hoc* fashion as they are needed. In the AD Model Builder version, log normal and beta distributions are preprogrammed for use in specifying prior information about Q_v for any abundance or survival index.

The user must specify which surveys have prior distributions, minimum and maximum legal bounds (q_{min} and q_{max}), the arithmetic mean $(\bar{q})$ and the arithmetic CV for the prior the distribution. Goodness of fit for Q_v values outside the bounds (q_{min}, q_{max}) are calculated:

$$L = \left| \begin{array}{l} 10000\left(Q_v - q_{\max}\right)^2 \; if \; Q_v \geq q_{\max} \\ 10000\left(q_{\min} - Q_v\right)^2 \; if \; Q_v \leq q_{\min} \end{array} \right.$$

Goodness of fit for Q_v values inside the legal bounds depend on whether the distribution of potential values is log normal or follows a beta distribution.

Lognormal case

Goodness of fit for lognormal Q_v values within legal bounds is:

$$L = 0.5\left[\frac{\ln\left(Q_v\right) - \tau}{\varphi}\right]^2$$

where the log scale standard deviation $\varphi = \sqrt{\ln\left(1 + CV\right)}$ and $\tau = \ln\left(\bar{q}\right) - \frac{\varphi^2}{2}$ is the mean of the corresponding log normal distribution.

Beta distribution case

The first step in calculation goodness of fit for Q_v values with beta distributions is to calculate the mean and variance of the corresponding "standardized" beta distribution:

$$\bar{q}' = \frac{\bar{q} - q_{\min}}{D}$$

and

$$Var\left(q'\right) = \left(\frac{\bar{q}\; CV}{D}\right)^2$$

where the range of the standardized beta distribution is $D=q_{max}-q_{min}$. Equating the mean and variance to the estimators for the mean and variance for the standardized beta distribution (the "method of moments") gives the simultaneous equations:

$$\bar{q}' = \frac{a}{a+b}$$

and

$$Var\left(q'\right) = \frac{ab}{\left(a+b\right)^2\left(a+b+1\right)}$$

where a and b are parameters of the standardized beta distribution.[22] Solving the simultaneous equations gives:

$$b = \frac{(\bar{q}' - 1)[Var(q') + (\bar{q}' - 1)\bar{q}']}{Var(q')}$$

and:

$$a = \frac{b\bar{q}'}{1 - \bar{q}'}$$

Goodness of fit for beta Q_v values within legal bounds is calculated with the NLL:

$$L = (a-1)\ln(Q'_v) + (b-1)\ln(1 - Q'_v)$$

where $Q'_v = Q_v / (Q_v - q_{\min})$ is the standardized value of the survey scaling parameter Q_v.

Prior information about relative abundance index scaling parameters (Q-ratios)

Constraints on "Q-ratios" can be used in fitting models if some information about the relative values of scaling parameters for two abundance indices is available. For example, ASMFC (2001, p. 46-47) assumed that the relative scaling parameters for recruit and post-recruit lobsters taken in the same survey was either 0.5 or 1. If both indices are from the same survey cruise (e.g. one index for new recruits and one index for old recruits in the same survey), then assumptions about q-ratios are analogous to assumptions about the average selectivity of the survey of the survey for new and old recruits.

Q-ratio constraints tend to stabilize and have strong effects on model estimates. ASMFC (2001, p. 274) found, for example, that goodness of fit to survey data, abundance and fishing mortality estimates for lobster changed dramatically over a range of assumed q-ratio values.

To use q-ratio information in the KLAMZ model, the user must identify two surveys, a target value for the ratio of their Q values, and a CV for differences between the models estimated q-ratio and the target value. For example, if the user believes that the scaling parameters for abundance index 1 and abundance index 3 is 0.5, with a CV=0.25 for uncertainty in the prior information then the model's estimate of the q-ratio is $\rho = Q_1/Q_3$. The goodness of fit calculation is:

$$L = 0.5\left(\frac{\ln(\rho/\tau)}{\sigma}\right)^2$$

where τ is the target value and the log scale standard deviation σ is calculated from the arithmetic CV supplied by the user.

Normally, a single q-ratio constraint would be used for the ratio of new and old recruits taken during the same survey operation. However, in KLAMZ any number of q-ratio constraints can be used simultaneously and the scaling parameters can be for any two indices in the model.

Surplus production modeling

Surplus production models can be fit internally to biomass and surplus production estimates in the model (Jacobson et al. 2002). Models fit internally can be used to constrain estimates of biomass and recruitment, to summarize results in terms of surplus production, or as a source of

22 If x has a standardized beta distribution with parameters a and b, then the probability of x is

$$P(x) = \frac{x^{a-1}(1-x)^{b-1}}{\Gamma(a,b)}.$$

information in tuning the model. The NLL for goodness of fit assumes normally distributed process errors in the surplus production process:

$$L = 0.5 \sum_{j=1}^{N_P} \left(\frac{\widetilde{P}_j - P_j}{\sigma} \right)^2$$

where N_p is the number of surplus production estimates (number of years less one), $\widetilde{P}_t$ is a predicted value from the surplus production curve, P_t is the assessment model estimate, and the standard deviation σ is supplied by the user based, for example, on preliminary variances for surplus production estimates.[23] Either the symmetrical Schaefer (1957) or asymmetric Fox (1970) surplus production curve may be used to calculate $\widetilde{P}_t$ (Quinn and Deriso 1999).

It may be important to use a surplus production curve that is compatible with recruitment patterns or assumptions about the underlying spawner-recruit relationship. More research is required, but the asymmetric shape of the Fox surplus production curve appears reasonably compatible with the assumption that recruitment follows a Beverton-Holt spawner-recruit curve (Mohn and Black 1998). In contrast, the symmetric Schaefer surplus production model appears reasonably compatible with the assumption that recruitment follows a Ricker spawner-recruit curve.

The Schaefer model has two log transformed parameters that are estimated in KLAMZ:

$$\widetilde{P}_t = e^{\alpha} B_t - e^{\beta} B_t^2$$

The Fox model also has two log transformed parameters:

$$\widetilde{P}_t = -e\left(e^{e^{\alpha}}\right) \frac{B_t}{e^{\beta}} \log\left(\frac{B_t}{e^{\beta}} \right)$$

See Quinn and Deriso (1999) for formulas used to calculate reference points (F_{MSY}, B_{MSY}, MSY, and K) for both surplus production models.

Catch/biomass

Forward simulation models like KLAMZ may tend to estimate absurdly high fishing mortality rates, particularly if data are limited. The likelihood constraint used to prevent this potential problem is:

$$L = 0.5 \sum_{t=0}^{N} \left(d_t^2 + q^2 \right)$$

where:

$$d_t = \left| \begin{array}{l} Ft - \Phi \; if \; Ft > \Phi \\ 0 \; otherwise \end{array} \right.$$

and

with the threshold value κ normally set by the user to about 0.95. Values for κ can be linked to

23 Variances in NLL for surplus production-biomass models are a subject of ongoing research. The advantage in assuming normal errors is that negative production values (which occur in many stocks, e.g. Jacobson et al. 2001) are accommodated. In addition, production models can be fit easily by linear regression of P_t on B_t and B_t^2 with no intercept term. However, variance of production estimate residuals increases with predicted surplus production. Therefore, the current approach to fitting production curves in KLAMZ is not completely satisfactory.

maximum F values using the modified catch equation described above. For example, to use a maximum fishing mortality rate of about $F \approx 4$ with M=0.2 and G=0.1 (maximum X=4+0.2-0.1=4.1), set $\kappa \approx F/X(1-e^{-X})$=4 / 4.1 (1-e^{-4})=0.96.

Uncertainty

The AD Model Builder version of the KLAMZ model automatically calculates variances for parameters and quantities of interest (e.g. R_t, F_t, B_t, F_{MSY}, B_{MSY}, $\overline{F}_{\mathrm{Re}cent}$, $\overline{B}_{\mathrm{Re}cent}$, $\overline{F}_{\mathrm{Re}cent} / F_{MSY}$, $\overline{B}_{\mathrm{Re}cent} / B_{MSY}$, etc.) by the delta method using exact derivatives. If the objective function is the log of a proper posterior distribution, then Markov Chain Monte Carlo (MCMC) techniques implemented in AD Model Builder libraries can be used estimate posterior distributions representing uncertainty in the same parameters and quantities.[24]

Bootstrapping

A FORTRAN program called BootADM can be used to bootstrap survey and survival index data in the KLAMZ model. Based on output files from a "basecase" model run, BootADM extracts standardized residuals:

$$r_{v,j} = \frac{\ln\left(I_{v,j} \Big/ \hat{I}_{v,j} \right)}{\sigma_{v,j}}$$

along with log scale standard deviations ($\sigma_{v,j}$, originally from survey CV's or estimated from goodness of fit), and predicted values $\left(\hat{I}_{v,j}\right)$ for all active abundance and survival observations. The original standardized residuals are pooled and then resampled (with replacement) to form new sets of bootstrapped survey "data":

$${}^{x}I_{v,j} = \hat{I}_{v,j} e^{r\sigma_{v,j}}$$

where r is a resampled residual. Residuals for abundance and survival data are combined in bootstrap calculations. BootADM builds new KLAMZ data files and runs the KLAMZ model repetitively, collecting the bootstrapped parameter and other estimates at each iteration and writing them to a comma separated text file that can be processed in Excel to calculate bootstrap variances, confidence intervals, bias estimates, etc. for all parameters and quantities of interest (Efron 1982).

Projections

Stochastic projections can be carried out using another FORTRAN program called SPROJDDF based on bootstrap output from BootADM. Basically, bootstrap estimates of biomass, recruitment, spawning biomass, natural and fishing mortality during the terminal years are used with recruit model parameters from each bootstrap run to start and carryout projections.[25] Given a user-specified level of catch or fishing mortality, the delay-difference equation is used to project stock status for a user-specified number of years. Recruitment during each projected year is based on

24 MCMC calculations are not available in the current version because objective function calculations use concentrated likelihood formulas. However, the C++ version of KLAMZ is programmed in other respects to accommodate Bayesian estimation.

25 At present, only Beverton-Holt recruitment calculations are available in SPROJDDF.

simulated spawning biomass, log normal random numbers, and spawner-recruit parameters (including the residual variance) estimated in the bootstrap run. This approach is similar to carrying out projections based on parameters and state variables sampled from a posterior distribution for the basecase model fit. It differs from most current approaches because the spawner-recruit parameters vary from projection to projection.

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APPENDIX B7: "West coast groundfish harvest rate policy workshop report", provided courtesy of the Pacific Fishery Management Council.

West Coast Groundfish Harvest Rate Policy Workshop
Alaska Fisheries Science Center, Seattle, Washington: March 20-23, 2000
Sponsored by the Scientific & Statistical Committee of
the Pacific Fishery Management Council

Panel Report

Stephen Ralston (chairman), James R. Bence, William G. Clark,
Ramon J. Conser, Thomas Jagielo, and Terrance J. Quinn II.

Scientific and Management Background

Through 1998 the policy of the Pacific Fishery Management Council (PFMC) was to set the Allowable Biological Catch (ABC) of a stock by applying the fishing mortality rate that produces Maximum Sustainable Yield (F_{MSY}) to an estimate of exploitable stock biomass. Policies of this kind are termed constant rate policies because, once the estimate of F_{MSY} is determined, the annual ABC is strictly proportional to estimates of exploitable biomass. However, owing to short data series and other technical issues, it generally has not been possible to directly estimate F_{MSY} reliably for any stock. Consequently, during the 1980s and into the early 1990s, one of several common surrogate or proxy estimates of F_{MSY} was used (e.g., $F_{0.1}$ or F=M).

Clark (1991) proposed the $F_{35\%}$ harvest rate as a more general and rational surrogate rate. $F_{35\%}$ is the fishing mortality rate that reduces the spawning potential *per recruit* to 35% of the unfished level. By reasonably assuming that fecundity is proportional to average weight, it is the rate of fishing that reduces the spawning biomass *per recruit* to 35% of what would exist if there were no fishing. Clark showed that this rate would produce a yield close to MSY for a range of life history parameters and productivity relationships that were intended to cover the great majority of well-studied groundfish stocks with long histories of exploitation (most of which were Atlantic stocks). He also showed that $F_{35\%}$ was very close to both $F_{0.1}$ and F=M when the schedules of recruitment and maturity coincided, and were sensibly higher or lower when they differed. However, a later paper extended the original analysis to cases with random and serially correlated recruitment variation (Clark 1993), and concluded that $F_{40\%}$ would be a better choice overall than $F_{35\%}$. Mace (1994) also recommended $F_{40\%}$ on the basis of deterministic calculations. The current scientific consensus now indicates that $F_{40\%}$ is an appropriate default harvest rate for stocks with unknown productivity parameters.

The PFMC adopted $F_{35\%}$ as its standard surrogate in 1992, and switched to $F_{40\%}$ for *Sebastes* only in 1997, based principally on the conclusions of Clark (1993) and Mace (1994). In 1998 it then adopted the so-called "40-10" rule under Amendment 11 to the groundfish FMP. The 40-10 rule represented a departure from prior constant rate harvest policies, wherein the target fishing mortality rate is reduced for stocks whose biomass is below 40% of the estimated unfished biomass (B_0).

Common Confusion Over Relative Biomass and Relative Biomass per Recruit

In addition to recommending the $F_{35\%}$ strategy, Clark (1991) suggested a more robust biomass-based strategy that consists of simply maintaining spawning biomass at around 40% of the estimated unfished level. Perhaps partly because of the shared "40%" level, it is often supposed that the $F_{40\%}$ harvest rate will reduce spawning biomass to 40% of unfished biomass, but that is only true for stocks with highly resilient spawner-recruit relationships. For less resilient stocks, $F_{40\%}$ will reduce biomass to a lower level, possibly much lower, while still providing a yield near MSY. That is possible because yield is not very sensitive to equilibrium biomass over a wide range of biomass levels, so a yield near MSY can be obtained even when biomass is well below B_{MSY}. It is this feature of yield curves that makes it possible for a rate like $F_{40\%}$ to perform well in terms of yield over a wide range of spawner-recruit productivity curves. For some curves $F_{40\%}$ is well above F_{MSY} and for some of the curves it is well below, but in none of the cases considered is it so far above or below F_{MSY} that yield is much lower than MSY.

For the most likely sort of groundfish spawner-recruit relationships (i.e., asymptotic curves such as the Beverton-Holt model), and if other forms of stock compensation are negligible, B_{MSY} is likely to lie in the range of 25-40% of unfished biomass. Therefore, even if F_{MSY} was known and was implemented for a stock, the resulting biomass level would generally be less than 40% of B_0 on average. For some stocks, recruitment variations alone might then result in biomass levels falling below 25% of the unfished level, which is the overfished threshold as implemented in Amendment 11 to the groundfish FMP. Thus, fishing at $F_{40\%}$, which can be well above (or below) F_{MSY}, can be expected to result in biomass levels that are occasionally or on average very low for some stocks. Thus, given the new requirement of biomass-based overfished thresholds (Department of Commerce 1998), the relationship between harvest rates and biomass levels becomes more critical.

Declines of Pacific Coast Stocks Fished at $F_{35\text{-}40\%}$

Ralston (1998) showed that a number of Pacific coast rockfish stocks declined to low levels during the last two decades, contributing to concerns about the wisdom of the $F_{35\%}$ policy. His findings, as well as analyses conducted by the GMT during the preparation of Amendment 11, led to a series of workshops, including this latest review. This panel received a number of papers dealing with the productivity of the stocks in question and considered arguments for and against retaining the $F_{35\%}/F_{40\%}$ rate (in conjunction with the 40-10 rule) for all stocks.

We believe there are at least three possible factors that are responsible for the observed declines in groundfish stocks:

1. Normal operation of the $F_{35\%}/F_{40\%}$ strategy.

As explained above, either an $F_{35\%}$ or $F_{40\%}$ harvest rate will often lead to biomass levels that are well below what many people commonly expect, even when the rate is no larger than F_{MSY}. When it is larger, as will happen for some stocks, resulting biomasses can be very low. The important point is that both F_{MSY} and the proxy rate are calculated to achieve a certain level of yield, not biomass. In addition, harvesting at $F_{35\%}/F_{40\%}$ should be viewed as a risk-neutral policy in that, being a compromise intermediate rate, some stocks will be over-exploited and some stocks will be under-exploited, with no penalty imposed for over-exploitation.

2. Higher than intended harvest rates.

Recent assessments show that in many cases, actual fishing mortality rates were well above $F_{35\%}$. This can happen in any fishery when quotas are set on the basis of current biomass estimates, which are subsequently revised downward in a later assessment.

3. Apparently low productivity of Pacific coast stocks.

The spawner-recruit estimates that have accumulated over the last twenty years on Pacific coast groundfish stocks indicate very low resiliency in the spawner-recruit relationships — at or below the lowest values estimated for well-studied stocks elsewhere in the world (Myers *et al.* 1999). It is not surprising then, that the estimated productivity of these stocks is in many instances lower than the range of values considered plausible by Clark (1991) in his derivation of the $F_{35\%}$ strategy.

Because these low productivity estimates are so common among Pacific coast groundfish stocks, and so uncommon elsewhere, there is some suspicion that they result from some unrecognized flaw common to all of the Pacific coast groundfish assessments. However, with the exception of discards (see below), the panel has no reason to doubt the accuracy of west coast groundfish stock assessments. The same methods and models have produced estimates of higher productivity elsewhere (e.g., in Alaska). For the time being, therefore, we believe that all of the assessment results should be taken at face value, and that the Council's harvest strategy should be reconsidered in light of the apparently low productivity of many of the stocks.

The reason for anomalously low productivity in this region is not certain, but it may well be linked to the climatic regime shift that occurred in the eastern Pacific ocean around 1977-78. Since then, ocean conditions have been generally more favorable for many Alaskan stocks and have been less favorable for many Pacific coast stocks. Sometime in the future conditions on the west coast are likely to change again. Still, there is no assurance that this will occur in the near future and so, in the interim, the PFMC should manage groundfish stocks according to their current productive capacity.

The panel reviewed results presented by Williams (see Appendix A), which suggest that discards of small fish could contribute to the perception of low groundfish productivity. To the extent that this occurs, its effect is to reduce apparent recruitments and therefore to make ground-fish stocks appear to be less resilient. This scenario depends on: (1) an increasing exploitation rate over time and (2) substantial unaccounted for discarding of the smallest fish captured. While groundfish exploitation rates have certainly risen, and substantial unaccounted for discards of small fish is likely in some fisheries, discards are generally not documented for these stock and cannot be quantified at present. Clearly more research on this issue is desirable and, in general,

the panel stresses that a full accounting of total catch is necessary for the PFMC to adequately manage any of the resources under its authority.

Panel Recommendations for Default Groundfish Harvest Rates

The panel reviewed the information presented by each presenter (see Appendix A), as well as other recently published material (e.g., Myers *et al.* 1999). Of particular importance were the works of Brodziak, Dorn, MacCall, and Parrish because each of these studies broadly re-analyzed the information presented in historical PFMC stock assessments in an attempt to estimate F_{MSY} for each stock and their F_{spr} equivalents (i.e., the spawning potential per recuit fishing mortality rate). Significantly, each of these studies indicated that in many instances groundfish productivity, as estimated from the results of stock assessments, is insufficient to support harvests at the $F_{35\%}$ or even $F_{40\%}$ rates.

With respect to the rockfishes (*Sebastes* spp.) the panel found the work of Dorn to be very compelling. His results showed that, when the genus is examined as a whole through the use of meta-analysis, west coast rockfish stocks (exclusive of Pacific ocean perch) have F_{MSY} rates that range between $F_{45\%} - F_{67\%}$ for risk-neutral models, assuming either the Beverton-Holt or Ricker models with lognormal or gamma errors (four cases). However, gamma error models fit the data more poorly than models with a lognormal error structure and, as a consequence, the panel supported the use of Dorn's lognormal analysis only. For that subset of cases, the estimated F_{MSY} rates ranged $F_{45\%} - F_{54\%}$ over the two recruitment models. The panel then adopted $F_{50\%}$ as a midpoint, risk-neutral, proxy for rockfish F_{MSY}. In addition, the panel recommends including the thorneyheads (genus *Sebastolobus*) with the rockfish in the setting of default harvest rate proxies.

The panel discussed results for Pacific whiting and concluded that the information base for that species was the best available for any west coast groundfish. Harvests are currently determined using the 40-10 policy in association with a fishing mortality rate equal to $F_{40\%}$. This rate is based on a separate and distinct meta-analysis of worldwide *Merluccius* productivity that was conducted as part of the last stock assessment (Dorn *et al.* 1999) and seems appropriate as a risk-neutral harvest policy. Consequently, the panel does not recommend any changes in harvest rate for Pacific whiting.

For flatfishes (including Dover sole), the panel concluded that resiliency is typically higher than in other taxa (e.g., Brodziak *et al.* 1997, Mace and Sissenwine 1993, Myers *et al.* 1999). As a consequence, the panel recommends using a default rate of $F_{40\%}$ for all flatfish species in the groundfish FMP. This rate is consistent with the general findings of Clark (1993) and Mace (1994).

For all other species in the groundfish FMP (including sablefish and lingcod) the panel recommends an intermediate harvest rate of $F_{45\%}$. This intermediate rate was selected as a sensible risk-neutral alternative that would afford increased protection to all the remaining groundfish stocks. However, the level of certainty in setting this default rate is very low. Consequently, the panel makes two recommendations

with respect to the estimation of groundfish productivity, i.e.,

(1) Assessment authors are encouraged to evaluate the resiliency of the specific stocks they model. When such analysis produces scientifically credible estimates of productivity, the analyst is encouraged to present those findings as part of their stock assessment. However, any productivity analysis should always include a measure of the uncertainty in the point estimates of management reference points (e.g., F_{MSY}, B_{MSY}, and B_0).

(2) A proper consideration of risk is essential in the setting of optimum yields for west coast groundfish stocks. Utilization of a risk-neutral harvest rate proxy (e.g., $F_{50\%}$ for *Sebastes* and *Sebastolobus*) implies that some stocks within the group are quite likely to be over-exploited. Similarly, calculation of an ABC using an unbiased stock-specific point estimate of F_{MSY} will result in overfishing if the estimate is, by chance, too high. It is the PFMC's responsibility to account for these risks of overfishing through the use of a precautionary approach in the establishment of optimum yields. In addition, the NMFS Guidelines specify that status determination criteria must specify a maximum fishing mortality rate threshold that is less than or equal to F_{MSY} (Department of Commerce 1998). While this issue is not specifically addressed in this report, the choice of the threshold should depend on the level of uncertainty associated with the estimate of F_{MSY} or its proxy.

In summary, panel recommendations with respect to risk-neutral default harvest rate F_{MSY} proxies for west coast groundfish are:

Pacific whiting	$F_{40\%}$
Sebastes & *Sebastolobus*	$F_{50\%}$
Flatfish	$F_{40\%}$
Other groundfish	$F_{45\%}$

Due to a lack of detailed life history and stock status information, it will not be possible to implement these recommendations for many stocks. In particular, the "remaining rockfish" management unit (PFMC 1999) includes a number of species for which the ABC has been set using the F=M harvest rate proxy (Rogers *et al.* 1996). Currently, the optimum yield (OY) of those species is reduced by 25% as a "precautionary adjustment" (PFMC 1999), amounting to an F=0.75M policy. The panel discussed the remaining rockfish category in light of results presented in MacCall's production model analysis (Appendix A), which indicated that 0.40M may be a better proxy for an optimal exploitation rate. However, due to the review panel's unwillingness to fully endorse production modeling as a viable means of estimating groundfish productivity (see below), the panel recommended that the PFMC establish F=0.75M as the default, risk-neutral policy for the remaining rockfish management category. This determination was consistent with results presented for Pacific ocean perch, for which F_{MSY}=0.80M. Even so, concern was expressed within the panel that a more conservative harvest rate might be warranted, such as that used by the North Pacific Fishery Management Council, which in similar swept-area applications assumes that q=1.0. In either case, given the high degree of uncertainty underlying the technical basis of this recommendation, and the real possibility that MacCall's findings are accurate, precautionary adjustments in setting the OY of the remaining rockfish are recommended.

The panel discussed the hardship to the fishing industry that the immediate application of these new, more restrictive, rates will cause. The National Standard Guidelines for implementa-tion of the Magnuson-Stevens Act specify (Department of Commerce 1998): "Overfishing occurs whenever a stock of stock complex is subjected to a rate or level of fishing mortality that jeopardizes the capacity of a stock or stock complex to produce MSY on a continuing basis." The PFMC may, therefore, wish to consider the propriety and legality of a short-term phase-in of these new rates to ameliorate the immediate impact to the groundfish industry.

Surplus Production Models

During the workshop, methods considering an examination of the relationship between surplus

production and stock biomass were discussed as potential alternatives to methods based on stock-recruit models for determining appropriate exploitation rates. The panel generally agreed that an examination of estimates of surplus production and their relationship with estimates of biomass or other variables is useful. However, the panel does not endorse the general replacement of a stock-recruitment based approach at this time, nor the requirement of using a biomass-based surplus production model as one approach for estimating MSY, F_{MSY} and B_{MSY} for all assessed stocks. The panel concluded that this is an area that could benefit from additional research.

There were three presentations dealing with biomass-based production model approaches on the agenda (Jacobson *et al*, MacCall, and Parrish; see Appendix A). The fundamental premise of these approaches was to use the output from a detailed age-structured model as an accurate representation of exploitable stock biomass (i.e., assume q = 1.0) and to estimate the relationship between catches and changes in biomass to determine production. Most of the panel concluded that this kind of approach has potential application when applied to estimates generated from age-structured or delay-difference assessments. This is possible because absolute stock biomass estimates are generally available from the assessment models and, by definition, estimated surplus production can be calculated from the time series of catch and estimated biomass. The disadvantage of this approach, however, is that the various biological processes underlying stock compensation are not directly addressed, whereas in age-structured approaches these processes can be treated explicitly. Whether surplus production is estimated internally within the model (e.g., Jacobson *et al.*) or externally after the fact (MacCall, Parrish), is an issue deserving of more study (see also results from Ianelli).

Although the full panel saw benefits to explicit consideration of biomass production implied by assessments, some panelists expressed significant reservations regarding the use of production models to determine F_{MSY} and related quantities. These reservations were largely based on the view that this approach discards important information contained in the original age-structured model results. For example, age-structure can influence production because young fish generally have higher weight-specific growth rates than older fish. As a result, the same biomass can lead to different levels of production, depending upon the age composition of the population. Likewise, changes in selectivity over time will change the amount of surplus production at a given biomass. Although such variation in surplus production could be dealt with as correlated process error (Jacobson *et al.*) this converts variation explained by the age-structured model into additional error. In any event, age-structured analyses can provide specific information on the nature of compensation (e.g., in individual growth, maturation, or recruitment), which is not possible from an examination of the aggregate surplus production-biomass relationship alone.

Other panelists argued that estimates of F_{MSY} from surplus production models might be more robust than those that depend upon solely on stock-recruitment relationships. The idea here is that (1) error in assessment model estimates of biomass may cancel-out because production estimates involve differencing model biomass estimates, and (2) potentially biased estimates of recruitment (e.g., discards of small fish) play a less critical role in the analysis. Simulations presented by MacCall at the second Groundfish Productivity Workshop in Monterey, CA suggested this was the case. However, given the few number of replicate simulations and the limited suite of scenarios in that paper, the panel did not view this work as definitive.

Estimation of B_0, B_{40} and Related Problems

Although variable rate biomass-based harvest policies were not the primary focus of the workshop, the newly implemented 40-10 harvest policy was, nonetheless, the subject of much discussion. While in practice it is possible to consider F_{MSY} proxies in isolation from biomass targets and thresholds, in principle these two subjects are inextricably linked.

The main concern about the 40-10 harvest policy is that it involves the calculation of two biomass reference points, i.e., the virgin biomass that would exist in the absence of fishing (B_0) and the exploited biomass that is 40% of that pristine level ($B_{40\%}$). Within the PFMC, it appears that parameter B_0 is usually obtained from a stock assessment model and estimates of what biomass may have been in the far past.

A number of problems are likely to occur in the estimation of this parameter. First, its estimated value may be far larger than any historical observed biomass due to vagaries of parameter estimation and the age composition of the population at the start of the data series (e.g., Pacific ocean perch; see Ianelli in Appendix A). In some cases, it may be justifiable to constrain the value of B_0 to be near the historical maximum or some other value, as long as a clear rationale is provided and the sensitivity of the constraint is examined.

A second problem is that models are frequently configured to assume that the age composition is at equilibrium at the start of the modeled period. If this assumption fails, then the estimate of parameter B_0 may be biased. Third, there is no guarantee that under any fishing mortality regime, including zero fishing, that the population will rebuild to this level. The reason for this is that the amount of recruitment needed to produce historical levels of spawning biomass may not occur in the future. Given that many West Coast stocks have been on a "one-way trip" downward, a sensible harvest policy would first reverse the decline, and then rebuild to a level that could be expected based on current and expected future conditions. Once that level of rebuilding is accomplished, it may then be possible to rebuild toward a level consistent with historical patterns.

Therefore, some alternatives for calculating B_0 that look toward the future instead of the past should probably be considered. Two clear alternatives involve determining: (1) whether a spawner-recruit model is used to project the population forward and (2) if not, what exact values of the recruitment time series are to be used in forecasting future biomass. If a spawner-recruit model is used, then it should be possible to determine pristine biomass and B_{MSY} as reference points automatically. These points can then be implemented in the harvest policy, as is done by the North Pacific Fishery Management Council. However, it is often quite difficult to assert that a reliable spawner-recruit relationship is known, so typically such a relationship would not be invoked. Nevertheless, it is often wise to provide for reduced recruitment at low spawning biomass levels, particularly if the stock has been fished down to a point where recruitment is believed to have been impacted. Some recent modeling efforts with ADMB and Bayesian considerations (e.g., Pacific hake) lend hope to better determining MSY parameters.

If a spawner-recruit relationship is not used, then a projection of future unfished equilibrium biomass can be made by multiplying contemporary recruitment values by the corresponding spawner biomass per recruit (SPR) function. For example, the average recruitment over the time series might be used with an SPR function at a fishing mortality of 0 to arrive at the expected equilibrium unfished biomass in the future, to be used as B_0. From this information $B_{40\%}$ could be obtained. This type of approach is especially appropriate if it is known there has been a change in stock productivity. A caveat to doing this, however, is that it can be very difficult to detect a change in productivity, so the rationale for restricting the time period must be carefully considered.

Whichever approach is used, it should be documented carefully and properly justified. The same methodology should be used for all biomass reference points and it should be clearly stated whether a reference point is based on SPR calculations that are fully independent of spawning biomass, or whether recruitments have been adjusted downward by a spawner-recruit relationship. We think justification for the calculation of biomass reference points should address consistency between the assumptions used in their derivation and those underlying F_{MSY} estimates or proxies.

We note that another type of calculation is required by the NMFS overfishing guidelines, which could lead to further confusion. Namely, a threshold level that provides for a 10-year rebuilding to a target level such as B_{MSY} must be found (Department of Commerce 1998). This level is also a function of the recruitment series used and depends on whether a spawner-recruit relation exists. Consequently, for consistency the same process that is used for determining other reference points should be used here. The PFMC has apparently been allowed to use $B_{25\%}$ for this threshold, but it is unclear how rebuilding plans, which are triggered when biomass drops below this value, will interface with the 40-10 rule, which in itself, is an automatic rebuilding plan. Other Councils are currently experiencing this confusion as well, so hopefully there will be more flexibility and clarity in the NMFS overfishing guidelines in the future.

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Some Relevant Unpublished Manuscripts

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APPENDIX B8: **Updated shell length/ meat weight relationships for use in the next assessment.**

For each ocean quahog assessment, biomass of meats per tow is calculated using a shell length/meat weight relationship for quahogs of any given length ($MW = e^aL^b$). Each of the assessment regions has its own set of alpha and beta parameters as meat weight at length varies by region. For the last several assessments (2000, 2004, 2007 and current), biomass of meats per tow for DMV and NJ has been calculated using SL/MW relationships from Murawski and Serchuk (1979). The clams they used were measured at sea and their meats were frozen for later weighing ashore.

During the 1997 NEFSC clam survey, quahogs from LI and GBK were measured and the meats weighed fresh on board the DEII to derive SL/MW relationships for those two areas. This new 1997 GBK relationship was used starting with the 2000 assessment. For the 2000 assessment, the parameters for LI were an average of the parameters derived from the fresh meats samples on the 1997 survey and those derived by Murawski and Serchuk (1979) from frozen meats (Table 1).

Since the 1997 NEFSC clam survey, fresh meat weights have also been collected during the 2002, 2005 and 2008 NEFSC clam surveys. We used only the lengths and fresh meat weights from these surveys to derive new SL/MW parameters for NJ, LI, SNE and GBK. Data was not collected from all regions every year, and no data was collected from SVA or DMV during any of those four surveys. We fit curves for each year the data was collected for each region, and then created an average curve for each region. These new relationships should give a more accurate and current estimate of biomass for the next assessment.

Table 1. Alpha and beta parameters for various SL/MW relationships by region and source. The years 1997, 2002, 2005 and 2005 are the years fresh meats were collected during the NEFSC clam survey, N refers to how many samples (clams) were used to fit the curve.

	SVA			DMV			NJ		
	alpha	beta	N	alpha	beta	N	alpha	beta	N
Murawski and Serchuk (1979)	-9.0423	2.7880		-9.0423	2.7880		-9.8472	2.9495	
1997									
2002							-9.4091	2.9320	117
2005							-10.0110	3.1144	155
2008							-9.6618	2.9689	324
average curves (data 1997+)							-9.6634	2.9927	
previous SARCs (2004,2007)	-9.0423	2.7880		-9.0423	2.7880		-9.8472	2.9495	

	LI			SNE			GBK		
	alpha	beta	N	alpha	beta	N	alpha	beta	N
Murawski and Serchuk (1979)	-9.1243	2.7750							
1997	-9.3102	2.8605	151				-8.8338	2.7611	72
2002				-9.0439	2.8238	158	-9.6670	2.9522	268
2005	-10.0380	3.1627	92	-9.6041	2.9108	71			
2008	-8.7270	2.5520	460	-9.5091	2.9104	243	-9.0576	2.7328	308
average curves (data 1997+)	-9.1962	2.7790		-9.3541	2.8729		-9.1276	2.7952	
previous SARCs (2000, 2004,2007)	-9.2336	2.8225		-9.1243	2.7750		-8.9691	2.7673	

The surveys in 1997, 2002, 2005 and 2008 collected SLMW data from freshly shucked meats.

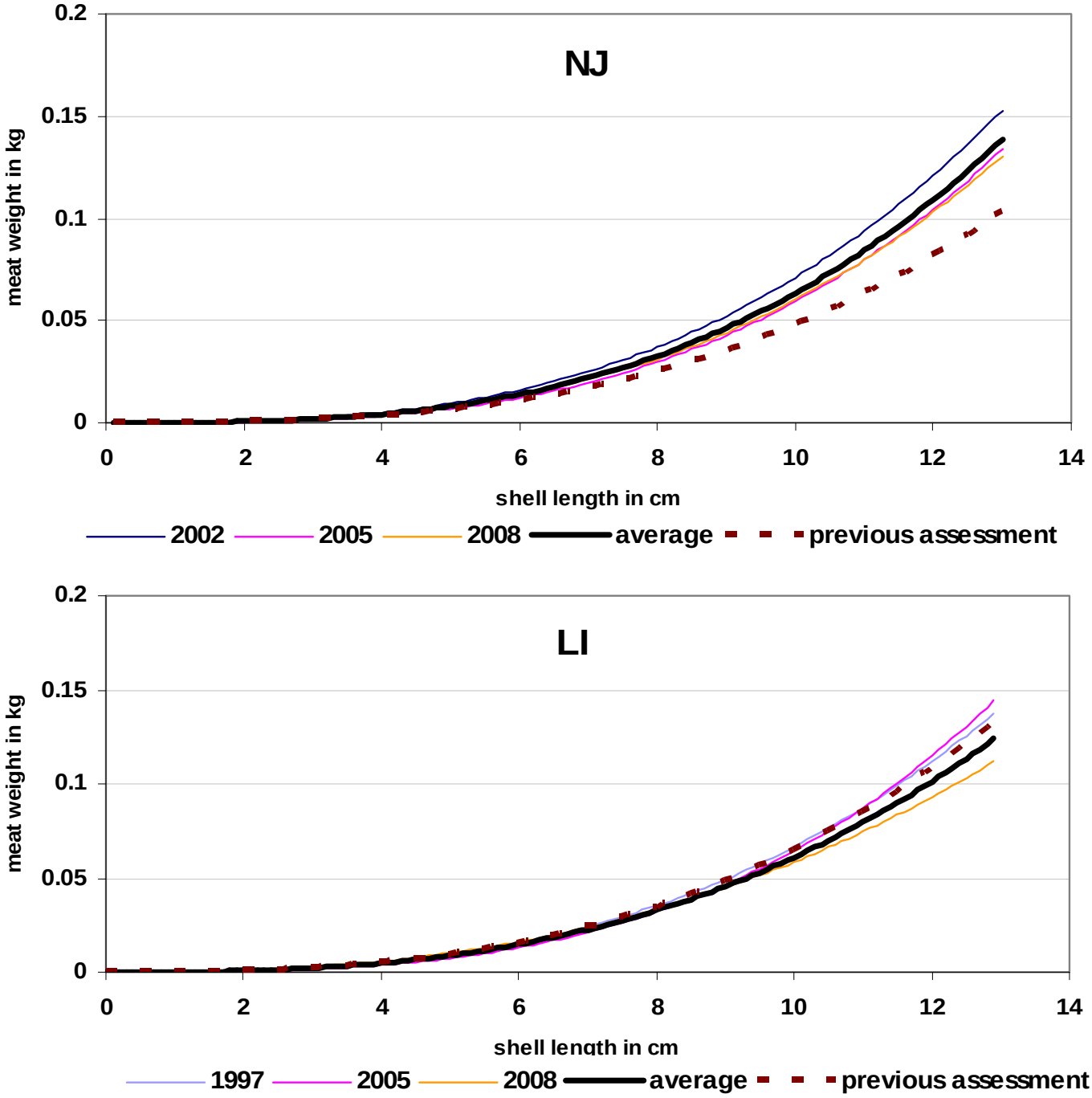

Figure 1. Shell length/ meat weight relationships for the NJ and LI assessment regions.

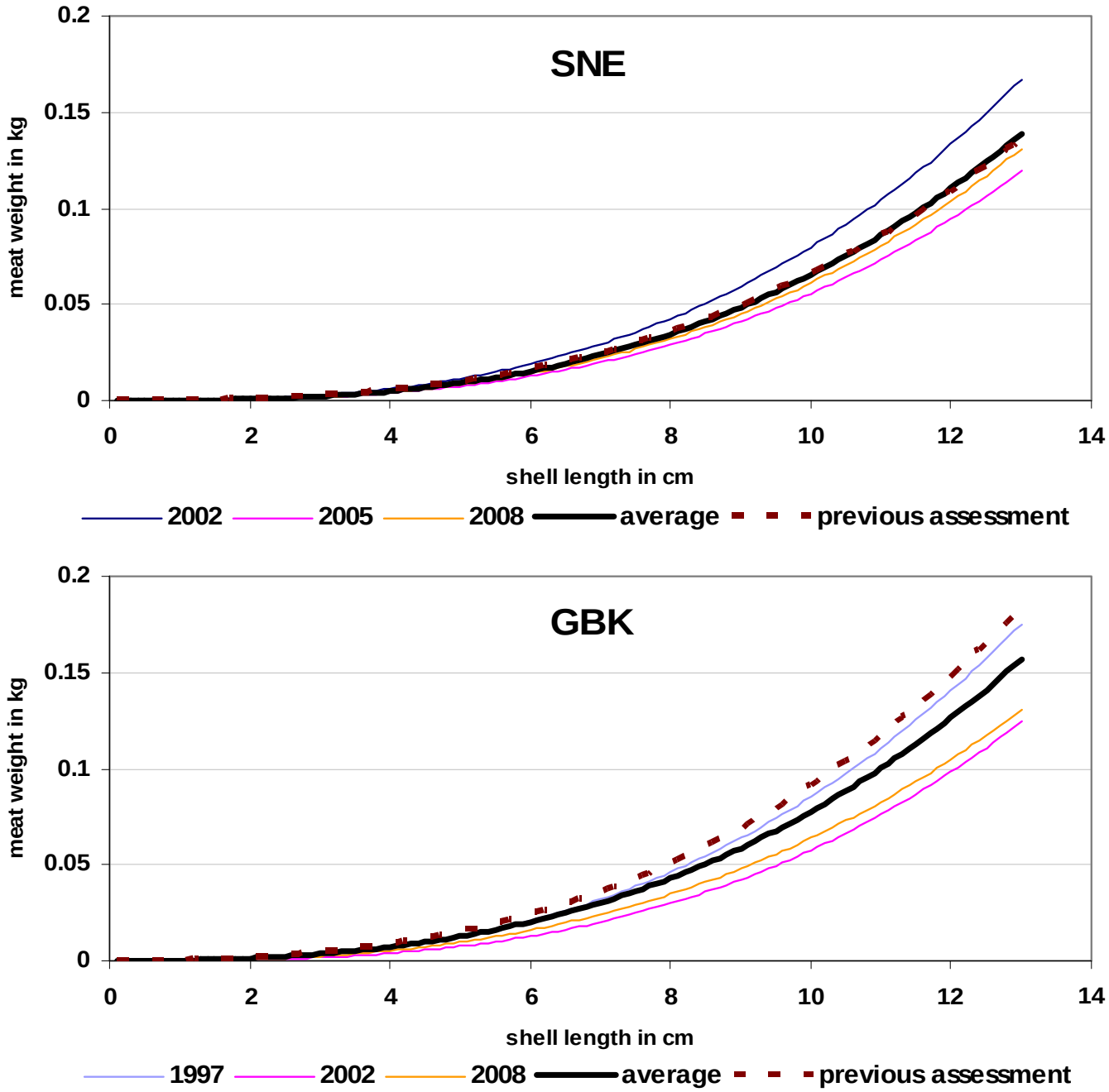

Figure 2. Shell length/ meat weight relationships for the SNE and GBK assessment regions

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WEAKFISH STOCK ASSESSMENT REPORT

A Report of the ASMFC Weakfish Technical Committee

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C2.0 Terms of Reference for Weakfish

1. Evaluate biases, precision, uncertainty, and sampling methodology of the commercial and recreational catch (including landings and discards) and effort.
2. Evaluate precision, geographical coverage, representation of stock structure, and relative accuracy of the fisheries independent and dependent indices of abundance. Review preliminary work on standardization of abundance indices.
3. Evaluate the ADAPT VPA catch at age modeling methods and the estimates of F, Z, spawning stock biomass, and total abundance of weakfish produced, along with the uncertainty and potential bias of those estimates. Review the severity of retrospective pattern.
4. Evaluate the index based methods and the estimates of F, ages 1+ stock biomass, surplus production, and time-varying natural mortality of weakfish produced, along with the uncertainty of those estimates. Determine whether these techniques could complement or substitute for age-based modeling for management advice.
5. Evaluate testing of fishing and additional trophic and environmental covariates and modeling of hypotheses using biomass dynamic models featuring multiple indices blended into a single index with and without a Steele-Henderson (Type III) predator-prey extension. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.
6. Evaluate AIC-based hypothes is testing of fishing and additional predation-competition effects using multi-index biomass dynamic models with and without prey-based, predator-based, or ratio dependent predator-prey extensions. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.
7. Review evidence for constant or recent systematic changes in natural mortality, productivity, and/or unreported removals.
8. Estimate biological reference points using equilibrium and non-equilibrium assumptions and evaluate stock status relative to these BRPs.
9. Review stock projections and impacts on the stock under different assumptions of fishing and natural mortality.
10. Make research recommendations for improving data collection and assessment.

C3.0 Executive Summary

C3.1 Major findings for TOR 1 –Evaluate biases, precision, uncertainty, and sampling methodology of the commercial and recreational catch (including landings and discards) and effort.

Weakfish fishery data were evaluated from four fishery sectors: commercial harvest, commercial discards, recreational harvest, and recreational discards. Commercial harvest data were obtained from state and federal harvest reporting systems. Commercial discards were estimated following the method of de Silva (2004) for key gear-species combinations found to be associated with relatively high rates of weakfish discards. A 100% discard mortality rate was assumed. Recreational harvest and discards were obtained from the Marine Recreational Fisheries Statistics Survey (MRFSS) conducted by the National Marine Fisheries Service. Harvest numbers and weight are directly available; discard numbers were estimated as the number of weakfish released alive times a discard mortality rate of 10% which is based on quantitative studies.

Harvest and discard estimates were stratified by region (north/south), year, and season (early = January to June and late = July to December). Commercial harvest was further stratified by state and gear. Where available, stratum specific biological data (length data and length-weight equations) were used to convert harvest and discard weights to number of weakfish removals at size. Where stratum specific data were not available (some commercial harvest strata), samples were substituted from the next most representative stratum. Numbers at size was then converted to numbers at age using region/year/season specific age-length keys. Numbers at age were summed across strata within a year to develop annual estimates of total weakfish catch at age.

Several sources of potential bias were identified that may result in uncertainty in annual catch at age estimates. These include inaccurate harvest/discard estimates as a result of under/over reporting or inappropriate survey methods; insufficient sample size to characterize length distributions; substitution of data from alternate cells in the catch at size characterization and age-length keys; errors in aging techniques or the scale-otolith age conversion; and others. Several of these sources are generic and not specific to weakfish. Attempts have been made to quantify some of these error sources; however, the extent of uncertainty associated with each of these sources, and their cumulative effect, remains largely unknown. Improvements in data collection from commercial landings have been instituted since 2000 that have greatly increased coverage and reliability of data.

C3.2 Major findings for TOR 2 – Evaluate precision, geographical coverage, representation of stock structure, and relative accuracy of the fisheries independent and dependent indices of abundance. Review preliminary work on standardization of abundance indices.

Five fishery independent age structured surveys were evaluated for use in the stock assessment. Surveys were evaluated relative to criteria such as geographic coverage, ability to accurately track weakfish abundance, and survey precision, among other factors. Catch per unit effort indices of abundance from three age-structured fishery independent surveys, including the Delaware Bay and SEAMAP trawl surveys and the North Carolina gillnet survey, were found to be suitable for use in the assessment. The North Carolina gillnet survey began in 2001, and this is the first time it has been included as a tuning index for weakfish. The NEFSC fall trawl survey, which has been used as a tuning index in previous weakfish stock assessments, was found to be unsuitable for use because of high interannual variability in catches, limited ability to capture weakfish greater

than 34 cm, and instances of negative mortality in year class catch curves. Similarly, a CPUE index based on two fall cruises of the New Jersey ocean trawl survey was found to provide little information on weakfish stock abundance; however, an alternate index based on the proportion of "positive" (*i.e.* non-zero) tows from the August cruise was found to be a suitable indicator of abundance. Ten young of year fishery independent surveys were also evaluated, one of which (Massachusetts trawl survey) was eliminated from further analysis due to exceptionally large coefficients of vavriation. Two fishery dependent indices of abundance were also included in the assessment. One is based on total catch per trip in the Mid-Atlantic private boat recreational fishery and encompasses an age aggregate index for ages 2+. The other is based on harvest per trip in the Mid-Atlantic private boat recreational fishery and is separated into age specific indices for ages 3, 4, 5, and 6+.

A team of researchers at Virginia Tech University has recently begun investigating the utility of standardizing weakfish abundance indices relative to spatial, temporal, and environmental factors using GLM and GAM methods. Although further evaluation of the methods and results is required by the Weakfish Technical Committee (WTC), preliminary results of the standardization analyses are presented in this report.

C3.3 Major findings for TOR 3 – Evaluate the ADAPT VPA catch at age modeling methods and the estimates of F, Z, spawning stock biomass, and total abundance of weakfish produced, along with the uncertainty and potential bias of those estimates. Review the severity of retrospective bias.

Age structured modeling was conducted using ADAPT VPA. Various runs were conducted using different sets of tuning indices. The different runs were evaluated with respect to model fit, residuals, and retrospective patterns. All runs produced consistent estimates of parameter values for the years 1982 to 2002. Trends in estimated parameters for the years 2003 to 2007 varied widely, and were confounded by a prominent retrospective pattern. Fishing mortality was generally underestimated, while biomass and abundance parameters were over estimated. Model runs that included fishery dependent indices had smaller retrospective bias and better model fit (lower mean squared residual (MSR)) than runs tuned solely with fishery independent indices. Runs that included young of year indices had extended retrospective patterns but inconsistent effect on MSR. The run tuned solely with fishery dependent indices produced the best fit and minimal retrospective pattern; but inclusion of fishery independent indices from New Jersey, Delaware, and North Carolina did not substantially increase the retrospective pattern and produced the second lowest MSR of all runs investigated. This run was therefore selected as the preferred run. These indices correspond well with changes in harvest, abundance, CPUE, and population age structure. Terminal year estimates were estimated as $F_{2007} = 0.51$ (unweighted, ages 4-5) and $SSB_{2007} = 7{,}236$ MT, although these were poorly estimated given the observed retrospective pattern. Attempts to correct for retrospective pattern were conducted but were not specifically endorsed by the WTC. Because ADAPT VPA calculates fishing mortality as F = Z-M, estimates of F are dependent on input values of natural mortality. The WTC has expressed concern regarding the assumption of constant natural mortality of M = 0.25 across all ages and years. To circumvent the concerns regarding input M, the WTC prefers to combine model estimated F rates and input M values to portray the trend in total mortality, Z. Following record low levels in the mid 1990s, total mortality increased dramatically and exceeded Z = 2.6 in 2003. Z has declined in recent years to $Z_{2007} = 0.76$, but values in recent years are likely underestimated given the observed retrospective pattern.

C3.4 Major findings of TOR 4 – Evaluate the index based methods and the estimates of F, ages 1+ stock biomass, surplus production, and time-varying natural mortality of weakfish produced, along with the uncertainty of those estimates. Determine whether these techniques could complement or substitute for age-based modeling for management advice.

Because of systematic retrospective bias exhibited in recent F and stock biomass (mt) estimates from the catch-at-age (ADAPT) model, the 2006 Weakfish Assessment (Kahn et al 2006, Uphoff 2006a; Crecco 2006) relied primarily on an index-based (ages 1+) approach to monitor temporal changes in weakfish biomass (mt) and fishing mortality (F) from 1981 to 2003. Given that the index-based approach produced F and weakfish stock biomass (mt) estimates that displayed a similar trend to that from the converged portion (1982-1999) of the 2006 VPA, F and biomass estimates were updated with this approach through 2008 using an annual blended index based on the recreational private boat cpue, as well as on the New Jersey and Delaware trawl indices. The index-based approach was also used to estimate the magnitude and trend in ages 1+ weakfish surplus production from 1981 to 2008.

Biomass weighted fishing mortality (FWt) estimates on ages 1+ weakfish from the index-based approach were high (FWt range: 0.69- 1.16) by most standards from 1981 to 1987. The magnitude of FWt estimates, however, rose even higher to beyond 1.0 from 1988 to 1991, and greatly exceeded our current overfishing threshold for weakfish (Fmsy = 0.53). The magnitude of FWt estimates declined steadily thereafter to below 0.60 in most years from 2000 to 2008. The ages 1+ fishing mortality rates weighted by number (FNt) were almost always lower in magnitude than the corresponding biomass weighted fishing rates. The ages 1+ numbers weighted (FNt) estimates from 1981 to 2008 followed a similar trend over time as the biomass weighted FWt estimates, but unlike the biomass weighted fishing rates (FWt), the FNt estimates fell abruptly after 1991 and remained below 0.30 from 1993 to 2008.

Weakfish ages 1+ biomass levels from the index-based approach exhibited wide contrast from 1981 to 2008. Weakfish ages 1+ biomass (mt) remained relatively high (14,200 and 41,500 mt) from 1981 to 1988 but biomass levels fell steadily to below 10,000 mt from 1989 to 1993. Weakfish coast-wide biomass rose again temporarily from 1994 to 1996, but biomass fell steadily thereafter to the lowest level in the time series in 2008 (1,333 mt). The time series of weakfish ages 1+ surplus production (SURPt) from 1981 to 2008 followed the same general trend as stock biomass. Weakfish surplus production remained relatively high from 1982 to 1986 and again in 1993 and 1994, but SURPt levels fell steadily after 1995 and remained very low in most years from 2001 to 2008 despite relatively low and stable fishing mortality. The unexpected drop in weakfish surplus production after 1999 coincided with a sharp rise in the coast-wide abundance of two potential predators: striped bass and spiny dogfish.

C3.5 Major findings of TOR 5 – Evaluate testing of fishing and additional trophic and environmental covariates and modeling of hypotheses using biomass dynamic models featuring multiple indices blended into a single index with and without a Steele-Henderson (Type III) predator-prey extension. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and

whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.

Since the index-based approach produced 1982-1998 ages 1+ F and weakfish stock biomass estimates that were similar to those over the converged portion (1982-1998) of the 2006 VPA, this approach was used to update ages 1+ F and stock biomass through 2008 using the recreational private boat cpue, as well as New Jersey and Delaware trawl indices. In addition, ages 1+ surplus production estimates were derived from 1981 to 2008 from which steady-state overfishing (Fmsy, Bmsy) thresholds were derived for Atlantic coast weakfish. Third, the age aggregated (ages 1+) Steele and Henderson (1984) (S-H) production model was updated through 2008 to further examine the joint effects of fishing and predation from striped bass (*Morone saxatilus*) and spiny dogfish (*Squalus acanthias*). The results from the S-H model were also used to estimate equilibrium and non-equilibrium *Fmsy* and *Bmsy* thresholds. Fourth, to provide a more thorough examination of the Predation Hypothesis, additional candidate predators such as bluefish (*Pomatomus saltatrix*) and summer flounder (*Paralichthys dentatus*) were also considered as candidate predators on weakfish, especially since both finfish predators have risen sharply inshore along the Atlantic coast after 1998. Finally, environmental disturbances have been proposed as a major process governing shifts in finfish production and recruitment (Hollowed et al 2000b), so environmental factors such as decadal shifts in mean sea surface water temperature and deviations in the winter North Atlantic Oscillation Index were also examined as potential explanatory variables.

The preponderance of statistical evidence given here supports the Predation Hypothesis involving enhanced predation by striped bass and spiny dogfish as the primary factor behind the recent and unexpected decline in weakfish productivity. Statistical evidence in support of the Predation Hypothesis consists of a significant ($P < 0.0001$) inverse correlation between declining weakfish biomass and surplus production from 1999 to 2008 and striped bass and spiny dogfish abundance from 1982 to 2004. Striped bass abundance along the Atlantic coast rose 10 fold from 1994 to 2006 (Kahn 2005), although the 2008 striped bass abundance estimate fell by over 40% since 2006. Similarly, spiny dogfish abundance has increased 10 fold since 1999 and has remained high thereafter. During this recent period (1999-2008) of declining weakfish productivity, fishing mortality (FW) and discard mortality (Fdisc) rates remained low and relatively stable, indicating that the recent drop in weakfish productivity did not coincide with rising exploitation. The strong positive correlation (Pearson $r = 0.91$, $P < 0.0001$) between the recent rise in weakfish juvenile mortality (Z0) and rising striped bass and spiny dogfish abundance further suggests that the recent emergence of a weakfish recruitment bottleneck at age 0 was largely due to enhanced predation by these two finfish predators. By contrast, discard mortality rates on small (< age 2) weakfish remained low and stable after 1999 during which juvenile mortality (Z0) rose steadily. Third, the residual patterns in all Logistics and Gompertz model runs that included only fishing effects (landings) produced inordinately low overfishing thresholds (Fmsy, Bmsy), poor precision around the estimates, and the residuals exhibited a pronounced serial correlation over time, clearly indicating model misspecification. However, when the predation term (Tpred), reflecting the joint predation by striped bass and spiny dogfish, was added to the models, the fit of the models to weakfish surplus production and biomass dramatically improved, the precision and magnitude of Fmsy and Bmsy rose to more plausible levels (Fmsy > 0.45), and, most importantly, the direction of the residuals over time shifted to a more random pattern and were therefore free of model misspecification.

When the equilibrium overfishing thresholds (Fmsy = 0.72, Bmsy = 17,009 mt) from the S-H model are considered, ages 1+ biomass weighted fishing mortality (FW) on weakfish exceeded the

estimated Fmsy threshold of 0.72 in most years from 1981 to 1992. Weakfish coast-wide biomass (mt) exceeded the biomass threshold (Bmsy = 17,009 mt) in 1981 and 1982, but biomass fell quickly below Bmsy thereafter. When more stringent management regulations were enacted after 1991, fishing mortality (FW) fell by 50 to 70% and biomass began to rise toward the Bmsy threshold. However, weakfish biomass fell unexpectedly after 1999 to the lowest level in the time series in 2007 despite the fact that fishing mortality rates remained below Fmsy in most years from 1998 to 2008 (exception: 2002).

Although most of the statistical and empirical evidence given here and elsewhere (see TOR #6 this assessment) supports the Predation Hypothesis, other factors such as unreported commercial and recreational landings, disease, toxins and parasitism cannot be ruled out at this time to explain the annual production loss of between 3,000 and 5,000 mt of weakfish. There is no evidence thus far that would link recent increases in disease, toxins and parasitism to the recent failure of weakfish. There has been a recent rise in sea surface water temperatures along the Atlantic coast (Oviatt 2004), but these analyses indicated that water temperature shifts were not significantly ($P < 0.05$) linked to recent increases in weakfish juvenile mortality (Z0), nor in the decline in weakfish surplus production and stock biomass. It is possible that an enormous upsurge in unreported weakfish landings and commercial and recreational discards took place between 1996 and 2008 to account for the estimated 3,000 to 4,000 mt annual loss of weakfish surplus production, but a recent upsurge in unreported landings seems unlikely for several reasons. First, if the sources of this rapid upsurge in unreported weakfish landings and discards are thus far unknown, it would be nearly impossible to remove this source of mortality without closing virtually all inshore fishing activity between North Carolina and Rhode Island. Second, if a recent rise in unreported landings and discards resulted in the recent weakfish stock collapse, we would expect that other finfish stocks with a similar temporal and spatial distribution as weakfish (i. e. Atlantic croaker and summer flounder) to be likewise depleted. But Atlantic croaker and summer flounder stocks have either grown or have remained relatively stable from 1998 to 2008. Third, if a recent rise in weakfish unreported landings caused the recent weakfish stock collapse, all of the statistical and empirical evidence presented elsewhere in this assessment on enhanced predation would have to be regarded as a mere coincidence. Finally, unreported landings in the order of 3,000 to 4,000 mt annually are equivalent to about 5 times the current (2007-2008) known landings and estimated discards used in this weakfish assessment. The possibility that such an astronomical rise in unreported landings and discards took place after 1998 and then remain unnoticed by port agents, enforcement and management agencies seems remote. The management implications associated with a rise in predatory mortality on weakfish are also discussed.

C3.6 Major findings of TOR 6 – Evaluate AIC-based hypothesis testing of fishing and additional predation-competition effects using multi-index biomass dynamic models with and without prey-based, predator-based, or ratio dependent predator-prey extensions. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.

The most reliable estimates of trends or values of F from the previous weakfish *Cynoscion regalis* assessment indicated it had been modest since at least 1995, while weakfish abundance and

surplus production declined to low levels, most likely due to increased natural mortality. Hypotheses featuring fishing, environmental conditions, forage abundance, competition, and predation were examined and two strong covariates emerged: Atlantic menhaden *Brevoortia tyrannus* (forage) abundance and predation- competition from striped bass *Morone saxatilis*.

In this 1981-2006 assessment of weakfish, biomass dynamics models were used to test multiple hypotheses about fishing alone or fishing plus interactions with striped bass (alone or influenced by Atlantic menhaden) as the cause of the recent failure of weakfish recovery. Spatial, temporal, and diet overlaps were sufficient for interactions. Logistic and Gompertz production functions were considered and six models were developed for each function. These models considered fishing alone and fishing in combination with five predator-competitor functions; three models considered prey-based Type I, II, and III functional responses, while two explicitly mimicked depensatory mortality by considering additional natural mortality solely as a function of striped bass biomass or as a function of striped bass biomass and the ratio of Atlantic menhaden to striped bass biomass.

We used three exploitable biomass indices (EBI; indices of weakfish 250 mm or greater) to evaluate biomass dynamics during 1981-2006: mid-Atlantic private/rental boat catch per trip (WRI; as biomass and estimated from MRFSS), DE (1990-2006), and NJ (1989-2006) trawl survey EBIs. Biomass dynamic models used total weight of aggregated harvest (NMFS estimates) and discards by both the commercial and recreational fisheries. The Weakfish Technical Committee (WTC) considered all estimated commercial discards to have died and updated ratio-based estimates of commercial discards using the general method of De Silva (2004), but used all years combined rather than annual estimates (latter were variable and imprecise). Discard sampling was not conducted until 1994 and market-related discard ratios estimated for 1994-2002 were used to estimate weakfish discards in prior years. Estimates after 1993 used total discard ratios (market + regulatory). Recreational discard losses were estimated as MRFSS number discarded*mean weight *mortality. The MRFSS does not estimate weight of released weakfish and discard mean weight (0.15 kg) estimated from MRFSS 2004-2007 headboat surveys was substituted for harvest mean weight used in the previous assessment. Weakfish hook-and-release experiments produced dichotomous mean mortality estimates (≈3% or 15%) and 10% release mortality was adopted by the WTC.

We used Akaike information criteria adjusted for small sample size, AIC_c, to evaluate the 12 hypotheses. AIC_c indicated a 98% chance that the Gompertz production model with a depensatory function relating D_t (weakfish biomass lost to striped bass predation-competition) to the biomass of age 2+ striped bass and the ratio of menhaden to striped bass biomass was best (Gompertz Depensatory Ratio model or GDR) given the data. This model explained 90% of the variation in EBI. Fishing only models were poor choices for describing biomass dynamics of the data, ranking seventh and ninth out of twelve.

Annual (year t) estimates of F_t and M_{pt} (instantaneous natural mortality rate due to striped bass predation-competition) were combined to estimate Z_{pt} (total mortality excluding residual M). Loss of weakfish biomass per unit of striped bass biomass (D_t / P_t) was estimated. Biomass estimates (B_t) provided a basis for estimating surplus production and production that accounted for losses due to striped bass predation and competition.

Equilibrium biological reference points (EBRPs) were estimated (F_{msy} and B_{msy}) and two approaches were used to estimate predator-competitor reference points (NBRPs) for mortality when predation-competition losses were included: total mortality at maximum sustained yield (Z_{msy}) and non-equilibrium F_{msy} (or $F_{psyt} = F_{msy} - M_{pt}$). The former simply involved renaming equilibrium

estimates of F_{msy} as Z_{msy} in models with predator-prey terms and comparing it to Z_{pt}. Although biomass dynamic models do not provide SSB thresholds explicitly, the early maturity of weakfish allowed B_t / K to serve as a proxy for MSP to compare to the target and threshold. Amendment 4 to the Interstate Fishery Management Plan for Weakfish (ASMFC 2002) lists SSB that is 20% of an unfished stock as a maximum spawning potential (MSP) threshold and 30% as a target.

BRPs were F_{msy} or Z_{msy} = 0.48 and B_{msy} = 18,941 mt based on r = 0.48, K = 51,521 mt. Jackknifing and bootstrapping indicated all parameters were precisely estimated and parameter values of the base run were very close to their medians, indicating minimal bias. Several approaches were used to investigate sensitivity of model parameters and estimates of Z_t, F_t, and B_t. Estimates of B_t were standardized to K (B_t / K) and estimates of Z_t and F_t were standardized to Z_{msy} in sensitivity analyses. Sensitivity of model parameters to data from the beginning or ending of the time-series was tested by removing blocks of data and rerunning the model. Biomass estimated in 1981 for the 1981-2006 time-series was greater than K and we ran a version of this model with B_{1981} constrained to be less than K for comparison. Sensitivity to different assumed recreational discard mean weights was tested as well because of their importance in estimating WRI.

Overall, differences in B_t / K, Z_t / Z_{msy}, and F / Z_{msy} stabilized by 1983 among all initial time-block removal treatments. Biomass dynamics of weakfish were portrayed similarly. We chose to keep the results of the unconstrained GDR (all years), but removed 1981 and 1982 estimates from consideration. Removing up to three years from the end of the time-series or changing mean weight of recreational discards had little impact.

Biomass dynamic modeling indicated weakfish biomass in 2006-2007 was depleted well below its threshold, the stock was not overfished based on equilibrium F_{msy}, but was subject to high natural mortality that eroded the safe level of fishing. The proxy for MSP (2007 MSP = 6%) was far below the 20% threshold in Amendment 4, while high negative values of F_{psyt} (non-equilibrium F_{msy}) and surplus production indicated that complete (and unlikely) elimination of harvest and bycatch would not be sufficient to end the decline. Production persisted at a modest level in recent years, although it was falling gradually. Depensatory mortality, driven by high striped bass biomass and a low ratio of Atlantic menhaden to striped bass (an indicator of low feeding success on striped bass' main prey), appeared the most likely explanation for increasing natural mortality that undermined recovery given the data, hypotheses, and models developed. The GDR indicated that as menhaden have become less abundant and striped bass more-so, striped bass searching has lead to increasing encounters with vulnerable-sized weakfish (up to 400 mm) searching for smaller prey-fish (anchovies and age 0 menhaden) that are also found in diets of the largest striped bass. High natural mortality of weakfish in recent years was derived from very low loss per striped bass applied over a large striped bass biomass and was independent of weakfish biomass over the years modeled. Fishing played a secondary role in recent biomass dynamics. Striped bass predation-competition dominated weakfish biomass dynamics after overharvesting had been controlled in the early 1990s. Predation-competition from striped bass has increasingly eroded weakfish surplus production and F_{msy} reference points and there seems little chance of restoring weakfish by manipulating its fisheries alone because F has become a low fraction of total mortality (20% by 2006). At this time, leverage for manipulating weakfish may mostly reside in the menhaden to bass ratio; however, it can be difficult to predict the effects of fishing or culling policies from fairly simple representations of predation processes.

Additional regression analyses reinforced the high potential for striped bass, menhaden, and weakfish linkage. Predicted mean weight of weakfish at 340 mm had undergone a significant decline during 1992-2006, and was negatively related to striped bass biomass and positively related

to the ratio of menhaden to striped bass biomass. This would be consistent with the expected effects of intense competition. Estimates of D_t / P_t were strongly related to field-based estimates of feeding success of striped bass in coastal VA and NC during winter. During 1959-2006, weakfish commercial harvest and the DE PSD Q+ length quality index closely followed the ratio of Atlantic menhaden to striped bass and correlations were strong ($\rho \approx 0.82$). These associations indicated that this ratio was important in dynamics of weakfish beyond the period covered by the GDR.

Additional Gompertz biomass dynamic models mimicked various commercial bycatch scenarios (additional losses increasing as functions of time, a constant multiple of bycatch estimates, or constant additional weight) that imposed additional losses after 1995 to reflect regulatory discards. Best models of bycatch scenarios invoked about the same additional biomass loss as estimated by GDR. The failure of recovery since the late 1990s cannot be attributed to overfishing unless bycatch and under-reported catches were much greater than estimated, growing from about 3-4 times the estimates in 1996 to 15-20 times by 2006. If results of hypothetical bycatch scenarios are taken at face value, then weakfish regulations created this massive boost in discards and represent a colossal management failure. Implementation of further conservation measures short of a coast-wide moratorium on all Atlantic coast fisheries would not minimize this nominal discard problem. There is no evidence available thus far of an Atlantic coast fishery capable of generating additional unreported weakfish losses of this magnitude.

C3.7 Major findings of TOR 7 –Review evidence for constant or recent systematic changes in natural mortality, productivity, and/or unreported removals.

During development of the 2006 weakfish stock assessment, the WTC noticed an unexpected decline in stock size at low levels of fishing mortality that had previously resulted in stock growth. Further investigation indicated that weakfish had been experiencing increased predation/competition that could be a major driving force in stock dynamics. This section presents updates on analyses investigating these multispecies hypotheses and evaluates additional sources of data that provide support for the theory that decreased production has contributed to recent stock declines.

One of the major concerns the WTC has expressed regarding the age structured modeling is the assumption of constant natural mortality across all ages and years. As an alternative, the WTC has investigated trends in relative fishing mortality which is not influenced by assumptions regarding M. Relative F, calculated as the ratio between annual harvest and an annual index of abundance, was rescaled to instantaneous rates using a scalar vector based on a short time series of F rates from the converged portion of the ADAPT VPA. Rescaled relative F follows a similar pattern to VPA estimated F from 1982 to 1998. Following 1998, however, F_{VPA} follows a nearly exponential increase while relative F remains stable at moderate levels. This discrepancy suggests that estimates of natural mortality in the ADAPT input are inaccurate.

In addition, biomass dynamic modeling was conducted to investigate possible environmental and ecosystem covariates that might be influencing weakfish stock dynamics. Two similar but independent analyses were pursued using simple (fishing only) and extended (fishing plus ecological covariates) production models. Production models incorporating only fishing effects provided poorer model fits and greater parameter uncertainty relative to the extended models. Of the extended models, the "predation" model indicated that weakfish natural mortality had increased during the last decade as the biomass of two predators (striped bass and spiny dogfish) had increased, while the "forage" model concluded that increased striped bass abundance in conjunction with a decline in their primary forage (menhaden) had resulted in increased weakfish mortality. In

both cases, the increased losses from predation/competition eroded weakfish productivity while fishing mortality remained relatively low.

Finally, the WTC investigated additional data sources that could indicate changes in weakfish productivity. Commercial landings, as a proxy for weakfish abundance, were correlated with the Atlantic Multidecadal Oscillation, a time series of sea surface temperatures from the North Atlantic that exhibit a 65 to 70 year oscillation. Strong correlations between the two time series over more than 70 years suggest that weakfish abundance may be influenced by environmental parameters such as temperature. Qualitative analysis extends the association several more decades. Weakfish food habit data obtained from the NEFSC Food Habits Database showed a shift in prey items from forage fish and large invertebrates to smaller invertebrates and an increased incidence of empty stomachs during the 1990s. The incidence of empty stomachs is strongly correlated with total mortality estimated by ADAPT VPA. These data are consistent with shifts in weakfish diets observed during the same period in the Chesapeake Bay, and suggest that weakfish productivity may have been compromised during the 1990s as primary prey items were less available.

Taken as a whole, there are several pieces of evidence that suggest that weakfish productivity underwent a shift during the 1990s, either directly (*e.g.* predation) or indirectly (*e.g.* shift in environmental conditions).

C3.8 Major findings of TOR 8 – Estimate biological reference points using equilibrium and non-equilibrium assumptions and evaluate stock status relative to these BRPs.

Weakfish are currently managed relative to reference points developed under Amendment 4 of the Weakfish Fishery Management Plan. Reference points were updated for this assessment using a spreadsheet based model using age-specific input values and length-weight-age relationships. Fishing mortality reference points were found by solving for F rates that provided spawning stock biomass of 30% (target) and 20% (threshold) relative to unfished stock. Assuming constant natural mortality of M = 0.25 and partial recruitment equal to the average of the most recent three years estimated by ADAPT, new fishing mortality reference points were estimated as $F_{target} = F_{30\%} = 0.28$ and $F_{threshold} = F_{20\%} = 0.42$, a decrease of 10% and 16% respectively relative to Amendment 4 reference points. Similarly threshold biomass declined nearly 10% to the new estimate of $SSB_{20\%}$ = 10,179 MT. Comparison of VPA based parameter estimates to these reference points indicates that weakfish are overfished and overfishing is occurring.

The WTC has expressed concern with a few aspects of the ADAPT VPA, including a prominent retrospective pattern in recent years and the use of a constant input natural mortality rate, both of which could affect reference point estimation. The retrospective pattern generally overestimates abundance for the last five years, resulting in underestimated partial recruitment. Using a partial recruitment vector from a more stable portion of the time series (1999 to 2001) decreased F reference points even further relative to Amendment 4. In addition, recent analyses indicate that natural mortality has increased substantially over the last decade to values approaching M = 1.0 or higher. Reference points calculated using an input of M = 0.8 and the more stable partial recruitment values provides estimates of a fishing mortality target of $F_{30\%} = 0.78$, and a fishing mortality threshold of $F_{20\%} = 1.20$. Under these assumptions, F_{2007} is below the target mortality rate, but is likely underestimated given the observed retrospective pattern.

In addition to the equilibrium reference points calculated based on ADAPT output, both of the biomass dynamic models investigated during this assessment produced estimates of equilibrium and

non-equilibrium MSY reference points. The full Steele-Henderson model with predation terms provided estimates of equilibrium F_{MSY} = 0.72 and B_{MSY} = 17,009 mt. The best fit model evaluated under the forage hypothesis resulted in equilibrium reference points of F_{MSY} = 0.48, and B_{MSY} = 18,941 MT. Nonequilibrium reference points calculated by both models indicate that F_{MSY} has declined in recent years as predation/competition has eroded the amount of weakfish productivity "available" to fishing. Total biomass (as proxy for SSB) has declined to less than 10% of MSY thresholds in 2007 for both models.

C3.9 Major findings of TOR 9 – Review stock projections and impacts on the stock under different assumptions of fishing and natural mortality.

Projections were conducted for each of the three main models investigated. Relative to output from the ADAPT VPA, biomass was projected for 25 years using the AgePro (version 3.1) module of the NFT Toolbox. Multiple runs were conducted assuming a range of fishing and natural mortality values assuming recruitment followed an empirical distribution of Age 1 numbers estimated through the full time series of the VPA. All projection runs resulted in increased biomass over time but reached different asymptotic values depending on assumptions regarding F and M. Under the assumption of constant M = 0.25 and a harvest moratorium, SSB is projected to increase to more than 275,000 MT by the year 2032; however, increasing mortality to M = 0.75 results in a biomass projection of 45,000 MT by 2013. Projections at given M values are lower if harvest is allowed.

For the predation hypothesis model, weakfish relative spawning stock biomass (TSSB) projections were made from 2010 to 2020 following the imposition of a simulated coast-wide moratorium (F = 0) to harvest beginning in 2009. Given the uncertainty surrounding the current and future trend in natural mortality (M) estimates, the following three scenarios that bracket a wide range of possibilities were examined with the weakfish Harvest Control Model (HCM) following a simulated 2009 moratorium to harvest: 1) M is fixed at 0.25 throughout the time series (1980-2020) as in ADAPT, there is no recent rise in trophic impacts on weakfish productivity, and recent (1999-2008) fishing mortality (F) has remained high (F > 1.0) as per ADAPT; 2) there is a moderate rise in M (from 0.25 to 0.65) from 1999 to 2020 due to predation but the magnitude of predatory mortality is less than predicted by the Steele-Henderson Model, and recent (1999-2008) F estimates have risen to moderate (0.7 to 1.0) levels; and 3) M on weakfish after 1997 has risen four-fold in magnitude (from 0.25 to 1.0) as per the Steele-Henderson Model, and fishing mortality (F) rates have remained relatively low (F < 0.50) from 1996 to 2008 as per the Index-based Analysis. In Scenario #1 under a relatively low and fixed natural mortality (M = 0.25) throughout the time series (1980-2020), the HCM predicted that a moratorium to all weakfish harvest (F = 0) enacted in 2009 and thereafter would lead to rapid TSSB recovery that would approached the Bmsy threshold by 2020. In scenario #2 under the assumption of a moderate rise in M from 0.25 to 0.65 after 1997, the HCM predicted that a moratorium (F = 0) enacted in 2009 would result in some measurable TSSB rebuilding by 2020, but the magnitude of stock growth would fall far short of the Bmsy threshold. In scenario #3 under a pronounced rise in M from 0.25 to 1.0 ostensibly due to enhanced predation, the HCM predicted that a moratorium to harvest in 2009 and thereafter would result in little if any TSSB rebuilding by 2020.

For the forage hypothesis model, jackknife and bootstrap estimates of parameter estimates from the best fit model and biomass in 2007 were projected to evaluate the effect of fishing restriction scenarios through 2015. Three fishery management scenarios were portrayed: an

approximation of the minimum F reduction in Amendment 4 ($F_{recover}$), F = 0.2 (a literal interpretation of Amendment 4), and a moratorium.

Stock recovery was not possible under $F_{recover}$ and F = 0.2, and there was about 1.4% chance of recovery under moratorium conditions for bootstrap runs and 0% chance for jackknife moratorium scenarios. It should be noted that estimates of F since 2003 have been at or below $F_{recover}$. Jackknifing and bootstrapping indicated greater than 90% chance that weakfish biomass would fall to zero by 2015 even under a moratorium if trophic conditions prevailing in 2006 continued. These projections are excessively grim and are considered a worst case scenario.

C3.10 Major findings of TOR 10 – Make research recommendations for improving data collection and assessment.

The list of prioritized research recommendations presented in the 2008 Weakfish Fishery Management Plan Review was updated by the WTC. Several recommendations were identified as completed or under investigation, while several new recommendations were identified and added to the list.

C4.0 Introduction

This is the first update to the weakfish stock assessment since 2006 when the assessment was peer reviewed through the Atlantic States Marine Fisheries Commission (ASMFC) External Peer Review process. The 2006 assessment updated the stock through the 2003 fishing season. The current assessment includes harvest data and survey indices through 2007.

C4.1 Management Unit Definition

Weakfish stocks on the U.S. Atlantic coast are managed through the ASMFC Interstate Fishery Management Plan (FMP) for Weakfish. Under this FMP, weakfish are managed as a single unit stock throughout their coastal range. Historically, all states from Massachusetts through Florida had a declared interest in the species. Currently, however, Massachusetts, Connecticut, South Carolina, Georgia, and Florida maintain *de minimus* status, and are therefore exempt from certain regulatory and monitoring requirements.

C4.2 Management History

The following is a brief review of the history of weakfish fishery management through the ASMFC. Additional details are provided in the various amendments and addenda to the original Weakfish Fishery Management Plan, which are available online at *www.asmfc.org*.

The first fishery management plan for weakfish was implemented by ASMFC in 1985 to address stock declines, bycatch concerns, the lack of sufficient data for management, and interstate user conflicts (Mercer 1985). The management measures under the FMP were voluntary, and no state implemented the full set of management provisions outlined in the FMP.

Amendment 1, adopted in 1991, established a target fishing mortality rate of $F_{20\%} = 0.34$ (Seagraves 1991). This target was to be achieved by a 52% reduction in directed harvest over the course of four years, as well as a 50% reduction in bycatch mortality in the penaeid shrimp fisheries by 1994. Although adoption of turtle excluder devices (TEDs) in the shrimp fishery led to bycatch reductions, none of the states with directed fisheries adopted the full complement of regulations recommended in the amendment.

Continued concern regarding the status of the weakfish stock was a major impetus for the development and passage of the Atlantic Coastal Fisheries Cooperative Management Act (1993),

which made compliance with ASMFC fishery management plans mandatory for member states. Following the Act's passage, the ASMFC approved Amendment 2 to the Weakfish FMP for implementation in April 1995 (ASMFC 1994). The provisions of Amendment 2 were mandatory and included harvest control strategies such as a 12" (305 mm) total length (TL) minimum size, maintenance of existing minimum mesh sizes, and a 50% shrimp trawl bycatch reduction requirement by 1996. Fishing mortality would be reduced in a stepwise fashion, with a 25% reduction in weakfish fishing mortality in 1995 followed by a 25% reduction in exploitation in 1996.

Following implementation of Amendment 2, below average fishery catch rates and spawning stock biomass continued, along with a lack of older fish. In response, Amendment 3 was developed to reduce fishing mortality to $F = 0.50$ by the year 2000, restore an expanded age structure, and restore fish to their full geographical range (ASMFC 1996). Commercial fisheries were regulated by a combination of season and area closures, mesh regulations to minimize harvest of fish less than 12" TL, and stricter requirements for bycatch reduction devices (BRDs). The minimum recreational requirements were a 12" TL minimum size limit and four fish possession limit. States were allowed to implement alternate size and bag limit regulations if they were conservationally equivalent to the minimum requirements. Bag limits were not required for minimum sizes of 16" TL or greater.

In 2000, a peer review of a stock assessment with data through 1998 indicated that weakfish biomass was high and fishing mortality rate was below the target of $F = 0.50$. Despite being ahead of schedule, it was recommended that low fishing mortality rates be continued to maintain an appropriate spawning biomass and promote expansion of stock size and age composition. Also as a result of the assessment, the WTC recognized several inconsistencies between management practices and stock dynamics. These could only be addressed through the development of a new FMP amendment. In the meantime, however, Addendum I to Amendment 3 was passed to maintain existing regulations until approval of the new amendment.

Weakfish stocks on the U.S. Atlantic coast are currently managed under Amendment 4 to the FMP (ASMFC 2002). Reference points established in Amendment 3 were too high to ensure sufficient spawning stock biomass, and the reference period used to develop recreational management measures represented an overexploited stock (insufficient abundance of older, larger individuals). In response to these concerns, Amendment 4, implemented in July 2003, established new fishing mortality and spawning stock biomass reference points, and adjusted the reference period to a period of greater stock health (1981 to 1985). Amendment 4 established new reference points: a fishing mortality target of $F_{target} = F_{30\%} = 0.31$; a fishing mortality threshold of $F_{threshold} = F_{20\%} = 0.5$; and a spawning stock biomass threshold of $SSB_{threshold} = SSB_{20\%} = 14{,}428$ metric tons (MT; 31.8 million pounds). A fishing mortality rate greater than $F = 0.5$ constitutes overfishing, and the stock is considered overfished if SSB is less than 14,428 MT. If it is determined that the weakfish stock is overfished, Amendment 4 requires ASMFC to implement measures to rebuild the population within six years (1½ generations).

Several addenda have been passed to improve management capabilities under Amendment 4. Addendum I was passed in December 2005 to modify biological sampling targets. Addendum III (May 2007) modified bycatch reduction requirements to maintain consistency with the South Atlantic Fishery Management Council. Of greater significance was passage of Addendum II in February 2007. A stock assessment conducted in 2006 showed a significant turn of events from previous assessment results (see full discussion in Section **C4.3**, **Assessment History**). Model results indicated that weakfish stocks were at historic low levels, and that fishing mortality was a relatively minor component of total mortality. Projection analyses indicated that even with a full moratorium on harvest, stock rebuilding would occur slowly at best without a significant decrease in

other sources of mortality. To minimize overall mortality without unduly penalizing fishermen, and to prevent expansion of the fishery in the event the stock begins to rebuild, Addendum II required that all states: 1) maintain current minimum sizes, 2) implement a recreational six fish bag limit (except South Carolina which was in the process of implementing a 10 fish limit), and 3) impose a 150 pound commercial bycatch trip limit (except *de minimus* states). Addendum II also established landings-based triggers to re-evaluate these criteria.

C4.3 Assessment History

Early stock assessment analyses for weakfish were conducted using a variety of virtual population models, such as the Murphy VPA (Vaughan et al 1991) and CAGEAN. The first peer reviewed assessment analyzed data through 1996 using Extended Survivor Analysis (XSA). The peer review was conducted in 1997 by the Stock Assessment Review Committee (SARC) at the 26th Northeast Regional Stock Assessment Workshop (SAW; NEFSC 1998a). The SARC had concerns with the XSA model runs and requested updated runs as well as exploratory CAGEAN and ADAPT model runs. These were conducted during the SAW, but there was insufficient time to fully review the results. As such, the SARC did not endorse the point estimates of F and SSB. Regardless, all models used indicated that SSB was increasing rapidly and fishing mortality rates were decreasing rapidly. SSB had increased an average of 22.5% per year since 1991, while F had decreased an average of 21.4% per year since 1990 (NEFSC 1998a). The SARC concluded that continuation of low fishing mortality rates and good recruitment would allow for age expansion to a point comparable to that observed in the early 1980s.

The subsequent assessment, including data through 1998, was peer reviewed at the 30th SAW/SARC in 1999 (NEFSC 2000). The stock was assessed using the ADAPT VPA as recommended by the 26th SARC. Ages in recent years were taken from otoliths, which required a conversion of scale-based ages from earlier years to otolith-based ages. The approved VPA run included only indices from the core abundance area (New York to North Carolina). The model indicated that fishing mortality rates had declined to 0.21 in 1998, well below both $F_{MAX} = 0.27$ and $F_{MSY} = 0.6$. In addition, SSB had increased to about 39,000 MT, approximately 55% of an unfished stock. The SARC did observe a noticeable retrospective pattern, which overestimated stock size and underestimated fishing mortality in the last few years. Regardless, the SARC 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.

A stock assessment update was conducted in 2002 (with data through 2000) using the SARC approved methodology (ADAPT VPA with tuning indices from the core area; Kahn 2002). The assessment showed that estimates of fishing mortality decreased further to $F = 0.12$, while SSB increased to over 50,000 MT. Although this assessment was not peer reviewed, the WTC expressed concern about a strong retrospective pattern that resulted in high levels of uncertainty in recent year estimates. The WTC recognized poor biological sampling of commercial catches, commercial discards, and recreational discards as a likely source of much of this error, especially when coupled with the assumption of error-free catch at age estimates used by ADAPT. Estimates of F and SSB were "corrected" by multiplying each parameter by the average amount each parameter changed in recent years with the addition of more data. Even so, the corrected estimate of $F = 0.23$ was substantially below $F_{Target} = 0.31$, and corrected SSB = 35,000 MT was more than double $SSB_{Threshold}$ = 14,428 MT.

In 2003, the Weakfish Stock Assessment Subcommittee (WSAS) began preparation for a

2004 peer review through the 40th SAW. Model results using the SARC approved methodology still exhibited a strong retrospective pattern, and results from both ADAPT VPA and biomass dynamic models indicated the stock was at very high levels (carrying capacity in the case of the biomass dynamic model; see Uphoff 2005c) with very low fishing mortality. The WTC was concerned that these results were not consistent with low catch rates and diminishing size structure being observed by commercial and recreational fishermen targeting weakfish.

For these reasons, the WSAS 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 40th SAW, 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 and fishery-dependent indices of abundance and catch at age).

The SARC agreed with the WSAS 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 (*e.g.* Cook 2005). The SARC indicated that it felt the problem was conflicting data, and expressed skepticism about the reliability of some survey indices, especially the Northeast Fishery 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 WSAS in the 2002 update process (ASMFC 2006a, Part A).

The stock assessment was completed in February 2006 and submitted to ASMFC for evaluation through the ASMFC External Peer Review process. The Peer Review Panel consisted of four fisheries biologists with expertise in population dynamics and stock assessment methods. The Panel did not endorse the statements regarding weakfish stock status and identified several issues that required additional work or attention by the WTC before the report would be suitable for management purposes (ASMFC 2006a, Part B). In particular, the Panel had concerns regarding stock structure, age composition data, and fishery discards.

The Weakfish Management Board directed the WTC to address the issues identified by the Peer Review Panel. Specifically, the Management Board tasked the WTC to further investigate stock structure and discards; determine agreements and disagreements among the assessment report, the peer review panel report, and the 40th SARC report; and provide an account of the implementation of recommendations from the 40th SARC.

In August 2006, the WTC provided a response to these tasks (ASMFC 2006a, Part C). Based on these responses, the WTC's analyses, and significant evidence, the Management Board accepted the following five points for management use:

1. The stock is declining;
2. Total mortality is increasing;
3. There is little evidence of overfishing occurring;
4. Something other than fishing mortality is causing the stock decline, and;
5. There is a strong chance that regulating the fishery will not, in itself, reverse the stock decline.

In December 2008, the NEFSC held the 2008 Northeast Data Poor Stocks Workshop (DPSW) to evaluate reference points for stocks with limited data. Although weakfish is not considered a data poor stock, the current assessment was reviewed as a work in progress. The intent of the review was

not a formal evaluation of the work (*i.e.* not a "Pass/Fail" evaluation), but a cursory review of the general data, methods, and preliminary results to provide guidance on ways to improve the analysis. The review panel expressed several concerns with the analysis, particularly with input data and lack of empirical data to support the species interaction hypotheses (Miller et al 2009). The WTC has reviewed the report and made appropriate modifications to the analyses for this final product. The weakfish portion of Miller et al (2009), along with the WTC's responses is provided in Appendix C-1.

C4.4 Life History

Weakfish, *Cynoscion regalis*, are estuarine dependent members of the drum family (Sciaenidae). Commonly occurring from Massachusetts to Florida, weakfish are most common in the Mid-Atlantic region from North Carolina to New York (Wilk 1979). Common migration patterns for weakfish include spring spawning movement into estuaries and bays and reverse movements out of the estuaries in the fall either offshore and/or to more southern regions to overwinter (Bigelow and Schroeder 1953, Wilk 1979). The spawning season is protracted and begins in the spring taking place in coastal estuaries and bays. Weakfish mature early (age-1) and the maximum recorded age using otoliths is seventeen years.

C4.4.1 Reproduction

Weakfish spawn in the nearshore and estuarine areas of the coast. In North Carolina, the spawning season occurs from March to September and peaks from April to June (Merriner 1976). Spawning further north occurs later and is less protracted. In Chesapeake Bay, spawning has been documented to occur from May to August (Lowerre-Barbieri et al. 1996). From Delaware Bay to New York spawning occurs from May to mid-July (Shepherd and Grimes 1984).

Early to mature, weakfish spawn multiple times in a season and have indeterminate fecundity (Lowerre-Barbieri et al. 1996). Reproductive work in Chesapeake Bay during 1991 and 1992 found that 90% of age-1 weakfish were mature. Batch fecundity ranged from 75,289 to 517,845 eggs/female and significantly increased with both total length and somatic weight (Lowerre-Barbieri et al. 1996). During 1999 and 2000, a study conducted in Delaware and Chesapeake Bays noted no increase in the size at maturity (168 mm) from that previously estimated despite a marked increase in the overall population size (Nye and Targett 2008). Similarly, most (97%) age-1 fish were mature. Both studies indicated that spawning frequency and batch fecundity vary by year and that these two variables act jointly to determine total egg production (Nye and Targett 2008). Nye and Targett (2008) also noted that despite maturing early, age-1 weakfish spawned less frequently, arrived later to the estuary and had lower batch fecundity than did older fish, likely resulting in an overly optimistic assumption about the contribution of age-1 fish to the overall reproductive success of the stock. This is currently amplified by the fact that larger, older fish comprise a small proportion of the overall population.

C4.4.2 Feeding Habits

Spatial and temporal variation in juvenile weakfish diet has been observed in studies conducted in Delaware Bay and Chesapeake Bay (Hartman and Brandt 1995, Grecay and Targett 1996, R. Latour, Virginia Institute of Marine Science, pers.comm). In Delaware Bay, Grecay 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).

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*). Section **C10.0** of this report examines changes in reported weakfish diet in detail.

C4.4.3 Age and Growth

Weakfish growth is rapid during the first year, and age-1 fish typically cover a wide range of sizes, a result of the protracted spawning season. After age-1, length becomes much less reliable as predictor of age due to an increasing overlap in lengths occurring over several age groups. Lowerre-Barbierri et al. (1995) found length at age to be similar between sexes with females attaining slightly greater length at age than males. Pooled across sexes, they reported observed TL's for weakfish collected in the spring (1989-1992) from Chesapeake Bay to be 176, 311,412, 510, 558, and 631 mm for ages 1-6 respectively. Growth was described using the Von Bertalanffy growth model ($r^2 = 0.98$; L_∞=919; K=0.19; t_0=-0.13). The L_∞ reported for other regions were similar: 893 mm TL for Delaware Bay (Villoso 1990) and 917 mm FL for North Carolina (Hawkins 1988) with the exception of Shephard and Grimes (1983) which reported lower L_∞ estimates for Chesapeake Bay (686 mm TL) and North Carolina (400 mm TL). The historical maximum age recorded using otoliths is 17 years. The fish was collected from Delaware Bay in 1985. The maximum age used in previous assessments considers Tmax to be 12 years (Kahn 2002). The world record weight for hook-and-line was recently captured on May 6, 2008 off of New York (8.67 kg). Weakfish have undergone large fluctuations in landings since the late 1800s, and there are reports from New England in the 1700s of decadal-scale abrupt shifts in abundance (Cushing 1982). Similar to landings, historic changes in the maximum size and age have been reported with weakfish typically obtaining their maximum size and age during periods of higher landings (Lowerre-Barbieri et al. 1995). More recent growth rates have slowed to the point that mean lengths at age of adults are several centimeters shorter than they were in the early 1990s (Lowerre-Barbieri et al. 1995, Kahn 2002). Weakfish weight at age plummeted in the 1990s by nearly half for 3-5 year-old weakfish.

C4.4.4 Natural Mortality

The 26th SARC recommended that M = 0.25 be used in modeling constant natural mortality and this value was used in prior assessments on weakfish (NEFSC 2000, Kahn 2002, ASMFC 2006a, Part A). A recent review of indirect, life history based methods to estimate natural mortality was conducted for weakfish along the Atlantic coast. This review found age-independent M rates ranging from 0.25 to 0.68 using various methods (Munyandorero 2008). The majority of the estimates ranged from 0.25 to 0.38. Where applicable, Tmax was set to age-12. Age-dependent estimates using the Lorenzen method resulted in M estimates ranging from 0.44 to 0.13*year^{-1} for age-1 to age-8 fish respectively.

The most recent age-structured VPA utilized data through 2003 and was run under the

assumption of constant M (0.25). Kahn et al. 2006). During the mid-1990's weakfish underwent a series of regulatory changes through Amendment 3 to the ASMFC weakfish FMP. After this period, the stock had an initial positive response including an increase in abundance, an expansion in the number of older fish in the population and an increase in the maximum observed age (up to age-12). Since that time, weakfish stocks have declined with landings currently at historic lows in both the recreational and commercial fisheries. These declines have occurred in spite of increased regulations to protect the fishery and no apparent increasing trend in either the effort of directed fisheries or the occurrence of discards. As a result, the WTC investigated possible causes for the recent decline in weakfish landings and age-structure. While the cause of the decline is not readily apparent, it does not appear that fishing mortality is not likely the primary culprit. Relative F estimates for the most recent years showed that fishing mortality had been low and stable from 1995 through 2003, in sharp contrast to the ADAPT estimates of a consistent increasing trend in F over that period. Consequently, the WTC does not support all results of the age-structured VPA under constant M assumptions, although the WTC does consider the VPA results with the reconstructed total mortality (based on adding the input M to the annual estimates of fishing mortality) to be reasonably accurate, along with the estimates of declining biomass through 2000. However, due to a retrospective bias extending back three years from the terminal estimate, the WTC regarded estimates for the last three years (2001-2003) as unstable and unreliable for manangement purposes. As a result of these analyses and conclusions, the WTC determined that the assumption of constant M was violated. Several investigations into possible systematic changes in M have been explored.

C4.4.5 Stock Definitions

The weakfish range extends along the Atlantic coast from Massachusetts to southern Florida, although strays are occasionally found as far as Nova Scotia, Canada and into the eastern Gulf of Mexico. Primary abundance occurs between New York and North Carolina. Within their range there is evidence of multiple stocks. Munyandorero (2006; see ASMFC 2006a, Part C) provides a concise but thorough overview of available information on weakfish stock structure. The following is an excerpt.

> Investigations of weakfish population structure along the US Atlantic coast have been undertaken through tagging, meristic, morphological, life history, genetic and otolith chemistry. The conclusions reached are conflicting. While Crawford et al. (1988), Graves et al. (1992) and Cordes and 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…

Crawford et al. (1988) recommended that weakfish be managed as separate northern and southern stocks, while Graves et al. (1992) recommended management of a single unit stock. The WTC reviewed the available information and reached the following conclusions.

- Evidence of stock structure exists
- Data is inadequate to define stock structure, and there is enough potential mixing that pinpointing the location of a north/south split is not possible at this time
- If a north to mid-Atlantic subpopulation is in serious decline, this does not warrant a north-south split based on conservation concerns (ASMFC 2006a, Part C).

Based on those recommendations, the ASMFC Weakfish FMP continues to manage Atlantic coast weakfish as a single unit stock throughout their coastal range.

C4.5 Habitat Description

Weakfish are found in shallow marine and estuarine waters along the Atlantic coast. They can be found in salinities as low as 6 ppt (Dahlberg 1972) and temperatures ranging from 17° to 26.5^{o} C (Merriner 1976).

Like many other North Atlantic species, weakfish exhibit a north-inshore/south-offshore migration pattern, although in the southern part of their range they are considered resident. Shepherd and Grimes (1983) observed that migrations occur in conjunction with movements of the 16-24° isotherms. Warming of coastal waters during springtime triggers a northward and inshore migration of adults from their wintering grounds in the Mid-Atlantic. The spring migration brings fish to nearshore coastal waters, coastal bays, and estuaries where spawning occurs.

Weakfish spawn in estuarine and nearshore habitats throughout their range. Principal spawning areas are from North Carolina to Montauk, NY, although spawning and presence of juveniles has been observed in the bays and inlets of Georgia and South Carolina (Lunz and Schwartz 1969, Mahood 1974, and Powles and Stender 1978, all as cited in Mercer 1985). Larval and juvenile weakfish generally inhabit estuarine rivers, bays, and sounds, but have been taken in freshwater (Thomas 1971) and as far as 70 km offshore (Berrien et al 1978). Mercer (1983) found that juveniles are most prevalent in shallow bays and navigation channels and are commonly associated with sand or sand/grass bottoms.

Weakfish form aggregations and move southward and offshore as temperatures decline in the fall. Important wintering grounds for the stock are located on the continental shelf from Chesapeake Bay to Cape Lookout, North Carolina (Merriner 1973, as cited in Mercer 1985).

C4.6 Fishery description

C4.6.1 Overview of fisheries

C4.6.1.1 Commercial Fishery

Records of commercial weakfish landings are available back to 1950 through the National Marine Fisheries Service (NMFS) website. From 1950 through the 1960s commercial landings ranged from about 2,000 to 4,000 metric tons (MT) per year (Figure C4.6-1). Beginning in 1970, reported landings exhibited a dramatic increase to a record high of more than 16,000 MT in 1980. From 1982 to 1988, landings fluctuated between approximately 8,000 and 10,000 MT. Since 1989, landings have declined continuously, except for a brief increase to about 4,000 MT in the mid- to late-1990s. Estimated commercial harvest in 2007 is the lowest on record at approximately 388 MT.

The general pattern for the commercial fishery is that in the winter, most landings occur in North Carolina as that state's fishery targets the overwintering aggregation off the North Carolina

coast. This seasonal fishery has accounted for the largest proportion of the commercial landings on the Atlantic coast. In spring, weakfish migrate back to spawning ares, primarily estuaries in North Carolina and the Mid-Atlantic states. Fishing occurs on the migrating fish along the coast and then concentrates on estuaries for the remainder of spring and summer, from Pamlico Sound in North Carolina through Peconic bay on eastern Long Island, New York. In mid-summer, some larger fish arrive in southern New England, including Rhode Island and Connecticut. With fall, weakfish leave estuaries and begin their fall migration south to the overwintering grounds, and are targeted as they move down the coast.

Three states - New Jersey, Virginia, and North Carolina - have consistently accounted for 70 to 90% of the coastwide commercial harvest since 1950 (Table C4.6-1; Figure C4.6-2). North Carolina has predominated with nearly 37% of the coastwide harvest over the last ten years, while Virginia and New Jersey have averaged 25.6% and 17.0% respectively. During this same time period, New York has accounted for nearly 10% of coastwide harvest.

From the mid 1950s to the early 1980s landings from the trawl fishery generally accounted for 50 to 70% of commercial landings (Figure C4.6-3). Beginning in the early 1980s, harvest from trawlers began a gradual decline, and recently have accounted for approximately 20% of total harvest. Conversely, between 1979 and 1987, landings from gillnets increased from around 10% of annual harvest to 45% of annual harvest, and have remained relatively stable since that time. Over the entire time period, pound nets and haul seines have each averaged between 10 and 20% of total harvest annually, despite declining trends.

Discarding of weakfish by commercial fishermen is known to occur, and discard mortality is assumed to be 100%. The first quantitative analysis of weakfish discards is provided in de Silva (2004). Most discarding occurs in conjunction with two gears (trawls and gillnets) and a limited number of target species. Prior to 1994, discards are assumed to have occurred for non-regulatory reasons because few regulations were in place to limit the fishery. Since 1994, both regulatory and non-regulatory discarding has occurred. Regardless, population removals as a result of commercial discarding appear to be minor relative to harvest, even in recent years as harvest has decreased.

From 1982 to 1990, estimates of biomass of discarded weakfish generally declined from around 600 MT to 200 MT, where it remained stable for several years (Table C4.6-2). With the implementation of state specific regulations in 1993, and mandatory coastwide measures in 1995, estimated discards increased dramatically to near 1,000 MT in the mid 1990s. Except for the time series maximum of approximately 1,150 MT in 2001, discards decreased steadily from 1996 to 2004, and have remained stable around 200 MT since that time. From 1982 to 1999, age 1 fish generally dominated the discards, with a few exceptions in the mid 1990s when age 0 fish outnumbered age 1 fish. Since 2000, however, discards of age 2 and 3 fish have exceeded those of age 1.

Changes in catch per unit effort (CPUE) over time can be indicative of changes in abundance/availability or a shift in target species. Where available, commercial weakfish CPUE was examined to evaluate trends in fishery performance over time. Data were evaluated for all trips where weakfish were harvested, as well as only trips that harvested more than 150 pounds (68 kg) of weakfish, in an attempt to include directed trips during an open season.

North Carolina has historically been the largest commercial harvester of weakfish, and CPUE data are available back to 1994. When all positive weakfish trips (only trips where weakfish were caught) are considered, all but two (flounder trawl and haul seine) of the eight fisheries exhibited a strong decline in CPUE (Figure C4.6-4). When only trips harvesting 150 or more pounds are considered, CPUE typically declines, though generally less severely than when trips with bycatch

allowances were included (Figure C4.6-4). In Virginia, commercial weakfish CPUE has fallen since the late 1990s, particularly since 2002 (Figure 4.6-5). Delaware gillnet fishery CPUE (positive trips) declined in the early 1990s, rebounded in the late 1990s, and has declined steadily since 2000 (Figure C4.6-6). Data are available from the Potomac River pound net fishery for 1976 to 1980 and 1988 to 2007 (Figure C4.6-7). Between 1976 and 1980, CPUE was high. By 1988, CPUE had dropped to less than one-third of peak levels, dropping below 20% of the 1980 peak between 1988 and 1993. CPUE rebounded to between 20 and 40% of the time series peak during 1994-2002 and then fell sharply (Figure C4.6-7).

Although there is some regional and temporal variability, commercial CPUE generally present a consistent pattern of recovery during the late 1990s and then a severe decline in the early 2000s. Commercial CPUE since the mid to late 1990s corresponds well with model estimates of population trends, fishery independent and fishery dependent abundance indices, and observed size and age structure. The WTC are not aware of any changes in regulations or fisherman behavior that would explain the recent decline in CPUE.

C4.6.1.2 Recreational Fishery

Recreational harvest statistics for the weakfish fishery are available on the NMFS Marine Recreational Fishery Statistics Survey (MRFSS) website for the period 1981 to 2007 (*www.st.nmfs.gov/st1/*). From 1981 to 1988, the number of weakfish caught and the number harvested fluctuated without trend between 2 million and around 11 million fish; however, during this same time period, harvested weight generally declined from around 7,259 MT to 2,722 MT (Figure C4.6-8). Nearly 90% of all fish caught were retained during these years.

From 1989 to 1993, catch (numbers) and harvest (numbers and weight) remained relatively stable. Catch fluctuated between 1.6 and 2.2 million fish, while harvest ranged between 0.95 and 1.8 million fish and 499 to 998 MT. Percentage of total catch that was harvested during this period decreased from around 90% to less than 50%.

In 1994, weakfish catches increased and averaged around 6 million fish until 2000. Harvest numbers increased to a lesser extent and fluctuated between approximately 1.5 and 2.5 million fish. Harvest weight also increased to 1,814 MT during this period. By 2003, catches and harvest had declined to at or near time series minimum and have remained relatively stable. In 2007, total catch was 2.01 million fish, with a harvest of 0.58 million fish and 313 MT. Since 1994, harvest has accounted for approximately 20-40% of all fish caught.

Recreational harvest has been dominated by the five Mid-Atlantic states between New Jersey and North Carolina (Table C4.6-3; Figure C4.6-9). New Jersey dominated landings in most years, averaging 35% of coastwide harvest across the time series. Virginia consistently produced greater than 20% of coastwide landings from 1981 to 1992 but has since declined, averaging about 10% over the last five years. Since 1995, several states have each had periods of substantial landings, with Delaware contributing 20-30% of total harvest for 1995-1998, Maryland accounting for approximately 25% from 1999 to 2001, and North Carolina averaging 22.5% from 2003 to 2007.

Recreational discard mortality is assumed to be 10% of all discarded fish based on catch-and-release experiments with weakfish and the closely related spotted seatrout (*Cynoscion nebulosis*; (*e.g.* Murphy et al 1995, Malchoff and Heins 1997, Swihart et al 2000, Duffy 2002, Gearhart 2002). Weakfish hook-and-release experiments produced dichotomous mean mortality estimates, either near 3% or 15%, and 10% release mortality was adopted by the WTC. From 1981 to 1989, harvested weakfish averaged 89% of total catch (numbers). Even with high landings, discard losses during this period were lowest of the time series, with all but one year having fewer than 100,000

fish discarded coastwide (Figure C4.6-10). Between 1989 and 1995, harvest fell to 27% of catch, and discard losses increased to more than 400,000 in 1995. Harvest rebounded slightly to 41% of catch in 1997 and 1998, but dropped back to between 20-40% since 1999. Despite relatively stable release rates since 1995, discard losses have varied greatly due to large interannual fluctuations in catch. Discard losses peaked at approximately 500,000 fish in 1996 and 2000, but have since decreased along with catch. For the last five years, discard losses have ranged between 135,000 and 225,000 fish.

Throughout the time series, total removals have been dominated by commercial and recreational harvest (Figure C4.6-11). Removals were greatest during the early portion of the time series, averaging 13,500 MT between 1981 and 1988. Between 1989 and 1993, removals dropped off quickly to 4,000 MT. The next few years showed a slight rebound to a peak of 6,500 MT in 1998. Since then, removals have declined continuously to the time series minimum of only 852 MT in 2007. Combined commercial and recreational discard losses were generally less than 5% of total removals prior to 1993. Discarding increased rapidly following implementation of management measures. Regardless, discard losses have averaged less than 20% of total removals since 1994.

C4.7 Current status

Throughout the 1980s and early 1990s, weakfish stocks experienced unsustainably high fishing mortality rates, which led to a decline in abundance into the 1990s. Fishing mortality rates declined during the early 1990s, and an increase in biomass was evident during the mid to late 1990s. The 2006 stock assessment indicates that fishing mortality has remained low under Amendment 4, yet weakfish biomass had dropped back to near historic low levels by 2003. Available evidence indicates that factors other than fishing mortality were the primary cause for biomass decline (ASMFC 2006a, Part A). A peer review of the stock assessment did not endorse the statements regarding weakfish stock status and identified several issues that required additional work or attention by the WTC before they would support its use for management purposes (ASMFC 2006a, part B). In particular, the review panel had concerns regarding stock structure, age composition data, and fishery discards. In August 2006, the WTC responded to the peer review panel's concerns (ASMFC 2006a, Part C). Based on these responses, the Technical Committee's analyses, and significant evidence, the Weakfish Management Board accepted the following five points for management use:

1. The stock is declining;
2. Total mortality is increasing;
3. There is little evidence of overfishing occurring;
4. Something other than fishing mortality is causing the stock decline, and;
5. There is a strong chance that regulating the fishery will not, in itself, reverse the stock decline.

C5.0 Evaluate biases, precision, uncertainty, and sampling methodology of the commercial and recreational catch (including landings and discards) and effort. (TOR #1)

C5. 1 Commercial

C5.1.1 Landings

Commercial landings data were taken from two sources. Where available, state-specific

harvest records collected through a mandatory reporting system were considered the most reliable source for landings. Unfortunately, not all states require mandatory reporting of weakfish harvest. In such cases, landings estimates were obtained from the NMFS commercial landings database, available through the NMFS Office of Science and Technology, Fisheries Statistics Division website (*http://www.st.nmfs.gov/st1/*). Although estimates are available from NMFS, it is not mandatory to report weakfish harvest to NMFS, so these records (like those of most species) may be incomplete. Discrepancies between NMFS reported harvest and state reported harvest under mandatory reporting suggest that NMFS harvest estimates for weakfish are a potential source of uncertainty. In an attempt to quantify the uncertainty between the two reporting systems, state reported landings from Delaware and Virginia were compared to federally reported landings in these two states for the period 2004 to 2006. Combined across all gears, NMFS reported landings for a given year differed from state landings by less than 10% in all instances except Virginia in 2006, when the difference exceeded 23% (Figure C5.1-1). However, when evaluated at the gear level, more than one-third of all year/state/gear combinations differed by more than 20%, and in three cases exceeded 100% (Figure C5.1-2). Generally speaking, then, annual estimates of weakfish harvest reported by state and federal agencies are relatively consistent when combined across all gears, but the allocation of landings by gear are less certain. Lack of state landings data and discrepancies between state and federal estimates are not confined to just weakfish, but are observed in most state managed species.

C5.1.1.1 Biological samples

Commercial biological samples include lengths, weights, and ages from state-specific port sampling programs. Commercial samples were combined with similar data from recreational and fishery independent sources to develop length-weight relationships and age-length keys (ALK) for use in the estimation of commercial catch at age.

Lengths

Commercial length data were used for two primary purposes: the development of length-weight equations and characterizing the distribution of commercial catches by length and age. Because a combination of both total length and fork length data were available, lengths were standardized to fork length measurements. A conversion factor was developed using data pooled across all sources in 2004 to 2006 that reported both total length and fork length. Total length (mm) was converted to fork length (mm) using the equation

$$FL = (TL + 5.8106) / 1.0437$$

Length-weight equations were developed as in the 2006 assessment (ASMFC 2006a, Part A). Length and weight data from all sources were pooled, and relationships were developed by region/year/season. Sample sizes and parameter estimates are presented in Table C5.1-1.

Characterization of fishery catch at size was conducted using similar procedures as the 2006 assessment (ASMFC 2006a, Part A), following methods described by Quinn and Deriso (1999). Commercial harvest estimates and length samples were stratified by region/year/season/state/gear. Landings not identified to specific gear were pooled at the region/year/season level and classified as "Other." In addition, fisheries (state-gear combination) with minimal landings (< 1% of region/year/season total) generally had insufficient sample size (see below) to characterize that fishery. These cells were also classified as "Other."

For northern region cells with significant landings (≥ 1% of region/year/season total) and sufficient sample size, harvest weight was converted to harvest number at size using predicted weight at length (from region/year/season specific length-weight equations) and observed length frequency distributions. Mean weight at length was estimated using the appropriate length-weight equation. Sample weight at length (average weight at length multiplied by number of samples at length) was then divided by total sample weight (weight at length summed across lengths) to determine weight distribution by size. These proportions were then multiplied by total harvest weight for that state/gear/season to determine harvest weight at length. Harvest weight at length was then converted to number at length by dividing weight by average weight at length.

For cells with significant harvest and insufficient sample size, the same methods were employed using length data borrowed from an appropriate substitution cell with sufficient sample size. Finally, the "Other" category was split into harvest from states with a 12" minimum size and states with a 16" minimum size. For each region/year/season/minimum size category, all available samples greater than the minimum size from the respective states were used to convert "Other" harvest weight to harvest number at size.

For the southern region, characterization of the fisheries was done slightly differently. Commercial sampling in North Carolina includes collection of both lengths and weights, so it was possible to develop an average fish weight by gear and season for each fishery. The average weight was divided into the harvest weight to estimate number harvested. The number harvested was then partitioned to catch at size using the length frequency distribution of the samples. Florida, the only other southern region state with commercial landings, collects no biological samples. Biological sample data from North Carolina were used as proxy information for Florida landings.

Results of the fishery specific catch at size analyses were combined across states and gears within a region to develop estimates of commercial harvest number at size by region/year/season.

Uncertainty in estimated harvest number at size could be introduced from two primary sources: 1) sample size of fish used to characterize a fishery, and 2) substitution of data from alternate strata for fisheries with insufficient sample size. These are discussed in more detail below.

Sample size and ratio of samples per metric ton of landings were used to evaluate adequacy of sampling intensity (Table C5.1-3). It was determined that a minimum sample of 30 lengths per stratum (region/year/season/state/gear) was required to adequately characterize a fishery. The minimum of 30 samples is much lower than sample sizes suggested in the literature for characterizing a population (Miranda 2007; Vokoun et al. 2001). Insufficient sampling would tend to introduce uncertainty into the catch at size estimates; however, these studies recommend sample sizes necessary to meet an objective (characterizing entire population) much different than the current analysis (characterizing harvest of specific gear). However, Miranda (2007) notes that distributions with a smaller size range require a smaller sample size. Considering minimum size limits and gear selectivity, the sample size required to characterize a fishery is likely lower than those to characterize an entire population. For example, Burns et al (1983) indicate that 100 fish per 200 MT of landings appears adequate to characterize many of the northeast U.S. groundfish stocks. ASMFC (2005) requires states to collect a minimum of 6 weakfish lengths per MT of landings, although not all states have been able to comply with this due to staff/funding constraints and difficulty obtaining samples. For this assessment, there are 141 cells (defined as region/year/season/state/gear combination with significant landings) that require expansion. All have more than 0.5 fish per MT (*i.e.* 100 fish per 200 MT), and only 13 (9.2%) have fewer than 6 fish per MT of landings. Sample size in the cells with less than 6/MT ranges from 30 to 419 fish. Also, 41 of the 141 cells have fewer than 30 samples and require substitution from alternate strata.

A higher minimum sample size requirement would increase the number of cells that require substitution, which could increase uncertainty in length and age distributions for those cells. Therefore, a sample size of 30 fish is considered an appropriate compromise between adequate lengths to fully characterize a fishery and the need to substitute data.

For strata with insufficient (< 30) length samples, data were usually substituted from the same region/year/season, but were sometimes substituted from another state and/or gear (Table C5.1-2). Care was taken to minimize differences in gear selectivity, and when necessary substituted data were truncated to account for differences in minimum size requirements between the two states. Regardless, the WTC recognizes that substituted data are not always representative of the stratum to which they are applied, resulting in uncertainty in the length frequency distribution of the catch. Of greatest concern are the geographic differences in fish size, coupled with the general lack of samples north of Delaware. Minimum size limits and average size of harvested fish were much larger in the northern portions of the range. When commercial samples from these states were insufficient and data were substituted from more southern states, the effect was an underestimation of the proportion of large fish in the harvest. In 2006, New Jersey began collecting biological samples from their commercial fisheries. By estimating length and age distributions of New Jersey commercial harvest using New Jersey data and data substituted from other cells, Brust (2009) shows that data substitution resulted in fish ages 1 to 5 being overestimated in the coastwide CAA by less than 5%, but fish ages 6 and older were underestimated by up to 52%. The 6+ plus group as a whole was underestimated by 9.9% in 2006 and nearly 32% in 2007 (Brust 2009). Improved sampling in the northern region since 2005 will better characterize landings from this region, decreasing uncertainty in catch at size estimates.

Ages

The principal use of age data is in the development of ALKs. Sample sizes of ages by year, season, and source are provided for recent years in Table C5.1-3. During the 1980s, ages were based on scale samples. During the 1990s, otoliths became the principal method for aging weakfish. For the 1998 stock assessment, scale-based ages in previous years were converted to otolith-based ages using a scale-otolith conversion matrix (similar to an ALK) based on direct comparison of approximately 2,300 samples (Daniel and Vaughan 1997; NEFSC 1998b). Uncertainty in either aging method, as well as in the scale-otolith conversion matrix would be propagated through the catch-at-age matrix.

Age-length data from all available sources (commercial, recreational, fishery independent) were pooled by region/year/season to develop stratum specific age-length keys (four keys per year) as described by Vaughan (2000). Length intervals with missing information in the keys were filled by either averaging age distribution for lengths above and below, substitution from another stratum, or interpolation. These filling procedures could lead to uncertainty in catch at age if the substituted age distributions are not representative of the cells into which they are substituted.

Once the age-length keys were complete, catch at size estimates by region/year/season were converted to catch at age using the appropriate age-length key. Catch at age estimates were pooled across regions and seasons to develop annual estimates of commercial catch at age.

C5.1.2 Discards

Discard mortality of weakfish by commercial fisheries was assumed to be 100%. The first quantitative analysis of weakfish commercial discards was provided by de Silva (2004). Most discarding occurs in conjunction with two gears (trawls and gillnets) and a limited number of target

species. Several methods to estimate discards were investigated, including effort based estimates, regression analysis, and ratio extrapolation. Effort data were not available for all states and years to develop effort based estimates. Regression estimation was conducted, but the predictive models fit poorly ($r^2 \sim 0.08$) and were considered inappropriate for use in the assessment. Ratio estimates work best when there is evidence of a positive linear relationship between the response and explanatory variable. Although there was no evidence of such a relationship in several of the gear-species combinations evaluated, it was determined that ratio extrapolation provided the most reliable estimates of discards from the methods investigated. With this method, discards were calculated using seasonal, annual, and multi-year (all years combined) ratios. It was determined that the multi-year estimates provided the most reliable estimates, and this method was selected as the final estimation methodology. Discards in the southern region (North Carolina to Florida) were considered insignificant, so commercial discards were only evaluated for the northern region. A full description of the methods is provided in de Silva (2004) and summarized below.

Data from the NEFSC observer database (1994 to present) were queried to obtain haul level estimates of weakfish discard weight and target species harvest weight for each gear/target species combination. Discard and harvest weight estimates were pooled across all years by gear and target species to develop gear/species specific weakfish discard ratios (calculated as summed weakfish discard weight divided by summed target species harvest weight for each gear and target species). The gear/species specific ratios were then applied to annual harvest estimates of that gear/species combination (from the NMFS commercial harvest website) to estimate total weakfish discard weight in that fishery. Weakfish discard length frequency data by gear (all species combined) were used to convert discard weight to discard numbers at size. Annual discard estimates by gear were partitioned into seasonal estimates by using the proportion of annual landings by season and gear from the NMFS landings database. Gear-species-season discards at size were summed across gears and species to determine total seasonal discards at size.

Prior to 1994, discards are assumed to have occurred for non-regulatory reasons only because few regulations were in place to limit the fishery. Unfortunately, observer data are not available prior to 1994. As such, data from 1994 forward were subset by regulatory and non-regulatory discards, and the method described above was used to develop discard ratios for non-regulatory discards only for 1994 forward. These ratios were applied to annual estimates of gear/target species harvest for 1981 to 1993 to estimate non-regulatory discards during this time period.

For the current assessment, commercial discard estimates were updated for 1994 forward in order to include additional years of data and significant updates to data used in the 2006 assessment. Using the general method of de Silva (2004), haul level data for the same gear and target species combinations were used to evaluate annual, multiyear (5 and 7 year), and all-year (1994 to 2007) weakfish discard ratios. Because of concerns with high interannual variability and uncomfortably large standard errors with estimates based on short time groupings, the all-year ratio estimation method was selected. Ratio-based estimates of weakfish discard weight were made for butterfish, long-fin squid, summer flounder and weakfish for trawl gear, and Atlantic croaker, bluefish, spiny dogfish, and weakfish in the gillnet fishery. Discard weight was converted to numbers at size using observed and re-sampled length frequencies, then converted to discard number at age using the appropriate region/year/season age-length key described in section **C5.1.1 Landings**. Catch at age estimates were summed across regions and seasons to determine annual commercial discard catch at age.

There are several potential sources of uncertainty with the commercial discard estimates. Sample size of observed hauls with weakfish discards is low in many years for some gear-species

combinations (Table C5.1-4), and discard ratios vary greatly (see *e.g.* Table 13 in de Silva 2004). Combining data across years improves sample size but may mask temporal trends in discarding. In addition, in order to convert discard weight to number, biological data from discarded fish were pooled across species for a given gear type, and/or resampled from observed fish to attain a minimum sample size (30 fish). For both concerns, larger sample sizes would provide more reliable estimates of discard rates.

Another source of uncertainty is the gear-species combinations used. The initial analysis found 14 gear-species combinations with substantial weakfish discards (de Silva 2004). However, many of these species are often captured together. To minimize the potential for duplicate counting of discards when discard ratios were multiplied by total harvest of each of the gear-target species combinations, principal component analysis was conducted to identify species groupings. Discard ratios were then multiplied by harvest of only one of the species within a group. Selecting a different species from the group would result in different estimates of total discards.

During the NEFSC DPSW in December 2008, reviewers were concerned that the methods used to estimate discards could result in substantial uncertainty. However, the reviewers could not agree on the direction of the bias. At least one reviewer was concerned that discards were overestimated because multiplying a discard ratio for a given target species by total harvest of that species includes harvest when that species was not the target species (*i.e.* harvest estimates applied to ratios were overestimated). This was addressed to some degree by minimizing the number of gear-species combinations through PCA. On the other hand, a second reviewer was concerned that the number of gear-species combinations was too limited and may have missed historic fisheries with large weakfish discards. The WTC is aware of these potential sources of bias. Unfortunately, the methods used to estimate weakfish commercial discards are constrained in many ways by the amount of available data. The methodology has been used in other ASMFC assessments (i.e. Atlantic croaker) and is a more comprehensive analysis than most assessments attempt at quantifying discards.

C5.2 Recreational

C5.2.1. Landings

Recreational landings data were obtained from the NMFS Marine Recreational Fishery Statistics Survey (MRFSS) database, which is available through the NMFS Office of Science and Technology, Fisheries Statistics Division website (*http://www.st.nmfs.gov/st1/*). MRFSS provides estimates for three subcategories of catch, including observed harvest (Type A), unobserved harvest (*e.g.* filleted before observation, discarded dead; Type B1) and discarded alive (Type B2). Estimates of harvest were developed for each region/year/season combination as a sum of observed and unobserved harvest (Type A + B1). Because sand seatrout and weakfish are indiscernible except through genetic analysis, MRFSS estimates in Florida are for the *Cynoscion* complex of weakfish, sand seatrout, and their hybrids. Estimates for true weakfish in Florida were found by multiplying MRFSS estimates by the proportion of true weakfish observed by Tringali et al. (2006) before combining with other southern region states.

Precision in recreational catch and harvest estimates are calculated as a percent standard error (PSE). Lower values indicate better precision than higher values, and most commonly caught species generally have PSEs less than 20% (NMFS 1999). As such, estimates with PSE values less than 20% are generally considered "acceptable" (NEFSC 1998b). Estimates of weakfish harvest were relatively precise (Table C5.2-1), with PSEs less than 15% for most years since 1982

(numbers) or 1983 (weight). Estimates of the number of weakfish discarded were less precise in the beginning of the time series, but PSEs have been below 20% for all but one year since 1986. However, a recent review of the survey identified several potential biases and inadequacies of the sampling and estimation methodologies (NRC 2006; see http://www.nap.edu/catalog.php?record_id=11616). These include the inability to interview anglers at private access sites; the increasing use of household cell phones which are unavailable to the telephone sampling frame; reliance on unverified assumptions; and differences in statistical properties of data collected through different survey methods. The effects of these biases on estimates of recreational catch, harvest, and discards can not easily be quantified, leading to uncertainty in MRFSS recreational estimates. This uncertainty applies to all catch types over the entire time series, which has been collected using the same general methodology throughout.

C5.2.1.1 Biological samples

Biological samples collected by MRFSS include lengths and weights of a subsample of Type A fish. No ages are collected from the recreational fishery. Recreational length-weight data were combined with similar data from commercial and fishery independent sources to develop length-weight relationships (see section **C5.1 Commercial**). Length data were also used to partition harvest into harvest at size. Because of small sample sizes (Table C5.1-3), length observations were pooled by region/year/season to expand harvest estimates at the same level of stratification. Unlike commercial data, estimates of recreational harvest in numbers are directly available from the MRFSS website. Catch at size was estimated as the proportion measured at size by stratum multiplied by the estimated harvest (number A+B1 fish) for that stratum. Number at size was converted to number at age using the appropriate region/year/season age-length key described in section **C5.1.1 Landings**. The number of length samples collected by MRFSS is above the criterion of 100 lengths per 200 MT of landings (Burns et al 1983).

C5.2.2 Discards

Estimates of the number of recreational weakfish discards (Type B2 fish) were obtained from the MRFSS database. Estimates in Florida were corrected for weakfish-sand seatrout hybridization using ratios reported by Tringali et al. (2006). In previous assessments, discard mortality was assumed to equal 20% of all discards. However, based on a review of available data, the WTC has decreased the discard mortality to 10% (*e.g.* Murphy et al 1995, Malchoff and Heins 1997, Swihart et al 2000, Duffy 2002, Gearhart 2002).

Prior to 2004, discarded fish were not sampled. Since 2004, MRFSS has collected lengths of discarded fish from the for-hire sector (party and charter boats). As such, this is the first assessment for which recreational discard length frequencies are available. Observed length frequencies were applied to discard mortality estimates to estimate the number of dead discards at size. For the northern region, this was done by year and season. Due to low sample size, southern region samples were pooled across seasons to develop annual length frequencies. Number at size was converted to number at age using the appropriate region/year/season age-length key described in section C5.1.1 Landings.

C5.3 Catch at Age Matrix

The catch-at-age matrix for 2004-2007 was developed using the same general procedures as previous assessments. Catch at size from the four major sources of removals (commercial harvest, commercial discards, recreational harvest, recreational discards) were converted to catch at age

using the appropriate region/year/season age-length key. Results were pooled across regions, seasons and sectors to estimate total annual removals at age (Table C5.3-1).

As described in each of the pertinent sections, there are several potential sources of uncertainty in the overall catch at age estimates. These include inaccurate harvest/discard estimates as a result of under/over reporting or inappropriate survey methods; insufficient sample size to characterize length distributions; substitution of data from alternate cells in the catch at size characterization and ALKs; errors in aging techniques or the scale-otolith age conversion; and others. Attempts have been made to quantify some of these error sources; however, the extent of uncertainty associated with each of these sources, and their cumulative effect, remains largely unknown. A persistent cumulative trend in either direction would result in inaccurate catch at age estimates and may influence assessment results. However, the sources of potential error and the methodologies used to develop the catch-at-age matrix in this assessment are similar to those used for other ASMFC species assessments that have passed peer review.

C6.0 Evaluate precision, geographical coverage, representation of stock structure, and relative accuracy of the fisheries independent and dependent indices of abundance. Review preliminary work on standardization of abundance indices. (TOR#2)

C6.1 Aged fishery independent surveys

C6.1.1 NEFSC Bottom Trawl Survey

The National Marine Fisheries Service (NMFS) Northeast Fishery Science Center (NEFSC) conducts seasonal trawl surveys between Nova Scotia and Cape Hatteras. Stratified random sampling is conducted using a #36 Yankee otter trawl equipped with roller gear and a 1.25 cm mesh codend liner. The survey covers a large portion of the geographic range of weakfish, including their "core" distribution area (NEFSC 2000) of New Jersey to North Carolina. Despite the extended latitudinal range, the survey is not capable of sampling in shallow waters, and few sites are conducted in waters less than 9 m. In addition, the survey does not sample the South Atlantic portion of the range.

Weakfish are infrequent in the winter, spring, and summer surveys, but are commonly intercepted in the fall during their offshore migration. Because weakfish are rarely caught in this survey north of New Jersey, the 30th SAW/SARC recommended developing an index of weakfish abundance using only strata from the south end of Long Island to Cape Hatteras during the fall survey. Indices at age are developed by applying annual survey specific length frequency data to the annual mean catch per tow and then applying either survey specific ALKs (when available) or the pooled northern region late season ALK (see section **C5.0)**. Because this survey occurs in the fall, true ages are increased by one year to develop an index of abundance on January 1 of the year following the survey (*e.g.* fall 1997 age 0 fish are treated as January 1, 1998 age 1 fish).

The annual mean catch per tow appears nearly cyclical, with relative peaks in abundance generally every 4 to 6 years (Figure C6.1-1). From 1982 through the mid 1990s, mean catch per tow cycled without trend, generally ranging between 40 and 120 fish per tow. Beginning in the mid 1990s, abundance gradually increased to a time series maximum of approximately 500 fish per tow in 2004. During 2005 – 2007, abundance decreased to about 200 fish per tow, but increased in 2008 to over 300 fish per tow. Standard errors (SE) follow a similar pattern as CPUE, with an overall cyclical pattern and a gradual increase beginning in the mid 1990s. During the early portion of the time series, SE varied between approximately 10 and 50, increasing to a peak of 90 in 2004.

Coefficient of variation (CV; SE as a ratio of the mean) has varied without trend between approximately 10 and 40% since 1990.

The survey index is dominated by age 1 fish (age 0 fish progressed to age 1), although fish have been observed out to age 6 (Figure C6.1-1). Age distribution was greatest in the early 1980s, but was truncated to predominantly ages 1-3 by the early 1990s. Age distribution expanded somewhat during the late 1990s as the stock began rebuilding as a result of management measures, but has since declined to primarily ages 1-4.

The WTC evaluated many of the age aggregated survey indices and found the NEFSC weakfish index performed poorly compared to others (see section **C10.0** of this report for a summary of the analyses). The timing of the survey, along with the highly contagious distribution of weakfish, leads to high variability between years and between tows within a year. Proportional stock density analysis indicates that the survey's ability to capture large weakfish, even when they are present, is poor. Using correlation analysis, it was found that the index was not well correlated with the other indices or the converged portion of the VPA, and was negatively correlated with harvest trends. Finally, catch curve analysis determined abundance of several year class increased over time (*i.e.* negative mortality; see Table 3 of ASMFC 2006a, Part A). Efforts to develop a more representative index of weakfish abundance (*e.g.* geometric mean, percent positive tows, etc.) were unsuccessful. Although the survey has several advantages, such as being the longest running and widest ranging fishery independent survey, the WTC concluded that the NEFSC fall survey is not suitable for use as an index of relative abundance in the assessment.

C6.1.2 New Jersey Ocean Trawl Program

New Jersey has conducted a stratified random trawl survey in nearshore ocean waters (to 27 meters depth) from Ambrose Channel (entrance to New York Harbor) to Cape Henlopen Channel (entrance to Delaware Bay) since 1988. The survey originated as bi-monthly cruises, but since 1991 has consisted of five cruises per year (January, April, June, August, and October). Strata are nearly identical to those used by NEFSC in this region (New Jersey's northern- and southern-most strata are truncated at New Jersey state boundaries). The gear used is a two-seam trawl with a 25 m headrope and 6.4 mm bar mesh codend liner. Due to funding constraints, several different vessels have been used to conduct the survey.

The geographic range of the survey is limited to nearshore ocean waters of the species distribution within the northern and southern borders of New Jersey. The survey occurs within the region sampled by the NEFSC trawl survey. The use of a smaller vessel, however, allows the New Jersey survey to provide better coverage in shallow waters.

The majority of weakfish are observed during the June, August and October cruises, although catches in June are inconsistent. During previous assessments, an index of weakfish abundance was developed using the August and October cruises. However, recent work has shown that the August-October index is a poor indicator of weakfish abundance (see section **C10.0** of this report). As with the NEFSC index, tow-level and annual mean catch rates show great variability, abundance of some year classes was shown to increase over time based on year class catch curves, and CPUE is not well correlated with harvest, other weakfish indices, or the converged portion of the VPA. Unlike the NEFSC index, however, the New Jersey index is capable of capturing large weakfish when present. Efforts to develop a more suitable index found that the percent of tows during the August cruise that captured weakfish was coherent with other indicators of weakfish abundance. For the current assessment then, the New Jersey index is based on the proportion of positive tows (PPT) from the August cruise. The index was aged using a combination of survey specific proportion at age data for

1995 to 1997 and DNREC 30-foot trawl survey proportion at age data from August cruises (supplemented with July and September data as needed) in all other years.

PPT was greatest between 1994 and 1997, with two-thirds of all tows containing weakfish in three of those years (Figure C6.1-2). PPT dropped precipitously to 28% in 1998 and remained generally between 25 and 50% until 2002. In 2003, PPT again dropped precipitously to the time series low of just 10.3%. Since then, PPT has rebounded slightly, with 32 to 35% of tows containing weakfish in three of the last four years. Binomial confidence intervals (90%) indicate the upper and lower CI differ from the mean by approximately 7 to 16% in all years.

Indices at age document the occurrence of strong 1993 and 1996 year classes as they appear in 1994 and 1997 and move through the age structure (Figure C6.1-2). As these cohorts increase in age, the proportion of fish ages 4 to 6+ was increased and exceeded 40% in three years between 1997 and 2000. Since 2002, however, the survey has been dominated (>= 73%) by age 1 and 2, with 4+ fish making up less than 5% of the annual catch..

Detailed investigation into the New Jersey Ocean Trawl Survey provides evidence that the percent of positive tows from the August cruise provides a reliable index of weakfish abundance. The WTC has therefore determined that the New Jersey trawl survey index is acceptable for use in the stock assessment.

C6.1.3 Delaware DFW Delaware Bay Trawl Survey

The Delaware Division of Fish and Wildlife has conducted a trawl survey within the Delaware Bay intermittently since 1966 (1966-1971, 1979-1984, and 1990 – present). The survey collects monthly samples (March through December) at nine fixed stations throughout the Delaware portion of the Bay. The net used has a 30.5 foot headrope and 2" stretch mesh codend. For the current assessment, only the 1981-1984 and 1990-present time series are evaluated. Weakfish abundance is calculated as an average number of age 1+ fish per nautical mile for June to October cruises, and the index is treated as a mid-year abundance (*i.e.* not progressed forward to January 1). Since 1991, length frequencies have been aged using survey specific age-length keys.

The geographic range of this survey is limited to the Delaware Bay, a very small portion of the weakfish stock range; however, Delaware Bay is known to be a major spawning ground for weakfish on the Atlantic coast (Nye et al 2008). As the survey occurs monthly for a large portion of the year, fish from a wide size and age distribution are available to the survey, from young of year to large older spawners.

Weakfish abundance was moderate in the early 1980s and early 1990s, ranging between approximately 15-30 fish/nm (Figure C6.1-3). Beginning in 1992, abundance increased sharply to a time series high of over 230 fish in 1996. Abundance decreased by more than half in 1997, and has exhibited a generally declining trend since that time. CV of the composite index showed relatively high variability from 1991 to 1995, ranging from 20 to 60%. Interannual variability in CV stabilized in 1995 and generally ranged from 19to 26% until 2001. Since 2001, CV has shown a slight increase, estimated at 33% in 2006.

Age structure advanced from primarily age 1 and 2 fish in the early 1990s to include ages 7 and 8 in 1998-2000 (Figure C6.1-3). Abundance of age 4+ fish accounted for 30 to 35% of the total index in 1997 and 1998 as the large 1993 year class moved through. Abundance of older ages has since declined to levels observed in the early 1990s, with 4+ fish accounting for less than 1% of the total.

The Delaware 30-foot trawl survey occurs in one of the major weakfish spawning areas and has been shown to capture a wide size and age range of weakfish throughout the year. Trends in

abundance correspond well with anecdotal and observed information from commercial and recreational fisheries and are coherent with other indicators of weakfish abundance. The WTC has determined that the Delaware 30-foot trawl survey provides a reliable index of weakfish abundance.

C6.1.4 SEAMAP Fall Survey

The Southeast Area Monitoring and Assessment Program (SEAMAP) has conducted three seasonal trawl surveys since 1989 between Cape Hatteras, NC and Cape Canaveral, FL. A stratified random design is employed to sample inner (4.6 to 9.1 m) and outer (9.1 to 18.2 m) depth strata using twin 75-foot (22.9 m) highrise mongoose trawls towed behind a double rigged St. Augustine shrimp trawler. The geographic range of the survey encompasses nearshore ocean waters south of Cape Hatteras, and SEAMAP is the only fishery independent survey conducted in the southern portion of the weakfish range. Unfortunately, catches of weakfish south of North Carolina are extremely small and of little value as an index of abundance. An index of abundance is therefore generated using only strata off North Carolina during fall cruises. Catch is aged using survey specific length and age data (where available) or southern region late season ALKs, which are primarily North Carolina data. Fall aged fish are progressed one age to estimate January 1 abundance in the following year.

From 1990 to 2001, the survey ranged from approximately 5 to 30 fish per tow, with the exceptions of 1992 with an index of less than 1 fish per tow, and 1993 and 1994 with indices of approximately 44 and 52 fish per tow (Figure C6.1-4). From 2002 to 2004, the index increased to between 35 and 60 fish per tow, before jumping drastically to nearly 500 fish per tow in 2005. In 2006, the index dropped back down to 45 fish per tow. Removing 2005 as an outlier, the index has increased gradually throughout the time series. CV for the survey is relatively high, ranging from 24% in 1996 to 76% in 1997, with an average of nearly 48% over the time series.

Age structure is truncated in the survey catch-at-age matrix, and the survey is driven primarily by age 1 and age 2 fish (Figure C6.1-4). Barring the 2006 (lagged year) index value, strongest recruitment (age 1) events occurred in 1995 and 2003. The 2006 index is anomalously high, with an age 3 index greater than the age 1 index in most years. Age 4+ fish generally constitute less than 1% of the total catch, with a maximum of 11.2% in 1998 and 7.7% in 1999 as the strong 1995 recruits moved through.

The SEAMAP survey index is highly variable, with CV's greater than most other available weakfish indices. Regardless, this is the only offshore survey in the southern region of the weakfish range, and the WTC determined that it should be considered for inclusion in stock assessment analyses.

C6.1.5 Pamlico Sound Independent Gillnet Study (PSIGNS)

In May 2001, the NCDMF began a gillnet survey in Pamlico Sound to provide fishery independent relative abundance indices for key estuarine species. This is the first weakfish stock assessment for which a sufficient time series has been available from this survey.

The survey is conducted throughout the year, providing a mean index of abundance in the survey year (*i.e.* indices at age are not progressed to January 1). Sampling uses a stratified random design based on area and water depth. Twice per month a deep-water and shallow-water sample are collected from each of 8 areas using a gillnet consisting of eight 27.4 m segments of 7.6, 8.9, 10.2, 11.4, 12.7, 14.0, 15.2, and 16.5 cm (3.0" to 6.5" by half inch) stretched mesh gill net. Nets are typically deployed within an hour of sunset and retrieved the next morning, for approximate soak

times of 12 h. This sampling design results in a total of approximately 32 gill net samples (16 deep and 16 shallow samples) being collected per month across both the Rivers and Sound. Catch rates of target species are calculated annually and expressed as an overall CPUE along with corresponding length class distributions. The overall CPUE provides a relative index of abundance showing availability of each species to the study, while the length distribution and aged CPUE estimates show the size structure of each species for a given year. The overall CPUE was defined as the mean number captured per sample and was further expressed as the number at length per sample, with a sample being one array of nets fished for 12 hours. Due to disproportionate sizes among stratum and region, the final CPUE estimate was weighted. For weakfish, the CPUE at age was calculated for 6-month periods (Jan-Jun and Jul-Dec) in the same manner as was done for the harvest catch at age (see section **C5.1**).

Total CPUE showed a decline for the first few years of the survey, stabilized for a few years, and declined again in the last years (Figure C6.1-5). CPUE in 2007 was approximately one-third of the value in the first year of the survey. No estimates of precision were provided for this survey.

Age 1 fish are not fully recruited to the gear, and in most years it appears age 2 are also under represented as the index for age 3 fish is generally equal to or greater than the index for age 2 in the previous year (Figure C6.1-5). One prominent feature of this survey is the tracking of an apparent strong 1999 year class as it progresses through the first years of the survey.

Although the time series is short and no estimates of survey variability are available, the survey appears to adequately track abundance of weakfish across years. The WTC accepted the survey for use in the stock assessment.

C6.2 Young of year fishery independent surveys

C6.2.1 Massachusetts DMF Trawl Survey

The Massachusetts Division of Marine Fisheries conducts a stratified random trawl survey in six depth zones (0-9.1, 9.1-18.3, 18.3-27.4, 27.4-36.6, 36.6-54.9, and >54.9 m) and five geographic regions within the state. Sampling has been conducted twice per year (May and September) since 1978. Survey gear consists of a two-seam whiting trawl with a 11.9 m headrope and a 12.7 mm stretch mesh codend liner. Weakfish, primarily young of year, are most commonly observed during the fall survey in the three regions south of Cape Cod. Arithmetic mean catch per tow is used as an index of young of year abundance in the survey year.

The MA DMF trawl survey area encompasses nearshore ocean and estuarine areas within Massachusetts state boundaries. Like the New Jersey trawl survey, the survey area overlaps a portion of the NEFSC trawl survey area, but a smaller vessel allows more comprehensive sampling of shallow waters. Although large numbers of weakfish have been observed in Cape Cod Bay and Massachusetts Bay (Collette and Klein-MacPhee 2002), these waters are generally considered the northern extent of the weakfish range.

Mean annual catch per tow is consistently under 2 fish, with only three exceptions since 1981 (Table C6.2-1, Figure C6.2-1). Abundance generally declined from 1981 to 1984. In 1985, abundance increased more than 100-fold to the time series high of more than 15 fish per tow. Recruitment was again relatively high in 1986 (2.7 fish per tow), before dropping back to near zero levels for 1987 to 1994. Since 1994, abundance has shown a general upward trend, while at the same time exhibiting greater interannual variability. The second highest index value of 2.9 fish per tow occurred in 2006, before dropping back to just 0.2 fish per tow in 2007.

Standard errors were high and exhibited a similar trend as mean abundance. The CV was

generally greater than 60%, and exceeded 90% in eight years. Because of the low catch rates and high variability, the WTC has determined that this index provides little information on the abundance of weakfish. The survey was not used in the assessment.

C6.2.2 Rhode Island Trawl Survey

The Rhode Island Division of Fish and Wildlife seasonal trawl survey was initiated in 1979 to monitor recreationally important finfish stocks in Narragansett Bay, Rhode Island Sound, and Block Island Sound. The survey employs a stratified random design and records aggregate weight by species, frequency, individual length measurements, and various physical data. In 1990, a monthly component was added to the survey, which includes 13 fixed stations in Narragansett Bay. Sampling is conducted using a two seam trawl net with a 12.2 m headrope and 6.4 mm codend liner. Calibration studies have been conducted to ensure continuity of the catch series when changes in vessel and gear were required. An index of weakfish abundance is calculated as the geometric mean number per tow.

CPUE was relatively high in 1981 and 1982, but dropped off in 1983 and remained below 2.5 fish per tow until 1995 (Table C6.2-1, Figure C6.2-1). In 1996 and 1997, YOY abundance increased to more than 6 fish per tow, before dropping off to previous levels for 1998 to 2002. Since 2003, recruitment has shown great interannual variability, ranging from the time series high of 16.5 in 2003 to the second lowest value of 0.17 in 2006. CV of the log mean values are generally between 10 and 30%, with only two values (1995, 2006) exceeding 40%.

The RI survey occurs in a very small portion of the weakfish range and is outside the core area described by NEFSC (2000). However, several episodes of strong recruitment have been observed, and CV is moderate. For these reasons, the WTC has determined that the Rhode Island YOY survey is suitable for use in the assessment.

C6.2.3 Connecticut DEP Long Island Sound Trawl Survey

Since 1984, the Connecticut DEP has conducted spring and fall trawl surveys in the Connecticut portion of Long Island Sound between the New York/Connecticut border in the west and New London, CT in the east. Survey effort consists of three spring cruises conducted during April, May and June, and three fall cruises during September/October. Stratified random sampling is employed based on four depth zones and three bottom types. Survey gear consists of a 14 x 9.1 m high-rise otter trawl with 5 mm codend mesh. The survey catches mostly YOY and age 1 weakfish as defined by examination of length frequencies. Indices of abundance for age 0 and age 1+ are developed as geometric mean catch per tow, but only the YOY index was considered for this assessment.

Sampling is limited to Long Island Sound. The Sound encompasses a very small portion of the weakfish range, but may serve as a primary nursery habitat in this region.

From 1984 to 1998, the YOY index varied without trend, and generally ranged from approximately 3 to 10 fish per tow, with relatively strong year classes (10-15 fish per tow) occurring in five years (Table C6.2-1; Figure C6.2-2). In 1999, recruitment increased sharply and has remained above 30 fish per tow in all years except 2005 and 2006. Time series highs of more than 63 fish per tow occurred in 2000 and 2007, while minimum catches of approximately 1 fish or less occurred in 1984, 1986, and 2006. CV of the YOY index has exhibited a generally negative trend over the time series.

NEFSC (2000) recommended that this survey not be used as an index of abundance because it occurs outside the core area of weakfish abundance. However, large recruitment events have been

observed in this area over the last ten years, suggesting it may provide prime nursery habitat. In addition, precision of the YOY catches is strong. For these reasons, the WTC concluded that the Long Island Sound YOY index was suitable for use in the assessment.

C6.2.4 NYDEC Peconic Bay Juvenile Trawl Survey

The New York Division of Fish, Wildlife and Marine Resources has conducted a juvenile trawl survey in the Peconic Bay estuary of Long Island since 1985. Weakfish was the primary target species when the survey was initiated, and Peconic Bay was selected for the survey area because of its importance as a weakfish spawning ground. Random sampling occurs weekly between May and October using a semi-balloon shrimp trawl with a 4.9 m headrope and 12.7 mm stretch mesh codend liner. The survey samples mainly young of year weakfish, and a YOY index has historically been calculated as a geometric mean catch per tow over all sampling months. In 2005 and 2006, technical difficulties constrained sampling to May – July (2005) and July – October (2006), so a revised index using only July and August has been calculated. The two indices (all months and July-August) show a similar increasing trend and are well correlated ($r = 0.96$). The July-August index provides higher estimates of abundance and appears to be more variable between years, although CV is lower for the July-August index than for all months combined.

The July-August index ranges from less than one to more than 30 fish per tow (Table C6.2-1; Figure C6.2-3). Despite large interannual variations, there appears to be a gradual increase in recruitment over the time series. Strong year classes occurred in 1991, 1996, and 2005 (time series high). Standard error of the catch has increased over the time series as well; however, 95% confidence limits around the mean are moderate. The lower bound averages approximately 35% less than the mean value, while the average upper bound is approximately 50% larger than the mean.

Because this survey is conducted outside the apparent core area, NEFSC (2000) recommended that this survey not be used as an index of abundance. However, the survey was developed specifically to monitor trends in weakfish populations on an important spawning ground, and some strong year classes have been observed. Precision of the survey is acceptable. For these reasons, the WTC has used the Peconic Bay YOY survey in the assessment.

C6.2.5 Delaware DFW Delaware Bay Juvenile Trawl Survey

In addition to their 30-foot trawl survey, the Delaware DFW conducts a fixed station survey in Delaware Bay targeting juvenile finfish. Sampling is conducted monthly from April through October using a semi-balloon otter trawl. The net has a 5.2 m headrope and a 12.7 mm stretch mesh codend liner. Weakfish are a significant component of the catch, with the greatest majority of these weakfish (more than 99% in some years) being young of the year. A YOY index is calculated as the geometric mean number per tow during the June to October cruises.

As with the Delaware 30-foot trawl index, the survey is restricted to Delaware Bay. Although this encompasses only a small portion of the geographic range of weakfish, Delaware Bay is known to provide significant spawning and nursery habitat for the species.

Throughout this time series, recruitment indices have generally fallen between 5 and 15 fish per tow, with only 2 values below and three values above this range (Table C6.2-1, Figure C6.2-3). Weak recruitment occurred in 1983 and 1988, with less than 5 fish per tow, while the two strongest recruitment events of 20.1 and 16.8 fish per tow occurred in 1991 and 2005, respectively. Average recruitment over the time series has been approximately 10.8 fish per tow. The index indicates three general stanzas in recruitment since 1981. From 1981 to 1990, recruitment was generally below the long term average. In 1991, recruitment increased to the time series high, beginning a decade of

above-average recruitment. In 2001, recruitment dropped below average and has remained there for five of the last seven years. Estimated means appear precise, with 95% confidence limits typically only 25% to 35% larger or smaller than the mean value.

The Delaware young of year survey occurs within the core area of weakfish abundance and encompasses a major spawning/nursery area for the species during months when weakfish are present. The survey has captured the occurrence of several strong year classes with good precision. The WTC has used this survey in the stock assessment.

C6.2.6 Maryland DNR Chesapeake Bay and Coastal Bays Juvenile Trawl Surveys

The Maryland Department of Natural Resources conducts two juvenile trawl surveys: one in the lower eastern region of Maryland's portion of Chesapeake Bay from 1980 to the present, and one in the Atlantic coastal bays from 1972 to the present. Both surveys sample fixed stations using a 4.9 m semi-balloon otter trawl with a 12.7 mm stretch mesh codend liner. The coastal bays project samples monthly from April through October, while the Chesapeake survey runs monthly from May through October. Due to non-standardized survey methods during the early portions of both surveys, only data from 1989 onward are used to calculate YOY abundance indices. Indices are calculated as geometric mean catch per tow.

Both surveys are confined to Maryland state waters which constitute only a small portion of the weakfish range. Regardless, both survey areas are sheltered estuarine environments and may provide suitable spawning and nursery habitat for the species.

The Chesapeake index steadily increased from 0.4 fish per tow in 1989 to the time series high of 8.1 fish per tow in 2001 (Table C6.2-1, Figure C6.2-4). The index has steadily declined to less than 2 fish per tow in 2007. The coastal bays index appears stable between 0.9 and 1.9 fish per tow during1989-1994. In 1995, recruitment increased to 4.4 fish per tow and then decreased gradually to 2.6 in 2001. During the period 1989 to 2001, interannual variability was minor with few exceptions. Beginning in 2001, interannual variability increased dramatically. The weakest recruitment of the time series occurred in 2002, followed in 2003 by the time series high of 5.6 fish per tow. Coefficients of variation have ranged from 3-22% for the Chesapeake Bay survey and 4-15% for the coastal bays survey (except 30% in 2008).

Both surveys occur within the core region of weakfish abundance during months when weakfish would be present. Precision is uncertain, but the WTC has determined both are suitable for use in the assessment.

C6.2.7 Virginia Institute of Marine Science Chesapeake Bay Trawl Survey

The Virginia Institute of Marine Science (VIMS) has conducted a trawl survey in lower Chesapeake Bay since 1955. Over time there have been several changes to sampling strategy and survey area. Currently, sampling is conducted using a 9.1 m semi-balloon otter trawl with a 6.4 mm codend liner. Sampling occurs monthly throughout the year using stratified random sampling in the mainstem bay and fixed stations in tributaries. Young of year are identified through examination of length frequencies (monthly ranges), and an index of recruitment is computed as the geometric mean catch per tow during August to October from the three major tributaries.

The geographic region covered by the survey includes the Virginia portion of the Chesapeake Bay and lower portions of its three main tributaries (James, York, and Rappahannock Rivers). Although sampling does occur in the main stem, catches of weakfish are generally minimal in the Bay, so the index is limited to the three tributaries. Few large weakfish are present year round, but the estuaries provide suitable nursery grounds for juveniles.

Recruitment varies widely over the time series, ranging from less than 5 fish per tow to more than 35 fish per tow (Table C6.2-1, Figure C6.2-4). Interannual variability is often large, particularly in the early portion of the time series, with the maximum and minimum indices occurring in consecutive years (1985, 1986). From 1986 to 1990, the survey shows a rapid increase from 4.7 to 30.0 fish per tow, followed by a sharp drop back to 7.0 fish per tow by 1994. Recruitment rebounded slightly through 1999, but generally has been declining since.

No estimates of survey variability are available for the current index; however, 95% CIs for an index that includes Bay and River stations (data not shown) indicate good precision which has improved as the survey progressed. Sine 1989, CIs have generally been within 25 to 40% of the observed mean value. It could be expected that precision of the "river only" index would be greater, as catches of weakfish are less variable in the rivers than the Chesapeake Bay.

The VIMS trawl survey occurs within the core region of weakfish abundance during months when weakfish would be present. Precision is uncertain, but proxy data indicate low to moderate variability. The WTC has determined that this survey is suitable for use in the assessment.

C6.2.8 North Carolina DMF Pamlico Sound Juvenile Trawl Survey

The North Carolina Division of Marine Fisheries conducts a juvenile trawl survey in Pamlico Sound. Sampling is conducted in June and September using a stratified random design. Survey gear consists of twin 9.1 m mongoose trawl nets with 19.1 mm codend mesh. Data from these surveys are used to develop an ages 1+ index (June) and a YOY index (September), both based on length frequency analysis. Indices are calculated as geometric mean catch per tow.

Between 1987 and 1999, the YOY index ranged from approximately 1 to 60 fish per tow. Catch was characterized by large interannual fluctuations, but shows a consistent increase over that period from the time series low in 1987 to the time series high in 1999 (Table C6.2-1, Figure C6.2-5). From 1999 to 2002, recruitment dropped rapidly from 60 to 4 fish per tow. Since 2002, the index indicates a modest rebound to approximately 13 fish per tow in 2007. Since 2000, interannual variability is much less pronounce than in the early portion of the time series.

From 1987 to 1999, the ages 1+ index varies without trend between approximately 5 and 20 fish per tow. The index does capture some of the larger year classes observed in the YOY index, and peaks in 2000 (one year after the YOY peak) at 51.5 fish per tow. The index drops rapidly back to less than 5 fish per tow and remains stable for the remainder of the time series except for the 2006 index of nearly 30 fish per tow.

Variability for both indices was moderately large, with 95% CIs in many years deviating by more than 40% from the mean value. For the YOY index, variability decreased as abundance increased. For both surveys, variability has increased in recent years.

The survey area encompasses only a small portion of the weakfish range and survey variability for both indices is moderate. However, the survey occurs within a prime weakfish spawning/nursery ground and provides the only recruitment index in the southern portion of the range. The WTC has included both the YOY and 1+ indices in the assessment.

C6.3 Other fishery independent surveys

In addition to the fisheries independent surveys listed above for consideration as tuning indices, weakfish biological data are also obtained from two other trawl surveys in the Mid-Atlantic region. The Chesapeake Bay Multispecies Monitoring and Assessment Program (ChesMMAP) is a large mesh trawl survey that began in 2002 (Bonzek et al 2007). Sampling is conducted bi-monthly from March through November, targeting juvenile to adult finfish throughout the main stem of the

Chesapeake Bay. The Northeast Area Monitoring and Assessment Program (NEAMAP) Trawl Survey is a nearshore ocean trawl survey operating between Cape Hatteras, NC and Montauk, NY (Bonzek et al 2008). The intent of this survey is to sample nearshore ocean waters which have typically been under represented in NEFSC trawl surveys. A pilot survey was conducted in 2006, and bi-annual (spring and fall) cruises have occurred since.

Although the time series from these surveys are not yet sufficient to provide trends, weakfish biological data collected during survey cruises have been made available for use in the stock assessment. When a sufficient time series has been attained, the WTC will evaluate these surveys as candidates for tuning indices.

Recently, information was obtained on weakfish juvenile abundance collected through the Georgia Coastal Resources Division Ecological Monitoring Trawl Survey (EMS). Sampling occurs at 43 fixed stations throughout six coastal sounds using a 12.2 m flat otter trawl equipped with 4.8 cm stretch mesh in the codend. The weakfish index is calculated from sound and creek strata during monthly cruises between July and October. Juveniles are identified by monthly length cutoffs. The survey began in 2003, so the time series is currently too short. When additional years of data are available, the WTC will re-evaluate this survey.

C6.4 Fishery dependent surveys

Historically, a fishery dependent index of weakfish abundance was developed using recreational catch per "directed trip" (trips where weakfish was identified as a target species; *cf.* NEFSC 1998b, 2000). During the review of the 2000 assessment, the SARC expressed concerns about fishery dependent indices, and they were removed from the assessment. Potential sources of bias in fishery dependent indices include non-random distribution of effort, and hyperstability of the index (Hilborn and Walters 1992).

In 2006, a revised recreational index was developed that the WTC feels largely addresses the concerns expressed by the 30th SARC. Estimates of catch include all weakfish (A+B1+B2) captured by the recreational private/rental boat mode in state waters of the mid-Atlantic region (New York to Virginia). As described in Crecco (2005a), the private/rental mode is highly mobile and capable of catching weakfish over a large range of sizes. Catches were constrained to the mid-Atlantic region because private/rental boat catches from this region have accounted for greater than 60% of annual recreational catch. Two estimates of effort were used to convert catch to CPUE. The first uses all private/rental boat trips in state waters of the mid-Atlantic region (Crecco 2005a). The second, detailed by Brust (2004), estimates effort as the number of private/rental boat trips in mid-Atlantic state waters that captured any of a suite of species typically associated with catches of weakfish. The two indices were highly correlated ($r > 0.98$). Although the 2006 assessment used the index based on the suite of associated species, the current assessment uses the index based on all mid-Atlantic private/rental boat trips because of its ease of calculation without loss of information, and because it provides consistency between assessment methodologies.

The methods described above provide an index of total catch per trip, including both harvested and discarded fish. Based on assumptions regarding availability to the fishery (gear and area) at size, this index is considered to represent ages 1+. Age specific indices for harvested fish were developed by dividing total harvest by the effort index (Mid-Atlantic private/rental boat trips) and applying the age structure of the recreational harvest of 2+ fish. When the index is lagged forward, this method provides indices at age for 3 through 6+.

Both indices (age aggregated 1+ and age specific 3 to 6+) are developed using data from mid-Atlantic state waters. This region encompasses the primary distribution of weakfish within its

range. Although all sizes and ages of weakfish are present in this region, younger fish are not considered to be captured or harvested by the fishery, so the indices are only representative of mature fish.

CPUE of all ages varied greatly from 0.2 to 0.8 during 1981 to 1988 (Figure C6.3-1). From 1989 to 1993, CPUE was stable between 0.15 and 0.2, following which it increased rapidly to a peak of 0.61 in 1996. Since that time, CPUE has exhibited a consistent decline to the time series minimum of 0.09 in 2007.

During the early portion of the time series, age structure of harvested fish was dominated by fish aged 2 and 3 (Figure C6.4-1). By the mid 1990s, minimum size limits and expanding age structure increased the influence of older fish, and 4 year old fish were the most prevalent age group between 1996 and 1999, with 5 year olds dominant in 2000. Since 2001, older ages have fallen off to near zero levels, and harvest consists mostly of ages 2 and 3.

Estimates of precision are available for the different components of the indices (catch, harvest, and effort); however, there are no direct estimates of survey precision. Discussions on uncertainty in catch and effort estimates are presented in section **C5.2**.

Fishery dependent indices were employed in some of the early stock assessments, but were dropped in subsequent years due to concerns over their reliability. During the 2006 assessment, the WTC developed a new recreational index that covers the entire core area of the population range based on a highly mobile fleet whose effort is not specific to weakfish. The index covers the entire core area of the population range, is highly mobile, but effort is not specific to weakfish. Results are coherent with other indicators of weakfish population size, as well as harvest and the converged portion of the VPA (see Section **C10.0** of this report). The WTC has therefore determined that the fishery dependent indices are suitable for use in the stock assessment.

C6.5 Discussion

Of all surveys reviewed, only NEFSC and MA YOY were excluded from further analysis due to concerns over information content. The remaining indices include age specific indices from four fishery independent surveys (NJ, DE, NC gillnet, SEAMAP), nine fishery independent young of year surveys, and age specific indices from the recreational fishery. Although previous peer reviews recommend limiting indices to the core population area, the WTC has determined that three YOY surveys from outside the core area provide reliable information on weakfish recruitment, and these surveys have been included in modeling investigations.

The informative value of different tuning indices have received a lot of consideration during recent reviews of the weakfish assessment, either as a work in progress (SAW 40, 2008 DPSW) or the completed work (ASMFC 2006a). Each review panel had varying opinions of the indices, but a common recommendation from all three is that the indices need further evaluation. The WTC has attempted to address these concerns as well as possible (*e.g.* ASMFC 2006a, Parts A and C; Technical Committee response to 2008 DPSW panel report (Appendix C-1)), and a summary of recent work is presented in Section **C10.0** of this report. Most recently, the DPSW panel recognized and supported much of the work done by the WTC to address previous concerns, but still identified several issues with the indices (Miller et al 2009, bullets d through f). These concerns are summarized below, as are the WTC's responses to the panel report.

In bullet d) of the panel report, the panel recognizes that decisions to exclude indices were based on "several valid reasons." They continue, however, by stating that the remaining indices may not be indicative of true population dynamics because they "were restricted to limited spatial areas within the overall weakfish stock area." Further, they were concerned that similar trends in

remaining age-aggregate indices could be due to these indices containing little information. The WTC recognizes that the remaining fishery independent surveys are localized; however, the review panel supported the exclusion of the only coastwide index (NEFSC) for "valid reasons." In addition, the three remaining "aged" fishery independent surveys occur in the state with the most commercial landings (NC gillnet), the largest recreational harvest (NJ), and a principal spawning area (DE 30-foot). When considered on an age-aggregated basis, all three surveys show the same general trend, as does the recreational index which encompasses the entire core region. These findings suggest that the core area is adequately covered. In response to the concern over limited data content, the WTC points out that the trends in indices reflect changes in commercial CPUE, population trends from previous assessments, and observed age structure from fishery dependent and independent sampling.

Bullet e) of the panel report identifies concerns regarding the recreational index. In particular, the Panel expressed concerns that the MRFSS index was one of few that exhibited any clear pattern, which could be due to anglers switching methods as abundance of a more favorable species (striped bass) increased. The WTC responds that the pattern observed in the MRFSS index is coherent with three other aged surveys. Only two aged indices – NEFSC and SEAMAP - showed trends different from MRFSS and the others. Further, NEFSC was found to be uninformative, and the SEAMAP index exhibits moderate to large CVs. Although no alternative estimators were provided in the panel report, one suggestion made during the face to face meeting was found to produce the same general trend as the MRFSS index presented.

In bullet f), the review panel recognizes that although four surveys (NJ, DE, NC gillnet, MRFSS) show coherence when age aggregated, VPA results vary greatly when using only fishery dependent or fishery independent tuning indices. Preliminary evaluation of this issue was presented in the WTC response; however, further work identified incorrect age data in the MRFSS index used for preliminary runs. Additional information on VPA runs is presented in section **C7.0**.

In summary, the WTC recognizes that many of the concerns expressed over the last few years regarding the tuning indices are valid concerns. Significant time has been spent investigating and evaluating each of these concerns. The result is a set of tuning indices that the WTC considers is representative of weakfish population dynamics throughout the core region.

Another topic that has been gaining popularity in fisheries stock assessment is the process of standardizing abundance indices relative to temporal, spatial, and environmental factors. The three most recent weakfish peer reviews have all recommended that available weakfish indices of abundance be scrutinized to better understand their information content. During sampling, factors external to population dynamics may contribute "noise" that, if not accounted for, is generally attributed to fluctuations in population size. Standardizing indices using methods such as generalized linear models (GLM) and generalized additive models (GAM) seeks to identify factors that contribute to survey variability and minimize their influence, in order to better observe the true population signal.

Recently, a team of researchers from Virginia Polytechnic Institute and State University (Virginia Tech) has begun investigating the use of CPUE standardization for weakfish (Winter et al 2009). Raw data from nearly all abundance indices (fishery independent and fishery dependent) considered by the WTC were analyzed using GLM and (where appropriate) GAM. Preliminary results were presented to the WTC in April 2009. The WTC was encouraged by the results, but many questions remained. For example, indices investigated by Winter et al (2009) were not all directly comparable to indices considered by the WTC, in particular the New Jersey (CPUE all cruises vs percent positive tows during August cruise) and fishery dependent indices (Mid-Atlantic private boat CPUE vs CPUE for all sub-regions and modes). In the case of the fishery dependent

index, factors were included to account for sub-region and mode, but the discrepancies made comparisons difficult. Also, preliminary results did not include summary statistics of the various factors investigated for each index. It was not possible, therefore, for the WTC to evaluate which factors were significant or the proportion of total variability explained by each factor.

The researchers from Virginia Tech were receptive to comments and suggestions from the WTC, and the full report, updated with responses to some of the WTC's concerns (*e.g.* p-values for different factors), is included with the materials for SAW 48 as a supplementary report for weakfish (entitled "Winter et al 2009 – Weakfish CPUE standardization.doc"). However, given the WTC's concerns regarding the preliminary results, and the short timeline available to complete this assessment for peer review in June 2009, the WTC has determined that there was insufficient time to incorporate standardized indices into the current assessment. The WTC supports continuing this work to address the research recommendation and for consideration in future stock assessments.

C7.0 Evaluate the ADAPT VPA catch at age modeling methods and the estimates of F, Z, spawning stock biomass, and total abundance of weakfish produced, along with the uncertainty and potential bias of those estimates. Review the severity of retrospective pattern. (TOR #3)

C7.1 Introduction

Age structured modeling was conducted using ADAPT VPA (version 2.8, available for download at *http://nft.nefsc.noaa.gov*) as recommended by the 30th SARC (NEFSC 2000). Development of catch at age inputs from four fishery sectors (commercial harvest, commercial discards, recreational harvest, and recreational discards) is discussed in detail in section **C5.0** (see Table C5.3-1). A total of 38 abundance indices were considered suitable for use in the assessment (section **C6.0** and Figures C6.1-1 through C6.4-1), including 24 age specific fishery independent indices (four surveys age 1 to 6+), nine fishery independent young of year surveys, four age specific fishery dependent indices (MRFSS 3 to 6+), and one fishery dependent age aggregated index (MRFSS 2+). Specific configuration of the model, such as biological specifications and options selected, is presented in the supplementary report entitled "Weakfish ADAPT final run output.txt."

Multiple runs were conducted using various combinations of tuning indices and/or modified input values (sensitivity runs). All model runs were reviewed and the preferred model run was selected based on evaluation of model fit, residuals, retrospective patterns, and other considerations.

C7.2 General findings

Regardless of tuning indices selected, all runs using baseline data (calculated CAA, constant M = 0.25) provided consistent, virtually identical results through 2002. Trends in estimated parameters for the years 2003 to 2007 varied widely, as did the magnitude and duration of retrospective patterns. Model runs that included fishery dependent indices produced more conservative results (lower abundance, higher F), lower mean squared residual (MSR), and less severe retrospective patterns than those tuned solely with fishery independent indices (Table C7.2-1; Figure 7.2-1 through 7.2-3). When YOY indices were used to tune the model, the magnitude and duration of retrospective patterns were greater than when YOY indices were excluded, but exclusion of YOY indices did not necessarily result in lower MSR.

Significant consideration was given to selection of tuning indices. Section **C10.0** provides a detailed summary of work conducted to evaluate indices based on criteria such as size structure of the catch, and coherence with other indices, landings and the converged portion of the VPA. This

method resulted in the NEFSC fall trawl survey being dropped from consideration, and the New Jersey ocean trawl index being changed from average CPUE to proportion positive tows (PPT). Further work was conducted to investigate combinations of indices that produced consistent results while minimizing the retrospective pattern. Some committee members cautioned that this method of index selection might appear subjective, and recommended an alternate method of index selection. The proposed method developed a list of criteria against which each index could be scored, including survey design, spatial and temporal coverage, and catch characteristics. Indices receiving the highest total scores would be selected as tuning indices. An exploratory implementation of this method selected NEFSC ages 1-2, DE 30 foot trawl age 1, NC gillnet age 1, SEAMAP ages 1-6+, and the RI, CT, NY, DE, MD, and NC YOY indices for use in tuning the model. During the converged portion of the time series (1982 to 2002), ADAPT results using these tuning indices were consistent with results from other runs with other tuning indices (Figure C7.2-4); however, the WTC was concerned that the magnitude and duration of the retrospective pattern in recent years were greater using this method than observed in other runs (Figure C7.2-5), and MSR for this run was among the highest observed (Table C7.2-1). In addition, the WTC was concerned that this method selected two ages from the NEFSC survey, which the WTC had excluded based on poor performance relative to established criteria, and all 6 ages of the SEAMAP survey, which is confounded by large interannual variability and an extreme outlier in 2005 that likely influences estimates in recent years. Based on these concerns, the WTC did not pursue this method any further for this assessment. It is recognized that alternative criteria and/or a different cut-off score for "acceptable" indices might influence the results, but additional work was postponed until completion of the ongoing assessment.

The WTC also investigated the use of tuning indices that were standardized relative to spatial, temporal, and environmental factors using methods such as GLM and GAM (Winter et al 2009; see section **C6.5**). Exploratory model runs using standardized indices were conducted in ADAPT VPA and compared to results using non-standardized indices. Winter et al (2009) indicate that GAM methods were superior to GLM for modeling weakfish catch rates. Model runs were therefore conducted using GAM standardized indices where possible, and GLM only for those indices that could not be standardized using GAM. Because the MRFSS harvest index was not standardized, comparisons were restricted to runs using only fishery independent indices. Also, indices developed by Winter et al (2009) were not all directly comparable to indices considered by the WTC, in particular the New Jersey (CPUE all cruises vs. proportion positive tows during August cruise). Regardless, ADAPT results using standardized indices were very similar to comparable runs using non-standardized indices and produced somewhat smaller retrospective patterns (Figures C7.2-6 and C7.2-7). As noted in section **C6.5**, although the WTC is encouraged by these findings, initial results of the index standardization analysis were presented too late in the development of this assessment report to be fully evaluated. The above discussion is based on examples of preliminary work that have not been fully endorsed by the WTC. The WTC will address their concerns regarding the index standardization analyses and continue to investigate the use of standardized tuning indices for future assessments.

The 2008 DPSW panel expressed concern over the inconsistency of model results between runs using only fishery dependent tuning indices and runs using only fishery independent indices. In the preliminary results presented during the 2008 DPSW, the New Jersey, Delaware, North Carolina gillnet, and MRFSS CPUE indices all showed a similar pattern in abundance when aggregated across ages; however, when disaggregated and used to tune the VPA, the fishery dependent results were inconsistent with the results using fishery independent indices. These results suggested that

one or more of the indices performed poorly at tracking weakfish age structure. The DPSW reviewers suggested that the WTC investigate this discrepancy. Preliminary results of these investigations were presented in the WTC response to the review panel's comments (see Appendix C-1). Additional evaluation shows, however, that the results presented at the DPSW were inaccurate due to incorrect age allocation of the fishery dependent indices. Updated results show that runs using only NJ, DE, or MRFSS indices produce consistent results (Figure C7.2-8).

C7.3 Preferred model run

C7.3.1 Model output

The previous (2006) assessment selected a model run tuned with only the fishery dependent indices, mainly because it produced a greatly reduced retrospective pattern relative to other runs. For this current assessment, the fishery dependent only run still produced the smallest retrospective (Figures C7.2-2 and C7.2-3) and best model fit (Table C7.2-1); however, including indices from the New Jersey ocean trawl, Delaware 30-foot trawl, and North Carolina gillnet surveys produces nearly identical trends in recent years with only marginal increases in the retrospective pattern. MSR nearly doubled relative to the fishery dependent only run, but is still lower than all other runs investigated. Minor improvements were made to the output by expressing index values to four decimal places and including preliminary estimates of abundance in 2008 for all selected tuning indices where available (only NC gillnet data were not). This run is hereafter referred to as the "final" or "preferred" run. Index values are shown in Table C7.3-1. Selected parameter estimates are presented in Figures C7.3-1 and C7.3-2, and the full model output is provided in the supplementary report entitled "Weakfish ADAPT final run output.txt."

Following the 2006 assessment, the WTC received criticism for not including any fishery independent tuning indices. The preferred run for the current (2009) assessment addresses this concern by including fishery independent indices from New Jersey, Delaware, and North Carolina along with the fishery dependent tuning indices. However, another criticism received by the WTC following the 2006 assessment was the selection of indices that produced desired results. As all the tuning indices used in the preferred run of the current assessment exhibit the same trend, this criticism may reappear. The 2008 DPSW suggested that these indices were coherent because they contained little information about weakfish abundance (Miller et al 2009). The WTC argues that these surveys were selected, not because they produce the desired result, but because they most accurately reflect other indicators of stock dynamics. These indices correspond well with changes in harvest, abundance, CPUE, and population age structure. It is doubtful that indices that lack useful information would track these parameters so closely. In addition, parameter estimates derived using tuning indices selected based on scored criteria were nearly identical to results of the preferred run for the years 1982 to 2002. The retrospective pattern based on scored criteria was longer and of much greater magnitude than the preferred run.

Estimates of fishing mortality (unweighted average, ages 4-5) from the preferred run approached or exceeded F = 1.5 in most years between 1982 and 1988 (Table C7.3-2, Figure C7.3-1). Between 1989 and 1994, F values generally varied around F = 1.0 before dropping rapidly to the time series low of F = 0.32 in 1995. A nearly exponential increase ensued during most of the following decade, peaking in 2003 at the time series high of F = 2.39. Fishing mortality rates decline after 2003, and the terminal year estimate of $F_{2007} = 0.51$ (80% CI range 0.38 to 0.82) is the second lowest value in the time series. A prominent retrospective pattern (Figure C7.3-2) indicates estimates since 2003 may be inaccurate (see section **C7.3.2**).

Between 1982 and 1985, spawning stock biomass varied generally between 8,000 and 13,000 MT, increased rapidly to approximately 20,000 MT in 1986-1987, then returned to its previous range during 1988 to 1993 (Figure C7.3-1). A gradual increase followed until SSB reached a relative peak around 19,000 MT in 1997. Since that time, SSB has undergone a gradual decline to the time series low of 1,987 MT in 2005 before rebounding slightly in the last two years. SSB in the terminal year is estimated as SSB_{2007} = 7,236 MT. Bootstrapping indicates an 80% CI range for SSB in 2007 from 5,300 to 10,733 MT. A prominent retrospective pattern (Figure C7.3-2) makes these estimates unreliable.

From 1982 to 1984, total abundance was estimated to range between 80 and 100 million individuals per year (Figure C7.3-1). Abundance increased rapidly to the time series high of 120 million in 1986, then declined by more than 70% in just 4 years to less than 50 million individuals in 1989. A steady increase was observed between 1989 and 1994, peaking at around 90 million. A nearly continuous decline has been observed since that time, with the time series low of just 10.6 million individuals estimated in 2005. Recent years' estimates are slightly higher, and the terminal year is estimated as N_{2007} = 32.8 million with an 80% CI range of 11.95 to 90.05 million. A prominent retrospective pattern (Figure C7.3-2) makes these estimates unreliable.

Recruitment to age 1 followed a similar trend as total abundance (Figure C7.3-1). The time series high of 70.9 million in 1986 was followed by a sharp decrease to just 20.8 million four years later. Recruitment increased steadily to around 49.3 million in 1994 and has since declined, with the time series low of just 4.5 million recruits estimated in 2005. Recent years' estimates are slightly higher, and the terminal year is estimated as R_{2007} = 16.4 million with an 80% CI range of 3.8 to 59.1 million. A prominent retrospective pattern (Figure C7.3-2) makes these estimates unreliable.

Survey residuals tend to show a strong serial pattern, particularly for the younger ages (Figure C7.3-3). Abundance at age of the New Jersey and Delaware trawl surveys is generally overestimated by the model during the early years of the surveys, shifting to an underestimation during the mid to late 1990s. The same pattern is observed for the recreational harvest index of abundance (aggregated ages 2-6+). In contrast, the recreational harvest indices are underestimated by the model during the first decade of the time series, but the early 1990s, residuals have shifted to negative values before increasing back to positive values since 2002.

C7.3.2 Retrospective pattern

As in previous stock assessments (NEFSC 2000, Kahn 2002, ASMFC 2006a, Part A) reliability of estimates in recent years is diminished by the presence of a prominent retrospective pattern. For the current assessment, fishing mortality is underestimated for the most recent five to seven years, while SSB, total abundance, and recruitment are all overestimated (Figure C7.3-2). The degree of error is substantial, with fishing mortality in 2003 increasing 130% from F_{2003} = 1.04 when 2003 is the terminal year to F_{2003} = 2.39 when data through 2007 are included. Similarly, SSB_{2003} decreases by nearly 75% from 8,282 MT to 2,190 MT between terminal years 2003 and 2007. Improvements to the ADAPT VPA model implemented in 2007 (version 2.6) allow users to combine retrospective analysis with bootstrapping (ADAPT version history, available at *http://nft.nefsc.noaa.gov*). Results of such an analysis further elaborate the degree of uncertainty in recent year estimates. Frequency distributions of bootstrapped estimates of F_{2005} from terminal year 2005 and terminal year 2007 do not overlap, and those of SSB overlap only slightly (Figure C7.3-4). Estimated 80% confidence intervals do not overlap for either parameter (Table C7.3-3).

Despite a significant retrospective pattern, the 30th SARC determined that results from the ADAPT model could be used to establish reference points and provide guidance to management

(NEFSC 2000). Quantitative analysis of the retrospective pattern observed in this assessment shows that estimates differ by more than 100% between the time they are made in the terminal year and five years later when estimates converge (Figure C7.3-2). In addition, bootstrap confidence intervals of terminal year estimates are not sufficient to capture the uncertainty (Figure C7.3-4, Table C7.3-3). The WTC has concluded that ADAPT parameter estimates in recent years are unreliable and should not be used for management at face value. An attempt to correct parameter estimates for the retrospective pattern is presented below. The WTC does not necessarily endorse the corrected estimates, but is presenting them here for review and discussion by the 48th SAW/SARC.

An attempt was made to correct parameter estimates in recent years using the observed retrospective pattern. For this analysis, it is assumed that parameter estimates in 2003 using data through terminal year 2007 have "converged" and are robust to additional years of data being added in the future, and that the retrospective pattern observed since 2003 continues into the future. Violation of these assumptions will invalidate the following retrospective correction analysis.

The estimate of F_{2003} calculated for terminal year 2007 was divided by estimates of F_{2003} from each of the other terminal years 2003 to 2006 to calculate a ratio of change for each terminal year relative to the year it would converge. Assuming four years of additional data are required for a terminal year estimate to converge, estimates of F_{2004} to F_{2007} from terminal year 2007 were multiplied by the appropriate ratio of change before convergence. For example, the ratio of F_{2003} calculated in 2007 relative to 2005 (2.39/2.09 = 1.1460) indicates how much F_{2003} changed in the last two years before stabilizing. F_{2005} is expected to converge in terminal year 2009, or two years after the current estimate with data through 2007 (F_{2005} = 2.07). A corrected estimate of F_{2005} in 2009 was calculated by multiplying the estimate of F_{2005} from 2007 by the ratio of change in the last two years before convergence. Specifically, $convergedF_{2005} = 2.07 * 1.146 = 2.37$.

Similar methods were used to correct estimates of SSB, January 1 abundance, and recruitment.

Results of the retrospective correction analysis are presented in Figure C7.3-5. Correcting for the retrospective pattern produces higher estimates of F and lower estimates of SSB, total abundance, and R during the period 2003 to 2007. As noted above, these results are dependent on the assumption that estimates from fishing year 2003 have stabilized with data through 2007 and that the observed retrospective pattern continues into the future. There is no way to validate these assumptions without additional years of data.

C7.3.3 Total mortality

Within ADAPT VPA, calculation of fishing mortality is constructed around the Baranov catch equation (NEFSC 2003). Given abundance of age a+1 in year t+1 and catch of age a in year t, total mortality on age a in year t is calculated iteratively using the equation

$$\frac{N_{a+1,t+1}}{C_{a,t}} = \frac{Z_{a,t}}{F_{a,t}} * \frac{e^{-Z_{a,t}}}{1-e^{-Z_{a,t}}} .$$

Fishing mortality on age a in year t is then estimated as

$$F_{a,t} = Z_{a,t} - M .$$

Following this procedure, estimates of fishing mortality are dependent on assumptions regarding natural mortality. Incorrect assumptions (*i.e.* inaccurate input values) for M would produce inaccurate estimates of fishing mortality. No direct estimates of natural mortality are available for weakfish, so the WTC has historically operated on the assumption of constant M = 0.25 across all years and ages. Alternative assumptions would produce different fishing mortality trends than presented above. If M is indeed constant but of a different magnitude, F estimates would follow a similar trajectory but be scaled up or down from the current estimates. Of greater concern is natural mortality that is not constant across time. Assumptions of constant natural mortality when M is actually increasing over time would lead to overestimates of fishing mortality. Alternatively, a negative trend in M would result in F being underestimated.

To circumvent these issues regarding assumptions about M, it is possible to increase the ADAPT estimated fishing mortality rates by the value of input natural mortality to evaluate trends in total mortality. Since natural mortality was assumed constant at all ages for all years (M = 0.25), the trend in total mortality is the same as the trend in F scaled up by 0.25 (Figure C7.3-1). Z declines from values between 1.5 and 2.0 in the early 1980s to the time series low of 0.57 in 1995. In the years following, Z increased nearly exponentially to a peak of more than 2.6 in 2003. Correcting for the retrospective pattern seen in F, total mortality appears to level off at approximately 2.5 to 2.6 before dropping by about 45% in 2007 (Figure C7.3-5).

C7.3.4 Other age-based models

ADAPT VPA operates under the assumption that catch at age is known without error. If catch at age is mis-specified in the input data, the errors are carried through to the results of abundance and fishing mortality at age. If errors in the CAA matrix are known or assumed to be large, other age structured models that are more robust to CAA uncertainty should be explored.

For weakfish, there are several known sources of uncertainty in the CAA. These include conversion of scale-based ages from the 1980s and early 1990s to otolith-based ages used since around 1994, and the substitution of commercial length frequency data to characterize fisheries in states with insufficient data (see section **C5.0**). During the development of the 2002 stock assessment update, the WTC began investigating the use of Integrated Catch at Age Analysis (ICA), which incorporates a statistical error model and is not dependent on error-free CAA (de Silva 2002). Due to loss of staff, this methodology was not pursued beyond the initial investigation.

The most recent peer reviews of the weakfish stock assessment, either as a work in progress (SAW 40, 2008 DPSW) or the completed work (ASMFC 2006a), have focused on concerns with input data. The 2008 DPSW review panel recommended the WTC investigate the use of a forward projecting statistical catch at age model which is robust to errors in the CAA (Miller et al 2009). On that recommendation, the WTC has begun investigating the use of the Age Structured Assessment Program (ASAP) available through the NEFSC toolbox. Trends in fishing mortality and SSB generally follow similar patterns as those estimated in ADAPT (Figure C7.3-6). To date, only preliminary runs have been completed, and the results presented should be considered as examples. The WTC has not had sufficient time to fine tune the model inputs, nor evaluate the reasons for, or the implications of, the discrepancies between the two models. The WTC will continue investigating the use of ASAP and/or other age-structured models for use in future stock assessments.

C7.4 Discussion

Fishing mortality estimated using ADAPT VPA exhibited a generally declining trend between 1982 and 1995, followed a nearly exponential increase during the next decade, and then dropping off in recent years. SSB generally declined through the 1980s, exhibited some rebuilding during the mid to late 1990s, then declined to record lows by 2005. A number of exploratory modeling runs using available data indicate good precision of estimates between 1982 and about 2002. From 2003 to 2007, different sets of tuning indices produced different parameter estimates, and all were compromised by a prominent retrospective pattern. Fishing mortality is generally underestimated, while SSB, January 1 stock abundance, and recruitment are overestimated. Results after 2002 are therefore considered unreliable. An attempt was made to correct for the observed retrospective pattern, but the results have not been endorsed by the WTC.

Implementation of ADAPT for this and previous stock assessments has been conducted under the assumption of constant natural mortality. Violation of this assumption results in inaccurate fishing mortality estimates. The WTC has strong reservations about this assumption and has expressed concern regarding the trend in fishing mortality calculated through ADAPT. As an alternative, the WTC prefers to combine calculated F estimates with input M rates to portray a trend in total mortality, Z. The trend in Z is identical to the trend in F scaled upward by 0.25, and is also influenced by the retrospective pattern in recent years. Regardless, estimates of Z prior to 2002 are not limited by assumptions regarding M and are therefore considered more accurate.

ADAPT VPA has long been the accepted modeling approach for weakfish based on the recommendation of the 26th SARC (NEFSC 1998b). The subsequent peer review (NEFSC 2000) determined that estimates based on ADAPT VPA were suitable for management. In recent years, however, the WTC has doubted the utility of ADAPT for a number of reasons. First, the severe retrospective pattern indicates that parameter estimates in the last five years or more are unreliable. A combined bootstrap and retrospective analysis provides evidence that bootstrapping is insufficient to capture the uncertainty in terminal year estimates. As such, the WTC considers 2002 or 2003 as the most recent year of "acceptable" parameter estimates. Estimates that are now more than five years old are of limited utility for determining future management strategies. Secondly, calculations within ADAPT are conditional upon certain assumptions, either hard programmed (*e.g.* error-free catch at age) or user defined (*e.g.* constant M). The WTC is aware of several sources of uncertainty in the weakfish catch at age matrix, including the scale to otolith conversion, substitution of commercial samples, and commercial discards estimates. The extent of error is unknown, but could be substantial, making ADAPT estimates unreliable. In addition, without direct evidence suggesting otherwise, the WTC has historically assumed natural mortality was constant over all ages and years. This assumption was called into question during development of the 2004 stock assessment when it was observed that stock size began declining under low fishing pressure. Additional work (presented in later sections) provides evidence that suggests weakfish productivity has changed over the available time series, which may be affecting our evaluation of the stock and appropriate management strategies.

The WTC has been tasked to evaluate weakfish stock status in order to support sustainable management of the stock. Without an approved alternative analytical method, results of ADAPT VPA modeling are presented for review and discussion by the 48th SARC. However, given the concerns presented above, the WTC does not believe the terminal year estimates are suitable for management purposes. Beginning with the 2004 assessment, alternative analytical methods have been (and will continue to be) investigated in an attempt to find a more suitable model that is not constrained by (*i.e.* not dependent on, or at least more robust to violations of) assumptions

underlying the current model. The WTC also intends to continue evaluating available data sources to improve historical and future input data. The WTC welcomes specific comments from the SARC on how to improve the input data and model parameterization, and/or comments on appropriate alternative analytical methods. The goal is to find a suitable method that is capable of accurately depicting recent and historical trends in abundance and mortality rates to allow ASMFC the ability to manage for the long term sustainability of the stock.

C8.0 Evaluate the index based methods and the estimates of F, ages 1+ stock biomass, surplus production, and time-varying natural mortality of weakfish produced, along with the uncertainty of those estimates. Determine whether these techniques could complement or substitute for age-based modeling for management advice. (TOR #4)

C8.1 INTRODUCTION

Our ability to assess the current status of Atlantic coast weakfish has been continually plagued by a pronounced retrospective bias associated with all previous ADAPT model runs. Previous ADAPT runs made with trawl-based tuning indices (Kahn et al 2006) exhibited severe systematic retrospective bias that underestimated F and overestimated stock size in the most recent years by as much as 80%. The degree of retrospective bias from ADAPT was so severe that the results gave the false impression that weakfish stock size had actually risen to record high levels by 2002 despite the presence of record low coast-wide landings after 1999. Such a large systematic bias greatly confounds our ability to determine whether or not weakfish abundance has fallen and whether or not the stock is overfished. The degree of retrospective bias from ADAPT was reduced to some extent when the model was tuned specifically to recreational catch-effort data from the MRFSS (Kahn et al 2006).

Because of the aforementioned limitations thus far in the catch-at-age approach, the 2006 Assessment (Kahn et al 2006, Uphoff 2006a; Crecco 2006) relied primarily on an index-based (ages 1+) method to monitor temporal changes in weakfish biomass (mt) and fishing mortality (F) from 1981 to 2003. In the 2006 assessment, relative weakfish abundance was represented solely by the MRFSS recreational cpue from the Mid-Atlantic private boat sector of the recreational fishery. More recently, Uphoff (2008) reported that trends in the MRFSS recreational indices as well as trawl based indices from the States of New Jersey (August data only) and Delaware were highly correlated ($P < 0.01$) with trends in coast-wide landings and with ages 1+ weakfish abundance from the converged portion (1981-2001) of the most recent VPA model run. Since the index-based approach produced a trend in ages 1+F that closely followed the F trend from the converged (1982-2000) portion of ADAPT without having to rely on the often untested assumption of a fixed M (i. e. F = Z-fixed M), a blended index was derived here based on the recreational cpue, New Jersey and Delaware trawl indices and used to update ages 1+ F and stock biomass estimates (mt) from 1981 to 2008.

Results from the last assessment (Kahn et al 2006) revealed that overfishing (F > Fmsy) had occurred on weakfish from about 1981 to 1991. More stringent fishery regulations were imposed on the weakfish recreational and commercial fisheries in 1992 and 1993 after which fishing mortality (F) rates fell to well below Fmsy from 1996 to 2003. Following a drop in F, coast-wide weakfish abundance initially rose from 1992 until about 1998, but stock size dropped unexpectedly thereafter to the lowest levels in the time series by 2007 despite low and stable fishing mortality (F) rates during this period. Because F levels remained relatively low and stable after 1999, additional analyses have begun to find evidence for trophic interactions involving enhanced striped bass

(*Morone saxatilus*) and spiny dogfish (*Squalus acanthias*) predation or a more complex trophic triangle among weakfish, striped bass and Atlantic menhaden (*Brevoortia tyrannus)* (see sections **C9.0** through **C11.0**).

C8.2 METHODS

C8.2.1 Recreational Abundance Index

A weakfish relative abundance index in numbers (RelNt) was developed annually from 1981 to 2008 (Table C8.2-1) as a recreational catch-effort ratio:

$$\textbf{RelNt = MIDN / Effort.} \qquad \textbf{(1)}$$

The recreational catches (MIDN, numbers) (type A, B1 and B2) and fishing effort estimates (Effort in millions of trips) in equation (1) were taken from the MRFSS annual surveys from the private boat sector within the Mid-Atlantic subregion (Virginia to New York). Weakfish catch and fishing effort data were confined to the Mid-Atlantic subregion because private boats catches from this subregion represent the major (> 60%) component of the total coast-wide recreational weakfish catches with acceptably high relative precision about the mean catch estimates after 1984 (annual CV values < 0.13). Moreover, the private boat fishery is highly mobile and capable of catching weakfish of all sizes throughout their range.

A second time series (1981-2008) of weakfish relative abundance in weight (RelWt) was also derived as a ratio of recreational catches (A, B1, B2) in weight (MIDW, mt) to fishing effort (Effort) from the Mid-Atlantic private boat fishery (Table C8.2-1). Since the MRFSS has obtained weight (kg) data from only the harvest (A, B1), the average weight of released weakfish (the B2 component) was assumed to be constant (0.15 kg) each year based on length frequencies (cm) of several thousand released weakfish in the coast-wide head-boat fishery from 2004 to 2007. The weakfish lengths (cm) from this fishery were converted to weight (kg) using the coast-wide length-weight equation for weakfish.

The proposed recreational abundance indices for weakfish (RelNt and RelWt) are fishery dependent and thus partially included in the total (sport, commercial and discards) coast-wide landings. However, the problem of colinearity between the recreational indices and total coast-wide landings should be relatively minor for several reasons. First, auto-correlation between the relative abundance indices (RelNt and RelWt) and total recreational and commercial weakfish landings is minimized by the fact that Mid-Atlantic private boat recreational catches (type A, B1 and B2) rather than harvest (type A and B1) was used to derive the RelNt. The recreational catches are usually three to four times higher each year than the harvest after 1990. Second, the private boat catches (A, B1, B2) in the MRFSS were further divided by private boat fishing effort (Et) in which the effort trend from 1981 to 2003 is inversely correlated ($r = -0.39$, $P < 0.08$) with the trend in total coast-wide harvest. Finally, the time series (1982 to 2002) of recreational abundance indices (RelNt, RelWt) was shown to be highly correlated (Pearson $r = 0.84$, $P < 0.009$) with ages 1+ weakfish stock size from the converged portion (1982 to 2003) of the most recent VPA model run (Uphoff 2008).

Additional age 1+ weakfish abundance indices are available in weight (mean kg/tow) and number (mean N/tow) from the New Jersey (NJ DFW) and Delaware (DE FW) inshore trawl surveys (Table C8.2-2). Uphoff (2008) noted that the time series (1990-2007) of Delaware weakfish trawl indices were highly correlated ($P < 0.01$) to the recreational indices (RelWt), to coast-wide weakfish landings and to age1+ abundance from the converged portion (1990-2000) of the most recent

ADAPT VPA model run. By contrast, the New Jersey age 1+ indices from 1989 to 2007 were poorly correlated (P <0.54) to landings and the converged portion of the VPA. Uphoff (2008) found that the New Jersey trawl indices could be made more informative about trends in coast-wide abundance if the August proportion of positive weakfish (converted to a weight index by multiplying by an estimate of mean weight of exploitable sized weakfish) was used instead of the geometric mean number/tow index from August and October. For this reason, the August indices of positive tows from the New Jersey survey were used to index weakfish relative abundance. Given that these trawl surveys spatially overlap the recreational private boat RelNt indices, the most representative coast-wide weakfish abundance index was chosen as a blended (scaled and averaged) index based on the recreational private boat indices (RelNt, RelWt) (Table C8.2-1), the Delaware trawl indices and the revised August New Jersey trawl indices (Table C8.2-2). The rationale for blending the three indices together seemed justified since trends in the weakfish indices from the three surveys were highly correlated from 1981 to 2007 (Figures C8.2-1 and C8.2-2). Before the three data sets indices could be combined into a coast-wide index **(WkNt, WkWt**), the time series from each survey had to be standardized to equivalent abundance units. Equivalent units were established in a three-step process. First, the long-term mean abundance index was derived separately for the recreational, Delaware and New Jersey abundance indices. Second, a scalar was derived as a ratio of the long-term average Delaware and New Jersey indices to the long-term average recreational private boat index (catch/trip). Finally, each annual index from the Delaware and New Jersey time series was then multiplied times the respective scalar, thereby transforming the magnitude of the Delaware and New Jersey trawl indices to relative units of the recreational private boat indices (Table C8.2-3). Note that the Delaware and New Jersey indices began in 1990 and 1989, respectively and are not yet available for 2008. For this reason, the 1981 – 1988 and 2008 recreational private boat indices were used to reflect coast-wide weakfish abundance during those years.

C8.2.2 Relative Fishing Mortality (RelFt) and Scaled F Estimates

In this analysis, relative fishing mortality estimates (RelFt) were derived on ages 1+ weakfish from 1981 to 2008. The theoretical foundation of the relative F approach is based on a simple re-arrangement of the Baranov catch equation (Ricker 1975, page 13, equation 1.17) with respect to F:

$$\mathbf{F = Harvest / Mean\ Stock\ Size,} \qquad \mathbf{(2)}$$

where: mean relative stock size in equation 2 is typically expressed as the average of relative abundance indices in years t and t+1. In this analysis, RelFt estimates were based on the ratio of coast-wide annual (commercial and recreational landings plus discards) landings (numbers) of ages 1+ weakfish in year t (Catchnt) to the corresponding blended weakfish relative abundance index (WkNt, WkNt+1) in year t and t+1:

$$\mathbf{RelFnt = Catchnt / [(WkNt + WkNt{+}1)/2].} \qquad \mathbf{(3)}$$

Equation (3) is very similar to the equation introduced earlier by Sinclair (1998) except that he used relative exploitation:

$$\mathbf{Relu = Catchnt/ RelNt} \qquad \mathbf{(4)}$$

In this analysis, average (ages 1+) relative fishing mortality rates in weight and number

(RelFnt and RelFwt) (Table C8.2-4) were estimated from 1981 to 2008 via equation (3). RelFwt values were expressed by the ratio of annual total coast-wide (commercial and recreational plus discards) weakfish landings in weight (Catchw, mt) to the blended relative abundance indices in weight (WkWt) (Table C8.2-4). The 1981-2007 coast-wide weakfish landings (Catchn) in numbers (N*1000) and weight (Catchw) of age 1+ fish were taken directly from age aggregate landings data. The 2008 recreational landings and discard estimates are available from the MRFSS, but the 2008 commercial landings and discard estimates are not yet available. To estimate total commercial and recreational landings and discards (Catchn, Catchw) indirectly in 2008, the 2008 recreational landings were divided by the average ratio of recreational landings to total landings in 2006 and 2007. The same ratio approach was also used to estimate total weakfish discards in 2008.

The relative F estimates from equation 3 do not consider temporal and spatial shifts in the age structure, so this approach is designed only to address average annual shifts in ages 1+ F across time (1981-2008). Thus, the RelFt values are uninformative about year-class and age-specific changes in F over the time series. The strength of the relative F method, however, is in its simplicity and intuitive appeal, allowing scientists to evaluate the relative accuracy of tuning indices and how they might affect the magnitude and trend in F estimates. Most importantly, since RelFt estimates are expressed as a ratio of annual harvest to mean relative abundance, the trends in relative F are not confounded by the often untested assumption of constant natural mortality (M = 0.25) used explicitly to derive F estimates (F = Z – 0.25) in the ADAPT and Forward Projection models.

The next step in this analysis was to transform the ages 1+ relative fishing mortality rates (RelFnt and RelFwt) from 1981 to 2008 into units of instantaneous fishing mortality (F). This transformation was based on two scalars consisting of the average ratios of F to relative F (RelFnt, RelFwt) across some known time period. The instantaneous fishing mortality (F) rates used for scaling were taken directly from the 2009 ADAPT runs that was configured with tuning indices from the Delaware and New Jersey trawl surveys and North Carolina gillnet survey, as well as from the recreational cpue indices. Since a severe retrospective bias in F was present for all ADAPT runs during recent years (2004-2007), a block of annual F (ages 1+) from 1982 to 1985 was chosen for scaling because the magnitude and trend in F estimates from the converged portion of the model were robust to changes to the tuning indices in ADAPT model runs. Finally, since ages 1+ F estimates for weakfish based on ADAPT were derived assuming a constant natural mortality (F = Z-0.25) across all ages and years, it was assumed that the chosen M of 0.25 on ages 1+ weakfish was a reasonable approximation of average natural mortality over a narrow period (from 1982 to 1985), without having to make the more problematic assumption that M was fixed over the entire (1982-2007) time series.

The first scalar used to transform relative weighted F in numbers (RelFnt) to units of instantaneous F (FNt) consisted of the ratio between the long-term (1982-1985) average numbers weighted F from the most recent ADAPT VPA (Fvpa = 0.58, Se = 0.07) for ages 1+ and the corresponding (1982-1985) RelFn values (Table C8.2-5). The resulting scalar for converting RelFn in numbers to units of FN was 0.0000067. The second scalar from the same ADAPT run was used to convert relative biomass weighted F (RelFw) estimates into units of instantaneous biomass weighted fishing mortality (FWt) (Table C8.2-5). This second scalar consisted of the average biomass weighted F estimates (ages 1+) from 1982 to 1985 (Fvpa = 0.89, SE = 0.12). The resulting scalar used to convert RelFw to units of FW was 0.000024. The time series (1981-2008) of fishing mortality (F) rates in weight (FW) and number (FN) (Table C8.2-4) were derived by multiplying the annual relative fishing rates (RelFn, RelFw) by the corresponding fixed scalar.

C8.2.3 Stock Biomass and Surplus Production

Average stock biomass (Biowt, mt) and average stock numbers (Biont*1000) of ages 1+ weakfish were estimated from 1981 to 2008 (Table C8.2-6). The biomass series (Biowt) was derived by dividing the annual coast-wide weakfish harvest and discards in mt (Catchw) (Table C8.2-4) by the average biomass weighted F on age 1+ weakfish (FWt). The coast-wide number of weakfish (Biont) was determined by dividing coast-wide harvest in numbers and discards (Catchn) by the corresponding ages 1+ F in numbers (FNt) (Table C8.2-6).

A time series (1981-2008) of surplus production estimates in year t (SURPt) was also derived (Table C8.2-6). As in Jacobson et al (2002) and more recently in Walters et al (2008), the SURPt values were expressed each year by subtracting weakfish biomass in year t (Biowt) from the biomass in year t+1 (Biowt+1), and then adding the coast-wide harvest (mt) (Catchw):

$$\mathbf{SURPt = Biowt+1 - Biowt + Catchw.} \qquad \mathbf{(5)}$$

To examine whether or not overfishing has adversely affected weakfish surplus production (SURPt), the average (ages 1+) biomass weighted F estimates (FWt) lagged one year (t-1) (Table C8.2-4) were linearly regressed against surplus production (SURPt) from 1981 to 2008. If overfishing has recently occurred, the slope of the regression should be negative and statistically significant ($P < 0.05$).

C8.3 RESULTS

C8.3.1 Scaled Fishing Mortality (F)

Biomass weighted fishing mortality (FWt) estimates on ages 1+ weakfish were high (FWt range: 0.69- 1.16) by most standards from 1981 to 1987 (Table C8.2-4). The magnitude of FWt estimates, however, rose even higher to beyond 1.0 from 1988 to 1991 (Table C8.2-4, Figure C8.3-1), and greatly exceeded our current overfishing threshold for weakfish (Fmsy = 0.53). The magnitude of FWt estimates declined steadily thereafter to below 0.60 in most years from 2000 to 2008. The ages 1+ fishing mortality rates weighted by number (FNt) were almost always lower in magnitude than the corresponding biomass weighted fishing rates (FWt) (Table C8.2-4, Figure C8.3-1). The ages 1+ FNt estimates from 1981 to 2008 followed a similar trend over time as the biomass weighted FWt estimates (Table C8.2-4), but unlike the biomass weighted fishing rates (FWt), the FNt estimates fell abruptly after 1991 and remained below 0.30 from 1993 to 2008.

Weakfish biomass levels (Biow) have exhibited wide contrast from 1981 to 2008 (Table C8.2-6, Figure C8.3-2). Weakfish ages 1+ biomass (Biow, mt) remained relatively high (14,200 and 41,500 mt) from 1981 to 1988 but biomass levels fell steadily to below 10,000 mt from 1989 to 1993 (Table C8.2-6). Weakfish coast-wide biomass rose again temporarily from 1994 to 1996, but weakfish biomass fell steadily thereafter to the lowest level in the time series in 2008 (1,333 mt).

The time series of weakfish surplus production (SURPt) from 1981 to 2008 followed the same general trend as stock biomass (Biow) (Table C8.2-6, Figure C8.3-2). Weakfish surplus production remained relatively high from 1982 to 1986 and again in 1993 and 1994 (Table C8.2-6), but SURPt levels fell steadily after 1995 and remained very low in most years from 2001 to 2008 despite relatively low and stable fishing mortality (Figure C8.3-1). The unexpected drop in weakfish surplus production after 1999 coincided with, and may be attributed to, a sharp rise in the abundance of two potential predators: striped bass and spiny dogfish.(Figures C8.3-3 and C8.3-4). Since the

time series of striped bass and spiny dogfish abundance is highly correlated over time (Pearson r = 0.87, P < 0.0001), there is no way to statistically separate their potential predatory effects on weakfish. As a result, a scaled and blended index of striped bass and dogfish abundance (Tpred) (Figure C8.3-4) was developed and used instead of individual striped bass and spiny dogfish indices in the investigation of trophic linkages to weakfish stock dynamics (section **C9.0)**

C8.4 Management and Scientific Implications

Like virtually all single-species stock assessments conducted along the Atlantic coast, natural mortality (M) of age 1+ weakfish was initially assumed to be constant (M = 0.25) in previous Yield-per-Recruit and VPA model runs. Results from the index-based analysis indicated that ages 1+ F estimates remained low and steady from 1999 to 2008 despite a systematic rise in ages 1+ total mortality (Z) after 1998 as indicated by ADAPT. These conflicting trends between Z and F strongly suggest that natural mortality (M) has recently risen systematically (Figure C8.4-1), and was therefore the primary cause for the recent weakfish stock failure along the Atlantic coast. The management consequences of assuming a fixed M when the annual M values actually rises systematically over time can be serious (Swain and Chouinard 2008). As noted in this weakfish assessment, by holding M constant, the resulting ages 1+ fishing mortality rates (F) on weakfish would have exceeded 1.40 in recent years based on the 2009 VPA run despite the recent precipitous drop in landings and tuning indices. 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.

C9.0 Evaluate testing of fishing and additional trophic and environmental covariates and modeling of hypotheses using biomass dynamic models featuring multiple indices blended into a single index with and without a Steele-Henderson (Type III) predator-prey extension. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice. (TOR #5)

C9.1 INTRODUCTION

Our ability to assess the current status of Atlantic coast weakfish has been continually plagued by a pronounced retrospective bias associated with all previous ADAPT model runs. Previous ADAPT runs made with trawl-based tuning indices (Kahn et al 2006) exhibited severe retrospective bias that underestimated F and overestimated stock size in the most recent years by as much as 80%. The degree of retrospective bias from ADAPT was so severe that the results gave the false impression that weakfish stock size had actually risen to record high levels by 2002 despite the presence of record low coast-wide landings after 1999. Such a large systematic bias greatly confounds our ability to determine whether or not weakfish abundance has fallen and whether or not the stock is overfished. The degree of retrospective bias from ADAPT was reduced to some extent when the model was tuned specifically to recreational catch-effort data from the MRFSS (Kahn et al

2006).

Because of the aforementioned limitations to the catch-at-age approach, the 2006 Assessment (Kahn et al 2006, Uphoff 2006a; Crecco 2006) relied primarily on an index-based (ages 1+) method to monitor temporal changes in weakfish biomass (mt) and fishing mortality (F) from 1981 to 2003. This index-based approach was used to update ages 1+ F and stock biomass estimates through 2008. The index-based analysis for weakfish is described in detail in section **C8.0** of this assessment and was used as the foundation for the following biomass dynamic analyses. Many of the tables and figures presented in section **C8.0** are referred to within this section.

Results from the last assessment (Kahn et al 2006) revealed that overfishing (F > Fmsy) had occurred on weakfish from about 1981 to 1991. As a result, more stringent regulations were imposed on the weakfish recreational and commercial fisheries in 1992 and 1993 after which fishing mortality (F) rates fell to well below Fmsy from 1996 to 2003. Following a drop in F, coast-wide weakfish abundance initially rose from 1992 until about 1998, but stock size dropped unexpectedly thereafter to the lowest levels in the time series by 2007 despite low and stable fishing mortality (F) rates during this period. Because F levels were low after 1999, other studies (Crecco 2006; Uphoff 2006a) have begun to find evidence that linked enhanced striped bass (*Morone saxatilus*) predation to the decline in weakfish productivity after 1998. Striped bass abundance along the Atlantic coast has risen to record high levels after 1998 (Nelson 2007) coincident with the recent failure in weakfish productivity. Larger (> 50 cm) striped bass are known to consume a wide variety of finfish prey including weakfish (Overton et al 2008; Hartman and Brandt 1995; Walter and Austin 2003), and striped bass are known to overlap the temporal and spatial distribution of weakfish north of Cape Hatteras NC (Rudershausen et al 2005). Since predation is generally regarded as a major force structuring marine fish communities (Bax 1991), the age aggregated Steele and Henderson (1984) (S-H) production model was updated through 2008 to further examine the joint effects of fishing and predation on Atlantic coast weakfish. Also, to provide a more thorough examination of the the hypothesis of increased predation on weakfish, additional candidate predators such as spiny dogfish (*Squalus acanthias*), bluefish (*Pomatomus saltatrix*) and summer flounder (*Paralichthys dentatus)* were also considered, especially since all three finfish predators have risen sharply inshore along the Atlantic coast after 1998. Finally, environmental disturbances have been proposed as a major process governing finfish production and recruitment (Hollowed et al 2000b), so environmental factors such as decadal shifts in mean sea surface water temperature and deviations in the winter North Atlantic Oscillation Index were also examined as potential explanatory variables.

The following analysis focuses on the direct interaction between potential predators and weakfish and is hereafter referred to as the "predation hypothesis" to distinguish it from the more indirect "forage hypothesis" presented in section **C10.0**.

C9.2 METHODS

C9.2.1 Recreational Abundance Index

As described in section **C8.2**, a weakfish relative abundance index in numbers (RelNt) was developed annually from 1981 to 2008 (Table C8.2-1) as a recreational catch-effort ratio:

$$\textbf{RelNt = MIDN / Effort.} \qquad \textbf{(1)}$$

C9.2.2 Relative Fishing Mortality (RelFt) and Scaled F Estimates

In this analysis, relative fishing mortality estimates (RelFt) were derived on ages 1+ weakfish from 1981 to 2008. The theoretical foundation of the relative F approach is based on a simple re-arrangement of the Baranov catch equation (Ricker 1975, page 13, equation 1.17) with respect to F:

$$\mathbf{F = Harvest / Mean\ Stock\ Size,} \qquad \mathbf{(2)}$$

where: mean relative stock size in equation 2 is typically expressed as the average of relative abundance indices in years t and t+1. In this analysis, RelFt estimates were based on the ratio of coast-wide annual (commercial and recreational landings plus discards) landings (numbers) of ages 1+ weakfish in year t (Catchnt) to the corresponding blended weakfish relative abundance index (WkNt, WkNt+1) in year t and t+1:

$$\mathbf{RelFnt = Catchnt / [(WkNt + WkNt{+}1)/2].} \qquad \mathbf{(3)}$$

Equation (3) is very similar to the equation introduced earlier by Sinclair (1998) except that he used relative exploitation:

$$\mathbf{Relu = Catchnt/ RelNt} \qquad \mathbf{(4)}$$

Additional description of these methods and equations is presented in section **C8.2** of this report.

C9.2.3 Stock Biomass and Surplus Production

Average stock biomass (Biowt, mt) and average stock numbers (Biont*1000) of ages 1+ weakfish were estimated from 1981 to 2008 (Table C8.2-6). The biomass series (Biowt) was derived by dividing the annual coast-wide weakfish harvest and discards in mt (Catchw) (Table C8.2-4) by the average biomass weighted F on age 1+ weakfish (FWt). The coast-wide number of weakfish (Biont) was determined by dividing coast-wide harvest in numbers and discards (Catchn) by the corresponding ages 1+ F in numbers (FNt) (Table C8.2-6).

A time series (1981-2008) of surplus production estimates in year t (SURPt) was also derived (Table C8.2-6). As in Jacobson et al (2002) and more recently in Walters et al (2008), the SURPt values were expressed each year by subtracting weakfish biomass in year t (Biowt) from the biomass in year t+1 (Biowt+1), and then adding the coast-wide harvest (mt) (Catchw):

$$\mathbf{SURPt = Biowt{+}1 - Biowt + Catchw.} \qquad \mathbf{(5)}$$

Additional description of these methods and equations is presented in section **C8.2** of this report.

C9.2.4 Overfishing Thresholds (Fmsy, Bmsy)

Surplus production estimates have been used to monitor trends in per capita stock productivity for many exploited finfish populations (Jacobson et al 2002).). Walters et al (2008) noted that few stock assessments conducted thus far have examined the temporal trends in surplus production against biomass and how these trends may relate to the degree of density dependence and to the presence of enhanced trophic and environmental effects. Having a time series (1981-2008) of

weakfish surplus production (SURPt) and ages 1+ stock biomass in year t (Biowt) (Table C8.2-6), updated F_{msy} and B_{msy} thresholds were estimated for weakfish using the dynamic Gompertz external surplus production model (Quinn and Deriso 1999; Jacobson et al 2002). The Gompertz form was selected over the more widely used logistics equation because Yoshimoto and Clarke (1993) reported that, under simulation conditions, the Gompertz model produced more realistic (positive) and stable overfishing thresholds than the logistics model. In the asymmetrical Gompertz model, surplus production estimates (SURPt) from 1981-2008 were regressed against weakfish biomass (Biowt) and the product of the log weakfish biomass and biomass (LogBiowt*Biowt) in a two variable linear regression model without a y-axis intercept:

$$\mathbf{SURPt = a*Biowt + b * [(LogBiowt)*Biowt],} \qquad \mathbf{(6)}$$

where: K – theoretical carrying capacity (mt) = exp (a / b);
MSY- maximum sustainable yield (mt) = (-b * K)/2.72;
B_{msy} – stock size (mt) at MSY = K / 2.72;
F_{msy} – instantaneous fishing mortality at MSY= MSY / B_{msy};
Fcoll – instantaneous fishing mortality at stock collapse = F_{msy} *2.72.

Surplus production and stock biomass estimates are often plagued by moderate to high measurement errors (Quinn and Deriso 1999). For this reason, the Gompertz model (equation 6) was fitted as a linear robust regression model using the least trimmed squares regression (LTS) objective function as recommended by Rousseeuw and Van Driessen (2000). The parameter estimates (a, b) and resulting reference points (F_{msy}, N_{msy}, Fcoll) from the production model (equation 6) were derived from the ROBUSTREG procedure contained in the Statistical Analysis System (SAS 2002). The parameter estimates (a, b) and their standard errors based on least squares (LS) are highly prone to the presence of outliers. With robust linear regression like LTS, outlying observations are identified and automatically down-weighted, resulting in higher precision and more robust parameter estimates (a, b) over those derived from ordinary least squares.

In all model runs with equation 6, residual plots were examined against time (year) to check for the presence of serial correlations. A serial correlation in the residuals would suggest model misspecification, implying that additional biotic and abiotic factors other than fishing may be affecting weakfish surplus production (equation 6). To test for potential joint effects of fishing (F), environmental and trophic variables on weakfish surplus production, additional explanatory variable were added to equation 6 such as striped bass, bluefish, summer flounder and spiny dogfish abundance in a stepwise multiple regression (see section below for details). In these multiple regression models, residual plots across time were also examined to detect for the presence of serial correlations in the residuals. If enhanced predation is adversely affecting weakfish surplus production, the extended production models with predation effects should generate more precise parameter estimates (a, b) in equation (6) and more importantly, the residual plots should exhibit little if any serial correlation over time.

C9.2.5 Screening for Potential Trophic and Environmental Effects

There is a vast array of potential finfish predators that could prey on weakfish and thus undermine weakfish surplus production and stock size in recent years. Due to the recent and unexpected drop in weakfish stock size since 1999 under relatively low (F < 0.32) fishing mortality (Kahn et al 2006), other factors such as enhanced predation or temporal shifts in environmental

factors may have both played a measurable role in the recent decline in weakfish productivity. In this report, a predation hypothesis was tested for weakfish based on the potential trophic interactions among weakfish, and one or more candidate finfish predators. To be included as a candidate predator in this analysis, each potential finfish predator must overlap the temporal and spatial distribution of weakfish, have a documented history of consuming weakfish and, most importantly, the relative abundance of the candidate predators must have been at high abundance during some portion of the time series (1981-2007) in which weakfish abundance is being evaluated. Four finfish predators including striped bass (*Morone saxatilus*), spiny dogfish (*Squalus acanthias*), bluefish (*Pomatomus saltatrix*) and summer flounder (*Paralichthys dentatus)* were considered as candidate predators on weakfish, especially since all four have risen sharply inshore along the Atlantic coast after 1998. Bluefish, striped bass, summer flounder and spiny dogfish are major inshore finfish predators that have recently risen sharply in abundance along the Atlantic coast coincident with the drop in weakfish productivity. Moreover, these finfish predators overlap the spatial and temporal distribution of weakfish, and all are considered, to some extent, as potential candidate predators on weakfish (Overton et al 2008; Latour et al 2007; Bowman et al 2000). Uphoff (2003) has hypothesized that the predator-prey interaction between striped bass and Atlantic menhaden (*Brevoortia tyrannus*) may play an important role on the degree to which striped bass may have recently switched prey preference from menhaden to weakfish. For this reason, a time series (1981-2007) of menhaden abundance was also included as an additional explanatory variable in the stepwise regressions. Striped bass is regarded as a voracious predator from the Mid and North Atlantic on menhaden, gizzard shad and herring (Hartman 1993). Larger (> 70 cm) striped bass, however, have been reported to eat spot, flounder and weakfish in Chesapeake Bay (Overton et al 2008; Hartman and Brandt 1995; Walter and Austin 2003). A recent study in Chesapeake Bay (Latour et al 2007) reported that medium to large (> 38 cm) summer flounder fed extensively on ages 0 and 1 weakfish. Bluefish and spiny dogfish also prey upon a variety of finfishes including weakfish throughout the Atlantic coast (Bowman et al 2000; Stehlik 2007).

Annual coast-wide abundance of striped bass (ages 8+) in numbers (Table C9.2-1) has been estimated from 1981 to 2006 by the Statistical Catch-at-Age Model (SCAM) (Nelson 2007). Since the striped bass VPA underestimates recent (2004-2006) stock size estimates, a second time series (1981-2008) of coast-wide striped bass abundance (mean catch/trip) was derived as a ratio of striped bass recreational catch from the MRFSS private boat sector to private boat fishing effort (trips) within the Mid and North Atlantic sub-regions (Kahn 2007). This recreational time series is believed to be very informative about trends in coast-wide striped bass abundance since the MRFSS indices were highly correlated to ages 8+ abundance from the converged portion (1982-2001) of SCAM. For this reason, the MRFSS relative abundance indices were scaled to units of ages 8+ striped bass abundance from SCAM and used to reflect striped bass abundance in all subsequent analyses. Annual abundance changes in spiny dogfish, summer flounder and bluefish from 1981 to 2007 were indexed here as catch/trip based on the coast-wide recreational catches in number (A, B1, B2) and coast-wide effort (trips) from the private boat fishery in the MRFSS surveys (Table C9.2-1). These trends in coast-wide recreational cpue of summer flounder, dogfish and bluefish were assumed to be informative about coast-wide trends in these stocks from 1981 to 2008. A time series (1981-2005) of ages 1+ menhaden abundance (numbers) (Table C9.2-1) was taken from the most recent menhaden stock assessment (ASMFC 2006b).

Pearson correlation (r) and stepwise multiple regression models were used to screen for several key biotic and abiotic factors that may be linked statistically ($P < 0.05$) to changes in weakfish surplus production (SURPt) as well as to changes in the blended coast-wide biomass

indices (**WkWt**) 1981 to 2008 (Table C8.2-3). To examine whether surplus production and weakfish relative abundance may be linked to trophic and environmental factors, each of the response variables was related to the time series of the four candidate finfish predators (striped bass, spiny dogfish, summer flounder and bluefish) on weakfish (Table C9.2-1) and two environmental variables (deviations in the mean sea surface water temperature and deviations in the winter North Atlantic Oscillation index (NAO)) in thestepwise regression models. Annual deviations in the winter NAO indices from 1981 to 2008 (Table C9.2-1) were taken from the NOAA web site reported in Collie et al (2008). A time series (1976-2008) of average summer (July-September) sea surface water temperatures (C) was taken from a continuous temperature recorder in Long Island Sound located at the Millstone Nuclear Power Station, Waterford CT (Table C9.2-1). Temperature effects in the stepwise model (equation 1) were expressed as annual deviations from the long-term (1981-1998) mean temperature (devtemp) recorded prior to 1999.

Since enhanced predation should negatively impact weakfish productivity, the stepwise model selected a candidate predator when the slope (b) for predation effects is negative and statistically significant ($P < 0.05$). The stepwise regression models were run in SAS (SAS 2002) using the PROC REG procedure. Temperature effects in the stepwise model (equation 1) were expressed as annual deviations from the long-term (1976-1998) mean temperature (devtemp) recorded prior to 1999. In addition, deviations in the winter North Atlantic Oscillation (NAO) index (**Environj**) from 1981 to 2008 (Table C9.2-1) was also included as a potential explanatory variable in a stepwise linear regression model:

$$\mathbf{WkNt = a + b^{*}\ FWt + \ c\ ^{*}\ (Pred,\ Environj).} \qquad \mathbf{(7)}$$

Unlike enhanced predatory effects that are assumed to be negative on weakfish surplus production, climatic effects such as rising sea surface temperature could have a positive or negative impact on weakfish productivity. Climatic disturbances (**Environj**) in the form of a decadal rise in water temperature have been proposed as a major process restructuring ecological systems in southern New England (Oviatt 2004). A rise in temperature may cause stress and direct mortality or perhaps alter the competitive advantage of weakfish with other finfishes, thereby restructuring the inshore finfish community. Temporal changes in the NAO are thought to influence wind fields and levels of precipitation over the North Atlantic, thus affecting finfish larval drift, their subsequent recruitment and ecosystem productivity (Collie et al 2008).

There has been a coast-wide commercial and recreational fishery on weakfish for at least a hundred years (Kahn et al 2006). Thus, the time series (1981-2008) of ages 1+ biomass weighted (FWt) fishing mortality rates (Table C8.2-4) was also included as an explanatory variable in the weakfish stepwise (equation 7) regression models. Since an assessment of fishing effects on weakfish is an important priority, fishing mortality was always included (INCLUDE statement in PROC REG procedure) in the stepwise regression models regardless of its level of statistical significance. If fishing mortality effects on weakfish is excessive, the slope for fishing mortality effect will be negative and statistically significant ($P < 0.05$).

Statistical support for the predation and environmental hypotheses would be evident if the slope (c) for predation and environmental effects in equation (7) was negative and statistically significant ($P < 0.05$). This would imply that enhanced trophic and environmental factors have eroded weakfish abundance and surplus production independently of fishery effects. Moreover, if the slope for predation effects is significant, the inclusion of an extra predation term in the Gompertz model (equation 6) can greatly enhance the precision around the (a) and (b) parameters, thus

allowing more precise estimates of Fmsy and Bmsy thresholds. To test for potential joint effects of fishing (F) and trophic interactions on weakfish productivity, residual plots against time were examined for the presence of serial correlations. As a result, a negative and statistically significant slope estimate in equation 8 would imply enhanced predation effects on weakfish surplus production. To test for potential joint effects of fishing (F) and trophic interactions on weakfish productivity, residual plots across time were derived for each production model with and without predation. If predation is an important variable, the plot of residuals from the Gompertz models without predation terms (equation 6) should exhibit a pronounced serial correlation over time, indicating model misspecification. By contrast, the residuals from production models that include predation effects (equation 6) should exhibit no serial correlation over time.

C9.2.6 Age 0 Mortality

One major problem in quantifying predation or other environmental effects on weakfish is pinpointing the life history period 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, juvenile weakfish are particularly at risk to an array of potential finfish predators. Several recent predation studies (Beck 1997; Wahle 2003) have shown that a systematic rise in mortality during the juvenile stage may produce a demographic bottleneck that can constrict 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 has occurred for age 0 weakfish, a time series of relative weakfish juvenile mortality (Z0) was derived for the 1981 to 2006 year-classes. The Z0 estimate by year-class was expressed by a log ratio between coast-wide age 1 abundance (N1t+1) in year t+1 to the mean coast-wide juvenile abundance index (N0t) in year t:

$$\mathbf{Z0 = - \log (N1t+1 / N0t).} \qquad \mathbf{(8)}$$

Weakfish juvenile abundance surveys (N0) have been conducted along the Atlantic coast from Rhode Island to North Carolina from 1981-2007 (Table C9.2-2). Eight juvenile surveys (Table C9.2-28) were used to construct average coast-wide juvenile indices (N0) from 1981 to 2007. Details of the various surveys are presented in section **C6.0**. Note that juvenile weakfish indices are not yet available in 2008. The abundance indices were expressed as the geometric mean catch per tow, resulting in indices of varying magnitude. As a result, before the indices from the eight surveys were combined into a coast-wide average index (N0), 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 (1982-2007) 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 into units of the New Jersey indices (Table C9.2-2). Finally, the coast-wide geometric mean index (N0) was derived as the grand mean across the scaled indices from 1982-2004 (Table C9.2-3).

A time series (1982-2007) of weakfish age 1 abundance (in millions of fish) used in the numerator of equation (8) was taken directly from the preferred 2009 ADAPT VPA model run (Table C9.2-3). The N1 estimates were derived independently of the relative juvenile abundance

estimates (Table C9.2-2) since none of the juvenile indices were used to tune the VPA. Note that the juvenile abundance indices in the denominator of equation (8) are expressed in relative units so that the juvenile total mortality rates (Z0) are expressed as relative mortality estimates. Also, note that the current VPA generated age 1 abundance estimates (t+1) for the 1981 to 2006 year-classes during the years 1982 to 2007. The age 1 abundance (N1) values from the VPA were larger than the coast-wide juvenile index (Table C9.2-2) resulting in some negative values of mortality (Z0) via equation (8a). To generate positive juvenile mortality rates (Z0), all age 1 abundance data (N1) were reduced in magnitude by dividing N1 by an arbitrary value of 100.0.

To examine the hypothesis that the recent emergence of a recruitment bottleneck between age 0 and age 1 was linked to shifts in trophic and environmental factors, a time series of candidate predators such as striped bass, bluefish and spiny dogfish and summer flounder abundance (Table C9.2-1) were included as explanatory variables in a stepwise regression where the time series of Z0 estimates was the response variable (Table C9.2-3). In addition, environmental variables such as mean summer sea surface water temperature and deviations in the winter North Atlantic Oscillation (NAO) index (environ term) from 1981 to 2007 (Table C9.2-1) were also considered as additional explanatory variables in the stepwise model. Finally, a recent rise in juvenile mortality (Z0) could be due to a rise in weakfish discards from the commercial and recreational fisheries. To test this hypothesis, the annual contribution of fishing mortality due to discards number (Fdisn) was derived as the fraction of annual discards in number to total weakfish landings in number times the current numbers weighted F (FN) (Table C8.2-4). The time series (1981-2007) of Fdisn estimates were included in the stepwise model as a final explanatory variable.

C9.2.7 Stock-Recruitment Effects

In this report, the shape and residual pattern of the weakfish stock-recruitment (S-R) relationship was explored with the Ricker (1975) model:

$$\mathbf{Rec0 = A * Biow * exp\ (B * Biow),} \qquad \mathbf{(9)}$$

where: A = the magnitude of compensatory reserve;
B = the coefficient of compensatory density-dependent mortality;
Rec0 = blended coast-wide index of weakfish age 0 recruits (Table C9.2-3);
Biow = average weakfish biomass (mt) (Table C8.2-6).

The parameter estimates (A, B) from the S-R model (equation 9) were derived from the NLIN procedure (Marquardt Algorithm) contained in the Statistical Analysis System (SAS 2002).

Given the likely presence of outliers in the S-R data, the Ricker S-R model was 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 SAS (2002). This re-weighting scheme is designed to detect outliers, thereby allowing the down weighting of S-R 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 (A, Kp) and at the same time generating globally converged parameter estimates. As suggested by Holland and Welsch (1978), a range of threshold estimates was 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 and minimum variance. The

two-step re-weighting approach always produced converged estimates (global estimates) that were within 10% of the parameter estimates (A, Kp) derived by the nonlinear least squares approach. However, the standard errors about the estimates based on iterative re-weighting were always 30 to 45% lower than the standard errors from the least squares method.

To examine for potential predatory and environmental effects on the weakfish S-R model, the nonlinear Ricker S-R model (equation 9) was linearized and included in the stepwise model that included an additional exponent (c) reflecting potential predation (pred) and environmental (environ) effects:

$$\textbf{Log (Rec0/Biow) = A * Biow * exp (B * Biow) * exp(c*pred, environ).} \qquad \textbf{(10)}$$

As in the surplus production analyses, potential explanatory variables included striped bass, summer flounder, bluefish and spiny dogfish abundance (Table C9.2-1) as well as environmental variables (environ) including annual sea surface water temperature and deviations in the winter North Atlantic Oscillation (NAO) index (Table C9.2-1). Statistical evidence consistent with predation would exist if additional exponents (c) for predation effects in equation (10) were negative and statistically significant ($P < 0.05$). Further statistical support for the predation hypotheses would be evident, if the serial correlation in residuals evident in the basic Ricker S-R model (equation 9) was minimized following the addition of predation effects to the linearized S-R model (equation 10).

C9.2.8 Steele-Henderson (S-H) Production Model

The last weakfish assessment (Kahn et al 2006) reported statistical evidence of a strong linkage between enhanced striped bass predation and the recent unexpected decline in coast-wide weakfish. As a result, the age aggregated Steele-Henderson (S-H) production model (Steele and Henderson 1984) was updated to link fishing and predation effects to the recent drop in weakfish productivity. 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 1997b). The Steele-Henderson (S-H) model incorporates compensatory stock dynamics of the prey with fishing effects plus a sigmoid type III functional response by the predator. The Type III response adds a degree of realism to the model since it may lead to either prey stability at low to intermediate predator abundance, or to critical depensation of the prey at low prey abundance (Spencer and Collie 1997b; Collie and DeLong 1999). The age aggregated Steele-Henderson (S-H) production model was used to estimate equilibrium and time varying overfishing thresholds (F_{msy}, N_{msy}) for weakfish in the presence of a significant ($P < 0.05$) predatory 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 weakfish 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) of the S-H model requires a high degree of contrast in the time series (1981-2007) of stock sizes. The S-H model was originally configured as a logistics production model with an added sigmoid function that reflected the foraging response by the predator. Previous simulation studies (Yoshimoto and Clarke 1993) have indicated that the Gompertz asymmetrical model produced more realistic (positive values of F_{msy}) and robust parameter estimates than the logistics model. As a result, the surplus production portion of the S-H model was converted from the logistics to the

Gompertz form:

Biowt+1= Biowt+log(K)*r*Biowt*(1-(log(Biowt)/log(K)))- Catchw-

(c*Pred*(Biowt)2)/(A**2+(Biowt)**2)]** **(11)**

where: Biowt+1 =weakfish stock biomass (mt) in year t+1 (Table C8.2-6);
Biowt = weakfish stock biomass (mt) in year t (Table C8.2-6);
Pred = abundance of the selected predator during year t (Table C9.2-1);
Catchw = harvest and discards (mt) of weakfish in year t (Table C8.2-4);
K = estimated carrying capacity of weakfish biomass;
r = intrinsic rate of weakfish population increase;
c = per capita consumption rate of the predator;
A = weakfish biomass (mt) at which predator satiation takes place.

All parameter estimates (r, K, c and A) from the S-H model (equation 11) 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 (Biowt, Biowt+1) (Table C8.2-6) and the abundance of one or more predators (Pred) by nonlinear least squares regression methods. The choice of one or more predators in equation (9) depends on the outcome of the stepwise regression procedure. Given the likely presence of measurement errors in the input data, the S-H model was fitted as a nonlinear robust regression using the iterative reweighted least squares method outlined by Holland and Welsch (1978).

As indicated by Spencer and Collie (1997b), the S-H model (equation 11) represents the merger of two models; one is a conventional Gompertz discrete time dynamic production model with only fishing effects (Catchw) present:

Biowt+1 = Biowt+log(K)*r*Biowt*(1-(log(Biowt)/log(K)))- Catchw **(12)**

whereas the other model expresses additional predatory effects via the Type III functional response:

$$[(c*Pred*(Biow_t)^2)/(A^2+(Biowt)^2)] \quad (13)$$

To more fully examine whether or not fishing mortality (Ft) alone might account for the systematic decline in weakfish abundance, the discrete time Gompertz portion (equation 12) of the full S-H model (equation 11) was fitted separately to weakfish stock sizes (Biowt, Biowt+1) from 1981 to 2007 using nonlinear robust regression methods. If fishing effects are largely responsible for the current decline in weakfish, the predicted stocks sizes should closely follow the observed abundances (high coefficient of determination), the resulting parameter estimates (r, K) from the discrete Gompertz model (equation 12) should differ significantly ($P < 0.05$) from zero, and the resulting equilibrium F_{msy} threshold estimate (r) should at least approximate previous F_{msy} estimates for weakfish based on previous stock assessment (Kahn et al 2006). If the discrete time model fails in this regard, then the full S-H model with predation effects (equation 11) was then applied to

determine whether or not the extended model could fulfill all three of the aforementioned requirements.

Uphoff (2006) noted that if the predation parameter estimates (c, A) from the S-H model are sufficiently robust and precise, then a time series of weakfish biomass consumed (Dt in mt) annually by predation (Pred) can be derived in the form:

$$\mathbf{Dt = [(c*Pred*(Biowt)**2) / (A^2 + (Biowt)^2]} \qquad \mathbf{(14)}$$

Once (Dt) is estimated via equation (14), the weakfish instantaneous consumption rate associated with predation (Mpt) can be derived annually for an annual predator:

$$\mathbf{Mpt = Dt /[(Biowt+Biowt+1)/2)] \ .} \qquad \mathbf{(15)}$$

In the dynamic Gompertz production model without predation (equation 12), the equilibrium F_{msy} threshold is solely expressed by the intrinsic rate (r) parameter, whereas B_{msy} is expressed by the carrying capacity (K) divided by 2.72 (Quinn and Deriso 1999). Since temporal shifts in striped bass predation are absent from the traditional dynamic models, the overfishing definitions (F_{msy}, B_{msy}) in these models are fixed in time. However, in the non-equilibrium S-H model (equation 11) the ability to identify steady-state conditions is far more difficult because predation effects are transient over time. In the non-equilibrium S-H model, weakfish surplus production and predation-induced mortality (Mpt) can vary greatly across years, resulting in time varying F_{msy} and B_{msy} thresholds. The degree of temporal variation in F_{msy} and B_{msy} depends on the magnitude and trend in predator abundance, the predator consumption exponent (c) and on the prey stock size (A) at which the consumption threshold of predation takes place in equation (11). Thus, the annual $F_{msy\ t}$ values from the S-H model are not fixed in time but rather are a function of the fixed intrinsic rate (r) minus the time varying predator consumption rate (Mpt):

$$\mathbf{F_{msy\ t} = r * exp(- Mpt) \ .} \qquad \mathbf{(16)}$$

Similarly, the weakfish biomass threshold (B_{msy}) can vary over time depending on the amount of weakfish biomass consumed annually by the predator (Dt):

$$\mathbf{B_{msy} = [K- Dt] / 2.72 \ .} \qquad \mathbf{(17)}$$

Although weakfish overfishing thresholds ($F_{msy\ t}$, $B_{msy\ t}$) derived from the S- H model are time varying, equilibrium reference points can be approximated as the long-term (1981-2007) mean F_{msy} t and $B_{msy\ t}$.

C9.3 RESULTS

C9.3.1 Scaled Ages 1+ Fishing Mortality (F), Biomass and Surplus Production

Estimates of rescaled relative F, weakfish stock biomass, and weakfish surplus production

are presented in section **C8.3** of this report.

Juvenile and Age 1 Abundance Including Juvenile Mortality (Z0)

The relative magnitude of weakfish juvenile indices varied greatly across the eight surveys for the 1981 to 2007 year-classes (Table C9.2-2). When the eight juvenile data sets were scaled and blended into a mean coast-wide index (Rec0) (Table C9.2-3), the coast-wide index (Rec0) showed a fairly persistent rise after 1995 (Figure C9.3-1). Although coast-wide average recruitment (Rec0) remained high in most years from 1998 to 2007, a pronounced drop in weakfish ages 1+ abundance (Nvpa) from the 2009 VPA was clearly evident from 1998 to 2006 (Table C9.2-3). Abundance of age 1 weakfish (Nvpa) from the most recent VPA run varied without trend from 1982 to about 1998, then age 1 abundance fell steadily thereafter in sharp contrast to the general rise in juvenile recruitment after 1995 (Figure C9.3-1). The resulting weakfish juvenile mortality (Z0) estimates increased in magnitude for the 1999 to 2006 year-classes (Table C9.2-3), and were positively correlated (Pearson $r = 0.88$, $P < 0.0001$) to the blended predation index (Tpred) consisting of striped bass and spiny dogfish (Figure C9.3-2). These findings strongly suggest that recent rise in juvenile natural mortality since 1998 is consistent with the recent emergence of a trophic bottleneck that may have interfered with the flow of recruitment from age 0 to ages 1+.

C9.3.2 Correlation Matrix and Stepwise Regression

The time series of weakfish surplus production (SURPt) (Table C8.2-6), age aggregated biomass index (WkW) (Table C8.2-3) and weakfish juvenile mortality (Z0) from 1981 to 2008 (Table C9.2-3) was examined in a correlation matrix against each of the ten potential explanatory variables (Table C9.2-1). The correlation matrix revealed that striped bass and spiny dogfish abundance were always inversely related ($P < 0.001$) to weakfish surplus production and weakfish biomass and positively related to juvenile mortality (Z0) (Table C9.3-1, Figure C9.3-2). The time series of menhaden abundance, a major finfish prey of both striped bass and spiny dogfish, was positively related ($P < 0.001$) to weakfish surplus production and weakfish biomass and inversely related to juvenile weakfish mortality (Table C9.3-1), suggesting that a recent drop in menhaden abundance may have enhanced the predation risk of weakfish by spiny dogfish and striped bass. None of the other potential explanatory variables, including fishing mortality (FWt) and discard effects, were correlated across all three (SURPt, WkW, Z0) response variables. These findings strongly suggest that the recent failure in weakfish productivity after 1998 was linked mainly to enhanced predation from striped bass and spiny dogfish, the magnitude of which appeared to be enhanced by the recent coast-wide decline in Atlantic menhaden.

The time series of weakfish juvenile mortality rates (Z0) for the 1981 to 2006 year-classes (Table C9.2-3, Figure C9.3-2) were used as a response variable in the stepwise regression model against the blended striped bass and dogfish abundance index (Tpred) as well as with the five additional explanatory variables (Table C9.2-1). The stepwise model selected only the blended predation index (Tpred) as the most significant ($P < 0.0001$) explanatory variable that was closely tied (Pearson $r = 0.84$) to the rise in weakfish juvenile mortality (Z0) (Table C9.3-2, Figure C9.3-2). No other explanatory variable was selected at the $P < 0.05$ level in the stepwise model. It is important to note that estimated weakfish coast-wide discards (Disn) and discard-related fishing mortality (FDisn) (Table C8.2-4) were included in the models but were not positively correlated ($P < 0.05$) to juvenile mortality (Z0). These findings suggest that fisheries-related activities were not tied directly to the recent rise in weakfish juvenile mortality. The emergence of an age 0 recruitment bottleneck after 1996 appeared to be linked mainly to enhanced predation from striped bass and

spiny dogfish and rather than to enhanced fisheries discards.

Weakfish surplus production (SURP) and coast-wide relative biomass (WKW) (Tables C8.2-3 and C8.2-6) from 2001 to 2008 were also related to six explanatory variables in stepwise regression models where biomass weighted fishing mortality (FWt) effects were always included (Table C9.3-2). Once again, the stepwise model always selected the blended predation index (Tpred), comprised of striped bass and spiny dogfish, as the most significant ($P < 0.0001$) explanatory variable (Table C9.3-2). The Tpred variable was inversely related (Pearson $r = -0.80$) to the post 1999 decline in weakfish surplus production (Figure C8.3-3) and biomass. Although biomass weighted fishing mortality effects (FW) were independent ($P < 0.45$) of weakfish surplus production (SURP), FWt were negative and statistically significant ($P <0.007$) on weakfish relative biomass (WKW) but only in the presence of predation (Tpred) (Table C9.3-2). The biotic mechanism(s) behind the apparent negative interaction of predation and fishing mortality on weakfish biomass are not clear at this time. Most of the statistical evidence given thus far suggests that the recent (1999-2008) failure in weakfish surplus production and stock biomass was due primarily to enhanced predation from striped bass and spiny dogfish.

C9.3.4 Gompertz Production Models With and Without Predation Effects

The external Gompertz surplus production model (equation 6) was fitted by linear robust regression to weakfish stock biomass in year t and t+1 (Biot, Biot1) (Table C8.2-6). This model with only fishing effects (Catchw) did provide a reasonably good fit (r**2 = 0.54) to the biomass time series with statistically significant ($P < 0.05$) r and K parameter estimates (Table C9.3-3). The resulting overfishing thresholds (Fmsy, Bmsy) of 0.26 and 57,388 mt, respectively, were much different than earlier estimates (Fmsy = 0.41, Bmsy = 25,400 mt) based on the Steele-Henderson model in 2005. Moreover, the resulting production model consistently over predicted weakfish surplus production from 1998 to 2008 by 20% to 350% (Figure C9.3-3), indicating the presence of a distinct serial pattern in the residuals. This systematic trend in the direction of the residuals usually indicates that the model is lacking an informative parameter(s).

When the blended predation parameter (Tpred) was added to the basic Gompertz model, the fit to surplus production greatly improved (r**2 = 0.84) and the parameters (r, K, C) were estimated with much higher precision than the production model without predation (Table C9.3-3). The slope representing predation effects (C) was negative and statistically significant ($P <0.0001$) which is consistent with the significant inverse relationship (Pearson $r = -0.87$, $P < 0.0001$) between weakfish surplus production and blended predation from 1981 to 2008 (Figure C8.3-4). The resulting equilibrium overfishing thresholds (Fmsy, Bmsy) from the extended Gompertz model were 0.54 and 19,289 mt, respectively (Table C9.3-3). Moreover, unlike the strong serial correlation in the residuals noted from the basic Gompertz model (Figure C9.3.3), there is little if any systematic residual pattern from the extended Gompertz model that includes predation effects of striped bass and spiny dogfish (Figure C9.3-4).

C9.3.5 Stock –Recruitment Effects

The Ricker S-R (equation 9) model was applied to relative coast-wide juvenile recruitment (Rec 0) (Table C9.2-3) and weakfish biomass (Biow, mt) (Table C8.2-6) from 1981-2007 using iterative re-weighted nonlinear least squares regression. The dome-shaped Ricker stock-recruitment (S-R) model was a good fit ($r^2 = 0.52$) to the S-R data from 1981 to 2007 (Figure C9.3-5). The steep ascending limb and pronounced dome indicated the strong presence of density-dependent mortality and the ability of weakfish to compensate for relatively high levels ($F < 0.8$) of fishing mortality.

The resulting parameter estimates were precise (A = 0.016, SE = 0.004, B = -0.00015, SE = 0.000027), both of which differed significantly (P <0.05) from zero (Table C9.3-4). However, the residual plot over time indicated the presence of a distinct serial correlation (Figure C9.3-6). The residuals were mostly negative from 1981 to 1994 but the direction of the residuals shifted abruptly in a positive direction thereafter (Figure C9.3-6), indicating model misspecification.

When the linearized version of the Ricker S-R model (equation 10) was fitted to several candidate predators (blended predators, summer flounder and bluefish) and environmental factors (deviations in summer sea surface temperatures and deviations in the NAO index) the stepwise model chose the blended predators (tpred) as the only significant explanatory variable to the S-R model. When tpred was added as a second explanatory variable to the S-R model, the extended model explained 81% of the recruitment variation, all three parameter estimates (A, B, c) became highly significant (P <0.0001) (Table C9.3-4) and, most importantly, the anomalous residual pattern seen in the basic Ricker S-R models virtually disappeared when Tpred was added to the model (Figure C9.3-7). These finding are consistent with the Predation Hypothesis, indicating that the transmission of age 0 weakfish recruits to the adult stock has been recently impeded due to enhanced predation from striped bass and spiny dogfish.

C9.3.6 Steele-Henderson Model

The discrete version of the Gompertz surplus production model with only fishing effects (equation 12) was fitted by nonlinear least squares and iterative re-weighted least squares regression to weakfish stock biomass in year t and t+1 (Biowt, Biowt+1) (Table C8.2-6). Like the results from the Gompertz external model (Table C9.3-3), this model provided a good fit (r**2 = 0.68) to the biomass data with statistically significant (P < 0.05) r and K parameter estimates (Table C9.3-6). The resulting overfishing thresholds (Fmsy, Bmsy) 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 2008 by 10 to 250% (Figure C9.3-8), indicating that the basic Gompertz model was plagued by substantial process error. The model (equation 12) applied 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 (Fmsy, Bmsy) (Table C9.3-6), but severe process error nevertheless persisted in the residuals from 1998 to 2008. Even when the Logistics form of the surplus production model was used instead of the Gompertz, the same serial residual pattern persisted over time, indicating that the residual problem was not due to the configuration of the production model. Given the clear residual problem associated with the basic Gompertz and Logistics models, the surplus production models without predation were not used to estimate overfishing thresholds for weakfish.

The full Steele-Henderson (S-H) production model (equation 11) was applied to weakfish stock biomass in year t and t+1 (Biot, Biot1) and to the blended predators (Tpred) by iterative re-weighting (Tables C9.3-5 and C9.3-6). The S-H models provided a very good fit (r**2 = 0.84 for unweighted and 0.94 with iterative reweighting) to the biomass and predation data (Figure C9.3-10) with statistically significant (P < 0.05) r, K, c and A parameter estimates (Table C9.3-6). Overall, the S-H models was not only a better fit to weakfish biomass than the Gompertz model with only fishing effects (Table C9.3-6), but the r and K parameters were estimated with much higher precision. The resulting weakfish equilibrium overfishing thresholds (Fmsy, Bmsy) from the nonlinear least squares S-H models for weakfish was 0.72 and 17,009 mt, respectively (Table C9.3-6). Moreover, unlike the severe residual pattern evident from 1998 to 2008 based on the basic Gompertz models (Figure C9.3-3), there is little if any systematic residual pattern from the S-H model fitted by iterative re-

weighting (Figure C9.3-9). Finally, the time series (1981-2008) of predation-based natural mortality (Mp) from striped bass and spiny dogfish (Tpred) was derived via equations (14) and (15) (Table C9.3-5). The predatory mortality rates (Mp) rose two to four fold in magnitude after 1997 coincident with a steady drop in weakfish biomass and surplus production. This inverse relationship between Mp and weakfish biomass is consistent with the presence of depensatory density-dependent predation mortality and, if persistent over time, could be highly destabilizing to future weakfish stock rebuilding. These findings strongly suggest that the recent rise in weakfish ages 1+ natural mortality (MWt) coupled with the failure in weakfish productivity (SURPt) are tied directly to the increase in striped bass and spiny dogfish predation mortality (Mp).

The estimated instantaneous consumption rates (Mp) of weakfish by striped bass and spiny dogfish rose steadily in magnitude from 0.22 in 1986 to 1.64 in 2002 and then remained relatively high thereafter (Table C9.3-5, Figure C9.3-11). The estimated biomass (mt) of weakfish consumed (Dt) annually exceeded 4000 mt in most years from 1995 to 2008 during which weakfish biomass fell by 80% after 2000 (Table C9.3-5). Despite the systematic decline in weakfish biomass after 1999, the magnitude of weakfish biomass (Dt, mt) consumed by striped bass and spiny dogfish remained relatively high, equal or exceeding the weakfish coast-wide biomass after 2001 (Table C9.3-5, Figure C9.3-11). Moreover, weakfish biomass consumed (Dt) annually and the instantaneous consumption rates (Mp) from 1997 to 2008 (Table C9.3-5) were 30% to 300% higher than the coast-wide landings and fishing mortality (F) rates, respectively, on weakfish during those years (Table C9.3-7, Figure C9.3-12). The effects of enhanced predatory consumption (Mp) coupled with moderate fishing mortality (FW) on weakfish from 2000 to 2008 easily exceeded the Fmsy threshold, resulting in the systematic decline in weakfish surplus production and biomass from 2001 to 2008 (Figure C8.3-2).

When the equilibrium overfishing thresholds (Fmsy = 0.72, Bmsy = 17,009 mt) from the S-H model (Table C9.3-6) are considered, ages 1+ biomass weighted fishing mortality (FW) on weakfish exceeded the estimated Fmsy threshold of 0.72 in most years from 1981 to 1992 (Figure C9.3-13). Weakfish coast-wide biomass (mt) exceeded the biomass threshold (Bmsy = 17,009 mt) in 1981 and 1982, but biomass fell quickly below Bmsy thereafter (Figure C9.3-14). When more stringent management regulations were enacted after 1991, fishing mortality (FW) fell by 50 to 70% (Figure C9.3-13) and biomass began to rise toward the Bmsy threshold (Figure C9.3-14). However, weakfish biomass fell unexpectedly after 1999 to the lowest level in the time series in 2007 (Figure C9.3-14) despite the fact that fishing mortality rates remained below Fmsy in most years (Figure C9.3-13) from 1998 to 2008 (exception: 2002).

Because predatory consumption rates (Mp) on weakfish increased about five fold after 2000 (Figure C9.3-10), the overfishing definitions (F_{msy}, B_{msy}) did not remain fixed over time (Table C9.3-5). The non-equilibrium F_{msy} levels via equation (16) remained relatively stable around 0.60 to 0.78 from 1981 to 1989 during which weakfish biomass and surplus production remained high and stable (Figure C8.3-2). However, when predatory consumption rates (Mp) rose after 1997 (Figure C9.3-11) and surplus production dropped, annual $F_{msy\ t}$ thresholds dropped sharply from around 0.60 to 0.70 in the 1980's to below 0.30 after 2000 (Table C9.3-5) in concert with a steep rise in striped bass and spiny dogfish abundance (Figure C8.3-3) and predation related mortality rates (Mp) (Figure C9.3-11). In contrast, non-equilibrium biomass thresholds (B_{msy}) were more robust to rising Mp (Table C9.3-5). The B_{msy} thresholds remained relatively stable at between 14,000 and 15,000 mt from 1995 to 2008 despite rising Mp and Dt (Figures C9.3-11 and C9.3-12).

C9.4 Scientific and Management Implications

The preponderance of statistical evidence given here supports the Predation Hypothesis involving enhanced predation by striped bass and spiny dogfish as the primary factor behind the recent and unexpected decline in weakfish productivity. Statistical evidence in support of the Predation Hypothesis consists of a significant (P <0.0001) inverse correlation between declining weakfish biomass and surplus production from 1999 to 2008 and striped bass and spiny dogfish abundance from 1982 to 2004. Striped bass abundance along the Atlantic coast rose 10 fold from 1994 to 2006 (Kahn 2005), although the 2008 striped bass abundance estimate fell by over 40% since 2006. Similarly, spiny dogfish abundance has increased 10 fold since 1999 and has remained high thereafter. During this recent period (1999-2008) of declining weakfish productivity, fishing mortality (FW) and discard mortality (Fdisc) rates remained low and relatively stable, indicating that the recent drop in weakfish productivity did not coincide with rising exploitation. The strong positive correlation (Pearson r = 0.91, P <0.0001) between the recent rise in weakfish juvenile mortality (Z0) and rising striped bass and spiny dogfish abundance further suggests that the recent emergence of a weakfish recruitment bottleneck at age 0 was largely due to enhanced predation by these two finfish predators. By contrast, discard mortality estimates of small (< age 2) weakfish remained low and stable after 1999 during which juvenile mortality (Z0) rose steadily. Third, the residual patterns in all logistics and Gompertz model runs that included only fishing effects (landings) produced inordinately low overfishing thresholds (Fmsy, Bmsy), poor precision around the estimates, and the residuals exhibited a pronounced serially correlation over time, clearly indicating model misspecification. However, when the predation term (Tpred), reflecting the joint predation by striped bass and spiny dogfish, was added to the models, the fit of the models to weakfish surplus production and biomass dramatically improved, the precision and magnitude of Fmsy and Bmsy rose to more plausible levels (Fmsy > 0.45), and, most importantly, the direction of the residuals over time shifted to a more random pattern and were therefore free of model misspecification. Finally the instantaneous consumption rates (Mp) on ages 1+ weakfish derived empirically from the Steele-Henderson model from 1999 to 2008 were closely correlated (Pearson r = 0.81, P < 0.0002) to the rise in ages 1+ total mortality (Z) derived independently from the most recent ADAPT VPA run. This strongly suggests that the recent increase in ages 1+ total mortality (Z) in the VPA was not due to a rise in fishing mortality (F) but rather to enhanced predation by striped bass and spiny dogfish..

Although results from regression and production models alone do not demonstrate causality, recent empirical evidence is consistent with the Predation Hypothesis involving striped bass and spiny dogfish. Striped bass are known to consume finfish prey up to 60% of their own body length (Manooch 1973). Unlike fluke and bluefish that prey mainly on small (< 30 cm) fish, large (> 76 cm) striped bass can easily prey on larger (> 40 cm) weakfish, whereas smaller stripers (< 65 cm) can effectively prey on juvenile and age 1 weakfish. Recent food habits studies of spiny dogfish show that dogfish prey on a wide variety of finfishes including weakfish (Stenlik 2007; unpublished NEAMAP data). Striped bass and spiny dogfish seem to prey more commonly on menhaden and river herring (*Alosa* spp) (Uphoff 2003: Bowman et al 2000). But coast-wide menhaden and river herring abundance has fallen dramatically after 1995 (citation), thereby enhancing the prospects that striped bass and spiny dogfish would recently switch to alternative prey such as weakfish and summer flounder. Within the forage hypothesis model (section **C10.0**) it was shown that the decline in menhaden was likely responsible for the recent rise in striped bass predation on weakfish after 1996. Moreover, 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 2005b). 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. Spiny dogfish are found coast-wide from Canada to Florida (Stehnik 2007), but their post 1998 rise inshore has occurred mostly along the Mid and North Atlantic subregions, thus allowing spiny dogfish to spatially overlap weakfish. The predator assemblage south of Cape Lookout, NC is largely a different field of predators such as channel bass, spotted seatrout, king mackerel, and cobia. This well -defined spatial match between prey (weakfish) and predators (striped bass and spiny dogfish) 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; Rudershausen 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).

Although most of the statistical and empirical evidence given here and elsewhere (Uphoff 2009, this assessment) supports the Predation Hypothesis, other factors such as unreported commercial and recreational landings and discards, disease, toxins and parasitism cannot be ruled out at this time to explain the annual production loss of between 3,000 and 5,000 mt of weakfish. At this time, there is no evidence that would link recent increases in disease, toxins and parasitism to the recent failure of weakfish. There has been a recent rise in sea surface water temperatures along the Atlantic coast (Oviatt 2004), but this analyses indicated that water temperature shifts were not significantly ($P < 0.05$) linked to recent increases in weakfish juvenile mortality (Z0), nor in the decline in weakfish surplus production and stock biomass. It is possible that an enormous upsurge in unreported weakfish landings and commercial and recreational discards took place between 1996 and 2008 to account for the estimated 3,000 to 4,000 mt annual loss of weakfish surplus production, but a recent upsurge in unreported landings seems unlikely for several reasons. First, if the sources of this rapid upsurge in unreported weakfish landings and discards are thus far unknown, it would be nearly impossible to remove this source of mortality without closing virtually all inshore fishing activity between North Carolina and Rhode Island. Second, if a recent rise in unreported landings and discards resulted in the recent weakfish stock collapse, we would expect that other finfish stocks with a similar temporal and spatial distribution as weakfish (i. e. Atlantic croaker and summer flounder) to be likewise depleted. But Atlantic croaker and summer flounder stocks have either grown or have remained relatively stable from 1998 to 2008. Third, if a recent rise in weakfish unreported landings caused the recent weakfish stock collapse, all of the statistical and empirical evidence presented elsewhere in this assessment on enhanced predation would have to be regarded as a mere coincidence. Finally, unreported landings in the order of 3,000 to 4,000 mt annually are equivalent to about 5 times the current (2007-2008) known landings and estimated discards used in this weakfish assessment. The possibility that such an astronomical rise in unreported landings and discards took place after 1998 and then remain hidden from port agents, enforcement and management agencies seems remote.

The strong inverse relationship between predatory consumption rates (Mp) rates and weakfish biomass since 1997 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 makes stock rebuilding via the implementation of fisheries management

measures an exceedingly difficult task. As indicated by Spencer and Collie 1997b), 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 Fmsy of 0.72 in all but one year since 1996. Under depensatory predation, the weakfish stock would be 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 low 2008 stock biomass level of 1,333 mt which is about 92% below the new steady-state Bmsy level of 17,100 mt. The phenomenon of depensatory mortality, if driven largely by striped bass and spiny dogfish predation, could lead to a persistent and perhaps irreversible failure in weakfish productivity unless striped bass and spiny dogfish 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 findings of compensatory density-dependent survival during or before the juvenile stage. Compensatory mortality was manifested here by the steep ascending limb and pronounced dome on the Ricker stock-recruitment curve, as well as by the recent and persistent rise in the coast-wide juvenile recruitment (N0) after 1995. This apparent recruitment surge took place despite the pronounced rise in juvenile mortality (Z0) and drop in weakfish biomass after 1998. 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 on weakfish harvest, would reduce the 2008 numbers weighted FN of 0.19 and landings to near zero. A coast-wide moratorium would also reduce the high total mortality (Z) levels on ages 1+ weakfish by about 25% based on the average 2004-2008 ratio of fishing (F) to total mortality (F / Z) (refer to Table C9.3-5). 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 1981 to 1989, 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, however, the F/Z ratios fell quickly to below 0.30 in most years, thereby greatly reducing the likelihood that management measures imposed after 2003 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. 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 single-species stock assessments conducted along the Atlantic coast, natural mortality (M) of age 1+ weakfish was initially assumed to be constant (M = 0.25) in previous Yield-

per-Recruit and VPA model runs. Results from the index-based analysis (section **C8.0**) indicated that ages 1+ F estimates remained low and steady from 1999 to 2008 despite a systematic rise in ages 1+ total mortality (Z) after 1998 as indicated by ADAPT. These conflicting trends between Z and F strongly suggest that natural mortality (M) has recently tripled in magnitude, and was therefore the primary cause for the recent weakfish stock failure along the Atlantic coast. The management consequences of assuming a constant M when the annual M values actually rises systematically over time can be serious (Swain and Chouinard 2008). 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 1.40 in 2007 based on the 2009 VPA run despite the recent precipitous drop in landings and tuning indices. 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 mainly 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.

C10.0 Evaluate AIC-based hypothesis testing of fishing and additional predation-competition effects using multi-index biomass dynamic models with and without prey-based, predator-based, or ratio dependent predator-prey extensions. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice. (TOR #6)

C10.1 Introduction

Weakfish fisheries were subject to increasingly restrictive management during the early and mid-1990s (ASMFC 2002). In the late 1980s, states south of VA did not have length limits; commercial minimum length limits were 229 TL in VA and NJ, 250 mm in MD and DE, and 305 mm in NY, CT, and RI (Vaughan et al. 1991; see Figure 1 for map). Recreational fisheries were generally subject to the same length limits, although NJ and CT did not have any. States required a general permit for harvest and there were some gear and area restrictions at that time (Vaughan et al. 1991). After imposition of Amendment 3 in 1996 (a coastwide 305 mm length limit or its equivalent allowing at least 1-2 years of spawning, trawl and gill net mesh restrictions, recreational bag limits, reductions in directed commercial fishery effort, and fish excluder devices to minimize bycatch in southern shrimp fisheries; ASMFC 1996), weakfish were thought to be on the path to recovery, but by 2000-2003 recreational and commercial landings of weakfish along the Atlantic Coast approached all-time lows as population estimates derived from age structured or biomass dynamic models (De Silva 2002; Kahn 2002; Uphoff 2005b) reached all-time highs. This disconnection between weakfish stock assessment results, management actions, and fishery performance lead the WTC to consider external factors that lie outside of the purview of traditional single-species assessments.

The most reliable estimates of trends or values of F during 1981-2003 indicated it had been modest since at least 1995, while weakfish abundance and surplus production declined to low levels (Crecco 2006; Kahn et al. 2006; Uphoff 2006a). This lead to formulation of a hypothesis that the

most likely cause of declining weakfish fortune was increased natural mortality (Kahn et al. 2006). Decline of weakfish in response to increasing M was a default postulate reflecting the difficulty in finding evidence of increasing F, constant M, or recurring year-class failure. Fishing is often blamed for declines in fish stocks, but there is a preponderance of evidence that many other factors cause stock fluctuations (Deriso et al. 2008). During 2004-2006, a series of analyses (summarized below; Crecco 2006; Kahn et al. 2006; Uphoff 2006a) investigated hypotheses that weakfish population dynamics, growth, and survival were negatively affected by fishing, environmental conditions, forage abundance, competition and predation. As a result of these analyses, two strong covariates emerged: Atlantic menhaden (forage) abundance, and predation-competition from striped bass. Other candidate predator-competitors in particular were suggested by these analyses, but were less consistently identified as factors and often co-varied with striped bass (Crecco 2006; Uphoff 2006a).

Crecco (2006) and Uphoff (2006a) 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. This predator-prey model was a biomass dynamic model with a sigmoidal type III predation function added to estimate additional predation losses (Collie and Spencer 1993). When applied generally, this predator-prey model reproduced rapid shifts in abundance exhibited by marine fish populations (Steele and Henderson 1984) and was useful in exploring the role of spiny dogfish *Squalus acanthias* predation on Georges Bank haddock *Melanogrammus aeglefinus* recovery and management (Spencer and Collie 1995; 1997b). Collie and DeLong (1999) applied delay-difference biomass dynamics models with additional species interaction terms (competition and Type I, II, or III predator functional responses) to evaluate multispecies interactions in the Georges Bank fish community.

Biomass dynamic models are the simplest full stock assessment method (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 - an index of relative abundance, landings (both in weight), and, in the case of species interactions, biomass or indices of predator-competitors. The stock is considered as undifferentiated biomass and age, size, and sex structure are ignored (Haddon 2001). When species interactions are not considered, variation in exploitation and biomass is important for fitting the model - length of the time-series is not (Hilborn and Walters 1992). In some situations, biomass dynamic models do not perform well (National Research Council 1998) and fisheries scientists usually prefer age-structured models (Hilborn and Walters 1992). However, biomass dynamic models may provide as good or better estimates of management parameters at a fraction of the cost (Hilborn and Walters 1992; Walters and Kitchell 2001).

The WTC has pursued biomass dynamic modeling since the late 1990s (Uphoff 2005a) as a contrasting approach to age-structured models of weakfish dynamics. Three potential problems impacting age structured assessment approaches for weakfish have been identified by the WTC: (1) estimates of F and stock were based on scale ages up through the mid-1990s and otolith ages afterwards (these aging methods may not assign the same ages), (2) age structure was not sampled well in all regions over many years (states from NJ and north had ages assigned from states south even though larger weakfish were more common and higher size limits were in place), and (3) retrospective bias of most recent year estimates from ADAPT and ASAP has been severe (NEFSC 1998b; NEFSC 2000; Kahn et al. 2006; section **C7.0** of this report), leading to questions about the ability of these techniques to supply current management advice (Walters and Martell 2004). Hilborn and Walters (1992) suggested a pragmatic approach of using both age structured and surplus production modeling when data are available because they are fundamentally different approaches to

answer many of the same stock assessment questions. 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 (Hilborn and Walters 1992).

Biomass dynamic models could also be easily modified with predator-prey terms for hypothesis testing. In Crecco (2006) and Uphoff (2006a), estimates of M and F from the Steele-Henderson model were used for investigating whether F alone or additional non-equilibrium changes in M associated with predation-competition could be influencing weakfish. Crecco (2006) largely explored biomass dynamics using a Gompertz (asymmetric) production function, while Uphoff (2006a) employed a logistic-based (symmetric) version.

In this approach to assessing 1981-2006 weakfish biomass dynamics, we have evaluated the relative merits of multiple hypotheses examining whether fishing alone or fishing and interactions with striped bass influenced biomass dynamics of weakfish. Included in this analysis is an evaluation of the two types of production functions used by Crecco (2006) and Uphoff (2006a). In addition to fishing only models, Type I, II, and III prey-dependent predation functions (Ginzburg and Akçakaya 1992) were considered as were two additional depensatory mortality functions. Detection of moderate to severe depensatory mortality is important because stocks that are subject to it often undergo sudden and persistent drops in surplus production and shifts into domains of population behavior that are unresponsive to management (Spencer 1997; Walters and Kitchell 2001). A decline toward extinction may even be possible (Hilborn and Walters 1992).

Platt (1964) offered that strong inference and rapid progress in science can be derived from (1) devising alternative hypotheses; (2) devising crucial experiments with alternative, exclusive outcomes (as much as possible); (3) carrying out the experiments to get as clean a result as possible; and (4) making sequential hypotheses from possibilities that remain. Burnham and Anderson (2001) described the use of Kullback-Leibler information as a basis for making valid inference from analyses of empirical ecological data and to objectively answer "What model should be used to approximate reality given the data at hand?" Three general principles guide model-based inference of ecological data: simplicity and parsimony, multiple working hypotheses, and strength of evidence (Burnham and Anderson 2001). As part of a new paradigm for stock assessment in the 21st century, Sainsbury (1998) advocated formulating multiple hypotheses about stock status and evaluating them with empirical data. Deriso et al. (2008) presented a framework for evaluating the cause of fishery declines by incorporating covariates into assessment models to evaluate fishing effects versus other natural or human impacts. This framework has several components: identifying factors potentially affecting the stock, developing a stock assessment model that incorporates them, fitting the model, testing hypotheses, and evaluating impact of the factors (Deriso et al. 2008). We have applied this outline to structure this investigation of weakfish population dynamics during 1981-2006. Identification of factors was largely carried out in the previous assessment (Crecco 2006; Kahn et al. 2006; Uphoff 2006a) and the remaining components related to model development and hypothesis testing are the focus of this assessment of 1981-2006 weakfish dynamics.

The following analysis investigates both direct and indirect trophic interactions between weakfish and potential predators and/or competitors. Hereafter it will be referred to as the "forage hypothesis" to distinguish it from the more direct "predation hypothesis" presented in section **C9.0**.

C10.2 Summary of Previous Analyses of External Factors Affecting Weakfish

C10.2.1 Predator-competitors and climatic variables

Crecco (2006) and Uphoff (2006a) used diet literature review, statistical analyses, and empirical models to examine major processes (predation, competition, environmental forcing, and forage availability) affecting weakfish biomass dynamics. This empirical approach provided a bridge between the common assumption that predation and competition impose a constant rate of mortality that allows the stock to be assessed in isolation from its environment (single species assessment; Bax 1998; Sainsbury 1998) and more complex process-based multispecies models (Whipple et al. 2000).

Crecco (2006) and Uphoff (2006a) explored whether shifts in weakfish landings, growth, natural mortality, and distribution or their proxies coincided with climatic factors, forage or predator-competitor abundance or biomass using correlation and regression analyses, inferring competition for forage if test results were logical and statistically significant. Potential predator-competitors (striped bass, summer flounder *Paralichthys dentatus* , bluefish *Pomatomus saltatrix*, spiny dogfish *Squalus acanthias*, and Atlantic croaker *Micropogonias undulatus*), major forage species for piscivores along the mid-Atlantic (bay anchovy *Anchoa mitchilli*, Atlantic menhaden, and spot *Leiostomus xanthurus*), and major climatic features (North Atlantic Oscillation or NAO, and water temperature) were evaluated as candidates for inclusion in assessment models. Analytical strategies of Crecco (2006) and Uphoff (2006a) consistently and strongly indicated striped bass was a major predator – competitor of weakfish.

Crecco (2006) used three stepwise multiple regression analyses ($P < 0.05$) to test linkages of biotic and abiotic factors to changes in weakfish surplus production (SP), M on ages 1+, and juvenile M. Explanatory variables in these analyses included F, abundance or relative abundance of candidate finfish predator-competitors, average annual surface water temperature, and deviations in the winter NAO (both lagged 1 or 2 years to coincide with weakfish recruitment to the adult stock). Coast-wide biomass of spiny dogfish, and coast-wide striped bass, summer flounder, and bluefish stock abundance estimates from 1982 to 2004 were taken from stock assessments. In the case of the Atlantic croaker, assessment estimates were available from 1982 to 2002, so 1982-2004 total recreational catch of croaker (MRFSS A, B1, B2 estimates) was used as a coast-wide index (1982-2002 recreational catches were highly correlated with abundance estimates; Crecco 2006).

Stepwise regressions selected striped bass abundance as the only ($P < 0.001$) explanatory variable inversely related to SP and juvenile M, while summer flounder biomass was inversely related to M of ages 1+ (Crecco 2006). Correlation analyses indicated that SP, M on ages 1+, and juvenile M were also inversely associated ($P < 0.05$) with striped bass abundance estimated from tagging (SP, M on ages 1+, and juvenile M) and to Atlantic croaker relative abundance (SP and juvenile M). Due to substantial co-linearity among striped bass, summer flounder, and Atlantic croaker indicators, stepwise models were unable to select second or third significant variables (Crecco 2006).

Uphoff (2006a) applied three approaches to evaluate the potential of striped bass, Atlantic croaker, bluefish, summer flounder, and spiny dogfish to influence weakfish biomass dynamics. First, long-term trends (1950-2003) in Atlantic Coast commercial landings (excluding spiny dogfish) were correlated with weakfish landings to determine if significant ($P \leq 0.10$) negative associations existed. Second, biomass estimates from single species assessments (generally 1981-2003) were used in a species-specific predator-weakfish model (logistic biomass dynamic model with a Type III predation function). Deterministic runs were made and model fit, 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 using the rescaled relative F technique (landings divided by average indices scaled into VPA numbers or biomass weighted F; Kahn et al. 2006) were used to evaluate the candidates. 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 (Uphoff 2006a). Uphoff (2006a) developed a "scorecard" for each candidate predator-competitor based on these three components. Ten criteria were considered - four based on statistical or model results, and six on surveyed diet literature. The 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), spiny dogfish (5.2), and Atlantic croaker (2.6).

Uphoff (2006a) conducted additional explorations of the role of forage and striped bass on long-term commercial landings and the DE PSD Q+ index (a length-frequency index; discontinuous 1966-2003 time-series; Appendix C-2) with correlation, categorical or stepwise regression analyses (Neter and Wasserman 1974; Rose et al. 1986; Freund and Littel 2000). The DE PSD Q+ index was considered an indicator of long-term weakfish dynamics because of its significant and positive associations with recreational trophy citations, commercial and recreational yield, MRFSS catch per trip, and inshore-offshore recreational harvest distribution (Uphoff 2004; Kahn et al. 2006; Appendix C-2). Forage abundance indices and a long-term indicator of striped bass biomass (LBI, a Chesapeake Bay egg presence-absence index of spawning biomass; Uphoff 1997) were independent variables. Indices of Atlantic menhaden, spot, and bay anchovy relative abundance from NC, VA, MD, DE, and NJ, age 0 abundance estimates of Atlantic menhaden along the Atlantic Coast, and LBI were used in correlation and stepwise regression analyses with DE PSD Q+ (Uphoff 2006a).

A negative relationship of striped bass and weakfish landings that reflected high or low periods of underlying weakfish productivity was described by categorical regression of period and striped bass landings ($P < 0.001$; see Figures 18 and 19 in Uphoff 2006a). This approach predicted high (1973-1989) and low (1950-1972 and 1989+) periods of weakfish landings as abrupt time-blocks. Stepwise regressions described a negative relationship of weakfish landings with LBI and a positive relationship with Atlantic menhaden abundance or $\log_e$-transformed juvenile menhaden indices during 1955-2002 ($P < 0.0001$). In general, menhaden accounted for most variation in weakfish landings and striped bass accounted for a lesser amount in these stepwise regressions and reproduced the time-blocks of weakfish landings described by the categorical regression of striped bass landings and productivity period (Uphoff 2006a).

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 forage species and sporadically in NJ and NC (Uphoff 2006a). Stepwise regressions of striped bass biomass and juvenile menhaden abundance against DE PSD Q+ indicated a significant positive influence of forage and negative influence of LBI (Uphoff 2006a).

Changes in weakfish landings and size distribution were logically consistent with trends in menhaden relative abundance and striped bass biomass over a broad geographic area and time-span. These changes, coupled with little evidence of a rise in F, supported the development of hypotheses linking the aborted weakfish recovery to striped bass and menhaden.

C10.3 Background for Weakfish, Striped Bass, and Atlantic Menhaden Interactions

C10.3.1 Spatial and Temporal Distributions

Weakfish occur along the Atlantic coast from FL to MA, straying as far north as Nova Scotia (Mercer 1985; Figure C10.3-1). They are most abundant from NC to NY. Adult weakfish disperse

from continental shelf wintering grounds located from lower Chesapeake Bay to Cape Lookout, NC, as waters warm in spring and initiate an inshore, northerly migration (Mercer 1985; ASMFC 2002). They enter sounds, bays, and estuaries in early spring and may stay through summer, although most in northern states return to the ocean (Mercer 1985). In fall, a general southerly movement back to overwintering grounds occurs (Mercer 1985).

Striped bass are found along the entire Atlantic coast from Canada to FL, but are most abundant between ME and NC. Migratory stocks of striped bass are mainly composed of fish spawned in Hudson River (NY) or Chesapeake Bay, but Chesapeake Bay fish predominate and are associated with most of the production along the Atlantic coast (Boreman and Austin 1985; Dorazio et al. 1994; Richards and Rago 1999). After spawning in tidal-fresh Chesapeake Bay rivers during April-May, most immature and mature striped bass females and some males migrate to spend summer in coastal waters of the mid-Atlantic (VA-NY) and New England (CT-ME; Dorazio et al. 1994). Immature females largely leave the Chesapeake Bay after ages 2-3 and reside in coastal waters until they mature. Remaining, mostly male, striped bass reside within estuarine waters of Chesapeake Bay throughout summer-fall (Dorazio et al. 1994; Secor and Piccoli 2007). In fall, southward movement of coastal fish begins (Dorazio et al. 1994). During November-March, some striped bass from Chesapeake Bay and the coastal contingent overwinter in the inshore zone between Cape Henry, VA, and Cape Lookout, NC (Setzler et al. 1980).

Atlantic menhaden are distributed from FL to ME during summer, with older, larger individuals found further northward and smaller, younger fish in the southern half of the range (Ahrenholz 1991). In September, the northerly portion of the population begins to migrate southwards. By December, these fish are in waters off the NC coast. These fish are followed by large numbers of young-of-year menhaden (Ahrenholz 1991). Although described as disappearing and dispersing from inshore waters by late January (Ahrenholz 1991), menhaden were dominant in striped bass diets in nearshore waters from the VA Capes to the Outer Banks of NC during December-March, 2000-2007 (Overton et al. 2008). During March-early April, schools move rapidly northward and have redistributed by June (Ahrenholz 1991). This general pattern of movement is closely approximated by weakfish (ASMFC 2004) and striped bass (Walter et al. 2003).

C10.3.2 Trophic Ecology of Weakfish and Striped Bass

Weakfish feed throughout the water column (Wilk 1979; Mercer 1985). Young weakfish feed primarily on mysid shrimp and anchovies, while older weakfish feed on the clupeid species that are abundant in a given area. Cannibalism has been reported. Functional morphology indicates that weakfish are upper midwater feeders (Wilk 1979; Mercer 1985).

Striped bass evolved as a schooling species to take advantage of the great energy resource of clupeids along the Atlantic Coast of North America (Stevens 1979). Striped bass actively select for Atlantic menhaden, but will feed on other species when menhaden are not sufficiently abundant (Overton 2003; ASMFC 2004; Rudershausen et al. 2005).

In the mid-Atlantic region, bay anchovy represented the prey most consumed by age 0 weakfish and age 1 striped bass as they initiated piscivory, but both switched mostly to similar, larger prey (clupeids, primarily Atlantic menhaden) within a year (Mercer 1985; Taylor 1987; Hartman and Brandt 1995). Juvenile menhaden and spot would be most relevant to weakfish and younger, smaller striped bass making the transition through piscivory because of size selectivity and gape limitations (Stein et al. 1988; Juanes 1994).

Weakfish expanded the size of items in their diet least of 18 species examined by Scharf et

al. (2000) as they grew. They largely remain dependent on items in the size range of bay anchovy, and juvenile menhaden or spot; maximum fish prey size of an 800 mm weakfish was about 180 mm (Scharf et al. 2000). Walter and Austin (2003) and Overton et al. (2008) found that large striped bass fed on small pelagic prey (bay anchovy and juvenile clupeids) as well as large prey items. The upper 99% quantile of prey fish total length that can be eaten by 800 mm striped bass is approximately 280 mm and a 1,200 mm striped bass can eat a 415 mm fish (F. Juanes, University of Massachusetts, personal communication). Minimum prey size is below 50 mm for the entire size ranges of weakfish and striped bass.

Weakfish have been specifically identified as a minor food item in some, but not all, striped bass diet studies, particularly in the mid-Atlantic region (Manooch 1973; Wilk 1979; Mercer 1985; Walter and Austin 2003; Overton et al. 2008). Weakfish were encountered in striped bass diets during fall or winter (a period of heavy pelagic feeding) in the mid-Atlantic region and accounted for no more than 1-3% of striped bass diet by weight or volume (Manooch 1973: Walter and Austin 2003; Overton et al. 2008). These overwintering striped bass feed mostly on Atlantic menhaden and bay anchovy (Manooch 1973; Austin and Walter 2003; Overton et al. 2008).

Early switching of weakfish and striped bass to a fish diet indicates that both are specialized piscivores (Persson and Brönmark 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 at a feeding stage where they experience intense competition (Persson and Brönmark 2002). Species undergoing ontogenetic diet shifts face a risk of delayed transitions among feeding stages if food resources are limited and competition is intense (Persson and Brönmark 2002). Competing individuals and predators may hinder one another's feeding activities, leading to starvation or they may eat one another (including cannibalism; Yodzis 1994). 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 (Persson and Brönmark 2002). Individuals not reaching size advantage over prey may become stunted at size where consumption balances metabolic requirements (Bax 1998; Persson and Brönmark 2002).

C10.4 Current Stock Assessment

C10.4.1 Harvest and Discard Estimates

Biomass dynamic models employed in this analysis used total weight of aggregated harvest and discards by both the commercial and recreational fisheries. Commercial landings and MRFSS recreational harvest estimates were obtained from the National Marine Fisheries Service (www.st.nmfs.gov/st1/).

De Silva (2004) provided the first quantitative analysis of weakfish discards based on ratios of discards to harvest in 1994-2003 NMFS at sea observer data. The NMFS 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. Only hauls that were completely sampled for discards were evaluated. During 1994-2003, discard data were primarily available for gillnets (287 trips and 615 hauls; 89% landed in VA, NC or NJ) and otter trawls (196 trips and 519 hauls; 41% landed in MD and 49% in NJ, NY, or RI.). The annual number of gillnet trips where weakfish discards were observed ranged from 5 to 68. Annual number of otter trawl trips where weakfish discards were observed ranged from 4 to 45 (De Silva 2004).

The WTC adopted De Silva's (2004) technique for estimates of weakfish discards using

annual (1994-2003) discard ratios (weakfish to target species) and NMFS commercial landings of a suite of target species for the previous assessment (Kahn et al. 2006). The ratio of at sea observations of weight of discarded weakfish to weight 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 (De Silva 2004). 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 (De Silva 2004). The WTC assumed that gear-specific relationships among the target species in the data set reflected the entire fishery.

For the current assessment, the WTC considered all estimated commercial discards to have died and chose to update de Silva's (2004) method for estimating of commercial discards (section **C5.0** of this report). Haul level data for the same gear and target species combinations were used to evaluate annual, multiyear (5 or 7 year blocks), and all-year (1994 to 2007) weakfish discard ratios. Because of concerns with high interannual variability and uncomfortably large standard errors of estimates based on short time groupings, the WTC elected to use the all-year ratio estimates. Ratio-based estimates of weakfish discard weight were made for butterfish, long-fin squid, summer flounder and weakfish fisheries for trawl harvests, and Atlantic croaker, bluefish, spiny dogfish, and weakfish gillnet harvests.

Discard sampling was not conducted until 1994 and market-related discard ratios estimated for 1994-2002 were used to estimate weakfish discards in prior years (De Silva 2004). 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. Discard estimates after 1993 used total discard ratios to estimate weakfish discards for all states.

Recreational discard losses were calculated as the product of estimates of MRFSS number discarded, mean weight, and discard mortality rate. The MRFSS does not estimate weight of released weakfish and we used mean weight of weakfish recreational discards (0.15 kg) estimated from MRFSS 2004-2007 headboat survey length-frequencies (D. Kahn, DE Fish and Wildlife, personal communication). This constant mean weight approximated the mode of six annual regional (VA and north or NC and south) mean discard weight estimates for headboat catches (range = 0.14-0.17 kg, three ≈ 0.15 kg). Weakfish hook-and-release experiments produced dichotomous mean mortality estimates either near 3% or 15% (Table C10.4-1) and 10% release mortality was adopted by the WTC.

C10.4.2 Biomass Indices

Exploitable biomass was defined as biomass of weakfish 250 mm and larger. Weakfish of this size and age (1+) and larger were heavily exploited in the past (305 mm length limit or its equivalent was not mandatory until 1994; ASMFC 2002) and represented spawning biomass (Nye et al. 2008). Size limits and gear restrictions imposed since Amendment 2 (1994) have greatly lessened vulnerability of weakfish less than 305 mm to directed harvest.

C10.4.2.1 1981-2003 Assessment's Exploitable Biomass Indices

Crecco (2006) and Uphoff (2006a) evaluated fishery and predator-weakfish biomass dynamics primarily with recreational catch (harvested and released) per effort index of relative biomass (WRI). DE and NJ trawl survey exploitable biomass indices (Kahn et al. 2006) provided additional indicators of weakfish exploitable stock biomass. These trawl surveys and WRI were positively correlated ($r \geq 0.71$; $P \leq 0.003$), but trawl survey indices were not used extensively in the

assessment (Crecco 2006; Kahn et al. 2006; Uphoff 2006a). A third trawl survey index (Northeast Fisheries Science Center fall survey) was rejected because exploitable-sized weakfish were poorly represented, it did not correlate consistently with other indices, cohort catch curves produced positive "mortality" rates, and the 40th SARC expressed concerns about high inter-annual variability (WTC 2006; Kahn et al. 2006; Uphoff 2006a).

C10.4.2.2 1981-2006 Indices of Exploitable Biomass

Three exploitable biomass indices (EBI) were used: one fishery-dependent and two fishery-independent. The fishery-dependent index (WRI) was estimated as weakfish recreational private/rental harvest and release weight per trip in all areas of the mid-Atlantic (VA-NY; Crecco 2009). A MRFSS directed trip index of weakfish catch (harvest and releases) per trip (Brust 2004) was very closely correlated (Pearson $r = 0.96$, $P < 0.0001$) with the more general WRI (Crecco 2005a), so it was not considered further in analyses.

The WRI represented the only EBI available for the entire 1981-2006 time series. Harvest weight estimates were obtained directly from the MRFSS. Weight of recreational releases was estimated from numbers released (MRFSS type B2) multiplied by average weight of released weakfish from the coastwide head-boat fishery (assumed constant at 0.15 kg; see section **C10.4**). Effort equaled MRFSS annual estimates of all trips for the private/rental boat sector. The WRI used previously by Crecco (2006) and Uphoff (2006a) assigned harvest mean weights to released weakfish and these weights were much higher than indicated by headboat releases (3-15 times higher; median ≈ 5-times).

The private boat fishery is highly mobile and capable of catching weakfish of all sizes. Weakfish 250 mm TL and larger have constituted the majority of weakfish measured by the MRFSS. Weakfish less than 250 mm and as large as 890 mm were consistently present in historic MRFSS length-frequencies (1979-1998; M. Gibson, RI Fish and Wildlife personal communication; these years are not currently available at the MRFSS website); smaller fish are still present in 2004-2008 length-frequencies available online (www.st.nmfs.gov/st1/), but larger weakfish were consistently present out to about 580 mm in 2008.

Summed mid-Atlantic catches (1981-2006) indicated that nearly all weakfish (numbers) were caught during three MRFSS waves: May-June (24%), July-August (42%), and September-October (33%). Summed mid-Atlantic private boat trips were distributed similarly: May-June (27%), July-August (40%), and September-October (22%). The Mid-Atlantic subregion of the MRFSS represented the major (38-76%, median = 62%) component of annual coastwide recreational weakfish catches.

A general recommendation for data in stock assessment is that information only be used once (Cotter et al. 2004). In the case of the WRI and this assessment, the same information is contained in both the landings and the index. However, division by effort in the index reduces the direct dependency in the data (V. Crecco, CT DEP, personal communication).

Delaware and New Jersey trawl survey EBI were used as additional indicators of weakfish stock biomass. Essential data for calculating EBI for all years included annual biomass (kg) or an estimate of central tendency of abundance of all weakfish caught per tow, annual length-frequencies (cm length groups), and an equation that predicted weight (kg) at length (cm). Each agency provided catch-per-effort data and length-frequencies for the available time-series. We assumed all length-frequencies represented random samples of lengths. Annual length-frequency was converted to weight-frequency by multiplying each cm-group catch by predicted mean weight (kg) of that cm-group.

Predicted weight-at-length was derived from annual regressions of $\log_e$-transformed weight (kg) on $\log_e$-transformed length (cm). Individual lengths and weights were available from the NEFSC fall trawl survey annually for 1992-2006 (Uphoff 2005a; G. Shepherd, NMFS, personal communication). We used a weight-length regression (converted from pounds and inches to kg and cm) to estimate weight-at-length during 1989-1991 based on combined data from NC during 1982-1983 and 1988-1989, NY during 1988-1989, and coastwide data collected by VIMS (Virginia Institute of Marine Science) during 1988 (D. Vaughan, NMFS, personal communication).

A DE EBI was derived for each year as $B_{DE} * (B_E / B_A)$; where B_{DE} is the kg per nautical mile estimated for all weakfish by DE; B_E equaled the summed weight of weakfish $\geq$ 25 cm created by multiplying cm length-group specific catch per tow by predicted weight; and B_A equaled the summed weight for all weakfish derived by the same procedure described for $B_{E.}$

DE data was available for 1966-1971, 1979-1984, and 1990-2006. We excluded years prior to 1990 because (1) recreational catches were only available since 1981, (2) the time series was discontinuous, (3) survey vessels had changed between 1984 and 1990, and (4) 1981-2003 DE EBI were poorly correlated with WRI, but 1990-2003 were significantly and positively correlated with WRI and NJ EBI (Uphoff 2006a).

Mean kg per nautical mile estimates were provided by DE for all weakfish captured. Twenty minute tows with a 9.1-m headrope trawl were made at nine fixed stations located along the eastern side of Delaware Bay each month during March-December (Michels and Greco 2004). May-October catches during 1990-2007 comprised 98% of weakfish biomass captured. May accounted for approximately 55% of biomass; June and July, 10-12%' and August-October 5-8%; CVs of annual estimates of biomass per nautical mile ranged from 16-54% with a median of 22%. A single CV (1994) was in excess of 40%, while 19 of 24 were below 25% (S. Michels, DEDFW, personal communication).

NJ has conducted a stratified random survey along its coast during 1988-2006 (catch data only were available for 1988) and provided a complete file of catch and length-frequency data. February, April, and June data were pared from estimates because of very high frequencies of zero catches (Uphoff 2005a). Indices and weight-at-length distributions were calculated from August and October rounds, the convention used for NJ age-structured indices in early versions of ADAPT (NEFSC 2000).

Uphoff (2005a) calculated three estimators of central tendency and their standard deviations for 1989-2003 NJ data: arithmetic mean (NJAM), the proportion of positive tows (NJPT), and mean $\log_e$-transformed catches + 1 (NJLN); NJLN was converted to a geometric mean (NJGM) to express central tendency. These estimates of central tendency were based on catches of all sizes of weakfish.

Precision of NJPT and NJLN during 1989-2003 was comparable (CV = 7.7-29.1%) and trends in survey time-series were similar when NJPT and NJLN were used as indicators of central tendency (Uphoff 2005a). NJAM was less precise (CV = 22.9-60.9%) and indicated different trends in abundance. Uphoff (2005a) chose NJGM as the estimator of central tendency for the NJ trawl survey because its precision was comparable or better than NJPT and it indicated the same trend in relative abundance as NJPT.

An annual NJ length-frequency was converted to weight-at-length distribution (described above). A mean weight of all weakfish sampled (MWT) was estimated by dividing the summed weight frequency by the summed numeric frequency. The exploitable fraction (B_E / B_A) of the sampled biomass was estimated as the summed weight of all weakfish $\geq$ 25 cm divided by the summed weight of all weakfish. An index of exploitable biomass (NJ EB) was then derived for each year as NJ EB = NJGM * MWT* (B_E / B_A).

The three EBI were different in scale and we used z-transformations to place them on the same scale to view relative trends and added two to each value to positively rescale them. Z-transformations were based upon means and standard deviations during 1990-2006 - years in common among all three surveys. This approach removed the need for multiple index scalars since all were in common scale.

Once these Z+2 EB indices time-series were plotted together, two general trends emerged (Figure C10.4-1). There was (1) general agreement between WRI Z+2 and DE Z+2 indices, and (2) NJ Z+2 (GM based) exhibited considerably more inter-annual variation and disagreed substantially with the others during 1998-1999 (much lower) and 2004-2005 (much higher; Figure C10.4-1). Substitution of NJ PT for NJ GM in the NJ EBI was a slight improvement at best.

This disagreement among indices led us to re-examine whether all three EBI were suitable indicators of stock status and, if not, could they be calculated from a different estimator of central tendency to conform to accepted indicators. This search for conformity implied a single stock of weakfish, even though evidence exists for multiple stocks (Munyandorero 2006). This assumption of a single stock was consistent with how assessments of weakfish have been conducted in the past and how the species has been managed (NEFSC 1998a; NEFSC 2000; ASMFC 2002; Kahn et al. 2006).

This re-examination considered four concepts for suitability: precision, accuracy, consistency among indices, and coherency (inter-annual variation of an index). Coefficient of variation (CV) was used to evaluate precision of trawl surveys and PSEs (proportional standard error; standard error of an estimate as a percentage of the estimate; Personal communication from the National Marine Fisheries Service, Fisheries Statistics Division) were used for MRFSS components. A CV (or PSE) of 40% was considered a threshold for rejection and CVs of 25% or less was desired. Estimates of CV were based on weight or count-based estimators or proportions of samples with weakfish of all sizes. Precision of WRI could not be determined directly, but PSE of harvest weight, catch, and effort were available from the MRFSS survey. Accuracy criteria were to reflect the true status of the population which, of course, was not readily answered. We used trends in 1981-2006 fishery losses (harvest and discard weight in commercial and recreational fisheries) as one indicator and biomass estimated from the converged portion (years exhibiting minimal or no retrospective bias - 982-2000) of the "best" VPA run (20) from Kahn et al. (2006) as another. Consistency was indicated by how well indices correlated among themselves. Coherency was indicated by correlation with a grand mean of Z+2 transformed WRI, DE EBI and NJ EBI. We considered the grand mean a form of smoothing. Correlation coefficients and P (< 0.05) were considered in evaluations of accuracy and consistency. Evaluation of coherency involved correlations with a grand mean of the indicators which rendered P meaningless. Coherency was indicated by similar, high correlations among indices. As a first step, the four available EBI (WRI, DE, NJGM, and NJPT) were evaluated with the four criteria.

Precision of the components of the WRI were within the desirable range. Proportional standard errors (PSE) of trips ranged from 2.6-6.7%, harvest weight PSEs ranged from 8.8-21.8%, and catch PSEs fell between 5.2% and 19.8%. Precision of NJ estimators of central tendency was largely described previously. The NJAM exceeded the 40% CV threshold and was generally above the desirable criterion (CV > 25%). Neither NJLN nor NJPT estimators had CVs above the threshold, but both had values outside the desired range. All but one DE trawl survey estimate of mean kg per nautical mile during 1990-2006 had a CV < 40% and 5 of the 17 remaining estimates had CVs between 25% and 40% (S Michels, DEDFW, personal communication).

In general, WRI and DE met the remaining criteria, while NJ EB indices performed poorly. None of the four EBI correlated at $P < 0.05$ with the converged portion of the VPA, but DE EBI and

NJ GM EBI correlated at $P < 0.1$, and WRI correlated at $P < 0.2$ (Table 2). WRI and DE EBI were significantly correlated with Atlantic coast fishery removals, but neither of the NJ EBIs were. The WRI and DE EBI were significantly correlated with each other and each was significantly correlated with one of the NJ EBI; the NJ indices were significantly correlated with one another. WRI and DE EBI ($\rho = 0.82$-0.95) were more strongly associated with both sets of grand means than either NJ EBI (NJGM, $\rho = 0.70$ and NJPT, $\rho = 0.49$; Table C10.4-2). We concluded that WRI and DE EBI were suitable indicators of weakfish biomass, but that other estimators should be considered for NJ.

We began by looking at trends in each month (August and October) used in the NJ EBI. We concentrated on the proportion of positive tows (PT) as an indicator of central tendency because its precision should not have been as sensitive to reductions in sample size (from N = 71-79 for two months to N = 34-41). I calculated the proportion of positive tows and its 95% confidence interval by using the normal distribution to approximate the binomial probability distribution (Ott 1977). Proportion of zero catches and its inverse, NJPT in this case, tend to be robust to errors and biases in sampling when zeros are frequently encountered (Green 1979; Mangel and Smith 1990). Presence-absence indices have been more robust indicators of yellowtail snapper and age 0 white sturgeon abundance (Bannerot and Austin 1983; Counihan et al. 1999), California sardine (Mangel and Smith 1990) and upper Chesapeake Bay striped bass egg abundance (a spawning stock biomass indicator; Uphoff 1997), and longfin squid fishery performance (Lange 1991) than catch per effort. Marine surveys often contain substantial proportions of zeros and log-normal distributions of positive catches that cause high variability and low precision of sample means (Pennington 1983; 1996). The use of more efficient estimators is one way of increasing survey precision (Pennington 1983; 1996; McConnaughey and Conquest 1993).

CVs of NJ PT for either month were generally within the threshold (CV < 40%), with one exception (August 2003). All October NJ PT CVs were less than 25%, while two August NJ PT CVs were greater than 25%, but less than 40%. CVs were usually higher in August (14 of 18 were greater than October), but confidence intervals of August or October NJPT were significantly different from 0 at $P = 0.05$ in all years.

We explored two options for calculating EBI from NJ August or October PT. The first option was EBI = PT * EWT and the second was EBI = PT * E * EWT, where E is the exploitable fraction (number of weakfish $\leq$ 250 mm / total number) for the August or October sampling round and EWT is mean weight of exploitable weakfish in either round. Criteria for evaluating EBI indices described previously were used to evaluate the four potential indices.

August round PT * EWT (Figure C10.4-2) met all criteria. It was significantly correlated with WRI, DE EBI, total fishery losses, and the converged portion of the VPA at $P < 0.05$ (Table C10.4-3). Correlations of this index with the grand mean of WRI, DE, and this NJ EBI (August round PT * EWT; all indices z-transformed +2; z-transformation based on 1990-2006) were similar ($\rho = 0.89$, 0.90, and 0.86, respectively). Other indices only met a single criterion each and were not as strongly correlated with their respective grand means (Table C10.4-3). We selected August round PT * EWT as NJ EBI and it is denoted as NJA EBI.

C10.4.3 Striped Bass and Atlantic Menhaden Biomass

Versions of predator-prey models required estimates of striped bass and Atlantic menhaden biomass. The annual sum of SCAM (Statistical Catch at Age Model) biomass estimates for 2 year-old and older striped bass were used as predator-competitor biomass during 1982-2006 (NEFSC 2008). Striped bass of this size would also be important predators of menhaden and would be capable of consuming weakfish (Hartman and Brandt 1995; Walter et al. 2003; Uphoff 2003; Uphoff

2006a; Overton et al. 2008). Atlantic menhaden biomass estimates (ages 1+ during 1981-2005) from a forward projection model were used (ASMFC 2006b).

Neither the striped bass nor menhaden assessments provided time-series that matched the weakfish time-series; 1981 was missing for striped bass and 2006 was missing for Atlantic menhaden. We applied two techniques to fill in these missing estimates. The approaches used to make these estimates will be described briefly.

We developed a known-biomass production model (MacCall 2002) for ages 2+ striped bass using SCAM estimates of biomass, landings, and discard biomass (assumed to be 10% of landings) to estimate biomass. We used SCAM biomass estimates for 1982-2006 (G. Nelson, MA DMF, personal communication; NEFSC 2008), and recreational and commercial landings for 1981-2006 to generate an estimate of biomass in 1981. Parameters r, K, and the model scalar (see modeling section) based on 1982-2006, combined with estimates of fishery losses would allow for an estimate of biomass in 1981. Biomass estimates from the known-biomass production model had a modest bias (10%) and we adjusted output downwards by this amount. Biomass of striped bass in 1981 was estimated to be 8,789 mt (Figure C10.4-3).

We used categorical regression (Neter and Wasserman 1974; Rose et al. 1986) to estimate 2006 biomass of ages 1+ menhaden (from the forward projection model; ASMFC 2006b) from landings. We constructed annual age 1+ biomass estimates by multiplying abundance at age by mean weight derived from length-at-age and weight-at-length equations in ASMFC (2006b). A categorical variable was used to split the 1955-2005 time-series into periods where bait landings were not estimated (reduction landings only during 1955-1984; coded 0; ASMFC 2006b) and the remaining period where bait estimates were added to reduction landings (coded 1). This model provided a significant fit ($R^2 = 0.70$, $P < 0.0001$; Figure C10.4-4) and serial patterning of residuals was not indicated. All parameters were significant at $P < 0.04$ and the relationship was described by the equation: $B_t = (L_t * 2.484) + (217{,}798 * C) - 221{,}922$; where B_t = biomass in year t in MT, L_t = landings in MT, and C = bait landings category. Biomass of ages 1+ Atlantic menhaden in 2006 was predicted to be 449,784 MT (Figure C10.4-4).

C10.4.4 Weakfish Biomass Dynamic Models

Biomass dynamic models provided a stock assessment framework that allowed the incorporation of covariates for hypothesis testing. Both standard and predator-prey production models evaluated the effect of fishing, while the latter allowed for estimation of predation-competition impacts as well. The small size (250 mm) that defined exploitable biomass of weakfish allowed for use of biomass dynamic models without lags necessary in some delay-difference predator-prey models of other species (Collie and DeLong 1999; Overholtz et al. 2008).

Harvest estimates are summarized in Table C10.4-4; indices, striped bass biomass, Atlantic menhaden biomass, and the ratio of menhaden to striped bass biomass are summarized in Table C10.4-5.

WRI (1981-2006), DE EBI (1990-2006), and NJA EBI (1989-2006) were used as indices of relative abundance. Each set of indices were *Z*-transformed based on means and standard deviations from a common time period (1990-2006) and 2 was added to remove negative values (designated as ZEBI). This transformation allowed for a single index scalar (a nuisance parameter). Landings during 1981-2006 included both directed harvest and discards by the recreational and commercial fisheries. Biomass estimates for 2 year-old and older striped bass and ages 1+ Atlantic menhaden were used.

Two types of production functions were considered: the symmetric Schaefer (logistic) model and the asymmetric Gompertz model (Haddon 2001). Previously, Uphoff (2006a) used the Schaefer model as the basis for exploration fishing and predation-competition effects on weakfish, while Crecco (2006) used the Gompertz version.

Biomass dynamics based on the Schaefer model were described by the following discrete time-step equation:

$$\mathbf{B_t = B_{t-1} + rB_{t-1}\,(1-(B_{t-1} / K)) - H_{t-1} - D_{t-1} + \varepsilon;} \qquad (1)$$

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; D_{t-1} was predation-competition losses from striped bass and ε observation error (described below; Hilborn and Walters 1992: Collie and Spencer 1993; Spencer and Collie 1995; Haddon 2001).

The Gompertz predator-prey model was expressed as

$$\mathbf{B_t = B_{t-1} + \log_e(K)\;\; r\, B_{t-1}\, [1-((\log_e (B_{t-1})) / (\log_e(K))] - H_{t-1} - D_{t-1} + \varepsilon} \qquad (2)$$

Quinn and Deriso 1999; Haddon 2001; Crecco 2006). Parameter definitions are the same as equation 1.

In all, a dozen models were formulated, a half-dozen for each production function. Fishing only hypotheses were modeled by omitting functions that estimated D_{t-1}. Predator-prey functions described Type I, II, and III predator functional responses, as well as two functions designed to mimic depensation. The Type I, II, and III responses assumed that rates of consumption by striped bass were dependent on weakfish density (prey-dependent trophic function; Ginzburg and Akçakaya 1992).

The Type I functional response estimated D_{t-1} as

$$\mathbf{c\, P_{t-1}\, B_{t-1}}; \qquad (3)$$

where c = a positive constant; P_{t-1} = striped bass biomass in the previous year; and B_{t-1} = biomass of weakfish in the previous year (Yodzis 1994; Collie and DeLong 1999). Estimates of D_{t-1} from the Type II function were estimated as

$$\mathbf{(cP_{t-1}B_{t-1}) / (1 + a\, B_{t-1})}; \qquad (4)$$

where c is a positive constant and a is a coefficient that combines search time and probability of capture (Yodzis 1994; Collie and DeLong 1999). The Type III function estimated D_{t-1} as

$$\mathbf{[(dP_{t-1}(B_{t-1})^2)/) / (A^2+(B_{t-1})^2)]}\ ; \qquad (5)$$

where d is maximum per capita consumption by striped bass biomass (P_{t-1}); and A is weakfish biomass where predator satiation begins (Collie and Spencer 1993; Spencer and Collie 1995).

Depensatory models did not include B_{t-1} to estimate D_{t-1}. The simplest depensatory model describes D_{t-1} solely as a function of striped bass (predator-competitor) biomass:

$$\boldsymbol{c}\mathbf{P_{t-1}}. \tag{6}$$

A final depensatory model expressed D_{t-1} as a function of the ratio of striped bass biomass to the biomass of its main prey, Atlantic menhaden (R_{t-1}) and striped bass (predator-competitor) biomass:

$$\mathbf{P_{t-1}}(\boldsymbol{c}/\ \mathbf{R_{t-1}}). \tag{7}$$

This equation expresses D_{t-1} solely as a function of striped bass attack success on its main prey, Atlantic menhaden and weakfish were alternative prey. Ratio-dependence is another approach to functional response theory, and the ratio summarizes interference of predators with one another and spatial restriction in foraging (Ginzburg and Akçakaya 1992; Walters and Juanes 1993; Yodzis 1994; Walters and Martell 2004). Natural systems may be closer to ratio-dependence than prey-dependence (Ginzburg and Akçakaya 1992).

The Haddon (2001) spreadsheet version of a biomass dynamic model was easy to adapt to predator-prey formulations because it estimated weakfish biomass as a first step and then estimated an annual scalar (q_t) as B_t / $ZEBI_t$ (Haddon 2001). Estimating weakfish biomass first allowed striped bass biomass to be used directly rather than converting them to weakfish index equivalents. The geometric mean of annual estimates of q_t (GM q_t) was used to predict $ZEBI_t$. as geometric mean GM q_t * B_t. Biomass was estimated directly for 1981 (B_{1981}) and then the model estimated subsequent years (Haddon 2001).

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 2001) was used to estimate predator-prey model parameters that minimized the sum of observation errors (observed $\log_e ZEBI_t$ - predicted $\log_e ZEBI_t)^2$ (Haddon 2001). The spreadsheet version of the model combined with the genetic algorithm provided a great deal of flexibility for trying different model variations.

C10.4.5 Hypothesis Testing

We used Akaike information criteria adjusted for small sample size, AIC_c, to evaluate the 12 models describing hypotheses that related changes in weakfish biomass to fishing alone or to fishing + predation-competition losses due to striped bass (Burnham and Anderson 2001):

$$\mathbf{AIC_c = -2(log\text{-}likelihood) + 2K + [(2K(K+1)) / (n\text{-}K\text{-}1)]}; \tag{8}$$

where n is sample size and K is the number of model parameters. We calculated lognormal likelihoods as described by equations 3.29 and 3.30 in Haddon (2001). We rescaled AIC_c values to Δ_i, ($AIC_{c\,i}$ – minimum AIC_c) where i is an individual model (Burnham and Anderson 2001). The Δ_i values provide a quick "strength of evidence" comparison and ranking of models and hypotheses. Values of $\Delta_i \leq 2$ have substantial support, while those > 10 have essentially no support. We used Δ_i to calculate Akaike weights that approximated the probability that model i constituted the best model of the hypothesis given the data. The best model or subset of models were used to generate parameters of interest, estimate variability, and explore model sensitivity. An R^2 was calculated for

the best model or models to describe goodness of fit (Burnham and Anderson 2001).

Once a best model (or models) was chosen, residuals were examined to see if they were normally distributed with a mean of zero and to see if serial trends were present. We used "BestFit" software (Palisade Corporation 1997) to examine whether residuals were normally distributed. Bestfit identifies a distribution that most likely produced the data using Chi-square, Kolmogorov-Smirnov, and Anderson-Darling statistics and provides graphical output for viewing the selected distributions (Palisade Corporation 1997). A regression of residuals against year explored whether a significant linear trend (autocorrelation) was present; other types of serial patterns were evaluated visually. If residual patterns were acceptable, precision of parameters (see below) was estimated. If parameters were significantly different from zero, the model was accepted.

All 12 models had the same number of observations (n = 60), while parameters varied from 4-6. Trial versions of biomass dynamic models indicated one data point (NJA ZEBI in 2003) would account for a large portion (nearly half in some cases) of the sums of squares (SSQ) and this point was eliminated from SSQ calculations in all models.

C10.4.6 Model Output and Biological Reference Points

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

$$\mathbf{F_t = H_{t-1} / [(B_t + B_{t-1}) / 2]} \qquad (9)$$

(Ricker 1975). An equivalent instantaneous natural mortality rate associated with striped bass predation and competition was estimated as:

$$\mathbf{M_{pt} = D_{t-1} / [(B_t + B_{t-1}) / 2]}. \qquad (10)$$

Total instantaneous annual mortality of weakfish (Z_{pt}) due to due to fishing and striped bass predation equaled $F_t + M_{pt}$. This estimate did not account for other sources of natural mortality not associated with food web effects represented by striped bass. The ratio of F_t to Z_{pt} provided an indicator of "leverage" for recovery by managing the weakfish fishery and bycatch (V. Crecco, CT DEP, personal communication).

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

$$\mathbf{D_t / P_t}. \qquad (11)$$

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 and applied to fishing only models:

$$\mathbf{B_t - B_{t-1} + H_{t-1}} \qquad (12)$$

(Prager 1994; Jacobson et al. 2002). The second estimate of SP (SP_{Dt}) explicitly accounted for losses due to striped bass predation and competition:

$$\mathbf{B_t - B_{t-1} - D_{t-1} + H_{t-1}}. \quad (13)$$

This approach defined surplus production similarly to Overholtz et al. (2008). Correlation analysis was used to estimate associations of SP or SP_{Dt} with F_t and/or M_{pt} to explore the possible linkage of these parameters to changes in surplus production.

Production (recruitment to exploitable biomass + individual weight gain) was estimated by adding estimates of D_{t-1} (if necessary) to the basic production equation described by Walters and Hilborn (2005):

$$\mathbf{P_{t-1} = B_t - B_{t-1} + H_{t-1} + D_{t-1}}. \quad (14)$$

Estimates of F were compared to relative F estimates rescaled into F (Sinclair 1998; Crecco 2006) using:

$$\mathbf{F_{Rt} = (H_{t-1} * q_{mean}) / ((mean\ ZEBI_t + mean\ ZEBI_{t-1}) / 2)}; \quad (15)$$

where F_R is an F estimated by rescaling harvest in year t into ZEBI units using the geometric mean of q_t estimated from the biomass dynamic model and then dividing this product into the average of Z+2 transformed WRI, DE EBI, and NJA ZEBI in years t and t+1. This comparison allowed me to view the impact of the observation error assumption on trends in F based on indices.

Equilibrium biological reference points (EBRPs) could be estimated for relevant Schaefer (F_{msy} = r / 2 and B_{msy} = K / 2) and Gompertz models (F_{msy} = r $F_{collapse}$ = r * 2.72, B_{msy} = K / 2.72; Quinn and Deriso 1999; Crecco 2006). Nonequilibrium reference points were also developed for predator-prey models that accounted for additional losses from striped bass that were assumed to be responsible for all but F and residual natural mortality (denoted as M_1 by Overholtz et al. 2008).

Two approaches were used to estimate reference points (NBRPs) for mortality when predation-competition losses were included: total mortality at maximum sustained yield (Z_{msy}) and annual non-equilibrium F_{msy} (F_{psyt}). The former simply involved renaming equilibrium estimates of F_{msy} as Z_{msy} in models with predator-prey terms and comparing it to Z_{pt}. This estimate of total mortality at MSY would be constant across time and would serve as a benchmark for total mortality changes.

Annual F that provided MSY after accounting for predation was estimated as

$$\mathbf{F_{psyt} = F_{msy} - M_{pt}}. \quad (16)$$

In this case, the predator or predator-prey conditions related to total mortality is considered "off-limits" for management and yield can only be influenced by managing the fishery.

Amendment 4 to the Interstate Fishery Management Plan for Weakfish (ASMFC 2002) lists SSB that is 20% of an unfished stock as a maximum spawning potential (MSP) threshold and 30% as a target. Although biomass dynamic models do not provide SSB thresholds explicitly, the biology of weakfish allows for MSP proxies. We used B_t / K as a proxy for MSP to compare to the target and threshold. Minimum size criterion for inclusion as exploitable biomass ($\geq$ 250 mm) was greater than minimum size at 50% maturity (168 mm; all age 1; Nye et al. 2008). Maturity was complete by

230 mm and all age 2 fish were mature; however, age 1 weakfish produced far fewer spawns per season than ages 2 and older (weakfish are indeterminate batch spawners; Lowerre-Barbieri et al. 1998; Nye et al. 2008). Examination of 2001-2003 length at age data (J. Brust NJDEP, personal communication) indicated that age 1 weakfish mostly ranged between 130-300 mm and age 2 ranged from 200-400 mm. Inclusion of age 1 weakfish in B_t as well as annual variation in batch fecundity could introduce some bias in trying to directly interpret it as a proxy for egg deposition, but these would be problems for equilibrium-based calculations from spawner biomass per recruit or equilibrium yield models as well.

C10.4.7 Variability and Model Sensitivity

Jackknifing and bootstrapping (Efron and Tibshirani 1993) were used to describe variability of parameters estimated in the final 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; Efron and Tibshirani 1993; Prager 1994; Haddon 2001). Percentile confidence intervals of model parameters and the time-series of B_t, F_t, M_{pt} and Z_t / Z_{msy} were estimated.

First, jackknifing was conducted manually using Evolver. Then bootstrapping followed using Excel's Solver. In order for Solver to run, parameter constraints were imposed based on, but broader than, jackknife minimums and maximums.

Several approaches were used to investigate sensitivity of model parameters and estimates of Z_t, F_t, and B_t. All sensitivity analyses were deterministic. Estimates of B_t were standardized to K (B_t / K) and estimates of Z_t and F_t were standardized to Z_{msy}. These standardizations were used in an attempt to minimize likely correlations between r, K, and B_0 that would affect comparisons based on absolute values of Z_t, F_t, and B_t.

Sensitivity of model parameters to data from the beginning or ending of the time-series was tested by removing blocks of data and rerunning the model. Time blocks of years up through 1989 were removed from the beginning of the model. These time blocks at the beginning encompassed data where WRI was the only time-series until the first fishery-independent data were available. Up to three of the most recent years were removed from the time-series that began in 1981. Biomass estimated in 1981 for the 1981-2006 time-series was greater than K and we ran a version of this model with B_{1981} constrained to be less than K for comparison (Prager 1994). B_t / K estimates were compared in addition to model parameters in this last exercise.

C10.5 Results

C10.5.1 Hypothesis Testing

AIC_c scores indicated a 98% chance that the Gompertz production model with a depensatory function relating M_{pt} to the biomass of age 2+ striped bass and the ratio of menhaden to striped bass biomass was best (hereafter, Gompertz depensatory ratio model or GDR; equations 2 and 7) for describing the hypothesis given the data (Table C10.5-1). This model fit the data well and explained 90% of the variation in ZEBI (Figure C10.5-1). Remaining models will not be considered, but it is interesting to note that the second and third ranked models were also Gompertz functions with predator-prey functions. Fishing only models were poor choices for describing biomass dynamics of the data, ranking seventh and ninth out of twelve (Table C10.5-1).

The mean of residuals of the GDR model was close to zero (0.0045) and the normal distribution was either the first or second choice of the three procedures in BestFit (a triangular distribution was first in two). A linear serial trend in residuals was not indicated by regression (r^2 = 0.014, P = 0.37). There were periods of larger positive (1986-1989) and negative (1993-1995) residuals, but residuals of remaining years were mixed (Figure C10.5-2).

C10.5.2 Model Parameters

Model parameters were estimated as r = 0.48, K = 51,521 mt, c = 0.29, and B_{1981} = 82,472 mt. The EBRPs were F_{msy} or Z_{msy} = 0.48, $F_{collapse}$ = 1.31, and B_{msy} = 18,941 mt. Jackknifing and bootstrapping indicated all parameters were precisely estimated (Tables C10.5-2 and C10.5-3). Parameter values of the base run were very close to their medians, indicating minimal bias. Bootstrapping required constraints and changes to base settings of Excel after preliminary trials indicated repeated failure of Solver without them. Constraints were r, 0.40 – 0.55; K, 45,000 - 55,000 MT: B_{1981}, 65,000 - 90,000 MT; and c, 0.2 - 0.4. Iterations were set at 1,000, precision to 0.1, tolerance to 0.5%, and convergence to 0.1.

The initial biomass estimate (1981) exceeded the estimate of K. We applied the penalty function for large values of B_0 described by Prager (1994), but this did not change results substantially. We proceeded through a series of sensitivity analyses to determine a course of action. Estimates of other model parameters were influenced by the estimate of B_0. Estimates of r (and dependent reference points) declined steadily as time-blocks were removed and B_0 declined (Table C10.5-4). Estimates of K and c displayed a down and then up trajectory as years were removed. When models were initiated progressively later from 1981-1984, K declined from 51,000 mt to 46,000 mt and then rose at a quicker rate to 78,000 mt as 1985-1989 became the initial years in the time-series. Coefficient c declined steadily from 0.29 to 0.20 as 1981-1987 time-blocks were removed and then rose rapidly to 0.35 when the GDR model was initiated in 1988 or 1989. Constraining B_{1981} to be less than K lowered r and raised K substantially and resulted in a slight drop in c. The mean of the squared residuals was lowest with the unconstrained 1981-2006 data set (Table C10.5-4). The net impact of these parameter changes was judged by comparing time-series of B_t / K, Z_t / Z_{msy}, and F_t / Z_{msy}.

Sensitivity analyses indicated that B_0 / K exceeded one when the time-series was initiated in 1981 or 1982 and was close to one when initiated in 1983 or 1984 (Figure C10.5-3). Remaining treatments did not exhibit B_0 / K estimates exceeding or approaching one. Estimates of B_t / K converged for the remainder of the time-series by the late-1980s. Constraining B_{1981} to less than K resulted in convergence of B_t / K with the unrestrained GDR by 1983 (Figure C10.5-3).

Estimates of Z_t / Z_{msy} were similar in trend among time-block treatments (initial years removed) until the most recent three years of the time-series, when a bifurcation between a steady increase and a plateau occurred (Figure C10.5-4). The unconstrained, constrained, 1982+ time-series, and 1989+ time-series (additional indices present) treatments produced similar increasing trends to Z_{2006} / Z_{msy} ≈ 3.0-3.5,while remaining treatments indicated a leveling of Z_{2006} / Z_{msy} ≈ 2.0-2.5. There were mixed rankings of values of Z_t / Z_{msy} among treatments, but similar trends for 1985-2003. There was more variability among treatments of years prior to 1985 (Figure C10.5-4).

Trends in F_t / Z_{msy} were similar among time-block removal treatments (Figure C10.5-5). Ranking of treatments was mixed across the time-series; there was not any one treatment that produced a consistent higher or lower value. Time-series based on the unconstrained B_{1981} and B_{1981} constrained to be less than K converged by1983. Treatments using all years of data generally produced highest values of F_t / Z_{msy} in 2006 (0.66 for no constraint on B_{1981} / K and 0.63 for B_{1981} <

K). Over all treatments, F_t / Z_{msy} during 2006 varied between 0.45 and 0.66, with a median of 0.53 (Figure C10.5-5).

Overall, differences in B_t / K, Z_t / Z_{msy}, and F_t / Z_{msy} stabilized by 1983 when the entire time-series was used and B_{1981} was constrained. Biomass dynamics of weakfish were portrayed similarly among all initial time-block removal treatments. We chose to keep the results of the unconstrained GDR, but did not use 1981 and 1982 estimates. Prager (1994) recommended not using the first 2-4 years of production model estimates to draw inferences about biomass.

Removing up to three of the most recent years from the time-series had little impact on parameter estimates, except for B_{1981}. Estimates of r increased from 0.48 when all data were used, to 0.50 when 2006 or 2005-2006 were removed; estimates of K did not change; B_{1981} declined from 82,472 to 68,286 MT, and c fell from 0.292 to 0.291. Collectively, these changes did little to alter depiction of biomass dynamics after 1982 (illustrated by B_t / K; Figure C10.5-6).

Assigning larger mean weights to recreational discards influenced parameter estimates (Table C10.5-5), but did not discernibly alter estimates of B_t unless weights above 0.3 kg or harvest mean weights were assigned (Figure C10.5-7). We accepted the use of 0.15 kg for recreational discard weights in the WRI and fishery loss estimates. Only the run with 0.4 kg mean weight had B_t estimates that were consistently higher than other runs. Using harvest mean weight produced similar estimates of B_t as the remaining treatments during 1982-1991, but estimates were consistently higher afterwards (Figure C10.5-7). The timing of the separation of estimates based on harvest weights was similar to the timing of increasing minimum length limits beginning with Amendment 1 (ASMFC 2002); size limits would have increased recreational discarding and the impact of assumed discard weights.

Biomass was about 33,600 mt in 1983 and fell to 6,300 mt by 1990; B_t rose to 18,000 mt in 1996-1997 and then began to decline again, reaching a brief plateau at 5,000-6,000 mt during 2003-2005 (Figure C10.5-8). The pace of loss accelerated in 2006 (4,000 mt) and 2007 (3,000 mt). Landings exhibited the same general trend as B_t. Initial losses to striped bass (D_t) were very low and rose slowly to less than 1,000 mt during 1983-1993 and then began to accelerate to about 5,100 mt by 1997. Estimates of D_t fluctuated between 4,100 and 5,900 mt during 1998-2006 while B_t was dropping. Fishery-related loss estimates fell below D_t in 1999 and were less than 25% of D_t since 2004 (Figure C10.5-8). Jackknife and bootstrap estimates of B_t (Figures C10.5-9 and C10.5-10) and D_t (Figures C10.5-11 and C10.5-12) indicated little bias and estimated precision of B_t and D_t from these techniques was high.

Estimates of F_t were near Z_{msy} (0.48) during 1983-1985, then peaked quickly at 1.3 during 1988, then fell steadily to below Z_{msy} in 1992 (Figure C10.5-13). F_t remained below Z_{msy} during 1992-1997 and then rose slowly to a level slightly above Z_{msy} during 1998-2002. After 2002, F_t dropped sharply and remained at about 50-70% of Z_{msy} through 2006 (Figure C10.5-13).

Estimates of M_{pt} rose from a very low level in the mid-1980s to estimates equivalent to F_t and Z_{msy} by 1999 (Figure C10.5-13). After 1999, M_{pt} accelerated rapidly and reached 1.4 in 2006 (Figure C10.5-13).

Comparing F_t to equilibrium F_{msy} and non-equilibrium F_{psyt} may provide the best depiction of how D_t influenced biomass dynamics (Figure C10.5-14). After 1992, F_t remained below or near what would be equilibrium F_{msy} (or Z_{msy}); however, increasing losses to striped bass were continuously eroding F_{psyt} (Figure C10.5-14). This erosion was reflected in continuous estimates of negative SP_{Dt} after 1996 (Figure C10.5-15). Negative surplus production (in absolute terms) exceeded landings since 2002 and biomass since 2006. Surplus production was generally in excess of 6,000 mt during 1983-1994 (Figure C10.5-15). Surplus production was significantly associated

($P < 0.01$) with F_t and M_{pt}, but the modest positive correlation ($\rho = +0.54$) with F_t was somewhat counter-intuitive (although F should increase surplus production initially in an unfished or lightly fished stock) while the association with M_{pt} was logical, negative, and strong ($\rho = -0.90$).

Production varied between 5,000 – 9,000 mt during 1984-2006 and was highest during periods when biomass was greater than production (1984-1988 and 1992-2002; Figure 20). High biomass during the mid-1980s was not reflective of concurrent production and its deterioration was not particularly reflective of F_t, or M_{pt}. Production was close to biomass when total mortality rates were high during 1989-1991 (due to fishing) and 2003-2006 (due to predation-competition; Figure C10.5-15). Production appeared to decline when proxy MSP (B_t / K) fell below 20% (Figure C10.5-16).

Estimates of proxy MSP (B_t / K) fell continuously after 1983 from 65% of K to reach about 12% by 1990 (Figure C10.5-16). F_t / Z_{pt} indicated that this early decline would have been attributed to F (ratio was near 1). Fishing mortality reductions lead to rebuilding to approximately 35% of MSP by 1996-1997 (target level in Amendment 4 = 30%; ASMFC 2002), while F_t / Z_{pt} was falling from near 1.0 to approximately 0.5. Spawning potential continuously fell after 1997, reaching 8% MSP in 2006 and 6% MSP in 2007 as F_t / Z_{pt} declined from about 0.5 in 1997-2002 to less than 0.2 in 2006 (Figure C10.5-16).

Estimates of F_t from the GDR were usually close to estimates of relative F_t, except during 1986-1989 (Figure C10.5-17). A plot of catchability over time ($q_t = ZEBI_t / B_t$) suggests this was a period of elevated catchability in the WRI (Figure C10.5-18). However, catchability was higher for the 1989 NJA EBI as well, indicating that there may have been regional aggregation of weakfish as biomass was declining as opposed to nonrandom behavior by recreational anglers alone (Figure C10.5-18).

Jackknife and bootstrap estimates of F_t were precise, with slightly more variability in estimates during 2005-2006 (Figures C10.5-19 and C10.5-20, respectively). Estimates of F_{2006} were likely to be between 0.23 and 0.32 based on jackknife 90% percentile confidence intervals, while bootstrapping indicated F_t was likely between 0.31 and 0.40. Median and original values of F_{2006} (0.31) were very close, indicating minimal bias.

As with F_t, jackknife and bootstrap estimates of M_{pt} were precise, with slightly more variability in estimates during 2005-2006 (Figures C10.5-21 and C10.5-22). Estimates of M_{p2006} were likely to be between 1.3 and 1.7 based on jackknife 90% percentile confidence intervals, while bootstrapping indicated M_{p2006} was likely between 1.0 and 1.4. Median estimates of M_{p2006} from both techniques and original values (1.41) were the same.

Estimated loss of weakfish biomass per biomass of striped bass (D_t / P_t) was very low (0.004 – 0.005 mt / mt) during 1983-1987 when the menhaden to striped bass biomass ratio (R_t) was at its zenith (70-80 mt / mt; Figure C10.5-23). A rapid drop in R_t to 20-30 during 1988-1993 produced a rise in estimated D_t / P_t to 0.01-0.02. By 1997, R_t became asymptotically low at less than 10, while modeled estimates of D_t / P_t became asymptotically high between 0.04 and 0.06 (Figure 28). These changes in R_t produced two striped bass functional response regimes (Figure C10.5-24). During 1983-1993, while weakfish biomass (B_t) fell from 34,000 to 6,300 mt, D_t / P_t was low – between 0.004 and 0.01. A rapid transition followed and during 1997-2006, D_t / P_t increased to 0.04-0.06 as B_t fell from 18,000 to 4,000 mt (Figure C10.5-24).

C10.6 Verification of Predation-Competition Loss Estimation

C10.6.1 Long-term Evidence of the Influence of the Menhaden to Striped Bass Ratio on Weakfish

A long-term view (1959-2006) of the influence of the menhaden to striped bass ratio on weakfish was developed from an index-based menhaden to bass ratio, Atlantic Coast weakfish commercial harvest, and a Delaware Bay trawl survey length-frequency index (DE PSD Q+; Appendix C-2). Chesapeake Bay menhaden to bass index ratios (CR_t) for 1959-2006 were developed from presence-absence of age 0 menhaden in the MD seine survey (E. Durell, MD DNR, personal communication; see http://www.dnr.state.md.us/fisheries/juvindex/index.html for survey description) and striped bass egg presence-absence indices in MD spawning areas (an index of mature striped bass female biomass; Uphoff 1997). Presence-absence of menhaden in the MD survey was significantly related to estimates of coastal menhaden abundance (all ages) two years later from the ASMFC assessment (linear regression, 1959-2005, $r^2 = 0.58$, $P < 0.0001$; J. Uphoff, MD DNR, unpublished analysis). As with Skud (1982), my basic premise was that a truly competitive relationship between striped bass and weakfish would transcend deficiencies of weakfish landings or indices. A strong correlation existed between weakfish landings and the fishery-independent DE PSD Q+ index ($\rho = 0.94$), indicating both reflected similar processes.

Trends in both sets of menhaden to striped bass biomass ratios (R_t and CR_t) tracked each other closely, falling from their highest levels in the early 1980s to asymptotic lows in the mid-1990s (Figure C10.6-1). Chesapeake Bay menhaden to bass index ratios (CR_t) indicated low attack success in the 1960s and a sudden rise in 1971. Higher ratios were maintained until the early 1990s (Figure C10.6-1). A linear regression of 1982-2005 estimates (strictly based on assessment results as opposed to 1981 and 2006 predictions) was significant ($r^2 = 0.80$, $P < 0.001$) and CR_t could be converted to R_t with the equation:

$$R_t = (67.3 * CR_t) - 5.8.$$

During 1959-2006, weakfish commercial harvest and the DE PSD Q+ index closely followed CR_t (Figure C10.6-2) and correlations with the CR_t were strong ($\rho \approx 0.82$ in both cases). These associations indicated that this ratio has been important in dynamics of weakfish beyond the period covered by the GDR.

C10.6.2 Short-term Evidence of the Influence of Striped Bass on Weakfish Condition

Competition between striped bass and weakfish would be expected to influence condition of weakfish. An estimate of mean weight at a standard length provides a measure of body condition (Deriso et al. 2008).

Individual lengths and weights were available from the NEFSC fall survey annually for 1992-2006 (Uphoff 2005; G. Shepherd, NMFS, personal communication). These samples are collected by standardized trawl in a stratified random design during a restricted period of time (NEFSC 2007) and should minimize gear, location, and seasonal variation of weight. Predicted weight-at-length was derived from annual regressions of $\log_e$-transformed weight (kg) on $\log_e$-transformed length (cm) that were applied to NJ trawl survey length-frequencies to derive exploitable biomass (*described previously in Trawl Survey Indices of Exploitable Biomass*). Generally, weakfish < 400 mm were well represented in the length-weight samples, although larger weakfish were present in the length-weight samples. Fits of the $\log_e$-transformed length and weight data were very good ($r^2 \approx 0.99$, except 2003 $r^2 = 0.97$). The length used to designate quality

weakfish in PSD analysis (Appendix C-2), 340 mm, was used as a standard length and predicted weight at this length was used as a body condition indicator (hereafter, Q_{wt}; Table C10.6-1). Regression analysis tested whether Q_{wt} had undergone a significant linear decline since 1992, and whether it was negatively related to striped bass biomass or the ratio of menhaden to striped bass biomass (R_t). Inspection of residuals of these regressions indicated that 2003 Q_{wt} was a potential outlier and a separate set of analyses was run with this year removed.

Regression analysis (all years and with 2003 removed; r^2 range = 0.41-0.61 and P range = 0.0009-0.01) indicated that Q_{wt} had undergone a significant decline during 1992-2006, and that the decline in Q_{wt} was significantly related with both striped bass biomass and the ratio of menhaden to striped bass biomass (Table C10.6-2). Predicted body mass at 340 mm was negatively related to striped bass biomass and positively related to the ratio of menhaden to striped bass biomass. These relationships were consistent with expected effects of intense competition between weakfish and striped bass and with parameterization of the GDR model.

C10.6.3 Comparison of Model and Field Estimates of Weakfish Consumption by Striped Bass

Unlike harvest, we do not have a natural mortality "body count" to unequivocally estimate D_t or M_{pt}. However, estimates of D_t / P_t can be judged by comparing them to estimates of maximum consumption (C_{max}) and annual consumption of food (C_t; weight of all prey, fish and invertebrates, consumed) per weight of striped bass on an annual basis from bioenergetic models of striped bass. These estimates have been made for Chesapeake Bay during 1955-1959 (Griffin 2001), 1990-1992 (Hartman and Brandt 1995a; 1995b), and 1998-2001 (Overton 2003). Bioenergetics models of Griffin (2001) and Overton (2003) were the same as Hartman and Brandt (1995a; 1995b), but used diet and temperature data for their respective time periods. We confined comparisons of C_t to striped bass 2 years-old and older on day 365 (full year's consumption); these studies estimated C_t for ages 2-6. These comparisons allow for an estimate of the scale of individual consumption of weakfish by striped bass and may answer the questions "Are weakfish a major or minor item?" or "Are the estimates of D_t / P_t excessively high (in excess of C_{max} or C_t)?"

Overton et al. (2008) provided a time-series of striped bass diet trends to compare with D_t / P_t. This study examined 1,154 striped bass during winter in the Atlantic Ocean off the VA and NC coasts. These striped bass were collected during 1994-1996, 2000, 2002-2003, and 2005-2007 and their diets were dominated by Atlantic menhaden and bay anchovy (Overton et al. 2008). Percent of striped bass stomachs with food (SS_t) was relevant to GDR estimates of D_t / P_t. In the GDR, decreased feeding success by striped bass (indicated independently by SS_t) should lead to increased search times and more encounters with weakfish since the diet is dominated by other species and includes a large contribution by menhaden, i.e., D_t / P_t would be negatively related to SS_t and influenced by R_t. Linear regression with categorical variables and a slope shift coefficient was used to analyze these relationships (Freund and Littel 2000). The annual ratio of menhaden biomass to striped bass biomass (R_t) was converted to a categorical variable (MR_t), 0 for ratios 10 and above (1994-1996) and 1 for ratios below 10 (remaining years). Categories were used to minimize confounding with D_t / P_t estimates derived from the model that used these ratios to estimate D_t. A slope shift coefficient was estimated because the time trend in the functional response plot indicate two periods with different slopes that would have to be accounted for within the span of Overton et al. (2008) data (see Figure C10.5-24). The slope shift coefficient was estimated by including a variable equal to the product of SS_t, multiplied by MR_t (Freund and Littel 2000). These variables were tested for inclusion by multiple regression and those significant at $P < 0.05$ were retained. An inverse transformation of the dependent variable, D_t / P_t, was used to place the small ratios on a

larger scale.

Estimates of D_t / P_t were quite low when compared to estimates of weight of annual food consumption (fish and invertebrates) per weight of striped bass (C_t) in three Chesapeake Bay bioenergetics studies (Table C10.6-3; Hartman and Brandt 1995; Griffin 2001; Griffin and Margraf 2003; Overton 2003). Estimates of C_t ranged from 4.1 to 7.9, with a median of 5.5, while D_t / P_t fell between 0.004 and 0.06. Estimates of C_t were about half to a third of C_{max} (Hartman and Brandt 1995). Weakfish were a minor diet item subject to a very low level of consumption by striped bass (if all losses were strictly defined as consumption). This low level of consumption was applied over an increasingly large striped bass biomass that was engaging in higher search times (reflected by low R_t) for their main prey after 1996. Encounters with weakfish may have been facilitated by diet, spatial, and temporal overlap with striped bass, resulting in disproportionately high D_t as weakfish biomass declined. There was no feedback between D_t and P_t since weakfish were such a minor diet item.

Estimates of D_t / P_t were strongly related to estimates of feeding success of striped bass in coastal VA and NC during winter, indicated by SS_t (as a percentage). The relationship was described by linear regression with a slope shift coefficient:

$$D_t / P_t = 1 / [(0.204*(SS_t)) - (0.16*(MR_t* SS_t)) + 15.35]; (R^2 = 0.89, P = 0.0013);$$

where MR_t is the category for menhaden to striped bass biomass ratio (Tables C10.6-4 and C10.6-5).

C10.7 Could Bycatch be Responsible for the Recent Decline?

Weakfish Peer Review and Data Poor Workshop panels in 2006 and 2008, respectively, suggested that 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. We fit Gompertz biomass dynamic models to the three ZEBI's, harvest, estimated discards, and an additional loss term (L_F) mimicking various commercial bycatch scenarios (L_F increasing as linear, quadratic, or exponential functions of time, a constant multiple of bycatch estimates, or as a constant additional weight). Additional losses were imposed after 1995 to reflect regulatory discards. Results were contrasted with the GDR and a Gompertz model using estimated fishery losses without additions. We used AIC_c to calculate Akaike weights that approximated the probability that model i constituted the best model of the hypothesis given the data.

Akaike weights (Table C10.7-1) indicated that there was approximately a 91% chance that the GDR provided the best model of the dynamics of the data, while there was a 6% chance when L_F was constant (an additional 3,495 MT since 1996), and a 3% chance when L_F increased as a quadratic function of time (increasing from an 2,273 MT in 1996 to 5,054 MT in 2006). These two bycatch scenarios invoke about the same biomass of L_F as D_t estimated by GDR (Figure C10.7-1). In general, bycatch estimates started at about 2-3 times the estimates of 1996 and ended up 15 to 20 times higher by 2006 (Figure C10.7-1). Each of these three scenarios provided an excellent fit to the EBI ($R^2 \approx 0.90$). Remaining models had near zero odds of explaining the data compared to these three.

C10.8 Discussion

Biomass dynamic modeling indicated the weakfish stock was not overfished in 2006 based on equilibrium F_{msy}, but was subject to high natural mortality that eroded the safe level of fishing. The

B_t / K proxy for MSP in 2007 (6%) was far below the 20% MSP threshold in Amendment 4, while high negative values of F_{psyt} (non-equilibrium F_{msy}) and surplus production indicated that complete (and unlikely) elimination of harvest and bycatch would not be sufficient to end the decline. Production persisted at a modest level in recent years, although it was falling gradually. Depensatory mortality, driven by high striped bass biomass and a low ratio of Atlantic menhaden to striped bass (an indicator of low feeding success on striped bass' main prey), appeared the most likely explanation for increasing natural mortality that undermined recovery given the data, hypotheses, and models developed. Fishing played a secondary role in recent biomass dynamics. Striped bass predation-competition dominated weakfish biomass dynamics after overharvesting had been controlled in the early 1990s.

Regression analyses and biomass dynamic models indicated high potential for striped bass, menhaden, and weakfish linkage. Spatial, temporal, and diet overlaps were sufficient for interactions to occur. Modeling indicated high mortality of weakfish from striped bass was derived from very low loss of weakfish per striped bass applied over a large striped bass biomass. Predation-competition from striped bass has increasingly eroded weakfish surplus production and F_{msy} reference points and there seems little chance of restoring weakfish by manipulating its fisheries alone because F has become a low fraction of total mortality (≈20% by 2006). At this time, leverage for manipulating weakfish may mostly reside in the menhaden to bass ratio.

Projections of future weakfish biomass under these conditions indicated little ability to generate recovery by manipulating the fishery and suggest extirpation in a time frame shorter than a moratorium could be implemented. These predictions should be greatly qualified. We believe these predictions indicate (1) a need for broader multispecies or ecosystem-based management perspective on weakfish and (2) managing the weakfish fishery alone is unlikely to have much impact unless conditions contributing to high M_{pt} (very high striped bass biomass and low menhaden-bass ratio) lessen. The possibility of extirpation is far more uncertain than indicated by projections because of high uncertainty of model specifications. Forecasts, starting with B_{2007}, have gone beyond weakfish biomass limits represented by data (up through 2006), which the model has fit well, and continued to represent predation-competition as unchanged. This representation of a constant predation-competition function in GDR projections becomes highly uncertain as weakfish densities drop and implies lack of refuge for weakfish, as well as uniform (and suicidal) linkage of weakfish migratory and feeding behavior with striped bass. Munyandorero's (2006) summarization of weakfish population structure literature would suggest the South Atlantic region could serve as a refuge since sub-populations below NC are not subject to commercial fisheries or predation from the migratory striped bass that roam from ME to NC. The Atlantic Coast weakfish stock appears to be a mosaic of regional sub-populations with enough exchange for genetic uniformity (Munyandorero 2006).

It can be difficult to predict the effects of fishing or culling policies from fairly simple representations of predation processes (Bax 1998; Yodzis 2001; Hollowed et al. 2000a). Hollowed et al. (2000b) cited Beverton's argument that only first order symmetric interactions in which a predator and fishery exploit the same prey species are predictable. First order asymmetric interactions, in which the predator (striped bass) and fishery (weakfish) exploit different interacting prey species (striped bass mainly exploit menhaden in this case and their exploitation of weakfish is a function of their failure to do this successfully) are unpredictable (Hollowed et al. 2000a; Yodzis 2001). Further complicating projections of weakfish biomass dynamics was the reliance on single species, constant M-based biomass estimates for striped bass and Atlantic menhaden biomass as the principle drivers of M_{pt}. There is evidence for both species that their natural mortality has changed systematically (tag-based estimates for striped bass and multispecies modeling for menhaden; Jiang

et al. 2007; ASMFC Multispecies Technical Committee 2008; Gauthier et al. 2008; NEFSC 2008) and mortality could be linked between these species (Uphoff 2003). Systematic and linked changes in M would alter estimates of striped bass and menhaden biomass which drive D_t and M_{pt} in the GDR.

Atlantic menhaden is a buffer species that absorbs predatory pressure from other species when they are sufficiently abundant (ASMFC 2004). Striped bass actively select for Atlantic menhaden, but will feed on other species when menhaden are not sufficiently abundant (Overton 2003; ASMFC 2004; Ruderhausen et al. 2005). Striped bass appear capable of limiting their prey populations along the Atlantic Coast at high biomass that existed during the mid-1990s into 2006 (Hartman 2003; Uphoff 2003; Heimbuch 2008). The GDR indicated that as menhaden have become less abundant and striped bass more-so, striped bass searching has lead to increasing encounters with vulnerable-sized weakfish searching for smaller prey-fish (anchovies and age 0 menhaden) that are also found in diets of the largest striped bass (Walter and Austin 2003; Overton et al. 2008).

Hypothesis testing indicated that depensatory mortality associated with high striped bass and low menhaden biomass has reduced productivity of weakfish along the Atlantic coast. Depensation may not be common, but it is not so rare a possibility that it should be considered implausible (Walters and Kitchell 2001). Movement from high to low equilibrium states can be induced by depensatory mortality (Collie and Spencer 1993). 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). 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 effect of striped bass on weakfish in the GDR was minimal when R_t was above 20, but it intensified (more weakfish died) as R_t fell lower and their biomass dynamics were dominated by striped bass. Switching behavior by large predators may cause unexpected, sequential depletion in prey (Walters et al. 2005). In the early to mid-1990s, important forage species (Atlantic menhaden, spot, and bay anchovy) dropped to low levels (Uphoff 2006a; ASMFC Multispecies Technical Committee 2008) and depletion of weakfish followed in the late 1990s.

Hartman and Brandt (1995) found that striped bass in Chesapeake Bay during 1990-1992 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.

Estimates of D_t in the GDR represented combined effects of direct striped bass predation, as well as cannibalism and starvation induced by intense competition (Yodzis 1994). Estimates of D_t may have included covarying species effects as well, but striped bass predation-competition should predominate.

Weakfish are cannibalistic; weakfish juveniles were specifically mentioned in weakfish diet studies reviewed by Mercer (1985). They comprised about 20% of weakfish diet by weight in Delaware Bay during 1985 (Taylor 1987). Atlantic coast biomass estimates indicated striped bass were at low abundance and menhaden were at high abundance during 1985 ($R_t \approx 80$), but menhaden were not abundant in DE and NJ surveys (Uphoff 2006a). Weakfish were not specifically mentioned in diets of weakfish sampled in Chesapeake Bay during 1990-1992 ($R_t \approx 20$; Hartman and Brandt

1995), but other weakfish comprised 8% of ages 0-5 weakfish diets in Chesapeake Bay during 2002-2003 ($R_t \approx 5$; Bonzek et al. 2004).

Diminished size quality (indicated by PSD Q+: Appendix C-2) and condition (indicated by Q_{wt}) were both related to the biomass of striped bass and Atlantic menhaden (Uphoff 2006a; see section **C10.6-2**). Substantial decreases in mean weight at age during 1981-2003 in the VPA catch-at-age matrix also occurred (Uphoff 2006a); however, interpretation is confounded by the potential biases as aging shifted from scales to otoliths at the same time. Natural selection balances predation risk and the need to obtain food for growth and fear of being eaten may reduce foraging and growth (Walters and Juanes 1993; Walters and Martell 2004; Trussell et al. 2006) and even lead to starvation (Pine et al 2009). Mortality due to starvation is a size-dependent process and represents an alternative (albeit final) response to reduced growth and stunting during food shortages (Ney 1990; Persson and Brönmark 2002). Starvation may be more common than generally perceived (Ney 1990) and it might be expected once the bioenergetic limits of reduced growth were breached.

Comparisons of trends in biomass and production from biomass dynamic modeling and length quality (Appendix C-2) suggested large scale shifts in biomass and production relationships from the early-to-mid 1980s to current conditions. High biomass during the mid-1980s was not reflective of concurrent production, and its deterioration during 1983-1985 was not reflective of F_t ($\approx F_{msy}$) or M_{pt} (near 0). The DE PSD Quality+ index (Appendix C-2, Figure 1) indicates that larger weakfish were more common during this earlier period. The grand mean of standardized YOY indices has increased since the 1980s (see Figure 32 in Kahn et al. 2006) as size quality has diminished. This suggests a trade-off between high growth–low recruitment and low growth –high recruitment that coincided with a drop in R_t after 1987 (see Figure 30).

Models used in this investigation were of modest complexity because of the basic limitations of the observational data. Simplified procedures allowed for the testing of overfishing and predation-competition hypotheses. The strategy of including striped bass predation-competition was chosen in light of realization that dynamics of predation, competition, environmental regime shifts, and habitat alteration or deterioration could take over once overharvesting had been controlled (Link 2002). Stationarity of ecological conditions or constancy of M cannot always be an acceptable default assumption (Sainsbury 1998) and simply presuming that ceasing exploitation on an overfished stock will result in recovery ignores the uncertainty imposed by ecological systems (Link 2002). Evaluating effects of fishing on ecosystems often has to be pursued by testing hypothetical conceptual models and multispecies models are essential (Hollowed et al. 2000a). An important element of these models is that they must include predators having the greatest impact on commercial and recreational species (Hollowed et al. 2000a).

The GDR was a step in an inferential process described by Platt (1964), Burnham and Anderson (2001); and Deriso et al. (2008). Hypothesis testing in subsequent years may yield other or additional environmental or ecological factors or functional descriptions. The "right" functional form of predator-prey models can be difficult to define (Yodzis 1994), but typical parameters in age structured single species models (constant M, stock-recruitment steepness, selectivity) are sometimes confounded and difficult to estimate (Magnusson and Hilborn 2007). The exact form of the predation-competition term may not crucial. Management benchmarks from the "top" three models (GDR, bass only, and Type II; all Gompertz functions) were very close (F_{msy} range = 0.48-0.51 and B_{msy} range = 19,000-20,000 mt), as were estimates of current status (B_t / K in 2006-2007 range = 6-9%). Management advice would not vary appreciably since Z_{2006} / Z_{msy} ranged from 2.4 to 3.6 while F_{2006} / Z_{p2006} was between 0.18 and 0.20, i.e, total mortality was very high and mostly due to striped bass predation-competition.

Overfishing was the cause of the decline of weakfish in the mid-to-late 1980s, and management was successful in preventing it from reoccurring, at least by standard equilibrium benchmarks. The failure of recovery since the late 1990s cannot be attributed to overfishing unless bycatch and under-reported catches were much greater than those estimated, growing from about 3-4 times the estimates in 1996 to 15-20 times by 2006. If results of hypothetical bycatch scenarios are taken at face value, then weakfish regulations created a massive boost in discards and represent a colossal management failure. Implementation of further conservation measures short of a coast-wide moratorium on many Atlantic coast fisheries would not minimize this nominal discard problem. There is no evidence available thus far of an Atlantic coast fishery capable of generating additional unreported weakfish discards of this magnitude (Uphoff 2006b).

Equilibrium MSY limit reference points, consistent with the precautionary approach (Restrepo et al. 1998) and estimated from the GDR, indicated that F had been near or below F_{msy} for the last 15 years, but B_t fell well below 20% of K that approximated the MSP limit in Amendment 4 (ASMFC 2002). Biomass continued falling even as F_t fell to about 50-60% of F_{msy} after 2003. Fishing mortality rates in this range should have been associated with stock expansion or stability under constant M conditions and these equilibrium-based biological reference points had little utility when M was rising rapidly. Biological reference points can be sensitive with species subject to predation and cannibalism and may represent moving targets (Collie and Gislason 2001). Multispecies simulation models indicated it was impossible to derive a single fixed value for F_{msy}, but stock-recruitment and yield curves may be distorted if multispecies effects are disregarded and predictions of recovery can be much too optimistic (Hollowed et al. 2000a). Moustahfid et al. (2009) added predation loss estimates as a "fleet" in an age structured assessment of Atlantic mackerel and noted that their inclusion altered model outputs and biological reference points. Non-equilibrium F_{psyt} was informative; however, it was not developed until predation-competition effects were well established and is likely to be retrospective in practice. Once understood, non-equilibrium F_{psyt} does provide short-term understanding of underlying productivity and its relationship to fishing. In the case of the GDR, by 2000 rising M_{pt} ate away all F that represented a safe level of fishing.

The biomass dynamic approach applied here and by others (Collie and Spencer 1993; Spencer and Collie 1995; Spencer and Collie 1997a; Collie and DeLong 1999, Crecco 2006; Uphoff 2006a) is fundamentally different from biomass or age structured multispecies models employed by Hollowed et al. (2000b), Overholtz et al. (2008), and Moustahfid et al. (2009), and ASMFC (NEFSC 2006). The biomass dynamic approach applied here generates estimates of predation-competition losses through their predation-competition terms, predator-competitor biomass or trends (Crecco (2006) used relative abundance), and fits to observed indices or estimates of "victim" biomass. Estimates of individual or stock level consumption were not applied; however, estimates individual of consumption by striped bass were needed for verifying results or they could define constraints on generated estimates. Other methods require predator abundance and annual consumption rates or estimates of consumption by age (Hollowed et al. 2000b; ASMFC 2003; Overholtz et al. 2008). Size, and prey type preference parameters, evacuation rates, biomass of "other food" not explicitly modeled, and annual or semi-annual diets may also be needed (ASMFC 2003; Moustahfid et al. 2009).

It is not possible to look at all predation processes in an ecosystem (Bax 1998; Sainsbury 1998; Walters and Martell 2004) and we concentrated on a limited representation of the food web. The GDR is a minimal realistic model that represents first-order indirect (triangular) interactions (Bax 1998). Triangular food webs have been proposed as a mechanism promoting stability in marine ecosystems (Bax 1998). The virtue of a minimum-realistic approach is tractability in analyzing and

parameterizing (Yodzis 2001). Its shortcoming is that there is no *a priori* way to know how much complexity is the minimum and that can only be judged by comparing results to more complex ecosystem models. A minimum-realistic approach was used to develop an understanding of how culling seals might affect the hake (*Merluccius paradoxis* and *Merluccius capensis*) fishery in the Benguila ecosystem (Punt and Butterworth 1995; Yodzis 2001).

The GDR supports the existence of a dominance hierarchy, as defined by Skud (1982), between striped bass (dominant) and weakfish (subordinate). A dominant species is defined as the more abundant of two species that interact and whose densities are maintained at distinctly different levels (Skud 1982). Ignoring dominance hierarchy can lead to erroneous interpretation of factors contributing to changes in subordinate species abundance. Weakfish landings (1929-2006; Figure C10.8-1), GDR biomass (Figure C10.8-1), and DE PSD Q+ (see Figure 1 in Appendix C-2) maintained distinct high or low levels related to striped bass and R_t. Changes in abundance of dominant species are positively correlated with environmental factors that improve survival, while abundance of subordinate species depends on density of the dominant species (Skud 1982). Most variation in year-class strength of striped bass in Chesapeake Bay (the largest spawning area on the Atlantic Coast) can be explained by freshwater flow and temperature during March through May (Rutherford and Houde 1995; North et al. 2005) although recent high biomass and trophic demand also reflect imposition of conservative management measures (Richards and Rago 1999; Uphoff 2003). Variation in weakfish abundance is strongly related to striped bass and its success in feeding on menhaden (indexed through R_t). An assumption of constant M ignores dominance hierarchy and attributes all changes in weakfish biomass to fishing, which would grossly overestimate the potential for recovery by managing the weakfish fishery alone.

More complex fisheries ecosystem models of other species complexes in other geographic regions (Ecopath with Ecosim or EWE) have highlighted processes similar to those displayed by the GDR. Large dominant fishes may be successful due partly to "cultivation effects", where adults crop down forage species that are potential competitors of their own juveniles (striped bass cropping weakfish in this case; Walters and Kitchell 2001). Dominance is a result of not only being able to acquire resources but also by insuring the best possible trophic conditions for the dominant species' young. This hierarchy leads to the risk of persistent depensatory effects (low juvenile survival) that develop with some time lag following periods of adult stock depletion (Walters and Kitchell 2001). EWE modeling of multiple systems has indicated that trophic interactions in combination with widespread fishing at F_{msy} may cause considerable change in community structure and MSY than those predicted from single-species assessments (Walters et al. 2005). The most critical predictions of change in juvenile mortality rates from EWE simulations of the Gulf of Mexico ecosystem arose when abundant predators caused high mortality of prey without those rates being represented as high proportions of predator diets (Walters et al. 2008).

Experience with Northwest Atlantic Cod *Gadus morhua* provides a case history to consider when contemplating weakfish biomass dynamics presented here. Excessive fishing mortality was the predominant factor in depletion of these stocks of cod and led to their collapse by the early 1990s (Shelton et al. 2006). Despite imposition of severe catch restrictions for over a decade, stocks largely failed to recover at predicted rates due to declines in productivity. Factors contributing to this decline in productivity (in rank order) were increased natural mortality, decreased body growth, reduced recruitment was implicated in a few cases, and continued fishing and bycatch were important (Shelton et al. 2006). An increase in M is now routinely incorporated into several cod stock assessments (Chouinard et al. 2005). Changes in estimated M in southern Gulf of St. Lawrence cod matched fluctuations of in grey seal abundance, although some inconsistencies in seal diet data

conflicted with this hypothesis (Chouinard et al. 2005). Productivity of this stock of cod has declined to the point where the population is no longer viable even in the absence of fishing (Swain and Chouinard 2008).

C11.0 Review evidence for constant or recent systematic changes in natural mortality, productivity, and/or unreported removals. (TOR #7)

C11.1 Introduction

A period of stock rebuilding appears to have occurred during the early 1990s, as evidenced by expansion of the population age structure, increases in commercial and recreational harvest and CPUE, and positive trends in abundance and biomass in subsequent stock assessments. However, during development of the 2006 weakfish stock assessment (ASMFC 2006a, Part A), the WTC was confronted with anomalous patterns in weakfish abundance estimates and modeling results. Preliminary runs of age structured and simple biomass models indicated that the stock had rebuilt to record levels. In contrast, observed size and age structure had diminished, CPUE and harvest were down, and fishermen were complaining about lack of fish. Management measures had not changed, and there was no evidence of increased harvest or discard rates. Discrepancies in trends of fishing mortality estimated by VPA and relative F analysis provided further evidence that the modeling results were suspect. The WTC searched for possible causes of model uncertainty and the unexpected reversal in stock growth. Additional modeling exercises, including biomass modeling with environmental and ecological covariates, suggested that the anomalous pattern was due to an incorrect assumption of constant natural mortality, and that stock declines were a result of increased interaction with principal predator/competitor species (ASMFC 2006a, Parts A and C).

Results of the 2006 stock assessment were vetted through the ASMFC external peer review process (ASMFC 2006a, Part B). The peer review panel did not accept the WTC's evaluation of the stock, suggesting that the age structured modeling could be improved by addressing concerns with the input data. These issues were addressed as well as possible by the WTC prior to and during the development of the 2009 stock assessment. Regardless, preliminary age structured and simple biomass modeling attempts still produce unrealistic results. As such, biomass modeling with ecological and environmental covariates was again pursued, with updates and improvements made over the previous (2006) methods.

An interim review of the current stock assessment as a work in progress was conducted during the 2008 DPSW in Woods Hole, MA. Modeling results presented during the DPSW indicate that fishing mortality is a minor component of total mortality, suggesting trophic or environmental forces as major influences on weakfish stock dynamics. The reviewers had similar comments as the 2006 panel regarding the advanced modeling efforts, citing concerns with input data and lack of empirical evidence to support the predation/competition hypotheses (Miller et al 2009). Although the panel appears to agree that weakfish populations have decreased, they contend that any number of sources could have been the driving force of the decline, such as environmental factors or unreported/underestimated removals. The following is a review of a suite of analyses searching for evidence of declines in productivity due to non-fishing sources.

C11.2 Review of previous findings

C11.2.1 Relative F

By the early 1990s, harvest (total removals) had fallen to less than half the levels seen during most of the previous decade (Figure C11.2-1). As a result (presumably), biomass began a period of rebuilding around 1993. During the mid to late 1990s, harvest was increasing slightly, but with the increase in stock size, F rates had fallen to all time low levels between 1995 and 1997. Regardless, a dramatic drop in biomass was observed between 1997 and 1998, and the decline has continued to recent years. Harvest levels remained relatively stable and were similar to those that had presumably led to stock increases earlier in the decade, but now were causing a rapid stock decline. As biomass declined, predicted trends in fishing mortality from age structured modeling (ADAPT VPA) indicated a continuous, almost exponential increase in fishing mortality between 1995 and 2002, regardless of tuning indices used. (Trends after 2002 are unreliable due to retrospective patterns.) The WTC expressed concern that trends in F from the VPA might be influenced by the assumption of constant natural mortality, M. Relative F analysis was considered as an alternative method to calculate trends in fishing mortality. Relative F evaluates the stock as a whole, and is not influenced by changes in age structure. Some of the benefits of relative F are that it is simple and intuitive, easy to calculate, and not constrained by assumptions about natural mortality. The following is a summary of the relative F analysis presented in section **C8.0** of this report.

Relative F is calculated as

$$F_{rel} = \frac{H_t}{0.5*(I_t + I_{t+1})}$$

where H_t is total harvest in year t, and I are index values of abundance in year t and t+1. Total harvest biomass was estimated as the sum of commercial harvest from state and NMFS reporting programs, commercial discards using the methods of de Silva (2004), recreational harvest (A+B1) from MRFSS, and 10% of total recreational discards (from MRFSS) multiplied by the MRFSS estimated average weight of discarded fish of 0.15 kg. Three surveys – MRFSS 1+ CPUE, New Jersey Ocean Trawl Survey positive tows, and Delaware DFW 30-foot Delaware Bay Trawl Survey – were standardized to a common scale and averaged to develop a single index of weakfish abundance. Additional information on these data and their sources are presented in sections **C5.0** (harvest) and **C6.0** (indices).

In order to directly compare relative F estimates with estimates derived from the VPA, the relative F time series was rescaled using a span of years during the converged portion of the VPA which is robust to the number and source of tuning indices used. This method assumes relative stability of M, which is one of the main concerns of the WTC regarding ADAPT; however, the assumption is only applied for a short time period (5 years) which is much more realistic than M remaining constant over decades.

To rescale the relative F vector, the ratio between F_{VPA} (biomass weighted 1+) and F_{rel} was found for the years 1982-1985. The average ratio for these years was computed and applied to F_{rel} for all years in the time series.

Comparison of trends in rescaled F_{rel} and F_{VPA} (biomass weighted ages 1+) show similar patterns between 1982 and 1998 (Figure C11.2-2). Fishing mortality was relatively high during the

1980s, generally ranging from 0.7 to 1.3. In the early 1990s, both trends begin a gradual decline to time series lows of 0.23 (F_{VPA}) and 0.35 (F_{rel}) in 1995 before gradually increasing to between 0.5 and 0.6 by 1999. Beginning in 1999, the trends differ greatly. F_{rel} exhibits relative stability, generally ranging between 0.4 and 0.6 until the end of the time series. In contrast, F_{VPA} continues to increase rapidly, reaching a time series high of 1.57 in 2003. F_{VPA} declines after 2003; however a strong retrospective pattern (Figure C7.3-2) suggests F_{VPA} estimates in recent years are greatly underestimated.

Harvest and abundance data used for the VPA and relative F analyses are essentially identical. Trajectories of biomass harvested and biomass indices used for relative F are strongly correlated with the corresponding age aggregated numbers based inputs for the VPA. Identical input data and divergent results suggest violation of model assumptions or mis-specification of one of the models. An underlying assumption of the ADAPT model is that catch at age is known without error. Section **C5.0** of this document identifies several sources of uncertainty with the overall catch at age, which likely lead to this assumption being violated through inaccurate harvest information and/or poorly specified age distribution. Error in the age structure would not be expected to produce such divergent trends in fishing mortality from the two methods. Inaccurate harvest information could produce the observed discrepancy, but only if there was a systematic increase in the amount of unrecorded landings. Several recent peer reviews (ASMFC 2006a, Part B; Miller et al 2009) suggest unreported landings could be contributing to the anomalous patterns. In an attempt to quantify the amount of unrecorded removals required to mimic trends in total mortality observed in the VPA, two alternative applications of the "forage model" were run using estimates of unrecorded removals that were constant or increasing over time (see section **C10.0** of this report). A constant level of approximately 3,500 MT of additional removals since 1995, or an increase from 2,300 to 5,000 MT additional removals between 1995 and 2007 both resulted in strong model fits ($R^2 \sim 0.90$; see Figure C10.7-1); however, neither performed as well as the "best" model fit which incorporated menhaden and striped bass abundance. In addition, the implications of these unrecorded removal scenarios are drastic. Total recorded removals in 2007 were less than 1,000 MT while estimated unrecorded removals were three to five times greater, suggesting our understanding of fleet dynamics is minimal and reporting requirements are ineffective. Also, the 2006 and DPSW review panels suggested the increases could be due to additional discarding. Such large increases in discarding rates following implementation of management measures imply a complete failure of management to constrain removals.

Another possible cause for the discrepancy in results between the VPA and relative F could be misspecification of input data. Without direct evidence to the contrary, natural mortality is often considered constant throughout time. This simplistic assumption may hold over short time periods, but is likely unrealistic over decades. Regardless, the baseline VPA model runs investigated for this and all previous stock assessments assumed constant natural mortality in weakfish of M = 0.25. ADAPT estimation procedures incorporate the input natural mortality rate, and model results are therefore influenced by the rate selected. Incorrect assumptions in the input M matrix would lead to inaccurate estimates of population parameters by the ADAPT model. In contrast, relative F calculations are independent of any assumptions regarding natural mortality rate. Discrepancies in F trends calculated using the two methods may indicate invalid assumptions regarding M.

C11.2.2 Biomass modeling

Due to concerns over the uncertainty in recent years' estimates from age structured modeling (*i.e.* retrospective patterns), as well as concerns regarding the assumptions upon which the ADAPT

runs were constructed (in particular, constant M and error free CAA), the WTC investigated biomass dynamic modeling. Methods are described in detail in sections **C9.0** and **C10.0**, and are summarized below.

Two similar but alternative models were developed to investigate both simple and extended biomass models. Three indices of weakfish abundance (two fishery independent and one fishery dependent) were standardized and combined into a single composite index to depict trends in biomass. Simple models used basic (Schaefer and/or Gompertz) formulations of surplus production models with fishing as the only source of mortality, while the extended models incorporated ecological and environmental covariates as well. One model (the "predation hypothesis") investigated trends in potential weakfish predators as well as several environmental variables. In contrast, the "forage model" investigated links between weakfish abundance and trends in potential predators/competitors and forage species. Suitable covariates for inclusion in the extended models were selected using correlation analysis and stepwise regression. For a particular covariate to be retained for further analysis, statistical results had to be significant and logical (*e.g.* negative correlation for predator, positive correlation for forage). The predation model implemented a Type III predator-prey functional response (Steele-Henderson), while the forage model investigated a suite of response functions. "Best" models were selected using model fits, parameter estimation, trends in residuals, and/or Akaike Information Criterion (AIC).

Detailed results of the predation hypothesis are presented in section **C9.0**, while the forage hypothesis is discussed in detail in section **C10.0**. The following is a summary of findings for the two models.

Predation model

Simple biomass dynamic modeling using the Gompertz formulation produced significant parameter estimates and strong model fit ($r^2 = 0.68$). However, uncertainty around parameter estimates was moderate, and a prominent systematic trend in residuals suggests process error (missing terms) in the population model (Figure C9.3-8).

Correlation analysis and stepwise regression were used to investigate potential links between key environmental and ecological covariates and trends in weakfish abundance. Results of the statistical analysis indicate that indices of abundance based on recreational CPUE for two key predator species (striped bass and spiny dogfish) are negatively correlated with weakfish biomass (Table C9.3-1). Including these indices in a biomass dynamic model with a Type III functional response increased model fit ($r^2 = 0.94$) and greatly reduced uncertainty in parameter estimates (Table C9.3-6). Additionally, residuals from the run including predation appear much more random (Figure C9.3-9).

Forage model

Statistical analyses identified significant correlations between weakfish and both striped bass abundance and menhaden abundance that might influence weakfish stock dynamics. These terms were included in biomass dynamic modeling that evaluated six different mortality hypotheses (fishing only, 3 functional responses, 2 depensatory responses) in both Schaefer and Gompertz formulations, for a total of 12 competing models. AIC showed that the model including fishing mortality, striped bass abundance, and a depensatory response to the ratio of menhaden to striped bass performed best, while the two models with fishing mortality only ranked 7th and 9th out of 12.

Results of these two models (predation hypothesis and forage hypothesis) suggest that a simple biomass dynamic model based on trends in weakfish abundance and harvest is insufficient for

characterizing population dynamics. Inclusion of predation terms greatly improves model results, providing trends that more accurately reflect field observations, and strongly support a systematic increase in M in the past decade or more. Comparison of Mp from the predation model and M calculated from $Z_{VPA} - F_{rel}$ show a very similar pattern (Figure C11.2-3). In addition, the fishing only models produce population parameters that are unrealistic for a species with a life history such as weakfish. Weakfish grow quickly and mature at a young age, suggesting that the population growth parameter, r, would be relatively high. The Gompertz model when only fishing mortality is assumed estimates r = 0.26 and a carrying capacity (K) over 150,000 MT (Table C9.3-3). This relatively slow growth rate and extreme carrying capacity seem unlikely for a species that exhibits rapid growth and early maturation. Including ecological covariates in the biomass models results in r and K estimates which are much more realistic for a species like weakfish.

It has been argued that spatial and temporal overlap of striped bass and weakfish are limited to the fall during southward migrations, and therefore opportunity for predation events is uncommon. The WTC recognizes that weakfish are not a principal prey item of striped bass. A comparison of estimated predation mortality and striped bass bioenergetic data indicates weakfish are a minor component of striped bass diet (see section **C10.6.3**). Regardless, with the dramatic increase in striped bass abundance over the last decade, even low consumption rates would lead to substantial increases in weakfish predation mortality.

C11.3 Additional analyses to investigate changes in productivity

C11.3.1 Environmental

North Atlantic sea surface temperatures have been found to exhibit a 65 to 70 year oscillation (Schlesinger and Ramankutty 1994; Enfield et al 2001). Kerr (2000) termed this oscillation the Atlantic Multidecadal Oscillation (AMO) to distinguish it from the atmospheric North Atlantic Oscillation (NAO). Since its discovery, the AMO has been linked to the occurrence and severity of climatic events throughout the northern hemisphere such as rainfall/drought, severity of Atlantic hurricanes, and North American and European summer climate (Enfield et al 2001). A concise, albeit "unofficial," review of the AMO is available from Wikipedia, the open access online encyclopedia (*http://en.wikipedia.org/wiki/*).

Data for the AMO are available from the National Oceanic and Atmospheric Administration (NOAA) Earth System Research Laboratory at *http://www.cdc.noaa.gov/data/timeseries/AMO* as both an annual index (1948 to 2007) and a 10-year moving average index (1861 to 2002). The longest official time series available for weakfish consists of reported commercial harvest from 1950 to 2007. Although many factors affect reported harvest (*e.g.* abundance, effort, gear selectivity, data collection methods), the harvest time series was considered proxy data for weakfish abundance. Trends in the smoothed AMO index were compared to commercial weakfish harvest estimates from 1950 to 2002. Pearson correlation was conducted using SAS Proc Corr (SAS 1990). To evaluate potential temporal delays in population response, correlations were conducted with the harvest time series lagged backward 0 to 10 years.

Using reported weakfish commercial harvest as proxy data for weakfish abundance, graphical analysis suggests that weakfish abundance increases when the AMO is in a negative phase (Figure C11.3-1). In 1950, the AMO was in a positive phase, but began a gradual decline in the mid 1950s, crossing into a negative phase in the mid 1960s, and continuing to a relative minimum a decade later. The AMO index then increased steadily until the early 1980s, leveled off for a number of years, before resuming an increasing trend in 1990. The index entered a positive phase around

1994, and continued to increase through 2002 (the last year available for the 10 year average). In contrast, weakfish landings, and presumably abundance, were low from 1950 to the 1960s. Rapid increases in harvest began in 1970 while the AMO was in a negative phase. Shortly after the AMO reached a minimum, landings peaked and rapidly dropped off. As the AMO entered the recent positive phase, landings have dropped to record low levels.

Correlations between the 10 year moving average AMO index and commercial weakfish landings were negative and highly significant ($P < 0.001$) for all comparisons (Table C11.3-1). The strongest association ($r = -0.83$) was found when a five year time lag was incorporated. Peaks in commercial landings and record numbers of citation size fish (Uphoff 2004) are reported during the late 1970s and early 1980s when the AMO was at its lowest point. Despite commencing an upward trend in the mid 1970s, a slight stabilization of the index coincided (when a five year lag is incorporated) with management measures, perhaps boosting their effectiveness. The index resumed a positive trend, and the transition from a negative to a positive phase occurred around the time of the unexpected reversal in weakfish stock growth.

The longest official index available for weakfish is the NMFS commercial harvest record extending from 1950 to present. However, Joseph (1972) provides a landings history back to 1929. Correlation analysis with the longer time series provides weaker, but still significant negative relationships between AMO and weakfish landings. The strongest correlations ($r \sim -0.38$, $P < 0.01$) occur with a 7 to 9 year time lag (Table C11.3-1). Between 1929 and 1935, AMO was increasing in a positive phase after which it stabilized. Landings during this period were decreasing, and reached relative low levels a short time after the AMO stabilized. Landings began to rebound shortly after the AMO began a declining trend around 1959.

Quantitative analysis is not possible prior to 1929, but a qualitative analysis suggests the correlations hold for earlier periods as well. Collette and Klein-MacPhee (2002) indicate periods of high landings in southern New England and into Cape Cod Bay during the early part of the 20th century. This harvest boom occurred as the AMO switched from a positive to negative phase, and during portions of the next 3 decades while the AMO remained negative. Landings from this region were not consistently high throughout this period, but were alternately abundant and scarce. This would not be unexpected for a species fished heavily near the extent of its range. High fishing pressure would diminish the stock, but during a period of high productivity, populations could rebuild quickly.

Crecco (2009) investigated localized temperature variation but found no correlation with weakfish abundance. Water temperature data from other New England stations (*e.g.* Oviatt 2004, Nixon et al 2004) show similar patterns to that observed by Crecco, but with localized differences in magnitude and duration of anomalies. The AMO incorporates data from the entire North Atlantic which may be more representative for a stock distributed over a wide range. In addition, Crecco evaluated correlations in temperature and abundance over a shorter time scale and without a smoothing function. Interannual variability in temperature, particularly when confined to a single site, may partially mask long term trends, making correlations harder to detect.

Decadal scale influence of climate and other variables on marine populations has been gaining recognition in the last two decades (*cf* Hare and Able 2007, Introduction pp 31-32). In some recent examples, Oviatt (2004) found that increased winter water temperatures led to a decline in boreal demersal species such and winter flounder (*Pseudopleuronectes americanus*) and windowpane flounder (*Scopthalmus aquosus*), followed closely by an increase in demersal decapod crustaceans. Increased winter temperatures were also strongly correlated with increases in abundance of grey snapper (*Lutjanus griseus*) in Texas (Tolan and Fisher 2007). Perhaps most

relevant to weakfish, Hare and Able (2007) identify a link between winter water temperature and juvenile abundance of Atlantic croaker (*Micropogonias undulatus*), another member of the scianid family. The authors propose a conceptual model whereby above average winter water temperatures result in high juvenile survival which leads to "outbursts" in Atlantic croaker populations. These studies provide evidence that productivity and stock dynamics can be significantly influenced by environmental factors such as water temperature. It is recognized that factors other than abundance influence harvest levels, but strong correlations and qualitative associations between weakfish harvest and the AMO over the last century provide support that oceanic conditions may have a significant influence on weakfish stock dynamics and may help explain the unexpected decline in productivity over the last decade. It is not possible from the available data to determine whether oceanic conditions are acting on weakfish directly (*e.g.* recruitment strength), indirectly (*e.g.* changes in predator and/or prey abundance), or a combination of both.

C11.3.2 Food habits

The NEFSC Food Habits Database contains information on type and prevalence of prey items for key species, obtained by analysis of stomach contents collected during seasonal trawl surveys. Preliminary review of the database showed that fall surveys from 1991 to present have used consistent methodology and have collected sufficient sample size of weakfish. Prey types were grouped into broad diet categories (amphipod/isopod, crab/shrimp, forage fish, empty, other), and the annual percent composition in all observed stomachs was calculated for each prey category. Three year moving averages were calculated to smooth out interannual variation in prevalence, and SAS Proc Corr (SAS 1990) was used to evaluate correlations between prey prevalence and trends in total mortality (biomass weighted, ages 1-5) estimated by ADAPT VPA. These correlations were conducted for 1-inch size intervals from 5" to 12", and for all sizes combined, for the years 1991 (first year of food habits data) to 2002 (most recent year of VPA results without retrospective pattern).

Negative correlations with Z_{VPA} were generally observed with percent composition of crab/shrimp and forage fish, while the percent composition of amphipod/isopod and the prevalence of empty stomachs were positively correlated (Table C11.3-2; Figure C11.3-2). More than half of these correlations were strong ($P < 0.10$), and only the 12" category was not significantly correlated. The "empty" category produced the greatest number of significant correlations (6 out of 9 size groups) and often produced the strongest correlations with Z. One interpretation of these data is that primary prey items (forage fish and large invertebrates) became scare, so weakfish shifted to other, less optimal prey (small invertebrates) or were unable to find suitable forage (empty stomachs).

The 2008 DPSW review panel expressed concern over the lack of empirical data to support the hypothesis that weakfish productivity has been compromised since the late 1990s. The data presented above provide direct empirical evidence that weakfish forage opportunities may have diminished during the mid to late 1990s. Decreased forage abundance and/or sub-optimal prey types could affect weakfish productivity through increased natural mortality or changes to other parameters (*e.g.* growth rates). Empirical data are also provided through a comparison of two studies on weakfish diet composition in Chesapeake Bay (Hartman and Brandt 1995; R. Latour, Virginia Institute of Marine Science, pers. comm.). The studies occurred approximately a decade apart on either side of the time period during which natural mortality was suspected to have increased. These studies indicate shifts in diet composition (Figure C11.3-3) that are consistent with those observed in the NEFSC food habits database. Weakfish initiate piscivory at an early age, beginning with bay anchovy at age 0 and moving to larger prey such as menhaden and spot by age 1

to 2. In Chesapeake Bay, Hartman and Brandt (1995) indicated that greater than 80% of weakfish diet from age 0 to 2 consisted of these principal prey items, with the majority of the remainder (5 to 20%) made up of invertebrates (Figure C11.3-3). In contrast, by 2002, 30% or more of weakfish diets age 0 to 5 consisted of invertebrate species, with forage fish accounting for less than 40% of the total (R. Latour, Virginia Institute of Marine Science, pers. comm.). 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. During both periods, weakfish 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 C11.3-3). 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.

Early switching to a fish diet indicates that weakfish is a specialized piscivore (Persson and Brönmark 2002). Early switching requires high growth rate, which implies high densities of proper forage and safe foraging opportunities. Species undergoing ontogenetic diet shifts face a risk of delayed transitions among feeding stages if food resources are limited and competition is intense (Persson and Brönmark 2002). If unfavorable conditions prevail, then growth is reduced, often leading to ontogenetic bottlenecks where individuals fail to gain size advantage on larger prey items in the next stage. Unfavorable conditions can also result in increased vulnerability to competitors and predators. Competing individuals and predators may hinder one another's feeding activities, leading to starvation or they may eat one another (including cannibalism; Yodzis 1994). The two Chesapeake Bay studies and the NEFSC food habits database provide evidence that weakfish forage opportunities diminished during the late 1990s. Shifts in diet composition may have resulted in decreased productivity of the stock through decreased growth rates and/or increased competitive or predatory mortality.

C11.3.3 Size at Age

Another possible indicator of changing productivity would be changes in size at age over time. Preliminary investigations were conducted using separate time series of scale- and otolith-based age data, but are not presented here. Although some data produce significant trends (both positive and negative) in size at age during the time series, the WTC was concerned that the availability of samples could be affecting the results. Due to geographic differences in size (*e.g.* between southern and northern Mid-Atlantic states), annual differences in the source and sample size by source can affect estimated average size when evaluated on a regional or stockwide basis. Additionally, no single data source had a consistent time series of sufficient sample size at age to evaluate on an individual basis. The WTC intends to continue investigating potential changes in size at age by identifying alternative data sources and analytical methods.

In addition, trends in size (weight) at age were used to evaluate trends in natural morality (Lorenzen 1996). Preliminary investigations indicate that trends in natural mortality are well correlated with VPA-based estimates of total mortality. However, as mentioned above, the WTC is concerned that trends in weight at age may be affected by interannual variability in the location and

intensity of sampling, so formal results are not presented in this report.

C11.4 Discussion

Following the 2006 stock assessment, the WTC concluded that fishing mortality for weakfish had remained stable at low levels in recent years, but that the population was near all time low levels due to increased predation/competition with striped bass. These conclusions received criticism from the 2006 peer review panel and the 2008 DPSW review panel. The panels identified concerns with input data and lack of empirical data to support the conclusions. The DPSW panel expressed the opinion that any number of factors could have contributed to the trends in fishing mortality observed from the VPA, such as unreported harvest, changing fishing practices, or environmental/ecological influences other than predation. Responses to concerns regarding input data are discussed elsewhere in this report (*e.g.* sections **C5.0** and **C6.0**). This section of the report investigates a variety of data types and sources that might support or discount the hypothesis that weakfish productivity has declined.

The WTC recognizes that population dynamics of any species are influenced by a wide variety of anthropogenic, environmental, and ecological forces, both direct and indirect. As these forces work in concert, enhancing or diminishing the effects of each other, it is not possible to predict the magnitude or direction of their cumulative impact. This section is not meant to be a comprehensive review of all the possible influences on the weakfish stock, but attempts to investigate some of the more direct factors. The overwhelming majority of data types investigated lend support to the WTC's evaluation that factors other than fishing mortality were influencing stock dynamics.

- During the period 1995 to 2003, total mortality (biomass weighted Z_{VPA} ages 1 to 5) increased by 375% from 0.48 to 1.82 (Figure C11.3-2)
- During the period 1995 to 2002, F_{rel} estimates, which are independent of natural mortality, remained relatively stable and were much lower than F_{VPA} estimates based on assumptions of known constant M. (Figure C11.2-2)
- Biomass models that incorporated ecological interactions with key predators/competitors and forage species resulted in improved model fits and more realistic population parameters than models based on fishing mortality alone. (Tables C9.3-3; compare Figures C9.3-8 and C9.3-9).
- Estimates of unrecorded removals necessary to replicate F_{VPA} trends were unrealistically high and imply complete management failure. (Figure C10.7-1)
- Trends in north Atlantic sea surface temperature tracked through the AMO are strongly correlated with weakfish harvest trends over the last century. (Figure C11.3-1)
- Two independent studies identify shifts in prey types during the 1990s. (Table C11.3-1; Figures C11.3-3)
- The incidence of empty weakfish stomachs observed in the NEFSC food habits database correlates strongly with the increase in total mortality (Z_{VPA}) as fishing mortality (F_{rel}) remains stable (Figure C11.3-2)

The WTC admits that the conclusion in the 2006 stock assessment that weakfish declined solely as a result of striped bass predation may have been overly myopic; however, given the additional evidence presented in this section, the WTC still contends that productivity of the weakfish stock was compromised beginning in the mid to late 1990s due to one or more of the factors discussed

above. Potential changes in productivity include decreased growth rates, starvation, competition, or increased vulnerability to predation, among others. Regardless of the source, the overall effect was a decline in weakfish stock biomass which analyses suggest was not attributable to fishing pressure.

C12.0 Estimate biological reference points using equilibrium and non-equilibrium assumptions and evaluate stock status relative to these BRPs. (TOR #8)

C12.1 Thompson-Bell reference points

Weakfish are currently managed relative to reference points developed under Amendment 4 of the Weakfish Fishery Management Plan. Reference points were updated during the 2002 assessment (Kahn 2002) using a spreadsheet based model, but the updated reference points were not adopted because it was not a benchmark stock assessment. For the current benchmark assessment, reference points have been calculated using the spreadsheet model. The model requires input values for partial recruitment at age, natural mortality, and age specific maturity. Weight at age is calculated using relationship described by Vaughan 1999. SSB is calculated as the summed product of number and weight at age. Fishing mortality reference points were found by solving for F rates that provided spawning stock biomass of 30% (target) and 20% (threshold) relative to unfished stock (J. Uphoff, MD DNR, pers comm.).

Reference points established under Amendment 4 assumed equilibrium conditions with a constant natural mortality rate of M = 0.25. Implementing the spreadsheet method described above and updating partial recruitment values to the average of the most recent three years (2005-2007) allows direct comparison of reference points during the two periods. Amendment IV established new reference points for fishing mortality target of $F_{\text{target}} = F_{30\%} = 0.31$, and a fishing mortality threshold of $F_{\text{threshold}} = F_{20\%} = 0.5$. Updating partial recruitment to the average of 2005-2007 decreases reference points estimates to $F_{30\%} = 0.28$ and $F_{20\%} = 0.42$. The Shepherd spawner–recruit function was also updated using recent data, providing a new SSB threshold estimate of $SSB_{20\%}$ = 13,108 MT. Comparison of VPA based parameter estimates indicates that fishing mortality on weakfish has been above the equilibrium target of $F_{30\%}$ in all years since 1981, and above the $F_{20\%}$ threshold in every year except 1995 (Figure C12.1-1). SSB has exceeded the $SSB_{20\%}$ threshold during only two time periods: 1986-1987 and 1994-1997. SSB_{2007} is approximately 55% of the threshold, but is likely overestimated based on the observed retrospective pattern.

Reference point estimates are conditional on the input values, such as partial recruitment at age. Given the pronounced retrospective pattern that overestimates abundance at age (see section **C7.0**), estimates of partial recruitment in recent years are likely to be underestimated. Consequently, fishing mortality reference points are likely to be overestimated. The retrospective pattern diminishes to near zero in approximately five years, so using partial recruitment vectors from an earlier time period may provide more stable estimates of reference points. Using average partial recruitment at age from 1999 to 2001 provides estimates of a fishing mortality target of $F_{30\%} = 0.25$, and a fishing mortality threshold of $F_{20\%} = 0.36$. The associated SSB threshold is estimated at $SSB_{20\%}$ = 10,179 MT.

Another factor that could affect the reference point estimates is the assumption of constant natural mortality. The WTC has expressed concern regarding the assumption that M has remained constant throughout the time period analyzed. Recent analyses indicate that natural mortality may have increased dramatically over the last decade, to levels as high as M = 1.0 or higher. Reference points were also calculated using a natural mortality rate of M = 0.8. Using average partial

recruitment at age from 1999 to 2001 provides estimates of a fishing mortality target of $F_{30\%} = 0.78$, and a fishing mortality threshold of $F_{20\%} = 1.20$. Under these assumptions, fishing mortality on weakfish declined below the threshold for most years between 1989 and 2001, and was below the $F_{30\%}$ target between 1995 and 1999. F_{2007} is below the target mortality rate, but is likely underestimated given the observed retrospective pattern.

As noted above, reference points assume natural mortality is constant over time. Sensitivity of reference points can be investigated under different assumptions of constant natural mortality, but more advanced techniques are required to investigate the influence of systematic changes in M. Both of the biomass dynamic models investigated during this assessment produced estimates of both equilibrium and non-equilibrium reference points, and are discussed below.

C12.2 Reference points calculated by the "Predation Hypothesis"

The following is a summary of the reference point and stock status analyses conducted under the predation hypothesis. Additional details are presented in the full discussion of the predation hypothesis in section **C8.0** of this report.

C12.2.1 Equilibrium conditions

Updated F_{msy} and B_{msy} thresholds were estimated for weakfish using the dynamic Gompertz external surplus production model (Quinn and Deriso 1999; Jacobson et al 2002). Surplus production estimates from 1981-2008 were regressed against weakfish biomass (Biowt) and the product of the log weakfish biomass and biomass (LogBiowt*Biowt) in a two variable linear regression model without a y-axis intercept (see equation (6) in section **C8.0**) using the ROBUSTREG procedure contained in the Statistical Analysis System (SAS 2002). Equilibrium F_{MSY} threshold is expressed by the intrinsic growth rate (r) parameter, whereas B_{MSY} is expressed by the carrying capacity (K) divided by 2.72 (Quinn and Deriso 1999). Since temporal shifts in striped bass predation are absent from the traditional dynamic models, the overfishing definitions (F_{msy}, B_{msy}) in these models are fixed in time.

The analysis was conducted for the two scenarios of fishing mortality only and fishing mortality plus predation. The "fishing only" model provided a reasonably good fit (r**2 = 0.54) to the biomass time series with statistically significant ($P < 0.05$) r and K parameter estimates (Table C9.3-3). The resulting overfishing thresholds were $F_{MSY} = 0.26$ and $B_{MSY} = 57{,}388$ MT. However, a distinct serial pattern was present in the residuals, and the model consistently over predicted weakfish surplus production from 1998 to 2008 by 20% to 350% (Figure C9.3-3). This systematic trend in the direction of the residuals usually indicates that the model is lacking an informative parameter(s).

Addition of a predation term resulted in greatly improved model fit (r**2 = 0.84), and the parameters (r, K, C) were estimated with much higher precision than the production model without predation (Table C9.3-3). Resulting equilibrium overfishing thresholds (Fmsy, Bmsy) from the extended Gompertz model were 0.54 and 19,289 mt, respectively (Table C9.3-3). Moreover, the strong serial correlation in the residuals noted from the basic Gompertz model was minimized when predation effects of striped bass and spiny dogfish were included (Figure C9.3-4).

Within the predation hypothesis, reference points were also calculated using the discrete Gompertz model with and without predation (see equations 12 and 11 of section **C9.0**) using iterative reweighting. The discrete model with predation is referred to in this report as the Steele-Henderson (S-H) model. As above, the equilibrium F_{MSY} threshold is expressed by the intrinsic

growth rate (r) parameter, and B_{MSY} is expressed by the carrying capacity (K) divided by 2.72 (Quinn and Deriso 1999).

When only fishing effects are evaluated, this model provided a good fit (r**2 = 0.82) to the biomass data with statistically significant ($P < 0.05$) r and K parameter estimates. Overfishing thresholds were calculated as $F_{MSY} = 0.32$ and $B_{MSY} = 25{,}259$ MT. As with the external model, the discrete model with fishing effects only consistently over predicted weakfish biomass from 1998 to 2008 by 10 to 250% (Figure C9.3-8), indicating that the basic Gompertz model was plagued by substantial process error.

The full Steele-Henderson (S-H) production model fit with iterative reweighting provided a very good fit (r**2 = 0.94) to the biomass and predation data with statistically significant ($P < 0.05$) r, K, c and A parameter estimates (Table C9.3-6). Overall, the S-H models was not only a better fit to weakfish biomass than the Gompertz model with only fishing effects, but the r and K parameters were estimated with much higher precision. The resulting weakfish equilibrium overfishing thresholds were estimated at $F_{MSY} = 0.72$ and $B_{MSY} = 17{,}009$ mt (Table C9.3-6). Moreover, unlike the severe residual pattern evident from 1998 to 2008 based on the basic Gompertz models, there is little if any systematic residual pattern from the S-H model fitted by iterative re-weighting (Figure C9.3-9).

Given the clear residual problem associated with the Gompertz models estimated without predation, these models were not used to evaluate weakfish stock status. Relationships between parameter estimates and reference points evaluated through the Steele-Henderson model are shown in Figures C9.3-13 (fishing mortality) and C9.3-14 (biomass). Fishing mortality fell below the F_{MSY} = 0.72 1993 and has remained below the threshold in all years except 2002. SSB fell below B_{MSY} = 17,009 MT threshold in 1983, falling to less that one-third the threshold in 1990. Biomass subsequently rebounded to more than 80% of the threshold in 1995 and 1996, but has since declined.

C12.2.2 Non-equilibrium conditions

The non-equilibrium S-H model allows the direct calculation of the amount of mortality attributable to predation, M_p. Because predation effects are transient over time, weakfish surplus production and predation-induced mortality can vary greatly across years, resulting in time varying F_{msy} and B_{msy} thresholds. The degree of temporal variation in F_{msy} and B_{msy} depends on the magnitude and trend in predator abundance, the predator consumption exponent (c) and on the prey stock size (A) at which the consumption threshold of predation takes place in equation (11). Thus, the annual F_{MSY} values from the S-H model are not fixed in time but rather are a function of the fixed intrinsic rate (r) and the time varying predator consumption rate. Similarly, weakfish biomass threshold (B_{MSY}) can vary over time depending on the amount of weakfish biomass consumed annually (see equations 16 and 17 in section **C9.0**).

Calculation of non-equilibrium reference points indicates that F_{MSY} has decreased substantially from above 0.75 on the early 1980s to less than 0.25 in most years since 2001 (Table C9.3-5). Estimated fishing mortality rates dropped below the threshold in 1995 and 1996; however, as excessive predation has eroded stock size and associated surplus production, F has exceeded the non-equilibrium F_{MSY} estimates in all years since 1997 (Figure C12.2-1).

B_{MSY} has shown greater stability than F_{MSY}, but still declined from over 20,000 MT in 1981 to less than 15,000 MT between 1997 and 2002, before rebounding in recent years (Table C9.3-5). Stock size as a proportion of maximum spawning potential exceeded 25% MSP between 1981 and 1986, and again between 1994-1996, but has since decreased drastically to less than 5% of an

unfished stock since 2005 (Table C9.3-5).

C12.3 Reference points calculated by the "Forage Hypothesis"

The following is a summary of the reference point and stock status analyses conducted under the forage hypothesis. Additional details are presented in the full discussion of the forage hypothesis in section **C10.0** of this report.

C12.3.1 Equilibrium conditions

The forage hypothesis investigated both Schaefer and Gompertz forms of the surplus production function. For both methods, equilibrium reference points are estimated from the fit population parameter values. However, the Gompertz production function with depensatory predation mortality as a function of striped bass biomass and the striped bass:menhaden ratio provided the best model fit, so reference points are evaluated for that model only. For the Gompertz model, equilibrium reference points are calculated as $F_{MSY} = r$ and $B_{MSY} = K / 2.72$.

The best model fit resulted in parameter estimates of r = 0.48, K = 51,521 MT, c = 0.29, and B_{1981} = 82,472 MT, providing equilibrium reference points of $F_{MSY} = 0.48$, $F_{collapse} = 1.31$, and B_{MSY} = 18,941 MT. Jackknife and bootstrap procedures indicated that parameters were estimated with good precision (Tables C10.5-2 and C10.5-3). Estimated fishing mortality rates exceeded F_{MSY} for the first decade of the time series, but dropped below the threshold in 1992 (Figure C10.5-13). By 1995, F had fallen to approximately 60% of the threshold, but subsequently began a steady increase to a level slightly above F_{MSY} during 1998-2002. After 2002, F_t dropped sharply and remained at about 50-70% of the threshold through 2006 (Figure C10.5-13).

Biomass has been below the SSB_{MSY} threshold in all years except 1996. Estimates of proxy MSP, calculated as B_t / K, fell continuously after 1983 from 65% of K to reach about 12% by 1990 (Figure C10.5-16).

C12.3.2 Non-equilibrium conditions

Two methods were implemented to calculate nonequilibrium reference points from the forage hypothesis model. The first simply involved renaming F_{MSY} to $Z_{MSY} = 0.48$ and comparing it to annual estimates of total (fishing plus predation) mortality (Z_{pt}). This estimate of total mortality at MSY is constant across time and serves as a benchmark for total mortality changes. Estimates of M_{pt} rose from a very low level in the mid-1980s to estimates equivalent to F_t and Z_{msy} by 1999 (Figure C10.5-13). After 1999, M_{pt} accelerated rapidly and reached 1.4 in 2006 (Figure C10.5-13). When combined with estimates of fishing mortality, it is evident that total mortality ($Z_{pt} = F + M_{pt}$) has exceeded the equilibrium reference point throughout the time series, except for a brief period in the mid 1990s.

The second method used to calculate non-equilibrium reference points was to directly calculate F_{MSY} under non-equilibrium conditions as equilibrium F_{MSY} minus predation mortality (see equation 16 of section **C10.0**). Mortality associated with the predator or predator-prey conditions is considered "off-limits" for management and yield can only be influenced by managing the fishery. Low level of predation mortality occurring in the 1980s and early 1990s resulted in F_{PSYt} levels close to Z_{MSY} (Figure C10.5-14). As fishing mortality rate declined, estimates of F_t fell below the non-equilibrium threshold between 1993 and 1996. However, increasing losses due to striped bass began to erode F_{PSYt}. Consequently, the amount of production available for harvest has declined, and fishing mortality has remained above the threshold in all years since 1997.

C12.4 Discussion

Reference point estimates and stock status determinations for the different models are compared in Table C12.4-1. Under equilibrium conditions, the ADAPT VPA estimate of F_{2007} (0.51) exceeds the overfishing threshold ($F_{20\%}$ = 0.42), indicating that overfishing is occurring. However, the two production models provide estimates of F_{2007} that are lower than their respective F_{MSY} thresholds. Under these scenarios, overfishing is not occurring on weakfish. For all three models under equilibrium condition, SSB_{2007} is below the respective biomass threshold, indicating that weakfish are overfished.

It is not possible to calculate non-equilibrium reference points from the VPA; however assuming non-equilibrium conditions, both production models indicate that weakfish are overfished and overfishing is occurring. The transition from overfishing not occurring under equilibrium to overfishing occurring in a non-equilibrium system can be attributed to the erosion of "available" production due to increased weakfish removals through predation and/or competition over the last decade. Although fishing mortality is above the threshold level, both models indicate that the F/Z ratio has been below 50% since the mid to late 1990s.

The 2006 peer review panel did not accept the WTC's determination that stock size had declined due to increased natural mortality as a result of predation (ASMFC 2006a, Part B). In fact, little attention was given to weakfish stock status at all during that review. In the current analysis, all models investigated indicate that weakfish stocks are depressed. Regardless of the many factors influencing the stock that may have led to this condition, it is imperative that the 48th SARC evaluate the determination of stock status so that the Weakfish Management Board can be apprised of the situation. Within this document the WTC has presented a suite of hypotheses that attempt to explain the mechanisms that have resulted in the current low biomass, the implications of which will help determine future management actions. The WTC requests guidance from the SARC on the utility of the different models and how they may be improved in order to ensure sustainable management of the stock. However, before management measures can be considered, the true status of the stock must be determined. With the models investigated in this stock assessment, it is evident that the weakfish stock declined substantially and is at or near all time low levels.

C13.0 Review stock projections and impacts on the stock under different assumptions of fishing mortality. (TOR #9)

C13.1 ADAPT projections

Projections were conducted relative to output from the ADAPT VPA using the AgePro (version 3.1) module of the NFT Toolbox. Biomass was projected for 25 years starting from the terminal year estimates of the preferred ADAPT VPA run (see section **C7.0** of this report). Values for biological parameters (*e.g.* weights at age, partial recruitment, etc) were the same as those used in the terminal year of ADAPT input. Multiple runs were conducted assuming a range of fishing and natural mortality values. Recruitment followed an empirical distribution of Age 1 numbers estimated through the full time series of the VPA.

All projection runs resulted in increased biomass over time and appeared to reach asymptotic values (Figure C13.1-1). However, different assumptions regarding F and M resulted in a wide range of projected biomass. Under the assumption of constant M = 0.25 and a harvest moratorium, SSB is projected to increase to more than 275,000 MT by the year 2032. Allowing harvest to occur at a level F = 0.25 (slightly below the fishing mortality target of $F_{30\%}$ = 0.28) the stock is expected to

peak at approximately 135,000 MT by 2019.

The WTC has expressed concern that natural mortality has increased in recent years. Under this scenario, projections based on M = 0.25 would be overestimated. Recent analyses have indicated that natural mortality may be as high as M = 1.0 or more. To investigate the potential for stock rebuilding under similar conditions, projections were conducted assuming a constant M = 0.75 (Figure C13.1-1). Under a harvest moratorium, SSB is expected to increase to approximately 45,000 MT by 2013. Allowing fishing mortality to occur at a level of F = 0.25, rebuilding would occur to 36,000 MT by 2013.

C13.2 Projections based on the "Predation Hypothesis"

C13.2.1 Introduction

Results from previous weakfish assessments (Kahn et al 2002, 2006) revealed that overfishing (F > Fmsy) had occurred on weakfish from about 1981 to 1991. As a result, more stringent fishery regulations were imposed on the weakfish recreational and commercial fisheries in 1992 and 1993 and fishing mortality (F) rates from 1996 to 2003 fell to well below Fmsy. Following a drop in F, coast-wide weakfish abundance initially rose about five fold from 1992 until about 1998, but stock size dropped unexpectedly thereafter to the lowest levels in the time series by 2007. Because F levels were relatively low and stable after 1999, follow-up studies (see TOR #5-7) have begun to find evidence for a Trophic Hypothesis involving enhanced striped bass (*Morone saxatilus*) and spiny dogfish (*Squalus acanthias*) predation or a more complex trophic triangle among weakfish, striped bass and Atlantic menhaden (*Brevoortia tyrannus)*. Under non-steady-state conditions that might arise from a recent increase in natural mortality (M), the projected pace of weakfish stock rebuilding following a moratorium to harvest (F = 0) might occur very slowly if at all as was reported recently for Atlantic cod (*Gadus morhua*) in the Gulf of St. Lawrence (Swain and Chouinard 2008). As shown recently by Walters et al (2008), the rate of stock rebuilding of depleted finfish stocks can be highly unpredictable especially under a shifting environment. By contrast, if M for weakfish is assumed to be fixed over time at 0.25 as was assumed in the ADAPT model, and, if future recruitment remains average, a moratorium to harvest (F = 0) should result in rapid stock rebuilding to Bmsy over a relative short time horizon. Clearly, the degree to which additional management promotes rapid (< 15 years) weakfish stock rebuilding in the future depends, among other things, on whether or not future M remains low and fixed over time.

In this section, weakfish relative spawning stock biomass (TSSB, mt) projections were made from 2010 to 2020 following the imposition of a simulated coast-wide moratorium (F = 0) to harvest beginning in 2009. The projection model was an extension of the Harvest Control Model (HCM) (Rugolo and Crecco 1993) used to forecast decadal shifts in striped bass (*Morone saxatilus*) biomass under several management scenarios. Given the uncertainty surrounding the current and future trend in M estimates, the following three scenarios that bracket a wide range of possibilities were examined with the weakfish HCM following a simulated 2009 moratorium to harvest: 1) M is fixed at 0.25 throughout the time series (1980-2020) as in ADAPT, there is no recent rise in trophic impacts on weakfish productivity, and recent (1999-2008) fishing mortality (F) have remained high (F > 1.0) as per ADAPT; 2) there is a moderate rise in M (from 0.25 to 0.65) from 1999 to 2020 due to predation but the magnitude of predatory mortality is less than predicted by the Steele-Henderson Model, and recent (1999-2008) F estimates have risen to moderate (0.7 to 1.0) levels; and 3) M on weakfish after 1997 has risen four-fold in magnitude (from 0.25 to 1.0) as per the Steele-Henderson Model, and fishing mortality (F) rates have remained relatively low (F < 0.50) from 1996 to 2008 as

per the Index-based Analysis.

C13.2.2 Methods

A modification to the Harvest Control Model (HCM) was used to project relative changes in weakfish total spawning stock biomass (TSSB) from 2010 to 2020 following the imposition of a simulated moratorium to harvest (F = 0) beginning in 2009. The theory and application of the HCM are fully described for striped bass catch quota projections (Rugolo and Crecco 1993). The entire model is programmed in SAS (SAS 2002) (Appendix C-3). The model output (ages 1+ TSSB) in the HCM was expressed as year (1980-2020) and age (ages 1-10) effects. A small percentage of weakfish do survive beyond age 10, but the average contribution of ages 11+ weakfish to annual SSB was relatively modest from 1982 to 2007. The HCM is configured to operate off several age-related (ages 1 to 10) vectors that reflect somatic growth, maturity, natural mortality and partial recruitment (PR).

In this analysis, three scenarios are offered to examine how potential shifts in future natural mortality (M) levels might affect rebuilding of weakfish TSSB following the imposition of a simulated 2009 moratorium (F = 0) to harvest (Table 1). The scenarios are: 1) M is fixed at 0.25 throughout the time series (1980-2020) as in ADAPT, there is no recent rise in M due enhanced trophic impacts on weakfish productivity, unreported weakfish landings and discards have recently (1999-2008) increased steadily to where they now comprise over 80% of the total fishery losses and recent (1999-2008) fishing mortality (F) has remained high (F > 1.0) as per ADAPT; 2) there is a moderate rise in M (from 0.25 to 0.65) from 1999 to 2020 due to enhanced predation but the magnitude of predatory mortality is less than predicted by the Steele-Henderson Model, there is a more modest rise in recent (1998-2008) unreported landings and recent (1999-2008) F estimates have risen and remain moderately high (0.7 to 1.0) from 1999 to 2008; and 3) M levels on weakfish from 1999 to 2020 have exhibited a four- fold rise (0.25 to 1.0) in magnitude as per the Steele-Henderson Model, unreported landings have remained low and steady from 1999 to 2008 and fishing mortality (F) rates have remained relatively low (F < 0.50) from 1996 to 2008 as per the Index-based Analysis. In all of the above scenarios, input parameters in the weakfish HCM include fully recruited fishing mortality (F), somatic growth in average length (mm) and weight (kg) by age, percentage maturity by age, the partial recruitment vectors (PR) by age to the fisheries, either a fixed M of 0.25 (scenario #1) or a systematic rise in natural mortality (Scenarios #2 and 3) from 1997 to 2020. The fully recruited (ages 4+) F estimates, PR vectors, age at maturity and age-specific somatic growths in weight were taken directly from the preferred ADAPT run. Since retrospective bias in ADAPT severely underestimated recent (> 2004) F estimates, the fully recruited F in 2004 was repeated for the years 2005 to 2008. Discard mortality among sublegal weakfish was expressed in the HCM as 0.10 *F. Annual estimates of relative TSSB were summed in the HCM as the product of relative abundance by age, the average age-specific weight (kg) and the average age-specific maturity vector. The pace of future TSSB rebuilding among the three scenarios was measured arbitrarily against a relative Bmsy value that was scaled directly from the estimated equilibrium Bmsy threshold for weakfish from surplus production modeling.

In the HCM, a time series vector (1970-2020) of recruitment relative abundance is the primary variable that dictates temporal changes in future weakfish TSSB abundance by age. Thus relative TSSB in numbers from the HCM across the time series is expressed in recruitment units. There is a time series (1982-2007) of coast-wide average age 0 indices (Rec 0) that are expressed as mean catch/effort from nine State surveys from 1982 to 2007 (see section **C9.0**). However, the relatively high and stable age 0 indices occurring from 1997 to 2006 have thus far not translated into

high subsequent abundance of ages 1 and older fish, This mismatch in year-class abundance between subsequent age groups suggests the emergence of a recruitment bottleneck in recent years that has constricted the flow of recruitment via a rise in natural mortality (see sections **C7.0** to **C9.0**). Thus, instead of using average age 0 recruitment to express coast-wide recruitment in the HCM, a time series (1970-2008) of trial values of age 1 weakfish relative indices were imputed to the HCM so that the resulting trend in model-based TSSB from 1981 to 2008 closely approximated (Pearson correlation r> 0.90) the observed trend in ages 1+ relative biomass from the Index-based Analysis (refer to section **C8.0**). All available trawl data have indicated that coast-wide weakfish abundance was at peak levels from the mid 1970's until the early 1980's. To simulate these high TSSB levels from 1980 to 1985, a series (1970-1980) of dominant year-classes and relatively low fishing mortalities (F < 0.4) were imputed in the HCM. Note that projected TSSB each year is composed of 10 age groups. For this reason, it was necessary to begin recruitment to the HCM in 1970 so that the first full year of simulated TSSB composed of 10 age groups began in 1980. The long-term (1982-2008) average recruitment (long-term mean = 16.5) level was used to express age 1 recruitment from 2009 to 2020 (Table 1). We are primarily interested in comparing how a simulated moratorium (F = 0) might affect future stock rebuilding among three scenarios with a fixed M versus a rising future M. Thus, the use of constant recruitment from 2009 to 2020 would not alter the relative trajectory in projected TSSB rebuilding from 2010 to 2020. Random or log normal variation (stochastic) of future (2009-2020) recruitment using the observed recruitment time series (1982-2007) could have been used in TSSB projections from the HCM, but stochastic variation in future recruitment would have complicated the interpretation of TSSB trends during the rebuilding phase (2010-2020) of each scenario.

C13.2.3 Results and Discussion

In Scenario #1 under a relatively low and fixed natural mortality (M = 0.25) throughout the time series (1980-2020), the HCM predicted that a moratorium to all weakfish harvest (F = 0) enacted in 2009 and thereafter would lead to rapid TSSB recovery that would approached the Bmsy threshold by 2020 (Figure C13.2-1). Since M was assumed to have remained fixed at 0.25, reducing a high F (> 2.0) to zero from 2009 to 2020 exerts an enormous leverage on enhanced survival (high F/Z ratio) with which to rapidly rebuild TSSB from 2010 to 2020. However, it is important to note that in Scenario #1, we have assumed that the primary cause of the post 1999 weakfish stock collapse is overfishing mainly resulting in a systematic rise in unreported landings and discards from some yet unknown recreational and commercial fisheries. Since reported weakfish landings and estimated discards have fallen from 5500 mt in 1996 to about 800 mt by 2008, these unreported landings would need to have increased from about 1,000 mt to 5,000 mt. from 1998 to 2008 to have caused the post 1999 weakfish stock decline. Thus, in order to enact a complete moratorium on all weakfish landings in 2009, nearly all inshore fisheries activities from North Carolina to Rhode Island would have to cease from 2009 to 2020.

In scenario #2 under the assumption of a moderate rise in M from 0.25 to 0.65 after 1997, the HCM predicted that a moratorium (F = 0) enacted in 2009 would result in some measureable TSSB rebuilding by 2020, but the magnitude of stock growth would fall far short of the Bmsy threshold (Figure C13.2-2). Since M rose more moderately in this Scenario from 0.25 to 0.65, a moratorium to harvest exerts somewhat less leverage on survival (lower F/Z ratio) than Scenario #1 in order to rebuild TSSB over a 10 year horizon. Note that in Scenario #2, we have assumed that both a moderate rise in M (from 0.25 to 0.65) after 1996 coupled with a more modest systematic increase in unreported landings resulted in the post 1999 stock collapse. Thus, like Scenario #1, to enact an

effective moratorium to harvest of both reported and unreported weakfish landings, we would have to close nearly all inshore fisheries from North Carolina to Rhode Island from 2009 to 2020.

In scenario #3 under a pronounced rise in M from 0.25 to 1.0 ostensibly due to enhanced predation, the HCM predicted that a moratorium to harvest in 2009 and thereafter would result in little if any TSSB rebuilding by 2020 (Figure C13.2-3). Given the pronounced rise in M coupled with low and steady F from 1996 to 2008 in Scenario #3, a moratorium to harvest after 2008 provides very little leverage to enhanced survival (lowest F/Z ratio) on which measurable stock rebuilding depends. In Scenario #3, we note that fishing mortality (F) was largely driven by reported weakfish landings which have fallen by 80% from 1996 to 2008. By contrast, unreported weakfish landings and discards are assumed to have remained relatively low and steady from 1996 to 2008 during which M rose four-fold (0.25 to 1.0). In this Scenario, we have assumed that high M would persist throughout the rebuilding time frame (2010-2020). It is important to note that if M of 1.0 associated largely with enhanced predation during the recovery period (2010-2020) should drop back to the pre-1997 level of 0.25, measureable weakfish stock rebuilding would likely occur after 2010 without further management restrictions, although the imposition of a moratorium during this period would accelerate the pace of stock rebuilding.

C13.3 Projections based on the "Forage Hypothesis"

Jackknife and bootstrap estimates of GDR parameters and biomass in 2007 were projected with equations 2 and 7 to evaluate the effect of fishing restriction scenarios through 2015. Projections explored whether conditions set forward in Amendment 4 (ASMFC 2002) for stock rebuilding could be achieved by managing the weakfish fishery alone. Under overfished conditions (two consecutive years above the F threshold), F was to drop below 0.2 and SSB was to be rebuilt to 30% of an unfished stock (ASMFC 2002). While these conditions were based on VPA-based estimates and reference points, approximations could be developed from the GDR. The probability of biomass growing to 15,000 mt (approximating 30% of K) by 2015 under reduced fishing could be determined. Three fishery management scenarios were portrayed: an approximation of the minimum F reduction in Amendment 4 ($F_{recover}$), F = 0.2 (a literal interpretation of Amendment 4), and a moratorium.

Projections of biomass used each jackknife or bootstrap trial estimate of r, K, c, R_t, and B_{2007}. Estimates of H_t equaled U_t * B_t during 2008-2015, where U_t was an assumed exploitation rate derived from F_{2006} for 2007-2008, a transition F for 2009, and a 2010-2015 target F (specified above) and biomass. Striped bass biomass and R_t were constant at the 2006 estimate (88,000 mt and 5, respectively). In trials where cuts in F were imposed, it was assumed that cuts in F would not occur until fall 2009, so biomass trajectory during 2008 was based on a continuation of 2007 conditions. Cuts were instituted in 2009 (F drops by a maximum of half to mimic regulations imposed by fall) and maintained until 2015. $F_{recover}$ was estimated as

$$\mathbf{(0.2 / 0.31)\ *\ Fmsy}; \qquad (17)$$

where 0.2 equaled the maximum F allowed for rebuilding under Amendment 4 and 0.31 equaled target. F_{msy} was represented by the median of the bootstrap and jackknife trials (0.48), therefore $F_{recover}$ was approximated as 0.31. Losses under moratorium conditions (U_m) during 2010-2015 were approximated as

$$\mathbf{(1-(H_w / H_t))\ *\ U_{2006}}; \qquad (18)$$

where H_w = losses of weakfish from all harvest and weakfish commercial fishery discards in 2006; H_t total losses of weakfish in all fisheries, and U_{2006} was estimated exploitation rate in 2006. U_m was approximated as 0.056. The probability of B_{2015} reaching 15,000 mt was estimated separately for jackknife or bootstrap-based projections.

Stock recovery was not possible under $F_{recover}$ and F = 0.2, and there was about 1.4% chance of recovery under moratorium conditions for bootstrap runs and 0% chance for jackknife moratorium scenarios. It should be noted that estimates of F since 2003 have been below or at $F_{recover}$. Prospects for weakfish are grim under these projections; jackknifing and bootstrapping indicated 98% and 91% chances (respectively) that weakfish biomass would fall to zero by 2010 if trophic conditions prevailing in 2006 continued. Instances where extirpation did not occur were associated with estimates of B_{2007} in excess of 4,600 mt. If high predation-competition losses estimated for 2006 prevail, a moratorium cannot be initiated in time according to the great majority of trials. By 2015, extirpation becomes a virtual certainty even under a moratorium

C14.0 Make research recommendations for improving data collection and assessment. (TOR #10)

The following list of prioritized research needs updates the list provided in the 2008 Weakfish Fishery Management Plan Review. New research recommendations identified by the WTC are presented in ALLCAPS. Comments regarding existing recommendations are shown in *italics*.

High Priority

INVESTIGATE ALTERNATIVE AGE BASED MODELS WHICH ALLOW ERROR IN CATCH AT AGE ESTIMATES (*E.G.* STATISTICAL CATCH AT AGE) AND/OR ARE LESS PRONE TO RETROSPECTIVE PATTERNS (*E.G.* EXTENDED SURVIVORS ANALYSIS).

EVALUATE CONSUMPTION OF WEAKFISH BY PREDATORS USING A MORE ADVANCED MULTISPECIES MODEL, SUCH AS THE ASMFC MSVPA MODEL OR ECOPATH WITH ECOSIM, TO VALIDATE ESTIMATES CALCULATED BY PRODUCTION MODELS WITH PREDATION-COMPETITION EXTENSIONS.

DEVELOP A BIOENERGETICS MODEL FOR WEAKFISH THAT ENCOMPASSES A BROADER RANGE OF AGES THAN HARTMAN AND BRANDT (1995). USE IT TO EVALUATE DIET AND GROWTH DATA.

INITIATE MONITORING OF WEAKFISH, STRIPED BASS, AND SPINY DOGFISH DIETS OVER A BROAD REGIONAL AND SPATIAL SCALE.

Collect catch and effort data including size and age composition of the catch, determine stock mortality throughout the range, and define gear characteristics. In particular, increase length-frequency sampling, particularly in fisheries from Maryland and further north.

Derive alternative estimates of discard mortality rates and the magnitude of discards for all commercial gear types from both directed and non-directed fisheries. In particular, quantify trawl bycatch, refine estimates of mortality for below minimum size fish, and focus on factors such as

distance from shore and geographical differences. Improved estimates of discard mortality would best be obtained through increased observer coverage.

Conduct an age validation study. *This work has been completed in Lowerre-Barbieri et al (1995)*.

Identify stocks and determine coastal movements and the extent of stock mixing, including characterization of stocks in over-wintering grounds. Most direct method would be to develop a coastwide tagging program. *Otolith samples have been obtained by Old Dominion University, but funding has not been available for processing.*

Conduct spatial and temporal analysis of the fishery independent survey data. The analysis should assess the impact of the variability of the surveys in regards to gear, time of year and geographic coverage on their (survey) use as stock indicators. *Research is ongoing by Dr. Yan Jiao of Virginia Tech. University. See Winter et al 2009 in the supplementary material for this peer review.*

Analyze the spawner-recruit relationship and examine the relationships between parental stock size and environmental factors on year-class strength. *Work is currently underway by Weakfish Stock Assessment Subcommittee*.

Develop latitudinal/seasonal/gear specific age length keys for the Atlantic coast. Increase sample sizes to consider gear specific keys.

Medium Priority
Examine geographic and temporal differences in growth rate (length and weight at age).

Define reproductive biology of weakfish, including size at sexual maturity, maturity schedules, fecundity, and spawning periodicity. Continue research on female spawning patterns: what is the seasonal and geographical extent of "batch" spawning; do females exhibit spawning site fidelity? *This work has been completed by Nye et al 2008 and Lowerre-Barbieri et al 1996*.

Compile data on larval and juvenile distribution from existing databases in order to obtain preliminary indications of spawning and nursery habitat location and extent.

Conduct hydrophonic studies to delineate weakfish spawning habitat locations and environmental preferences (temperature, depth, substrate, etc.) and enable quantification of spawning habitat.

Continue studies on mesh-size selectivity; up-to-date (1995) information is available only for North Carolina's gill net fishery. Mesh-size selectivity studies for trawl fisheries are particularly sparse. *Gillnet selectivity has been investigated by Swihart et al (2000). Can also be obtained from the NC PSIGNS survey*.

Assemble socio-demographic-economic data as it becomes available from ACCSP.

Continue studies on recreational hook-and-release mortality rates, including factors such as depth, warmer water temperatures, and fish size in the analysis. Studies are needed in deep and warm water conditions. Further consideration of release mortality in both the recreational and

commercial fisheries is needed, and methods investigated to improve survival among released fish.

Low Priority

Define restrictions necessary for implementation of projects in spawning and over-wintering areas and develop policies on limiting development projects seasonally or spatially.

Document the impact of power plants and other water intakes on larval, post larval and juvenile weakfish mortality in spawning and nursery areas, and calculate the resultant impact to adult stock size. *Data are available for power plants in the Delaware Bay area and North Carolina. Data should be compiled and evaluated.*

Determine the onshore versus offshore components of the weakfish fishery.

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Tables

Table C4.6-1. Commercial landings (MT) and percent of annual total by state and year. A) Landings (MT); B) Percent of total.

A)

Year	MA	RI	CT	NY	NJ	DE	MD	VA	NC	FL	Total
1981	18.1	109.8	12.4	615.9	1,701.1	477.0	153.5	1,121.2	7,662.9	86.3	11,958.2
1982	10.4	80.2	11.6	570.2	940.5	587.2	113.0	974.9	5,466.9	79.9	8,834.8
1983	3.1	74.3	19.4	385.6	985.5	409.1	176.9	1,176.1	4,642.0	53.4	7,925.4
1984	2.2	76.0	14.2	219.8	1,248.1	354.9	147.4	956.6	5,892.6	57.1	8,968.9
1985	1.4	74.0	12.8	175.2	1,374.4	449.4	143.4	944.5	4,454.9	60.0	7,690.0
1986	2.6	57.9	6.2	163.2	1,455.4	328.2	152.7	904.5	6,490.7	49.3	9,610.7
1987	0.8	35.7	13.4	149.3	949.9	262.1	166.4	890.3	5,220.2	55.8	7,743.9
1988	1.7	8.8	1.1	56.5	1,058.2	240.7	377.7	668.2	6,845.6	52.2	9,310.7
1989	0.9	4.4	1.0	46.9	661.6	240.5	337.4	465.0	4,588.5	77.7	6,423.9
1990	0.8	11.2	0.6	9.0	439.2	278.1	300.4	547.7	2,631.8	62.2	4,281.0
1991	0.9	11.3	9.7	50.6	532.6	225.6	148.9	480.7	2,408.0	74.8	3,943.1
1992	1.4	13.7	1.6	76.2	426.7	164.4	174.8	249.5	2,205.6	67.1	3,381.0
1993	0.5	4.5	0.7	40.1	378.5	88.3	82.5	493.5	1,954.7	65.5	3,108.8
1994		8.2	5.0	45.1	315.4	118.8	63.9	587.1	1,583.0	81.5	2,808.0
1995	0.2	23.9	2.9	78.2	393.4	127.6	31.5	673.6	1,865.8	22.8	3,219.9
1996	0.0	19.7	3.1	165.7	372.9		60.2	719.9	1,804.3	2.0	3,147.8
1997	0.0	14.1	5.0	152.7	470.1	253.5	87.4	706.7	1,615.3	5.3	3,310.1
1998	0.2	35.0	6.6	225.2	818.6	250.7	110.9	845.5	1,521.4	6.8	3,820.9
1999	1.2	57.3	10.1	222.2	585.7	199.7	101.4	759.3	1,187.3	7.9	3,132.1
2000	0.2	85.9	3.6	160.0	486.0	149.1	94.5	618.2	847.8	4.3	2,449.6
2001	0.1	49.7	3.1	262.5	379.9	85.1	84.3	508.9	889.2	4.9	2,267.7
2002	0.4	55.7	4.6	233.1	391.5	78.4	50.5	518.9	829.3	2.6	2,165.0
2003	0.2	28.7	1.4	65.5	154.3	41.5	21.5	208.4	385.0	1.2	907.7
2004	0.0	17.4	2.8	80.9	92.8	23.3	19.8	161.9	310.9	1.2	711.0
2005		18.9	2.8	49.8	29.2	32.1	16.2	176.9	191.2	3.3	520.4
2006	3.9	20.2	3.2	69.3	93.7	15.6	23.2	85.2	164.6	2.7	481.6
2007	0.2	9.3	0.9	39.3	74.6	11.1	12.6	156.7	79.6	3.5	387.8

Table C4.6-1 (continued). Commercial landings (MT) and percent of annual total by state and year. A) Landings (MT); B) Percent of total.

B)	Year	MA	RI	CT	NY	NJ	DE	MD	VA	NC	FL
	1981	0.15	0.92	0.1	5.15	14.23	3.99	1.28	9.38	64.08	0.72
	1982	0.12	0.91	0.13	6.45	10.65	6.65	1.28	11.03	61.88	0.9
	1983	0.04	0.94	0.24	4.87	12.43	5.16	2.23	14.84	58.57	0.67
	1984	0.02	0.85	0.16	2.45	13.92	3.96	1.64	10.67	65.7	0.64
	1985	0.02	0.96	0.17	2.28	17.87	5.84	1.86	12.28	57.93	0.78
	1986	0.03	0.6	0.06	1.7	15.14	3.41	1.59	9.41	67.54	0.51
	1987	0.01	0.46	0.17	1.93	12.27	3.38	2.15	11.5	67.41	0.72
	1988	0.02	0.09	0.01	0.61	11.37	2.59	4.06	7.18	73.52	0.56
	1989	0.01	0.07	0.02	0.73	10.3	3.74	5.25	7.24	71.43	1.21
	1990	0.02	0.26	0.01	0.21	10.26	6.5	7.02	12.79	61.48	1.45
	1991	0.02	0.29	0.25	1.28	13.51	5.72	3.78	12.19	61.07	1.9
	1992	0.04	0.41	0.05	2.25	12.62	4.86	5.17	7.38	65.24	1.98
	1993	0.02	0.14	0.02	1.29	12.18	2.84	2.65	15.87	62.88	2.11
	1994	0	0.29	0.18	1.61	11.23	4.23	2.28	20.91	56.37	2.9
	1995	0.01	0.74	0.09	2.43	12.22	3.96	0.98	20.92	57.95	0.71
	1996	0	0.63	0.1	5.26	11.85	0	1.91	22.87	57.32	0.06
	1997	0	0.43	0.15	4.61	14.2	7.66	2.64	21.35	48.8	0.16
	1998	0.01	0.92	0.17	5.89	21.42	6.56	2.9	22.13	39.82	0.18
	1999	0.04	1.83	0.32	7.09	18.7	6.38	3.24	24.24	37.91	0.25
	2000	0.01	3.51	0.15	6.53	19.84	6.09	3.86	25.24	34.61	0.18
	2001	0	2.19	0.14	11.58	16.75	3.75	3.72	22.44	39.21	0.22
	2002	0.02	2.57	0.21	10.77	18.08	3.62	2.33	23.97	38.3	0.12
	2003	0.02	3.16	0.15	7.22	17	4.57	2.37	22.96	42.41	0.13
	2004	0	2.45	0.39	11.38	13.05	3.28	2.78	22.77	43.73	0.17
	2005	0	3.63	0.54	9.57	5.61	6.17	3.11	33.99	36.74	0.63
	2006	0.81	4.19	0.66	14.39	19.46	3.24	4.82	17.69	34.18	0.56
	2007	0.05	2.4	0.23	10.13	19.24	2.86	3.25	40.41	20.53	0.9

Table C4.6-2. Estimated commercial discards of weakfish by year.

		Number at age (thousands)							Total N
Year	MT	0	1	2	3	4	5	6+	(thousands)
1982	604.54	1,231.2	2,020.8	1,173.3	54.4	4.2	1.4	0.3	4,485.5
1983	417.29	338.2	1,573.2	769.2	120.3	4.6	1.1	0.2	2,806.7
1984	681.34	1,431.7	2,911.0	845.0	83.8	5.6	1.3	0.3	5,278.7
1985	386.55	664.1	1,363.0	572.2	112.6	5.6	1.3	0.3	2,719.2
1986	431.77	339.1	1,578.3	778.0	127.1	7.7	1.8	0.3	2,832.4
1987	364.28	226.6	1,280.0	809.8	93.3	6.9	1.7	0.4	2,418.5
1988	326.02	66.5	1,209.9	698.0	89.8	34.3	8.2	1.5	2,108.3
1989	311.71	434.5	1,459.2	508.9	27.0	4.7	1.1	0.2	2,435.7
1990	190.16	107.7	615.1	353.6	81.4	4.6	1.0	0.2	1,163.6
1991	200.09	112.8	692.2	365.5	64.1	19.0	4.4	0.9	1,259.0
1992	216.62	280.0	873.6	364.0	35.2	3.3	0.8	0.2	1,557.1
1993	416.35	126.9	1,205.4	586.8	53.9	5.5	1.2	0.2	1,980.0
1994	989.57	219.6	378.5	934.1	355.7	127.7	3.0	0.9	2,019.4
1995	715.83	810.6	576.6	421.3	125.0	42.3	1.3	0.5	1,977.7
1996	985.21	1,026.3	643.9	403.4	608.1	138.3	28.9	1.4	2,850.4
1997	864.41	81.8	488.9	550.9	229.7	318.1	35.3	8.6	1,713.3
1998	762.07	262.1	300.5	296.3	309.3	79.7	70.2	26.9	1,345.1
1999	746.42	2,347.4	416.1	104.1	123.7	183.6	15.9	15.1	3,205.9
2000	548.36	0.6	249.3	618.2	465.0	133.0	40.4	6.0	1,512.6
2001	1,138.70	1,153.2	2,124.5	734.4	305.0	297.4	104.1	34.7	4,753.5
2002	470.07	1,310.0	2,093.7	456.0	155.0	25.6	5.8	2.4	4,048.6
2003	238.22	2.9	19.6	157.6	127.5	34.4	8.2	3.7	353.9
2004	209.44	17.2	98.5	276.6	110.9	5.5	4.6	2.5	515.9
2005	173.91	0.6	33.6	307.3	202.0	24.2	0.4	0.2	568.2
2006	199.00	54.5	230.1	187.8	245.5	30.9	0.4	0.0	749.1
2007	193.27	43.1	288.7	548.4	254.3	16.3	3.3	0.0	1,154.1

Table C4.6-3. Recreational harvest and percent of annual total by state and year. A) Harvest (thousands of fish); B) Percent of total. *Florida values have been corrected for sand seatrout and weakfish-sand seatrout hybrids.

A)

Year	MA	RI	CT	NY	NJ	DE	MD	VA	NC	SC	GA	FL*	Total
1981	5.95	18.37	18.71	275.12	1,028.79	122.74	177.76	7,484.78	204.23	2.58	2.43	0.00	9,341.46
1982	0.00	18.61	11.77	88.23	104.07	217.82	440.15	715.89	200.05	17.34	0.00	11.65	1,825.58
1983	2.73	74.61	6.36	36.93	2,857.09	1,009.90	595.29	354.85	387.87	6.81	17.21	69.64	5,419.29
1984	2.24	0.00	1.56	20.13	1,026.04	593.11	104.06	782.85	489.47	7.84	0.00	103.34	3,130.63
1985	0.00	17.09	2.87	89.54	812.84	365.69	305.80	505.22	217.67	61.79	4.81	8.92	2,392.24
1986	0.00	4.60	7.32	34.58	2,500.62	914.49	1,947.39	2,418.05	611.36	78.32	18.13	27.16	8,562.01
1987	0.00	0.00	0.78	7.45	1,666.62	638.34	824.88	1,015.41	624.16	18.84	10.80	13.58	4,820.87
1988	0.00	0.00	0.00	13.22	642.03	974.71	1,163.77	2,297.05	438.15	1.83	0.00	20.92	5,551.68
1989	0.00	0.00	0.00	6.44	303.29	254.17	226.51	357.86	190.19	6.81	8.25	30.08	1,383.59
1990	0.00	0.41	0.00	3.06	216.39	179.84	370.53	286.46	91.30	8.03	2.27	18.54	1,176.81
1991	0.00	0.00	18.70	28.07	545.67	366.46	221.24	351.95	140.83	19.62	4.95	24.97	1,722.45
1992	0.00	9.62	0.43	5.28	311.66	100.56	137.26	265.65	35.49	23.50	1.75	14.71	905.91
1993	0.00	0.00	2.46	12.61	203.92	235.31	238.77	108.39	106.74	7.36	14.75	31.57	961.88
1994	0.00	0.00	0.00	1.87	591.57	300.21	332.85	169.74	177.97	46.86	0.72	46.23	1,668.01
1995	0.00	1.57	0.00	22.31	671.85	406.73	88.70	226.68	62.48	29.90	22.44	11.95	1,544.60
1996	0.00	0.00	0.00	16.32	1,104.25	633.92	183.41	193.86	90.70	5.70	5.41	7.55	2,241.13
1997	0.00	1.42	0.52	112.99	1,028.33	647.53	162.90	557.81	184.95	2.04	44.20	18.29	2,760.97
1998	0.62	0.00	2.18	21.39	920.56	455.60	290.05	463.53	191.18	15.84	0.72	6.44	2,368.11
1999	0.00	2.30	1.61	18.35	583.88	224.31	340.10	229.21	127.16	3.94	1.68	26.18	1,558.71
2000	0.00	0.71	7.34	42.41	760.28	311.55	475.35	286.75	71.25	5.59	4.18	30.28	1,995.68
2001	0.00	2.30	0.72	28.13	736.07	72.45	302.72	175.87	158.61	0.00	3.32	11.14	1,491.32
2002	0.00	1.42	1.80	24.96	492.88	121.88	100.47	178.11	90.17	90.25	0.85	16.67	1,119.45
2003	0.11	0.30	0.44	9.23	151.10	20.12	41.05	86.11	153.75	4.16	1.57	6.28	474.24
2004	0.00	0.00	0.00	7.60	183.65	6.97	29.65	103.18	237.40	153.59	9.82	10.50	742.34
2005	0.00	1.01	0.00	0.36	1,053.01	19.03	22.16	30.35	163.27	129.58	5.76	18.28	1,442.80
2006	0.00	3.30	0.00	9.12	417.53	11.16	0.47	58.81	153.70	7.12	3.50	19.62	684.33
2007	0.00	0.00	0.00	7.12	209.31	4.18	10.32	44.49	114.33	71.23	4.71	26.05	491.74

Table C4.6-3 (continued). Recreational harvest and percent of annual total by state and year. A) Harvest (thousands of fish); B) Percent of total. *Florida values have been corrected for sand seatrout and weakfish-sand seatrout hybrids.

B)

Year	MA	RI	CT	NY	NJ	DE	MD	VA	NC	SC	GA	FL*
1981	0.06	0.20	0.20	2.95	11.01	1.31	1.90	80.12	2.19	0.03	0.03	0.00
1982	0.00	1.02	0.64	4.83	5.70	11.93	24.11	39.21	10.96	0.95	0.00	0.64
1983	0.05	1.38	0.12	0.68	52.72	18.64	10.98	6.55	7.16	0.13	0.32	1.29
1984	0.07	0.00	0.05	0.64	32.77	18.95	3.32	25.01	15.63	0.25	0.00	3.30
1985	0.00	0.71	0.12	3.74	33.98	15.29	12.78	21.12	9.10	2.58	0.20	0.37
1986	0.00	0.05	0.09	0.40	29.21	10.68	22.74	28.24	7.14	0.91	0.21	0.32
1987	0.00	0.00	0.02	0.15	34.57	13.24	17.11	21.06	12.95	0.39	0.22	0.28
1988	0.00	0.00	0.00	0.24	11.56	17.56	20.96	41.38	7.89	0.03	0.00	0.38
1989	0.00	0.00	0.00	0.47	21.92	18.37	16.37	25.86	13.75	0.49	0.60	2.17
1990	0.00	0.03	0.00	0.26	18.39	15.28	31.49	24.34	7.76	0.68	0.19	1.58
1991	0.00	0.00	1.09	1.63	31.68	21.28	12.84	20.43	8.18	1.14	0.29	1.45
1992	0.00	1.06	0.05	0.58	34.40	11.10	15.15	29.32	3.92	2.59	0.19	1.62
1993	0.00	0.00	0.26	1.31	21.20	24.46	24.82	11.27	11.10	0.77	1.53	3.28
1994	0.00	0.00	0.00	0.11	35.47	18.00	19.95	10.18	10.67	2.81	0.04	2.77
1995	0.00	0.10	0.00	1.44	43.50	26.33	5.74	14.68	4.04	1.94	1.45	0.77
1996	0.00	0.00	0.00	0.73	49.27	28.29	8.18	8.65	4.05	0.25	0.24	0.34
1997	0.00	0.05	0.02	4.09	37.25	23.45	5.90	20.20	6.70	0.07	1.60	0.66
1998	0.03	0.00	0.09	0.90	38.87	19.24	12.25	19.57	8.07	0.67	0.03	0.27
1999	0.00	0.15	0.10	1.18	37.46	14.39	21.82	14.71	8.16	0.25	0.11	1.68
2000	0.00	0.04	0.37	2.12	38.10	15.61	23.82	14.37	3.57	0.28	0.21	1.52
2001	0.00	0.15	0.05	1.89	49.36	4.86	20.30	11.79	10.64	0.00	0.22	0.75
2002	0.00	0.13	0.16	2.23	44.03	10.89	8.97	15.91	8.05	8.06	0.08	1.49
2003	0.02	0.06	0.09	1.95	31.86	4.24	8.66	18.16	32.42	0.88	0.33	1.33
2004	0.00	0.00	0.00	1.02	24.74	0.94	3.99	13.90	31.98	20.69	1.32	1.41
2005	0.00	0.07	0.00	0.02	72.98	1.32	1.54	2.10	11.32	8.98	0.40	1.27
2006	0.00	0.48	0.00	1.33	61.01	1.63	0.07	8.59	22.46	1.04	0.51	2.87
2007	0.00	0.00	0.00	1.45	42.56	0.85	2.10	9.05	23.25	14.49	0.96	5.30

Table C5.1-1. Sample size and parameter estimates for weakfish length-weight equations.

Region	Year	Season	N	a	b
North	2004	Early	1,553	1.593E-08	3.0510
North	2004	Late	2,290	3.612E-08	2.9091
North	2005	Early	810	1.599E-08	3.0394
North	2005	Late	3,466	8.150E-08	2.7815
North	2006	Early	1,031	1.472E-08	3.0707
North	2006	Late	3,112	7.565E-08	2.8010
North	2007	Early	1,631	5.396E-08	2.8649
North	2007	Late	3,889	4.376E-08	2.8852
South	2004	Early	568	4.386E-08	2.8971
South	2004	Late	596	2.991E-08	2.9554
South	2005	Early	480	2.309E-08	2.9913
South	2005	Late	574	2.889E-08	2.9510
South	2006	Early	488	1.538E-08	3.0586
South	2006	Late	475	3.604E-08	2.9178
South	2007	Early	298	1.648E-08	3.0575
South	2007	Late	266	4.537E-08	2.8882

Table C5.1-2. Commercial biological sample substitution matrix.

	2004 Early	2004 Late	2005 Early	2005 Late	2006 Early	2006 Late	2007 Early	2007 Late
MA Trawl	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	NY Trawl PLUS MD and NJ Trawl 16+	Combined in "Other"	Combined in "Other"
MA Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
RI Float trap	2003 RI Float trap E and L combined	2003 RI Float trap E and L combined	Combined in "Other"	Direct	Combined in "Other"	Direct	Combined in "Other"	Combined in "Other"
RI Trawl	VA Pound 16+	MD Trawl 16+ AND VA Pound 16+	VA 2006 Early Pound 16+	MD 2005 Late Trawl 16+	NJ Pound 16+	NY Trawl PLUS MD and NJ Trawl 16+	NJ Trawl 16+	NJ Trawl PLUS pound 16+
RI Gill	Combined in "Other"	VA Gill 16+ AND 2005 VA Gill 16+	Combined in "Other"	VA Gill 16+	Combined in "Other"	NJ Gill 16+	Combined in "Other	NY Gill 16+
RI Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
CT Trawl	Combined in "Other"	MD Trawl 16+ AND VA Pound 16+	Combined in "Other"	Combined in "Other"	NJ Pound 16+	Combined in "Other"	Combined in "Other"	Combined in "Other"
CT Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
NY Gill	VA Gill 16+	VA Gill 16+ AND 2005 VA Gill 16+	VA Gill 16+	NY Gill PLUS VA Gill 16+	Direct	Direct	Combined in "Other"	Direct
NY Trawl	VA Pound 16+	MD Trawl 16+ AND VA Pound 16+	VA 2006 Early Pound 16+	Direct	NJ Pound 16+	NY Trawl PLUS MD and NJ Trawl 16+	Combined in "Other"	NJ 2007 Late Trawl PLUS Pound 16+
NY Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
NJ Gill	VA Gill 13+	VA Gill 13+	VA Gill 13+	MD Gill 13+	Direct	Direct	Direct	Direct
NJ Pound	Combined in "Other"	Combined in "Other"	VA Pound 13+	Combined in "Other"	Direct	Direct	Direct	Direct
NJ Trawl	VA Pound 13+	MD Trawl 12+	Combined in "Other"	MD Trawl 12+	NJ Pound 13+	Direct	Direct	Direct
NJ Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
DE Gill	Direct	VA Gill 12+	Direct	VA Gill 12+	Direct	VA Gill 12+	DE 2006 Early Gill	Combined in "Other"
DE Hand	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
DE Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"

Table C5.1-2 (continued). Commercial biological sample substitution matrix.

MD Gill	Combined in "Other"	Combined in "Other"	Combined in "Other"	Direct	Combined in "Other"	Direct	Combined in "Other"	Direct
MD Pound	Combined in "Other"	MD Pound PLUS VA Pound 12+	Combined in "Other"	Direct	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
MD Trawl	Combined in "Other"	Direct	Combined in "Other"	Direct	Combined in "Other"	Direct	Combined in "Other"	Direct
MD Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
VA Gill	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
VA Haul sne	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
VA Pound	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
VA Other	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"	Combined in "Other"
NC Bch sne	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
NC Est GN	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
NC LH Sne	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
NC Sink gill	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
NC Pound	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
NC Trawl	Direct	Direct	Direct	Direct	Direct	Direct	Direct	Direct
NC Other	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data
SC Other	No harvest	No harvest	No harvest	No harvest	No harvest	No harvest	No harvest	No harvest
GA Other	No harvest	No harvest	No harvest	No harvest	No harvest	No harvest	No harvest	No harvest
FL Other	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data	Combined NC data

Table C5.1-3. Biological sampling intensity by state and season for recent years.

			AGES			LENGTHS		
Year	Season	Source2	Comm	Recr	Fish Ind	Comm	Recr	Fish Ind
2004	Early	MA						
		RI						
		CT						
		NY					1	
		NJ	11				6	
		DE	46			782		
		MD	12					
		VA	354			1408	126	
		NC	353			5129	31	
		SC					11	
		GA						
		FL					15	
		NEFSC						
		CHESMAP			263			271
		NEAMAP						
		SEAMAP			246			246
2004	Early Total		776		509	7319	190	517
2004	Late	MA						
		RI	4					
		CT						
		NY					4	
		NJ	46				61	
		DE			533		13	533
		MD	136			105	44	
		VA	55			1848	45	
		NC	295			3312	142	
		SC					38	
		GA					15	
		FL					14	
		NEFSC			476			563
		CHESMAP			811			837
		NEAMAP						
		SEAMAP			316			316
2004	Late Total		532		2136	5265	376	2249
2005	Early	MA						
		RI					1	
		CT						
		NY						
		NJ	13				2	
		DE	43			573	12	
		MD	18					
		VA	217			1000	294	
		NC	295			5529	37	
		SC						
		GA					25	
		FL					13	
		NEFSC						
		CHESMAP			99			99
		NEAMAP						
		SEAMAP			185			185
2005	Early Total		586		284	7102	383	284

Table C5.1-3 (continued). Biological sampling intensity by state and season for recent years.

			AGES			LENGTHS		
Year	Season	Source2	Comm	Recr	Fish Ind	Comm	Recr	Fish Ind
2005	Late	MA						
		RI	60			59		
		CT						
		NY				400	6	
		NJ	135				131	
		DE			601		29	601
		MD	260			301	30	
		VA	244			3046	51	
		NC	293			3266	117	
		SC					31	
		GA					4	
		FL					25	
		NEFSC			594			648
		CHESMAP			1005			1020
		NEAMAP						
		SEAMAP			286			285
2005	Late Total		932		2486	7013	424	2554
2006	Early	MA						
		RI	5					
		CT						
		NY					1	
		NJ	43			350	17	
		DE	79			117	5	
		MD						
		VA	361			1738	51	
		NC	800			6337	95	
		SC					3	
		GA						
		FL					37	
		NEFSC						
		CHESMAP			167			171
		NEAMAP						
		SEAMAP			120			121
2006	Early Total		1283		287	8542	209	292
2006	Late	MA						
		RI	38			38	3	
		CT						
		NY				41	4	
		NJ	257			380	69	
		DE			481		11	481
		MD	180			176		
		VA	253			3540	8	
		NC	696			2246	149	
		SC					75	
		GA					1	
		FL					13	
		NEFSC			1171			1198
		CHESMAP			550			557
		NEAMAP			494			494
		SEAMAP			197			197
2006	Late Total		1386		2893	6383	330	2927

Table C5.1-3 (continued). Biological sampling intensity by state and season for recent years.

			AGES			LENGTHS		
Year	Season	Source2	Comm	Recr	Fish Ind	Comm	Recr	Fish Ind
2007	Early	MA						
		RI						
		CT						
		NY						
		NJ	350			370	8	
		DE	159				2	
		MD						
		VA	280			997	4	
		NC	298			3455	14	
		SC					25	
		GA					4	
		FL					11	
		NEFSC						
		CHESMAP			120			120
		NEAMAP						
		SEAMAP						
2007	Early Total		1087		120	4822	68	120
2007	Late	MA						
		RI						
		CT						
		NY				61		
		NJ	193			183	30	
		DE			446		6	446
		MD	276			204	7	
		VA	142			1831	5	
		NC	270			2653	65	
		SC					150	
		GA					5	
		FL					27	
		NEFSC			682			748
		CHESMAP			434			434
		NEAMAP			564			572
		SEAMAP						
2007	Late Total		881		2126	4932	295	2200

Table C5.1-4. Number of observed commercial hauls with weakfish discards by year and gear type.

Year	Gillnet	Otter trawl	Other	Total
1994	92	19		111
1995	158	140		298
1996	99	137	1	236
1997	63	33	1	96
1998	68	5		73
1999	26	29		55
2000	41	12		53
2001	26	53		79
2002	15	42		57
2003	6	53		59
2004	7	110		117
2005	4	34		38
2006	1	35	1	36
2007	5	77	1	82
Total	611	779	4	1390

Table C5.2-1. Coastwide recreational weakfish harvest and discards and percent standard error (PSE). Estimates from Florida are not corrected for weakfish/sand seatrout hybrids.

Year	Harvest (A+B1) Numbers	NumPSE	Harvest (A+B1) Pounds	LbsPSE	Discard (B2) Numbers	DiscPSE
1981	9,344,461	35.4	16,105,028	15.7	284,848	49.2
1982	1,854,090	14.8	8,285,326	15.3	190,580	48.8
1983	5,642,950	12.8	11,730,619	12.3	273,319	35.3
1984	3,520,811	13.7	7,013,781	21.1	248,229	27.3
1985	2,419,670	9.4	5,489,026	10.6	356,154	34.9
1986	8,664,122	8.3	10,141,786	9.2	2,309,464	16.4
1987	4,871,532	11	6,749,890	14.3	848,274	18.9
1988	5,626,268	11.9	6,331,649	11.2	820,115	34.6
1989	1,495,391	8	2,177,237	7.7	179,177	16.6
1990	1,232,253	6.8	1,347,260	8.1	439,555	12.3
1991	1,812,691	6.8	2,130,563	7.4	788,789	17.7
1992	960,151	7.5	1,398,980	8.5	707,658	12.6
1993	1,079,275	7.3	1,102,340	9.3	1,139,284	10.2
1994	1,826,495	7.7	1,795,517	9.5	3,102,455	7.1
1995	1,588,079	8	1,855,548	8.7	4,108,344	6.9
1996	2,269,330	8.4	2,925,392	10.5	5,036,968	7.1
1997	2,815,654	7.6	3,692,716	7.5	4,016,709	5.8
1998	2,386,345	6.7	4,044,974	7.6	3,311,050	7
1999	1,651,554	7	3,143,427	8	2,826,435	6.6
2000	2,089,202	7.6	4,154,794	9.2	4,870,876	5.8
2001	1,526,583	7.1	2,722,630	7.9	3,708,952	5.8
2002	1,171,889	9.2	2,192,607	8.1	2,117,876	6.7
2003	497,571	8.9	864,962	11.5	1,600,485	8.5
2004	777,856	11.8	926,962	12.4	1,888,567	9.1
2005	1,503,540	10.9	1,587,378	11.5	2,344,871	10
2006	745,135	11.2	919,662	14.1	2,416,228	9.9
2007	584,569	14.4	692,392	15.8	1,427,669	13.7

Table C5.3-1. Weakfish catch at age matrix (thousands of fish) for all four fishery sectors combined.

	Age					
Year	1	2	3	4	5	6+
1982	9,914.20	12,967.00	5,473.00	2,778.20	721.60	639.50
1983	8,004.00	12,869.10	5,822.70	2,780.00	568.20	424.10
1984	10,444.20	14,736.90	6,521.10	3,045.30	484.50	254.50
1985	14,153.20	11,262.30	3,246.10	1,171.00	212.90	55.10
1986	18,610.70	15,778.40	4,942.40	1,823.70	264.10	52.10
1987	16,256.30	14,343.10	4,347.10	1,485.20	145.40	11.00
1988	8,161.90	16,140.80	10,545.30	6,092.00	1,050.50	70.70
1989	3,705.00	5,304.90	4,333.50	2,922.30	626.20	84.60
1990	9,510.10	4,890.10	2,093.60	1,204.80	591.40	89.10
1991	9,795.90	5,825.60	2,750.00	1,373.60	463.40	57.30
1992	5,179.50	6,046.00	2,211.00	1,255.00	527.80	65.00
1993	4,974.80	6,357.00	2,179.80	1,138.60	401.10	48.20
1994	3,761.88	4,347.41	3,561.04	1,563.46	204.05	39.79
1995	4,336.27	3,727.71	3,566.71	1,637.76	198.10	54.28
1996	2,498.83	2,689.49	5,033.34	3,174.23	1,379.29	100.14
1997	1,716.38	2,394.16	2,913.23	5,522.01	1,523.11	410.19
1998	1,270.62	2,138.27	3,983.12	2,019.16	2,928.80	909.47
1999	1,412.62	1,300.41	2,256.60	3,326.01	725.65	1,145.02
2000	1,376.99	1,727.14	1,985.75	1,663.66	1,528.22	403.05
2001	2,420.66	2,953.08	1,474.09	1,219.89	658.73	485.92
2002	2,591.74	1,070.50	2,695.67	823.88	388.16	231.49
2003	335.59	949.93	959.71	718.40	209.46	254.18
2004	852.25	1,511.95	667.85	115.80	49.72	38.35
2005	334.26	1,771.52	1,255.15	191.46	10.19	27.12
2006	747.26	637.33	959.20	252.90	15.49	11.94
2007	616.61	1,148.02	507.63	135.20	25.23	5.78

Table C6.2-1. Young of year indices of abundance.

Year	MA CPUE	MA CV	RI CPUE	RI RelCV	CT 0 CPUE	CT 0 RelCV	CT Fall 1+ CPUE	CT Fall 1+ RelCV	CT Spring 1+ CPUE	NY CPUE	NY 95% CI
1981	1.87	90.65	5.04	12.71							
1982	0.77	45.59	4.11	13.23							
1983	1.14	67.91	0.79	21.39							
1984	0.15	85.24	0.38	39.39	1.00	0.17	0.53	0.18	0.02		
1985	15.19	100.00	2.37	16.38	6.19	0.10	0.24	0.23	0.00		
1986	2.67	85.00	0.70	21.73	13.16	0.09	0.24	0.24	0.10		
1987	0.00	0.00	0.33	37.25	0.63	0.22	0.11	0.36	0.02	0.6	0.40 - 0.82
1988	0.00	0.00	0.90	29.22	3.49	0.12	0.06	0.41	0.05	0.1	0.05 - 0.17
1989	1.40	94.95	0.42	34.23	8.69	0.11	0.02	0.70	0.04	1.4	0.91 - 1.96
1990	0.00	0.00	2.45	21.14	5.56	0.10	0.08	0.34	0.07	0.6	0.34 - 0.80
1991	0.07	63.83	1.66	26.62	11.95	0.08	0.31	0.28	0.28	20.6	15.30 - 27.73
1992	0.02	100.00	2.35	20.56	3.05	0.12	0.18	0.30	0.12	3.3	2.17 - 4.73
1993	0.00	0.00	1.30	27.13	4.08	0.09	0.12	0.28	0.10	1.0	0.68 - 1.46
1994	0.16	86.60	1.57	26.78	11.19	0.08	0.06	0.41	0.04	8.3	5.53 - 12.35
1995	0.63	66.44	0.09	60.52	5.22	0.12	0.70	0.18	0.18	1.6	1.16 - 2.14
1996	0.26	92.32	6.34	20.24	15.23	0.08	0.56	0.21	0.19	24.5	15.72 - 37.86
1997	0.57	75.87	6.69	16.08	12.38	0.08	0.89	0.16	0.42	18.8	12.97 - 26.91
1998	0.28	80.00	1.39	24.91	5.02	0.12	0.28	0.29	0.37	1.0	0.69 - 1.44
1999	0.32	53.45	1.00	23.98	30.93	0.07	0.39	0.22	0.45	8.4	5.87 - 11.94
2000	1.10	69.55	2.10	19.30	63.31	0.07	0.30	0.26	0.18	15.9	10.33 - 24.13
2001	1.11	74.24	2.39	21.41	40.09	0.07	0.52	0.21	0.27	16.2	10.70 - 24.24
2002	0.06	100.00	2.07	19.74	41.35	0.06	0.16	0.31	0.16	12.2	7.75 - 18.82
2003	1.82	52.41	16.54	13.15	49.41	0.07	0.07	0.57	0.04	7.0	4.45 - 10.76
2004	0.08	100.00	0.40	35.92	58.98	0.06	0.21	0.24	0.15	5.5	3.67 - 8.11
2005	0.74	71.90	8.64	16.29	25.86	0.07	0.12	0.39	0.27	32.0	15.60 - 64.51
2006	2.93	83.59	0.16	43.05	1.05	0.25	0.29	0.37	0.14	8.7	5.66 - 13.13
2007	0.16		4.83	16.80	63.93	0.06	0.06	0.54	0.11	12.07	8.03 - 17.94

Table C6.2-1 (continued). Young of year indices of abundance.

Year	DE CPUE	DE 95% CI	MD Ches Bay CPUE	MD Coast CPUE	VIMS CPUE	NC 0 CPUE	NC 0 95% CI	NC 1+ CPUE	NC 1+ 95% CI
1981	5.98	4.39 - 8.05			6.02				
1982	11.49	9.15 - 14.36			10.95				
1983	4.47	3.43 - 5.76			10.85				
1984	6.67	4.90 - 8.96			6.05				
1985	9.35	7.11 - 12.22			37.04				
1986	12.94	10.02 - 16.64			4.62				
1987	5.98	4.45 - 7.95			17.85	1.01	0.40 - 1.88	14.07	8.70 - 22.40
1988	4.73	3.35 - 6.53			21.72	23.8	12.89 - 43.29	12.09	7.10 - 20.15
1989	11.11	8.40 - 14.6	0.44	0.87	21.27	4.04	2.18 - 6.99	3.51	2.08 - 5.60
1990	8.73	6.52 - 11.59	0.95	1.72	30.01	9.23	4.50 - 18.01	5.98	3.78 - 9.21
1991	20.07	15.73 - 25.54	0.78	1.89	15.32	3.77	1.62 - 7.70	4.67	2.87 - 7.29
1992	14.72	11.06 - 19.49	3.24	1.81	15.91	14.34	8.68 - 23.29	6.3	3.61 - 10.57
1993	14.79	11.01 - 19.75	1.59	0.91	15.42	1.67	0.83 - 2.91	20.63	14.09 - 30.00
1994	11.47	8.43 - 15.49	2.33	1.84	7.04	20.6	12.10 - 34.61	16.56	10.54 - 25.73
1995	13.49	10.01 - 18.06	5.95	4.44	11.00	14.3	8.85 - 22.75	14.32	8.67 - 23.26
1996	11.93	8.67 - 16.30	6.40	3.18	7.42	38.01	25.67 - 56.04	9.69	6.02 - 15.27
1997	15.40	11.14 - 21.16	4.28	3.06	14.82	16.57	10.54 - 25.73	19.05	11.66 - 30.76
1998	11.35	8.61 - 14.87	5.87	2.80	9.95	31.41	19.46 - 50.34	3.05	1.41 - 5.81
1999	13.51	10.23 - 17.76	3.26	2.76	16.25	59.65	43.93 - 80.86	15.91	10.57 - 23.73
2000	14.16	10.55 - 18.9	6.54	2.34	11.09	35.22	24.27 - 50.92	51.52	34.21 - 77.35
2001	7.57	5.60 - 10.12	8.10	2.56	11.52	5.17	2.98 - 8.59	9.9	5.48 - 17.34
2002	5.96	4.25 - 8.23	3.92	0.61	8.59	4	2.16 - 6.93	3.78	2.02 - 6.58
2003	10.44	7.58 - 14.26	4.89	5.64	5.42	10.98	7.20 - 16.52	3.22	1.83 - 5.29
2004	8.39	6.00 - 11.59	1.62	3.39	10.47	9.4	5.44 - 15.81	4.35	2.06 - 8.36
2005	16.84	12.02 - 23.45	3.55	4.98	7.10	9.05	5.22 - 15.24	4.31	2.04 - 8.30
2006	5.35	3.93 - 7.19	2.41	1.50	6.20	7.4	3.80 - 13.70	29.71	17.63 - 49.64
2007	13.70	10.07 - 18.52	1.64	2.32	14.37	12.93	7.28 - 22.44	4.3	2.41 - 7.22

Table C7.2-1. Comparison of model fit statistics for various sensitivity runs and the preferred model run. Tuning indices used are as follows: Base = All available aged and young of year indices; FI+YOY = All aged fishery independent indices plus all YOY indices; FI only = All aged fishery independent indices only; FD+YOY = All fishery dependent aged indices plus all YOY indices; FD only = Fishery dependent indices only; Scores = indices selected by assigning scores to a set of survey criteria; Preferred = the WTC preferred model run

	Base	FI+YOY	FI only	FD+YOY	FD only	Scores	Preferred
Residual Sum of Squares	1042.95	939.705	572.098	460.518	78.3876	676.472	446.968
Number of Residuals	633	512	293	340	121	367	359
Number of Parameters	6	6	6	6	6	6	6
Degrees of Freedom	627	506	287	334	115	361	353
Mean Squared Residual	1.66339	1.85712	1.99337	1.3788	0.681631	1.87388	1.2662
Standard Deviation	1.28973	1.36276	1.41187	1.17422	0.82561	1.3689	1.12526

Table C7.3-1. Index values for tuning indices used in the preferred ADAPT VPA model run. DE = Delaware 30-foot trawl, NJ = New Jersey ocean trawl, NC = Norh Carolina gillnet, MRFSS = recreational harvest, Rec = recreational catch.

Year	Index DE 1	DE 2	DE 3	DE 4	DE 5	DE 6+	NJ 1	NJ 2	NJ 3	NJ 4	NJ 5	NJ 6+
1982	4.71	7.33	3.02	1.45	0.24	0.33	0	0	0	0	0	0
1983	5.39	8	3.34	1.57	0.2	0.18	0	0	0	0	0	0
1984	2.99	5.32	2.42	1.15	0.13	0.07	0	0	0	0	0	0
1985	0	0	0	0	0	0	0	0	0	0	0	0
1986	0	0	0	0	0	0	0	0	0	0	0	0
1987	0	0	0	0	0	0	0	0	0	0	0	0
1988	0	0	0	0	0	0	0	0	0	0	0	0
1989	0	0	0	0	0	0	0	0	0	0	0	0
1990	0	0	0	0	0	0	0.0356	0.026	0.0057	0.0024	0.0003	0
1991	27.1588	3.6382	0.6383	0	0	0	0.0745	0.0523	0.0099	0.0023	0.0001	0
1992	21.1512	2.6081	0.0318	0.0318	0	0	0.0892	0.3206	0.0585	0.0007	0	0
1993	50.2801	25.422	3.9002	0.5169	0	0	0.0405	0.1103	0.0445	0.0181	0.0014	0
1994	113.4222	68.536	23.6497	0.9653	0	0	0.2556	0.1489	0.0316	0	0	0
1995	75.2633	53.4866	15.7403	5.4251	0.0764	0	0.0398	0.1607	0.0702	0.0082	0	0
1996	44.0437	48.3017	111.1071	23.8182	6.387	0.1331	0.0543	0.0434	0.1177	0.0273	0.0107	0.0009
1997	33.4142	25.0014	13.5633	34.5991	2.9623	0.474	0.1323	0.0342	0.0466	0.1339	0.0116	0
1998	23.3566	24.6266	20.3749	11.5955	20.7062	1.3252	0.0087	0.044	0.0679	0.035	0.0821	0.0113
1999	42.0661	20.1604	17.031	6.7402	2.5878	4.0331	0.0222	0.0396	0.0302	0.0053	0.0015	0.0012
2000	97.8468	50.3831	23.6377	5.7446	0.6592	0.9418	0.0397	0.1255	0.1011	0.2153	0.0112	0.0002
2001	13.1139	42.6313	18.7718	5.5263	0.5702	0.0877	0.0047	0.0788	0.1381	0.0582	0.0137	0.0029
2002	89.3537	23.3228	27.9672	3.8367	0.4039	0	0.1398	0.0546	0.0239	0.0029	0	0
2003	50.1607	13.9835	1.2183	0.4237	0	0	0.0416	0.0407	0.0021	0.0003	0	0
2004	26.3628	22.4048	0.1089	0	0	0	0.0591	0.1735	0.0726	0.0129	0.0002	0
2005	12.0686	14.5354	2.3872	0	0	0	0.0977	0.0934	0.0099	0.0008	0	0
2006	58.3793	37.1285	10.0217	0.7848	0	0	0.0746	0.0747	0.0161	0	0	0
2007	23.4698	17.4276	2.0963	0.1644	0	0	0.0926	0.0122	0.0009	0.0001	0	0
2008	40.9767	4.4014	0.5579	0	0	0	0.0591	0.1735	0.0726	0.0129	0.0002	0

Table C7.3-1 (continued). Index values for tuning indices used in the preferred ADAPT VPA model run. DE = Delaware 30-foot trawl, NJ = New Jersey ocean trawl, NC = Norh Carolina gillnet, MRFSS = recreational harvest, Rec = recreational catch.

Year	Index NC 1	NC 2	NC 3	NC 4	NC 5	NC 6+	MRFSS 3	MRFSS 4	MRFSS 5	MRFSS 6+	Rec 2-6+
1982	0	0	0	0	0	0	0.1865	0.2176	0.2131	0.1066	0.2021
1983	0	0	0	0	0	0	0.0084	0.0588	0.0671	0.063	0.5109
1984	0	0	0	0	0	0	0.1771	0.1631	0.1165	0.0326	0.3001
1985	0	0	0	0	0	0	0.1033	0.0919	0.0632	0.0172	0.2777
1986	0	0	0	0	0	0	0.0933	0.0758	0.0525	0.0175	0.7681
1987	0	0	0	0	0	0	0.3885	0.1329	0.0664	0.0102	0.5178
1988	0	0	0	0	0	0	0.2272	0.1262	0.0707	0.0101	0.5738
1989	0	0	0	0	0	0	0.1498	0.1915	0.129	0.0291	0.1673
1990	0	0	0	0	0	0	0.0527	0.0527	0.0341	0.0093	0.1557
1991	0	0	0	0	0	0	0.0574	0.0309	0.0177	0.0044	0.1778
1992	0	0	0	0	0	0	0.053	0.0485	0.0265	0.0088	0.1928
1993	0	0	0	0	0	0	0.037	0.0328	0.0287	0.0082	0.1651
1994	0	0	0	0	0	0	0.03	0.0258	0.0172	0.0043	0.3849
1995	0	0	0	0	0	0	0.0355	0.0659	0.0304	0	0.5079
1996	0	0	0	0	0	0	0.0271	0.0588	0.0407	0.0045	0.6054
1997	0	0	0	0	0	0	0.0137	0.0504	0.1054	0.0321	0.5177
1998	0	0	0	0	0	0	0.0151	0.0605	0.0958	0.0302	0.5297
1999	0	0	0	0	0	0	0.0162	0.0647	0.1024	0.0323	0.4101
2000	0	0	0	0	0	0	0.0105	0.0264	0.0632	0.0474	0.4585
2001	0.148	1.4886	0.3694	0.1459	0.1127	0	0.0109	0.0274	0.0328	0.0711	0.2818
2002	0.1729	0.2674	1.0873	0.2423	0.0569	0.0089	0.0368	0.0263	0.0158	0.0158	0.2379
2003	0.1454	0.3669	0.345	0.6802	0.0583	0	0.0051	0.0462	0.0205	0.0154	0.1162
2004	0.2269	0.5946	0.584	0.1885	0.0722	0	0.0094	0.0047	0.0047	0.0047	0.1154
2005	0.1719	0.4862	0.6925	0.1747	0.0343	0.0009	0.0135	0.0058	0.0021	0.002	0.2243
2006	0.1974	0.3766	0.5534	0.213	0.0543	0.0008	0.0489	0.0384	0.0058	0.0001	0.1704
2007	0.1597	0.2564	0.179	0.114	0.052	0.0008	0.0084	0.0196	0.0088	0.0009	0.0884
2008	0	0	0	0	0	0	0.0113	0.0054	0.0026	0.0003	0.13

Table C7.3-2. Fishing mortality estimates from preferred run of ADAPT VPA.

	Ages 4-5 Unweighted	N Wtd	B Wtd	C Wtd
1982	1.4015	1.4015	1.4015	1.4015
1983	1.6135	1.6135	1.6135	1.6135
1984	2.2116	2.2116	2.2116	2.2116
1985	1.3506	1.3506	1.3506	1.3506
1986	1.7287	1.7287	1.7287	1.7287
1987	0.6618	0.6618	0.6618	0.6618
1988	1.7955	1.7955	1.7955	1.7955
1989	1.1127	1.1127	1.1127	1.1127
1990	0.7645	0.7645	0.7645	0.7645
1991	0.8351	0.8351	0.8351	0.8351
1992	1.0256	1.0256	1.0256	1.0256
1993	1.3097	1.3097	1.3097	1.3097
1994	0.9848	0.9848	0.9848	0.9848
1995	0.3237	0.3237	0.3237	0.3237
1996	0.5304	0.5304	0.5304	0.5304
1997	0.5634	0.5634	0.5634	0.5634
1998	0.7208	0.7208	0.7208	0.7208
1999	0.6693	0.6693	0.6693	0.6693
2000	0.8219	0.8219	0.8219	0.8219
2001	1.0366	1.0366	1.0366	1.0366
2002	1.3437	1.3437	1.3437	1.3437
2003	2.3915	2.3915	2.3915	2.3915
2004	2.1654	2.1654	2.1654	2.1654
2005	2.0733	2.0733	2.0733	2.0733
2006	1.3175	1.3175	1.3175	1.3175
2007	0.5101	0.4784	0.4828	0.4779

	Ages 1-5 Unweighted	N Wtd	B Wtd	C Wtd
1982	0.9182	0.5433	0.7299	0.6642
1983	1.0231	0.5748	0.9195	0.7473
1984	1.446	0.7622	1.2146	1.0228
1985	0.8896	0.4477	0.6967	0.5392
1986	1.0526	0.5097	0.758	0.5867
1987	0.5303	0.4564	0.4913	0.4651
1988	1.2169	0.9216	1.2525	1.0414
1989	0.7895	0.5474	0.8092	0.7008
1990	0.6204	0.5395	0.6247	0.5453
1991	0.7222	0.5875	0.7466	0.6137
1992	0.7219	0.3824	0.749	0.5401
1993	0.7349	0.3042	0.5366	0.4193
1994	0.5337	0.1994	0.3409	0.3293
1995	0.2635	0.21	0.2348	0.2365
1996	0.3328	0.2403	0.3384	0.3145
1997	0.3699	0.3003	0.3954	0.4159
1998	0.4395	0.3655	0.501	0.4669
1999	0.4402	0.3837	0.5308	0.4768
2000	0.5525	0.4075	0.6482	0.5633
2001	0.7686	0.6293	0.8023	0.6666
2002	1.0447	0.9078	1.1444	0.988
2003	1.4689	0.6538	1.5688	1.4585
2004	1.2523	0.4319	0.6936	0.666
2005	1.2511	0.579	0.8919	0.9359
2006	0.8194	0.2876	0.5168	0.6327
2007	0.3309	0.1288	0.2034	0.2287

Table C7.3-2. Bootstrap mean and 80% confidence intervals for estimates of fishing mortality and spawning stock biomass in 2005 using different terminal years.

Fishing mortality in 2005

Terminal year	Mean	Lower 80% CI	Upper 80% CI
2005	0.39	0.2475	0.5333
2006	0.94	0.6487	1.2235
2007	2.04	1.9222	2.1584

Spawning stock biomass (MT) in 2005

Terminal year	Mean	Lower 80% CI	Upper 80% CI
2005	4,889	3,738	6,040
2006	3,045	2,574	3,516
2007	2,265	2,182	2,347

Table C8.2-1. Mid-Atlantic private boat catch of weakfish in number (MIDN*1000), catch in weight (MIDW, mt) private boat effort (MIDEF, Trips*1000), abundance index in number (RELNT) and weight (RELWT).

YEAR	midN	midw	midef	RelNt	RELWT
1981	4592	5549.44	6032	0.76	0.92
1982	1218	2712.6	6028	0.20	0.45
1983	4328	4066.56	8472	0.51	0.48
1984	2365	2285.49	7881	0.30	0.29
1985	1942	1748.5	6994	0.28	0.25
1986	7683	3501.05	10003	0.77	0.35
1987	4429	2566.2	8554	0.52	0.30
1988	5085	2481.36	8862	0.57	0.28
1989	1130	810.72	6756	0.17	0.12
1990	1227	551.67	7881	0.16	0.07
1991	1549	784.17	8713	0.18	0.09
1992	1331	621.36	6904	0.19	0.09
1993	1445	437.5	8750	0.17	0.05
1994	3498	908.7	9087	0.38	0.10
1995	4358	1115.53	8581	0.51	0.13
1996	5354	1680.36	8844	0.61	0.19
1997	5035	1750.5	9725	0.52	0.18
1998	4571	1984.9	8630	0.53	0.23
1999	3254	1428.3	7935	0.41	0.18
2000	5192	2038.32	11324	0.46	0.18
2001	3376	1318.02	11982	0.28	0.11
2002	2272	1050.61	9551	0.24	0.11
2003	1311	451.44	11286	0.12	0.04
2004	1279	332.52	11084	0.12	0.03
2005	2631	821.1	11730	0.22	0.07
2006	2066	484.92	12123	0.17	0.04
2007	1109	251.02	12551	0.09	0.02
2008	1510	374.18	11693	0.13	0.03

Table C8.2-2. Delaware trawl survey index of weakfish in mean number (DEN) and weight (DEW) per tow and New Jersey trawl survey index of weakfish in mean number (NJN) and weight (NJW) per tow.

YEAR	DEN	DEW	NJN	NJW
1989			0.10	0.12
1990	4.27	1.15	0.08	0.08
1991	2.68	2.63	0.12	0.08
1992	29.80	3.80	0.31	0.12
1993	93.00	11.31	0.17	0.07
1994	74.70	14.84	0.41	0.12
1995	189.80	23.93	0.32	0.14
1996	77.00	42.56	0.28	0.14
1997	78.69	20.04	0.39	0.19
1998	50.55	21.44	0.22	0.10
1999	81.36	12.66	0.22	0.08
2000	67.59	20.68	0.53	0.13
2001	32.31	13.58	0.18	0.10
2002	15.62	14.43	0.24	0.07
2003	22.52	4.65	0.04	0.02
2004	16.93	4.30	0.13	0.06
2005	47.93	2.91	0.23	0.06
2006	19.69	5.20	0.11	0.03
2007	23.07	2.80	0.14	0.04

Table C8.2-3. Scaled Delaware trawl survey index in number (ADEN) and weight (ADEW) per tow, scaled New Jersey trawl survey index in number (ANJN), and weight (ANJN), recreational cpue in number (RelNt) and weight (RelWt) and the blended indicies in number (WKN) and weight (WKW), 1981 to 2008.

YEAR	ADEN	ANJN	RelNt	WKN	ANJW	RELWT	ADEW	WKW
1981			0.761	0.761		0.920		0.920
1982			0.202	0.202		0.450		0.450
1983			0.511	0.511		0.480		0.480
1984			0.300	0.300		0.290		0.290
1985			0.278	0.278		0.250		0.250
1986			0.768	0.768		0.350		0.350
1987			0.518	0.518		0.300		0.300
1988			0.574	0.574		0.280		0.280
1989		0.159	0.167	0.163	0.258	0.120		0.189
1990	0.029	0.127	0.156	0.104	0.172	0.070	0.018	0.087
1991	0.018	0.191	0.178	0.129	0.172	0.090	0.042	0.101
1992	0.203	0.493	0.193	0.296	0.258	0.090	0.061	0.136
1993	0.632	0.270	0.165	0.356	0.151	0.050	0.181	0.127
1994	0.508	0.652	0.385	0.515	0.258	0.100	0.237	0.198
1995	1.291	0.509	0.508	0.769	0.301	0.130	0.383	0.271
1996	0.524	0.445	0.605	0.525	0.301	0.190	0.681	0.391
1997	0.535	0.620	0.518	0.558	0.409	0.180	0.321	0.303
1998	0.344	0.350	0.530	0.408	0.215	0.230	0.343	0.263
1999	0.553	0.350	0.410	0.438	0.172	0.180	0.203	0.185
2000	0.460	0.843	0.459	0.587	0.280	0.180	0.331	0.263
2001	0.220	0.286	0.282	0.263	0.215	0.110	0.217	0.181
2002	0.106	0.382	0.238	0.242	0.151	0.110	0.231	0.164
2003	0.153	0.064	0.116	0.111	0.043	0.040	0.074	0.052
2004	0.115	0.207	0.115	0.146	0.129	0.030	0.069	0.076
2005	0.326	0.366	0.224	0.305	0.129	0.070	0.047	0.082
2006	0.134	0.175	0.170	0.160	0.065	0.040	0.083	0.063
2007	0.157	0.223	0.088	0.156	0.086	0.020	0.045	0.050
2008			0.129	0.129		0.032		0.032

Table C8.2-4. Coast-wide weakfish harvest and discards in number (CATCHN) and weight (CATCHW, MT), relative fishing mortality in number (RELFN) and weight (RelWt)), ages 1+l fishing mortality in numbers (FN) and weight (FW) and fishing mortality rates due to discards in number (FDISN) and weight (FDISW), 1981 to 2008.

YEAR	CATCHN	HARVN	DISN	CATCHW	HARVW	DISCARDS	RELFN	RELFW	FN	FW	FDISW	FDISN
1981	49379	44693.2	4685.8	20199	19269.1	929.9	102517.3	29487.6	0.687	0.708	0.033	0.065
1982	32493	27988.7	4504.3	13408	12593.3	814.7	91155.2	28834.4	0.611	0.692	0.042	0.085
1983	30043	27210.7	2832.3	13849	13247.3	601.7	74093.5	35971.4	0.496	0.863	0.038	0.047
1984	35487	30184.0	5303.0	13010	12151.1	858.9	122844.4	48185.2	0.823	1.156	0.076	0.123
1985	30045	27290.8	2754.2	10720	10180.1	539.9	57461.9	35733.3	0.385	0.858	0.043	0.035
1986	41471	38409.5	3061.5	14863	14211.5	651.5	64504.2	45732.3	0.432	1.098	0.048	0.032
1987	36587	34080.3	2506.7	11311	10805.5	505.5	67035.7	39003.5	0.449	0.936	0.042	0.031
1988	42062	39871.7	2190.3	12690	12182.4	507.6	114155.0	54115.1	0.765	1.299	0.052	0.040
1989	16977	14523.4	2453.6	7865	7411.9	453.1	127118.6	57034.1	0.852	1.369	0.079	0.123
1990	18379	17171.7	1207.3	5165	4892.5	272.5	157820.3	54900.1	1.057	1.318	0.070	0.069
1991	20266	18930.9	1335.1	5206	4909.4	296.6	95359.6	43816.6	0.639	1.052	0.060	0.042
1992	15284	13659.3	1624.7	4320	4015.3	304.7	46879.4	32799.3	0.314	0.787	0.056	0.033
1993	15100	13010.5	2089.5	4011	3608.4	402.6	34677.5	24635.1	0.232	0.591	0.059	0.032
1994	12442	8954.2	3487.8	4659	3622.3	1036.7	19379.5	19835.1	0.130	0.476	0.106	0.036
1995	12936	9560.1	3375.9	4840	4062.1	777.9	19996.5	14623.5	0.134	0.351	0.056	0.035
1996	13819	8818.2	5000.8	5536	4474.6	1061.4	25534.7	15960.8	0.171	0.383	0.073	0.062
1997	13631	10622.8	3008.2	5911	4985.8	925.2	28239.7	20897.0	0.189	0.502	0.079	0.042
1998	12762	10486.2	2275.8	6468	5655.8	812.2	30190.0	28905.1	0.202	0.694	0.087	0.036
1999	9771	4815.7	4955.3	5347	4557.7	789.3	19072.0	23853.9	0.128	0.572	0.085	0.065
2000	8922	6938.3	1983.7	4956	4334.2	621.8	21005.5	22313.3	0.141	0.536	0.067	0.031
2001	9853	4734.6	5118.4	4698	3503	1195.0	39064.1	27270.1	0.262	0.654	0.166	0.136
2002	8922	4667.8	4254.2	3662	3159.8	502.2	50568.9	33866.6	0.339	0.813	0.111	0.162
2003	2905	2395.7	509.3	1563	1300.6	262.4	22633.0	24345.8	0.152	0.584	0.098	0.027
2004	3350	2655.9	694.1	1350	1112.1	237.9	14854.4	17111.7	0.100	0.411	0.072	0.021
2005	3637	2843.0	794.0	1450	1240.8	209.2	15641.5	20080.3	0.105	0.482	0.070	0.023
2006	2698	1723.5	974.5	1134	898.7	235.3	17093.1	20100.4	0.115	0.482	0.100	0.041
2007	2513	1222.8	1290.2	967	752.2	214.8	17630.0	23508.9	0.118	0.564	0.125	0.061
2008	1348	957.6	390.4	811	556.4	254.6	10444.1	25343.8	0.070	0.608	0.191	0.020

Table C8.2-5. Method of scaling relative fishing mortality rates in numbers (RelNt) and weight (RelWt) from 1981 to 2008 (Table 4) to units of instantaneous fishing mortality (F). Annual instantaneous F estimates (ages 1+, number weighted and biomass weighted) were taken from the ADAPT VPA run, tuned to the Delaware trawl, SEAMAP trawl indices and cpue from the Mid-Atlantic recreational fishery from 1982 to 1985. The scalars to convert relative F to units of instantaneous F (FNt, FWt) from 1981 to 2008 (Tables 4) were derived from the average (1982-1985) F estimates from the VPA to the corresponding relative F estimates.

Number Weighted F (ages 1+)

YEAR	VPA FN	RelFnt
1982	0.54	91,155
1983	0.58	74,094
1984	0.76	122,844
1985	0.45	57,462
Average	**0.58**	**86,389**

Scalar = 0.58 / 86,389 = 0.0000067

Biomass Weighted F (ages 1+)

Year	VPA FW	RelFwt
1982	0.73	28,834
1983	0.92	35,971
1984	1.22	48,185
1985	0.70	35,733
Average	**0.89**	**37,131**

Scalar = 0.89 /37,131 = 0.000024

Table C8.2-6. Coast-wide age 1+ weakfish stock size in number (BION*1000), weight (BIOW, MT), harvest in weight (MT) and surplus production (SURP, MT), 1981 to 2008.

YEAR	CATCHW	BIOW	BION	SURP
1981	20199	28541.7	71890.3	11032.3
1982	13408	19375.0	53202.7	10074.7
1983	13849	16041.7	60518.5	9057.3
1984	13010	11250.0	43116.1	14260.0
1985	10720	12500.0	78040.0	11761.7
1986	14863	13541.7	95958.1	13404.7
1987	11311	12083.3	81460.3	8998.5
1988	12690	9770.8	54994.6	8665.0
1989	7865	5745.8	19933.2	6039.2
1990	5165	3920.0	17381.4	6195.6
1991	5206	4950.6	31719.7	5743.4
1992	4320	5487.9	48660.9	5616.1
1993	4011	6784.0	64991.3	7013.9
1994	4659	9786.9	95823.7	8662.6
1995	4840	13790.6	96554.4	5501.5
1996	5536	14452.1	80773.9	2869.9
1997	5911	11786.0	72043.1	3448.6
1998	6468	9323.6	63092.9	6484.3
1999	5347	9339.9	76466.1	5261.7
2000	4956	9254.6	63394.8	2879.6
2001	4698	7178.2	37645.8	2025.2
2002	3662	4505.4	26333.2	1831.6
2003	1563	2675.0	19157.0	2175.2
2004	1350	3287.2	33660.1	1071.5
2005	1450	3008.8	34704.8	791.9
2006	1134	2350.7	23558.4	497.2
2007	967	1713.9	21274.8	586.4
2008	811	1333.3	19264.0	810.7

Table C9.2-1. Abundance indices of striped bass (STRIPrec2) from the MRFSS catch/trip, striped bass abundance from SCAM, summer flounder abundance (FLUKE), bluefish abundance (BLUE), dogfish abundance (DGFISH), mean deviation in sea surface temperature (devtemp, C) and the deviations in the winter North Atlantic Oscillation Index (NAO), 1981-2008.

YEAR	STRIP2	STRIPREC2	fluke	BLUE	DGFISH	devTEMP	NAO
1981	463	160.59	0.64	0.67	0.20	0.50	2.05
1982	463	283.40	0.54	0.69	0.20	0.08	0.80
1983	333	292.84	1.12	0.70	0.36	-0.01	3.42
1984	245	226.72	1.23	0.53	0.84	0.09	1.60
1985	232	321.18	0.63	0.54	0.10	0.59	-0.63
1986	337	406.20	0.77	0.71	1.50	-0.21	0.50
1987	412	321.18	0.83	0.78	0.90	0.09	-0.75
1988	495	755.73	0.62	0.43	1.30	-0.41	0.72
1989	628	774.62	0.10	0.46	2.30	0.09	5.08
1990	1375	1180.83	0.38	0.53	0.90	0.29	3.96
1991	1918	1719.28	0.61	0.38	1.90	0.39	1.03
1992	2329	2427.78	0.55	0.38	1.50	-0.51	3.28
1993	2621	2635.60	0.93	0.27	1.40	0.29	2.67
1994	3052	5308.99	0.66	0.26	1.50	0.49	3.03
1995	3496	6584.28	0.64	0.26	0.70	0.69	3.96
1996	3865	7500.60	0.75	0.25	0.43	-0.91	-3.78
1997	4498	9730.00	0.67	0.29	0.86	-0.81	-0.17
1998	4372	9918.93	0.86	0.24	0.69	0.39	0.72
1999	4421	8955.38	0.90	0.24	0.82	1.29	1.70
2000	4982	9153.76	0.70	0.31	0.60	0.89	2.80
2001	6934	7084.95	0.84	0.36	2.70	0.89	-1.89
2002	7133	8360.24	0.53	0.33	2.80	1.19	0.76
2003	7669	8483.05	0.64	0.36	3.40	0.69	0.20
2004	8028	9304.90	0.63	0.46	4.40	0.39	-0.07
2005	6927	9824.46	0.77	0.41	4.10	0.69	0.12
2006	5915	12110.54	0.56	0.38	4.45	0.89	-1.09
2007	5915	8039.06	0.59	0.41	4.36	0.69	2.80
2008	5915	5923.02	0.71	0.43	3.73	1.09	2.11

Table C9.2-2. Weakfish juvenile indices from Rhode Island (RI0), Connecticut (CT0), New York (NY0), New Jersey (NJ0), Delaware (DE0), Maryland (MD0), Virginia (VA0) and North Carolina (NC0), 1981 to 2007

YEAR	RI0	CT0	NY0	NJ0	DE0	MD0	VA0	NC0
1981	39.00				5.98		6.02	
1982	19.60				11.49		10.95	
1983	3.10				4.47		10.85	
1984	5.00	1.00			6.67		6.05	
1985	19.20	6.19			9.35		37.04	
1986	2.00	13.16			12.94		4.62	
1987	1.30	0.63	0.60		5.98		17.85	12.14
1988	10.90	3.49	0.10	25.71	4.73		21.72	105.50
1989	1.20	8.69	1.40	43.37	11.11	0.44	21.27	14.20
1990	27.30	5.56	0.60	14.71	8.73	0.95	30.00	50.20
1991	25.40	11.95	20.60	27.09	20.07	0.78	15.32	36.90
1992	14.50	3.05	3.30	5.95	14.72	3.24	15.91	42.70
1993	7.50	4.08	1.00	23.88	14.79	1.59	15.42	8.70
1994	15.20	11.19	8.30	37.14	11.47	2.33	7.04	68.10
1995	0.30	5.22	1.60	77.48	13.49	5.95	11.00	38.20
1996	116.10	15.23	24.50	46.27	11.93	6.40	7.42	72.40
1997	88.80	12.38	18.80	21.75	15.40	4.28	14.82	32.80
1998	13.20	5.02	1.00	3.04	11.35	5.87	9.95	70.40
1999	3.70	30.93	8.40	25.32	13.51	3.26	16.25	100.00
2000	9.40	63.31	15.90	0.09	14.16	6.54	11.09	63.00
2001	19.30	40.10	16.20	21.68	7.57	8.10	11.52	30.30
2002	8.40	41.35	12.20	39.31	5.96	3.92	8.59	22.00
2003	198.00	49.41	7.00	72.72	10.44	4.89	5.42	23.90
2004	1.88	58.98	5.50	68.38	8.39	1.62	10.47	28.80
2005	129.50	25.86	32.00	70.64	16.84	3.55	7.10	28.80
2006	0.36	1.05	8.70	9.03	5.35	2.41	6.20	39.10
2007		63.93			13.70	1.64		56.80

Table C9.2-3. Scaled weakfish juvenile indices from Rhode Island (ARI0), Connecticut (ACT0), New York (ANY0), New Jersey (NJ0), Delaware (ADE0), Maryland (AMD0), Virginia (AVA0), and North Carolina (ANC0), blended age 0 index (REC0), age 1 abundance (NVPA) and juvenile relative mortality (z0), 1981-2007.

YEAR	ARI0	ACT0	ANY0	NJ0	ADE0	AMD0	AVA0	ANC0	REC0	nvpa	z0
1981	43.29				18.54		15.35		25.73	45	1.74
1982	21.76				35.62		27.92		28.43	41	1.94
1983	3.44				13.86		27.67		14.99	44.1	1.22
1984	5.55	1.66			20.68		15.43		13.88	63.4	0.78
1985	21.31	10.28			28.99		94.45		48.25	70.9	1.92
1986	2.22	21.85			40.11		11.78		18.04	54.4	1.20
1987	1.44	1.05	2.13		18.54		45.52	8.98	12.94	26.3	1.59
1988	12.10	5.79	0.36	25.71	14.66		55.39	78.07	27.44	20.8	2.58
1989	1.33	14.43	4.97	43.37	34.44	4.11	54.24	10.51	20.92	26.7	2.06
1990	30.30	9.23	2.13	14.71	27.06	8.87	76.50	37.15	25.74	29.5	2.17
1991	28.19	19.84	73.13	27.09	62.22	7.29	39.07	27.31	35.52	35.6	2.30
1992	16.10	5.06	11.72	5.95	45.63	30.26	40.57	31.60	23.36	36.7	1.85
1993	8.33	6.77	3.55	23.88	45.85	14.85	39.32	6.44	18.62	49.3	1.33
1994	16.87	18.58	29.47	37.14	35.56	21.76	17.95	50.39	28.46	23.4	2.50
1995	0.33	8.67	5.68	77.48	41.82	55.57	28.05	28.27	30.73	29.7	2.34
1996	128.87	25.28	86.98	46.27	36.98	59.78	18.92	53.58	57.08	16.2	3.56
1997	98.57	20.55	66.74	21.75	47.74	39.98	37.79	24.27	44.67	11.4	3.67
1998	14.65	8.33	3.55	3.04	35.19	54.83	25.37	52.10	24.63	9.4	3.27
1999	4.11	51.34	29.82	25.32	41.88	30.45	41.44	74.00	37.29	12.5	3.40
2000	10.43	105.10	56.45	0.09	43.90	61.08	28.28	46.62	43.99	6.3	4.25
2001	21.42	66.57	57.51	21.68	23.47	75.65	29.38	22.42	39.76	6	4.19
2002	9.32	68.64	43.31	39.31	18.48	36.61	21.90	16.28	31.73	6.4	3.90
2003	219.78	82.02	24.85	72.72	32.36	45.67	13.82	17.69	63.61	8.8	4.28
2004	2.09	97.91	19.53	68.38	26.01	15.13	26.70	21.31	34.63	4.7	4.30
2005	143.75	42.93	113.60	70.64	52.20	33.16	18.11	21.31	61.96	6.7	4.53
2006	0.40	1.74	30.89	9.03	16.59	22.51	15.81	28.93	15.74	4.5	3.55
2007		106.12			42.47	15.32		42.03	51.49		

Table C9.3-1. Pearson correlation (r) matrix relating the three weakfish response variables consisting of surplus production (SURPT) blended weakfish biomass WkW) and juvenile mortality (Z0) with each of the 10 potential explanatory variables. The P value indicates the level of statistical significance.

	Response Variable					
Explanatory Variable	**SURPt**		**WkW**		**Z0**	
	r	**P**	**r**	**P**	**r**	**P**
Striprec2	-0.75	0.0001*	- 0.44	0.02*	0.88	0.0001*
Fluke	-0.30	0.13	0.20	0.31	- 0.18	0.37
Menhaden	0.76	0.0001*	0.47	0.01*	- 0.68	0.0001*
Bluefish	0.44	0.02**	0.47	0.01**	-0.57	0.002**
Spiny Dogfish	- 0.64	0.0003*	-0.65	0.0003*	0.57	0.002*
DeviationsTemperature	-0.30	0.12	-0.32	0.10	- 0.36	0.07
NAO	0.23	0.25	-0.002	0.99	-0.43	0.03*
FWt	0.51	0.0007**	0.06	0.74	-0552	0.003**
Discards	.		.		–0.22	0.27
Fdisn	.		.		0.01	0.95

- *** a statistically significant (P < 0.05) correlation with dependent variable.**
- **** a statistically significant (P < 0.05) correlation with dependent variable but in an implausible direction.**

Table C9.3-2. Three stepwise regression models with weakfish surplus production (SURPt), weakfish biomass (WkW) and juvenile mortality (Z0) used as response variables. The explanatory variables included a blended predator index (Tpred) of striped bass and spiny dogfish, bluefish, and fluke as well as mean annual water temperature, deviations in the North Atlantic Oscillation Index, fishing mortality (FWt) and discards (Disn). Given below are the explanatory 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 associated with each explanatory variable.

	Response Variables											
Variable	**SURPt**				**WkWt**				**Z0**			
	b	**SEb**	**r**	**P**	**b**	**SEb**	**r**	**P**	**b**	**SEb**	**r**	**P**
Tpred	-0.84	0.16	-0.80	0.0001	-0.37	0.08	0.79	0.0002	0.29	0.04	0.84	0.0001
FWt					-0.44	0.15	0.79	0.007				

Table C9.3-3. Parameter estimates (r, K, C) and steady-state overfishing thresholds (Fmsy, Fcoll, Bmsy) for weakfish derived from the Gompertz Surplus Production Model (see equation 6 in text) with and without the inclusion of blended predation from striped bass and spiny dogfish. These models were fitted by nonlinear iterative reweighting least squares, 1981-2008. The standard error (SE) is given for each parameter estimate, as well as the coefficient of determination (r**2).

	Straight Gompertz		***Gompertz with Predation***	
Parameters	**Mean**	**SE**	**Mean**	**SE**
r	0.26	0.10	0.54	0.05
K	156,095 mt	50,116 mt	52,466 mt	4,722 mt
C*			-0.31	0.05
r2**	0.54		0.84	
	Equilibrium Overfishing Thresholds			
Fmsy	0.26	0.10	0.54	0.05
Fcoll	0.71	0.27	1.47	0.25
Bmsy	57,388 mt	18,394 mt	19,289 mt	1,736 mt

- *** C = exponent for predation effects**

Table C9.3-4. Parameter estimates (A,B) of the Ricker Stock-Recruitment (S-R) model for weakfish (see equation 6 in text) with and without the inclusion of blended predation from striped bass and spiny dogfish. These alpha (A) and beta (B) parameters were estimated by nonlinear iterative reweighting least squares, 1981-2008. The standard error (SE) is given for each parameter estimate, as well as the coefficient of determination (r**2).

	Straight Ricker		***Ricker with Predation***	
Parameters	**Mean**	**SE**	**Mean**	**SE**
A	0.016	0.004	0.0.0048	0.002
B	-0.00015	0.000027	-0.00008	0.00003
C*			0.00011	0.00004
r2**	0.52		0.81	

- *** C = exponent for predation effects**

Table C9.3-5. Estimated weakfish predation losses (DT, MT), predation mortality (MP), fishing mortality (FW), total mortality (Z), fishing to total mortality ratio (FZ ratio), non-equilibrium FMSY, non-equilibrium BMSY, fraction maximum spawning potential (MSP) based on the Steele-Henderson production model with blended predation from striped bass and weakfish.

YEAR	DT	MP	FW	Ztotal	FZRATIO	FMSY	FCOLL	BMSY	msp
1981	467.83	0.02	0.71	0.83	0.86	0.78	2.12	20274.63	0.42
1982	532.68	0.03	0.69	0.82	0.84	0.77	2.10	20209.78	0.35
1983	810.11	0.06	0.86	1.02	0.84	0.75	2.04	19932.36	0.24
1984	1521.90	0.13	1.16	1.38	0.84	0.70	1.90	19220.57	0.27
1985	355.71	0.03	0.86	0.98	0.87	0.77	2.10	20386.76	0.29
1986	2817.90	0.22	1.10	1.42	0.77	0.64	1.73	17924.56	0.26
1987	1700.80	0.16	0.94	1.19	0.79	0.68	1.85	19041.67	0.21
1988	2492.42	0.32	1.30	1.72	0.76	0.58	1.57	18250.05	0.12
1989	3216.11	0.67	1.37	2.13	0.64	0.41	1.11	17526.36	0.08
1990	1185.29	0.27	1.32	1.68	0.78	0.61	1.65	19557.18	0.11
1991	2798.77	0.54	1.05	1.69	0.62	0.46	1.26	17943.69	0.12
1992	2831.72	0.46	0.79	1.35	0.58	0.50	1.36	17910.75	0.15
1993	3182.35	0.38	0.59	1.08	0.55	0.54	1.47	17560.12	0.21
1994	5304.54	0.45	0.48	1.03	0.46	0.51	1.38	15437.92	0.30
1995	5088.90	0.36	0.35	0.81	0.43	0.55	1.51	15653.57	0.31
1996	5200.18	0.40	0.38	0.88	0.44	0.53	1.45	15542.29	0.25
1997	7015.28	0.66	0.50	1.27	0.40	0.41	1.11	13727.19	0.20
1998	6435.14	0.69	0.69	1.48	0.47	0.40	1.08	14307.32	0.20
1999	6124.05	0.66	0.57	1.33	0.43	0.41	1.12	14618.42	0.20
2000	5866.88	0.71	0.54	1.35	0.40	0.39	1.06	14875.58	0.16
2001	7287.29	1.25	0.65	2.00	0.33	0.23	0.62	13455.18	0.10
2002	5879.04	1.64	0.81	2.55	0.32	0.15	0.42	14863.43	0.06
2003	3602.11	1.21	0.58	1.89	0.31	0.24	0.65	17140.36	0.07
2004	5663.27	1.80	0.41	2.31	0.18	0.13	0.36	15079.20	0.07
2005	4990.96	1.86	0.48	2.44	0.20	0.12	0.34	15751.50	0.05
2006	4076.38	2.01	0.48	2.59	0.19	0.11	0.29	16666.09	0.04
2007	2034.71	1.34	0.56	2.00	0.28	0.21	0.57	18707.76	0.03
2008	1061.03	0.80	0.61	1.50	0.40	0.36	0.98	19681.44	0.03

Table C9.3-6. Parameter estimates (r, K, c, A) and weakfish steady-state overfishing thresholds (Fmsy, Fcoll, Bmsy) derived from the Steele-Henderson Production Model (see equation 9 in text) with blended predation from striped bass and spiny dogfish fitted by nonlinear least squares and by nonlinear iterative reweighting least squares, 1981-2008. The standard error (SE) is given for each parameter estimate, as well as the coefficient of determination (r**2).

	Least Squares		***Iterative Reweighting***	
Parameters	**Mean**	**SE**	**Mean**	**SE**
r	0.75	0.22	0.72	0.09
K	49,095 mt	13,620 mt	46,264 mt	5,410 mt
c	1.35	0.53	1.28	0.23
A	5,364 mt	1,810 mt	4,034 mt	715 mt
r2**	0.84		0.94	
	Equilibrium Overfishing Thresholds			
Fmsy	0.75	0.22	0.72	0.09
Fcoll	2.04	0.60	1.96	0.25
Bmsy	18,050 mt	5,001 mt	17,009 mt	1,989 mt

Table C9.3-7. Weakfish losses due to predation (DT, MT), coast-wide harvest (HARVW, MT) and discards (discards, MT), 1981-2008.

YEAR	DISCARDS	HARVW	DT
1981	929.9	19269.1	467.83
1982	814.7	12593.3	532.68
1983	601.7	13247.3	810.11
1984	858.9	12151.1	1521.90
1985	539.9	10180.1	355.71
1986	651.5	14211.5	2817.90
1987	505.5	10805.5	1700.80
1988	507.6	12182.4	2492.42
1989	453.1	7411.9	3216.11
1990	272.5	4892.5	1185.29
1991	296.6	4909.4	2798.77
1992	304.7	4015.3	2831.72
1993	402.6	3608.4	3182.35
1994	1036.7	3622.3	5304.54
1995	777.9	4062.1	5088.90
1996	1061.4	4474.6	5200.18
1997	925.2	4985.8	7015.28
1998	812.2	5655.8	6435.14
1999	789.3	4557.7	6124.05
2000	621.8	4334.2	5866.88
2001	1195.0	3503.0	7287.29
2002	502.2	3159.8	5879.04
2003	262.4	1300.6	3602.11
2004	237.9	1112.1	5663.27
2005	209.2	1240.8	4990.96
2006	235.3	898.7	4076.38
2007	214.8	752.2	2034.71
2008	254.6	556.4	1061.03

Table C10.4-1. Summary of catch-and-release mortality experiment results.

Study	Malchoff and Heins 1997	Swihart et al. 2000	Swihart 2000	Gearhart 2002
Location	NY, Great South Bay	VA, Chesapeake Bay Mouth	VA, Chesapeake Bay Mouth	NC estuaries
Dates	Aug 14 – Sept 8, 1995	June, 1995	July 29-Aug 5, 1999	2000-2001, April-Nov
Mode	Pier	Boat	Pier	Boat
Bait type	Bait and artificial (no difference)	Bait	Artificials	Artificials and bait
Water temperature range °C	22 – 27	Mean = 23	26.7 - 30.8 Mean =28.7	13.8 -26.5
Mortality range	0 – 6.5%			0.0-32.1%
Mean mortality	2.6%, 95% CI = 0.6-7.0%	2%	14.5%	10.4-17.6% (high vs low salinity)
Fish Size	Mostly sublegal (< 405 mm) 315-425 mm	235-338 mm, mean = 283 mm	225-382 mm, mean = 298 mm	Not reported
N	90; 4 trials	360	145	180
Trial	72 h	Up to 23 days All deaths < 61 h	96 h	72 h

Table C10.4-2. Correlations of initial weight based indices (based on Uphoff 2006a) and other indicators of stock status. Shading indicates P < 0.05. EBI = exploitable biomass index. WRI = MRFSS catch per trip index. NJGM EBI is based on geometric mean counts during August and October rounds. NJPT EBI is based on proportion of tows with weakfish during August and October rounds. VPA B = biomass from converged portion of "best" VPA in Kahn (2006; run 20). Losses = all estimated fishery-related losses. All data are for 1981-2006, but N varies due to time-series.

		WRI	DE EBI	NJGM EBI	NJPT EBI	VPA B
DE EBI	r	0.75				
	P	0.0005				
	N	17				
NJGM EBI	r	0.21	0.48			
	P	0.39	0.05			
	N	18	17			
NJPT EBI	r	0.59	0.43	0.59		
	P	0.01	0.09	0.01		
	N	18	17	18		
VPA B	r	0.34	0.54	0.56	0.20	
	P	0.16	0.09	0.06	0.45	
	N	19	11	12	12	
Losses	r	0.91	0.54	-0.24	0.09	0.33
	P	<0.0001	0.02	0.32	0.72	0.17
	N	26	17	18	18	19
Grand NJGM	r	0.82	0.92	0.70		
Grand NJPT	r	0.92	0.85		0.49	

Table C10.4-3. Correlations of revised NJ EBI with indicators of weakfish biomass. Shading indicates P < 0.05. EBI = exploitable biomass index. WRI = MRFSS catch per trip index. P = proportion of tows with weakfish, W = mean weight of all weakfish, E = exploitable fraction of weight. VPA B = biomass from converged portion of "best" VPA in Kahn (2006; run 20). Losses = all estimated fishery-related losses. Grand Mean = average of Z+2 transformed WRI, DE EBI, and each of the trial NJ EBI. All data are for 1989-2006, but N varies due to time-series.

		NJ Index			
Indicators	Correlation	Aug P*W	Aug P*W*E	Oct P*W	Oct P*W*E
WRI	r	0.54674	0.6212	0.33207	0.5899
	P	0.0189	0.0059	0.1782	0.01
	N	18	18	18	18
DE	r	0.63182	0.42801	0.57519	0.35851
	P	0.0065	0.0865	0.0157	0.1576
	N	17	17	17	17
Losses	r	0.70675	0.42839	0.25463	0.09881
	P	0.001	0.0761	0.3079	0.6965
	N	18	18	18	18
VPA B	r	0.65561	0.07927	0.34805	0.319
	P	0.0206	0.8065	0.2676	0.3122
	N	12	12	12	12
Grand mean	r	0.84511	0.81844	0.78084	0.78199

Table C10.4-4. Harvest-related loss estimates used for biomass dynamic models.

Year	Commercial harvest MT	Commercial discards MT	Recreational harvest MT	Recreational discards mt	Total Harvest Losses
1981	11,958.5	925.5	7305.2	4.3	20,193.4
1982	8,835.3	811.8	3758.2	2.9	13,408.2
1983	7,926.6	597.6	5321.0	4.1	13,849.3
1984	8,969.2	855.2	3181.4	3.7	13,009.5
1985	7,690.0	534.6	2489.8	5.3	10,719.7
1986	9,610.8	616.9	4600.3	34.6	14,862.6
1987	7,743.9	492.8	3061.7	12.7	11,311.1
1988	9,310.7	495.3	2872.0	12.3	12,690.3
1989	6,424.3	450.4	987.6	2.7	7,864.9
1990	4,281.2	265.9	611.1	6.6	5,164.8
1991	3,943.0	284.8	966.4	11.8	5,206.0
1992	3,381.0	294.1	634.6	10.6	4,320.3
1993	3,108.8	385.5	500.0	17.1	4,011.5
1994	2,808.0	990.2	814.4	46.5	4,659.2
1995	3,219.9	716.3	841.7	61.6	4,839.5
1996	3,148.0	985.8	1326.9	75.6	5,536.3
1997	3,310.3	864.9	1675.0	60.3	5,910.5
1998	3,820.7	762.5	1834.8	49.7	6,467.7
1999	3,132.2	746.9	1425.8	42.4	5,347.3
2000	2,449.7	548.7	1884.6	73.1	4,956.1
2001	2,267.8	1,139.4	1235.0	55.6	4,697.8
2002	2,165.1	470.4	994.6	31.8	3,661.8
2003	907.8	238.4	392.3	24.0	1,562.5
2004	691.2	209.6	420.5	28.3	1,349.6
2005	520.4	174.0	720.0	35.2	1,449.6
2006	481.7	199.1	417.2	36.2	1,134.2

Table C10.4-5. Rescaled indices (z-transformed based on common time period + 2 to remove values less than zero), striped bass and Atlantic menhaden biomass estimates (mt), and their ratio used as inputs in weakfish biomass dynamic models.

Year	MRFSS	NJ	DE	Bass Biomass	Menhaden Biomass	Menhaden:Bass
1981	15.4			8789	686249	78.1
1982	7.7			10020	877809	87.6
1983	8.2			7159	551590	77.0
1984	5.0			7534	555868	73.8
1985	4.2			9003	704552	78.3
1986	6.0			11302	778336	68.9
1987	5.2			15597	898324	57.6
1988	4.7			21091	507110	24.0
1989	2.1	2.6		28134	844023	30.0
1990	1.3	1.8	0.9	33116	862413	26.0
1991	1.6	1.7	1.0	37770	805784	21.3
1992	1.6	2.7	1.1	47590	806223	16.9
1993	1.1	1.5	1.8	55335	1062836	19.2
1994	1.9	2.7	2.2	65038	743954	11.4
1995	2.3	3.0	3.0	82562	854719	10.4
1996	3.3	3.1	4.8	96180	739099	7.7
1997	3.2	4.2	2.7	105788	636684	6.0
1998	4.0	2.0	2.8	96550	575810	6.0
1999	3.1	1.6	2.0	95744	482853	5.0
2000	3.1	2.8	2.7	100037	522825	5.2
2001	2.0	2.3	2.1	98382	643163	6.5
2002	2.0	1.5	2.1	100019	491495	4.9
2003	0.8	0.2	1.2	99256	693816	7.0
2004	0.7	1.3	1.2	95472	475986	5.0
2005	1.3	1.2	1.1	94438	496006	5.3
2006	0.9	0.5	1.3	88366	449784	5.1

Table C10.5-1. AIC_c for 12 modeled hypotheses. N = 60 for all models.

Hypothesis	Production fc	Predator-prey fc	K	AICc	Delta AICc	Aikike wt
Depensatory	Gompertz	Bass biomass & menhaden:bass	5	-31.9	0	0.978
Depensatory	Gompertz	Bass biomass only	5	-23.3	8.6	0.013
Predator-prey	Gompertz	Type II weakfish:bass	6	-22.5	9.4	0.009
Depensatory	Schaefer	Bass biomass & menhaden:bass	5	0.8	32.7	0.000
Predator-prey	Schaefer	Type II weakfish:bass	6	10.2	42.1	0.000
Predator-prey	Schaefer	Type III weakfish:bass	6	10.3	42.3	0.000
Fishing	Schaefer	None	4	13.6	45.6	0.000
Predator-prey	Schaefer	Type I weakfish:bass	5	14.1	46.1	0.000
Fishing	Gompertz	None	4	16.1	48.0	0.000
Depensatory	Schaefer	Bass biomass	5	16.3	48.2	0.000
Predator-prey	Gompertz	Type III weakfish:bass	6	19.7	51.6	0.000
Predator-prey	Gompertz	Type I weakfish:bass	5	26.9	58.8	0.000

Table C10.5-2. Parameter estimates and their distribution based on jackknifing. N = 60. Original is the base run with all data.

Statistic	r	K	B_{1981}	c
Mean	0.482	51,481	81,631	0.288
Median	0.482	51,521	82,471	0.292
Maximum	0.484	51,788	82,473	0.292
Minimum	0.469	48,698	70,164	0.242
5th%	0.481	51,511	79,980	0.272
95th%	0.483	51,555	82,472	0.292
Original	0.482	51,521	82,472	0.292

Table C10.5-3. Parameter estimates and their distribution based on bootstrapping. N = 500, excluding the initial run. Original is the base run with all data.

Statistic	r	k	B_{1981}	c
Mean	0.48190	51522	82472	0.2924
Median	0.48190	51521	82472	0.2924
Min	0.48190	51521	82466	0.2923
Max	0.48195	51529	82472	0.2924
5th%	0.48190	51521	82472	0.2924
95th%	0.48192	51525	82472	0.2924
Original	0.48190	51521	82472	0.2924

Table C10.5-4. Estimates of model parameters when blocks of years were removed from the beginning of the time-series or biomass in 1981 was constrained to be less than K. MSR = mean of the squared residuals.

Treatment	year 0	r	K	Bo	c	MSR
All	1981	0.48	51521	82472	0.29	0.087
1982 +	1982	0.47	48672	70534	0.24	0.090
1983 +	1983	0.47	47339	47392	0.25	0.089
1984 +	1984	0.46	47748	42343	0.27	0.112
1985 +	1985	0.44	45979	35334	0.21	0.106
1986 +	1986	0.34	59197	30501	0.20	0.110
1987 +	1987	0.35	58297	21384	0.20	0.106
1988 +	1988	0.35	59365	18352	0.24	0.108
1989 +	1989	0.33	78130	12951	0.35	0.089
Constrain K	1981	0.31	72800	67907	0.26	0.102

Table C10.5-5. Changes in GDR model parameters when different weights were assigned to released MRFSS weakfish catch in the WRI and recreational discard losses.

Weight assigned (kg)	r	k	Bo	c
Base - 0.15	0.48	51521	82472	0.29
Discard = 0.2	0.44	56783	62349	0.28
Discard = 0.3	0.44	58539	55852	0.31
Discard = 0.4	0.41	57148	79360	0.27
Harvest wt (median=0.7)	0.43	61322	49899	0.34

Table C10.6-1. Predicted mean weight-at-age of 340 mm weakfish sampled in the NEFSC fall trawl survey during 1992-2006. Slopes and intercepts describe ln-transformed total length (cm).

Year	Intercept	Slope	Predicted kg
1992	-11.60	3.02	0.39
1993	-11.82	3.08	0.38
1994	-11.34	2.95	0.39
1995	-11.38	2.94	0.37
1996	-11.35	2.93	0.36
1997	-11.39	2.94	0.35
1998	-11.32	2.93	0.37
1999	-11.47	2.98	0.38
2000	-11.25	2.90	0.36
2001	-11.54	2.99	0.37
2002	-11.67	3.02	0.36
2003	-10.54	2.67	0.32
2004	-11.35	2.92	0.35
2005	-11.39	2.93	0.35
2006	-11.49	2.96	0.35

Table C10.6-2. Statistics and parameter estimates of predicted weight of weakfish at 340 mm (Q_{wt}) during 1992-2006. Models tested for linear trend of Q_{wt} with year (Model = Year), striped bass biomass estimates (Model = Bass), and the ratio of menhaden to striped bass biomass (Model = Ratio). Statistics and parameter estimates are presented for all years and with a potential outlier removed.

Model	r^2	*P*	Slope	Slope SE	Intercept	Intercept SE
			All years			
Year	0.57	0.0011	-0.003	0.0007	6.36	1.45
Bass	0.51	0.0027	$-7.2\ 10^{-07}$	$1.94\ 10^{-07}$	0.43	0.02
Ratio	0.41	0.0107	0.0025	0.0008	0.34	0.01
			2003 removed			
Year	0.62	0.0009	-0.00256	0.0006	5.48	1.16
Bass	0.61	0.0010	$-6.3\ 10^{-07}$	$1.46\ 10^{-07}$	0.42	0.01
Ratio	0.56	0.0020	0.0024	0.0006	0.35	0.01

Table C10.6-3. Estimates of annual biomass of food consumed per biomass of striped bass (Ct) from three Chesapeake Bay bioenergetics studies. Years = years included in bioenergetics estimates; 1955-1959 = Griffin (2001) and Griffin and Margraf (2003); 1990-1992 = Hartman and Brandt (1995a: 1995b); and 1998-2000 = Overton (2003). Shading highlights minimum, maximum, and median estimates of Ct.

Years	1955-1959	1990-1992	1998-2000	1998-2001	1998-2002
Region	Bay	Bay	Upper Bay	Middle Bay	Lower Bay
Age	Ct	Ct	Ct	Ct	Ct
2	6.2	6.3	4.1	6.2	7.9
3	5.6	5.6	5.1	6.1	5.3
4	5.1	5.5	5.6	6.2	6.3
5	4.7	5.4	5.5	6.2	5.9
6	4.3	5.0	5.1	4.2	4.9

Table C10.6-4. Annual percentage of striped bass stomachs with food (SSt; Overton et al. 2008) and model estimates of weakfish biomass consumed per striped bass biomass (Dt / Pt) annually. MR_t = time category assigned to menhaden: striped bass (> 10 = 1 or < 10 = 0).

Year	SSt (%)	MRt	Dt / Pt
1994	99	0	0.026
1995	100	0	0.028
1996	74	0	0.038
2000	84	1	0.056
2002	77	1	0.060
2003	84	1	0.042
2005	23	1	0.059
2006	28	1	0.056
2007	81	1	0.057

Table C10.6-5. Summary of results of final multiple regression models describing the relationships of annual weakfish biomass consumed (Dt / Pt) and percentage of striped bass stomachs with food (SSt; Overton et al. 2008)

Dependent	1 / (Dt / Pt)
Independent	SSt
Slope	0.20442
Slope SE	0.041
Intercept	15.35
Intercept SE	3.31
Category	N/A
Category SE	N/A
Slope shift	-0.16
Slope Shift SE	0.03
P	0.0013
R^2	0.89

Table C10.7-1. Comparison of AIC_c for Gompertz production models excluding and including additional loss terms that describe striped bass related or additional bycatch estimates. GDR is the model selected to represent biomass dynamics.

Model	Loss fc	K	AICc	Delta AICc	Aikike wt
GDR	Bass biomass & menhaden:bass	5	-41	0	0.908
Added Bycatch	Constant * estimated bycatch	5	17	58	0.000
Fishery Only	No additional losses	5	16	57	0.000
Added Bycatch	Power fc since 1996	6	-24	16	0.000
Added Bycatch	Linear since 1996	6	-27	14	0.001
Added Bycatch	Quadratic since 1996	7	-34	7	0.031
Added Bycatch	Constant addition since 1996	5	-35	5	0.060

Table C11.3-1. Results of correlation analysis between Atlantic Multidecadal Oscillation (AMO) and weakfish commercial landings. A) NMFS landings only; B) Landings from Joseph (1972) for years 1929 to 1949 and NMFS landings 1950 to present.

A)		No lag	1 yr lag	2 yr lag	3 yr lag	4 yr lag	5 yr lag	6 yr lag	7 yr lag	8 yr lag	9 yr lag	10 yr lag
	r	-0.62416	-0.68332	-0.73352	-0.77764	-0.8094	-0.8303	-0.81922	-0.78916	-0.74835	-0.68789	-0.61433
	P	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001
	N	54	55	56	57	58	58	58	58	57	56	55

B)		No lag	1 yr lag	2 yr lag	3 yr lag	4 yr lag	5 yr lag	6 yr lag	7 yr lag	8 yr lag	9 yr lag	10 yr lag
	r	-0.15067	-0.23159	-0.2832	-0.31004	-0.33631	-0.34733	-0.35973	-0.37929	-0.38121	-0.37908	-0.35678
	P	0.2131	0.0555	0.0193	0.0101	0.0054	0.0043	0.0032	0.002	0.0019	0.0022	0.0044
	N	70	69	68	68	67	66	65	64	64	63	62

Table C11.3-2. Correlation of NEFSC weakfish food habit data with biomass weighted Z for ages 1-5 estimated by ADAPT. * = strong negative correlation at α = 0.10; ** = strong positive correlation α = 0.10.

		Amphi_Iso	Crab_ Shrimp	Empty	Forage
5"	r	0.5184	-0.7412	0.7706	0.7413
	P	0.1023	0.009*	0.0055**	0.009**
	N	11	11	11	11
6"	r	0.6154	-0.7411	0.7178	-0.6568
	P	0.0438**	0.0091*	0.0129**	0.0281*
	N	11	11	11	11
7"	r	0.6320	-0.5050	0.6198	-0.2746
	P	0.037**	0.1131	0.0419**	0.4138
	N	11	11	11	11
8"	r	0.8188	-0.1846	0.1784	-0.7272
	P	0.0021**	0.5868	0.5996	0.0112*
	N	11	11	11	11
9"	r	0.7016	-0.6440	0.4331	-0.5495
	P	0.0161**	0.0325*	0.1834	0.08*
	N	11	11	11	11
10"	r	0.6694	-0.5237	0.8578	-0.8543
	P	0.0243**	0.0983*	0.0007**	0.0008*
	N	11	11	11	11
11"	r	0.2441	-0.4096	0.7312	-0.1466
	P	0.4695	0.2110	0.0106**	0.6672
	N	11	11	11	11
12"	r	-0.1972	0.0005	0.1715	-0.2782
	P	0.5610	0.9989	0.6142	0.4075
	N	11	11	11	11
All	r	0.6764	-0.6086	0.7993	-0.7305
	P	0.0223**	0.0469*	0.0032**	0.0107*
	N	11	11	11	11

Table C12.4-1. Comparison of reference point estimates and weakfish stock status determinations under equilibrium and non-equilibrium conditions from ADAPT VPA, the full Steele-Henderson production model incorporating predation, and the Gompertz production model incorporating predation-competition losses as a function of striped bass biomass and the menhaden:striped bass ratio (GDR). Fishing mortality rates are numbers weighted for ADAPT and biomass weighted for Steele-Henderson and GDR.

		Equilibrium Conditions			**Non-equilibrium Conditions**	
Model	**Reference point**	**Threshold value**	**2007 value exceeds threshold***		**Threshold value**	**2007 value exceeds threshold***
ADAPT VPA	$F_{20\%}$	0.42	Yes		--	--
	$SSB_{20\%}$	10,179 MT	Yes		--	--
Steele-Henderson	F_{MSY}	0.72	No		0.36	Yes
	SSB_{MSY}	17,009 MT	Yes		19,681 MT	Yes
	MSP				20%	Yes
GDR	F_{MSY}	0.48	No		Z_{MSY} = 0.48	Yes
	SSB_{MSY}	18,941 MT	Yes		--	--
	MSP				20%	Yes

* "Exceeds" interpreted here as $F_{2007} > F_{threshold}$ or $SSB_{2007} < SSB_{threshold}$

Table C13.2-1. Relative weakfish recruitment (Rec0), fully recruited fishing mortality (F) and natural mortality (M) used in the HCM projections of weakfish SSB among the three scenarios from 1980 to 2020.

		Scenario 1		Scenario 2		Scenario 3	
	Rec0	F	M	F	M	F	M
1980	230	1.29	0.25	1.29	0.25	1.29	0.25
1981	55.7	1.73	0.25	1.73	0.25	1.73	0.25
1982	18.4	1.40	0.25	1.40	0.25	1.4	0.25
1983	35	1.61	0.25	1.61	0.25	1.61	0.25
1984	33.9	2.21	0.25	2.21	0.25	2.21	0.25
1985	18.3	1.35	0.25	1.35	0.25	1.35	0.25
1986	8	1.73	0.25	1.73	0.25	1.73	0.25
1987	5.9	2.66	0.25	2.66	0.25	2.66	0.25
1988	7.4	1.81	0.25	1.81	0.25	1.81	0.25
1989	1.9	2.12	0.25	2.12	0.25	2.12	0.25
1990	5.7	2.76	0.25	2.76	0.25	2.76	0.25
1991	5.5	0.84	0.25	0.84	0.25	0.84	0.25
1992	43.4	1.03	0.25	1.03	0.25	1.03	0.25
1993	88.6	0.71	0.25	0.71	0.25	0.71	0.25
1994	28.5	0.98	0.25	0.98	0.25	0.98	0.25
1995	10.7	0.32	0.25	0.32	0.25	0.32	0.25
1996	17.1	0.53	0.25	0.53	0.25	0.53	0.25
1997	14.7	0.56	0.25	0.56	0.25	0.56	0.25
1998	4.6	0.72	0.25	0.45	0.25	0.45	0.25
1999	17.3	2.67	0.25	1.02	0.65	0.37	0.65
2000	4	2.82	0.25	1.17	0.65	0.52	0.75
2001	9.8	2.04	0.25	1.19	0.65	0.54	0.75
2002	11.7	2.34	0.25	0.99	0.65	0.34	0.95
2003	3.6	2.38	0.25	1.03	0.65	0.38	1.00
2004	4.6	2.06	0.25	1.11	0.65	0.46	1.00
2005	12	2.52	0.25	1.17	0.65	0.52	1.00
2006	5.7	2.47	0.25	1.12	0.65	0.47	1.00
2007	21.5	2.13	0.25	1.08	0.65	0.43	1.00
2008	8.5	2.20	0.25	1.08	0.65	0.43	1.00
2009	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2010	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2011	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2012	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2013	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2014	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2015	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2016	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2017	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2018	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2019	16.5	2.20	0.25	1.08	0.65	0.43	1.00
2020	16.5	2.20	0.25	1.08	0.65	0.43	1.00

Figures

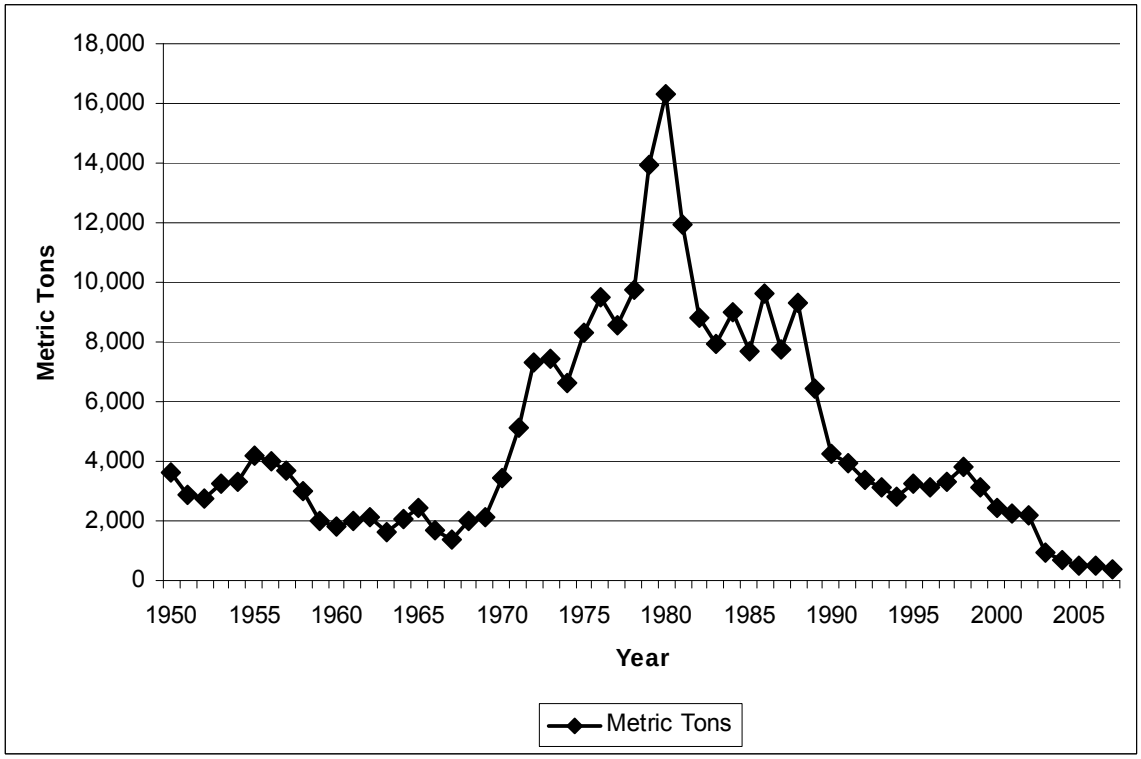


Figure C4.6-1. Commercial harvest of weakfish on the Atlantic coast.

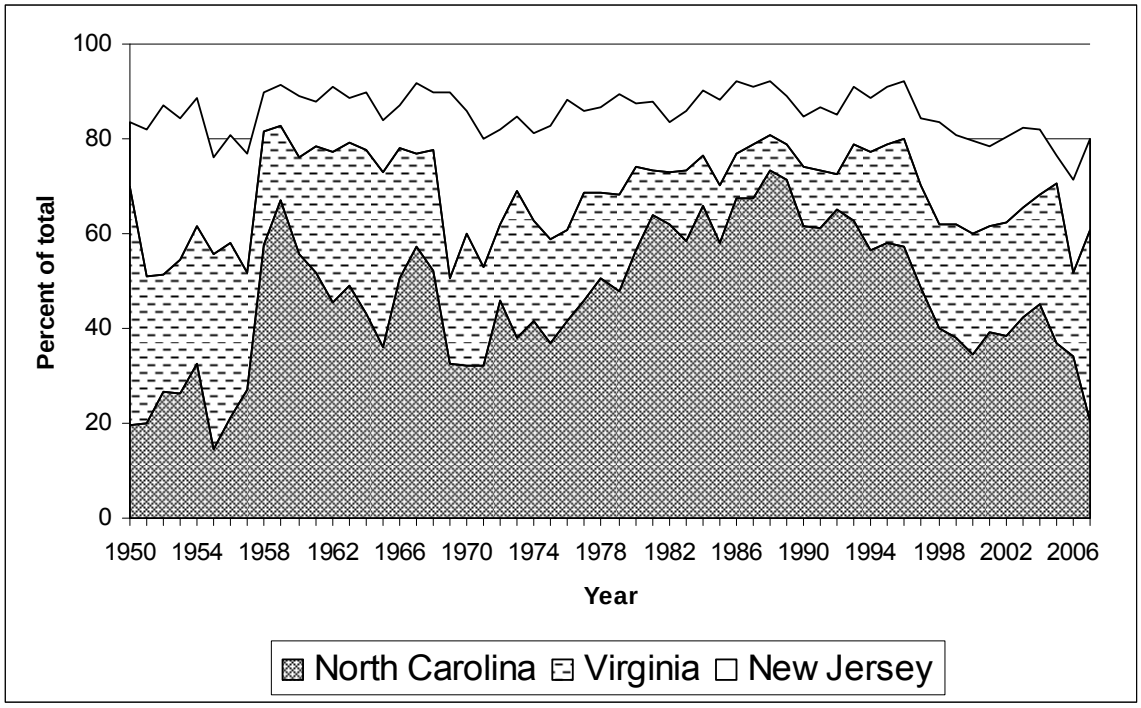


Figure C4.6-2. Proportion of annual commercial weakfish harvest by dominant states.

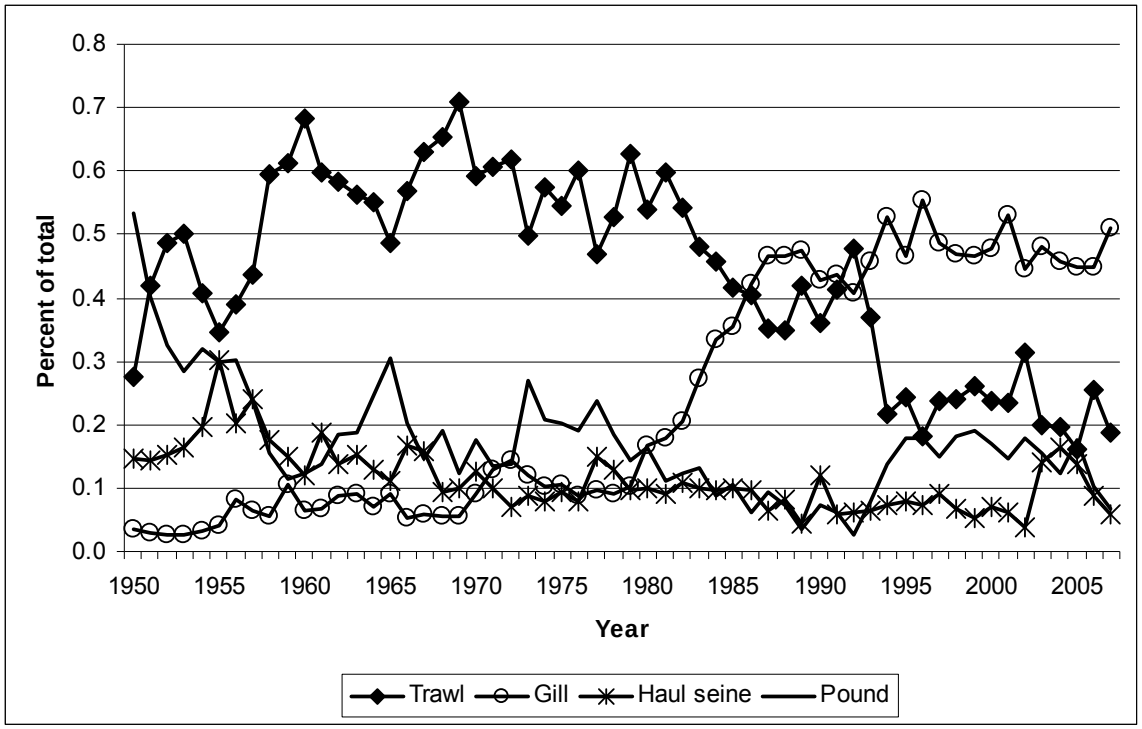


Figure C4.6-3. Proportion of annual coastwide commercial weakfish harvest by dominant gears.

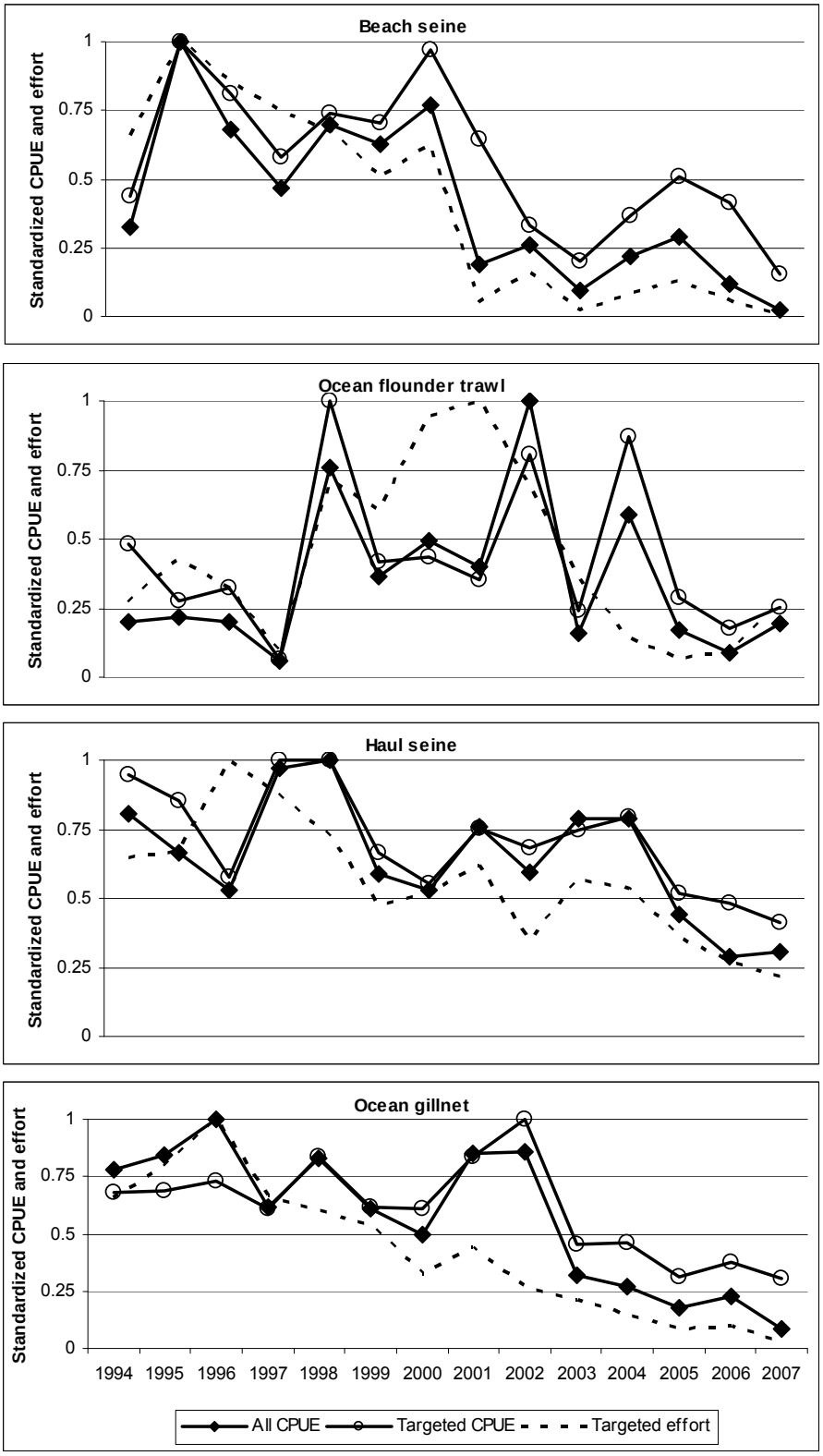


Figure C4.6-4. Standardized commercial CPUE and effort from eight North Carolina fisheries. All CPUE = CPUE from all positive trips; "Targeted" = trips with greater than 150 lbs of weakfish.

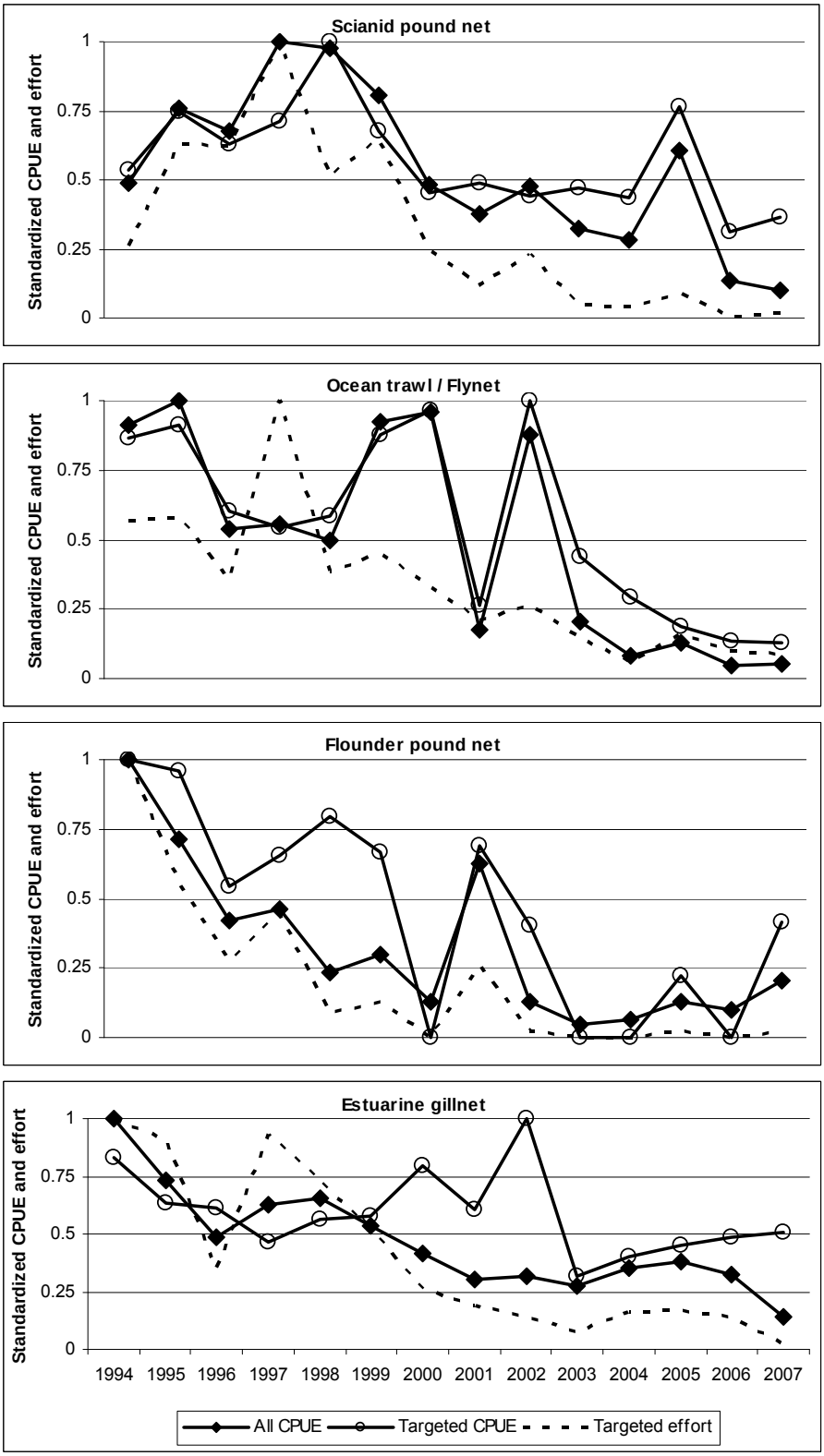


Figure C4.6-4 (continued). Standardized commercial CPUE and effort from eight North Carolina fisheries. All CPUE = CPUE from all positive trips; "Targeted" = trips with greater than 150 lbs of weakfish.

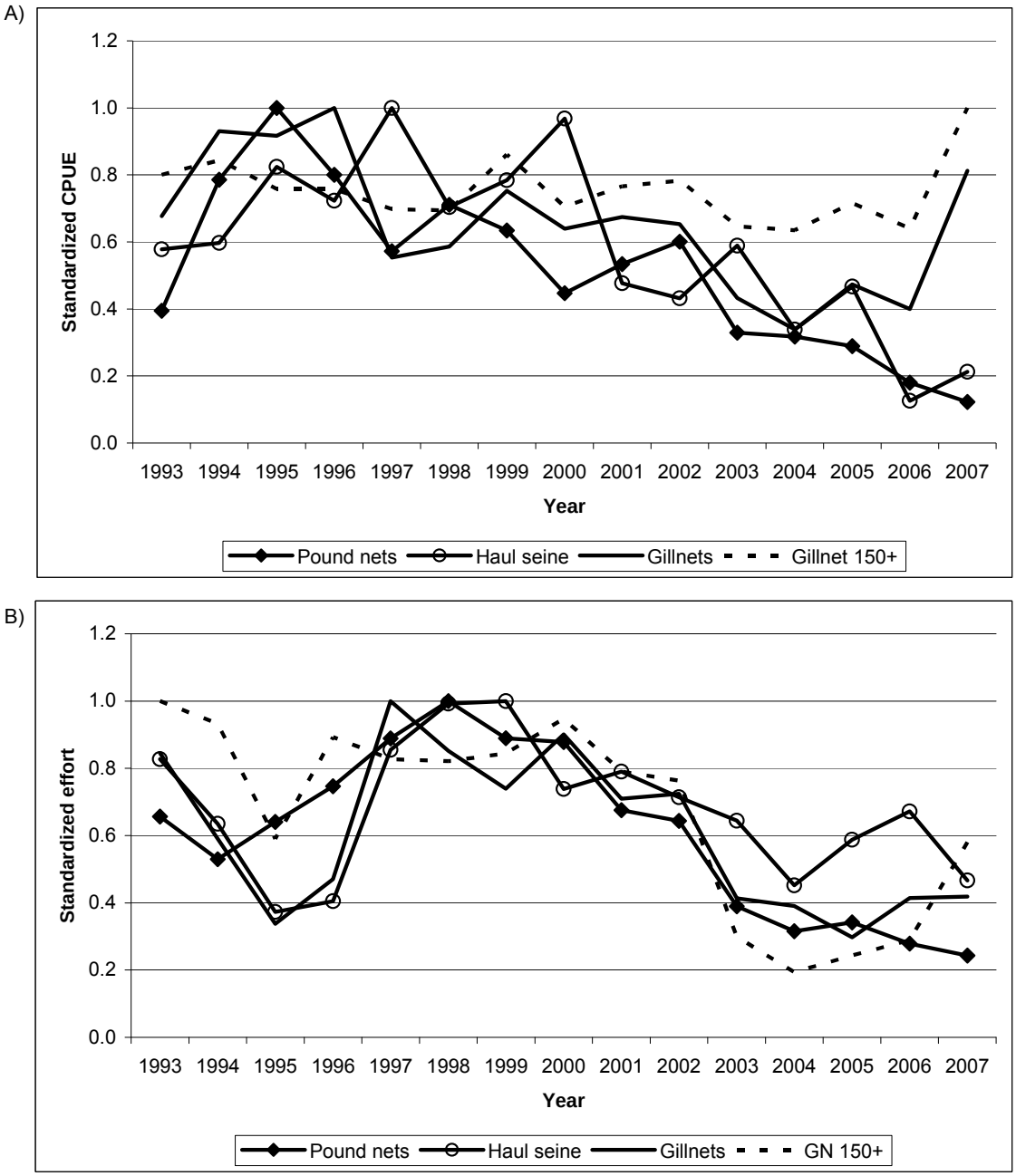


Figure C4.6-5. Standardized commercial CPUE and effort from three Virginia fisheries. A) CPUE; B) Effort. GN 150+ = gillnet trips with 150 pounds or more of weakfish.

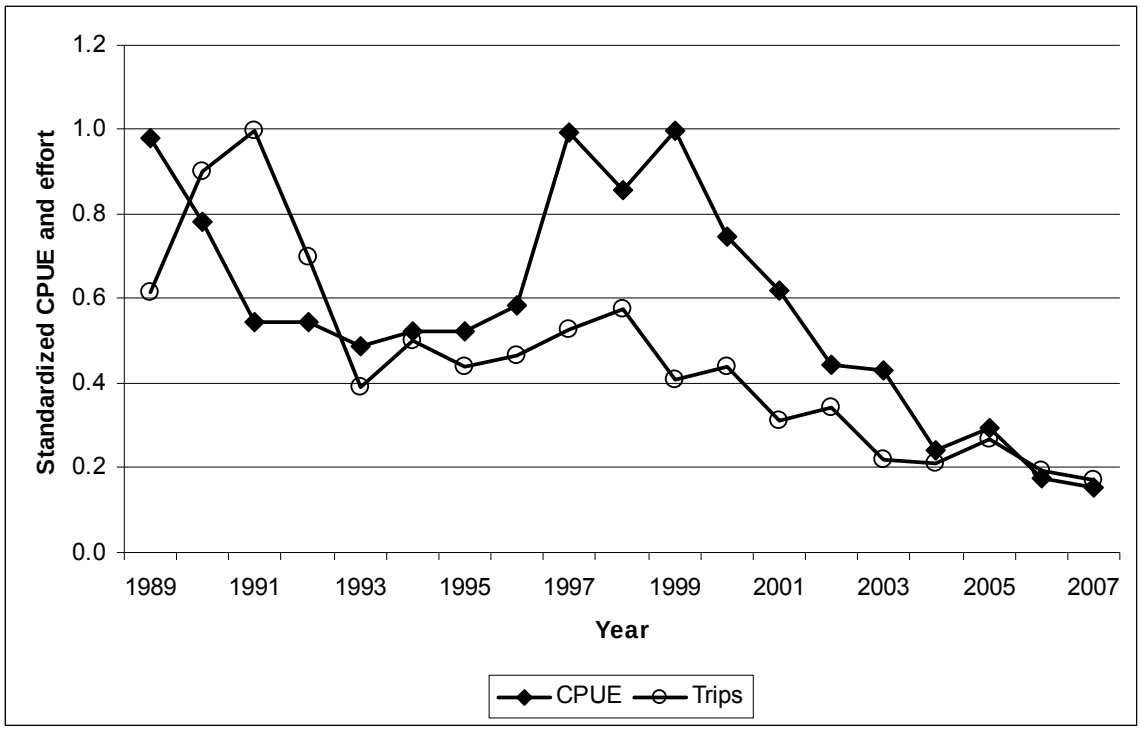


Figure C4.6-6. Standardized commercial CPUE and effort from Delaware's gillnet fishery.

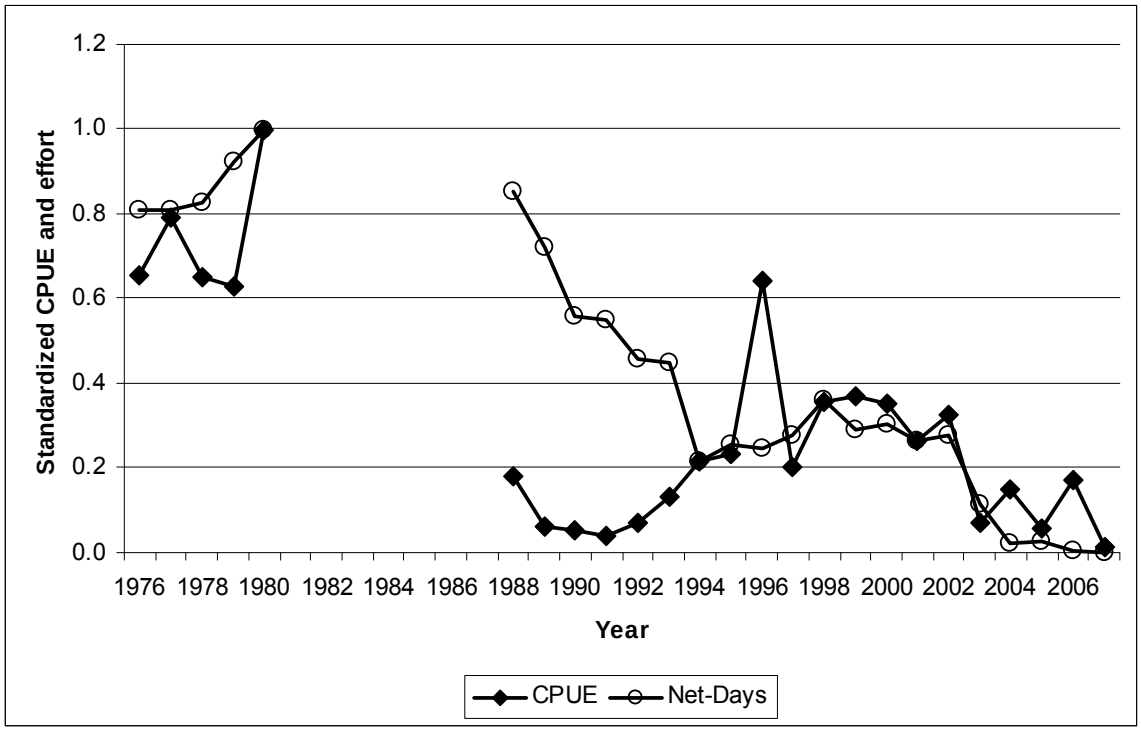


Figure C4.6-7. Standardized commercial CPUE and effort from the Potomac River pound net fishery.

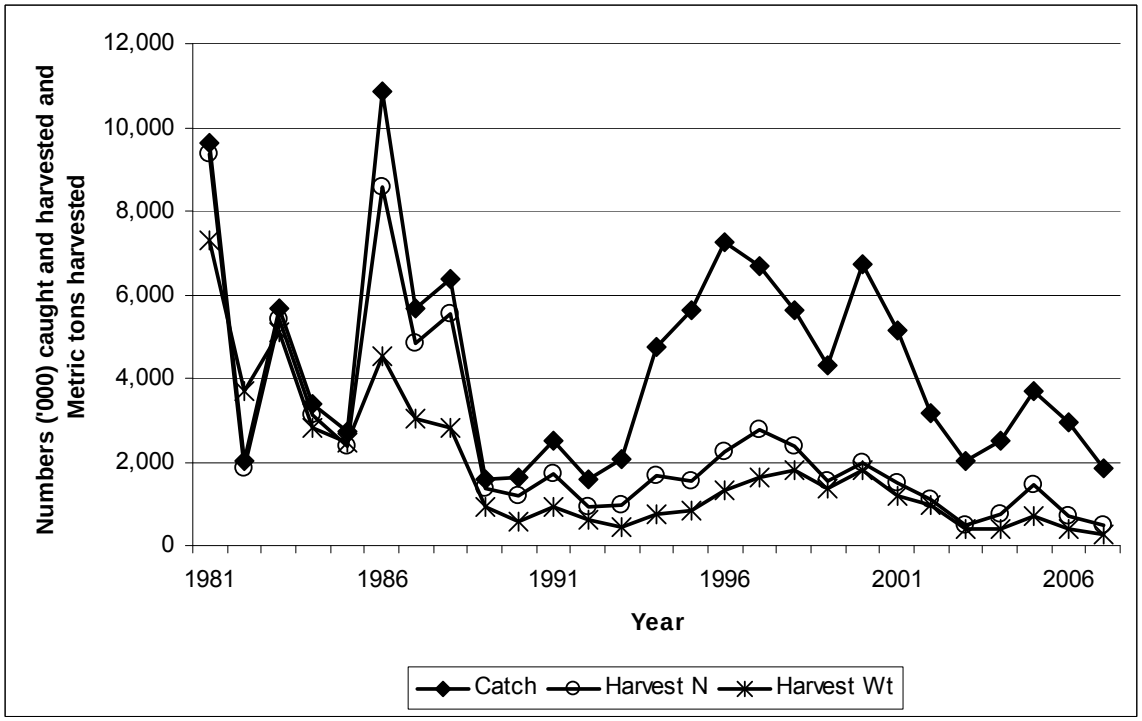


Figure C4.6-8. Recreational catch (thousands), harvest numbers (thousands) and harvest weight (MT) of weakfish on the Atlantic coast.

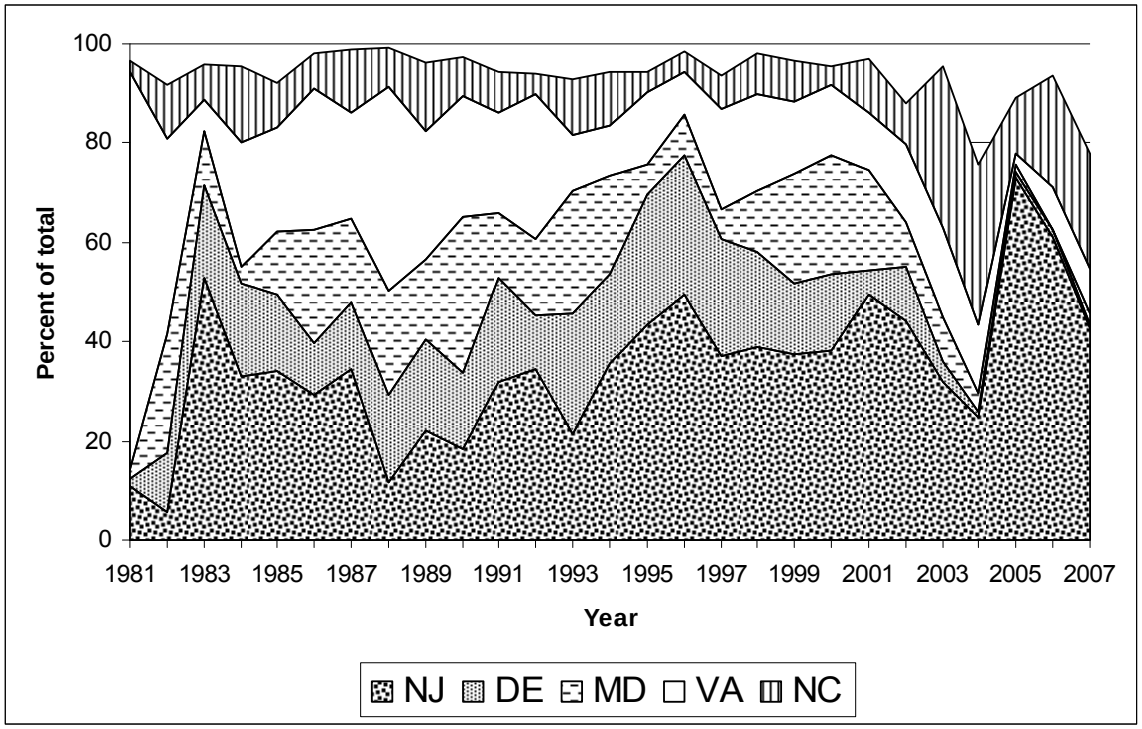


Figure C4.6-9. Proportion of annual recreational weakfish harvest by dominant states.

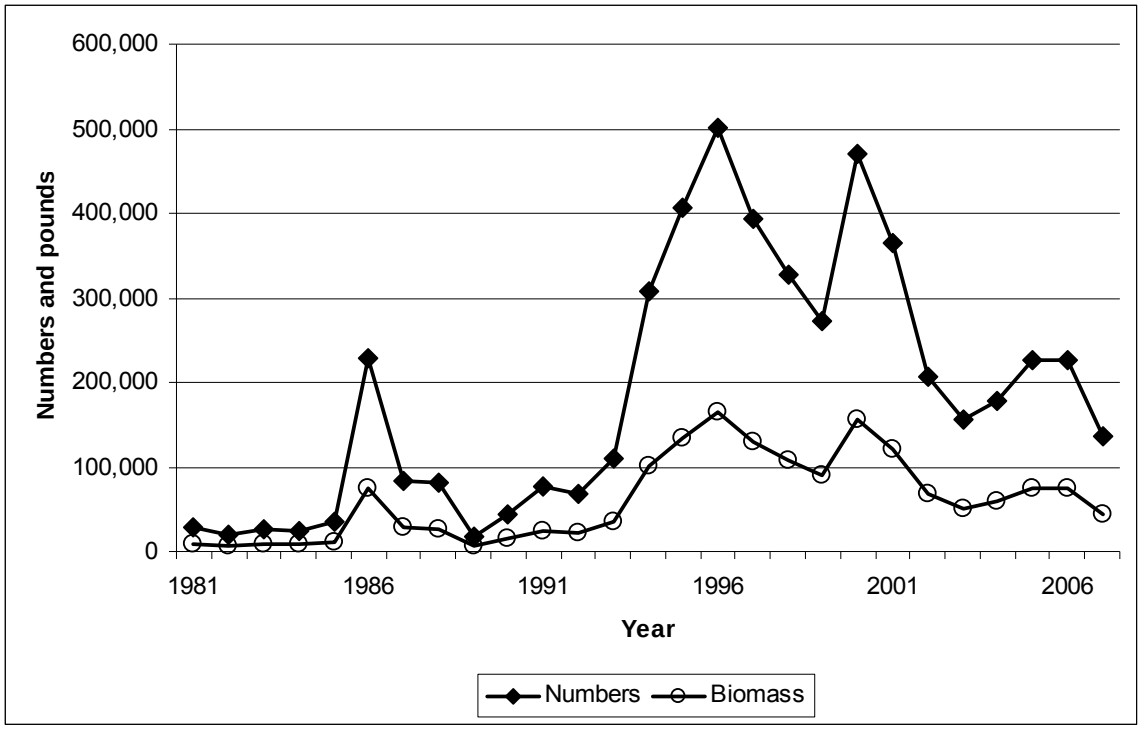
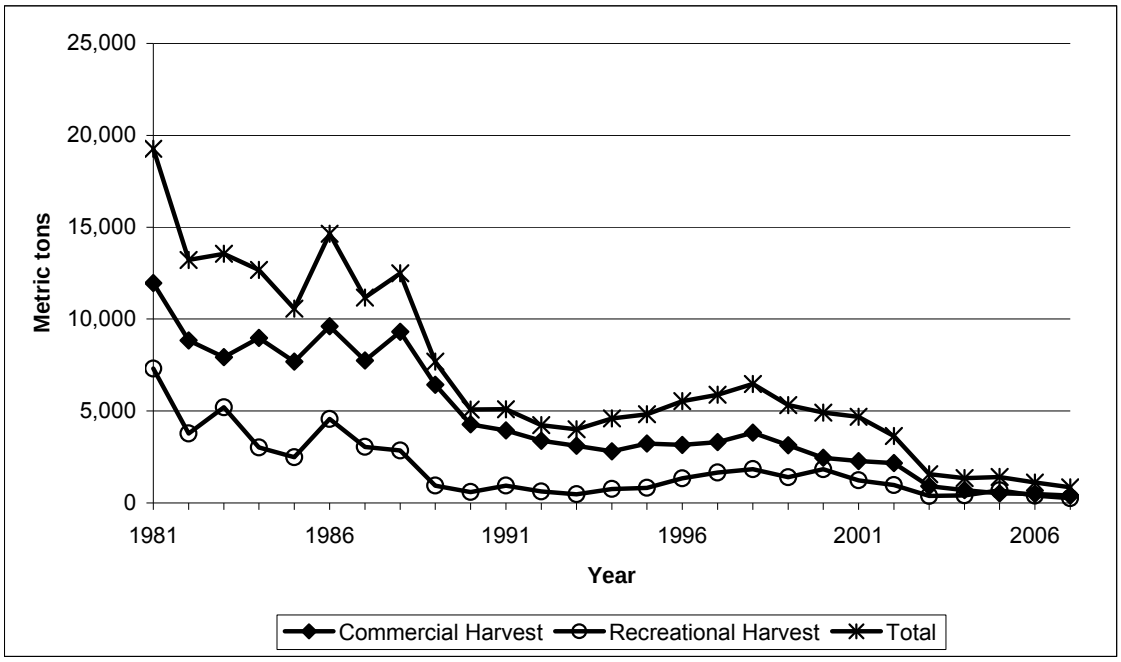


Figure C4.6-10. Recreational discard losses assuming a 10% discard mortality rate.

A)



B)

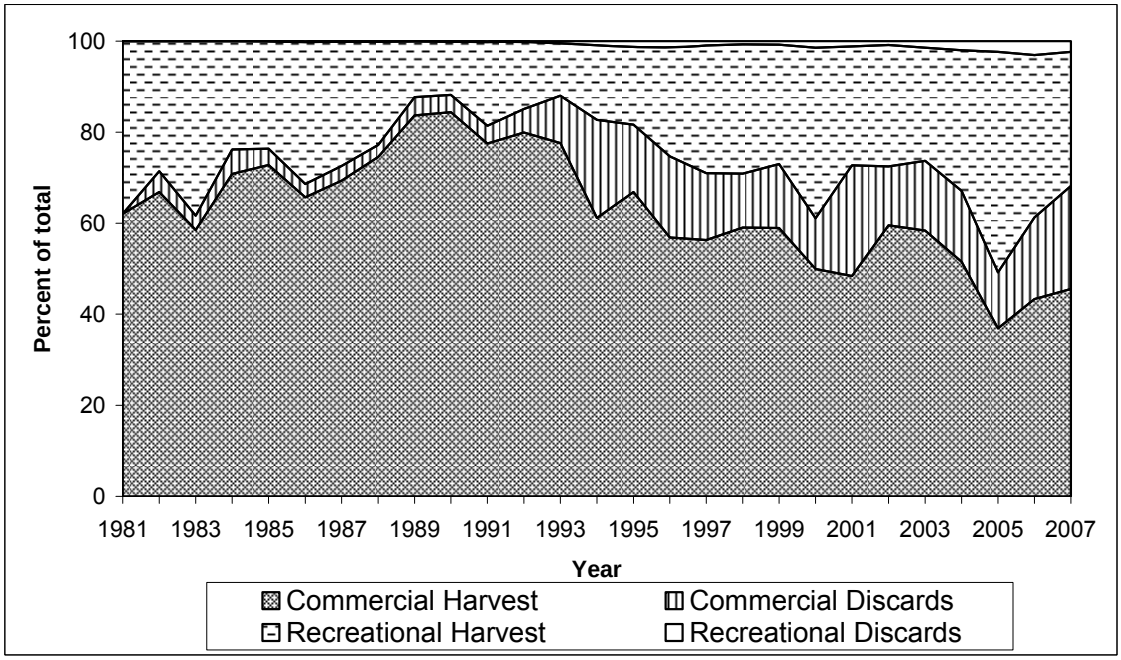
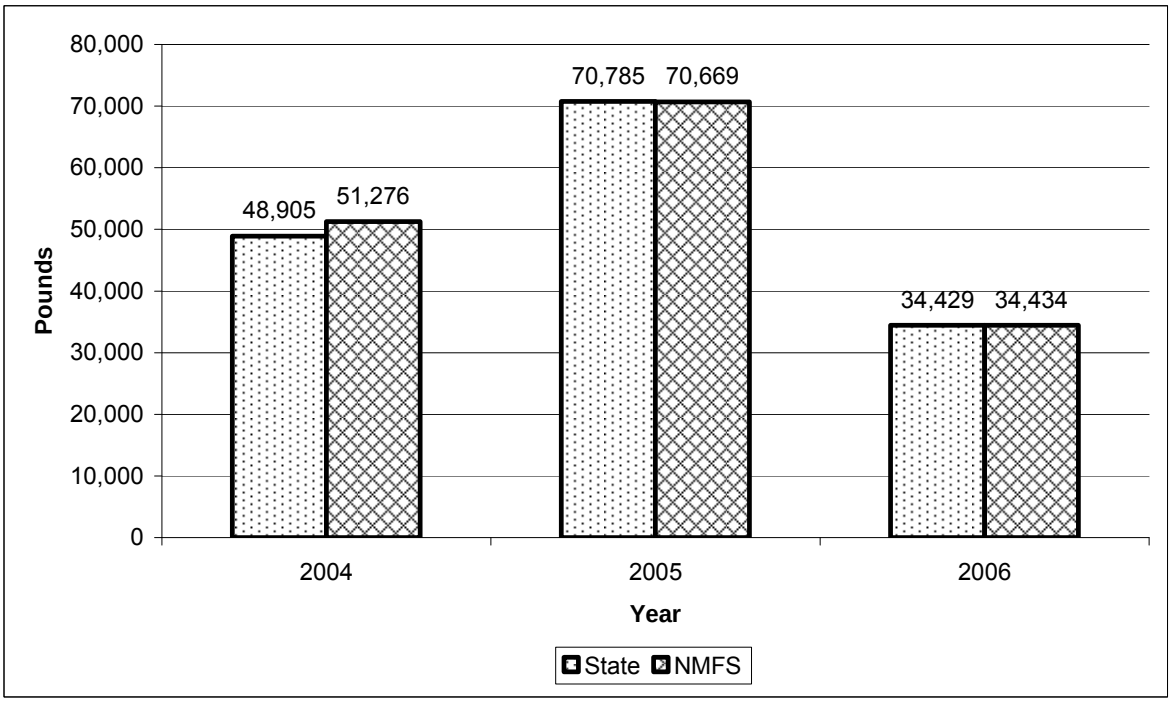


Figure C4.6-11. Total weakfish removals. A) Harvest weight (metric tons) for the two principal sectors and all four sectors combined; B) Percent of total biomass removals by sector.

A)



B)

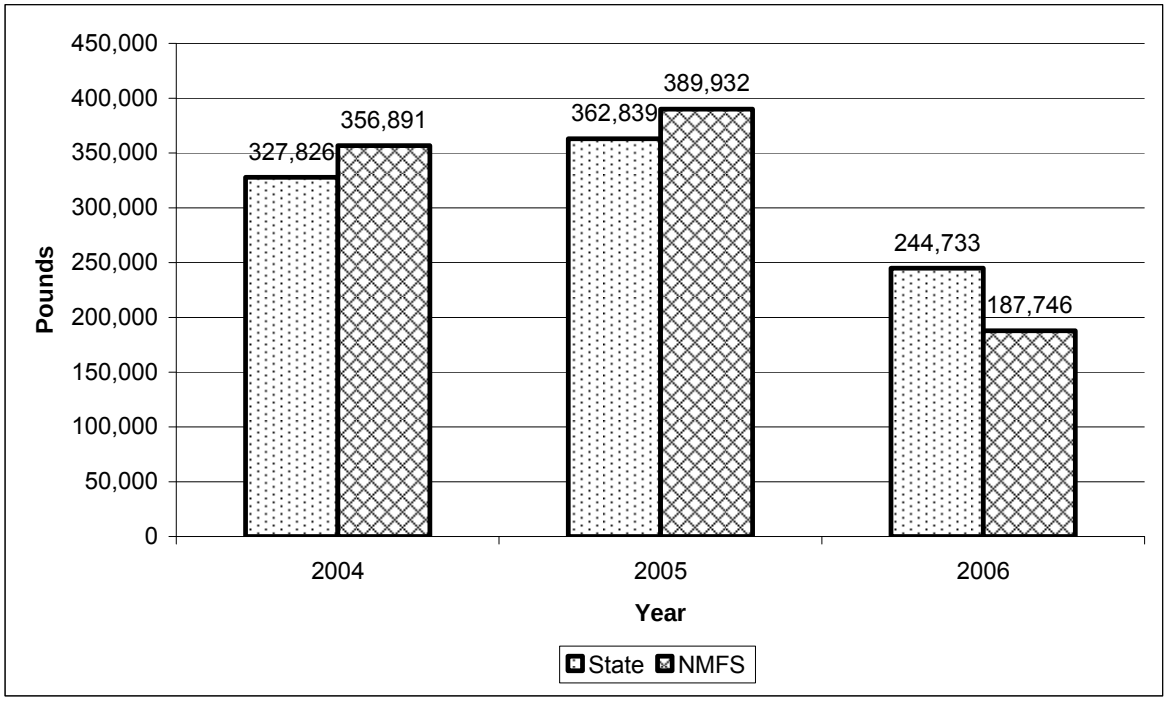
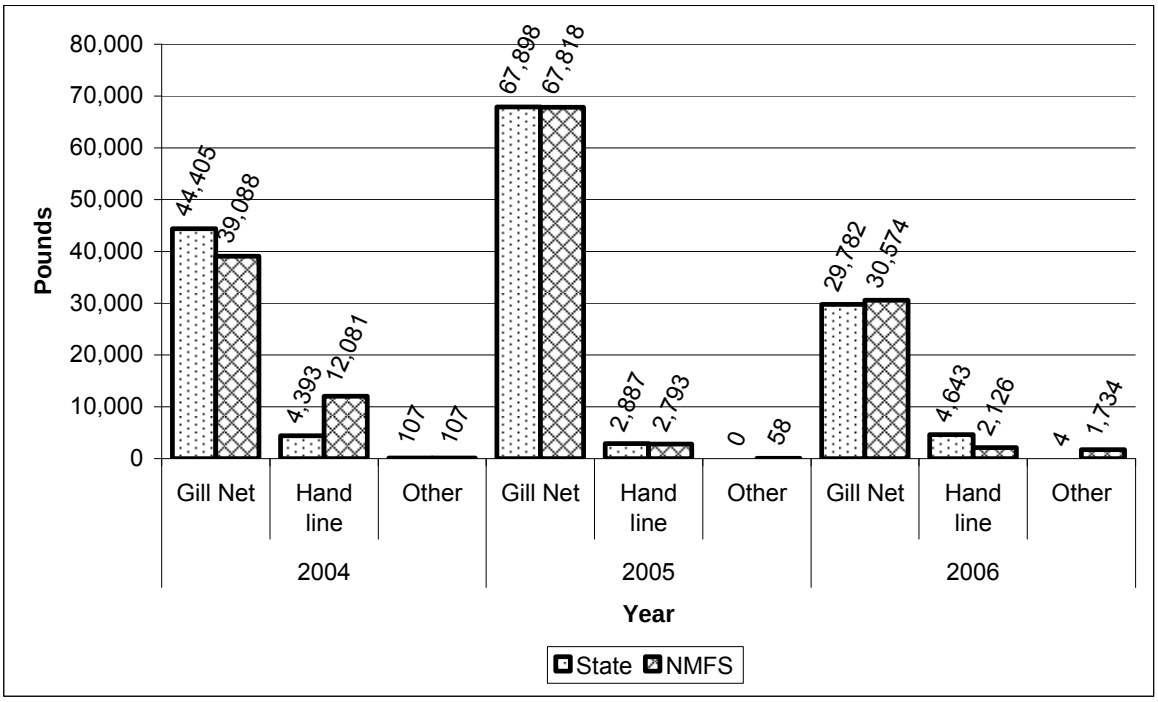


Figure C5.1-1. Comparison of state and federally reported landings on an annual basis for A) Delaware and B) Virginia.

A)



B)

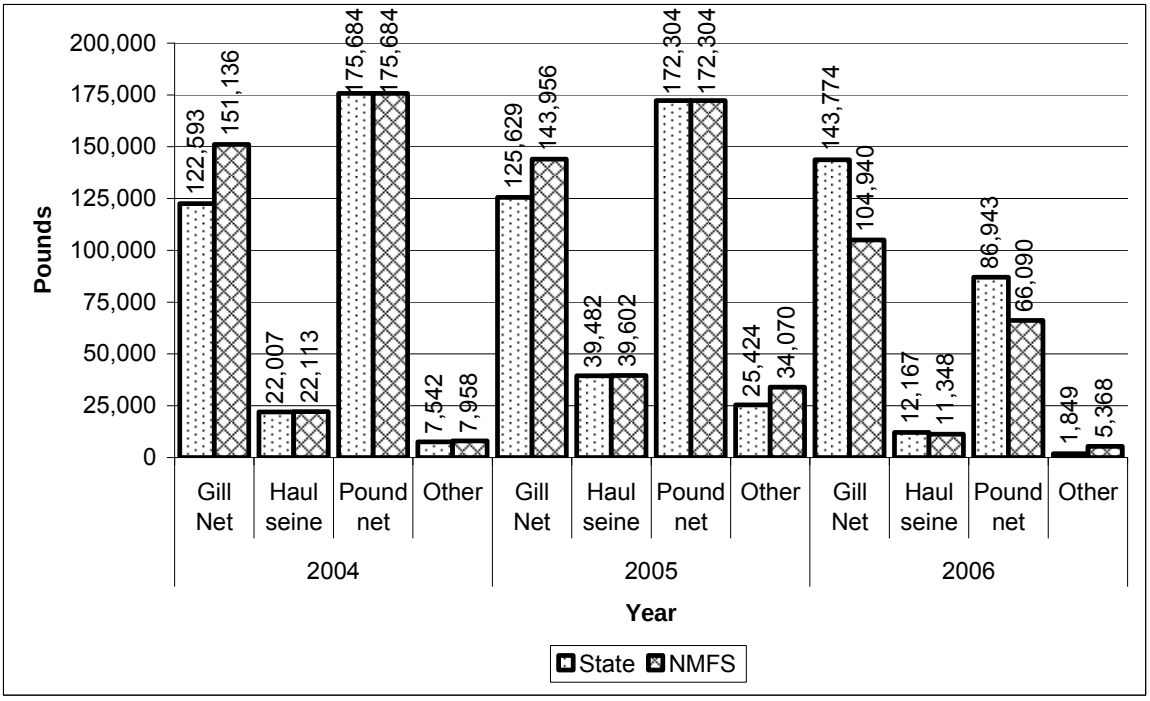


Figure C5.1-2. Comparison of state and federally reported data by gear for A) Delaware and B) Virginia

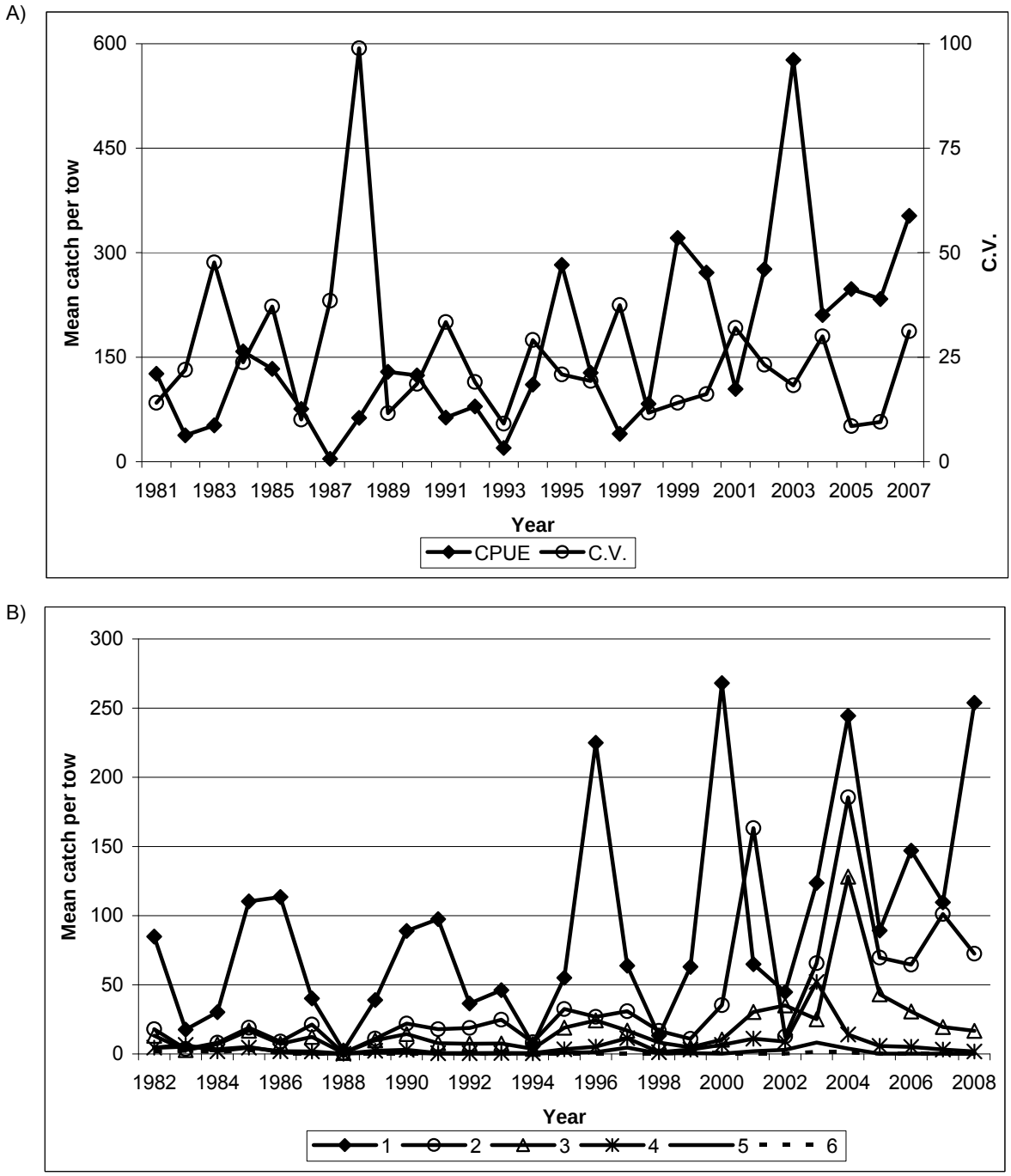


Figure 6.1-1. Results of the NEFSC fall trawl survey. A) Age aggregated CPUE and CV in true survey year; B) Age specific CPUE progressed one year and age

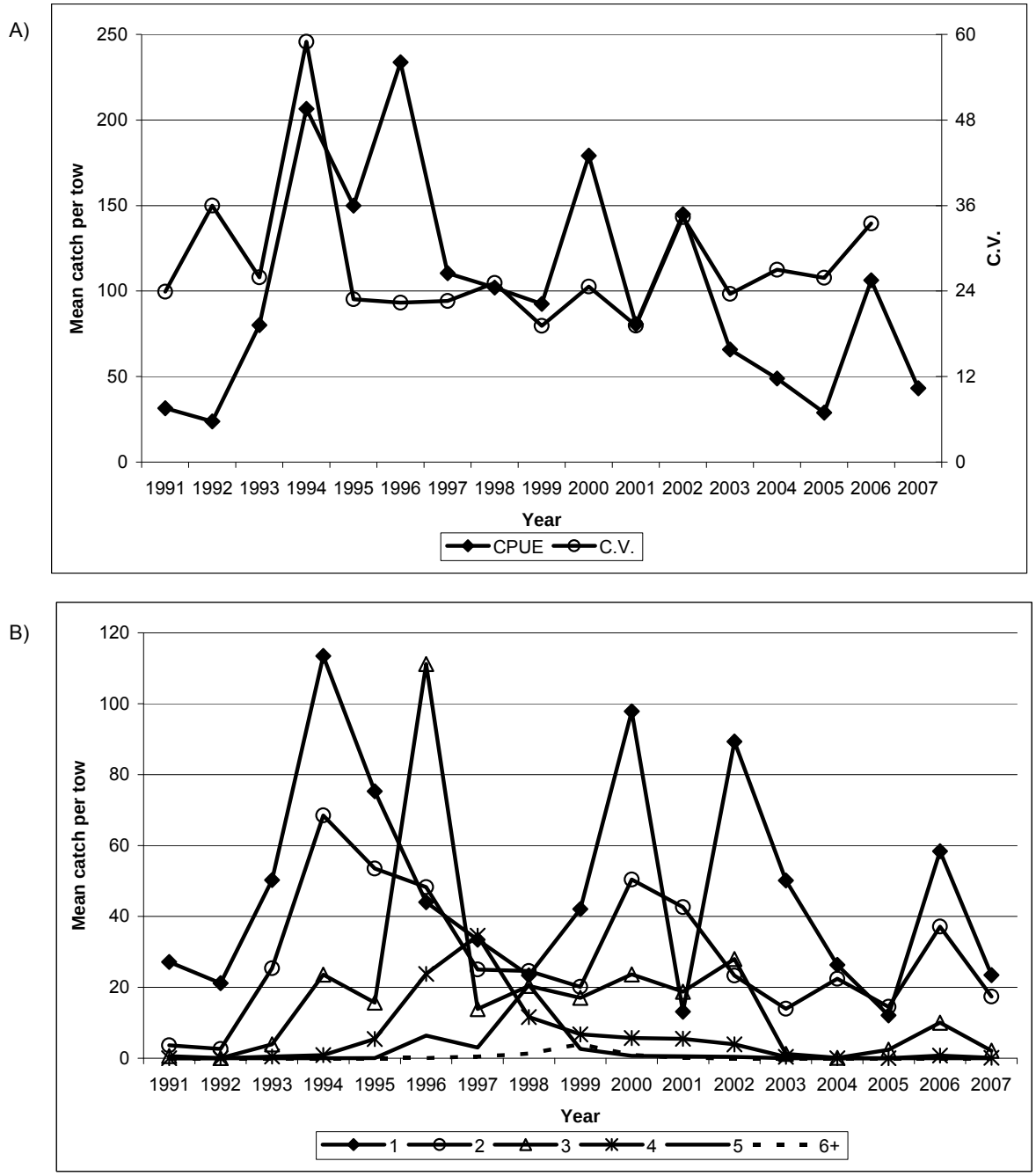


Figure 6.1-2. Results of the New Jersey ocean trawl survey August cruise. A) Age aggregated PPT with 90% confidence intervals; B) Age specific PPT. Survey is not lagged, so all values are in the true survey year.

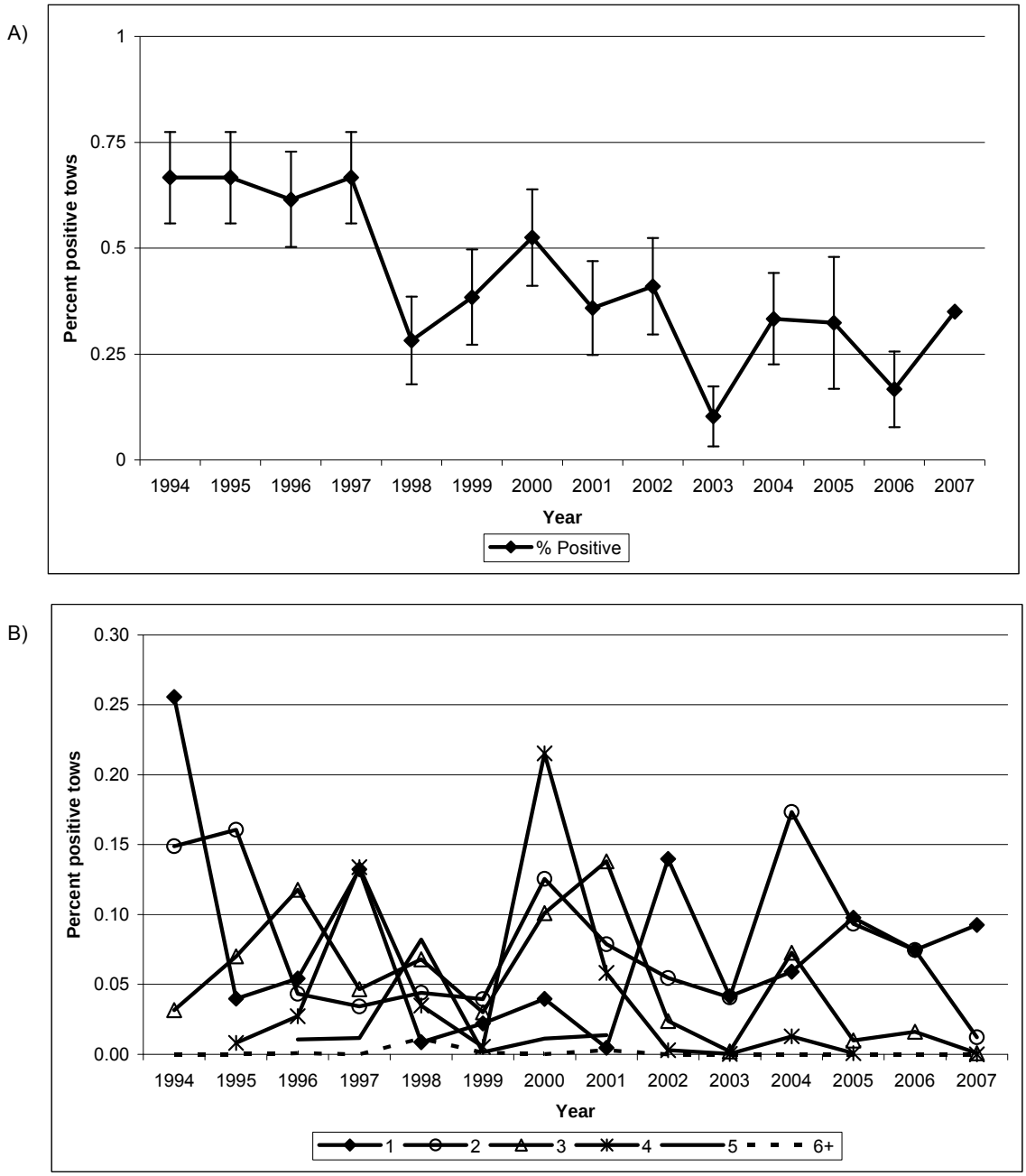


Figure 6.1-3. Results of the DNREC Delaware Bay 30-foot trawl survey. A) Age aggregated CPUE (ages 1+) and CV (ages 0+); B) Age specific CPUE. Survey is not lagged, so all values are in the true survey year.

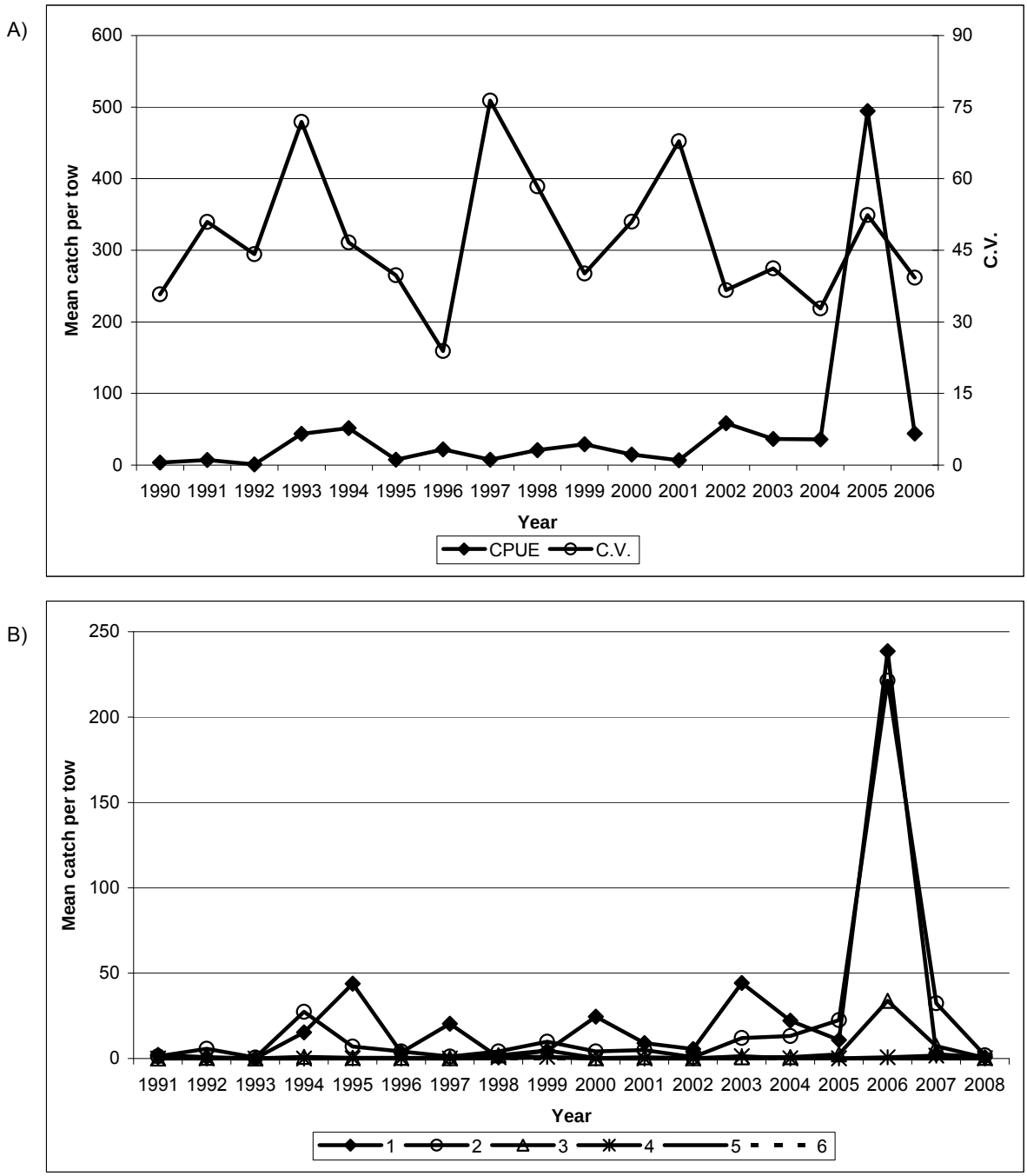


Figure 6.1-4. Results of the SEAMAP trawl survey in North Carolina waters. A) Age aggregated CPUE and CV in true survey year; B) Age specific CPUE progressed one year and age.

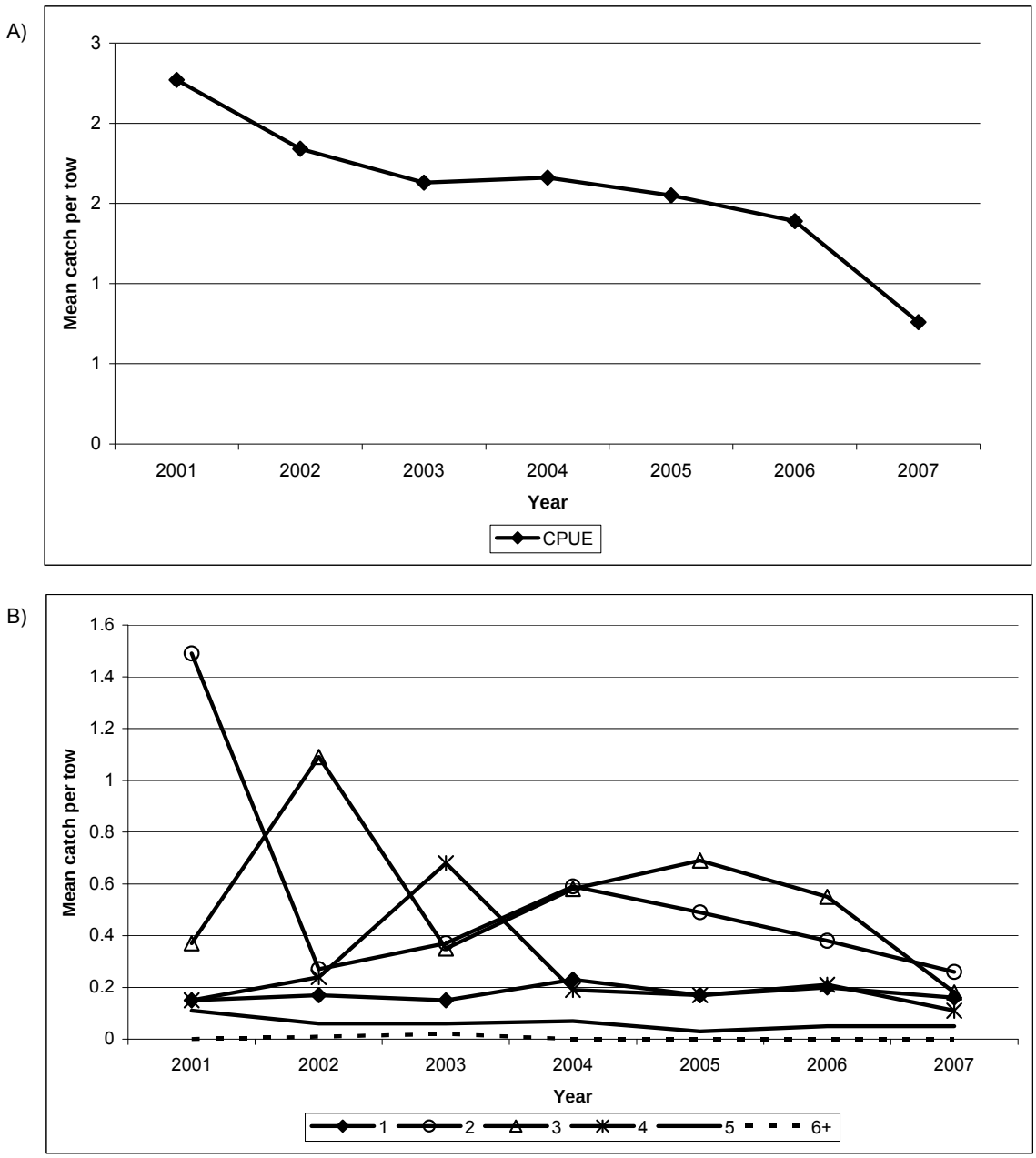


Figure 6.1-5. Results of the NCDMF gillnet survey in Pamlico Sound. A) Age aggregated CPUE; B) Age specific CPUE. Survey is not lagged, so all values are in the true survey year.

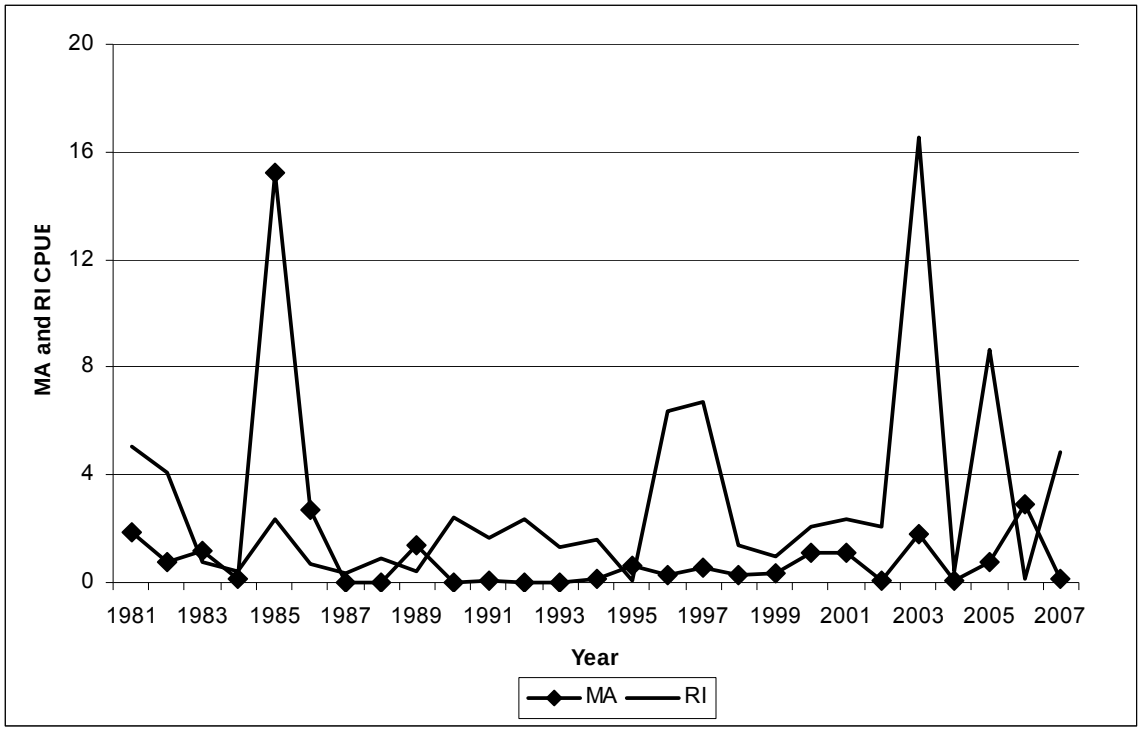


Figure C6.2-1. Massachusetts and Rhode Island YOY indices of abundance.

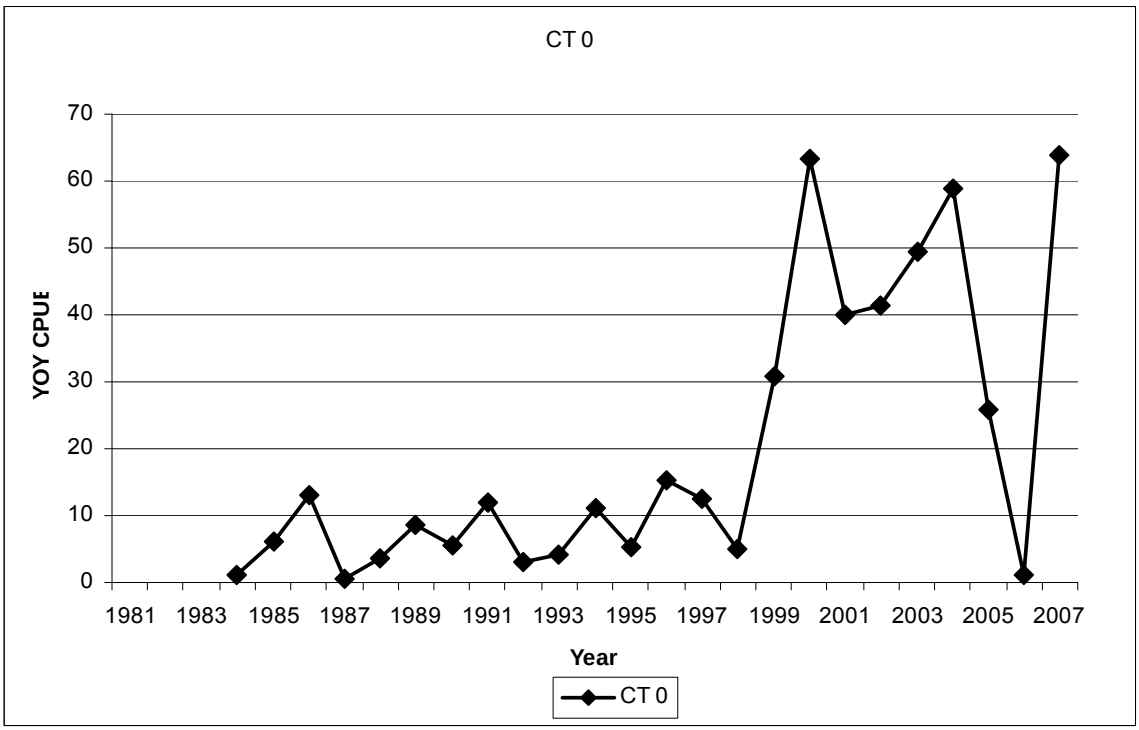


Figure C6.2-2. Connecticut YOY and 1+ indices of abundance.

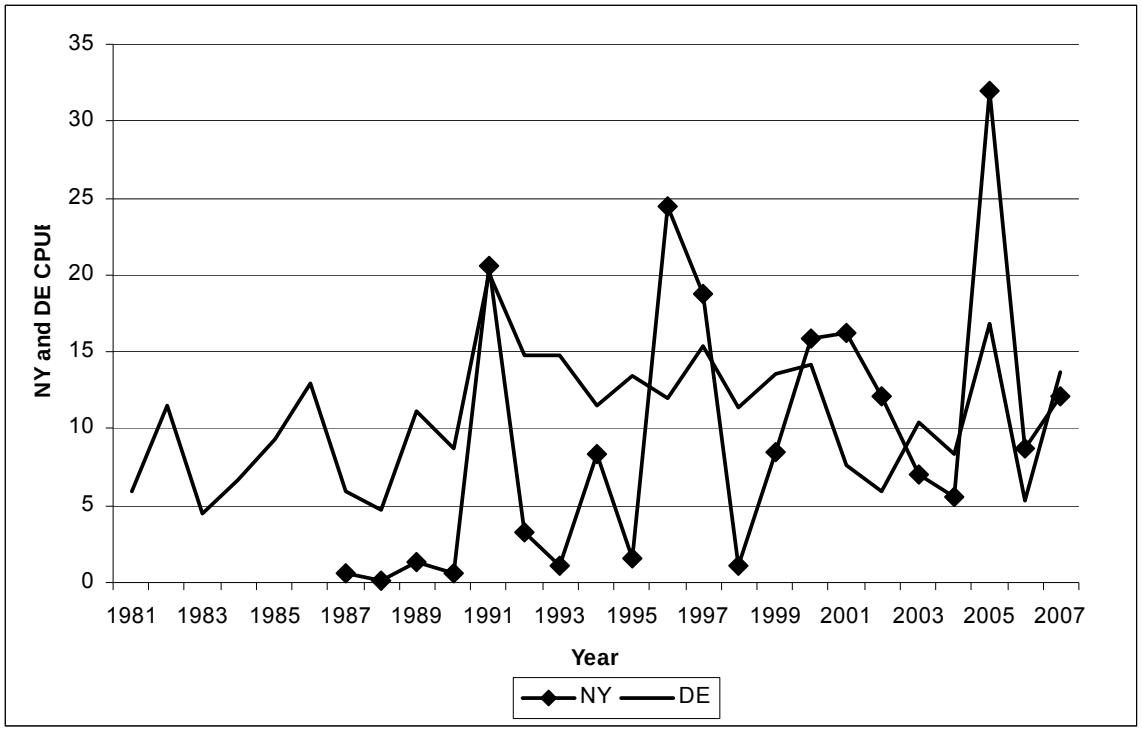


Figure C6.2-3. New York and Delaware YOY indices of abundance.

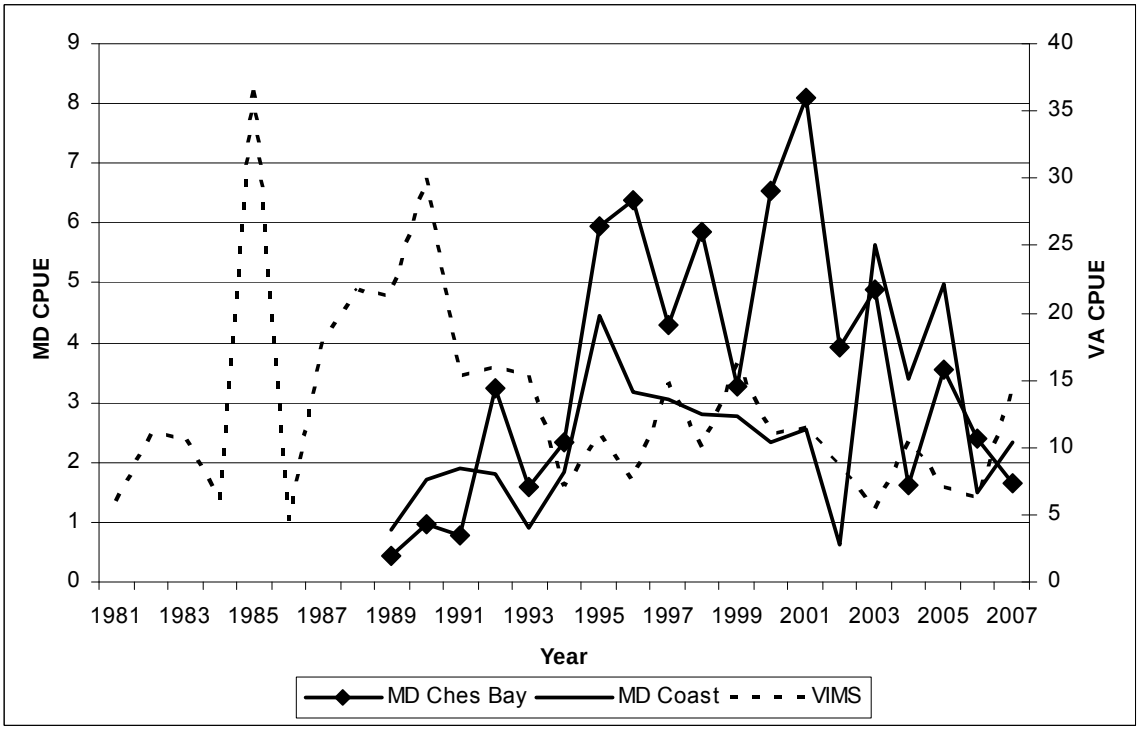


Figure C6.2-4. Maryland and Virginia YOY indices of abundance.

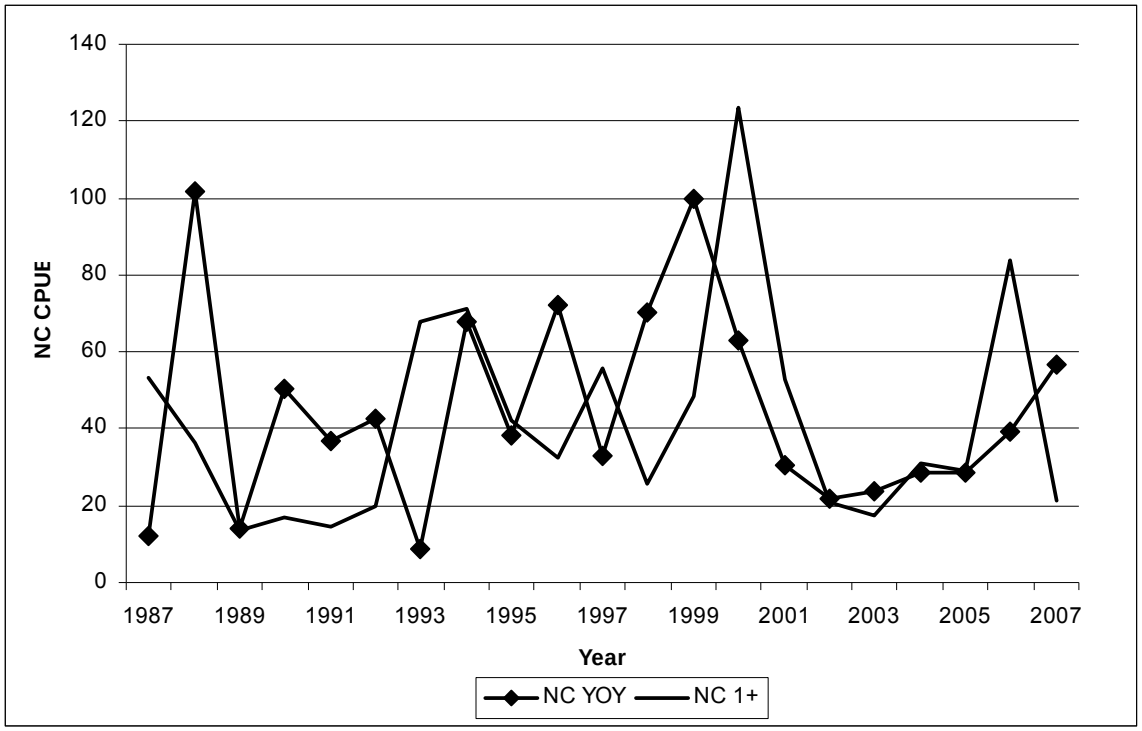


Figure C6.2-5. North Carolina YOY and 1+ indices of abundance.

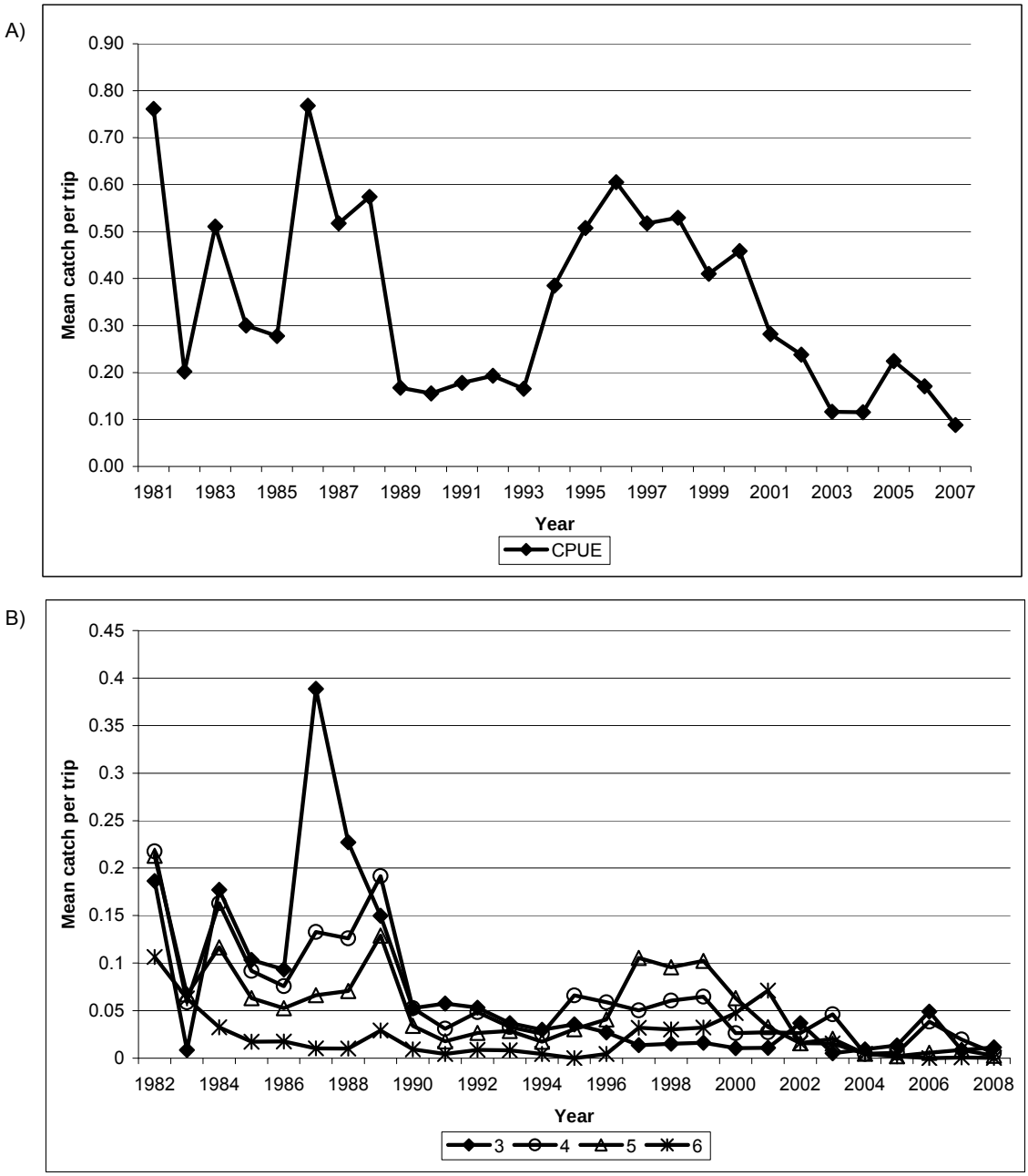


Figure C6.4-1. Fishery dependent (MRFSS) indices of abundance. A) Age aggregated (1+) catch per trip in true fishing year; B) Age specific catch per trip progressed forward one year and age.

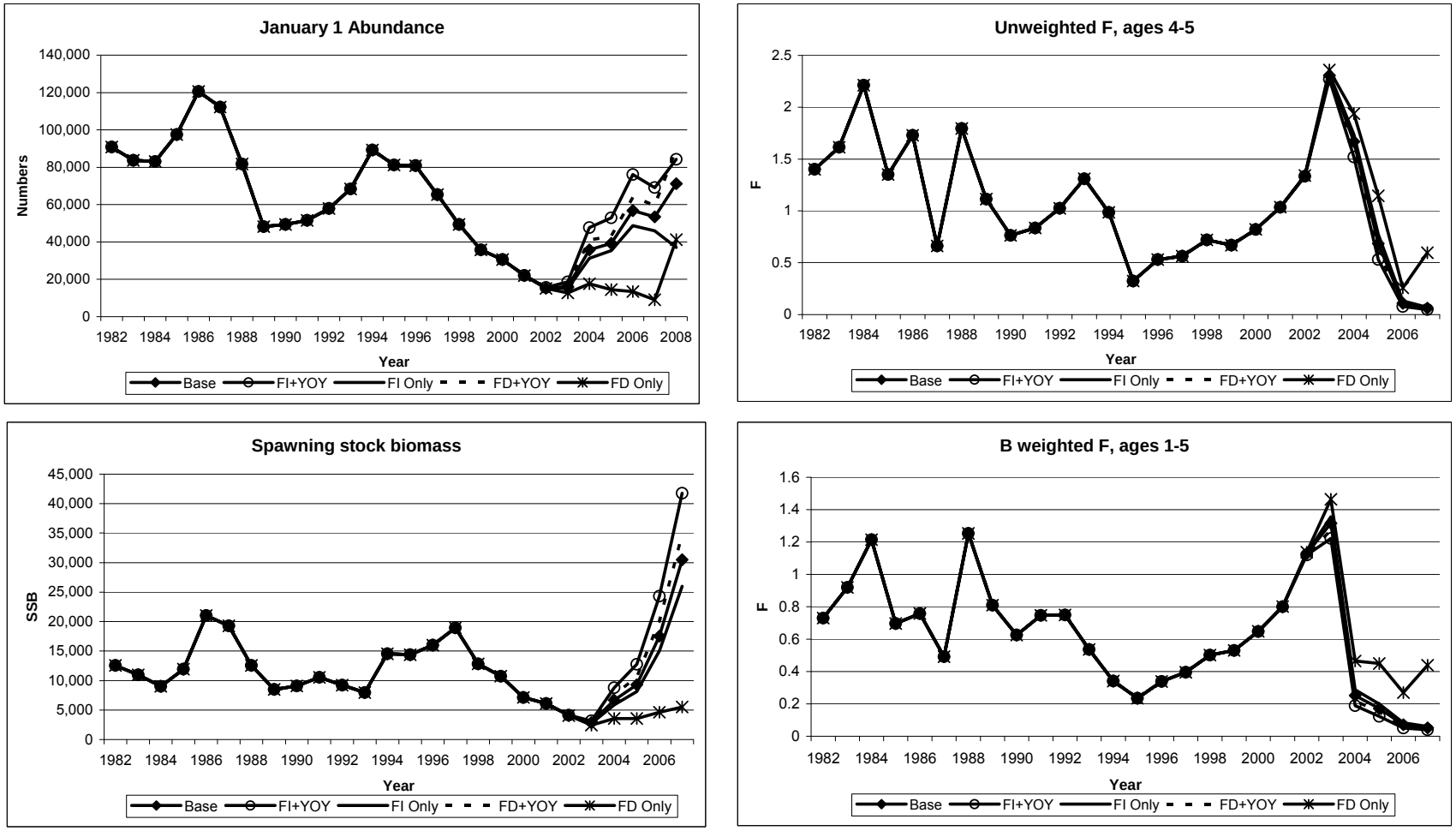


Figure C7.2-1. Summary results of preliminary ADAPT VPA runs. Tuning indices used are as follows: Base = All available aged and young of year indices; FI+YOY = All aged fishery independent indices plus all YOY indices; FI only = All aged fishery independent indices only; FD+YOY = All fishery dependent aged indices plus all YOY indices; FD only = Fishery dependent indices only.

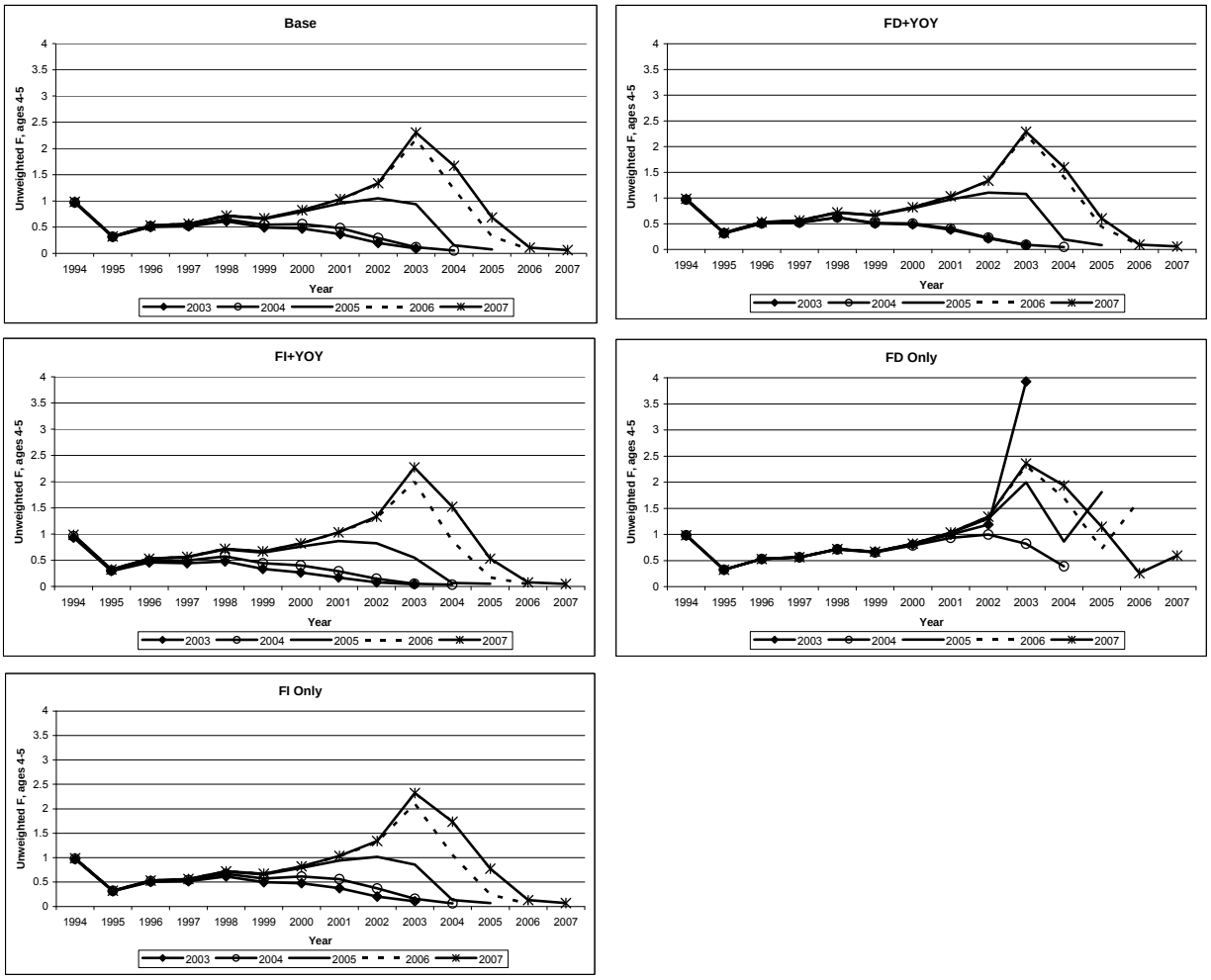


Figure C7.2-2. Retrospective patterns in fishing mortality (unweighted average age 4-5) from preliminary ADAPT VPA runs. Tuning indices used are as follows: Base = All available aged and young of year indices; FI+YOY = All aged fishery independent indices plus all YOY indices; FI only = All aged fishery independent indices only; FD+YOY = All fishery dependent aged indices plus all YOY indices; FD only = Fishery dependent indices only.

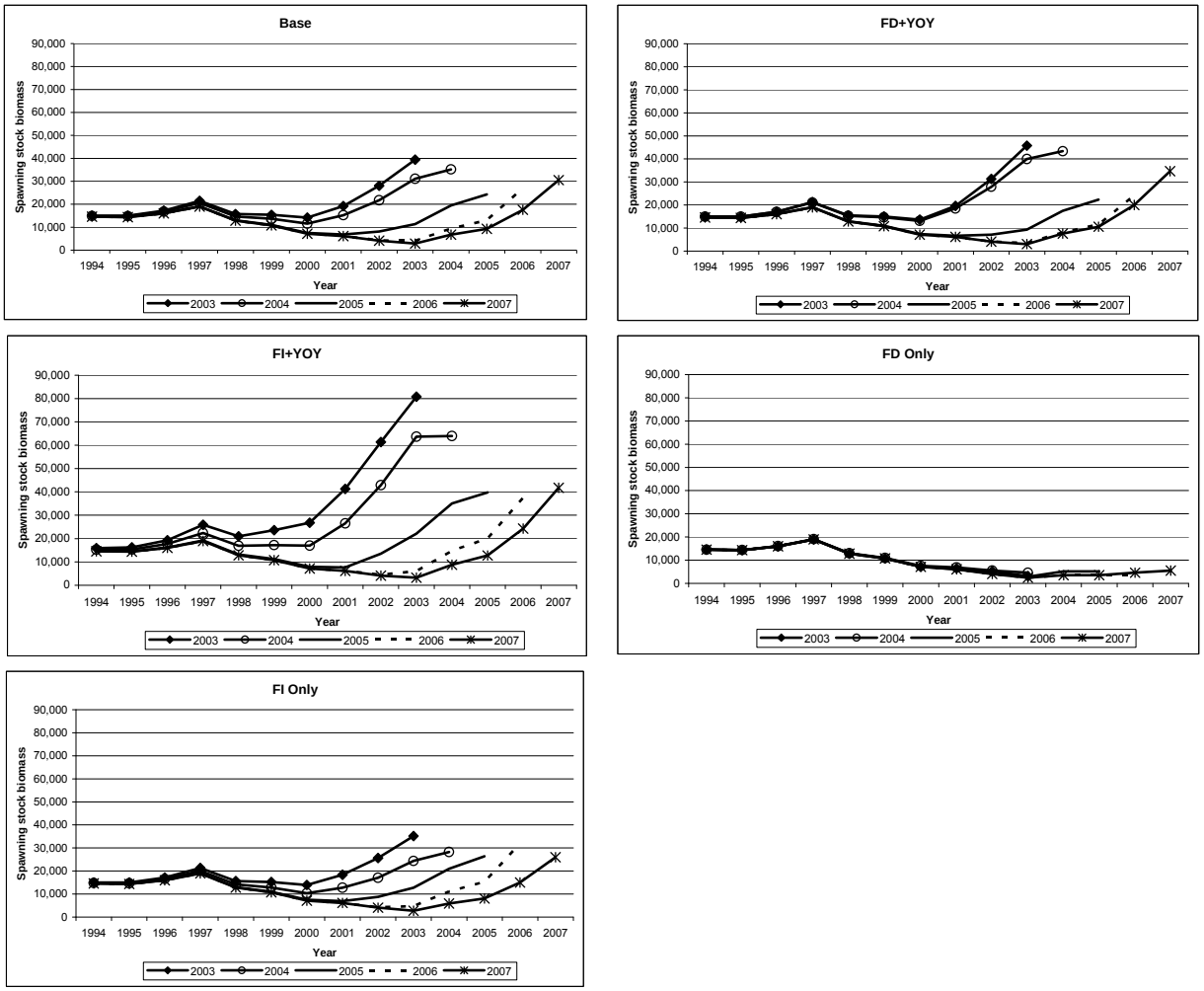


Figure C7.2-3. Retrospective patterns in spawning stock biomass from preliminary ADAPT VPA runs. Tuning indices used are as follows: Base = All available aged and young of year indices; FI+YOY = All aged fishery independent indices plus all YOY indices; FI only = All aged fishery independent indices only; FD+YOY = All fishery dependent aged indices plus all YOY indices; FD only = Fishery dependent indices only.

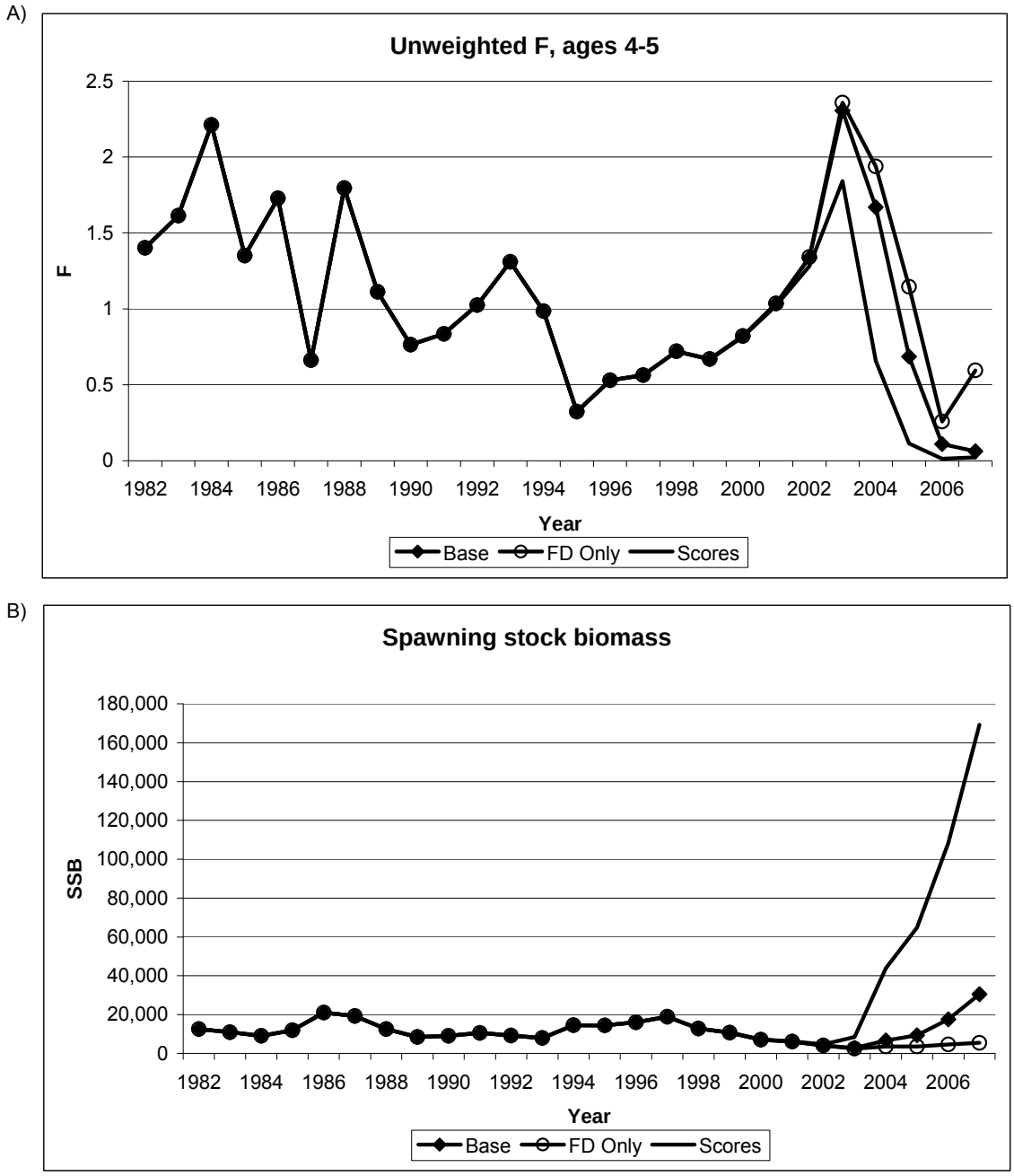


Figure C7.2-4. Summary results of ADAPT VPA run using indices selected from scored criteria (Scores) compared to Base and Fishery Dependent only runs. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass. Indices used in the Scores run include NEFSC ages 1-2, DE 30 foot trawl age 1, NC gillnet age 1, SEAMAP ages 1-6+, and RI, CT, NY, DE, MD, and NC YOY indices.

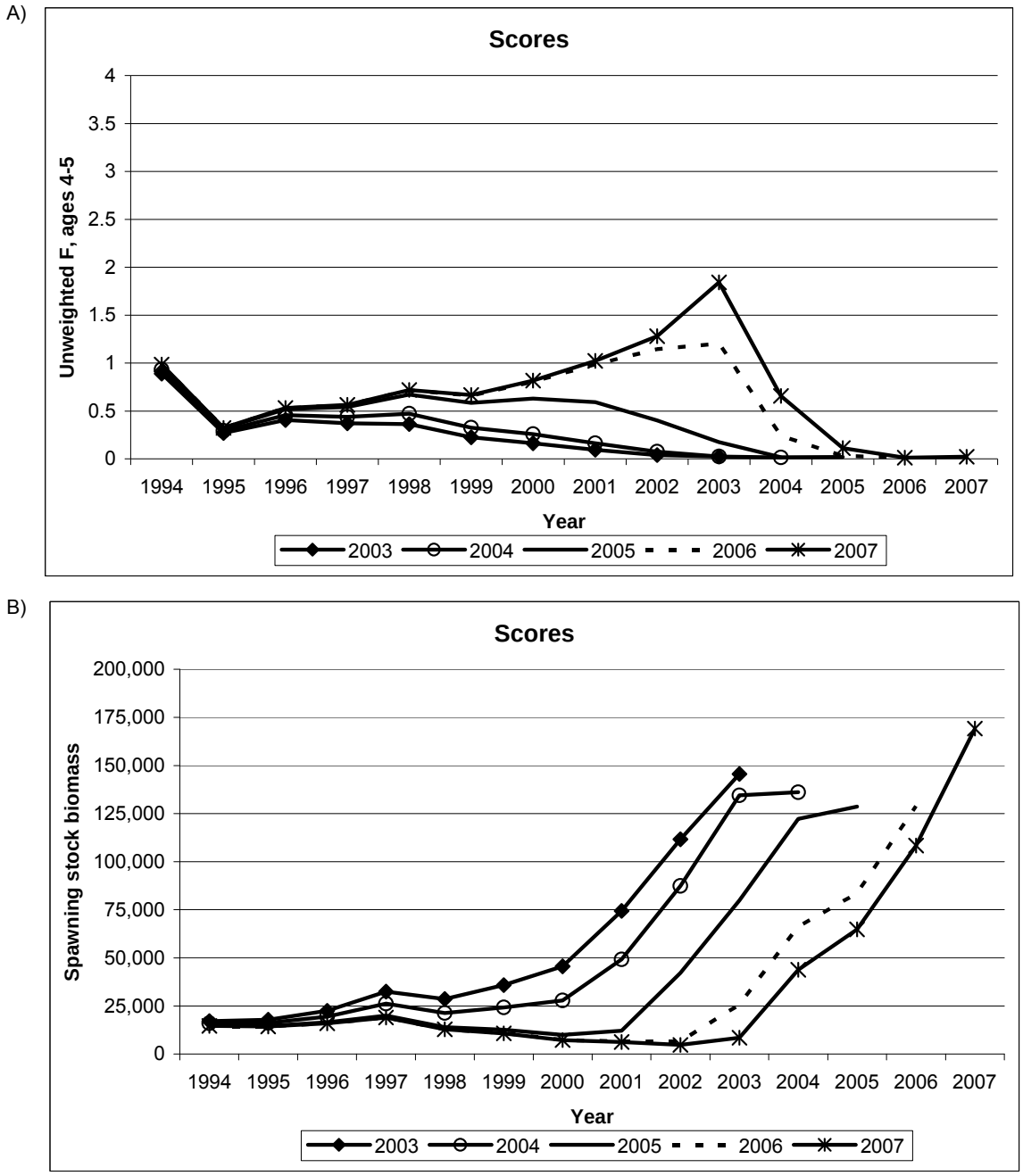


Figure C7.2-5. Retrospective results of ADAPT VPA run using indices selected from scored criteria. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass. Patterns are more severe (longer and greater magnitude) than those presented in Figures C7.2-2 and C7.2-3. Notice the difference in scale between 7.2-5B and 7.2-3.

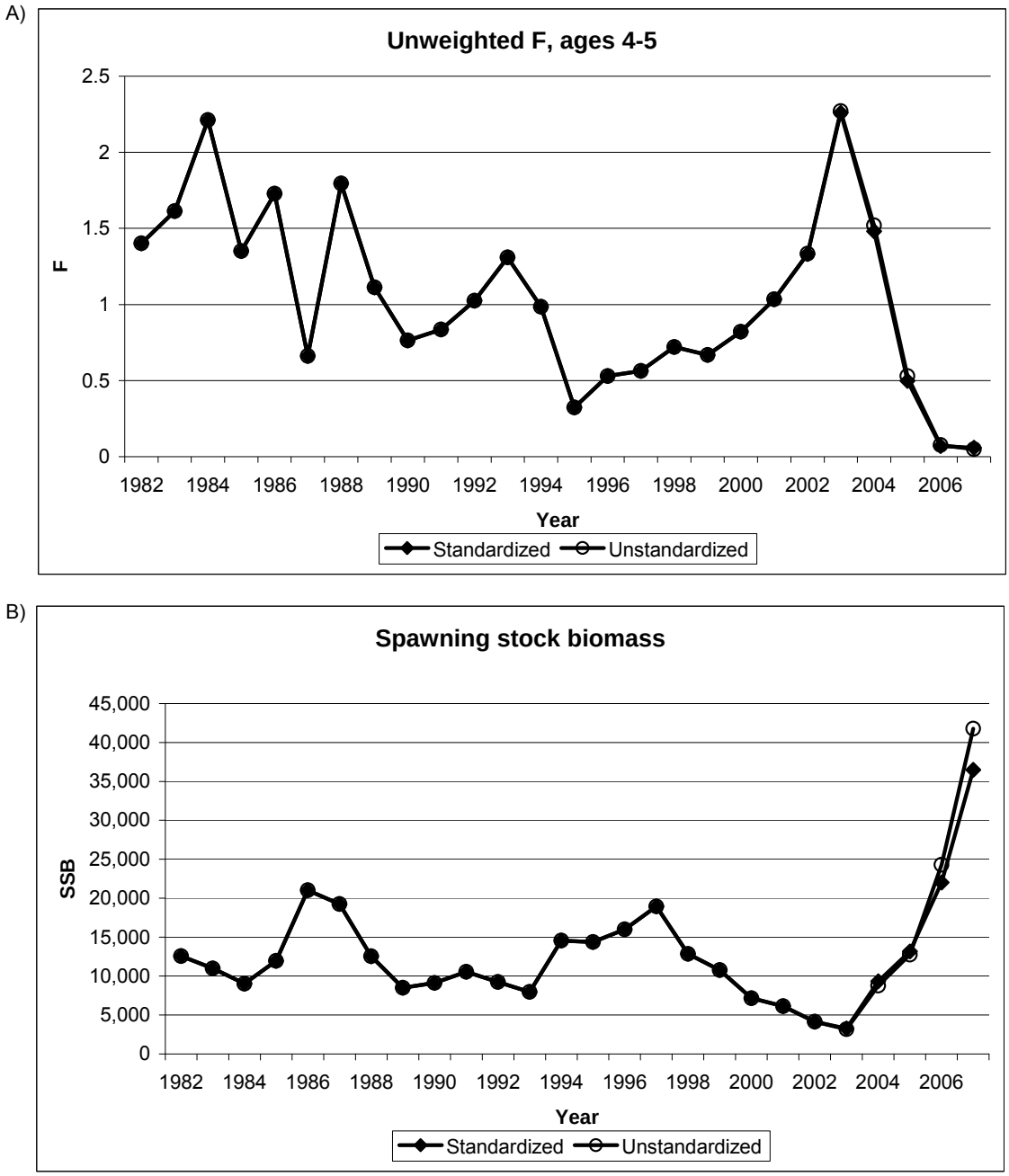


Figure C7.2-6. Comparison of ADAPT VPA results using standardized and non-standardized tuning indices for all age fishery independent surveys and all YOY surveys. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass.

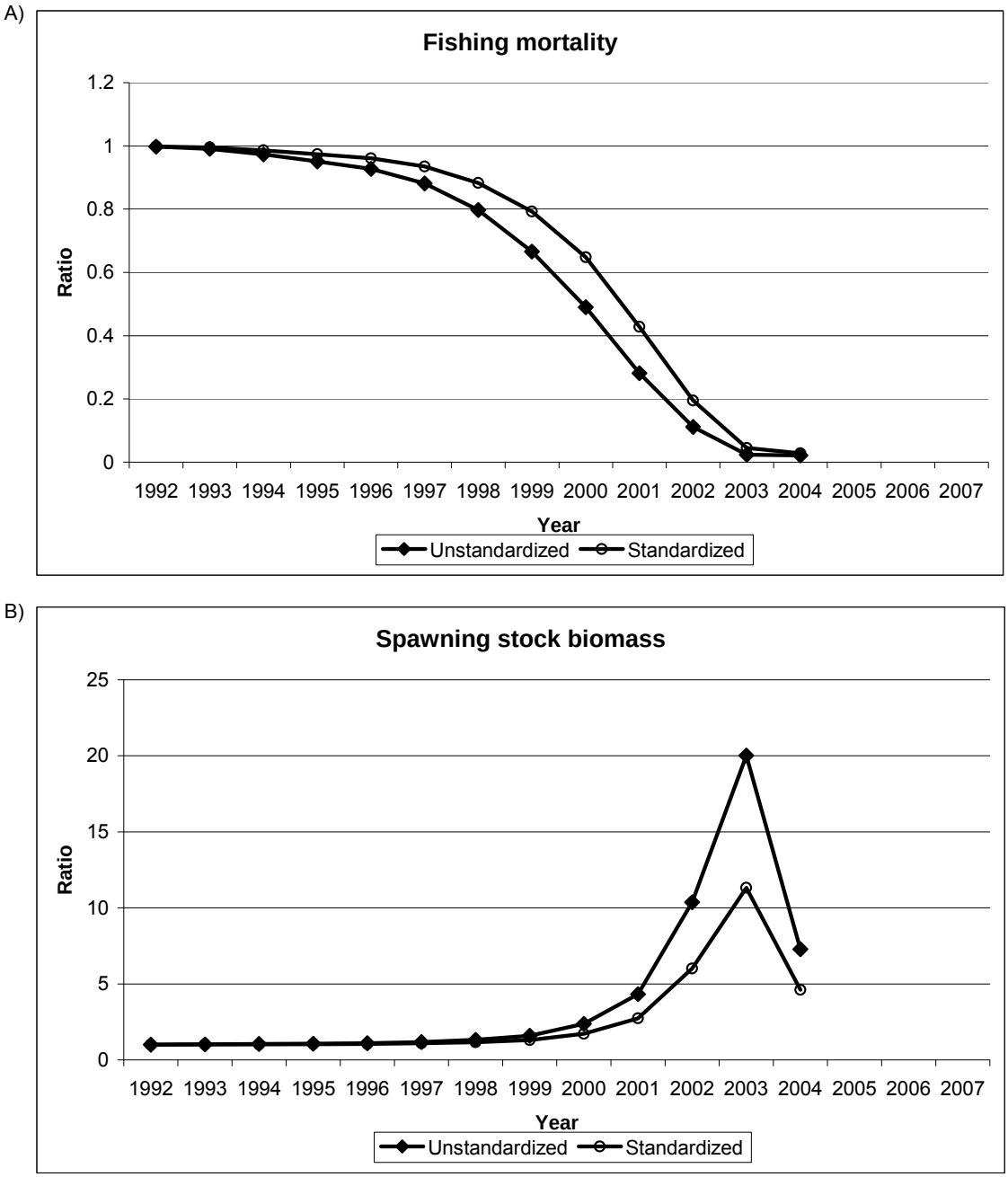


Figure C7.2-7. Comparison of typical retrospective pattern, expressed as the ratio of parameter estimates in terminal year 2004 relative to terminal year 2007, using standardized and non-standardized tuning indices for all aged fishery independent surveys and all YOY surveys. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass.

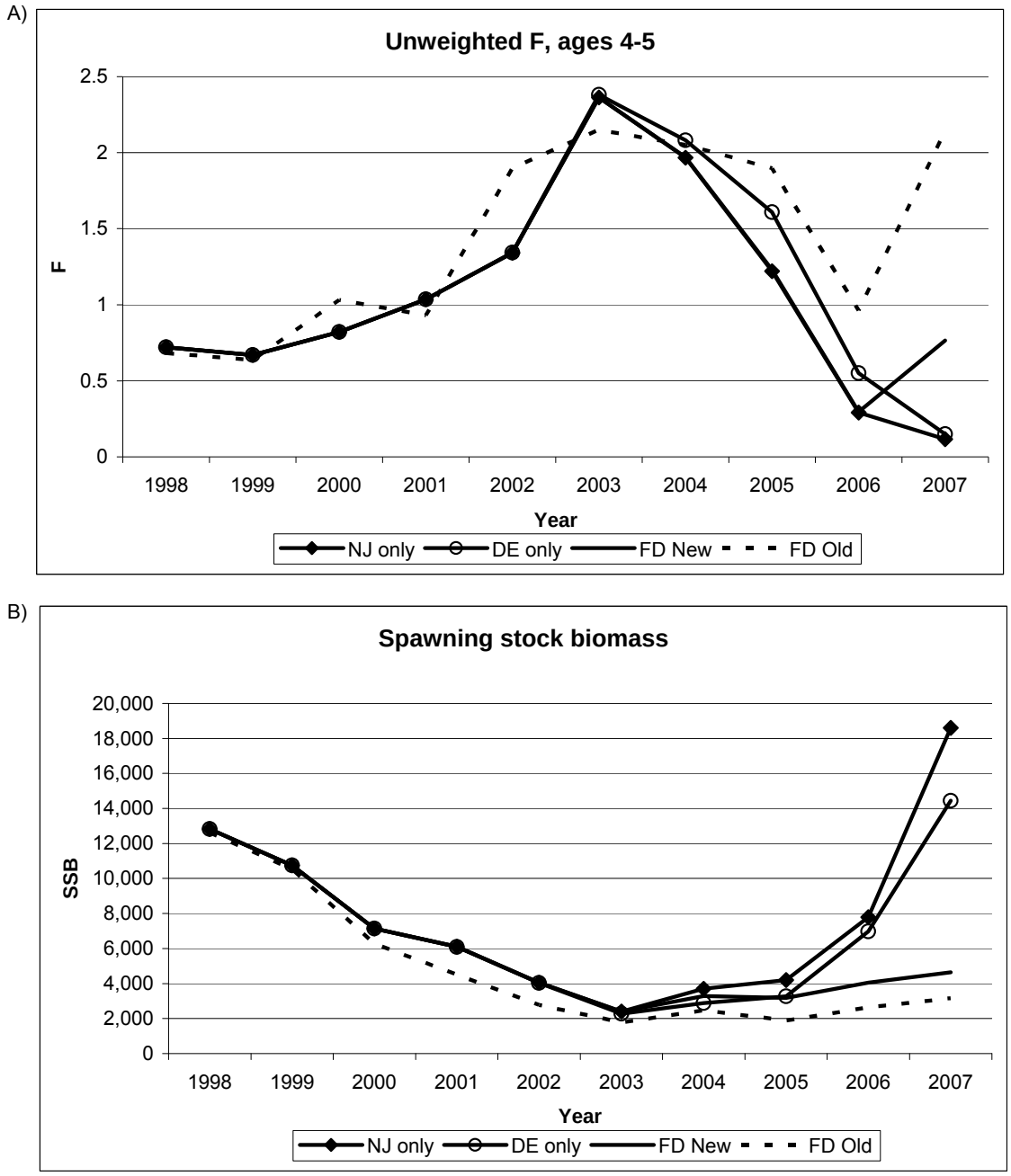


Figure C7.2-8. Comparison of ADAPT VPA results for the last 10 years using tuning indices from individual surveys. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass.

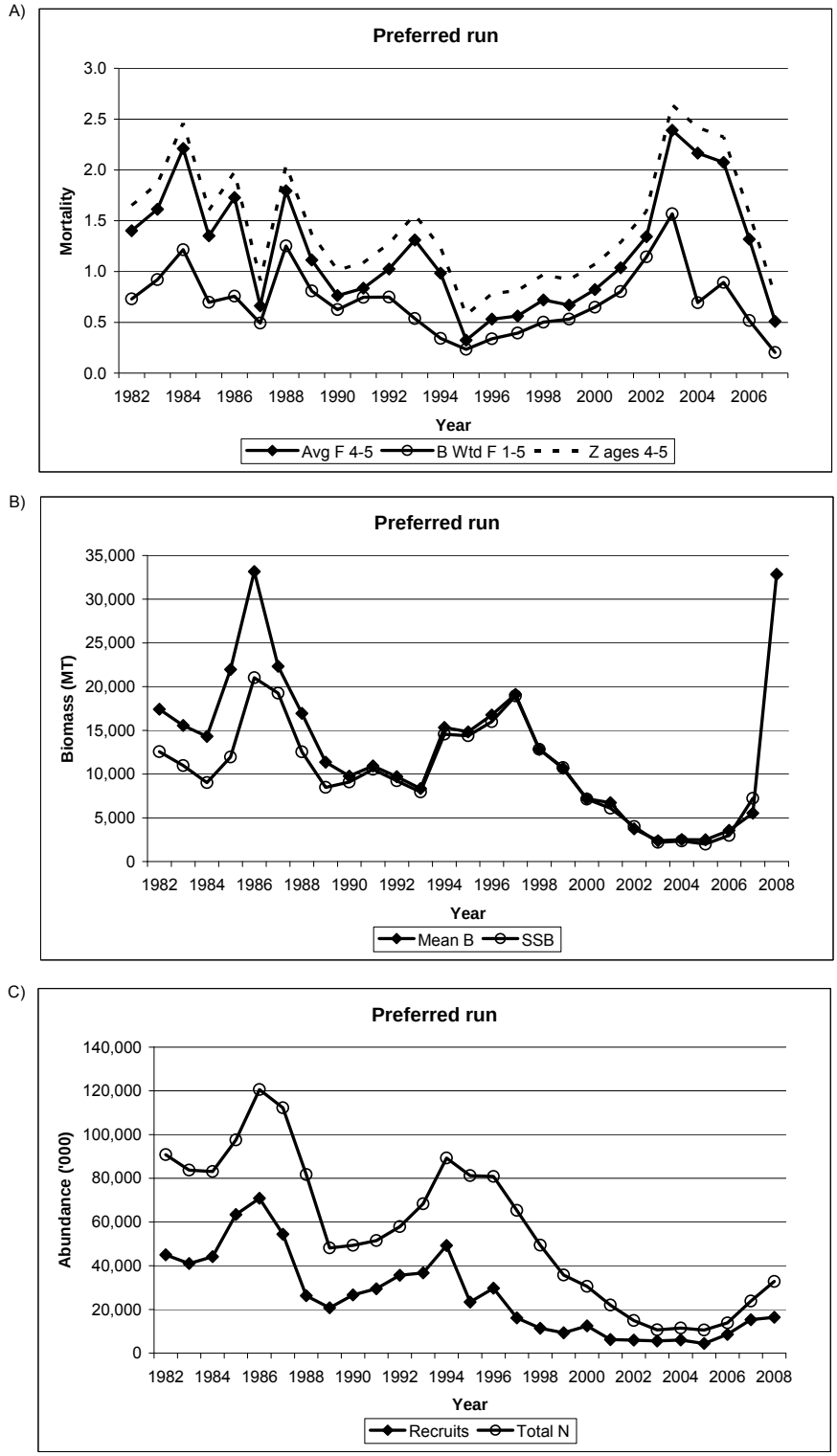


Figure C7.3-1. Summarized parameter estimates from the preferred ADAPT VPA run. A) Fishing mortality; B) Biomass; C) Abundance.

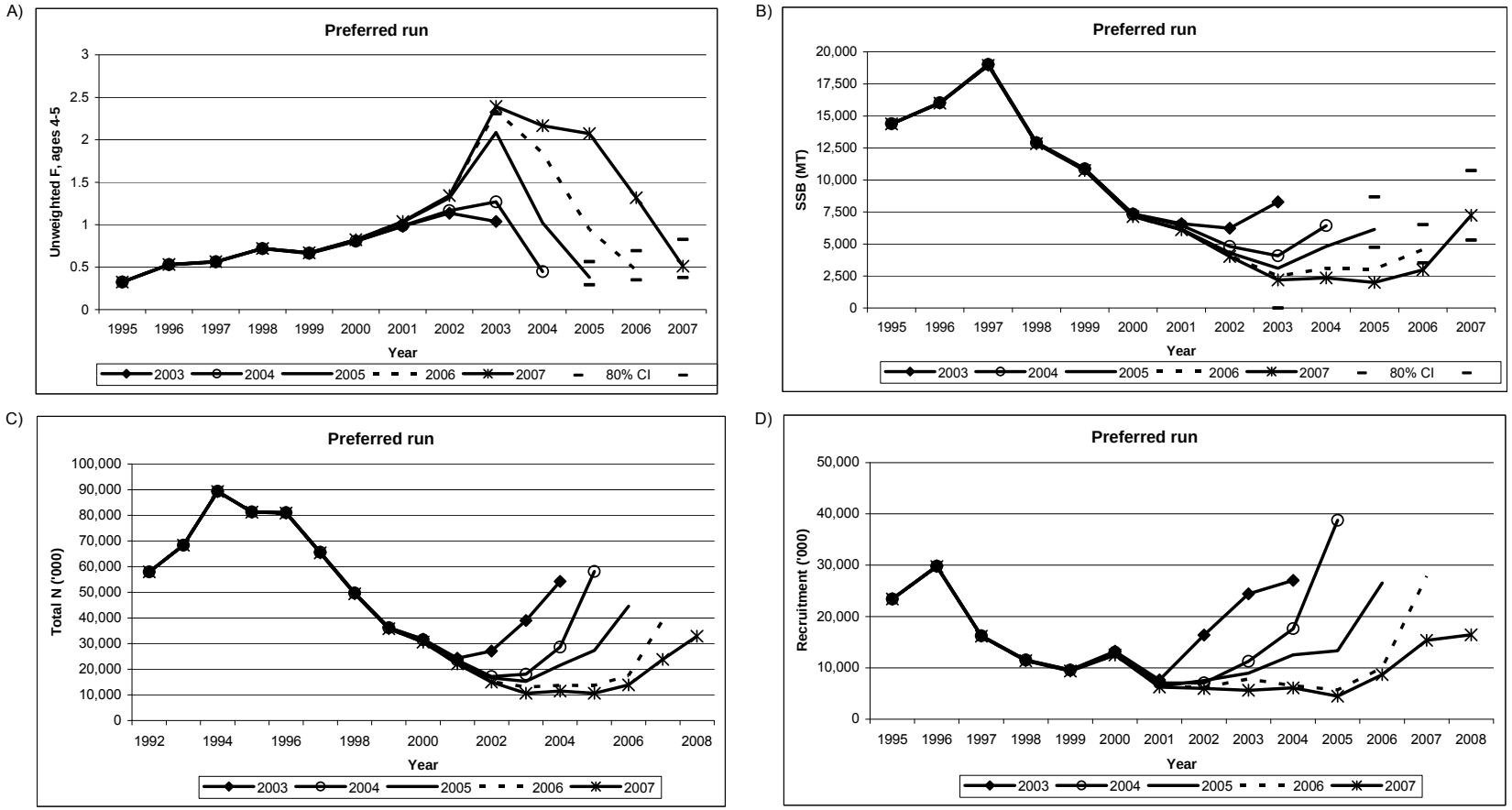


Figure C7.3-2. Retrospective results for the preferred ADAPT VPA run. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass; C) Total January 1 stock abundance; D) Recruitment to age 1.

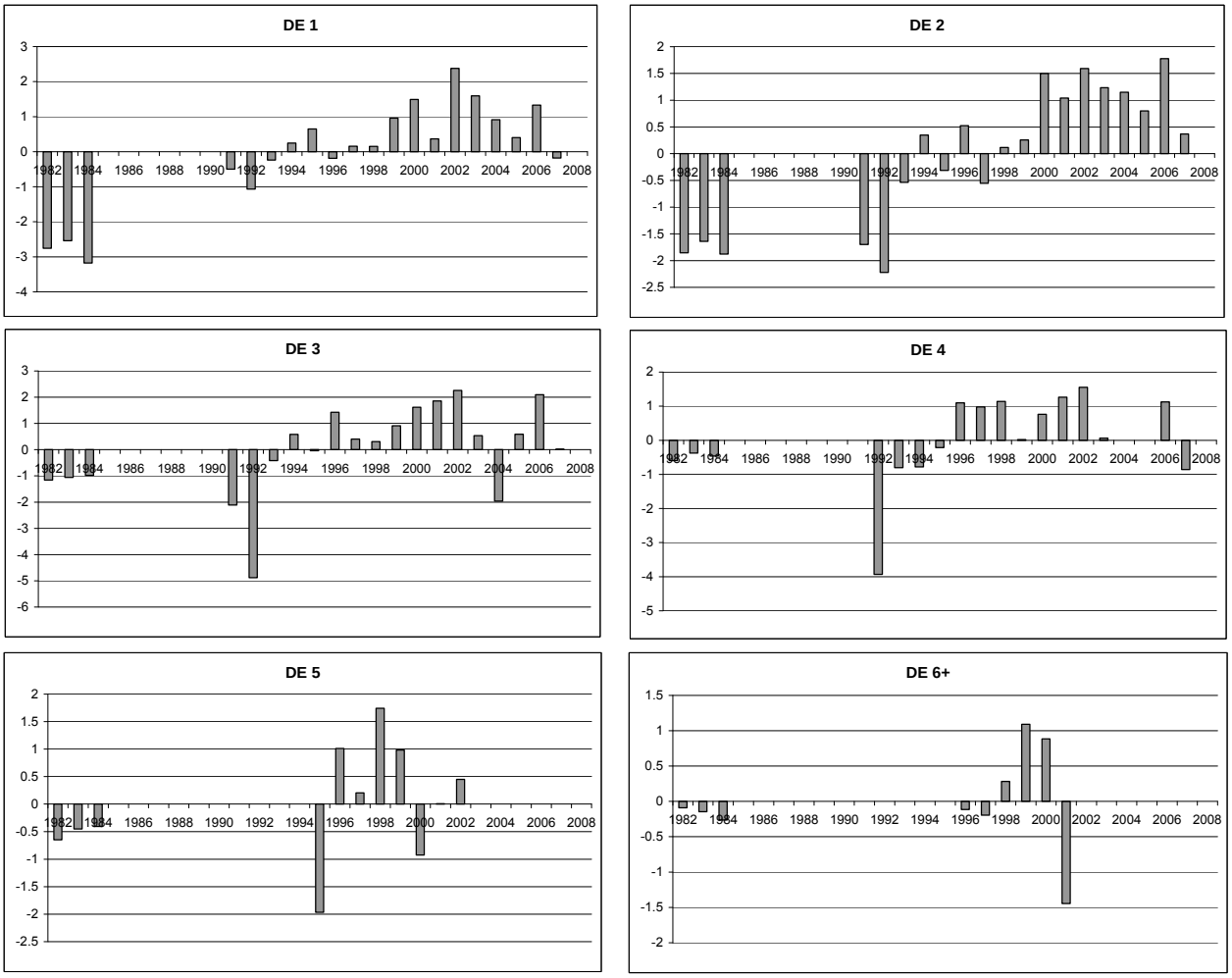


Figure C7.3-3. Survey residuals from the preferred ADAPT VPA run using Delaware and New Jersey trawl survey indices, North Carolina gillnet survey indices, and fishery dependent indices.

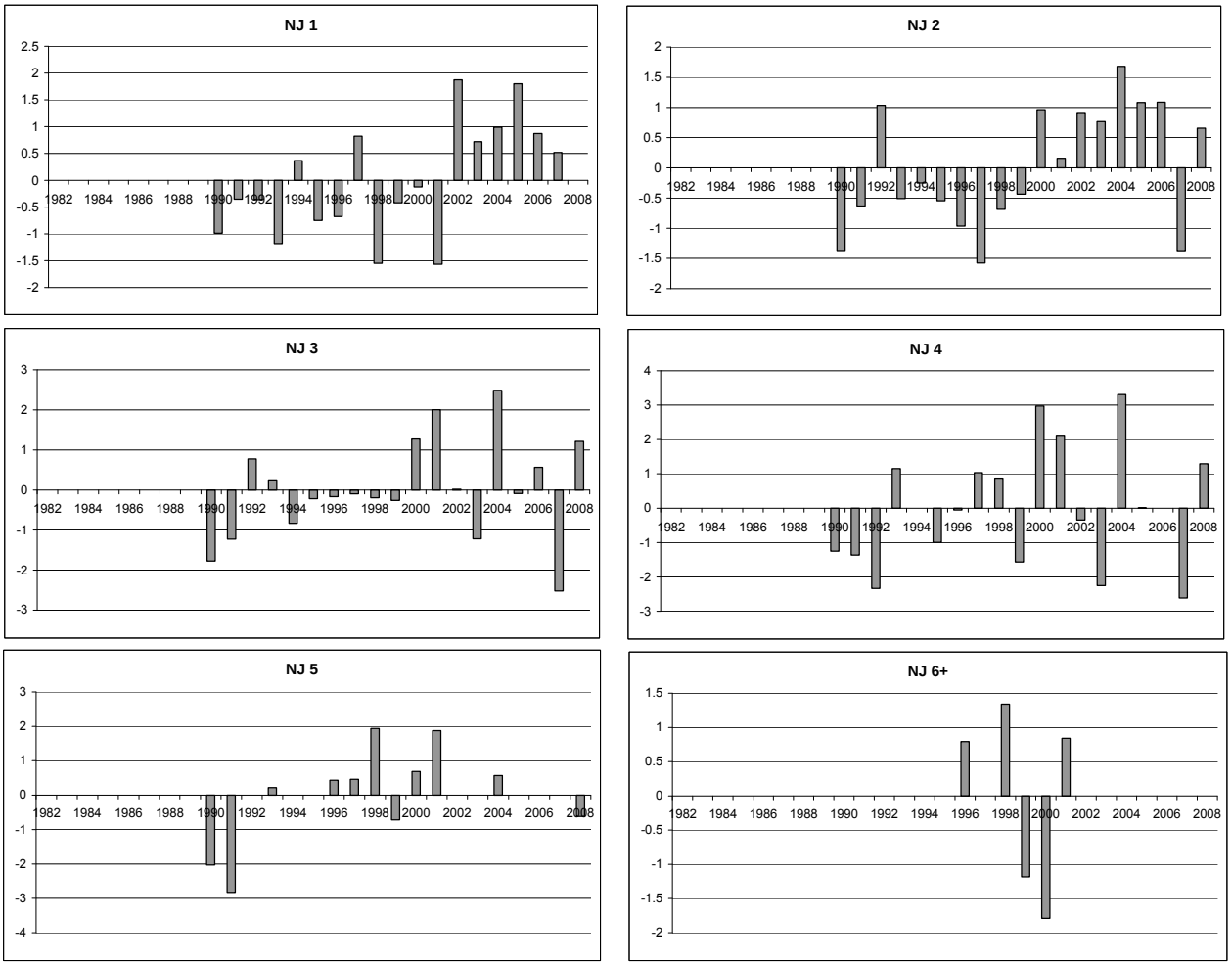


Figure C7.3-3 (continued). Survey residuals from the preferred ADAPT VPA run using Delaware and New Jersey trawl survey indices, North Carolina gillnet survey indices, and fishery dependent indices.

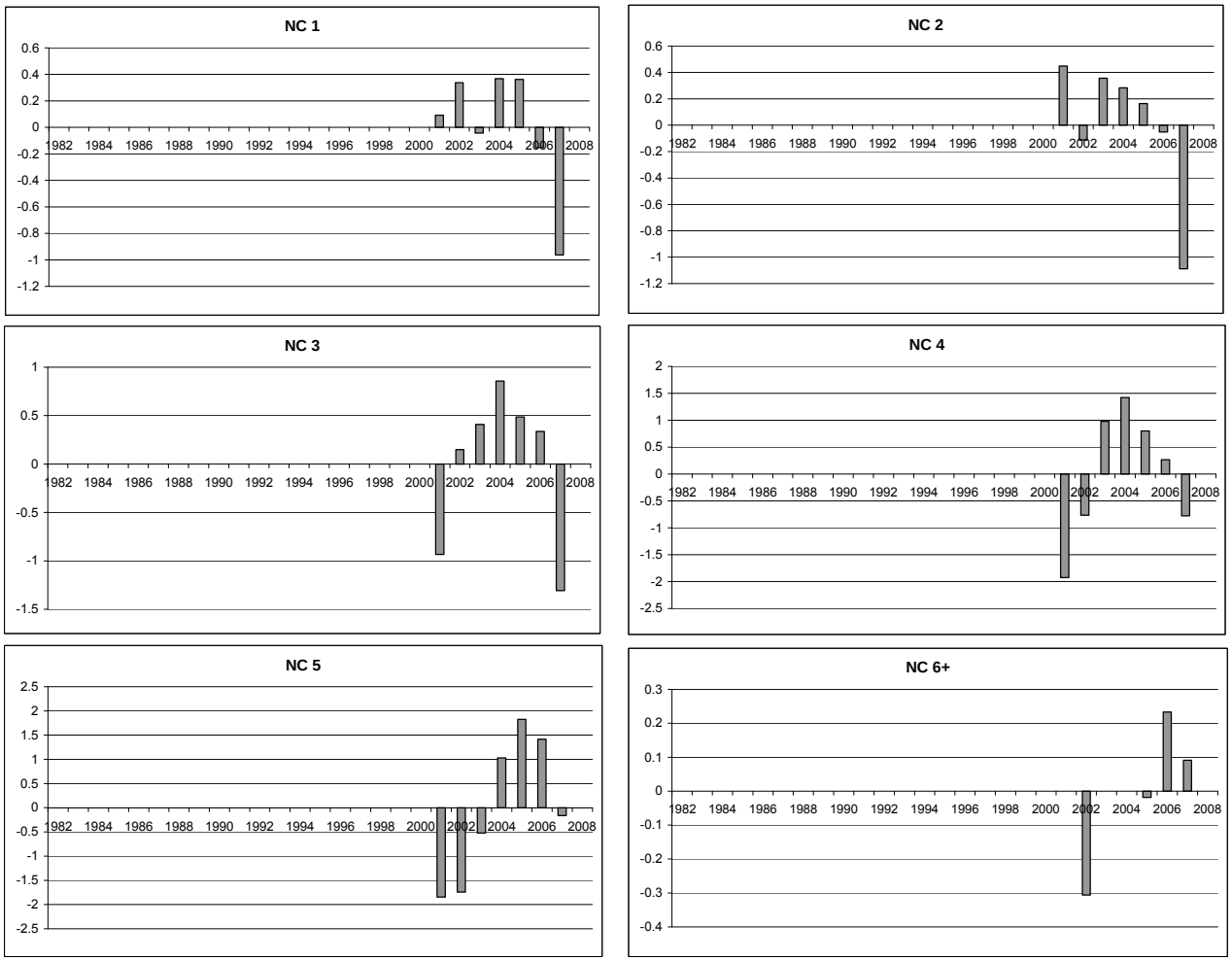


Figure C7.3-3 (continued). Survey residuals from the preferred ADAPT VPA run using Delaware and New Jersey trawl survey indices, North Carolina gillnet survey indices, and fishery dependent indices.

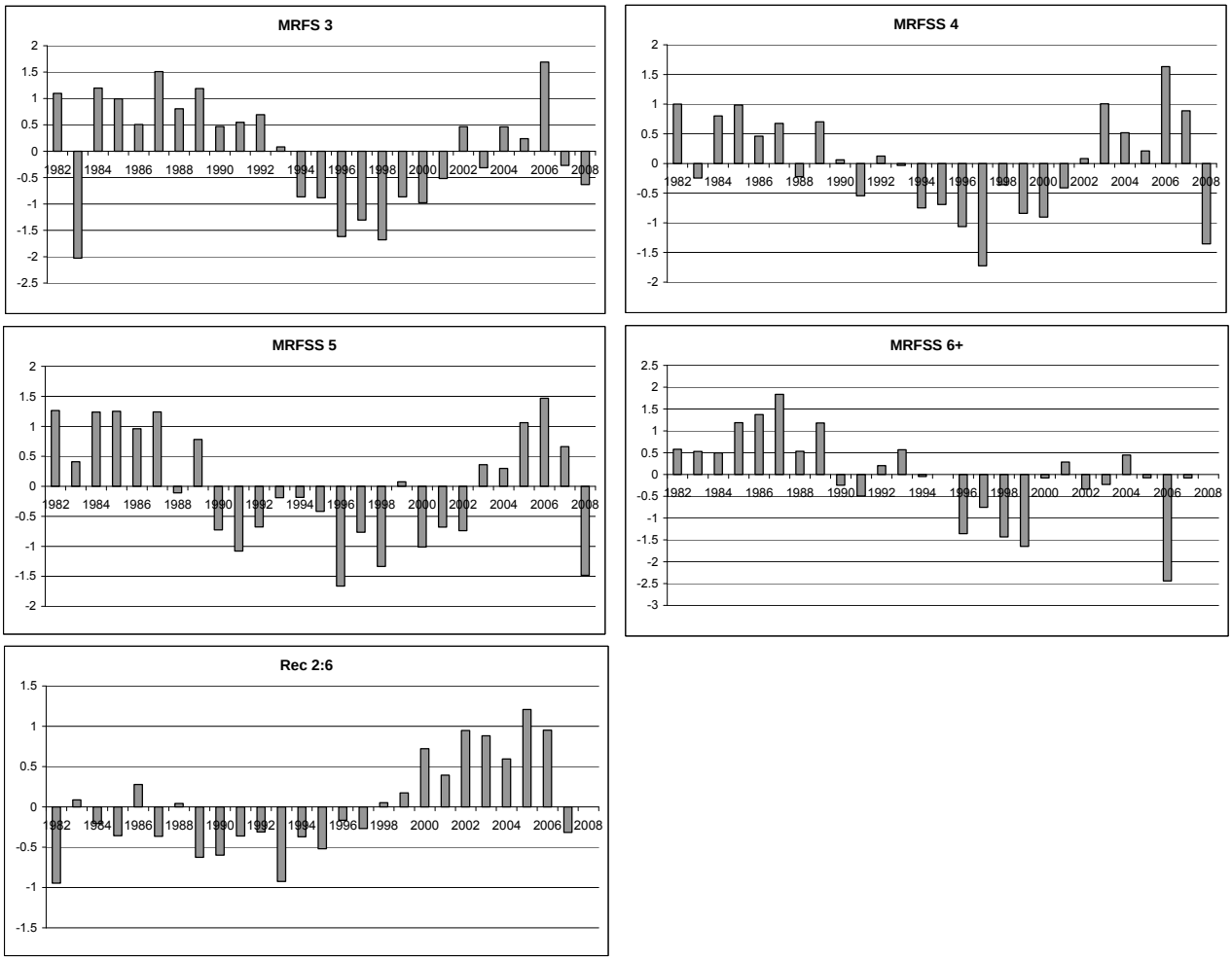


Figure C7.3-3 (continued). Survey residuals from the preferred ADAPT VPA run using Delaware and New Jersey trawl survey indices, North Carolina gillnet survey indices, and fishery dependent indices.

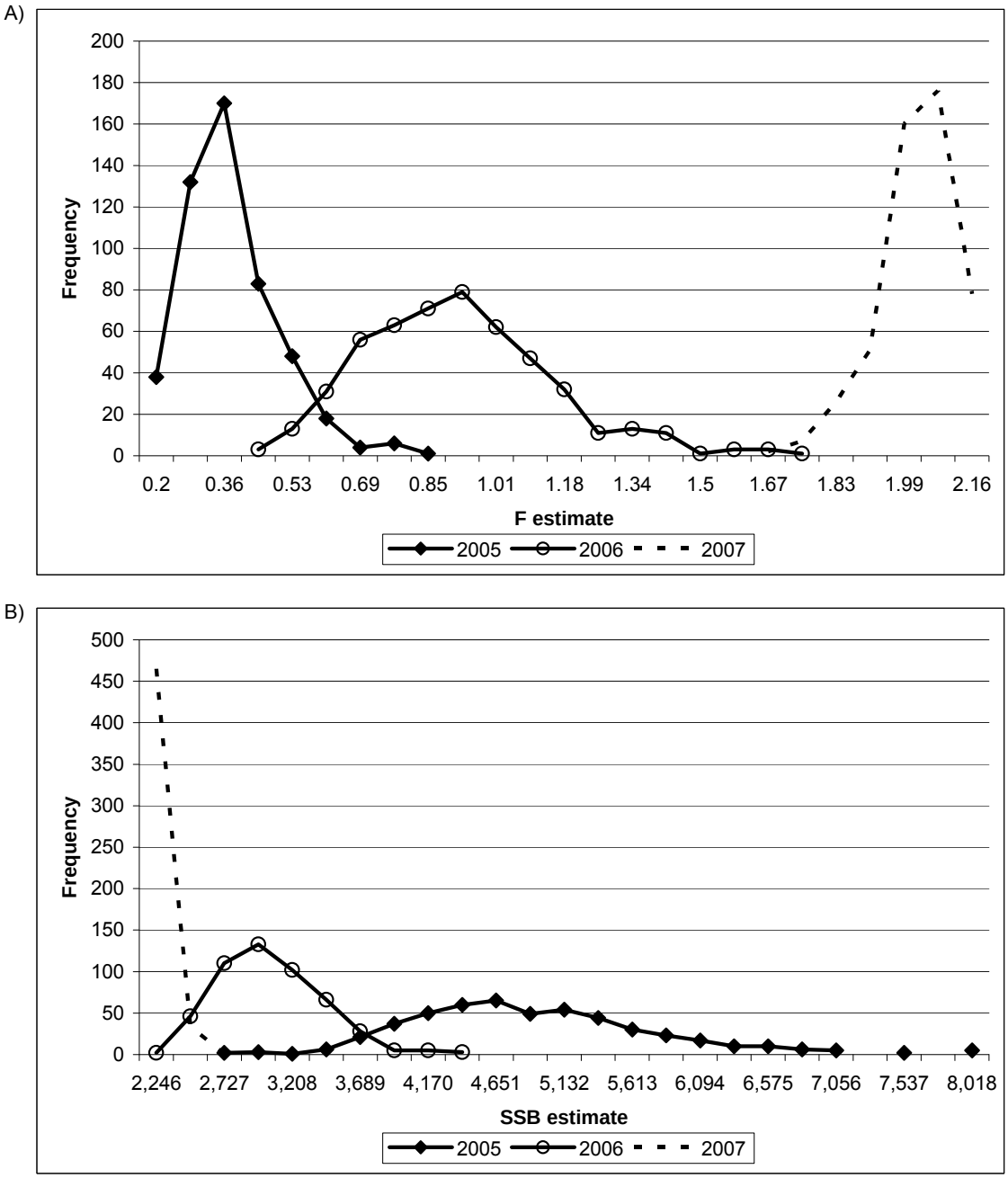


Figure C7.3-4. Frequency distributions of parameter estimates for a range of terminal years based on 500 bootstrap iterations per year. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass.

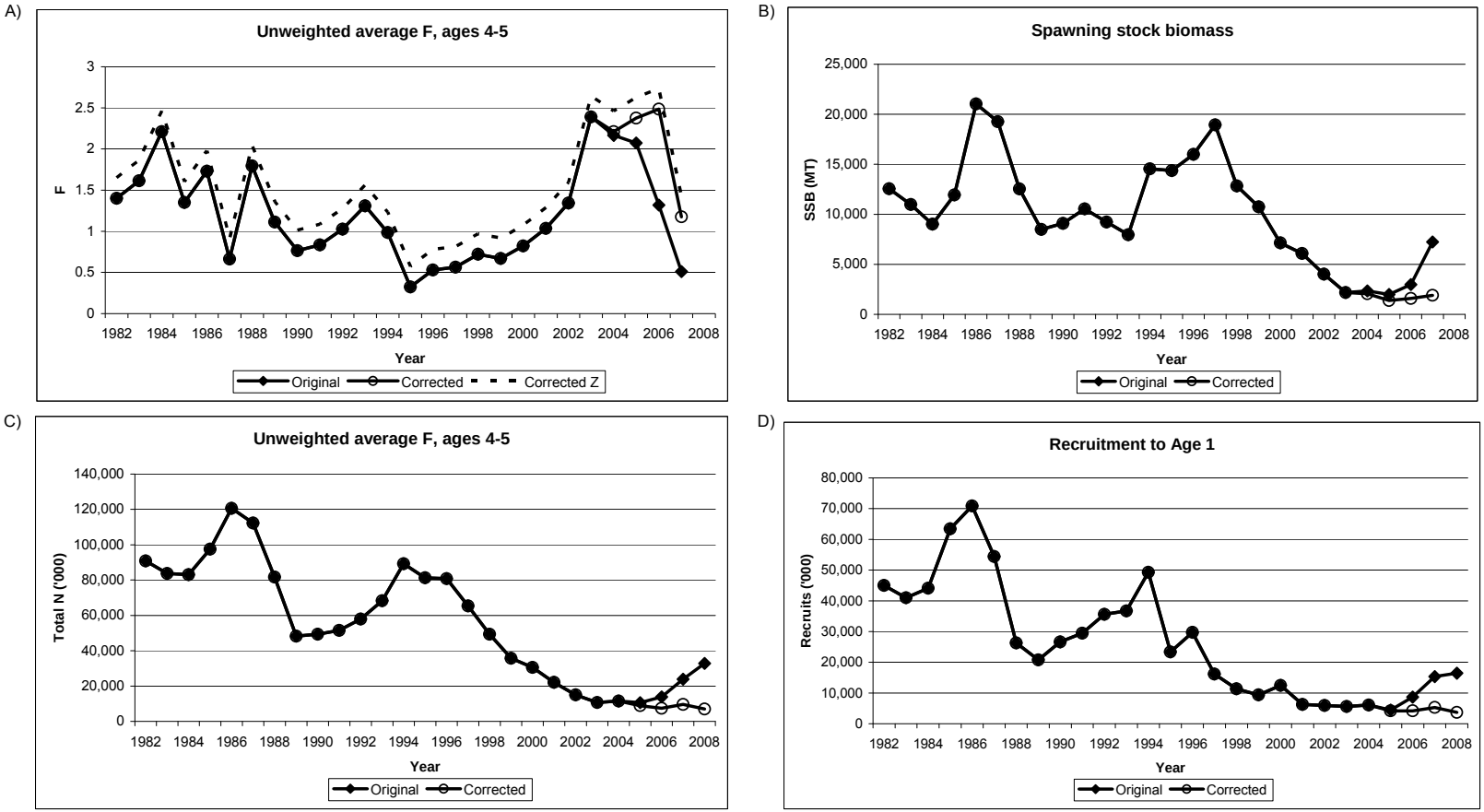


Figure C7.3-5. Retrospective corrected parameter estimates from ADAPT VPA. A) Unweighted average fishing mortality, ages 4-5; B) Spawning stock biomass; C) Total January 1 stock abundance; D) Recruitment to age 1.

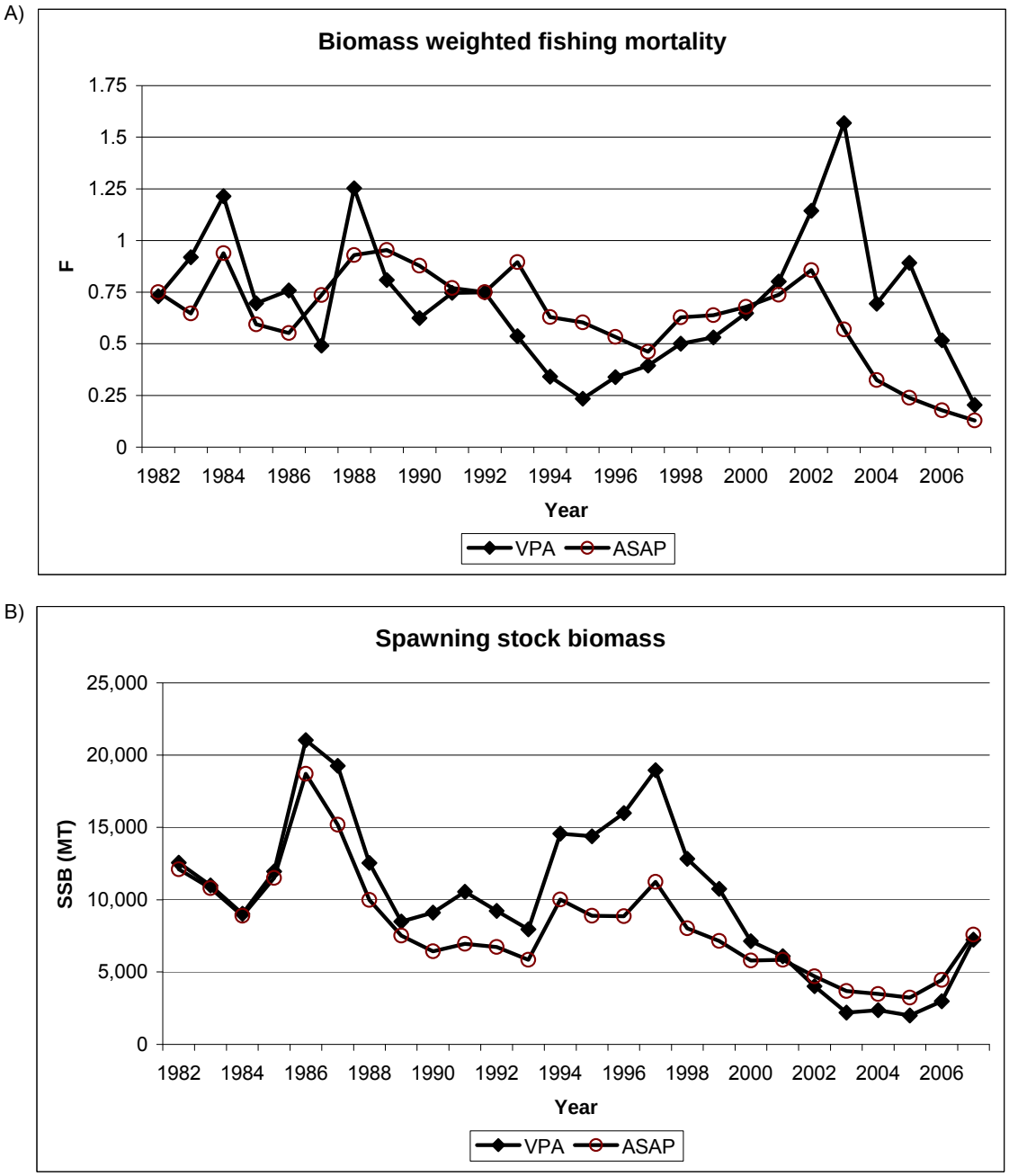


Figure C7.3-6. Comparison of results from the preferred ADAPT run and preliminary ASAP model runs. A) Biomass weighted fishing mortality; B) Spawning stock biomass.

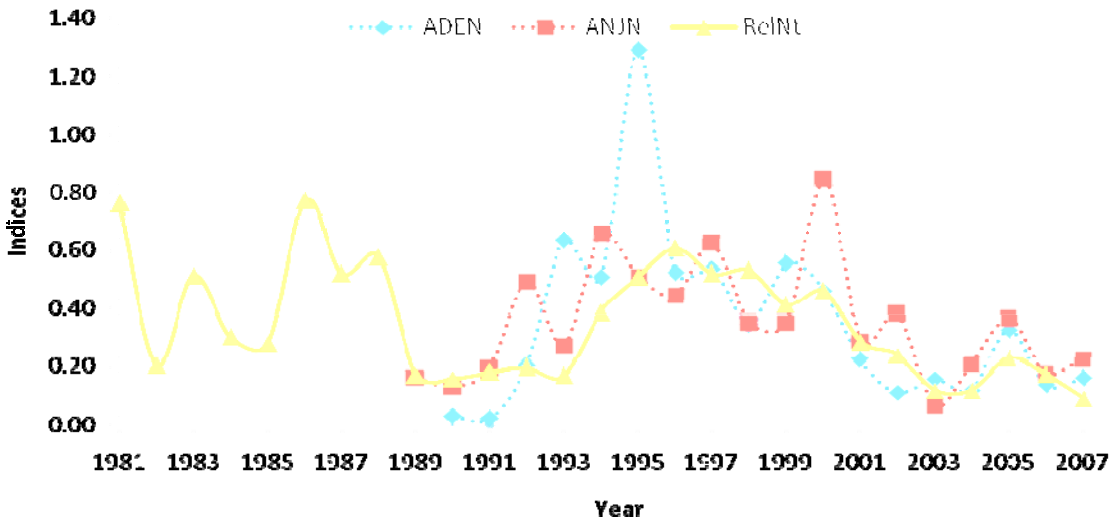


Figure C8.2-1. Plot of scaled New Jersey (ANJN), Delaware (ADEN) and recreational CPUE (RelNt) weakfish indices in number, 1981-2007.

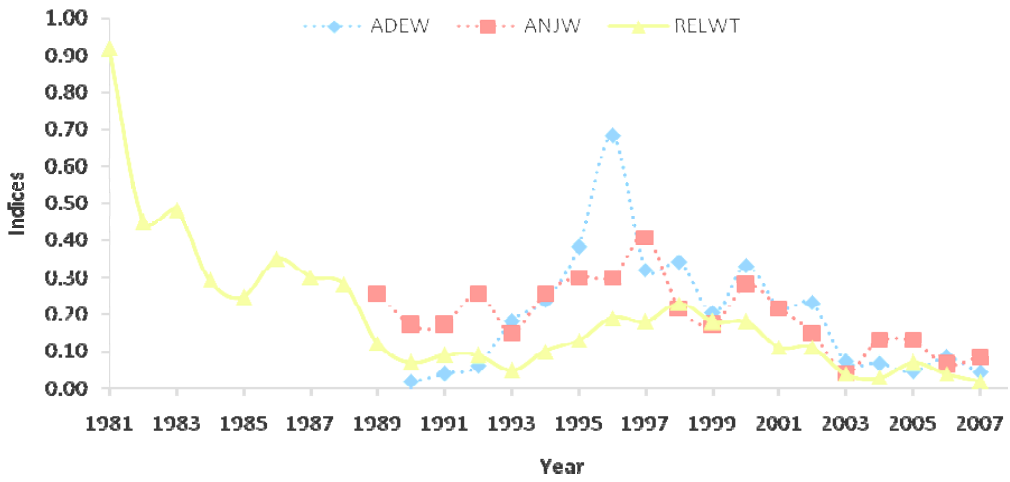


Figure C8.2-2. Plot of scaled New Jersey (ANJW), Delaware (ADEW) and recreational CPUE (RelWt) weakfish indices in weight, 1981-2007.

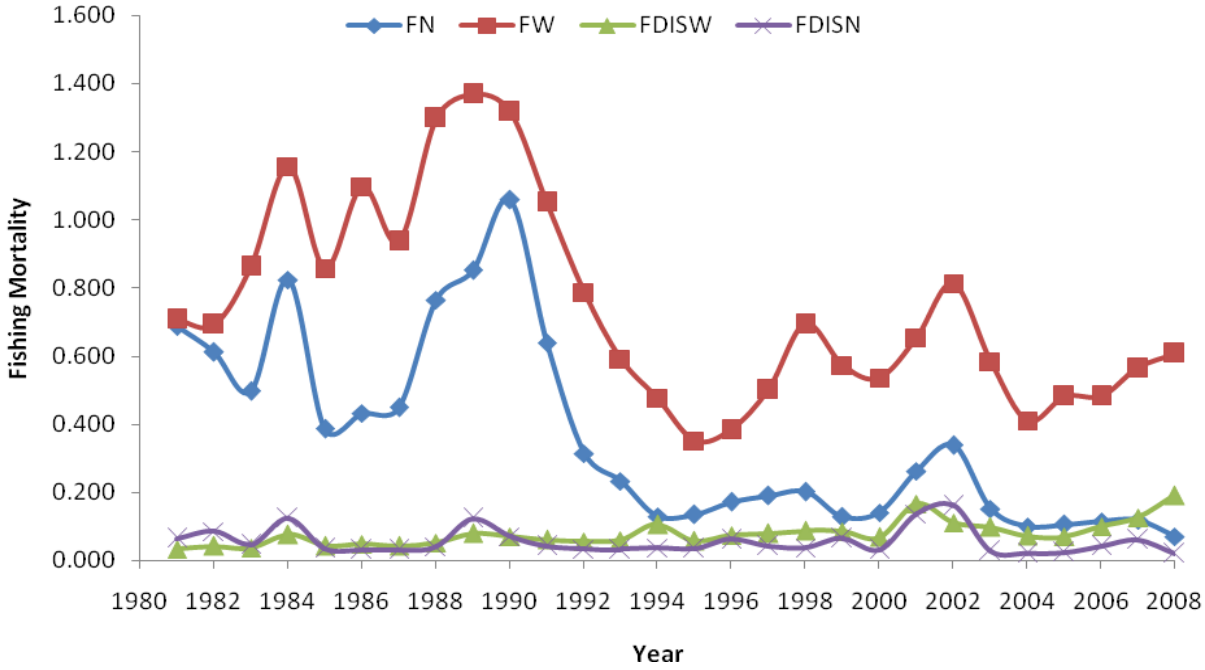


Figure C8.3-1. Plot of ages 1+ fishing mortality (FW, FN) on weakfish and fishing mortality due to discards (Fdisw, Fdisn).

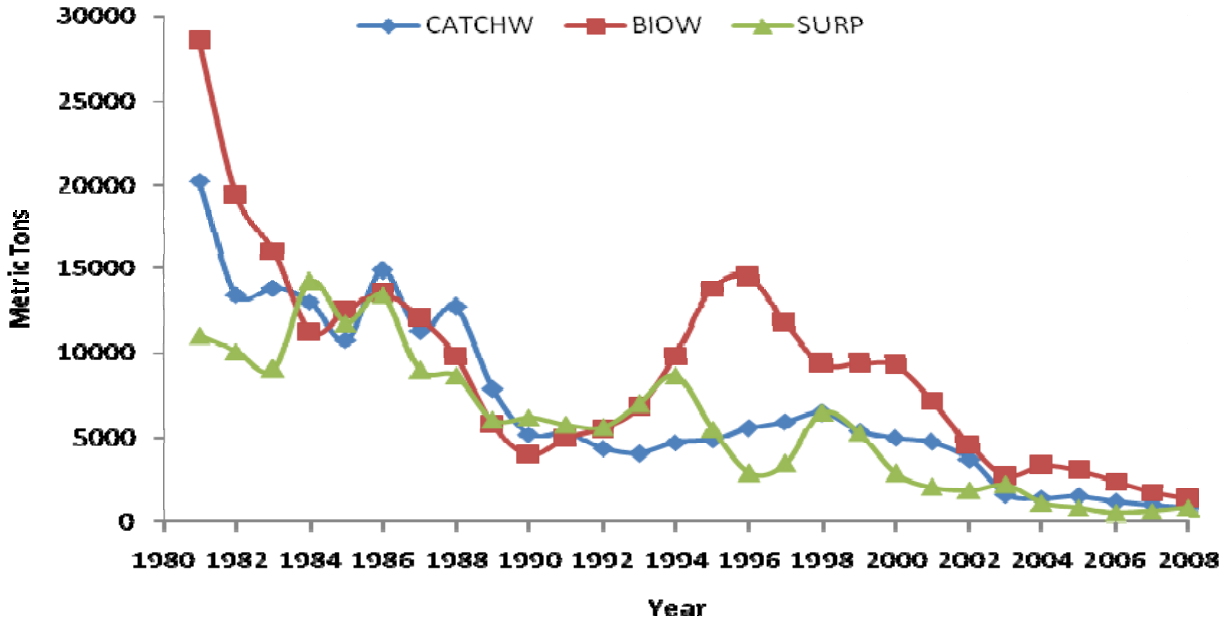


Figure C8.3-2. Plot of ages 1+ weakfish biomass (BIOW,mt), surplus production (SURP,mt) and coast-wide landings (CATCHW, mt), 1981-2008.

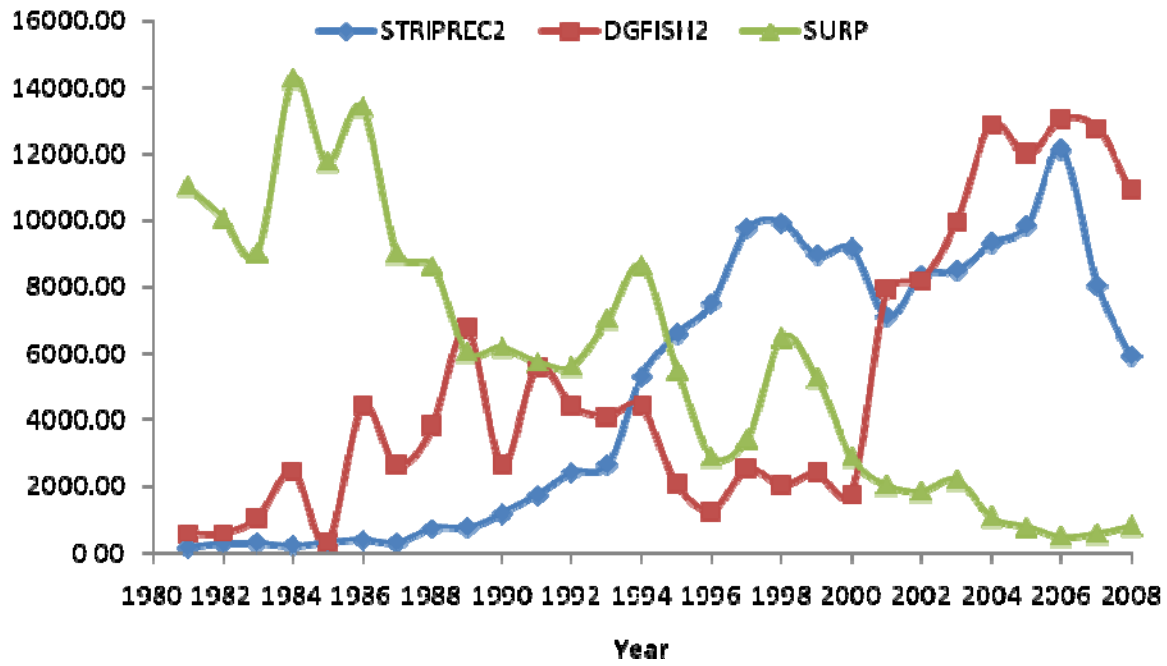


Figure C8.3-3. Relationship between weakfish surplus production and striped bass and spiny dogfish relative abundance, 1981-2007.

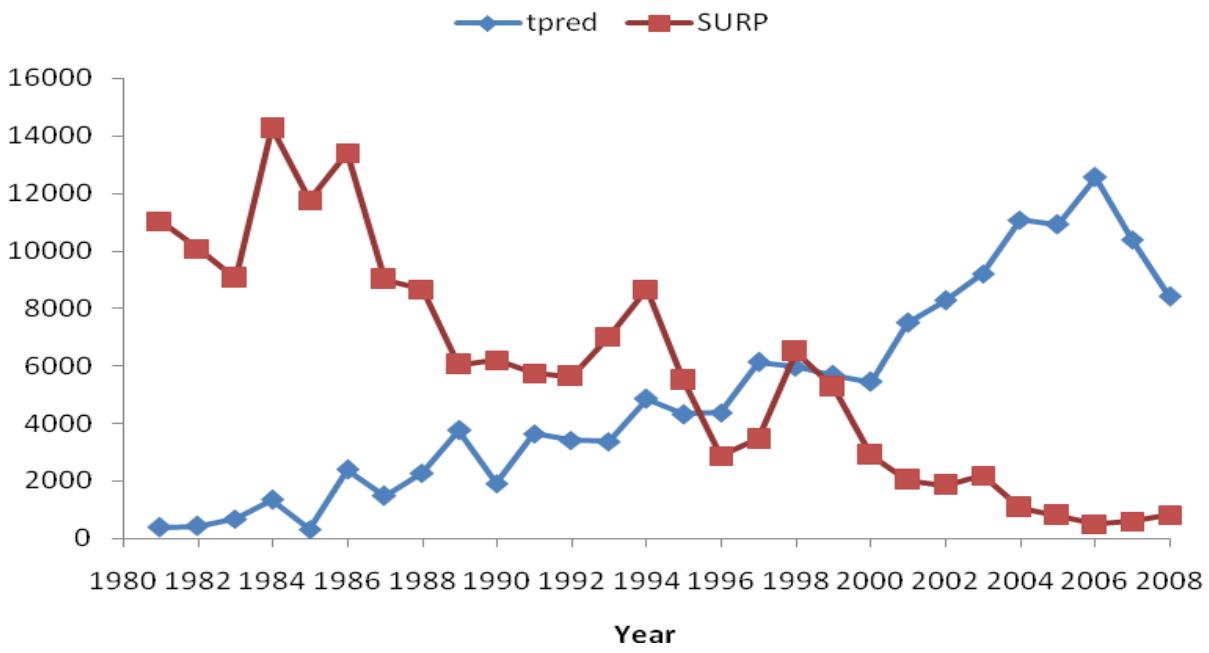


Figure C8.3-4. Relationship between weakfish surplus production and blended predation from striped bass and spiny dogfish, 1981-2008 .

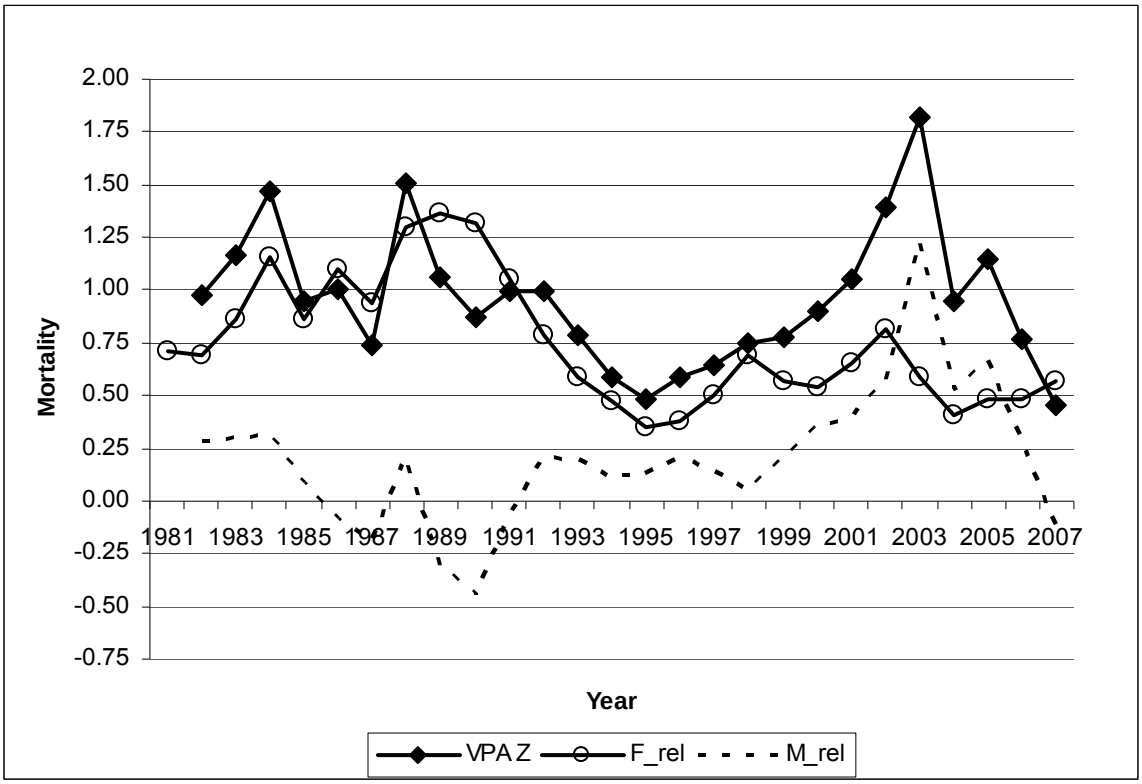


Figure C8.4-1. Comparison of biomass weighted Z from ADAPT and biomass weighted F from relative F calculations to indicate the expected trend in natural mortality.

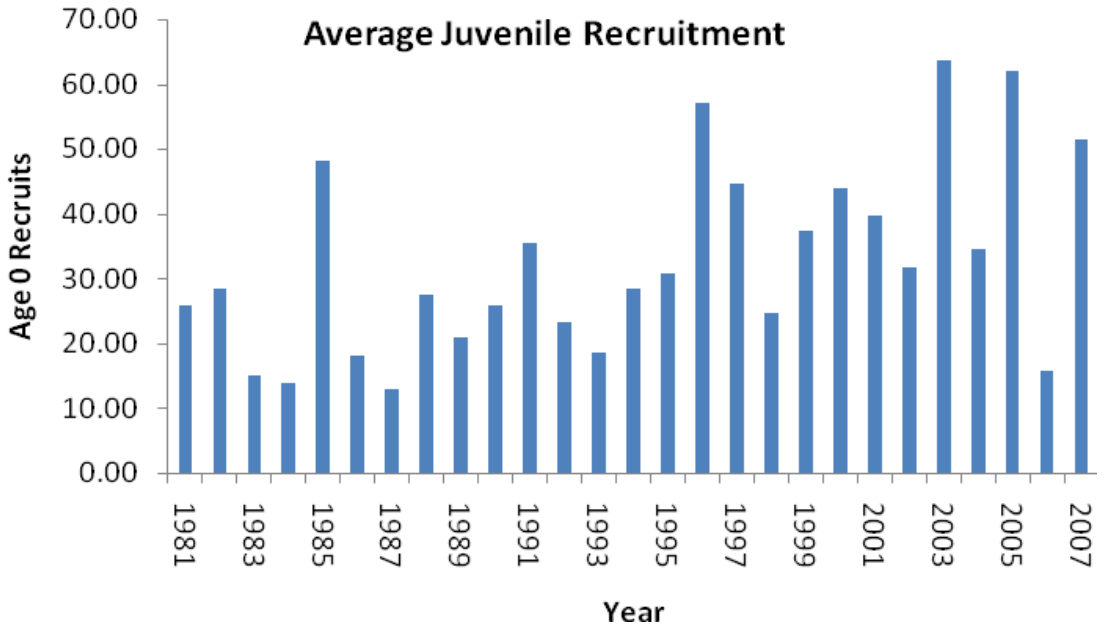


Figure C9.3-1. Plot of relative juvenile abundance of weakfish based on the average of nine juvenile indices, 1981-2007.

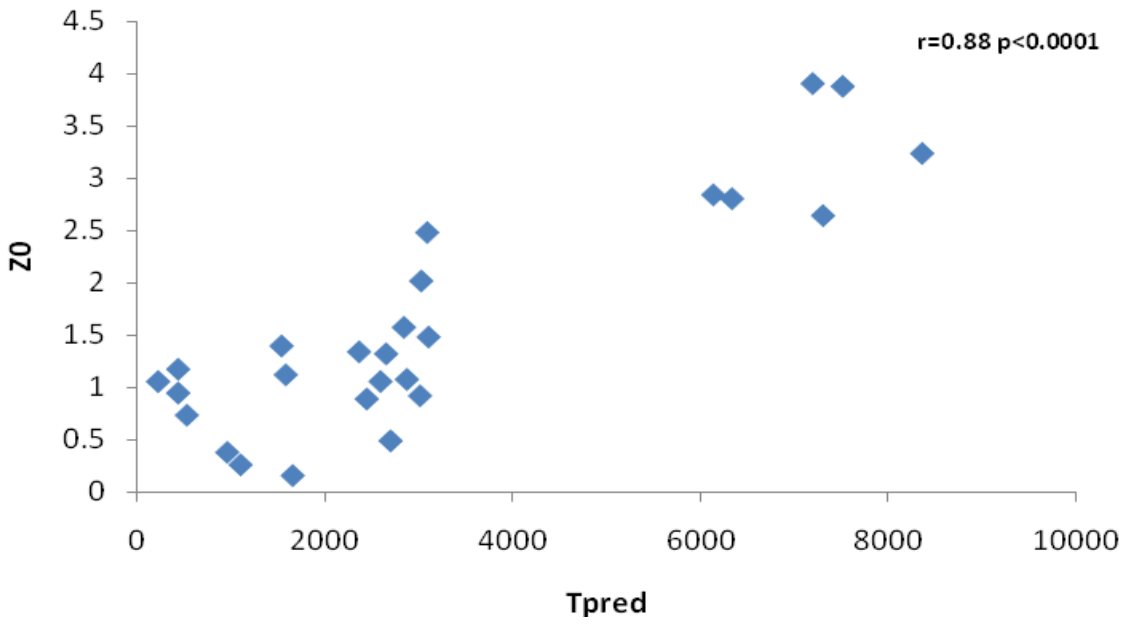


Figure C9.3-2. Relationship between weakfish juvenile mortality (z0) and average predator abundance (Tpred) based on striped bass and spiny dogfish, 1981-2007.

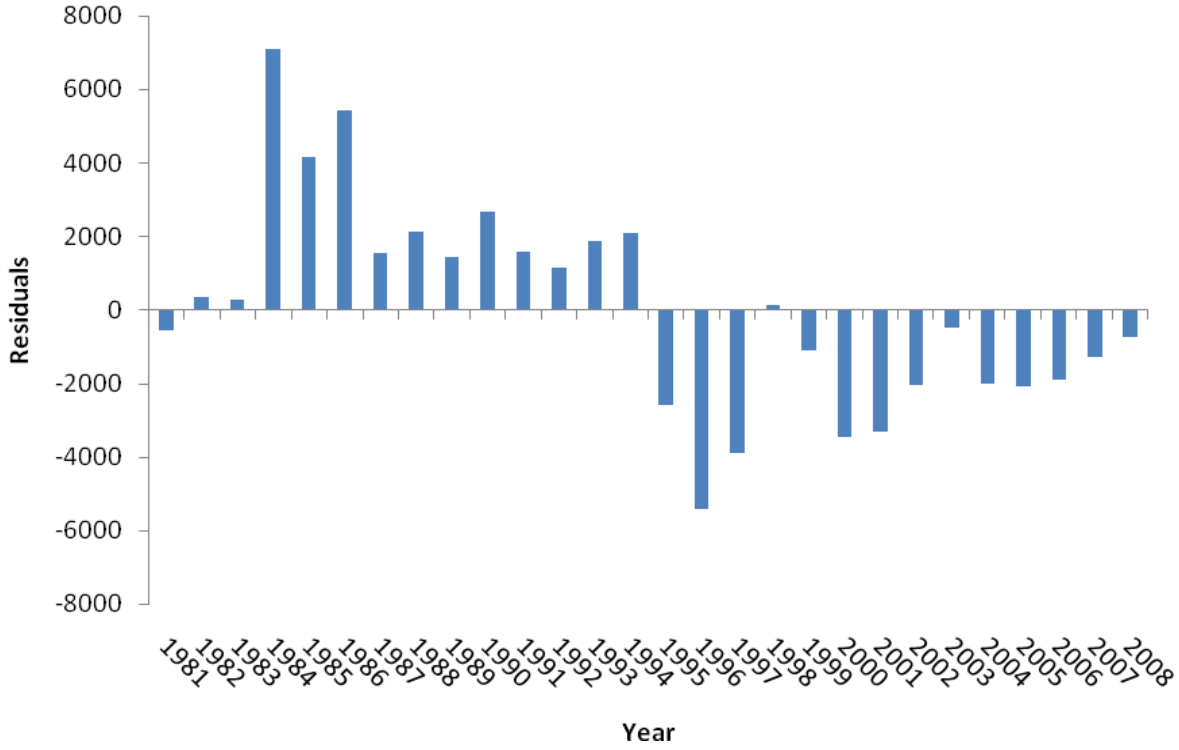


Figure C9.3-3. Anamalous residual pattern from Gompertz External model for weakfish without predation, 1981-2008.

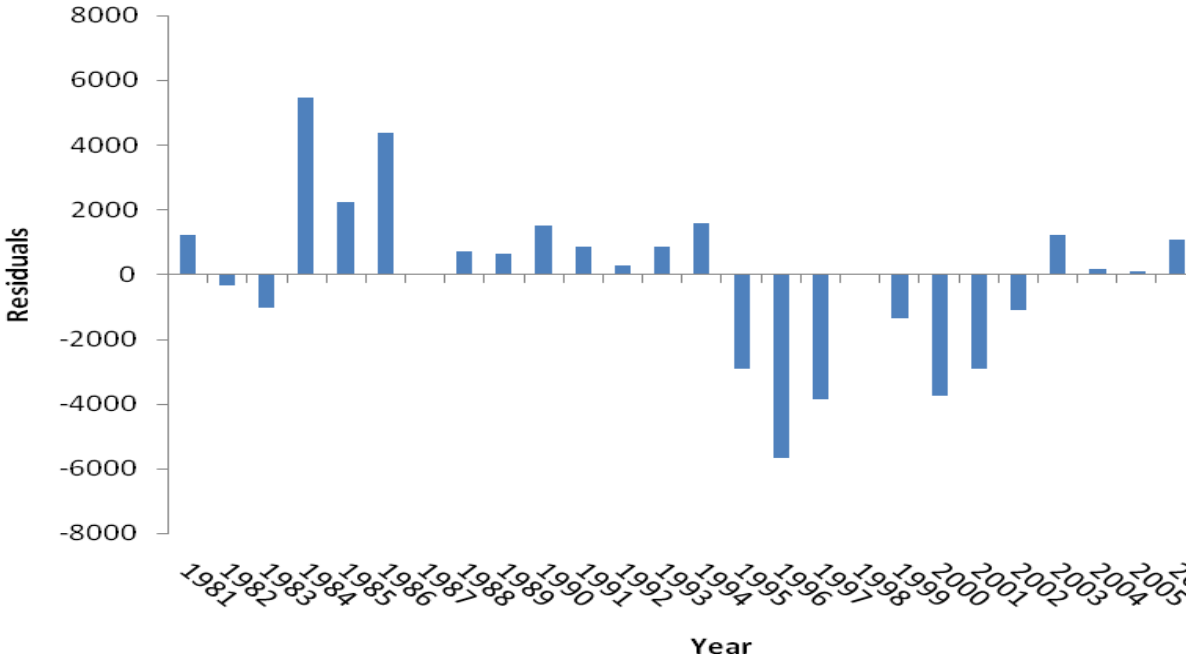


Figure C9.3-4. Residual pattern from Gompertz model for weakfish with blended dogfish and striped bass predation effects, 1981-2008.

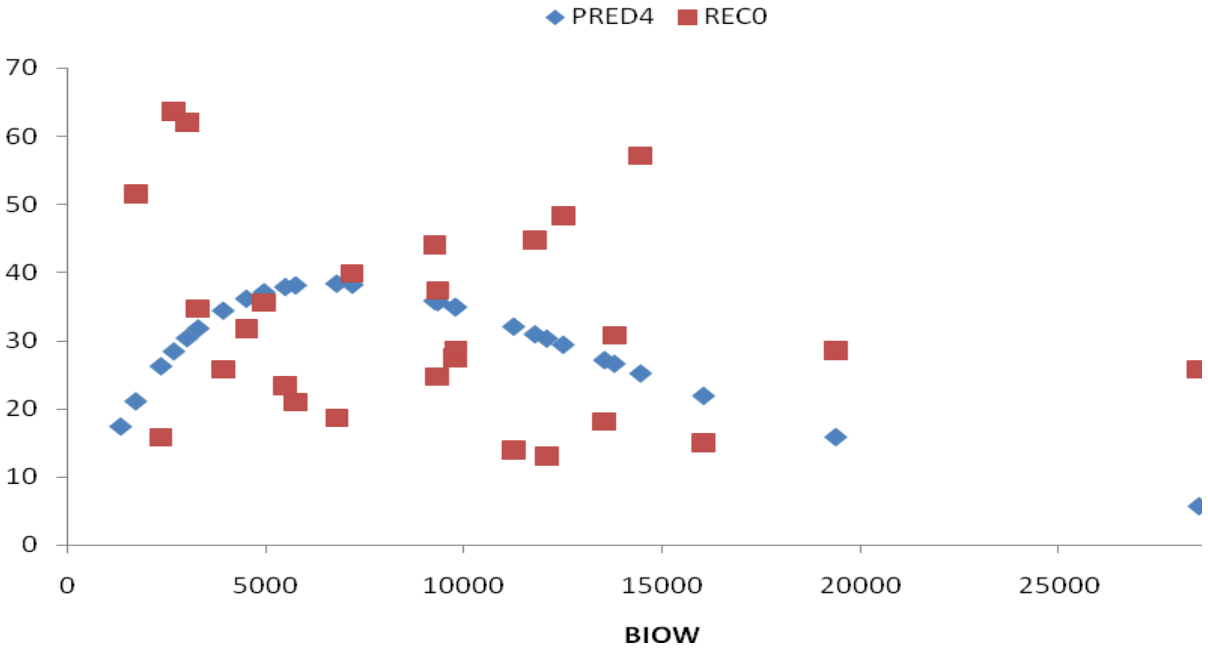


Figure C9.3-5. Ricker stock-recruitment fit to average coast-wide recruitment and weakfish biomass

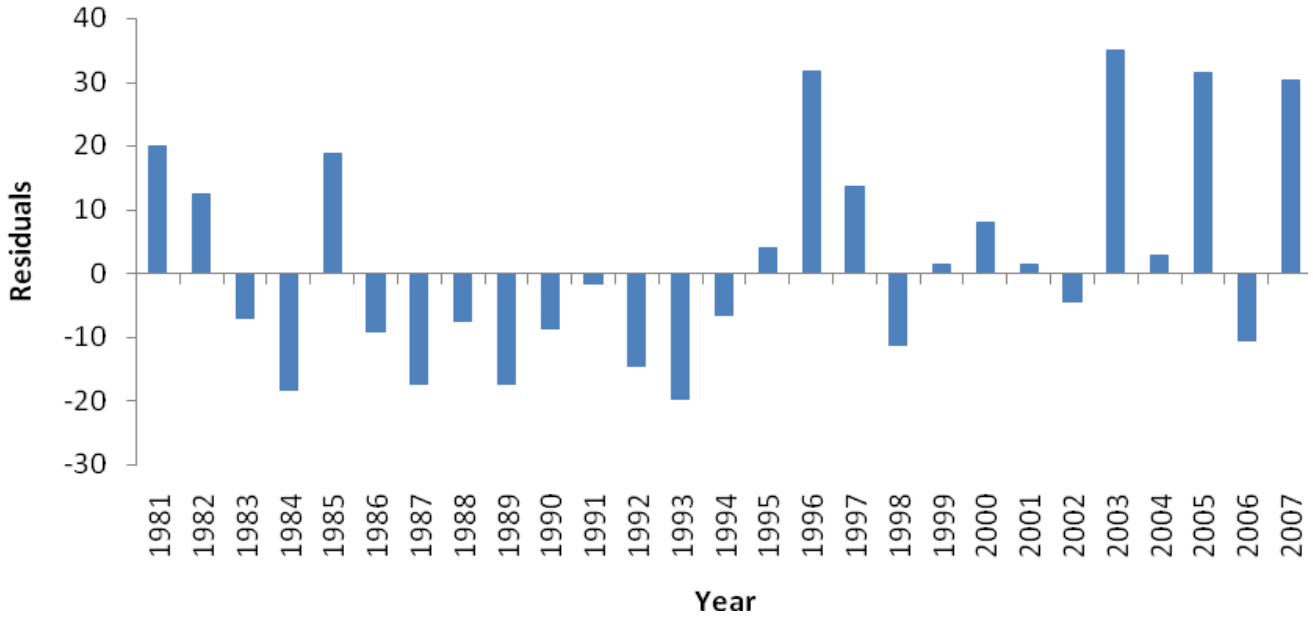


Figure C9.3-6. Serial correlation of residuals for the Ricker S-R model for weakfish without predation, 1981-2008.

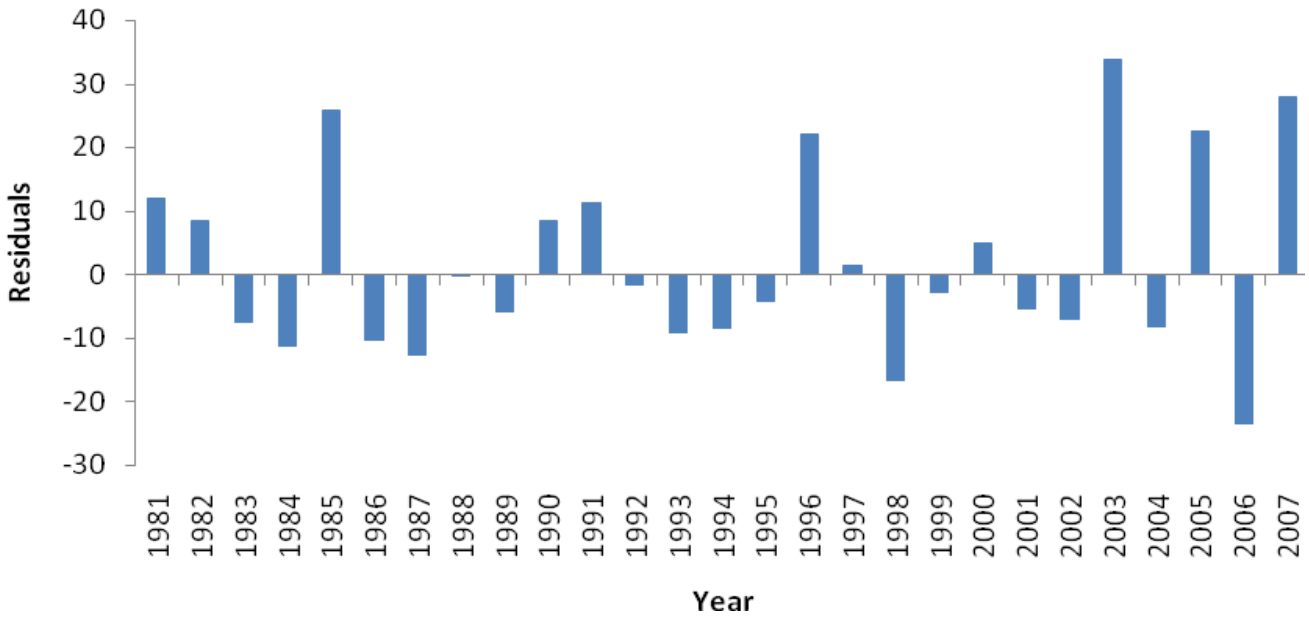


Figure C9.3-7. Residual plot for the Ricker S-R model with striped bass and spiny dogfish predation, 1981-2007.

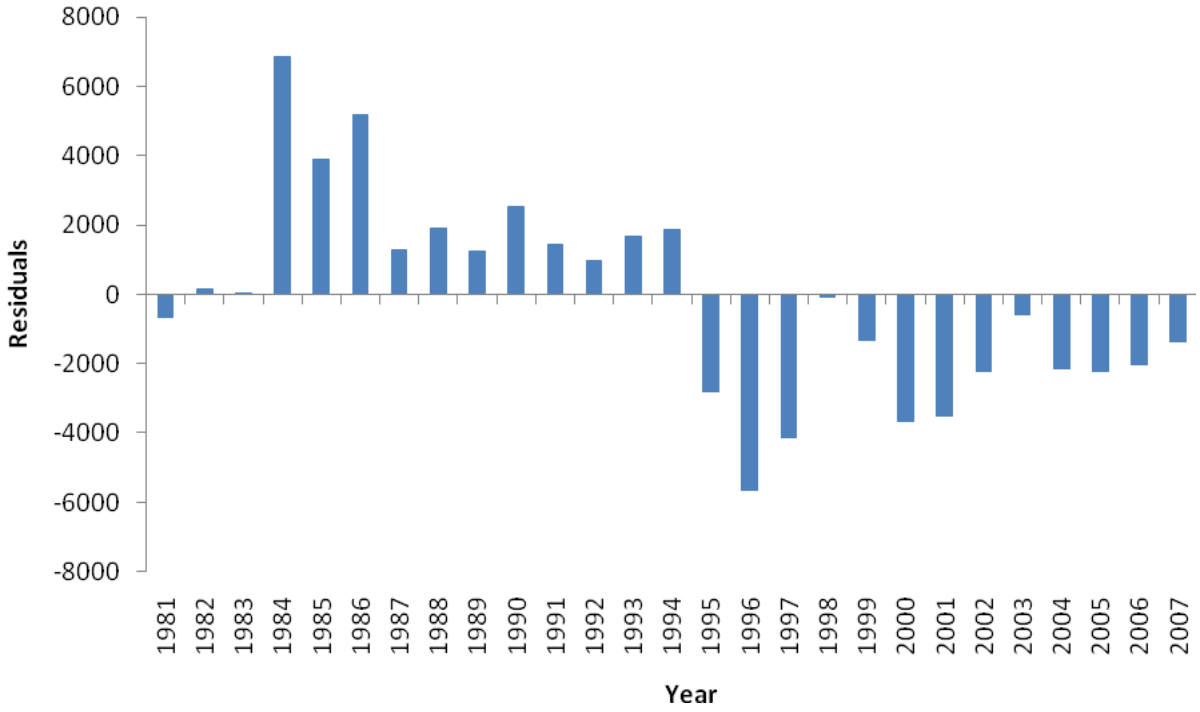


Figure C9.3-8. Residual plot for discrete Gompertz Production Model on weakfish without predation, 1981-2008.

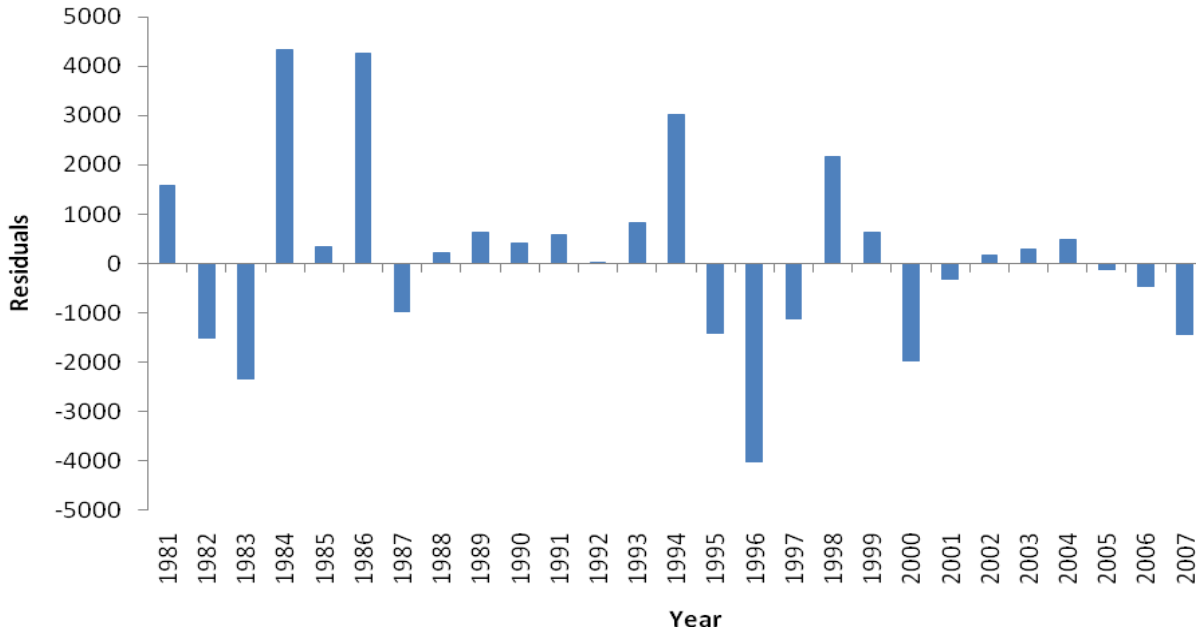


Figure C9.3-9. Residual plot for Steele-Henderson model for weakfish with predation from spiny dogfish and striped bass, 1981-2008.

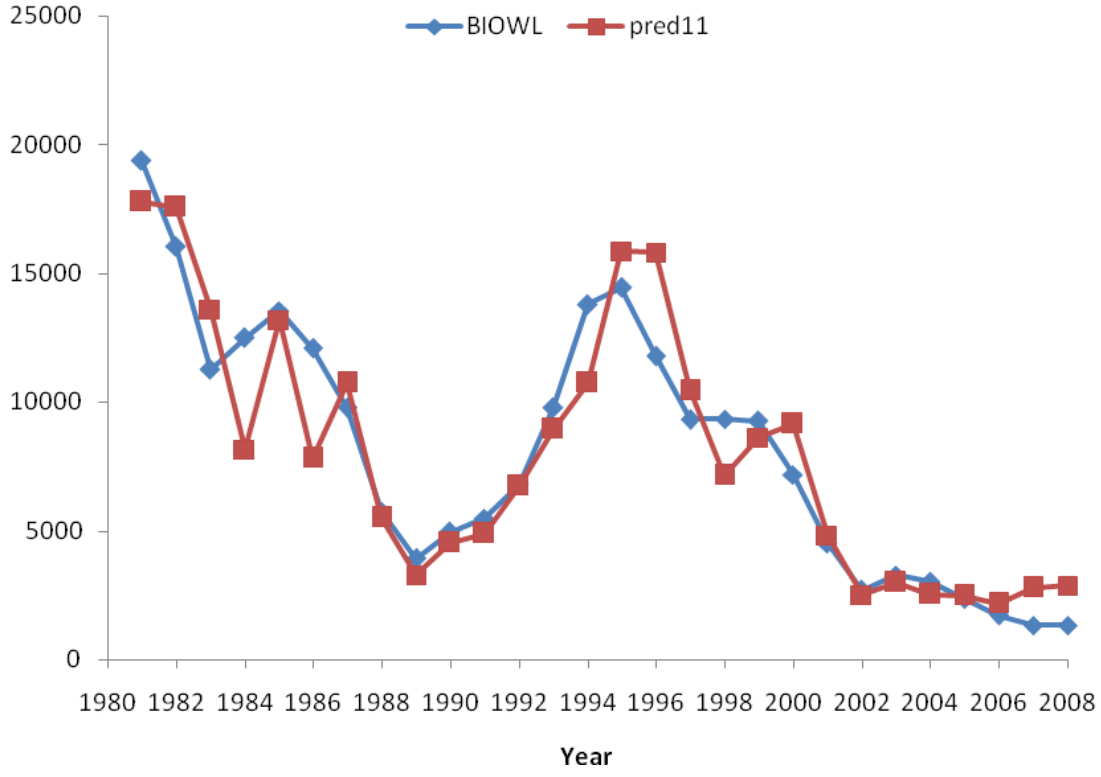


Figure C9.3-10. Plot of observed weakfish biomass and predicted biomass from the Steele-Henderson Model.

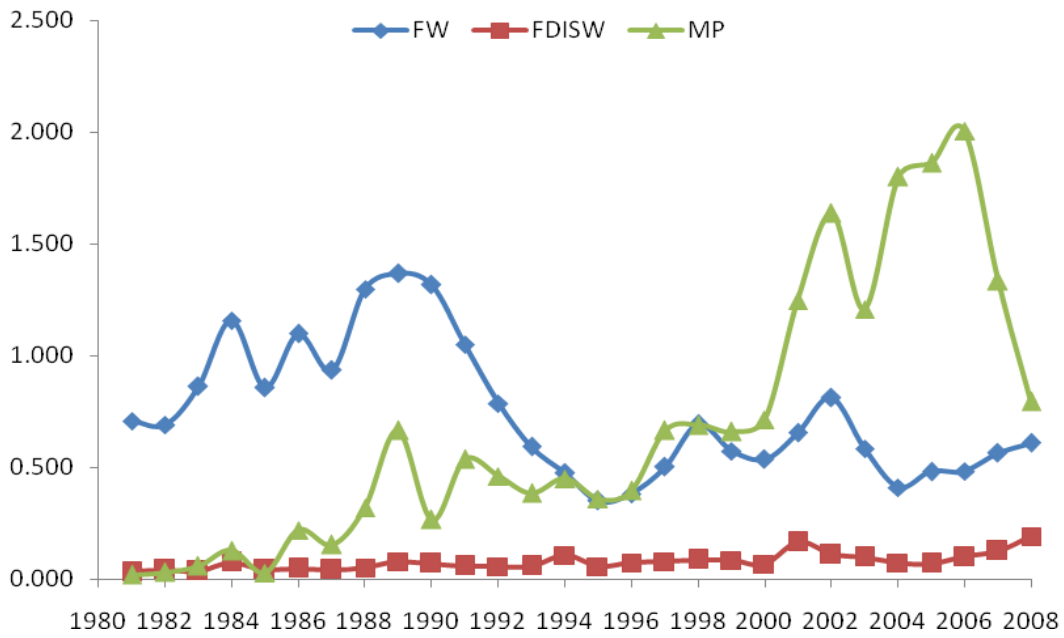


Figure C9.3-11. Plot of predation mortality (MP), total fishing mortality (FW) and discard fishing mortality

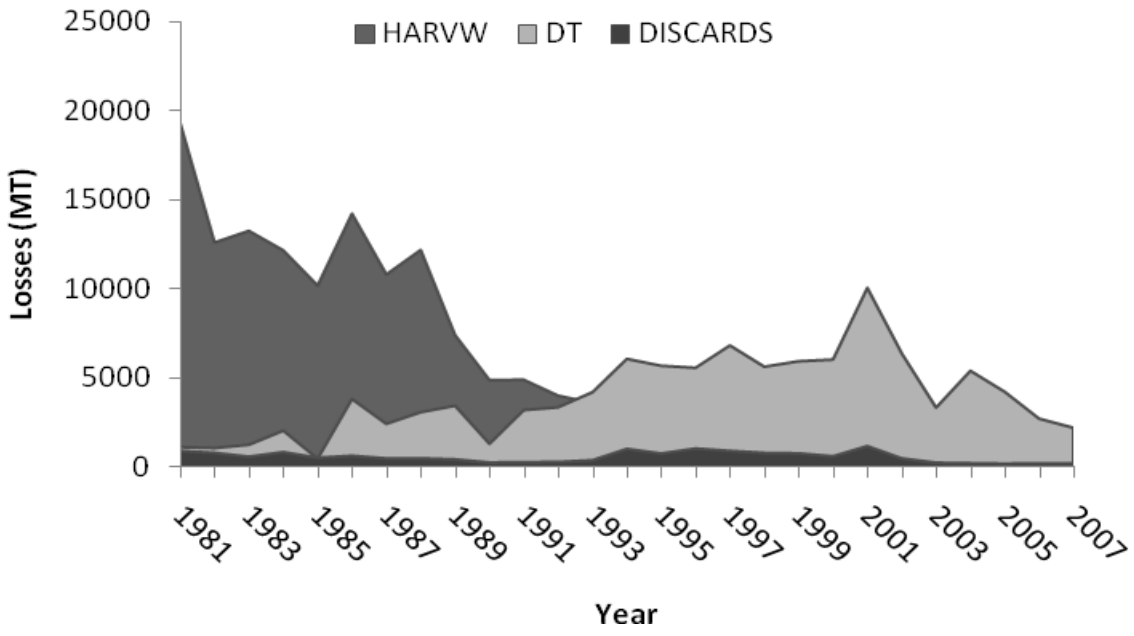


Figure C9.3-12. Contribution of predation losses (DT) to total harvest (HARVW) and discards (DISCARDS).

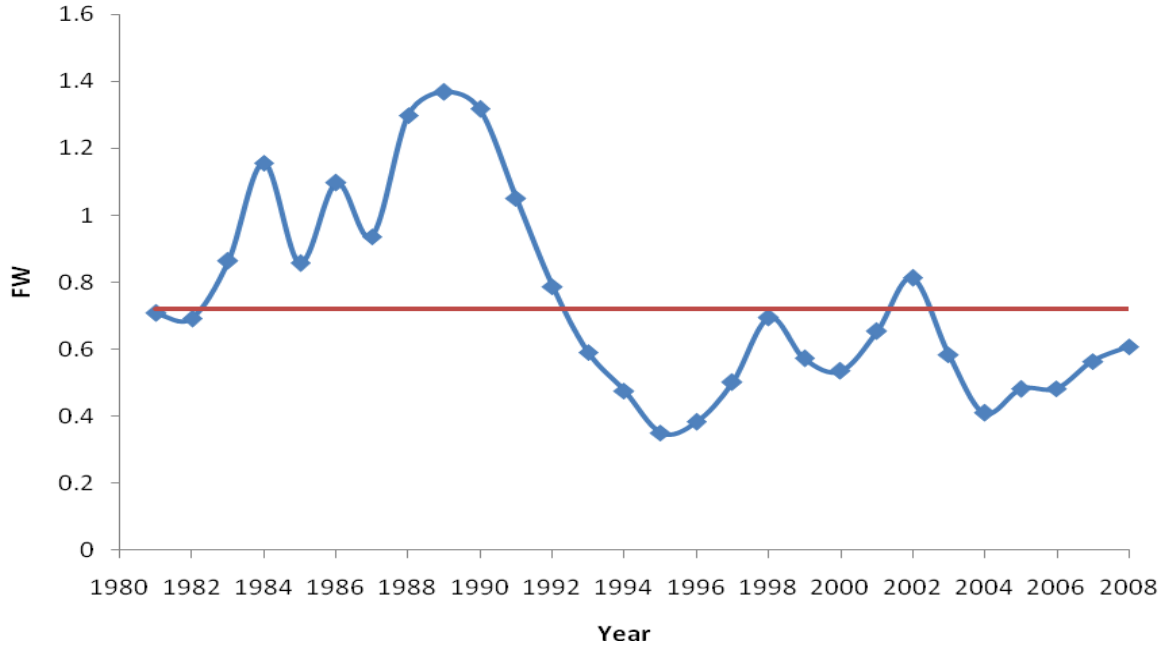


Figure C9.3-13. Relationship between ages 1+ biomass weighted F on weakfish and equilibrium FMSY from 1981 to 2008.

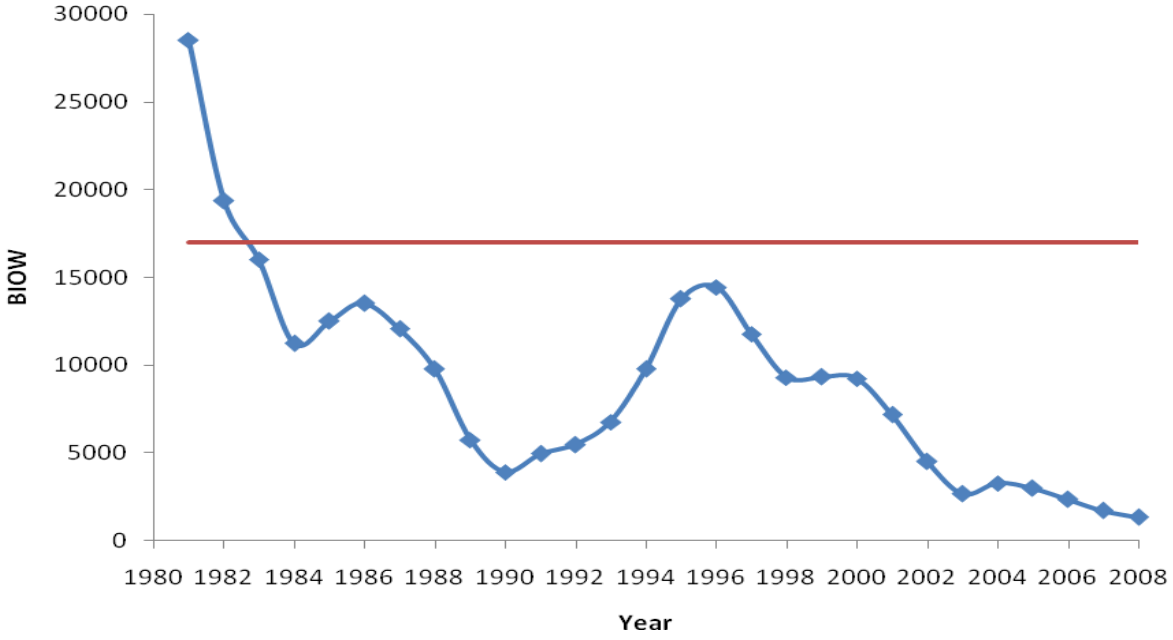


Figure C9.3-14. Relationship between weakfish biomass (MT) and equilibrium BMSY from 1981-2008.

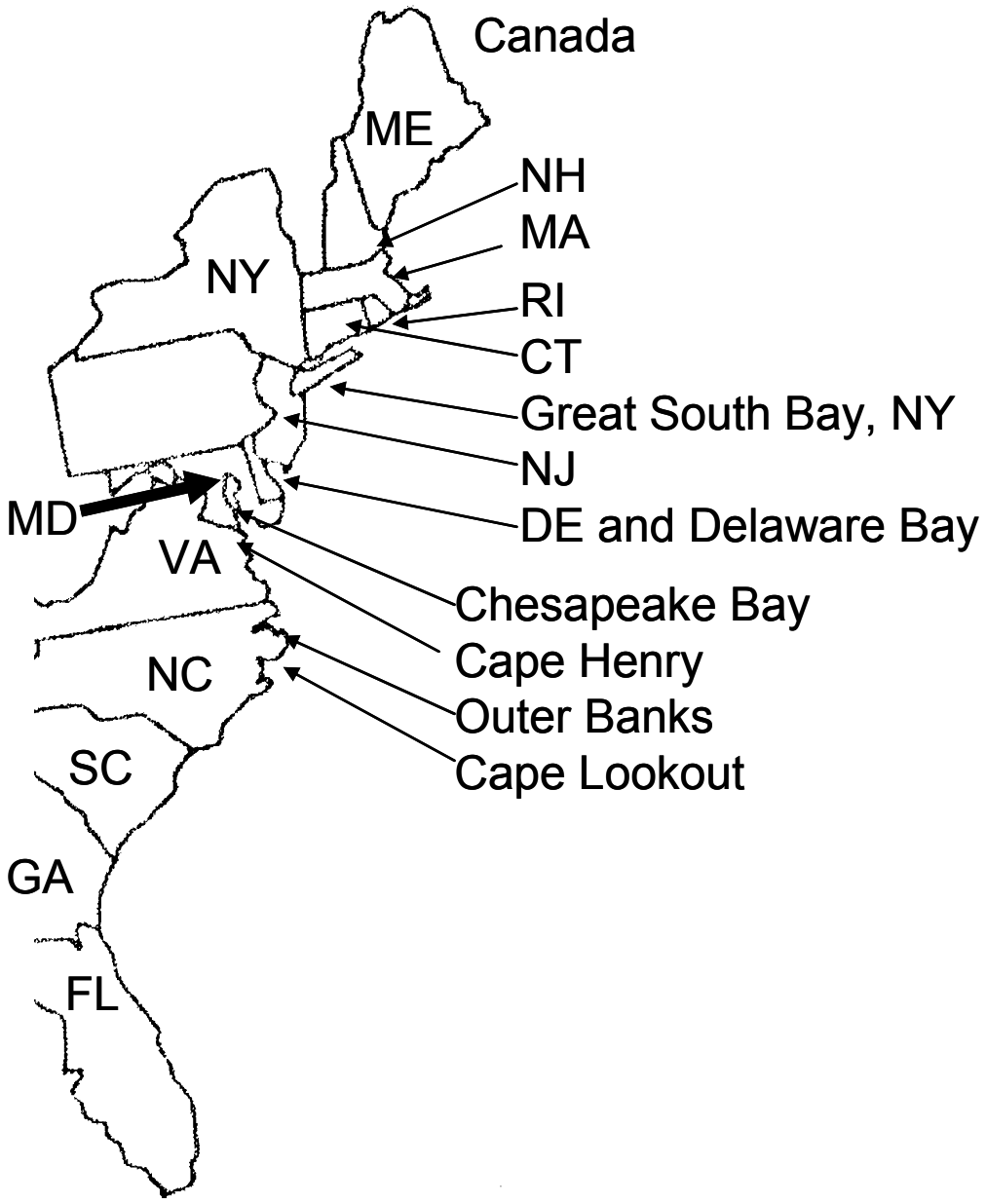


Figure C10.3-1. The Atlantic Coast of the United States. Prominent states and locations are listed. States from FL to MA participate in weakfish management. (Apologies to Canada)

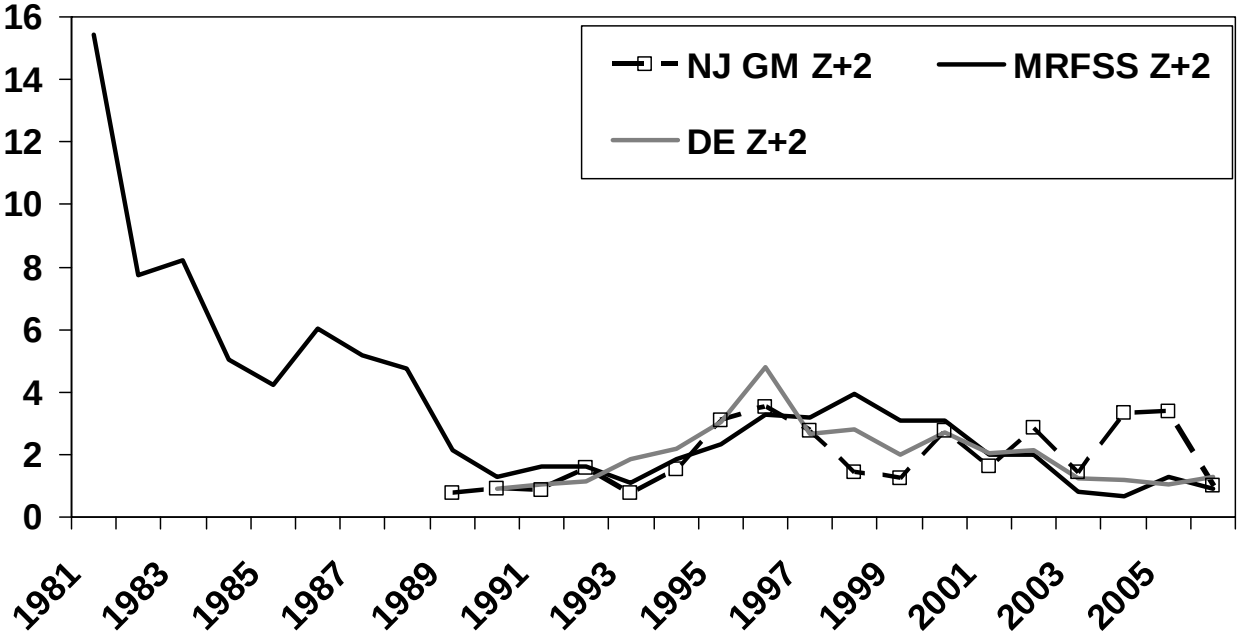


Figure C10.4-1. Standardized indices (Z+2) from 1981-2003 assessment: NJ, MRFSS, and DE.

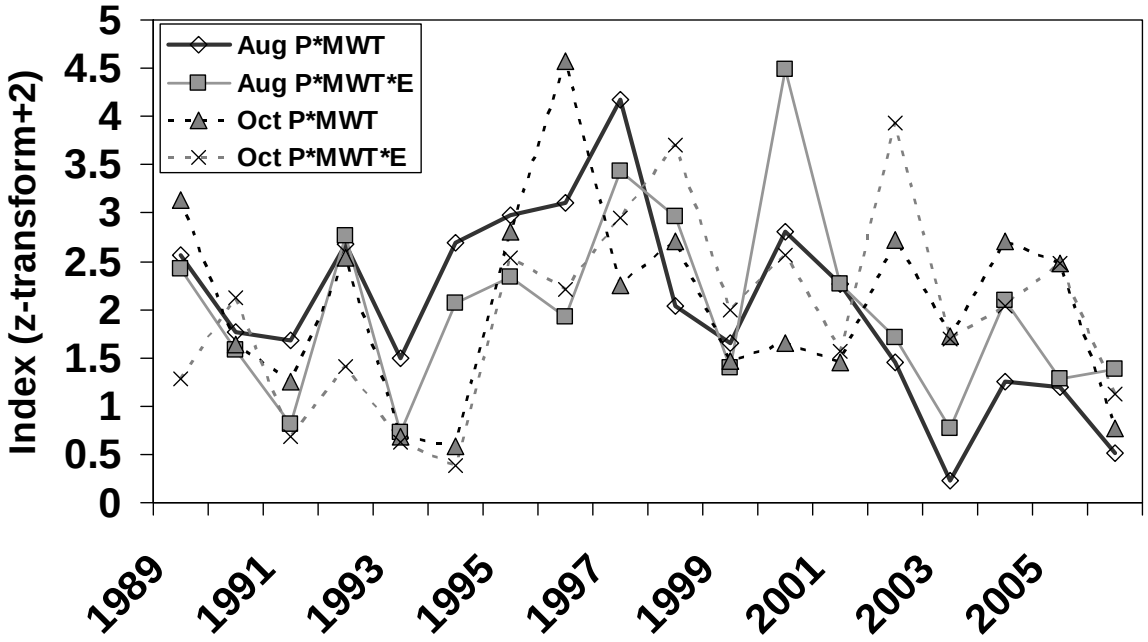


Figure C10.4-2. Proportion of trawls with weakfish based exploitable biomass indices based on August or October rounds of the NJ survey. P = proportion of trawls with weakfish; MWT = mean weight of all weakfish; and E = exploitable fraction of total trawl biomass.

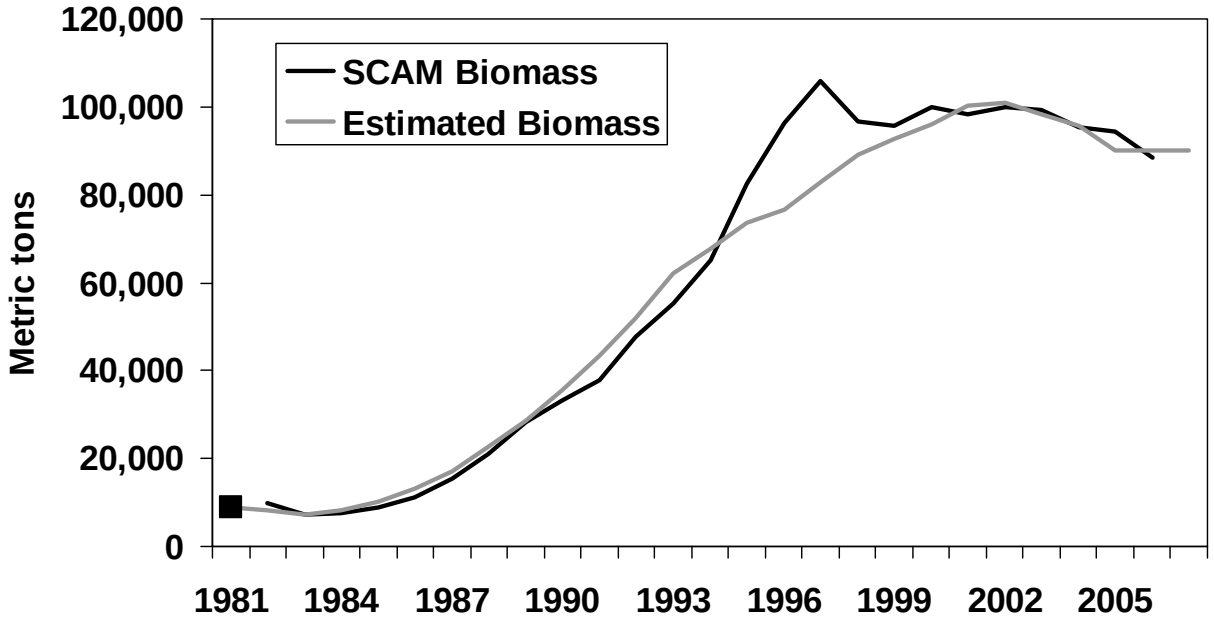


Figure C10.4-3. Striped bass biomass estimated from SCAM and known biomass production model. Square indicates estimate for 1981.

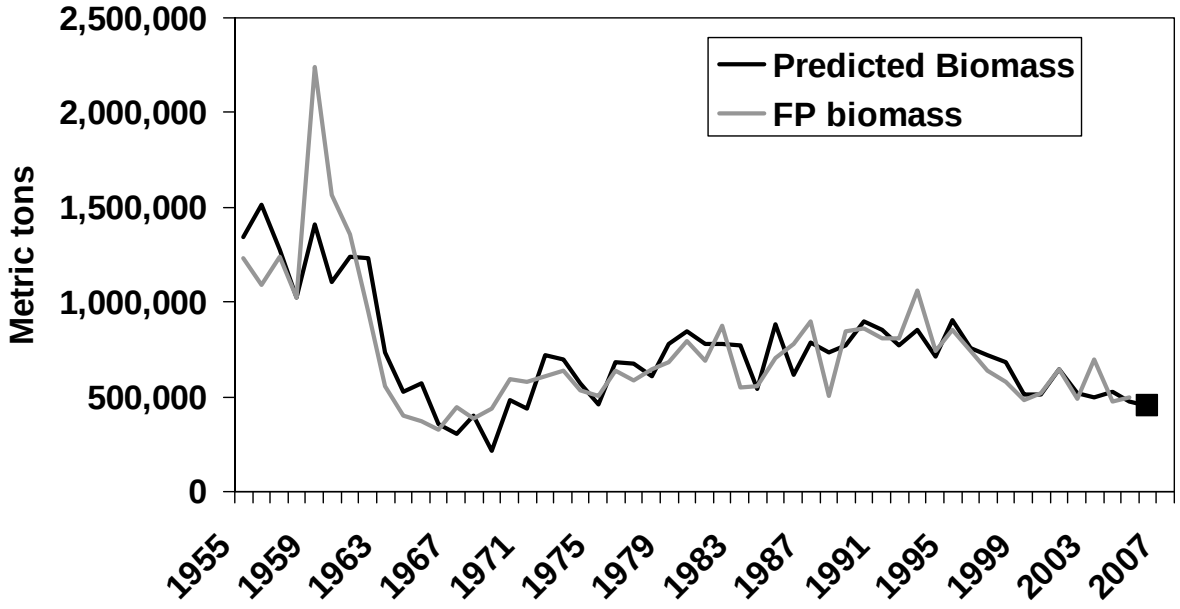


Figure C10.4-4. Biomass of ages 1+ Atlantic menhaden estimated from forward projection (FP) and predicted from categorical regression. Square indicates estimate for 2006.

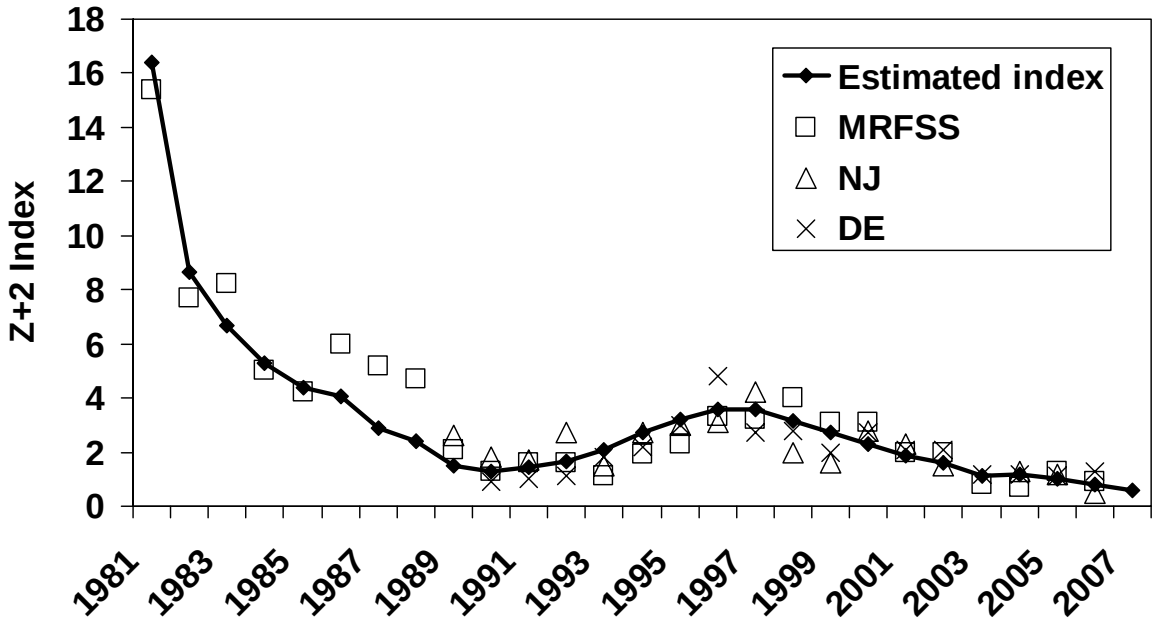


Figure C10.5-1. Observed and estimated indices. Indices have been z-transformed to the 1990-2006 time series and had 2 added to eliminate negative values.

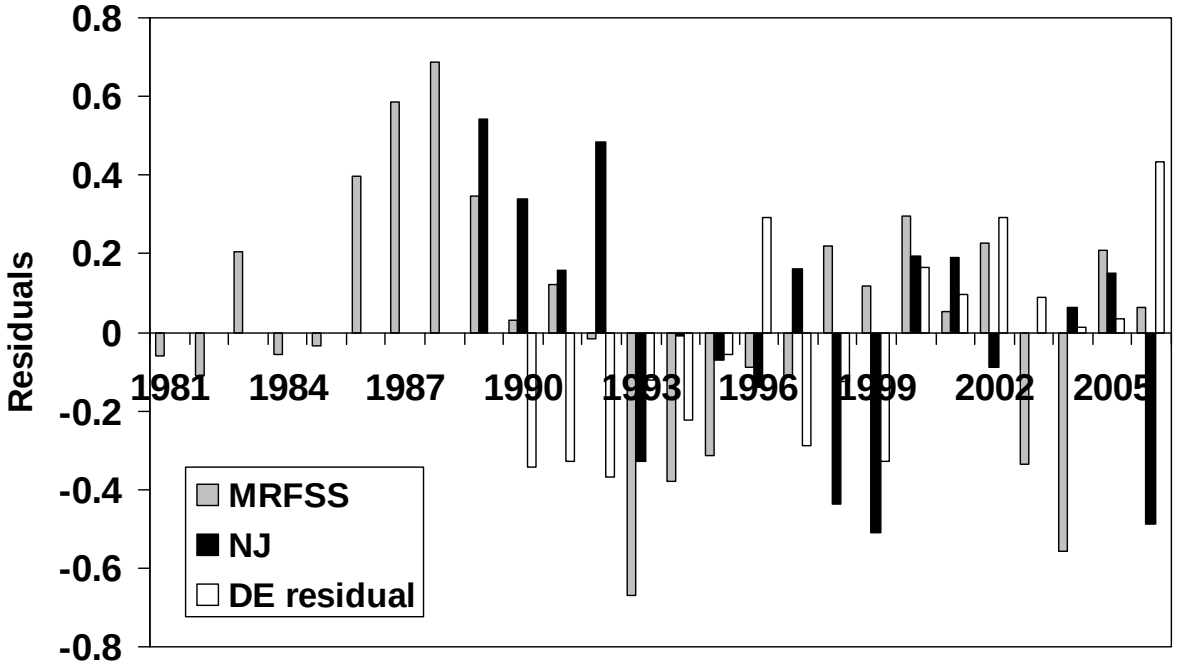


Figure C10.5-2. Residuals of the GDR versus year.

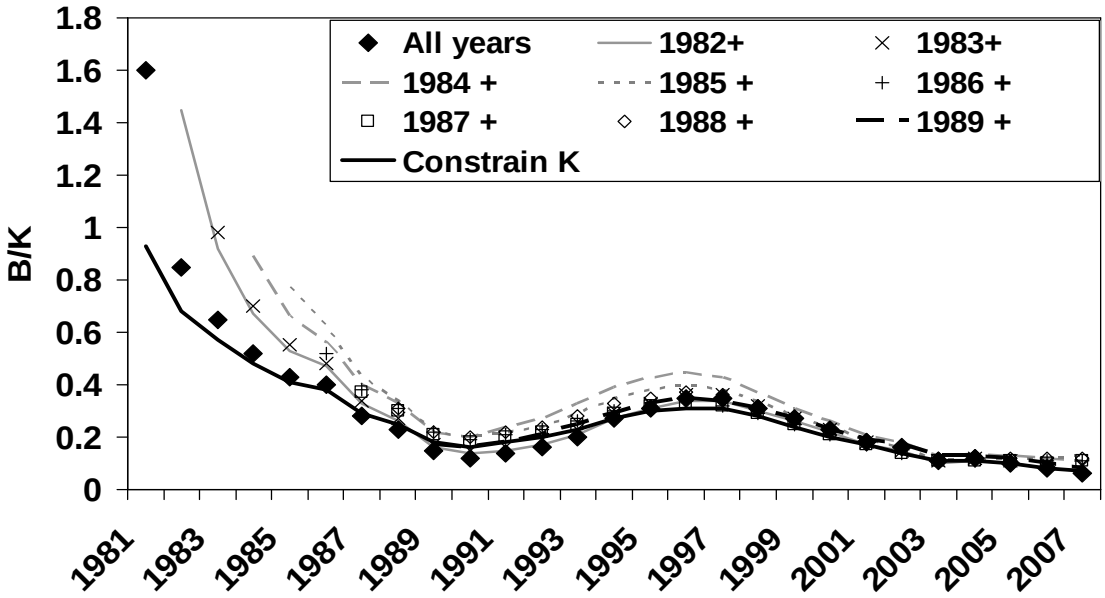


Figure C10.5-3. Estimates of biomass / unfished stock biomass (B_t / K) for sensitivity runs with time-blocks removed from the beginning or B_{1981} constrained to be less than K. Start of time-series is indicated by year in legend.

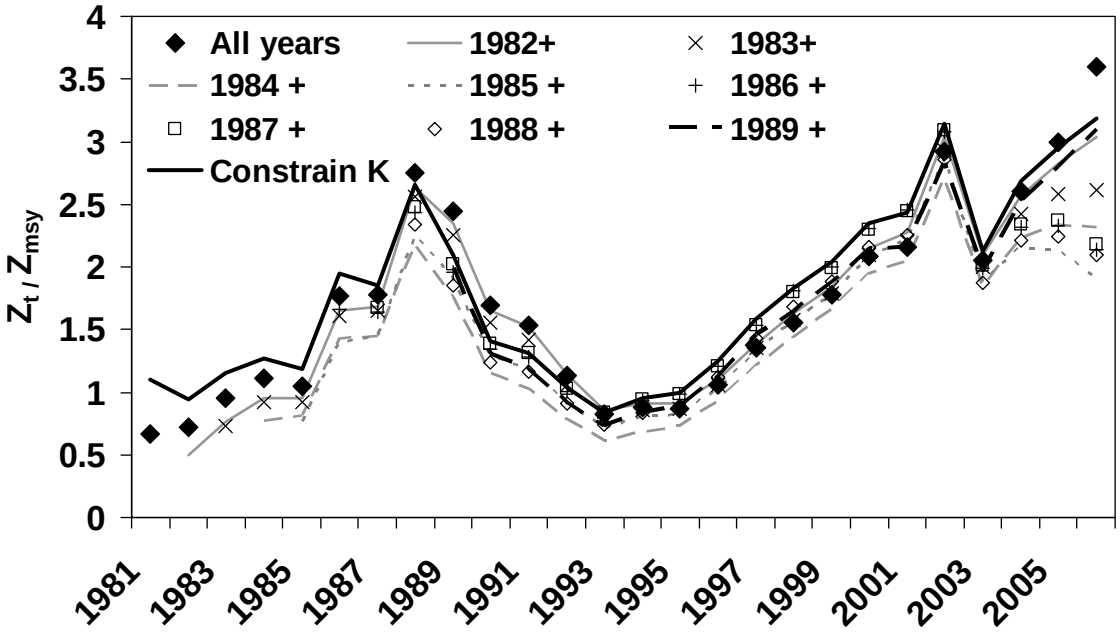


Figure C10.5-4. Estimates of Z_t / Z_{msy} for sensitivity runs with time-blocks removed from the beginning or B_0 constrained to be less than K. Start of time-series is indicated by year in legend.

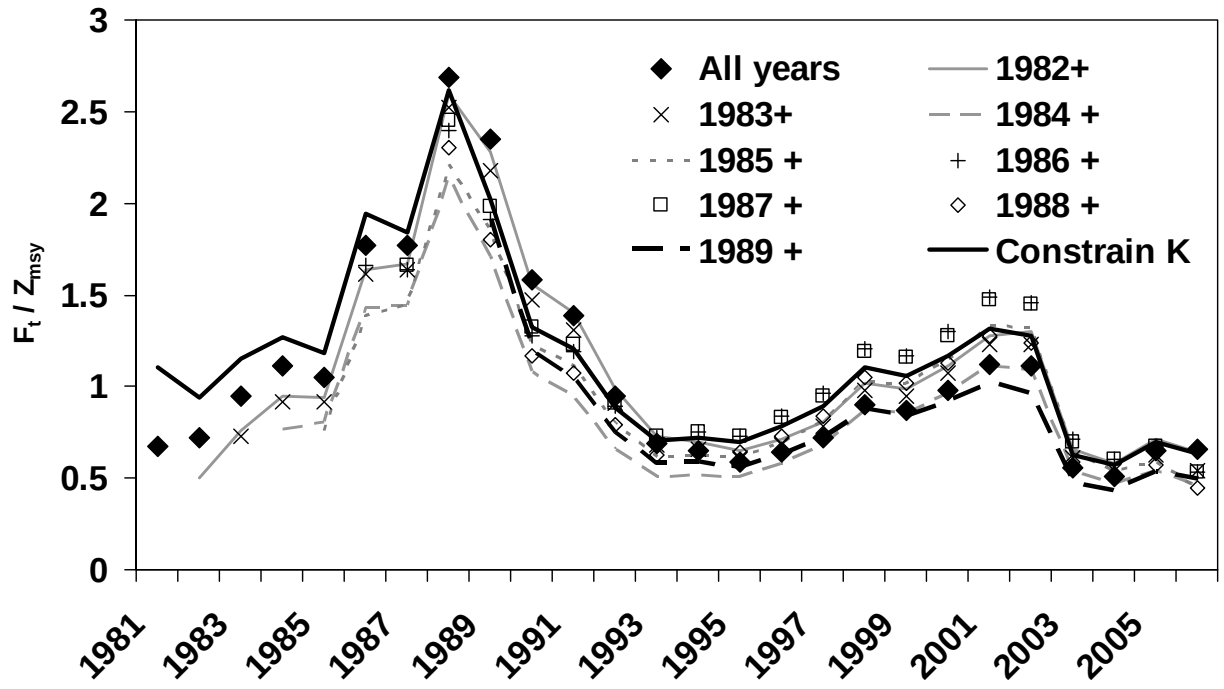


Figure C10.5-5. Estimates of F_t / Z_{msy} for sensitivity runs with time-blocks removed from the beginning or B_0 constrained to be less than K. Start of time-series is indicated by year in legend.

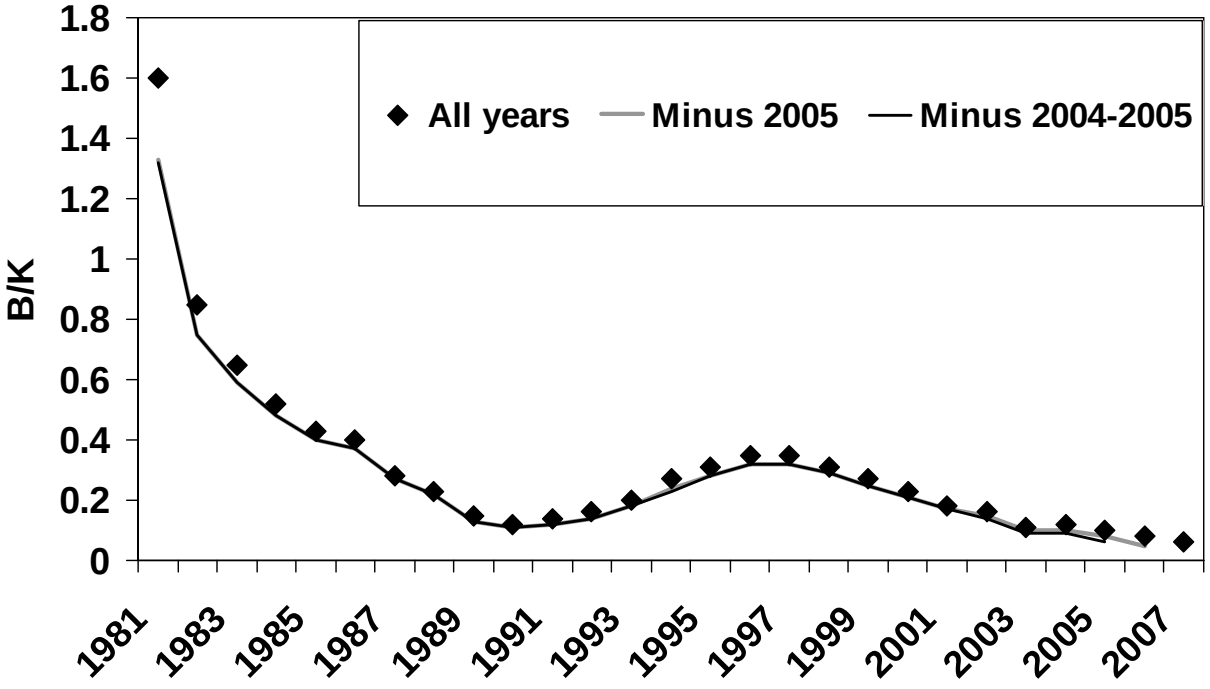


Figure C10.5-6. Estimates of biomass / unfished stock biomass (B / K) for sensitivity runs with time-blocks removed from the end of the time series.

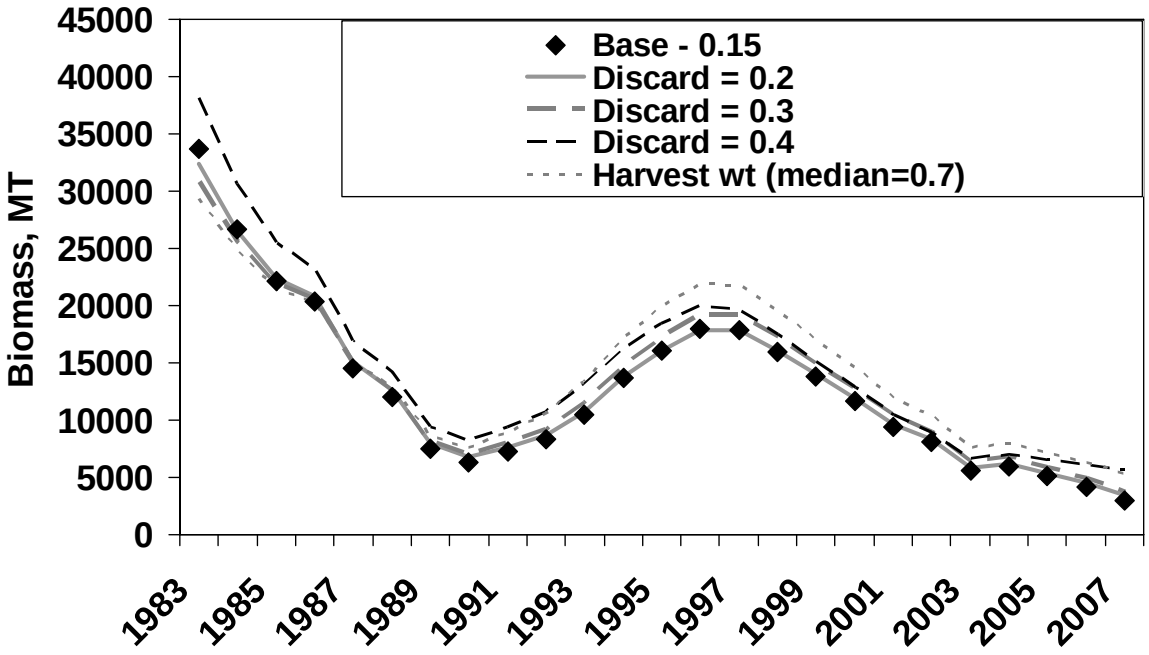


Figure C10.5-7. Estimates of biomass for sensitivity runs with different mean weights applied to recreational discards in the MRFSS index and fishery losses.

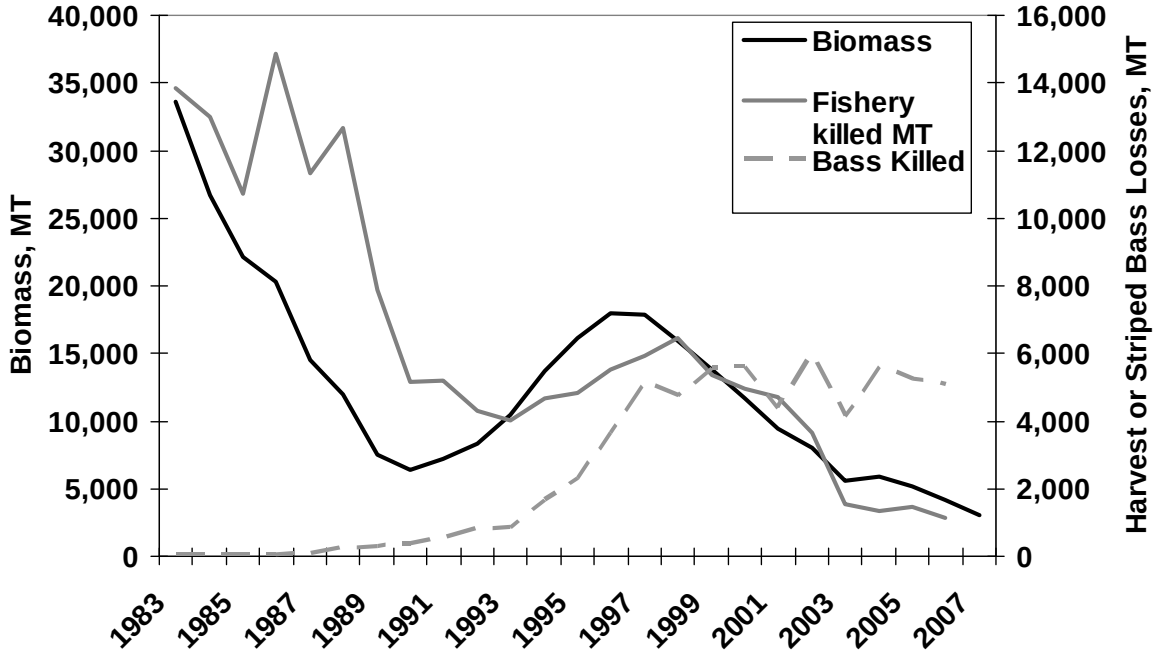


Figure C10.5-8. Trends in estimates of weakfish biomass and losses due to fisheries and striped bass predation-competition. Note separate axes for biomass and harvest or striped bass related losses.

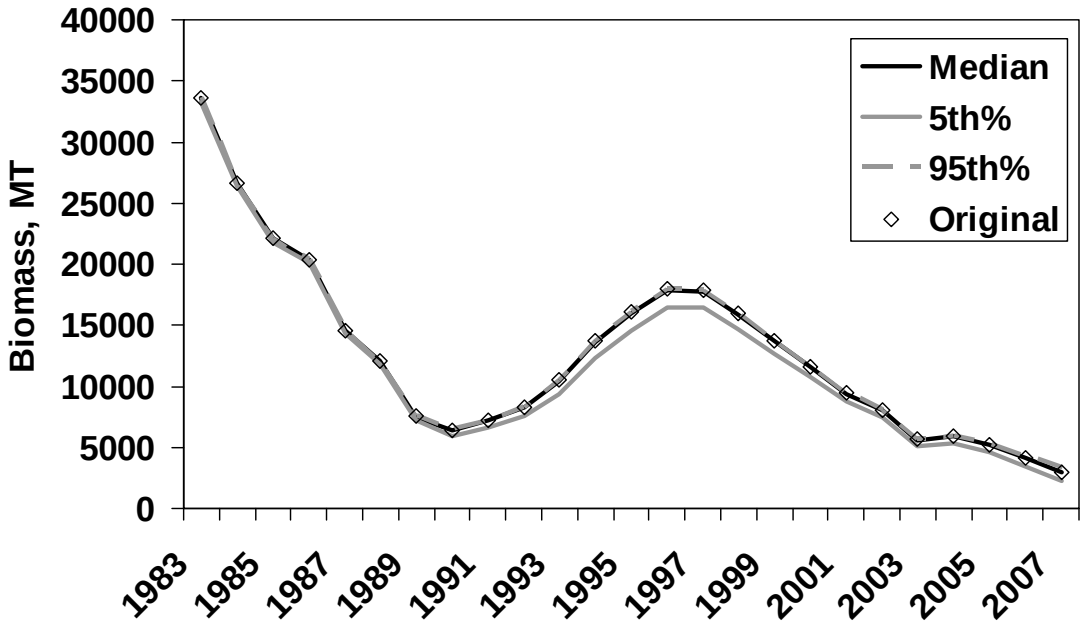


Figure C10.5-9. Jackknife estimates of weakfish biomass 90% confidence intervals.

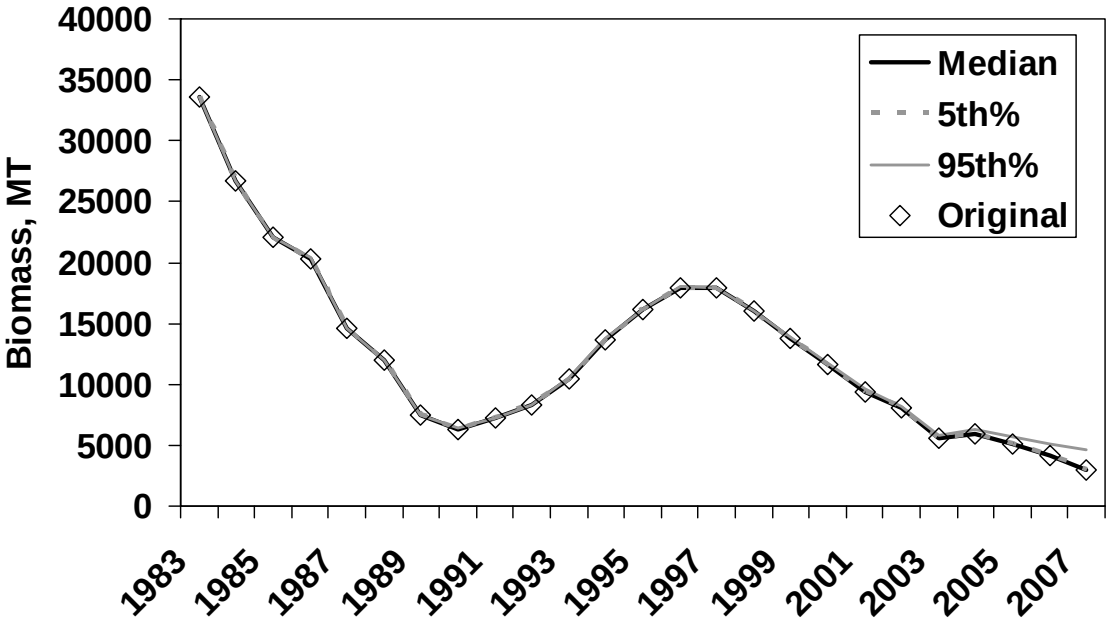


Figure C10.5-10. Bootstrapped estimates (N = 500) of biomass 90% confidence intervals.

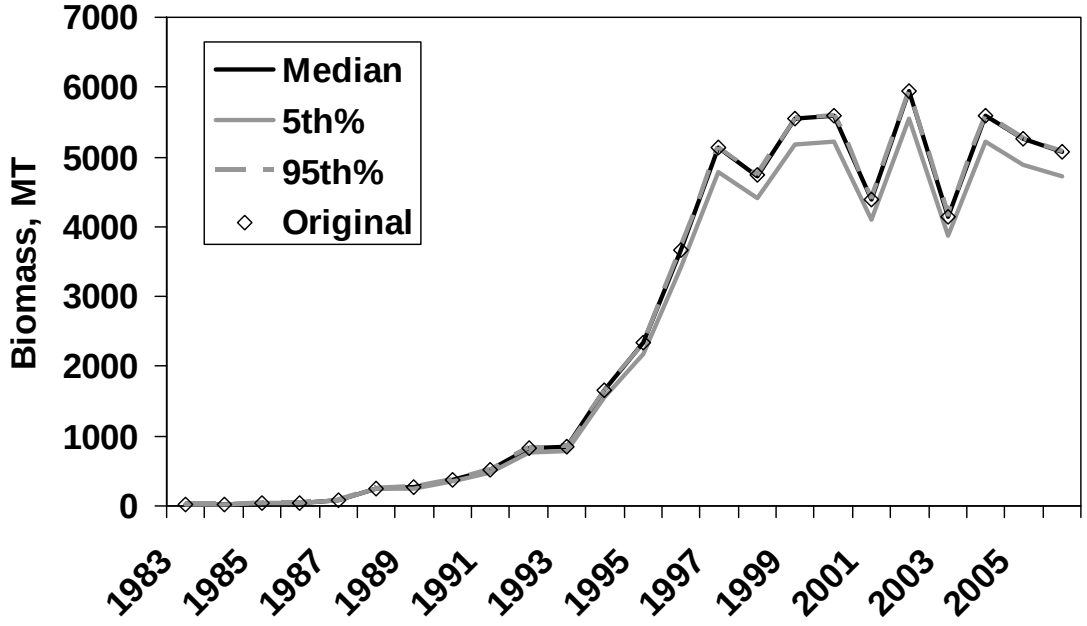


Figure C10.5-11. Jackknife estimates of 90% confidence intervals of weakfish biomass lost to striped bass.

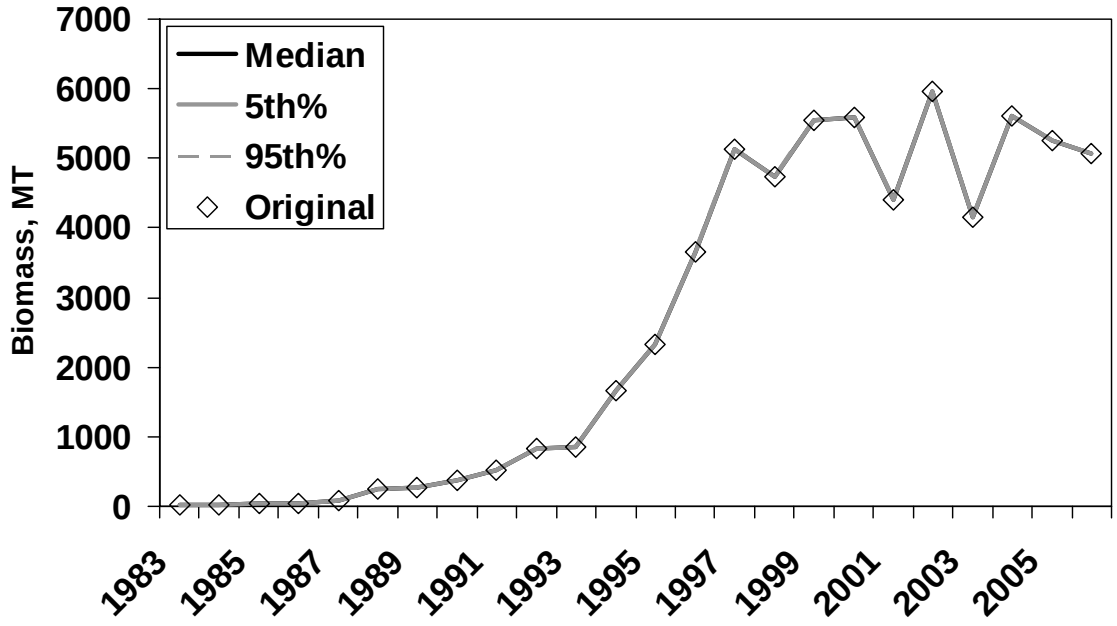


Figure C10.5-12. Bootstrapped estimates (N = 500) of 90% confidence intervals of weakfish biomass lost to striped bass.

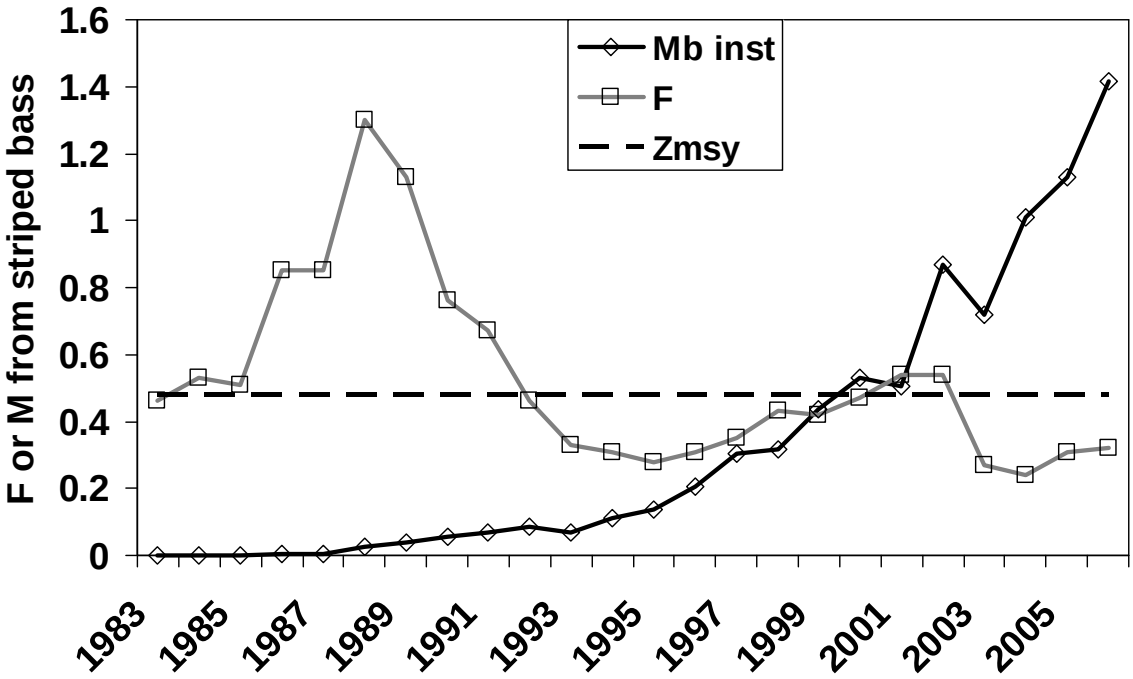


Figure C10.5-13. Estimates of F and M_p and Z_{msy}.

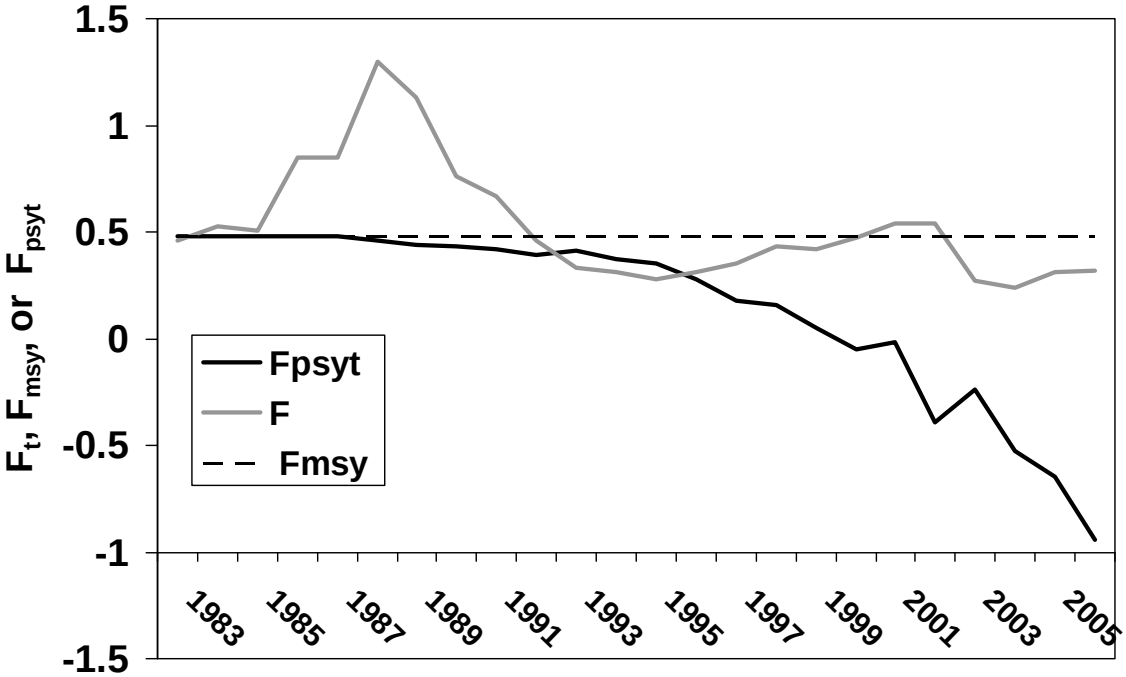


Figure C10.5-14. Estimates of F, equilibrium F_{msy}, or non-equilibrium Fpsyt (= $F_{msy} - M_{pt}$).

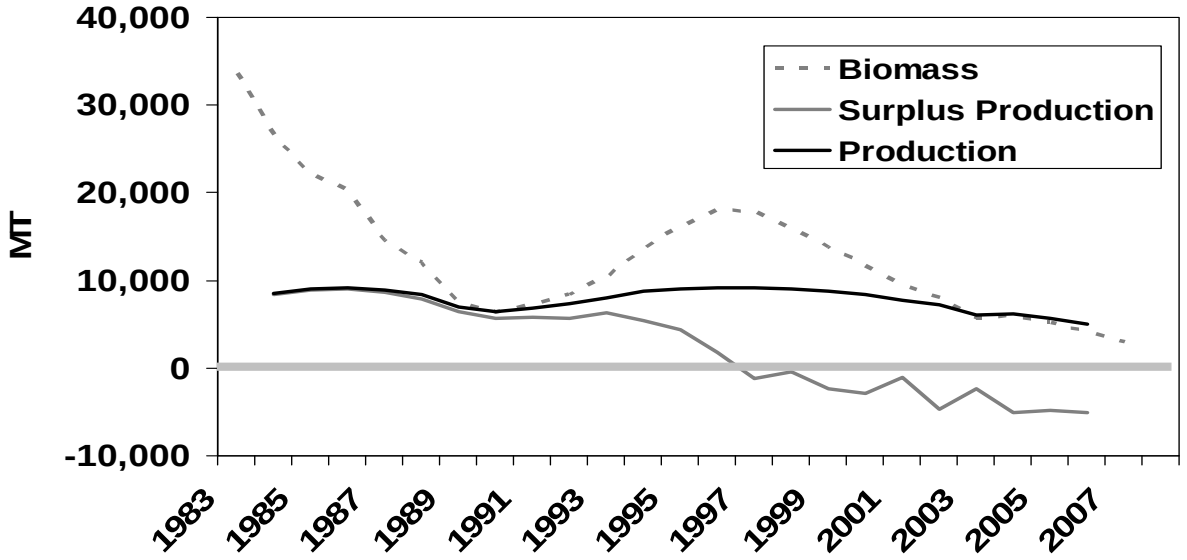


Figure C10.5-15. Biomass, surplus production (less striped bass related losses), and production.

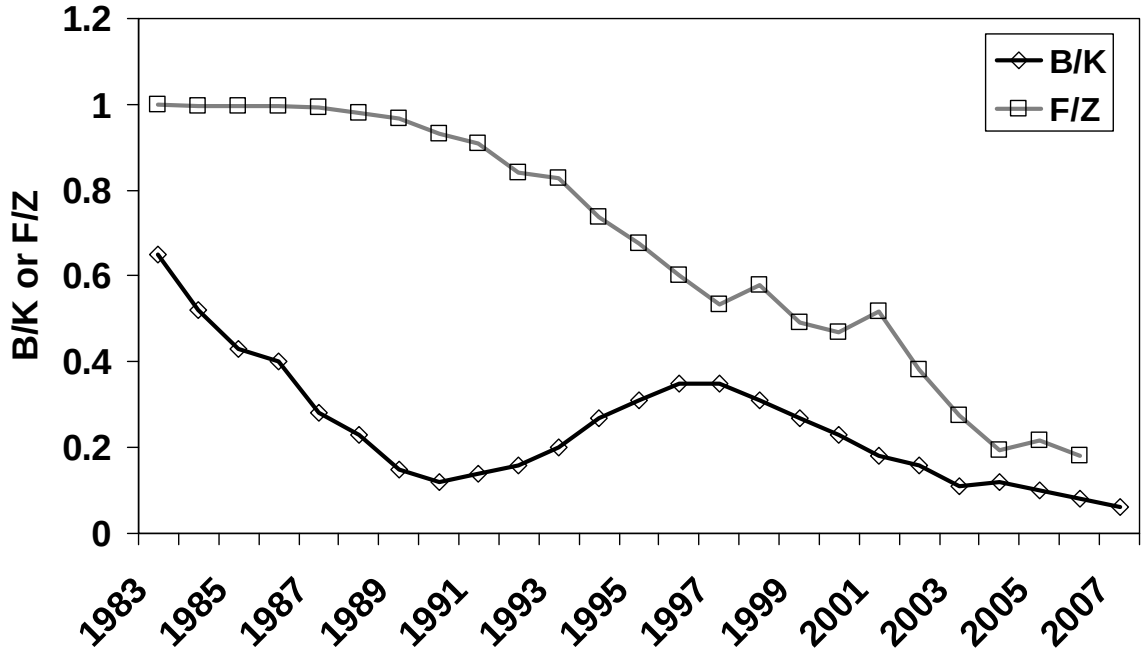


Figure C10.5-16. Ratios of biomass to unfished stock size (B/K) and fishing mortality (F) to total mortality ($Z = F + M_{pt}$).

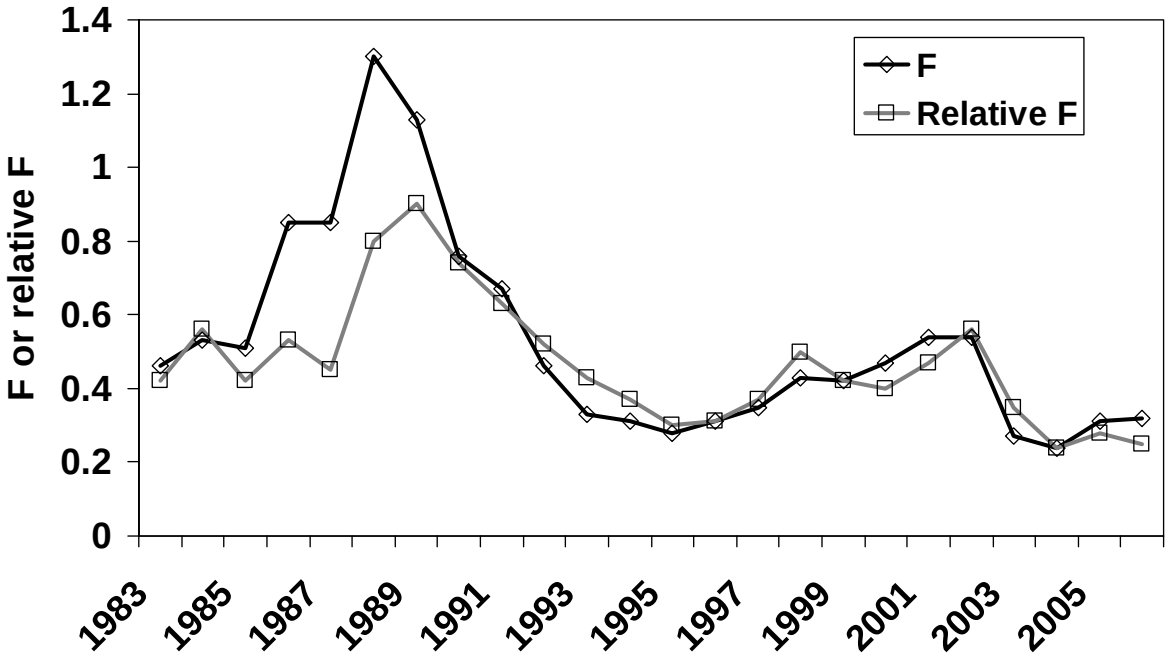


Figure C10.5-17. F estimated by the GDR model and relative F.

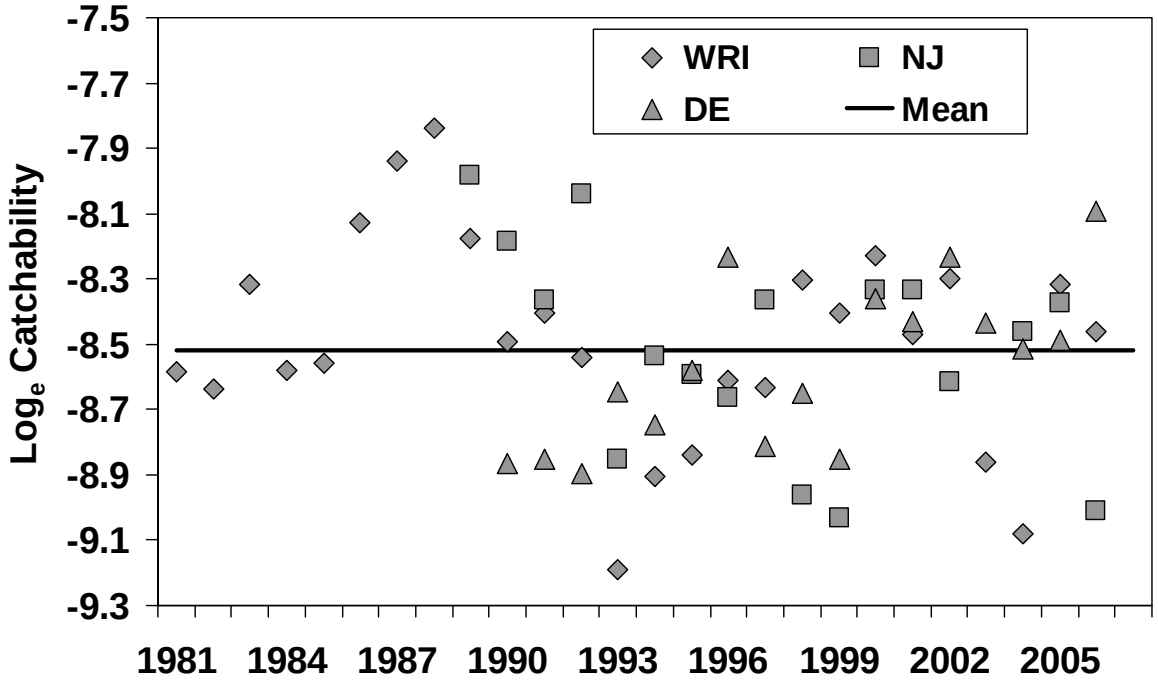


Figure C10.5-18. Trends in $\log_e$-transformed catchability for the three exploitable biomass indices. Mean is for the all indices combined.

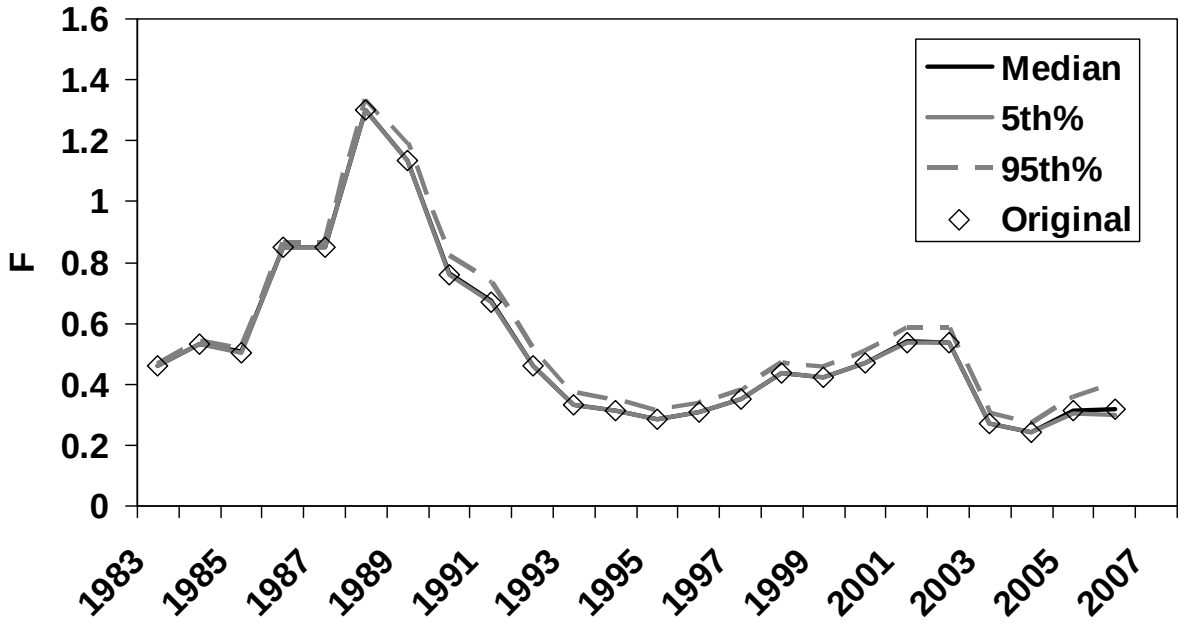


Figure C10.5-19. Jackknife estimates of the 90% confidence interval of F.

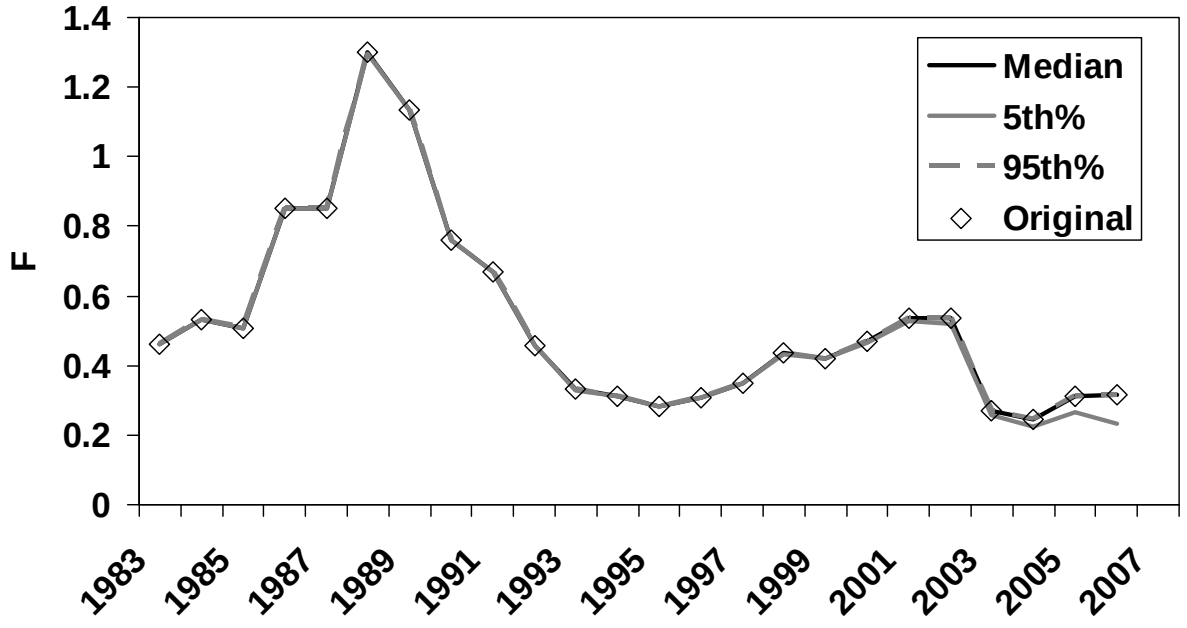


Figure C10.5-20. Bootstrap (N = 500) estimates of the 90% confidence interval of F.

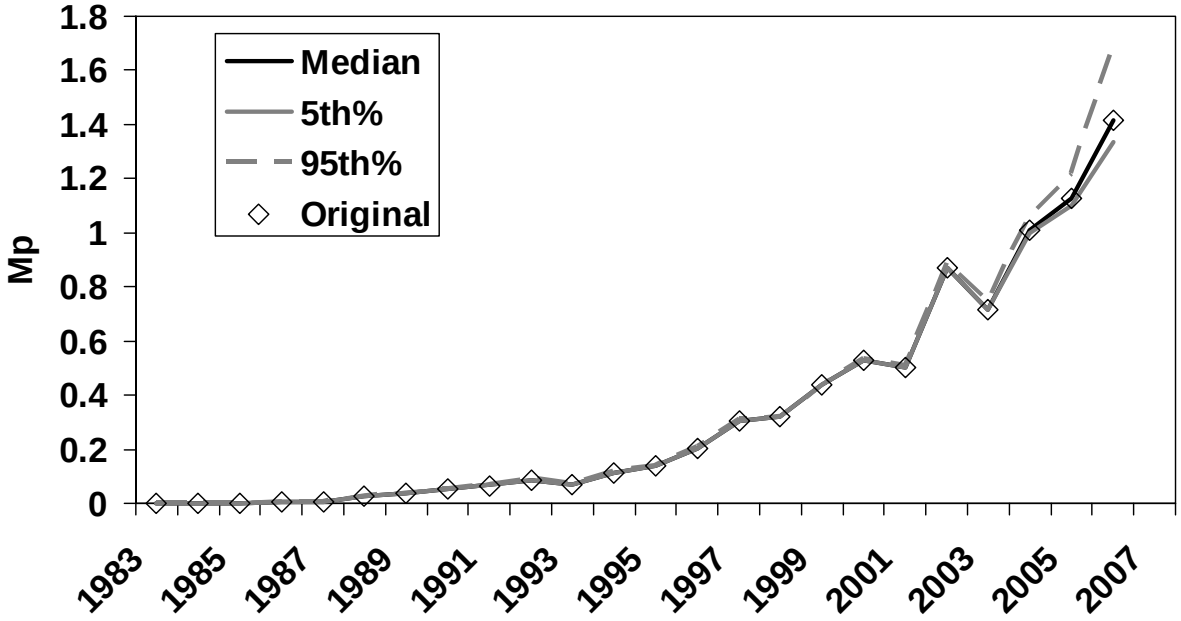


Figure C10.5-21. Jackknife estimates of the 90% confidence interval of M_p.

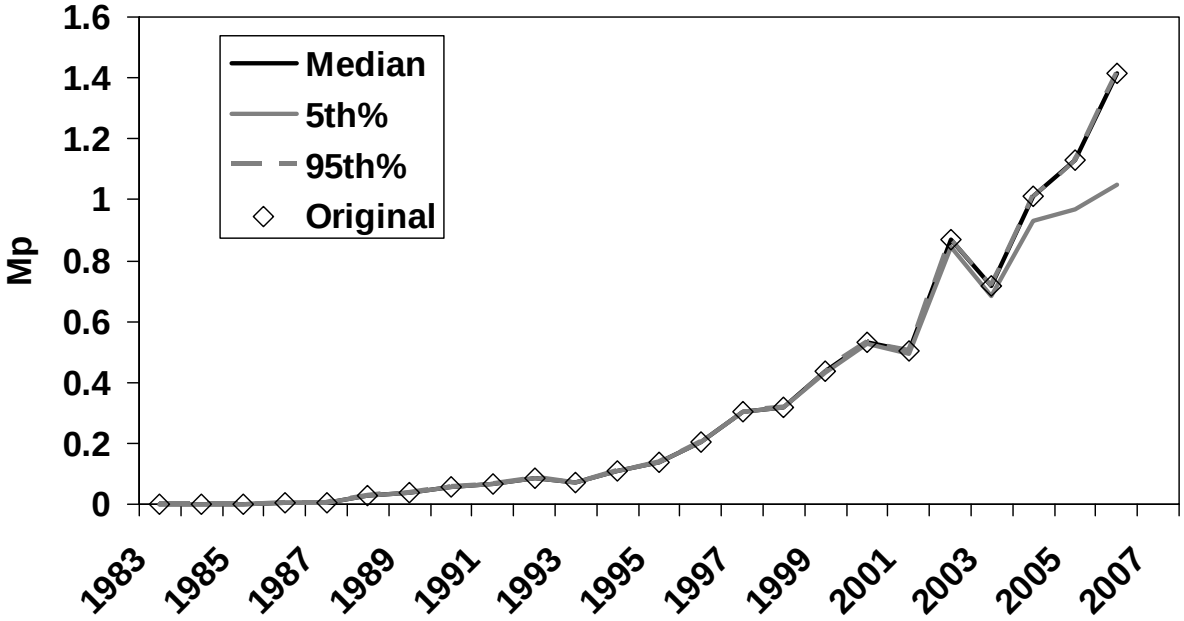


Figure C10.5-22. Bootstrap (N = 500) estimates of the 90% confidence interval of M_p.

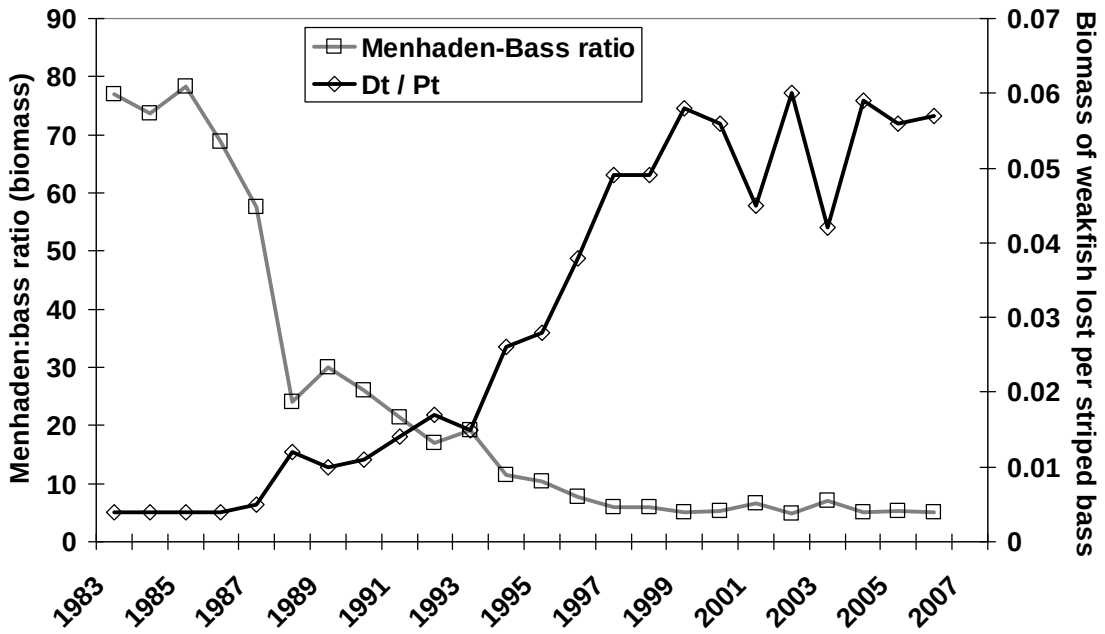


Figure C10.5-23. Ratio of menhaden to striped bass and estimates of weakfish loss per striped bass (D_t / P_t; both based on biomass).

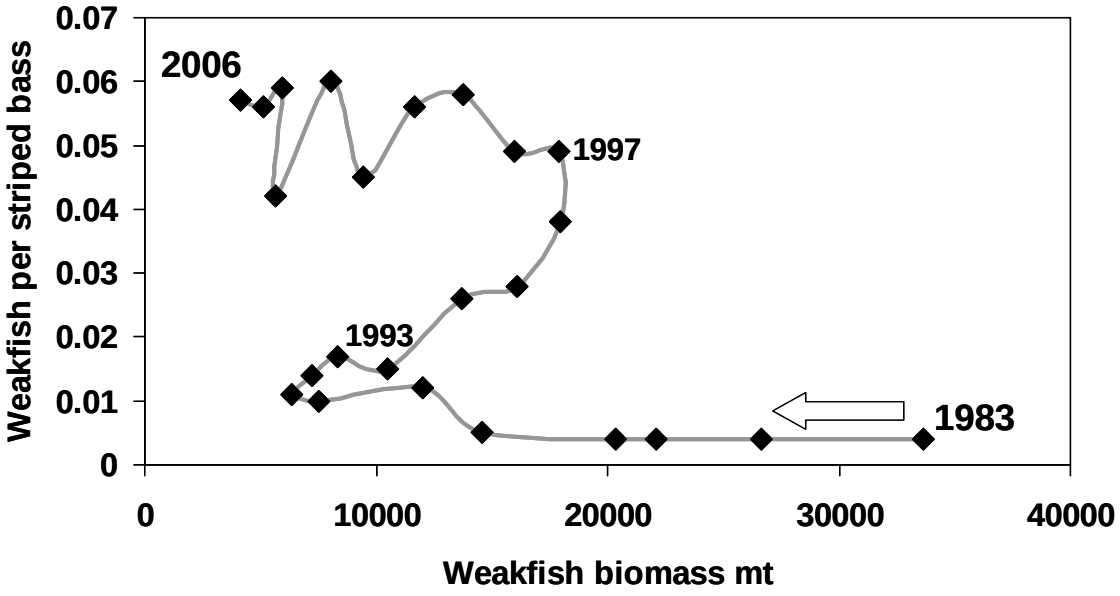


Figure C10.5-24. Weakfish loss per striped bass (biomass) versus weakfish biomass. Note shift in functional response between 1983-1993 and 1997-2006.

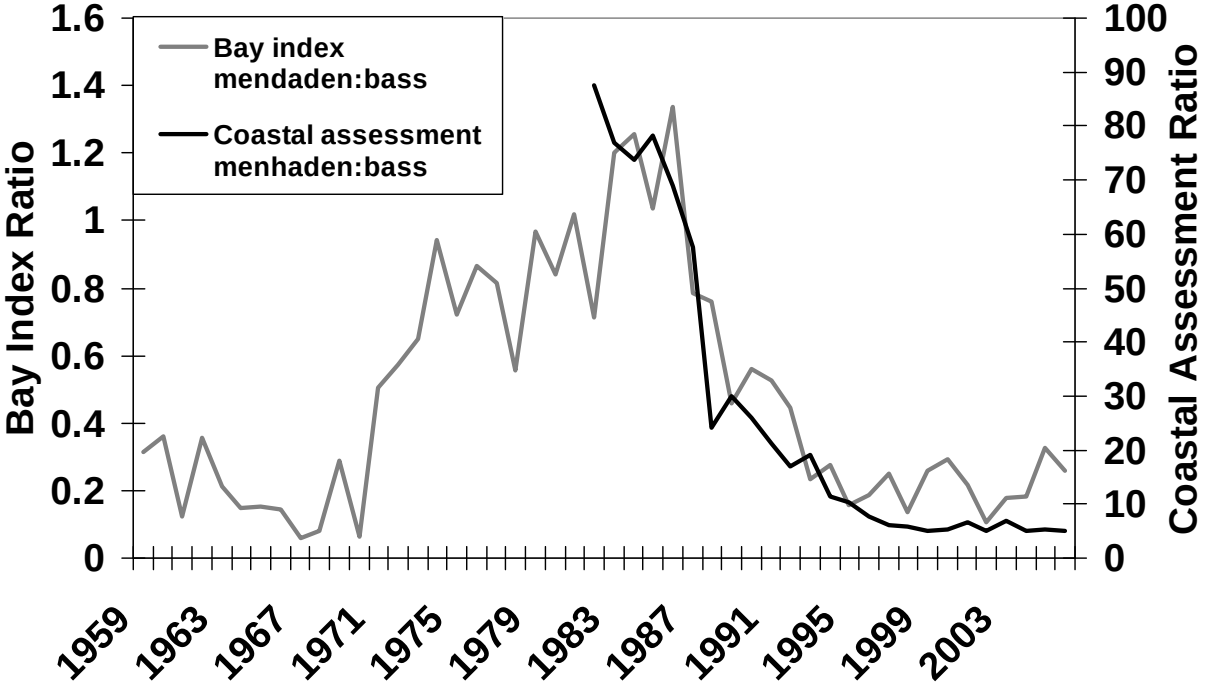


Figure C10.6-1. Ratio of ages 1+ menhaden biomass to ages 2+ striped bass biomass derived from coastal stock assessments and from Chesapeake Bay indices during 1959-2006.

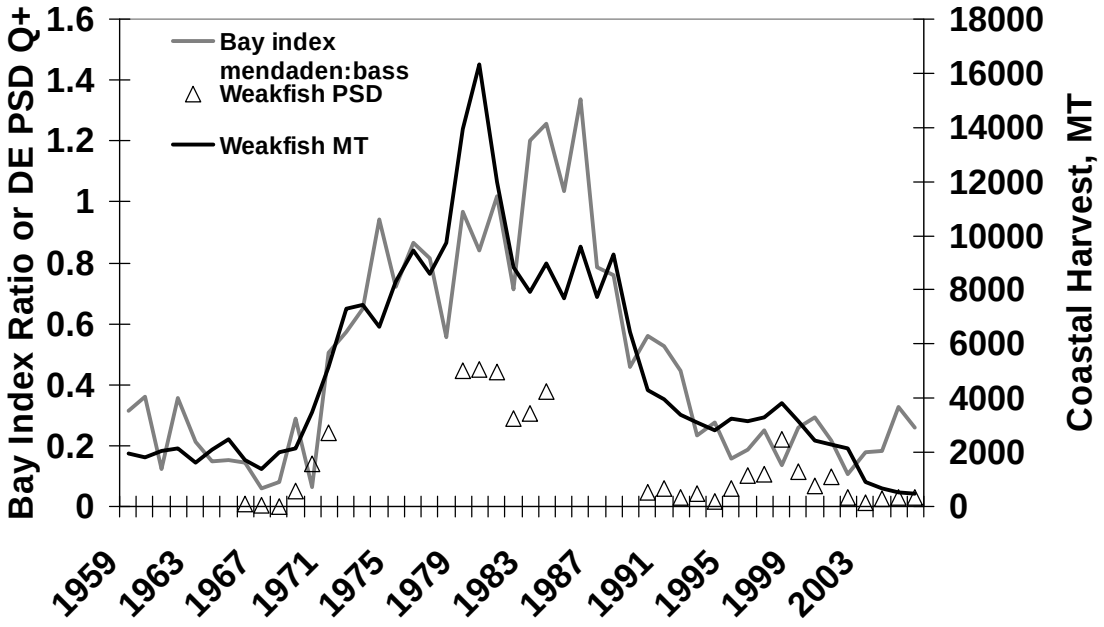


Figure C10.6-2. Chesapeake Bay menhaden to striped bass ratio index, weakfish commercial harvest, and DE PSD Q+ indices during 1959-2006.

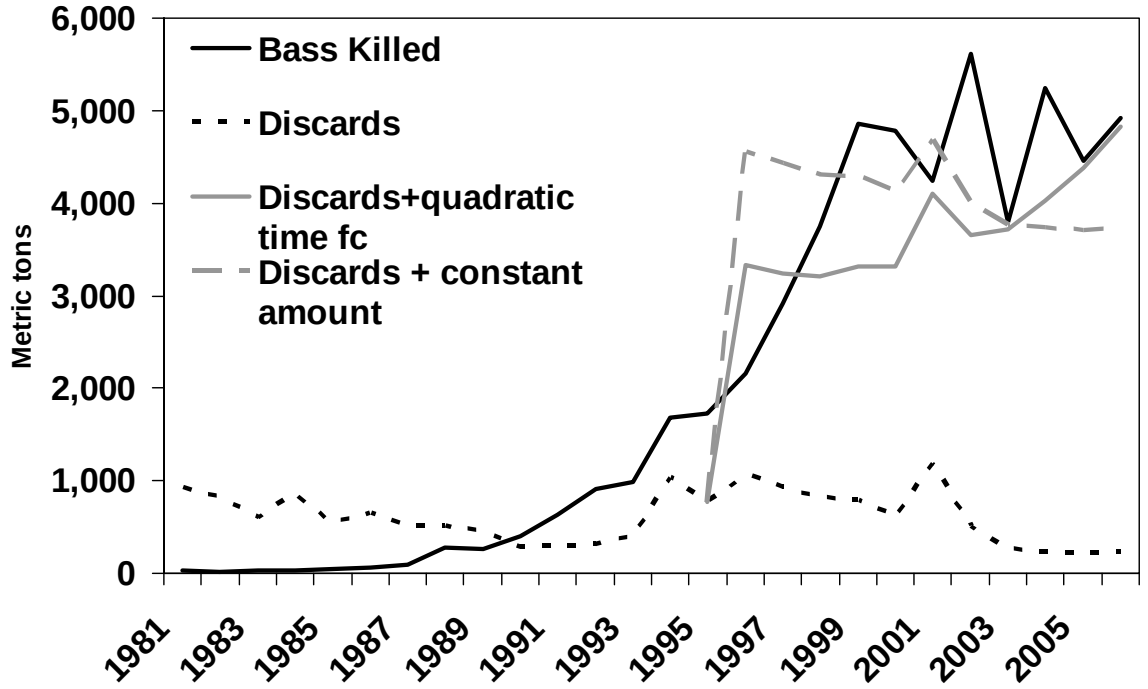


Figure C10.7-1. Comparison of model estimates of weakfish killed by striped bass or as extra bycatch / unreported harvest (starts in 1996) and discard estimates.

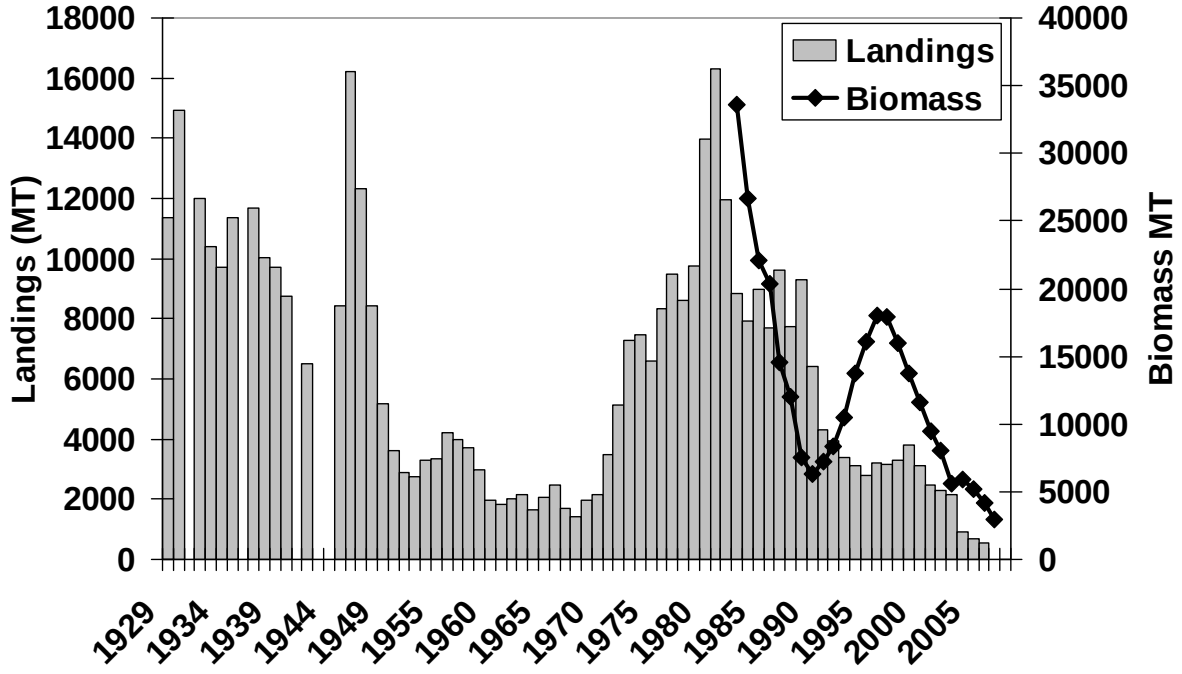


Figure C10.8-1. Commercial landings of weakfish during 1929-2006 and biomass during 1983-2006. Landings for 1929-1950 were estimated from Joseph (1972; Figure 4).

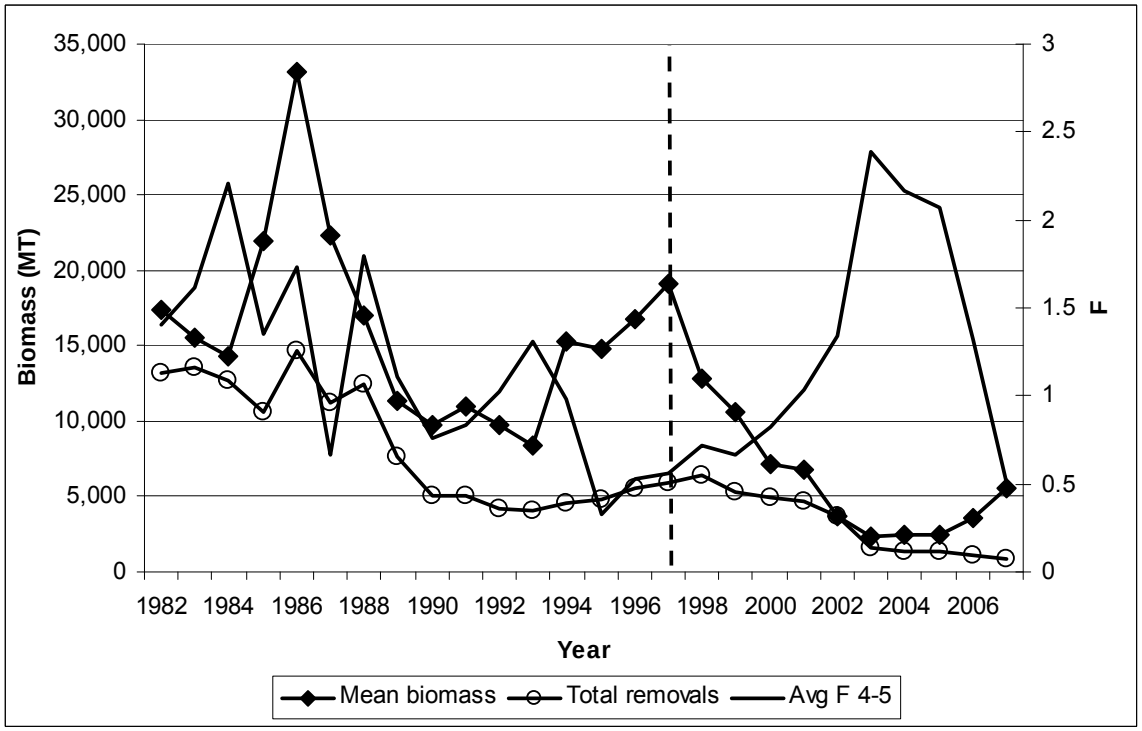


Figure C11.2-1. Relation between mean stock biomass, total removals, and fishing mortality (unweighted average, ages 4-5). The dashed vertical line indicates the year weakfish abundance began to decline in the absence of increased removals.

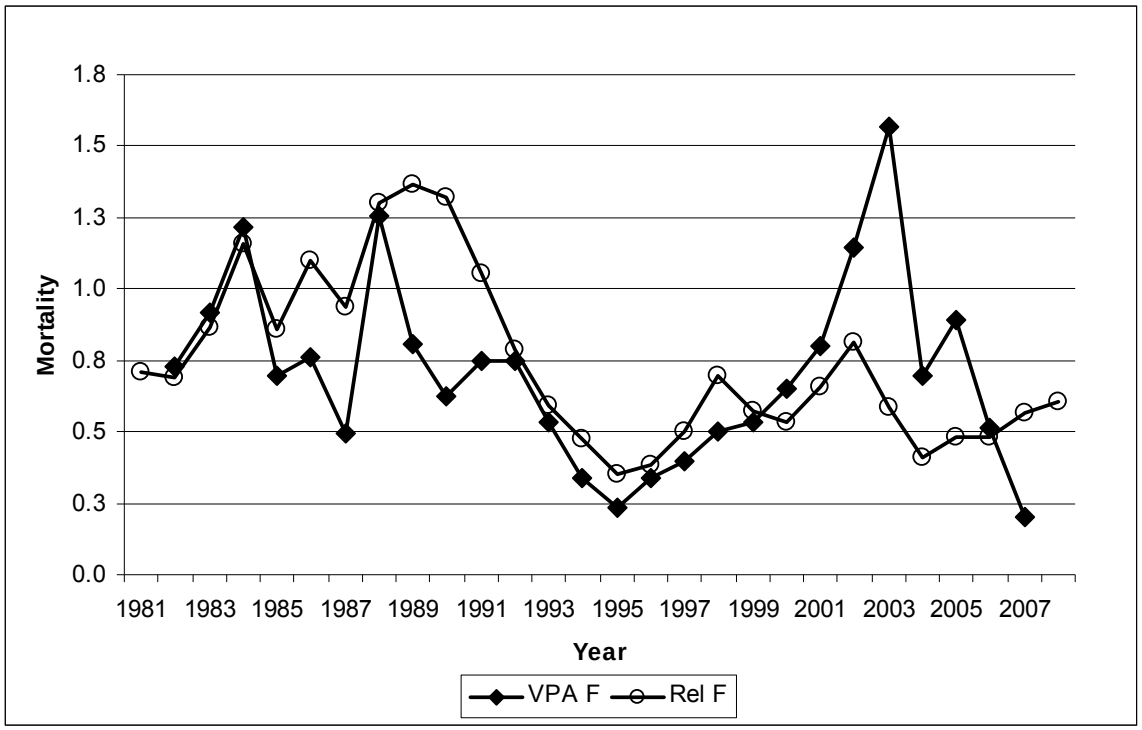


Figure C11.2-2. Comparison of fishing mortality rates calculated by ADAPT VPA and relative F calculations. Fishing mortality expressed as biomass weighted F on ages 1+.

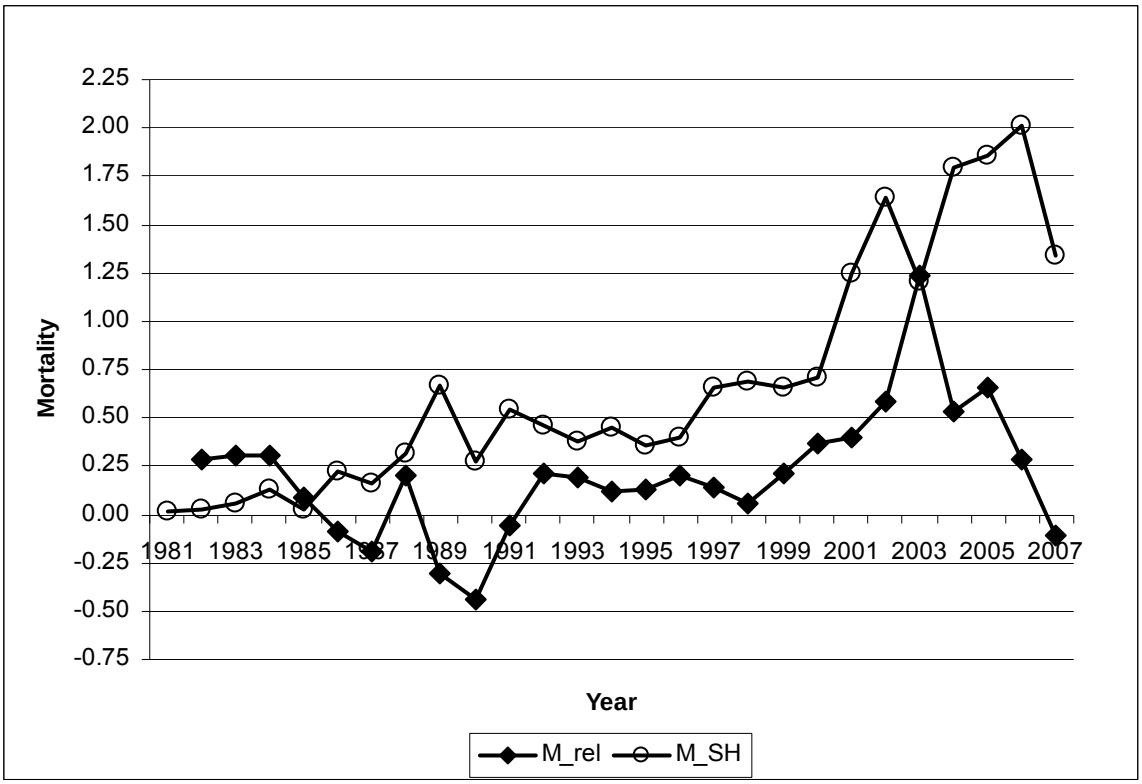
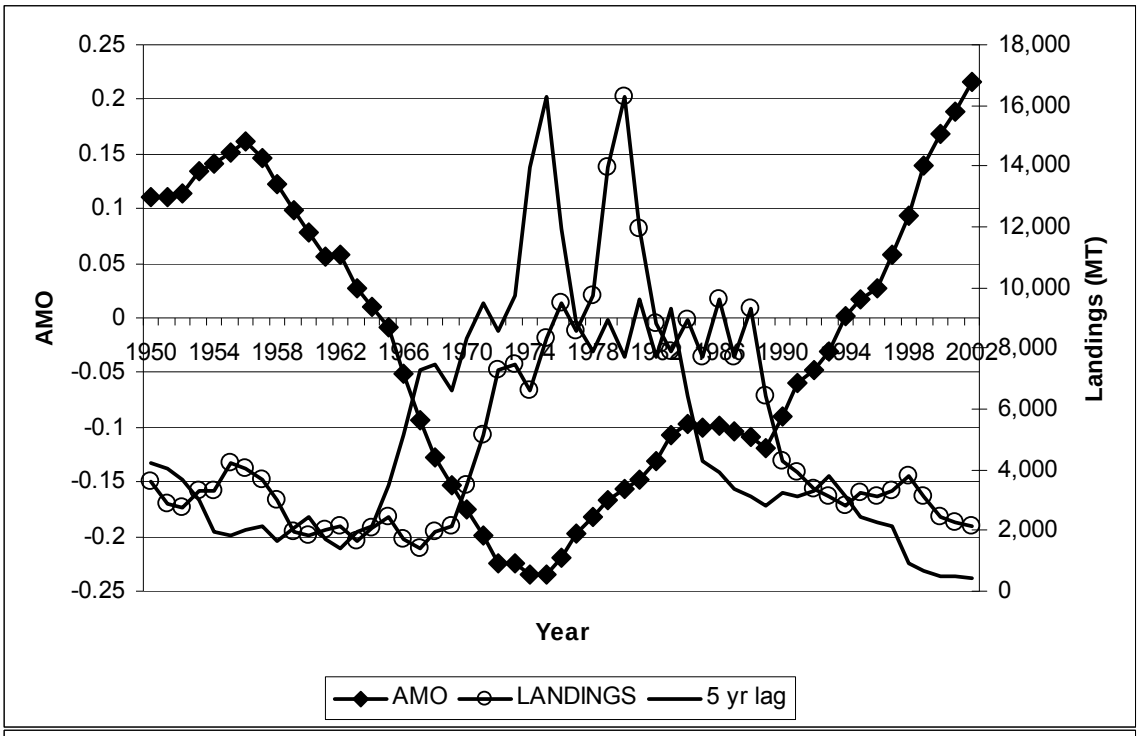


Figure C11.2-3. Comparison of natural mortality as a result of predation from the full Steele-Henderson model (M_SH) with M calculated as $Z_{VPA} - F_{rel}$.

A)



B)

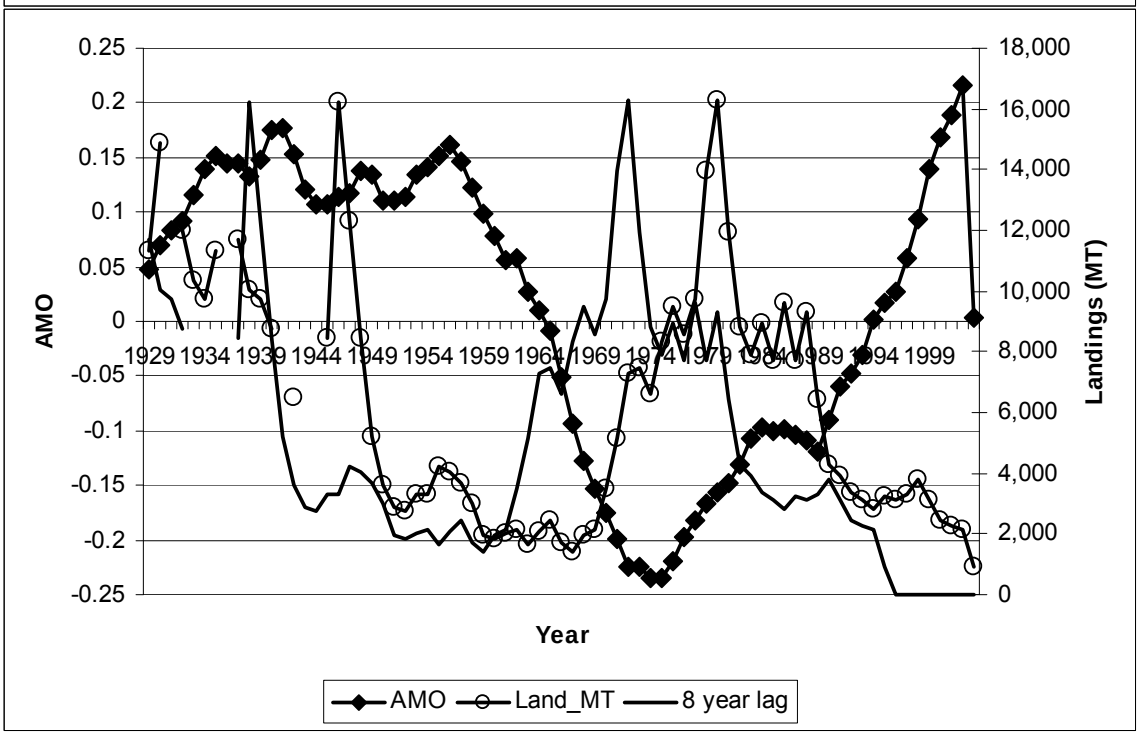


Figure C11.3-1. Relation of Atlantic Multidecadal Oscillation (AMO) to reported weakfish commercial landings. A) NMFS landings only; B) Joseph (1972) plus NMFS landings

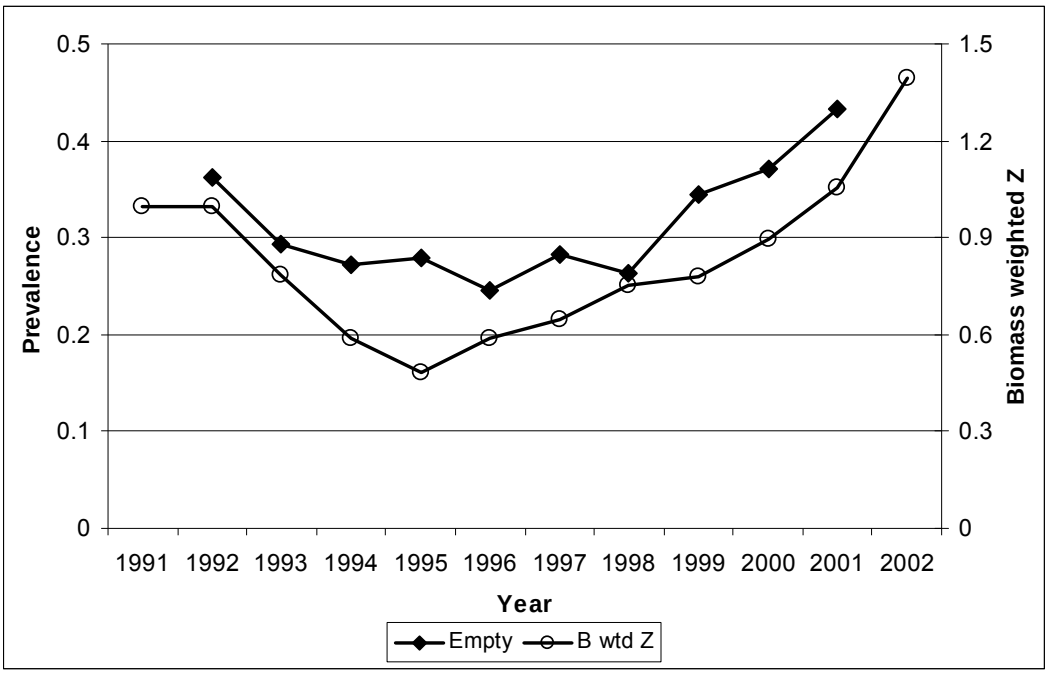


Figure C11.3-2. Relationship between the prevalence of empty stomachs (3 year average) observed in weakfish from the NEFSC food habits database and biomass weighted Z (ages 1-5) estimated by ADAPT.

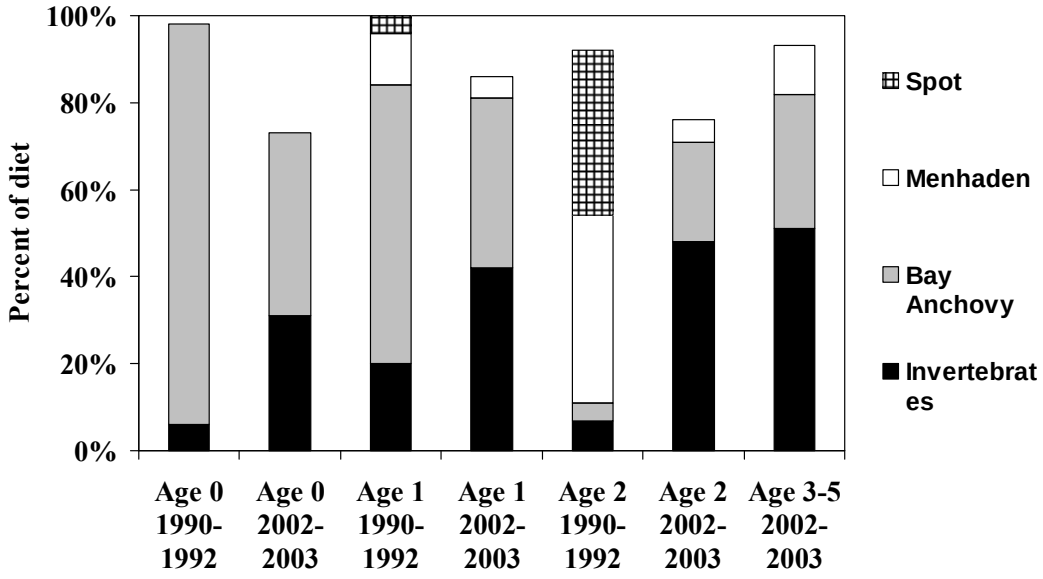


Figure C11.3-3. Comparison of age-specific weakfish diets (by weight) in Chesapeake during 1990-1992 and 2002-2003. Data for 1990-1992 were from Hartman and Brandt (1995) and data for 2002-2003 were provided by R. Latour (VIMS).

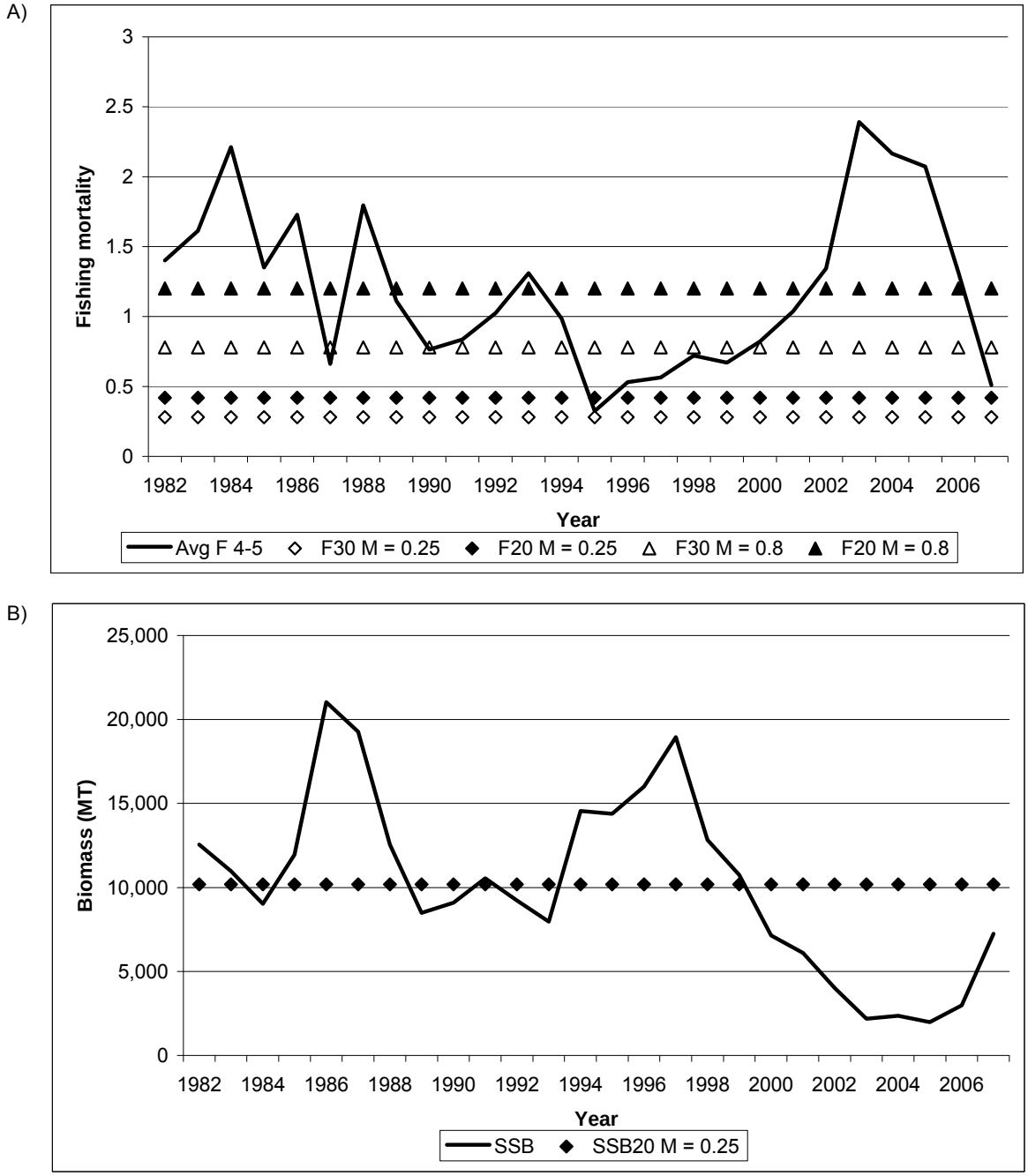


Figure C12.1-1. Weakfish stock status relative to equilibrium Thompson Bell reference points. A) Fishing mortality; B) Spawning stock biomass.

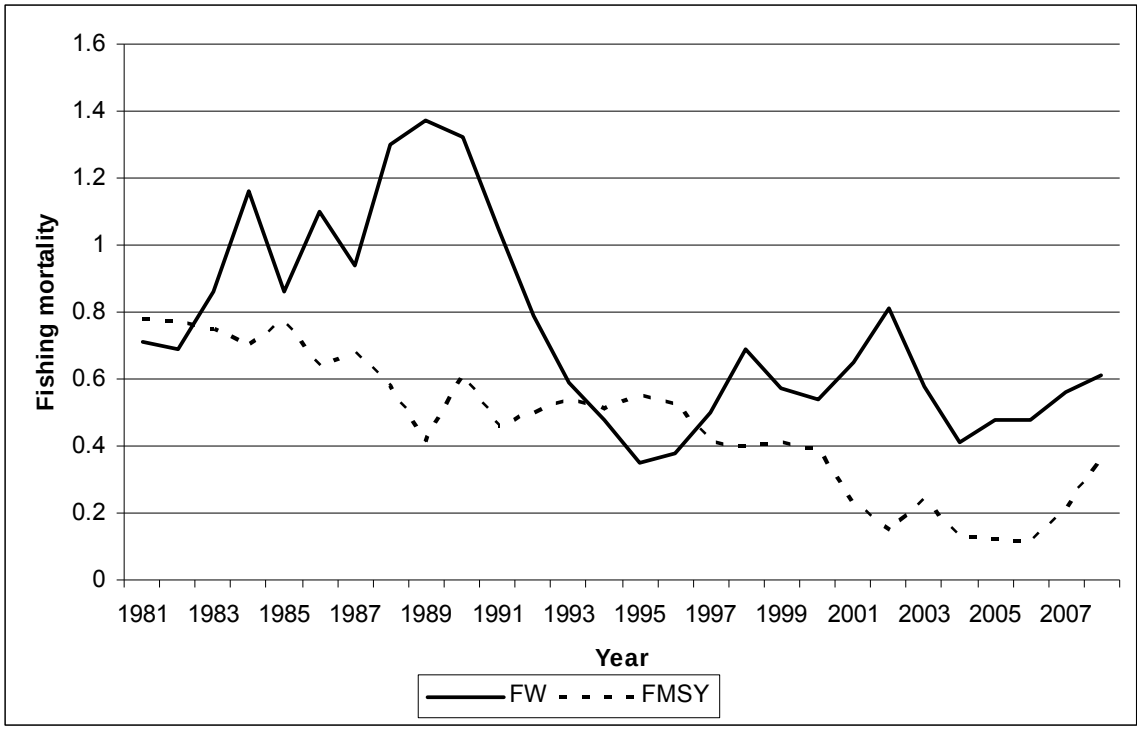


Figure C12.2-1. Weakfish stock status relative to non-equilibrium fishing mortality reference points calculated using the Steele-Henderson model.

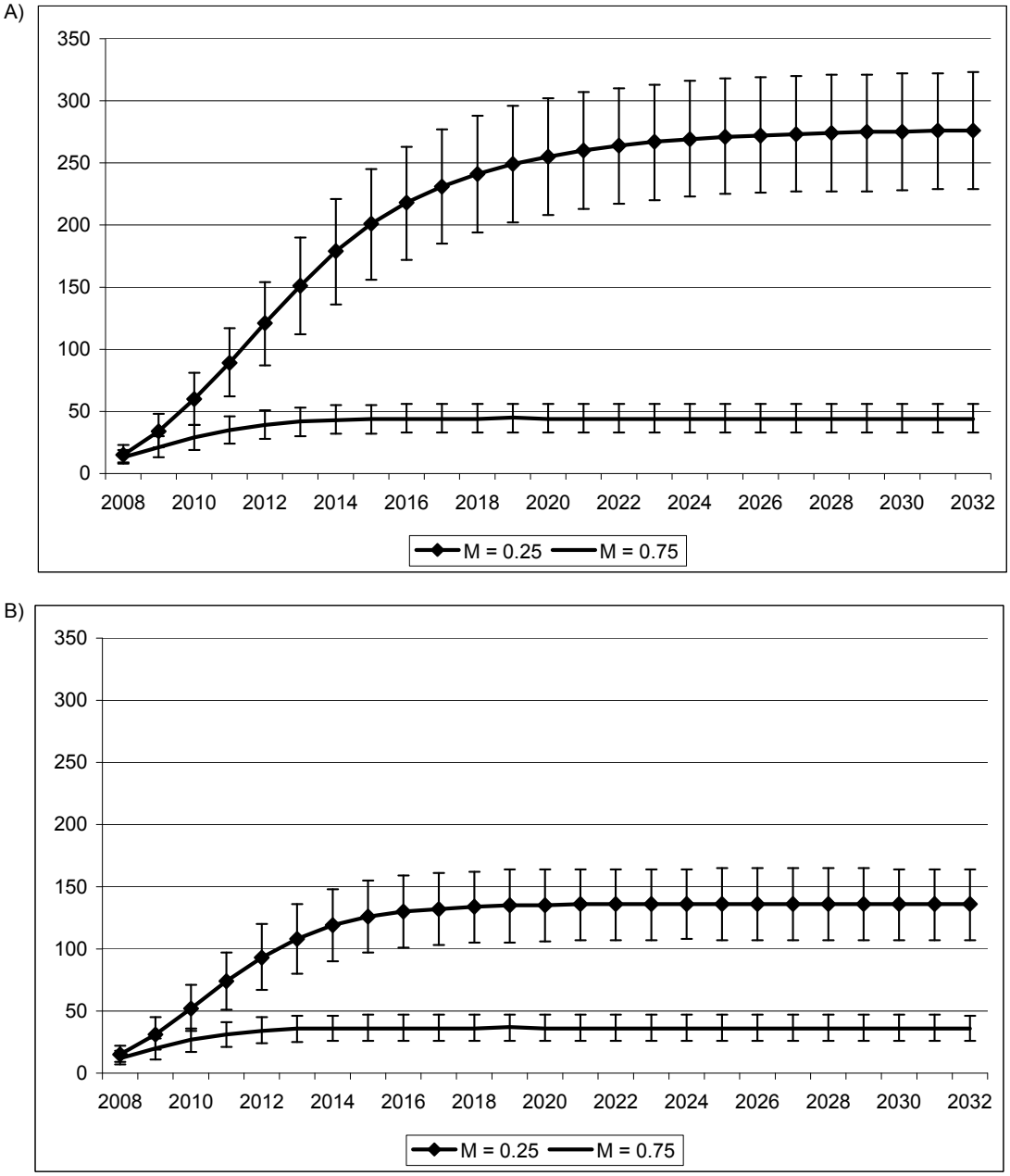


Figure C13.1-1. Biomass projections under varying scenarios of fishing and natural mortality. Projections are calculated relative to ADAPT VPA terminal year estimates. A) Harvest moratorium; B) F = 0.25.

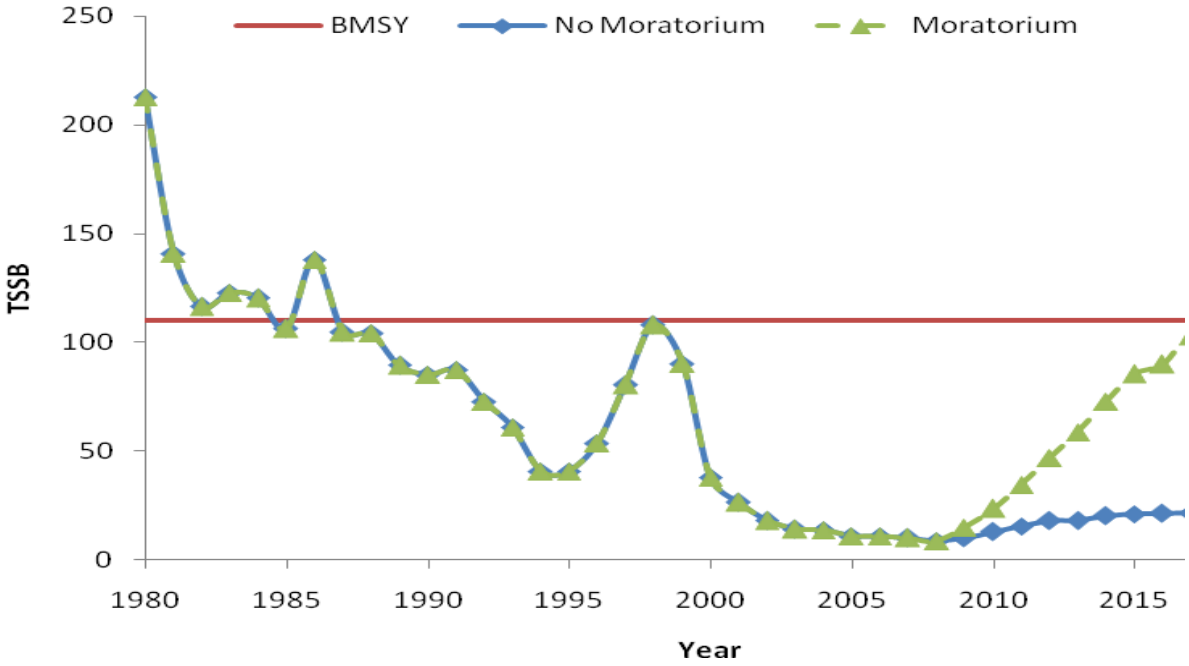


Figure C13.2-1. Effects of a simulated moratorium to harvest (F=0) in 2009 on rebuilding future weakfish TSSB under scenario #1, fixed M=0.25, approximate BMSY is 110.

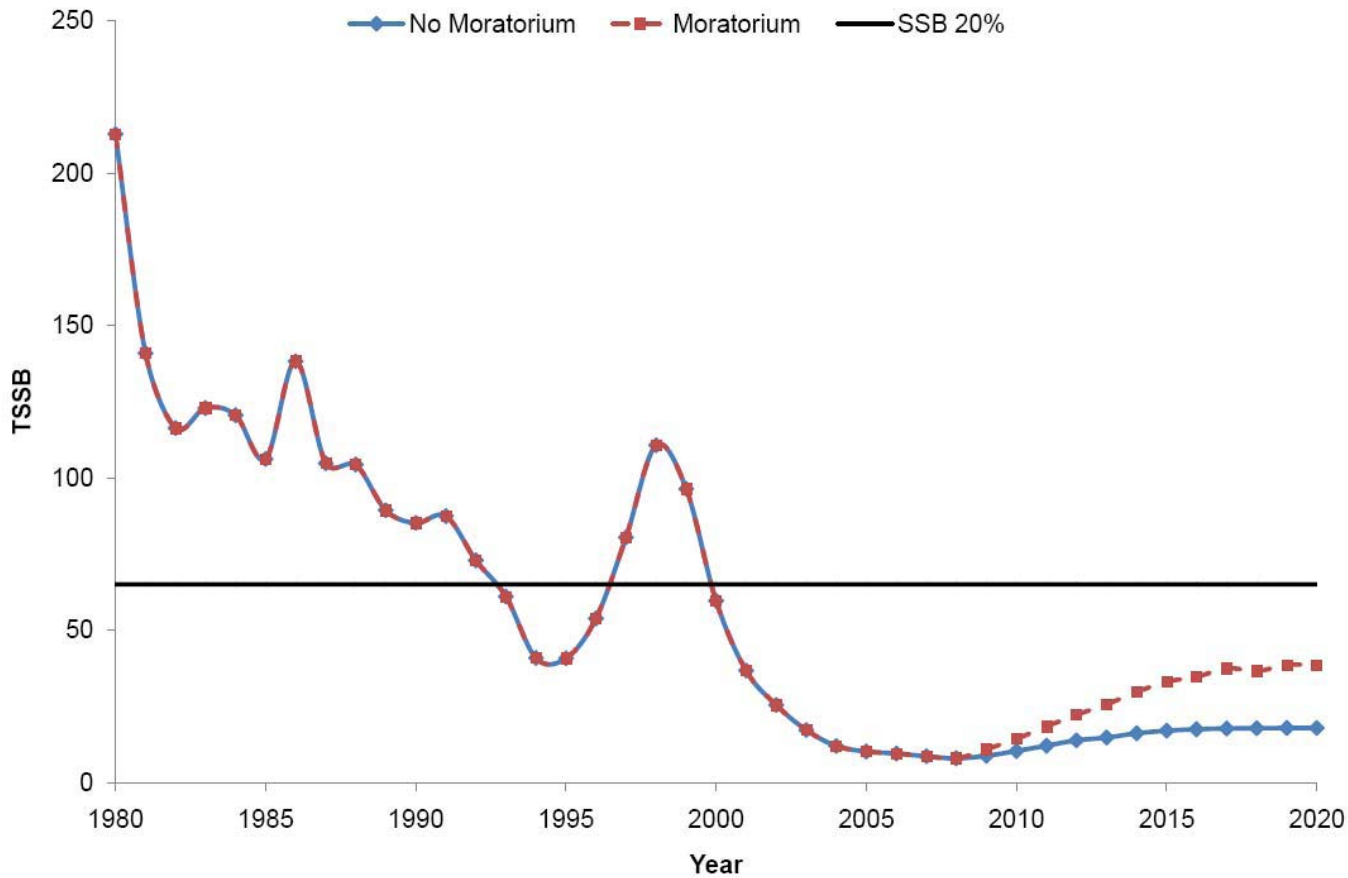


Figure C13.2-2. Effects of a simulated moratorium to harvest (F=0) in 2009 on rebuilding future weakfish TSSB under scenario #2, M rises from 0.25 to 0.65. The projection is based on the assumption that M = 0.25 from 1981-1998, followed by a rise to M=0.65 thereafter. All values in the figure, including SSB20% have been scaled, so they are indicative of relative trends in biomass in relation to SSB20%, and not absolute biomass. SSB20% was estimated assuming constant natural mortality of M = 0.25. Projections were conducted based on results of the Steele-Henderson model described in section C9.0 of the stock assessment report.

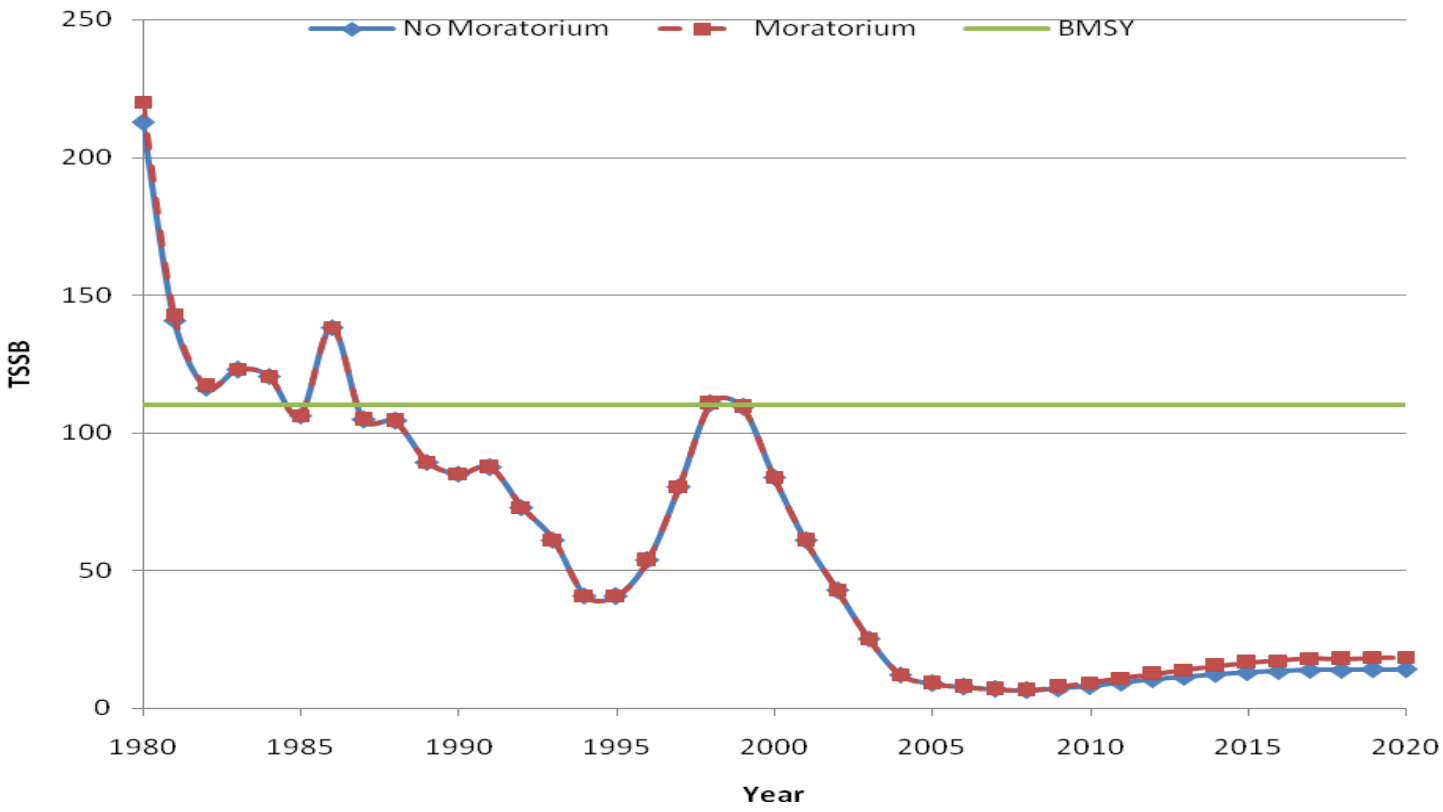


Figure C13.2-3. Effects of a simulated moratorium to harvest (F=0) in 2009 on rebuilding future weakfish TSSB under scenario #3, M rises to 0.25 to 1.0, approximate BMSY is 110.

Appendixes

to the 2009 Weakfish Assessment
C-1 – C-5

SAW/SARC 48
June 1-4, 2009
Woods Hole, MA

Appendix C-1	**Weakfish Technical Committee's response to Data Poor Meeting comments**
Appendix C-2	**Proportional Stock Density Indices for Weakfish**
Appendix C-3	**SAS-based application of the Harvest Control Model to conduct weakfish stock projections.**
Appendix C-4	**Index Standardization**
Appendix C-5	**Preferred Run ADAPT output**

C-1 Appendix

Appendix C-1 was written in February 2009 by the Weakfish Technical Committee (i.e., Assessment Scientists), and is a response to written comments from the Peer Review of the December 2008 NE Data Poor Stock Meeting

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Weakfish portion of the 2008 Data Poor Workshop Report (Miller et al 2009) and the Weakfish Technical Committee's response to the Panel's comments.

SAW/SARC 48
June 1-4, 2009
Woods Hole, MA

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ASMFC Weakfish Technical Committee Report
Weakfish Management Board
February 5, 2009

The Weakfish Stock Assessment was presented to the Data Poor Workshop (DPW) by Jeff Brust, chair of the ASMFC Weakfish Stock Assessment Subcommittee (SAS) in December. In preparation for the meeting, the Review Panel (Panel) was provided with access to a range of working papers (ASMFC Weakfish TC 2008a, b, c, Uphoff 2008) that outlined the approach taken in several key aspects of the assessment. The Panel, although aware of past assessment methodologies and information most likely did not have sufficient time to review those materials. Weakfish was allotted close to four hours of time for review.

The Weakfish Technical Committee (TC) and SAS held a conference call on January 22, 2009 to discuss the DPW report which was released on January 20th. Overall, the TC/SAS believes the DPW reviewers to be too general in their comments without specific recommendations regarding how to improve the current assessment. Many of their "concerns" have been thoroughly vetted through past assessments as well as during current analysis. Also, the time frame required to perform a proper review was lacking in this case, including both for SAS Chair and the Panel. This did not allow the SAS Chair to discuss previous assessments which could have answered many of the reviewers concerns and thus in the end may not have been concerns at all. The following comments, **in bold**, represent the consensus view of the TC/SAS on the DPW process.

3.1 Background

The stock assessment for weakfish (*Cynoscion regalis*) that is being conducted by the Atlantic States Marine Fisheries Commission's technical committee (ASMFC - TC) was presented to the Panel by Mr. Jeff Brust, chair of the ASMFC weakfish TC on the afternoon of Thursday December 11th. The discussion on this first afternoon focused on the application of an age structured virtual population analysis to the weakfish stock. The remainder of the presentation, which focused on biomass dynamic models of weakfish that include covariates, was given on Friday morning.

In preparation for the meeting, the review Panel was provided with access to a range of working papers (ASMFC Weakfish TC 2008a, b, c, Uphoff 2008) that outlined the approach taken in several key aspects of the assessment.

The Panel did not have time to provide a full and careful consideration of all elements of the assessment including the quality of all data inputs and the appropriateness of the inferences drawn. Thus, the comments that follow should not be considered as representing a detailed peer review of the weakfish assessment. However, the Panel considered that it had adequate time to provide some general overview comments which we hope will be of help to the ASMFC in providing guidance to the weakfish TC as it seeks to complete its assessment. Discussions between Panel members and the TC chair were open and cordial.

- **The SAS Chair did not have sufficient time to fully elaborate on the data sources and analytical methods of the stock assessment. Many of the Panel's comments are based on insufficient information due to lack of time.**
- **There was no review of previous assessments and methodologies which could have assisted the Panel during their deliberations.**
- **The DPW review should have been conducted as a brief review of highly condensed data and methodologies and not a hyper critical review trashing all of the data.**

- **The TC/SAS did not ask for a general overview from the DPW. There were specific questions, in terms of assistance, which the Panel never answered.**
- **The whole DPW process cost the SAS a lot in lost time and didn't return much advice. The Board should be concerned about lost time of their state's employees.**
- **Due to time constraints, the Panel saw a lot of what the last peer review saw. It is not too surprising that the reviews are similar. It would have been useful to have had something reflecting more of the new work rather than the old.**

3.2. Virtual Population Analysis

At their core stock assessments examine the consequences of observations under a suite of assumptions to explain the dynamics of the stock. Thus, it is critical that the assessment team be confident of the observations entering the assessment model. Errors and uncertainties in the observations on which the assessment is based can lead to spurious patterns in the inferred dynamics that may not be reflective of the true underlying dynamics.

- **Not only should the TC/SAS be confident of the observations, it is also important that the TC/SAS is confident in the assumptions on which the model is constructed. One of the main concerns the TC has with VPA is the assumption of constant M. This is not a new concern with this committee, nor is it constrained to just weakfish. Members of the TC have been opposed to the use of this assumption for years, if not decades, for weakfish as well as striped bass, lobster, and others. As noted above, if the data your model is based upon are not valid, results of the model could be suspect. This extends to the assumptions the model is based upon as well. Therefore, errors and uncertainties in the assumptions on which the assessment is based can lead to spurious patterns in the inferred dynamics that may not be reflective of the true underlying dynamics. Unfortunately, the assumptions rarely receive the same level of scrutiny as the data.**

The Panel expressed serious concerns over the reliability of input data used in the weakfish Virtual Population Analysis (VPA). The Panel concluded that until apparent inconsistencies in the input data are more fully explored, the TC's conclusion that the lack of fit of the VPA to the observations is due solely to an increasing natural mortality (M) rate is premature. The Panel recognizes that increasing M could be a possibility. This has been observed in other stocks at low population sizes (e.g. northwest Atlantic Cod) where predator – prey dynamics can maintain prey at low levels of abundance. However, before concluding that M is increasing, it is essential that the TC fully address the data input issues. The Panel does not consider that the VPA results are indicative of a pattern of increasing M to the exclusion of other plausible explanations.

- **The data used for the current run of the VPA have changed little since the first peer reviewed weakfish assessment. In fact, data have gotten better – otolith based ages, expanded commercial sampling, fishery specific CAA. The VPA model passed one peer review (the 30th SARC) with no problems, yet now the data are no longer valid in this Panel's perspective. The results of past VPAs were used for management with the apparent positive result of a partial stock rebuilding. If the data are as bad as this review suggests, can we believe any of the trends and management results from previous assessments? (One would think not.) On the other hand, if the past**

assessments are valid, then why are they no longer valid when the data have actually improved?

- **The TC/SASC has never said that the lack of fit of the VPA is due solely to an increasing natural mortality (M). It has been documented that a rise in M helps explain why the VPA shows a rise in total mortality as suggested by the VPA.**
- **If the Panel disagrees with a dynamic M, why didn't the Panel recommend using only a constant M? Could it be that they realize the outputs would not show a clear picture of what is actually happening with the weakfish stock.**
- **The TC/SAS believes it has fully addressed all data input "issues" to the best of their ability. These are the same issues that affect all assessments and are not solely restricted to the weakfish assessment.**
- **If there were other plausible explanations that the Panel is aware of and the TC/SAS has not attempted to analyze, why do they not share them with us at this time?**
- **Would it be useful to the Board if the TC/SAS were to provide a table that lists or provides examples of other assessments that use the same types of data or data grinding processes as the weakfish TC/SAS? For instance, which assessments use NMFS landings (all), MRFSS estimates (all), bycatch from observer data (?), MRFSS index (At least three other ASMFC assessments feature MRFSS indices: Atlantic croaker, striped bass and bluefish), regional indices (all assessments are likely to).**

The concerns noted by the Panel centered on the following issues:

a) Reliability of catch information: While the Panel did not have sufficient time to examine the catch records in detail, there was some suggestion from the presentation that catches in some fisheries may have been underestimated substantially. For example, the TC chair and the Panel discussed uncertainties in the NC landings, particularly with regard to allocation to different gear types. It is important that not only the total catch is known, but that it is allocated accurately to the different sectors given the different biological catch characteristics in those sectors.

- **There seems to be no factual basis for concern of the catch information. Harvest weight is taken from NMFS and/or state weigh-out sources as with any other assessed species. The TC/SAS recognizes there is some uncertainty with all commercial and recreational landings estimates. Regardless, they are commonly used throughout ASMFC assessment work and should be of no concern to the Panel.**
- **The issue of uncertainties of the North Carolina landings seems unwarranted according to their sampling methodologies. The TC/SAS believes the sampling of North Carolina's commercial fisheries to be exceptional.**
- **Patrick Campfield from ASMFC has provided analysis in the past on potential bias in NMFS landings versus individual state weigh-outs.**
- **Trends in recreational and commercial landings follow similar patterns, even though they are determined using very different methods.**
- **What specific information was presented that suggested the catch was underestimated? The Panel does not provide sufficient information for us to fully evaluate this concern.**

b) Expansion of discard estimates based on catch per haul of targeted species on observed vessels to total discard for the fleet is likely biased: Related to the concerns expressed over the reliability of the

catch data, similar concerns were expressed over the reliability of the discard data. The Panel suggested alternative approaches to the TC chair that might ameliorate these concerns.

- **As with the harvest information, it appears that the concern of bias in discard estimates is more speculative than factual. The DPW Panel does not seem to be aware of Janaka de Silva's 2004 paper where he investigated a variety of discard estimation methodologies, with the current method selected as the most reliable estimates of discards. He performed similar analysis during the last assessment of Atlantic croaker, an approved ASMFC assessment.**
- **It is possible for the SAS to:**
- **Provide a table of the number of hauls or trips sampled by year/gear/target species to show low number of instances of weakfish discards at this fine of analysis.**
- **Summarize findings of regression method, trip based method, etc. Conduct sensitivity runs of VPA.**
- **Reference and summarize Jim Uphoff's report that estimates amount of "unaccounted catch" necessary to fit trend. The estimates of discards would have to be enormous (15 to 20 times higher than the original estimate in recent years).**
- **Are there any other species with better, or even different, estimate of discards? Doubtful, the TC/SAS believes these estimates to be the best available.**
- **Reviewers could not agree on direction of bias in estimates, so where does that leave us?**
- **The Panel should have provided the alternative approaches mentioned in this section.**

c) Reliability of catch at age information: The catch and discard tonnage are partitioned in the catch at age matrix. The key assumption of the VPA is that the catch at age is known with no or negligible error. For weakfish, catch at age is not fully described and estimates from one region and one sector have to be applied to other regions and sectors to provide a full catch at age matrix. The Panel concluded that the catch at matrix is of unknown precision.

- **The TC/SAS recognizes that substitution affects precision and agrees that there may be errors in the CAA matrix. However, weakfish is not the only species where substitutions are made to develop catch estimates (e.g. bluefish). Substitutions were made in the past and passed the 30th SARC. Issues such as this require a great deal of review of past techniques. The SAS has already completed some work in this area including comparison of NJ CAA with substitutions of different data. Have other species' assessments utilized similar methodology of applying estimates from one sector to another when necessary? Did these estimates pass peer review?**
- **The SAS may also, time permitting, attempt to develop a 1993+ VPA to eliminate scale-otolith conversion concerns.**
- **The data concerns are minimal in recent years and continue to improve with the addition of commercial ages from NJ and NY.**
- **One option is to justify not using VPA (or any age structured assessment) until have sufficient number of years with better sampling from NJ north (started in 2004, so ignore VPA until 2014?).**
- **Try SCAM or other age based assessment modeling? Would have been completed but too much time wasted on the DPW.**

d) Spatial and temporal coverage of the indices: Although the VPA could have used more than 40 separate indices, many were found to be inappropriate by the TC for several valid reasons. However, the fishery-independent indices that were selected did not cover the entire population area, but rather were restricted to limited spatial areas within the overall weakfish stock area. Such indices may not reflective of the entire population. If such indices are used, the implicit assumption is that each index represents a constant proportion of the overall population across the entire time series. When this assumption is not met, the overall results of the assessment are likely not reliable. While the TC spent considerable effort selecting those indices whose aggregate trends were comparable, the Panel remained concerned that these indices may have been coherent because they contained little information, rather than because they are reliable indices of population abundance.

- **There are 44 indices available, including five (5) fishery dependent (MRFSS 3-6+ and 2+) as well as 12 from NEFSC (1-6+) and SEAMAP (1-6+) that were omitted. The remainder, including nine (9) young-of-year, the NJ (6), DE (6), and NC (6) indices are all localized indices. The panel apparently recognized that the NEFSC survey (the only coastwide FI index) was inappropriate, so there is no "coastwide survey" besides the MRFSS CPUE index (see (e) below). The TC/SAS does believe that the indices used cover the core area as suggested from the SAW 40 review. Also, NC has the majority of commercial landings, NJ the majority of recreational landings, and DE is a major spawning area. These three State's have the only aged FI surveys within core area and all show the same trend.**
- **The trends in FI indices are similar to trends in MRFSS and commercial CPUE.**
- **The Panel says these indices may be coherent because they contain little information. Once again there is no factual evidence that would warrant such a determination from the Panel. Only 2 aged indices (NEFSC and SEAMAP) showed a different trend, and their removal from the analysis was justified and approved by the Panel. Also, the trends observed in these indices are also observed in commercial and recreational harvest trends and commercial CPUE. Are all these data sources uninformative? Does the Panel suggest there is better data out there somewhere? If we cannot use these data, we have no data left to perform an assessment.**
- **The assessed trends of indices are too directional to be uninformative. There isn't that much inter-annual variation in recruited ages as we have them configured. The MRFSS index suggests that in the long-term they may be more of a one-way trip. Basically, they are a steady decline with a blip at recovery. However a recent paper by Magnusson and Hilborn (Magnusson, A. and R. Hilborn. 2007. What makes fisheries data informative? Fish and Fisheries. 8:337-358) found that one-way trip data was surprisingly informative.**
- **Dr. Yan Jiao's work of standardizing the indices may assist in proving the worth of the indices. Another way of measuring the indices is to compare them against the converged portion of VPA.**
- **DE and NJ are at the center of the core area and both indices correlate with FD indices.**

e) MRFSS CPUE index: The use of a MRFSS index is not inherently inappropriate and the assessment team appeared to be aware of potential issues in the use of such indices. However, the Panel noted particular concerns given that the MRFSS index was one of the few that exhibited any

clear signal or contrast. When such indices dominate the input data set, these concerns become magnified. The Panel was appreciative of the efforts by the TC that have been made since the previous assessment to improve the index but still had concerns over the reliability of this index. For example, the index could have declined because anglers switched the rigging of tackle used to favor striped bass. The MRFSS weakfish CPUE would be expected to decline for this reason alone, particularly as all private and party boat trips were used as the index of effort. The Panel could not suggest a better estimator of effort for use in the calculations given the time available. The Panel remained concerned over the reliability of this fishery-dependent index, particularly given its pivotal role in the VPA.

- **The MRFSS index was one of four (4) indices that showed similar signals, along with other indicators as mentioned above. To be fair, commercial CPUE was not covered during DPW. Only two "aged" indices (NEFSC and SEAMAP) did not show this trend.**
- **Once again there is no recommendation on how to make the MRFSS index better or why it is unreliable. During the workshop, one participant suggested a revised calculation methodology. Preliminary analysis suggests this modification did not change the overall pattern.**
- **The thought that anglers have "switched rigging" has caused the decrease in CPUE is not a very good example of why the index may be flawed. In many areas, the same methodologies are used for catching both species. It would be possible to evaluate this assumption by recalculating the trend with directed striped bass trips removed to see how the trend responds?**
- **The DPW Panel is mistaken in its assumption that party boat trips were used to develop the MRFSS index of effort.**
- **MRFSS VPA run preferred mainly because it gave the shortest retrospective pattern, not because it had the trend the TC/SAS was looking for. It also has a relatively large sample size and incorporates the entire core area (not localized like FI indices).**
- **It is the pattern of the index that is important!**

f) Coherence of fishery-independent indices: The Panel was troubled by the apparent coherence of the aggregate fishery independent indices used as input to the model compared to the different trajectories estimated as output by the VPA when different groupings of these indices were used as inputs. The Panel considered that the differences between the coherence of the input time series and the model outputs may reflect differences in the age-specific catchabilities and thus abundances monitored by these surveys. The Panel felt that detailed exploration of this apparent discrepancy should be conducted.

- **The TC/SAS agreed that this was the most useful of the DPW concerns and recommendations.**
- **All FI runs show a similar pattern although the Panel seemed to be concerned about the difference in FI vs. FD runs for indices that show same aggregate trend. Jeff Brust looked into these differences. The short retrospective of the FD run is driven by the MRFSS 3 to 6+ indices. Using just the MRFSS 2+ gives the same VPA result as using FI indices. Catchability does change when regulations change. This was described in the plot of the residuals from these indices in the 2006 assessment. The TC/SAS will continue exploration of this and provide results in assessment.**

- **Need to take all indices of a given age (all age 1, all age 2, etc) and compare them. Maybe something will jump out at us.**

g) Weights at age: The Panel noted substantial discrepancies in the weights at age in the catch at age (e.g., age-4 weakfish in one year were heavier than age-5 fish in the subsequent year). These discrepancies could be a consequence of estimation of the catch at age for one fleet using catch at age data from a different fleet.

- **How bad is this compared to other species? All assessments have this and it will affect estimates of biomass but not estimates of stock size (in numbers) or mortality rates. We are currently looking into historic data in attempt to find any problems with the WAA. It could also reflect real phenomena from heavy fishing or poor feeding. Plus there might be some effect of mixed ageing currencies (scales converted to otolith ages and actual otolith ages).**
- **Someone should look at length at age over time as well. Might provide insight into changing productivity over time.**

Overall the review Panel believed that the conclusion that a time varying M was the principal explanation for the pattern of low biomass and high F's observed in the MRFSS tuned VPA was unwarranted. The review Panel felt that other alternative explanations, even assuming inputs were correct, including missing catch, changing catchabilities and inappropriateness of information in the input surveys should be fully explored before the results of the VPA can be used as a spring board to suggest the need to explain an increasing pattern in M. The Panel noted that many of these concerns had been raised by the previous peer review team and has yet to be adequately addressed. Given the nature of the concerns regarding the catch at age, the assessment team should consider a statistical catch at age approach rather than VPA.

- **The results of the VPA provided in this assessment are not intended as spring board to explain an increasing pattern in M. The VPA estimates total mortality (Z) based on changes in age structure, then divides Z into F & M by subtracting the input M. Relative F analysis showed that the increase in Z was not due to F, so the SAS initiated analysis to determine an increase in M. The TC/SAS is aware of the limitations of the VPA such as the assumption of constant M, retrospective patterns and known error in CAA but it is still useful for total mortality estimates and stock size estimates.**
- **The 30th SARC reviewed data through 1998. Regardless of indices used, all runs of current model show same trends through 2001 or 2002. Therefore, the entire converged portion of the VPA output has passed peer review. Could changes to the inputs affect the converged portion of VPA? It is not apparent from the Panel's comments if they think trends prior to 2000 are inaccurate.**
- **With regard to missing catch, this could really be any unaccounted losses, such as harvest, mortality, bycatch, etc. The TC/SAS has developed estimates of the amount of unaccounted losses necessary to improve the model fit. In recent years, unaccounted losses would need to be 15-20 times our current discard estimates – an amount that would be difficult to overlook in our sampling.**
- **With regard to changing survey catchabilities, biomass and abundance estimates from the converged portion of the VPA – which is not dependent on survey data –clearly**

indicate a declining abundance. The decline can not therefore be entirely due to trends in catchability.

- **With regard to inappropriate info in the indices, there is evidence that abundance is declining outside the influence of survey data (such as commercial CPUE from VA and NC), as explained in the above bullet as well as sensitivity runs made without any tuning indices during the 2006 assessment The 2004 peer review recommended using a recreational CPUE index because of problems with the trawl surveys. We followed their recommendations.**
- **So they just trashed ALL of our VPA input data, yet suggest a model that uses the exact same inputs. The SCAM has more flexibility, but enough to compensate for what they consider to be such inappropriate data? A model can only do so much with the data you give it. The SAS initiated development of an ASAP model, but haven't been able to continue development because of the preparation for and dealing with the DPW review.**
- **There were no comments regarding Relative F modeling which is a good predictor of trends in fishing mortality without the reliance on actual measured abundance. It also is not dependent on an estimate of M which the SAS considers its strong point.**
- **It is puzzling that the panel never recommends performing the analysis with a constant M.**
- **Tuning indices only affect estimates in the most recent years. The TC feels the panel was too focused on indices.**

3.3. Biomass dynamic modeling

The Panel was very interested in the results of the biomass dynamic models that were presented during the meeting. The Panel felt that they were an interesting exploration of potential ecological mechanisms acting on weakfish. However, if such models are to form the foundation for management there needs to be compelling and direct empirical evidence for the mechanisms being hypothesized In general, the Panel considered that such evidence was lacking. The Panel was further concerned that the implications of the results for management (e.g., if surplus production in weakfish is truly negative currently, then no viable weakfish fishery is possible) had not been fully considered by the TC.

- **Biomass modeling shows fisheries are not responsible for the most recent decline in weakfish. Evidence of potential predation/competition assists this hypothesis. Jim Uphoff and Vic Crecco have developed biomass models that take into account the various potential causes including environmental, predation and competition factors.**
- **There is empirical evidence that is in synch with the modeling results. Members of the SAS performed a large-scale review of diet literature as well as spatial and temporal overlap. Statistical analysis of the literature was used to determine main candidates for modeling. This analysis was not available to the DPW but it was in the last assessment. It will also be presented at the final review in June.**
- **The TC/SAS has fully considered the implications of these findings, but really that's not the job of the TC/SAS as the implications are not a technical or analytical issue. The job of the TC/SAS is to determine the status of the weakfish population. It's up to the managers to discuss the implications of the findings and decide what needs to be**

done. The alternative is to pretend that fishing mortality is the only factor influencing stock dynamics, and fishermen need to bear the brunt of "fixing" the population.

- **The SAS has presented a convincing case (hypothesis) that biotic factors in the form of enhanced predatory mortality, rather than overfishing, was the main reason for the recent unexpected and steady decline of weakfish abundance from 2000 to 2007. The recent trend in overall juvenile weakfish recruitment has been high since the late 1990's, but these dominant year-classes no longer translate through the age structure in subsequent years, suggesting the recent emergence of a demographic bottleneck in recruitment. Unless there has been a steady rise in weakfish juvenile discards since 1999, the emergence of a demographic bottleneck is consistent with enhanced predation on smaller weakfish. Has any other single species stock assessment conducted thus far examined the recruitment data to see whether such a phenomenon might exist?**

The Panel again noted the central role of the MRFSS index in determining the results of the biomass dynamic modeling. When an index with a strong, almost exponentially declining pattern is used to drive a model, any variable that shows an opposite trend will appear as a strong covariate in model fits – particularly given the latitude in parameters implied by the assumption of the form of a type III functional response. However, such correlations obviously do not imply causation. Under such circumstances, the Panel noted that documentation of weakfish consumption by striped bass needs to be more fully documented to provide the causation strongly implied by the assumptions of the models presented to the Panel. The TC needs to consider the pattern of spatial and temporal overlap of the two species and the influence of this dynamic on the levels of consumption required. Such consideration appeared lacking from the material presented to the Panel.

- **MRFSS was only one of three equally weighted indices used in the two biomass models. The empirical evidence is not limited to predation. The TC/SAS will continue to look into forage data (e.g. menhaden and FHDBS), environmental data (oscillations, buoy data), changes in growth parameters (Vaughan data), changes in M (Lorenzen), and indices of forage (SVDBS engraulids and clupeids).**
- **The SAS has developed hypotheses, presented data, and evaluated plausibility. Regardless of cause, weakfish are at low abundance. The Panel even implies they agree with this statement in several of their responses. With regard to predation, there does not need to be a large number of weakfish eaten per predator, or even a trend in number eaten, because the trend in predators has increased, and there is an increased abundance of predators, even small consumption of weakfish might affect the stock.**
- **The temporal/spatial overlap of the two species is considerable, particularly during fall migration and overwintering areas.**
- **The SAS has produced an update on discards which gives a brief description of the relative level of consumption.**

The assumption of a type III functional response appears arbitrary. There are several valid alternatives that have been used in other predator-prey models – ranging from type I and II, to foraging arena concepts (Walters and Juanes 1993, Walters and Martell 2004). Each of the different functional responses would have extremely different consequences for the dynamics of weakfish inferred by such models. It was not apparent from the material presented that an adequate exploration of this aspect of the biomass dynamic models with covariates had been undertaken.

- **This comment is largely due to the time limitations of the workshop. The first day (2+ hours) was spent discussing data inputs, VPA, and relative F. The second day, due to time limitations, the Panel considered both biomass models in less than an hour, so they may not have picked up on the differences between the predator hypothesis and the forage hypothesis.**
- **Regardless, the forage hypothesis investigated 4 functional responses, including Type I, II, III and depensatory. Type III actually performed poorly, so the SAS has already addressed this. It will be thoroughly discussed text of the final assessment.**
- **Residual errors were clarified through adding a predation factor. Through analysis, the SAS has not been able to falsify the striped bass/dogfish predation factor.**

The Panel noted that when a resource is in a depleted condition, such as in the case of weakfish, a number of factors can be responsible for maintaining the stock in the depressed state. Examples in the literature of "predator pits" preventing recovery in predator - prey models have been reported (Bundy and Fanning 2005). There is a continuing debate in Atlantic Canada on the role of grey seals maintaining Atlantic cod at their low level of abundance (Chouinard et al. 2005, Trzcinski et al. 2006). However, the mechanisms maintaining the prey species at low levels of abundance and the mechanism that caused the reduced abundance in the first place are not necessarily the same thing. Thus, for weakfish, predation may be maintaining the population at low levels, without having contributed to the original decline of the stock.

- **The TC/SAS agrees that the weakfish stock is at a low level and that the influence(s) that is keeping it there could be different than what put it there in the first place. The question is what put it there in the first place, especially since we were starting to rebuild (and have lots of different data sources that indicate this). To change from rebuilding to declining stock, total mortality would have had to increase and/or recruitment decrease. Relative F indicates that fishing mortality was probably not the leading cause.**
- **The SAS looked at other factors such as recruitment (environmental?) or mortality (other anthropogenic or natural).**
- **The DPW Panel appears to concede that the stock is at low levels. If it is due to predation, we probably can't have a viable fishery until that predation pressure is removed. These are the same implications they accused us of not fully considering?**
- **If the Panel agrees the stock is at low levels, they are implicitly "accepting" the index and harvest data, thereby contradicting many of their concerns.**

The Panel felt that the attempts of the TC to develop a minimum realistic model (MRM) for weakfish trophic interactions, as recommended by Plagányi (2007), were laudable. However, the Panel also felt that the biomass dynamic models were not yet at the stage to provide a reliable basis for the determination of weakfish stock status.

- **The DPW Panel believes that the input data and the results of the VPA are inappropriate,**
-

- **the biomass models are unreliable, and SCAM uses same data as VPA. So what is left? Weakfish has become the ultimate example of "data poor" after the last 2 reviews. The data has been shot down as unacceptable and ASMFC is worse off now than prior to the last assessment. Back to the drawing board?**

The Panel did not have sufficient time to provide responses to a number of specific questions raised by the TC themselves (ASMFC Weakfish TC 2008a). However, the Panel noted that it has provided guidance on several questions. Most importantly, perhaps for the management of weakfish, the Panel feels that the VPA is not yet sufficiently developed or its results sufficiently explored to support the conclusion of an increasing pattern in M. While the Panel appreciated the spirit of the exploration of ecological mechanisms to explain a pattern of increasing M, these analyses are not of sufficient reliability, given concerns over the MRFSS index and the lack of empirical evidence for the hypothesized predator-prey interaction involving striped bass and weakfish, to be a current assessment tool of the weakfish resource.

- **It appears on several levels that the Assessment presentation needs adjustment. The SAS may need to make some concessions on the cause of stock decline by generalizing, but it can do this without saying it is all due to F. Also, the focus should be on the findings of all the models – i.e. that the stock appears to be at low levels currently. The Panel appears to agree that the stock is depressed, which is a positive step since the last peer review didn't even agree that we were in a depressed state. The Board needs to know that the stock is in a depressed state, so we need to figure out how to get the Panel to make a statement about status regardless of how that status came about.**
- **It will not take much to address the concerns of this Panel. In a few places the SAS will need to do some additional exploratory work with models and such. Much of the work will come from just expanding the text of the assessment to acknowledge the Panel's concerns and to reference and summarize other reports where that work was done.**

Appendix C-2

Proportional Stock Density Indices for Weakfish

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Amendment 4 requires that descriptors of age or size structure be reported to the Management Board as part of each stock assessment. Proportional stock densities (PSDs) are standardized methods for analyzing length-frequency data that quantify size structure of a fish population (typically gamefish) into categories of sizes that are of interest to recreational anglers (Gablehouse 1984). These techniques allow comparisons of size quality across species. Used commonly for freshwater stock assessment, they have been applied on a limited basis in marine management; Maryland DNR uses them routinely to assess size quality of several species of estuarine and marine gamefish (J. Uphoff, MD DNR, personal observation). Length-frequency data, indexed by PSDs, 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).

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 (Kahn et al. 2006). Sample sizes in the DE and NJ surveys were large enough for precise estimates of PSD Q (Kahn et al. 2006). The normal distribution approximation of the binomial distribution was used to calculate the SD (Ott 1977). The DE Quality+ PSD represents the longest fishery-independent data set available for weakfish. The DE PSD Q+ index was significantly ($P < 0.05$) 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 (Uphoff 2004). Proportion of recreational harvest in bays and sounds was negatively associated with DE PSD Q+ (Uphoff 2004).

The PSD Q+ size quality indices for DE (1966-1971, 1979-1984, and 1990-2006; Appendix 1, Figure 1) and NJ (1989-2006; August and October; Appendix 1, Figure 2) indicated that size quality of weakfish in recent years was truncated at smaller lengths. 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 (Appendix 1, Figure 1). Values in the early 1980s were twice as high as this peak. Values were lower during 1966-1968 than current indices (Appendix 1, Figure 1).

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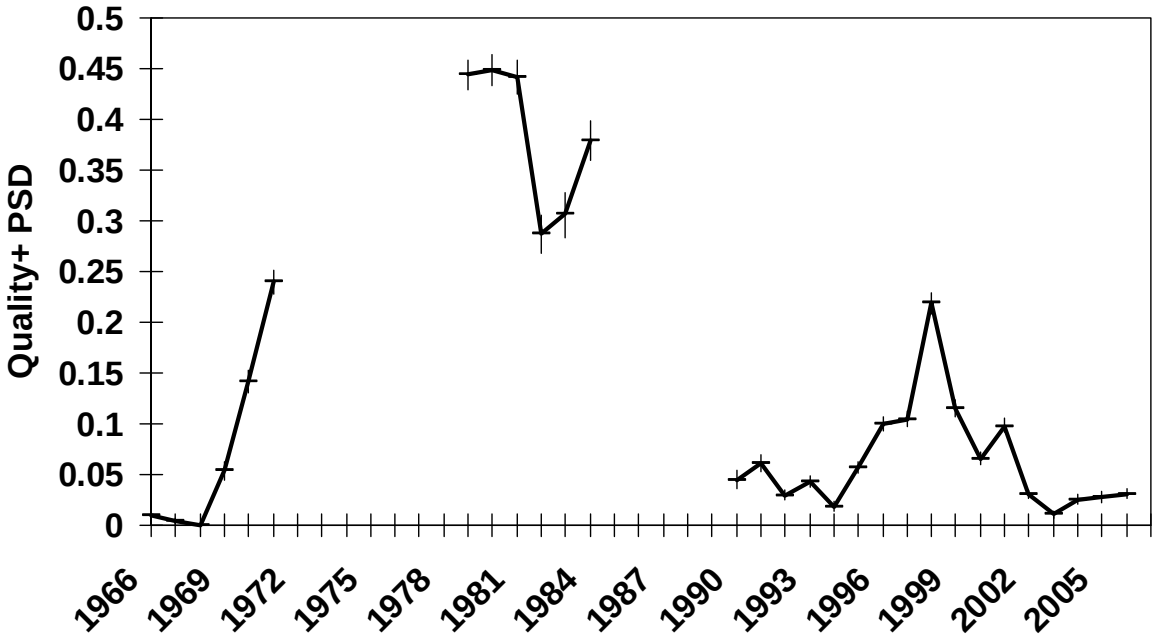
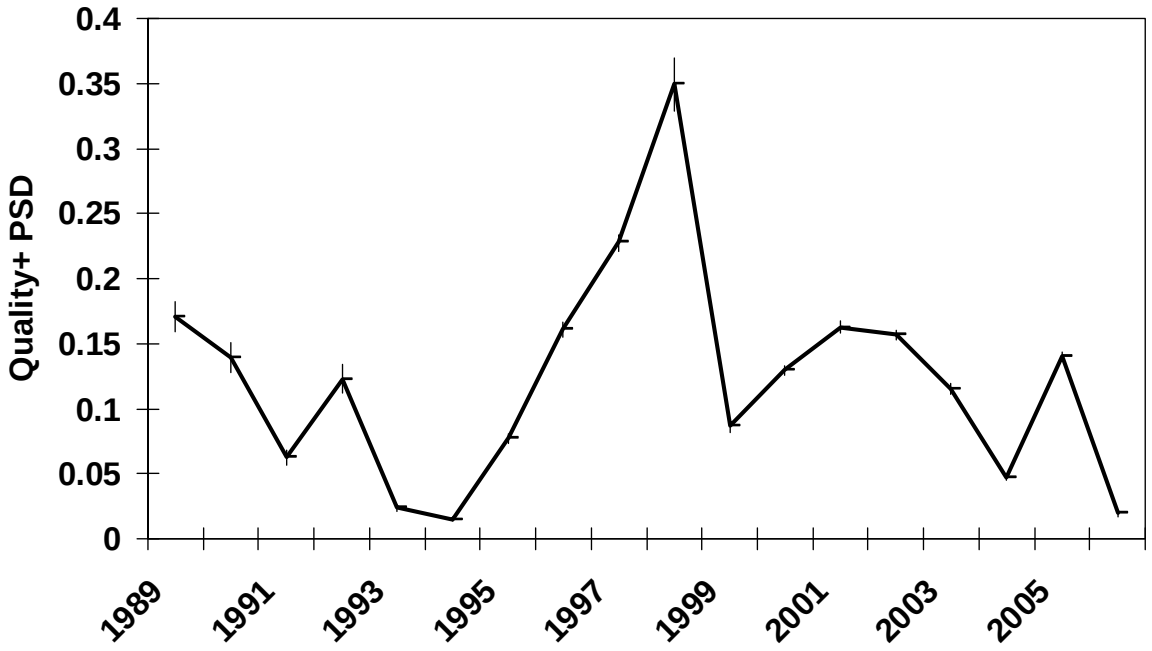


Appendix C-2, Figure 1. 95% CI of Delaware survey Quality+ PSD and trend



Appendix C-2, Figure 2. NJ survey Quality+ PSD 95% CI and trend

Appendix C-3

SAS-based application of the Harvest Control Model to conduct weakfish stock projections

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```
*-APPENDIX C-3-----------------------------------------------------------------
                                  ----------*
| Run-Specific Macro Variables HARVEST CONTROL MODEL FOR WEAKFISH FIXED M OF
0.25, 1980-2020                                                  |
*-----------------------------------------------------------------------------*;
%let MaxAge = 10;                /* Maximum age Considered                   */

%let A = 0.75;                  /* A in Stock Recruitment Relationship-not used
 */
%let B = 1.5;                   /* B in Stock Recruitment Relationship-not used
  */
%let K = 4000;                 /* K in Stock Recruitment Relationship-not used
*/

*-----------------------------------------------------------------------------*
| Read in Annual Data:                                                        |
|    |
|             CstF..     Coast     F-     BASED     ON     recent     ADAPT     run
                     |
|                      Juv1..        age        1        recruitment        (REC0)
     |
|
|
     M   .. NATURAL MORTALITY RATE-TIME VARYING
|
*-----------------------------------------------------------------------------*;
data YearClas;
  input Year  CstF REC0  ;
  if _n_ = 1 then call symput('FrstYear',put(Year,4.));
*-----------------------------------------------------*
Year BayF  CstF REC0 BayR CstR BayD  CstD  JUV2 JUV3 JUV4  M RECC RECH
*-----------------------------------------------------*;
cards;

1970  0.44 0.44 160   24  24  0.00  0.00  0.7 0.16 . 0.15 . .
1971  0.22 0.72 140    24  24  0.00  0.00  1.8 0.16 . 0.15 . .
1972  0.24 0.64 80    24  24  0.00  0.00  1.5 0.16 . 0.15 . .
1973  0.22 0.82  190   24  24  0.00  0.00 0.3 0.16 . 0.15 . .
1974  0.22 0.82 60    24  24  0.00  0.00  0.9 0.16 . 0.15 . .
1975  0.42 0.42 150    24  24  0.00  0.00  1.3 0.16 . 0.15 . .
1976  0.24 0.74 170    24  24  0.00  0.00  0.7 0.16 . 0.15 . .
1977  0.40 0.90 140    24  24  0.00  0.00  1.8 0.16 . 0.25 . .
1978  0.35 0.95 170   24  24  0.00  0.00  1.8 0.16 . 0.25 . .
1979  1.21 1.21 130    24  24  0.00  0.00  0.3 0.01 . 0.25 . .
1980  1.29  1.29 230   24  24  0.00  0.00  0.9 0.01 . 0.25 . .
1981  1.73  1.73 55.7   12  17  0.00  0.00  1.3  0.01 1393 0.25 56 53
1982  1.40  1.40 18.4   12  17  0.00  0.00  2.1  0.30 2785 0.25 56 11
1983  1.61  1.61 35.0   14  24  0.00  0.00  2.4  0.08 2441 0.25 56 11
1984  2.21  2.21 33.9   14  24  0.00  0.00  2.3  0.73 2964 0.25 59 8
1985  1.35  1.35 18.3   14  24  0.00  0.00  0.3 0.05 2876 0.25 32 0
1986  1.73  1.73 8.0   14  29  0.00  0.00  0.4 0.70 3642 0.25 161 0
1987  2.66  2.66 5.9   14  31  0.00  0.00  2.4  1.03 4740 0.25 82 0
1988  1.81  1.81 7.4   14  33  0.00  0.00  5.0  0.88 5972 0.25 82 0
1989  2.12  2.12 1.9   14  33  0.00  0.00  3.2  2.58 8100 0.25 30 0
1990  2.76  2.76 5.7   18  34  0.00  0.00  2.9  2.50 7342 0.25 171 0.7
1991  0.84  0.84 5.5   18  34  0.00  0.00  0.55 1.18 6790 0.25 243 71
1992  1.03  1.03 43.4   18  34  0.00  0.00  1.7 2.92 8129 0.25 356 92
1993  0.71  0.71 88.6  18  32  0.00  0.00  1.9 4.45 10042 0.25 543 66
1994  0.98  0.98 28.5   18  32  0.00  0.00  2.6  2.40 6391 0.25 664 154
1995  0.32  0.32 10.7   18  29  0.00  0.00  2.4 4.95 7838 0.25 1593 221
```

```
1996  0.53  0.53 17.1  18  29  0.00  0.00  1.3 4.25 7876 0.25 1411 238
1997  2.56  2.56  14.7  18  29  0.00  0.00  3.3 2.53 9105 0.25 1604 144
1998  2.72  2.72  4.6  18  28  0.00  0.00  2.0 3.28 9518 0.25 1132 122
1999  2.67  2.67  17.3  18  28  0.00  0.00  4.0 4.75 6302 0.25 1337 88
2000  2.82  2.82  4.0  18  26  0.00  0.0  0.5 4.4 . 0.25 1120 99
2001  2.04  2.04  9.8 18  26  0.00 0.00 . . . 0.25 1254 52
2002  2.34  2.34  11.7   18  26  0.0  0.00 . . . 0.25 1591 67
2003  2.38  2.38   3.6  18  26 0.0  0.00 . . . 0.25 1666 78
2004  2.06  2.06   4.6  18   26  0.0  0.0 . . . 0.25 1807 63
2005  2.52  2.52   12.0   18  26  0.0  0.0 . . . 0.25 1584 51
2006  2.47  2.47   5.7   18   26  0.0  0.0 . . . 0.25 1266 91
2007  2.13  2.13   21.5   18   26  0.0  0.0 . . . 0.25 . 103
2008  2.20  2.20   8.5   18   26  0.0  0.0 . . . 0.25 . 38
2009  2.20  2.20   13.0     18   26   0.0 0.0 . . . 0.25 .  .
2010  2.20  2.20   13.0     18   26    0.0 0.0 . . . 0.25 . .
2011  2.20  2.20   12.0     18   26   0.0  0.0 . . .  0.25 . .
2012  2.20  2.20   21.0     18   26   0.0  0.0 . .  .  0.25 . .
2013  2.20  2.20    9.0    18   26   0.0  0.0 . . .   0.25 . .
2014  2.20  2.20    12.0    18   26   0.0   0.0 . . .   0.25 . .
2015  2.20  2.20    22.0    18   26   0.0   0.0 . . .  0.25 . .
2016   2.2  2.2     22.0    18   26   0.0   0.0  . . .  0.25 . .
2017  2.20  2.20    9.0    18   26   0.0  0.0 . . .   0.25 . .
2018  2.20  2.20    12.0    18   26   0.0   0.0 . . .   0.25 . .
2019  2.20  2.20    22.0    18   26   0.0   0.0 . . .  0.25 . .
2020   2.2  2.2     22.0    18   26   0.0   0.0  . . .  0.25 . .
;
run;

*-----------------------------------------------------------------------*
| Read in Population Parameters:
|   AGE .....AGE GROUP
|   MigR ... Migration Rate (not used in this version)                  |
|   MatR ... Maturation Rate (fraction mature)                          |
|   RcrR ... Partial Rec Rate (fraction recruited to the fishery)
|   PERR .....FRACTION AGES 8+
*-----------------------------------------------------------------------*;
data PopParms;
  input Age MigR MatR rcrbr RCRBbR RCRCR RCRCcR PERR;
*----------------------------------------*
Age    MigR  MatR  RcrBr RCRBbR RCRCR RCRCcR PERR
*----------------------------------------*;
cards;
0    1.00    0.00  0.0    0.0    0.0   0.0   0.0
1    0.00    0.9  0.34   0.16    0.34  0.16   0.0
2    0.0     1.0  0.58   0.47   0.58  0.47   0.0
3    0.0     1.0  0.75   0.73   0.75  0.73   0.0
4    0.0     1.0  1.0    1.0   1.0  1.0  0.0
5    0.0     1.0  1.0    1.0    1.0  1.0  0.0
6    0.0     1.0  1.0    1.0     1.0  1.0  0.0
7    0.0     1.0  1.0    1.0     1.0  1.0  0.0
8    0.0     1.00  1.00   1.0     1.0  1.0  1.0
9    0.0     1.00  1.00   1.0     1.0  1.0   1.0
10   0.0    1.00  1.00   1.0     1.0  1.0   1.0


;
run;

*-----------------------------------------------------------------------*
| Read in Age at Length Data:                                           |
```

```
|   Lgth ... Length (inches)                                                |
|   Age .... Age (years)                                                    |
*---------------------------------------------------------------------------*;
data LngthAge;
  input Lgth Age;
  if _n_ = 1 then call symput('MinLngth',put(Lgth,2.));
  else call symput('MaxLngth',put(Lgth,2.));
*--------------------------
Lgth  Age
*--------------------------;
cards;
12   2.0
14   2.5
17   3.2
18   4.0
24   4.9
26   5.7
28   6.8
29   7.2
30   7.5
32   7.3
31   7.1
33   7.7
34   8.1
36   9.0
;
run;

%put;
%put FrstYear = &FrstYear;
%put MaxAge   = &MaxAge;
%put MinLngth = &MinLngth;
%put MaxLngth = &MaxLngth;


*---------------------------------------------------------------------------*
| The Model                                                                 |
*---------------------------------------------------------------------------*;
data Model (keep = Year BAYR CSTR M BayF CstF Juv1 BayR CstR JUV2 JUV3 JUV4 REC
RECC RECH
                   LREC JUV5 LREC2 TSSB TBES TCES TN_Bay TN_CST TBAYCAT TBAYLD
TCatch TYIELD STK MT TCOASTYD TCSTCAT WEIGHT Z FZ)
     Debug (keep = Year BAYR CSTR Age Migrate
                   N_Bay ZB1 ZB2 FB1 FB2
                   BayF1 BayF2 BayDeath BayCatch BAYIELD BES
                   N_Cst ZC1 ZC2 FC1 FC2
                   CstF1 CstF2 FPrimeC1 FPrimeC2 CstDeath CstCatch
                   COASTYLD Wt SSB BES CES );


array Mig{0:&MaxAge};                     /* Migration Rate                 */
array Mat{0:&MaxAge};                     /* Maturation Rate                */
array RcrB{0:&MaxAge};                     /* PARTIAL REC BAY               */
array RcrBb{0:&MaxAge};                     /* PARTIAL REC BAY              */
array RcrC{0:&MaxAge};                     /* PARTIAL REC CST               */
array RcrCc{0:&MaxAge};                     /* PARTIAL REC CST              */
ARRAY PER{0:&MAXAGE};
array L_Age{&MinLngth:&MaxLngth};         /* Age-Length Key                 */
array BayPop{0:&MaxAge};                  /* Bay Population Array           */
array CstPop{0:&MaxAge};                  /* Coast Population Array         */
```

```
A=&A; B=&B; K=&K;                          /* Stock Recruitment Params       */

do until (EOP);                            /* Read in PopParms Data Set      */
  set PopParms  end=EOP;
   Mig{Age}=MigR;
  Mat{Age}=MatR;
  RcrB{Age}=RcrBr;
  Rcrbb{Age} = Rcrbbr;
  RcrC{Age}=RcrCr;
  Rcrcc{Age}= Rcrccr;
  PER{AGE} = PERR;

  if Age = &MaxAge then EOP=1;
  end;

do until (EOL);                            /* Read in Age at Length Key      */
  set LngthAge end=EOL;
  L_Age{Lgth}=Age;
  end;

do Age = 0 to &MaxAge;                     /* Nobody home                    */
  BayPop{Age}=0;
  CstPop{Age}=0;
  end;

** ASSUMPTION: REC0 of Age 0 fish at beginning of year 0   ;
**             i.e. before mortality (natural and discard)                 ;

do until (EOY);

  set YearClas end=EOY;                    /* Read a year class obs          */
  if Juv1 = . then Juv1 = 0;
  IF JUV2 = . THEN JUV2 = 0;
  IF JUV3 = . THEN JUV3 =0;
  REC = JUV1;
  LREC = LOG(REC);
  JUV5 = JUV4*0.002;
    LREC2 = LOG(JUV1);
  Disc_Bay = 0.1 * BayD;                   /* Bay Discard Mortality Rate     */
  Disc_Cst = 0.1 * CstD;                   /* Cst Discard Mortality Rate     */
  do Age = 10 to 0 by -1;                  /* COMPUTE F's AND Z's            */

    M_Age = L_Age{BayR};                   /* Mean age at min length (Bay)   */
    D_Age = M_Age - Age;                   /* Delta age                      */
    if 0 < D_Age < 0.5 then do;            /* Sublegal during all or part of */
      Ratio = D_Age / 0.5;                 /*  the first half the the year.  */
      FPrimeB1 = Disc_Bay * Ratio;        /* F prime 1st half the year       */
      FPrimeB2 = 0;                        /* F prime 2nd half the year      */
      BayF1 = BayF*0.5 * (1.0-Ratio);         /* F 1st half the year         */
      BayF2 = BayF*0.5;                       /* F 2nd half the year         */
      end;
    else if D_Age >= 0.5 then do;          /* Sublegal during all or part of */
      if D_Age > 1.0 then D_Age = 1.0;     /*  the second half the the year. */
      Ratio = (D_Age-0.5) / 0.5;
      FPrimeB1 = Disc_Bay;               /* F prime 1st half the year        */
      FPrimeB2 = Disc_Bay * Ratio;        /* F prime 2nd half the year       */
      BayF1 = 0;                           /* F 1st half the year            */
      BayF2 = BayF*0.5 * (1.0-Ratio);       /* F 2nd half the year           */
      end;
```

```
else do;                                /* Legal During whole year        */
  FPrimeB1 = 0;                         /* F prime 1st half the year      */
  FPrimeB2 = 0;                         /* F prime 2nd half the year      */
  BayF1 = BayF*0.5;                        /* F 1st half the year         */
  BayF2 = BayF*0.5;                        /* F 2nd half the year         */
  end;

M_Age = L_Age{CstR};                    /* Mean age at min length (Coast) */
D_Age = M_Age - Age;                    /* Delta age                      */
if 0 < D_Age < 0.5 then do;             /* Sublegal during all or part of */
  Ratio = D_Age / 0.5;                  /*  the first half the the year.  */
  FPrimeC1 = Disc_Cst * Ratio;         /* F prime 1st half the year       */
  FPrimeC2 = 0;                         /* F prime 2nd half the year      */
  CstF1 = CstF*0.5 * (1.0-Ratio);          /* F 1st half the year         */
  CstF2 = CstF*0.5;                        /* F 2nd half the year         */
  end;
else if D_Age >= 0.5 then do;           /* Sublegal during all or part of */
  if D_Age > 1.0 then D_Age = 1.0;      /*  the second half the the year. */
  Ratio = (D_Age-0.5) / 0.5;
  FPrimeC1 = Disc_Cst;                 /* F prime 1st half the year       */
  FPrimeC2 = Disc_Cst * Ratio;         /* F prime 2nd half the year       */
  CstF1 = 0;                            /* F 1st half the year            */
  CstF2 = CstF*0.5 * (1.0-Ratio);          /* F 2nd half the year         */
  end;
else do;                                /* Legal During whole year        */
  FPrimeC1 = 0;                         /* F prime 1st half the year      */
  FPrimeC2 = 0;                         /* F prime 2nd half the year      */
  CstF1 = CstF*0.5;                        /* F 1st half the year         */
  CstF2 = CstF*0.5;                        /* F 2nd half the year         */
  end;

  if year> 1993
  then
FB1 = BayF1*RcrBb{Age} + FPrimeB1*RcrBb{Age}; /* Bay F 1st half of year*/
  else FB1=BAYF1*RCRB{AGE} + FPRIMEB1*RCRB{AGE};
IF YEAR>1993 THEN
FB2 = BayF2*RcrBb{Age} + FPrimeB2*RcrBb{Age}; /* Bay F 2nd half of year*/
  ELSE FB2= BAYF2*RCRB{AGE}+FPRIMEB1*RCRB{AGE};
  IF YEAR> 1993
  THEN
FC1 = CstF1*RcrCc{Age} + FPrimeC1*RcrCc{Age}; /* Cst F 1st half of year*/
  ELSE FC1= CSTF1*RCRC{AGE}+FPRIMEC1*RCRC{AGE};
  IF YEAR>1993
  THEN
FC2 = CstF2*RcrCc{Age} + FPrimeC2*RcrCc{Age}; /* Cst F 2nd half of year*/
  ELSE FC2= CSTF2*RCRC{AGE}+FPRIMEC2*RCRC{AGE};


If AGE < 3 THEN ZB1= (0.10/2)+FB1;
ELSE ZB1 = (M/2) + FB1;
  IF AGE < 3 THEN ZB2 = (0.10/2) + FB2;      /* Bay Z 1st half of year */
ELSE ZB2 = (M/2) + FB2;                      /* Bay Z 2nd half of year */
  IF AGE < 3 THEN ZC1 = (0.10/2) + FC1;
ELSE ZC1 = (M/2) + FC1;                    /* Cst Z 1st half of year   */
  IF AGE < 3 THEN ZC2 = (0.10/2) + FC2;
ELSE ZC2 = (M/2) + FC2;                    /* Cst Z 2nd half of year   */

                                           /* BIRTH, DEATH, MIGRATION  */
```

```
if Age > 0 then BayPop{Age}=BayPop{Age-1}; /* Celebrate Bay Birthday   */
else BayPop{Age} = juv1;                   /* New Year Class           */
Survive = exp(-ZB1*0.5) * BayPop{Age};     /* Bay survivors 1st half   */
Deaths1 = BayPop{Age} - Survive;           /* Deaths 1st half of year  */
Catch1  = (BayF1/ZB1) * Deaths1;           /* Catch 1st half of year   */
Migrate = Survive * (1.0-Mig{Age});        /* Nbrs migrate to coast    */
Survive = Survive - Migrate;               /* Bay Surv after migrate   */
BayPop{Age} = exp(-ZB2*0.5) * Survive;     /* Bay Surv at end of year  */
Deaths2 = Survive - BayPop{Age};           /* Deaths 2nd half of year  */
Catch2  = (BayF2/ZB2) * Deaths2;           /* Catch 2nd half of year   */

BayDeath = Deaths1 + Deaths2;             /* Bay death and destruction */
BayCatch = Catch1  + Catch2;               /* Bay catch                */

if Age > 0 then CstPop{Age}=CstPop{Age-1}; /* Celebrate Coast Birthday */
Survive = exp(-ZC1*0.5) * CstPop{Age};     /* Cst survivors 1st half   */
Deaths1 = CstPop{Age} - Survive;           /* Deaths 1st half of year  */
Catch1  = (CstF1/ZC1) * Deaths1;           /* Catch 1st half of year   */
Survive = Survive + Migrate;               /* Cst Surv after immigrate */
CstPop{Age} = exp(-ZC2*0.5) * Survive;     /* Cst Surv at end of year  */
Deaths2 = Survive - CstPop{Age};           /* Deaths 2nd half of year  */
Catch2  = (CstF2/ZC2) * Deaths2;           /* Catch 2nd half of year   */

                                           /* ACCUMULATE TOTALS        */
CstDeath = Deaths1 + Deaths2;              /* Cst death and destruction*/
CstCatch = Catch1  + Catch2;               /* Cst catch                */
 Catch = BayCatch + CstCatch;               /* Cst & Bay Catch         */

N_Bay = BayPop{Age};                       /* debug                    */
N_Cst = CstPop{Age};                       /* debug                    */

Wt = 9.1*(1-exp(-0.15*(Age-0.68)))**3.0;   /* Weight at Age            */
SSB = (CstPop{Age}+BayPop{Age})*Wt*Mat{Age};/* Spawning stock biomass  */
BES = BAYPOP{AGE}*PER{AGE}*wt;  /*  EXPLOITABLE STOCK BIOMASS */
CES = CSTPOP{AGE}*WT*PER{AGE};  /*  EXPLOIABLE STOCK BIOMASS */
BAYIELD = WT* BAYCATCH;
COASTYLD= WT*CSTCATCH;
YIELD = BAYIELD + COASTYLD;

TSSB + SSB;       /*  Total Spawning Stock Biomass/*

TBES + BES;
TCES + CES;
  STK =TBES+TCES;
TBayCat + BayCatch;
TCstCat + CstCatch;
TCatch + Catch;
TN_Cst + N_Cst;
TN_BAY + N_BAY;
TBAYLD+ BAYIELD;
TCOASTYD+COASTYLD;
TYIELD+YIELD;
MT = 78.31+999.27*TYIELD;
IF TCATCH > 0 THEN
WEIGHT = TYIELD/TCATCH;
  PROP = TCES/TN_CST;
  Z = CSTF+M;
  FZ = CSTF/Z;
output Debug;
```

```
  end;  /* do Age */

  output Model;

  TSSB=0;   TBayCat=0; TBAYLD=0; TYIELD=0;
  TCatch=0; TCstCat=0; TCOASTYD=0;
  TN_Cst =0; TBES=0; TCES=0; TN_BAY=0;

end;  /* do Year */

stop;
run;


*-------------------------------------------------------------------------*
| Print the Results                                                       |
*-------------------------------------------------------------------------*;
title1 " TABLE 1.weakfish PROJECTION (TOTAL STOCK) MODEL FOR coast-wide stock
1980-2020, M IS FIXED";

title2 "Age-Specific Model Parameters";
proc print data=PopParms NOOBS;
  id Age;
run;


title2 "HARVEST CONTROL Summary Statistics, M FIXED, STOCK BASED THE 1970-2008
REC 0";
proc print data=Model NOOBS;
where year > (&frstyear + &maxage-1);
   VAR YEAR CSTF  M Juv1 tssb TN_CST TCES WEIGHT  Z FZ ;
run;


PROC CHART DATA= MODEL;
WHERE YEAR> (&FRSTYEAR + &MAXAGE -1);
VBAR YEAR/DISCRETE SUMVAR = Tssb;
TITLE'FIGURE 1. SIMULATED WEAKFISH SSB UNDER CURRENT MINIMUM SIZE LIMITS';
TITLE2'FROM COASTAL POPULATION UNDER A FIXED M FROM 1980 TO 2020';
RUN;

PROC CHART DATA = MODEL;
WHERE YEAR> (&FRSTYEAR + &MAXAGE -1);
  VBAR YEAR/DISCRETE SUMVAR = TCES;
   TITLE' FIGURE 2.TREND IN PROPORTION OF AGES 8+ WEAKFISH UNDER CURRENT SIZE
LIMITS)';
    TITLE2'FROM THE COASTAL POPULATION UNDER A FIXED M FROM 1980 TO 2020 ';
        RUN;

/*

PROC CHART DATA= MODEL;
WHERE YEAR> (&FRSTYEAR + &MAXAGE -1);
VBAR YEAR/DISCRETE SUMVAR = TBES;
TITLE'FIGURE 1. SIMULATED REC CATCH (#-RELATIVE UNITS) OF PRE-MIGRANT BASS';
TITLE2'FROM UPPER BAY,1982 TO 2004 UNDER TIME VARYING M FROM 1982 TO 2011';
title3' sex ratio of landings assumed to be 80% males';
RUN;

PROC CHART DATA = MODEL;
```

```
WHERE YEAR> (&FRSTYEAR + &MAXAGE -1);
  VBAR YEAR/DISCRETE SUMVAR = TN_BAY;
   TITLE' FIGURE 2.TREND IN TOTAL STRIPER population in number';
    TITLE2'FROM UPPER BAY MARYLAND FROM 1982 TO 2011 ';
  RUN;

PROC NLIN METHOD = MARQUARDT HOUGAARD DATA= MODEL;
WHERE YEAR> (&FRSTYEAR + &MAXAGE-1);
PARMS A = 0.5, B = -0.0000002;
BOUNDS A>0, B<0;
 RJ= A*TSSB*exp(-b*tssb);
MODEL LREC = LOG(RJ);
OUTPUT OUT = SHEP P =PRED R=RESID;
QUIT;
DATA SHEP;
SET SHEP;
RP = EXP(PRED);
PROC PLOT DATA= SHEP;
PLOT rec*TSSB = 'O' RP*TSSB = 'P'/OVERLAY;
QUIT;
PROC MEANS DATA= MODEL;
VAR JUV1 JUV2 JUV3 JUV4 JUV5 REC;
Run;

title2 "Year Class Statistics-BAY";
proc print data=Debug;
  by Year;
  id Age;
  var N_Bay ZB1 ZB2 BAYF1 BAYF2 BayDeath BayCatch Migrate
      BAYIELD BES SSB;
  sum N_Bay BayDeath BayCatch Migrate BAYIELD
      BES CES  SSB;
  format N_Bay BayDeath BayCatch Migrate
         BAYIELD BES  SSB  9.5;
run;

title2 "Year Class Statistics-COAST";
proc print data=Debug;
  by Year;
  id Age;
  var  N_Cst ZC1 ZC2 CSTF1 CSTF2 CstDeath CstCatch COASTYLD CES  SSB ;
  sum  N_Cst CstDeath CstCatch COASTYLD CES  SSB;
  format  N_Cst CstDeath CstCatch COASTYLD CES SSB  9.5;
run;
```

Appendix C-4

Weakfish
CPUE estimates calculated with GLM and GAM standardizations

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Abstract

Current management of weakfish in the US is hampered by disparities among the various survey catch rate indices that are used for stock assessment. To improve consistency among the indices we calculated standardizations of catch rate based on geographic and environmental data collected in each survey, using generalized linear models (GLM) and generalized additive models (GAM). Results of the analyses showed that GAM gives better fit for modeling weakfish catch rates than GLM. Comparisons by cross-correlation showed that standardization with GAM or GLM improved consistency (positively or negatively) among most surveys when high or low years were correlated. Juvenile weakfish surveys tended to have higher positive correlation than adult surveys.

Introduction

Weakfish (*Cynoscion regalis*) is a commercially and recreationally valuable migratory fish that inhabits the western Atlantic coastal waters from Nova Scotia to northern Florida. Current management of weakfish in the US is hampered by disparities among the survey catch rate indices collected by various state agencies that are included in the stock assessments. The disparities include different gear types that are used, different variables that are measured, and different methods for summarizing data; e.g., arithmetic vs. geometric mean catches. Many spatial, temporal, and environmental factors are known to influence the catch rates of fishes, and standardizing catch rate indices in relation to these factors improves their utility in stock assessments (Harley et al. 2001; Maunder and Punt 2004). Our objective in this paper is therefore to develop standardizations of weakfish catch rate indices using generalized linear and/or generalized additive models, in an effort to increase the consistency of the catch rate indices among different survey databases.

Methods

Catch-per-unit-effort (CPUE) estimates of weakfish are compiled from survey data provided by state, inter-state, and federal agencies along the Atlantic coast. All available survey databases comprise differences in the time range covered, sampling intensity, the gear that was used, and the environmental and geographic variables that were measured along with weakfish catches. Therefore, available survey databases were used opportunistically to calculate weakfish CPUE estimates on the scale of inter-annual (yearly) variation and the scale of annual (usually monthly) variation. CPUE estimates were calculated by numbers of weakfish using four methods: arithmetic mean ($\overline{N}$ where N is the number of weakfish caught per sample for, e.g., a given year in a given survey), geometric mean ($\exp(\overline{\log(N+1)})-1$, Gottschall et al. 2008), generalized linear model GLM (McCullagh and Nelder 1989), and generalized additive model GAM (Wood 2006).

GLM and GAM were calculated using the environmental and geographic variables of the surveys as predictor variables. Geographic coordinates (i.e., latitude and longitude), when available, were particularly important because Moran's Index tests (Moran, 1950) calculated on several survey data sets showed strong spatial autocorrelation among catch samples. Variables that were not significant in the GLM or GAM models (model coefficients $p \geq 0.10$) were removed by backward selection. Variables were also excluded if they were missing from >20% of samples, or reduced the total number of useable samples by >20% (e.g., two variables might be missing from just 15% of samples each, but if they were non-overlapping so as to reduce the total number of useable samples by 30%, then the lesser significant of the two would be excluded). Because of high proportions of zero catches in many surveys, GLM and GAM were calculated using a delta-lognormal approach (Lo et al. 1992), as follows: presence or absence (0/1) of weakfish catch, and catch numbers of only

positive catch, were first analyzed separately by either GLM or GAM. Presence / absence was modeled on the binomial distribution, while positive catches were log-transformed and modeled on the normal distribution. Fit of the log positive catches to the normal distribution was verified by chi-square tests. The year (or month) effects in both the presence/absence and the positive catch models were extracted by setting all other variables to their median (Maunder and Punt 2004) and calculating the sum of variables × their GLM (or GAM) coefficients. The presence / absence proportions and positive catch model predictions were then back-transformed to linear domain, and multiplied together to give relative estimates of yearly (or monthly) average CPUE. GLM and GAM values were scaled to the arithmetic means of each survey. Year and month were always treated as categorical variables in the analyses. Environmental and geographic variables were treated as continuous unless otherwise noted in the survey summaries, below. For each survey, the goodness of fit of the GLM and GAM were compared by the Akaike information criterion (AIC; Akaike 1973).

Results

Catch rate standardization

GLM and GAM yearly standardizations of catch rate indices are summarized in Table 1 for the three coast-wide surveys (NMFS, MRFSS, and SEAMAP), in Table 2 for the adult surveys (RI, NJ, DE, MD and NC), and in Table 3 for juvenile surveys (CT, NY, DE, VIMS, NC and Maryland 'blue crab'). Time series of yearly log-transformed standardized CPUE are also plotted together in Figures 1 and 2.The designation of an 'adult' survey actually means that all age classes were targeted. Each survey is separately described below and all versions of its yearly and monthly indices (arithmetic mean, geometric mean, GLM, GAM) are listed in the Appendix. Numbers of samples in the Appendix tables refer to useable samples, i.e., after data were edited for errors or omissions. Significance values of GLM and GAM variables for each survey, together with R^2 of the models, are also summarized as tables in the Appendix.

NMFS

The NMFS fall survey by the NEFSC includes 1894 samples from 1972 to 2006 (Tables A1 and A2), taken with a Yankee #36 bottom trawl (18 m headrope, 24 m footrope). Samples were geographically referenced by latitude and longitude (range: 34.90 to 41.60° N and 69.52 to 76.07° W), and depth and bottom temperature were measured in the survey as environmental variables. Both depth and bottom temperature were included in the GLM and GAM analyses. However, bottom temperature was not recorded for 328 samples, reducing the total number of data entries analyzed to 1566. Significance values of model variables are given in Table A3.

Log catch rate generally increased from south to north and decreased from west to east (i.e., nearshore to offshore) (Fig. 3).

MRFSS

The MRFS survey includes 13477 samples for weakfish from 1981 to 2007 (Tables A4 and A5). Catch and effort data are collected by telephone interview and angler intercept sampling (see www.st.nmfs.noaa.gov/st1/recreational/overview/overview.html for procedures and data access). Samples, coming from recreational fishers, did not have latitude and longitude data but were referenced to the state where the catch was taken. We used state as an ordinal index from south to north: FL = 1, GA = 2, SC = 3, NC = 4, VA = 5, MD = 6, DE = 7, NJ = 8, NY = 9, CT = 10, RI = 11, MA = 12, NH = 13, ME = 14, and treated this index as a continuous variable in the GLM and

GAM analyses. The approximate distance from shore of the catch was categorized in the MRFS survey as "inland", "ocean ≤3 mi.", "ocean >3 mi.", and in a few cases "ocean ≤10 mi.", or "ocean >10 mi.". We likewise transferred these categories to an ordinal index as distances 1, 2, 3, 6.67, and 10 respectively, and treated this index as a continuous variable. However, for GAM this index had to be excluded because it resulted in too few distinct values for smoothing. The one other variable available for analysis of these data (and used for both GLM and GAM) was the fishing mode, consisting of categories "beach/bank", "man made", "private/rental", "party/charter", "shore", and "charter". Significance values of model variables are given in Table A6.

The along-coast state index showed a parabolic relationship to log catch rate: catch increased south-to-north until approximately the mid-Atlantic, the decreased again further north (Fig. 4).

SEAMAP

The SEAMAP survey database includes 4388 samples from 1990 to 2006 (Tables A7 and A8), collected from tows of paired 22.9-m mongoose-type Falcon trawl nets (ASMFC 2000). Samples ranged from 28.76 to 35.23° N and 75.59 to 81.44° W, and environmental variables included depth, bottom temperature, and bottom salinity. However, bottom temperature was excluded from GLM as non-significant, and both bottom temperature and depth were excluded from GAM as non-significant. Three samples were removed from analyses due to missing data entries. Significance values of model variables are given in Table A9.

Log catch rate increased south-to-north from approximately 31°N to 33°N, and generally decreased west to east (Fig. 5).

RI fall trawl survey

The RI fall trawl survey collected 565 samples from 1979 to 2007 (Tables A10 and A11). This survey did not report any zero catches, and therefore the presence / absence component of the GLM and GAM analyses was not used. Latitude and longitude were not given but data were indexed by area (Narragansett Bay, Rhode Island Sound, Block Island Sound), and 11 strata nested within areas. Numbers of samples per strata were very uneven (6 strata had <10 data, 4 strata had between 15 and 69 data, and one stratum had 389 data). Therefore strata were not used for analysis, but area was used as a categorical variable. Since both other available variables (year and month) were categorical too, only GLM was calculated. Significance values of model variables are given in Table A12.

NJ trawl survey

The New Jersey trawl survey collected 3430 samples from 1988 to 2006 (Tables A13 and A14). Samples were referenced by start latitude and longitude and end latitude and longitude, which were averaged per sample. Average latitude and longitude ranged from 38.52 to 40.48° N and from 73.74 to 75.05° W. Other recorded variables included start depth, end depth, minimum depth, and maximum depth, which were averaged per trawl, and tow duration. Average depth was significant and used in both GLM and GAM analyses, while tow duration was not significant. Significance values of variables in the model are given in Table A15.

Log catch rate increased south to north and decreased west to east (Fig. 6).

DE 30-ft trawl survey

The Delaware 30-ft trawl survey collected 2246 samples from 1966 to 2007. Years 1972-1973, 1975-1978, and 1985-1989 were skipped (Tables A16 and A17). Beginning and end latitude and

longitude were averaged per sample, and average latitude and longitude ranged from 38.80 to 39.39° N and from 75.02 to 75.48° W. Variables depth, tow duration, bottom salinity and bottom temperature were included in both GLM and GAM analyses. Surface salinity, surface temperature, surface DO, and bottom DO were excluded from the analyses as non-significant. Variables of weather, tide, and sea state were not examined. Significance values of model variables are given in Table A18.151 samples were removed from analyses due to missing data entries.

Log catch rate increased and decreased variably with latitude, albeit over a survey range of only 0.6 degrees. Log catch rate decreased with longitude from 75.3°W to 75.1°W (Fig. 7).

MD coastal bay survey

The Maryland coastal bay survey included 956 samples from 1972 to 2007. Samples were taken by beach seine (BCHS), bottom trawl (BTRW), and 25-ft trawl (T25). BTRW accounted for 879 of the samples (92%) and therefore only samples by this method were used, to prevent a confounding gear factor. Bottom trawls were not taken in the years 1974, 1983, and 1988 (only beach seines were taken), and therefore these years are excluded from the analyses (Tables A19 and A20). This survey did not report any zero catches, and therefore the presence / absence component of the GLM and GAM analyses was not used. The 20 different sample sites were the only catch information besides date included in the data file, and were used as a categorical variable. Since only categorical variables were available for analysis, only GLM was used. Significance values of model variables are given in Table A21.

NC Pamlico Sound Gillnet survey

The North Carolina Pamlico Sound survey collected 2142 samples from 2001 to 2007 (Tables A22 and A23). Sample latitude and longitude ranged from 35.05 to 35.82° N and 75.47 to 76.52° W. Environmental variables measured in the surveys included depth, bottom temperature, DO, and salinity, and surface temperature, DO, and salinity. Bottom temperature, bottom DO and surface temperature were not significant and excluded from GLM, while bottom DO and surface temperature were not significant and excluded from GAM. Significance values of variables used in the models are given in Table A24. 45 samples were removed from analyses due to missing data entries.

Log catch rate decreased slightly with latitude increasing from 35.6 to 35.8°N. Log catch rate increased slightly west to east from 76.4°W to 75.8°W, then decreased from 75.8°W to 75.5°W (Fig. 8).

Connecticut trawl survey

The Connecticut trawl survey collected 3492 samples from 1989 to 2008, except 1991 (Tables A25 and A26). Sample latitude and longitude ranged from 40.87 to 41.33° N and from 71.19 to 73.71° W. Environmental variables in the surveys included depth, tow duration, surface temperature, salinity, DO and conductivity, and bottom temperature, salinity, DO and conductivity. Bottom type was also scored in the survey but was not considered for analysis. Bottom DO and conductivity and surface DO and conductivity had too few measures to be included in analyses. Bottom salinity and bottom temperature were not significant and excluded from GLM, while bottom salinity was not significant and excluded from GAM. 294 additional samples were removed from analyses due to missing data entries in the environmental variables. Significance values of variables used in the models are given in Table A27.

Log catch rate varied intermittently with latitude over the range of the survey (40.8°N to

41.4°N). Log catch rate first increased slightly then decreased with longitude west to east (Fig. 9).

NY juvenile trawl survey

The New York juvenile trawl survey collected 8092 samples from 1987 to 2007 (Tables A28 and A29). Latitude and longitude were not included in the data file, but samples were referenced to 77 stations. No other environmental variables were included in the data file. Since only the categorical variables year, month, and station were available, only GLM was calculated. Significance values of model variables are given in Table A30.

DE 16-ft trawl survey

The Delaware 16-ft trawl survey, targeted on juvenile fish, includes 7228 samples from 1980 to 2007 (Tables A31 and A32). Beginning latitude and longitude and ending latitude and longitude were averaged per sample. Average latitude and longitude ranged from 38.88 to 39.75° N and 75.21 to 75.60° W. Variables included depth, tow duration, surface temperature, surface salinity, and surface dissolved oxygen (DO). Surface DO and surface salinity were not significant and excluded from GLM, while depth, duration, and surface DO were not significant and excluded from GAM. 757 samples were removed from analyses due to missing data entries. Significance values of model variables are given in Table A33.

Log catch rate decreased south to north from 39.3°N to 39.5°N, and decreased west to east from 75.5°W to 75.4°W (Fig. 10).

VIMS y-o-y trawl survey

The VIMS young-of-year trawl survey collected 20877 samples from 1988 to 2007 (Tables A34 and A35). Sample latitude and longitude ranged from 36.85 to 38.19° N and from 75.73 to 76.98° W. Variables included trawl depth, Secchi depth (water transparency), temperature, salinity and DO, and, as a categorical variable, the river the sample was taken from. Salinity and Secchi depth were not significant and were excluded from GLM and GAM analyses. 1043 samples were removed from analyses due to missing data entries. Significance values of model variables are given in Table A36.

Log catch rate increased and decreased variably with latitude, and generally decreased with longitude over the range of the survey (Fig. 11).

NC juvenile trawl survey

The North Carolina juvenile trawl survey collected 1685 samples from 1987 to 2007 (Tables A37 and A38). Sample latitude and longitude ranged from 34.95 to 36.08° N and from 75.52 to 76.97° W. This survey did not report any zero catches, and therefore the presence / absence component of the GLM and GAM analyses was not used. Survey variables included depth, surface temperature, DO and salinity, and bottom temperature, DO and salinity. Bottom temperature, surface temperature and surface salinity, as well as latitude, were not significant and excluded from the GLM analysis. Depth, bottom temperature, surface temperature and surface salinity were excluded from the GAM analysis. Significance values of model variables are given in Table A39.

Log catch rate peaked at the intermediate latitude over the range of the survey (approx. 35°N to 36.1°N). Log catch rate increased with longitude from 77°W to 75.8°W, then decreased with longitude from 75.8°W to 75.5°W (Fig. 12).

Maryland Chesapeake "blue crab" survey

The Maryland "blue crab" survey collected 1190 samples from 1989 to 2008 (Tables A40 and A41). Latitude and longitude were not included in the data file, but samples were assigned to nine areas. The areas were treated as categorical variables. The survey also recorded wind speed, salinity, depth and temperature measurements. However, all combinations of these environmental variables were either not significant or excluded too many samples through missing data. Only the categorical variables year, month, and area were analyzed, and therefore only GLM was calculated. The survey had too few zero catches (<7%) to derive a presence / absence model that converged, and therefore only a single CPUE model in the normal distribution was calculated. Catches were log-transformed as log(N + 1) instead of log(N) to include the zero catches that were recorded. Significance values of model variables are given in Table A42.

Comparison between GLM and GAM models

AIC values calculated from GAM were lower than AIC values from GLM for all survey CPUE indices in which the two models could be compared, except one (Delaware 16 ft. trawl; Table 4). As noted for some individual surveys, above, GAM cannot be calculated when only categorical data are available to include in the model. In all cases in Table 4, the presence/absence models were more improved by using GAM vs. GLM than the positive catch models (for example, for the NMFS survey the ratio of GLM over GAM AIC for presence/absence is 11998.2/10206.5 ≅ 1.18; an improvement of 18%, the ratio of GLM over GAM AIC for positive catch is 12433.3/11628.0 ≅ 1.07; an improvement of 7%). The results suggest that GAM is better than GLM for modeling catch rates of weakfish. Therefore GAM was used in the following consistency analyses.

Consistency among indices

Log catch rate generally increased south-to-north and decreased west-to-east over surveys spanning at least 2° of latitude or longitude respectively. Spatially more restricted surveys were more variable.

Cross-correlation analyses of yearly CPUE indices showed no consistent pattern of positive or negative correlation among adult surveys, for either geometric mean or GAM/GLM standardized CPUE indices. GAM/GLM standardized CPUEs more frequently showed negative correlations than geometric mean CPUEs, especially in the SEAMAP and Maryland surveys. Most correlations were not very strong, with only 4 geometric correlations and 3 GAM/GLM correlations > ±0.5 (Tables 5 and 6). However, pair-wise plots of the adult surveys' CPUE indices suggest that correlations are stronger when only salient years are considered, i.e., those years in which catches were higher or lower than usual. Figure 13 shows the geometric and GAM/GLM pair-wise plots of the NMFS survey vs. the other adult surveys; in effect the data corresponding to the first row or column of Tables 5 and 6. On each plot a subsample was selected by eye, consisting of a block of the data that were highest or lowest on the x-axis and highest or lowest on the y-axis. These data are shaded in black on each plot and fitted with a linear regression. Subsamples were selected separately for the geometric and GAM/GLM indices of each survey pair, but with the constraint that they had to have the same number of data, and could only be selected by making one 'cut' along each axis. For example (Figure 13, top row), low NMFS CPUE co-occurred with any size of MRFSS CPUE, but high NMFS CPUE only co-occurred with low MRFSS CPUE. The linear regressions indicate that indices of salient years correlated more strongly with GAM/GLM standardization than with geometric mean for the NMFS survey vs. MRFSS, SEAMAP, RI, NJ, and DE, and less strongly or indifferently for the NMFS survey vs. MD and NC. Among all adult survey pairs, GAM/GLM indices of salient years correlated more strongly than geometric indices of salient years in 14 of 28

comparisons, and correlated more poorly in 4 comparisons (Tables 7 and 8). More strongly correlated GAM/GLM indices occurred especially in the NMFS and MRFSS surveys.

Among juvenile surveys, cross-correlations were positive for geometric indices in 13 of 15 comparisons, and positive for GAM/GLM standardized indices in 14 of 15 comparisons (Tables 9 and 10).GAM/GLM standardized indices were more strongly positive than the corresponding geometric indices in 10 of the 15 comparisons.

Discussion

The 14 surveys analyzed in this study presented a high diversity in their patterns of inter-annual CPUE variability. Coast-wide and adult surveys showed relatively little consistency in their tendencies to be positively or negatively correlated with each other, for both geometric mean and GAM/GLM standardized indices. When only years of higher or lower than usual CPUE were considered, GAM/GLM correlations were typically stronger than geometric mean correlations, suggesting that exceptionally strong or weak yearly catches occur in relation to geographic and environmental conditions, while average yearly catches don't. Of the three broadest surveys, NMFS and SEAMAP were positively correlated with each other over salient (high/low) years, while both were negatively correlated with MRFSS. Although it can't be verified from the available data, a possible explanation for this pattern is that weakfish were further inshore some years and more likely to be taken in the recreational fishery, while further offshore in other years and more likely to be captured in the surveys.

Juvenile surveys showed much more consistently positive cross-correlations than adult surveys. This suggests that the inconsistency among adult surveys may largely be due to variability in the age class distributions that different surveys – using different equipment – capture. In addition, some of the state-wide surveys that are more restricted to nearshore embayments may have captured local subpopulations (e.g., Thorrold et al. 2001) that are more distinct from each other than the overall weakfish stock. More detailed analyses of size/age distributions within the different surveys will likely be useful in resolving these questions.

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Table 1. Summary of yearly GLM and GAM CPUE standardizations for coast-wide weakfish surveys.

Year	NMFS		MRFSS		SEAMAP	
	GLM	GAM	GLM	GAM	GLM	GAM
1972	45.696	160.871				
1973	134.668	95.469				
1974	79.345	62.627				
1975	93.467	120.150				
1976	208.078	252.409				
1977	104.461	89.489				
1978	125.966	162.478				
1979	132.597	160.136				
1980	190.607	115.854				
1981	75.968	82.151	0.098	0.148		
1982	136.621	83.983	0.045	0.064		
1983	103.618	123.044	0.101	0.136		
1984	298.990	265.316	0.182	0.089		
1985	292.836	201.726	0.236	0.108		
1986	131.395	108.814	0.207	0.173		
1987	29.230	28.933	0.145	0.131		
1988	14.217	10.200	0.115	0.080		
1989	145.636	133.904	0.078	0.066		
1990	121.816	83.889	0.071	0.054	16.558	44.204
1991	87.236	64.502	0.080	0.073	10.587	34.687
1992	49.265	42.708	0.056	0.053	9.514	35.278
1993	34.159	43.714	0.102	0.082	4.268	17.896
1994	216.526	190.233	0.129	0.143	4.980	20.190
1995	388.507	384.356	0.123	0.131	8.612	30.480
1996	176.093	185.752	0.199	0.169	11.245	35.217
1997	46.998	35.781	0.168	0.183	5.431	20.529
1998	54.462	47.842	0.161	0.179	8.195	32.341
1999	342.856	340.655	0.121	0.174	6.143	22.384
2000	214.908	191.984	0.166	0.196	11.188	33.146
2001	93.215	105.926	0.102	0.120	17.469	46.824
2002	204.121	240.488	0.102	0.115	4.637	20.967
2003	707.552	593.502	0.043	0.050	11.559	40.473
2004	255.548	263.958	0.098	0.115	13.528	41.899
2005	241.607	256.965	0.079	0.109	7.881	32.597
2006	600.830	849.290	0.052	0.091	10.070	38.306
2007			0.033	0.062		

Table 2. Summary of yearly GLM and GAM CPUE standardizations for states' adult weakfish surveys.

Year	RI	NJ		DE		MD	NC	
	GLM	GLM	GAM	GLM	GAM	GLM	GLM	GAM
1966				246.840	121.522			
1967				136.597	93.710			
1968				63.825	81.686			
1969				169.587	144.138			
1970				234.109	152.384			
1971				368.205	210.422			
1972						4.604		
1973						5.698		
1974				64.298	27.112			
1975						52.877		
1976						86.563		
1977						4.564		
1978						29.689		
1979	12.145			26.813	51.640	11.244		
1980	46.050			17.057	37.644	22.791		
1981	58.100			13.978	32.445	104.490		
1982	57.910			36.012	54.971	39.662		
1983	16.858			32.232	60.836			
1984	38.980			23.216	24.105	5.391		
1985	34.956					7.583		
1986	11.610					50.215		
1987	19.358					156.013		
1988	19.568	81.915	8.351			9.765		
1989	10.137	44.501	30.721			15.357		
1990	64.315	26.260	7.088	19.419	39.224	25.501		
1991	34.391	49.852	32.367	74.751	57.253	51.628		
1992	31.508	22.886	30.354	17.617	43.255	31.418		
1993	26.020	28.200	23.772	80.430	99.656	21.295		
1994	87.130	53.636	54.054	49.284	82.335	16.630		
1995	9.295	101.073	153.508	181.130	199.226	16.672		
1996	165.436	148.785	277.406	237.120	293.148	6.487		
1997	108.072	141.378	114.073	140.978	198.421	4.512		
1998	29.996	19.049	21.592	316.731	196.982	9.783		
1999	20.648	37.108	39.913	164.496	245.189	18.591		
2000	32.200	39.814	45.519	125.841	260.535	11.708		
2001	58.833	53.517	31.719	149.125	163.924	21.899	2.678	1.873
2002	33.456	197.973	195.490	204.106	116.800	9.294	1.342	1.468
2003	246.854	92.913	65.222	93.277	142.299	5.881	1.018	1.420
2004	19.180	245.019	220.879	59.774	81.985	8.068	1.058	1.163
2005	105.235	131.107	152.807	32.859	45.505	5.760	1.167	1.292
2006	6.941	22.579	32.730	82.233	105.427	14.340	0.891	0.905
2007	78.251			103.979	91.555		0.506	0.539
2008								

Table 3. Summary of yearly GLM and GAM CPUE standardizations for states' juvenile weakfish surveys.

Year	CT		NY	DE		MD(blue crab)	VIMS		NC	
	GLM	GAM	GLM	GLM	GAM	GLM	GLM	GAM	GLM	GAM
1980				12.116	9.889					
1981				19.766	17.541					
1982				24.776	24.957					
1983				13.530	12.612					
1984				18.437	28.226					
1985				2.249	24.066					
1986				28.425	27.733					
1987			3.904	20.623	18.753				13.669	13.430
1988			0.977	18.167	22.226		14.276	17.422	17.277	16.481
1989	1.348	0.609	0.152	23.922	24.670	10.276	16.201	11.433	11.305	11.776
1990	0.628	0.554	0.201	25.242	23.848	14.574	14.377	8.567	17.817	15.807
1991			28.780	38.577	40.395	19.394	9.363	8.318	9.872	9.387
1992	0.982	0.585	6.328	40.047	43.430	30.694	15.450	21.378	19.690	18.275
1993	0.836	0.652	1.432	38.966	47.640	22.144	12.418	13.834	14.354	15.324
1994	1.125	1.240	16.677	54.135	61.629	23.681	12.342	13.859	18.676	19.551
1995	0.377	0.460	6.622	45.430	47.161	53.642	16.084	17.908	19.755	19.520
1996	0.760	0.883	56.713	36.791	38.679	46.622	13.819	14.767	23.117	23.806
1997	1.755	1.198	1.705	47.419	52.432	44.013	12.915	15.169	24.530	25.922
1998	0.283	0.914	1.103	32.846	37.557	38.454	10.304	9.545	28.397	27.731
1999			51.880	33.377	36.227	52.043	15.731	17.293	30.380	32.394
2000	1.678	1.672	100.122	63.135	50.196	56.037	16.717	22.554	26.094	26.742
2001	1.552	1.407	42.470	21.618	20.362	69.271	22.344	19.102	14.857	15.245
2002	1.316	0.986	95.765	20.555	18.364	32.382	15.444	14.499	12.362	11.846
2003	0.845	2.001	47.478	31.441	31.460	32.456	11.634	13.983	12.803	13.024
2004	1.181	1.298	6.749	26.766	29.277	15.945	10.504	8.708	18.064	17.821
2005	1.114	1.020	10.111	52.974	49.500	32.193	8.801	8.766	15.963	15.160
2006	0.335	0.440	5.206	17.794	17.814	7.629	13.868	15.705	22.869	23.041
2007	1.530	1.541	105.074	43.132	38.389	5.307	14.628	13.584	18.849	18.415
2008	0.060	0.337				8.355				

Table 4. Comparison of AIC from GLM and GAM for surveys in which both models were used. The lowest AIC by row (for positive catch or presence/absence) is highlighted in yellow.

Surveys	GAM		GLM	
	Positive catch	Present/Absent	Positive catch	Present/Absent
MRFSS	11628.0	10206.5	12433.3	11998.2
NMFS	3495.9	1866.2	4204.8	2026.7
SEAMAP	7580.2	5260.9	7695.5	5741.7
NJ	4351.8	2683.6	4459.0	2895.0
DE 30	5368.0	1358.4	5466.9	1429.9
CT	2547.5	2028.3	2576.0	2171.8
DE 16	14766.8	4216.0	14680.9	4343.2
VIMS	27277.8	15451.7	27541.2	16781.5
NC juv.	4764.1	*	4837.9	*
NC gill.	1902.0	2414.4	1928.0	2524.7

Table 5. Cross-correlation of annual CPUEs of coastal wide and adult weakfish surveys averaged by geometric mean. Negative correlations are highlighted in yellow.

NMFS	MRFSS	SEAMAP	RI	NJ	DE	MD	NC
1	-0.238	0.235	0.125	0.202	0.202	0.470	-0.600
-0.238	1	-0.209	-0.068	0.154	0.530	0.014	0.805
0.235	-0.209	1	0.414	0.148	0.088	0.265	0.189
0.125	-0.068	0.414	1	0.280	0.290	0.233	0.029
0.202	0.154	0.148	0.280	1	0.491	0.065	-0.278
0.202	0.530	0.088	0.290	0.491	1	0.003	0.374
0.470	0.014	0.265	0.233	0.065	0.003	1	0.655
-0.600	0.805	0.189	0.029	-0.278	0.374	0.655	1

Table 6. Cross-correlation of annual CPUEs of coast-wide and adult weakfish surveys standardized by GAM, or GLM if GAM was not appropriate. Negative correlations are highlighted in yellow.

NMFS	MRFSS	SEAMAP	RI	NJ	DE	MD	NC
1	-0.145	0.206	0.278	0.157	0.226	-0.254	-0.597
-0.145	1	-0.383	-0.091	0.322	0.688	0.074	0.495
0.206	-0.383	1	0.152	-0.054	-0.159	-0.195	0.153
0.278	-0.091	0.152	1	0.294	0.213	-0.158	0.180
0.157	0.322	-0.054	0.294	1	0.259	-0.172	-0.170
0.226	0.688	-0.159	0.213	0.259	1	-0.334	0.544
-0.254	0.074	-0.195	-0.158	-0.172	-0.334	1	0.155
-0.597	0.495	0.153	0.180	-0.170	0.544	0.155	1

Table 7. Cross-correlation of salient annual CPUEs of coastal wide and adult weakfish surveys averaged by geometric mean. Negative correlations are highlighted in yellow.

NMFS	MRFSS	SEAMAP	RI	NJ	DE	MD	NC
1	-0.769	0.787	0.747	-0.590	-0.598	0.980	0.599
-0.769	1	-0.589	-0.515	0.590	0.692	-0.898	0.898
0.787	-0.589	1	0.887	0.642	0.748	0.609	0.295
0.747	-0.515	0.887	1	-0.393	0.627	0.875	0.063
-0.590	0.590	0.642	-0.393	1	0.710	-0.623	0.680
-0.598	0.692	0.748	0.627	0.710	1	0.152	0.528
0.980	-0.898	0.609	0.875	-0.623	0.152	1	0.941
0.599	0.898	0.295	0.063	0.680	0.528	0.941	1

Table 8. Cross-correlation of salient annual CPUEs of coast-wide and adult weakfish surveys standardized by GAM, or GLM if GAM was not appropriate. Negative correlations are highlighted in yellow.

NMFS	MRFSS	SEAMAP	RI	NJ	DE	MD	NC
1	-0.839	0.908	0.867	-0.913	-0.979	-0.960	-0.597
-0.839	1	-0.611	-0.750	0.932	0.890	-0.912	0.896
0.908	-0.611	1	0.884	0.754	0.755	-0.399	0.595
0.867	-0.750	0.884	1	-0.363	0.625	-0.516	0.258
-0.913	0.932	0.754	-0.363	1	0.606	-0.619	0.404
-0.979	0.890	0.755	0.625	0.606	1	-0.671	0.805
-0.960	-0.912	-0.399	-0.516	-0.619	-0.671	1	0.962
-0.597	0.896	0.595	0.258	0.404	0.805	0.962	1

Table 9. Cross-correlation of annual CPUEs of juvenile weakfish surveys averaged by geometric mean. Negative correlations are highlighted in yellow.

CT	NY	DE	MD (blue crab)	VIMS	NC
1	0.364	0.149	0.093	0.182	-0.346
0.364	1	0.070	0.040	0.202	0.312
0.149	0.070	1	0.065	-0.064	0.325
0.093	0.040	0.065	1	0.478	0.458
0.182	0.202	-0.064	0.478	1	0.267
-0.346	0.312	0.325	0.458	0.267	1

Table 10. Cross-correlation of annual CPUEs of juvenile weakfish surveys standardized by GAM or GLM if GAM was not appropriate. Negative correlations are highlighted in yellow.

CT	NY	DE	MD (blue crab)	VIMS	NC
1	0.445	0.132	0.160	0.020	-0.248
0.445	1	0.007	0.217	0.323	0.113
0.132	0.007	1	0.195	0.091	0.336
0.160	0.217	0.195	1	0.583	0.412
0.020	0.323	0.091	0.583	1	0.359
-0.248	0.113	0.336	0.412	0.359	1

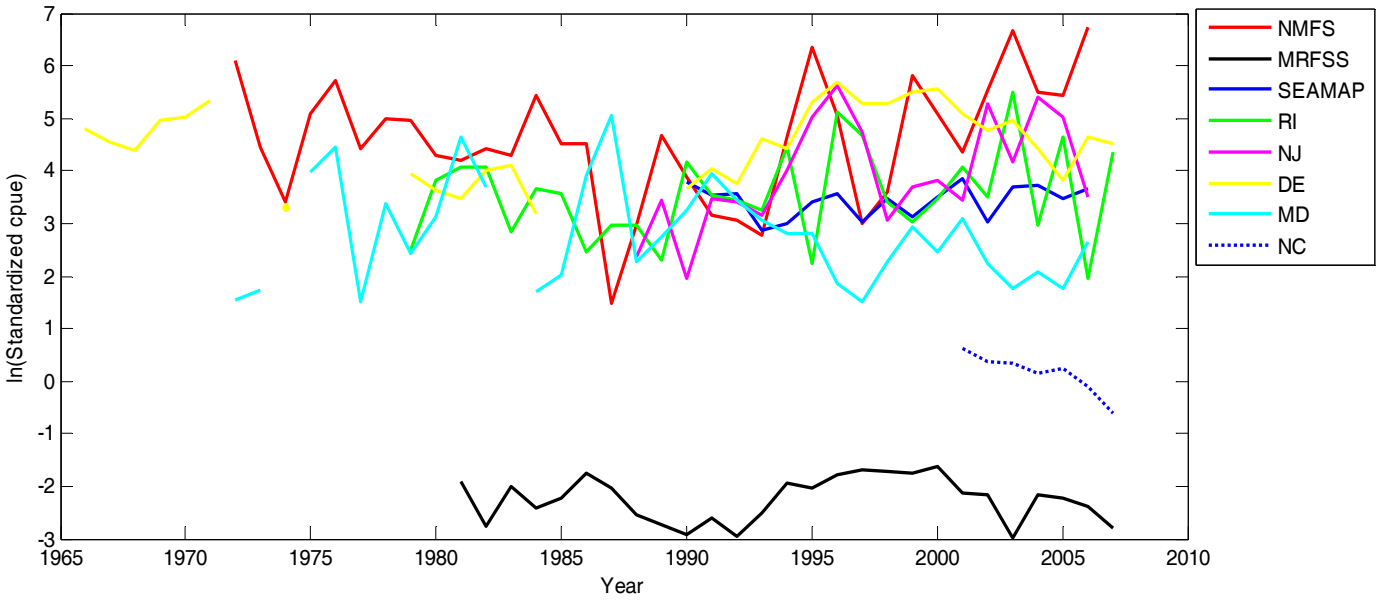

Figure 1. Summary of the standardized abundance indices (coast-wide and state adult surveys).

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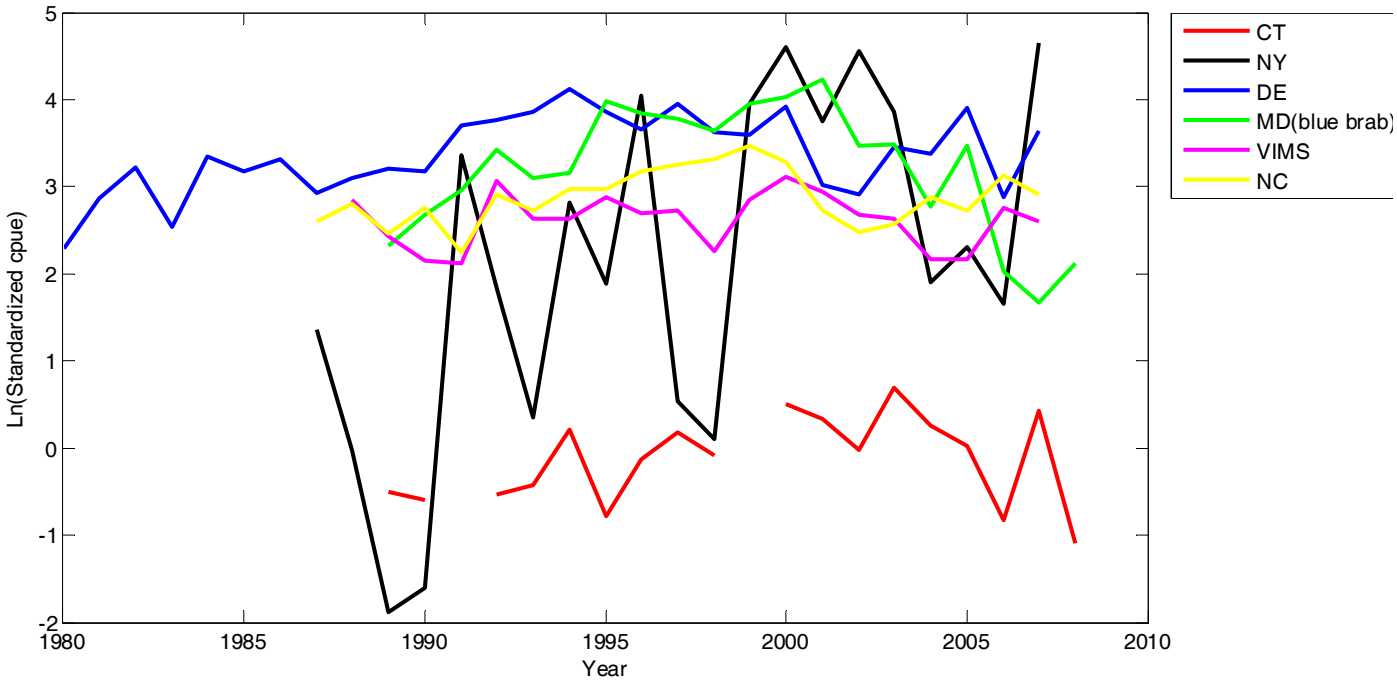

Figure 2. Summary of the standardized abundance indices (state juvenile surveys)

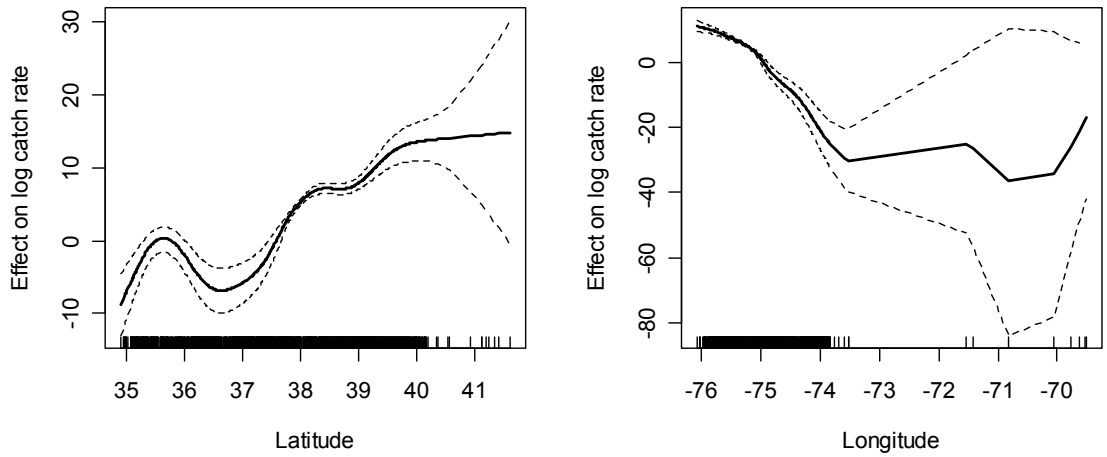

Figure 3. NMFS survey: Effect of latitude and longitude on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

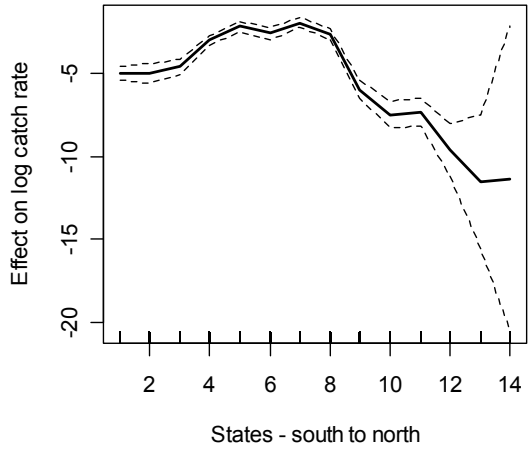

Figure 4. MRFS survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

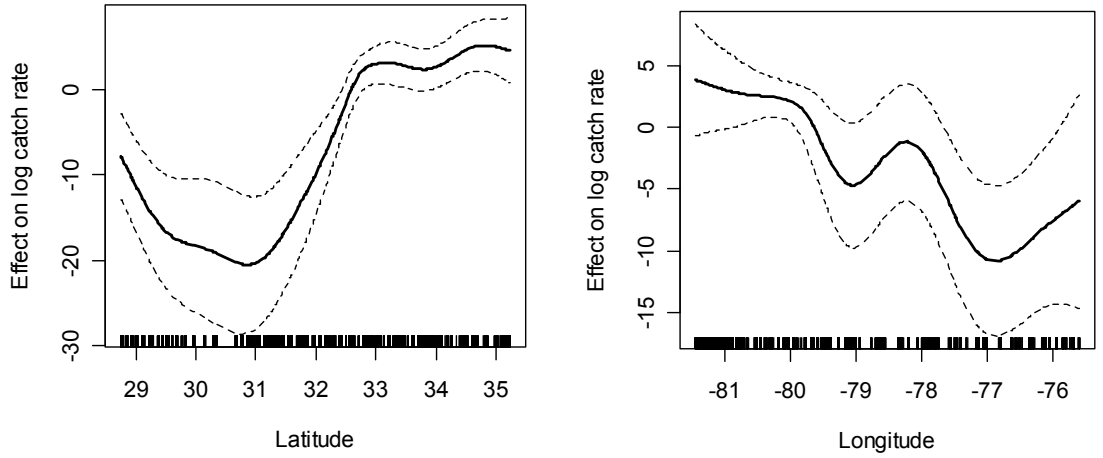

Figure 5. SEAMAP survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals

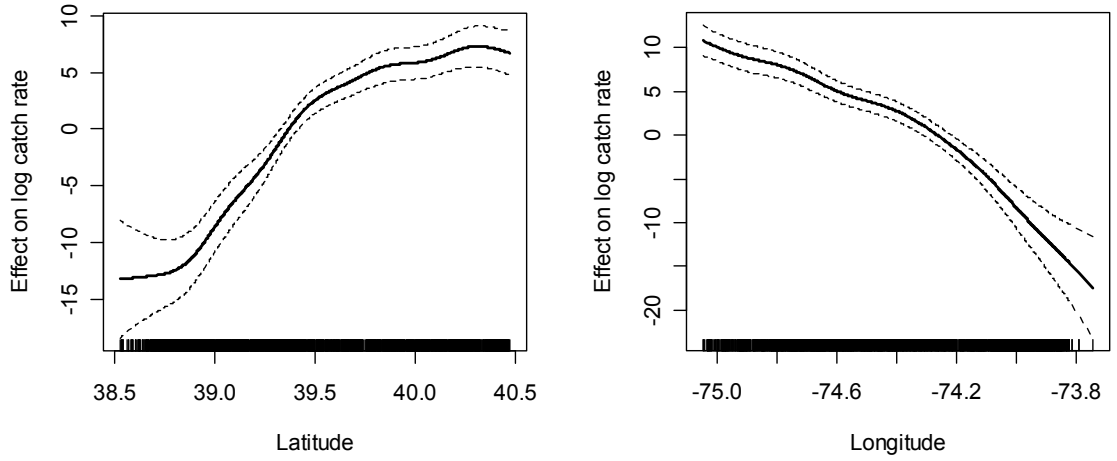

Figure 6. New Jersey survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

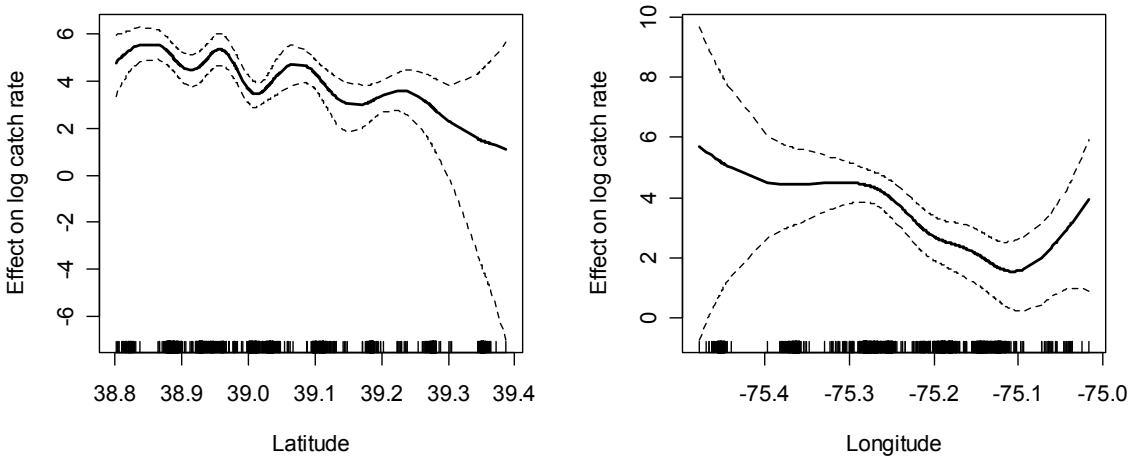

Figure 7. Delaware 30 ft. trawl survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

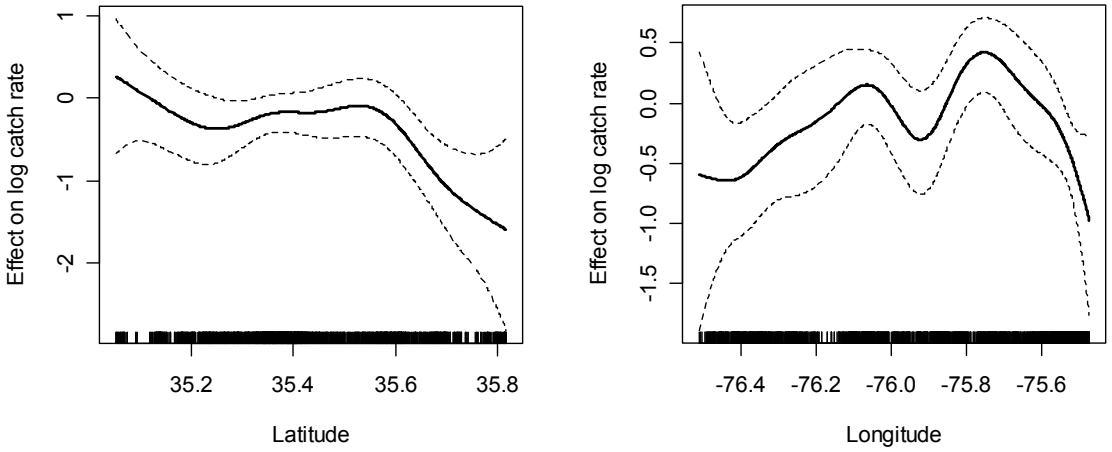

Figure 8. North Carolina gillnet survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

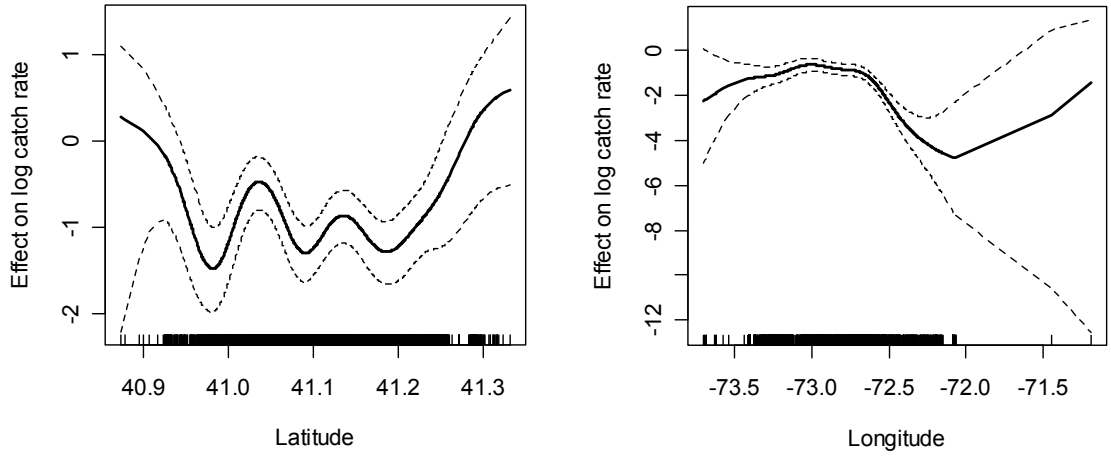

Figure 9. Connecticut survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

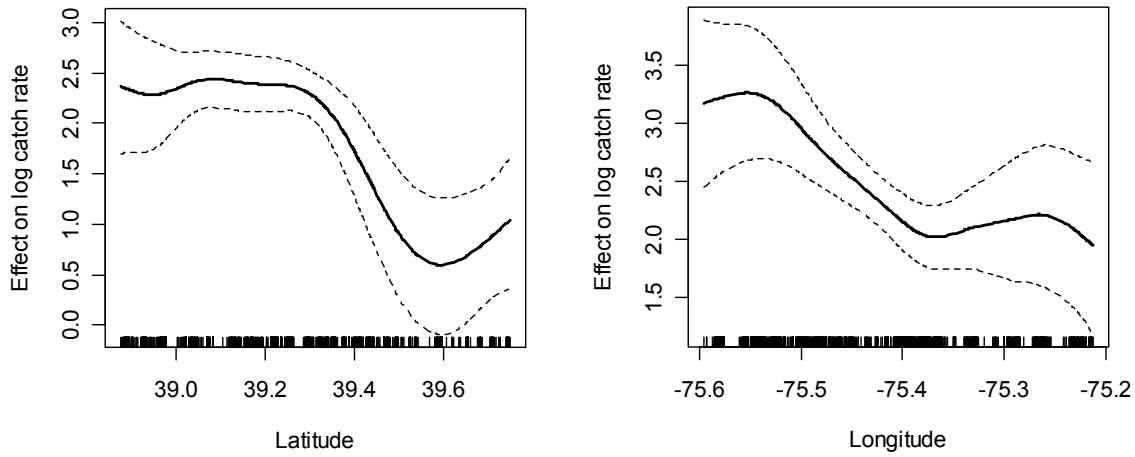

Figure 10. Delaware 16 ft. trawl survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

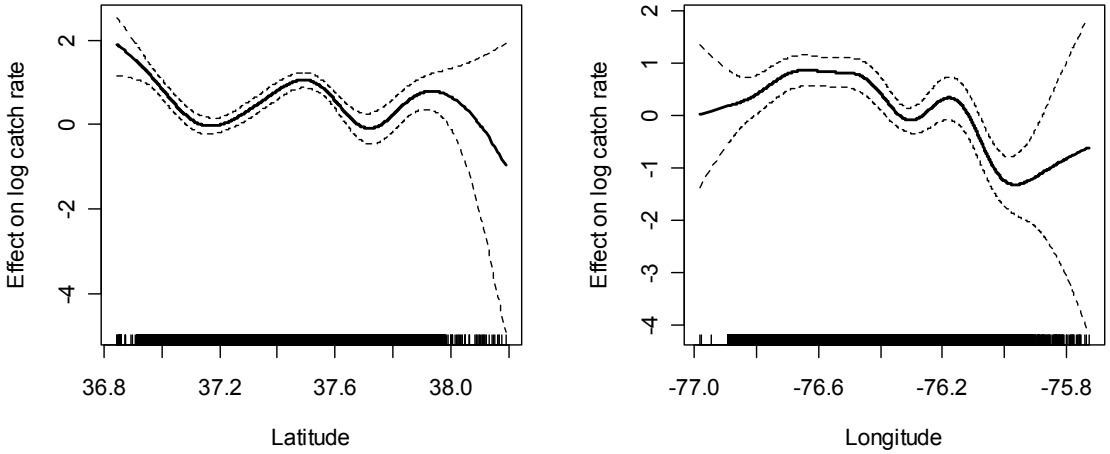

Figure 11. VIMS survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

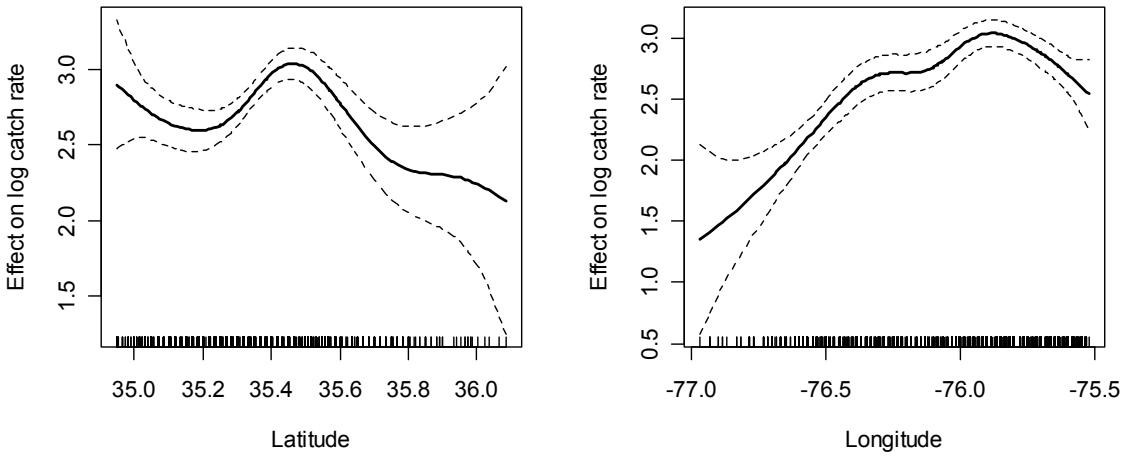

Figure 12. North Carolina juvenile survey: Effect of the along-coast state index on log catch rate. Dotted lines around the trend line represent the 95% confidence intervals.

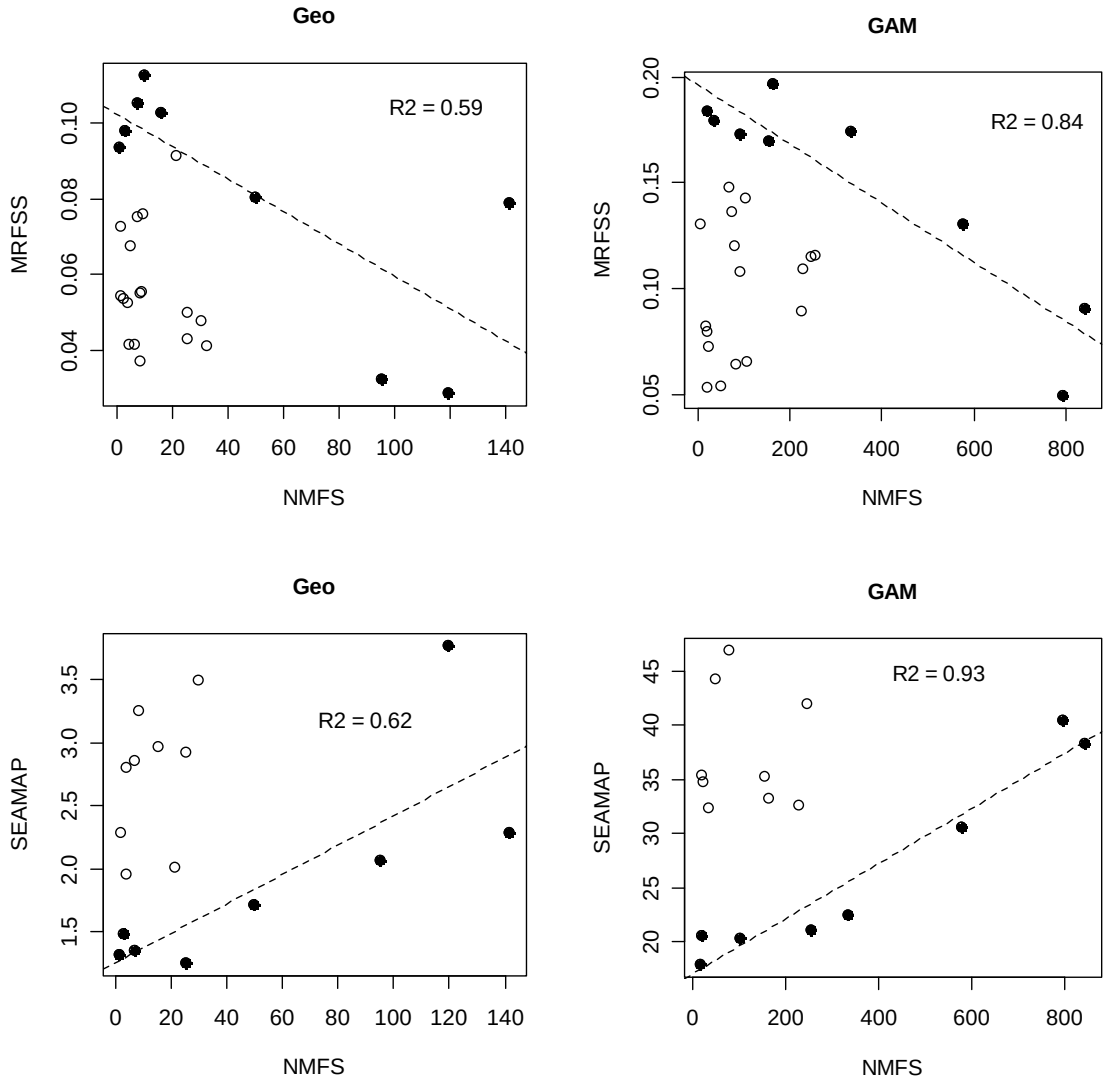

Figure 13. Pairwise plots of NMFS CPUE indices vs . those of other adult surveys, with black shading indicating the salient (high or low) yearly indices that are most likely to have significant correlations (shown as dotted lines).

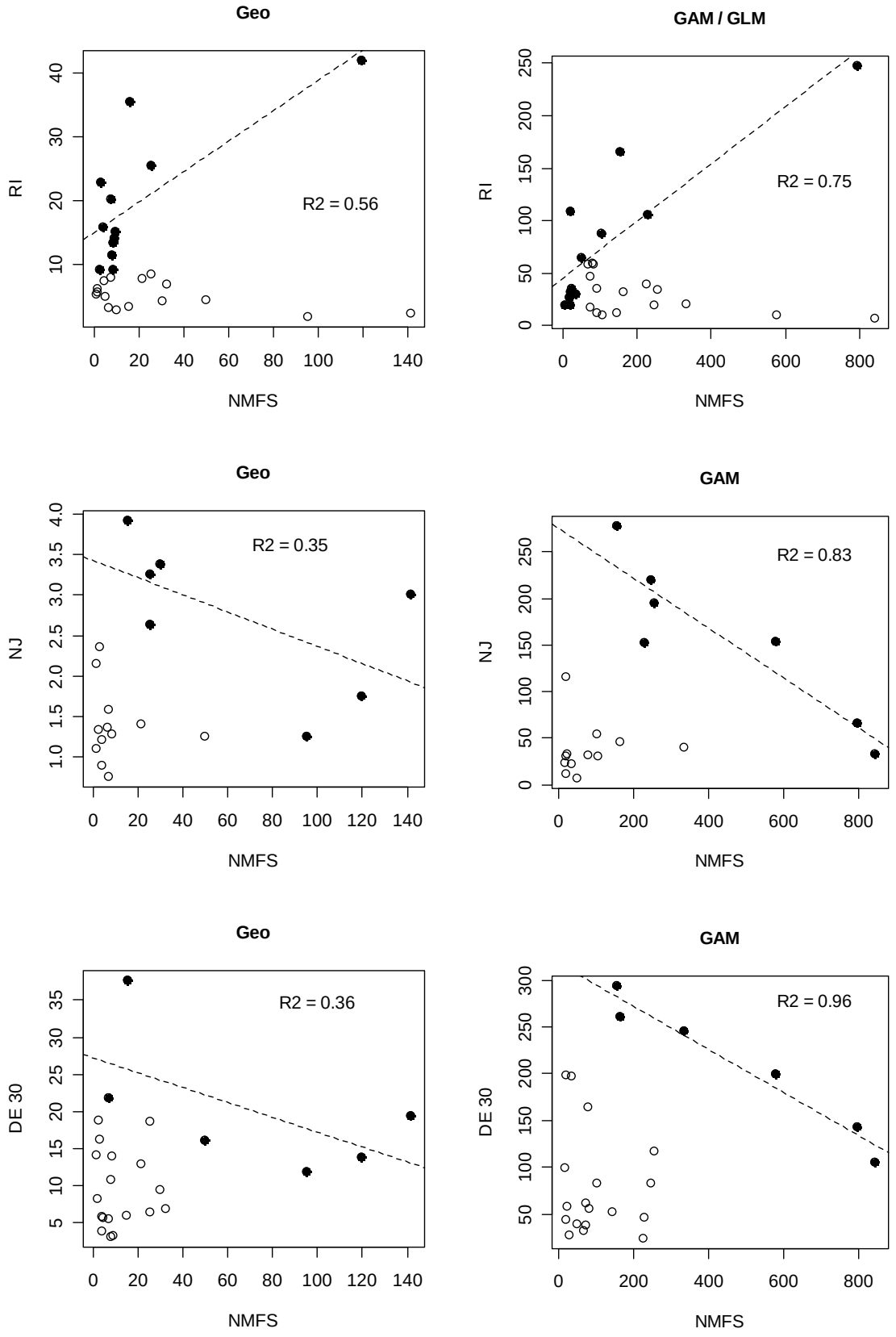

Figure 13 cont.

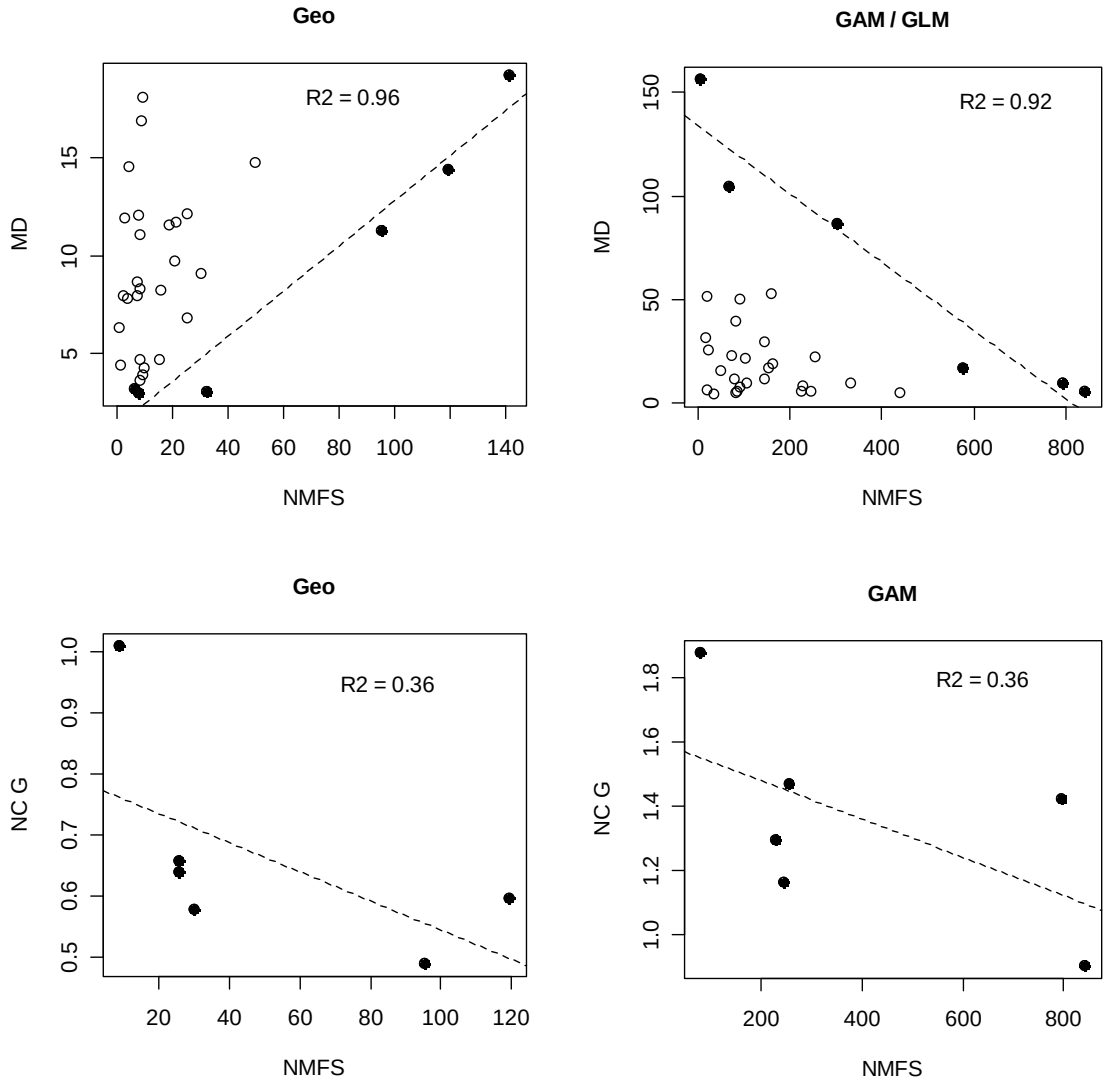

Figure 13 cont.

Appendix.

Table A1. NMFS survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1972	80	57.438	8.592	45.69604	160.8706
1973	72	86.653	7.797	134.6679	95.46888
1974	55	149.891	2.520	79.34548	62.62724
1975	63	59.333	8.419	93.46729	120.1499
1976	49	187.388	20.770	208.0777	252.4089
1977	48	196.167	9.291	104.4608	89.48942
1978	49	296.959	19.023	125.9659	162.4776
1979	47	160.681	15.126	132.5966	160.1355
1980	37	105.459	8.001	190.6066	115.854
1981	50	191.280	9.102	75.96795	82.15111
1982	32	39.313	8.250	136.6211	83.98266
1983	53	55.604	4.663	103.618	123.0438
1984	43	210.209	32.264	298.9904	265.3157
1985	23	30.391	8.129	292.8357	201.726
1986	36	54.833	9.888	131.3955	108.814
1987	13	4.077	0.610	29.23047	28.93322
1988	14	131.214	1.282	14.21662	10.19952
1989	18	47.111	6.323	145.6362	133.9036
1990	18	176.722	3.969	121.816	83.88865
1991	55	90.473	2.283	87.23552	64.50227
1992	31	111.677	4.125	49.26488	42.70775
1993	51	13.451	1.369	34.1585	43.71365
1994	52	141.135	7.231	216.5262	190.2331
1995	40	460.200	141.957	388.5071	384.3564
1996	47	168.915	15.652	176.0929	185.7516
1997	51	37.078	2.903	46.99831	35.78076
1998	47	123.872	7.162	54.46245	47.84231
1999	49	384.612	49.758	342.8563	340.655
2000	51	374.176	21.311	214.9083	191.9837
2001	52	154.788	8.679	93.21527	105.926
2002	53	328.717	25.609	204.1207	240.488
2003	48	586.625	119.729	707.5523	593.5019
2004	44	266.136	30.171	255.5483	263.9581
2005	47	329.383	25.628	241.6068	256.9651
2006	48	290.625	95.527	600.8297	849.2897

Table A2. NMFS survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
9	1113	189.801	70.755	47.367	44.103
10	326	193.699	69.682	34.166	42.889
11	121	40.471	27.985	34.468	52.790
12	6	458.167	187.079	742.952	719.171

Table A3. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the NMFS CPUE. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		< 0.001	< 0.001	< 0.001	< 0.001
Latitude		< 0.001	< 0.001	0.002	0.425
Depth		< 0.001	0.004	0.043	< 0.001
Bottom temperature		< 0.001	< 0.001	0.635	0.485
Month	9	0.016	0.310	0.026	0.299
	10	0.012	0.323	0.010	0.466
	11	0.004	0.074	0.002	0.191
	12	n/a	n/a	n/a	n/a
Year	1972	0.038	0.407	0.003	0.310
	1973	< 0.001	< 0.001	0.005	< 0.001
	1974	< 0.001	< 0.001	< 0.001	< 0.001
	1975	0.006	0.008	0.014	0.021
	1976	0.015	0.025	0.047	0.126
	1977	< 0.001	0.003	< 0.001	0.022
	1978	< 0.001	0.022	< 0.001	0.041
	1979	< 0.001	0.003	< 0.001	0.015
	1980	< 0.001	< 0.001	0.002	< 0.001
	1981	< 0.001	0.002	< 0.001	0.012
	1982	< 0.001	0.147	0.002	0.989
	1983	< 0.001	< 0.001	< 0.001	< 0.001
	1984	0.003	0.129	0.097	0.418
	1985	0.002	0.048	0.152	0.473
	1986	< 0.001	< 0.001	< 0.001	0.008
	1987	< 0.001	< 0.001	< 0.001	< 0.001
	1988	< 0.001	< 0.001	< 0.001	< 0.001
	1989	< 0.001	< 0.001	0.002	0.003
	1990	< 0.001	< 0.001	< 0.001	< 0.001
	1991	< 0.001	< 0.001	< 0.001	< 0.001
	1992	< 0.001	< 0.001	< 0.001	< 0.001
	1993	< 0.001	< 0.001	< 0.001	< 0.001
	1994	< 0.001	< 0.001		< 0.001
	1995	0.031	0.877		0.671
	1996	< 0.001	0.002		0.011
	1997	< 0.001	< 0.001		< 0.001
	1998	< 0.001	< 0.001		< 0.001
	1999	0.018	0.136		0.326
	2000	< 0.001	0.001		0.006
	2001	< 0.001	< 0.001		< 0.001
	2002	0.001	0.023		0.059
	2003	0.454	0.679		0.799
	2004	0.004	0.011		0.058
	2005	0.003	0.013		0.033
	2006	n/a	n/a	n/a	n/a
R^2		0.380	0.283	0.251	0.219
Composite R^2		0.242		0.181	

Table A4. MRFSS yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1981	462	0.162	0.076	0.098	0.148
1982	498	0.081	0.037	0.045	0.064
1983	496	0.122	0.068	0.101	0.136
1984	537	0.063	0.041	0.182	0.089
1985	552	0.092	0.055	0.236	0.108
1986	475	0.288	0.113	0.207	0.173
1987	451	0.184	0.094	0.145	0.131
1988	441	0.148	0.073	0.115	0.080
1989	467	0.054	0.042	0.078	0.066
1990	468	0.099	0.053	0.071	0.054
1991	465	0.091	0.054	0.080	0.073
1992	457	0.068	0.041	0.056	0.053
1993	451	0.090	0.054	0.102	0.082
1994	485	0.198	0.075	0.129	0.143
1995	478	0.139	0.079	0.123	0.131
1996	467	0.179	0.103	0.199	0.169
1997	481	0.162	0.098	0.168	0.183
1998	474	0.191	0.105	0.161	0.179
1999	476	0.124	0.080	0.121	0.174
2000	483	0.150	0.091	0.166	0.196
2001	486	0.080	0.055	0.102	0.120
2002	488	0.066	0.050	0.102	0.115
2003	500	0.053	0.029	0.043	0.050
2004	483	0.067	0.048	0.098	0.115
2005	651	0.063	0.043	0.079	0.109
2006	645	0.043	0.032	0.052	0.091
2007	660	0.039	0.027	0.033	0.062

Table A5. MRFSS wave (bi-monthly) CPUE averages.

Wave	N	Arith. Mean	Geo. Mean	GLM	GAM
1	273	0.028	0.015	0.030	0.085
2	2167	0.018	0.013	0.027	0.028
3	2944	0.117	0.067	0.104	0.089
4	3030	0.111	0.073	0.094	0.115
5	2905	0.238	0.115	0.241	0.193
6	2158	0.038	0.027	0.054	0.039

Table A6. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the MRFSS CPUE. Blank spaces indicate that the area index was not included as a variable in GAM. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
State index		< 0.001	< 0.001	< 0.001	< 0.001
Area index				< 0.001	< 0.001
Mode	beach/bank	n/a	n/a	n/a	n/a
	charter	< 0.001	< 0.001	< 0.001	< 0.001
	man made	0.633	< 0.001	0.205	< 0.001
	party	< 0.001	0.986	< 0.001	0.435
	party/charter	< 0.001	0.008	< 0.001	< 0.001
	private/rental	< 0.001	< 0.001	< 0.001	< 0.001
	shore	0.013	< 0.001	0.003	0.078
Wave	1	n/a	n/a	n/a	n/a
	2	0.203	< 0.001	0.037	0.322
	3	0.017	< 0.001	0.003	< 0.001
	4	0.005	< 0.001	0.027	< 0.001
	5	0.480	< 0.001	< 0.001	< 0.001
	6	0.038	0.066	0.016	0.039
Year	1981	n/a	n/a	n/a	n/a
	1982	0.124	0.012	0.312	0.008
	1983	0.166	0.184	0.465	0.528
	1984	0.582	0.036	0.872	0.007
	1985	0.024	0.286	0.171	0.699
	1986	0.557	0.090	0.746	0.005
	1987	0.005	0.003	0.119	< 0.001
	1988	0.027	0.568	0.330	0.513
	1989	< 0.001	0.029	< 0.001	< 0.001
	1990	< 0.001	0.594	< 0.001	0.064
	1991	< 0.001	0.203	0.002	0.007
	1992	< 0.001	0.569	< 0.001	0.065
	1993	< 0.001	0.153	0.020	0.007
	1994	0.006	< 0.001	0.115	< 0.001
	1995	< 0.001	< 0.001	0.033	< 0.001
	1996	0.048	< 0.001	0.580	< 0.001
	1997	0.018	< 0.001	0.175	< 0.001
	1998	0.001	< 0.001	0.065	< 0.001
	1999	< 0.001	< 0.001	0.007	< 0.001
	2000	0.006	< 0.001	0.045	< 0.001
	2001	< 0.001	< 0.001	0.002	< 0.001
	2002	< 0.001	< 0.001	< 0.001	< 0.001
	2003	< 0.001	0.003	< 0.001	< 0.001
	2004	< 0.001	< 0.001	0.001	< 0.001
	2005	< 0.001	< 0.001	0.008	< 0.001
	2006	< 0.001	0.004	0.001	< 0.001
	2007	< 0.001	0.017	< 0.001	< 0.001
R^2		0.393	0.313	0.215	0.191
Composite R^2		0.052		0.014	

Table A7. SEAMAP yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1990	231	21.381	2.801	16.558	44.204
1991	233	18.339	2.283	10.587	34.687
1992	234	24.906	1.960	9.514	35.278
1993	234	37.333	1.318	4.268	17.896
1994	234	13.107	1.354	4.980	20.190
1995	234	19.000	2.282	8.612	30.480
1996	232	17.289	2.966	11.245	35.217
1997	234	15.991	1.487	5.431	20.529
1998	234	44.893	2.854	8.195	32.341
1999	234	16.415	1.712	6.143	22.384
2000	234	14.239	2.012	11.188	33.146
2001	306	24.072	3.244	17.469	46.824
2002	303	10.904	1.258	4.637	20.967
2003	302	28.805	3.759	11.559	40.473
2004	302	62.288	3.495	13.528	41.899
2005	302	147.818	2.922	7.881	32.597
2006	302	30.639	2.062	10.070	38.306

Table A8. SEAMAP monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
4	953	62.313	2.447	75.677	30.922
5	512	40.693	3.111	31.156	38.149
7	1242	22.899	1.994	39.225	56.561
8	220	15.695	2.103	26.837	36.876
9	19	3.579	1.244	31.441	22.655
10	1215	14.113	2.203	25.092	26.905
11	224	86.906	2.421	16.771	34.131

Table A9. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the SEAMAP CPUE. Blank spaces indicate that depth was not included as a variable in the GAM. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		< 0.001	< 0.001	< 0.001	< 0.001
Latitude		< 0.001	< 0.001	< 0.001	0.060
Depth				< 0.001	0.638
Bottom salinity		< 0.001	< 0.001	< 0.001	< 0.001
Month	4	n/a	n/a	n/a	n/a
	5	0.307	0.200	0.109	0.014
	7	< 0.001	0.029	< 0.001	0.057
	8	0.570	0.435	0.342	0.332
	9	0.926	0.253	0.652	0.959
	10	0.043	0.232	0.048	0.083
	11	0.637	0.027	0.851	0.020
Year	1990	n/a	n/a	n/a	n/a
	1991	0.435	0.258	0.217	0.269
	1992	0.789	0.041	0.458	0.054
	1993	0.044	< 0.001	0.050	< 0.001
	1994	0.047	< 0.001	0.029	< 0.001
	1995	0.494	0.007	0.296	0.021
	1996	0.401	0.442	0.056	0.668
	1997	0.183	< 0.001	0.237	< 0.001
	1998	0.843	< 0.001	0.410	0.003
	1999	0.053	0.002	0.016	0.017
	2000	0.348	0.212	0.231	0.498
	2001	0.795	0.886	0.641	0.249
	2002	0.155	< 0.001	0.179	< 0.001
	2003	0.663	0.036	0.986	0.501
	2004	0.570	0.044	0.389	0.540
	2005	0.168	< 0.001	0.230	< 0.001
	2006	0.046	< 0.001	0.008	< 0.001
R^2		0.256	0.146	0.202	0.061
Composite R^2		0.114		< 0.001	

Table A10. Rhode Island fall trawl survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM
1979	11	9.636	3.337	12.145
1980	26	32.346	11.364	46.050
1981	44	63.773	15.124	58.100
1982	40	33.725	13.310	57.910
1983	22	9.682	4.893	16.858
1984	10	31.200	6.788	38.980
1985	32	37.156	9.107	34.956
1986	21	5.048	2.802	11.610
1987	8	8.375	5.288	19.358
1988	14	32.571	5.524	19.568
1989	10	5.300	3.054	10.137
1990	19	60.263	15.766	64.315
1991	17	58.294	9.069	34.391
1992	20	25.400	7.332	31.508
1993	14	18.214	6.142	26.020
1994	13	49.000	20.215	87.130
1995	4	2.750	2.310	9.295
1996	17	235.824	35.453	165.436
1997	27	138.185	22.898	108.072
1998	17	32.588	7.896	29.996
1999	17	8.882	4.349	20.648
2000	21	17.857	7.636	32.200
2001	19	42.737	13.990	58.833
2002	20	16.800	8.431	33.456
2003	32	259.875	41.927	246.854
2004	9	8.778	4.236	19.180
2005	29	177.828	25.480	105.235
2006	7	2.143	1.798	6.941
2007	25	59.200	16.995	78.251

Table A11. Rhode Island fall trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM
9	468	72.511	12.556	59.317
10	96	22.125	5.333	29.040
11	1	1.000	1.000	7.279

Table A12. Significance (*p*-values) of predictor variables used in the GLM for the Rhode Island survey CPUE. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. The last line of the table is the R^2 of the model.

Variable		GLM
		Positive catch
Area	BIS	n/a
	NB	< 0.001
	RIS	< 0.001
Month	9	n/a
	10	0.980
	11	0.727
Year	1979	n/a
	1980	0.065
	1981	0.025
	1982	0.024
	1983	0.662
	1984	0.149
	1985	0.121
	1986	0.941
	1987	0.603
	1988	0.557
	1989	0.817
	1990	0.029
	1991	0.169
	1992	0.199
	1993	0.344
	1994	0.012
	1995	0.794
	1996	< 0.001
	1997	0.003
	1998	0.242
	1999	0.490
	2000	0.182
	2001	0.041
	2002	0.167
	2003	< 0.001
	2004	0.594
	2005	0.002
	2006	0.530
	2007	0.011
R^2		0.252

Table A13. New Jersey trawl survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1988	68	65.971	2.144	81.915	8.351
1989	192	42.411	1.366	44.501	30.721
1990	171	14.865	0.900	26.260	7.088
1991	189	58.481	1.332	49.852	32.367
1992	191	15.298	1.210	22.886	30.354
1993	187	66.658	1.095	28.200	23.772
1994	186	78.957	1.576	53.636	54.054
1995	188	91.479	3.008	101.073	153.508
1996	189	119.233	3.910	148.785	277.406
1997	187	62.684	2.360	141.378	114.073
1998	188	20.622	0.761	19.049	21.592
1999	186	58.527	1.247	37.108	39.913
2000	187	67.374	1.409	39.814	45.519
2001	186	57.677	1.279	53.517	31.719
2002	188	164.473	3.252	197.973	195.490
2003	188	85.287	1.742	92.913	65.222
2004	187	179.882	3.381	245.019	220.879
2005	186	158.199	2.636	131.107	152.807
2006	186	129.484	1.252	22.579	32.730

Table A14. New Jersey trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
1	348	0.086	0.042	0.025	0.154
2	171	0.029	0.011	0.231	0.037
3	23	0.000	0.000	0.000	0.000
4	662	0.363	0.137	0.073	0.316
5	24	10.958	2.067	16.494	17.015
6	675	93.287	1.429	9.004	12.166
7	39	35.103	1.652	17.765	5.937
8	630	135.479	2.955	38.750	46.250
9	102	174.078	7.447	67.021	272.652
10	644	161.120	8.586	502.652	166.633
11	86	93.302	4.751	54.984	162.056
12	26	6.731	0.824	3.538	27.321

Table A15. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the New Jersey survey CPUE. Blank spaces indicate that month 3 (March) never had positive catch. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		< 0.001	< 0.001	< 0.001	< 0.001
Latitude		< 0.001	< 0.001	0.002	< 0.001
Depth		0.017	< 0.001	< 0.001	< 0.001
Month	1	n/a	n/a	n/a	n/a
	2	0.729	0.041	0.574	0.017
	3		1.000		0.958
	4	0.519	< 0.001	0.553	< 0.001
	5	0.053	< 0.001	0.042	< 0.001
	6	0.001	< 0.001	0.002	< 0.001
	7	0.036	0.001	0.638	0.133
	8	< 0.001	< 0.001	< 0.001	< 0.001
	9	< 0.001	< 0.001	< 0.001	< 0.001
	10	< 0.001	< 0.001	< 0.001	< 0.001
	11	< 0.001	< 0.001	< 0.001	< 0.001
	12	0.006	< 0.001	0.009	< 0.001
Year	1988	n/a	n/a	n/a	n/a
	1989	0.181	0.045	0.493	0.073
	1990	0.803	0.920	0.951	0.845
	1991	0.120	0.062	0.164	0.044
	1992	0.268	0.016	0.687	0.093
	1993	0.396	0.050	0.962	0.122
	1994	0.009	0.084	0.025	0.149
	1995	< 0.001	< 0.001	0.004	< 0.001
	1996	< 0.001	< 0.001	< 0.001	< 0.001
	1997	0.031	< 0.001	< 0.001	< 0.001
	1998	0.541	0.042	0.727	0.071
	1999	0.036	0.121	0.067	0.148
	2000	0.040	0.034	0.079	0.032
	2001	0.096	0.126	0.131	0.080
	2002	< 0.001	< 0.001	< 0.001	< 0.001
	2003	0.003	0.088	0.004	0.048
	2004	< 0.001	0.003	< 0.001	0.001
	2005	< 0.001	< 0.001	< 0.001	< 0.001
	2006	0.087	0.147	0.065	0.049
R^2		0.395	0.383	0.306	0.325
Composite R^2		0.079		0.062	

Table A16. Delaware 30-ft trawl survey yearly CPUE averages.

Year	N	Arith.mean	Geo.mean	GLM	GAM
1966	56	202.054	44.818	246.840	121.522
1967	75	140.960	10.987	136.597	93.710
1968	37	143.892	6.471	63.825	81.686
1969	40	170.075	19.729	169.587	144.138
1970	37	142.351	25.265	234.109	152.384
1971	38	203.184	33.142	368.205	210.422
1974	18	52.833	18.744	64.298	27.112
1979	91	27.209	5.987	26.813	51.640
1980	92	24.099	3.174	17.057	37.644
1981	98	15.194	3.226	13.978	32.445
1982	41	38.390	10.817	36.012	54.971
1983	38	31.921	5.721	32.232	60.836
1984	45	17.489	6.825	23.216	24.105
1990	55	25.400	5.862	19.419	39.224
1991	72	52.648	7.626	74.751	57.253
1992	86	54.372	3.816	17.617	43.255
1993	82	113.976	14.067	80.430	99.656
1994	71	223.971	5.617	49.284	82.335
1995	86	212.174	19.299	181.130	199.226
1996	76	346.760	37.109	237.120	293.148
1997	83	129.195	16.359	140.978	198.421
1998	80	158.150	21.722	316.731	196.982
1999	85	142.482	17.036	164.496	245.189
2000	66	198.697	12.876	125.841	260.535
2001	69	96.087	13.932	149.125	163.924
2002	68	194.672	19.482	204.106	116.800
2003	63	83.516	14.248	93.277	142.299
2004	83	69.519	9.741	59.774	81.985
2005	85	42.447	6.467	32.859	45.505
2006	90	129.764	11.618	82.233	105.427
2007	89	82.438	8.809	103.979	91.555

Table A17. Delaware 30-ft trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
1	24	0.000	0.000	0.000	0.000
2	21	0.000	0.000	0.000	0.000
3	162	0.013	0.009	3.255	6.542
4	187	8.770	1.555	48.150	45.741
5	211	135.029	15.089	149.129	124.141
6	219	204.124	25.726	116.181	118.988
7	210	249.548	33.683	89.830	114.896
8	251	196.799	41.033	107.451	109.719
9	233	167.589	49.680	181.007	189.873
10	201	101.164	35.929	272.733	212.557
11	209	15.635	3.703	81.732	111.310
12	167	2.192	0.357	31.395	42.694

Table A18. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the Delaware survey CPUE. Blank spaces indicate that depth was not used as a variable in GAM, and positive catches were not observed in months 1 and 2 (January and February). "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		< 0.001	0.002	0.013	< 0.001
Latitude		< 0.001	< 0.001	0.010	< 0.001
Depth				< 0.001	0.276
Duration		0.005	0.270	< 0.001	0.601
Bottom temperature		< 0.001	< 0.001	< 0.001	< 0.001
Bottom salinity		0.151	0.006	0.005	0.003
Month	1		n/a		n/a
	2		1.000		0.998
	3	n/a	1.000	n/a	0.980
	4	0.385	1.000	0.289	0.974
	5	0.096	1.000	0.029	0.973
	6	0.104	1.000	0.038	0.975
	7	0.102	1.000	0.049	0.977
	8	0.100	1.000	0.048	0.977
	9	0.047	1.000	0.018	0.975
	10	0.035	1.000	0.010	0.972
	11	0.108	1.000	0.089	0.973
	12	0.421	1.000	0.423	0.975
Year	1966	n/a	n/a	n/a	n/a
	1967	0.467	0.571	0.701	0.499
	1968	0.401	0.484	0.172	0.094
	1969	0.629	0.813	0.988	0.163
	1970	0.620	0.209	0.930	0.731
	1971	0.208	0.750	0.939	0.938
	1974	0.004	0.266	< 0.001	0.110
	1979	0.006	0.932	0.002	0.428
	1980	0.001	0.104	< 0.001	0.031
	1981	< 0.001	0.039	< 0.001	0.010
	1982	0.075	0.131	0.250	0.166
	1983	0.160	0.412	0.195	0.925
	1984	< 0.001	0.110	< 0.001	0.119
	1990	0.015	0.018	0.013	0.004
	1991	0.055	0.132	0.015	0.061
	1992	0.020	0.024	0.039	0.006
	1993	0.636	0.381	0.432	0.165
	1994	0.372	0.203	0.153	0.043
	1995	0.084	0.418	0.358	0.100
	1996	0.006	0.160	0.037	0.623
	1997	0.122	0.781	0.444	0.529
	1998	0.134	0.599	0.482	0.806
	1999	0.024	0.808	0.120	0.270
	2000	0.027	0.485	0.158	0.656
	2001	0.327	0.736	0.936	0.556
	2002	0.949	0.708	0.457	0.335
	2003	0.447	0.156	0.444	0.114
	2004	0.517	0.035	0.412	0.017
	2005	0.037	0.011	0.021	0.003
	2006	0.940	0.106	0.936	0.045
	2007	0.658	0.081	0.465	0.008
R^2		0.402	0.531	0.324	0.485
Composite R^2		0.050		0.049	

Table A19. Maryland coastal bay survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM
1972	3	6.333	4.667	4.604
1973	1	3.000	3.000	5.698
1975	17	29.412	8.299	52.877
1976	14	17.143	9.650	86.563
1977	7	5.571	3.854	4.564
1978	22	54.955	11.514	29.689
1979	9	6.222	4.662	11.244
1980	1	12.000	12.000	22.791
1981	16	53.750	18.054	104.490
1982	12	18.667	11.014	39.662
1984	6	6.167	3.060	5.391
1985	10	6.100	3.632	7.583
1986	4	5.250	4.244	50.215
1987	4	31.000	6.310	156.013
1989	21	4.095	3.203	9.765
1990	39	13.744	7.782	15.357
1991	36	16.583	7.896	25.501
1992	31	44.516	14.478	51.628
1993	28	8.036	4.414	31.418
1994	48	20.125	7.944	21.295
1995	48	77.417	19.122	16.630
1996	49	24.286	8.204	16.672
1997	44	33.932	11.881	6.487
1998	45	29.444	8.599	4.512
1999	40	109.600	14.689	9.783
2000	38	43.526	11.685	18.591
2001	37	47.919	16.818	11.708
2002	17	15.176	6.814	21.899
2003	57	43.158	14.328	9.294
2004	49	22.959	9.039	5.881
2005	56	31.964	12.071	8.068
2006	29	23.828	11.264	5.760
2007	41	20.098	8.355	14.340

Table A20. Maryland coastal bay survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM
4	3	1.000	1.000	6.690
5	8	1.250	1.181	2.249
6	26	3.308	2.018	8.068
7	242	71.789	19.689	20.410
8	310	30.397	12.085	13.280
9	200	12.410	6.482	27.263
10	84	5.476	3.255	29.250
11	6	6.833	3.769	25.253

Table A21. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the Maryland coastal bay survey CPUE. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last line of the table is the R^2 of the model.

Variable		GLM
		Positive catch
Site	T001	n/a
	T002	0.237
	T003	0.431
	T004	0.335
	T005	0.011
	T006	0.014
	T007	< 0.001
	T008	< 0.001
	T009	< 0.001
	T010	< 0.001
	T011	< 0.001
	T012	< 0.001
	T013	< 0.001
	T014	< 0.001
	T015	< 0.001
	T016	< 0.001
	T017	< 0.001
	T018	< 0.001
	T019	< 0.001
	T020	< 0.001
Month	4	n/a
	5	0.745
	6	0.424
	7	< 0.001
	8	< 0.001
	9	0.003
	10	0.051
	11	0.075
Year	1972	n/a
	1973	0.056
	1975	0.642
	1976	0.835
	1977	0.230
	1978	0.477
	1979	0.194
	1980	0.472
	1981	0.645
	1982	0.407
	1984	0.272
	1985	0.552
	1986	0.696
	1987	0.688
	1989	0.066
	1990	0.913
	1991	0.595
	1992	0.639

	1993	0.228
	1994	0.446
	1995	0.336
	1996	0.924
	1997	0.799
	1998	0.855
	1999	0.842
	2000	0.951
	2001	0.524
	2002	0.493
	2003	0.437
	2004	0.909
	2005	0.659
	2006	0.934
	2007	0.756
R^2		0.517

Table A22. North Carolina gillnet survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
2001	230	1.661	1.008	2.678	1.873
2002	306	1.529	0.658	1.342	1.468
2003	316	1.241	0.596	1.018	1.420
2004	317	1.366	0.576	1.058	1.163
2005	297	1.300	0.639	1.167	1.292
2006	317	1.082	0.490	0.891	0.905
2007	314	0.481	0.284	0.506	0.539

Table A23. North Carolina gillnet survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
2	94	0.096	0.062	0.187	0.249
3	188	0.559	0.290	0.632	0.845
4	185	3.341	1.459	3.166	2.756
5	221	2.330	1.022	1.883	1.783
6	220	0.664	0.339	0.666	0.640
7	222	0.563	0.286	0.569	0.670
8	222	0.671	0.414	0.774	0.828
9	206	1.015	0.648	1.243	1.162
10	216	1.639	0.944	1.777	1.602
11	213	1.080	0.628	1.173	1.217
12	110	0.864	0.381	0.748	1.068

Table A24. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the North Carolina gillnet survey CPUE. Blank spaces indicate that bottom temperature was not included as a variable in GLM. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		0.001	< 0.001	0.902	0.009
Latitude		0.207	< 0.001	0.004	< 0.001
Depth		0.079	< 0.001	0.883	0.003
Bottom temperature		0.716	< 0.001		
Bottom salinity		0.073	0.006	0.156	0.007
Surface DO		0.316	0.018	0.051	< 0.001
Surface salinity		0.055	< 0.001	0.280	< 0.001
Month	2	n/a	n/a	n/a	n/a
	3	0.202	0.018	0.248	0.003
	4	0.001	< 0.001	0.002	< 0.001
	5	0.019	< 0.001	0.016	< 0.001
	6	0.525	0.100	0.473	0.026
	7	0.762	0.048	0.730	0.046
	8	0.677	0.013	0.621	0.005
	9	0.373	0.001	0.333	< 0.001
	10	0.071	< 0.001	0.053	< 0.001
	11	0.156	< 0.001	0.194	< 0.001
	12	0.074	0.017	0.083	0.006
Year	2001	n/a	n/a	n/a	n/a
	2002	0.680	0.026	0.335	< 0.001
	2003	0.454	0.002	0.825	< 0.001
	2004	0.624	< 0.001	0.367	< 0.001
	2005	0.619	0.002	0.515	< 0.001
	2006	0.206	< 0.001	0.151	< 0.001
	2007	0.004	< 0.001	0.004	< 0.001
R^2		0.176	0.176	0.127	0.118
Composite R^2		0.155		0.096	

Table A25. Connecticut trawl survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1989	155	1.143	0.295	1.348	0.609
1990	60	0.270	0.205	0.628	0.554
1992	155	0.624	0.397	0.982	0.585
1993	239	0.625	0.395	0.836	0.652
1994	240	2.065	0.611	1.125	1.240
1995	200	0.367	0.270	0.377	0.460
1996	200	0.639	0.402	0.760	0.883
1997	155	0.679	0.535	1.755	1.198
1998	133	0.277	0.181	0.283	0.914
1999	26	0.000	0.000	0.000	0.000
2000	199	2.394	0.761	1.678	1.672
2001	199	1.449	0.618	1.552	1.407
2002	199	1.258	0.571	1.316	0.986
2003	200	1.100	0.553	0.845	2.001
2004	199	1.310	0.569	1.181	1.298
2005	200	1.094	0.519	1.114	1.020
2006	120	0.376	0.240	0.335	0.440
2007	200	2.002	0.684	1.530	1.541
2008	119	0.034	0.024	0.060	0.337

Table A26. Connecticut trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
4	501	0.004	0.003	0.006	0.521
5	709	0.061	0.043	0.138	0.606
6	672	0.213	0.150	0.256	0.342
7	43	0.122	0.087	0.137	0.086
8	52	0.181	0.133	0.126	0.131
9	596	2.527	1.409	3.285	1.193
10	534	3.031	1.414	2.000	1.818
11	86	1.703	1.320	1.771	1.703
12	5	1.000	0.741	1.122	2.311

Table A27. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the CT trawl survey CPUE. Blank spaces indicate where bottom salinity was not included in GAM or positive catch GLM, and positive catches were not recorded in 1988 or 1999. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		< 0.001	< 0.001	0.040	< 0.001
Latitude		0.024	< 0.001	0.028	
Depth		0.004	< 0.001	0.036	0.096
Duration		< 0.001	0.101	< 0.001	0.095
Bottom temperature		0.001	< 0.001		< 0.001
Bottom salinity					0.066
Month	4	n/a	n/a	n/a	n/a
	5	0.799	0.995	0.947	0.169
	6	0.766	0.401	0.956	0.294
	7	0.420	0.173	0.292	0.738
	8	0.862	0.119	0.854	0.414
	9	0.357	0.703	0.344	0.327
	10	0.301	0.227	0.374	0.002
	11	0.673	0.009	0.821	< 0.001
	12	0.522	< 0.001	0.702	< 0.001
Year	1988		n/a		
	1989	n/a	1.000	n/a	n/a
	1990	< 0.001	1.000	0.005	0.131
	1992	< 0.001	1.000	< 0.001	0.001
	1993	< 0.001	1.000	< 0.001	< 0.001
	1994	0.007	1.000	< 0.001	< 0.001
	1995	< 0.001	1.000	0.006	< 0.001
	1996	< 0.001	1.000	< 0.001	< 0.001
	1997	< 0.001	1.000	< 0.001	< 0.001
	1998	< 0.001	1.000	< 0.001	< 0.001
	1999		1.000		0.976
	2000	0.180	1.000	0.212	< 0.001
	2001	< 0.001	1.000	0.004	< 0.001
	2002	< 0.001	1.000	0.001	< 0.001
	2003	0.015	1.000	0.012	< 0.001
	2004	< 0.001	1.000	< 0.001	< 0.001
	2005	< 0.001	1.000	< 0.001	< 0.001
	2006	< 0.001	1.000	< 0.001	0.044
	2007	0.016	1.000	0.030	< 0.001
	2008	0.046	1.000	0.053	0.240
R^2		0.268	0.533	0.229	0.488
Composite R^2		0.196		0.174	

Table A28. New York juvenile trawl survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM
1987	354	0.862	0.328	3.904
1988	426	0.254	0.106	0.977
1989	420	3.274	0.574	0.152
1990	430	1.058	0.264	0.201
1991	398	25.894	4.440	28.780
1992	411	10.727	1.205	6.328
1993	414	2.056	0.428	1.432
1994	428	27.182	1.725	16.677
1995	376	3.005	0.914	6.622
1996	409	88.814	5.004	56.713
1997	379	29.900	2.745	1.705
1998	395	2.382	0.515	1.103
1999	400	17.235	2.221	51.880
2000	420	67.183	3.899	100.122
2001	414	46.498	3.249	42.470
2002	415	54.860	2.604	95.765
2003	392	23.862	1.363	47.478
2004	408	22.532	2.127	6.749
2005	182	66.349	2.714	10.111
2006	244	36.033	4.776	5.206
2007	377	59.488	3.470	105.074

Table A29. New York juvenile trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM
4	8	0.000	0.000	0.000
5	1221	0.034	0.012	0.109
6	1351	10.495	0.453	11.642
7	1453	69.522	4.561	55.257
8	1391	55.884	6.055	74.277
9	1272	16.237	2.310	7.006
10	1339	3.370	0.683	7.304
11	57	0.053	0.032	0.000

Table A30. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the New York juvenile survey CPUE. The blank space indicates that no positive catches were recorded in month 4 (April). "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last line of the table is the R^2 of the model.

Variable		GLM	
		Positive catch	Present/Absent
Station	1	n/a	n/a
	2	< 0.001	< 0.001
	3	0.001	< 0.001
	4	0.103	0.344
	5	0.364	0.054
	6	0.953	0.205
	7	0.651	0.014
	8	0.461	0.370
	9	< 0.001	< 0.001
	10	< 0.001	< 0.001
	11	0.528	0.383
	12	0.295	0.011
	13	0.085	0.003
	14	< 0.001	< 0.001
	15	< 0.001	< 0.001
	16	< 0.001	< 0.001
	17	< 0.001	< 0.001
	18	< 0.001	< 0.001
	19	< 0.001	0.016
	20	0.012	0.004
	21	0.018	0.124
	22	< 0.001	< 0.001
	23	< 0.001	< 0.001
	24	< 0.001	< 0.001
	25	0.388	0.028
	26	< 0.001	< 0.001
	27	< 0.001	< 0.001
	28	< 0.001	< 0.001
	29	< 0.001	< 0.001
	30	< 0.001	< 0.001
	31	< 0.001	< 0.001
	32	< 0.001	0.004
	33	0.165	0.607
	34	0.032	0.868
	35	0.114	0.332
	36	0.181	0.929
	37	0.073	0.516
	38	< 0.001	< 0.001
	39	< 0.001	< 0.001
	40	< 0.001	< 0.001
	41	0.031	0.330
	42	0.289	0.032
	43	< 0.001	< 0.001
	44	< 0.001	< 0.001

	45	< 0.001	< 0.001
	46	0.001	< 0.001
	47	0.160	0.314
	48	< 0.001	< 0.001
	49	< 0.001	< 0.001
	50	< 0.001	< 0.001
	51	0.035	0.725
	52	0.001	0.128
	53	< 0.001	< 0.001
	54	< 0.001	< 0.001
	55	< 0.001	< 0.001
	56	< 0.001	< 0.001
	57	0.178	0.992
	58	0.068	0.407
	59	0.017	0.019
	60	0.037	0.005
	61	0.003	< 0.001
	62	< 0.001	< 0.001
	63	< 0.001	< 0.001
	64	< 0.001	< 0.001
	65	< 0.001	< 0.001
	66	< 0.001	< 0.001
	67	< 0.001	< 0.001
	68	< 0.001	0.104
	69	0.013	0.028
	70	< 0.001	< 0.001
	71	0.146	0.876
	72	0.002	0.085
	73	< 0.001	< 0.001
	74	< 0.001	< 0.001
	75	< 0.001	0.007
	76	0.051	0.795
	77	0.078	0.429
Month	4		n/a
	5	n/a	0.969
	6	< 0.001	0.953
	7	< 0.001	0.939
	8	< 0.001	0.936
	9	< 0.001	0.941
	10	0.005	0.947
	11	0.838	0.961
Year	1987	n/a	n/a
	1988	0.339	< 0.001
	1989	< 0.001	0.004
	1990	0.272	0.269
	1991	< 0.001	< 0.001
	1992	< 0.001	< 0.001
	1993	0.019	0.545
	1994	< 0.001	< 0.001
	1995	< 0.001	< 0.001
	1996	< 0.001	< 0.001
	1997	< 0.001	< 0.001
	1998	0.009	< 0.001

	1999	< 0.001	< 0.001
	2000	< 0.001	< 0.001
	2001	< 0.001	< 0.001
	2002	< 0.001	< 0.001
	2003	< 0.001	< 0.001
	2004	< 0.001	< 0.001
	2005	< 0.001	< 0.001
	2006	< 0.001	< 0.001
	2007	< 0.001	< 0.001
R^2		0.440	0.474
Composite R^2		0.100	

Table A31. Delaware 16-ft. trawl survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1980	193	12.843	3.919	12.116	9.889
1981	189	15.452	4.117	19.766	17.541
1982	229	18.811	5.788	24.776	24.957
1983	226	10.454	3.355	13.530	12.612
1984	185	30.351	5.544	18.437	28.226
1985	133	7.271	1.431	2.249	24.066
1986	203	30.722	6.404	28.425	27.733
1987	176	18.589	4.288	20.623	18.753
1988	165	21.540	3.393	18.167	22.226
1989	138	16.291	4.102	23.922	24.670
1990	193	32.026	8.739	25.242	23.848
1991	276	31.347	7.460	38.577	40.395
1992	275	31.640	7.591	40.047	43.430
1993	168	34.196	7.271	38.966	47.640
1994	267	51.864	10.188	54.135	61.629
1995	264	55.727	10.011	45.430	47.161
1996	279	50.838	6.302	36.791	38.679
1997	279	59.194	8.436	47.419	52.432
1998	317	26.097	5.472	32.846	37.557
1999	231	32.709	6.372	33.377	36.227
2000	257	43.371	10.074	63.135	50.196
2001	240	25.609	4.533	21.618	20.362
2002	275	26.344	3.526	20.555	18.364
2003	275	28.585	4.465	31.441	31.460
2004	253	26.292	3.639	26.766	29.277
2005	240	65.038	10.772	52.974	49.500
2006	273	14.326	3.298	17.794	17.814
2007	272	34.728	7.247	43.132	38.389

Table A32. Delaware 16-ft. trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
4	866	0.099	0.040	0.258	0.434
5	886	2.608	0.812	5.095	6.062
6	962	20.490	3.356	15.665	15.165
7	936	88.764	29.840	81.128	76.784
8	955	57.216	23.366	59.375	57.151
9	954	39.119	14.906	40.799	47.053
10	912	12.798	4.867	18.774	23.383

Table A33. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the Delaware 16-ft. trawl survey CPUE. Blank spaces indicate that depth and tow duration were not included as variables in the GAM. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		0.001	0.005	< 0.001	< 0.001
Latitude		< 0.001	< 0.001	< 0.001	< 0.001
Depth				0.010	0.788
Duration				0.017	0.775
Surface temperature		< 0.001	< 0.001	< 0.001	0.002
Surface salinity		0.002	< 0.001	0.544	< 0.001
Month	4	n/a	n/a	n/a	n/a
	5	0.003	< 0.001	0.002	< 0.001
	6	< 0.001	< 0.001	< 0.001	< 0.001
	7	< 0.001	< 0.001	< 0.001	< 0.001
	8	< 0.001	< 0.001	< 0.001	< 0.001
	9	< 0.001	< 0.001	< 0.001	< 0.001
	10	< 0.001	< 0.001	< 0.001	< 0.001
Year	1980	n/a	n/a	n/a	n/a
	1981	0.002	0.233	0.010	0.747
	1982	< 0.001	0.003	< 0.001	0.042
	1983	0.300	0.078	0.469	0.335
	1984	< 0.001	0.141	< 0.001	0.579
	1985	< 0.001	0.983	< 0.001	0.425
	1986	< 0.001	0.005	< 0.001	0.031
	1987	< 0.001	0.410	0.002	0.852
	1988	< 0.001	0.246	< 0.001	0.185
	1989	< 0.001	0.080	0.002	0.444
	1990	< 0.001	0.228	< 0.001	0.947
	1991	< 0.001	0.004	< 0.001	< 0.001
	1992	< 0.001	< 0.001	< 0.001	0.001
	1993	< 0.001	< 0.001	< 0.001	0.001
	1994	< 0.001	< 0.001	< 0.001	< 0.001
	1995	< 0.001	< 0.001	< 0.001	< 0.001
	1996	< 0.001	< 0.001	< 0.001	0.005
	1997	< 0.001	< 0.001	< 0.001	0.001
	1998	< 0.001	0.002	< 0.001	0.076
	1999	< 0.001	< 0.001	< 0.001	0.022
	2000	< 0.001	< 0.001	< 0.001	< 0.001
	2001	< 0.001	0.133	< 0.001	0.838
	2002	< 0.001	0.758	< 0.001	0.307
	2003	< 0.001	0.124	< 0.001	0.762
	2004	< 0.001	0.069	< 0.001	0.914
	2005	< 0.001	0.025	< 0.001	0.347
	2006	< 0.001	0.424	0.008	0.860
	2007	< 0.001	0.019	< 0.001	0.109
R^2		0.322	0.499	0.297	0.471
Composite R^2		0.180		0.161	

Table A34. VIMS trawl survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1988	710	21.259	1.704	14.276	17.422
1989	716	18.894	1.490	16.201	11.433
1990	709	16.422	1.445	14.377	8.567
1991	650	9.752	1.206	9.363	8.318
1992	658	15.895	1.965	15.450	21.378
1993	589	19.474	1.558	12.418	13.834
1994	662	9.956	1.544	12.342	13.859
1995	635	12.898	2.076	16.084	17.908
1996	1152	13.773	1.677	13.819	14.767
1997	1189	12.373	1.727	12.915	15.169
1998	1254	12.510	1.407	10.304	9.545
1999	1321	14.760	2.002	15.731	17.293
2000	1351	13.928	2.054	16.717	22.554
2001	1107	27.046	2.778	22.344	19.102
2002	1088	14.619	1.893	15.444	14.499
2003	1194	11.899	1.486	11.634	13.983
2004	1222	10.639	1.408	10.504	8.708
2005	1211	7.701	1.386	8.801	8.766
2006	1193	10.293	1.808	13.868	15.705
2007	1223	12.303	1.900	14.628	13.584

Table A35. VIMS trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
1	1072	0.006	0.003	0.037	0.888
2	1562	0.002	0.001	0.013	0.499
3	1143	0.001	0.001	0.022	0.087
4	1661	0.337	0.109	0.798	1.840
5	1828	2.231	0.813	4.292	5.972
6	1850	3.518	1.047	4.757	3.885
7	1809	20.837	3.617	14.780	7.634
8	1754	37.365	7.987	31.658	16.285
9	1861	49.988	10.823	45.306	36.644
10	1869	27.544	5.995	30.718	40.378
11	1781	9.389	2.181	13.893	28.001
12	1644	1.207	0.251	1.916	10.311

Table A36. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the VIMS CPUE. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. Last two lines of the table are the R^2 of each model, and the composite R^2 resulting from multiplying the respective two components of the delta-lognormal method, i.e., positive catch × presence/absence.

Variable		GAM		GLM	
		Positive catch	Present/Absent	Positive catch	Present/Absent
Longitude		< 0.001	< 0.001	0.094	< 0.001
Latitude		< 0.001	< 0.001	0.007	0.114
Depth		< 0.001	< 0.001	< 0.001	< 0.001
DO		< 0.001	< 0.001	0.009	0.279
Temperature		< 0.001	< 0.001	< 0.001	< 0.001
River	AT	n/a	n/a	n/a	n/a
	CL	0.260	0.289	0.444	0.970
	CP	0.668	0.084	0.576	0.096
	GW	0.754	0.102	0.736	0.536
	JA	0.053	0.165	0.234	0.816
	JE	0.796	0.570	0.297	0.301
	MB	0.363	0.003	0.624	0.125
	ME	0.174	0.013	0.363	0.119
	MN	0.102	0.003	0.423	0.018
	MS	0.906	0.027	0.483	0.144
	MW	0.500	0.185	0.881	0.406
	PK	0.111	0.070	0.476	0.326
	PM	0.545	0.106	0.574	0.637
	PO	0.132	0.449	0.200	0.213
	RA	0.163	0.334	0.403	0.725
	YK	0.169	0.045	0.644	0.259
Month	1	n/a	n/a	n/a	n/a
	2	0.597	0.968	0.679	0.341
	3	0.942	0.047	0.913	0.119
	4	0.411	0.009	0.667	< 0.001
	5	0.491	< 0.001	0.787	< 0.001
	6	0.295	< 0.001	0.471	< 0.001
	7	0.735	< 0.001	0.986	< 0.001
	8	0.748	< 0.001	0.478	< 0.001
	9	0.315	< 0.001	0.180	< 0.001
	10	0.332	< 0.001	0.194	< 0.001
	11	0.542	< 0.001	0.295	< 0.001
	12	0.991	< 0.001	0.740	< 0.001
Year	1988	n/a	n/a	n/a	n/a
	1989	0.634	0.003	0.900	0.057
	1990	0.297	< 0.001	0.702	< 0.001
	1991	< 0.001	0.234	< 0.001	0.322
	1992	0.032	< 0.001	0.152	< 0.001
	1993	0.009	0.379	0.028	0.340
	1994	0.003	0.220	0.006	0.323
	1995	0.003	< 0.001	0.010	0.002
	1996	0.004	0.125	0.024	0.415
	1997	< 0.001	< 0.001	< 0.001	0.001

	1998	< 0.001	0.753	< 0.001	0.774
	1999	0.010	0.005	0.045	0.018
	2000	0.007	< 0.001	0.017	< 0.001
	2001	0.287	0.032	0.647	0.051
	2002	< 0.001	0.021	0.003	0.078
	2003	< 0.001	0.012	< 0.001	0.279
	2004	< 0.001	0.790	< 0.001	0.654
	2005	< 0.001	0.042	< 0.001	0.448
	2006	< 0.001	< 0.001	< 0.001	0.002
	2007	< 0.001	0.010	< 0.001	0.028
R^2		0.293	0.427	0.264	0.375
Composite R^2		0.158		0.118	

Table A37. North Carolina juvenile trawl survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM	GAM
1987	85	12.741	7.084	13.669	13.430
1988	120	17.042	9.110	17.277	16.481
1989	93	12.677	6.512	11.305	11.776
1990	69	19.768	13.095	17.817	15.807
1991	65	12.846	8.315	9.872	9.387
1992	72	18.472	14.603	19.690	18.275
1993	77	15.143	10.678	14.354	15.324
1994	86	20.547	14.831	18.676	19.551
1995	88	19.068	14.949	19.755	19.520
1996	83	23.373	17.938	23.117	23.806
1997	87	21.977	17.494	24.530	25.922
1998	71	22.676	18.398	28.397	27.731
1999	104	31.202	27.250	30.380	32.394
2000	101	22.455	19.541	26.094	26.742
2001	67	17.239	11.482	14.857	15.245
2002	63	13.127	9.260	12.362	11.846
2003	77	13.039	9.714	12.803	13.024
2004	67	17.836	13.352	18.064	17.821
2005	67	17.687	12.145	15.963	15.160
2006	78	21.462	16.400	22.869	23.041
2007	65	20.323	14.887	18.849	18.415

Table A38. North Carolina juvenile trawl survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM	GAM
1	1	1.000	1.000	3.320	1.763
3	35	1.629	1.458	2.487	2.688
6	701	16.693	11.894	16.589	17.348
7	53	37.604	32.360	31.234	30.614
9	785	21.146	15.519	22.472	23.907
10	50	21.680	19.257	24.307	23.615
12	60	5.767	3.456	5.110	5.584

Table A39. Significance (*p*-values) of predictor variables used in the GAMs and GLMs for the North Carolina juvenile survey CPUE. Blank spaces indicate that depth was not used as a variable in the GAM, and latitude was not used as a variable in the GLM. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. The last line of the table is the R^2 of each model.

Variable		GAM	GLM
		Positive catch	Positive catch
Longitude		< 0.001	< 0.001
Latitude		< 0.001	
Depth			< 0.001
Bottom DO		0.002	0.005
Surface DO		< 0.001	0.005
Month	1	n/a	n/a
	3	0.674	0.843
	6	0.021	0.038
	7	0.005	0.009
	9	0.009	0.017
	10	0.010	0.017
	12	0.247	0.354
Year	1987	n/a	n/a
	1988	0.144	0.085
	1989	0.379	0.241
	1990	0.319	0.099
	1991	0.033	0.065
	1992	0.061	0.024
	1993	0.414	0.707
	1994	0.017	0.042
	1995	0.017	0.016
	1996	< 0.001	< 0.001
	1997	< 0.001	< 0.001
	1998	< 0.001	< 0.001
	1999	< 0.001	0.003
	2000	< 0.001	< 0.001
	2001	0.506	0.618
	2002	0.492	0.643
	2003	0.866	0.784
	2004	0.111	0.106
	2005	0.519	0.377
	2006	0.002	0.003
	2007	0.086	0.061
R^2		0.311	0.272

Table A40. Maryland Chesapeake “blue crab” survey yearly CPUE averages.

Year	N	Arith. Mean	Geo. Mean	GLM
1989	18	4.944	3.071	10.276
1990	37	19.108	5.189	14.574
1991	26	24.846	6.175	19.394
1992	53	26.736	10.259	30.694
1993	44	31.955	6.020	22.144
1994	36	29.694	9.110	23.681
1995	99	29.131	12.113	53.642
1996	65	68.062	14.857	46.622
1997	63	40.778	14.022	44.013
1998	58	68.362	20.506	38.454
1999	55	35.018	14.424	52.043
2000	84	57.190	14.369	56.037
2001	88	53.182	20.174	69.271
2002	69	39.971	9.418	32.382
2003	60	25.400	10.985	32.456
2004	60	7.167	3.587	15.945
2005	59	28.729	7.421	32.193
2006	71	13.563	3.973	7.629
2007	87	6.586	2.478	5.307
2008	58	4.690	1.928	8.355

Table A41. Maryland Chesapeake “blue crab” survey monthly CPUE averages.

Month	N	Arith. Mean	Geo. Mean	GLM
5	16	1.750	1.566	4.570
6	141	39.624	8.155	17.362
7	262	44.805	14.229	43.673
8	331	41.610	11.209	48.918
9	328	21.448	6.665	29.435
10	105	5.952	2.841	11.791
11	7	2.000	1.784	1.442

Table A42. Significance (*p*-values) of predictor variables used in the GLM for the Maryland "blue crab" survey CPUE. "n/a" entries indicate that a parameter was not reported for one category of a categorical variable, in order to avoid over-fitting the model. The last line of the table is the R^2 of the model.

Variable		GLM
		Positive catch
Site	CHR	n/a
	CPR	0.110
	EBY	0.663
	FBY	0.535
	LCP	0.531
	NTK	0.933
	PAX	0.009
	POC	< 0.001
	TNG	< 0.001
Month	5	n/a
	6	< 0.001
	7	< 0.001
	8	< 0.001
	9	< 0.001
	10	0.008
	11	0.767
Year	1989	n/a
	1990	0.341
	1991	0.102
	1992	0.001
	1993	0.029
	1994	0.021
	1995	< 0.001
	1996	< 0.001
	1997	< 0.001
	1998	< 0.001
	1999	< 0.001
	2000	< 0.001
	2001	< 0.001
	2002	< 0.001
	2003	< 0.001
	2004	0.200
	2005	< 0.001
	2006	0.014
	2007	0.136
	2008	0.339
R^2		0.433

Appendix C-5

Weakfish

ADAPT Output

SAW/SARC 48
June 1-4, 2009
Woods Hole, MA

VPA Version 2.8.0

Model ID: Weakfish SASC preferred run

Input File: C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND RETRO_00.DAT
Date of Run: 21-APR-2009 Time of Run: 14:12

Levenburg-Marquardt Algorithm Completed 16 Iterations
Residual Sum of Squares = 446.968

Number of Residuals = 359
Number of Parameters = 6
Degrees of Freedom = 353
Mean Squared Residual = 1.26620
Standard Deviation = 1.12526

Number of Years = 26
Number of Ages = 6
First Year = 1982
Youngest Age = 1
Oldest True Age = 5

Number of Survey Indices Available = 38
Number of Survey Indices Used in Estimate = 23

VPA Classic Method - Auto Estimated Q's

Stock Numbers Predicted in Terminal Year Plus One (2008)

Age	Stock Predicted	Std. Error	CV
1	16442.584	0.190205E+05	0.115679E+01
2	11406.889	0.683918E+04	0.599566E+00
3	3737.354	0.173831E+04	0.465117E+00
4	1060.126	0.480622E+03	0.453364E+00
5	200.106	0.953091E+02	0.476292E+00
6	4.679	0.537012E+01	0.114759E+01

Catchability Values for Each Survey Used in Estimate

INDEX	Catchability	Std. Error	CV
1	0.211918E-02	0.690956E-03	0.326049E+00
2	0.250557E-02	0.718644E-03	0.286818E+00
3	0.141990E-02	0.541978E-03	0.381702E+00
4	0.132513E-02	0.421512E-03	0.318091E+00
5	0.892300E-03	0.279400E-03	0.313123E+00
6	0.792329E-03	0.193343E-03	0.244019E+00
7	0.359433E-05	0.842737E-06	0.234463E+00
8	0.789323E-05	0.180624E-05	0.228835E+00
9	0.578221E-05	0.164373E-05	0.284273E+00
10	0.333437E-05	0.147753E-05	0.443120E+00
11	0.185515E-05	0.821036E-06	0.442572E+00
12	0.149946E-05	0.932948E-06	0.622190E+00
13	0.315053E-04	0.562930E-05	0.178678E+00
14	0.157929E-03	0.311494E-04	0.197237E+00

15	0.455255E-03	0.137916E-03	0.302942E+00
16	0.849967E-03	0.384431E-03	0.452289E+00
17	0.112275E-02	0.627731E-03	0.559104E+00
18	0.701428E-04	0.801330E-05	0.114243E+00
34	0.568621E-05	0.113335E-05	0.199316E+00
35	0.197591E-04	0.314498E-05	0.159167E+00
36	0.573293E-04	0.105223E-04	0.183542E+00
37	0.641095E-04	0.121778E-04	0.189953E+00
38	0.183228E-04	0.213521E-05	0.116533E+00

```
-- Non-Linear Least Squares Fit --

Default Tolerances Used

Scaled Gradient Tolerance      =    6.055454E-06
Scaled Step Tolerance          =    3.666853E-11
Relative Function Tolerance    =    3.666853E-11
Absolute Function Tolerance    =    4.930381E-32

VPA Method Options

- Catchability Values Estimated as an Analytic Function of N
- Catch Equation Used in Cohort Solution
- Plus Group Backward Calculation Method Used
- Arithmetic Average Used in F-Oldest Calculation
- F-Oldest Calculation in Years Prior to Terminal Year
  Uses Fishing Mortality in Ages  4 to  4
- Calculation of Population of Age 1 In Year 2008
  =  Stock Estimate

Stock Estimates

Age   1
Age   2
Age   3
Age   4
Age   5
Age   6

Full F in Terminal Year               =   0.5570

F in Oldest True Age in Terminal Year =   0.5570

Full F Calculated Using Average Method
```

Age	Input Partial Recruitment	Calc Partial Recruitment	Fishing Mortality	Used In Full F	Comments
1	0.120	0.083	0.0464	YES	Stock Estimate in T+1
2	0.280	0.428	0.2383	YES	Stock Estimate in T+1
3	0.570	0.627	0.3495	YES	Stock Estimate in T+1
4	1.000	0.831	0.4631	YES	Stock Estimate in T+1
5	1.000	1.000	0.5570		Input PR * Full F

Catch At Age - Input Data

AGE	1982	1983	1984	1985	1986
1	9914.2	8004.0	10444.2	14153.2	18610.7
2	12967.0	12869.1	14736.9	11262.3	15778.4
3	5473.0	5822.7	6521.1	3246.1	4942.4
4	2778.2	2780.0	3045.3	1171.0	1823.7
5	721.6	568.2	484.5	212.9	264.1
6	639.5	424.1	254.5	55.1	52.1

AGE	1987	1988	1989	1990	1991
1	16256.3	8161.9	3705.0	9510.1	9795.9
2	14343.1	16140.8	5304.9	4890.1	5825.6
3	4347.1	10545.3	4333.5	2093.6	2750.0
4	1485.2	6092.0	2922.3	1204.8	1373.6
5	145.4	1050.5	626.2	591.4	463.4
6	11.0	70.7	84.6	89.1	57.3

AGE	1992	1993	1994	1995	1996
1	5179.5	4974.8	3761.9	4336.3	2498.8
2	6046.0	6357.0	4347.4	3727.7	2689.5
3	2211.0	2179.8	3561.0	3566.7	5033.3
4	1255.0	1138.6	1563.5	1637.8	3174.2
5	527.8	401.1	204.1	198.1	1379.3
6	65.0	48.2	39.8	54.3	100.1

AGE	1997	1998	1999	2000	2001
1	1716.4	1270.6	1412.6	1377.0	2420.7
2	2394.2	2138.3	1300.4	1727.1	2953.1
3	2913.2	3983.1	2256.6	1985.7	1474.1
4	5522.0	2019.2	3326.0	1663.7	1219.9
5	1523.1	2928.8	725.7	1528.2	658.7
6	410.2	909.5	1145.0	403.0	485.9

AGE	2002	2003	2004	2005	2006
1	2591.7	335.6	852.3	334.3	747.3
2	1070.5	949.9	1511.9	1771.5	637.3
3	2695.7	959.7	667.8	1255.2	959.2
4	823.9	718.4	115.8	191.5	252.9
5	388.2	209.5	49.7	10.2	15.5
6	231.5	254.2	38.4	27.1	11.9

Catch At Age - Input Data

AGE	2007
1	616.6
2	1148.0
3	507.6
4	135.2
5	25.2
6	5.8

Weight At Age - Input Data

AGE	1982	1983	1984	1985	1986
1	0.1420	0.1210	0.1590	0.1420	0.1890
2	0.2790	0.2540	0.2940	0.4480	0.4850
3	0.5210	0.4850	0.5670	1.1400	1.2830
4	0.8210	1.5040	1.1860	2.6890	2.7130
5	1.4110	2.3710	1.6670	2.5760	2.9550
6	3.0330	2.8620	2.5360	3.0550	3.1730

AGE	1987	1988	1989	1990	1991
1	0.1250	0.1290	0.1260	0.1000	0.1050
2	0.2940	0.2560	0.2670	0.2310	0.3630
3	0.5670	0.5390	0.5720	0.6210	0.7480
4	1.1860	1.1180	1.0970	1.1270	1.2050
5	1.6670	1.8810	1.7960	1.6740	1.6870
6	2.5360	3.0260	3.3480	2.2070	2.1570

AGE	1992	1993	1994	1995	1996
1	0.0850	0.0760	0.1180	0.1080	0.1040
2	0.3130	0.2040	0.2570	0.2050	0.2100
3	0.6660	0.3940	0.4460	0.3330	0.3510
4	1.0970	0.6350	0.6750	0.4860	0.5220
5	1.5590	0.9110	0.9320	0.6620	0.7170
6	2.0170	1.2080	1.2060	0.8530	0.9300

AGE	1997	1998	1999	2000	2001
1	0.1860	0.1200	0.1050	0.1050	0.2000
2	0.3000	0.2300	0.2410	0.2410	0.3800
3	0.4380	0.3740	0.4280	0.4280	0.4500
4	0.5960	0.5470	0.6570	0.6570	0.7200
5	0.7700	0.7420	0.9150	0.9150	1.0900
6	0.9560	0.9530	1.1910	1.1910	2.3900

AGE	2002	2003	2004	2005	2006
1	0.2600	0.2200	0.2280	0.2790	0.2910
2	0.3200	0.3100	0.3440	0.3540	0.3310
3	0.5700	0.4000	0.4480	0.4420	0.4820
4	0.7200	0.6600	0.7850	0.5070	0.7250
5	0.9000	1.5300	1.6150	1.2320	0.9100
6	1.3900	2.5700	4.1650	3.7100	4.8610

Weight At Age - Input Data

AGE	2007
1	0.2150
2	0.3450
3	0.4680
4	0.8400
5	0.9700
6	3.8980

JAN-1 Weights at Age - Input Data

AGE	1982	1983	1984	1985	1986
1	0.1060	0.0780	0.0950	0.0770	0.1520
2	0.2120	0.1900	0.1890	0.2670	0.2620
3	0.3070	0.3680	0.3790	0.5790	0.7580
4	0.4830	0.8850	0.7580	1.2350	1.7590
5	1.0760	1.4000	1.5830	1.7500	2.8190
6	3.0330	2.8600	2.5360	3.0600	3.1730

AGE	1987	1988	1989	1990	1991
1	0.0870	0.0900	0.1090	0.0600	0.0360
2	0.2360	0.1790	0.1860	0.1040	0.2150
3	0.5240	0.3980	0.3830	0.4070	0.5430
4	1.2340	0.7960	0.7690	0.8650	0.9710
5	2.1270	1.4940	1.4170	1.3990	1.4460
6	2.5360	3.0260	3.3480	1.9450	1.9250

AGE	1992	1993	1994	1995	1996
1	0.0270	0.0360	0.0690	0.0710	0.0660
2	0.1810	0.1320	0.1810	0.1530	0.1520
3	0.4770	0.2920	0.3460	0.2650	0.2760
4	0.8750	0.5090	0.5560	0.4070	0.4330
5	1.3260	0.7690	0.8000	0.5720	0.6170
6	1.7900	1.0580	1.0670	0.7550	0.8220

AGE	1997	1998	1999	2000	2001
1	0.1390	0.0780	0.0590	0.0590	0.0430
2	0.2390	0.1700	0.1660	0.1660	0.1820
3	0.3660	0.2980	0.3290	0.3290	0.4250
4	0.5150	0.4570	0.5380	0.5380	0.7510
5	0.6810	0.6420	0.7830	0.7830	1.1340
6	0.8620	0.8460	1.0510	1.0510	1.5480

AGE	2002	2003	2004	2005	2006
1	0.0920	0.0670	0.1160	0.1010	0.1330
2	0.2650	0.2490	0.2790	0.2370	0.2870
3	0.5200	0.5440	0.5120	0.4310	0.5000
4	0.8360	0.9240	0.8020	0.6740	0.7620
5	1.1910	1.3560	1.1340	0.9530	1.0640
6	1.5660	1.8090	1.4930	1.2570	1.3920

JAN-1 Weights at Age - Input Data

AGE	2007	2008
1	0.1740	0.1740
2	0.3880	0.3880
3	0.6750	0.6750
4	1.0150	1.0150
5	1.3880	1.3880
6	1.7760	1.7760

SSB Weight At Age - Input Data

AGE	1982	1983	1984	1985	1986
1	0.0950	0.0700	0.0860	0.0690	0.1370
2	0.2120	0.1900	0.1890	0.2670	0.2620
3	0.3070	0.3680	0.3790	0.5790	0.7580
4	0.4830	0.8850	0.7580	1.2350	1.7590
5	1.0760	1.3950	1.5830	1.7480	2.8190
6	3.0330	2.8620	2.5360	3.0550	3.1730

AGE	1987	1988	1989	1990	1991
1	0.0780	0.0810	0.0980	0.1000	0.1100
2	0.2360	0.1790	0.1860	0.1800	0.3100
3	0.5240	0.3980	0.3830	0.5400	0.6800
4	1.2340	0.7960	0.7690	1.0400	1.1200
5	2.1270	1.4940	1.4170	1.5800	1.6000
6	2.5360	3.0260	3.3480	2.3900	2.3300

AGE	1992	1993	1994	1995	1996
1	0.0900	0.0800	0.1200	0.1100	0.1000
2	0.2600	0.1800	0.2300	0.1900	0.1900
3	0.6000	0.3600	0.4100	0.3100	0.3200
4	1.0200	0.5900	0.6300	0.4600	0.4900
5	1.4800	0.8600	0.8900	0.6300	0.6800
6	2.1900	1.3300	1.3200	0.9400	1.0200

AGE	1997	1998	1999	2000	2001
1	0.1900	0.1200	0.1100	0.1100	0.0860
2	0.2800	0.2100	0.2100	0.2100	0.2680
3	0.4100	0.3500	0.3900	0.3900	0.5470
4	0.5700	0.5200	0.6200	0.6200	0.8990
5	0.7400	0.7100	0.8700	0.8700	1.2970
6	1.0300	1.0400	1.3100	1.3100	1.7170

AGE	2002	2003	2004	2005	2006
1	0.1500	0.1250	0.1720	0.1480	0.1870
2	0.3580	0.3550	0.3640	0.3080	0.3660
3	0.6400	0.6880	0.6220	0.5230	0.6000
4	0.9740	1.0930	0.9310	0.7820	0.8790
5	1.3400	1.5360	1.2750	1.0720	1.1920
6	1.7190	1.9910	1.6410	1.3830	1.5290

SSB Weight At Age - Input Data

AGE	2007
1	0.2500
2	0.4950
3	0.8060
4	1.1610
5	1.5420
6	1.9330

Natural Mortality - Input Data

AGE	1982	1983	1984	1985	1986
1	0.2500	0.2500	0.2500	0.2500	0.2500
2	0.2500	0.2500	0.2500	0.2500	0.2500
3	0.2500	0.2500	0.2500	0.2500	0.2500
4	0.2500	0.2500	0.2500	0.2500	0.2500
5	0.2500	0.2500	0.2500	0.2500	0.2500
6	0.2500	0.2500	0.2500	0.2500	0.2500

AGE	1987	1988	1989	1990	1991
1	0.2500	0.2500	0.2500	0.2500	0.2500
2	0.2500	0.2500	0.2500	0.2500	0.2500
3	0.2500	0.2500	0.2500	0.2500	0.2500
4	0.2500	0.2500	0.2500	0.2500	0.2500
5	0.2500	0.2500	0.2500	0.2500	0.2500
6	0.2500	0.2500	0.2500	0.2500	0.2500

AGE	1992	1993	1994	1995	1996
1	0.2500	0.2500	0.2500	0.2500	0.2500
2	0.2500	0.2500	0.2500	0.2500	0.2500
3	0.2500	0.2500	0.2500	0.2500	0.2500
4	0.2500	0.2500	0.2500	0.2500	0.2500
5	0.2500	0.2500	0.2500	0.2500	0.2500
6	0.2500	0.2500	0.2500	0.2500	0.2500

AGE	1997	1998	1999	2000	2001
1	0.2500	0.2500	0.2500	0.2500	0.2500
2	0.2500	0.2500	0.2500	0.2500	0.2500
3	0.2500	0.2500	0.2500	0.2500	0.2500
4	0.2500	0.2500	0.2500	0.2500	0.2500
5	0.2500	0.2500	0.2500	0.2500	0.2500
6	0.2500	0.2500	0.2500	0.2500	0.2500

AGE	2002	2003	2004	2005	2006
1	0.2500	0.2500	0.2500	0.2500	0.2500
2	0.2500	0.2500	0.2500	0.2500	0.2500
3	0.2500	0.2500	0.2500	0.2500	0.2500
4	0.2500	0.2500	0.2500	0.2500	0.2500
5	0.2500	0.2500	0.2500	0.2500	0.2500
6	0.2500	0.2500	0.2500	0.2500	0.2500

Natural Mortality - Input Data

AGE	2007
1	0.2500
2	0.2500
3	0.2500
4	0.2500
5	0.2500
6	0.2500

Proportion of Natural Mortality Before Spawning = 0.4000
Proportion of Fishing Mortality Before Spawning = 0.4000

Maturity - Input Data

AGE	1982	1983	1984	1985	1986
1	0.9000	0.9000	0.9000	0.9000	0.9000
2	1.0000	1.0000	1.0000	1.0000	1.0000
3	1.0000	1.0000	1.0000	1.0000	1.0000
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	1987	1988	1989	1990	1991
1	0.9000	0.9000	0.9000	0.9000	0.9000
2	1.0000	1.0000	1.0000	1.0000	1.0000
3	1.0000	1.0000	1.0000	1.0000	1.0000
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	1992	1993	1994	1995	1996
1	0.9000	0.9000	0.9000	0.9000	0.9000
2	1.0000	1.0000	1.0000	1.0000	1.0000
3	1.0000	1.0000	1.0000	1.0000	1.0000
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	1997	1998	1999	2000	2001
1	0.9000	0.9000	0.9000	0.9000	0.9000
2	1.0000	1.0000	1.0000	1.0000	1.0000
3	1.0000	1.0000	1.0000	1.0000	1.0000
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	2002	2003	2004	2005	2006
1	0.9000	0.9000	0.9000	0.9000	0.9000
2	1.0000	1.0000	1.0000	1.0000	1.0000
3	1.0000	1.0000	1.0000	1.0000	1.0000
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

Maturity - Input Data

AGE	2007
1	0.9000
2	1.0000
3	1.0000
4	1.0000
5	1.0000
6	1.0000

Input Partial Recruitment

AGE	
1	0.1200
2	0.2800
3	0.5700
4	1.0000
5	1.0000

Input F-Plus Ratio

YEAR	
1982	1.0000
1983	1.0000
1984	1.0000
1985	1.0000
1986	1.0000
1987	1.0000
1988	1.0000
1989	1.0000
1990	1.0000
1991	1.0000
1992	1.0000
1993	1.0000
1994	1.0000
1995	1.0000
1996	1.0000
1997	1.0000
1998	1.0000
1999	1.0000
2000	1.0000
2001	1.0000
2002	1.0000
2003	1.0000
2004	1.0000
2005	1.0000
2006	1.0000
2007	1.0000

SURVEY - INPUT DATA

INDEX	1	2	3	4	5
SURVEY TAG	DEDFW	DEDFW	DEDFW	DEDFW	DEDFW
AGE	1	2	3	4	5
TIME	MEAN	MEAN	MEAN	MEAN	MEAN
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1	1	1
1982	4.7100	7.3300	3.0200	1.4500	0.2400
1983	5.3900	8.0000	3.3400	1.5700	0.2000
1984	2.9900	5.3200	2.4200	1.1500	0.1300
1985	0.0000	0.0000	0.0000	0.0000	0.0000
1986	0.0000	0.0000	0.0000	0.0000	0.0000
1987	0.0000	0.0000	0.0000	0.0000	0.0000
1988	0.0000	0.0000	0.0000	0.0000	0.0000
1989	0.0000	0.0000	0.0000	0.0000	0.0000
1990	0.0000	0.0000	0.0000	0.0000	0.0000
1991	27.1588	3.6382	0.6383	0.0000	0.0000
1992	21.1512	2.6081	0.0318	0.0318	0.0000
1993	50.2801	25.4220	3.9002	0.5169	0.0000
1994	113.4222	68.5360	23.6497	0.9653	0.0000
1995	75.2633	53.4866	15.7403	5.4251	0.0764
1996	44.0437	48.3017	111.1071	23.8182	6.3870
1997	33.4142	25.0014	13.5633	34.5991	2.9623
1998	23.3566	24.6266	20.3749	11.5955	20.7062
1999	42.0661	20.1604	17.0310	6.7402	2.5878
2000	97.8468	50.3831	23.6377	5.7446	0.6592
2001	13.1139	42.6313	18.7718	5.5263	0.5702
2002	89.3537	23.3228	27.9672	3.8367	0.4039
2003	50.1607	13.9835	1.2183	0.4237	0.0000
2004	26.3628	22.4048	0.1089	0.0000	0.0000
2005	12.0686	14.5354	2.3872	0.0000	0.0000
2006	58.3793	37.1285	10.0217	0.7848	0.0000
2007	23.4698	17.4276	2.0963	0.1644	0.0000
2008	0.0000	0.0000	0.0000	0.0000	0.0000

SURVEY - INPUT DATA

INDEX	6	7	8	9	10
SURVEY TAG	DEDFW	NJDEP	NJDEP	NJDEP	NJDEP
AGE	6	1	2	3	4
TIME	MEAN	JAN-1	JAN-1	JAN-1	JAN-1
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1	1	1
1982	0.3300	0.0000	0.0000	0.0000	0.0000
1983	0.1800	0.0000	0.0000	0.0000	0.0000
1984	0.0700	0.0000	0.0000	0.0000	0.0000
1985	0.0000	0.0000	0.0000	0.0000	0.0000
1986	0.0000	0.0000	0.0000	0.0000	0.0000
1987	0.0000	0.0000	0.0000	0.0000	0.0000
1988	0.0000	0.0000	0.0000	0.0000	0.0000
1989	0.0000	0.0000	0.0000	0.0000	0.0000
1990	0.0000	0.0356	0.0260	0.0057	0.0024
1991	0.0000	0.0745	0.0523	0.0099	0.0023
1992	0.0000	0.0892	0.3206	0.0585	0.0007
1993	0.0000	0.0405	0.1103	0.0445	0.0181
1994	0.0000	0.2556	0.1489	0.0316	0.0000
1995	0.0000	0.0398	0.1607	0.0702	0.0082
1996	0.1331	0.0543	0.0434	0.1177	0.0273
1997	0.4740	0.1323	0.0342	0.0466	0.1339
1998	1.3252	0.0087	0.0440	0.0679	0.0350
1999	4.0331	0.0222	0.0396	0.0302	0.0053
2000	0.9418	0.0397	0.1255	0.1011	0.2153
2001	0.0877	0.0047	0.0788	0.1381	0.0582
2002	0.0000	0.1398	0.0546	0.0239	0.0029
2003	0.0000	0.0416	0.0407	0.0021	0.0003
2004	0.0000	0.0591	0.1735	0.0726	0.0129
2005	0.0000	0.0977	0.0934	0.0099	0.0008
2006	0.0000	0.0746	0.0747	0.0161	0.0000
2007	0.0000	0.0926	0.0122	0.0009	0.0001
2008	0.0000	0.0591	0.1735	0.0726	0.0129

SURVEY - INPUT DATA

INDEX	11	12	13	14	15
SURVEY TAG	NJDEP	NJDEP	NCGill	NCGill	NCGill
AGE	5	6	1	2	3
TIME	JAN-1	JAN-1	MEAN	MEAN	MEAN
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1	1	1
1982	0.0000	0.0000	0.0000	0.0000	0.0000
1983	0.0000	0.0000	0.0000	0.0000	0.0000
1984	0.0000	0.0000	0.0000	0.0000	0.0000
1985	0.0000	0.0000	0.0000	0.0000	0.0000
1986	0.0000	0.0000	0.0000	0.0000	0.0000
1987	0.0000	0.0000	0.0000	0.0000	0.0000
1988	0.0000	0.0000	0.0000	0.0000	0.0000
1989	0.0000	0.0000	0.0000	0.0000	0.0000
1990	0.0003	0.0000	0.0000	0.0000	0.0000
1991	0.0001	0.0000	0.0000	0.0000	0.0000
1992	0.0000	0.0000	0.0000	0.0000	0.0000
1993	0.0014	0.0000	0.0000	0.0000	0.0000
1994	0.0000	0.0000	0.0000	0.0000	0.0000
1995	0.0000	0.0000	0.0000	0.0000	0.0000
1996	0.0107	0.0009	0.0000	0.0000	0.0000
1997	0.0116	0.0000	0.0000	0.0000	0.0000
1998	0.0821	0.0113	0.0000	0.0000	0.0000
1999	0.0015	0.0012	0.0000	0.0000	0.0000
2000	0.0112	0.0002	0.0000	0.0000	0.0000
2001	0.0137	0.0029	0.1480	1.4886	0.3694
2002	0.0000	0.0000	0.1729	0.2674	1.0873
2003	0.0000	0.0000	0.1454	0.3669	0.3450
2004	0.0002	0.0000	0.2269	0.5946	0.5840
2005	0.0000	0.0000	0.1719	0.4862	0.6925
2006	0.0000	0.0000	0.1974	0.3766	0.5534
2007	0.0000	0.0000	0.1597	0.2564	0.1790
2008	0.0002	0.0000	0.0000	0.0000	0.0000

SURVEY - INPUT DATA

INDEX	16	17	18	19	20
SURVEY TAG	NCGill	NCGill	NCGill	SEAFALL	SEAFALL
AGE	4	5	6	1	2
TIME	MEAN	MEAN	MEAN	JAN-1	JAN-1
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1	1	1
1982	0.0000	0.0000	0.0000	0.0000	0.0000
1983	0.0000	0.0000	0.0000	0.0000	0.0000
1984	0.0000	0.0000	0.0000	0.0000	0.0000
1985	0.0000	0.0000	0.0000	0.0000	0.0000
1986	0.0000	0.0000	0.0000	0.0000	0.0000
1987	0.0000	0.0000	0.0000	0.0000	0.0000
1988	0.0000	0.0000	0.0000	0.0000	0.0000
1989	0.0000	0.0000	0.0000	0.0000	0.0000
1990	0.0000	0.0000	0.0000	0.0000	0.0000
1991	0.0000	0.0000	0.0000	2.1500	1.3200
1992	0.0000	0.0000	0.0000	0.8500	5.6700
1993	0.0000	0.0000	0.0000	0.2000	0.6200
1994	0.0000	0.0000	0.0000	15.2600	27.4000
1995	0.0000	0.0000	0.0000	43.7800	7.0200
1996	0.0000	0.0000	0.0000	3.6400	4.1600
1997	0.0000	0.0000	0.0000	20.3400	1.1500
1998	0.0000	0.0000	0.0000	0.7100	4.0900
1999	0.0000	0.0000	0.0000	4.5400	9.8700
2000	0.0000	0.0000	0.0000	24.5400	4.2100
2001	0.1459	0.1127	0.0000	8.9600	5.0100
2002	0.2423	0.0569	0.0089	5.6000	0.9300
2003	0.6802	0.0583	0.0000	44.3300	12.0000
2004	0.1885	0.0722	0.0000	22.2200	13.2200
2005	0.1747	0.0343	0.0009	10.8900	22.4300
2006	0.2130	0.0543	0.0008	238.5600	221.3300
2007	0.1140	0.0520	0.0008	2.6200	32.4200
2008	0.0000	0.0000	0.0000	0.6600	1.9000

SURVEY - INPUT DATA

INDEX	21	22	23	24	25
SURVEY TAG	SEAFALL	SEAFALL	SEAFALL	SEAFALL	RI
AGE	3	4	5	6	1
TIME	JAN-1	JAN-1	JAN-1	JAN-1	JAN-1
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1	1	1
1982	0.0000	0.0000	0.0000	0.0000	5.0400
1983	0.0000	0.0000	0.0000	0.0000	4.1100
1984	0.0000	0.0000	0.0000	0.0000	0.7900
1985	0.0000	0.0000	0.0000	0.0000	0.3800
1986	0.0000	0.0000	0.0000	0.0000	2.3700
1987	0.0000	0.0000	0.0000	0.0000	0.7000
1988	0.0000	0.0000	0.0000	0.0000	0.3300
1989	0.0000	0.0000	0.0000	0.0000	0.9000
1990	0.0000	0.0000	0.0000	0.0000	0.4200
1991	0.0500	0.0100	0.0000	0.0000	2.4500
1992	0.4200	0.0900	0.0100	0.0100	1.6600
1993	0.1100	0.0100	0.0100	0.0000	2.3500
1994	1.0300	0.0100	0.0100	0.0100	1.3000
1995	0.4800	0.1800	0.0300	0.0000	1.5700
1996	0.4500	0.0700	0.0000	0.0100	0.0900
1997	0.3100	0.0300	0.0000	0.0100	6.3400
1998	1.8400	0.7500	0.0800	0.0100	6.6900
1999	4.7200	0.9600	0.5900	0.0600	1.3900
2000	0.2700	0.0500	0.0100	0.0100	1.0000
2001	0.7400	0.0400	0.0100	0.0100	2.1000
2002	0.3300	0.0100	0.0100	0.0100	2.3900
2003	0.7700	1.4000	0.0100	0.0100	2.0700
2004	0.9100	0.1100	0.2200	0.0100	16.5400
2005	2.4400	0.0900	0.0000	0.0000	0.4000
2006	33.7400	0.7500	0.0000	0.0000	8.6400
2007	7.3000	1.7800	0.0000	0.0000	0.1600
2008	0.3900	0.1800	0.0000	0.0000	4.8300

SURVEY - INPUT DATA

INDEX	26	27	28	29	30
SURVEY TAG	CT	NY	DEDFW	MDDNR1	MDDNR2
AGE	1	1	1	1	1
TIME	JAN-1	JAN-1	JAN-1	JAN-1	JAN-1
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1	1	1
1982	0.0000	0.0000	5.9800	0.2400	1.8500
1983	0.0000	0.0000	11.4900	0.2200	0.0000
1984	0.0000	0.0000	4.4700	1.3200	0.0000
1985	1.0000	0.0000	6.6700	0.1300	0.5200
1986	6.1900	0.0000	9.3500	1.6600	1.4000
1987	13.1600	0.0000	12.9400	0.4400	0.8500
1988	0.6300	0.6000	5.9800	0.3600	0.5600
1989	3.4900	0.1100	4.7300	0.2300	0.0000
1990	8.6900	1.3800	11.1100	0.4400	0.8700
1991	5.5600	0.5500	8.7300	0.9500	1.7200
1992	11.9500	20.6400	20.0700	0.7800	1.8900
1993	3.0500	3.2600	14.7200	3.2400	1.8100
1994	4.0800	1.0300	14.7900	1.5900	0.9100
1995	11.1900	8.3300	11.4700	2.3300	1.8400
1996	5.2200	1.6000	13.4900	5.9500	4.4400
1997	15.2300	24.4900	11.9300	6.4000	3.1800
1998	12.3800	18.7500	15.4000	4.2800	3.0600
1999	5.0200	1.0300	11.3500	5.8700	2.8000
2000	30.9300	8.4300	13.5100	3.2600	2.7600
2001	63.3100	15.8800	14.1600	6.5400	2.3400
2002	40.0900	16.1800	7.5700	8.1000	2.5600
2003	41.3500	12.1700	5.9600	3.9200	0.6100
2004	49.4100	7.0100	10.4400	4.8900	5.6400
2005	58.9800	5.5200	8.3900	1.6200	3.3900
2006	25.8600	31.9800	16.8400	3.5500	4.9800
2007	1.0500	8.7000	5.3500	2.4100	1.5000
2008	63.9300	12.0700	13.7000	1.6400	2.3200

SURVEY - INPUT DATA

INDEX	31	32	33	34	35
SURVEY TAG	VIMS	NCDMF	NCDMF	MRFSS	MRFSS
AGE	1	1	2	3	4
TIME	JAN-1	JAN-1	JAN-1	JAN-1	JAN-1
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1	1	1
1982	6.0200	0.0000	0.0000	0.1865	0.2176
1983	10.9500	0.0000	0.0000	0.0084	0.0588
1984	10.8500	0.0000	0.0000	0.1771	0.1631
1985	6.0500	0.0000	0.0000	0.1033	0.0919
1986	37.0400	0.0000	0.0000	0.0933	0.0758
1987	4.6200	0.0000	0.0000	0.3885	0.1329
1988	17.8500	12.1400	53.1400	0.2272	0.1262
1989	21.7200	101.5000	36.3800	0.1498	0.1915
1990	21.2700	14.2000	13.6400	0.0527	0.0527
1991	30.0100	50.2000	17.1800	0.0574	0.0309
1992	15.3200	36.9600	14.7400	0.0530	0.0485
1993	15.9100	42.7100	19.9200	0.0370	0.0328
1994	15.4200	8.7000	67.7100	0.0300	0.0258
1995	7.0400	68.0600	71.4300	0.0355	0.0659
1996	11.0000	38.2100	42.2800	0.0271	0.0588
1997	7.4200	72.3900	32.6200	0.0137	0.0504
1998	14.8200	32.7900	55.7000	0.0151	0.0605
1999	9.9500	70.4400	25.8100	0.0162	0.0647
2000	16.2500	99.9000	48.3600	0.0105	0.0264
2001	11.0900	62.9900	123.6600	0.0109	0.0274
2002	11.5200	30.3000	52.7500	0.0368	0.0263
2003	8.5900	22.0000	20.6400	0.0051	0.0462
2004	5.4200	23.9300	17.3900	0.0094	0.0047
2005	10.4700	28.7500	31.2400	0.0135	0.0058
2006	7.1000	28.7600	29.0000	0.0489	0.0384
2007	6.2000	39.0900	83.8200	0.0084	0.0196
2008	14.3700	56.7700	21.5300	0.0113	0.0054

SURVEY - INPUT DATA

INDEX	36	37	38		
SURVEY TAG	MRFSS	MRFSS	rec		
AGE	5	6	2 - 6	NUMBERS	NUMBERS
TIME	JAN-1	JAN-1	MEAN	NUMBERS	NUMBERS
TYPE	NUMBERS	NUMBERS	NUMBERS	NUMBERS	NUMBERS
RETRO FLAG	1	1	1		

1982	0.2131	0.1066	0.2021
1983	0.0671	0.0630	0.5109
1984	0.1165	0.0326	0.3001
1985	0.0632	0.0172	0.2777
1986	0.0525	0.0175	0.7681
1987	0.0664	0.0102	0.5178
1988	0.0707	0.0101	0.5738
1989	0.1290	0.0291	0.1673
1990	0.0341	0.0093	0.1557
1991	0.0177	0.0044	0.1778
1992	0.0265	0.0088	0.1928
1993	0.0287	0.0082	0.1651
1994	0.0172	0.0043	0.3849
1995	0.0304	0.0000	0.5079
1996	0.0407	0.0045	0.6054
1997	0.1054	0.0321	0.5177
1998	0.0958	0.0302	0.5297
1999	0.1024	0.0323	0.4101
2000	0.0632	0.0474	0.4585
2001	0.0328	0.0711	0.2818
2002	0.0158	0.0158	0.2379
2003	0.0205	0.0154	0.1162
2004	0.0047	0.0047	0.1154
2005	0.0021	0.0020	0.2243
2006	0.0058	0.0001	0.1704
2007	0.0088	0.0009	0.0884
2008	0.0026	0.0003	0.0000

Additional Output Files

Population File C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R
Auxilliary File C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R
Covariance File C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R
Residuals File C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R
Log File C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R

Bootstrap Files

Bootstrap Stock Numbers C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R
Bootstrap Fishing Mortality C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R
Bootstrap Biomass C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R
Bootstrap Catchability C:\WEAKVPA\FINAL RUNS\UNLAGGED NJ\FINAL RUN\FINAL RUN BOOT AND R

Estimation Results

JAN-1 Population Numbers

AGE	1982	1983	1984	1985	1986
1	45006.	41012.	44143.	63406.	70885.
2	28860.	26376.	24931.	25244.	36997.
3	10964.	11224.	9394.	6722.	9886.
4	4051.	3800.	3706.	1739.	2422.
5	1052.	777.	590.	316.	351.
6	932.	580.	310.	82.	69.
Total	90865.	83769.	83073.	97508.	120610.

AGE	1987	1988	1989	1990	1991
1	54403.	26316.	20817.	26655.	29497.
2	38942.	28181.	13374.	12966.	12476.
3	15109.	17841.	8028.	5802.	5842.
4	3421.	7971.	4810.	2508.	2696.
5	335.	1374.	1031.	1231.	909.
6	25.	93.	139.	185.	112.
Total	112235.	81776.	48198.	49348.	51532.

AGE	1992	1993	1994	1995	1996
1	35646.	36736.	49285.	23406.	29734.
2	14433.	23219.	24247.	35078.	14430.
3	4665.	5987.	12530.	15075.	24047.
4	2166.	1717.	2765.	6649.	8621.
5	911.	605.	361.	804.	3746.
6	112.	73.	70.	220.	272.
Total	57932.	68337.	89257.	81232.	80849.

AGE	1997	1998	1999	2000	2001
1	16171.	11394.	9380.	12496.	6263.
2	20962.	11087.	7759.	6066.	8523.
3	8882.	14224.	6762.	4903.	3216.
4	14322.	4378.	7599.	3299.	2092.
5	3950.	6349.	1658.	3031.	1130.
6	1064.	1972.	2616.	799.	833.
Total	65352.	49404.	35774.	30594.	22057.

AGE	2002	2003	2004	2005	2006
1	5969.	5630.	6107.	4480.	8662.
2	2772.	2398.	4090.	4009.	3196.
3	4065.	1227.	1042.	1869.	1584.
4	1226.	854.	142.	238.	380.
5	578.	249.	61.	13.	23.
6	345.	302.	47.	34.	18.
Total	14954.	10662.	11489.	10643.	13864.

JAN-1 Population Numbers

AGE	2007	2008
1	15343.	16443.
2	6090.	11407.
3	1931.	3737.
4	408.	1060.
5	79.	200.
6	15.	5.
Total	23866.	32852.

Fishing Mortality Calculated

AGE	1982	1983	1984	1985	1986
1	0.2844	0.2477	0.3088	0.2887	0.3490
2	0.6944	0.7824	1.0608	0.6875	0.6456
3	0.8095	0.8582	1.4370	0.7706	0.8113
4	1.4015	1.6135	2.2116	1.3506	1.7287
5	1.4015	1.6135	2.2116	1.3506	1.7287
6	1.4015	1.6135	2.2116	1.3506	1.7287

AGE	1987	1988	1989	1990	1991
1	0.4078	0.4269	0.2234	0.5092	0.4648
2	0.5306	1.0057	0.5851	0.5473	0.7337
3	0.3895	1.0608	0.9134	0.5166	0.7423
4	0.6618	1.7955	1.1127	0.7645	0.8351
5	0.6618	1.7955	1.1127	0.7645	0.8351
6	0.6618	1.7955	1.1127	0.7645	0.8351

AGE	1992	1993	1994	1995	1996
1	0.1787	0.1655	0.0900	0.2337	0.0996
2	0.6299	0.3669	0.2253	0.1276	0.2352
3	0.7496	0.5227	0.3837	0.3088	0.2682
4	1.0256	1.3097	0.9848	0.3237	0.5304
5	1.0256	1.3097	0.9848	0.3237	0.5304
6	1.0256	1.3097	0.9848	0.3237	0.5304

AGE	1997	1998	1999	2000	2001
1	0.1274	0.1343	0.1858	0.1326	0.5652
2	0.1378	0.2445	0.2090	0.3845	0.4904
3	0.4576	0.3769	0.4676	0.6017	0.7142
4	0.5634	0.7208	0.6693	0.8219	1.0366
5	0.5634	0.7208	0.6693	0.8219	1.0366
6	0.5634	0.7208	0.6693	0.8219	1.0366

AGE	2002	2003	2004	2005	2006
1	0.6618	0.0696	0.1710	0.0879	0.1024
2	0.5647	0.5839	0.5331	0.6783	0.2539
3	1.3097	1.9079	1.2269	1.3426	1.1059
4	1.3437	2.3915	2.1654	2.0733	1.3175
5	1.3437	2.3915	2.1654	2.0733	1.3175
6	1.3437	2.3915	2.1654	2.0733	1.3175

Fishing Mortality Calculated

AGE	2007
1	0.0464
2	0.2383
3	0.3495
4	0.4631
5	0.5570
6	0.5570

Average Fishing Mortality For Ages 4- 5

Year	Average F	N Weighted	Biomass Wtd	Catch Wtd
1982	1.4015	1.4015	1.4015	1.4015
1983	1.6135	1.6135	1.6135	1.6135
1984	2.2116	2.2116	2.2116	2.2116
1985	1.3506	1.3506	1.3506	1.3506
1986	1.7287	1.7287	1.7287	1.7287
1987	0.6618	0.6618	0.6618	0.6618
1988	1.7955	1.7955	1.7955	1.7955
1989	1.1127	1.1127	1.1127	1.1127
1990	0.7645	0.7645	0.7645	0.7645
1991	0.8351	0.8351	0.8351	0.8351
1992	1.0256	1.0256	1.0256	1.0256
1993	1.3097	1.3097	1.3097	1.3097
1994	0.9848	0.9848	0.9848	0.9848
1995	0.3237	0.3237	0.3237	0.3237
1996	0.5304	0.5304	0.5304	0.5304
1997	0.5634	0.5634	0.5634	0.5634
1998	0.7208	0.7208	0.7208	0.7208
1999	0.6693	0.6693	0.6693	0.6693
2000	0.8219	0.8219	0.8219	0.8219

2001	1.0366	1.0366	1.0366	1.0366
2002	1.3437	1.3437	1.3437	1.3437
2003	2.3915	2.3915	2.3915	2.3915
2004	2.1654	2.1654	2.1654	2.1654
2005	2.0733	2.0733	2.0733	2.0733
2006	1.3175	1.3175	1.3175	1.3175
2007	0.5101	0.4784	0.4828	0.4779

Average Fishing Mortality For Ages 1- 5

Year	Average F	N Weighted	Biomass Wtd	Catch Wtd
1982	0.9182	0.5433	0.7299	0.6642
1983	1.0231	0.5748	0.9195	0.7473
1984	1.4460	0.7622	1.2146	1.0228
1985	0.8896	0.4477	0.6967	0.5392
1986	1.0526	0.5097	0.7580	0.5867
1987	0.5303	0.4564	0.4913	0.4651
1988	1.2169	0.9216	1.2525	1.0414
1989	0.7895	0.5474	0.8092	0.7008
1990	0.6204	0.5395	0.6247	0.5453
1991	0.7222	0.5875	0.7466	0.6137
1992	0.7219	0.3824	0.7490	0.5401
1993	0.7349	0.3042	0.5366	0.4193
1994	0.5337	0.1994	0.3409	0.3293
1995	0.2635	0.2100	0.2348	0.2365
1996	0.3328	0.2403	0.3384	0.3145
1997	0.3699	0.3003	0.3954	0.4159
1998	0.4395	0.3655	0.5010	0.4669
1999	0.4402	0.3837	0.5308	0.4768
2000	0.5525	0.4075	0.6482	0.5633
2001	0.7686	0.6293	0.8023	0.6666
2002	1.0447	0.9078	1.1444	0.9880
2003	1.4689	0.6538	1.5688	1.4585
2004	1.2523	0.4319	0.6936	0.6660
2005	1.2511	0.5790	0.8919	0.9359
2006	0.8194	0.2876	0.5168	0.6327
2007	0.3309	0.1288	0.2034	0.2287

Back Calculated Partial Recruitment

AGE	1982	1983	1984	1985	1986
1	0.2029	0.1535	0.1396	0.2138	0.2019
2	0.4954	0.4849	0.4797	0.5090	0.3735
3	0.5776	0.5319	0.6498	0.5705	0.4693
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	1987	1988	1989	1990	1991
1	0.6162	0.2377	0.2008	0.6660	0.5566
2	0.8018	0.5601	0.5258	0.7158	0.8786
3	0.5885	0.5908	0.8208	0.6758	0.8889
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	1992	1993	1994	1995	1996
1	0.1742	0.1264	0.0914	0.7218	0.1878
2	0.6142	0.2801	0.2287	0.3941	0.4435
3	0.7309	0.3991	0.3896	0.9540	0.5057
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	1997	1998	1999	2000	2001
1	0.2262	0.1863	0.2776	0.1614	0.5453
2	0.2446	0.3391	0.3123	0.4678	0.4731
3	0.8122	0.5229	0.6988	0.7321	0.6890
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

AGE	2002	2003	2004	2005	2006
1	0.4925	0.0291	0.0790	0.0424	0.0777
2	0.4202	0.2441	0.2462	0.3272	0.1927
3	0.9747	0.7978	0.5666	0.6475	0.8394
4	1.0000	1.0000	1.0000	1.0000	1.0000
5	1.0000	1.0000	1.0000	1.0000	1.0000
6	1.0000	1.0000	1.0000	1.0000	1.0000

Back Calculated Partial Recruitment

AGE	2007
1	0.0834
2	0.4277
3	0.6275
4	0.8314
5	1.0000
6	1.0000

JAN-1 Biomass

AGE	1982	1983	1984	1985	1986
1	4771.	3199.	4194.	4882.	10774.
2	6118.	5011.	4712.	6740.	9693.
3	3366.	4131.	3560.	3892.	7494.
4	1956.	3363.	2809.	2147.	4261.
5	1132.	1087.	933.	553.	989.
6	2828.	1658.	785.	250.	220.
Total	20171.	18450.	16993.	18465.	33431.

AGE	1987	1988	1989	1990	1991
1	4733.	2368.	2269.	1599.	1062.
2	9190.	5044.	2488.	1348.	2682.
3	7917.	7101.	3075.	2362.	3172.
4	4221.	6345.	3699.	2170.	2617.
5	712.	2054.	1461.	1722.	1315.
6	64.	280.	466.	361.	216.
Total	26838.	23192.	13457.	9562.	11065.

AGE	1992	1993	1994	1995	1996
1	962.	1323.	3401.	1662.	1962.
2	2612.	3065.	4389.	5367.	2193.
3	2225.	1748.	4335.	3995.	6637.
4	1895.	874.	1537.	2706.	3733.
5	1208.	465.	289.	460.	2311.
6	201.	77.	75.	166.	223.
Total	9104.	7551.	14026.	14356.	17060.

AGE	1997	1998	1999	2000	2001
1	2248.	889.	553.	737.	269.
2	5010.	1885.	1288.	1007.	1551.
3	3251.	4239.	2225.	1613.	1367.
4	7376.	2001.	4088.	1775.	1571.
5	2690.	4076.	1298.	2373.	1281.
6	917.	1668.	2749.	840.	1290.
Total	21492.	14757.	12202.	8345.	7329.

AGE	2002	2003	2004	2005	2006
1	549.	377.	708.	453.	1152.
2	735.	597.	1141.	950.	917.
3	2114.	668.	533.	806.	792.
4	1025.	789.	114.	160.	290.
5	688.	338.	69.	12.	25.
6	540.	547.	70.	42.	25.
Total	5650.	3316.	2636.	2423.	3201.

JAN-1 Biomass

AGE	2007	2008
1	2670.	2861.
2	2363.	4426.
3	1303.	2523.
4	414.	1076.
5	110.	278.
6	27.	8.
Total	6887.	11172.

Mean Biomass

AGE	1982	1983	1984	1985	1986
1	4951.	3909.	5377.	6961.	10079.
2	5210.	4178.	4084.	7339.	11854.
3	3522.	3290.	2573.	4802.	7816.
4	1627.	2591.	1633.	2331.	2862.
5	726.	835.	365.	406.	451.
6	1384.	752.	292.	125.	96.
Total	17421.	15556.	14324.	21965.	33158.

AGE	1987	1988	1989	1990	1991
1	4983.	2467.	2089.	1868.	2213.
2	7948.	4109.	2421.	2064.	2882.
3	6329.	5358.	2714.	2517.	2771.
4	2662.	3793.	2881.	1776.	1982.
5	366.	1100.	1011.	1295.	936.
6	42.	119.	255.	257.	148.
Total	22330.	16946.	11371.	9777.	10933.

AGE	1992	1993	1994	1995	1996
1	2464.	2285.	4930.	2004.	2610.
2	3004.	3535.	4960.	5989.	2401.
3	1964.	1643.	4139.	3846.	6587.
4	1342.	552.	1072.	2459.	3124.
5	802.	279.	193.	405.	1865.
6	128.	44.	49.	143.	176.
Total	9705.	8338.	15342.	14846.	16762.

AGE	1997	1998	1999	2000	2001
1	2505.	1135.	798.	1090.	857.
2	5213.	2012.	1499.	1083.	2288.
3	2789.	3952.	2065.	1412.	929.
4	5841.	1532.	3265.	1330.	847.
5	2082.	3015.	992.	1701.	693.
6	696.	1202.	2038.	584.	1120.
Total	19126.	12849.	10658.	7200.	6734.

AGE	2002	2003	2004	2005	2006
1	1018.	1060.	1136.	1061.	2125.
2	607.	504.	976.	925.	831.
3	1173.	201.	244.	413.	418.
4	441.	198.	42.	47.	139.
5	260.	134.	37.	6.	11.
6	239.	273.	74.	48.	44.
Total	3739.	2371.	2509.	2500.	3567.

Mean Biomass

AGE	2007
1	2855.
2	1662.
3	680.
4	245.
5	53.
6	41.
Total	5535.

Spawning Stock Biomass

AGE	1982	1983	1984	1985	1986
1	3107.	2117.	2732.	3174.	6878.
2	4194.	3316.	2789.	4633.	6775.
3	2203.	2651.	1813.	2587.	4902.
4	1011.	1596.	1049.	1132.	1931.
5	585.	514.	349.	291.	448.
6	1461.	787.	293.	132.	100.
Total	12560.	10982.	9026.	11949.	21033.

AGE	1987	1988	1989	1990	1991
1	2936.	1463.	1519.	1771.	2194.
2	6726.	3053.	1781.	1697.	2609.
3	6130.	4203.	1931.	2306.	2671.
4	2931.	2799.	2145.	1738.	1956.
5	495.	906.	847.	1296.	943.
6	45.	124.	270.	295.	170.
Total	19262.	12548.	8493.	9103.	10543.

AGE	1992	1993	1994	1995	1996
1	2432.	2240.	4646.	1910.	2327.
2	2639.	3266.	4611.	5731.	2258.
3	1877.	1582.	3987.	3737.	6254.
4	1326.	543.	1063.	2431.	3092.
5	809.	279.	196.	403.	1864.
6	147.	52.	57.	165.	203.
Total	9231.	7961.	14560.	14376.	15998.

AGE	1997	1998	1999	2000	2001
1	2378.	1055.	780.	1062.	350.
2	5026.	1911.	1356.	988.	1699.
3	2744.	3874.	1979.	1360.	1196.
4	5896.	1544.	3262.	1332.	1124.
5	2111.	3057.	999.	1717.	876.
6	791.	1391.	2373.	682.	855.
Total	18947.	12832.	10748.	7141.	6100.

AGE	2002	2003	2004	2005	2006
1	560.	557.	799.	521.	1266.
2	716.	610.	1088.	852.	956.
3	1394.	356.	359.	517.	553.
4	631.	325.	50.	73.	179.
5	409.	133.	30.	5.	15.
6	313.	209.	29.	18.	15.
Total	4024.	2190.	2355.	1987.	2983.

Spawning Stock Biomass

AGE	2007
1	3066.
2	2480.
3	1224.
4	356.
5	89.
6	21.
Total	7236.

Catch Biomass

AGE	1982	1983	1984	1985	1986
1	1408.	968.	1661.	2010.	3517.
2	3618.	3269.	4333.	5046.	7653.
3	2851.	2824.	3697.	3701.	6341.
4	2281.	4181.	3612.	3149.	4948.
5	1018.	1347.	808.	548.	780.
6	1940.	1214.	645.	168.	165.
Total	13116.	13803.	14756.	14621.	23404.

AGE	1987	1988	1989	1990	1991
1	2032.	1053.	467.	951.	1029.
2	4217.	4132.	1416.	1130.	2115.
3	2465.	5684.	2479.	1300.	2057.
4	1761.	6811.	3206.	1358.	1655.
5	242.	1976.	1125.	990.	782.
6	28.	214.	283.	197.	124.
Total	10745.	19870.	8976.	5925.	7761.

AGE	1992	1993	1994	1995	1996
1	440.	378.	444.	468.	260.
2	1892.	1297.	1117.	764.	565.
3	1473.	859.	1588.	1188.	1767.
4	1377.	723.	1055.	796.	1657.
5	823.	365.	190.	131.	989.
6	131.	58.	48.	46.	93.
Total	6136.	3680.	4443.	3394.	5330.

AGE	1997	1998	1999	2000	2001
1	319.	152.	148.	145.	484.
2	718.	492.	313.	416.	1122.
3	1276.	1490.	966.	850.	663.
4	3291.	1105.	2185.	1093.	878.
5	1173.	2173.	664.	1398.	718.
6	392.	867.	1364.	480.	1161.
Total	7170.	6278.	5640.	4382.	5027.

AGE	2002	2003	2004	2005	2006
1	674.	74.	194.	93.	217.
2	343.	294.	520.	627.	211.
3	1537.	384.	299.	555.	462.
4	593.	474.	91.	97.	183.
5	349.	321.	80.	13.	14.
6	322.	653.	160.	101.	58.
Total	3817.	2200.	1345.	1485.	1146.

Catch Biomass

AGE	2007
1	133.
2	396.
3	238.
4	114.
5	24.
6	23.
Total	927.

Catch Numbers

AGE	1982	1983	1984	1985	1986
1	9914.2	8004.0	10444.2	14153.2	18610.7
2	12967.0	12869.1	14736.9	11262.3	15778.4
3	5473.0	5822.7	6521.1	3246.1	4942.4
4	2778.2	2780.0	3045.3	1171.0	1823.7
5	721.6	568.2	484.5	212.9	264.1
6	639.5	424.1	254.5	55.1	52.1
Total	32493.5	30468.1	35486.5	30100.6	41471.4

AGE	1987	1988	1989	1990	1991
1	16256.3	8161.9	3705.0	9510.1	9795.9
2	14343.1	16140.8	5304.9	4890.1	5825.6
3	4347.1	10545.3	4333.5	2093.6	2750.0
4	1485.2	6092.0	2922.3	1204.8	1373.6
5	145.4	1050.5	626.2	591.4	463.4
6	11.0	70.7	84.6	89.1	57.3
Total	36588.1	42061.2	16976.5	18379.1	20265.8

AGE	1992	1993	1994	1995	1996
1	5179.5	4974.8	3761.9	4336.3	2498.8
2	6046.0	6357.0	4347.4	3727.7	2689.5
3	2211.0	2179.8	3561.0	3566.7	5033.3
4	1255.0	1138.6	1563.5	1637.8	3174.2
5	527.8	401.1	204.1	198.1	1379.3
6	65.0	48.2	39.8	54.3	100.1
Total	15284.3	15099.5	13477.7	13520.9	14875.2

AGE	1997	1998	1999	2000	2001
1	1716.4	1270.6	1412.6	1377.0	2420.7
2	2394.2	2138.3	1300.4	1727.1	2953.1
3	2913.2	3983.1	2256.6	1985.7	1474.1
4	5522.0	2019.2	3326.0	1663.7	1219.9
5	1523.1	2928.8	725.7	1528.2	658.7
6	410.2	909.5	1145.0	403.0	485.9
Total	14479.1	13249.5	10166.3	8684.7	9212.4

AGE	2002	2003	2004	2005	2006
1	2591.7	335.6	852.3	334.3	747.3
2	1070.5	949.9	1511.9	1771.5	637.3
3	2695.7	959.7	667.8	1255.2	959.2
4	823.9	718.4	115.8	191.5	252.9
5	388.2	209.5	49.7	10.2	15.5
6	231.5	254.2	38.4	27.1	11.9
Total	7801.5	3427.3	3235.9	3589.8	2624.1

Catch Numbers

AGE	2007
1	616.6
2	1148.0
3	507.6
4	135.2
5	25.2
6	5.8
Total	2438.4

Surplus Production

Average Adjustment Factor (Delta) = 1.0000

Year	Biomass	Delta Biomass	Catch Biomass	Surplus Production
1982	20171.083	-1721.376	13115.726	11394.350
1983	18449.707	-1456.223	13803.341	12347.118
1984	16993.483	1471.296	14755.539	16226.835
1985	18464.779	14965.989	14621.399	29587.388
1986	33430.768	-6592.756	23404.472	16811.716
1987	26838.012	-3646.337	10745.440	7099.103
1988	23191.675	-9734.783	19869.631	10134.849
1989	13456.893	-3894.793	8975.659	5080.867
1990	9562.100	1503.181	5925.206	7428.387
1991	11065.281	-1961.751	7760.802	5799.051
1992	9103.530	-1552.084	6135.862	4583.777
1993	7551.446	6474.078	3680.393	10154.471
1994	14025.524	330.412	4442.975	4773.387
1995	14355.936	2704.246	3393.641	6097.887
1996	17060.182	4431.459	5330.342	9761.801
1997	21491.641	-6734.272	7169.542	435.270
1998	14757.369	-2555.289	6278.386	3723.097
1999	12202.080	-3856.852	5640.437	1783.585
2000	8345.228	-1015.922	4382.023	3366.100
2001	7329.306	-1679.011	5027.275	3348.264
2002	5650.295	-2334.027	3817.324	1483.297
2003	3316.268	-680.319	2200.154	1519.835
2004	2635.949	-213.040	1344.697	1131.657
2005	2422.910	777.898	1485.377	2263.275
2006	3200.807	3686.450	1146.048	4832.498
2007	6887.257	4284.426	926.806	5211.232
2008	11171.683			

Summary of Survey Indices Used in the Estimate

INDEX	Survey Tag	Age	Time	Type	Catchability	Std. Error	CV
1	DEDFW	1	MEAN	NUMBER	0.2119E-02	0.6910E-03	0.3260E+00
2	DEDFW	2	MEAN	NUMBER	0.2506E-02	0.7186E-03	0.2868E+00
3	DEDFW	3	MEAN	NUMBER	0.1420E-02	0.5420E-03	0.3817E+00
4	DEDFW	4	MEAN	NUMBER	0.1325E-02	0.4215E-03	0.3181E+00
5	DEDFW	5	MEAN	NUMBER	0.8923E-03	0.2794E-03	0.3131E+00
6	DEDFW	6	MEAN	NUMBER	0.7923E-03	0.1933E-03	0.2440E+00
7	NJDEP	1	JAN-1	NUMBER	0.3594E-05	0.8427E-06	0.2345E+00
8	NJDEP	2	JAN-1	NUMBER	0.7893E-05	0.1806E-05	0.2288E+00
9	NJDEP	3	JAN-1	NUMBER	0.5782E-05	0.1644E-05	0.2843E+00
10	NJDEP	4	JAN-1	NUMBER	0.3334E-05	0.1478E-05	0.4431E+00
11	NJDEP	5	JAN-1	NUMBER	0.1855E-05	0.8210E-06	0.4426E+00
12	NJDEP	6	JAN-1	NUMBER	0.1499E-05	0.9329E-06	0.6222E+00
13	NCGill	1	MEAN	NUMBER	0.3151E-04	0.5629E-05	0.1787E+00
14	NCGill	2	MEAN	NUMBER	0.1579E-03	0.3115E-04	0.1972E+00
15	NCGill	3	MEAN	NUMBER	0.4553E-03	0.1379E-03	0.3029E+00
16	NCGill	4	MEAN	NUMBER	0.8500E-03	0.3844E-03	0.4523E+00
17	NCGill	5	MEAN	NUMBER	0.1123E-02	0.6277E-03	0.5591E+00
18	NCGill	6	MEAN	NUMBER	0.7014E-04	0.8013E-05	0.1142E+00
34	MRFSS	3	JAN-1	NUMBER	0.5686E-05	0.1133E-05	0.1993E+00
35	MRFSS	4	JAN-1	NUMBER	0.1976E-04	0.3145E-05	0.1592E+00
36	MRFSS	5	JAN-1	NUMBER	0.5733E-04	0.1052E-04	0.1835E+00
37	MRFSS	6	JAN-1	NUMBER	0.6411E-04	0.1218E-04	0.1900E+00
38	rec	2 - 6	MEAN	NUMBER	0.1832E-04	0.2135E-05	0.1165E+00

Survey Index: 1 Tag: DEDFW AGE = 1
Time = MEAN Type = NUMBER
Catchability = 0.211918E-02 % Variance Contribution = 9.0380
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.471000E+01	0.738853E+02	-0.275283E+01
1983	0.539000E+01	0.684657E+02	-0.254179E+01
1984	0.299000E+01	0.716663E+02	-0.317675E+01
1985	N/A	0.103885E+03	N/A
1986	N/A	0.113012E+03	N/A
1987	N/A	0.844810E+02	N/A
1988	N/A	0.405200E+02	N/A
1989	N/A	0.351415E+02	N/A
1990	N/A	0.395799E+02	N/A
1991	0.271588E+02	0.446618E+02	-0.497418E+00
1992	0.211512E+02	0.614345E+02	-0.106627E+01
1993	0.502801E+02	0.637032E+02	-0.236626E+00
1994	0.113422E+03	0.885392E+02	0.247672E+00
1995	0.752633E+02	0.393266E+02	0.649090E+00
1996	0.440437E+02	0.531771E+02	-0.188446E+00
1997	0.334142E+02	0.285450E+02	0.157499E+00
1998	0.233566E+02	0.200478E+02	0.152759E+00
1999	0.420661E+02	0.161125E+02	0.959645E+00
2000	0.978468E+02	0.220036E+02	0.149220E+01
2001	0.131139E+02	0.907614E+01	0.368024E+00
2002	0.893537E+02	0.829898E+01	0.237647E+01
2003	0.501607E+02	0.102129E+02	0.159158E+01
2004	0.263628E+02	0.105631E+02	0.914585E+00
2005	0.120686E+02	0.805700E+01	0.404066E+00
2006	0.583793E+02	0.154717E+02	0.132795E+01
2007	0.234698E+02	0.281381E+02	-0.181410E+00
2008	N/A	0.000000E+00	N/A

Survey Index: 2 Tag: DEDFW AGE = 2
Time = MEAN Type = NUMBER
Catchability = 0.250557E-02 % Variance Contribution = 6.9939
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.733000E+01	0.467906E+02	-0.185371E+01
1983	0.800000E+01	0.412148E+02	-0.163935E+01
1984	0.532000E+01	0.348077E+02	-0.187836E+01
1985	N/A	0.410478E+02	N/A
1986	N/A	0.612378E+02	N/A
1987	N/A	0.677316E+02	N/A
1988	N/A	0.402116E+02	N/A
1989	N/A	0.227189E+02	N/A
1990	N/A	0.223886E+02	N/A
1991	0.363820E+01	0.198952E+02	-0.169899E+01
1992	0.260810E+01	0.240493E+02	-0.222148E+01
1993	0.254220E+02	0.434173E+02	-0.535244E+00
1994	0.685360E+02	0.483542E+02	0.348806E+00
1995	0.534866E+02	0.732032E+02	-0.313808E+00
1996	0.483017E+02	0.286455E+02	0.522469E+00
1997	0.250014E+02	0.435367E+02	-0.554673E+00
1998	0.246266E+02	0.219172E+02	0.116556E+00

1999	0.201604E+02	0.155890E+02	0.257153E+00
2000	0.503831E+02	0.112544E+02	0.149890E+01
2001	0.426313E+02	0.150866E+02	0.103878E+01
2002	0.233228E+02	0.475010E+01	0.159127E+01
2003	0.139835E+02	0.407628E+01	0.123269E+01
2004	0.224048E+02	0.710606E+01	0.114833E+01
2005	0.145354E+02	0.654358E+01	0.798103E+00
2006	0.371285E+02	0.628982E+01	0.177545E+01
2007	0.174276E+02	0.120726E+02	0.367117E+00
2008	N/A	0.000000E+00	N/A

Survey Index: 3 Tag: DEDFW AGE = 3
Time = MEAN Type = NUMBER
Catchability = 0.141990E-02 % Variance Contribution = 12.3867
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.302000E+01	0.959993E+01	-0.115650E+01
1983	0.334000E+01	0.963323E+01	-0.105925E+01
1984	0.242000E+01	0.644333E+01	-0.979278E+00
1985	N/A	0.598131E+01	N/A
1986	N/A	0.865021E+01	N/A
1987	N/A	0.158488E+02	N/A
1988	N/A	0.141154E+02	N/A
1989	N/A	0.673682E+01	N/A
1990	N/A	0.575404E+01	N/A
1991	0.638300E+00	0.526027E+01	-0.210913E+01
1992	0.318000E-01	0.418784E+01	-0.488047E+01
1993	0.390020E+01	0.592142E+01	-0.417548E+00
1994	0.236497E+02	0.131776E+02	0.584832E+00
1995	0.157403E+02	0.163978E+02	-0.409246E-01
1996	0.111107E+03	0.266459E+02	0.142786E+01
1997	0.135633E+02	0.903976E+01	0.405735E+00
1998	0.203749E+02	0.150050E+02	0.305919E+00
1999	0.170310E+02	0.685167E+01	0.910544E+00
2000	0.236377E+02	0.468592E+01	0.161828E+01
2001	0.187718E+02	0.293052E+01	0.185718E+01
2002	0.279672E+02	0.292249E+01	0.225860E+01
2003	0.121830E+01	0.714240E+00	0.533992E+00
2004	0.108900E+00	0.772872E+00	-0.195968E+01
2005	0.238720E+01	0.132749E+01	0.586831E+00
2006	0.100217E+02	0.123151E+01	0.209652E+01
2007	0.209630E+01	0.206199E+01	0.165018E-01
2008	N/A	0.000000E+00	N/A

Survey Index: 4 Tag: DEDFW AGE = 4
Time = MEAN Type = NUMBER
Catchability = 0.132513E-02 % Variance Contribution = 6.1574
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.145000E+01	0.262682E+01	-0.594210E+00
1983	0.157000E+01	0.228320E+01	-0.374501E+00
1984	0.115000E+01	0.182466E+01	-0.461634E+00
1985	N/A	0.114891E+01	N/A
1986	N/A	0.139797E+01	N/A
1987	N/A	0.297395E+01	N/A
1988	N/A	0.449595E+01	N/A
1989	N/A	0.348012E+01	N/A
1990	N/A	0.208826E+01	N/A
1991	N/A	0.217973E+01	N/A
1992	0.318000E-01	0.162151E+01	-0.393165E+01
1993	0.516900E+00	0.115204E+01	-0.801444E+00
1994	0.965300E+00	0.210378E+01	-0.779051E+00
1995	0.542510E+01	0.670420E+01	-0.211698E+00
1996	0.238182E+02	0.793064E+01	0.109972E+01
1997	0.345991E+02	0.129877E+02	0.979823E+00
1998	0.115955E+02	0.371187E+01	0.113908E+01
1999	0.674020E+01	0.658549E+01	0.232210E-01
2000	0.574460E+01	0.268239E+01	0.761552E+00
2001	0.552630E+01	0.155948E+01	0.126517E+01
2002	0.383670E+01	0.812500E+00	0.155225E+01
2003	0.423700E+00	0.398061E+00	0.624205E-01
2004	N/A	0.708650E-01	N/A
2005	N/A	0.122394E+00	N/A
2006	0.784800E+00	0.254361E+00	0.112667E+01
2007	0.164400E+00	0.386844E+00	-0.855720E+00
2008	N/A	0.000000E+00	N/A

Survey Index: 5 Tag: DEDFW AGE = 5
Time = MEAN Type = NUMBER
Catchability = 0.892300E-03 % Variance Contribution = 2.4129
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.240000E+00	0.459426E+00	-0.649339E+00
1983	0.200000E+00	0.314233E+00	-0.451818E+00
1984	0.130000E+00	0.195478E+00	-0.407916E+00
1985	N/A	0.140655E+00	N/A
1986	N/A	0.136321E+00	N/A
1987	N/A	0.196049E+00	N/A
1988	N/A	0.522047E+00	N/A
1989	N/A	0.502152E+00	N/A
1990	N/A	0.690244E+00	N/A
1991	N/A	0.495165E+00	N/A
1992	N/A	0.459195E+00	N/A
1993	N/A	0.273277E+00	N/A
1994	N/A	0.184926E+00	N/A
1995	0.764000E-01	0.546038E+00	-0.196671E+01
1996	0.638700E+01	0.232052E+01	0.101247E+01
1997	0.296230E+01	0.241222E+01	0.205417E+00
1998	0.207062E+02	0.362540E+01	0.174247E+01

1999	0.258780E+01	0.967554E+00	0.983792E+00
2000	0.659200E+00	0.165913E+01	-0.923019E+00
2001	0.570200E+00	0.567015E+00	0.560103E-02
2002	0.403900E+00	0.257785E+00	0.449043E+00
2003	N/A	0.781663E-01	N/A
2004	N/A	0.204801E-01	N/A
2005	N/A	0.438978E-02	N/A
2006	N/A	0.104975E-01	N/A
2007	N/A	0.485526E-01	N/A
2008	N/A	0.000000E+00	N/A

Survey Index: 6 Tag: DEDFW AGE = 6
Time = MEAN Type = NUMBER
Catchability = 0.792329E-03 % Variance Contribution = 0.9592
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.330000E+00	0.361538E+00	-0.912754E-01
1983	0.180000E+00	0.208264E+00	-0.145849E+00
1984	0.700000E-01	0.911775E-01	-0.264313E+00
1985	N/A	0.323241E-01	N/A
1986	N/A	0.238797E-01	N/A
1987	N/A	0.131701E-01	N/A
1988	N/A	0.311981E-01	N/A
1989	N/A	0.602403E-01	N/A
1990	N/A	0.923408E-01	N/A
1991	N/A	0.543680E-01	N/A
1992	N/A	0.502153E-01	N/A
1993	N/A	0.291603E-01	N/A
1994	N/A	0.320208E-01	N/A
1995	N/A	0.132902E+00	N/A
1996	0.133100E+00	0.149539E+00	-0.116458E+00
1997	0.474000E+00	0.576872E+00	-0.196413E+00
1998	0.132520E+01	0.999686E+00	0.281878E+00
1999	0.403310E+01	0.135556E+01	0.109032E+01
2000	0.941800E+00	0.388507E+00	0.885482E+00
2001	0.877000E-01	0.371406E+00	-0.144337E+01
2002	N/A	0.136505E+00	N/A
2003	N/A	0.842182E-01	N/A
2004	N/A	0.140508E-01	N/A
2005	N/A	0.103564E-01	N/A
2006	N/A	0.715643E-02	N/A
2007	N/A	0.824995E-02	N/A
2008	N/A	0.000000E+00	N/A

Survey Index: 7 Tag: NJDEP AGE = 1
Time = JAN-1 Type = NUMBER
Catchability = 0.359433E-05 % Variance Contribution = 4.2063
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.161766E+00	N/A
1983	N/A	0.147412E+00	N/A
1984	N/A	0.158664E+00	N/A
1985	N/A	0.227900E+00	N/A
1986	N/A	0.254782E+00	N/A
1987	N/A	0.195543E+00	N/A
1988	N/A	0.945887E-01	N/A
1989	N/A	0.748213E-01	N/A
1990	0.356000E-01	0.958075E-01	-0.989996E+00
1991	0.745000E-01	0.106023E+00	-0.352853E+00
1992	0.892000E-01	0.128123E+00	-0.362106E+00
1993	0.405000E-01	0.132043E+00	-0.118182E+01
1994	0.255600E+00	0.177147E+00	0.366634E+00
1995	0.398000E-01	0.841277E-01	-0.748469E+00
1996	0.543000E-01	0.106875E+00	-0.677135E+00
1997	0.132300E+00	0.581249E-01	0.822478E+00
1998	0.870000E-02	0.409543E-01	-0.154913E+01
1999	0.222000E-01	0.337142E-01	-0.417827E+00
2000	0.397000E-01	0.449140E-01	-0.123398E+00
2001	0.470000E-02	0.225119E-01	-0.156648E+01
2002	0.139800E+00	0.214550E-01	0.187426E+01
2003	0.416000E-01	0.202377E-01	0.720554E+00
2004	0.591000E-01	0.219511E-01	0.990414E+00
2005	0.977000E-01	0.161042E-01	0.180282E+01
2006	0.746000E-01	0.311356E-01	0.873789E+00
2007	0.926000E-01	0.551475E-01	0.518277E+00
2008	0.591000E-01	0.591000E-01	-0.412020E-10

Survey Index: 8 Tag: NJDEP AGE = 2
Time = JAN-1 Type = NUMBER
Catchability = 0.789323E-05 % Variance Contribution = 4.0068
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.227799E+00	N/A
1983	N/A	0.208188E+00	N/A
1984	N/A	0.196790E+00	N/A
1985	N/A	0.199258E+00	N/A
1986	N/A	0.292026E+00	N/A
1987	N/A	0.307377E+00	N/A
1988	N/A	0.222438E+00	N/A
1989	N/A	0.105565E+00	N/A
1990	0.260000E-01	0.102342E+00	-0.137023E+01
1991	0.523000E-01	0.984749E-01	-0.632805E+00
1992	0.320600E+00	0.113919E+00	0.103470E+01
1993	0.110300E+00	0.183272E+00	-0.507765E+00
1994	0.148900E+00	0.191384E+00	-0.251006E+00
1995	0.160700E+00	0.276881E+00	-0.544049E+00
1996	0.434000E-01	0.113899E+00	-0.964856E+00
1997	0.342000E-01	0.165460E+00	-0.157650E+01
1998	0.440000E-01	0.875156E-01	-0.687628E+00

1999	0.396000E-01	0.612397E-01	-0.435966E+00
2000	0.125500E+00	0.478838E-01	0.963528E+00
2001	0.788000E-01	0.672743E-01	0.158135E+00
2002	0.546000E-01	0.218780E-01	0.914550E+00
2003	0.407000E-01	0.189310E-01	0.765425E+00
2004	0.173500E+00	0.322836E-01	0.168162E+01
2005	0.934000E-01	0.316417E-01	0.108241E+01
2006	0.747000E-01	0.252241E-01	0.108568E+01
2007	0.122000E-01	0.480692E-01	-0.137121E+01
2008	0.173500E+00	0.900372E-01	0.655955E+00

Survey Index: 9 Tag: NJDEP AGE = 3
Time = JAN-1 Type = NUMBER
Catchability = 0.578221E-05 % Variance Contribution = 6.1833
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.633936E-01	N/A
1983	N/A	0.649021E-01	N/A
1984	N/A	0.543188E-01	N/A
1985	N/A	0.388655E-01	N/A
1986	N/A	0.571641E-01	N/A
1987	N/A	0.873606E-01	N/A
1988	N/A	0.103158E+00	N/A
1989	N/A	0.464186E-01	N/A
1990	0.570000E-02	0.335504E-01	-0.177258E+01
1991	0.990000E-02	0.337790E-01	-0.122731E+01
1992	0.585000E-01	0.269751E-01	0.774113E+00
1993	0.445000E-01	0.346179E-01	0.251119E+00
1994	0.316000E-01	0.724498E-01	-0.829737E+00
1995	0.702000E-01	0.871642E-01	-0.216445E+00
1996	0.117700E+00	0.139042E+00	-0.166640E+00
1997	0.466000E-01	0.513596E-01	-0.972513E-01
1998	0.679000E-01	0.822465E-01	-0.191685E+00
1999	0.302000E-01	0.391010E-01	-0.258307E+00
2000	0.101100E+00	0.283483E-01	0.127154E+01
2001	0.138100E+00	0.185980E-01	0.200492E+01
2002	0.239000E-01	0.235026E-01	0.167678E-01
2003	0.210000E-02	0.709649E-02	-0.121766E+01
2004	0.726000E-01	0.602374E-02	0.248926E+01
2005	0.990000E-02	0.108076E-01	-0.877138E-01
2006	0.161000E-01	0.916087E-02	0.563878E+00
2007	0.900000E-03	0.111642E-01	-0.251807E+01
2008	0.726000E-01	0.216102E-01	0.121180E+01

Survey Index: 10 Tag: NJDEP AGE = 4
Time = JAN-1 Type = NUMBER
Catchability = 0.333437E-05 % Variance Contribution = 11.9491
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.135059E-01	N/A
1983	N/A	0.126716E-01	N/A
1984	N/A	0.123559E-01	N/A
1985	N/A	0.579695E-02	N/A
1986	N/A	0.807695E-02	N/A
1987	N/A	0.114061E-01	N/A
1988	N/A	0.265780E-01	N/A
1989	N/A	0.160385E-01	N/A
1990	0.240000E-02	0.836319E-02	-0.124837E+01
1991	0.230000E-02	0.898825E-02	-0.136301E+01
1992	0.700000E-03	0.722129E-02	-0.233371E+01
1993	0.181000E-01	0.572456E-02	0.115115E+01
1994	N/A	0.921814E-02	N/A
1995	0.820000E-02	0.221689E-01	-0.994556E+00
1996	0.273000E-01	0.287445E-01	-0.515612E-01
1997	0.133900E+00	0.477540E-01	0.103103E+01
1998	0.350000E-01	0.145963E-01	0.874583E+00
1999	0.530000E-02	0.253380E-01	-0.156460E+01
2000	0.215300E+00	0.110012E-01	0.297403E+01
2001	0.582000E-01	0.697527E-02	0.212151E+01
2002	0.290000E-02	0.408907E-02	-0.343607E+00
2003	0.300000E-03	0.284880E-02	-0.225087E+01
2004	0.129000E-01	0.472948E-03	0.330600E+01
2005	0.800000E-03	0.793219E-03	0.851259E-02
2006	N/A	0.126766E-02	N/A
2007	0.100000E-03	0.136139E-02	-0.261109E+01
2008	0.129000E-01	0.353485E-02	0.129456E+01

Survey Index: 11 Tag: NJDEP AGE = 5
Time = JAN-1 Type = NUMBER
Catchability = 0.185515E-05 % Variance Contribution = 4.8204
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.195174E-02	N/A
1983	N/A	0.144096E-02	N/A
1984	N/A	0.109371E-02	N/A
1985	N/A	0.586385E-03	N/A
1986	N/A	0.650769E-03	N/A
1987	N/A	0.621271E-03	N/A
1988	N/A	0.254990E-02	N/A
1989	N/A	0.191212E-02	N/A
1990	0.300000E-03	0.228403E-02	-0.202992E+01
1991	0.100000E-03	0.168708E-02	-0.282558E+01
1992	N/A	0.168968E-02	N/A
1993	0.140000E-02	0.112199E-02	0.221371E+00
1994	N/A	0.669503E-03	N/A
1995	N/A	0.149187E-02	N/A
1996	0.107000E-01	0.694935E-02	0.431596E+00
1997	0.116000E-01	0.732835E-02	0.459255E+00
1998	0.821000E-01	0.117792E-01	0.194160E+01

1999	0.150000E-02	0.307589E-02	-0.718130E+00
2000	0.112000E-01	0.562222E-02	0.689186E+00
2001	0.137000E-01	0.209550E-02	0.187760E+01
2002	N/A	0.107194E-02	N/A
2003	N/A	0.462215E-03	N/A
2004	0.200000E-03	0.112934E-03	0.571511E+00
2005	N/A	0.235065E-04	N/A
2006	N/A	0.432264E-04	N/A
2007	N/A	0.147097E-03	N/A
2008	0.200000E-03	0.371226E-03	-0.618494E+00

Survey Index: 12 Tag: NJDEP AGE = 6
Time = JAN-1 Type = NUMBER
Catchability = 0.149946E-05 % Variance Contribution = 1.7322
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.139805E-02	N/A
1983	N/A	0.869313E-03	N/A
1984	N/A	0.464359E-03	N/A
1985	N/A	0.122663E-03	N/A
1986	N/A	0.103765E-03	N/A
1987	N/A	0.379897E-04	N/A
1988	N/A	0.138708E-03	N/A
1989	N/A	0.208799E-03	N/A
1990	N/A	0.278135E-03	N/A
1991	N/A	0.168612E-03	N/A
1992	N/A	0.168192E-03	N/A
1993	N/A	0.108978E-03	N/A
1994	N/A	0.105523E-03	N/A
1995	N/A	0.330524E-03	N/A
1996	0.900000E-03	0.407639E-03	0.792013E+00
1997	N/A	0.159525E-02	N/A
1998	0.113000E-01	0.295655E-02	0.134078E+01
1999	0.120000E-02	0.392262E-02	-0.118444E+01
2000	0.200000E-03	0.119837E-02	-0.179040E+01
2001	0.290000E-02	0.124941E-02	0.842041E+00
2002	N/A	0.516680E-03	N/A
2003	N/A	0.453306E-03	N/A
2004	N/A	0.705272E-04	N/A
2005	N/A	0.504794E-04	N/A
2006	N/A	0.268238E-04	N/A
2007	N/A	0.227511E-04	N/A
2008	N/A	0.701670E-05	N/A

Survey Index: 13 Tag: NCGill AGE = 1
Time = MEAN Type = NUMBER
Catchability = 0.315053E-04 % Variance Contribution = 0.3000
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.109843E+01	N/A
1983	N/A	0.101786E+01	N/A
1984	N/A	0.106544E+01	N/A
1985	N/A	0.154444E+01	N/A
1986	N/A	0.168012E+01	N/A
1987	N/A	0.125596E+01	N/A
1988	N/A	0.602400E+00	N/A
1989	N/A	0.522440E+00	N/A
1990	N/A	0.588423E+00	N/A
1991	N/A	0.663976E+00	N/A
1992	N/A	0.913330E+00	N/A
1993	N/A	0.947059E+00	N/A
1994	N/A	0.131629E+01	N/A
1995	N/A	0.584659E+00	N/A
1996	N/A	0.790570E+00	N/A
1997	N/A	0.424371E+00	N/A
1998	N/A	0.298046E+00	N/A
1999	N/A	0.239541E+00	N/A
2000	N/A	0.327121E+00	N/A
2001	0.148000E+00	0.134933E+00	0.924368E-01
2002	0.172900E+00	0.123379E+00	0.337455E+00
2003	0.145400E+00	0.151833E+00	-0.432916E-01
2004	0.226900E+00	0.157039E+00	0.368014E+00
2005	0.171900E+00	0.119781E+00	0.361246E+00
2006	0.197400E+00	0.230014E+00	-0.152906E+00
2007	0.159700E+00	0.418322E+00	-0.962953E+00
2008	N/A	0.000000E+00	N/A

Survey Index: 14 Tag: NCGill AGE = 2
Time = MEAN Type = NUMBER
Catchability = 0.157929E-03 % Variance Contribution = 0.3656
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.294926E+01	N/A
1983	N/A	0.259781E+01	N/A
1984	N/A	0.219397E+01	N/A
1985	N/A	0.258729E+01	N/A
1986	N/A	0.385989E+01	N/A
1987	N/A	0.426920E+01	N/A
1988	N/A	0.253458E+01	N/A
1989	N/A	0.143200E+01	N/A
1990	N/A	0.141118E+01	N/A
1991	N/A	0.125401E+01	N/A
1992	N/A	0.151585E+01	N/A
1993	N/A	0.273664E+01	N/A
1994	N/A	0.304782E+01	N/A
1995	N/A	0.461408E+01	N/A
1996	N/A	0.180556E+01	N/A
1997	N/A	0.274417E+01	N/A
1998	N/A	0.138146E+01	N/A

1999	N/A	0.982593E+00	N/A
2000	N/A	0.709375E+00	N/A
2001	0.148860E+01	0.950925E+00	0.448156E+00
2002	0.267400E+00	0.299404E+00	-0.113048E+00
2003	0.366900E+00	0.256932E+00	0.356276E+00
2004	0.594600E+00	0.447902E+00	0.283313E+00
2005	0.486200E+00	0.412449E+00	0.164508E+00
2006	0.376600E+00	0.396454E+00	-0.513775E-01
2007	0.256400E+00	0.760949E+00	-0.108783E+01
2008	N/A	0.000000E+00	N/A

Survey Index: 15 Tag: NCGill AGE = 3
Time = MEAN Type = NUMBER
Catchability = 0.455255E-03 % Variance Contribution = 0.8624
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.307798E+01	N/A
1983	N/A	0.308866E+01	N/A
1984	N/A	0.206590E+01	N/A
1985	N/A	0.191776E+01	N/A
1986	N/A	0.277348E+01	N/A
1987	N/A	0.508153E+01	N/A
1988	N/A	0.452576E+01	N/A
1989	N/A	0.216000E+01	N/A
1990	N/A	0.184489E+01	N/A
1991	N/A	0.168658E+01	N/A
1992	N/A	0.134273E+01	N/A
1993	N/A	0.189856E+01	N/A
1994	N/A	0.422507E+01	N/A
1995	N/A	0.525756E+01	N/A
1996	N/A	0.854335E+01	N/A
1997	N/A	0.289838E+01	N/A
1998	N/A	0.481099E+01	N/A
1999	N/A	0.219682E+01	N/A
2000	N/A	0.150243E+01	N/A
2001	0.369400E+00	0.939598E+00	-0.933572E+00
2002	0.108730E+01	0.937026E+00	0.148742E+00
2003	0.345000E+00	0.229004E+00	0.409806E+00
2004	0.584000E+00	0.247803E+00	0.857269E+00
2005	0.692500E+00	0.425627E+00	0.486744E+00
2006	0.553400E+00	0.394852E+00	0.337570E+00
2007	0.179000E+00	0.661127E+00	-0.130656E+01
2008	N/A	0.000000E+00	N/A

Survey Index: 16 Tag: NCGill AGE = 4
Time = MEAN Type = NUMBER
Catchability = 0.849967E-03 % Variance Contribution = 1.9222
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.168490E+01	N/A
1983	N/A	0.146449E+01	N/A
1984	N/A	0.117038E+01	N/A
1985	N/A	0.736933E+00	N/A
1986	N/A	0.896685E+00	N/A
1987	N/A	0.190755E+01	N/A
1988	N/A	0.288380E+01	N/A
1989	N/A	0.223222E+01	N/A
1990	N/A	0.133945E+01	N/A
1991	N/A	0.139812E+01	N/A
1992	N/A	0.104007E+01	N/A
1993	N/A	0.738946E+00	N/A
1994	N/A	0.134941E+01	N/A
1995	N/A	0.430022E+01	N/A
1996	N/A	0.508688E+01	N/A
1997	N/A	0.833060E+01	N/A
1998	N/A	0.238087E+01	N/A
1999	N/A	0.422407E+01	N/A
2000	N/A	0.172054E+01	N/A
2001	0.145900E+00	0.100028E+01	-0.192512E+01
2002	0.242300E+00	0.521155E+00	-0.765871E+00
2003	0.680200E+00	0.255325E+00	0.979851E+00
2004	0.188500E+00	0.454543E-01	0.142239E+01
2005	0.174700E+00	0.785060E-01	0.799895E+00
2006	0.213000E+00	0.163153E+00	0.266605E+00
2007	0.114000E+00	0.248130E+00	-0.777755E+00
2008	N/A	0.000000E+00	N/A

Survey Index: 17 Tag: NCGill AGE = 5
Time = MEAN Type = NUMBER
Catchability = 0.112275E-02 % Variance Contribution = 2.9374
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.578078E+00	N/A
1983	N/A	0.395387E+00	N/A
1984	N/A	0.245963E+00	N/A
1985	N/A	0.176981E+00	N/A
1986	N/A	0.171528E+00	N/A
1987	N/A	0.246681E+00	N/A
1988	N/A	0.656871E+00	N/A
1989	N/A	0.631838E+00	N/A
1990	N/A	0.868507E+00	N/A
1991	N/A	0.623047E+00	N/A
1992	N/A	0.577788E+00	N/A
1993	N/A	0.343854E+00	N/A
1994	N/A	0.232685E+00	N/A
1995	N/A	0.687058E+00	N/A
1996	N/A	0.291982E+01	N/A
1997	N/A	0.303520E+01	N/A
1998	N/A	0.456170E+01	N/A

1999	N/A	0.121744E+01	N/A
2000	N/A	0.208761E+01	N/A
2001	0.112700E+00	0.713453E+00	-0.184539E+01
2002	0.569000E-01	0.324360E+00	-0.174056E+01
2003	0.583000E-01	0.983536E-01	-0.522967E+00
2004	0.722000E-01	0.257693E-01	0.103026E+01
2005	0.343000E-01	0.552349E-02	0.182614E+01
2006	0.543000E-01	0.132086E-01	0.141366E+01
2007	0.520000E-01	0.610918E-01	-0.161134E+00
2008	N/A	0.000000E+00	N/A

Survey Index: 18 Tag: NCGill AGE = 6
Time = MEAN Type = NUMBER
Catchability = 0.701428E-04 % Variance Contribution = 0.0350
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	N/A	0.320060E-01	N/A
1983	N/A	0.184371E-01	N/A
1984	N/A	0.807171E-02	N/A
1985	N/A	0.286157E-02	N/A
1986	N/A	0.211400E-02	N/A
1987	N/A	0.116591E-02	N/A
1988	N/A	0.276188E-02	N/A
1989	N/A	0.533291E-02	N/A
1990	N/A	0.817469E-02	N/A
1991	N/A	0.481306E-02	N/A
1992	N/A	0.444543E-02	N/A
1993	N/A	0.258149E-02	N/A
1994	N/A	0.283472E-02	N/A
1995	N/A	0.117655E-01	N/A
1996	N/A	0.132383E-01	N/A
1997	N/A	0.510690E-01	N/A
1998	N/A	0.884996E-01	N/A
1999	N/A	0.120004E+00	N/A
2000	N/A	0.343935E-01	N/A
2001	N/A	0.328796E-01	N/A
2002	0.890000E-02	0.120844E-01	-0.305863E+00
2003	N/A	0.745561E-02	N/A
2004	N/A	0.124388E-02	N/A
2005	0.900000E-03	0.916821E-03	-0.185177E-01
2006	0.800000E-03	0.633540E-03	0.233288E+00
2007	0.800000E-03	0.730347E-03	0.910925E-01
2008	N/A	0.000000E+00	N/A

Survey Index: 34 Tag: MRFSS AGE = 3
Time = JAN-1 Type = NUMBER
Catchability = 0.568621E-05 % Variance Contribution = 6.2394
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual

1982	0.186500E+00	0.623411E-01	0.109581E+01
1983	0.840000E-02	0.638246E-01	-0.202791E+01
1984	0.177100E+00	0.534169E-01	0.119859E+01
1985	0.103300E+00	0.382202E-01	0.994273E+00
1986	0.933000E-01	0.562150E-01	0.506636E+00
1987	0.388500E+00	0.859102E-01	0.150899E+01
1988	0.227200E+00	0.101446E+00	0.806307E+00
1989	0.149800E+00	0.456479E-01	0.118834E+01
1990	0.527000E-01	0.329934E-01	0.468309E+00
1991	0.574000E-01	0.332182E-01	0.546947E+00
1992	0.530000E-01	0.265272E-01	0.692120E+00
1993	0.370000E-01	0.340431E-01	0.832897E-01
1994	0.300000E-01	0.712470E-01	-0.864955E+00
1995	0.355000E-01	0.857170E-01	-0.881519E+00
1996	0.271000E-01	0.136734E+00	-0.161850E+01
1997	0.137000E-01	0.505069E-01	-0.130471E+01
1998	0.151000E-01	0.808810E-01	-0.167828E+01
1999	0.162000E-01	0.384518E-01	-0.864395E+00
2000	0.105000E-01	0.278777E-01	-0.976451E+00
2001	0.109000E-01	0.182892E-01	-0.517548E+00
2002	0.368000E-01	0.231124E-01	0.465129E+00
2003	0.510000E-02	0.697867E-02	-0.313617E+00
2004	0.940000E-02	0.592373E-02	0.461743E+00
2005	0.135000E-01	0.106282E-01	0.239183E+00
2006	0.489000E-01	0.900877E-02	0.169158E+01
2007	0.840000E-02	0.109788E-01	-0.267736E+00
2008	0.113000E-01	0.212514E-01	-0.631620E+00

Survey Index: 35 Tag: MRFSS AGE = 4
Time = JAN-1 Type = NUMBER
Catchability = 0.197591E-04 % Variance Contribution = 3.9789
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.217600E+00	0.800344E-01	0.100020E+01
1983	0.588000E-01	0.750905E-01	-0.244552E+00
1984	0.163100E+00	0.732198E-01	0.800898E+00
1985	0.919000E-01	0.343520E-01	0.984040E+00
1986	0.758000E-01	0.478630E-01	0.459755E+00
1987	0.132900E+00	0.675912E-01	0.676120E+00
1988	0.126200E+00	0.157498E+00	-0.221545E+00
1989	0.191500E+00	0.950421E-01	0.700568E+00
1990	0.527000E-01	0.495592E-01	0.614469E-01
1991	0.309000E-01	0.532633E-01	-0.544491E+00
1992	0.485000E-01	0.427925E-01	0.125202E+00
1993	0.328000E-01	0.339231E-01	-0.336667E-01
1994	0.258000E-01	0.546256E-01	-0.750127E+00
1995	0.659000E-01	0.131370E+00	-0.689881E+00
1996	0.588000E-01	0.170337E+00	-0.106364E+01
1997	0.504000E-01	0.282985E+00	-0.172540E+01
1998	0.605000E-01	0.864957E-01	-0.357451E+00
1999	0.647000E-01	0.150150E+00	-0.841873E+00
2000	0.264000E-01	0.651916E-01	-0.903967E+00
2001	0.274000E-01	0.413346E-01	-0.411157E+00
2002	0.263000E-01	0.242313E-01	0.819223E-01
2003	0.462000E-01	0.168816E-01	0.100675E+01
2004	0.470000E-02	0.280263E-02	0.517003E+00
2005	0.580000E-02	0.470052E-02	0.210185E+00
2006	0.384000E-01	0.751199E-02	0.163156E+01

2007	0.196000E-01	0.806740E-02	0.887698E+00
2008	0.540000E-02	0.209471E-01	-0.135560E+01

Survey Index: 36 Tag: MRFSS AGE = 5
Time = JAN-1 Type = NUMBER
Catchability = 0.573293E-04 % Variance Contribution = 5.2909
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.213100E+00	0.603143E-01	0.126219E+01
1983	0.671000E-01	0.445299E-01	0.410023E+00
1984	0.116500E+00	0.337989E-01	0.123746E+01
1985	0.632000E-01	0.181210E-01	0.124923E+01
1986	0.525000E-01	0.201106E-01	0.959565E+00
1987	0.664000E-01	0.191991E-01	0.124084E+01
1988	0.707000E-01	0.787992E-01	-0.108457E+00
1989	0.129000E+00	0.590901E-01	0.780749E+00
1990	0.341000E-01	0.705832E-01	-0.727495E+00
1991	0.177000E-01	0.521355E-01	-0.108028E+01
1992	0.265000E-01	0.522160E-01	-0.678244E+00
1993	0.287000E-01	0.346726E-01	-0.189054E+00
1994	0.172000E-01	0.206896E-01	-0.184720E+00
1995	0.304000E-01	0.461032E-01	-0.416440E+00
1996	0.407000E-01	0.214755E+00	-0.166327E+01
1997	0.105400E+00	0.226467E+00	-0.764837E+00
1998	0.958000E-01	0.364012E+00	-0.133492E+01
1999	0.102400E+00	0.950540E-01	0.744420E-01
2000	0.632000E-01	0.173743E+00	-0.101127E+01
2001	0.328000E-01	0.647571E-01	-0.680216E+00
2002	0.158000E-01	0.331260E-01	-0.740308E+00
2003	0.205000E-01	0.142838E-01	0.361301E+00
2004	0.470000E-02	0.348999E-02	0.297662E+00
2005	0.210000E-02	0.726420E-03	0.106156E+01
2006	0.580000E-02	0.133582E-02	0.146831E+01
2007	0.880000E-02	0.454571E-02	0.660568E+00
2008	0.260000E-02	0.114720E-01	-0.148439E+01

Survey Index: 37 Tag: MRFSS AGE = 6
Time = JAN-1 Type = NUMBER
Catchability = 0.641095E-04 % Variance Contribution = 5.2472
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.106600E+00	0.597736E-01	0.578519E+00
1983	0.630000E-01	0.371676E-01	0.527698E+00
1984	0.326000E-01	0.198537E-01	0.495921E+00
1985	0.172000E-01	0.524448E-02	0.118773E+01
1986	0.175000E-01	0.443650E-02	0.137234E+01
1987	0.102000E-01	0.162425E-02	0.183734E+01
1988	0.101000E-01	0.593049E-02	0.532429E+00
1989	0.291000E-01	0.892724E-02	0.118163E+01
1990	0.930000E-02	0.118917E-01	-0.245824E+00
1991	0.440000E-02	0.720905E-02	-0.493732E+00
1992	0.880000E-02	0.719106E-02	0.201914E+00
1993	0.820000E-02	0.465936E-02	0.565255E+00
1994	0.430000E-02	0.451166E-02	-0.480511E-01

1995	N/A	0.141316E-01	N/A
1996	0.450000E-02	0.174286E-01	-0.135404E+01
1997	0.321000E-01	0.682052E-01	-0.753665E+00
1998	0.302000E-01	0.126408E+00	-0.143167E+01
1999	0.323000E-01	0.167712E+00	-0.164718E+01
2000	0.474000E-01	0.512363E-01	-0.778251E-01
2001	0.711000E-01	0.534186E-01	0.285929E+00
2002	0.158000E-01	0.220907E-01	-0.335147E+00
2003	0.154000E-01	0.193812E-01	-0.229934E+00
2004	0.470000E-02	0.301540E-02	0.443830E+00
2005	0.200000E-02	0.215825E-02	-0.761511E-01
2006	0.100000E-03	0.114686E-02	-0.243961E+01
2007	0.900000E-03	0.972727E-03	-0.777085E-01
2008	0.300000E-03	0.300000E-03	-0.601915E-07

Survey Index: 38 Tag: rec AGE = 2 - 6
Time = MEAN Type = NUMBER
Catchability = 0.183228E-04 % Variance Contribution = 1.9749
Residual = LN(Observed) - LN(Predicted)

Year	Observed	Predicted	Residual
1982	0.202100E+00	0.520167E+00	-0.945387E+00
1983	0.510900E+00	0.468545E+00	0.865428E-01
1984	0.300100E+00	0.369041E+00	-0.206792E+00
1985	0.277700E+00	0.396881E+00	-0.357096E+00
1986	0.768100E+00	0.582127E+00	0.277232E+00
1987	0.517800E+00	0.745277E+00	-0.364167E+00
1988	0.573800E+00	0.549817E+00	0.426957E-01
1989	0.167300E+00	0.312897E+00	-0.626086E+00
1990	0.155700E+00	0.283159E+00	-0.598077E+00
1991	0.177800E+00	0.254934E+00	-0.360346E+00
1992	0.192800E+00	0.262921E+00	-0.310198E+00
1993	0.165100E+00	0.416130E+00	-0.924446E+00
1994	0.384900E+00	0.557280E+00	-0.370084E+00
1995	0.507900E+00	0.853909E+00	-0.519540E+00
1996	0.605400E+00	0.714092E+00	-0.165123E+00
1997	0.517700E+00	0.677484E+00	-0.268991E+00
1998	0.529700E+00	0.502793E+00	0.521319E-01
1999	0.410100E+00	0.344690E+00	0.173756E+00
2000	0.458500E+00	0.222913E+00	0.721180E+00
2001	0.281800E+00	0.189937E+00	0.394505E+00
2002	0.237900E+00	0.921340E-01	0.948606E+00
2003	0.116200E+00	0.480825E-01	0.882394E+00
2004	0.115400E+00	0.636639E-01	0.594786E+00
2005	0.224300E+00	0.670042E-01	0.120823E+01
2006	0.170400E+00	0.657861E-01	0.951740E+00
2007	0.884000E-01	0.121430E+00	-0.317465E+00
2008	N/A	0.000000E+00	N/A

Bootstrap Summary Report

Number of Bootstrap Repetitions Requested = 500
Number of Bootstrap Repetitions Completed = 500
Bootstrap Output Variable: Stock Estimates (2008)

	NLLS Estimate	Bootstrap Mean	Bootstrap Std Error	C.V. For NLLS Soln.
N 1	16443.	25024.	22636.	0.9046
N 2	11407.	12666.	7057.	0.5572
N 3	3737.	4081.	1660.	0.4068
N 4	1060.	1164.	585.	0.5025
N 5	200.	212.	112.	0.5300
N 6	5.	7.	6.	0.9056

	Bias Estimate	Bias Std. Error	Per Cent Bias	NLLS Estimate Corrected For Bias	C.V. For Corrected Estimate
N 1	8581.	1083.	52.1885	7861.	2.8794
N 2	1259.	321.	11.0380	10148.	0.6954
N 3	344.	76.	9.1983	3394.	0.4892
N 4	104.	27.	9.7640	957.	0.6113
N 5	12.	5.	6.0242	188.	0.5979
N 6	2.	0.	40.6860	3.	2.1480

	LOWER 80. % CI	UPPER 80. % CI
N 1	3841.	59080.
N 2	5384.	22460.
N 3	2138.	6142.
N 4	499.	1998.
N 5	89.	356.
N 6	1.	14.

Bootstrap Output Variable: Catchability Estimates

	NLLS Estimate	Bootstrap Mean	Bootstrap Std Error	C.V. For NLLS Soln.
Q 1	0.211918E-02	0.227085E-02	0.680531E-03	0.2997
Q 2	0.250557E-02	0.258029E-02	0.657196E-03	0.2547
Q 3	0.141990E-02	0.150457E-02	0.548043E-03	0.3643
Q 4	0.132513E-02	0.141311E-02	0.406719E-03	0.2878
Q 5	0.892300E-03	0.970657E-03	0.310111E-03	0.3195
Q 6	0.792329E-03	0.820719E-03	0.196189E-03	0.2390
Q 7	0.359433E-05	0.377379E-05	0.918097E-06	0.2433
Q 8	0.789323E-05	0.816576E-05	0.175158E-05	0.2145
Q 9	0.578221E-05	0.596933E-05	0.167484E-05	0.2806
Q 10	0.333437E-05	0.372262E-05	0.155389E-05	0.4174
Q 11	0.185515E-05	0.203206E-05	0.895587E-06	0.4407
Q 12	0.149946E-05	0.177698E-05	0.114970E-05	0.6470
Q 13	0.315053E-04	0.320050E-04	0.597824E-05	0.1868
Q 14	0.157929E-03	0.161713E-03	0.331682E-04	0.2051
Q 15	0.455255E-03	0.475327E-03	0.145131E-03	0.3053
Q 16	0.849967E-03	0.924423E-03	0.418767E-03	0.4530
Q 17	0.112275E-02	0.131778E-02	0.739879E-03	0.5615
Q 18	0.701428E-04	0.720016E-04	0.110620E-04	0.1536
Q 34	0.568621E-05	0.575464E-05	0.111705E-05	0.1941
Q 35	0.197591E-04	0.196796E-04	0.322614E-05	0.1639
Q 36	0.573293E-04	0.575328E-04	0.104855E-04	0.1823
Q 37	0.641095E-04	0.664651E-04	0.127805E-04	0.1923
Q 38	0.183228E-04	0.183956E-04	0.225698E-05	0.1227

	Bias Estimate	Bias Std. Error	Per Cent Bias	NLLS Estimate Corrected For Bias	C.V. For Corrected Estimate
Q 1	0.1517E-03	0.3118E-04	7.1569	0.1968E-02	0.3459
Q 2	0.7472E-04	0.2958E-04	2.9820	0.2431E-02	0.2704
Q 3	0.8468E-04	0.2480E-04	5.9635	0.1335E-02	0.4105
Q 4	0.8798E-04	0.1861E-04	6.6391	0.1237E-02	0.3288
Q 5	0.7836E-04	0.1431E-04	8.7815	0.8139E-03	0.3810
Q 6	0.2839E-04	0.8865E-05	3.5831	0.7639E-03	0.2568
Q 7	0.1795E-06	0.4184E-07	4.9931	0.3415E-05	0.2689
Q 8	0.2725E-06	0.7928E-07	3.4527	0.7621E-05	0.2298
Q 9	0.1871E-06	0.7537E-07	3.2361	0.5595E-05	0.2993
Q 10	0.3882E-06	0.7163E-07	11.6438	0.2946E-05	0.5274
Q 11	0.1769E-06	0.4083E-07	9.5362	0.1678E-05	0.5336
Q 12	0.2775E-06	0.5290E-07	18.5079	0.1222E-05	0.9409
Q 13	0.4997E-06	0.2683E-06	1.5861	0.3101E-04	0.1928
Q 14	0.3784E-05	0.1493E-05	2.3959	0.1541E-03	0.2152
Q 15	0.2007E-04	0.6552E-05	4.4090	0.4352E-03	0.3335
Q 16	0.7446E-04	0.1902E-04	8.7599	0.7755E-03	0.5400
Q 17	0.1950E-03	0.3422E-04	17.3716	0.9277E-03	0.7975
Q 18	0.1859E-05	0.5017E-06	2.6500	0.6828E-04	0.1620
Q 34	0.6843E-07	0.5005E-07	1.2034	0.5618E-05	0.1988
Q 35	-0.7948E-07	0.1443E-06	-0.4023	0.1984E-04	0.1626
Q 36	0.2034E-06	0.4690E-06	0.3548	0.5713E-04	0.1836
Q 37	0.2356E-05	0.5812E-06	3.6744	0.6175E-04	0.2070
Q 38	0.7283E-07	0.1010E-06	0.3975	0.1825E-04	0.1237

	LOWER 80. % CI	UPPER 80. % CI
Q 1	0.145102E-02	0.315938E-02
Q 2	0.177309E-02	0.350300E-02
Q 3	0.861406E-03	0.226706E-02
Q 4	0.941045E-03	0.197397E-02
Q 5	0.603069E-03	0.134357E-02
Q 6	0.588875E-03	0.106866E-02
Q 7	0.267697E-05	0.495628E-05
Q 8	0.591968E-05	0.104718E-04
Q 9	0.411532E-05	0.810935E-05
Q 10	0.197659E-05	0.583408E-05
Q 11	0.108615E-05	0.330985E-05
Q 12	0.656648E-06	0.334615E-05
Q 13	0.249530E-04	0.394452E-04
Q 14	0.121522E-03	0.205019E-03
Q 15	0.299199E-03	0.658288E-03
Q 16	0.472049E-03	0.149022E-02
Q 17	0.587057E-03	0.213450E-02
Q 18	0.585721E-04	0.876035E-04
Q 34	0.439891E-05	0.722621E-05
Q 35	0.160044E-04	0.238337E-04
Q 36	0.451248E-04	0.715270E-04
Q 37	0.506761E-04	0.827805E-04
Q 38	0.153827E-04	0.214638E-04

Bootstrap Output Variable: Fishing Mortality (2007)

	NLLS Estimate	Bootstrap Mean	Bootstrap Std Error	C.V. For NLLS Soln.
AGE 1	0.0464	0.0557	0.032226	0.5785
AGE 2	0.2383	0.2501	0.093221	0.3727
AGE 3	0.3495	0.3949	0.201140	0.5093
AGE 4	0.4631	0.5229	0.219453	0.4197
AGE 5	0.5570	0.6212	0.163984	0.2640
AGE 6	0.5570	0.6212	0.163984	0.2640

	Bias Estimate	Bias Std. Error	Per Cent Bias	NLLS Estimate Corrected For Bias	C.V. For Corrected Estimate
AGE 1	0.009267	0.001500	19.9549	0.0372	0.8670
AGE 2	0.011890	0.004203	4.9903	0.2264	0.4118
AGE 3	0.045404	0.009222	12.9899	0.3041	0.6614
AGE 4	0.059782	0.010173	12.9083	0.4033	0.5441
AGE 5	0.064133	0.007876	11.5133	0.4929	0.3327
AGE 6	0.064133	0.007876	11.5133	0.4929	0.3327

	LOWER 80. % CI	UPPER 80. % CI
AGE 1	0.023788	0.094236
AGE 2	0.148484	0.382338
AGE 3	0.199230	0.626814
AGE 4	0.285896	0.831382
AGE 5	0.440975	0.841022
AGE 6	0.440975	0.841022

Bootstrap Output Variable: Average F (2007) AGES 4 - 5

	NLLS Estimate	Bootstrap Mean	Bootstrap Std Error	C.V. For NLLS Soln.
AVG F	0.5101	0.5720	0.177343	0.3100
N WTD	0.4784	0.5403	0.203690	0.3770
B WTD	0.4828	0.5453	0.199536	0.3659
C WTD	0.4779	0.5383	0.203961	0.3789

	Bias Estimate	Bias Std. Error	Per Cent Bias	NLLS Estimate Corrected For Bias	C.V. For Corrected Estimate
AVG F	0.061958	0.008402	12.1466	0.4481	0.3957
N WTD	0.061952	0.009522	12.9498	0.4164	0.4891
B WTD	0.062470	0.009351	12.9382	0.4204	0.4747
C WTD	0.060465	0.009515	12.6529	0.4174	0.4886

	LOWER 80. % CI	UPPER 80. % CI
AVG F	0.377615	0.824861
N WTD	0.323682	0.817200
B WTD	0.334919	0.818263
C WTD	0.321282	0.814314

Bootstrap Output Variable: Biomass

JAN-1 Biomass (2008) Mean Biomass & SSB (2007)

	NLLS Estimate	Bootstrap Mean	Bootstrap Std Error	C.V. For NLLS Soln.
JAN-1	11172.	13511.	5152.	0.3813
MEAN	5535.	6042.	1915.	0.3169
SSB	7236.	7875.	2216.	0.2814

	Bias Estimate	Bias Std. Error	Per Cent Bias	NLLS Estimate Corrected For Bias	C.V. For Corrected Estimate
JAN-1	2339.	253.	20.9357	8833.	0.5833
MEAN	507.	89.	9.1632	5028.	0.3809
SSB	638.	103.	8.8192	6598.	0.3359

	LOWER 80. % CI	UPPER 80. % CI
JAN-1	7985.	20256.
MEAN	3873.	8677.
SSB	5300.	10733.

Plus Group Diagnostic Report

Calculation Method Selected = Backward

Year	Population Backward	Population Forward	F Forward	F Backward	Ratio
1982	932.	932.	1.401488	1.401496	1.000005
1983	580.	545.	1.886933	1.613466	0.855073
1984	310.	327.	1.886933	2.211595	1.172058
1985	82.	71.	1.886933	1.350613	0.715771
1986	69.	64.	2.130353	1.728682	0.811454
1987	25.	54.	0.257708	0.661775	2.567921
1988	93.	167.	0.637377	1.795549	2.817093
1989	139.	247.	0.484230	1.112728	2.297934
1990	185.	382.	0.303597	0.764521	2.518214
1991	112.	666.	0.102052	0.835058	8.182700
1992	112.	776.	0.099279	1.025611	10.330604
1993	73.	801.	0.070292	1.309665	18.631764
1994	70.	709.	0.065466	0.984819	15.043320
1995	220.	622.	0.103631	0.323722	3.123799
1996	272.	890.	0.135564	0.530377	3.912380
1997	1064.	2322.	0.221603	0.563406	2.542413
1998	1972.	3200.	0.383713	0.720849	1.878615
1999	2616.	4103.	0.375361	0.669256	1.782965
2000	799.	2857.	0.173012	0.821885	4.750451
2001	833.	2909.	0.208224	1.036577	4.978188
2002	345.	2152.	0.129287	1.343702	10.393179
2003	302.	1590.	0.198410	2.391292	12.052273
2004	47.	1033.	0.042879	2.163593	50.458683
2005	34.	776.	0.040222	2.060598	51.230496
2006	18.	582.	0.023370	1.280009	54.771862
2007	15.	448.	0.014739	0.551376	37.408845
2008	9.	381.	N/A	N/A	

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