Northeast Fisheries Science Center Reference Document 14-09

59th Northeast Regional Stock Assessment Workshop (59th SAW)

Assessment Report

by the Northeast Fisheries Science Center

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NOAA Fisheries, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543

U.S. DEPARTMENT OF COMMERCE
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Northeast Fisheries Science Center
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Table of Contents

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 Statistical 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 summary of the assessment results in a format useful to managers; an *Assessment Report* – a detailed account of the assessments for each stock; and the SARC panelist reports – 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 59th SARC was convened in Woods Hole at the Northeast Fisheries Science Center, July 15-18, 2014 to review benchmark stock assessments of: Gulf of Maine haddock (*Melanogrammus aeglefinus*) and sea scallop (*Placopecten magellanicus*). CIE reviews for SARC59 were based on detailed reports produced by NEFSC Assessment Working Groups. 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:

Text in this section is based on SARC-59 Review Panel reports (available at http://www.nefsc.noaa.gov/nefsc/saw/ under the heading "SARC-59 Panelist Reports").

For **Gulf of Maine haddock** all but one of the Terms of Reference (ToRs) were fully met and the assessment results from an ASAP model can be used as a basis for management. In 2013, overfishing was not occurring, and the stock was not overfished. The Panel recommended that future work could be done on estimation of the survival rate of discards in the recreational fishery and on the natural mortality rate. Given the continued changes in fishing practices, gear and location, along with the possibility of hyper-aggregation, fishery LPUE for GoM haddock is currently not a reliable indicator of stock status or dynamics.

For **sea scallop** all of the ToRs were fully met and the assessment results can be used as a basis for management. In 2013, overfishing was not occurring, and the stock was not overfished. Stock reconstructions were conducted appropriately using a statistical length-based model (CASA). The assessment used data collected with scallop dredges, a towed digital camera, and a video drop camera. The Panel felt that uncertainty in the assessment was underestimated and identified approaches to address this in the future.

Table 1. 59th Stock Assessment Review Committee Panel.

SARC Chairman (MAFMC SSC):

Mr. J.-J. Maguire
Sillery, Quebec CANADA
Email: jeanjacquesmaguire@gmail.com

SARC Panelists (CIE):

Dr. Panayiota Apostolaki
Technical advisor
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London, UK
Email: yiotaapost@yahoo.ie

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Table 2. Agenda, 59th Stock Assessment Review Committee Meeting.

July 15-18, 2014

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

AGENDA* (version: July 14, 2014)

TOPIC	PRESENTER(S)	SARC LEADER	RAPPORTEUR

Tuesday, July 15

10 – 10:30 AM
Welcome — **James Weinberg**, SAW Chair
Introduction — **J. –J. Maguire,** SARC Chair
Agenda
Conduct of Meeting

10:30 – 12:30 PM Assessment Presentation (A. GoM haddock)
Mark Terceiro TBD **Jon Deroba**

12:30 – 1:30 PM Lunch

1:30 – 3:30 PM Assesssment Presentation (A. GoM haddock)
Mark Terceiro TBD **Chuck Adams**

3:30 – 3:45 PM Break

3:45 – 5:45 PM SARC Discussion w/ Presenters (A. GoM haddock)
J. –J. Maguire, SARC Chair **Chuck Adams**

5:45 – 6 PM Public Comments

Wednesday, July 16

8:30 – 10:30 AM Assessment Presentation (B. scallop)
Dvora Hart TBD **Toni Chute**

10:30 – 10:45 AM Break

10:45 – 12:30 PM (cont.) Assessment Presentation (B. scallop)
Dvora Hart TBD **Toni Chute**

12:30 – 1:30 PM Lunch

1:30 – 3:30 PM	SARC Discussion w/presenters (B. scallop) **J. –J. Maguire,** SARC Chair	**Toni Chute**
3:30 – 3:45 PM	Public Comments	
3:45 -4 PM	Break	
4 – 6 PM	Revisit with presenters (A. GoM haddock) **J. –J. Maguire,** SARC Chair	**Brian Linton**

Thursday, July 17

8:30 – 10:30	Revisit with presenters (B. scallop) **J. –J. Maguire,** SARC Chair	**Alicia Miller**
10:30 – 10:45	Break	
10:45 – 12:15	Review/edit Assessment Summary Report (A. GoM haddock) **J. –J. Maguire,** SARC Chair	**Alicia Miller**
12:15 – 1:15 PM	Lunch	
1:15 – 2:45 PM	(cont.) edit Assessment Summary Report (A. GoM haddock) **J. –J. Maguire,** SARC Chair	**Tony Wood**
2:45 – 3 PM	Break	
3 – 6 PM	Review/edit Assessment Summary Report (B. scallop) **J. –J. Maguire,** SARC Chair	**Burton Shank**

Friday, July 18

9:00 AM – 5:00 PM SARC Report writing. (closed meeting)

*All 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. 59th SAW/SARC, List of Attendees

Name	Affiliation	Email
Adams Chuck	NEFSC	Charles.adams@noaa.gov
Brooks Liz	NEFSC	Liz.brooks@noaa.gov
Carr-Harris Andrew	SSB	Andrew.carr-harris@noaa.gov
Chang Jui-Han	NEFSC	jui-han.chang@noaa.gov
Cournane Jamie	NEFMC	jcournane@nefmc.org
Deroba Jon	NEFSC	jonathan.deroba@noaa.gov
Evans John	NEFSC	John.g.evans@noaa.gov
Hart Dvora	NEFSC	Deborah.hart@noaa.gov
Hogan Fiona	NEFMC	Fiona.hogan@noaa.gov
Jacobson Larry	NEFSC	Larry.jacobson@noaa.gov
Legault Chris	NEFSC	chris.legault@noaa.gov
Linton Brian	NEFSC	brian.linton@noaa.gov
Miller Alicia	NEFSC	Alicia.miller@noaa.gov
Nies Tom	NEFMC	tnies@nefmc.org
Nitschke Paul	NEFSC	paul.nitschke@noaa.gov
O'Brien Loretta	NEFSC	Loretta.O'Brien@noaa.gov
Odell Jackie	NSC	Jackie@northeastseafoodcoalition.org
Rago Paul	NEFSC	Paul.Rago@noaa.org
Serchuk Fred	NEFSC	fred.serchuk@noaa.gov
Sosebee Kathy	NEFSC	katherine.sosebee@noaa.gov
Tooley Mary Beth	NEFMC	mbtooley@live.com
Weinberg James	NEFSC	James.Weinberg@noaa.org
Whitmore Willie	GARFO	William.whitmore@noaa.gov
Wigley Susan	NEFSC	susan.wigley@noaa.gov
Wood Tony	NEFSC	anthony.wood@noaa.gov

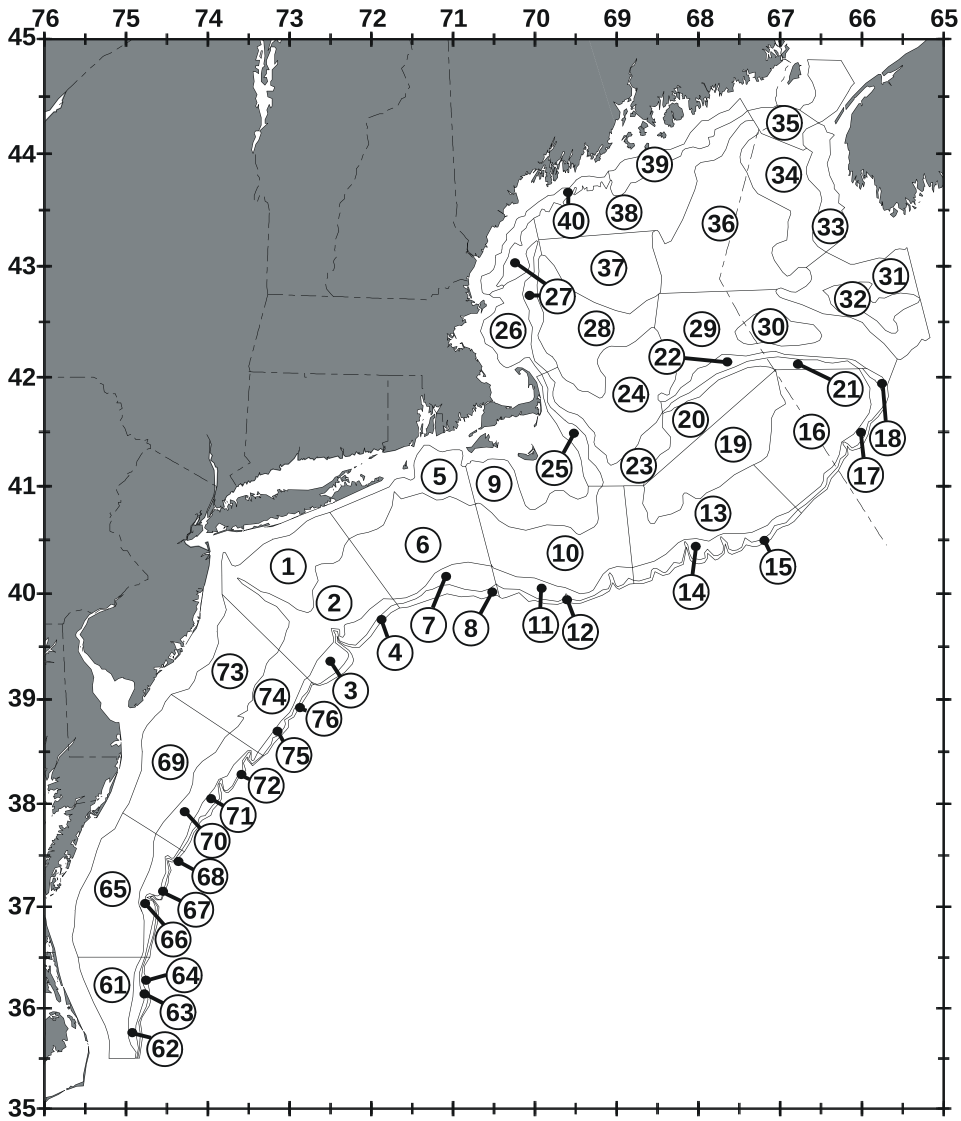

Figure 1. Offshore depth strata that have been sampled during Northeast Fisheries Science Center bottom trawl research surveys. Some of these may not be sampled presently.

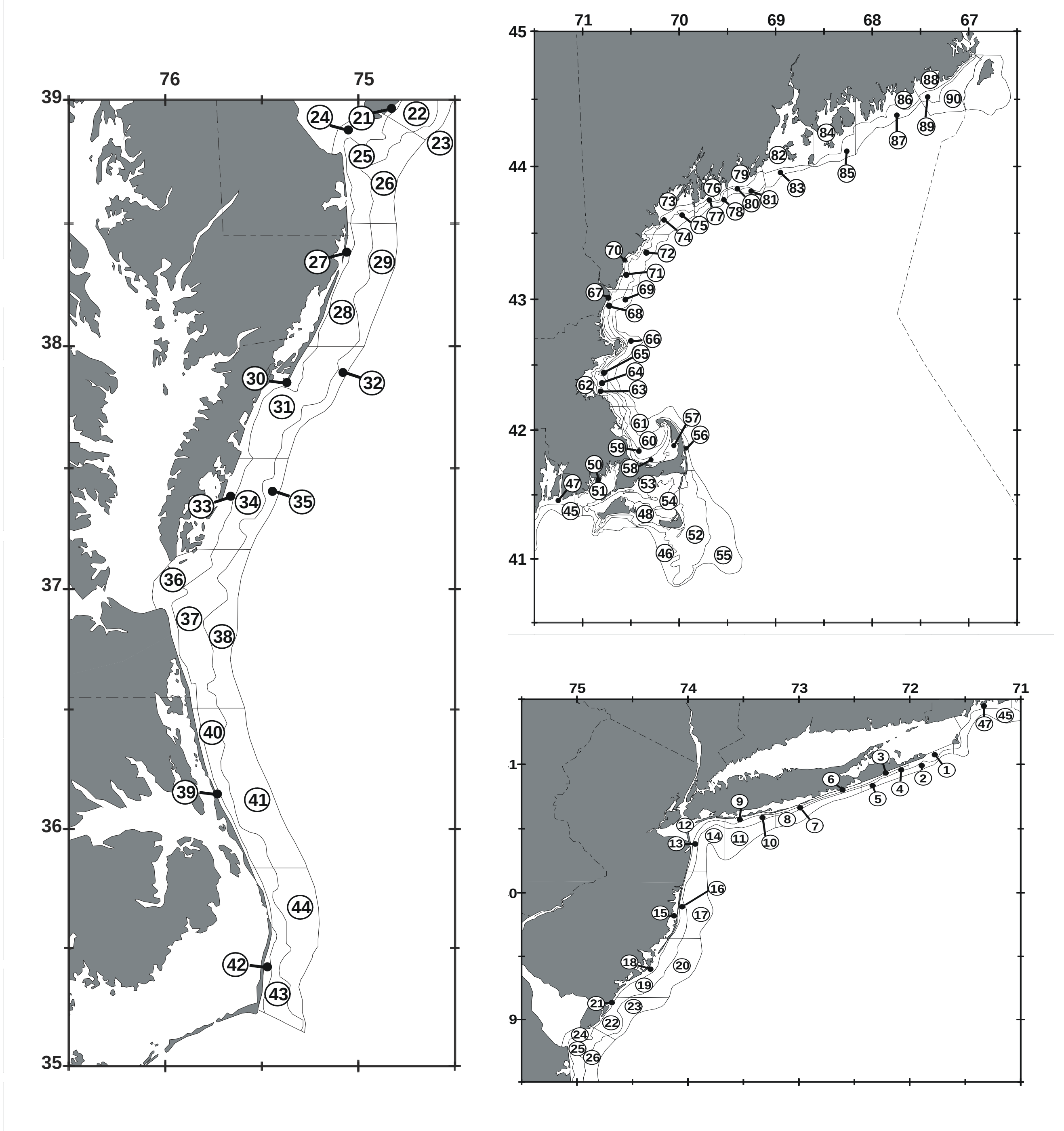

Figure 2. Inshore depth strata that have been sampled during Northeast Fisheries Science Center bottom trawl research surveys. Some of these may not be sampled presently.

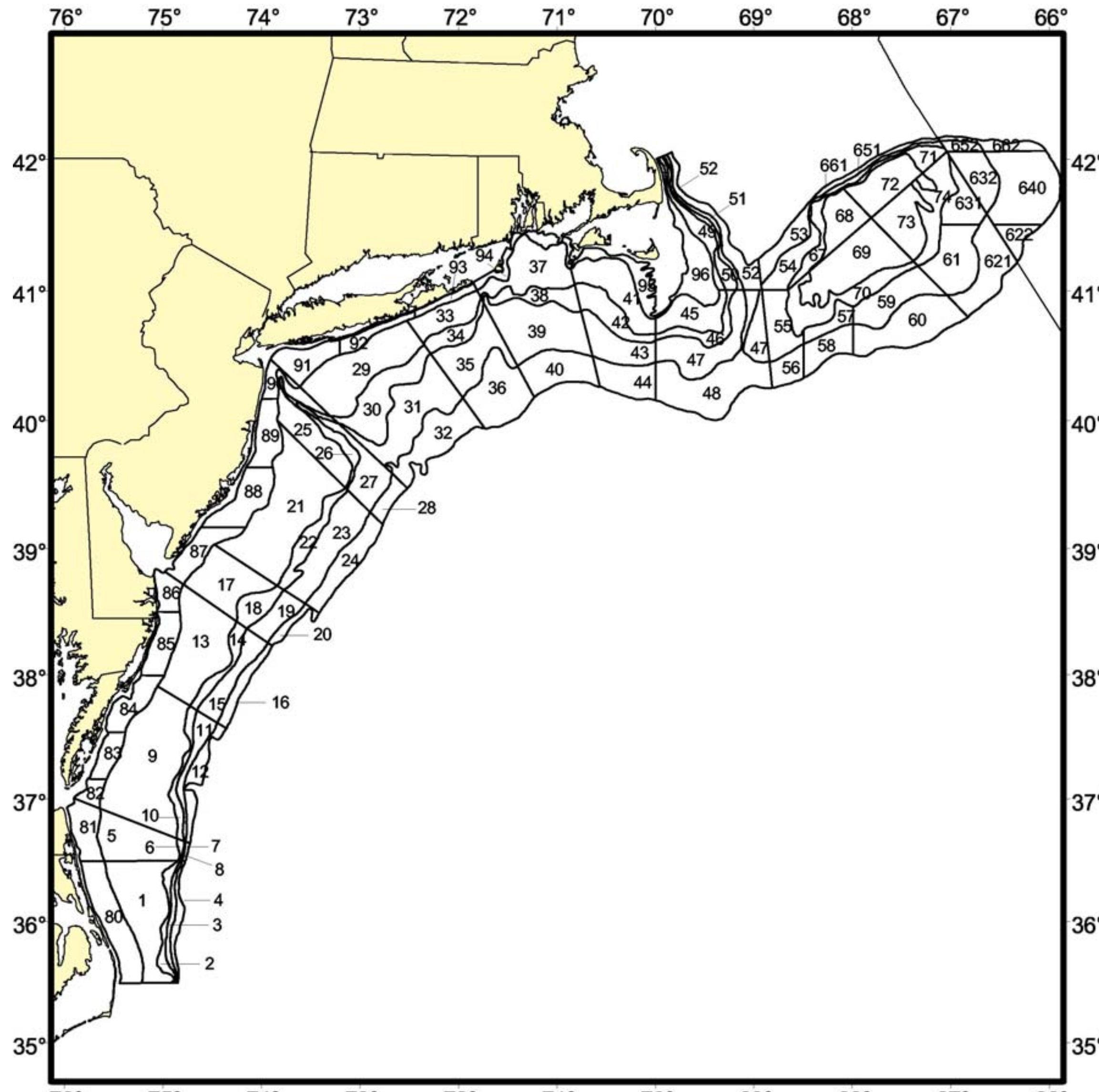

Figure 3. Depth strata sampled during Northeast Fisheries Science Center clam dredge research surveys.

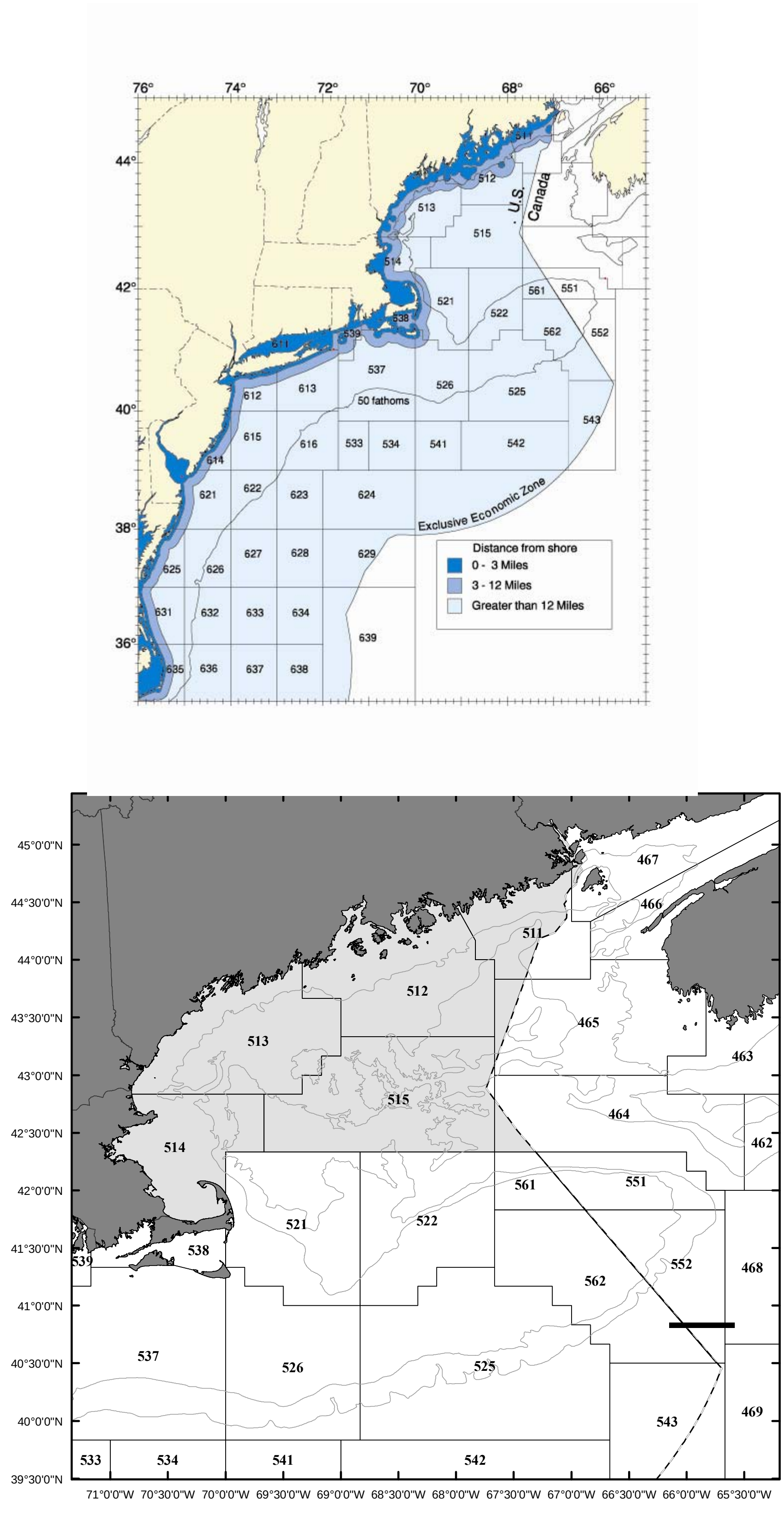

Figure 4. Statistical areas used for reporting commercial catches.

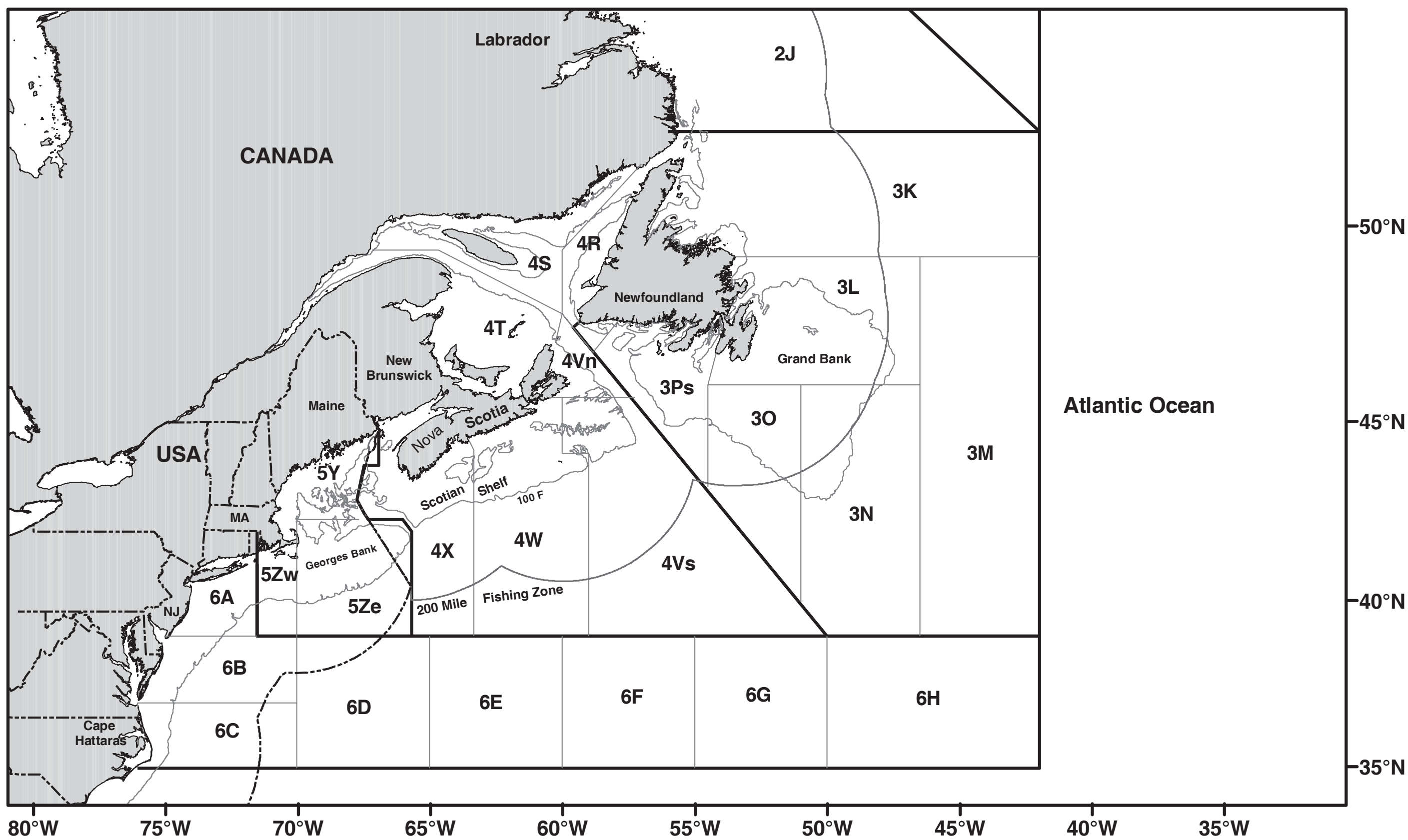

Figure 5. Catch reporting areas of the Northwest Atlantic Fisheries Organization (NAFO) for Subareas 3-6.

A. GULF OF MAINE HADDOCK BENCHMARK STOCK ASSESSMENT FOR 2014

Executive Summary

TOR 1 Estimate catch from all sources including landings and discards. Include recreational discards, as appropriate. Describe the spatial and temporal distribution of landings, discards, and fishing effort. Characterize the uncertainty in these sources of data. Investigate the utility of commercial or recreational LPUE as a measure of relative abundance.

Since 1977, fishery removals of Gulf of Maine haddock have ranged from 187 mt to 7,656 mt. Fishery removals over the past five years have ranged from 692 mt to 958 mt. Prior to 1989 there are no direct estimates of commercial discards but discards were hindcast back to 1982 by gear. Prior to 1981 there are no direct estimates of recreational removals and no attempt was made to hindcast recreational catch pre-1981. Over the assessment time series, commercial landings have been the dominant source of fishery removals, constituting 30-100% of the total catch. Commercial discards have been a small component of fishery removals with the exception of a period between 1993 and 1997 when trip limits were 1,000 lb or less. Recreational catch has varied annually from a low of 39 mt in 1981 to a high of 618 mt in 2007. Recreational catches have constituted between <1% and 65% of total annual removals, averaging 17% over the 1977-2013 period.

Currently both the commercial and recreational fisheries are primarily concentrated in the western Gulf of Maine region. Historically, the commercial trawl fishery was more broadly distributed with a large fraction of the landings coming from statistical area 515 in the central Gulf of Maine. The spatial trends in the fishery are partly in response to changes in the distribution of Gulf of Maine haddock, but also reflect changes in the trawl fleet composition and the effects of fishery regulations.

The SAW 59 WG evaluated standardized landings per unit effort (LPUE) indices from both the commercial and recreational fishery and considered their utility as indices of abundance within the Gulf of Maine haddock stock assessment. Over the longer term, there have been a number of regulatory changes (e.g. seasonal closures, trip limits, etc) which call into question the utility of LPUE as an index of haddock biomass. Based on these concerns, the SAW 59 WG recommended that the LPUE indices not be used in the SAW 59 assessment models. It should be noted that sensitivity models were developed that incorporated LPUE indices and these model results are similar to those of the base model, but model diagnostics highlight the poor explanatory power of the LPUE indices (described in Appendix A.2).

TOR 2 Present the survey data being used in the assessment (e.g., indices of relative or absolute abundance, recruitment, state surveys, age-length data, etc.). If available, consider whether tagging information could be used in estimation of stock size or exploitation rate. Characterize the uncertainty and any bias in these sources of data.

The Northeast Fisheries Science Center (NEFSC) spring and fall bottom trawl surveys began in 1968 and 1963 respectively, providing a long time series of fishery independent indices. The *aggregate indices of abundance (numbers/tow) and biomass (weight/tow) have fluctuated over time, primarily in response to episodic recruitment events. The time series is characterized by an early period of high abundance followed by a decline to time series lows in the late-1980s and early 1990s that also coincided with truncation in the population structure. Since the late 1990s the population has generally increased – first in response to the contribution of the 1998 year class, and most recently due to several moderate to strong recruitment events since 2010. These recent large year classes have lead to increases in survey indices that are at, or near, time series highs.*

The MADMF bottom trawl survey began in 1978, with two surveys (spring and fall) conducted annually. Age-specific Gulf of Maine haddock indices are not available for this survey. Indices-at-age were constructed by applying age-length info from the NEFSC survey The MADMF survey catches very few older fish and exhibits poor cohort tracking within the survey. Additionally, it shows only marginal cohesion with the NEFSC surveys.

The SAW 59 WG also evaluated data from the Maine-New Hampshire (MENH) inshore groundfish survey which began in the fall of 2000. Age-specific information are only available for the fall survey from 2005 onward, though work is currently being conducted to age available structures from the spring survey. Indices-at-age were constructed by augmenting the available age information with age-length info from the NEFSC survey. The degree of cohort tracking is greater in the MENH survey compared to the MADMF survey. However, similar to the MADMF survey, catches are dominated by young fish. Survey indices show poor agreement with NEFSC survey indices.

Model sensitivities were explored which incorporated the MADMF and MENH survey indices. Generally, model diagnostics for these sensitivities were poor with large residuals on state survey fits. The combination of residual patterns on the indices-at-age fits and poorly estimated selectivity at older ages suggests that there is limited utility in incorporating the older age classes from the state surveys in the tuning of the ASAP model. Attempts to fit only the age-1 indices resulted in neither improved model diagnostics nor markedly different model results. The state surveys were not included in the final population model.

The tagging component of this TOR is described under TOR 3.

TOR 3 Evaluate the hypothesis that haddock migration from Georges Bank influences dynamics of GOM stock. Consider role of potential causal factors such as density dependence and environmental conditions.

Several lines of evidence evaluated by the SAW 59 WG indicate that the mixing rates between the Gulf of Maine and Georges Bank stocks are low. The SAW 59 WG evaluated an in-depth review/analysis of stock mixing prepared by staff from NEFSC, the NEFMC Groundfish Plan Development Team (NEFMC GPDT 2013). The investigation had four primary themes:

- *Literature review of Gulf of Maine/Georges Bank exchange rates.*

- *Revisiting past assertions of recruitment synchrony between the Gulf of Maine and Georges Bank stocks.*
- *Year-class tracking in survey data and Gulf of Maine haddock assessment diagnostics.*
- *Analysis of the consequences of setting catch advice based on movement rate assumptions.*

Based on the work performed by the NEFSC and GPDT, the GPDT concluded that there was no technical basis for adjusting the quota between the two stocks based on the "spillover" of Georges Bank haddock into the Gulf of Maine stock. The SSC agreed with this conclusion noting the significant risk to the Gulf of Maine haddock resource that could occur should an adjustment to the quota be made, particularly given "…the lack of compelling empirical evidence." The SSC further noted that "if fishermen are observing abundance of haddock in the Gulf of Maine that does not seem to comport with the outcomes of the assessment, this might be due to a recent increase since the terminal year of the last assessment update (2010). If so, the appropriate response is to update the Gulf of Maine assessment to see if that change is detected."

In addition to the above work, the SAW 59 WG reviewed a re-analysis of Northeast Consortium Cooperative Haddock Tagging Program data. Between March 2005 and December 2008 the Northeast Consortium Cooperative Haddock Tagging (NCCHT) Program tagged 20,418 haddock in the Georges Bank and Gulf of Maine region. Of the releases 531 recoveries (168 released with two tags) were reported between 2005 and 2010. Miller and Palmer (2014) applied a finite-state continuous time model to the existing NCCHT data to provide estimates of mortality and movement rates. The results of the reanalysis showed poor precision of the natural mortality rate estimate, but the point estimate was consistently between 0.2 and 0.3 for reporting rates ≥ 0.3. The instantaneous migration rates implied greater movement of individuals into the Gulf of Maine than to the Georges Bank stock area given that they survive all sources of mortality and the estimates are not sensitive to the assumed reporting rate. With a reporting rate = 1, the migration rate estimates implied individuals starting in the Gulf of Maine had an approximately 94% probability of being in the Gulf or Maine 1 year later given they survived the interval. Individuals from the Georges Bank region had an approximately 86% probability of being in the Georges Bank region one year later given they survived the interval. Fishing mortality rate estimates were negatively correlated with reporting rates.

The SAW 59 WG found the mortality rates from the reanalysis consistent with other lines of information (e.g., catch-curve analyses, assessment model outputs), but felt that the mixing rate estimates were high and inconsistent with the analyses conducted by the GPDT. The authors stressed that the results are greatly affected by the location, size of fish, and timing of the releases. Many of the releases were near the stock boundaries and in areas closed to groundfishing. The proximity to the stock boundaries might cause migration rates to be greater than the general population if there are substantial portions further away from the stock boundaries and they move at similar speeds and directions. The SAW 59 WG did not feel that the tagging exercises conducted to date had been designed in a way that would allow annual interchange proportions to be estimated reliably.

The SAW 59 WG also examined sensitivity assessment models that allowed for estimation of mixing between stocks. These model results are described under TOR 4, but generally the *estimated annual percent mixing from Georges Bank to the Gulf of Maine from these models was low, and consistent with the PDT analysis. Stock structure cannot be specified conclusively with available information. Directed research designed to expressly determine between-stock movement rates is needed to definitely address the degree of mixing between the two stocks (see TOR8).*

TOR 4 Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series (integrating results from TOR-3), and estimate their uncertainty. Include a historical retrospective analysis to allow a comparison with previous assessment results and previous projections.

The VPA model used for the most recent assessment of Gulf of Maine haddock (2012 AOP) was updated to account for the changes to the data inputs as well as three additional years of catch and survey data. The changes to the input data included:

- *Revisions to commercial landings and reestimated landings-at-age.*
- *Revisions to the commercial discard fleets and reestimated discards-at-age.*
- *Revisions to the recreational catch to convert MRFSS catch to MRIP-equivalents.*
    - *Reestimated recreational landings-at-age.*
    - *Estimation of recreational discards-at-age (not included in prior assessments).*
- *Reestimated survey indices and indices-at-age.*
- *Updated maturity ogive.*

The updated VPA estimated the 2013 spawning stock biomass (SSB_{2013}) at 6,135 mt and average fishing mortality on ages 6-8 ($F_{6\text{-}8(2013)}$) at 0.82. The 2012 AOP VPA assessment estimated SSB_{2010} at 2,868 mt and $F_{6\text{-}8(2010)}$ at 0.82. Comparatively, the updated VPA now estimates SSB_{2010} at 3,070 mt and $F_{6\text{-}8(2010)}$ at 0.82. The general conclusions from the updated VPA are that the updates to the data inputs had only minor impacts on the model results and extending the time series through to 2013 did not change the historical perception of the resource. The more recent data does suggest that two strong year classes have been spawned since 2010. There has been an overall increase in the spawning stock biomass, primarily as result of the 2010 year class moving into the spawning population. The projections from the 2012 AOP update assumed the size of the 2010 year class to be equal to the geometric mean recruitment of the time series; based on the updated VPA, this assumption underestimated the year class size. ***The updated VPA is not the base model for this assessment.***

In this updated assessment, a statistical catch-at-age model (ASAP) represents the new preferred model. While the results of the GARM III and 2012 AOP assessments show that catch-at-age could be constructed to support a defensible VPA model, the amount of imputation required to construct the catch-at-age time series, primarily in the way of commercial discards and recreational catch, introduces questions as to whether this stock would be better assessed using a statistical catch-at-age model where it is not assumed that catch is known exactly. Additional support for exploring a statistical catch-at-age model include: the ability to explore alternative model formulations to counter/lend support to VPA results, and the ability to explicitly handle data uncertainty, particularly with respect to uncertainty in the survey data.

The SAW 59 WGs preferred ASAP model (ASAP_final_temp10) reflects the best model with which to evaluate stock status and provide catch advice. The assessment indicates that total SSB has ranged from 600 mt to 15,178 mt during the assessment time period, with current SSB in 2013 estimated at 4,153 mt (90% CI = 2,690 – 6,043 mt). Currently, total biomass is estimated at 7,749 mt (90% CI = 5,470 – 11,039 mt). The 2013 fully recruited fishing mortality (F_{full}) is estimated at 0.39 (90% CI =0.24 – 1.60).

A retrospective analysis over the years 2006 to 2013 indicated small retrospective error in both F and SSB with no consistent patterns for under/over-estimation. Over the last 7 years, retrospective error resulted in an average of 10% underestimation of SSB and 24% overestimation of fishing mortality. The SAW 59 WG recommended that no correction be made for the retrospective error given the small magnitude and lack of consistent patterns.

The time series mean recruitment (age-1) is around 2.6 million fish. Recruitment patterns of Gulf of Maine haddock are highly episodic, a feature common among many haddock stocks. Several moderate to strong year classes have been spawned in the last fifteen years, including the 1998, 2003, 2010 and most recently, the 2012 year class. ***The size of the 2012 year class is the largest source of uncertainty in this stock assessment, owing to the fact that the estimate is based entirely on only two survey data points.*** *A sensitivity ASAP model (ASAP_final_temp11) was brought forward which placed additional constraint on the estimation of the 2012 year class to illustrate the impacts of 2012 year class size uncertainty on catch projections (see TOR7).*

The SAW 59 WG also evaluated the results from three sensitivity models based on the SCAA statistical catch-at-age methodology (see Appendix 3). All three of the SCAA models achieved results similar to the ASAP_final_temp10 model. The first of the SCAA models considers haddock in the Gulf of Maine to be an isolated stock (SCAA no movement model), which is identical to the WGs preferred ASAP model. The other two incorporate movement into the Gulf of Maine, either permanent or temporary, by haddock from Georges Bank. Under both movement models, the amount of mixing is estimated to be low (<0.8% of the Georges Bank stock annually moving into the Gulf of Maine region). The evidence for such movement from these analyses point to scenarios involving limited movement being of similar plausibility to that of an isolated stock; however, mixing amongst the stocks has limited impact on assessment results. The WG discussed how to interpret the mixing parameter estimates coming from the SCAA movement models. The SCAA movement models do not incorporate specific information to inform the model about migration rates (e.g., tagging); as such, the mixing parameters don't represent actual mixing rates, rather the mixing parameters represent upper bounds on the amount of mixing that could be supported by the data. The mixing parameters are confounded by other parameters or data observation/process error.

Given the limited among of mixing supported by the SCAA models and the robustness of the assessment results to mixing assumptions, The SAW 59 WGs recommended the ASAP model, ASAP_final_temp10, as the preferred model with which to evaluate stock status and provide catch advice – this decision was supported by the SARC.

TOR 5 State the existing stock status definitions for "overfished" and "overfishing". Then update or redefine biological reference points (BRPs; point estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, F_{MSY} and MSY) and provide estimates of their uncertainty. If analytic model-based estimates are unavailable, consider recommending alternative measurable proxies for BRPs. Comment on the scientific adequacy of existing BRPs and the "new" (i.e., updated, redefined, or alternative) BRPs.

The existing MSY reference points based on a spawning potential ratio (SPR) of 40% were established at GARM III and updated as part of the 2012 AOP update. The overfishing definition is $F_{MSY\text{-}proxy} = F_{40\%} = 0.46$. Maximum sustainable yield and SSB_{MSY} were derived from the median values of long-term projections run at a constant harvest of $F_{40\%} = 0.46$. Projected recruitment was modeled from a cumulative density function (CDF) of VPA model estimated recruitment as well as a hindcast of recruitment between 1963 and 1977. Recruitment events that were a) associated with the large 1962 year class (considered a "bonanza" outlier), or b) when SSB was less than 3,000 mt, were excluded from the recruitment series. The resulting BRP estimates were: SSB_{MSY} = 4,904 mt (90% confidence interval of 2,272 – 10,604 mt), and MSY = 1,117 mt (90% confidence interval of 553 – 2,563 mt). A stock is considered to be overfished if spawning biomass is less than half of SSB_{MSY}; the existing overfished definition is ½ SSB_{MSY} = 2,452 mt.

New reference points were warranted given the changes in data inputs and the assessment model, as well as small changes in the fishery selectivity and weights-at-age. The WG concluded that because Gulf of Maine haddock recruitment events are highly episodic and not well described by traditional stock recruitment relationships, a MSY proxy approach to reference points was warranted. This is the same conclusion reached at GARM III.

The WG saw no compelling reason to select a different F_{MSY} proxy than the $F_{40\%}$ metric that had been adopted previously. While there were differences in the YPR inputs between the 2012 update and the current assessment, these differences were small. The resulting $F_{40\%}$ values were identical (0.46) to the $F_{MSY\text{-}proxy}$ value from the 2012 assessment. Stochastic long-term projections at $F_{40\%}$ were used to determine new recommended biomass-related reference points (proxies for both SSB_{MSY} and MSY). The projection inputs were identical to the YPR inputs.

The WG discussed various ways to project future recruitment. It found the GARM III method to be arbitrary (e.g., excluding very large and very small recruitment events) and instead opted to use a simpler method of using the CDF of the 1977-2011 age-1 recruitments from the preferred ASAP model. Age-1 recruitments in 2012 and 2013 were not included in the cumulative density function due to their greater variance. The resulting biomass reference points and their 90% confidence intervals are SSB_{MSY} = 4,108 mt (1,774 – 7,861 mt) and MSY = 955 mt (421 – 1,807 mt). The overfished biomass threshold of ½ SSB_{MSY} = 2,054 mt.

TOR 6 Evaluate stock status with respect to the existing model (from previous peer reviewed accepted assessment) and with respect to a new model developed for this peer review. In both cases, evaluate whether the stock is rebuilt (if in a rebuilding plan).

a. **When working with the existing model, update it with new data and evaluate stock status (overfished and overfishing) with respect to the existing BRP estimates.**

The existing reference points are $F_{MSY\text{-}proxy} = F_{40\%} = 0.46$, $SSB_{MSY} = 4{,}904$ mt (90% confidence interval of 2,272 – 10,604 mt) (½ SSB_{MSY}, or 2,452 mt), and MSY = 1,117 mt (90% confidence interval of 553 – 2,563 mt). The updated VPA model (Model 6, 2013_UPDATE) estimates 2013 SSB at 3,070 mt. This exceeds the existing overfished threshold of 2,452 mt; therefore, the stock is not overfished. The updated estimate of average fishing mortality on ages 6-8 ($F_{6\text{-}8}$) in 2013 is 0.82. This is greater than the overfishing limit of 0.46, and therefore, overfishing is occurring.

b. **Then use the newly proposed model and evaluate stock status with respect to "new" BRPs and their estimates (from TOR-5).**

The revised reference points are $F_{MSY\text{-}proxy} = F_{40\%} = 0.46$, $SSB_{MSY} = 4{,}108$ mt (90% confidence interval of 1,774 – 7,861 mt) (½ SSB_{MSY}, or 2,054 mt), and MSY = 955 mt (90% confidence interval of 421 – 1,807 mt). The ASAP_final_temp10 model estimates 2013 SSB at 4,153 mt. This is greater than the SSB_{MSY} level of 4,108 mt; therefore, the stock is rebuilt and not overfished. The estimate of 2011 fully recruited fishing mortality (F_{full}) is 0.39. This is less than the overfishing limit of 0.46, and therefore, overfishing is not occurring.

TOR 7 Develop approaches and apply them to conduct stock projections and to compute the statistical distribution (e.g., probability density function) of the OFL (overfishing level) (see Appendix to SAW TORs for definitions).

a. **Provide numerical annual projections (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. Use a sensitivity analysis approach in which a range of assumptions about the most important uncertainties in the assessment are considered (e.g., terminal year abundance, variability in recruitment, migration from Georges Bank).**

The short-term (2014-2017) projection method samples from a cumulative density function derived from ASAP estimated age-1 recruitment from 1977 and 2011 (identical to the recruitment series used for establishing reference points). No retrospective adjustment needed to be applied in the projections.

Due to the high degree of uncertainty of the size of the 2012 year class, two projection models were developed. The first is based on the preferred population model (ASAP_final_temp10) and the second is based on a sensitivity model that constrained the size of the 2012 year class (ASAP_final_temp11). Both projection models were run under two different assumptions of calendar year 2014 catch – harvest at FMSY (0.46) and an assumed 2014 catch of 500 mt. The fishing year 2014 Gulf of Maine haddock Annual Catch Limit (ACL) is set at 323 mt, though the ACL does not account for recreational *discards. The 500 mt estimate used in the projections was informed by the fishing year 2014 ACL and recent recreational discard amounts.*

Catch projections under both models range from 1,271 mt to 2,512 mt between 2015 and 2017. Under all scenarios, spawning stock biomass is not projected to drop below the target biomass level (SSB_{MSY}) through 2017. The increase in biomass above target biomass levels during the projection period reflects the contribution of the 2010 and 2012 year classes to the exploitable biomass.

Recent reviews of historical and contemporary tagging studies suggest that there is movement of fish between the Gulf of Maine and Georges Bank stocks, though there is considerable uncertainty regarding the degree of mixing. Several lines of evidence examined during the SAW/SARC59 assessment indicate that annual percent mixing from Georges Bank to the Gulf of Maine is low, though the mixing scenarios have similar statistical plausibility to that of an isolated stock. While mixing amongst the stocks has limited impacts on stock status, catch projections of the SCAA models (Appendix 3) under constant fishing mortality were found to be sensitive to limited movement for the case where the movement is permanent (SCAA migration model), but much less so when movement was modeled as non-permanent interchange (SCAA sabbatical model). The catch projection results from the most biologically realistic SCAA mixing model (i.e., allows mixing between stocks as opposed to unidirectional movement) are nearly identical to the SCAA model with no mixing and within the 90% confidence intervals of the projections from the preferred ASAP model (Figure A.203).

The SAW 59 WG noted that the evidence for mixing is not conclusive and that the mixing scenarios have similar statistical similar plausibility to that of an isolated stock. Given this, it concluded that the projections based on the ASAP_final_temp10 model should be used for management advice. The SARC agreed with this decision.

b. Comment on which projections seem most realistic. Consider the major uncertainties in the assessment as well as sensitivity of the projections to various assumptions.

Both the WG and SARC concluded that the projections based off the ASAP_final_temp10 model were the 'most realistic'. However, it should be stressed that the absolute size of the 2012 year class is the largest source of uncertainty in this assessment. The risks associated with management actions taken during 2015 – 2017 were examined by undertaking stock projections under two different assumptions of year class size. Under both scenarios the spawning stock biomass is projected to increase well above the target levels and catch can be sustained above MSY levels.

The differences in these two short-term projections in 2014 and 2015 are primarily due to the differences in the size of the 2010 year class between the two different models. However, as the projection horizon increases, and the contribution of the 2012 year class becomes more important and the divergence in catch advice becomes larger (> 600 mt). Based on the estimated selectivity patterns, the 2012 year class is predicted to be 50%

selected by the fishery in 2017 at age-5. Recent changes to the commercial minimum retention size may result in this year class recruiting to the fishery sooner.

The assumption of the catch in 2014 will have limited impacts on stock size and catch advice in the subsequent years, though the two assumed values (catch= $F_{MSY\text{-}proxy}$ and 500 mt) should be re-evaluated once additional information on 2014 catches is available.

c. Describe this stock's vulnerability (see "Appendix to the SAW TORs") to becoming overfished, and how this could affect the choice of ABC.

There are several factors that should be considered when setting catch advice for the Gulf of Maine haddock stock. While these uncertainties have been discussed previously, particular attention should be given to the factors below when determining the appropriate level of scientific uncertainty to prescribe to this stock assessment.

The mortality of haddock discarded in the recreational and commercial fishery is unknown. For trawl and gillnet gear, mortality is likely high and not substantially different than the assumption of 100% used in the assessment. While there is limited information available to suggest that mortality of haddock discarded in the commercial longline fishery may be lower than 100%, given the small magnitude of longline removals, the impacts of this assumption on the assessment results are likely small. However, given the large amount of recreational discards occurring in recent years, the model results and subsequent catch advice could be sensitive to the assumption of 50% discard mortality used in this assessment. While the assessment results were shown to be relatively insensitive to this assumption, it does have implications for management and catch allocation between the commercial and recreational fleets.

Several lines of evidence examined during the SAW/SARC59 assessment indicate that annual percent mixing from Georges Bank to the Gulf of Maine is low; however, stock structure and the specific degree of mixing cannot be specified conclusively with the available information. While the catch projections for the more biologically realistic mixing scenario (non-permanent interchange) were nearly identical to no-movement assumptions, the projections which assumed permanent movement of Georges Bank haddock into the Gulf of Maine were higher than the no movement scenarios. Setting catch advice higher on the presumption that permanent movement of Georges Bank haddock into the Gulf of Maine is occurring, if in fact it is not, could lead to overfishing of the Gulf of Maine stock (NEFMC GPDT 2013).

The absolute size of the 2012 year class is the largest source of uncertainty in this assessment. Based on the estimated selectivity patterns, this year class is predicted to be 50% selected by the fishery in 2017 at age-5. Recent changes to the commercial minimum retention size may result in this year class recruiting to the fishery sooner. Given the high uncertainty with respect to this year class size, the assessment should be updated if future estimates of its size differ significantly from those used in this assessment.

TOR 8 Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in most recent SARC reviewed assessment and review panel reports. Identify new research recommendations.

The SAW 59 WG reviewed the status of previous research recommendations and proposed new ones to address issues raised during the WG meeting. There were two research recommendation carried forward from GARM III. One of which is no longer relevant due to the switch from a virtual population analysis assessment model to a statistical catch-at-age model. The second one relates to the estimation of haddock discarded in the recreational fishery, a topic which has been partially addressed in TOR 1. The SAW 59 WG reiterated the need for directed field research on this topic.

The WG noted that the haddock tagging experiments conducted to date were not designed to address the issue of between-stock movement rates. Research designed to expressly determine between-stock movement rates is needed to reduce the uncertainty of analytical models that include these rates.

Additionally, the SAW 59 WG proposed five new research recommendations which have broad applicability to many northeast United States groundfish stocks. These include: methods to standardize CPUE indices, development of approaches to incorporate additional stock-recruitment models and autoregressive error into population models, and advance the application of multi-model inference and risk evaluation into the Northeast Region stock assessment process.

SAW 59 Terms of Reference for Gulf of Maine (GOM) haddock

1. Estimate catch from all sources including landings and discards. Include recreational discards, as appropriate. Describe the spatial and temporal distribution of landings, discards, and fishing effort. Characterize the uncertainty in these sources of data. Investigate the utility of commercial or recreational LPUE as a measure of relative abundance.

2. Present the survey data being used in the assessment (e.g., indices of relative or absolute abundance, recruitment, state surveys, age-length data, etc.). If available, consider whether tagging information could be used in estimation of stock size or exploitation rate. Characterize the uncertainty and any bias in these sources of data.

3. Evaluate the hypothesis that haddock migration from Georges Bank influences dynamics of GOM stock. Consider role of potential causal factors such as density dependence and environmental conditions.

4. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series (integrating results from TOR-3), and estimate their uncertainty. Include a historical retrospective analysis to allow a comparison with previous assessment results and previous projections.

5. State the existing stock status definitions for "overfished" and "overfishing". Then update or redefine biological reference points (BRPs; point estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, F_{MSY} and MSY) and provide estimates of their uncertainty. If analytic model-based estimates are unavailable, consider recommending alternative measurable proxies for BRPs. Comment on the scientific adequacy of existing BRPs and the "new" (i.e., updated, redefined, or alternative) BRPs.

6. Evaluate stock status with respect to the existing model (from previous peer reviewed accepted assessment) and with respect to a new model developed for this peer review. In both cases, evaluate whether the stock is rebuilt (if in a rebuilding plan).
   a. When working with the existing model, update it with new data and evaluate stock status (overfished and overfishing) with respect to the existing BRP estimates.
   b. Then use the newly proposed model and evaluate stock status with respect to "new" BRPs and their estimates (from TOR-5).

7. Develop approaches and apply them to conduct stock projections and to compute the statistical distribution (e.g., probability density function) of the OFL (overfishing level) (see Appendix to SAW TORs for definitions).

a. Provide numerical annual projections (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. Use a sensitivity analysis approach in which a range of assumptions about the most important uncertainties in the assessment are considered (e.g., terminal year abundance, variability in recruitment, migration from Georges Bank).
b. Comment on which projections seem most realistic. Consider the major uncertainties in the assessment as well as sensitivity of the projections to various assumptions.
c. Describe this stock's vulnerability (see "Appendix to the SAW TORs") to becoming overfished, and how this could affect the choice of ABC.

8. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in most recent SARC reviewed assessment and review panel reports. Identify new research recommendations.

Introduction

The 59th Stock Assessment Workshop Working Group (SAW 59 WG) prepared the assessment report. The working group convened June 2-6, 2014 at the Northeast Fisheries Science Center (NEFSC) in Woods Hole, MA. A complete list of working group participants can be found in Appendix A.1.

Assessment history

Prior to 2002, Gulf of Maine haddock assessments had been conducted (NEFSC 1986,NEFSC 2001) by comparing exploitation rates to biological reference points generated from a surplus production model. The 32nd Stock Assessment Review Committee (SARC) expressed concerns with this approach and suggested that other approaches be explored (NEFSC 2001). In 2002, the Gulf of Maine haddock stock was assessed as part of the first Groundfish Assessment Review Meeting (GARM I, NEFSC 2002a). The 2002 assessment compared survey biomass and exploitation rate indices from 1963 to 2001 (Table 1) to biological reference points (BRPs) generated by the Working Group on Re-estimation of Biological Reference Points for New England Groundfish (NEFSC 2002b). Reference points were established using the index-based model, An Index Method (AIM) available from the NOAA Fisheries Toolbox (http://nft.nefsc.noaa.gov/). The reference points were based on a maximum sustainable yield (MSY) level approximated from the average commercial landings between 1959 and 1966 – this period represented a stable period in the landings time series. The fishing mortality (F) reference point (F_{ref}) was set equal to the relative F (Eq. 1) where the replacement ratio equaled 1. The replacement ratio is equal to the biomass index in the current year divided by the average biomass indices from a 3-year centered mean. The biomass reference point (B_{ref}) was estimated by dividing the MSY proxy by F_{ref}. At GARM I, the proxy F_{ref} (exploitation rate index) and $B_{Threshold}$ (1/2 B_{ref}) were estimated at 0.23 and 11.09 kg/tow, respectively. The 2001 exploitation rate was estimated at 0.12 and the 3-year average fall survey biomass index was 10.31 kg/tow. Based on these estimates, the GARM I assessment concluded that the Gulf of Maine haddock stock was overfished, but overfishing was not occurring (Table 2).

(Equation 1)
$$relF_t = \frac{C_t}{\left(\frac{I_{t-1} + I_t + I_{t+1}}{3}\right)}$$

The stock was reassessed again in 2005 as part of GARM II (NEFSC 2005). The same index-based approach used in GARM I was applied in the 2005 assessment. As of 2004, the exploitation rate had increased to 0.18 and the fall biomass index had declined to 5.79 kg/tow. Consequently, stock status remained unchanged. The GARM II review noted the sensitivity of the assessment results to the exclusion of commercial discards and recreational catch. Additionally, the GARM II review made a research recommendation to explore the use of age-structured models in future assessments. Previous assessments had not utilized age-structured models because biological data (length frequencies, age and maturity sampling) were sparse during the late 80s and early- to mid-90s and considered inadequate for use in a virtual population assessment (VPA) analytic assessment (NEFSC 2001).

The Gulf of Maine haddock stock was next assessed in 2008 as part of GARM III (NEFSC 2008). That assessment made several major improvements to the input data and assessment methodologies applied to previous assessments. Notably, commercial discards and recreational landings were included and attempts were made to reconstruct the catch-at-age and survey indices-at-age from 1977 to 2007 from the available biological samples and using hindcasting/imputation procedures. An ADAPT-VPA model was applied to the data, which estimated the 2007 spawning stock biomass at 5,850 mt and average fishing mortality on ages 6-8 was estimated at 0.35. GARM III reference points were based on a yield per recruit analysis, with $F_{40\%}$ (0.43) selected as the fishing mortality reference point ($F_{MSY\text{-}proxy}$). Long-term stochastic projections under a harvest strategy of $F_{40\%}$ were used to estimate proxy values of SSB_{MSY} and MSY of 5,900 mt (1/2 B_{MSY}=$B_{threshold}$=2,950 mt) and 1,360 mt, respectively. As of GARM III, the stock was not overfished and overfishing was not occurring. The stock was considered fully rebuilt due to the fact that the GARM III assessment indicated that spawning stock biomass had exceeded the biomass threshold in 2000.

Most recently, the Gulf of Maine haddock stock was assessed in 2012 (NEFSC 2012). This assessment included data through 2010. The 2012 assessment results were peer reviewed by an Assessment Oversight Panel (AOP) and constituted an update of the benchmark assessment developed at GARM III. Relative to GARM III, the 2010 SSB had declined to 2,868 mt and fishing mortality had increased to 0.82. BRPs were revised using updated estimates of selectivity and weights-at-age. The updated BRPs are shown in Table 2. Based on the results of the 2012 update, the Gulf of Maine haddock stock was declared not overfished, but overfishing was occurring. It should be noted that the projected 2011 spawning stock biomass was estimated to decline below the biomass threshold.

Fisheries Management

Gulf of Maine haddock have been managed under two different management authorities in recent history. Prior to 1977 the stock was managed under an international treaty through the International Commission for the Northwest Atlantic Fisheries (ICNAF). The majority of management measures implemented under ICNAF applied to the greater Subarea 5 which consists of both the 5Y(Gulf of Maine) and 5Z (Georges Bank) Divisions. Fisheries management was primarily controlled through annual total allowable catches (TACs), minimum mesh sizes and spawning closures (Clark et al. 1982). As early as 1951, mesh size regulations were imposed which initially set the minimum codend mesh size at 4.1 inches (114 mm), though these were increased to 5.1 inches (130 mm) in 1974. In response to severe declines in haddock abundance noted during the late 1960s, a 12,000 mt TAC was first implemented in 1970, with subsequent reductions to 6,000 mt in 1972 and then a prohibition on targeted fishing by 1974. The TAC was quickly raised back to 6,000 mt in 1975 under the rationale that establishing some low TAC level would be more effective at controlling fishing mortality compared to prohibiting targeting fishing. Spawning closures were first implemented in 1970, though these were restricted to the Georges Bank region. The Magnuson Fishery Conservation and Management Act (MFMCA) was passed in 1977 and subsequently the management authority of New England groundfish stocks shifted to the New England Fishery Management Council (NEFMC).

The use of TACs continued under the NEFMC authority through 1982. In 1982, the “Interim” Groundfish Fisheries Management Plan (FMP) was implemented, replacing the quota system (TAC) with input controls such as mesh sizes (Table A.3) and minimum retention sizes (Table A.4). The initial Groundfish FMP was implemented in 1985 and largely carried forward the existing measures from the interim FMP. Amendment 4 to the FMP required the use of a Nordmore grate in the northern shrimp fishery as well as placing a prohibition on the retention of groundfish bycatch in the shrimp fishery. Beginning with Amendment 5 (1994), there was a concerted attempt to reduce fishing effort through a days-at-sea (DAS) reduction schedule. Additionally, Amendment 5 brought about mandatory vessel reporting in the way of the Vessel Trip Reports (VTRs). Effort controls were increased under Amendment 7 through further acceleration of the DAS reduction schedule, and the addition of seasonal and year round closures in the Gulf of Maine. Between 1994 and 1999 trip limits ranged from 500 lb to 50,000 lb, in addition to limits on the allowed landings/DAS (Table A.3). Several increases in the minimum mesh sizes occurred, most notably a shift to 6 inch (152 mm) mesh in 1994 followed by increases to 6.5 inch mesh for square rigged trawls in 1999, and a 6.5 inch (165 mm) requirement for all trawl gear in 2000 under Framework 33. In 2004, Amendment 13 implemented mandatory electronic reporting for all primary federally permitted seafood dealers. Amendment 13 also established reference point thresholds for the 18 groundfish stocks as well as formalized rebuilding plans for all overfished stocks (< ½ SSB_{MSY}) – at the time, this included Gulf of Maine haddock. Through 2010, a series of framework actions and interim rules placed additional restrictions on DAS usage and seasonal closures on the recreational fishery. The effort controls first adopted in 1994 were frequently changed, making it difficult to isolate the effects of individual regulations. The use of often-changing trip limits led to increased discard rates and may have contributed to high-grading. In response to perceived high bycatch of haddock in the herring fishery, Framework 43 implemented a haddock bycatch cap at 0.2% of the combined total allowable catch of Gulf of Maine and Georges Bank haddock.

In 2010, the groundfish fishery experienced a major management change with the passage of Amendment 16. Amendment 16, with the introduction of annual catch limits (ACLs), represented a return to the use of hard TACs. Additionally, 17 new groundfish sectors were approved and those vessels not members of a groundfish sector were subject to additional cuts in DAS and restrictive trip limits. Vessels fishing under the sector management system were exempt from DAS restrictions and instead, each sector was given a share of the total commercial groundfish sub-ACL. How the catch was divided up amongst sector vessels, or how catch was allocated throughout the year, was left to the discretion of the sector. One of the requirements of Amendment 16 was an increase in the overall level of observer coverage. This was accomplished using observers trained through the existing Northeast Fisheries Observer Program (NEFOP) as well as a new class of observers termed At-Sea Monitors (ASMs). The data collection protocols for ASMs were restricted to catch estimation and the collection of limited biological information (e.g., lengths). The recent shift to a catch share system in 2010 appears to have dramatically reduced discards but it is too soon to fully understand the overall impacts of the sector management system.

Since the passage of Amendment 16, two framework modifications have been made to the FMP with direct impacts on the management of Gulf of Maine haddock. Framework 46, implemented in August 2011, revised the haddock bycatch cap for the herring fishery to apply to only mid-water trawl gear and establish separate stock specific caps equal to 1% of the acceptable biological catch (ABC) levels of the respective stocks. Framework 48 reduced the commercial minimum size for several groundfish species, including haddock, which was reduced from 18 to 16 inches. The reduced minimum size became effective on July 1, 2013. Around the same time, the National Marine Fisheries Service (NMFS), through an emergency action, increased the recreational minimum retention size from 18 to 21 inches in an attempt to constrain recreational catches to the allocated sub-ACL. It should be noted that the current assessment will only include catch data through December 31, 2013, thus there is insufficient information for this assessment to make inferences about possible changes in selectivity brought about by the recent changes in minimum size regulations.

Biology

Distribution and stock structure

Haddock (*Melanogrammus aeglefinus*) is a demersal gadoid species whose range in United States (US) waters extends from the mid-Atlantic Bight north to the Canadian border (Collette et al. 2002). Globally, haddock occur on both sides of the North Atlantic Ocean, extending southward in the eastern Atlantic to the Bay of Biscay. Within the United States Exclusive Economic Zone (EEZ) there are two recognized stocks of haddock: Gulf of Maine and Georges Bank (Fig. A.1). The existing Gulf of Maine stock complex extends from the northern tip of Cape Cod east to the US/Canadian border and north to the coast of Maine (Fig. A.2, Cargnelli et al. 1999). Several meta-analyses of the life history parameters of haddock in the region have been conducted over the last four decades that generally support the current stock boundaries (Begg 1998, Beg et al. 1999). These investigations have highlighted differences in both the growth and maturation rates between the Gulf of Maine and Georges Bank stocks (Begg 1998, Begg et al. 1999). There are discreet spawning regions within the Gulf of Maine (Ames 1998) which may constitute localized metapopulations. The Gulf of Maine haddock stock may be composed of a seasonal migratory stock and non-migratory stock extending southward into the Nantucket Shoals region (Begg 1998).

Within the Gulf of Maine, haddock tend to move inshore in spring to spawn before returning to the deeper offshore waters in late summer. Peak spawning occurs during March and April and likely fluctuates inter-annually in response to water temperatures (Cargnelli et al.1999). Given that haddock are seldom found below 180 m, the various channels and basins within the Gulf of Maine likely serve as barriers to juvenile and adult dispersal (Begg 1998, Cargnelli et al. 1999). Many of the identified spawning areas of haddock are associated with gravel or sandy substrate (Colton 1972, Ames 1998, Cargnelli et al. 1999). Compared to the Georges Bank region, there are limited areas of suitable habitat in the Gulf of Maine (Clark et al. 1982, NEFSC 2012). This likely explains, in part, the large disparities in stock size between the Georges Bank and Gulf of Maine regions.

Recent reviews of historical and contemporary tagging studies (Begg 1998, NEFMC GPDT 2013) suggest that there is movement of fish between the Gulf of Maine and Georges Bank stocks, though there is considerable uncertainty regarding the degree of mixing. One recent study provided a crude approximation of 10% (Brodziak et al. 2008a), though subsequent work has shown that mixing rates of that magnitude are unlikely given the maintenance of strong cohort signals within the Gulf of Maine stock, the large disparities in stock sizes and the asynchronous recruitment between the two stocks (NEFMC GPDT 2013). While Brodziak et al. 2008b concluded that recruitment between the two stocks was synchronous based on an examination of NEFSC age-0 bottom trawl survey indices, this analysis was recently revisited (NEFMC GPDT 2013). The updated analysis concluded that the relationship reported in Brodziak et al. 2008b, while significant, was in fact weak, accounting for only 28% of the total recruitment variance. Using a longer time series of survey indices, the updated analysis concluded that there was no apparent synchrony in age-0 survey indices. Distribution of eggs has suggested that the Gulf of Maine and Georges Bank regions constitute distinct groups of haddock (Begg 1998). Survey distributions indicate spatial segregation between the areas of concentration within the two regions (Fig. A.2). The topic of exchange between the Gulf of Maine and Georges Bank regions will be explored in more depth under Terms of Reference 3.

Length-weight relationship

Beginning in 1992, the NEFSC bottom trawl surveys began using digital scales to record individual fish weights. Using these data, the benchmark GARM III assessment (NEFSC 2008) developed seasonal survey-based length-weight (LW) equations as the basis for converting catch weights to numbers-at-age (Equations 2-4). Updated survey-based length weight equations using data through 2013 were compared to the existing length weight equation. Both seasonal (spring/fall) and annual updates were evaluated and showed little difference from those established during GARM III (Fig. A.3). The use of a time-invariant LW equation is only appropriate if the LW relationship has remained stable over time. An examination of the time series of relative condition factor (Froese 2006) by season shows little evidence of pronounced temporal trends (Fig. A.4). Given the results of this comparison, the SAW/SARC 59 assessment will apply the same LW relationships established during GARM III.

(Equation 2) $W_{live\ (kg)} = 0.000007690 \cdot L_{fork\ (cm)}^{3.0622}$ *(spring)*
(Equation 3) $W_{live\ (kg)} = 0.000009870 \cdot L_{fork\ (cm)}^{3.0090}$ *(fall)*
(Equation 4) $W_{live\ (kg)} = 0.000009298 \cdot L_{fork\ (cm)}^{3.0205}$ *(annual)*

There are divergent opinions as to whether it is more appropriate to use a landings-based length-weight equation versus a survey-based length-weight equation to convert catch weights to numbers-at-age. Advocates for a landings-based derivation argue that since the fishery may catch larger (heavier) fish at length, there is the possibility that a survey-based length weight equation may be biased low, particularly at greater lengths. A survey-based approach may be preferred when a large portion of the catch is composed of discards (or some other fraction not sampled such as recreational landings) or when the catch weights-at-age are also used to estimate stock weights due to sparse sampling of older ages in the surveys (missing or highly variable estimates of weights-at-age). In the case of Gulf of Maine haddock, the arguments for a survey-based LW relationship are valid (large fraction of catches not from commercial landings and use of catch weights to estimate stock weights). Currently in the Northeast Region, fishery surveys are the only source of individual length-weight sampling.

The suitability of applying a survey-based LW equation to commercial landings was evaluated by applying the seasonal LW relationships in equations 2 and 3 to the observed length frequency distributions of commercial biological samples collected between 1977 and 2013. The estimated weights were then compared to the recorded sample weight and the distributions of differences were examined for the presence of bias. Examinations across years showed no evidence of strong temporal trends and across all market categories the interquartile ranges of the differences overlapped the equality line in the majority of years for both the ‘scrod’ and ‘large’ landings market categories (Fig. A.5). There was some indication that the estimated weights were greater than the recorded weights for the ‘large’ market category which could suggest that the survey LW relationships estimate heavier fish at length relative to the true relationship within the commercial landings. Interestingly, using the arguments made against the use of survey-based LW presented above, this is opposite of the expectation.

Since haddock are typically landed in gutted form, a more likely explanation for the discrepancies noted in the ‘large’ market category is that the current conversion factor for converting gutted haddock to its live weight equivalent is incorrectly specified. A small increase in the established conversion factor of 1.14 would be sufficient to lower the ratios such that the means were more closely aligned with the equality line. There has been an ongoing data collection effort by the NEFSC’s Cooperative Research Program to collect information needed to support a re-evaluation of the established conversion factors; however, this work is still in progress and preliminary results are not available.

Growth and maturity

Haddock in the Gulf of Maine reach a maximum size around 75 cm (≈ 5 kg). Comparison of Gulf of Maine and Georges Bank growth curves estimated from time series averages of NEFSC survey data show similar growth patterns (Fig. A.6). This runs contrary to the conclusions of a previous study (Begg et al. 1999), though the growth curves estimated in the Begg et al. (1999) study were highly variable over time and between regions. Gulf of Maine haddock von Bertalanffy growth parameters were reestimated using NEFSC survey data from 1970 to 2013 (Equations 5-6). A summary of the number of ages included in the analysis are presented in Table A.5.

(Equation 5) $L(t)=62.5\cdot(1-e^{-0.41(t-0.05)})$ *(spring)*

(Equation 6) $L(t)=65.8\cdot(1-e^{-0.38(t+0.62)})$ *(fall)*

Density-dependent growth has been observed within the Georges Bank haddock stock, with large cohorts experiencing slower growth (NEFSC 2012). Cohort specific growth was evaluated for four cohorts of Gulf of Maine haddock and compared to the 1997-2013 mean length-at-age (Fig. A.8). These comparisons do show that on average the mean length-at-age of known large cohorts (e.g., 1998 and 2003) tend fall below the 1997-2013 times series mean; however, the differences are not as large as has been observed in the Georges Bank stock. Given the differences observed for the Gulf of Maine stock, it does not appear that large cohorts require special consideration of density dependent growth when conducting stock projections.

Examination of monthly trends in the mean length of Gulf of Maine haddock landed in the commercial fishery suggests that the majority of somatic growth occurs between April and July, with little growth occurring January through March (Fig. A.9). Examination of mean survey lengths-at-age suggests that fish size-at-age has oscillated about the long-term mean with some indication of decreased growth over the past decade (Fig. A.10). There has been considerable variability in the sampling of Gulf of Maine haddock lengths and ages within the survey due in part to variable survey catches and sampling protocols (Fig. A.11). Some of the interannual variability observed in survey mean lengths-at-age is likely driven by sampling variability.

A logistic regression method (O'Brien et al. 1993) was used to fit maturity-at-age from the NEFSC spring survey data from 1977 to 2013. The number of maturity samples taken per year ranges from 1 to 364 (Table A.6). The trends in annual age-at-50% maturity ($A_{50\%}$; Fig. A.12), is not suggestive of any persistent temporal trends with only small variations around the time series average. Given the absence of persistent trends, and the occasional periods of low sampling, a decision was made to use the of a time-invariant maturity ogive to characterize the maturity schedule of Gulf of Maine haddock. The time series $A_{50\%}$ for male haddock was 1.85 and 2.39 for females (Fig. A.13). The corresponding length-at-50% maturity ($L_{50\%}$) was 30.2 cm and 36.5 cm, respectively. The input to the stock assessment model is based on the female maturity ogive presented in Table A.7. The approach is identical to that used for the GARM III assessment, with the only changes resulting from incorporation of an additional three years of survey data.

The GARM III assessment and subsequent assessment had assumed a spawning time of April 1. This is consistent with the peak period of spawning as inferred from egg distributions in the Gulf of Maine (Cargnelli et al.1999). This assessment will maintain an assumption of April 1 as the peak period of haddock spawning in the Gulf of Maine.

Natural mortality

Previous assessments of Gulf of Maine haddock have assumed a constant, age-invariant rate of instantaneous natural mortality (M) of 0.2 (e.g., NEFSC 2012, NEFSC 2008). While the accuracy of this assumption has not been thoroughly evaluated, it is consistent with the maximum ages observed in both fishery and survey data. Hoenig (1983) demonstrated that total mortality (Z) can be estimated as a function of the maximum observed age (t_{max}) in a population (ibid; Equation 7). This approach was further refined by Hewitt and Hoenig (2005; Equation 8). The maximum age observed in the survey was an age-18 fish in 1976, though more recently, in 2011 an age-22 fish was encountered in the commercial fishery. Generally, the maximum observed age in both the surveys and fishery has been increasing over time (Fig. A.14) – fish from the large 1998 year class continue to consistently be encountered in both surveys and the commercial fishery. Using the Hewitt and Hoenig (2005) approach, Z could be estimated at 0.23 assuming the maximum survey age of 18 or 0.19 assuming the maximum commercial fishery age of 22. The continued existence of the 1998 year class and general increase in the observed maximum age, suggest that total mortality, and by extension, natural mortality is low. Additionally, there is no evidence that natural mortality has increased over time as has been hypothesized for other gadoid stocks in the Gulf of Maine (NEFSC 2013). This assessment will maintain the assumption of a time invariant M of 0.2.

(Equation 7) $\ln(Z) = a + b*\ln(t_{max})$

(Equation 8) $Z = 4.22/t_{max}$

Ageing precision

Precision age testing for haddock is conducted six times a year; once for each season of the bottom trawl survey (spring and fall), and once for each quarter of the commercial samples. The precision tests are for both Georges Bank and Gulf of Maine stocks combined. Each precision test consists of a subsample of approximately 100 fish, and measures the consistency of age determination by the age reader. Two accuracy tests for Georges Bank haddock are generally conducted each year using the reference collection of Georges Bank samples (one prior to, and one after the production ageing). Lastly, an annual exchange of Georges Bank age samples is conducted with the Department of Fisheries and Oceans Canada (DFO) staff to compare age assignments between the two age readers (3-4 separate precision tests each year representing a range of sample sources/seasons; n ≈ 50-100 within each test).

Precision testing in the past year has demonstrated high consistency, with agreement levels between 95.9 and 99.0% (CVs range from 0.08 to 0.49%) for each test. The average precision level (5 tests) was 98.4% agreement and CV of 0.25%. These results exceed NEFSC standards for acceptable ageing consistency (>80% agreement, <5% CV); bias is assumed to be minimal in cases where the agreement level exceeds 90%. For samples collected during 2011 to 2013, the precision levels for all tests (17 tests) had an average agreement of 96.3% and an average CV of 0.51%. The best results showed nearly complete agreement (99.0%, 0.08% CV); the worst results were 90.7% agreement and a CV of 1.40%. Since 2011, the average accuracy level has been 93.6% agreement and a CV of 1.99% (5 tests); the best results were 96.4% agreement (May 2012) and a CV of 1.35% (August 2011); the worst results were 91.1% agreement (July 2013) and 2.86% CV (May 2012).

Historically, haddock age reading has been of high quality. Since regular testing began in 2004, precision levels have averaged 94.5% agreement with a CV of 0.73% in 75 tests. Accuracy tests have averaged 86.4% agreement and a CV of 2.26% (34 tests) in the same time period. The 2014 NEFSC/DFO exchange of Georges Bank samples yielded high precision levels. For the three tests conducted so far, the average results were 92.6% agreement and an average CV of 0.92%. One more test is still planned, on the 2014 Canadian spring survey samples (collected in February). Since 2010, when the current Canadian age reader began working with haddock, average annual precision levels in the exchange have been 78.4% (3.34% CV) in 2010, followed by 88.9% (2.43% CV) in 2011, 88.6% (1.81% CV) in 2012, and 81.0% (2.50% CV) in 2013. While the 2014 results were the best in the series and the 2010 results were worst, there is no clear overall trend. Among the five years (2010-2014), the best exchange result was 98.1% (0.38% CV) for NEFSC fall survey samples in the 2014 exchange. The worst result was 62.7% agreement (4.13% CV) for Canadian commercial samples in the 2010 exchange. None of the exchange comparisons revealed any bias. Only four times has the agreement level fallen below

NEFSC ageing standards (80%) in this time period; the CV level has been above NEFSC standards (5%) throughout.

The 2013 age samples were dominated by the 2013, 2012, 2010, 2006, and 2003 year classes. However, these strong year-classes were unlikely to have biased the age reader toward these age groups. Firstly, the QA/QC testing described above has demonstrated that the ages are accurate (as compared with the reference collection) and consistent (both by the NEFSC age reader and in comparison with the Canadian age reader). In addition, all samples are viewed at least twice to confirm the ages. Finally, difficult samples and fish with an atypical age/length combination are more closely examined.

Full testing results and an explanation of the statistics used can be found at http://www.nefsc.noaa.gov/fbp/QA-QC/hd-results.html.

TOR A.1. Estimate catch from all sources including landings and discards. Include recreational discards, as appropriate. Describe the spatial and temporal distribution of landings, discards, and fishing effort. Characterize the uncertainty in these sources of data. Investigate the utility of commercial or recreational LPUE as a measure of relative abundance.

Overview

In the recent period (1977 to present) total catch has ranged from 187 metric tons (mt) to 7,656 mt (Table A.8, Fig. A.15). Over the last decade, catch has averaged around 1,000 mt. Commercial landings are the predominant source of fishery removals, averaging 80% of the total catch between 1977 and 2013 – though recently, recreational catch has become an increasingly important source of fishery removals. Historical landing records of Gulf of Maine haddock extend back to 1930, though tabled values extend only until 1956 (Clark et al. 1982). The haddock fishery is a relatively new fishery compared to other New England groundfish fisheries with little exploitation prior to the 1900s. Haddock make a poor salt product, as such, the advent of the haddock fishery did not begin until developments in cold storage and distribution could support the fresh (unsalted) fish markets. The levels of commercial landings observed since 1977 are within the range of historical landings. Landings of Gulf of Maine haddock are considerably lower than those from the much larger Georges Banks stock (Fig. A.16).

With the exception of a period from 1994 to 1997, commercial discards of haddock have been less than 50 mt. While direct estimates of commercial discards only extend to 1989, low minimum retention sizes (Table A.4) likely limited the discarding of haddock in the commercial fishery pre-1989. Contemporary estimates of recreational catch from the Marine Recreational Fisheries Statistics Survey (MRFSS) extend back to 1981; however, there were the occasional saltwater angling surveys conducted between 1960 and 1979 which suggest that recreational catch pre-1981 was in the range of 250-400 mt (summarized in Clark et al. 1982). It is unclear whether these estimates represent recreational landings or total catch, though put in the context of commercial landings at the time, represent a minor component of fishery removals, historically.

Commercial landings

In 1982, the United Nations Convention on the Law of the Sea (UNCLOS) defined a country's Exclusive Economic Zone (EEZ) as a zone extending up to 200 nautical miles from a nation's coast. The EEZ defines the region where each country has sovereign rights to marine resources including fisheries. The geographic proximity of the US and Canada in the Gulf of Maine and Georges Bank Regions results in an overlap of each nation's EEZ. Given the importance of these areas with respect to resource extraction (among other reasons), the US and Canada both submitted cases to the International Court of Justice at The Hague, Netherlands seeking clarification. The Court issued a final ruling on October 12, 1984 formally delineating the US and Canadian EEZ. Hereafter, this demarcation line informally became known as the "Hague Line".

Within the Gulf of Maine, the US EEZ splits statistical areas 464, 465 and 467 (Fig. A.17). Prior to Hague line implementation, landings of haddock in US ports from these statistical areas could have been either from the Gulf of Maine or Scotian Shelf (Canadian) stocks. Current in-season management of Gulf of Maine haddock includes catch from these areas against fishery ACLs. Previous assessments have not included these catches. While landings from these statistical areas have been low since 1985 (≈ 3% of total landings, Fig. A.18), these landings have been included in the current assessment to maintain consistency with the existing ACL monitoring programs. No attempt was made to adjust landings prior to 1985 which is consistent with the approach used for Gulf of Maine cod (NEFSC 2013).

Since 1964, when modern catch statistics began, United States (US) domestic commercial landings of Gulf of Maine haddock have ranged from 122 mt to 5,593 mt (Tables A.8 and A.9). Beginning in the mid-1950s and extending until 1986, small amounts of haddock landings were reported by foreign vessels fishing in the Gulf of Maine. Foreign landings averaged less than 10% of the total stock landings during this time period and were dominated by Canadian landings (Clark et al. 1982).

Total US species landings are derived from the weighout reports of commercial seafood dealers and these data are generally considered a census of total landings. While un-reported landings are possible, no estimates exist to evaluate their magnitude. A secondary data source is required to apportion dealer landings to statistical area (stock) and assign basic information on fishing effort (e.g., gear, mesh, tow duration). Prior to 1994, the partitioning of stocks from total haddock landings was accomplished, in part, through a port-interview process conducted by port agents working for the National Marine Fisheries Service (NMFS). When trips were not interviewed, NMFS port agents would attribute area and fishing effort characteristics to the landings using personal knowledge of the fishery and/or information obtained during the interview process about vessels operating in the vicinity of the interviewed captain.

With the implementation of mandatory vessel trip reports (VTRs) in 1994, the port interview process ceased and the area and effort information was obtained directly from the VTRs. Unfortunately, the matching of dealer reports and VTRs has been problematic and secondary allocation procedures are needed to assign the area and effort information to dealer landings. Currently, a standardized procedure is used to assign area and effort from VTRs to dealer-reported landings from 1994 onward (Wigley et al. 2008). The product from this process is stored the NEFSC allocation (AA) database tables. Landings are matched to VTRs in a hierarchal manner, with landings matched at the top tier (level A, direct matching) having a higher confidence in the area and fishing effort attribution than those matched at the lower tiers. The matching rates have improved over time with over 80% of Gulf of Maine haddock landings being matched directly to VTRs since 2010 (Fig. A.19). While there is considerable variability in the matching success throughout the year (Fig. A.20), there are no clear seasonal trends as have been observed with other Gulf of Maine groundfish stocks (e.g., cod, NEFSC 2013). The overall precision associated with the allocation process, in terms of a CV ranges from 0.01 to 0.04 (Table A.10).

An additional area of uncertainty with stock landings stems from the misreporting and/or under reporting of statistical areas on VTRs. Federal regulations require that a separate VTR logbook sheet be filled out for each statistical area or gear/mesh fished. Vessels fishing in multiple statistical areas frequently under-report the number of statistical areas fished (Palmer and Wigley 2007, 2009 and 2012). Based on comparisons of VTR reports with vessel monitoring system (VMS) data, the impacts of this misreporting on Gulf of Maine haddock landings estimates could be potentially large (>20% underestimation error in terms of landings weight; Palmer and Wigley 2012). In all but 2004, the VMS-based methods estimated higher stock-level landings compared to VTR-based methods. However, a cross validation of the VMS allocation method with observer data between 2004 and 2011 suggests that, for Gulf of Maine haddock, VTR reports achieve stock allocations closer to the observer data more often than VMS-based methods (five out of the eight years compared). Additionally, VTR landings were frequently higher than the observed landings which would seem to invalidate the VMS-based results. While misreporting of stock landings does occur, given these conflicting results, it's difficult to quantify the possible impacts of VTR mis-reporting on the estimation of Gulf of Maine haddock landings. The error rates indicated by the VMS-based methods should be considered an extreme upper bound on the magnitude of error in the landings estimates.

For some species, there may be a component of the catch that does not get reported by seafood dealers. In the case of Gulf of Maine haddock, fish retained by the crew for home consumption are the largest component of commercial landings that would not be reported by seafood dealers. Estimates of home consumption can be derived from VTRs, but these estimates are likely underestimates of total home consumption landings due to incomplete reporting. From 1994 to 2013, home consumption landings averaged 1.6 mt/year, or approximately 0.3% of the total reported dealer landings (Table A.11). Even if these represent underestimates, it is unlikely that home consumption landings represent a significant source of fishery removals. Because of the low magnitude, home consumption estimates are not included in estimates of commercial landings.

Over the past five years, landings of Gulf of Maine haddock have exhibited consistent seasonal trends with peak landings occurring during the month of March (Fig. A.21). The sole exception to these patterns occurred in 2012 when there were large landings in both February and March. The commercial fishery is primarily conducted by vessels fishing trawl, gillnet and benthic longline (Fig. A.22). Gillnet gear contributed a larger fraction of the total landings early in the time series, but in the more recent period, constitutes less than 10% of the landings. The landings contribution of the benthic longline fleet has increased over time and currently ranks second in terms of landings. The primary gear for the exploitation of Gulf of Maine haddock has been, and remains, otter trawl. Over the 1977 to 2013 time period, otter trawl has averaged 77% of the total landings. During one of the GARM III working group meetings a fishing industry member reported that the trawl fishery had shifted to square-rigged mesh in the Gulf of Maine in order better target flatfish and that this shift had lead to a decline in the haddock selectivity of Gulf of Maine trawl fleet (e.g., Robertson and Stewart 1988). There is some evidence of this in the observer data where mesh type has been recorded since 1994. Diamond mesh was the predominant mesh type from 1995 to 1997, but there was a shift towards square mesh in beginning in 1998 (Table A.12), though since 2007, diamond mesh has again been the predominant mesh type in the Gulf of Maine trawl fishery. The mesh size requirements for square and diamond mesh have not always been identical (Table A.3), though a comparison of the distribution of observed mesh sizes does not indicate large discrepancies (Fig. A.24).

The ports of Gloucester, Portland and Boston have historically been the primary offload ports of Gulf of Maine haddock (Fig. A.25). Portland landings declined sharply in the early 2000s and Gloucester now accounts for over 50-70% of total commercial landings. Unlike other Gulf of Maine groundfish stocks like cod, there are no seasonal trends in the port-level landings (Fig. A.26). Cod landings are sensitive to rolling area closures that cycle clockwise around the western Gulf of Maine from May to June. Haddock landings appear to be less impacted by the seasonal closures.

From the 1980s to 2000 haddock landings came primarily from statistical area 515 in the central Gulf of Maine. However, over the last twenty years, landings have become increasingly concentrated in statistical area 514 in the western Gulf of Maine (Fig. A.27). The shift to statistical area 514 is consistent with an overall concentration of the groundfish fishery in the western Gulf of Maine that has been previously documented (NEFSC 2013). Similar to the seasonal port trends, there are no evident seasonal patterns in the statistical area landings (Fig. A.28).

Using the positional information provided on VTRs (Fig. A.29), annual Lorenz curves were estimated for both the commercial trawl and gillnet fishery based on the cumulative catch by ten minute square (following methods outlined in Wigley 1996). From the Lorenz curve an annual Gini index, or concentration index, can be estimated using Equation (9):

(Equation 9) $G = A/(A+B)$

where G is the Gini index, A is the area between 1:1 equality line and B is the area under the Lorenz curve.

Annual Gini indices were developed for both the commercial trawl, gillnet and benthic longline fleet based on the cumulative catch by ten minute square. Both gillnet and longline Gini indices show that these fleet have always been highly concentrated, though the level of concentration has increased between 1994 and 2013 (Fig. A.30); comparatively, the trawl Gini index has increased considerably over the time series from a concentration index of below 0.8 to current levels near 0.95. The concentration in the commercial trawl and gillnet fleet is characterized by a directional shift in the catch-weighted center (centroid) of fishing activity to the southwest (Fig. A.31). The longline fleet has only undergone small-scale changes in its distribution and there are no clear directional shifts in landings centroids. The current center of fishing activity is located in the western Gulf of Maine in the vicinity of 42.6° N x 70.0° W. The concentration of the haddock landings is also evident when comparing the haddock landings by ten minute square in 2013 to the aggregate VTR time series (Fig. A.32).

Landings of Gulf of Maine haddock were dominated by ton class 3 (51-150 tons) and 4 (151-500 tons) vessels until the early 2000s when landings by ton class 2 (5-50 tons) vessels increased sharply (Fig. A.33). It’s not clear exactly why the haddock landings increased for the ton class 2 vessels. Similar increases have been observed in the landings of other groundfish species such as cod, were trip limits had a greater impact on larger vessels (NEFSC 2013). In the case of haddock, while trip limits existed during this period (Table A.3), they were not as limiting as those of other stocks. It could be that since Gulf of Maine haddock has typically not been a target species, the haddock landings patterns are driven by factors un-related to haddock. Similar to port and statistical area, there are no clear seasonal patterns in ton class landings (Fig. A.34).

Commercial landings of Gulf of Maine haddock are classified by four primary market categories: snapper, scrod, large and unclassified (Fig. A.38). There are also medium and extra-large market categories that exist, but these are seldom used by the primary seafood buyers. The snapper market category constitutes fish in the 30-50 cm range and these are generally smaller than the scrod cull (Fig. A.36). The snapper market category largely disappeared in the late 1980s as the minimum retention size increased (Table A.13). However, in July 2013, the minimum retention size in the commercial fishery was reduced to 16 inches and the snapper market category has reemerged. Despite the differences in length frequency distribution, the sparseness of the biological sampling of this market category precludes being able to reliably characterize snapper landings. For this reason, the snapper market category has been combined with the scrod market category. Extra-large has been combined with the large market category and because of the absence of any biological sampling of the medium market category it has been treated as unclassified. Landings of both extra-large and medium market categories are minimal. There has been a general trend over time for landings to shift toward the scrod market category, particularly since 2000 (Fig. A.37). This shift is consistent with the decreases observed in the mean length-at-age noted previously (Fig. A.10). There was a tendency for scrod landings to peak during April and May, though the seasonal patterns changed considerably after July 2013 (Fig. A.38). It's unclear whether the change is due solely to the reductions in minimum size or whether this is reflective of a year class moving into the fishery.

Commercial landings: biosampling

Biological sampling (length and age) of Gulf of Maine haddock prior to 1977 was poor (Table A.14). Sampling intensities less than 200 mt per 100 lengths has traditionally been considered an unofficial NAFO/ICNAF standard (>200 mt/100 lengths). Since 1983 the sampling intensities dropped below that threshold and have remained there. Sampling intensities have been below 20 mt/100 lengths since 2003. Given that age sampling is conducted at the same time as length sampling (but lower density), it is not surprising that the sampling of age structures (otoliths) has followed similar trends as lengths. From 1982 onward the metric tons per 100 ages have been less than 1000 mt with sampling in the last five years less than 50 mt per 100 ages (Table A.15). While the overall sampling intensity for Gulf of Maine haddock has been good, there are a considerable number of calendar year quarters and market category cells with missing or limited biological samples (Table A.16).

For the GARM III assessment, commercial catch-at-age was estimated by aggregating lengths into 2 cm bins. For the AOP 2012 update, the additional years of catch-at-age (2007-2010, 2007 was reestimated due to changes in the landings data since GARM III) were estimated using 1 cm bins. For this assessment a complete update of the catch-at-age was conducted using 1 cm intervals for the entire time series. Catch-at-age was not stratified by gear type since the length frequency distributions of the landings are similar (Fig. A.39) and additional stratification would increase the level of imputation needed to construct the catch-at-age. This would have been particularly problematic given that sampling was non-existent or limited for sink gillnet and benthic longline in some years (Fig. A.40). Every attempt was made to maintain the market category/quarter stratification, consistent with the design of the biosampling program. However, when the availability of lengths for a particular market/quarter block was low, either a semiannual or annual time block was used. A criterion of 100 lengths per block was applied to the commercial landings for use as an objective basis to decide when it was necessary to bin across quarters. In situations where an annual time block was required, the annual LW relationship (Equation 4) was applied to convert landings in weight to landings-at-length in numbers. When sampling was maintained using quarterly or semi-annual time blocks the appropriate seasonal LW equation was applied (Equations 2 and 3). A summary of the amount of binning that was required is presented in Table A.16.

Total landings-at-age are presented in Table A.17. The bootstrapped generated CVs on the landings-at-age estimates are shown in Table A.18. CVs are generally less than 30% for those ages that make up the majority of the landings (Ages 4-8). Prior to 1984, the calculation of bootstrap CVs were not possible due to the inability to identify individual sampling events in the biosampling database. There is considerable uncertainty in the estimates of landings-at-age among some of the older ages, particularly beyond age-8 where the average CV begins to exceed 30%. Overall, younger ages have become less prevalent in the commercial landings with increases in the minimum retention size (Fig. A.41). There was a noted truncation of the age structure during the early- to mid-1990s, however since the late 1990s the age structure has expanded. The mean weights-at-age of the commercial landings have generally declined over time across all ages, though the declines are greater at older ages (Table A.19).

Commercial landings per unit effort (LPUE)

Commercial catch per unit effort (CPUE) indices have been applied in some groundfish stock assessments (e.g., Mayo et al 1994, NEFSC 2002a), however the practice has been largely discontinued due to major changes occurring in the Gulf of Maine groundfish fishery. The changes include measures to reduce fishing effort, closed areas, changes in mesh size and trip limits in addition to a switch in the fisheries-dependent data collection system from a landings interview/intercept program to a self reported logbook program (Table A.3). All of these issues affect the comparability of CPUEs estimated from post-1994 trends to those from the earlier time series and could cause a disconnect between CPUE and stock abundance. Additionally, these same issues would make standardization of a contemporary catch per unit effort (CPUE) index difficult. Similar issues with commercial catch rate indices have been previously noted (e.g. Harley et al. 2001, Maunder et al. 2006). Despite these concerns about the relationship of CPUE to stock abundance, it is informative to evaluate CPUE indices to gain an understanding of commercial catch patterns, even if these indices are not included in the assessment model.

The only accurate source of total fishery catch (retained and discarded) comes from the Northeast Fisheries Observer Program which began in 1989. This is a shorter time series than available for the dealer data; additionally, there are extended periods of low observer coverage in the twenty plus year time series. For this reason, landings per unit effort (LPUE) indices are likely to be a more informative source of commercial catch efficiency for Gulf of Maine groundfish. LPUE indices can be extended back to 1964 with the start of the modern commercial dealer data collection program.

An analytical dealer data set was created for the LPUE analysis. A description of the analytical set is included in Palmer (2012). Given that Gulf of Maine haddock landings are dominated by the trawl fleet, only data from commercial trawl trips were considered in this analysis. There is no way to accurately identify which trips in the dealer data constitute 'groundfish' trips with some probability of encountering haddock and which trips were engaged in other fisheries (e.g., fluke) with virtually no probability of encountering haddock. For this reason only trips that landed ≥ 1 lb haddock were included in the model. Nominal Gulf of Maine haddock commercial trawl LPUE (landings per days fished) shows very little trend since the mid-1980s after declining from a peak in 1980 (Fig. A.42).

Standardized LPUE indices were developed using a GLM model. The model included the following factors: year, area, ton class, quarter and depth zone. Factor levels were screened prior to inclusion in the model to evaluate those factors most appropriate for use based on their contribution to the overall haddock landings. The following factor levels were included:

- Area: 511, 512, 513, 514, 515
- Ton class: 23, 24, 25, 31, 32, 33, 41
- Quarter: 1-4
- Depth zone: 1-4

Only main effects were considered in the model to avoid confounding the interpretation of year effects. The year effects presumably provide information on changes in haddock abundance over time, but also likely absorb other factors not included in the model such as changes in technology, management measures and targeting behavior.

LPUE was log transformed (Fig. A.43) such that the linear LPUE model was:

(Equation 10) $$ln(U_i) = \alpha^{Y}_{y_i} + \alpha^{A}_{a_i} + \alpha^{T}_{t_i} + \alpha^{Q}_{q_i} + \alpha^{D}_{d_i} + \varepsilon_{y_i a_i t_i q_i d_i}$$

where U_i is landings per unit effort (days fished) and α^{Y}_{yi} is the coefficient for the year y_i, and similarly the coefficients for area (*A*), ton class (*T*), quarter (*Q*) and depth zone (*D*). Errors were assumed to have a log normal distribution.

The GLM model was run on the years 1977 to 2012. At the time the GLM model was developed the final 2013 commercial dealer data were not available. Standard levels were chosen for each factor as follows: year = 1982, area = 513, ton class = 31, quarter = 1, and depth zone = 4. Model coefficients were retransformed to linear scale after bias correction following Granger and Newbold (1977). To understand the influence of factors on the final GLM model, the model was developed using a stepwise selection process (PROC GLMSELECT, SAS Institute Inc.). While there is an indication of interactions among the factors, interaction terms were not included in this model because of the confounding effects on the interpretation of the year effects. Factors were added based on the adjusted AIC criterion (AIC_C). Factors entered the model in the following order: ton class, area, depth zone and quarter. Plots of the year coefficients (LPUE index) as factors are added to model show only marginal changes in the nominal LPUE index (Figure A.44). The final LPUE index is provided in Table A.20.

A comparison of the standardized LPUE index to the spawning stock biomass from the 2012 AOP update shows close agreement of the two series until 1994 (Fig. A.45). There were several moderate-to-strong recruitment events between 1993 and 1998 leading to a large increase in spawning biomass between 1994 and 2002 (NEFSC 2012). The LPUE index, while it increased marginally between 1994 and 2009, did not increase consistent with rate of increase in stock size. There was an apparent shift in relationship between LPUE and stock abundance/biomass in the mid-1990s such that after the mid-1990s, LPUE is not informative as an index of stock abundance (Fig. A.46). Based on these results, the commercial LPUE index will not be used in the Gulf of Maine haddock assessment model. This recommendation is consistent with the recommendations of recent SARCs (NEFSC 2013).

Commercial discards

Gulf of Maine Atlantic haddock are primarily discarded in the commercial fishery for three reasons: (1) fish are below the minimum retention size (too small), (2) fish are of poor quality, and (3) retention is prohibited (e.g., non-groundfish fisheries; Table A.21).

Direct sampling of the commercial fishery for discards has been conducted by fisheries observers since 1989. Beginning in May 2010, Amendment 16 created a new class of fisheries observers to support sector management of the northeast US groundfish fishery. These new observers were termed 'at-sea monitors', or ASMs. ASMs are deployed in the same manner as observers certified through the Northeast Fisheries Observer Program (NEFOP; Palmer et al. 2013), but they collect only basic information on fishery catches and length frequency distributions. Between 2010 and 2012, ASM coverage averaged approximately 20% of total groundfish trips whereas regular observer coverage (NEFOP) averaged about 6% (Palmer et al. 2013). A comparison of the estimated discard rates between ASM and NEFOP observers showed no statistical difference for the majority of gears and quarters examined (Wigley et al. 2012). The Gulf of Maine haddock ASM discard rates were statistically indistinguishable from the NEFOP discard rates as evidenced by the fact that the 95% confidence intervals of the difference between estimates include zero (Figs. A.47 – A.49). A comparison of the length frequency distributions showed only small differences (Fig. A.50) when the sampling was sufficient to make comparisons (Table A.22). Given these results, no distinction has been made between data collected by ASM and NEFOP observers with respect to discard estimation.

Beginning with the GARM III assessment, discards were estimated for five commercial gear types: large mesh (≥ 5.5") otter trawl, small mesh (<5.5") otter trawl, sink gillnet, benthic longline and midwater trawl. For this benchmark assessment we have reevaluated the gears for which discards will be estimated and included in the stock assessment model. Using data from 1989 to 2012 (2013 data were not available at the time of the analysis) we conducted a preliminary evaluation of discard estimates and available observer data for six commercial gears: benthic longline, large mesh otter trawl, small mesh otter trawl, shrimp trawl, large mesh (5.5" –

7.0”) sink gillnet and extra-large mesh (>8”) sink gillnet. It should be noted that the large mesh otter trawl gear includes standard otter trawl, Ruhle trawl and haddock separator trawl gears. Previous examinations of VTR data and observer data have shown that there are more trips observed that use these modified gear types than report the gear types on the VTR. This indicates that these gear types are not being accurately reported in the VTR data and no distinction can be made between the modified gear types and the standard otter trawl (NEFSC 2013). However, given that the use of these gear types did not begin until 2009 and the frequency of use is low in the Gulf of Maine, this should have negligible impacts on discard estimates. The mid-water trawl fleet was not included in this preliminary analysis; it was evaluated in a separate analysis which is described later in this section.

The preliminary estimates of discards by fleet showed that three gear types were responsible for the majority of Gulf of Maine haddock discards. Benthic longline, large mesh otter trawl and large mesh sink gillnet, are responsible for, on average, 89% of the Gulf of Maine haddock discards (Table A.23). While the discard estimates from the other gears exceeded 5% contribution to the annual total in some years, the CVs for these years were often large (Table A.24). The three primary gear types all had average CVs across the time series less than 0.5, while average CV of the three minor gear types all exceeded 0.5. The availability of length samples was also considered when deciding which gears to include in the final commercial discard estimates. The available length information for the three minor gear types was sparse and would have required extensive imputation to achieve estimates of discards-at-age (Table A.25). The three major gear types also have some years with limited length observations; the methods used to deal with these issues will be described later in this section. Considering the contribution to total discards, precision of the discard estimates and availability of length samples, this updated assessment will only include discard estimates from the benthic longline, large mesh otter trawl and large mesh sink gillnet gear.

In previous Gulf of Maine haddock assessments, discards had been estimated for the mid-water trawl fleet using the same Standardized Bycatch Reporting Methodology (SBRM) process used for other fleets (Wigley et al. 2007). This method uses observer data to estimate a ratio of discard of species of interest to the kept of all species ($d_{species}/k_{all}$) and then expands the estimated ratios by multiplying by the fleet-wide estimate of the retained catch of all species (K_{all}) that is obtained from dealer data. There are several problems with applying this approach to the mid-water trawl fleet. Since 2006, with the passage of FW 43 to the Multispecies FMP, category 1 herring vessels have been prohibited from discarding haddock. All haddock must be brought to shore, though the regulations prohibit the sale of these haddock for food. The prohibition on the food sale of these landings makes it unlikely that these landings would be reported in the dealer data as commercial landings. Therefore, to accurately account for the haddock removals from the mid-water trawl fleet, a bycatch (retained plus discarded catch) estimate should be used rather than a discard estimate. Secondly, the identification of stock area fished is problematic for paired midwater trawl trips owing to a complication within the analytical dealer database (AA tables). To accurately estimate haddock bycatch by mid-water trawl gear requires the use of VTR data to obtain K_{all} estimates. A separate analysis described in Palmer et al. (2014b) provides a description of the methods and summary of haddock bycatch by the mid-water trawl fleet between 1994 and 2012. Bycatch of haddock is primarily occurring in the Georges Bank region where annual bycatch estimates range from 0 – 281 mt; estimates for Gulf of Maine were less than, or equal to 4 mt annually. Over all years where haddock bycatch could be estimated, bycatch amount in the mid-water trawl fishery represented approximately 3.0% of the Georges Bank commercial landings and less than 0.1% of Gulf of Maine commercial landings. Bycatch amounts were estimated with moderate to poor precision with CVs ranging from 0.09 to 1.07. Given the small amounts of haddock bycatch occurring in the Gulf of Maine and the poor precision of these estimates, these removals will not be included in this assessment.

The total number of observed trips by gear type are presented in Table A.26. Final estimates of discards ranged from 2 mt in 1990 to a high of 378 mt in 1997 (Table A.27). Since the removal of restrictive trip limits (late 90s), the discarding of small fish seems to peak approximately three years after the spawning of moderate-to-strong year classes. While there are exceptions, large-mesh otter trawl is the major source of haddock discards. The resulting CVs on the discard estimates are variable on a gear-specific basis. At the aggregate level, CVs of total discards average 35%, however since 2010, and the addition of ASM coverage, CVs have been below 20%.

As a means of evaluating the accuracy of the discard estimation procedure, a check was conducted to attempt to estimate total landings using the same methodology used to estimate discards. Instead of estimating a d_{had}/k_{all} ratio, a k_{had}/k_{all} ratio is estimated. When compared to the total Gulf of Maine haddock landings, the results show close agreement with respect to scale and trends, lending support not only to the accuracy of the discard estimation procedure, but also corroborating the commercial landings estimate (Fig. A.51).

Commercial discards: biosampling

Observers collect length and age information from the discarded fraction of the catch (as well as on the retained catch); however, only length samples are currently available. ALKs were created using both commercial landings and NEFSC survey ALK corresponding to the appropriate season (spring/fall). Length sampling extends back to 1989 and with variable coverage over the times series. While the sampling intensity has exceeded the 200 mt/100 lengths threshold since 1992, there are many years with limited length sampling (Table A.29) requiring supplementing of the discard length frequencies be with survey lengths. The length distributions by gear are shown in Figure A.52 on an aggregate basis and by year in Figure A.53. Unlike the commercial landings length frequency distributions, the discard length frequencies vary considerably by gear.

In order to supplement the observer length frequency distributions with lengths from the NEFSC bottom trawl survey, we needed to first estimate gear-specific selectivity ogives for the discard gears under consideration. The selectivity ogives enable gear-specific sub-sampling from the survey length distributions. Gear selectivites were estimated using observer and survey length frequency distributions from 2009-2013. Since the generated selectivity ogives would be applied to length distributions from the Albatross IV time series, the length-frequency distributions from 2009-2013 (Bigelow series) needed to be calibrated using length-specific calibration factors (described under TOR 2). While applying selectivity ogives generated from a 2009-2013 reference period to early years is potentially problematic due to changes in minimum retention sizes, mesh sizes and possession limits, the reference time period was the only period in which sampling densities of the discard lengths were sufficient to achieve estimates of gear selectivity.

Survey length frequencies were first truncated using the minimum retention size for the year in question (Fig. A.54). Because minimum retention sizes are specified based on total length (in) and survey lengths are recorded in fork length (cm), the retention sizes were converted to fork length (cm) equivalents using Equation 11 (Livingstone 1957). While Pol et al. 2011 developed a similar fork length-total length relationship (Equation 12), the results are nearly identical. The Livingstone 1957 relationship had been used in previous assessments, so it was retained for this benchmark assessment.

(Equation 11) $L_{fork} = 0.944\ L_{total} + 0.58$ *(Livingstone 1957)*
(Equation 12) $L_{fork} = 0.95\ L_{total} + 0.65$ *(Pol et al. 2011)*

Using Pope's (1966) 'alternate tow' approach, the ratios of observed proportion-at-length discarded from the fishery to the proportion-at-length present in the survey are generated. Equation 13 (Wileman et al. 1996) is then fit to the aggregate ratios (across all years) to generate selectivity ogives based on logistic regression (Fig. A.55). A comparison of the estimated length frequency distributions to the observed distributions are shown in Figures A.56 – A.58. The estimated length frequencies pick up the major modes which likely correspond to incoming year classes; however the modal peaks are not identical in all years. A noted feature of the observed discards is that there is often some fraction of discards that are above the minimum retention size, a key violation of this method. While this is problematic, examination of the length distributions over time shows that these occurrences to be infrequent (Fig. A.53). At this time, this method provides the best means with which to impute discard length frequencies. The total number of survey lengths that were 'borrowed' are summarized in Table A.30.

(Equation 13)
$$r(l) = \left[\frac{\exp{(a+bl)}}{1+\exp{(a+bl)}}\right]$$

where:

r(l)i is the estimated selectivity at length, *l*
a and *b* are logit linear parameters

When estimating discards at length, attempts were made to maintain the separate semi-annual estimates so that the most appropriate seasonal LW equation could be applied. For some years and gear types this was not possible owing to limited sampling, as evidenced by the need to impute using survey lengths as described above. In these situations an annual time block was used to estimate discards-at-length and an annual LW equation was applied. A criterion of 30 lengths per block was used to provide an objective basis to decide when it was appropriate to bin across semesters. A summary of the time blocks applied in the estimation of discards-at-length is provided in Table A.29.

Commercial discard hindcasting: pre-1989

Direct observations of discards by fishery observers only exist from 1989 to present. The model formulations used in past assessments have started in 1977 owing to the availability of information on the age composition of commercial landings. For the GARM III, and the subsequent 2012 update, commercial discards were estimated in the pre-1989 period using a survey-scaling method (described in Palmer et al. (2008)). One important shortcoming of this approach is that it assumes that all fish from NEFSC surveys below the minimum retention size are selected by the fishery. As shown above, this is not a valid assumption. For this benchmark assessment we have instead applied a survey-filter method (also described in Palmer et al. (2008)). An additional benefit of this method is that it maintains consistency with the method used to impute length frequency distributions in the post-1989 period.

Attempts were made to hindcast discards for only large mesh otter trawl and large mesh gillnet. Because there was a limited longline fishery in the Gulf of Maine from 1982 to 1989 (Fig. A.22), no attempt was made to develop discard hindcast estimates for this gear type. The hindcast estimates could only be extended to 1982 because mesh size cannot be reliably identified in the dealer data prior to 1982. Given that the minimum mesh size was 5.125" from 1977 to 1982, the large mesh trawl gear as defined in the more recent period did not exist.

The survey-filter method requires information on survey numbers at length (N_i), estimates of gear selectivity at length (m_i), a scaling factor (q) and an estimate of total fishery effort (f). Assuming these are available, discard-at-length can be estimated using the following equations:

If:

(Equation 14.a) $C_i/f = q \bullet (N_i \bullet m_i)$, then
(Equation 14.b) $C_i = (q \bullet f) \bullet (N_i \bullet m_i)$ as above.

If :

(Equation 14.c) $K_i = C_i \bullet s_i$, and
(Equation 14.d) $D_i = C_i \bullet (1 - s_i)$, then
(Equation 14.e) $D_i = (q \bullet f) \bullet (N_i \bullet m_i) \bullet (1 - s_i)$, and
(Equation 14.f) $D_i / f = q \bullet [N_i \bullet m_i \bullet (1 - s_i)]$

where:
C_i is the catch retained by a given commercial mesh at length i
f is some estimate of total fishing effort
q is the proportionality constant
N_i is the abundance of fish in the survey at length i
m_i is the proportion of the available population retained by a given mesh at length i
s_i is the proportion of the retained catch kept at length i
K_i is the kept portion of the catch at length i
D_i is the discarded portion of the catch at length i

If it is assumed that the fish discarded pre-1989 were all less than the minimum size, the above equation can be simplified by setting s_i to 0. As noted above, there may be situations where this assumption does not hold, though it is likely valid for the majority of years. The gear selectivity ogives (m_i) developed above were retained for hindcast estimation. While it was not ideal to use a reference period so far removed from the hindcast period, the limited availability of observer length data necessitated this. As when generating the selectivity ogive, length-based calibration factors were used to convert Bigelow catches at length to Albatross IV equivalents.

By regressing the ratio of observed discards-at-length to the total fishing effort (K_{all} was used similar to the contemporary discard estimates) on the ratio of selectivity-adjusted survey numbers-at-length, the gear-specific scaling factor (q) can be estimated as the slope of the regression line (Equation 14.f, Fig. A.59).

Total discards estimated using the survey-filter approach have similar trends and scales to the direct estimates and are of similar magnitude to those achieved using the survey-scaling method developed in GARM III (Fig. A.60). Since there were no survey length observations within the selectivity window for 1988, no hindcast discard estimate could be generated. Neither the survey-scaling or survey-filter methods suggest that commercial discards were large during this period. For perspective, both approaches estimate total discards at less than 25 mt compared to average commercial landings exceeding 4,000 mt/year for the 1977-1988 period. Regardless of the method used to hindcast discards for the period pre-1989, the impacts on assessment results will be negligible.

Commercial discards-at-age and weights-at-age are presented in Tables A.31 and A.32 respectively. Bubble plots of commercial discards-at-age over time are shown in Fig. A.61.

Discard mortality

The GARM III assessment and the 2012 update both assumed 100% mortality of haddock discarded in the commercial fishery. While considerable work has been done on the mortality of haddock escaping trawl gear (e.g., Ingolfsson et al. 2007) very little work has been done on mortality of haddock captured (brought on board the fishing vessel) and then released. The act of capture and resulting stress of exposure to air, barotraumas, thermal shock, etc. will increase mortality (Hislop and Hemmings 1971, Davis 2002). A recent review of Atlantic cod mortality captured in New England waters estimated discard mortality at 30% for recreational hook and line, 33% for commercial longline gear, 75% for commercial otter trawl and 80% for sink gillnet (NEFSC 2013). Past studies have shown haddock to be less resilient to capture compared to other closely related gadoid species such as cod (Ingolfsson et al. 2007). It is expected that haddock mortalities are higher than those estimated for cod, though absent targeted studies, it is difficult to determine specific mortality rates. There is at least one known study from the Barents Sea that estimates haddock mortality in a pelagic longline fishery from 39-53% (Huse and Soldal 2002). It is however, difficult to apply these results to the Gulf of Maine region, given the differences in geography (seasonal temperature differences) and the gear type (pelagic vs. benthic gear). Given the current discard mortality estimates used for Gulf of Maine cod, and the lower resiliency of haddock, this assessment has assumed a mortality of 100% for haddock discarded by the commercial fishery. For otter trawl and sink gillnet, this is likely close to the true mortality. It's possible that the longline mortality is lower than the 100% assumption and subsequent work should be conducted in the region to better elucidate the true discard mortality.

Recreational landings

There is a large recreational fishery in the Gulf of Maine that, over the last decade, has accounted for approximately 29-86% of the total catch (Table A.8). Previous assessments have used data collected under the Marine Recreational Fisheries Statistical Survey (MRFSS). MRFSS data begin in 1981; however, there were the occasional saltwater angling surveys conducted between 1960 and 1979 that suggest recreational catch pre-1981 was in the range of 250-400 mt (summarized in Clark et al. 1982). It is unclear whether these estimates represent recreational landings or total catch, though put in the context of commercial landings at the time, would represent a much lower fraction of fishery removals compared to the recent period.

Beginning with this current assessment, MRFSS data have been reestimated using revised methodologies consistent with the new Marine Recreational Information Program (MRIP) which has replaced the MRFSS program (NMFS 2012). Since the existing data were collected under the MRFSS program, this assessment report will refer to these as MRFSS data. Beginning in 2012, recreational catch statistics were collected using the MRIP sampling design. The conversion of MRFSS data to MRIP estimates is described below. In general, both MRFSS and MRIP survey methods consist of site interviews to gather catch, effort and biological data from recreational anglers. There are three primary sampling modes: party/charter, private/rental and from shore. Sampling is conducted throughout the year in two-month waves; however in New England waters, wave 1 (January/February) has not been historically sampled. Sampling is stratified by state, mode and wave with samples allocated based on recent estimates of fishing pressure. Survey sampling sites are randomly selected from pre-determined access site lists. The interview procedures vary slightly by mode, but in general anglers are interviewed at assigned access sites on completion of fishing trips and/or during the course of the fishing trip for some party/charter sampling events. Interviews include collection of information on catch composition, effort and length and weight measurements from a random sample of fish from each species. Intercept data are combined with telephone surveys to provide total estimates of catch and effort (ASMFC 1994).

The MRFSS data collection program began in 1979, though estimates of recreationally caught haddock are not available until 1981. Recreational catch data are divided into three components: directly observed landings (A), unobserved landings (B1), and unobserved discards (B2). Catch types A and B1 are collectively referred to as the recreational harvest and B2 catch as recreational releases. Recreational catch is partitioned into Gulf of Maine and Georges Bank stocks using the annual site register lists; catches attributed to intercept/interview sites in Maine and New Hampshire as well as Massachusetts catches from Essex, Suffolk, and Plymouth counties are allocated to the Gulf of Maine stock and catches from Nantucket, Dukes and Bristol counties are assigned to the Georges Bank stock. Catches from Barnstable and Plymouth counties (Massachusetts) are split such that intercept sites bordering Cape Cod Bay are allocated to the Gulf of Maine stock and those on the east and south side of Cape Cod are allocated to the Georges Bank stock (note that there are a few exceptions to this rule where boat access sites occur on the south and east side, but it is known that vessels are catching haddock in Gulf of Maine waters).

Conversion of MRFSS data using MRIP methodologies

In 2012 NMFS released revised MRIP-converted estimates of MRFSS recreational catch extending back to 2004. The revised estimates were based on the application of the MRIP sampling design to the existing MRFSS data. For Gulf of Maine haddock, the revised MRIP estimates ranged from 49-107% of the MRFSS landings estimates and 56-144% of the MRFSS discard estimates (Table A.33). A working group convened in March 2012 recommended applying a ratio estimator to MRFSS data collected pre-2004 to convert the pre-2004 MRFSS data into scales consistent with the revised MRIP estimates. The WG recommended that the ratio estimator be based on the "ratio of means" (summed across all comparison years) rather than based on the "mean of ratios" for individual years (NMFS 2012). Consistent with the recommendations of the WG, that approach has been employed in the current assessment yielding a ratio estimator of 0.83 for AB1 catch and 0.95 for B2 catch (Table A.33).

Total recreational catch has been reestimated since GARM III due to minor updates to the MRFSS data and to accommodate the MRIP re-estimation. Updated catch estimates are presented in Table A.34. The MRFSS data collection program is a numbers based survey and conversion of MRFSS estimates to removals in terms of total biomass can be accomplished in several ways. Consistent with the methodologies used for other groundfish stocks (e.g., cod, NEFSC 2013), catch biomass estimates were developed by using annual length frequency distributions to generate numbers at length and subsequent application of the annual LW equation (Equation 4) to estimate total removals in terms of weight. Since the majority of the recreational catch occurs during the summer months, application of seasonal LW equations from either the spring or fall surveys was not appropriate.

A summary of recreational catch from 1981 to 2013 is presented in Table A.34. Recreational harvests have ranged from 0 to 573 mt, with harvests averaging approximately 290 mt over the past five years. CVs on the harvest estimates have been highly variable, though since 2004 CVs have been below 20%. Releases have ranged from 0 to 414 mt, with the releases increasing sharply over the past five years from only 49 mt in 2009 to the time series high of 414 mt in 2013. The CVs on releases are slightly higher than those of the harvest precision levels; however, they have been below 30% since 2004 with the exception of 2008. Overall the general precision of the release estimates is similar to the commercial discards.

Evaluation of VTR recreational information

While MRFSS/MRIP is the source for official recreational catch estimates, VTRs provide a useful source for understanding some of the finer spatial and temporal trends that cannot be easily determined from the MRFSS/MRIP data. They also help inform the validity of the MRFSS/MRIP sampling scheme and treatment of data. VTR data are only available for the federally permitted party (head boats) and charter modes. Early in the time series party/charter vessels were the predominant source of recreational catch, though the catch by private vessels has increased since 1997 (Fig. A.62). VTRs are not required for the private recreational fleet or party/charter vessels operating only within state waters (state permitted vessels), thus VTR-based estimates will underestimate the total recreational landings (Fig. A.63). While VTR estimates only provide a sub-component of the total recreational catch, a comparison of VTR catch trends to MRFSS/MRIP provides a validation of the accuracy of the MRFSS/MRIP data (Fig. A.63). The MRFSS program did not historically sample the New England region in wave 1 (January/February); however, an evaluation of VTR data indicates that ≤ 1% of the annual recreational catch occurs during wave 1, with the majority of recreational catch occurring during waves 3 and 4 (May to August; Fig. A.64).

Using the positional information from VTRs (Fig. A.65) the fine scale spatial characteristics of the federally permitted component of the recreational fleet can be described. Unlike the commercial trawl fishery, the recreational fishery has always been highly concentrated, with Gini indices ranging from 0.94 to 0.98 (Fig. A.66). There have been no large scale shifts in the center of recreational effort over time (Fig. A.67). The majority of VTR-reported recreational landings come almost exclusively from the western Gulf of Maine in the vicinity of the Western Gulf of Maine Closed Area (Fig. A.68). Approximately 90% of the total recreational catch comes from federal waters (> 3 mi from shore; Fig. A.69).

Recreational landings-at-age

The numbers-based estimates of recreational landings were converted to numbers-at-age using the length frequency information collected from the MRFSS/MRIP surveys and ALKs borrowed from the NEFSC survey. The length sampling of the recreational harvest was poor prior to 2002 (Table A.35). Generally, recreational harvest included only fish above the minimum retention size (Fig. A.70 and A.71). To supplement the length frequency distributions of the recreational harvest in the years before 2002, lengths were borrowed from the NEFSC surveys, using only lengths above the minimum retention size. Minimum retention sizes were converted to fork-length equivalents using Equation 11 prior to sub-sampling the survey lengths.

Recreational harvest (landings)-at-age are presented in Table A.36 and Figure A.72. The patterns are similar to the commercial landings-at-age with a truncation of the age structure in early 1990s followed by a sharp expansion and evidence of strong cohort signals, particularly of the 1998 and 2003 year classes. Recreational landing weights-at-age are presented in Tables A.37. Similar to the commercial landings, there is a noticeable decline in fish weights over time, particularly since 2000.

Recreational landings per unit effort (LPUE)

Using methods identical to those used to develop a LPUE index for the commercial trawl fleet, a recreational LPUE index was developed using VTR data from 1994 to 2013. LPUE was expressed in terms of number of fish caught per angler hour (number of anglers × fishing time). The un-standardized LPUE shows increasing trends since 1994, with slight declines over the last four years (Fig. A.73). The GLM model included the following factors: year, trip category (party/charter), area, quarter and depth zone. Factor levels were screened prior to inclusion in the model to evaluate those factors most appropriate for use based on their contribution to the overall haddock landings. The following factor levels were included:

- Area: 511, 512, 513, 514, 515
- Trip category: 2, 3
- Quarter: 1-4
- Depth zone: 1-4

Similar to the commercial trawl LPUE, recreational LPUE estimates were log transformed to normalize the distribution (Fig. A.74). Standard levels were chosen for each factor as follows: year = 1994, area = 514, trip category = 2, quarter = 1, and depth zone = 4.

Factors entered the model in the following order: trip category, depth zone, quarter and area. Plots of the year coefficients (LPUE index) as factors are added to model show only marginal changes in the nominal LPUE index (Figure A.75), with the largest effect coming from the incorporation of trip category. The final LPUE index is provided in Table A.38.

A comparison of the standardized LPUE index to the spawning stock biomass from the 2012 assessment updates shows poor agreement between the two series (Fig. A.76). Spawning stock biomass peaked in 2002 followed by a sharp decline, however recreational LPUE continued to increase until 2006, followed by a general decline until the end of the time series, with the exception of 2009. A scatter plot comparison of the two time series highlights the lack of relationship (Fig. A.77). The recreational fishery was less affected by regulatory changes during the 1994 to 2013 period, so it does not appear that regulatory effects may be responsible for the lack of an apparent relationship between the LPUE and SSB time series. It is possible that SSB is not a good index of the fraction of the resource exploitable by the recreational fishery. For example, the SSB peaked in 2002 when the large 1998 year class would have been four years old (approximate age of 100% maturity) – if these fish were not fully selected by the recreational fishery until they were six to eight years old, then this may explain the delay in the response of the LPUE. Partial recruitment patterns from the 2012 assessment update would seem to support this hypothesis. While there is general concern about the use of LPUE indices as indices of abundance for reasons previously highlighted, there may be utility in exploring the utility of the recreational LPUE index through a sensitivity model.

Recreational discards-at-age

With increases in the minimum recreational retention sizes, the contribution of recreational discards to total recreational catch has been increasing over time (Table A.8, Fig. A.15). In the GARM III, and subsequent 2012 update assessment, recreational discards were reported, but they were not included in the catch-at-age used in the assessment models. The primary reason for the exclusion of discards was the limited length frequency information available on recreational discards at the time the GARM III assessment was conducted. At-sea sampling of the party charter vessels did not begin until 2004 (i9 sampling). Since 2004, sampling has been highly variable, ranging from 14 to 2,343 lengths per year with sampling intensities ranging from 17.7 to 265.3 mt/100 lengths sampled (Table A.35). Overall, the sampling intensity of the recreational releases is below the level of the recreational harvest.

Because of the increasing importance of recreational discards over time, this benchmark assessment has attempted a hindcast of recreational discards using the available length frequency information and a variant of the survey-filter method used to hindcast commercial discards. Unlike commercial discards, estimates on the magnitude of recreational discards in terms of total numbers are already available from the MRFSS/MRIP data (Table A.34). The survey-filter method was needed only to re-construct the length frequency distributions of recreational discards back in time. Similar to commercial discards, the assumption was made that all discarding was done due to minimum retention sizes. This assumption appears to be valid for the recreational fishery, with very little discarding of legal-sized fish occurring from 2004 to 2013 (Figs. A.78 and A.79). Using the alternate-tow approach used for commercial discards, a gear selectivity ogive was constructed using NEFSC survey catch-at-length below the minimum recreational retention size (Fig. A.80). Because the sampling intensities of the recreational releases were reasonably good during the 2004-2008 period, this period was used as the reference period to construct the selectivity ogive. Using this periods removes any confounding effects of the Bigelow survey calibration when applied to the historical time period. The resulting selectivity ogive is shown in Figure A.81. Comparing the survey-filter length frequency distributions to the observed length frequency distributions showed reasonably close agreement (Fig. A.82).

Application of the generated selectivity ogive to the survey length distributions provided recreational length frequency distributions back to 1981. Unfortunately, due to the sparseness of the survey length information between 1981 and 1994, the number of available lengths were insufficient to characterize the annual length distributions. The survey lengths were aggregated over this period. Because recreational release estimates over this period were low (0-2.6 mt), any errors in the estimation of recreational release catch-at-age should have minimal impacts on the overall catch-at-age.

To our knowledge there are no available scientific studies on the discard mortality of haddock released from the recreational fishery. Efforts were made to reach out people familiar with the recreational haddock fishery to gain a better understanding of likely release mortality. Based on these informal communications, an assumption of 50% mortality seemed reasonable. This is higher than the 30% assumed for Gulf of Maine cod, and consistent with the observations that haddock is a more fragile fish when handled and likely to incur higher release mortality. This assessment will use a baseline assumption of 50% mortality of haddock released in the recreational fishery. Given the large magnitude of recreational releases in the later part of the assessment time series, the discard mortality assumptions for the recreational fishery may be important. Sensitivities of the base assessment model to alternate assumptions will be explored (Appendix 2).

Recreational discards-at-age are presented in Figure A.83 and Tables A.39 (baseline 50% mortality assumption) and Table A.40 (alternate 100% mortality assumption). Several cohorts are evident in the discards-at-age including the 1998, 2003 and 2010 year classes. Recreational release weights-at-age are presented in Table A.41.

Total catch-at-age and mean weight-at-age

Estimates of total catch-at-age were determined by summing the numbers-at-age across all of the catch components: commercial landings, commercial discards, recreational landings and recreational discards. An age-9^+ group was used in the construction of the catch-at-age. This decision was made based on the sparseness of information on older age classes in the middle of the time series during the period of severe truncation of the population age structure and increased CVs on commercial landings-at-age above age-8. The truncation began in the mid-1980s and persisted until the late 1990s. Total catch-at-age is presented in Tables A.42 (baseline 50% recreational release mortality), A.43 (alternate 0% recreational release mortality) and A.44 (alternate 100% recreational release mortality). Bubble plots of the total catch-at-age assuming the baseline 50% mortality are presented in Figure A.84. The updated catch-at-age estimates for this assessment agree closely with those from the 2012 update (Fig. A.85), despite the re-estimation of commercial landings-at-age, revised discard estimation procedures, revised recreational catch estimates and inclusion of recreational discards.

Mean catch weights-at-age were estimated by using a numbers weighted average of the individual catch component's mean weights-at-age. Estimated catch-weights under the range of recreational discard mortality assumptions are presented in Tables A.45 to A.47. Minor imputation of the catch weights at-age was required to fill in gaps in the youngest and oldest ages; a 5-year centered moving average was used to impute missing cells for all but the age-0 weights-at-age, where a time series average was applied. There is evidence of declines in the mean weights-at-age for fish older than age-5 over the last decade (Fig. A.86).

Estimation of January 1/spawning stock weights

Sampling of older age fish in the trawl surveys has historically been low, and use of survey-based weights-at-age to estimate January 1 and spawning stock weights for use as model inputs would require extensive imputation.

January 1 and spawning stock weights were estimated from catch weights using a method described in Rivard (1980, 1982). April 1 is the assumed spawning event in the base model. Given that there is little somatic growth between January 1 and the peak spawning period (Fig.A.9), spawning stock weights were set equal to January 1 weights-at-age. The Rivard method adjusts the catch mean weights-at-age, which are generally presumed to represent mid-year weights, back to January 1. Mean weights at the beginning of the year for a given age class are calculated as the geometric mean of the weight in the same year and of the same cohort in the previous year. No adjustments are made for the plus group calculation. Calculations for the initial and final years and ages are described in Rivard (1980,1982). Since the stock weights should reflect all fish in the population, the catch weights-at-age based on 100% assumption of recreational discard mortality were used to estimate January 1 and spawning stock weights. January 1/spawning stock weights are shown in Table A.48.

Catch-curve analyses

Catch curves were constructed for the aggregate fishery catches (commercial and recreational landings and discards) based on the methods of Robson and Chapman (1961). Catch curves were conducted on a cohort basis rather than an annual basis which removed the confounding effects of differential year class strength on the interpretation of catch curve results. Linear regressions were fit to the log transformed catches of ages 6-11 for the 1978 to 2006 year classes (Fig. A.87). While ages 6-11 may not precisely match the fully recruited ages, it offers a compromise between full selection and having sufficient ages to fit a reliable regression. The slope of the regressions provides a model-independent estimate of cohort Z. The analyses suggest time series Z estimates around 1.5 early in the time series, dropping to around 0.4 by 1990 and remaining at low levels until the end of the time series (Fig. A.88).

Catch curves can also be useful for making general inferences on the selectivity of both fisheries and surveys. While selectivities can be estimated from the fitting of stock assessment models, it is useful to have model-independent estimates of selectivity that can be used to validate model-based estimates and/or provide some *apriori* understanding of selectivity. A method described in Restrepo et al. (2007) uses the residuals from the log-transformed linear catch curve analysis to infer relative selectivity-at-age. Selectivities are estimated using the ratio of observed to predicted catch proportions and then rescaling the residuals from each curve so that the maximum positive residual equals 1. The distribution of selectivities-at-age from all cohorts was examined to evaluate the time series distributions of catch selectivity at age. While this approach masks any changes that may be occurring in the selectivity across time, it is useful for gaining a general understanding of catch and survey selectivities and evaluating whether there is strong evidence for the presence of domed-selectivity (i.e., lower selectivity at older ages). Examination of the residual patterns from total catch shows full selectivity not occurring until age-6 or 7 and then remaining relatively flat well into the age-9^+ group (Fig. A.89)

TOR A.2. Present the survey data being used in the assessment (e.g., indices of relative or absolute abundance, recruitment, state surveys, age-length data, etc.). If available, consider whether tagging information could be used in estimation of stock size or exploitation rate. Characterize the uncertainty and any bias in these sources of data.

There are three primary fishery independent surveys that operate semiannually in the Gulf of Maine: the NEFSC bottom trawl survey, Massachusetts Department of Marine Fisheries (MADMF) bottom trawl survey and the Maine-New Hampshire (MENH) inshore groundfish survey. All three surveys operate in both the spring and fall with the seasonal timing differing slightly between surveys. The NEFSC survey occurs the earliest of the three spring surveys with MADMF and MENH having similar timing. The MADMF survey occurs first in the fall with the NEFSC and MENH survey having similar timing.

NEFSC bottom trawl survey

The NEFSC spring and fall bottom trawl surveys began in 1968 and 1963, respectively. Together these two surveys represent the longest regional time series of fishery independent information. All previous Gulf of Maine haddock assessments used only the offshore survey strata (Fig. A.90). Given that haddock tend to be distributed in offshore waters outside of 3 miles (as evidenced by the recreational catch distributions, Fig. A.69), the offshore survey strata likely capture the major haddock distributions in the Gulf of Maine. The NEFSC spring survey in the Gulf of Maine occurs from late April to early May, with some annual variability, but no long-term shifts in survey timing (Fig. A.91). The mean depth of the spring survey is approximately 160 m, with no apparent shifts in the depth distribution over the survey time series. The time series trends of bottom temperature show considerable variability, though no indication of long-term trends (warming or cooling). Since 2010, temperatures have been warmer than average, though episodic departures above the mean were also observed in the early 1970s. The fall survey reaches the Gulf of Maine in late October to early November. Since the sampling design is identical between the spring and fall surveys, the mean depths are similar. Like the spring survey the fall time series trends of bottom temperature show no indication of long-term trends, though temperatures have been above average since 2010.

A frequent criticism of the NEFSC bottom trawl survey is that it does not cover the same areas where the commercial and recreational fisheries catch haddock, and thus 'misses' much of the haddock that exist in the Gulf of Maine. A comparison of the NEFSC spring and fall survey catches to commercial (total observed haddock catches by ten minute square) and recreational activity (total number of recreationally caught haddock by ten minute square) show close agreement between the location of survey and fishery catches (Fig. A.92).

The NEFSC bottom trawl survey has utilized three different vessels and three different door configurations throughout the time series of the survey (Table A.49). To maintain a consistent survey time series, survey indices are converted to 'Albatross IV/Polyvalent door' equivalents using several different conversion factors (Table A.50). The largest change in the survey time series occurred in 2009 when the FSV Albatross IV was decommissioned and replaced by the FSV Henry B. Bigelow. This resulted in changes not only to the vessel and doors, but also to the overall trawl gear and survey protocols (summarized in Table A.51). Calibration experiments to estimate survey differences were conducted in the spring and fall of 2008 (Brown 2009). The results of those experiments were peer reviewed by a panel of external (non-NMFS) experts and summarized in Miller et al. (2010). These results provide annual calibration coefficients both in terms of abundance (numbers) and biomass (weight). Further work by Brooks et al. (2010) developed length-specific abundance calibration coefficients for haddock. This method uses a segmented regression model where a constant conversion factor is applied to fish ≤ 18 cm and ≥ 51 cm, and a constantly decreasing linear regression is fit to fish between 18 and 51 cm (Fig. A.93). A comparison of the converted and unconverted spring and fall survey indices is presented in Figure A.94. It should be noted that while considerable focus has been placed on the Albatross/Bigelow calibration, the effects of door calibration are generally larger than those of the Albatross/Bigelow calibration, in all but the 2013 indices. As will be described below, there were a substantial number of juvenile fish encountered in both the spring and fall 2013 surveys which resulted in large differences between the unconverted and Albatross-converted Bigelow indices.

To evaluate differences in the day/night catchability of haddock, an analysis was conducted to determine whether there were appreciable differences in survey catchability between daytime and nighttime tows. The results showed that generally, catchability was slightly higher in the daytime tows. However, the trends between day and night tows were similar, and in most years the day/night survey indices fell within the 80% confidence interval (CI) of the aggregate index (Fig. A.95). Splitting by day and night would result in reduced tows and lost strata (Table A.52), which would increase the likelihood that survey indices could be influenced by a single large tow in any year. Given the loss of information that would occur by using only day/night indices, and because of the similarity in the trends, it is appropriate to use both day and night tows to calculate indices for the assessment.

The time series of aggregate survey indices are presented in Table A.53 and the corresponding CVs are presented in Table A.54. Bigelow year indices (2009-2013) are presented using both the station-haul-gear (SHG) criteria that was used in the Albatross IV survey protocols to determine representative hauls as well as the revised tow-operations-gear-acquisition (TOGA) criteria that has been used under the Bigelow survey protocols. The primary difference between the SHG and TOGA criteria is that the TOGA criteria takes advantage of the extensive sensor information collected on the net performance (bottom contact, wing spread, door spread, head rope height, etc.) to determine when a survey tow should be considered 'representative' and included in survey indices. The differences in survey indices between the protocols is variable, though in general, they reflect similar trends. Unconverted Bigelow indices are presented in Table A.55. Note that the unconverted Bigelow indices are only presented using the TOGA tow criteria.

Indices-at-age for both the spring and fall surveys are presented in Tables A.56-67. The tables are as follows:

- Tables A.56-58: Spring abundance (numbers-at-age) using SHG criteria (A.56), TOGA criteria (A.57) and Bigelow-series only (A.58).
- Tables A.59-61: Spring biomass (weight-at-age) using the SHG criteria (A.59), TOGA criteria (A.60) and Bigelow-series only (A.61).
- Tables A.62-64: Fall abundance (numbers-at-age) using SHG criteria (A.62), TOGA criteria (A.63) and Bigelow-series only (A.64).
- Tables A.65-67: Fall biomass (weight-at-age) using SHG criteria (A.65), TOGA criteria (A.66) and Bigelow-series only (A.67).

Plots of the spring and fall survey indices show very strong signals in both the spring and fall surveys, though the fall survey trends tend to be less variable and catches are generally larger (Fig. A.96). The fall survey likely better captures trends in the Gulf of Maine haddock resource since haddock migrate inshore during the spring to spawn, thus the spring survey is likely more susceptible to the timing of spawning and availability within the survey area.

Plots of the numbers-at-age for both the spring and fall survey are shown in Figure. A.97. The plots show approximately four periods of recruitment pulses, corresponding to the peaks in the aggregate survey indices. There was a strong 1963 year class that tracks well in both the fall and spring surveys as well as several moderate recruitment events during the mid-1970s. The period from the mid-1980s to the mid-1990s was characterized by poor recruitment. A strong year class was spawned in 1998, followed by a moderate year class in 2003. The 2010 year class appears to be moderate-to-strong and there are signs of another strong year class in 2012. The fall 2013 age-

0 index is the second largest index on record in the calibrated time series and could be indicative of a strong 2013 year class. It is, however, premature to make any inferences about the strength of the 2013 year class until subsequent observations are collected from additional surveys. Cohorts track well within the spring (Fig. A.98) and fall (Fig. A.99) surveys. Consistent with the earlier statements regarding the overall ability of the surveys to captures trends in the resource, the fall survey tends to do a better job tracking cohorts with strong tracking out to age-8 and strong cohesion between even the age-1 and age-8 indices. The spring survey still does a reasonable job of tracking cohort strength, though the confidence ellipses tend to be larger and the relationship less well defined compared to the fall survey.

There is a general trend towards declining weights-at-age over the last decades consistent with the trends observed in the survey mean lengths (Fig. A.100) and observed in the fishery (Fig. A.86).

NEFSC survey: spatial patterns

Since the 1970s the Gulf of Maine haddock resource has become increasingly concentrated in the western Gulf of Maine (Fig. A.101). There are indications that the haddock population is beginning to repopulate areas in the central and eastern Gulf of Maine over the last decade compared to the population lows during the 1990s. A time series of Gini indices were calculated following the techniques outlined in Wigley (1996). These results support the patterns shown in distribution plots and suggest an overall concentration of the resource over the last fifty years (Fig. A.102). These patterns are similar to the spatial aggregation that has occurred in the commercial fishery that were previously noted.

NEFSC survey: catch-curve analyses

Catch curves were constructed for the NEFSC spring (Fig. A.103) and fall surveys (Fig. A.104) using methods described earlier in this report. Catch curves were conducted on a cohort-basis to avoid the confounding effects of differential year class strength on the interpretation of catch curve results. Linear regressions were fit to the log transformed catches of ages 3-8. The slope of the regressions suggest time series Z estimates on the order of 1.0 early in the time series, declining to around 0.25 late in the time series, though there is considerable variation in both the spring (Fig. A.105) and fall (Fig. A.106) cohort Z estimates.

Selectivity patterns estimated from the catch curve residual patterns suggest nearly flat selectivity across all ages for the spring survey (Fig. A.107) and increasing selectivity until about age-3 in the fall survey (Fig. A.108). The residuals patterns for both surveys do not provide compelling evidence for domed selectivity.

By comparing the ratio of catch-at-age of fishery catch to surveys we can achieve a qualitative understanding of the selectivities of each (e.g., is the fishery likely to have lower selectivity at older ages relative to the survey). While these comparisons do not offer definitive estimates of overall selectivity, they are helpful for gaining an understanding of the relationships in a model-independent framework (Clark 2013). We've compared the proportion of fish age-6 and older caught in the NEFSC surveys relative to the fishery (Table A.68). The comparison suggest a higher proportion of age-6 fish and a lower proportion of older fish caught in the survey relative to the fishery. These results are likely skewed by the fact that fish are not fully selected to the fishery until age-6 or 7 whereas they are selected to the survey at much younger ages. While there is a general trend for lower selectivity of the older ages in the survey, this trend is not consistent. There are several years where the surveys show greater selectivity for the older ages relative to the fishery. Overall, the results are inconclusive, suggesting that at ages-6 and older there are no appreciable differences in selectivity between the fishery and the NEFSC surveys.

MADMF bottom trawl survey

The MADMF has conducted research bottom trawl surveys during the spring and fall since 1978. A complete description of the MADMF trawl survey is provided in King et al. (2010). The survey strata included in the MADMF survey covers the nearshore habitat within Massachusetts state waters in the southwestern Gulf of Maine (Fig. A.109). The MADMF surveys are conducted in relatively shallow waters (<85 m) and are limited in their spatial extent; as such, they do not provide an index of the total stock resource. Given the limited spatial extent, the MADMF survey may be more susceptible to resource availability due to timing of onshore/offshore seasonal movements (i.e., process error). The MADMF survey occurs in early late April to early May in the spring and in mid September in the fall (Fig. A.110). Similar to the NEFSC surveys there are no indications of time series trends in bottom water temperature. The majority of haddock encountered in the MADMF survey occur in the northern extent of the survey area to the south and north of Cape Ann, though the fall survey has encountered small, but regular, catches of haddock in Cape Cod Bay (Fig. A.111).

The indices from the MADMF survey are relatively flat, with the occasional spike in indices that may correspond to incoming year classes (Table A.69, Fig. A.112). Haddock caught in the MADMF survey have not been aged, though an examination of the catch distribution at length shows that the majority of haddock caught in the spring survey are between 15-30 cm and between 5- 20 cm in the fall (Fig. A.113). Comparing these length distributions to the growth curves from the NEFSC survey (Fig. A.7) would indicate that the MADMF survey is primarily picking up signals of age-1 in the spring and age-0 in the fall.

An ALK constructed from the NEFSC survey utilizing fish collected from both inshore and offshore strata was used to construct MADMF indices-at-age. The inclusion of the offshore strata was necessary to avoid considerable imputation in the age-at-length determination. Abundance (numbers/tow) indices-at-age for the spring and fall surveys are presented in Tables A.70 and A.71 and Figure A.114. Biomass indices-at-age have not been prepared due to the absence of a MADMF-specific LW relationship. Cohort tracking plots show poor tracking of cohorts in the spring survey (Fig. A.115) and only limited tracking between age-0 and age-1 in the fall survey (Fig. A.116).

Because of the limited overlap between the MADMF survey area and the Gulf of Maine haddock distribution, as well as the lack of ageing of the MADMF survey indices, this survey has not been used in previous stock assessment models. Given the noted differences in the timing of the

NEFSC and MADMF surveys, caution is warranted in the use of the MADMF survey indices-at-age in any assessment model.

MENH inshore groundfish trawl survey

The MENH inshore groundfish trawl survey has not been included in previous assessments. The MENH survey began in fall 2000 and has been conducted in the spring and fall annually in the nearshore waters of the Gulf of Maine (Fig. A.117; Sherman et al. 2005). The spring survey occurs around mid-May, slightly later than the NEFSC survey, and the fall survey in late-September/early-October at about the same time as the NEFSC survey (Fig. A.118).

The spatial distribution of catches shows widespread distribution of juvenile (≤ 30 cm) haddock; however, the highest concentrations of adult haddock are found in the southwest region of the Gulf of Maine, consistent with the NEFSC survey observations (Fig. A.119). It's unknown if the nearshore habitat in the eastern Gulf of Maine offers some sort of preferential juvenile habitat that is not exploited by adults or whether there fish in these regions don't survive to adulthood. The distribution plots over time do not show any noticeable trends in the spatial for either the spring (Fig. A.120) or fall (Fig. A.121) surveys. The time series of abundance and biomass indices show a noted increase at the end of the time series (Table A.72, Fig. A.122) which is similar to the increases observed in both the NEFSC and MADMF surveys and may be indicative of recent strong recruitment.

Haddock maturity samples have been taken since 2002 in the spring survey with sampling variable across the time series ranging from 9 to 176 fish per year (Table A.73). Given the limited sampling across time, a time series of maturation-at-age could not be constructed, but a time series averaged maturity ogive was constructed to compare to the NEFSC maturity ogive. The L_{50} for haddock captured in the MENH survey was approximately 28 cm and 20 cm for females and males respectively (Fig. A.123). This compares to 37 cm and 30 cm for males and females in the NEFSC spring survey. It is unknown whether these differences reflect true biological differences or whether there are differences in the macroscopic determination of maturity stage between the two surveys. To our knowledge no comparison has been performed to evaluate whether differences exist in the macroscopic determination of maturity between the two surveys.

The size frequencies from the MENH survey indicate that the survey catches similar size classes to that of the MADMF survey (Fig. A.124). The spring survey catches fish primarily between 15 and 30 cm and the fall survey between 5 and 25 cm. Since 2005, age samples have been collected from the MENH survey, though only the fall survey has been aged (Table A.73). While the age sampling protocol specifies that one otolith be sampled per every one cm length sample (1:1), this has not always been the case due to the lower priority haddock sampling receives in the MENH survey (*S. Sherman pers. comm.*). Owing to the small number of haddock sampled in some years, and the lower priority of haddock age sampling, the number of age samples available in any year ranges from 2 to 117 ages (Table A.73). Assuming growth was similar between the two surveys, the borrowing of the NEFSC ALKs would have minimal impacts. A comparison of the length frequency distributions-at-age shows moderate agreement at ages-0 and

1, though there is insufficient information to make any determination at ages-2 and older (Fig. A.125). Given the noted differences in the maturity-at-length, there is some concern with applying the NEFSC ALKs to the MENH survey.

Abundance (numbers/tow) indices-at-age for the MENH survey are presented in Tables A.75 and A.76 for the spring and fall surveys, respectively. The 1998, 2003 and 2010 year classes are evident in both the spring and fall surveys (Fig. A.126). Biomass indices-at-age have not been prepared because biomass indices-at-length were not readily available. The spring survey exhibits reasonable cohort tracking out to age-5 (Fig. A.127), though the cohort tracking in the fall survey is limited to between age-0 and age-1 (Fig. A.128).

For many of the same reasons discussed for the MADMF survey, caution should be used in using the MENH indices-at-age in an assessment model. The MENH spring survey does appear to have slightly better cohort tracking abilities compared to the MADMF surveys.

Inter-survey comparisons

Inter-survey comparisons show generally good agreement between the spring and fall NEFSC surveys, both in terms of abundance (numbers/tow, Fig. A.129) and biomass (weight/tow, Fig. A.130). Both surveys track reasonably well with the MADMF spring survey, though tracking with the MADMF fall and MENH surveys is poor. Comparison of age-specific indices shows some cohesion between the NEFSC, MADMF and MENH age-1 indices (Fig. A.131) as well as the fall age-0 indices (Fig. A.132); interestingly, there is generally poor agreement between the fall age-1 indices.

TOR A.3. Evaluate the hypothesis that haddock migration from Georges Bank influences dynamics of GOM stock. Consider role of potential causal factors such as density dependence and environmental conditions.

In March of 2013 the Associated Fisheries of Maine (AFM) submitted a problem statement (AFM 2013) to the NEFMC Groundfish Committee requesting that the Committee develop a management strategy that would consider the "spillover" of larger Georges Bank haddock into the smaller Gulf of Maine stock area (see Figure A.133 for a comparison of stock sizes). While the "spillover" concept was not explicitly defined, it was presumed to refer to the density-dependent expansion of an otherwise independent stock across its stock boundary. In response to the AFM problem statement the NEFMC passed the following motion at its April, 2013 meeting:

"To task the PDT and SSC to examine the issue of GB haddock spillover into the GOM stock area, provide an estimate of the amount of spillover when large year classes of GB haddock occur, and provide suggestions as to how the anticipated spill over of the strong 2010 year class can be used to adjust the GOM haddock ABC for FY 2013, 2014 and 2015."

This led to an in-depth review of the available scientific information by the staff from NEFSC, the NEFMC Groundfish Plan Development Team (GPDT) and the NEFMC Scientific and

Statistical Committee (SSC). The investigation had four primary themes which are summarized below. Further details on the analyses can be found in NEFMC GPDT (2013) and the SSC review of the GPDT analyses can be found in NEFMC SSC (2013).

Literature review of Gulf of Maine/Georges Bank exchange rates

There is extensive body of scientific literature on haddock tagging in the northwest Atlantic extending back to the early 1900s. Much of the work was performed with the objective of understanding stock structure and general movement patterns, so while the literature does indicate movement between the Georges Bank and Gulf of Maine regions, the research did not attempt to quantify the magnitude of the movement. More recent tagging studies focused tagging effort around the closed areas (Brodziak and Col 2006) and noted that the vast majority of haddock tagged on Georges Bank remained on the bank but some movement to the Gulf of Maine was observed. The Brodziak and Col (2006) report proposed a ten percent transfer from Georges Bank to the Gulf of Maine; however, any mention of an exchange rate is absent in a later report of the same study written after more tag returns had been received (Brodziak et al., 2008a). Based on the SSC's review of the tagging information it concluded that the Brodziak and Col (2006) ten percent transfer estimate is an upper bound and that the observed movement is likely reflective of movement across certain boundary areas and not indicative of processes operating across the broader stock areas (NEFMC SSC 2013).

There is little evidence from the NMFS trawl survey distribution that supports movement from one stock to the other. The deep central Gulf of Maine basin appears to provide a barrier to juvenile and adult dispersal (Fig. A.2, Begg 1998, Cargnelli et al. 1999), which would support the stock separation evident in the tagging studies. An examination of distribution plots from NEFSC bottom trawl surveys and observer catch data did not show clear evidence of the Georges Bank stock expanding beyond its stock boundaries following periods of strong recruitment (i.e., 2003 and 2010 year classes; Figs. A.134-136).

Based on the GPDT s review of the literature, it conclude that the exchange rates are not well characterized (NEFMC GPDT 2013). This conclusion was supported by the SSC (NEFMC SSC 2013).

Revisiting past assertions of recruitment synchrony between the Gulf of Maine and Georges Bank stocks

Brodziak et al (2008b) and others (Clark et al. 1982) have reported synchrony in the recruitment strength of Georges Bank and Gulf of Maine haddock stocks. A re-evaluation of the Brodziak et al. (2008b) analysis examining the concordance of haddock survey indices-at-ages from the Georges Bank and Gulf of Maine stock areas was conducted. Brodziak et al. (2008b) reported a significant correlation (R=0.53) between Georges Bank and Gulf of Maine age-0 NEFSC fall bottom trawl survey indices. While there is some synchrony of the recruitment events (covariance of Gulf of Maine recruitment and Georges Bank recruitment) it accounts for only 28% of the total variance, leaving 72% unexplained. Although the correlation is significant, any correlation above 0.3 would be statistically significant given the length of the time series (1963-2004).

Under the hypothesis that older Georges Bank fish spill-over into the Gulf of Maine, it might be expected that younger age Gulf of Maine abundance-at-age indices would not track cohorts at older ages within the Gulf of Maine as well as younger Georges Bank abundance-at-age indices do. Paired comparisons of Kendall rank correlation coefficients indicated that differences between Georges Bank and Gulf of Maine indices were statistically significant. Gulf of Maine indices had higher concordance with older Gulf of Maine indices than Georges Bank indices. Predictions of age-3 abundance in the Gulf of Maine from age-1 abundance of Georges Bank haddock appear to be highly uncertain. The abundance of age-1 fish on Georges Bank does not seem to be a reliable way to predict future cohort strength in the Gulf of Maine.

While the GPDT did note that there was some positive synchrony in year classes, the strength of the association was weak to moderate (explanatory power of only 4-26%), generally explaining a small amount of variation in Gulf of Maine recruitment. Furthermore, some correlation in year class strength could be due to similar environmental conditions influencing the recruitment dynamics of both stocks.

Year-class tracking in survey data and Gulf of Maine haddock assessment diagnostics

The GPDT noted that, if expansion of the Georges Bank stock across its border in response to strong cohorts does occur, then fish from the large 2003 year class on George Bank should have also appeared as a strong cohort in the Gulf of Maine assessment, obscuring cohort tracking in the Gulf of Maine haddock assessment. Spillover of just 1% of the large 2003 GB year class would have approximately doubled the size of the Gulf of Maine 2003 year class and obscured cohort signals within the survey indices. If spillover of Georges Bank haddock were occurring in these large quantities, particularly given the noted asynchrony of the 2003 year class) it would add considerable variability to survey indices, making the tracking of cohorts within the Gulf of Maine stock difficult. The Gulf of Maine haddock indices from the NEFSC bottom trawl surveys show very strong tracking of individual cohorts (Fig. A.96).

Moreover, diagnostic issues should be evident in the assessment model if “spillover” were occurring. Examination of the tracking of cohorts within survey indices at age as well as assessment model diagnostics (survey residuals, retrospective patterns) yielded no evidence to support a spillover of a detectable magnitude. Additionally, maturity, weights-at-age and selectivity difference between the two haddock stocks lends further support to stock separation. The SSC supported the PDT interpretation of these diagnostics.

Analysis of the consequences of setting catch advice based on movement rate assumptions

Projection scenarios conducted by the GPDT reveal that net movement rates greater than 2% of just the 2010 Georges Bank year class into the Gulf of Maine would quickly inundate the Gulf of Maine stock due to the inequalities in stock sizes. Assuming even a relatively small percentage of net movement into the Gulf of Maine would have large negative consequences for the Gulf of Maine stock if spillover is not occurring. These projections suggest that ad-hoc adjustments of quota for spillover would increase the risk of overfishing and spawning biomass declines for the Gulf of Maine stock in 2014 and beyond. The consequences of setting catch based on movement rates, if in fact movement was not occurring, would be severe for the Gulf of Maine stock. The magnitude of the difference in stock sizes and ACLs means that even a small assumed exchange rate could result in fishery catches many times the current Gulf of Maine ACL, and could even approach the entire estimated haddock biomass in the Gulf of Maine.

Conclusions of the GPDT investigation

Based on the work performed by the NEFSC and GPDT, the GPDT concluded that there was no technical basis for adjusting the quota between the two stocks based on the "spillover" of Georges Bank haddock into the Gulf of Maine stock. The SSC agreed with this conclusion noting the significant risk to the Gulf of Maine haddock resource that could occur should an adjustment to the quota be made, particularly given "*...the lack of compelling empirical evidence.*" The SSC further noted that "*if fishermen are observing abundance of haddock in the Gulf of Maine that does not seem to comport with the outcomes of the assessment, this might be due to a recent increase since the terminal year of the last assessment update (2010). If so, the appropriate response is to update the Gulf of Maine assessment to see if that change is detected.*" A GPDT examination of updated survey data suggests the 2010 Gulf of Maine year class may be stronger than the geometric mean assumption used in the 2012 AOP projections. The appropriateness of the geometric mean assumption is discussed in depth in Palmer et al. (2014).

Re-analysis of Northeast Consortium Cooperative Haddock Tagging Program data

Between March 2005 and December 2008 the Northeast Consortium Cooperative Haddock Tagging (NCCHT) Program tagged 20,418 haddock in the Georges Bank and Gulf of Maine region. Of the releases, 531 recoveries (168 released with two tags) were reported between 2005 and 2010 (Fig. A.137). A description of the study design and a summary study results are provided in Brodziak et al. (2008a) and CCCHFA (2009). While the study did have the stated design to describe movement between the Gulf of Maine and Georges Bank stocks, the primary focus of the study was to provide information on fish movements across the boundaries of four areas closed year-round to groundfishing. Tag releases were not distributed proportional to stock abundance or fishing effort, but released disproportionately inside closed areas (13,122, or 64%, of the releases were inside closed areas).

To date, no formal analysis of the tagging data has attempted to estimate movement rates. Miller and Palmer (2014) applied a finite-state continuous time model to the existing NCCHT data to generate estimates of mortality and movement rates. Overall, model fit was insensitive to the assumed reporting rates, with only an approximate two unit change of the maximized log-likelihood with changes to the assumed reporting rate. Parameter estimates other than those for annual and regional fishing mortality rates were not greatly affected by the assumed reporting rates over a wide range of values. There was poor precision of the natural mortality rate estimate, but the point estimate was consistently between 0.2 and 0.3 for reporting rates ≥ 0.3. The instantaneous migration rates implied greater movement of individuals into the Gulf of Maine than to the Georges Bank stock area given that they survive all sources of mortality and the estimates are not sensitive to the assumed reporting rate. With a reporting rate = 1, the migration rate estimates imply individuals starting in the Gulf of Maine have approximately 94% probability of being in the Gulf or Maine 1 year later given they survived the interval. Individuals starting on Georges Bank have approximately 86% probability of being on Georges Bank one year later given they survived the interval. Fishing mortality rate estimates were negatively correlated with reporting rates. In 2005, fishing mortality was estimated to be greater in the Gulf of Maine than Georges Bank whereas in years 2006 to 2008, estimates were similar for the two stock areas. In 2009 and 2010 estimates for the Georges Bank were greater than the Gulf of Maine.

The authors stressed that the results are greatly affected by the location, size of fish, and timing of the releases. Many of the releases were near the stock boundaries and in areas closed to groundfishing. The proximity to the stock boundaries might cause migration rates to be greater than the general population if there are substantial portions of the populations further away from stock boundaries and they move at similar speeds and directions. Releases in the closed areas may result in lower estimated fishing mortalities than the general population if the fish stay in the vicinity for some time which may be the reason for the lower fishing mortality estimated for unmixed individuals for the first 2 months after release.

The SAW 59 WG found the mortality rates consistent with other lines of information (e.g., catch-curve analyses, assessment model outputs), but felt that the mixing rate estimates were high and inconsistent with the analyses conducted by the GPDT. The SAW 59 WG did not feel that the tagging exercises conducted to date had been designed in a way that would allow annual interchange proportions to be reliably estimated. The SAW 59 WG also examined assessment models that allowed for estimation of mixing between stocks. These model results are described under TOR 4, but generally, the estimated annual percent mixing from Georges Bank to the Gulf of Maine from these models was low (<0.8%), and consistent with the GPDT analysis. Given the conflicting information provided by the NCCTP data, the SAW WG recommended that additional research designed to expressly determine between-stock movement rates is needed (see TOR8).

TOR A.4. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series (integrating results from TOR-3), and estimate their uncertainty. Include a historical retrospective analysis to allow a comparison with previous assessment results and previous projections.

Update of the 2012 AOP ADAPT-VPA model

There were substantial changes in the underlying data used for the current SAW/SARC 59 assessment compared to the data used in the GARM III and subsequent 2012 AOP assessments. The major changes include: reestimated landings-at-age, modifications to the fleets included in commercial discards, conversion from the MRFSS to MRIP recreational sampling program and calibration of historical MRFSS time series, inclusion of recreational discards and an assumption of 50% discard mortality, new estimates of weights-at-age that reflect landings and discards, minor revisions to the maturity ogive, and updates to the NEFSC survey indices. Additionally, there are three more years of catch and survey information that needed to be incorporated into the model. To fully understand how these data changes impact the VPA update, a bridge was constructed to transition from the 2012 AOP assessment model to a fully updated model.

The 2012 AOP assessment was conducted using the Adaptive Framework Virtual Population Analysis (ADAPT-VPA) model (NOAA Fisheries Toolbox ADAPT-VPA version 3.1.0, 2010). The most recent version of the ADAPT-VPA software is version 3.4.5 (2014). The differences between the VPA model versions primarily affect the usability and graphical interface; there should be no differences in the model calculations. The model formulation used in for the 2012 AOP assessment included an age-9^+ plus group with the 'backward' computation used to estimate the plus group stock size. For the SAW/SARC 59 model formulation, the 'combined' computation will be applied. The difference between the two methods relates to how the fishing mortality is calculated for the plus group. The backward method computes F on the plus-group as the product of the plus group ratio (α) and $F_{A-1,t}$. The stock size for the plus group is then sequentially calculated for years t=1 to T. The backward method can result in the predicted catch of the plus group not matching the observed catches, additionally, since all years are treated independently, impossible stock sizes can result (i.e., $N_{A,t} > \Sigma N_{A-1,t-1} + N_{A,t-1}$). When a low proportion of the population is in the plus group, the impacts of these issues are negligible. The combined computation method address the shortcomings of the backward computation by calculating consistent F and N that adhere to the catch equation and the input ratio of F between the oldest true age and the plus group age. The disadvantage of the combined method is that it disassociates the F on the oldest true age from the younger ages and can result in the F on the oldest true age and the plus group much higher or lower than the other ages. This approach is more appropriate when the plus group abundance is relatively large and the ratio of F between the oldest true age and the plus group age is well determined as is the case as it is in many of the years of the Gulf of Maine haddock time series. See the NOAA Toolbox ADAPT-VPA Version 3.0 Reference Manual for a full description of the methods.

Commercial landings and discards from 1977 to 2010 as well as recreational landings from 1981 to 2007 were accounted for in the model. Tuning indices included the NEFSC spring ages 1-6^+, and NEFSC fall ages 2-8^+ lagged forward by an age and a year (e.g., 2006 age-2 fish become 2007 age-3 fish in the model). The fully recruited F is determined as the unweighted average F on ages 6 to 8. The terminal year F on age-8 is estimated using the aggregate survivorship of age-6 and 7 fish from year t to t+1. In years prior to the terminal year, F on the oldest age is estimated using stock sizes from ages 6 and 7. Maturity-at-age was estimated from a time series average of the maturity observations from 1977 to 2010. Spawning stock biomass was calculated assuming an April 1 spawning period (0.25 into the calendar year). Natural mortality is assumed age and time invariant at M=0.2.

The general approach used to build the bridge from the 2012 AOP VPA to an updated SAW/SARC 59 VPA was as follows (model numbers correspond to the model summaries presented in Table A.77):

- Model 1: Re-run of the 2012 AOP update of the GARM III VPA model.
- Model 2: Update the ADAPT-VPA software to version 3.4.4.
- Model 3: Modify the plus-group calculation from 'backward' to the 'combined' method.
- Model 4: Include revised catch-at-age estimates of commercial landings, discard and recreational landing (harvest). Update catch WAA.
- Model 5: Include recreational discards-at-age assuming 50% discard mortality. Update catch WAA and stock WAA.
- Model 6: Add three additional years of catch data (2011-2013) and update all NEFSC survey indices-at-age. ***This model represents an updated VPA model.***

The results from the bridge building exercise are presented in Table A.78.

Updating the VPA software had no impact on model results (Fig. A.138-140). Using the combined method to handle the plus group calculations resulted in a rescaling of spawning stock biomass (Fig. A.138) and minor changes in the ages-6 to 8 average fishing mortality (F_{6-8}) time series (Fig. A.139). The combined method resulted in slight improvements in overall model fit as evidenced by the mean squared residual, CVs on the terminal ages and retrospective Mohn's rho values (Table A.78). Updating the catch data and incorporating the recreational discards increased the mean squared residuals and CVs, had variable impacts on the terminal population size and led to an overall decrease in the retrospective pattern; however, there was little impact on the overall assessment results in terms of spawning stock biomass, fishing mortality and age-1 recruitment.

Adding three additional years of data to extend the VPA through 2013 had minimal impacts on the model diagnostics and model results through 2010. The most notable feature of the 2013 update is the presence of what appear to be several strong year classes at the end of the time series, beginning with the 2010 year class (Fig. A.140). The 2012 AOP update did not provide a direct estimate of the 2010 year class owing to the sparseness of the information available to achieve a reliable estimate (a single survey observation). Instead, the 2012 AOP update applied a time series geometric mean to estimate the size of this year class in the t+1 year (see Palmer et al. 2014a for an evaluation of this assumption). With three more years of survey and catch data, it now appears that the 2010 year class was above average. Additionally, both the 2012 and 2013 year classes appear to be above average, though caution should be given to these estimates because of the limited observations available.

The overall retrospective pattern of the 2013 ADAPT-VPA model decreased relative to the previous models (Fig. A.141). The retrospective peels from the 2013 ADAPT-VPA update show variable model retrospective error, with retrospective peels of both spawning stock biomass (Fig. A.142), fishing mortality (Fig. A.143) and age-1 recruitment (Fig. A.144) having positive and negative relative differences over the 7-year peel and no consistent patterning. The survey fits to the final 2013 update of the ADAPT-VPA model do not exhibit strong residual patterns in the fits to either the spring (Fig. A.145.a) and fall (Fig. A.145.b-c) survey indices-at-age. Survey catchabilities (q) for the minimum area swept survey indices-at-age were relatively flat across ages in the spring survey and well below those of the fall survey (Fig. A.146). The fall survey qs increase with age, with the qs on the ages 5-8$^+$ indices exceeding 1 (fit as ages-6 to 9$^+$ lagged forward in the model). These patterns are nearly identical to those of both the GARM III (NEFSC 2008) and 2012 AOP VPA model (NEFSC 2012). While the q values on the older ages in fall survey are large, the uncertainty of these estimates is large. Partial recruitment patterns over the past five years are variable, but indicate that haddock in the Gulf of Maine do not fully recruit to the fishery until age-7 (Fig. A.147).

General conclusions from the updated 2013 ADAPT-VPA are:

- Use of the combined method for the plus group calculation had the largest impact on the overall assessment results, with a downward rescaling of spawning stock biomass and variable impacts on the time series of average fishing mortality.

- The updates to the data inputs had only minor impacts on the model results.

- Extending the time series through to 2013 did not change the historical perception of the resource. The more recent data does suggest that there are at least two strong year classes (2010 and 2012) that have been spawned over the past three years beginning in 2010. There has been an overall increase in the spawning stock biomass, primarily as result of the maturation of the 2010 year class. The projections from the 2012 AOP update assumed the size of the 2010 year class to be equal to the geometric mean recruitment of the time series (1.1 million fish). Based on the updated VPA, this assumption underestimated the year class size.

Development of an ASAP statistical catch-at-age model

The 32nd SAW WG concluded that "[t]*here is insufficient length and age sampling of US commercial landings to reliably estimate catch at age required to complete a VPA-based analytical assessment of this stock.*" (NEFSC 2001). While the results of the GARM III and 2012 AOP assessments show that catch-at-age could be constructed to support a defensible VPA model, the amount of imputation required to construct the catch-at-age time series, primarily in the way of commercial discards and recreational catch, brings up questions as to whether this stock would be better assessed using a statistical catch-at-age model where it is not assumed that catch is known exactly. Additional support for exploring a statistical catch-at-age model include: the ability to explore alternative model formulations to counter/lend support to VPA results, ability to estimate a stock-recruit relationship internal to the model, and the ability to explicitly handle data uncertainty, particularly with respect to uncertainty in the survey data.

The use of a statistical catch-at-age model for the Gulf of Maine haddock assessment was explored. More specifically, the statistical catch-at-age model, ASAP (Age Structured Assessment Program v3.0.17, Legault and Restrepo 1998), which can be obtained from the NOAA Fisheries Toolbox (http://nft.nefsc.noaa.gov/). ASAP is an age-structured model that uses forward computations assuming separability of fishing mortality into year and age components to estimate population sizes given observed catches, catch-at-age, and indices of abundance. Discards can be treated explicitly. The separability assumption is partially relaxed by allowing for fleet-specific computations and by allowing the selectivity-at-age to change in blocks of years. Weights are input for different components of the objective function which allows for configurations ranging from relatively simple age-structured production models to fully parameterized statistical catch-at-age models. The objective function is the sum of the negative log-likelihood of the fit to various model components. Catch-at-age and survey age composition are modeled assuming a multinomial distribution, while most other model components are assumed to have lognormal error. Specifically, lognormal error is assumed for: total catch in weight by fleet, survey indices, stock recruit relationship, and annual deviations in fishing mortality. Recruitment deviations are also assumed to follow a lognormal distribution, with annual deviations estimated as a bounded vector to force them to sum to zero (this centers the predictions on the expected stock recruit relationship). For more technical details, the reader is referred to the technical manual (Legault 2012).

Description of the SAW/SARC 59 ASAP base model

Model sensitivities

In evaluating the ASAP model for SAW/SARC 59, many model configurations were explored. In total, there were over 70 model runs conducted of the ASAP model. Overall, the variability in model results was small (Figure A.148), indicating that assessment results are robust to alternate assumptions and configurations ; however, there is considerable variability of the terminal estimates owing to model uncertainty in the estimation of two potentially large cohorts at the end of the time series. The nature of the sensitivity models fell into two different categories: 1) determining whether an alternate model formulation offered improved fit to the data; and 2) evaluating the sensitivity of the model with respect to a range of assumptions. Table A.79 provides a short summary of the number of models by sensitivity category as well as indicating where in this report a description of those sensitivity models can be found. The process of transitioning from the VPA to the ASAP model and evaluation and fine tuning of the ASAP model is described below. While attempts have been made to describe the development in a linear process, the model development process is inherently non-linear and fraught with dead-ends and second guessing of how the development process could have been better conducted.

Construction of a base ASAP model (ASAP_BASE)

An ASAP model for Gulf of Maine haddock was developed using past experience of ASAP model formulations for other groundfish stocks (e.g., Gulf of Maine cod, NEFSC 2013). Consistent with the VPA model formulation, the base ASAP model (ASAP_BASE) was constructed using an age-9 plus group and including data from only the years where catch-at-age data are available (1977-2013). The age-9^+ group decision is based primarily on the poor precision in estimating catch-at-age beyond age-8 in the commercial fishery (Table A.18) and sparseness of the survey observations at older ages (e.g., Fig. A.97). Unlike the VPA model which was run using calibrated survey indices based on the SHG station selection criteria, the base ASAP model was run using calibrated survey indices based on the TOGA station selection criteria for the Bigelow survey years (2009-2013). This change was done to keep the station selection criteria consistent with the Bigelow sampling protocols. The impacts of this change had negligible impacts on the assessment results (see Appendix A.2).

Fishery catches were modeled as a single fleet, with both commercial and recreational fleets combined. A sensitivity model exploring the treatment of commercial and recreational fleets separately is presented in Appendix 2. Three different fishery selectivity blocks were applied (1977-1988, 1989-2004, 2005-2013). The choice in selectivity blocks was informed by the previous experience with other Gulf of Maine groundfish stocks. The 1988/1989 split corresponds to the start of the at-sea observer program and the direct observation of fishery discards and length frequency information (though length sampling was sparse in the early years). Beginning around 1992, the magnitude of recreational catch began to increase. There were no major regulatory changes specifically in 2004 or 2005 that would give *apriori* expectation for a change in selectivity; however, the recreational minimum size dropped from 23” to 19” between 2002 and 2004 and there was a major change in the commercial dealer reporting system (paper to self-reported electronic) for all federally permitted dealers in 2005 which could have also impacted biological sampling of the commercial fishery. Perhaps more importantly, 2005 corresponds with the point when the declines in mean size-at-age stabilized; size-at-age has remained relatively stable since 2005 (e.g., Fig. A.86). In 2007, the commercial minimum size was reduced from 19” to 18”, with further reductions to 16” in July 2013. Minimum retention sizes were increased to 21” in the recreational fishery in 2013. The selectivity block assumptions were evaluated using several sensitivity models which are described in Appendix A.2.

Base on the partial recruitment patterns from the VPA selectivity-at-age was freely estimated with selectivity fixed at 1.0 for age-7, but allowed to be freely estimated at ages 8 and 9^+. The VPA partial recruitment patterns (Fig. A.147) were suggestive of limited doming of the fishery selectivity. While not conclusive, the catch-curve analysis also provided some indication of lower selectivity at older ages in the fishery (Fig. A.89). Similarly, the two NEFSC surveys were fixed at 1.0 on age-6, though the VPA catchability patterns (Fig. A.147) indicated that maximum selectivity may occur around age-5 in the spring survey and not until age-8 or older in the fall survey. The VPA catchability patterns are not dissimilar to the patterns observed in the catch curve analysis (Figs. A.107-108). The selectivity assumptions applied in the ASAP_BASE model were further evaluated using several sensitivity models described later in this report.

Like many haddock stocks, recruitment of Gulf of Maine haddock is highly episodic and not well described by traditional stock recruitment relationships. Given this, recruitment was modeled as deviations from the geometric mean (steepness fixed at 1.0). The ASAP model allows the deviations to be constrained by applying a penalty on the deviations. For the base model, the penalty function (lambda) was set at 0.2 and the CVs on the recruitment deviations were set at 0.5 for all years except the final three years, which were set at 0.1. This was an attempt to apply 'shrinkage' to the mean of the terminal year cohorts where there are limited observations available from which to accurately estimate year class size. This decision was based on past experience with Gulf of Maine haddock in a VPA model framework (Palmer et al. 2014a). The treatment of recruitment deviations was subsequently evaluated extensively with this work described later in this report.

The effective sample size (ESS) for the fishery was set at 80 and 15 and 20 for the NEFSC spring and fall surveys, respectively. CVs on the total catch were set at 0.15 for the period prior to recreational catch estimates (1977-1980), 0.10 for the period prior to direct discard estimates (1981-1988) and at 0.05 for the remainder of the catch time series (1989-2013). The CVs on the surveys were initially set equal to the bootstrapped CVs presented in Table A.54. The bootstrapped CVs characterize the sampling, or observation error, but additional process error may be present in the survey indices that are not reflected in the bootstrapped CVs. As with other model assumptions, ESS and CV assumptions were fully evaluated prior to the formulation of the final preferred ASAP model.

Diagnostics and results of the base ASAP model (ASAP_BASE)

A summary of basic model diagnostics for the ASAP_BASE model is provided in Table A.80. Root mean square error (RMSE) values are generally high (with the exception of the catch [Fleet1]), indicating over fitting of the data. Model fits to the fishery catches were good, and improved over time. Fits to catches early in the time period were variable, and not unexpected given larger CV placed on these catches owing to the higher uncertainty. There is some patterning of residuals over time, however the residuals are small. Generally, there is close agreement between modeled and observed catches (Fig. A.149).

The ESS of 80 assumed for the fishery catch-at-age appears reasonable for the later part of the time series, but is likely too high for the early parts of the time series (Fig. A.150), particularly the period in the late 1980s and early 1990s when there was severe truncation in the catch age structure (Fig. A.84) and low landings. Fits to the observed catch-at-age (Fig. A.151.a-c) were relatively good with several modes associated with moderate-to-strong year classes being picked up well (e.g., 1998 and 2003 year classes). There were no large residual runs indicative of year effects; there are however, some small year class effects associated the 1998 and 2003 year classes (Fig. A.152). Fits to the mean catch-at-age suggest that the catch-at-age is being fit too tightly early in the time series (Fig. A.153), consistent with the ESS fits.

Estimated fishery selectivities were flat-topped in the first selectivity block (1977-1988), but slightly domed in blocks 2 (1989-2004) and 3 (2005-2013; Fig. A.154). The selectivity estimates hit several parameter boundaries in blocks 1 and 3 (Table A.81) and will require additional fine tuning. The fishery selectivity parameters are well estimated with CVs ≤ 0.20 on most ages with the exceptions of the youngest and oldest ages. The selectivity trends with decreasing selectivity on the younger ages through time is consistent with management measures that have gradually increased mesh sizes and minimum retention sizes.

Fits to the NEFSC spring survey index exhibited no strong residual patterning (Fig. A.155). Overall, the model tracks the spring survey index well, though not unexpectedly, large residuals are observed for years with exceptionally low (e.g., 1990) and high (e.g., 2013) survey indices. The input ESS value of 15 were generally supported by the modeled estimates (Fig. A.156), though as noted with the fishery ESS values, the information content in the indices-at-age is variable over time and appears lower in the earlier third part of the time series. There is a decent fit of observed to predicted age compositions with an absence of year- or year class-effects and no large residual blocks (Fig. A.157). Similar to the ESS plot, the fits to the mean age suggest that the current ESS of 15 is likely too high, particularly in the first third of the time series (Fig. A.158).

The models fit the NEFSC fall survey reasonably well (Fig. A.159). Several large outliers are apparent, first around 2000 and most recently in 2011-2013. ESS values of 20 are generally consistent with the modeled estimates, though similar to the catch and spring model ESS estimates, there appears to be lower information content in the late 1980s/early 1990s (Fig. A.160). The fit to the age composition data was generally good, though there is some patterning particularly at the younger ages (< age-3) and in the plus group (Fig. A.161). The overall fit to the mean catch-at-age is reasonable though the RMSE values greater than 2 suggest that the ESS values should be lowered to account for higher uncertainty in the indices-at-age (Fig. A.162).

The NEFSC spring survey exhibits higher selectivity at younger ages relative to the fall survey (Fig. A.163). The selectivities are generally well estimated with CVs less than 0.20 for most ages, though the CV values do suggest that model estimates have hit boundaries for several ages (Table A.81). The selectivity patterns are generally consistent with the catchability patterns from the VPA (Fig. A.147) and the residual patterns from the catch curve analyses (Figs. A.107-108).

Survey catchabilities (q) are presented in Figure A.164. The NEFSC spring survey catchability estimate (q=0.26) is consistent with the catchability estimates from the VPA. The fall survey catchability estimate from the ASAP_BASE model (q=0.99) is considerably lower than the qs on the older ages in the VPA model. Profiling over a range of fall survey q values showed a model preference within the range of 0.8 to 1.0 (Fig. A.165). Within this range, there were minimal impacts on estimates of spawning stock biomass and fishing mortality. Since minimum area swept survey indices were used in the fitting of both the VPA and ASAP model, a q=0.99 could be suggestive that the survey is 100% efficient which is unlikely and indicative of model scaling problems. However, it should be noted that the minimum area swept scaling was performed using a wing spread footprint of 0.012 nm^2 per survey tow for the Albatross survey. If haddock herding occurs between the trawl doors, as has been reported in the literature (e.g., Engås and Godø 1989), survey catchabilities based on a wing spread footprint could represent upper bounds on survey catchability. A sensitivity model using door spread footprint assumption of 0.023 nm^2 per survey tow resulted in estimates of fall q of 0.48 and a spring q of 0.13 with no impacts on model fit or results (Table A.82). The true catchability of the NEFSC surveys are unknown, but with respect to the NEFSC fall catchability estimates, q values within the range of 0.48 to 0.99 are not suggestive of model scaling issues.

Estimates of spawning stock biomass recruitment are similar between the ASAP_BASE model and 2013 update of the VPA model (Fig. A.166). The peak of spawning stock biomass in the early 2000 period is scaled higher in the ASAP model, but the biomass in the early parts of the time series are of equivalent scales. The estimates of the 2010 and 2012 year class are considerably higher in the VPA model; this is expected owing to the shrinkage implemented in the ASAP_BASE model. The notable difference between the ASAP and the VPA models in the fishing mortality patterns. It should be noted that the basis that the fishing mortality basis varies between the two models, with ASAP fishing mortality expressed as the mortality on the fully recruited age classes (F_{full}) and the VPA expressed as the average mortality on ages 6-8 ($F_{6\text{-}8}$). However, the basis for expressing fishing mortality does not explain the differences between the two model runs. Another likely contributing factor to the differences in fishing mortality trends is the inherent difference between the estimation approaches of VPA and statistical catch-at-age models, specifically, the exact fitting of catch within the VPA framework. The largest differences between the two models occurs during the late 1980s/early 1990s when stock sizes were at time series lows (i.e., model variability in estimating fishing mortality on a small population).

Like the VPA model, the ASAP retrospective error was small (Fig. A.166, Table A.80). Coefficients of variation on SSB, F and recruitment have generally been less 0.2 except at the end of the time series where CVs approach or exceed 0.2 (Fig. A.167). Recruitment patterns for the Gulf of Maine haddock stock appear to have an auto-regressive nature with blocks of moderate-to-strong recruitment followed by period of poor recruitment leading to strong residual patterns in the deviations about the geometric mean (Fig. A.168). The periods of lowest recruitment correspond to periods of low spawning stock biomass (Fig. A.166).

Refinements of the ASAP base model

The model runs explored in the following section were intended to fine tune the base ASAP model and further explore issues related to model assumptions and initial configurations evident in the ASAP_BASE model diagnostics described above. Specifically, the model explorations described below address selectivity, the assumed precision of fishery catches, survey process error, and modeling of stock recruitment. Sensitivities not related to the transition from the ASAP_BASE model to the final SAW/SARC 59 model are described in detail in Appendix A.2. A summary of model diagnostics and results from these intermediate models is provided in Tables A.83 and A.85.

<u>ASAP_final_temp1 model</u>: The modeling of stock recruitment relationship, specifically, the amount of constraint applied to recruitment deviations was given considerable attention in several sensitivities described in depth in Appendix 2 and summarized briefly below. Of specific concern was the variability of the estimated sizes of the 2010 and 2012 year classes under a range of model configurations. The base model applied a penalty function (lambda) of 0.2 and set the CVs on the recruitment deviations at 0.5 for all years except the final three years, which were set at 0.1. Within ASAP, the CV value is converted to a variance and standard deviation that are used in the negative log likelihood calculations of the model minimization process

(Legault 2012). The tightening of CV bounds on the terminal recruitment estimates was an attempt to apply 'shrinkage' to the mean of the terminal year cohorts were there are limited observations available from which to accurately estimate year class size.

Several different methods of modeling recruitment deviations were explored: (1) setting the lambda to zero and allowing recruitment estimates to be unconstrained; (2) setting lambda at 1, but then applying some constraint on the recruitment deviations through the adjustment of the CV values. Under option 2, several different configurations were explored: a) hold CVs constant throughout the time series (no shrinkage); b) applying shrinkage over the terminal four years (e.g., those years not fully recruited to the surveys or fishery similar to the approach used the ASAP_BASE model); and c) applying shrinkage to only the terminal year.

A sensitivity analysis was conducted evaluating the retrospective performance of the ASAP model under the four different configurations described above. The performance of the model was evaluated back to 2000 to understand how well the model estimated the size of the large 1998 year class with only three years of information, which is identical to the current situation with respect to being able to estimate the size of the 2010 year class. While the results of the retrospective analysis were not conclusive, this analysis did suggest that within a Gulf of Maine haddock ASAP model, the 'no shrinkage' method offered a lower degree of recruitment estimation error compared to the other methods evaluated.

The specification of the recruitment deviations CVs will affect the level of constraint the model places on recruitment deviations. The sensitivity of the model to recruitment deviation CVs was evaluated by profiling across CV values from 0.6 to 2.4. Based on the profiles of the likelihoods, there is model preference for CVs on the order of 2.0; this is the point when the RMSE on the recruitment deviations approaches 1. Within this range, model results are relatively stable (e.g., 2013 SSB, 2010 and 2012 year class sizes and SSB and F retrospective patterns). One concern with the model runs at the high CVs are the fits to the survey indices; models with high recruitment deviation CVs tend to tightly fit both the 2013 spring and fall survey observations, both of which have large age-1 indices. Based on the model fits to large survey observations earlier in the time series, this degree of fit seems unlikely.

Based on the results of this exploratory work, the ASAP_BASE model was refined by setting a lambda value of 1 and holding the CVs constant at 2.0 for the entire time series (no shrinkage to the mean in the terminal years). Using this revised model configuration (ASAP_final_temp1), the ASAP model was further refined in an iterative fashion.

<u>ASAP_final_temp2 model</u>: Next, the CVs on the fishery catch were increased from the range of 0.05-0.15 used in the ASAP_BASE model (earlier years had higher CVs) to 0.10-0.20 in the ASAP_final_temp2 model (1977-1980=0.20, 1981-1988=0.15, 1989-2013=0.10). The low RMSE value in the ASAP_BASE model motivated this change. This change had minimal effects on the model results, but did increase the catch RMSE from 0.33 to 0.65. While the RMSE value was still less than 1, the revised CV levels approach the maximum level of uncertainty in the catch that is believable.

<u>ASAP_final_temp3 model</u>: The RMSE values on the fits to the survey indices were high (>1.5)

indicating an overfitting of the survey indices. The input survey CVs were initially set equal to the bootstrapped CVs presented in Table A.54. The bootstrapped CVs characterize the sampling error, but additional process error may be present in the survey indices that are not reflected in the bootstrapped CVs. In the ASAP_final_temp3 model, the CVs on the NEFSC spring and fall survey were increased by 0.3 and 0.2, respectively. The increase in survey CVs resulted in the model not fitting more recent survey observations as closely; this in turn impacted terminal estimates in the model with decreases in the estimated size of the 2010 and 2012 year classes and 2013 SSB and increases in 2013 fishing mortality. The RMSE values of the revised model were closer to 1.

ASAP_final_temp4 model: Comparison of input ESS values and model calculated ESS for both the catch-at-age and survey indices-at-age from the ASAP_BASE model indicated that a constant time series ESS value was not appropriate. In the ASAP_BASE model the input ESS on catch was set at 80 and the NEFSC spring and fall trawl surveys were set at 15 and 20, respectively. Using an iterative approach the input ESS values were adjusted in stanzas (5-14 year blocks) to approximately match the model calculated ESS. These adjustments had small impacts on the model results. It should be noted that an attempt was made to adjust the ESS using the stage-2 multiplier approach described in Francis (2011). However, the resulting ESSs were extremely small (e.g., catch ~ 20, surveys ~5) indicating that the information content of the age data was unreasonably low given the strong cohort signals present in the Gulf of Maine haddock data. This approach was not pursued further.

ASAP_final_temp5 model: This model run attempted to address some of the boundary solutions achieved in the selectivity-at-age estimation in the ASAP_BASE model by increasing the fully selected age in the catch selectivity blocks from age-7 to age-8. The fully selected age in the spring survey was adjusted from age-6 to age-4 and the in the fall survey from age-6 to age-8.

ASAP_final_temp6 model: The ASAP_final_temp5 model was still hitting a bound at 1 for the estimated age-9^+ selectivity in blocks 1 and 2 of the catch. In this step, the selectivities were fixed at 1 for these ages (flat top selectivity). A comparison between the selectivity-at-age estimates from the ASAP_BASE and ASAP_final_temp6 models is provided in Table A.84.

The ASAP_final_temp6 model was the preferred model brought forward to the SAW 59 WG. During the SAW 59 WG discussions, the WG remained concerned about the high CV applied to the recruitment deviations (2.0) and the lack of constraint this value provided. In particular, the WG expressed concern over the size of the 2012 year class in the ASAP_final_temp6 model – at 21.5 million fish it would be the largest year class in the assessment time series and more than 50% larger than the 1998 year class, the second largest year class. The WG felt a more conservative approach was warranted given that the size of the 2012 year class was based on only two survey observations (2013 age-1 spring and fall indices-at-age). Using the ASAP_final_temp6 model, the WG revisited some of the early recruitment sensitivities performed on the ASAP_BASE model to evaluate model results at lower CVs and the impacts of 'shrinkage' to the mean on terminal recruitments. Specifically, three different model sensitivities were conducted:

ASAP_final_temp7: The CV on the recruitment deviations was reduced from 2.0 to 1.0.

ASAP_final_temp8: The CV on the recruitment deviations was reduced from 2.0 to 1.0 for the years 1977 to 2010 and to 0.5 for the years 2011 to 2013.

ASAP_final_temp9: The CV on the recruitment deviations was reduced from 2.0 to 1.0 for the years 1977 to 2012 and to 0.5 for 2013.

The results from these sensitivity models are summarized in Table A.85. Overall, the WG was more comfortable with the lower CV on the recruitment deviations; however, the WG felt that the ASAP_final_temp8 ('shrinkage' on 2011-2013 recruitment estimates) put too much constraint on the 2010 year class estimate given that the model had six survey observations in addition to catch information, and therefore should have sufficient information with which to achieve a reliable estimate. The ASAP_final_temp9, which placed a lower CV on only the 2013 estimate (2012 year class at age-1), offered a compromise between the constant CV and 'shrinkage' approaches. Compared to the ASAP_final_temp7 model (constant CV of 1.0) the ASAP_final_temp9 model reduced the size of both the 2012 (46%; 16.7 million to 9.0 million fish) and 2010 (15%; 6.7 million to 5.7 million fish) year classes, and had lower SSB and higher F in 2013.

The WG acknowledged that both the ASAP_final_temp7 and ASAP_final_temp9 models were equally plausible and noted that the size of the 2012 year class represents the largest source of uncertainty in this assessment. However, for the purposes of selecting a 'preferred' model for use in determining stock status, the WG selected the ASAP_final_temp7 as the best option. The WG examined preliminary reference points based on both models and concluded that stock status determination was robust to model selection. The WG recommended that the approach used in the ASAP_final_temp9 model be carried through to catch projections to more fully capture assessment uncertainty (see TOR 7).

Prior to finalizing model selection, the WG made several minor adjustments to both the ASAP_final_temp7 and _temp9 model runs to address concerns with model estimated survey selectivity. There was concern among WG members that the 'saw tooth' nature of the estimated survey selectivity was biologically unrealistic (see Table A.84, the selectivity patterns of ASAP_final_temp7 and _temp9 were identical to those of the temp6 model shown). The WG opted to model survey selectivity as flat-topped with selectivity fixed at 1 for ages-4 and older in the spring survey and ages-6 and older in the fall survey. All other ages were freely estimated. This is identical to the approach taken in the SCAA models discussed later in this TOR. The changes in selectivity had only minor impacts on the model results (Table A.85).

The final models put forward by the SAW 59 WG are:

ASAP_final_temp10 (*preferred*): Recruitment CV set at 1.0 for the entire time series.
ASAP_final_temp11 (*projection sensitivity only*): Recruitment CV set at 1.0 for the years 1977 to 2012 and to 0.5 for 2013.

Diagnostics and results of the preferred ASAP model

RMSE values are improved over those of the ASAP_BASE model. The RMSE of the spring and fall surveys are within the 80% confidence interval of the root mean square error from a normal distribution (Fig. A.169). While the catch RMSE value is well below 1, this is common when fitting catch information and generally, not a concerning model diagnostic. Fits to catches early in the time period were variable, and expected given the larger CV placed on these catches owing to their higher uncertainty. There is some patterning of residuals over time, however the residuals are small and overall, there is close agreement between modeled and observed catches (Fig. A.170).

The ESS adjustments made to the ASAP_BASE model show improved agreement between the input ESS and model calculated ESS (Fig. A.171). Catch ESS varied from 20 in the 1988 to 1992 period to 140 between 2003 and 2013. Overall, fits to the observed catch-at-age were good (Fig. A.172.a-c), with very little residual patterning (Fig. A.173). Fits to the mean catch-at-age suggest that the catch-at-age are reasonably well estimated (Fig. A.174) though there are large residuals between 1989 and 1992 corresponding with a period of low catches and reduced biological sampling.

Fishery selectivities were flat-topped in the first (1977-1988) and second (1989-2004) selectivity blocks, but there is evidence of a slight dome in the third block (2005-2013) (Fig. A.175). As discussed in Appendix 2, the doming in block 3 may reflect the increasing contribution of the recreational fishery to the total catch late in the time series.

Similar to the ASAP_BASE model, the ASAP_final_temp10 model tracks the spring survey index well with no strong residual patterning (Fig. A.176). The adjusted ESS inputs agree with the model calculated ESS values (Fig. A.177). The spring survey input ESS varied from 5 in the 1987 to 1992 period to 25 between 2005 and 2013. There was no concerning patterning of indices-at-age residuals, and overall the model fit the indices-at-age well (Fig. A.178). With the exception of three large outliers (1987, 2002, 2010) the fits to the mean indices-at-age are reasonable (Fig. A.179).

Similarly, the fall survey index was fit well by the model (Fig. A.180). The adjusted ESS inputs agree with the model calculated ESS values (Fig. A.181), with fall ESS varying from 7 in the 1984 to 1997 period to 30 between 1998 and 2008. As with the spring survey indices-at-age, there was little patterning of indices-at-age residuals, with residuals being small overall (Fig. A.182). With the exception of four large outliers (1988, 1992, 2007, 2012) the fits to the mean indices-at-age are reasonable (Fig. A.183).

As with the ASAP_BASE model, the survey selectivities estimated in the ASAP_final_temp10 model show the NEFSC having greater selectivity for younger fish compared to the fall survey (Fig. A.184). The selectivities are generally well estimated with CVs less than 0.20 for most ages (Table A.84). The estimated survey catchability (q=0.25) is nearly identical to the ASAP_BASE model, though the fall survey catchability is about 7% lower (q=0.92) (Fig. A.185). As was discussed previously, while the fall q value approaches 1.0, given possible herding behavior and uncertainty in the true catchability of the survey gear, there is little indication of scaling concerns.

Recruitment of Gulf of Maine haddock is highly episodic. Since 1977 there have been several strong recruitment events. Excluding the 2012 year class, the 1998 year class is the largest observed year class within the assessment time series, estimated at approximately 13.5 million fish (Table A.86, Fig. A.186). The 1998 year class has persisted in the population and lead to a large buildup of fish in the 9^+ age class (Fig. A.187). More recently, moderate-to-large year classes were spawned in 2003 and 2010. The 2003 year class has just recently entered the 9^+ age class. The size of 2010 year class is particularly important in explaining the increase in stock size compared to the previous assessment update in 2012. For the 2012 AOP assessment, the 2010 year class represented the *t+1* year class (age-1 in 2011, one year beyond the 2010 terminal year of the assessment time series). Due to the limited amount of information available to estimate this year class (a single spring survey observation), this year class was estimated using the geometric mean (see Palmer et al. 2014a for a full discussion). The current estimated size of the 2010 year class is 6.7 million fish, approximately six times larger than the size assumed for the 2012 AOP update. The 2012 year class is estimated to be large, though the actual size of the year class is highly uncertain and represents the greatest area of uncertainty in this assessment.

Total SSB has ranged from 600 mt to 15,178 mt during the assessment time period, with current SSB in 2013 estimated at 4,153 mt (Table A.87, Fig. A.188). Total January 1 biomass in 2013 is estimated at 7,749 mt, with 2,158 mt of exploitable biomass. Fully recruited fishing mortality has ranged from 0.19 to 1.54 (Table A.88). The low fishing mortality on ages-1 through 5 is notable given that the maturity $A_{50\%}$ is approximately 2.4 (Fig. A.13); the current fishery selectivity pattern allows for two to three spawning events on average prior to entering the fishery. The fully recruited fishing mortality in 2013 is estimated at 0.39.

A retrospective analysis of model performance over the years 2006-2013 indicates retrospective error for both F and SSB of 0.30 and -0.15, respectively. The retrospective patterns show a model tendency to underestimate SSB and overestimate F in the earlier peels; more recently, the retrospective error has switched signs (Fig. A.189), suggesting a transient nature to the retrospective error. Overall, the retrospective error is small and the SAW 59 WG recommended that no adjustments be made for the retrospective error when determining stock status determination or when conducting stock projections.

The Hessian-based CVs on SSB have generally been around 0.10 for the majority of the assessment time series, with the exception of the terminal years where CVs increase to around 0.20 (Fig. A.190). Fishing mortality CVs were moderate (>0.20) early in the time series, but decreased over time below 0.15 until about 2010 before increasing at the end of the time series. Age-1 recruitment CVs have been highly variable with the smallest year classes having the greatest degree of uncertainty. The CVs on the 1998 and 2003 year classes are < 0.10 indicating that they are reasonably well estimated. The 2010 year class is less certain with a CV around 0.22 and the 2012 year class is highly uncertain with a CV >0.3. The WG discussed the Hessian-based CVs for the 2012 year class and felt that this value likely does not adequately capture the true uncertainty of this year class.

A MCMC simulation was performed to obtain posterior distributions of the SSB, total B, and F_{full} time series based on 1000 MCMC chains. Two MCMC chains of initial length of five million were simulated with every five thousandth value saved. The trace of each chain's saved draws suggests good mixing and a sufficient burn-in period (e.g., Fig. A.191 and A.192). The lagged autocorrelations showed decreasing correlation with increased lag with correlations ≤ 0.1 beyond lag-0 indicative of a well mixed MCMC chain (Fig. A.193 and A.194). From the MCMC distributions, 90% posterior probability intervals (PI) were calculated to provide a measure of uncertainty for the model point estimates. Time series plots of the SSB and F_{full} 90% PIs as well as plots of the posterior probability distributions for the SSB, January 1 biomass and F_{full} are shown in Figures A.195 through A.197. The 2013 ASAP_final_temp10 point estimates and the 90% PIs are reported in Table A.89.

The results from the SAW 59 preferred ASAP_final_temp10 model are similar to the model results from the initial ASAP_BASE model (Fig. A.198). The most notable differences are a small negative re-scaling of age-1 recruitment and SSB in the ASAP_final_temp10 model compared to ASAP_BASE. This in turn translates to a slight positive re-scaling of fully recruited fishing mortality.

Description of SCAA model results

Recent reviews of historical and contemporary tagging studies suggest that there is movement of fish between the Gulf of Maine and Georges Bank stocks, though there is considerable uncertainty regarding the degree of mixing. The SAW 59 WG evaluated the results from three sensitivity models which used the SCAA statistical catch-at-age methodology (described in depth in Appendix 3). The first of the SCAA models considers the haddock in the Gulf of Maine to be an isolated stock (SCAA no movement model), which is identical to the WGs preferred ASAP model. The other two models incorporate movement into that area, either permanent or temporary, by haddock from Georges Bank. The WG concluded that the most biologically realistic mixing scenario is one that allows for non-permanent interchange (mixing) between the stocks.

The SCAA permanent migration model estimates the annual proportion of Georges Bank fish moving into the Gulf of Maine region at 0.2% where as the SCAA sabbatical model (non-permanent interchange) estimated the movement at 0.75% annually. The statistical evidence for such movement from these analyses point to scenarios involving limited movement being of similar plausibility to that of an isolated stock; however, mixing amongst the stocks has limited impact on assessment results. All three of the SCAA models achieved similar results to the ASAP_final_temp10 model (Fig. A.199).

The SAW 59 WG discussed how to interpret the mixing parameter estimates coming from the SCAA movement models. The SCAA movement models do not incorporate specific information to inform the model about migration rates (e.g., tagging); as such, the mixing parameters don't represent actual mixing rates, rather the mixing parameters represent upper bounds on the amount of mixing that could be supported by the data. The mixing parameters are confounded by other parameters or data observation/process error. It's unclear how well the SCAA mixing models would perform on simulated data sets from an isolated population – i.e., would the movement models still estimate a non-zero mixing parameter? **Ultimately, the WG supported**

the use of the ASAP_final_temp10 model as the 'preferred' model, but felt that the SCAA projection results should be carried forward as sensitivities to inform catch advice decisions.

The 59th SARC supported the use of the ASAP_final_temp10 model as the preferred model on which to determine stock status and base management advice.

Historical assessment retrospective

A comparison between the results of the final ASAP model to the results of the 2012 AOP and GARM III VPA model is provided in Figure A.200. As discussed earlier when comparing the VPA and ASAP results, the fishing mortality patterns coming out of the two models are different, though these differences are primarily restricted to the pre-2000 period. It should be noted that the comparison of fishing mortality trends between the various assessments are not directly comparable because the calculation basis of the current ASAP model is not identical to the previous VPA models. The VPA outputs reflects an average F over ages 6 to 8 where as the ASAP output reflects the fully recruited fishing mortality. Both model types indicate high fishing mortality early in the time series until about 1998, after which fishing mortality has remained low. The scale and trends of population numbers and biomass has been consistent from assessment to assessment. The 2012 AOP estimates of the 1998 year class were slightly higher than the GARM III and the current SAW/SARC 59 estimates, but these estimation differences were minimized as the 1998 year class aged and contributed less to the overall Gulf of Maine haddock population. This historical "retrospective" examination of past model performance illustrates the general stability of the Gulf of Maine haddock assessment results.

TOR A.5. State the existing stock status definitions for "overfished" and "overfishing". Then update or redefine biological reference points (BRPs; point estimates or proxies for BMSY, BTHRESHOLD, FMSY and MSY) and provide estimates of their uncertainty. If analytic model-based estimates are unavailable, consider recommending alternative measurable proxies for BRPs. Comment on the scientific adequacy of existing BRPs and the "new" (i.e., updated, redefined, or alternative) BRPs.

The existing MSY reference points based on a spawning potential ratio (SPR) of 40% were established at GARM III (NEFSC 2008) and updated as part of the 2012 AOP update (NEFSC 2012). The inputs to the yield per recruit (YPR) analysis assumed fishery selectivity to be 'flat-topped' beyond the fully selected age (age-7), mean weight and partial recruitment patterns were calculated from an unweighted average of the most recent five years. Maturity and natural mortality were assumed to be time invariant. The overfishing definition was $F_{MSY\text{-}proxy} = F_{40\%} = 0.46$.

Maximum sustainable yield and SSB_{MSY} were derived from the median values of long-term projections (100 years) of the Age Structured Projection Model (AGEPRO, NOAA Fisheries Toolbox, http://nft.nefsc.noaa.gov/) run at a constant harvest of $F_{40\%} = 0.46$. Input vectors for the AGEPRO runs are the same as those used for the YPR analyses. Following on the methods employed in the GARM III assessment, projected recruitment was determined using the cumulative density function (CDF) of a recruitment series that included both VPA-estimated age-1 recruitment and hindcast recruitment estimates based on NEFSC fall bottom trawl survey age-1 indices. A linear regression was fit to VPA estimates of age-1 recruitment and NEFSC autumn bottom trawl survey indices of abundance of age-1 fish. Using the regression relationship, recruitment was estimated back to the 1962 year class. The 2008 GARM BRP Panel recommended a recruitment series that includes VPA estimated recruitment excluding recruitment estimates for years when SSB was less than 3,000 mt in addition to hindcast recruitment from 1962 to 1976 with the large 1962 year class removed (considered a "bonanza" outlier). The resulting BRP estimates were: SSB_{MSY} = 4,904 mt (90% CI of 2,272 – 10,604 mt), and MSY = 1,117 mt (90% CI of 553 – 2,563 mt). A stock is considered to be overfished if spawning biomass is less than half of SSB_{MSY}; the existing overfished definition is ½ SSB_{MSY} = 2,452 mt.

New reference points are warranted given the changes in data inputs and the assessment model, as well as small changes in the fishery selectivity and weights-at-age.

Ultimately, the WG concluded that because Gulf of Maine haddock recruitment events are highly episodic and not well described by traditional stock recruitment relationships, a MSY proxy approach to reference points was warranted. This is the same conclusion reached at GARM III. A yield per recruit analysis was conducted using 2009-2013 period as representative of future conditions. The WG reached this decision after an inspection of the weights-at-age noting that over this most recent five year periods weights have remained relatively stable. The inputs to the YPR analysis included the time invariant maturity ogive, the time and age invariant natural mortality value (M=0.2), the selectivity-at-age from the third selectivity block as well as the average catch and stock weights from 2009-2013 (Table A.90). The SAW/SARC 59 YPR inputs were not considerably different from those from the 2012 AOP update (Fig. A.201).

A stochastic YPR analysis which incorporated the empirical CVs of the input vectors (natural mortality CV assumed = 0.1) was conducted to better characterize the uncertainty in the proxy fishing mortality rate. After an examination of the YPR results (Table A.91), the WG saw no compelling reason to select a different F_{MSY} proxy than the $F_{40\%}$ metric that had been adopted previously. Because of the similarities in YPR inputs between the 2012 AOP and the current assessment, it is not surprising the $F_{40\%}$ values were identical: $F_{40\%}$ = 0.46 (90% CI of 0.36 – 0.54) (Table A.92).

Stochastic long-term projections (100 years) at $F_{40\%}$ were used to determine new recommended biomass-related reference points (proxies for both SSB_{MSY} and MSY). The projection inputs were identical to the YPR inputs.

The WG discussed various ways to project future recruitment. It found the GARM III method to be arbitrary (e.g., excluding very large and very small recruitment events) and instead opted to use a more straightforward method of using the CDF of the 1977-2011 age-1 recruitments as estimated by the preferred ASAP_final_temp10 model. The 2012 and 2013 recruitment observations were not included in the CDF due to the overall uncertainty in these estimates. *The WG did conduct long-term projections using the full 1977-2013 recruitment series, but these are* *for comparative purposes only.*

To approximate the distribution of the SSB and MSY distributions, the long term projections were made from 1000 estimates of numbers at age in 2014, which were estimated by performing MCMC simulation of the ASAP_final_temp10 model (described above under TOR 4). The 2014 age-1 estimates (*t+1*) were based on sampling from the empirical distribution of recruitment estimates from the full assessment times series (1977-2013). Long-term projections are insensitive to the 2014 starting numbers-at-age. All projections were conducted with the AGEPRO software (Age Structured Projection Model v4.2.2).

The resulting biomass reference points and their 90% CI corresponding to $F_{MSY\text{-}proxy} = F_{40\%} =$ 0.46 are SSB_{MSY} = 4,108 mt (1,774 – 7,861 mt) and MSY = 955 mt (421 – 1,807 mt). Table A.93 provides a comparison to the sensitivity reference points based on the CDF of 1977-2013 age-1 recruitments. The overfished biomass threshold is ½ SSB_{MSY}, or 2,054 mt.

The SAW 59 WG did discuss other methods for developing reference points for Gulf of Maine haddock. One such approach attempted to examine changes in stock productivity using a Ricker stock recruitment curve with a time-varying productivity parameter (Bell and Hare 2014). The goal of the approach was to examine if there were trends in stock productivity that could be used to inform forecasts of future abundance and in turn inform management decisions. The WG discussed this approach as well other similar approaches (e.g., time-varying/random walk parameter estimation, autoregressive processes) and noted that there was a need to first identify the underlying processes driving productivity shifts. Additionally, the WG noted that additional work was needed before these methods could be incorporated in the stock assessment process. Specifically, the WG noted that in developing these approaches, the robustness and utility of incorporating additional stock-recruitment models needs to be considered (see TOR 8).

TOR A.6. Evaluate stock status with respect to the existing model (from previous peer reviewed accepted assessment) and with respect to a new model developed for this peer review. In both cases, evaluate whether the stock is rebuilt (if in a rebuilding plan).

TOR A.6.a. When working with the existing model, update it with new data and evaluate stock status (overfished and overfishing) with respect to the existing BRP estimates.

The existing reference points are $F_{MSY\text{-}proxy} = F_{40\%}$ = 0.46, SSB_{MSY} = 4,904 mt (90% CI of 2,272 – 10,604 mt) (½ SSB_{MSY}, or 2,452 mt), and MSY = 1,117 mt (90% CI of 553 – 2,563 mt). The updated VPA model (Model 6, 2013_UPDATE) estimates 2013 SSB at 3,070 mt. This exceeds the existing overfished threshold of 2,452 mt; therefore, the stock is not overfished. The updated estimate of average fishing mortality on ages 6-8 ($F_{6\text{-}8}$) in 2013 is 0.82. This is greater than the overfishing limit of 0.46, and therefore, overfishing is occurring.

TOR A.6.b. Then use the newly proposed model and evaluate stock status with respect to “new” BRPs and their estimates (from TOR-5).

The revised reference points are $F_{MSY\text{-}proxy} = F_{40\%} = 0.46$ (90% CI of 0.36 – 0.54), SSB_{MSY} = 4,108 mt (90% CI of 1,774 – 7,861 mt) (½ SSB_{MSY}, or 2,054 mt), and MSY = 955 mt (90% CI of 421 – 1,807 mt). The ASAP_final_temp10 model estimates 2013 SSB at 4,153 mt. This is greater than the SSB_{MSY} level of 4,108 mt; therefore, the stock is rebuilt and not overfished. The estimate of 2011 fully recruited fishing mortality (F_{full}) is 0.39. This is less than the overfishing limit of 0.46, and therefore, overfishing is not occurring. The stock status determination is robust to model uncertainty (Fig. A.202).

TOR A.7. Develop approaches and apply them to conduct stock projections and to compute the statistical distribution (e.g., probability density function) of the OFL (overfishing level) (see Appendix to SAW TORs for definitions).

TOR A.7.a. Provide numerical annual projections (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. Use a sensitivity analysis approach in which a range of assumptions about the most important uncertainties in the assessment are considered (e.g., terminal year abundance, variability in recruitment, migration from Georges Bank).

Identical to the long-term projections used to determine SSB_{MSY} and MSY proxies, the short-term (2015-2017) projection method samples from a cumulative density function derived from ASAP estimated age-1 recruitment between 1977 and 2011. Age-1 recruitments in 2012 and 2013 were not included in the cumulative density function due to their greater variance. Note that the 2014 age-1 estimates ($t+1$) were based on sampling from the empirical distribution of recruitment estimates from the full assessment times series (1977-2013). The WG did examine the sensitivity of the short-term projections to variability in 2014-2017 recruitment assumptions and found results to be robust to the out year recruitment assumptions due to the small differences in median recruitment levels between the two assumptions (1.1 vs. 1.2 million fish, Table A.86) and the limited contribution of these year classes to the spawning stock biomass and fishery yield within the projection window. No retrospective adjustment needed to be applied in the projections.

Due to the high degree of uncertainty of the size of the 2012 year class, two projection models were developed. The first is based on the preferred population model (ASAP_final_temp10) and the second is based on a sensitivity model that constrained the size of the 2012 year class (ASAP_final_temp11). Both projection models were run under two different assumptions of calendar year 2014 catch – harvest at FMSY (0.46) and an assumed 2014 catch of 500 mt. The fishing year 2014 Gulf of Maine haddock Annual Catch Limit (ACL) is set at 323 mt, though the ACL does not account for recreational discards. The 500 mt estimate used in the projections was informed by the fishing year 2014 ACL and recent recreational discard amounts. Because fishing mortality is not allowed to exceed the overfishing limit (i.e., $F_{40\%}$), these projections provide an approach for defining the OFL. Results for the four projections (two models, each with two different 2014 catch assumptions) are provided in table A.94.

The New England Fishery Management Council's Scientific and Statistical Committee have traditionally applied a 75% control rule when recommending Acceptable Biological Catch

(ABC) levels. The above projections were repeated with harvest set at 75% $F_{MSY\text{-}proxy}$ (0.35) to demonstrate example projections for establishing ABC levels. Results for the four projections are provided in table A.95.

Recent reviews of historical and contemporary tagging studies suggest that there is movement of fish between the Gulf of Maine and Georges Bank stocks, though there is considerable uncertainty regarding the degree of mixing. Several lines of evidence examined during the SAW/SARC59 assessment indicate that annual percent mixing from Georges Bank to the Gulf of Maine is low (<0.8%), though the mixing scenarios have similar statistical plausibility to that of an isolated stock. While mixing amongst the stocks has limited impacts on stock status, catch projections of the SCAA models (Appendix 3) under constant fishing mortality were found to be sensitive to limited movement for the case where the movement is permanent (SCAA permanent migration model), but much less so when movement was modeled as non-permanent interchange (SCAA sabbatical model). The WG concluded that of the two mixing scenarios, the most biologically realistic is one that allows for non-permanent interchange between the stocks. The catch projection results from the most realistic SCAA mixing model (i.e., allows mixing between stocks as opposed to unidirectional movement) are nearly identical to the SCAA model with no mixing, with both being within the 90% confidence intervals of the projections from the preferred ASAP model (Figure A.203).

The SAW 59 WG noted that the evidence for mixing is not conclusive and that the mixing scenarios have similar statistical plausibility to that of an isolated stock. **Given this, it concluded that the projections based on the ASAP_final_temp10 model should be used as the preferred model for management advice. This decision was supported by the 59th SARC.**

TOR A.7.b. Comment on which projections seem most realistic. Consider the major uncertainties in the assessment as well as sensitivity of the projections to various assumptions.

The SAW 59 WG determined that the projections based off the ASAP_final_temp10 model were the 'most realistic'. However, it has stressed that the absolute size of the 2012 year class is the largest source of uncertainty in this assessment. The risks associated with management actions taken during 2015 – 2017 were examined by undertaking stock projections under two different assumptions of year class size. Under both scenarios, the spawning stock biomass is projected to increase well above the target levels and catch can be sustained above MSY levels.

The differences in these two short-term projections in 2014 and 2015 are primarily due to the differences in the size of the 2010 year class between the two different models. However, as the projection horizon increases, and the contribution of the 2012 year class becomes more important, the divergence in catch advice becomes larger (> 600 mt). Based on the estimated selectivity patterns, the 2012 year class is predicted to be 50% selected by the fishery in 2017 at age-5. Recent changes to the commercial minimum retention size may result in this year class recruiting to the fishery sooner.

The assumption of the catch in 2014 will have limited impacts on stock size and catch advice in the subsequent years, though the two assumed values (catch= $F_{MSY\text{-}proxy}$ and 500 mt) should be re-evaluated once additional information on 2014 catches are available.

TOR A.7.c. Describe this stock's vulnerability (see "Appendix to the SAW TORs") to becoming overfished, and how this could affect the choice of ABC.

There are several factors that should be considered when setting catch advice for the Gulf of Maine haddock stock. While these uncertainties have been discussed previously in this report, particular attention should be given to the factors below when determining the appropriate level of scientific uncertainty to prescribe to this stock assessment.

The mortality of haddock discarded in the recreational and commercial fishery is unknown. For trawl and gillnet gear, mortality is likely high and not substantially different than the assumption of 100% used in the assessment. While there is limited information available to suggest that mortality of haddock discarded in the commercial longline fishery may be lower than 100%, given the small magnitude of longline removals, the impacts of this assumption on the assessment results are likely small. However, given the large amount of recreational discards occurring in recent years, the model results and subsequent catch advice could be sensitive to the assumption of 50% discard mortality used in this assessment. While the assessment results were shown to be relatively insensitive to this assumption, it does have implications for management and catch allocation between the commercial and recreational fleets.

Several lines of evidence examined during the SAW/SARC59 assessment indicate that annual percent mixing from Georges Bank to the Gulf of Maine is low (<0.8%), though the mixing scenarios have similar statistical plausibility to that of an isolated stock. While the catch projections for the more biologically realistic mixing scenario (non-permanent interchange) were nearly identical to no-movement assumptions, the projections which assumed permanent movement of Georges Bank haddock into the Gulf of Maine were higher than the no movement scenarios. Setting catch advice higher on the presumption that permanent movement of Georges Bank haddock into the Gulf of Maine is occurring, if in fact it is not, could lead to overfishing of the Gulf of Maine stock (NEFMC GPDT 2013).

The absolute size of the 2012 year class is the largest source of uncertainty in this assessment. Based on the estimated selectivity patterns, this year class is predicted to be 50% selected by the fishery in 2017 at age-5. Recent changes to the commercial minimum retention size may result in this year class recruiting to the fishery sooner. Given the high uncertainty with respect to this year class size, the assessment should be updated if future estimates of its size differ significantly from those used in this assessment.

TOR A.8. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in most recent SARC reviewed assessment and review panel reports. Identify new research recommendations.

The SAW 59 WG reviewed the status of previous research recommendations and proposed new ones to address issues raised during the WG meeting.

GARM III

- Inverse variance weighting should be investigated as a means to compute the current year's fishing mortality as it has superior statistical characteristics than either the unweighted or weighted (by population) numbers.
  - This research recommendation is no longer relevant for the Gulf of Maine haddock assessment due to the switch from a virtual population analysis assessment model to a statistical catch-at-age model.

- Research should be undertaken on the estimation of the survivorship of haddock released in the recreational fishery.
  - This research recommendation has been partially addressed through the estimation of recreational discards described under TOR 1 and through the model sensitivity runs explored under TOR 4. Directed field studies are needed to better inform the assessment.

SAW 59 WG

- In the Northeast Region, frequent changes in management and the multispecies nature of the fishery hinder the ability to develop useful indices of abundance from fishery data. In stock assessments over the last decade, these problems have resulted in the development of standardized CPUE indices that have no demonstrated utility as indices of stock abundance for assessments of cod, haddock, white hake, yellowtail flounder, summer flounder, scup, or winter flounder. The qualitative properties of the fishery data are generally well described in the assessments. The SAW 59 WG recommends investigation of approaches to consider year/area and other interactions (e.g., at a finer scale than statistical area) in the hopes of developing more useful fishery-based indices of abundance. Given the considerable investment of time that may be required and its potential utility for a range of assessments, this work may be best pursued as research outside the Terms of Reference for any single stock assessment.

- Develop approaches for and evaluate the robustness and utility of incorporating additional stock-recruitment models (e.g., Ricker, time-varying/random walk parameter estimation, autoregressive processes) into the population models used in Northeast Region assessments.

- Develop approaches for and evaluate the robustness and utility of incorporating autoregressive error (e.g., AR1 processes in recruitment and catch-at-age resulting from sampling) in fishery and survey data into the population models used in Northeast Region assessments.

- The SAW 59 WG notes the further advice from the Council SSCs is needed to advance the application of multi-model inference and risk evaluation in Northeast Region stock assessments.

- Practical and logistic problem aside, in the future it would be beneficial to conduct the multiple Northeast Region haddock assessments at the same time, to facilitate comparability of the data and analytical results, especially if models include movement between the stock units.

- The haddock tagging experiments conducted to date were not designed to address the issue of between-stock movement rates. Research designed to expressly determine between-stock movement rates is needed to reduce the uncertainty of analytical models that include these rates.

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Tables

Table A.1. Summary of model inputs and formulations used to assess the Gulf of Maine haddock stock since 2002. There was no accepted analytical assessment prior to 2002.

Year	Meeting	Model	Starting year	Catch data series				Survey series		Plus group
				Commercial landings	Commercial discards	Recreational landings	Recreational discards	NEFSC fall	NEFSC spring	
2002	GARM I	AIM	1963	1963-2001				1963-2001		N/A
2005	GARM II	AIM	1963	1963-2004				1963-2004		N/A
2008	GARM III	VPA	1977	1977-2007	1977-2007	1982-2007		1977-2007	1977-2008	9+
2012	AOP	VPA	1977	1977-2010	1977-2010	1982-2010		1977-2010	1977-2011	9+

Table A.2. Summary of the results of the Gulf of Maine haddock assessments since 2002 and the resulting stock status determinations based on the biological reference points at the time of the assessment. *Notes:* [1]*F_{ref} = relative F where replacement ratio = 1, B_{ref} = MSY/F_{ref}. Replacement ratio: the biomass index in the current year divided by the average biomass indices from a 3 year centered mean.* [2]*YPR = Yield per recruit, based on 5yr averages of WAA, MAA and partial recruitment, F SPR basis = 40%.* [3]*The stock was projected to become overfished in the first year of the projections*

Year	Meeting	$SSB_{terminal}$	Exploitation rate/$F_{terminal}$	Exploitation rate/F note	Reference point basis	B_{ref}	F_{ref}	MSY_{proxy}	Stock status
2002	GARM I	10.31 kg/tow	0.12	Landings/3-yr fall survey avg.	MSY derived avg. 1959-1966 landings[1]	22.17 kg/tow	0.230	5,100 mt	Overfished, overfishing not occuring
2005	GARM II	5.79 kg/tow	0.18	Landings/3-yr fall survey avg.	MSY derived avg. 1959-1966 landings 1	22.17 kg/tow	0.230	5,100 mt	Overfished, overfishing not occuring
2008	GARM III	5,850 mt	0.35	$F_{avg6-8, N-Weighted}$	YRP[2]	5900 mt	0.430	1,360 mt	Not overfished, overfishing is not occuring
2012	AOP	2,868 mt	0.82	F_{avg6-8}	YRP[2]	4,904 mt	0.460	1,177 mt	Not overfished, overfishing is occuring[3]

Table A.3. Summary of major regulatory actions that have affected the Gulf of Maine haddock fishery since 1973. For a more detailed summary of regulatory actions, see Nies (2011).

Date	Regulatory action	Cod end minimum mesh size (in)	Miscellaneous	Closures	Differential DAS Counting
01/01/73		4.5			
01/01/77	Groundfish FMP	5.125			
01/01/82					
01/01/83		5.5			
01/01/89					
04/01/92	Shrimp trawl fishery: Nordmore grate regulation, groundfish bycatch prohibited				
05/01/94	Amendment 5	6.0	500 lb trip limit		DAS monitory w/ reduction schedule, mandatory reporting
05/01/96	Amendment 7		Trip limit raised to 1,000 lb/trip		Accelerated DAS reduction
05/01/97	Framework 20				
09/01/97			Trip limits raised to 1,000 lb/day with 10,000/trip maximum		
05/01/98	Framework 25			WGOM (Jeffreys Ledge, Stellwagen Bank)	
06/25/98					
09/01/98			Trip limits raised to 3,000 lb/day with 30,000/trip maximum		
02/01/99	Framework 26			Additional month-block closures for February to April	
05/01/99	Framework 27	6.5 square/6.0 diamond	Trip limit lowered to 2,000 lb/day with 20,000 lb/trip		
05/28/99					
08/03/99	Interim rule				
11/05/99			Trip limit raised to 5,000 lb/day with 50,000 lb/trip		
01/05/00	Framework 31			Additional month-block closures for February	
06/01/00	Framework 33	6.5 square/6.5 diamond			
10/26/00			Daily trip limit removed, total trip limit of 50,000 lb/trip remains in effect		
11/01/00				One month closure of Cashes Ledge	
05/01/02	Interim rule		Trip limits lowered to 3,000 lb/day with 30,000/trip maximum	Additional month-block closures for May - June 2003; Cashes Ledge Closed year round	20% reduction in DAS
06/01/02	Revised interim rule				
07/01/02			Daily limit suspended, 30,000/trip through 9/30/2002 then 50,000/trip thereafter		
08/01/02	Emergency rule				
03/13/03			Haddock possession limit suspended until May 1		
05/01/03			Trip limits lowered to 3,000 lb/day with 30,000/trip maximum		
05/01/04	Amendment 13		Trip limits suspended for remainder of 2004	WGOM, Cashes Ledge and rolling closures continued	Further reduction in DAS
05/01/06	Emergency rule				
11/22/06	FW 42				DAS counted 2:1 in inshore GOM
08/15/06	FW 43		Haddock cap for herring fishery implmented (set at 0.2% of the combined GOM/GBK haddock TAC)		
05/01/09	Interim rule				
05/01/10	Amendment 16			Some changes to rolling closures for sector vessels	DAS counted in 24 -hour blocks; no differential DAS counting except as AMs
05/01/11	Framework 45			Whaleback closure April 1 - June 30 (commercial and recreational)	
09/14/11	Framework 46		Changes to herring haddock cap (1% GOM haddock ABC)		
05/01/12	Framework 47				
05/01/13	Framework 48		Changes to minimum sizes for both commercial and recreationalfisheries (effective July 1)		

Table A.4. Commercial and recreational fishery minimum retention size limits for Gulf of Maine haddock, from 1977 to 2013. Prior to 1977 there were no federal minimum size limits for either fishery. *Note that minimum sizes were changed throughout the year or corresponding to the start of the groundfish fishing year (May 1 – April 30), thus the year/length relationships are approximate.*

Year	Commercial minimum size limit (total length, inches)	Recreational minimum size limit (total length, inches)	Management action
1977	16	15	Groundfish Fishery Management Plan
1978	16	15	
1979	16	15	
1980	16	15	
1981	16	15	
1982	16	15	
1983	17	15	Interim Groundfish Fishery Management Plan
1984	17	15	
1985	17	15	
1986	17	15	
1987	19	17	Amendment 1
1988	19	17	
1989	19	19	
1990	19	19	
1991	19	19	
1992	19	19	
1993	19	19	
1994	19	19	Amendment 5
1995	19	19	
1996	19	19	
1997	19	19	
1998	19	19	
1999	19	19	
2000	19	19	
2001	19	19	
2002	19	23	Framework 33
2003	19	21	Framework 22
2004	19	19	Amendment 13
2005	19	19	
2006	19	19	
2007	18	19	Emergency action (August 10, 2007 through August 10, 2008)
2008	18	19	
2009	18	18	Amendment 16
2010	18	18	
2011	18	18	
2012	18	18	January 6, 2012-April 20, 2012 recreational set at 19 inches as part of AM
2013	16	21	Framework 48, implemented on July 1, 2013

Table A.5. Summary of the number of haddock otoliths sampled from the Northeast Fisheries Science Center (NEFSC) bottom trawl surveys from 1970 to 2013 by season, stock and age. Otoliths that have not been aged are not included in this summary.

Age	Gulf of Maine		Georges Bank	
	Spring	Fall	Spring	Fall
0	1	277		2039
1	541	491	2481	2676
2	542	608	3005	2307
3	505	623	3010	2433
4	342	500	1986	1593
5	262	379	1402	910
6	157	279	931	785
7	112	202	909	587
8	93	169	563	359
9	57	62	307	190
10	44	71	198	83
11	21	38	40	36
12	15	22	33	36
13	13	12	23	14
14	9	7	11	4
15		3	1	1
16			1	1
17			1	
18	1			

Table A.6. Number of Gulf of Maine haddock maturity samples taken from the Northeast Fisheries Science Center (NEFSC) spring survey from 1977 to 2013 by year.

Year	Unknown	Male	Female	Total
1977	6	62	81	149
1978	1	7	21	29
1979		6	13	19
1980	4	28	27	59
1981	12	49	54	115
1982		37	39	76
1983		24	40	64
1984		18	16	34
1985	2	25	38	65
1986		7	19	26
1987	1	1	3	5
1988		6	3	9
1989		7	3	10
1990		1		1
1991		1	3	4
1992	3	1	5	9
1993	1	11	7	19
1994	2	5	13	20
1995	1	5	15	21
1996		4	6	10
1997		26	34	60
1998		5	6	11
1999	18	26	33	77
2000	2	43	38	83
2001	3	35	34	72
2002	7	29	83	119
2003	3	43	71	117
2004	2	10	29	41
2005	1	16	16	33
2006	1	34	56	91
2007		18	19	37
2008		30	27	57
2009	1	33	83	117
2010		40	52	92
2011	5	29	52	86
2012	5	88	128	221
2013	19	167	178	364

Table A.7. Gulf of Maine haddock female maturity ogive. The time series average incorporated data collected the Northeast Fisheries Science Center (NEFSC) spring survey between 1977 and 2013.

Age	**Proportion mature**	**Lower 95% CI**	**Upper 95% CI**
1	0.04	0.02	0.05
2	0.28	0.24	0.33
3	0.81	0.77	0.84
4	0.98	0.97	0.99
5	1.00	1.00	1.00
6	1.00	1.00	1.00
7	1.00	1.00	1.00
8	1.00	1.00	1.00
9	1.00	1.00	1.00
10	1.00	1.00	1.00
11	1.00	1.00	1.00
12	1.00	1.00	1.00
13	1.00	1.00	1.00
14	1.00	1.00	1.00

Table A.8. Estimates of Gulf of Maine haddock catch (mt) by fleet (commercial, recreational) and disposition (landed, discarded) from 1977 to 2013. **Recreational discard estimates do not account for post-release mortality.** *Missing values indicate that estimates are not available for those years.*

Year	US recreational discards (mt)	US recreational harvest (mt)	US commercial discards (mt)	US commercial landings (mt)	Foreign landings (mt)	Total removals	
						100% mortality of recreational discards	50% mortality of recreational discards
1977				3,230.1	26.0	3,256.1	3,256.1
1978				4,382.5	641.0	5,023.5	5,023.5
1979				4,130.6	257.0	4,387.6	4,387.6
1980				6,317.6	203.0	6,520.6	6,520.6
1981	0.0	38.2		5,713.3	513.0	6,264.5	6,264.5
1982	0.0	23.0	6.4	5,634.3	1,278.0	6,941.7	6,941.7
1983	0.0	52.7	6.5	5,593.4	2,003.0	7,655.6	7,655.6
1984	0.6	52.3	11.0	2,792.8	1,245.0	4,101.7	4,101.4
1985	0.0	21.6	16.5	2,259.1	791.0	3,088.2	3,088.2
1986	0.2	51.8	16.4	1,628.9	225.0	1,922.3	1,922.2
1987	0.0	39.2	23.9	846.3	0.0	909.4	909.4
1988	1.3	20.1		418.0	0.0	439.4	438.8
1989	2.6	13.1	5.0	265.1	0.0	285.9	284.6
1990	0.1	5.3	2.0	465.0	0.0	472.4	472.4
1991	0.0	0.3	2.8	443.5	0.0	446.6	446.6
1992	0.0	0.0	8.0	313.4	0.0	321.4	321.4
1993	0.0	0.6	13.3	193.0	0.0	206.9	206.9
1994	0.9	3.3	61.1	121.9	0.0	187.1	186.7
1995	27.4	124.1	87.7	178.2	0.0	417.4	403.7
1996	6.4	5.7	78.2	253.8	0.0	344.2	341.0
1997	10.5	30.2	378.7	623.7	0.0	1,043.2	1,037.9
1998	7.0	45.6	16.6	922.6	0.0	991.9	988.4
1999	9.8	17.8	2.3	569.1	0.0	599.0	594.1
2000	60.4	128.1	27.9	799.3	0.0	1,015.7	985.5
2001	86.8	169.3	12.9	1,006.8	0.0	1,275.8	1,232.4
2002	177.3	135.3	18.6	1,009.2	0.0	1,340.4	1,251.8
2003	257.4	173.9	17.7	1,026.4	0.0	1,475.4	1,346.7
2004	72.9	312.6	11.7	947.2	0.0	1,344.4	1,307.9
2005	72.0	538.1	25.0	977.7	0.0	1,612.7	1,576.7
2006	131.0	447.4	31.5	622.5	0.0	1,232.4	1,166.9
2007	91.4	572.7	46.9	677.9	0.0	1,388.9	1,343.2
2008	144.1	536.6	10.3	542.7	0.0	1,233.6	1,161.6
2009	48.8	408.6	12.3	500.3	0.0	970.0	945.6
2010	37.1	314.0	3.0	622.6	0.0	976.7	958.1
2011	22.4	228.8	5.6	498.6	0.0	755.3	744.2
2012	107.3	251.2	17.7	416.6	0.0	792.7	739.1
2013	413.9	241.1	32.3	212.0	0.0	899.4	692.4

Table A.9. Historical estimates of Gulf of Maine haddock catch (mt) by fleet (commercial, recreational) and disposition from 1956 to 1976. *Estimates of both United States (US) and foreign fleet commercial landings are shown. No estimates of recreational catch or commercial discards are available in the historical period.*

Year	US recreational discards (mt)	US recreational harvest (mt)	US commercial discards (mt)	US commercial landings (mt)	Foreign landings (mt)	Total (mt)
1956				7278.0	29.0	7307.0
1957				6141.0	25.0	6166.0
1958				7082.0	285.0	7367.0
1959				4497.0	163.0	4660.0
1960				4541.0	383.0	4924.0
1961				5297.0	56.0	5353.0
1962				5003.0	107.0	5110.0
1963				4742.0	47.0	4789.0
1964				5378.8	70.0	5448.8
1965				4154.7	159.0	4313.7
1966				4524.0	1125.0	5649.0
1967				4852.2	589.0	5441.2
1968				3417.3	120.0	3537.3
1969				2404.6	290.0	2694.6
1970				1435.8	105.0	1540.8
1971				1190.2	112.0	1302.2
1972				912.2	27.0	939.2
1973				525.9	49.0	574.9
1974				628.8	207.0	835.8
1975				1180.2	83.0	1263.2
1976				1834.5	91.0	1925.5

Table A.10. Coefficients of variation (CV) associated with the landings allocation procedure (AA tables, Wigley et al. 2008) for Gulf of Maine haddock commercial landings.

Year	Overall	A-level		
		B	C	D
1994	0.01	0.03	0.04	0.27
1995	0.01	0.02	0.03	0.15
1996	0.01	0.03	0.03	0.14
1997	0.01	0.03	0.07	0.19
1998	0.01	0.04	0.09	0.13
1999	0.01	0.05	0.06	0.25
2000	0.01	0.03	0.08	0.26
2001	0.01	0.03	0.06	0.18
2002	0.01	0.02	0.07	0.26
2003	0.01	0.04	0.05	0.25
2004	0.01	0.03	0.10	0.06
2005	0.01	0.04	0.07	0.08
2006	0.00	0.02	0.05	0.07
2007	0.00	0.02	0.18	0.09
2008	0.03	0.08	0.24	0.16
2009	0.01	0.05	0.03	0.09
2010	0.02	0.08	0.06	0.25
2011	0.02	0.15	0.20	0.22
2012	0.04	0.15	0.22	0.14
2013	0.01	0.06	0.11	0.24
Average	0.01	0.05	0.09	0.17

Table A.11. Estimates of total United States landings of Gulf of Maine haddock associated with 'non-dealer' transactions from 1994 to 2013. These estimates are obtained from information reported on Vessel Trip Reports (VTRs).

Year	Future sale	Home consumption	Legal sized unmarketable fish (LUMF)	Sold/used for bait	Total	Total dealer reported landings	Percentage of reported dealer landings
1994		0.33			0.33	121.9	0.3
1995		0.81			0.81	178.2	0.5
1996		1.77			1.77	253.8	0.7
1997		0.74			0.74	623.7	0.1
1998		1.25			1.25	922.6	0.1
1999		0.54		0.00	0.54	569.1	0.1
2000		1.82		0.00	1.82	799.3	0.2
2001		2.42		0.01	2.43	1006.8	0.2
2002	0.27	2.56			2.83	1009.2	0.3
2003		2.82			2.82	1026.4	0.3
2004	0.62	2.12		0.02	2.77	947.2	0.3
2005	0.84	1.50		0.02	2.36	977.7	0.2
2006	0.23	1.61			1.89	622.5	0.3
2007		2.30			2.30	677.9	0.3
2008	0.11	0.82			0.93	542.7	0.2
2009	0.02	0.75			0.76	500.3	0.2
2010	0.16	1.66	0.01	0.01	1.85	622.6	0.3
2011	0.46	2.56	0.04	0.01	3.08	498.6	0.6
2012	0.33	1.93	0.01		2.26	416.6	0.5
2013	0.40	0.90	0.00	0.00	1.31	212.0	0.6
Average	0.35	1.56	0.02	0.01	1.74	626.5	0.3

Table A.12. Proportions of observed hauls of otter trawl gear in the Gulf of Maine by mesh type from 1989 to 2013.

Year	Proportion of observed hauls				
	Combination	Diamond	Square	Square/wrapped	Unknown
1989					1.00
1990					1.00
1991					1.00
1992					1.00
1993					1.00
1994		0.37			0.63
1995		0.88	0.12		
1996		0.89	0.11		
1997		0.94	0.06		
1998		0.53	0.48		
1999	0.02	0.43	0.25	0.30	
2000		0.46	0.14	0.40	
2001		0.33	0.20	0.47	
2002	0.01	0.41	0.40	0.18	0.00
2003	0.01	0.59	0.35	0.05	0.00
2004		0.34	0.64		0.01
2005	0.01	0.35	0.63	0.00	0.00
2006		0.49	0.48	0.00	0.03
2007	0.00	0.63	0.36	0.00	
2008		0.68	0.32		
2009		0.61	0.39		
2010	0.01	0.73	0.26		0.00
2011		0.74	0.26	0.00	
2012		0.59	0.41	0.00	
2013		0.67	0.33		

Table A.13. Summary of biological sampling and commercial landings of Gulf of Maine haddock snapper market category fish from 1971 to 2013.

Length samples (no. lengths)					Commercial landings (mt)				
Year	Quarter				Year	Quarter			
	1	2	3	4		1	2	3	4
1971					1971	0.52			
1972					1972				3.39
1973					1973	2.79	1.40	3.41	4.40
1974					1974	14.55	4.77		0.30
1975	64				1975	1.09	3.38	6.53	27.37
1976					1976	1.82	0.44	17.44	148.62
1977		155		52	1977	98.96	85.89	48.68	18.21
1978				89	1978	1.89	4.35	28.90	23.34
1979					1979	18.74	6.38	0.54	1.23
1980		68			1980	3.79	37.60	39.32	6.65
1981		316	406		1981	8.31	32.36	29.47	27.88
1982	221				1982	15.76	1.04	0.33	1.40
1983					1983	0.01	0.09	0.05	
1984					1984				0.20
1985					1985		1.10	0.54	0.03
1986					1986		0.00		
1987					1987		0.25		
1988					1988				
1989					1989				
1990					1990				
1991					1991				
1992					1992				
1993					1993				
1994					1994				
1995					1995				
1996					1996		0.01		0.00
1997					1997				
1998					1998				
1999					1999				
2000					2000		0.03		
2001					2001				
2002					2002				
2003					2003				
2004					2004				
2005					2005				
2006					2006				
2007					2007				
2008					2008				
2009					2009				
2010					2010				
2011					2011				
2012					2012				
2013			122	115	2013			2.5	3.8

Table A.14. Total number of Gulf of Maine haddock lengths taken from commercial landings by quarter and year between 1969 and 2013. Sampling intensity is expressed as metric tons landings per 100 lengths sampled (*200 metric tons per 100 lengths is an unofficial NAFO/ICNAF standard*).

Year	Quarter				Total lengths	Commercial landings (mt)	Metric tons/100 lengths
	1	2	3	4			
1969		93	341	92	526	2404.6	457.2
1970						1435.8	
1971	86			183	269	1190.2	442.5
1972			74	115	189	912.2	482.7
1973	99		627	205	931	525.9	56.5
1974	207	47			254	628.8	247.6
1975	64	100			164	1180.2	719.6
1976	30		74	108	212	1834.5	865.3
1977	382	708	839	569	2498	3230.1	129.3
1978	372	357	379	203	1311	4382.5	334.3
1979	309		124	166	599	4130.6	689.6
1980	51	494	359	201	1105	6317.6	571.7
1981	53	410	771	1019	2253	5713.3	253.6
1982	576	53	1634	345	2608	5634.3	216.0
1983	561	1176	1759	699	4195	5593.4	133.3
1984	187	173	967	504	1831	2792.8	152.5
1985	700	799	871	735	3105	2259.1	72.8
1986	516	476	1075	475	2542	1628.9	64.1
1987	376	181	689	591	1837	846.3	46.1
1988	352	50	142	202	746	418.0	56.0
1989	291		65	247	603	265.1	44.0
1990	75	50		150	275	465.0	169.1
1991	57	146	395	425	1023	443.5	43.4
1992	228		53	130	411	313.4	76.3
1993	103	110	125		338	193.0	57.1
1994		100	52	516	668	121.9	18.3
1995	256				256	178.2	69.6
1996	77	92	84	527	780	253.8	32.5
1997	120	379	855	613	1967	623.7	31.7
1998	998	160	234	348	1740	922.6	53.0
1999	117		514	313	944	569.1	60.3
2000	965	572	496	495	2528	799.3	31.6
2001	881	201	379	1573	3034	1006.8	33.2
2002	1278	412	247	423	2360	1009.2	42.8
2003	1277	564	1998	1556	5395	1026.4	19.0
2004	2919	2027	363	1052	6361	947.2	14.9
2005	2137	774	1462	1521	5894	977.7	16.6
2006	2121	1222	1022	1131	5496	622.5	11.3
2007	1598	708	1784	1016	5106	677.9	13.3
2008	1355	938	699	436	3428	542.7	15.8
2009	1816	715	277	419	3227	500.3	15.5
2010	1618	453	322	770	3163	622.6	19.7
2011	1664	832	453	652	3601	498.6	13.8
2012	1990	859	291	430	3570	416.6	11.7
2013	1898	1457	883	929	5167	212.0	4.1

Table A.15. Total numbers of Gulf of Maine haddock ages sampled from commercial landings by quarter between 1965 and 2013.

Year	Quarter				Total ages	Commercial landings (mt)	Metric tons/100 ages
	1	2	3	4			
1965		35	209	102	346	4154.7	1200.8
1966		35	84	14	133	4524.0	3401.5
1967	88	185		53	326	4852.2	1488.4
1968	50	59		35	144	3417.3	2373.1
1969		20	46	15	81	2404.6	2968.7
1970						1435.8	
1971						1190.2	
1972			20	20	40	912.2	2280.6
1973	20		38	40	98	525.9	536.6
1974	40	20			60	628.8	1048.0
1975	15	25			40	1180.2	2950.4
1976	19		20	35	74	1834.5	2479.1
1977	112	195	232	220	759	3230.1	425.6
1978	120	135	89	49	393	4382.5	1115.1
1979	78		25	61	164	4130.6	2518.6
1980	17	97	88	46	248	6317.6	2547.4
1981	14	120	185	227	546	5713.3	1046.4
1982	123	14	359	91	587	5634.3	959.9
1983	155	304	302	153	914	5593.4	612.0
1984	47	52	276	125	500	2792.8	558.6
1985	190	204	230	180	804	2259.1	281.0
1986	118	136	232	116	602	1628.9	270.6
1987	76	38	175	199	488	846.3	173.4
1988	104		32	39	175	418.0	238.9
1989	91		16	42	149	265.1	177.9
1990	43	16		37	96	465.0	484.3
1991	16	32	117	87	252	443.5	176.0
1992	40		15	83	138	313.4	227.1
1993	20	42	49		111	193.0	173.9
1994		26	21	196	243	121.9	50.2
1995	86				86	178.2	207.2
1996	25	13	22	109	169	253.8	150.2
1997	23	101	199	145	468	623.7	133.3
1998	127	45	64	166	402	922.6	229.5
1999	33		143	105	281	569.1	202.5
2000	303	181	171	168	823	799.3	97.1
2001	242	72	121	393	828	1006.8	121.6
2002	555	138	24	158	875	1009.2	115.3
2003	411	178	739	473	1801	1026.4	57.0
2004	783	348	33	82	1246	947.2	76.0
2005	441	170	485	560	1656	977.7	59.0
2006	1078	433	581	480	2572	622.5	24.2
2007	783	338	888	515	2524	677.9	26.9
2008	685	535	373	218	1811	542.7	30.0
2009	993	443	218	248	1902	500.3	26.3
2010	941	192	184	339	1656	622.6	37.6
2011	961	504	236	180	1881	498.6	26.5
2012	880	533	214	243	1870	416.6	22.3
2013	1347	865	461	536	3209	212.0	6.6

Table A.16. Total numbers of Gulf of Maine haddock lengths sampled from commercial landings by market category, quarter and year between 1977 and 2013. *Cells shaded in grey indicate where lengths were aggregated semi-annually. Cells shaded blue indicate where lengths were aggregated annually. Aggregation occurred when length sampling was insufficient; a general criterion of 100 lengths/block was used to determine sampling sufficiency.*

Year	Large (1470)				Scrod (1475)			
	1	2	3	4	1	2	3	4
1977		197	358		382	511	481	569
1978	149	35	200		223	322	179	203
1979	195		124	100	114			66
1980		319	102		51	175	257	201
1981		52	257	638	53	358	514	381
1982	103		1361	104	473	53	273	154
1983	249	868	1317	496	312	308	340	203
1984		79	828	391	187	94	139	113
1985	347	597	573	651	353	202	298	84
1986	283	234	868	271	233	242	207	204
1987	214	102	614	405	162	79	75	186
1988	91		100	202	261	50	42	
1989	192		65	118	99			129
1990	34			100	41	50		50
1991		146	216	213	57		179	212
1992	121			19	107		53	111
1993	Combined 1992 & 1994 and ran annual				103	56	125	
1994		100	52	297				219
1995	62				194			
1996	77		84	427		92		100
1997	120	255	497	355		124	358	258
1998	309	111	78	313	689	49	156	35
1999	117		300	211			214	102
2000	488	313	339	208	477	259	157	287
2001	528	93	313	726	353	108	66	847
2002	930	210		262	348	202	247	161
2003	792	348	1282	1043	485	216	716	513
2004	1898	942	101	601	1021	1085	262	451
2005	1421	325	675	752	716	449	787	769
2006	1193	687	453	617	928	535	569	514
2007	817	348	1016	616	781	360	768	400
2008	789	472	351	141	566	466	348	295
2009	1248	409	142	243	568	306	135	176
2010	1018	214	187	614	600	239	135	156
2011	1050	362	237	344	614	470	216	308
2012	1262	376	171	213	728	483	120	217
2013	1208	706	345	413	690	751	538	516

Table A.17. Total Gulf of Maine haddock commercial landings-at-age (numbers) from 1977 to 2013.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Age15	Age16	Age17	Age18	Age19	Age20	Age21	Age22	Total
1977	0	39,755	1,762,962	53,167	366,967	184,629	189,299	0	0	0	0	0	0	2,411	0	0	0	0	0	0	0	0	0	2,599,190
1978	0	0	374,650	2,291,417	172,388	363,003	208,654	10,580	0	0	0	5,290	0	0	0	0	0	0	0	0	0	0	0	3,425,982
1979	0	0	67,315	559,608	1,576,962	183,133	99,093	45,294	10,898	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2,542,303
1980	0	0	884,750	104,084	755,832	1,366,770	143,816	95,570	27,794	0	0	25,756	0	0	0	0	0	0	0	0	0	0	0	3,404,372
1981	0	2,068	1,598,228	717,686	292,045	340,692	541,941	91,639	116,490	13,327	0	0	0	3,433	9,995	0	0	0	0	0	0	0	0	3,727,544
1982	0	30,106	605,235	1,508,516	618,180	100,219	300,546	476,719	107,236	35,008	19,261	5,206	10,366	0	5,849	0	0	0	0	0	0	0	0	3,822,447
1983	0	0	7,577	818,079	967,850	786,711	147,856	252,137	346,411	54,803	38,544	16,725	5,117	0	0	0	0	0	0	0	0	0	0	3,441,810
1984	0	0	63,736	44,622	588,124	253,782	359,988	61,607	64,176	132,508	3,153	2,775	5,415	2,161	0	0	0	0	0	0	0	0	0	1,582,047
1985	0	0	22,128	319,344	82,516	354,183	151,463	241,319	47,220	19,629	33,631	492	589	0	0	0	0	0	0	0	0	0	0	1,272,514
1986	0	0	0	166,503	340,987	75,330	111,567	84,708	101,115	10,694	3,792	0	0	0	0	0	0	0	0	0	0	0	0	894,696
1987	0	0	3,745	25,377	95,767	46,124	33,013	55,332	32,964	10,723	4,387	252	0	0	0	0	0	0	0	0	0	0	0	307,684
1988	0	0	0	11,539	11,895	52,410	53,781	7,538	13,744	2,772	1,232	0	0	0	0	0	0	0	0	0	0	0	0	154,911
1989	0	0	15,537	2,643	40,660	18,301	22,676	13,959	707	943	0	0	0	0	0	0	0	0	0	0	0	0	0	115,426
1990	0	0	2,018	142,445	1,686	28,564	17,479	27,146	3,794	0	0	0	0	0	0	0	0	0	0	0	0	0	0	223,132
1991	0	0	5,579	15,722	58,569	28,391	27,857	12,628	5,811	3,140	0	0	0	0	0	0	0	0	0	0	0	0	0	157,697
1992	0	0	7,753	92,057	36,323	19,083	2,246	1,134	0	1,895	0	0	0	0	0	0	0	0	0	0	0	0	0	160,491
1993	0	0	10,844	34,040	22,484	9,718	10,571	4,586	1,567	595	186	155	0	0	0	0	0	0	0	0	0	0	0	94,746
1994	0	0	6,274	30,211	10,445	1,674	7,045	3,469	1,138	206	83	153	0	0	0	0	0	0	0	0	0	0	0	60,698
1995	0	0	0	4,993	34,162	8,163	5,440	4,003	4,345	261	686	2,091	0	0	0	0	0	0	0	0	0	0	0	64,144
1996	0	0	3,273	57,790	46,874	14,339	3,775	6,579	5,240	990	0	0	0	0	0	0	0	0	0	0	0	0	0	138,860
1997	0	0	2,281	82,457	117,766	55,455	12,429	4,454	923	790	398	157	0	0	0	0	0	0	0	0	0	0	0	277,110
1998	0	0	11,630	21,006	115,275	180,018	51,089	16,925	8,321	5,514	1,299	547	0	0	0	0	0	0	0	0	0	0	0	411,624
1999	0	0	0	35,907	63,674	93,190	66,255	37,073	6,863	3,851	0	571	1,119	202	0	0	0	0	0	0	0	0	0	308,705
2000	0	0	3,872	36,032	85,996	54,166	108,783	62,046	27,905	14,516	3,111	1,835	1,944	824	1,616	0	0	0	0	0	0	0	0	402,646
2001	0	0	8,684	156,376	106,988	81,810	75,155	71,243	35,344	13,040	6,148	1,284	0	0	0	392	0	0	0	0	0	0	0	556,464
2002	0	0	0	12,751	185,844	92,068	92,509	28,044	60,738	41,761	13,112	3,282	181	0	287	0	0	0	0	0	0	0	0	530,577
2003	0	0	0	2,641	30,433	344,788	69,131	53,244	18,050	28,358	26,095	4,186	1,045	903	178	0	0	0	0	0	0	0	0	579,052
2004	0	0	0	1,847	18,877	42,616	357,654	41,117	24,824	7,245	13,814	17,603	2,279	157	0	0	0	0	0	0	0	0	0	528,033
2005	0	0	0	1,129	17,851	42,303	69,285	316,249	37,353	28,808	9,659	8,093	7,127	1,014	381	0	0	0	0	0	0	0	0	539,252
2006	0	0	0	8,099	294	20,587	36,028	39,908	202,196	23,052	9,071	1,915	3,875	2,126	606	79	0	0	0	0	0	0	0	347,836
2007	0	0	150	1,532	98,378	5,417	26,574	21,756	47,784	192,507	16,300	5,278	1,129	881	1,369	60	0	0	0	0	0	0	0	419,115
2008	0	0	705	21,476	9,102	187,543	1,793	19,203	13,666	20,349	76,643	3,202	3,071	122	0	0	0	0	0	0	0	0	0	356,875
2009	0	0	0	2,184	15,258	5,387	146,364	2,645	18,364	8,603	12,695	61,509	2,519	1,391	275	78	0	0	0	0	0	0	0	277,272
2010	0	0	576	1,600	20,417	28,718	10,619	191,506	2,415	11,366	8,332	10,475	58,600	2,617	292	106	0	81	0	0	0	0	0	347,720
2011	0	0	145	1,474	1,993	26,562	27,024	11,333	122,042	1,241	11,041	9,097	5,616	36,061	1,192	362	103	47	207	0	0	0	18	255,558
2012	0	0	200	8,048	5,398	6,047	33,255	23,118	8,336	92,765	1,506	6,613	5,131	6,442	22,413	944	468	36	123	0	0	0	0	220,843
2013	0	0	1,392	37,916	15,902	7,342	3,697	19,648	11,240	3,166	24,596	757	1,797	929	1,537	5,305	263	73	24	0	0	0	0	135,584

Table A.18. Coefficients of variation (CV) associated with the estimates of Gulf of Maine haddock commercial landings-at-age from 1984 to 2013. Precision estimates of commercial numbers at age could not be estimated prior to 1984. CVvalues greater than 0.3 are shaded grey.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Age15	Age16	Age17	Age18	Age19	Age20	Age21	Age22
1984			0.25	0.11	0.08	0.09	0.03	0.09	0.13	0.09	0.19	0.33	0.20	0.26									
1985			0.18	0.11	0.15	0.07	0.08	0.04	0.12	0.17	0.16	1.27	0.79										
1986				0.08	0.06	0.05	0.04	0.05	0.08	0.21	0.26												
1987			0.35	0.14	0.06	0.07	0.07	0.06	0.10	0.13	0.25	0.68											
1988				0.32	0.26	0.18	0.26	0.27	0.46	0.60	0.72												
1989			0.60	0.75	0.23	0.29	0.19	0.19	0.91	0.87													
1990			0.84	0.23	0.87	0.33	0.55	0.47	0.80														
1991			0.58	0.37	0.13	0.14	0.15	0.16	0.26	0.67													
1992			0.84	0.17	0.34	0.52	0.71	0.97		1.06													
1993			0.17	0.15	0.15	0.37	0.37	0.31	0.45	0.85	0.88	1.22											
1994			0.16	0.08	0.15	0.27	0.28	0.19	0.34	1.03	0.87	1.00											
1995				0.70	0.13	0.42	0.32	0.38	0.32		0.82	0.46											
1996			0.61	0.25	0.25	0.26	0.33	0.35	0.63	0.69													
1997			1.10	0.14	0.12	0.13	0.26	0.17	0.39	0.29	0.64	1.10											
1998			0.83	0.23	0.12	0.12	0.18	0.35	0.32	0.59	1.23	1.38											
1999				0.26	0.11	0.13	0.18	0.24	0.37	0.51		1.18	0.93	1.23									
2000			0.49	0.23	0.16	0.12	0.14	0.17	0.21	0.46	0.51	0.91	0.65	1.01	0.58								
2001			0.43	0.09	0.09	0.16	0.10	0.13	0.20	0.35	0.57	0.90				1.01							
2002				0.38	0.08	0.13	0.10	0.20	0.13	0.18	0.22	0.42	1.37		1.39								
2003				0.70	0.17	0.04	0.11	0.14	0.19	0.13	0.18	0.44	0.35	0.71	1.27								
2004				0.65	0.47	0.14	0.03	0.12	0.18	0.23	0.23	0.18	0.41	0.94									
2005				0.61	0.25	0.14	0.09	0.03	0.13	0.14	0.24	0.28	0.23	0.68	0.95								
2006				0.26	0.76	0.16	0.13	0.09	0.03	0.12	0.17	0.30	0.22	0.34	0.51	1.34							
2007			1.36	0.51	0.08	0.36	0.14	0.14	0.10	0.04	0.19	0.26	0.50	0.57	0.60	1.28							
2008			1.20	0.34	0.31	0.06	0.44	0.15	0.19	0.17	0.08	0.25	0.50	1.02									
2009				0.55	0.22	0.33	0.05	0.37	0.13	0.18	0.14	0.09	0.29	0.50	0.85	1.35							
2010			1.29	0.97	0.21	0.16	0.25	0.06	0.38	0.18	0.19	0.21	0.13	0.39	0.64	1.38		1.32					
2011			1.35	0.76	0.46	0.15	0.11	0.22	0.05	0.42	0.18	0.17	0.22	0.12	0.45	0.65	1.47	1.48	0.96				1.69
2012			0.77	0.26	0.27	0.25	0.13	0.13	0.21	0.07	0.38	0.22	0.23	0.22	0.14	0.45	0.68	1.43	1.40				
2013			0.50	0.10	0.09	0.13	0.16	0.07	0.09	0.16	0.05	0.32	0.18	0.27	0.22	0.12	0.46	0.83	1.26				
Average			0.70	0.35	0.23	0.19	0.20	0.21	0.27	0.38	0.39	0.59	0.45	0.59	0.69	0.95	0.87	1.26	1.21				1.69

Table A.19. Mean weights-at-age (kg) of commercially landed Gulf of Maine haddock from 1977 to 2013.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Age15	Age16	Age17	Age18	Age19	Age20	Age21	Age22
1977		0.113	0.757	1.163	2.008	2.558	3.358							4.686									
1978			0.777	1.234	1.684	2.438	3.108	4.642				6.088											
1979			0.774	1.155	1.805	2.261	2.659	2.775	3.587														
1980			0.76	1.168	1.852	2.389	3.354	3.602	4.562			4.204											
1981		0.56	0.685	1.516	1.978	2.64	3.024	3.657	4.18	3.841				3.95	3.984								
1982		0.376	0.623	0.995	2.139	2.598	3.107	3.647	4.13	4.347	4.09	4.642	4.81		3.412								
1983			0.862	1.205	1.728	2.377	2.969	3.372	3.717	4.152	4.316	4.397	3.528										
1984			0.949	1.305	1.809	2.324	3.165	3.928	4.505	4.11	3.95	5.138	2.527	4.642									
1985			1.139	1.102	1.901	2.342	2.653	3.588	4.09	4.479	3.917	6.226	4.976										
1986				1.233	1.464	2.353	2.498	3.061	3.636	4.745	4.191												
1987			1.111	1.805	2.064	2.424	2.608	3.27	4.239	5.007	5.646	6.798											
1988				1.123	1.614	2.558	2.577	3.868	4.606	4.893	5.821												
1989			1.34	2.067	1.835	2.319	2.865	3.548	4.666	4.244													
1990			0.833	1.541	3.331	2.456	3.044	3.734	3.547														
1991			1.637	1.916	2.657	3.027	2.958	3.35	4.433	3.881													
1992			1.415	1.783	1.978	2.656	3.067	2.079		3.45													
1993			1.085	1.635	2.043	2.44	3.015	3.393	3.358	2.948	4.662	3.95											
1994			1.188	1.712	2.162	2.927	2.644	3.254	3.273	2.985	4.707	3.907											
1995				1.854	2.083	2.553	3.614	4.357	5.209	4.825	4.286	6.222											
1996			1.696	1.451	1.884	2.213	3.202	2.494	2.404	3.252													
1997			1.245	2.166	1.975	2.631	3.275	3.168	3.969	4.048	4.508	2.488											
1998			1.225	1.528	1.909	2.25	2.856	3.358	3.162	2.834	2.947	4.871											
1999				1.34	1.615	1.773	1.932	2.294	3.052	3.246		3.368	3.299	4.329									
2000			1.266	1.223	1.547	1.775	2.022	2.421	2.735	2.821	3.625	2.924	3.584	4.514	3.901								
2001			1.153	1.379	1.532	1.825	2.233	2.259	2.467	2.378	2.729	2.24				3.517							
2002				1.227	1.413	1.667	2.179	2.625	2.361	2.597	2.8	3.589	5.788		3.144								
2003				1.028	1.359	1.551	1.851	2.197	2.541	2.593	2.572	2.46	2.843	2.134	4.073								
2004				1.036	1.407	1.429	1.774	1.897	2.11	2.366	2.146	2.295	2.35	3.501									
2005				1.053	1.236	1.591	1.555	1.809	2.047	2.192	2.594	2.316	2.839	2.497	2.488								
2006				1.146	1.329	1.493	1.778	1.638	1.814	2.01	2.164	2.437	2.248	2.332	2.344	2.611							
2007			0.812	1.162	1.236	1.238	1.625	1.681	1.671	1.755	1.864	2.123	3.029	2.398	2.11	3.004							
2008			1.061	1.164	1.238	1.39	1.489	1.792	1.772	1.658	1.786	1.982	2.074	2.987									
2009				1.132	1.242	1.385	1.728	1.677	1.968	2.14	1.986	2.031	2.343	1.775	2.662	2.814							
2010			1.13	0.883	1.16	1.456	1.651	1.762	2.155	2.163	2.237	2.077	2.089	2.253	3.512	2.488		3.745					
2011			0.812	1.165	1.212	1.494	1.696	1.885	2.006	1.987	2.165	2.14	2.023	2.243	2.145	2.521	2.611	2.869	3.198				5.061
2012			0.965	1.14	1.286	1.475	1.564	1.82	1.916	1.995	2.616	2.186	2.113	2.063	2.206	2.727	2.674	3.287	2.869				
2013			0.816	1.014	1.298	1.464	1.635	1.72	1.893	1.937	2.049	1.93	2.243	2.261	2.079	2.226	2.217	2.548	2.611				

Table A.20. Gulf of Maine haddock commercial otter trawl landings per unit effort index (LPUE) from 1977 to 2012. *Note that 2013 commercial landings data were not available at the time the LPUE analysis was conducted.*

Year	Index	Std. Error	Variance	Lower 95% CL	Upper 95% CL
1977	1.064	0.063	0.004	0.938	1.203
1978	1.250	0.066	0.004	1.096	1.420
1979	1.296	0.069	0.005	1.130	1.479
1980	1.488	0.062	0.004	1.315	1.677
1981	1.198	0.066	0.004	1.050	1.362
1982	1.000				
1983	0.820	0.057	0.003	0.733	0.915
1984	0.422	0.057	0.003	0.376	0.471
1985	0.296	0.056	0.003	0.265	0.330
1986	0.230	0.056	0.003	0.206	0.256
1987	0.118	0.060	0.004	0.104	0.132
1988	0.070	0.071	0.005	0.061	0.080
1989	0.062	0.100	0.010	0.050	0.075
1990	0.080	0.087	0.008	0.067	0.094
1991	0.074	0.082	0.007	0.063	0.087
1992	0.046	0.085	0.007	0.039	0.054
1993	0.045	0.088	0.008	0.038	0.053
1994	0.031	0.110	0.012	0.025	0.038
1995	0.032	0.077	0.006	0.028	0.037
1996	0.049	0.068	0.005	0.043	0.056
1997	0.083	0.067	0.005	0.072	0.094
1998	0.111	0.063	0.004	0.098	0.125
1999	0.073	0.070	0.005	0.063	0.083
2000	0.099	0.064	0.004	0.088	0.112
2001	0.147	0.057	0.003	0.132	0.165
2002	0.148	0.054	0.003	0.133	0.164
2003	0.185	0.052	0.003	0.166	0.204
2004	0.220	0.053	0.003	0.198	0.244
2005	0.193	0.053	0.003	0.174	0.214
2006	0.204	0.054	0.003	0.183	0.226
2007	0.227	0.055	0.003	0.204	0.252
2008	0.205	0.055	0.003	0.184	0.229
2009	0.227	0.058	0.003	0.202	0.253
2010	0.160	0.063	0.004	0.141	0.180
2011	0.129	0.056	0.003	0.116	0.144
2012	0.081	0.055	0.003	0.072	0.090

Table A.21. Fractional breakdown of the observed discards of Gulf of Maine haddock by discard reason from 1989 to 2013.

Year	Unknown/ other	No market	Poor quality	Regulatory, no retention	Regulatory, too small
1989	0.51	0.00	0.49	0.00	0.00
1990	0.67	0.00	0.33	0.00	0.00
1991	0.71	0.00	0.29	0.00	0.00
1992	0.82	0.00	0.18	0.00	0.00
1993	0.72	0.14	0.14	0.00	0.00
1994	0.49	0.01	0.00	0.42	0.08
1995	0.09	0.03	0.01	0.56	0.31
1996	0.13	0.03	0.06	0.30	0.48
1997	0.00	0.45	0.00	0.46	0.09
1998	0.83	0.00	0.01	0.00	0.16
1999	0.00	0.07	0.77	0.10	0.06
2000	0.00	0.53	0.21	0.00	0.26
2001	0.00	0.00	0.03	0.00	0.96
2002	0.00	0.05	0.10	0.01	0.84
2003	0.00	0.11	0.08	0.01	0.80
2004	0.00	0.08	0.11	0.32	0.49
2005	0.01	0.02	0.17	0.02	0.78
2006	0.00	0.00	0.03	0.01	0.97
2007	0.00	0.01	0.34	0.01	0.63
2008	0.00	0.00	0.02	0.04	0.93
2009	0.00	0.00	0.17	0.03	0.80
2010	0.00	0.00	0.08	0.08	0.84
2011	0.00	0.00	0.05	0.61	0.34
2012	0.01	0.05	0.06	0.07	0.81
2013	0.00	0.01	0.06	0.14	0.79

Table A.22. Number of Gulf of Maine haddock length observations recorded by certified observers (NEFOP) and at-sea monitors (ASM) by year and gear type. The gear codes listed are as follows: longline (010), otter trawl (050), sink gillnet (100) and mesh type codes are: large mesh (LM), extra-large mesh (ELM).

Year	NEGEAR	Mesh category	NEFOP lengths	ASM lengths
2010	010		1	10
2010	050	LM	25	58
2010	100	ELM	0	1
2010	100	LM	6	31
2011	010		2	51
2011	050	LM	111	418
2011	100	ELM	0	1
2011	100	LM	43	120
2012	010		52	138
2012	050	LM	578	922
2012	100	ELM	0	1
2012	100	LM	305	327
2013	010		16	6
2013	050	LM	282	2032
2013	100	ELM	0	1
2013	100	LM	118	169

Table A.23. Fraction of the total Gulf of Maine haddock estimated discards based on ***preliminary estimates of commercial discards*** by gear type from 1989 to 2012. Gears contributing greater than 5% of the total observed discards in any year are shaded grey. *Note that the 2013 estimates are missing because these data were not available at the time of this analysis.*

Year	Benthic longline	Otter trawl: Small mesh (<5.5")	Otter trawl: Large mesh (5.5" - 7.9")	Shrimp trawl	Sink gillnet: Large mesh (5.5" - 7.9")	Sink gillnet: Extra-large mesh (≥8")	Major gear type contribution	Total estimated discards (mt)
1989	0.00	0.00	0.41	0.03	0.57	0.00	0.97	5.1
1990	0.00	0.00	0.06	0.30	0.64	0.00	0.70	2.9
1991	0.14	0.00	0.33	0.07	0.46	0.00	0.93	3.0
1992	0.00	0.00	0.78	0.10	0.12	0.00	0.90	8.9
1993	0.00	0.00	0.67	0.10	0.23	0.00	0.90	14.8
1994	0.00	0.00	0.84	0.04	0.13	0.00	0.96	63.3
1995	0.00	0.01	0.91	0.02	0.06	0.00	0.97	90.1
1996	0.00	0.01	0.73	0.02	0.21	0.02	0.94	83.3
1997	0.00	0.00	1.00	0.00	0.00	0.00	1.00	379.9
1998	0.00	0.00	0.82	0.00	0.18	0.00	1.00	16.6
1999	0.00	0.06	0.44	0.00	0.50	0.00	0.94	2.5
2000	0.00	0.00	0.73	0.00	0.25	0.02	0.98	28.5
2001	0.00	0.25	0.41	0.00	0.34	0.00	0.75	17.3
2002	0.00	0.02	0.37	0.00	0.56	0.04	0.93	20.0
2003	0.28	0.01	0.41	0.00	0.24	0.07	0.93	19.1
2004	0.04	0.05	0.60	0.01	0.23	0.08	0.86	13.6
2005	0.58	0.00	0.25	0.00	0.10	0.06	0.93	26.8
2006	0.20	0.00	0.68	0.01	0.06	0.04	0.95	33.3
2007	0.37	0.01	0.08	0.01	0.50	0.03	0.95	49.3
2008	0.34	0.06	0.34	0.00	0.25	0.00	0.93	11.0
2009	0.30	0.02	0.18	0.03	0.46	0.01	0.94	13.1
2010	0.07	0.08	0.42	0.15	0.21	0.07	0.70	4.3
2011	0.11	0.42	0.23	0.00	0.22	0.02	0.56	9.9
2012	0.16	0.04	0.36	0.25	0.19	0.00	0.71	26.1
2013								
Average	0.11	0.04	0.50	0.05	0.28	0.02	0.89	

Table A.24. ***Preliminary estimates of the coefficients of variation*** (CV) for the Gulf of Maine haddock commercial discard (mt) estimates from 1989 to 2013 by gear. *Note that the 2013 estimates are missing because these data were not available at the time of this analysis.*

Year	Benthic longline	Otter trawl Small mesh (<5.5")	Otter trawl Large mesh (5.5" - 7.9")	Shrimp trawl	Sink gillnet Large mesh (5.5" - 7.9")	Sink gillnet Extra-large mesh (≥8")	Total
1989		0.83	0.84	0.80	0.49		0.72
1990			1.05	0.81	0.43		0.61
1991	1.19		0.56	0.84	0.31		0.44
1992			0.66	0.26	0.24		0.58
1993			0.53	0.21	0.33		0.44
1994			0.38	0.21	0.43		0.35
1995		0.33	0.37	0.25	0.40		0.35
1996		4.07	0.66	0.67	0.54	0.63	0.53
1997			0.96	0.72	1.04		0.95
1998			0.37		0.66		0.33
1999		0.47	1.05		0.53		0.63
2000			0.54		0.50	0.48	0.44
2001		0.70	0.65		0.35		0.37
2002		0.63	0.33		0.39	0.83	0.22
2003	0.45	0.57	0.18		0.23	0.30	0.16
2004	0.37	0.60	0.25	0.69	0.20	0.29	0.17
2005	0.26	0.32	0.18	0.52	0.22	0.19	0.16
2006	0.36	0.43	0.49	0.43	0.24	0.56	0.35
2007	0.39	0.37	0.31	0.69	0.83	0.64	0.39
2008	0.47	0.16	0.41	0.84	0.28	0.62	0.23
2009	0.81	0.73	0.34	0.40	0.27	3.01	0.31
2010	0.40	0.61	0.28	0.41	0.25	0.19	0.16
2011	0.30	0.45	0.11	0.00	0.08	0.14	0.20
2012	0.32	0.25	0.11	0.72	0.06	0.16	0.19
2013							
Average	0.48	0.72	0.48	0.53	0.39	0.62	0.39

Table A.25. Summary of the number of lengths collected from Gulf of Maine haddock discarded in the commercial fishery by gear type and semester from 1989 to 2013.

Year	Benthic longline		Otter trawl				Shrimp trawl		Sink gillnet				Total
			Small mesh (<5.5")		Large mesh (5.5" -				Large mesh (5.5" -		Extra-large mesh		
	1	2	1	2	1	2	1	2	1	2	1	2	
1989					1	8	1						10
1990													
1991										1			1
1992					10	23	7			1			41
1993					8	44	48	1	2	1			104
1994					8	17	88	32	1	18			164
1995				16	217	218	136		7	6			600
1996			21	3	56	32	5	36	25	8		4	190
1997					946	3	7			2	1		959
1998					10				2	2			14
1999				6		5				18			29
2000						17			6	2			25
2001			1		24	18			5				48
2002				40	10	49			35	3			137
2003	105		5	22	96	116			39	43	6	13	445
2004	23			121	41	195	1		55	38	3	26	503
2005	207	7		18	223	237			5	72	9	15	793
2006	140		4	3	219	101	111	2		3		2	585
2007	299			8	124	125	7		13	10	1		587
2008	63				33	185			3	3			287
2009	127			10	80	27		1	91	1			337
2010	11			3	25	58		18	3	34		1	153
2011	36	17		327	78	451	3		53	101		1	1067
2012	137	53	5	5	306	1244	11		130	438	1		2330
2013	22		123	77	1636	720	18		84	116		1	2797

Table A.26. Total number of Gulf of Maine commercial trips (statistical areas 464, 465, 467, 511-515) observed from 1989 to 2013, summarized by gear type. *The 2010-2013 numbers include trips observed by both at-sea monitors and observers.*

Year	Benthic longline	Otter trawl, large mesh (5.5" - 7.9")	Sink gillnet, large mesh (5.5" - 7.9")	Total
1977				
1978				
1979				
1980				
1981				
1982				
1983				
1984				
1985				
1986				
1987				
1988				
1989		38	105	143
1990		26	120	146
1991	3	48	801	852
1992	11	44	896	951
1993	3	17	560	580
1994		6	82	88
1995		24	62	86
1996		11	39	50
1997		5	31	36
1998		6	78	84
1999		27	70	97
2000		80	70	150
2001		112	39	151
2002	1	150	62	213
2003	18	251	254	523
2004	10	251	587	848
2005	58	499	505	1062
2006	36	203	109	348
2007	36	225	92	353
2008	20	254	130	404
2009	35	410	271	716
2010	52	615	1080	1747
2011	80	1014	1382	2476
2012	113	1123	1166	2402
2013	33	642	495	1170

Table A.27. ***Final estimates*** of Gulf of Maine haddock commercial discards (mt) by gear from 1977 to 2013 by gear. Estimates from 1989 to 2011 were estimated using an approach consistent with the Standardized Bycatch Report Methodology (Wigley et al., 2007). Estimates from 1977 to 1988 were hindcast using an approach documented in this report. *Note that hindcast discard estimates could not be obtained pre-1982, and no attempt was made to hindcast discards by longline gear.*

Year	Benthic longline	Otter trawl, large mesh (5.5" - 7.9")	Sink gillnet, large mesh (5.5" - 7.9")	Total
1977				
1978				
1979				
1980				
1981				
1982		0.5	5.9	6.4
1983		1.8	4.6	6.5
1984		4.4	6.6	11.0
1985		9.2	7.3	16.5
1986		8.5	7.9	16.4
1987		10.8	13.1	23.9
1988				
1989		2.1	2.9	5.0
1990		0.2	1.9	2.0
1991	0.4	1.0	1.4	2.8
1992	0.0	7.0	1.1	8.0
1993	0.0	9.9	3.4	13.3
1994		53.1	8.0	61.1
1995		82.0	5.8	87.7
1996		60.9	17.4	78.2
1997		378.5	0.3	378.7
1998		13.7	3.0	16.6
1999		1.1	1.2	2.3
2000		20.8	7.1	27.9
2001		7.1	5.8	12.9
2002		7.5	11.2	18.6
2003	5.3	7.9	4.6	17.7
2004	0.5	8.1	3.1	11.7
2005	15.5	6.8	2.7	25.0
2006	6.7	22.6	2.1	31.5
2007	18.5	3.7	24.7	46.9
2008	3.7	3.8	2.8	10.3
2009	4.0	2.3	6.0	12.3
2010	0.3	1.8	0.9	3.0
2011	1.0	2.3	2.2	5.6
2012	4.5	8.7	4.5	17.7
2013	1.3	28.4	2.6	32.3

Table A.28. ***Final coefficients of variation*** (CV) for the Gulf of Maine haddock commercial discard (mt) estimates from 1977 to 2013 by gear. *CVs are not available for hindcast discards (pre-1989).*

Year	Benthic longline	Otter trawl, large mesh (5.5" - 7.9")	Sink gillnet, large mesh (5.5" - 7.9")	Total
1977				
1978				
1979				
1980				
1981				
1982				
1983				
1984				
1985				
1986				
1987				
1988				
1989		0.84	0.49	0.45
1990		1.05	0.43	0.40
1991	1.19	0.56	0.31	0.31
1992		0.66	0.24	0.58
1993		0.53	0.33	0.40
1994		0.38	0.43	0.33
1995		0.37	0.40	0.35
1996		0.66	0.54	0.53
1997		0.96	1.04	0.95
1998		0.37	0.66	0.32
1999		1.05	0.53	0.56
2000		0.54	0.50	0.42
2001		0.65	0.35	0.39
2002		0.33	0.39	0.27
2003	0.45	0.18	0.23	0.17
2004	0.37	0.25	0.20	0.18
2005	0.26	0.18	0.22	0.17
2006	0.36	0.49	0.24	0.36
2007	0.39	0.31	0.83	0.47
2008	0.47	0.41	0.28	0.24
2009	0.81	0.34	0.27	0.30
2010	0.40	0.28	0.25	0.19
2011	0.30	0.11	0.08	0.08
2012	0.34	0.11	0.06	0.10
2013	0.28	0.16	0.09	0.14
Average	0.47	0.47	0.38	0.35

Table A.29. Length sampling of Gulf of Maine haddock commercial discards from 1989 to 2013 by gear type and semester. Sampling intensity is expressed as metric tons discards per 100 lengths sampled (*200 metric tons per 100 lengths is an unofficial NAFO/ICNAF standard*). Cells shaded grey and blue indicate where discards at length were estimated using annual time blocks. Cells shaded green indicate where discards at length were estimated using semester time blocks. Blue shaded cells indicate where length sampling was determined to be insufficient and augmented with survey length frequencies. A general criterion of 30 lengths/block was used to determine sampling sufficiency.

Year	Benthic longline		Otter trawl, large mesh (5.5" - 7.9")		Sink gillnet, large mesh (5.5" - 7.9")		Total lengths	Total discards (mt)	Metric tons/100 lengths
	1	2	1	2	1	2			
1989			1	8			9	5.0	55.4
1990							0	2.0	
1991						1	1	2.8	280.0
1992			10	23		1	34	8.0	23.6
1993			8	44	2	1	55	13.3	24.2
1994			8	17	1	18	44	61.1	138.8
1995			217	218	7	6	448	87.7	19.6
1996			56	32	25	8	121	78.2	64.7
1997			946	3		2	951	378.7	39.8
1998			10		2	2	14	16.6	118.9
1999				5		18	23	2.3	10.1
2000				17	6	2	25	27.9	111.4
2001			24	18	5		47	12.9	27.5
2002			10	49	35	3	97	18.6	19.2
2003	105		96	116	39	43	399	17.7	4.4
2004	23		41	195	55	38	352	11.7	3.3
2005	207	7	223	237	5	72	751	25.0	3.3
2006	140		219	101		3	463	31.5	6.8
2007	299		124	125	13	10	571	46.9	8.2
2008	63		33	185	3	3	287	10.3	3.6
2009	127		80	27	91	1	326	12.3	3.8
2010	11		25	58	3	34	131	3.0	2.3
2011	36	17	78	451	53	101	736	5.6	0.8
2012	137	53	306	1244	130	438	2308	17.7	0.8
2013	22		1636	720	84	116	2578	32.3	1.3

Table A.30. Summary of length observations borrowed from Northeast Fisheries Science Center (NEFSC) bottom trawl surveys by gear type. See report text for a description of the method used to sub-sample gear-specific lengths from the survey length distributions. The grey shaded cells indicate years when the survey lengths were applied for the estimation of commercial discards at length.

Year	Benthic longline	Otter trawl, large mesh (5.5" - 7.9")	Sink gillnet, large mesh (5.5" - 7.9")
1977	99	348	249
1978	41	94	79
1979	10	40	27
1980	4	53	25
1981	26	89	63
1982	7	21	16
1983	6	38	21
1984	12	33	27
1985	34	61	54
1986	1	1	1
1987	5	5	5
1988	0	0	0
1989	6	11	10
1990		2	1
1991	1	4	4
1992	2	7	3
1993	19	49	38
1994	5	13	11
1995	29	46	44
1996	29	42	38
1997	44	89	70
1998	31	53	44
1999	79	291	188
2000	438	739	670
2001	224	282	268
2002	110	154	135
2003	50	70	64
2004	67	83	83
2005	58	101	86
2006	191	258	244
2007	48	107	86
2008	176	359	313
2009	28	48	41
2010	7	17	10
2011	39	108	79
2012	279	655	533
2013	165	795	456

Table A.31. Total Gulf of Maine haddock commercial discards-at-age (numbers) from 1977 to 2013. *Note that commercial discard estimates are not available pre-1982.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Age15	Total
1977	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1978	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1979	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1980	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1981	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1982	0	301	12,883	1,385	0	0	0	0	0	0	0	0	0	0	0	0	14,569
1983	110	10,807	4,266	5,183	0	0	0	0	0	0	0	0	0	0	0	0	20,366
1984	0	1,070	18,321	4,267	1,585	0	0	0	0	0	0	0	0	0	0	0	25,243
1985	0	881	7,054	19,572	2,549	47	0	0	0	0	0	0	0	0	0	0	30,103
1986	0	3,588	10,765	10,765	3,588	3,588	0	0	0	0	0	0	0	0	0	0	32,294
1987	0		15,705	7,437	4,657	632	0	0	0	0	0	0	0	0	0	0	28,431
1988	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1989	0	1,168	5,456	458	497	0	0	0	0	0	0	0	0	0	0	0	7,579
1990	0	6,931	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6,931
1991	0	3,130	1,531	501	0	0	0	0	0	0	0	0	0	0	0	0	5,162
1992	0	1,819	5,339	2,314	220	29	0	0	0	0	0	0	0	0	0	0	9,721
1993	0	3,654	9,207	2,175	422	129	351	0	146	95	0	0	0	95	32	0	16,306
1994	69	6,417	16,161	13,226	3,005	1,650	2,076	2,138	573	0	0	245	0	0	0	0	45,560
1995	406	1,983	42,355	32,723	11,912	1,283	530	350	0	0	0	0	0	0	0	0	91,542
1996	0	2,577	19,546	66,865	8,339	1,769	211	404	274	134	0	0	0	0	0	0	100,119
1997	0	821	3,970	75,257	128,867	32,670	5,881	2,145	1,776	553	50	173	0	0	0	0	252,163
1998	965	5,681	7,890	2,360	8,247	2,601	10	0	0	0	0	0	0	0	0	0	27,754
1999	95	3,127	825	632	121	174	128	41	23	12	4	0	0	0	0	0	5,182
2000	0	1,867	32,786	11,083	1,942	734	259	26	11	0	0	3	0	0	0	0	48,711
2001	0	250	4,587	10,752	1,031	209	248	126	2	0	0	0	0	0	0	0	17,205
2002	47	420	1,069	3,644	13,998	1,677	620	104	454	60	0	0	0	0	0	0	22,093
2003	0	112	1,606	2,283	3,959	11,282	1,117	180	123	57	46	4	1	0	0	0	20,770
2004	0	1,251	311	2,048	1,303	2,129	5,394	466	208	18	36	22	2	0	0	0	13,188
2005	0	193	7,692	728	5,858	4,022	4,445	6,986	591	81	10	8	1	0	0	0	30,615
2006	0	80	700	22,304	1,565	3,661	2,189	3,776	5,679	345	22	0	0	0	0	0	40,321
2007	0	7,838	15,443	7,320	37,095	455	1,380	919	1,019	2,022	37	15	0	0	0	0	73,543
2008	0	96	5,695	3,741	310	5,677	89	151	85	120	109	0	0	0	0	0	16,073
2009	0	62	396	3,720	2,895	629	4,774	100	412	22	115	281	3	3	0	0	13,412
2010	27	734	792	484	1,037	412	191	593	0	1	46	4	22	0	0	0	4,343
2011	19	3,040	4,516	1,033	82	1,003	290	41	491	1	0	13	4	5	2	0	10,540
2012	8	1,010	26,796	4,636	965	37	485	85	18	285	5	3	0	8	14	0	34,355
2013	1,175	18,376	12,217	31,242	1,521	232	17	127	42	4	36	2	1	0	1	5	64,998

Table A.32. Mean weights-at-age (kg) of commercially discarded Gulf of Maine haddock from 1977 to 2013. *Note that commercial discard estimates are not available pre-1982.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Age15
1977																
1978																
1979																
1980																
1981																
1982		0.389	0.438	0.454												
1983	0.048	0.181	0.314	0.610												
1984		0.308	0.420	0.504	0.521											
1985		0.315	0.502	0.562	0.644	0.507										
1986		0.507	0.507	0.507	0.507	0.507										
1987			0.789	0.901	0.901	0.979										
1988																
1989		0.240	0.713	0.799	0.911											
1990		0.295														
1991		0.347	0.821	0.916												
1992		0.448	0.867	0.995	1.078	1.185										
1993		0.364	0.649	0.934	1.931	2.032	3.664		4.111	4.461				4.640	4.286	
1994	0.092	0.362	0.632	1.319	2.281	3.212	3.279	4.234	3.681			3.950				
1995	0.017	0.256	0.717	1.019	1.459	2.575	3.122	3.255								
1996		0.330	0.500	0.752	1.363	2.098	2.326	2.114	2.117	2.488						
1997		0.358	0.759	1.458	1.396	1.823	2.359	2.567	2.565	2.930	3.738	2.632				
1998	0.021	0.250	0.648	0.644	0.790	0.783	1.773									
1999	0.072	0.233	0.468	0.676	0.950	1.026	1.864	2.347	4.111	3.775	3.665					
2000		0.257	0.533	0.659	0.834	0.764	1.148	2.809	3.154			3.331				
2001		0.242	0.667	0.777	0.880	0.997	0.846	0.941	1.639							
2002	0.068	0.121	0.398	0.730	0.892	1.020	1.064	1.066	0.961	1.257						
2003		0.318	0.525	0.593	0.752	0.957	0.996	1.481	1.070	1.858	2.370	1.769	2.814			
2004		0.191	0.381	0.696	0.909	0.962	1.048	1.117	1.863	2.212	1.343	2.365	1.889			
2005		0.173	0.484	0.703	0.828	0.884	0.983	1.033	0.978	1.267	1.336	1.634	2.018			
2006		0.287	0.405	0.701	0.694	0.853	0.858	0.897	1.012	0.943	1.310					
2007		0.243	0.490	0.585	0.752	0.802	0.832	0.843	0.889	0.976	1.170	1.194				
2008		0.289	0.459	0.637	0.899	0.788	0.829	0.851	0.830	0.869	0.985					
2009		0.304	0.565	0.774	0.927	0.944	1.019	0.950	0.976	1.544	1.182	1.319	1.971	1.563		
2010	0.086	0.253	0.450	0.763	0.826	0.887	0.851	1.011		1.353	0.875	1.385	1.261			
2011	0.064	0.268	0.511	0.715	0.975	0.803	0.957	1.011	0.956	1.702	1.466	1.221	1.184	1.398	1.783	
2012	0.064	0.248	0.473	0.668	0.772	1.160	0.936	1.135	1.306	0.982	1.014	1.601		1.236	1.529	
2013	0.076	0.262	0.442	0.651	0.785	0.855	1.134	1.036	1.143	1.567	1.349	1.329	1.575		1.466	1.468

Table A.33. Annual ratios of Marine Recreational Fisheries Statistical Survey (MRFSS) and Marine Recreational Information Program (MRIP) Gulf of Maine haddock catch estimates and aggregate time series ratios (ratio of means) using the 2004 – 2011 period of overlap.

	MRFSS		MRIP			MRIP/MRFSS Ratio	
Year	Estimated recreational harvest, A + B1 (000s fish)	Estimated recreational releases, B2 (000s fish)	Estimated recreational harvest, A + B1 (000s fish)	Estimated recreational releases, B2 (000s fish)	Year	Harvest (A+B1)	Releases (B2)
2004	278.5	142.4	199.0	80.4	2004	0.71	0.56
2005	444.7	116.2	355.2	101.8	2005	0.80	0.88
2006	277.9	164.2	296.8	175.1	2006	1.07	1.07
2007	398.2	105.4	402.8	110.6	2007	1.01	1.05
2008	358.5	124.3	342.7	178.4	2008	0.96	1.44
2009	311.6	72.0	265.4	65.4	2009	0.85	0.91
2010	391.5	72.6	190.3	47.0	2010	0.49	0.65
2011	166.3	38.7	139.8	35.5	2011	0.84	0.92
Sum	**2,627**	**835.7**	**2,192.1**	**794.1**	**Ratio**	**0.83**	**0.95**

Table A.34. Estimates of Gulf of Maine haddock recreational catch in numbers (000s) and weight (mt). ***Recreational release estimates do not include any assumptions about discard mortality.***

	Recreational harvest (A+B1)			Recreational releases (B2)		
Year	Numbers (000s)	CV	Weight (mt)	Numbers (000s)	CV	Weight (mt)
1981	19.2	0.34	38.2	0.0		0.0
1982	16.3	0.49	23.0	0.1	1.00	0.0
1983	30.5	0.26	52.7	0.0		0.0
1984	26.2	0.32	52.3	1.6	0.75	0.6
1985	16.2	0.36	21.6	0.1	1.00	0.0
1986	29.1	0.32	51.8	0.4	0.75	0.2
1987	15.7	0.29	39.2	0.0		0.0
1988	6.4	0.31	20.1	2.8	0.58	1.3
1989	5.0	0.42	13.1	4.9	0.44	2.6
1990	1.5	0.48	5.3	0.3	1.00	0.1
1991	0.2	1.46	0.3	0.0		0.0
1992	0.0		0.0	0.0		0.0
1993	0.3	1.99	0.6	0.0		0.0
1994	2.0	0.52	3.3	1.6	0.61	0.9
1995	92.7	0.64	124.1	41.3	0.65	27.4
1996	3.5	0.34	5.7	8.2	0.35	6.4
1997	16.8	0.43	30.2	15.0	0.33	10.5
1998	23.6	0.31	45.6	9.1	0.36	7.0
1999	10.1	0.24	17.8	15.9	0.27	9.8
2000	67.6	0.24	128.1	96.1	0.22	60.4
2001	100.8	0.14	169.3	106.8	0.17	86.8
2002	69.7	0.18	135.3	163.6	0.21	177.3
2003	100.2	0.11	173.9	248.2	0.15	257.4
2004	199.0	0.16	312.6	80.4	0.22	72.9
2005	355.2	0.16	538.1	101.8	0.28	72.0
2006	296.8	0.10	447.4	175.1	0.14	131.0
2007	402.8	0.15	572.7	110.6	0.12	91.4
2008	342.7	0.13	536.6	178.4	0.42	144.1
2009	265.4	0.14	408.6	65.4	0.12	48.8
2010	190.3	0.15	314.0	47.0	0.23	37.1
2011	139.8	0.14	228.8	35.5	0.17	22.4
2012	167.5	0.19	251.2	189.7	0.14	107.3
2013	147.0	0.09	241.1	507.1	0.08	413.9

Table A.35. Length sampling intensity of recreationally caught Gulf of Maine haddock by catch type and year from 1981 to 2013. Sampling intensity is expressed as metric tons of landings per 100 lengths sampled (*200 metric tons per 100 lengths is an unofficial NAFO/ICNAF standard*). In some years recreational length frequencies were supplemented using length observations from the Northeast Fisheries Science Center (NEFSC); see text for a description of the methods used to sub-sample from the survey length distributions. Due to the limited number of survey lengths available to characterize recreational releases between 1981 and 1994, an aggregate length frequency distribution was applied (grey-shaded cells).

	Landings (A)				Releases (B2)			
Year	Lengths sampled	Harvest (A+B1, mt)	Metric tons/100 lengths	NEFSC survey supplement	Lengths sampled	Releases (B2, mt)	Metric tons/100 lengths	NEFSC survey supplement
1981	13	38.2	293.9	216		0.0		12
1982	2	23.0	1148.2	170		0.0		2
1983	10	52.7	527.4	166		0.0		0
1984	16	52.3	326.8	63		0.6		3
1985	7	21.6	308.4	262		0.0		4
1986	0	51.8		54		0.2		0
1987	6	39.2	652.9	35		0.0		1
1988	2	20.1	1006.2	20		1.3		0
1989	3	13.1	436.9	19		2.6		6
1990	0	5.3		6		0.1		1
1991	0	0.3		4		0.0		1
1992	0	0.0		4		0.0		2
1993	0	0.6		11		0.0		17
1994	4	3.3	81.4	10		0.9		5
1995	153	124.1	81.1			27.4		28
1996	25	5.7	22.9	53		6.4		31
1997	21	30.2	143.9	58		10.5		40
1998	62	45.6	73.6	54		7.0		29
1999	32	17.8	55.6	130		9.8		71
2000	34	128.1	376.7	167		60.4		445
2001	25	169.3	677.4	376		86.8		236
2002	119	135.3	113.7			177.3		551
2003	210	173.9	82.8			257.4		151
2004	2146	312.6	14.6		101	72.9	72.1	
2005	3269	538.1	16.5		140	72.0	51.4	
2006	2473	447.4	18.1		228	131.0	57.5	
2007	2082	572.7	27.5		143	91.4	63.9	
2008	2321	536.6	23.1		106	144.1	135.9	
2009	2366	408.6	17.3		56	48.8	87.1	
2010	1727	314.0	18.2		14	37.1	265.3	
2011	1484	228.8	15.4		29	22.4	77.2	
2012	1753	251.2	14.3		539	107.3	19.9	
2013	1019	241.1	23.7		2343	413.9	17.7	

Table A.36. Total Gulf of Maine haddock harvest (A+B1)-at-age (numbers) from 1977 to 2013. *Note that recreational catch estimates are not available pre-1981.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Age15	Age16	Age17	Age18	Total
1977	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1978	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1979	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1980	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1981	0	0	6,474	3,934	1,630	2,286	3,123	570	899	122	84	0	0	31	92	0	0	0	0	19,245
1982	0	0	2,454	9,499	2,497	363	426	805	116	76	94	0	16	0	5	0	0	0	0	16,351
1983	0	0	544	13,261	8,458	4,562	768	817	1,642	239	132	85	7	0	0	0	0	0	0	30,515
1984	0	0	6,301	936	8,287	2,876	4,986	591	637	1,507	0	30	19	0	0	0	0	0	0	26,170
1985	0	0	1,018	10,694	880	1,963	581	719	176	68	138	0	10	0	0	0	0	0	0	16,247
1986	0	539	0	6,263	14,207	2,418	2,461	1,644	1,367	204	0	0	0	0	0	0	0	0	0	29,103
1987	0	0	1,119	1,855	5,705	2,053	1,422	1,593	871	642	447	0	0	0	0	0	0	0	0	15,707
1988	0	0	0	266	445	2,342	1,783	97	1,305	145	0	0	0	0	0	0	0	0	0	6,383
1989	0	0	368	61	1,191	1,007	1,288	1,045	57	0	0	0	0	0	0	0	0	0	0	5,017
1990	0	0	0	576	0	256	128	320	256	0	0	0	0	0	0	0	0	0	0	1,536
1991	0	0	43	115	30	7	4	4	0	0	0	0	0	0	0	0	0	0	0	203
1992	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1993	0	0	43	78	59	71	35	0	0	0	0	0	0	0	0	0	0	0	0	286
1994	0	0	515	1,044	150	95	109	68	0	0	0	18	0	0	0	0	0	0	0	1,999
1995	0	0	13,408	48,402	29,610	718	303	303	0	0	0	0	0	0	0	0	0	0	0	92,744
1996	0	0	121	1,560	1,245	255	69	129	85	18	9	0	0	11	0	0	0	0	0	3,502
1997	0	0	202	5,490	7,938	2,012	586	279	89	114	35	10	0	0	0	0	0	0	0	16,755
1998	0	0	875	1,663	8,434	9,962	1,638	508	290	179	18	0	0	0	0	0	0	0	0	23,567
1999	0	0	0	1,484	1,576	3,072	2,461	1,173	263	85	8	0	27	0	0	0	0	0	0	10,149
2000	0	0	554	5,256	16,907	9,977	19,140	9,986	3,895	1,119	382	201	64	84	0	0	0	0	0	67,565
2001	0	0	3,395	34,438	21,048	13,816	10,855	8,892	5,101	2,026	916	191	0	0	0	94	0	0	0	100,772
2002	0	0	0	344	22,898	12,712	13,956	3,854	7,722	6,162	1,471	558	0	0	23	0	0	0	0	69,700
2003	0	0	18	352	4,326	64,000	12,713	7,466	2,564	4,205	3,682	516	273	133	0	0	0	0	0	100,248
2004	0	185	0	1,418	11,735	20,645	129,566	17,109	8,614	2,091	4,159	2,720	697	13	0	0	0	0	0	198,952
2005	0	0	1,671	1,880	21,593	34,865	59,620	203,276	15,698	9,339	2,128	3,263	1,651	197	46	0	0	0	0	355,227
2006	0	0	40	30,887	1,233	24,087	32,982	38,394	149,418	12,705	3,890	608	1,546	795	202	27	0	0	0	296,814
2007	0	0	638	3,164	162,193	5,450	24,982	20,350	38,260	133,950	9,118	3,480	178	551	468	24	0	0	0	402,806
2008	0	0	783	14,041	7,237	167,160	1,605	19,335	15,741	21,632	87,855	4,089	3,192	52	0	0	0	0	0	342,722
2009	0	0	617	12,399	27,983	8,209	141,524	2,558	13,182	4,995	9,486	41,845	1,080	1,377	156	14	0	0	0	265,425
2010	0	0	272	1,752	15,644	21,283	7,892	100,695	1,062	4,866	3,989	4,816	26,701	1,067	184	45	0	14	0	190,282
2011	0	0	2,456	1,976	2,537	22,135	19,859	4,846	57,999	713	4,201	3,959	2,966	15,412	566	124	18	9	26	139,802
2012	0	9	7,854	24,983	10,188	6,127	32,080	14,431	5,383	49,563	438	2,998	1,871	2,743	8,487	201	78	40	6	167,480
2013	0	9	1,358	45,812	16,253	8,189	3,974	18,810	16,395	3,426	30,583	0	0	0	0	2,164	0	0	0	146,973

Table A.37. Mean weights-at-age (kg) of recreationally landed Gulf of Maine haddock from 1977 to 2013. *Note that recreational catch estimates are not available pre-1981.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Age15	Age16	Age17	Age18
1977																			
1978																			
1979																			
1980																			
1981			0.747	1.424	1.943	2.771	3.346	3.729	4.647	3.299	7.943			3.950	3.479				
1982			0.836	1.144	1.691	2.559	2.728	3.143	3.289	4.701	4.040		4.310		2.654				
1983			0.725	1.109	1.619	2.136	3.034	3.803	4.138	4.186	4.630	4.074	2.782						
1984			0.662	1.141	1.638	2.344	3.204	3.415	4.182	3.903		4.825	2.531						
1985			0.855	0.971	1.396	1.824	2.366	3.479	4.171	4.730	3.960		5.014						
1986		0.507		1.287	1.514	2.141	2.379	3.126	3.647	3.998									
1987			0.951	1.447	1.978	2.353	2.624	3.344	4.108	5.329	7.276								
1988				1.442	1.875	2.633	2.735	4.115	5.141	5.208									
1989			1.238	1.994	1.889	2.788	2.845	3.476	3.048										
1990				2.097		1.773	2.654	4.697	6.945										
1991			1.081	1.178	2.099	2.296	2.296	2.296											
1992																			
1993			1.085	1.398	1.960	2.689	2.808												
1994			1.147	1.485	1.971	2.835	2.720	2.977				2.913							
1995			1.281	1.203	1.543	2.115	2.654	2.184											
1996			1.313	1.366	1.705	1.915	2.725	2.502	2.237	3.418	3.479			2.782					
1997			1.211	1.740	1.556	2.271	2.864	3.267	3.601	3.735	3.632	2.531							
1998			1.401	1.591	1.819	1.919	2.431	3.989	2.857	2.291	2.913								
1999				1.353	1.602	1.673	1.911	2.081	2.454	2.573	3.188		2.913						
2000			1.269	1.302	1.627	1.857	1.990	2.193	2.413	2.282	3.671	2.866	3.582	4.115					
2001			1.131	1.294	1.433	1.798	2.225	2.205	2.594	2.444	3.521	2.371				3.479			
2002				1.534	1.604	1.806	2.136	2.478	2.125	2.325	2.358	2.586			3.188				
2003			0.916	1.359	1.559	1.631	1.815	1.970	2.137	2.139	2.216	1.872	2.396	1.971					
2004		0.121		0.976	1.392	1.395	1.558	1.693	1.834	1.939	1.881	2.180	2.133	3.331					
2005			0.718	1.039	1.195	1.438	1.394	1.545	1.770	1.849	2.506	2.109	2.469	2.309	2.531				
2006			0.595	1.084	1.199	1.410	1.554	1.446	1.578	1.748	1.911	2.234	1.911	1.923	2.242	2.654			
2007			0.679	1.050	1.196	1.204	1.462	1.605	1.490	1.621	1.773	1.797	2.754	1.980	2.245	3.048			
2008			0.823	1.129	1.274	1.438	1.677	1.759	1.719	1.629	1.785	2.094	1.896	3.083					
2009			0.678	0.953	1.163	1.285	1.548	1.471	1.779	1.926	1.717	1.818	2.171	1.610	2.544	2.782			
2010			1.113	0.852	1.135	1.387	1.485	1.680	2.077	2.002	2.046	1.889	1.946	2.162	2.220	2.531		3.788	
2011			0.800	1.093	1.188	1.345	1.486	1.627	1.735	1.793	1.919	1.826	1.792	1.974	1.912	2.428	2.654	2.913	2.412
2012		0.393	0.853	1.075	1.164	1.342	1.365	1.626	1.659	1.777	2.352	1.970	2.005	1.762	1.981	2.685	2.720	3.331	2.913
2013		0.507	0.870	1.129	1.461	1.830	1.477	1.662	2.162	1.727	2.189					1.871			

Table A.38. Gulf of Maine haddock recreational VTR landings per unit effort index (LPUE) from 1994 to 2013.

Year	Index	Std. Error	Variance	CV	Lower 95% CL	Upper 95% CL
1994	1.000			0.000		
1995	1.865	0.087	0.008	0.047	1.566	2.205
1996	1.271	0.090	0.008	0.070	1.063	1.509
1997	2.170	0.087	0.008	0.040	1.823	2.563
1998	2.830	0.086	0.007	0.030	2.382	3.338
1999	2.536	0.086	0.007	0.034	2.134	2.991
2000	3.673	0.083	0.007	0.023	3.109	4.311
2001	2.846	0.082	0.007	0.029	2.416	3.329
2002	3.571	0.081	0.007	0.023	3.038	4.170
2003	3.167	0.081	0.007	0.026	2.693	3.700
2004	5.442	0.080	0.006	0.015	4.635	6.348
2005	6.406	0.080	0.006	0.012	5.462	7.467
2006	6.467	0.080	0.006	0.012	5.514	7.538
2007	5.454	0.080	0.006	0.015	4.650	6.356
2008	4.986	0.080	0.006	0.016	4.248	5.814
2009	6.154	0.080	0.006	0.013	5.242	7.178
2010	4.467	0.080	0.006	0.018	3.805	5.209
2011	4.229	0.080	0.006	0.019	3.602	4.934
2012	4.407	0.080	0.006	0.018	3.755	5.140
2013	3.851	0.080	0.006	0.021	3.279	4.493

Table A.39. Total Gulf of Maine haddock recreational dead discards-at-age (numbers) from 1977 to 2013 ***using an assumption of 50% mortality of recreational releases***. *Note that recreational catch estimates are not available pre-1981.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Total
1977	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1978	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1979	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1980	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1981	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1982	0	20	24	14	0	0	0	0	0	0	0	0	0	0	0	58
1983	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1984	0	132	623	48	0	0	0	0	0	0	0	0	0	0	0	803
1985	0	8	19	17	0	0	0	0	0	0	0	0	0	0	0	44
1986	0	151	54	0	0	0	0	0	0	0	0	0	0	0	0	205
1987	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1988	0	305	471	637	0	0	0	0	0	0	0	0	0	0	0	1,413
1989	0	222	1,826	315	80	0	0	0	0	0	0	0	0	0	0	2,443
1990	0	76	15	41	0	0	0	0	0	0	0	0	0	0	0	132
1991	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1992	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1993	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1994	0	38	731	50	0	0	0	0	0	0	0	0	0	0	0	819
1995	0	739	15,505	4,430	0	0	0	0	0	0	0	0	0	0	0	20,674
1996	0	212	588	3,290	0	0	0	0	0	0	0	0	0	0	0	4,090
1997	0	852	883	3,550	2,199	0	0	0	0	0	0	0	0	0	0	7,484
1998	0	152	3,357	68	782	185	0	0	0	0	0	0	0	0	0	4,544
1999	0	2,198	2,963	1,447	477	339	341	165	0	0	0	0	0	0	0	7,930
2000	0	488	31,429	13,712	1,932	213	274	0	0	0	0	0	0	0	0	48,048
2001	0	0	12,866	33,558	4,495	935	1,090	483	0	0	0	0	0	0	0	53,427
2002	0	0	1,303	11,082	52,593	10,686	3,328	127	1,516	1,043	108	0	0	0	0	81,786
2003	0	0	9,192	1,671	15,388	86,835	7,525	2,077	814	580	0	0	0	0	0	124,082
2004	0	398	1,592	8,834	1,042	6,592	20,090	978	402	99	157	14	0	0	0	40,198
2005	0	2,909	24,213	2,593	3,999	3,623	5,150	8,381	40	4	0	0	0	0	0	50,912
2006	0	2,381	1,434	62,284	5,367	4,328	514	1,385	9,684	137	17	7	13	2	0	87,553
2007	0	0	8,703	5,319	34,998	77	1,479	187	795	3,671	27	30	0	10	0	55,296
2008	0	1,683	11,293	16,559	2,442	47,245	1,546	3,487	84	1,086	3,735	17	7	0	0	89,184
2009	0	0	2,354	21,927	4,934	844	2,245	31	33	0	78	234	0	10	0	32,690
2010	0	1,678	5,085	9,926	2,508	1,836	411	1,494	0	26	0	105	420	0	0	23,489
2011	0	3,279	12,043	66	122	1,212	195	118	474	5	42	60	0	162	0	17,778
2012	0	1,623	75,709	11,174	3,437	79	1,398	130	65	1,096	21	11	26	13	88	94,870
2013	0	5,595	22,010	202,497	14,427	2,874	1,471	2,493	0	0	2,209	0	0	0	0	253,576

Table A.40. Total Gulf of Maine haddock recreational dead discards-at-age (numbers) from 1977 to 2013 ***using an assumption of 100% mortality of recreational releases***. *Note that recreational catch estimates are not available pre-1981.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14	Total
1977	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1978	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1979	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1980	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1981	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1982	0	40	47	27	0	0	0	0	0	0	0	0	0	0	0	114
1983	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1984	0	263	1,246	95	0	0	0	0	0	0	0	0	0	0	0	1,604
1985	0	16	38	33	0	0	0	0	0	0	0	0	0	0	0	87
1986	0	302	108	0	0	0	0	0	0	0	0	0	0	0	0	410
1987	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1988	0	609	942	1,274	0	0	0	0	0	0	0	0	0	0	0	2,825
1989	0	444	3,651	629	160	0	0	0	0	0	0	0	0	0	0	4,884
1990	0	152	30	82	0	0	0	0	0	0	0	0	0	0	0	264
1991	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1992	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1993	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1994	0	75	1,462	99	0	0	0	0	0	0	0	0	0	0	0	1,636
1995	0	1,477	31,010	8,860	0	0	0	0	0	0	0	0	0	0	0	41,347
1996	0	423	1,175	6,580	0	0	0	0	0	0	0	0	0	0	0	8,178
1997	0	1,704	1,766	7,100	4,398	0	0	0	0	0	0	0	0	0	0	14,968
1998	0	303	6,713	135	1,564	370	0	0	0	0	0	0	0	0	0	9,085
1999	0	4,396	5,925	2,894	953	677	682	330	0	0	0	0	0	0	0	15,857
2000	0	976	62,857	27,423	3,863	426	547	0	0	0	0	0	0	0	0	96,092
2001	0		25,731	67,115	8,990	1,870	2,180	965	0	0	0	0	0	0	0	106,851
2002	0		2,606	22,163	105,185	21,371	6,655	253	3,032	2,085	216	0	0	0	0	163,566
2003	0		18,384	3,342	30,775	173,669	15,049	4,153	1,628	1,159	0	0	0	0	0	248,159
2004	0	796	3,184	17,668	2,084	13,184	40,180	1,956	803	198	313	27	0	0	0	80,393
2005	0	5,818	48,425	5,186	7,998	7,246	10,299	16,761	79	7	0	0	0	0	0	101,819
2006	0	4,762	2,867	124,568	10,734	8,656	1,028	2,770	19,368	273	33	13	25	4	0	175,101
2007	0		17,405	10,637	69,996	153	2,958	374	1,589	7,341	54	59	0	19	0	110,585
2008	0	3,366	22,586	33,118	4,884	94,489	3,092	6,974	168	2,172	7,469	33	14	0	0	178,365
2009	0		4,708	43,854	9,867	1,688	4,489	61	65	0	155	467	0	19	0	65,373
2010	0	3,355	10,170	19,851	5,015	3,671	821	2,988	0	52	0	210	839	0	0	46,972
2011	0	6,557	24,085	131	243	2,423	390	235	947	9	83	120	0	323	0	35,546
2012	0	3,245	151,417	22,348	6,874	157	2,795	259	130	2,191	41	21	51	25	175	189,729
2013	0	11,189	44,020	404,994	28,853	5,748	2,941	4,985	0	0	4,417	0	0	0	0	507,147

Table A.41. Mean weights-at-age (kg) of recreationally released Gulf of Maine haddock from 1977 to 2013. *Note that recreational catch estimates are not available pre-1981.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14
1977															
1978															
1979															
1980															
1981															
1982		0.357	0.428	0.429											
1983															
1984		0.350	0.412	0.429											
1985		0.347	0.379	0.455											
1986		0.380	0.467												
1987															
1988		0.331	0.412	0.540											
1989		0.307	0.520	0.741	0.865										
1990		0.403	0.621	0.765											
1991															
1992															
1993															
1994		0.288	0.528	0.916											
1995		0.327	0.620	0.869											
1996		0.418	0.650	0.831											
1997		0.350	0.499	0.720	0.890										
1998		0.269	0.766	0.979	0.836	0.927									
1999		0.313	0.651	0.767	0.899	0.891	0.861	0.799							
2000		0.272	0.591	0.680	0.893	0.943	0.951								
2001			0.726	0.829	0.901	0.937	0.812	0.856							
2002			0.535	0.873	1.085	1.206	1.371	1.490	1.301	1.425	1.531				
2003			0.592	0.964	1.005	1.085	1.062	1.173	0.972	1.226					
2004		0.196	0.595	0.761	1.107	0.912	0.980	1.108	1.131	1.152	1.082	1.259			
2005		0.294	0.606	0.580	0.857	0.866	0.913	0.913	1.533	1.589					
2006		0.237	0.504	0.729	0.624	0.863	1.286	1.058	0.964	1.432	1.687	1.911	1.720	1.773	
2007			0.672	0.768	0.845	1.016	0.878	1.182	1.079	0.992	1.358	1.410		1.502	
2008		0.148	0.535	0.743	0.929	0.874	0.845	0.907	1.485	1.083	1.085	1.653	1.972		
2009			0.726	0.702	0.766	0.859	1.020	1.387	1.361		1.244	1.278		1.418	
2010		0.393	0.548	0.758	0.958	0.966	1.113	1.439		1.589		1.589	1.470		
2011		0.408	0.571	1.017	1.061	0.883	1.444	1.194	1.630	2.076	2.076	1.779		1.966	
2012		0.243	0.527	0.669	0.746	1.134	0.897	1.244	2.037	1.103	3.422	2.296	3.201	2.105	3.006
2013		0.351	0.573	0.818	1.065	0.692	1.141	1.168			2.179				

Table A.42. Total catch-at-age (numbers, 000s of fish) of Gulf of Maine haddock from 1977 to 2013 with an age 9^+ group ***assuming 50% mortality of recreational releases.*** ***<u>This formulation is used as the 'base' case.</u>*** **Only ages 1 through the 9^+group are used as assessment model inputs.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9^+
1977	0.0	39.8	1763.0	53.2	367.0	184.6	189.3	0.0	0.0	2.4
1978	0.0	0.0	374.7	2291.4	172.4	363.0	208.7	10.6	0.0	5.3
1979	0.0	0.0	67.3	559.6	1577.0	183.1	99.1	45.3	10.9	0.0
1980	0.0	0.0	884.8	104.1	755.8	1366.8	143.8	95.6	27.8	25.8
1981	0.0	2.1	1604.7	721.6	293.7	343.0	545.1	92.2	117.4	27.1
1982	0.0	30.4	620.6	1519.4	620.7	100.6	301.0	477.5	107.4	75.9
1983	0.1	10.8	12.4	836.5	976.3	791.3	148.6	253.0	348.1	115.7
1984	0.0	1.2	89.0	49.9	598.0	256.7	365.0	62.2	64.8	147.6
1985	0.0	0.9	30.2	349.6	85.9	356.2	152.0	242.0	47.4	54.6
1986	0.0	4.3	10.8	183.5	358.8	81.3	114.0	86.4	102.5	14.7
1987	0.0	0.0	20.6	34.7	106.1	48.8	34.4	56.9	33.8	16.5
1988	0.0	0.3	0.5	12.4	12.3	54.8	55.6	7.6	15.0	4.1
1989	0.0	1.4	23.2	3.5	42.4	19.3	24.0	15.0	0.8	0.9
1990	0.0	7.0	2.0	143.1	1.7	28.8	17.6	27.5	4.1	0.0
1991	0.0	3.1	7.2	16.3	58.6	28.4	27.9	12.6	5.8	3.1
1992	0.0	1.8	13.1	94.4	36.5	19.1	2.2	1.1	0.0	1.9
1993	0.0	3.7	20.1	36.3	23.0	9.9	11.0	4.6	1.7	1.2
1994	0.1	6.5	23.7	44.5	13.6	3.4	9.2	5.7	1.7	0.7
1995	0.4	2.7	71.3	90.5	75.7	10.2	6.3	4.7	4.3	3.0
1996	0.0	2.8	23.5	129.5	56.5	16.4	4.1	7.1	5.6	1.2
1997	0.0	1.7	7.3	166.8	256.8	90.1	18.9	6.9	2.8	2.3
1998	1.0	5.8	23.8	25.1	132.7	192.8	52.7	17.4	8.6	7.6
1999	0.1	5.3	3.8	39.5	65.8	96.8	69.2	38.5	7.1	5.9
2000	0.0	2.4	68.6	66.1	106.8	65.1	128.5	72.1	31.8	25.7
2001	0.0	0.3	29.5	235.1	133.6	96.8	87.3	80.7	40.4	24.1
2002	0.0	0.4	2.4	27.8	275.3	117.1	110.4	32.1	70.4	68.0
2003	0.0	0.1	10.8	6.9	54.1	506.9	90.5	63.0	21.6	70.3
2004	0.0	1.8	1.9	14.1	33.0	72.0	512.7	59.7	34.0	51.1
2005	0.0	3.1	33.6	6.3	49.3	84.8	138.5	534.9	53.7	71.8
2006	0.0	2.5	2.2	123.6	8.5	52.7	71.7	83.5	367.0	61.0
2007	0.0	7.8	24.9	17.3	332.7	11.4	54.4	43.2	87.9	371.1
2008	0.0	1.8	18.5	55.8	19.1	407.6	5.0	42.2	29.6	225.3
2009	0.0	0.1	3.4	40.2	51.1	15.1	294.9	5.3	32.0	146.8
2010	0.0	2.4	6.7	13.8	39.6	52.2	19.1	294.3	3.5	134.2
2011	0.0	6.3	19.2	4.5	4.7	50.9	47.4	16.3	181.0	93.3
2012	0.0	2.6	110.6	48.8	20.0	12.3	67.2	37.8	13.8	204.4
2013	1.2	24.0	37.0	317.5	48.1	18.6	9.2	41.1	27.7	76.9

Table A.43. Total catch-at-age (numbers, 000s of fish) of Gulf of Maine haddock from 1977 to 2013 with an age 9^+ group ***assuming 0% mortality of recreational releases.*** ***<u>This formulation is used for model sensitivity only</u>.*** Only *ages 1 through the* 9^+*group are used as assessment model inputs.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9$^+$
1977	0.0	39.8	1763.0	53.2	367.0	184.6	189.3	0.0	0.0	2.4
1978	0.0	0.0	374.7	2291.4	172.4	363.0	208.7	10.6	0.0	5.3
1979	0.0	0.0	67.3	559.6	1577.0	183.1	99.1	45.3	10.9	0.0
1980	0.0	0.0	884.8	104.1	755.8	1366.8	143.8	95.6	27.8	25.8
1981	0.0	2.1	1604.7	721.6	293.7	343.0	545.1	92.2	117.4	27.1
1982	0.0	30.4	620.6	1519.4	620.7	100.6	301.0	477.5	107.4	75.9
1983	0.1	10.8	12.4	836.5	976.3	791.3	148.6	253.0	348.1	115.7
1984	0.0	1.1	88.4	49.8	598.0	256.7	365.0	62.2	64.8	147.6
1985	0.0	0.9	30.2	349.6	85.9	356.2	152.0	242.0	47.4	54.6
1986	0.0	4.1	10.8	183.5	358.8	81.3	114.0	86.4	102.5	14.7
1987	0.0	0.0	20.6	34.7	106.1	48.8	34.4	56.9	33.8	16.5
1988	0.0	0.0	0.0	11.8	12.3	54.8	55.6	7.6	15.0	4.1
1989	0.0	1.2	21.4	3.2	42.3	19.3	24.0	15.0	0.8	0.9
1990	0.0	6.9	2.0	143.0	1.7	28.8	17.6	27.5	4.1	0.0
1991	0.0	3.1	7.2	16.3	58.6	28.4	27.9	12.6	5.8	3.1
1992	0.0	1.8	13.1	94.4	36.5	19.1	2.2	1.1	0.0	1.9
1993	0.0	3.7	20.1	36.3	23.0	9.9	11.0	4.6	1.7	1.2
1994	0.1	6.4	23.0	44.5	13.6	3.4	9.2	5.7	1.7	0.7
1995	0.4	2.0	55.8	86.1	75.7	10.2	6.3	4.7	4.3	3.0
1996	0.0	2.6	22.9	126.2	56.5	16.4	4.1	7.1	5.6	1.2
1997	0.0	0.8	6.5	163.2	254.6	90.1	18.9	6.9	2.8	2.3
1998	1.0	5.7	20.4	25.0	132.0	192.6	52.7	17.4	8.6	7.6
1999	0.1	3.1	0.8	38.0	65.4	96.4	68.8	38.3	7.1	5.9
2000	0.0	1.9	37.2	52.4	104.8	64.9	128.2	72.1	31.8	25.7
2001	0.0	0.3	16.7	201.6	129.1	95.8	86.3	80.3	40.4	24.1
2002	0.0	0.4	1.1	16.7	222.7	106.5	107.1	32.0	68.9	66.9
2003	0.0	0.1	1.6	5.3	38.7	420.1	83.0	60.9	20.7	69.7
2004	0.0	1.4	0.3	5.3	31.9	65.4	492.6	58.7	33.6	50.9
2005	0.0	0.2	9.4	3.7	45.3	81.2	133.4	526.5	53.6	71.8
2006	0.0	0.1	0.7	61.3	3.1	48.3	71.2	82.1	357.3	60.9
2007	0.0	7.8	16.2	12.0	297.7	11.3	52.9	43.0	87.1	367.4
2008	0.0	0.1	7.2	39.3	16.6	360.4	3.5	38.7	29.5	220.4
2009	0.0	0.1	1.0	18.3	46.1	14.2	292.7	5.3	32.0	146.4
2010	0.0	0.7	1.6	3.8	37.1	50.4	18.7	292.8	3.5	133.6
2011	0.0	3.0	7.1	4.5	4.6	49.7	47.2	16.2	180.5	93.0
2012	0.0	1.0	34.9	37.7	16.6	12.2	65.8	37.6	13.7	203.2
2013	1.2	18.4	15.0	115.0	33.7	15.8	7.7	38.6	27.7	74.7

Table A.44. Total catch-at-age (numbers, 000s of fish) of Gulf of Maine haddock from 1977 to 2013 with an age 9^{+} group ***assuming 100% mortality of recreational releases. <u>This formulation is used for model sensitivity only</u>.*** Only *ages 1 through the 9^{+}group are used as assessment model inputs.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9^{+}
1977	0.0	39.8	1763.0	53.2	367.0	184.6	189.3	0.0	0.0	2.4
1978	0.0	0.0	374.7	2291.4	172.4	363.0	208.7	10.6	0.0	5.3
1979	0.0	0.0	67.3	559.6	1577.0	183.1	99.1	45.3	10.9	0.0
1980	0.0	0.0	884.8	104.1	755.8	1366.8	143.8	95.6	27.8	25.8
1981	0.0	2.1	1604.7	721.6	293.7	343.0	545.1	92.2	117.4	27.1
1982	0.0	30.4	620.6	1519.4	620.7	100.6	301.0	477.5	107.4	75.9
1983	0.1	10.8	12.4	836.5	976.3	791.3	148.6	253.0	348.1	115.7
1984	0.0	1.3	89.6	49.9	598.0	256.7	365.0	62.2	64.8	147.6
1985	0.0	0.9	30.2	349.6	85.9	356.2	152.0	242.0	47.4	54.6
1986	0.0	4.4	10.9	183.5	358.8	81.3	114.0	86.4	102.5	14.7
1987	0.0	0.0	20.6	34.7	106.1	48.8	34.4	56.9	33.8	16.5
1988	0.0	0.6	0.9	13.1	12.3	54.8	55.6	7.6	15.0	4.1
1989	0.0	1.6	25.0	3.8	42.5	19.3	24.0	15.0	0.8	0.9
1990	0.0	7.1	2.0	143.1	1.7	28.8	17.6	27.5	4.1	0.0
1991	0.0	3.1	7.2	16.3	58.6	28.4	27.9	12.6	5.8	3.1
1992	0.0	1.8	13.1	94.4	36.5	19.1	2.2	1.1	0.0	1.9
1993	0.0	3.7	20.1	36.3	23.0	9.9	11.0	4.6	1.7	1.2
1994	0.1	6.5	24.4	44.6	13.6	3.4	9.2	5.7	1.7	0.7
1995	0.4	3.5	86.8	95.0	75.7	10.2	6.3	4.7	4.3	3.0
1996	0.0	3.0	24.1	132.8	56.5	16.4	4.1	7.1	5.6	1.2
1997	0.0	2.5	8.2	170.3	259.0	90.1	18.9	6.9	2.8	2.3
1998	1.0	6.0	27.1	25.2	133.5	193.0	52.7	17.4	8.6	7.6
1999	0.1	7.5	6.8	40.9	66.3	97.1	69.5	38.6	7.1	5.9
2000	0.0	2.8	100.1	79.8	108.7	65.3	128.7	72.1	31.8	25.7
2001	0.0	0.3	42.4	268.7	138.1	97.7	88.4	81.2	40.4	24.1
2002	0.0	0.4	3.7	38.9	327.9	127.8	113.7	32.3	71.9	69.2
2003	0.0	0.1	20.0	8.6	69.5	593.7	98.0	65.0	22.4	70.8
2004	0.0	2.2	3.5	23.0	34.0	78.6	532.8	60.6	34.4	51.4
2005	0.0	6.0	57.8	8.9	53.3	88.4	143.6	543.3	53.7	71.8
2006	0.0	4.8	3.6	185.9	13.8	57.0	72.2	84.8	376.7	61.2
2007	0.0	7.8	33.6	22.7	367.7	11.5	55.9	43.4	88.7	374.8
2008	0.0	3.5	29.8	72.4	21.5	454.9	6.6	45.7	29.7	230.1
2009	0.0	0.1	5.7	62.2	56.0	15.9	297.2	5.4	32.0	147.1
2010	0.0	4.1	11.8	23.7	42.1	54.1	19.5	295.8	3.5	134.7
2011	0.0	9.6	31.2	4.6	4.9	52.1	47.6	16.5	181.5	93.5
2012	0.0	4.3	186.3	60.0	23.4	12.4	68.6	37.9	13.9	205.7
2013	1.2	29.6	59.0	520.0	62.5	21.5	10.6	43.6	27.7	79.1

Table A.45. Gulf of Maine haddock mean weights-at-age (kg) of the total catch from 1977 to 2013 with an age 9^+ group ***assuming 50% mortality of recreational releases. <u>This formulation is used as the 'base' case</u>.*** Mean catch weights-at-age in the 9^+ group were estimated using a numbers weighted approach. Cells shaded grey were imputed using a 5-year centered moving average, cells shaded blue were imputed using a time series average. **Only ages 1 through the 9^+group are used as assessment model inputs.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9$^+$
1977	0.061	0.113	0.757	1.163	2.008	2.558	3.358	3.709	3.587	4.686
1978	0.061	0.113	0.777	1.234	1.684	2.438	3.108	4.642	4.075	6.088
1979	0.061	0.337	0.774	1.155	1.805	2.261	2.659	2.775	3.587	4.724
1980	0.061	0.468	0.760	1.168	1.852	2.389	3.354	3.602	4.562	4.204
1981	0.061	0.560	0.685	1.516	1.978	2.641	3.026	3.657	4.184	3.917
1982	0.061	0.376	0.620	0.995	2.137	2.598	3.106	3.646	4.129	4.293
1983	0.048	0.181	0.667	1.200	1.727	2.376	2.969	3.373	3.719	4.215
1984	0.061	0.313	0.816	1.233	1.803	2.324	3.166	3.923	4.502	4.073
1985	0.061	0.315	0.980	1.068	1.859	2.339	2.652	3.588	4.090	4.153
1986	0.061	0.503	0.507	1.192	1.456	2.265	2.495	3.062	3.636	4.592
1987	0.061	0.350	0.856	1.592	2.008	2.402	2.609	3.272	4.236	5.279
1988	0.061	0.331	0.412	1.100	1.623	2.561	2.582	3.871	4.652	5.180
1989	0.061	0.251	1.126	1.779	1.824	2.343	2.864	3.543	4.545	4.244
1990	0.061	0.296	0.831	1.543	3.331	2.450	3.041	3.745	3.762	4.189
1991	0.061	0.347	1.459	1.880	2.657	3.027	2.958	3.350	4.433	3.881
1992	0.061	0.448	1.192	1.764	1.973	2.654	3.067	2.079	3.757	3.450
1993	0.061	0.364	0.885	1.592	2.041	2.436	3.035	3.393	3.422	3.657
1994	0.092	0.362	0.787	1.589	2.186	3.062	2.788	3.620	3.410	3.721
1995	0.017	0.275	0.802	1.156	1.774	2.525	3.526	4.133	5.209	5.665
1996	0.061	0.337	0.674	1.073	1.803	2.196	3.148	2.473	2.387	3.164
1997	0.061	0.354	0.891	1.802	1.662	2.330	2.977	2.985	3.063	3.607
1998	0.021	0.250	0.975	1.448	1.827	2.212	2.843	3.376	3.152	2.988
1999	0.072	0.266	0.611	1.309	1.608	1.765	1.926	2.281	3.033	3.295
2000	0.061	0.260	0.607	1.022	1.535	1.773	2.013	2.390	2.696	3.101
2001	0.061	0.242	0.889	1.260	1.490	1.811	2.210	2.243	2.483	2.532
2002	0.068	0.121	0.473	1.025	1.340	1.631	2.143	2.598	2.303	2.644
2003	0.061	0.318	0.583	0.887	1.230	1.468	1.770	2.134	2.425	2.513
2004	0.061	0.185	0.560	0.809	1.373	1.358	1.681	1.820	2.027	2.208
2005	0.061	0.286	0.583	0.815	1.139	1.464	1.443	1.684	1.954	2.297
2006	0.061	0.238	0.474	0.840	0.745	1.359	1.644	1.507	1.683	2.008
2007	0.061	0.243	0.560	0.777	1.121	1.203	1.510	1.625	1.578	1.714
2008	0.061	0.156	0.544	0.995	1.207	1.341	1.339	1.700	1.740	1.758
2009	0.061	0.304	0.699	0.809	1.135	1.282	1.625	1.563	1.877	1.947
2010	0.086	0.350	0.609	0.785	1.129	1.406	1.563	1.731	2.131	2.069
2011	0.064	0.341	0.588	1.029	1.191	1.401	1.602	1.801	1.915	2.113
2012	0.064	0.246	0.538	0.954	1.106	1.406	1.451	1.742	1.815	1.979
2013	0.076	0.283	0.550	0.870	1.267	1.498	1.486	1.658	2.051	2.104
2009-2013 average	0.070	0.305	0.597	0.890	1.166	1.399	1.545	1.699	1.958	2.043

Table A.46. Gulf of Maine haddock mean weights-at-age (kg) of the total catch from 1977 to 2013 with an age 9^+ group ***assuming 0% mortality of recreational releases. <u>This formulation is used for model sensitivity only</u>.*** Mean catch weights-at-age in the 9^+ group were estimated using a numbers weighted approach. Cells shaded grey were imputed using a 5-year centered moving average, cells shaded blue were imputed using a time series average. **Only ages 1 through the 9^+group are used as assessment model inputs.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	$Age9^+$
1977	0.061	0.113	0.757	1.163	2.008	2.558	3.358	3.709	3.587	4.686
1978	0.061	0.113	0.777	1.234	1.684	2.438	3.108	4.642	4.075	6.088
1979	0.061	0.337	0.774	1.155	1.805	2.261	2.659	2.775	3.587	4.724
1980	0.061	0.468	0.760	1.168	1.852	2.389	3.354	3.602	4.562	4.204
1981	0.061	0.560	0.685	1.516	1.978	2.641	3.026	3.657	4.184	3.917
1982	0.061	0.376	0.620	0.995	2.137	2.598	3.106	3.646	4.129	4.293
1983	0.048	0.181	0.667	1.200	1.727	2.376	2.969	3.373	3.719	4.215
1984	0.061	0.308	0.819	1.233	1.803	2.324	3.166	3.923	4.502	4.073
1985	0.061	0.315	0.981	1.068	1.859	2.339	2.652	3.588	4.090	4.153
1986	0.061	0.507	0.507	1.192	1.456	2.265	2.495	3.062	3.636	4.592
1987	0.061	0.354	0.856	1.592	2.008	2.402	2.609	3.272	4.236	5.279
1988	0.061	0.347	0.844	1.130	1.623	2.561	2.582	3.871	4.652	5.180
1989	0.061	0.240	1.178	1.882	1.826	2.343	2.864	3.543	4.545	4.244
1990	0.061	0.295	0.833	1.543	3.331	2.450	3.041	3.745	3.762	4.189
1991	0.061	0.347	1.459	1.880	2.657	3.027	2.958	3.350	4.433	3.881
1992	0.061	0.448	1.192	1.764	1.973	2.654	3.067	2.079	3.757	3.450
1993	0.061	0.364	0.885	1.592	2.041	2.436	3.035	3.393	3.422	3.657
1994	0.092	0.362	0.796	1.590	2.186	3.062	2.788	3.620	3.410	3.721
1995	0.017	0.256	0.853	1.171	1.774	2.525	3.526	4.133	5.209	5.665
1996	0.061	0.330	0.675	1.080	1.803	2.196	3.148	2.473	2.387	3.164
1997	0.061	0.358	0.945	1.825	1.669	2.330	2.977	2.985	3.063	3.607
1998	0.021	0.250	1.009	1.449	1.833	2.213	2.843	3.376	3.152	2.988
1999	0.072	0.233	0.468	1.329	1.613	1.768	1.931	2.288	3.033	3.295
2000	0.061	0.257	0.620	1.112	1.547	1.776	2.015	2.390	2.696	3.101
2001	0.061	0.242	1.015	1.332	1.511	1.819	2.228	2.251	2.483	2.532
2002	0.068	0.121	0.398	1.125	1.400	1.673	2.167	2.602	2.325	2.665
2003	0.061	0.318	0.529	0.862	1.319	1.547	1.834	2.167	2.482	2.523
2004	0.061	0.182	0.381	0.889	1.381	1.403	1.709	1.831	2.038	2.214
2005	0.061	0.173	0.526	0.978	1.164	1.490	1.464	1.697	1.954	2.297
2006	0.061	0.287	0.415	0.953	0.956	1.403	1.646	1.514	1.702	2.009
2007	0.061	0.243	0.500	0.781	1.154	1.204	1.527	1.627	1.582	1.721
2008	0.061	0.289	0.558	1.101	1.247	1.403	1.559	1.772	1.741	1.773
2009	0.061	0.304	0.634	0.938	1.174	1.308	1.629	1.564	1.877	1.949
2010	0.086	0.253	0.799	0.854	1.140	1.422	1.573	1.732	2.131	2.071
2011	0.064	0.268	0.617	1.030	1.195	1.414	1.603	1.806	1.916	2.114
2012	0.064	0.249	0.561	1.039	1.181	1.407	1.463	1.744	1.814	1.983
2013	0.076	0.262	0.516	0.961	1.353	1.645	1.552	1.689	2.051	2.102
2009-2013 average	0.070	0.267	0.625	0.964	1.209	1.439	1.564	1.707	1.958	2.044

Table A.47. Gulf of Maine haddock mean weights-at-age (kg) of the total catch from 1977 to 2013 with an age 9^+ group ***assuming 100% mortality of recreational releases.*** ***<u>This formulation is used for model sensitivity only</u>.*** Mean catch weights-at-age in the 9^+ group were estimated using a numbers weighted approach. Cells shaded grey were imputed using a 5-year centered moving average, cells shaded blue were imputed using a time series average. **Only ages 1 through the 9^+group are used as assessment model inputs.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
1977	0.061	0.113	0.757	1.163	2.008	2.558	3.358	3.709	3.587	4.686
1978	0.061	0.113	0.777	1.234	1.684	2.438	3.108	4.642	4.075	6.088
1979	0.061	0.337	0.774	1.155	1.805	2.261	2.659	2.775	3.587	4.724
1980	0.061	0.468	0.760	1.168	1.852	2.389	3.354	3.602	4.562	4.204
1981	0.061	0.560	0.685	1.516	1.978	2.641	3.026	3.657	4.184	3.917
1982	0.061	0.376	0.620	0.995	2.137	2.598	3.106	3.646	4.129	4.293
1983	0.048	0.181	0.667	1.200	1.727	2.376	2.969	3.373	3.719	4.215
1984	0.061	0.316	0.813	1.232	1.803	2.324	3.166	3.923	4.502	4.073
1985	0.061	0.316	0.980	1.068	1.859	2.339	2.652	3.588	4.090	4.153
1986	0.061	0.498	0.507	1.192	1.456	2.265	2.495	3.062	3.636	4.592
1987	0.061	0.351	0.856	1.592	2.008	2.402	2.609	3.272	4.236	5.279
1988	0.061	0.331	0.412	1.073	1.623	2.561	2.582	3.871	4.652	5.180
1989	0.061	0.259	1.082	1.693	1.822	2.343	2.864	3.543	4.545	4.244
1990	0.061	0.297	0.830	1.543	3.331	2.450	3.041	3.745	3.762	4.189
1991	0.061	0.347	1.459	1.880	2.657	3.027	2.958	3.350	4.433	3.881
1992	0.061	0.448	1.192	1.764	1.973	2.654	3.067	2.079	3.757	3.450
1993	0.061	0.364	0.885	1.592	2.041	2.436	3.035	3.393	3.422	3.657
1994	0.092	0.361	0.780	1.588	2.186	3.062	2.788	3.620	3.410	3.721
1995	0.017	0.286	0.769	1.143	1.774	2.525	3.526	4.133	5.209	5.665
1996	0.061	0.342	0.674	1.067	1.803	2.196	3.148	2.473	2.387	3.164
1997	0.061	0.353	0.849	1.779	1.656	2.330	2.977	2.985	3.063	3.607
1998	0.021	0.251	0.949	1.446	1.822	2.211	2.843	3.376	3.152	2.988
1999	0.072	0.280	0.628	1.290	1.603	1.762	1.921	2.275	3.033	3.295
2000	0.061	0.262	0.602	0.963	1.524	1.771	2.011	2.390	2.696	3.101
2001	0.061	0.242	0.840	1.207	1.471	1.802	2.193	2.234	2.483	2.532
2002	0.068	0.121	0.495	0.981	1.299	1.595	2.120	2.593	2.282	2.624
2003	0.061	0.318	0.587	0.902	1.180	1.412	1.715	2.104	2.372	2.502
2004	0.061	0.187	0.576	0.790	1.364	1.321	1.654	1.808	2.017	2.202
2005	0.061	0.290	0.593	0.747	1.118	1.439	1.424	1.672	1.954	2.297
2006	0.061	0.238	0.486	0.803	0.698	1.321	1.641	1.499	1.664	2.006
2007	0.061	0.243	0.589	0.775	1.095	1.202	1.493	1.623	1.573	1.707
2008	0.061	0.152	0.540	0.937	1.175	1.293	1.223	1.640	1.740	1.744
2009	0.061	0.304	0.710	0.771	1.102	1.260	1.620	1.562	1.876	1.946
2010	0.086	0.368	0.583	0.774	1.118	1.391	1.553	1.729	2.131	2.067
2011	0.064	0.364	0.581	1.029	1.188	1.389	1.602	1.797	1.915	2.113
2012	0.064	0.245	0.533	0.901	1.053	1.404	1.439	1.741	1.817	1.975
2013	0.076	0.296	0.558	0.850	1.220	1.390	1.438	1.630	2.051	2.106
2009-2013 average	0.070	0.315	0.593	0.865	1.137	1.367	1.531	1.692	1.958	2.041

Table A.48. Gulf of Maine haddock mean January 1/spawning stock weights-at-age (kg) from 1977 to 2013 with an age 9^+ group. Weights were estimated from catch weights using Rivard (1980, 1982) approach based on the catch weights under a 100% mortality assumption. **Only ages 1 through the* 9^+*group are used as assessment model inputs.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
1977	0.045	0.043	0.593	0.967	1.822	2.321	2.856	3.539	3.648	4.686
1978	0.026	0.083	0.296	0.967	1.400	2.213	2.820	3.948	3.888	6.088
1979	0.022	0.143	0.296	0.947	1.492	1.951	2.546	2.937	4.081	4.724
1980	0.020	0.169	0.506	0.951	1.463	2.077	2.754	3.095	3.558	4.204
1981	0.025	0.185	0.566	1.073	1.520	2.212	2.689	3.502	3.882	3.917
1982	0.035	0.151	0.589	0.826	1.800	2.267	2.864	3.322	3.886	4.293
1983	0.019	0.105	0.501	0.863	1.311	2.253	2.777	3.237	3.682	4.215
1984	0.027	0.123	0.384	0.907	1.471	2.003	2.743	3.413	3.897	4.073
1985	0.021	0.139	0.557	0.932	1.513	2.054	2.483	3.370	4.006	4.153
1986	0.025	0.174	0.400	1.081	1.247	2.052	2.416	2.850	3.612	4.592
1987	0.026	0.146	0.653	0.898	1.547	1.870	2.431	2.857	3.602	5.279
1988	0.030	0.142	0.380	0.958	1.607	2.268	2.490	3.178	3.902	5.180
1989	0.028	0.126	0.598	0.835	1.398	1.950	2.708	3.025	4.195	4.244
1990	0.026	0.135	0.464	1.292	2.375	2.113	2.669	3.275	3.651	4.189
1991	0.023	0.146	0.658	1.249	2.025	3.175	2.692	3.192	4.075	3.881
1992	0.025	0.165	0.643	1.604	1.926	2.656	3.047	2.480	3.548	3.450
1993	0.025	0.149	0.630	1.378	1.898	2.192	2.838	3.226	2.667	3.657
1994	0.052	0.148	0.533	1.186	1.866	2.500	2.606	3.315	3.402	3.721
1995	0.004	0.162	0.527	0.944	1.678	2.349	3.286	3.395	4.342	5.665
1996	0.025	0.076	0.439	0.906	1.436	1.974	2.819	2.953	3.141	3.164
1997	0.030	0.147	0.539	1.095	1.329	2.050	2.557	3.065	2.752	3.607
1998	0.006	0.124	0.579	1.108	1.800	1.914	2.574	3.170	3.067	2.988
1999	0.038	0.077	0.397	1.106	1.523	1.792	2.061	2.543	3.200	3.295
2000	0.031	0.137	0.411	0.778	1.402	1.685	1.882	2.143	2.477	3.101
2001	0.043	0.122	0.469	0.852	1.190	1.657	1.971	2.120	2.436	2.532
2002	0.031	0.086	0.346	0.908	1.252	1.532	1.955	2.385	2.258	2.624
2003	0.035	0.147	0.267	0.668	1.076	1.354	1.654	2.112	2.480	2.502
2004	0.028	0.107	0.428	0.681	1.109	1.249	1.528	1.761	2.060	2.202
2005	0.031	0.133	0.333	0.656	0.940	1.401	1.372	1.663	1.880	2.297
2006	0.031	0.121	0.375	0.690	0.722	1.215	1.537	1.461	1.668	2.006
2007	0.039	0.122	0.374	0.614	0.938	0.916	1.404	1.632	1.536	1.707
2008	0.027	0.096	0.362	0.743	0.954	1.190	1.213	1.565	1.681	1.744
2009	0.025	0.136	0.329	0.645	1.016	1.217	1.447	1.382	1.754	1.946
2010	0.042	0.150	0.421	0.741	0.928	1.238	1.399	1.674	1.825	2.067
2011	0.033	0.177	0.462	0.775	0.959	1.246	1.493	1.671	1.820	2.113
2012	0.030	0.125	0.441	0.724	1.041	1.292	1.414	1.670	1.807	1.975
2013	0.042	0.138	0.370	0.673	1.048	1.210	1.421	1.532	1.890	2.106
2009-2013 average	0.034	0.145	0.404	0.712	0.999	1.240	1.435	1.586	1.819	2.041

Table A.49. Summary of vessels and trawl doors used in the Northeast Fisheries Science Center (NEFSC) spring and fall surveys from 1963 to 2013. When survey indices are calibrated to single time series, the calibration is based on Albatross IV, Polyvalent door equivalents. *Note that the spring survey did not begin until 1968.*

Year	Spring	Autumn	Door
1963		Albatross IV	BMV
1964		Albatross IV	BMV
1965		Albatross IV	BMV
1966		Albatross IV	BMV
1967		Albatross IV	BMV
1968	Albatross IV	Albatross IV	BMV
1969	Albatross IV	Albatross IV	BMV
1970	Albatross IV	Albatross IV	BMV
1971	Albatross IV	Albatross IV	BMV
1972	Albatross IV	Albatross IV	BMV
1973	Albatross IV	Albatross IV	BMV
1974	Albatross IV	Albatross IV	BMV
1975	Albatross IV	Albatross IV	BMV
1976	Albatross IV	Albatross IV	BMV
1977	Albatross IV	Delaware II	BMV
1978	Albatross IV	Delaware II	BMV
1979	Albatross IV/Delaware II	Albatross IV/Delaware II	BMV
1980	Albatross IV/Delaware II	Delaware II	BMV
1981	Delaware II	Albatross IV/Delaware II	BMV
1982	Delaware II	Albatross IV	BMV
1983	Albatross IV	Albatross IV	BMV
1984	Albatross IV	Albatross IV	BMV
1985	Albatross IV	Albatross IV	Polyvalent
1986	Albatross IV	Albatross IV	Polyvalent
1987	Albatross IV/Delaware II	Albatross IV	Polyvalent
1988	Albatross IV	Albatross IV/Delaware II	Polyvalent
1989	Delaware II	Delaware II	Polyvalent
1990	Delaware II	Delaware II	Polyvalent
1991	Delaware II	Delaware II	Polyvalent
1992	Albatross IV	Albatross IV	Polyvalent
1993	Albatross IV	Delaware II	Polyvalent
1994	Delaware II	Albatross IV	Polyvalent
1995	Albatross IV	Albatross IV	Polyvalent
1996	Albatross IV	Albatross IV	Polyvalent
1997	Albatross IV	Albatross IV	Polyvalent
1998	Albatross IV	Albatross IV	Polyvalent
1999	Albatross IV	Albatross IV	Polyvalent
2000	Albatross IV	Albatross IV	Polyvalent
2001	Albatross IV	Albatross IV	Polyvalent
2002	Albatross IV	Albatross IV	Polyvalent
2003	Delaware II	Albatross IV	Polyvalent
2004	Albatross IV	Albatross IV	Polyvalent
2005	Albatross IV	Albatross IV	Polyvalent
2006	Albatross IV	Albatross IV	Polyvalent
2007	Albatross IV	Albatross IV	Polyvalent
2008	Albatross IV	Albatross IV	Polyvalent
2009	Henry B. Bigelow	Henry B. Bigelow	PolyIce oval
2010	Henry B. Bigelow	Henry B. Bigelow	PolyIce oval
2011	Henry B. Bigelow	Henry B. Bigelow	PolyIce oval
2012	Henry B. Bigelow	Henry B. Bigelow	PolyIce oval
2013	Henry B. Bigelow	Henry B. Bigelow	PolyIce oval

Table A.50. Summary of survey calibration coefficients for converting survey index values to Albatross IV, Polyvalent door equivalent units.

Calibration type	Index	Length (cm)	Calibration coefficient	CV	Source
Deleware II to Albatross IV	Biomass (weight)	*NA*	0.790	*NA*	Forrester et al., 1997
	Abundance (numbers)	*NA*	0.820	*NA*	
BMV door to Polyvalent door	Biomass (weight)	*NA*	1.510	*NA*	
	Abundance (numbers)	*NA*	1.490	*NA*	
Bigelow to Albatross IV	Biomass (weight), spring	*NA*	0.878	*NA*	Miller et al. 2010
	Biomass (weight), fall	*NA*	1.489	*NA*	
	Abundance (numbers)	≤ 18	2.626	0.07	Brooks et al. 2010
		19	2.581	0.07	
		20	2.535	0.07	
		21	2.489	0.07	
		22	2.444	0.06	
		23	2.398	0.06	
		24	2.352	0.06	
		25	2.307	0.06	
		26	2.261	0.06	
		27	2.216	0.06	
		28	2.170	0.05	
		29	2.124	0.05	
		30	2.079	0.05	
		31	2.033	0.05	
		32	1.988	0.05	
		33	1.942	0.04	
		34	1.896	0.04	
		35	1.851	0.04	
		36	1.805	0.04	
		37	1.759	0.04	
		38	1.714	0.03	
		39	1.668	0.03	
		40	1.623	0.03	
		41	1.577	0.03	
		42	1.531	0.03	
		43	1.486	0.03	
		44	1.440	0.03	
		45	1.394	0.04	
		46	1.349	0.04	
		47	1.303	0.04	
		48	1.258	0.05	
		49	1.212	0.05	
		50	1.166	0.06	
		≥ 51	1.164	0.06	

Table A.51. Summary of the differences in survey protocols between the RV Albatross IV (1963-2008) and the FSV Henry B. Bigelow (2009 - present) surveys. Adapted from Brooks et al. (2010).

Measure	**FSV Henry B Bigelow**	**RV Albatross IV**
Tow speed	3.0 knots SOG	3.8 knots SOG
Tow duration	20min	30 mins
Headrope height	3.5-4m	1-2m
Ground gear (cookies, rock hoppers, etc.)	Rockhopper Sweep Total Length-25.5m Center- 8.9m length, 16” rockhoppers. Wings- 8.2m each 14” rockhoppers	Roller Sweep Total Length-24.5m Center-5m length, 16” rollers. Wings- 9.75m each, 4” cookies.
Mesh	Poly webbing Forward Portion of trawl (jibs, upper and lower wing ends, 1st&2nd side panels, 1st bottom belly)12cm,4mm Square aft to codend:6cm, 2.5mm Codend: 12cm, 4mm dbl. Codend Liner: 2.54cm, knotless	Nylon webbing Body of trawl= 12.7cm Codend- 11.5cm Liner (codend and aft portion of top belly)-1.27cm knotless
Net design	4 Seam, 3 Bridle	Yankee 36 (recent years)
Door type	550 kg PolyIce oval	450 kg polyvalent
Other comments	Wing End to Door distance= 36.5m	Wing End to Door Distance= 9m

Table A.52. Summary of the sampling of Northeast Fisheries Science Center (NEFSC) Gulf of Maine offshore survey strata (26-28, 36-40) broken down by season (spring/fall) and time of day (day/night) between 1963 and spring 2013. The day/night classification is based on sunrise/sunset (zenith angle of 90°50'). This summary applies SHG tow selection criteria between 1963 and 2008 and the TOGA criteria since 2009. *Note that the spring survey did not begin until 1968.*

	Spring						Fall					
Year	Combined		Day		Night		Combined		Day		Night	
	Strata	Stations	Strata	Stations	Strata	Stations	Strata	Stations	Strata	Stations	Strata	Stations
1963							8	41	6	14	8	27
1964							8	37	8	12	8	25
1965							8	38	8	17	8	21
1966							8	37	7	19	7	18
1967							8	38	6	13	8	25
1968	8	39	6	21	8	18	8	39	7	16	8	23
1969	8	40	7	18	7	22	8	40	7	14	8	26
1970	8	40	5	13	7	27	8	42	8	17	8	25
1971	8	46	8	21	8	25	8	44	8	15	8	29
1972	8	43	8	24	8	19	8	43	6	17	7	26
1973	8	37	8	16	8	21	8	43	7	15	8	28
1974	8	40	8	22	8	18	8	46	7	21	7	25
1975	7	43	6	17	6	26	8	54	6	22	7	32
1976	8	53	7	25	7	28	8	44	6	15	8	29
1977	8	56	8	34	8	22	8	60	7	21	8	39
1978	8	52	8	28	8	24	8	101	8	47	7	54
1979	8	61	8	39	7	22	8	103	8	43	8	60
1980	8	39	8	19	6	20	8	41	8	18	8	23
1981	8	42	8	28	8	14	8	42	8	21	8	21
1982	8	42	7	25	7	17	8	43	8	19	8	24
1983	8	42	8	30	5	12	8	36	6	13	7	23
1984	8	39	8	24	8	15	8	40	6	16	7	24
1985	8	37	8	21	7	16	8	42	8	16	8	26
1986	8	41	7	20	8	21	8	42	6	16	7	26
1987	8	36	6	21	6	15	8	40	7	18	8	22
1988	8	42	8	26	7	16	8	41	7	20	7	21
1989	8	41	7	23	8	18	8	40	6	17	6	23
1990	8	41	7	19	8	22	8	41	7	18	8	23
1991	8	39	8	23	7	16	8	42	7	16	8	26
1992	8	41	8	25	7	16	8	40	7	16	8	24
1993	8	40	8	21	7	19	8	40	8	22	7	18
1994	8	42	8	28	7	14	8	40	7	14	8	26
1995	8	42	8	21	7	21	8	45	7	16	8	29
1996	8	41	8	20	8	21	8	42	8	20	7	22
1997	8	42	8	25	8	17	8	41	8	20	8	21
1998	8	63	8	30	8	33	8	56	7	27	8	29
1999	8	41	7	20	8	21	8	60	7	30	8	30
2000	8	42	7	26	7	16	8	41	7	14	8	27
2001	8	41	8	25	7	16	8	43	7	22	7	21
2002	8	44	8	23	8	21	8	38	8	21	8	17
2003	8	41	5	17	7	24	8	40	8	15	7	25
2004	8	39	8	25	7	14	8	37	6	13	7	24
2005	8	40	8	24	5	16	8	40	7	15	8	25
2006	8	48	8	25	8	23	8	47	7	20	7	27
2007	8	39	8	19	8	20	8	42	7	17	7	25
2008	8	40	8	21	8	19	8	42	8	15	8	27
2009	8	55	8	33	7	22	8	40	6	13	7	27
2010	8	51	7	27	8	24	8	37	7	14	8	23
2011	8	43	7	23	7	20	8	35	8	18	8	17
2012	8	60	8	34	8	26	8	48	8	21	8	27
2013	8	53	7	28	8	25	8	48	8	17	8	31
Average		**44**		**24**		**20**		**45**		**19**		**26**

Table A.53. Northeast Fisheries Science Center (NEFSC) spring and fall bottom trawl survey indices for Gulf of Maine haddock from 1963 to 2013. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents using both the SHG and TOGA tow selection criteria. *Note that the spring survey did not begin until 1968.*

Year	Calibrated SHG tow criteria				Calibrated, Bigelow TOGA tow criteria			
	Spring		Fall		Spring		Fall	
	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow
1963			69.549	50.697				
1964			14.176	18.386				
1965			17.434	17.739				
1966			10.742	13.103				
1967			12.186	16.871				
1968	6.066	8.107	8.564	17.307				
1969	3.719	6.607	5.451	12.721				
1970	0.906	1.784	2.918	7.354				
1971	0.878	2.523	2.879	8.159				
1972	0.862	0.882	1.984	3.050				
1973	1.312	1.623	4.165	8.591				
1974	1.437	1.061	2.687	3.347				
1975	2.770	3.482	5.533	8.618				
1976	8.326	6.350	6.035	8.041				
1977	6.799	6.725	8.296	8.755				
1978	1.356	1.434	9.775	21.659				
1979	2.870	3.878	6.174	15.568				
1980	2.212	2.672	7.152	9.836				
1981	3.612	3.545	4.456	10.874				
1982	2.047	2.555	2.627	4.164				
1983	3.678	3.571	2.598	5.219				
1984	1.095	1.144	1.696	3.893				
1985	1.773	1.882	4.079	6.149				
1986	0.707	1.284	0.623	1.392				
1987	0.092	0.062	1.035	2.646				
1988	0.187	0.301	0.335	1.476				
1989	0.083	0.124	0.283	0.631				
1990	0.024	0.001	0.145	0.432				
1991	0.074	0.066	0.142	0.120				
1992	0.193	0.272	0.211	0.092				
1993	0.450	0.204	0.866	0.474				
1994	0.402	0.255	0.325	0.218				
1995	0.806	0.351	0.977	1.099				
1996	0.305	0.338	2.407	3.543				
1997	1.935	1.223	2.688	2.424				
1998	0.197	0.113	3.130	2.920				
1999	4.267	1.109	6.730	4.910				
2000	3.610	1.815	16.589	14.033				
2001	2.364	3.205	9.960	11.981				
2002	5.704	2.793	3.920	4.835				
2003	3.191	3.908	4.733	5.359				
2004	1.061	1.199	5.704	7.171				
2005	0.862	0.971	4.132	3.932				
2006	3.151	2.661	3.910	3.945				
2007	0.770	0.675	5.153	4.393				
2008	1.689	1.394	2.266	3.146				
2009	1.521	2.705	1.867	1.154	1.531	2.573	2.017	1.203
2010	1.126	2.349	3.320	2.552	1.630	3.713	2.662	1.339
2011	1.236	1.324	4.885	4.143	1.233	1.259	4.898	4.145
2012	2.720	2.720	5.397	2.880	2.977	2.926	5.397	2.880
2013	12.359	6.218	36.088	12.571	12.380	6.221	36.088	12.571

Table A.54. Coefficients of variation (CV) for the Northeast Fisheries Science Center (NEFSC) spring and fall bottom trawl survey indices for Gulf of Maine haddock from 1963 to 2013. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents using both the SHG and TOGA tow selection criteria. *Note that the spring survey did not begin until 1968.*

Year	Calibrated SHG tow criteria				Calibrated, Bigelow TOGA tow criteria			
	Spring		Fall		Spring		Fall	
	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow
1963			0.27	0.15				
1964			0.33	0.17				
1965			0.32	0.20				
1966			0.32	0.26				
1967			0.22	0.24				
1968	0.28	0.24	0.15	0.15				
1969	0.20	0.21	0.23	0.22				
1970	0.23	0.24	0.21	0.21				
1971	0.44	0.41	0.31	0.31				
1972	0.34	0.57	0.24	0.34				
1973	0.24	0.34	0.20	0.30				
1974	0.38	0.42	0.55	0.31				
1975	0.27	0.45	0.26	0.31				
1976	0.35	0.34	0.23	0.28				
1977	0.31	0.37	0.32	0.28				
1978	0.40	0.29	0.18	0.19				
1979	0.22	0.22	0.20	0.22				
1980	0.38	0.41	0.33	0.24				
1981	0.25	0.22	0.18	0.22				
1982	0.31	0.36	0.35	0.28				
1983	0.40	0.42	0.28	0.27				
1984	0.41	0.41	0.26	0.28				
1985	0.38	0.29	0.40	0.28				
1986	0.46	0.50	0.41	0.36				
1987	0.36	0.50	0.32	0.27				
1988	0.52	0.58	0.65	0.71				
1989	0.75	0.82	0.38	0.47				
1990	0.54	0.88	0.37	0.36				
1991	0.53	0.61	0.60	0.71				
1992	0.59	0.95	0.53	0.59				
1993	0.45	0.69	0.72	0.81				
1994	0.34	0.37	0.42	0.86				
1995	0.46	0.45	0.54	0.43				
1996	0.31	0.37	0.37	0.43				
1997	0.40	0.50	0.36	0.28				
1998	0.41	0.44	0.51	0.42				
1999	0.39	0.36	0.30	0.25				
2000	0.41	0.41	0.45	0.41				
2001	0.56	0.65	0.25	0.26				
2002	0.51	0.32	0.35	0.35				
2003	0.25	0.27	0.20	0.23				
2004	0.35	0.39	0.25	0.28				
2005	0.39	0.46	0.19	0.15				
2006	0.45	0.43	0.26	0.21				
2007	0.37	0.33	0.30	0.23				
2008	0.49	0.33	0.31	0.30				
2009	0.30	0.28	0.33	0.26	0.36	0.36	0.35	0.27
2010	0.36	0.37	0.26	0.38	0.35	0.38	0.47	0.62
2011	0.35	0.37	0.46	0.40	0.34	0.38	0.46	0.41
2012	0.44	0.44	0.59	0.71	0.41	0.42	0.58	0.70
2013	0.45	0.32	0.22	0.39	0.44	0.34	0.23	0.38
Average	**0.39**	**0.43**	**0.34**	**0.34**				

Table A.55. Uncalibrated FSV Bigelow survey indices and coefficients of variation (CV) for the Northeast Fisheries Science Center (NEFSC) spring and fall bottom trawl survey indices for Gulf of Maine haddock from 2009 to 2013. The TOGA tow selection criteria were used in the calculations of these indices.

Year	Indices				Coefficients of variation (CV)			
	Spring		Fall		Spring		Fall	
	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow	Mean number/tow	Mean weight (kg)/tow
2009	2.046	2.362	3.611	1.541	0.30	0.28	0.38	0.30
2010	2.071	3.262	5.783	3.309	0.33	0.40	0.35	0.48
2011	2.356	1.106	6.722	5.250	0.35	0.39	0.50	0.43
2012	5.555	2.570	10.494	4.287	0.39	0.42	0.54	0.72
2013	29.942	6.997	79.720	17.985	0.44	0.33	0.24	0.37

Table A.56. Northeast Fisheries Science Center (NEFSC) spring survey abundance indices-at-age (numbers/tow) from 1968 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the SHG tow selection criteria.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1968	0.000	0.000	0.000	0.030	0.259	4.342	0.929	0.164	0.212	0.077	0.000	0.053	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.054	0.019	0.244	2.643	0.694	0.029	0.036	0.000	0.000	0.000	0.000	0.000
1970	0.000	0.000	0.000	0.000	0.000	0.000	0.131	0.633	0.123	0.019	0.000	0.000	0.000	0.000	0.000
1971	0.000	0.000	0.000	0.000	0.000	0.000	0.024	0.024	0.673	0.124	0.032	0.000	0.000	0.000	0.000
1972	0.000	0.584	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.178	0.061	0.016	0.023	0.000	0.000
1973	0.000	0.149	0.764	0.000	0.000	0.000	0.000	0.054	0.000	0.007	0.319	0.019	0.000	0.000	0.000
1974	0.000	0.900	0.088	0.312	0.000	0.022	0.000	0.000	0.031	0.016	0.000	0.070	0.000	0.000	0.000
1975	0.000	0.015	1.973	0.155	0.409	0.000	0.008	0.008	0.065	0.130	0.000	0.000	0.000	0.000	0.008
1976	0.000	5.110	0.115	1.763	0.140	0.961	0.058	0.043	0.000	0.000	0.000	0.000	0.000	0.129	0.008
1977	0.000	1.043	3.383	0.033	1.382	0.399	0.560	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1978	0.000	0.085	0.682	0.355	0.030	0.204	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1979	0.000	0.292	0.097	0.702	1.268	0.233	0.135	0.091	0.052	0.000	0.000	0.000	0.000	0.000	0.000
1980	0.000	1.053	0.153	0.178	0.547	0.219	0.025	0.000	0.000	0.037	0.000	0.000	0.000	0.000	0.000
1981	0.000	1.115	1.094	0.549	0.239	0.349	0.167	0.025	0.034	0.041	0.000	0.000	0.000	0.000	0.000
1982	0.000	0.000	0.404	0.933	0.372	0.142	0.069	0.103	0.013	0.013	0.000	0.000	0.000	0.000	0.000
1983	0.085	1.395	0.133	1.087	0.307	0.486	0.000	0.109	0.054	0.000	0.000	0.022	0.000	0.000	0.000
1984	0.000	0.019	0.570	0.054	0.299	0.108	0.000	0.000	0.045	0.000	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.013	0.320	1.078	0.055	0.155	0.083	0.050	0.020	0.000	0.000	0.000	0.000	0.000	0.000
1986	0.000	0.051	0.000	0.162	0.362	0.000	0.036	0.073	0.023	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.000	0.036	0.025	0.031	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.043	0.000	0.000	0.015	0.119	0.000	0.000	0.010	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.000	0.036	0.012	0.000	0.012	0.024	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.012	0.012	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.000	0.014	0.007	0.007	0.045	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.000	0.085	0.000	0.000	0.109	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.000	0.261	0.146	0.000	0.000	0.029	0.015	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.000	0.074	0.182	0.116	0.030	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1995	0.000	0.441	0.240	0.079	0.023	0.000	0.000	0.000	0.023	0.000	0.000	0.000	0.000	0.000	0.000
1996	0.000	0.000	0.037	0.146	0.123	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1997	0.000	0.775	0.210	0.257	0.601	0.070	0.022	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	0.000	0.080	0.046	0.000	0.062	0.009	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.000	3.724	0.087	0.160	0.029	0.224	0.044	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2000	0.000	1.032	1.170	0.973	0.137	0.124	0.043	0.130	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.000	0.073	0.126	1.093	0.492	0.174	0.193	0.072	0.043	0.098	0.000	0.000	0.000	0.000	0.000
2002	0.000	3.299	0.206	0.600	1.415	0.098	0.027	0.022	0.036	0.000	0.000	0.000	0.000	0.000	0.000
2003	0.000	0.359	0.208	0.091	0.108	2.017	0.203	0.121	0.036	0.037	0.012	0.000	0.000	0.000	0.000
2004	0.000	0.115	0.000	0.154	0.022	0.095	0.625	0.036	0.000	0.015	0.000	0.000	0.000	0.000	0.000
2005	0.000	0.010	0.172	0.000	0.099	0.081	0.219	0.253	0.000	0.029	0.000	0.000	0.000	0.000	0.000
2006	0.000	0.179	0.076	1.651	0.318	0.104	0.019	0.201	0.545	0.058	0.000	0.000	0.000	0.000	0.000
2007	0.000	0.156	0.085	0.028	0.242	0.000	0.028	0.029	0.028	0.160	0.000	0.015	0.000	0.000	0.000
2008	0.000	0.043	0.564	0.406	0.000	0.303	0.000	0.027	0.052	0.097	0.197	0.000	0.000	0.000	0.000
2009	0.000	0.030	0.070	0.363	0.191	0.037	0.581	0.000	0.051	0.023	0.042	0.123	0.011	0.000	0.000
2010	0.000	0.103	0.013	0.016	0.116	0.091	0.044	0.478	0.000	0.040	0.024	0.039	0.152	0.000	0.011
2011	0.000	0.527	0.199	0.010	0.000	0.025	0.174	0.000	0.193	0.000	0.018	0.024	0.025	0.042	0.000
2012	0.000	0.505	1.374	0.183	0.050	0.000	0.159	0.083	0.042	0.194	0.000	0.029	0.000	0.032	0.070
2013	0.000	9.438	0.807	1.627	0.135	0.036	0.027	0.116	0.080	0.016	0.070	0.000	0.000	0.000	0.007

Table A.57. Northeast Fisheries Science Center (NEFSC) spring survey abundance indices-at-age (numbers/tow) from 1968 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the TOGA tow selection criteria.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1968	0.000	0.000	0.000	0.030	0.259	4.342	0.929	0.164	0.212	0.077	0.000	0.053	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.054	0.019	0.244	2.643	0.694	0.029	0.036	0.000	0.000	0.000	0.000	0.000
1970	0.000	0.000	0.000	0.000	0.000	0.000	0.131	0.633	0.123	0.019	0.000	0.000	0.000	0.000	0.000
1971	0.000	0.000	0.000	0.000	0.000	0.000	0.024	0.024	0.673	0.124	0.032	0.000	0.000	0.000	0.000
1972	0.000	0.584	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.178	0.061	0.016	0.023	0.000	0.000
1973	0.000	0.149	0.764	0.000	0.000	0.000	0.000	0.054	0.000	0.007	0.319	0.019	0.000	0.000	0.000
1974	0.000	0.900	0.088	0.312	0.000	0.022	0.000	0.000	0.031	0.016	0.000	0.070	0.000	0.000	0.000
1975	0.000	0.015	1.973	0.155	0.409	0.000	0.008	0.008	0.065	0.130	0.000	0.000	0.000	0.000	0.008
1976	0.000	5.110	0.115	1.763	0.140	0.961	0.058	0.043	0.000	0.000	0.000	0.000	0.000	0.129	0.008
1977	0.000	1.043	3.383	0.033	1.382	0.399	0.560	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1978	0.000	0.085	0.682	0.355	0.030	0.204	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1979	0.000	0.292	0.097	0.702	1.268	0.233	0.135	0.091	0.052	0.000	0.000	0.000	0.000	0.000	0.000
1980	0.000	1.053	0.153	0.178	0.547	0.219	0.025	0.000	0.000	0.037	0.000	0.000	0.000	0.000	0.000
1981	0.000	1.115	1.094	0.549	0.239	0.349	0.167	0.025	0.034	0.041	0.000	0.000	0.000	0.000	0.000
1982	0.000	0.000	0.404	0.933	0.372	0.142	0.069	0.103	0.013	0.013	0.000	0.000	0.000	0.000	0.000
1983	0.085	1.395	0.133	1.087	0.307	0.486	0.000	0.109	0.054	0.000	0.000	0.022	0.000	0.000	0.000
1984	0.000	0.019	0.570	0.054	0.299	0.108	0.000	0.000	0.045	0.000	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.013	0.320	1.078	0.055	0.155	0.083	0.050	0.020	0.000	0.000	0.000	0.000	0.000	0.000
1986	0.000	0.051	0.000	0.162	0.362	0.000	0.036	0.073	0.023	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.000	0.036	0.025	0.031	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.043	0.000	0.000	0.015	0.119	0.000	0.000	0.010	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.000	0.036	0.012	0.000	0.012	0.024	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.012	0.012	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.000	0.014	0.007	0.007	0.045	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.000	0.085	0.000	0.000	0.109	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.000	0.261	0.146	0.000	0.000	0.029	0.015	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.000	0.074	0.182	0.116	0.030	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1995	0.000	0.441	0.240	0.079	0.023	0.000	0.000	0.000	0.023	0.000	0.000	0.000	0.000	0.000	0.000
1996	0.000	0.000	0.037	0.146	0.123	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1997	0.000	0.775	0.210	0.257	0.601	0.070	0.022	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	0.000	0.080	0.046	0.000	0.062	0.009	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.000	3.724	0.087	0.160	0.029	0.224	0.044	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2000	0.000	1.032	1.170	0.973	0.137	0.124	0.043	0.130	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.000	0.073	0.126	1.093	0.492	0.174	0.193	0.072	0.043	0.098	0.000	0.000	0.000	0.000	0.000
2002	0.000	3.299	0.206	0.600	1.415	0.098	0.027	0.022	0.036	0.000	0.000	0.000	0.000	0.000	0.000
2003	0.000	0.359	0.208	0.091	0.108	2.017	0.203	0.121	0.036	0.037	0.012	0.000	0.000	0.000	0.000
2004	0.000	0.115	0.000	0.154	0.022	0.095	0.625	0.036	0.000	0.015	0.000	0.000	0.000	0.000	0.000
2005	0.000	0.010	0.172	0.000	0.099	0.081	0.219	0.253	0.000	0.029	0.000	0.000	0.000	0.000	0.000
2006	0.000	0.179	0.076	1.651	0.318	0.104	0.019	0.201	0.545	0.058	0.000	0.000	0.000	0.000	0.000
2007	0.000	0.156	0.085	0.028	0.242	0.000	0.028	0.029	0.028	0.160	0.000	0.015	0.000	0.000	0.000
2008	0.000	0.043	0.564	0.406	0.000	0.303	0.000	0.027	0.052	0.097	0.197	0.000	0.000	0.000	0.000
2009	0.000	0.032	0.089	0.421	0.219	0.040	0.548	0.000	0.014	0.017	0.037	0.106	0.009	0.000	0.000
2010	0.000	0.103	0.013	0.008	0.131	0.129	0.051	0.787	0.000	0.052	0.045	0.059	0.237	0.000	0.016
2011	0.000	0.555	0.199	0.010	0.000	0.025	0.174	0.000	0.162	0.000	0.018	0.024	0.025	0.042	0.000
2012	0.000	0.532	1.590	0.193	0.049	0.000	0.162	0.084	0.043	0.194	0.000	0.029	0.000	0.032	0.070
2013	0.000	9.459	0.807	1.627	0.135	0.036	0.027	0.116	0.080	0.016	0.070	0.000	0.000	0.000	0.007

Table A.58. Uncalibrated Northeast Fisheries Science Center (NEFSC) FSV Bigelow spring survey abundance indices-at-age (numbers/tow) from 2009 to 2013 for Gulf of Maine haddock. The TOGA tow selection criteria were used in the calculations of these indices.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
2009	0.000	0.077	0.140	0.554	0.245	0.045	0.692	0.000	0.060	0.027	0.049	0.145	0.012	0.000	0.000
2010	0.000	0.258	0.024	0.012	0.165	0.156	0.061	0.920	0.000	0.061	0.052	0.069	0.276	0.000	0.019
2011	0.000	1.385	0.407	0.015	0.000	0.029	0.205	0.000	0.189	0.000	0.021	0.028	0.029	0.049	0.000
2012	0.000	1.313	3.142	0.298	0.071	0.000	0.196	0.100	0.051	0.230	0.000	0.034	0.000	0.038	0.082
2013	0.000	22.971	1.960	4.257	0.269	0.043	0.036	0.172	0.101	0.018	0.104	0.000	0.000	0.000	0.011

Table A.59. Northeast Fisheries Science Center (NEFSC) spring survey biomass indices-at-age (weight/tow) from 1968 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the SHG tow selection criteria. *Note that biomass indices are not used in the current assessment.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1968	0.000	0.000	0.000	0.018	0.141	5.064	1.740	0.348	0.377	0.158	0.000	0.260	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.012	0.013	0.367	4.461	1.632	0.062	0.061	0.000	0.000	0.000	0.000	0.000
1970	0.000	0.000	0.000	0.000	0.000	0.000	0.333	1.151	0.261	0.039	0.000	0.000	0.000	0.000	0.000
1971	0.000	0.000	0.000	0.000	0.000	0.000	0.055	0.055	1.805	0.500	0.108	0.000	0.000	0.000	0.000
1972	0.000	0.028	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.541	0.173	0.071	0.069	0.000	0.000
1973	0.000	0.012	0.279	0.000	0.000	0.000	0.000	0.096	0.000	0.020	1.133	0.083	0.000	0.000	0.000
1974	0.000	0.053	0.037	0.275	0.000	0.030	0.000	0.000	0.122	0.039	0.000	0.505	0.000	0.000	0.000
1975	0.000	0.001	0.980	0.175	0.820	0.000	0.022	0.037	0.390	1.030	0.000	0.000	0.000	0.000	0.027
1976	0.000	0.776	0.045	2.092	0.195	2.421	0.050	0.153	0.000	0.000	0.000	0.000	0.000	0.577	0.043
1977	0.000	0.063	1.671	0.048	2.537	0.927	1.479	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1978	0.000	0.007	0.275	0.501	0.051	0.600	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1979	0.000	0.017	0.047	0.553	1.930	0.468	0.355	0.292	0.217	0.000	0.000	0.000	0.000	0.000	0.000
1980	0.000	0.144	0.096	0.241	1.117	0.591	0.122	0.000	0.000	0.362	0.000	0.000	0.000	0.000	0.000
1981	0.000	0.114	0.519	0.560	0.378	1.076	0.605	0.081	0.094	0.118	0.000	0.000	0.000	0.000	0.000
1982	0.000	0.000	0.189	0.977	0.517	0.262	0.161	0.317	0.061	0.072	0.000	0.000	0.000	0.000	0.000
1983	0.003	0.128	0.040	1.181	0.456	0.948	0.000	0.460	0.274	0.000	0.000	0.082	0.000	0.000	0.000
1984	0.000	0.002	0.180	0.047	0.520	0.239	0.000	0.000	0.156	0.000	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.001	0.125	0.899	0.079	0.311	0.190	0.197	0.080	0.000	0.000	0.000	0.000	0.000	0.000
1986	0.000	0.005	0.000	0.225	0.583	0.000	0.094	0.275	0.103	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.000	0.004	0.019	0.039	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.003	0.000	0.000	0.035	0.231	0.000	0.000	0.031	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.000	0.014	0.009	0.000	0.038	0.063	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.000	0.001	0.007	0.007	0.050	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.000	0.006	0.000	0.000	0.267	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.000	0.026	0.059	0.000	0.000	0.077	0.043	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.000	0.005	0.072	0.140	0.039	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1995	0.000	0.017	0.083	0.070	0.027	0.000	0.000	0.000	0.154	0.000	0.000	0.000	0.000	0.000	0.000
1996	0.000	0.000	0.005	0.099	0.234	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1997	0.000	0.057	0.097	0.186	0.710	0.126	0.047	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	0.000	0.014	0.024	0.000	0.055	0.020	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.000	0.293	0.053	0.200	0.078	0.412	0.074	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2000	0.000	0.087	0.377	0.527	0.220	0.219	0.092	0.293	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.000	0.004	0.049	0.995	0.832	0.299	0.434	0.244	0.117	0.231	0.000	0.000	0.000	0.000	0.000
2002	0.000	0.198	0.089	0.508	1.570	0.195	0.083	0.050	0.100	0.000	0.000	0.000	0.000	0.000	0.000
2003	0.000	0.015	0.075	0.081	0.078	2.813	0.357	0.268	0.118	0.074	0.029	0.000	0.000	0.000	0.000
2004	0.000	0.008	0.000	0.077	0.024	0.101	0.881	0.076	0.000	0.032	0.000	0.000	0.000	0.000	0.000
2005	0.000	0.000	0.046	0.000	0.108	0.137	0.271	0.355	0.000	0.055	0.000	0.000	0.000	0.000	0.000
2006	0.000	0.012	0.019	0.974	0.223	0.151	0.025	0.287	0.890	0.080	0.000	0.000	0.000	0.000	0.000
2007	0.000	0.012	0.027	0.015	0.230	0.000	0.036	0.047	0.043	0.231	0.000	0.034	0.000	0.000	0.000
2008	0.000	0.005	0.150	0.234	0.000	0.406	0.000	0.044	0.082	0.152	0.322	0.000	0.000	0.000	0.000
2009	0.000	0.003	0.029	0.337	0.295	0.059	1.246	0.000	0.138	0.061	0.147	0.360	0.029	0.000	0.000
2010	0.000	0.012	0.007	0.016	0.176	0.171	0.091	1.130	0.000	0.167	0.058	0.102	0.392	0.000	0.027
2011	0.000	0.065	0.080	0.011	0.000	0.051	0.361	0.000	0.441	0.000	0.054	0.072	0.071	0.120	0.000
2012	0.000	0.076	0.737	0.229	0.066	0.000	0.382	0.246	0.108	0.479	0.000	0.101	0.000	0.069	0.228
2013	0.000	1.883	0.503	2.328	0.278	0.121	0.080	0.373	0.308	0.060	0.261	0.000	0.000	0.000	0.025

Table A.60. Northeast Fisheries Science Center (NEFSC) spring survey biomass indices-at-age (weight/tow) from 1968 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the TOGA tow selection criteria. *Note that biomass indices are not used in the current assessment.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1968	0.000	0.000	0.000	0.018	0.141	5.064	1.740	0.348	0.377	0.158	0.000	0.260	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.012	0.013	0.367	4.461	1.632	0.062	0.061	0.000	0.000	0.000	0.000	0.000
1970	0.000	0.000	0.000	0.000	0.000	0.000	0.333	1.151	0.261	0.039	0.000	0.000	0.000	0.000	0.000
1971	0.000	0.000	0.000	0.000	0.000	0.000	0.055	0.055	1.805	0.500	0.108	0.000	0.000	0.000	0.000
1972	0.000	0.028	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.541	0.173	0.071	0.069	0.000	0.000
1973	0.000	0.012	0.279	0.000	0.000	0.000	0.000	0.096	0.000	0.020	1.133	0.083	0.000	0.000	0.000
1974	0.000	0.053	0.037	0.275	0.000	0.030	0.000	0.000	0.122	0.039	0.000	0.505	0.000	0.000	0.000
1975	0.000	0.001	0.980	0.175	0.820	0.000	0.022	0.037	0.390	1.030	0.000	0.000	0.000	0.000	0.027
1976	0.000	0.776	0.045	2.092	0.195	2.421	0.050	0.153	0.000	0.000	0.000	0.000	0.000	0.577	0.043
1977	0.000	0.063	1.671	0.048	2.537	0.927	1.479	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1978	0.000	0.007	0.275	0.501	0.051	0.600	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1979	0.000	0.017	0.047	0.553	1.930	0.468	0.355	0.292	0.217	0.000	0.000	0.000	0.000	0.000	0.000
1980	0.000	0.144	0.096	0.241	1.117	0.591	0.122	0.000	0.000	0.362	0.000	0.000	0.000	0.000	0.000
1981	0.000	0.114	0.519	0.560	0.378	1.076	0.605	0.081	0.094	0.118	0.000	0.000	0.000	0.000	0.000
1982	0.000	0.000	0.189	0.977	0.517	0.262	0.161	0.317	0.061	0.072	0.000	0.000	0.000	0.000	0.000
1983	0.003	0.128	0.040	1.181	0.456	0.948	0.000	0.460	0.274	0.000	0.000	0.082	0.000	0.000	0.000
1984	0.000	0.002	0.180	0.047	0.520	0.239	0.000	0.000	0.156	0.000	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.001	0.125	0.899	0.079	0.311	0.190	0.197	0.080	0.000	0.000	0.000	0.000	0.000	0.000
1986	0.000	0.005	0.000	0.225	0.583	0.000	0.094	0.275	0.103	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.000	0.004	0.019	0.039	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.003	0.000	0.000	0.035	0.231	0.000	0.000	0.031	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.000	0.014	0.009	0.000	0.038	0.063	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.000	0.001	0.007	0.007	0.050	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.000	0.006	0.000	0.000	0.267	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.000	0.026	0.059	0.000	0.000	0.077	0.043	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.000	0.005	0.072	0.140	0.039	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1995	0.000	0.017	0.083	0.070	0.027	0.000	0.000	0.000	0.154	0.000	0.000	0.000	0.000	0.000	0.000
1996	0.000	0.000	0.005	0.099	0.234	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1997	0.000	0.057	0.097	0.186	0.710	0.126	0.047	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	0.000	0.014	0.024	0.000	0.055	0.020	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.000	0.293	0.053	0.200	0.078	0.412	0.074	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2000	0.000	0.087	0.377	0.527	0.220	0.219	0.092	0.293	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.000	0.004	0.049	0.995	0.832	0.299	0.434	0.244	0.117	0.231	0.000	0.000	0.000	0.000	0.000
2002	0.000	0.198	0.089	0.508	1.570	0.195	0.083	0.050	0.100	0.000	0.000	0.000	0.000	0.000	0.000
2003	0.000	0.015	0.075	0.081	0.078	2.813	0.357	0.268	0.118	0.074	0.029	0.000	0.000	0.000	0.000
2004	0.000	0.008	0.000	0.077	0.024	0.101	0.881	0.076	0.000	0.032	0.000	0.000	0.000	0.000	0.000
2005	0.000	0.000	0.046	0.000	0.108	0.137	0.271	0.355	0.000	0.055	0.000	0.000	0.000	0.000	0.000
2006	0.000	0.012	0.019	0.974	0.223	0.151	0.025	0.287	0.890	0.080	0.000	0.000	0.000	0.000	0.000
2007	0.000	0.012	0.027	0.015	0.230	0.000	0.036	0.047	0.043	0.231	0.000	0.034	0.000	0.000	0.000
2008	0.000	0.005	0.150	0.234	0.000	0.406	0.000	0.044	0.082	0.152	0.322	0.000	0.000	0.000	0.000
2009	0.000	0.003	0.039	0.395	0.344	0.066	1.177	0.000	0.032	0.045	0.135	0.312	0.026	0.000	0.000
2010	0.000	0.012	0.007	0.008	0.195	0.267	0.106	1.997	0.000	0.199	0.108	0.160	0.617	0.000	0.039
2011	0.000	0.067	0.080	0.011	0.000	0.051	0.364	0.000	0.365	0.000	0.054	0.073	0.072	0.122	0.000
2012	0.000	0.084	0.875	0.240	0.066	0.000	0.398	0.255	0.114	0.488	0.000	0.103	0.000	0.071	0.232
2013	0.000	1.885	0.503	2.329	0.278	0.121	0.080	0.373	0.308	0.060	0.261	0.000	0.000	0.000	0.025

Table A.61. Uncalibrated Northeast Fisheries Science Center (NEFSC) FSV Bigelow spring survey biomass indices-at-age (weight/tow) from 2009 to 2013 for Gulf of Maine haddock. Stations were selected using the TOGA tow selection criteria. *Note that biomass indices are not used in the current assessment.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
2009	0.000	0.005	0.041	0.360	0.262	0.051	1.038	0.000	0.113	0.050	0.120	0.298	0.024	0.000	0.000
2010	0.000	0.022	0.009	0.009	0.181	0.237	0.094	1.737	0.000	0.173	0.094	0.139	0.535	0.000	0.034
2011	0.000	0.113	0.108	0.011	0.000	0.041	0.291	0.000	0.290	0.000	0.043	0.058	0.057	0.096	0.000
2012	0.000	0.122	1.008	0.207	0.057	0.000	0.286	0.181	0.080	0.346	0.000	0.072	0.000	0.051	0.162
2013	0.000	2.297	0.637	2.955	0.269	0.073	0.055	0.272	0.202	0.034	0.185	0.000	0.000	0.000	0.020

Table A.62. Northeast Fisheries Science Center (NEFSC) fall survey abundance indices-at-age (numbers/tow) from 1963 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the SHG tow selection criteria.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1963	35.425	12.307	1.770	2.954	7.037	4.850	1.721	1.287	1.067	0.686	0.362	0.000	0.000	0.085	0.000
1964	0.081	6.194	1.562	0.525	0.989	2.220	1.612	0.851	0.000	0.143	0.000	0.000	0.000	0.000	0.000
1965	0.108	0.666	7.688	5.009	0.305	1.347	1.368	0.646	0.246	0.051	0.000	0.000	0.000	0.000	0.000
1966	0.019	0.162	0.424	6.438	2.390	0.252	0.653	0.327	0.065	0.012	0.000	0.000	0.000	0.000	0.000
1967	0.000	0.000	0.000	1.562	7.924	1.865	0.515	0.068	0.203	0.050	0.000	0.000	0.000	0.000	0.000
1968	0.000	0.000	0.000	0.000	0.235	6.295	1.383	0.237	0.315	0.099	0.000	0.000	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.040	0.026	0.032	4.061	1.016	0.182	0.011	0.029	0.000	0.000	0.000	0.054
1970	0.000	0.048	0.000	0.000	0.000	0.133	0.151	1.918	0.656	0.000	0.011	0.000	0.000	0.000	0.000
1971	0.268	0.000	0.000	0.000	0.011	0.000	0.170	0.347	1.901	0.139	0.044	0.000	0.000	0.000	0.000
1972	0.000	1.190	0.000	0.000	0.000	0.048	0.000	0.000	0.000	0.461	0.201	0.084	0.000	0.000	0.000
1973	1.129	0.022	0.960	0.000	0.371	0.018	0.000	0.059	0.007	0.139	1.177	0.170	0.000	0.022	0.093
1974	0.022	1.660	0.103	0.502	0.065	0.000	0.000	0.000	0.000	0.000	0.000	0.306	0.000	0.000	0.030
1975	0.888	0.227	1.850	0.499	1.494	0.000	0.085	0.035	0.000	0.000	0.041	0.064	0.288	0.062	0.000
1976	1.633	1.794	0.097	1.249	0.159	0.921	0.000	0.174	0.000	0.000	0.000	0.000	0.008	0.000	0.000
1977	0.104	3.085	3.425	0.127	1.045	0.120	0.287	0.000	0.000	0.000	0.000	0.020	0.000	0.082	0.000
1978	0.180	0.069	1.550	5.668	0.148	0.762	1.158	0.125	0.000	0.000	0.000	0.000	0.045	0.000	0.071
1979	0.796	0.406	0.088	1.109	2.735	0.477	0.428	0.122	0.000	0.000	0.000	0.013	0.000	0.000	0.000
1980	3.953	0.509	0.292	0.000	0.298	1.062	0.636	0.181	0.129	0.093	0.000	0.000	0.000	0.000	0.000
1981	0.000	0.598	0.470	1.019	0.337	0.800	0.802	0.122	0.309	0.000	0.000	0.000	0.000	0.000	0.000
1982	0.386	0.037	0.586	0.906	0.375	0.049	0.000	0.096	0.096	0.000	0.096	0.000	0.000	0.000	0.000
1983	0.026	0.533	0.051	0.675	0.503	0.401	0.177	0.068	0.164	0.000	0.000	0.000	0.000	0.000	0.000
1984	0.000	0.210	0.534	0.000	0.292	0.000	0.413	0.000	0.034	0.214	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.089	0.396	2.794	0.017	0.192	0.134	0.389	0.000	0.000	0.068	0.000	0.000	0.000	0.000
1986	0.000	0.015	0.000	0.076	0.354	0.101	0.018	0.000	0.059	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.029	0.000	0.152	0.102	0.094	0.061	0.301	0.178	0.000	0.118	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.000	0.000	0.041	0.023	0.092	0.000	0.064	0.115	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.059	0.059	0.019	0.012	0.082	0.033	0.019	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.009	0.024	0.000	0.056	0.000	0.000	0.000	0.038	0.019	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.053	0.047	0.000	0.000	0.042	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.043	0.145	0.000	0.023	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.099	0.467	0.226	0.030	0.030	0.015	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.206	0.047	0.000	0.000	0.000	0.000	0.000	0.036	0.000	0.000	0.000	0.036	0.000	0.000	0.000
1995	0.000	0.094	0.604	0.184	0.036	0.036	0.000	0.000	0.000	0.000	0.000	0.000	0.023	0.000	0.000
1996	0.043	0.127	0.195	1.062	0.618	0.068	0.114	0.071	0.036	0.000	0.036	0.000	0.000	0.036	0.000
1997	0.214	1.328	0.030	0.385	0.578	0.061	0.090	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	1.466	0.241	0.416	0.130	0.431	0.303	0.070	0.049	0.025	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.548	3.229	0.594	0.829	0.253	0.478	0.513	0.169	0.059	0.026	0.032	0.000	0.000	0.000	0.000
2000	0.333	0.661	11.306	1.686	1.303	0.425	0.580	0.221	0.074	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.196	0.240	2.297	4.862	0.719	0.808	0.301	0.193	0.293	0.051	0.000	0.000	0.000	0.000	0.000
2002	0.014	0.121	0.014	0.541	2.454	0.342	0.144	0.000	0.215	0.031	0.000	0.045	0.000	0.000	0.000
2003	0.853	0.000	0.267	0.072	0.504	2.466	0.351	0.053	0.000	0.144	0.023	0.000	0.000	0.000	0.000
2004	0.073	0.348	0.029	0.546	0.250	0.828	3.234	0.124	0.156	0.000	0.027	0.090	0.000	0.000	0.000
2005	0.188	0.110	1.593	0.067	0.147	0.300	0.407	1.143	0.088	0.058	0.000	0.019	0.013	0.000	0.000
2006	0.230	0.264	0.083	1.781	0.027	0.205	0.108	0.290	0.848	0.048	0.008	0.000	0.000	0.017	0.000
2007	0.015	1.065	0.848	0.221	2.128	0.061	0.014	0.163	0.114	0.500	0.000	0.025	0.000	0.000	0.000
2008	0.000	0.000	0.404	0.111	0.000	1.045	0.000	0.161	0.114	0.110	0.281	0.000	0.041	0.000	0.000
2009	0.815	0.225	0.080	0.171	0.012	0.038	0.343	0.000	0.043	0.000	0.031	0.108	0.000	0.000	0.000
2010	1.158	0.094	0.072	0.140	0.277	0.593	0.217	0.548	0.000	0.016	0.000	0.030	0.177	0.000	0.000
2011	0.495	1.191	0.835	0.196	0.097	0.850	0.221	0.136	0.747	0.000	0.054	0.000	0.000	0.064	0.000
2012	1.467	0.526	3.291	0.010	0.000	0.020	0.028	0.028	0.000	0.028	0.000	0.000	0.000	0.000	0.000
2013	21.582	5.057	1.815	6.762	0.415	0.260	0.090	0.028	0.000	0.000	0.080	0.000	0.000	0.000	0.000

Table A.63. Northeast Fisheries Science Center (NEFSC) fall survey abundance indices-at-age (numbers/tow) from 1963 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the TOGA tow selection criteria.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1963	35.425	12.307	1.770	2.954	7.037	4.850	1.721	1.287	1.067	0.686	0.362	0.000	0.000	0.085	0.000
1964	0.081	6.194	1.562	0.525	0.989	2.220	1.612	0.851	0.000	0.143	0.000	0.000	0.000	0.000	0.000
1965	0.108	0.666	7.688	5.009	0.305	1.347	1.368	0.646	0.246	0.051	0.000	0.000	0.000	0.000	0.000
1966	0.019	0.162	0.424	6.438	2.390	0.252	0.653	0.327	0.065	0.012	0.000	0.000	0.000	0.000	0.000
1967	0.000	0.000	0.000	1.562	7.924	1.865	0.515	0.068	0.203	0.050	0.000	0.000	0.000	0.000	0.000
1968	0.000	0.000	0.000	0.000	0.235	6.295	1.383	0.237	0.315	0.099	0.000	0.000	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.040	0.026	0.032	4.061	1.016	0.182	0.011	0.029	0.000	0.000	0.000	0.054
1970	0.000	0.048	0.000	0.000	0.000	0.133	0.151	1.918	0.656	0.000	0.011	0.000	0.000	0.000	0.000
1971	0.268	0.000	0.000	0.000	0.011	0.000	0.170	0.347	1.901	0.139	0.044	0.000	0.000	0.000	0.000
1972	0.000	1.190	0.000	0.000	0.000	0.048	0.000	0.000	0.000	0.461	0.201	0.084	0.000	0.000	0.000
1973	1.129	0.022	0.960	0.000	0.371	0.018	0.000	0.059	0.007	0.139	1.177	0.170	0.000	0.022	0.093
1974	0.022	1.660	0.103	0.502	0.065	0.000	0.000	0.000	0.000	0.000	0.000	0.306	0.000	0.000	0.030
1975	0.888	0.227	1.850	0.499	1.494	0.000	0.085	0.035	0.000	0.000	0.041	0.064	0.288	0.062	0.000
1976	1.633	1.794	0.097	1.249	0.159	0.921	0.000	0.174	0.000	0.000	0.000	0.000	0.008	0.000	0.000
1977	0.104	3.085	3.425	0.127	1.045	0.120	0.287	0.000	0.000	0.000	0.000	0.020	0.000	0.082	0.000
1978	0.180	0.069	1.550	5.668	0.148	0.762	1.158	0.125	0.000	0.000	0.000	0.000	0.045	0.000	0.071
1979	0.796	0.406	0.088	1.109	2.735	0.477	0.428	0.122	0.000	0.000	0.000	0.013	0.000	0.000	0.000
1980	3.953	0.509	0.292	0.000	0.298	1.062	0.636	0.181	0.129	0.093	0.000	0.000	0.000	0.000	0.000
1981	0.000	0.598	0.470	1.019	0.337	0.800	0.802	0.122	0.309	0.000	0.000	0.000	0.000	0.000	0.000
1982	0.386	0.037	0.586	0.906	0.375	0.049	0.000	0.096	0.096	0.000	0.096	0.000	0.000	0.000	0.000
1983	0.026	0.533	0.051	0.675	0.503	0.401	0.177	0.068	0.164	0.000	0.000	0.000	0.000	0.000	0.000
1984	0.000	0.210	0.534	0.000	0.292	0.000	0.413	0.000	0.034	0.214	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.089	0.396	2.794	0.017	0.192	0.134	0.389	0.000	0.000	0.068	0.000	0.000	0.000	0.000
1986	0.000	0.015	0.000	0.076	0.354	0.101	0.018	0.000	0.059	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.029	0.000	0.152	0.102	0.094	0.061	0.301	0.178	0.000	0.118	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.000	0.000	0.041	0.023	0.092	0.000	0.064	0.115	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.059	0.059	0.019	0.012	0.082	0.033	0.019	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.009	0.024	0.000	0.056	0.000	0.000	0.000	0.038	0.019	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.053	0.047	0.000	0.000	0.042	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.043	0.145	0.000	0.023	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.099	0.467	0.226	0.030	0.030	0.015	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.206	0.047	0.000	0.000	0.000	0.000	0.000	0.036	0.000	0.000	0.000	0.036	0.000	0.000	0.000
1995	0.000	0.094	0.604	0.184	0.036	0.036	0.000	0.000	0.000	0.000	0.000	0.000	0.023	0.000	0.000
1996	0.043	0.127	0.195	1.062	0.618	0.068	0.114	0.071	0.036	0.000	0.036	0.000	0.000	0.036	0.000
1997	0.214	1.328	0.030	0.385	0.578	0.061	0.090	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	1.466	0.241	0.416	0.130	0.431	0.303	0.070	0.049	0.025	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.548	3.229	0.594	0.829	0.253	0.478	0.513	0.169	0.059	0.026	0.032	0.000	0.000	0.000	0.000
2000	0.333	0.661	11.306	1.686	1.303	0.425	0.580	0.221	0.074	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.196	0.240	2.297	4.862	0.719	0.808	0.301	0.193	0.293	0.051	0.000	0.000	0.000	0.000	0.000
2002	0.014	0.121	0.014	0.541	2.454	0.342	0.144	0.000	0.215	0.031	0.000	0.045	0.000	0.000	0.000
2003	0.853	0.000	0.267	0.072	0.504	2.466	0.351	0.053	0.000	0.144	0.023	0.000	0.000	0.000	0.000
2004	0.073	0.348	0.029	0.546	0.250	0.828	3.234	0.124	0.156	0.000	0.027	0.090	0.000	0.000	0.000
2005	0.188	0.110	1.593	0.067	0.147	0.300	0.407	1.143	0.088	0.058	0.000	0.019	0.013	0.000	0.000
2006	0.230	0.264	0.083	1.781	0.027	0.205	0.108	0.290	0.848	0.048	0.008	0.000	0.000	0.017	0.000
2007	0.015	1.065	0.848	0.221	2.128	0.061	0.014	0.163	0.114	0.500	0.000	0.025	0.000	0.000	0.000
2008	0.000	0.000	0.404	0.111	0.000	1.045	0.000	0.161	0.114	0.110	0.281	0.000	0.041	0.000	0.000
2009	0.888	0.258	0.092	0.188	0.012	0.040	0.348	0.000	0.052	0.000	0.031	0.108	0.000	0.000	0.000
2010	1.625	0.034	0.021	0.050	0.168	0.291	0.112	0.240	0.000	0.000	0.000	0.026	0.095	0.000	0.000
2011	0.508	1.191	0.835	0.196	0.097	0.850	0.221	0.136	0.747	0.000	0.054	0.000	0.000	0.064	0.000
2012	1.467	0.526	3.291	0.010	0.000	0.020	0.028	0.028	0.000	0.028	0.000	0.000	0.000	0.000	0.000
2013	21.582	5.057	1.815	6.762	0.415	0.260	0.090	0.028	0.000	0.000	0.080	0.000	0.000	0.000	0.000

Table A.64. Uncalibrated Northeast Fisheries Science Center (NEFSC) FSV Bigelow fall survey abundance indices-at-age (numbers/tow) from 2009 to 2013 for Gulf of Maine haddock. Stations were selected using the TOGA tow selection criteria.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
2009	2.135	0.496	0.138	0.229	0.015	0.048	0.361	0.000	0.050	0.000	0.029	0.111	0.000	0.000	0.000
2010	3.234	0.080	0.109	0.253	0.399	0.754	0.321	0.441	0.000	0.000	0.000	0.030	0.163	0.000	0.000
2011	1.149	1.996	1.095	0.216	0.098	0.903	0.241	0.163	0.745	0.000	0.051	0.000	0.000	0.066	0.000
2012	3.844	1.148	5.371	0.012	0.000	0.023	0.032	0.032	0.000	0.032	0.000	0.000	0.000	0.000	0.000
2013	56.007	10.560	2.949	9.111	0.501	0.370	0.107	0.029	0.000	0.000	0.087	0.000	0.000	0.000	0.000

Table A.65. Northeast Fisheries Science Center (NEFSC) fall survey biomass indices-at-age (weight/tow) from 1963 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the SHG tow selection criteria. *Note that biomass indices are not used in the current assessment.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1963	0.997	3.025	1.418	3.936	12.441	11.214	4.862	4.428	3.738	2.375	1.574	0.000	0.000	0.690	0.000
1964	0.004	1.322	1.134	0.719	1.905	4.854	4.281	3.376	0.000	0.792	0.000	0.000	0.000	0.000	0.000
1965	0.010	0.118	2.985	4.535	0.542	2.920	3.413	2.100	0.875	0.241	0.000	0.000	0.000	0.000	0.000
1966	0.002	0.053	0.255	5.847	3.025	0.653	1.851	1.129	0.247	0.041	0.000	0.000	0.000	0.000	0.000
1967	0.000	0.000	0.000	1.518	10.124	2.887	1.196	0.230	0.722	0.195	0.000	0.000	0.000	0.000	0.000
1968	0.000	0.000	0.000	0.000	0.371	11.415	3.207	0.733	1.176	0.405	0.000	0.000	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.071	0.040	0.060	8.750	2.988	0.473	0.034	0.080	0.000	0.000	0.000	0.225
1970	0.000	0.011	0.000	0.000	0.000	0.215	0.335	4.803	1.942	0.000	0.049	0.000	0.000	0.000	0.000
1971	0.004	0.000	0.000	0.000	0.021	0.000	0.557	1.322	5.517	0.609	0.129	0.000	0.000	0.000	0.000
1972	0.000	0.320	0.000	0.000	0.000	0.100	0.000	0.000	0.000	1.477	0.765	0.387	0.000	0.000	0.000
1973	0.027	0.004	0.667	0.000	0.883	0.071	0.000	0.220	0.018	0.491	4.841	0.740	0.000	0.118	0.511
1974	0.000	0.484	0.160	0.836	0.193	0.000	0.000	0.000	0.000	0.000	0.000	1.472	0.000	0.000	0.202
1975	0.020	0.076	1.670	1.024	3.413	0.000	0.277	0.109	0.000	0.000	0.239	0.243	1.315	0.233	0.000
1976	0.056	0.814	0.131	2.547	0.449	3.162	0.000	0.853	0.000	0.000	0.000	0.000	0.030	0.000	0.000
1977	0.008	0.902	3.273	0.237	2.613	0.375	0.961	0.000	0.000	0.000	0.000	0.064	0.000	0.323	0.000
1978	0.005	0.029	1.139	10.821	0.483	2.775	5.050	0.729	0.000	0.000	0.000	0.000	0.271	0.000	0.358
1979	0.023	0.136	0.125	2.043	8.674	1.946	1.938	0.597	0.000	0.000	0.000	0.088	0.000	0.000	0.000
1980	0.133	0.175	0.339	0.000	0.800	3.752	2.624	0.721	0.609	0.684	0.000	0.000	0.000	0.000	0.000
1981	0.000	0.169	0.358	1.653	0.882	2.399	3.167	0.667	1.580	0.000	0.000	0.000	0.000	0.000	0.000
1982	0.008	0.012	0.551	1.318	0.906	0.134	0.000	0.386	0.447	0.000	0.403	0.000	0.000	0.000	0.000
1983	0.002	0.113	0.039	1.119	1.046	1.049	0.668	0.460	0.724	0.000	0.000	0.000	0.000	0.000	0.000
1984	0.000	0.059	0.375	0.000	0.798	0.000	1.615	0.000	0.202	0.845	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.031	0.355	3.198	0.024	0.417	0.322	1.454	0.000	0.000	0.348	0.000	0.000	0.000	0.000
1986	0.000	0.008	0.000	0.097	0.666	0.309	0.044	0.000	0.267	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.001	0.000	0.170	0.164	0.190	0.146	0.803	0.597	0.000	0.575	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.000	0.000	0.119	0.060	0.294	0.000	0.310	0.693	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.015	0.067	0.035	0.032	0.278	0.113	0.090	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.000	0.007	0.000	0.120	0.000	0.000	0.000	0.162	0.143	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.002	0.018	0.000	0.000	0.100	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.001	0.053	0.000	0.038	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.002	0.141	0.192	0.040	0.059	0.040	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.003	0.009	0.000	0.000	0.000	0.000	0.000	0.106	0.000	0.000	0.000	0.101	0.000	0.000	0.000
1995	0.000	0.024	0.464	0.255	0.068	0.121	0.000	0.000	0.000	0.000	0.000	0.000	0.166	0.000	0.000
1996	0.002	0.052	0.141	1.197	1.129	0.135	0.322	0.220	0.100	0.000	0.137	0.000	0.000	0.110	0.000
1997	0.003	0.305	0.020	0.406	1.185	0.223	0.283	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	0.030	0.064	0.386	0.245	1.033	0.655	0.205	0.214	0.087	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.017	0.738	0.389	1.047	0.435	0.775	0.985	0.232	0.123	0.060	0.110	0.000	0.000	0.000	0.000
2000	0.012	0.167	6.903	1.860	2.049	0.987	1.323	0.569	0.163	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.003	0.035	1.882	5.298	1.016	1.616	0.682	0.531	0.804	0.114	0.000	0.000	0.000	0.000	0.000
2002	0.001	0.027	0.006	0.467	2.855	0.562	0.326	0.000	0.463	0.051	0.000	0.078	0.000	0.000	0.000
2003	0.021	0.000	0.177	0.086	0.635	3.473	0.546	0.072	0.000	0.284	0.065	0.000	0.000	0.000	0.000
2004	0.001	0.069	0.015	0.427	0.330	1.015	4.598	0.263	0.224	0.000	0.056	0.171	0.000	0.000	0.000
2005	0.006	0.009	0.720	0.037	0.120	0.356	0.575	1.723	0.197	0.115	0.000	0.042	0.033	0.000	0.000
2006	0.005	0.051	0.034	1.514	0.018	0.333	0.147	0.416	1.258	0.113	0.024	0.000	0.000	0.032	0.000
2007	0.000	0.216	0.459	0.178	2.193	0.078	0.030	0.232	0.154	0.807	0.000	0.046	0.000	0.000	0.000
2008	0.000	0.000	0.214	0.126	0.000	1.570	0.000	0.267	0.167	0.171	0.561	0.000	0.070	0.000	0.000
2009	0.031	0.040	0.039	0.144	0.007	0.038	0.530	0.000	0.071	0.000	0.041	0.214	0.000	0.000	0.000
2010	0.051	0.022	0.036	0.106	0.297	0.680	0.209	0.775	0.000	0.043	0.000	0.052	0.281	0.000	0.000
2011	0.012	0.296	0.523	0.217	0.123	1.055	0.266	0.143	1.279	0.000	0.104	0.000	0.000	0.126	0.000
2012	0.073	0.132	2.436	0.019	0.000	0.038	0.045	0.062	0.000	0.076	0.000	0.000	0.000	0.000	0.000
2013	1.429	1.627	1.280	7.043	0.577	0.259	0.117	0.057	0.000	0.000	0.181	0.000	0.000	0.000	0.000

Table A.66. Northeast Fisheries Science Center (NEFSC) fall survey biomass indices-at-age (weight/tow) from 1963 to 2013 for Gulf of Maine haddock. For the FSV Bigelow series (2009-present), indices have been converted to RV Albatross IV equivalents and stations selected using the TOGA tow selection criteria. *Note that biomass indices are not used in the current assessment.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
1963	0.997	3.025	1.418	3.936	12.441	11.214	4.862	4.428	3.738	2.375	1.574	0.000	0.000	0.690	0.000
1964	0.004	1.322	1.134	0.719	1.905	4.854	4.281	3.376	0.000	0.792	0.000	0.000	0.000	0.000	0.000
1965	0.010	0.118	2.985	4.535	0.542	2.920	3.413	2.100	0.875	0.241	0.000	0.000	0.000	0.000	0.000
1966	0.002	0.053	0.255	5.847	3.025	0.653	1.851	1.129	0.247	0.041	0.000	0.000	0.000	0.000	0.000
1967	0.000	0.000	0.000	1.518	10.124	2.887	1.196	0.230	0.722	0.195	0.000	0.000	0.000	0.000	0.000
1968	0.000	0.000	0.000	0.000	0.371	11.415	3.207	0.733	1.176	0.405	0.000	0.000	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.071	0.040	0.060	8.750	2.988	0.473	0.034	0.080	0.000	0.000	0.000	0.225
1970	0.000	0.011	0.000	0.000	0.000	0.215	0.335	4.803	1.942	0.000	0.049	0.000	0.000	0.000	0.000
1971	0.004	0.000	0.000	0.000	0.021	0.000	0.557	1.322	5.517	0.609	0.129	0.000	0.000	0.000	0.000
1972	0.000	0.320	0.000	0.000	0.000	0.100	0.000	0.000	0.000	1.477	0.765	0.387	0.000	0.000	0.000
1973	0.027	0.004	0.667	0.000	0.883	0.071	0.000	0.220	0.018	0.491	4.841	0.740	0.000	0.118	0.511
1974	0.000	0.484	0.160	0.836	0.193	0.000	0.000	0.000	0.000	0.000	0.000	1.472	0.000	0.000	0.202
1975	0.020	0.076	1.670	1.024	3.413	0.000	0.277	0.109	0.000	0.000	0.239	0.243	1.315	0.233	0.000
1976	0.056	0.814	0.131	2.547	0.449	3.162	0.000	0.853	0.000	0.000	0.000	0.000	0.030	0.000	0.000
1977	0.008	0.902	3.273	0.237	2.613	0.375	0.961	0.000	0.000	0.000	0.000	0.064	0.000	0.323	0.000
1978	0.005	0.029	1.139	10.821	0.483	2.775	5.050	0.729	0.000	0.000	0.000	0.000	0.271	0.000	0.358
1979	0.023	0.136	0.125	2.043	8.674	1.946	1.938	0.597	0.000	0.000	0.000	0.088	0.000	0.000	0.000
1980	0.133	0.175	0.339	0.000	0.800	3.752	2.624	0.721	0.609	0.684	0.000	0.000	0.000	0.000	0.000
1981	0.000	0.169	0.358	1.653	0.882	2.399	3.167	0.667	1.580	0.000	0.000	0.000	0.000	0.000	0.000
1982	0.008	0.012	0.551	1.318	0.906	0.134	0.000	0.386	0.447	0.000	0.403	0.000	0.000	0.000	0.000
1983	0.002	0.113	0.039	1.119	1.046	1.049	0.668	0.460	0.724	0.000	0.000	0.000	0.000	0.000	0.000
1984	0.000	0.059	0.375	0.000	0.798	0.000	1.615	0.000	0.202	0.845	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.031	0.355	3.198	0.024	0.417	0.322	1.454	0.000	0.000	0.348	0.000	0.000	0.000	0.000
1986	0.000	0.008	0.000	0.097	0.666	0.309	0.044	0.000	0.267	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.001	0.000	0.170	0.164	0.190	0.146	0.803	0.597	0.000	0.575	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.000	0.000	0.119	0.060	0.294	0.000	0.310	0.693	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.015	0.067	0.035	0.032	0.278	0.113	0.090	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.000	0.007	0.000	0.120	0.000	0.000	0.000	0.162	0.143	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.002	0.018	0.000	0.000	0.100	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.001	0.053	0.000	0.038	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.002	0.141	0.192	0.040	0.059	0.040	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.003	0.009	0.000	0.000	0.000	0.000	0.000	0.106	0.000	0.000	0.000	0.101	0.000	0.000	0.000
1995	0.000	0.024	0.464	0.255	0.068	0.121	0.000	0.000	0.000	0.000	0.000	0.000	0.166	0.000	0.000
1996	0.002	0.052	0.141	1.197	1.129	0.135	0.322	0.220	0.100	0.000	0.137	0.000	0.000	0.110	0.000
1997	0.003	0.305	0.020	0.406	1.185	0.223	0.283	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	0.030	0.064	0.386	0.245	1.033	0.655	0.205	0.214	0.087	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.017	0.738	0.389	1.047	0.435	0.775	0.985	0.232	0.123	0.060	0.110	0.000	0.000	0.000	0.000
2000	0.012	0.167	6.903	1.860	2.049	0.987	1.323	0.569	0.163	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.003	0.035	1.882	5.298	1.016	1.616	0.682	0.531	0.804	0.114	0.000	0.000	0.000	0.000	0.000
2002	0.001	0.027	0.006	0.467	2.855	0.562	0.326	0.000	0.463	0.051	0.000	0.078	0.000	0.000	0.000
2003	0.021	0.000	0.177	0.086	0.635	3.473	0.546	0.072	0.000	0.284	0.065	0.000	0.000	0.000	0.000
2004	0.001	0.069	0.015	0.427	0.330	1.015	4.598	0.263	0.224	0.000	0.056	0.171	0.000	0.000	0.000
2005	0.006	0.009	0.720	0.037	0.120	0.356	0.575	1.723	0.197	0.115	0.000	0.042	0.033	0.000	0.000
2006	0.005	0.051	0.034	1.514	0.018	0.333	0.147	0.416	1.258	0.113	0.024	0.000	0.000	0.032	0.000
2007	0.000	0.216	0.459	0.178	2.193	0.078	0.030	0.232	0.154	0.807	0.000	0.046	0.000	0.000	0.000
2008	0.000	0.000	0.214	0.126	0.000	1.570	0.000	0.267	0.167	0.171	0.561	0.000	0.070	0.000	0.000
2009	0.033	0.046	0.044	0.157	0.007	0.040	0.536	0.000	0.085	0.000	0.041	0.214	0.000	0.000	0.000
2010	0.085	0.005	0.010	0.037	0.205	0.345	0.111	0.333	0.000	0.000	0.000	0.047	0.162	0.000	0.000
2011	0.013	0.296	0.523	0.217	0.123	1.055	0.266	0.143	1.279	0.000	0.104	0.000	0.000	0.126	0.000
2012	0.073	0.132	2.436	0.019	0.000	0.038	0.045	0.062	0.000	0.076	0.000	0.000	0.000	0.000	0.000
2013	1.429	1.627	1.280	7.043	0.577	0.259	0.117	0.057	0.000	0.000	0.181	0.000	0.000	0.000	0.000

Table A.67. Uncalibrated Northeast Fisheries Science Center (NEFSC) FSV Bigelow fall survey biomass indices-at-age (weight/tow) from 2009 to 2013 for Gulf of Maine haddock. Stations were selected using the TOGA tow selection criteria. *Note that biomass indices are not used in the current assessment.*

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14+
2009	0.094	0.098	0.072	0.214	0.011	0.053	0.620	0.000	0.094	0.000	0.043	0.243	0.000	0.000	0.000
2010	0.182	0.012	0.062	0.225	0.491	0.952	0.348	0.686	0.000	0.000	0.000	0.063	0.289	0.000	0.000
2011	0.034	0.558	0.773	0.270	0.140	1.258	0.327	0.190	1.441	0.000	0.111	0.000	0.000	0.148	0.000
2012	0.177	0.262	3.591	0.020	0.000	0.041	0.048	0.067	0.000	0.082	0.000	0.000	0.000	0.000	0.000
2013	3.366	3.031	1.823	8.470	0.617	0.319	0.127	0.055	0.000	0.000	0.178	0.000	0.000	0.000	0.000

Table A.68. Ratio of NEFSC spring and fall survey proportions-at-age to fishery proportion-at-age for ages 6 to age 9^{+}. Cells shaded grey indicate where the survey proportions-at-age were greater than the proportions observed in the fishery. Unshaded cells indicate where the fishery proportions-at-age were greater relative to the survey. Missing values indicate either where no information was available from the survey (no fish age 6^{+}) or the fishery proportions at age were zero.

Year	NEFSC spring/fishery compare				NEFSC fall/fishery compare			
	Age6	Age7	Age8	Age9+	Age6	Age7	Age8	Age9+
1977	1.01			0.00	0.75			20.93
1978					0.89	1.89		3.52
1979	0.76	1.13	2.67		1.19	0.74	0.00	
1980	0.83	0.00	0.00	6.74	1.25	0.53	1.31	1.02
1981	0.90	0.81	0.84	4.40	0.93	0.84	1.67	0.00
1982	1.12	1.05	0.57	0.81	0.00	0.67	2.99	4.22
1983	0.00	2.01	0.73	0.91	2.52	0.57	1.00	0.00
1984	0.00	0.00	9.87	0.00	1.10	0.00	0.51	1.40
1985	1.77	0.67	1.36	0.00	0.74	1.35	0.00	1.04
1986	0.77	2.03	0.54	0.00	0.65	0.00	2.37	0.00
1987					2.08	0.74	0.00	1.71
1988	0.00	0.00	5.48	0.00	0.00	3.88	3.51	0.00
1989	1.70	0.00	0.00	0.00	1.08	0.98	0.00	0.00
1990					0.00	1.19	4.04	
1991								
1992								
1993	1.68	0.00	0.00	0.00				
1994					0.00	1.53	0.00	12.28
1995	0.00	0.00	4.21	0.00	0.00	0.00	0.00	6.03
1996					1.71	0.61	0.40	3.82
1997	1.63	0.00	0.00	0.00	1.63	0.00	0.00	0.00
1998					0.80	1.69	1.71	0.00
1999	1.74	0.00	0.00	0.00	1.12	0.66	1.24	1.48
2000	0.50	2.69	0.00	0.00	1.33	0.90	0.69	0.00
2001	1.27	0.51	0.60	2.33	0.96	0.66	2.01	0.59
2002	0.81	2.27	1.69	0.00	0.84	0.00	1.97	0.72
2003	1.35	1.16	0.99	0.42	1.66	0.36	0.00	1.02
2004	1.19	0.59	0.00	0.28	1.14	0.38	0.83	0.41
2005	2.52	0.75	0.00	0.64	1.36	0.99	0.76	0.58
2006	0.19	1.71	1.05	0.67	0.66	1.54	1.02	0.53
2007	1.11	1.43	0.69	1.01	0.18	2.57	0.89	0.96
2008	0.00	0.51	1.42	1.06	0.00	1.63	1.65	0.82
2009	1.22	0.00	0.28	0.76	1.05	0.00	1.43	0.84
2010	0.96	0.97	0.00	1.10	5.57	0.78	0.00	0.86
2011	2.79	0.00	0.68	0.89	1.29	2.29	1.14	0.35
2012	1.27	1.17	1.65	0.83	1.60	2.85	0.00	0.53
2013	1.43	1.38	1.42	0.59	7.68	0.53	0.00	0.82
Cells ≥ 1	**16**	**11**	**10**	**6**	**18**	**11**	**14**	**12**
Total	**29**	**28**	**28**	**28**	**34**	**33**	**32**	**32**
Fraction ≥ 1	**0.55**	**0.39**	**0.36**	**0.21**	**0.53**	**0.33**	**0.44**	**0.38**

Table A.69. Massachusetts Department of Marine Fisheries (MADMF) spring and fall survey indices and coefficients of variation (CV, italicized values in parentheses) from 1978 to 2013 for Gulf of Maine haddock.

Year	Abundance (numbers/tow)				Biomass (kg/tow)			
	Spring		Fall		Spring		Fall	
1978	0.536	*(0.36)*	0.029	*(0.89)*	0.291	*(0.37)*	0.000	*(0.85)*
1979	0.038	*(0.56)*	13.811	*(0.18)*	0.033	*(0.69)*	0.100	*(0.19)*
1980	2.305	*(0.31)*	10.747	*(0.25)*	0.284	*(0.35)*	0.318	*(0.28)*
1981	2.372	*(0.3)*	0.950	*(0.54)*	0.333	*(0.33)*	0.253	*(0.62)*
1982	0.042	*(0.75)*	25.608	*(0.43)*	0.005	*(0.73)*	0.256	*(0.43)*
1983	2.193	*(0.22)*	0.498	*(0.39)*	0.286	*(0.37)*	0.011	*(0.38)*
1984	0.267	*(0.62)*	0.000		0.221	*(0.66)*	0.000	
1985	0.177	*(0.72)*	0.000		0.136	*(0.71)*	0.000	
1986	0.000		0.172	*(0.58)*	0.000		0.000	*(0.61)*
1987	0.017	*(0.9)*	0.756	*(0.31)*	0.002	*(0.9)*	0.004	*(0.25)*
1988	0.021	*(0.87)*	0.105	*(0.72)*	0.041	*(0.86)*	0.000	*(0.71)*
1989	0.043	*(0.87)*	0.014	*(0.87)*	0.010	*(0.85)*	0.003	*(0.88)*
1990	0.000		0.100	*(0.71)*	0.000		0.001	*(0.71)*
1991	0.000		0.068	*(0.62)*	0.000		0.002	*(0.82)*
1992	0.016	*(0.79)*	0.182	*(0.53)*	0.001	*(0.83)*	0.003	*(0.49)*
1993	0.000		0.705	*(0.36)*	0.000		0.007	*(0.38)*
1994	0.025	*(0.72)*	0.244	*(0.42)*	0.002	*(0.7)*	0.022	*(0.69)*
1995	0.172	*(0.57)*	0.158	*(0.36)*	0.007	*(0.54)*	0.006	*(0.45)*
1996	0.000		1.692	*(0.33)*	0.000		0.026	*(0.4)*
1997	0.153	*(0.49)*	0.323	*(0.33)*	0.006	*(0.45)*	0.007	*(0.71)*
1998	0.054	*(0.55)*	9.052	*(0.28)*	0.001	*(0.65)*	0.108	*(0.28)*
1999	0.895	*(0.46)*	1.829	*(0.56)*	0.086	*(0.46)*	0.083	*(0.36)*
2000	0.290	*(0.36)*	0.135	*(0.57)*	0.052	*(0.37)*	0.017	*(0.54)*
2001	0.000		0.951	*(0.62)*	0.000		0.025	*(0.63)*
2002	0.516	*(0.15)*	0.034	*(0.87)*	0.483	*(0.22)*	0.003	*(0.9)*
2003	0.406	*(0.55)*	0.818	*(0.52)*	0.583	*(0.51)*	0.056	*(0.55)*
2004	0.354	*(0.16)*	0.507	*(0.43)*	0.580	*(0.17)*	0.224	*(0.58)*
2005	2.449	*(0.55)*	0.515	*(0.29)*	3.591	*(0.55)*	0.015	*(0.46)*
2006	1.263	*(0.32)*	0.046	*(0.56)*	1.501	*(0.4)*	0.026	*(0.63)*
2007	0.728	*(0.48)*	0.205	*(0.44)*	1.088	*(0.43)*	0.114	*(0.69)*
2008	0.358	*(0.49)*	0.984	*(0.68)*	0.558	*(0.52)*	1.372	*(0.65)*
2009	0.230	*(0.48)*	0.185	*(0.32)*	0.371	*(0.47)*	0.060	*(0.51)*
2010	0.131	*(0.49)*	1.172	*(0.28)*	0.196	*(0.49)*	0.223	*(0.68)*
2011	0.090	*(0.61)*	6.676	*(0.55)*	0.012	*(0.67)*	0.915	*(0.76)*
2012	0.000		0.337	*(0.56)*	0.000		0.005	*(0.63)*
2013	0.159	*(0.5)*	6.191	*(0.33)*	0.013	*(0.53)*	0.134	*(0.31)*
Avg	**0.453**	***(0.52)***	**2.383**	***(0.49)***	**0.299**	***(0.54)***	**0.122**	***(0.56)***
Min	**0.000**	***(0.15)***	**0.000**	***(0.18)***	**0.000**	***(0.17)***	**0.000**	***(0.19)***
Max	**2.449**	***(0.9)***	**25.608**	***(0.89)***	**3.591**	***(0.9)***	**1.372**	***(0.9)***

Table A.70. Massachusetts Department of Marine Fisheries (MADMF) spring survey abundance indices-at-age (numbers/tow) from 1978 to 2013 for Gulf of Maine haddock.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14
1978	0.000	0.000	0.450	0.086	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1979	0.000	0.013	0.000	0.000	0.000	0.025	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1980	0.386	1.842	0.000	0.014	0.025	0.039	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1981	0.000	2.175	0.108	0.058	0.028	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1982	0.000	0.042	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1983	0.051	1.983	0.073	0.057	0.029	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1984	0.000	0.029	0.081	0.081	0.056	0.020	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.000	0.000	0.177	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1986	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.000	0.017	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1988	0.000	0.000	0.000	0.000	0.000	0.021	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.021	0.021	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.000	0.016	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.000	0.025	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1995	0.000	0.172	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1996	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1997	0.000	0.153	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	0.030	0.025	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1999	0.000	0.895	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2000	0.000	0.178	0.070	0.042	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2002	0.000	0.160	0.005	0.072	0.194	0.055	0.012	0.018	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2003	0.000	0.000	0.000	0.009	0.005	0.348	0.030	0.011	0.000	0.000	0.004	0.000	0.000	0.000	0.000
2004	0.000	0.029	0.000	0.000	0.000	0.005	0.321	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2005	0.000	0.000	0.000	0.074	0.081	0.206	0.508	1.420	0.000	0.160	0.000	0.000	0.000	0.000	0.000
2006	0.000	0.184	0.000	0.172	0.073	0.117	0.020	0.139	0.519	0.039	0.000	0.000	0.000	0.000	0.000
2007	0.000	0.017	0.000	0.011	0.191	0.000	0.084	0.155	0.095	0.167	0.000	0.009	0.000	0.000	0.000
2008	0.000	0.025	0.000	0.009	0.000	0.167	0.000	0.023	0.021	0.019	0.070	0.025	0.000	0.000	0.000
2009	0.000	0.021	0.000	0.000	0.019	0.022	0.099	0.004	0.011	0.007	0.016	0.030	0.000	0.000	0.000
2010	0.000	0.026	0.000	0.017	0.000	0.000	0.000	0.066	0.000	0.000	0.009	0.000	0.009	0.000	0.004
2011	0.000	0.090	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2012	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2013	0.000	0.159	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Table A.71. Massachusetts Department of Marine Fisheries (MADMF) fall survey abundance indices-at-age (numbers/tow) from 1978 to 2013 for Gulf of Maine haddock.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12	Age13	Age14
1978	0.029	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1979	13.811	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1980	9.815	0.932	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1981	0.068	0.373	0.509	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1982	25.608	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1983	0.444	0.054	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1984	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1985	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1986	0.172	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1987	0.756	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1988	0.105	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1989	0.000	0.014	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1990	0.100	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1991	0.050	0.018	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1992	0.182	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1993	0.705	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1994	0.124	0.120	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1995	0.123	0.034	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1996	1.692	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1997	0.270	0.053	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1998	8.966	0.086	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1999	1.554	0.275	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2000	0.012	0.123	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2001	0.878	0.074	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2002	0.017	0.017	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2003	0.793	0.000	0.000	0.000	0.004	0.021	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2004	0.090	0.319	0.000	0.000	0.000	0.010	0.079	0.010	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2005	0.424	0.074	0.017	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2006	0.000	0.021	0.000	0.020	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	0.000	0.000
2007	0.106	0.025	0.000	0.000	0.031	0.006	0.000	0.000	0.010	0.027	0.000	0.000	0.000	0.000	0.000
2008	0.000	0.000	0.049	0.041	0.000	0.549	0.000	0.045	0.094	0.033	0.148	0.000	0.025	0.000	0.000
2009	0.084	0.076	0.000	0.000	0.000	0.000	0.025	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2010	1.049	0.000	0.000	0.000	0.000	0.057	0.010	0.032	0.000	0.000	0.000	0.012	0.012	0.000	0.000
2011	2.242	4.433	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2012	0.337	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2013	6.171	0.020	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Table A.72. Maine-New Hampshire (MENH) inshore groundfish survey Gulf of Maine haddock indices and coefficients of variation (CV) from 2000 to 2013. *Note that the spring survey did not begin until 2001.*

Year	Abundance (numbers/tow)				Biomass (kg/tow)			
	Spring	***CV***	**Fall**	***CV***	**Spring**	***CV***	**Fall**	***CV***
2000			4.12	*0.71*			0.71	*1.74*
2001	0.02	*2.00*	3.14	*1.02*	0.00	*2.00*	0.15	*1.06*
2002	4.33	*0.71*	0.29	*0.92*	1.20	*0.58*	0.02	*1.23*
2003	0.70	*0.92*	5.94	*0.94*	0.49	*1.04*	0.55	*0.73*
2004	1.67	*0.71*	2.65	*0.71*	0.26	*0.60*	0.21	*0.80*
2005	0.77	*0.66*	5.75	*0.18*	0.37	*0.71*	0.76	*1.34*
2006	1.58	*1.47*	1.18	*1.27*	0.33	*0.81*	0.43	*2.22*
2007	0.63	*0.50*	0.44	*1.08*	0.38	*0.66*	0.02	*0.53*
2008	0.43	*0.75*	0.68	*0.59*	0.40	*0.75*	0.02	*0.53*
2009	0.61	*0.60*	3.99	*0.67*	0.10	*0.70*	0.17	*0.56*
2010	0.85	*0.69*	10.86	*0.64*	0.19	*0.71*	0.46	*0.68*
2011	6.54	*1.00*	8.02	*0.78*	0.52	*0.88*	0.30	*0.71*
2012	6.56	*2.18*	12.65	*0.67*	0.65	*2.29*	0.78	*0.68*
2013	1.88	*0.41*	24.44	*0.46*	0.26	*0.60*	1.37	*0.57*

Table A.73. Summary of Gulf of Maine haddock age and maturity samples (individual fish) taken by the Maine-New Hampshire (MENH) inshore groundfish survey from 2000 to 2013. *Note that the spring survey did not begin until 2001, though no maturity samples were collected in this first year.*

Season	Year	Maturity	Ages
Fall	2000		
	2001		
	2002		
	2003		
	2004		
	2005	30	6
	2006	53	38
	2007	2	2
	2008	7	5
	2009	3	3
	2010	122	52
	2011	121	54
	2012	203	85
	2013	287	117
Spring	2001		
	2002	100	
	2003	50	
	2004	33	
	2005	40	
	2006	77	
	2007	61	
	2008	37	
	2009	9	
	2010	58	
	2011	176	
	2012	101	
	2013	130	

Table A.74. Summary of age-length information used to construct the Maine-New Hampshire (MENH) inshore groundfish survey indices-at-age for Gulf of Maine haddock. *Note that in spring 2001, only a single 3 cm fish was caught in the MENH survey, there was no corresponding age information from the NEFSC spring survey for this length bin.*

Season	Year	Source ALK		Total ages
		ME/NH	NEFSC	
Fall	2000		80	80
	2001		65	65
	2002		2	2
	2003		67	67
	2004		20	20
	2005	6	81	87
	2006	35	113	148
	2007	1	7	8
	2008	4		4
	2009	3	30	33
	2010	52	62	114
	2011	54	102	156
	2012	85	56	141
	2013	116	256	372
Spring	2001			
	2002		127	127
	2003		63	63
	2004		16	16
	2005		20	20
	2006		70	70
	2007		43	43
	2008		45	45
	2009		40	40
	2010		26	26
	2011		56	56
	2012		134	134
	2013		227	227

Table A.75. Maine-New Hampshire (MENH) spring survey abundance indices-at-age (numbers/tow) from 2001 to 2013 for Gulf of Maine haddock.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12
2001	0.016	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2002	0.000	3.127	0.115	0.371	0.653	0.027	0.000	0.000	0.025	0.013	0.000	0.000	0.000
2003	0.000	0.000	0.198	0.032	0.121	0.348	0.002	0.000	0.000	0.000	0.000	0.000	0.000
2004	0.000	1.493	0.013	0.064	0.000	0.018	0.083	0.000	0.000	0.000	0.000	0.000	0.000
2005	0.000	0.431	0.113	0.013	0.018	0.027	0.050	0.102	0.000	0.013	0.000	0.000	0.000
2006	0.000	1.356	0.011	0.066	0.008	0.007	0.014	0.036	0.076	0.003	0.000	0.000	0.000
2007	0.000	0.289	0.038	0.039	0.140	0.000	0.023	0.000	0.030	0.068	0.000	0.000	0.000
2008	0.000	0.165	0.039	0.016	0.013	0.115	0.000	0.017	0.002	0.003	0.049	0.010	0.000
2009	0.000	0.551	0.008	0.000	0.016	0.006	0.025	0.003	0.002	0.000	0.000	0.002	0.000
2010	0.000	0.787	0.000	0.000	0.004	0.004	0.019	0.024	0.000	0.002	0.002	0.000	0.014
2011	0.040	6.431	0.020	0.000	0.000	0.000	0.011	0.010	0.027	0.000	0.000	0.000	0.000
2012	0.010	5.742	0.794	0.008	0.000	0.000	0.005	0.000	0.000	0.000	0.000	0.000	0.000
2013	0.194	1.506	0.089	0.078	0.000	0.000	0.000	0.000	0.000	0.000	0.010	0.000	0.000

Table A.76. Maine-New Hampshire (MENH) fall survey abundance indices-at-age (numbers/tow) from 2000 to 2013 for Gulf of Maine haddock.

Year	Age0	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9	Age10	Age11	Age12
2000	2.499	1.211	0.256	0.088	0.065	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2001	2.903	0.057	0.128	0.051	0.003	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2002	0.140	0.145	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2003	5.640	0.065	0.016	0.003	0.020	0.172	0.020	0.003	0.000	0.005	0.000	0.000	0.000
2004	1.622	1.008	0.000	0.000	0.000	0.009	0.015	0.000	0.000	0.000	0.000	0.000	0.000
2005	5.182	0.000	0.236	0.012	0.016	0.059	0.075	0.143	0.004	0.020	0.000	0.000	0.000
2006	0.553	0.095	0.026	0.424	0.008	0.008	0.004	0.015	0.045	0.000	0.000	0.000	0.000
2007	0.356	0.079	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2008	0.679	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2009	3.959	0.029	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2010	10.846	0.010	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2011	7.708	0.315	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2012	12.178	0.453	0.022	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2013	22.848	1.526	0.054	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

Table A.77. Summary of the Gulf of Maine haddock ADAPT-VPA model formulations used to build a 'bridge' from the GARM III ADAPT-VPA model to the SAW/SARC 59 update. *The (+1) notation indicates that the survey index was lagged forward a year and an age in the model (e.g., age-1 in 1981 becomes age-2 in 1982).*

Model	Type	Version	Model description	Years	Catch	Plus group handling	Partial recruitment	Survey input style	Survey Indices		
										NEFSC	
									Tow evaluation	Spring	Fall
1	VPA	v3.1.0	AOP 2012	1977-2010	Commercial catch, recreational landings (no discards)	Backward calculation	N/A	Age-specific indices	SHG	1-6+	1-8+ (+1)
2	VPA	v.3.4.5	Software update to v3.4.4	1977-2010	Commercial catch, recreational landings (no discards)	Backward calculation	N/A	Age-specific indices	SHG	1-6+	1-8+ (+1)
3	VPA	v.3.4.5	Combined method	1977-2010	Commercial catch, recreational landings (no discards)	Combined method	N/A	Age-specific indices	SHG	1-6+	1-8+ (+1)
4	VPA	v.3.4.5	Fully updated catch info (except recrecreatioal discards), catch WAA matrix updated assuming 0% recreational discard mortality	1977-2010	Commercial catch, recreational landings (no discards)	Combined method	N/A	Age-specific indices	SHG	1-6+	1-8+ (+1)
5	VPA	v.3.4.5	Recreational discards added to catch, catch and stock WAA updated	1977-2010	Commercial catch, recreational catch	Combined method	N/A	Age-specific indices	SHG	1-6+	1-8+ (+1)
6	VPA	v.3.4.5	Add 2011-2013 catch data, update surveys indices (1977-2013)	1977-2013	Commercial catch, recreational catch	Combined method	N/A	Age-specific indices	SHG	1-6+	1-8+ (+1)

Table A.78. Summary of the Gulf of Maine haddock ADAPT-VPA model formulations used to build a 'bridge' from the GARM III ADAPT-VPA model to the SAW/SARC 59 update. Differences in model formulations are summarized in Table A.77.

Model		**1**	**2**	**3**	**4**	**5**	**6**
Model description		**AOP 2012**	**Software update**	**Combined method**	**Updated catch**	**Recreational discard**	**2013 update**
Mean squared residual		1.29	1.29	1.28	1.49	1.48	1.44
CVs on t+1 population numbers	Age2	0.67	0.67	0.66	0.72	0.71	0.86
	Age3	0.52	0.52	0.52	0.56	0.57	0.62
	Age4	0.53	0.53	0.53	0.57	0.60	0.52
	Age5	0.42	0.42	0.42	0.43	0.44	0.50
	Age6	0.47	0.47	0.50	0.47	0.48	0.50
	Age7	0.54	0.54	0.58	0.55	0.56	0.53
	Age8	0.90	0.90	0.71	0.64	0.69	0.88
	Age 9^+	1.14	1.14				
Terminal estimates	$F_{6\text{-}8,\ 2010}$	0.82	0.82	0.58	0.37	0.42	0.36
	$F_{6\text{-}8,\ 2013}$						0.82
	SSB_{2010} (mt)	2,868	2,868	3,146	3,631	3,230	3,070
	SSB_{2013} (mt)						6,135
Retro (Mohns Rho) *7 year 'peels'	F6-8	0.98	0.98	0.70	0.66	0.65	0.52
	SSB	-0.22	-0.22	-0.06	-0.03	-0.04	-0.03
	Recruits (Age1)	4.71	4.71	4.47	3.49	3.08	0.55

Table A.79. Summary of the ASAP model sensitivity runs and location of the run descriptions within this report.

Report location	Order	ASAP sensitivity runs
TOR 4	1	NEFSC fall BTS q-profile
TOR 4	2	Swept area determination (wing vs. door)
Appendix 2	1	Profiling over a range of natural mortality assumptions
Appendix 2	2	SHG/TOGA tow evaluation criteria
Appendix 2	3	Abundance vs. biomass survey indices
Appendix 2	4	Treatment of the Bigelow survey years as independent survey ind
Appendix 2	5	Model performance when tuned to NEFSC survey indices alone
Appendix 2	6	Inclusion of state surveys (MADMF, MENH)
Appendix 2	7	Inclusion of commercial and recreational LPUE indices
Appendix 2	8	Recreational discard mortality
Appendix 2	9	Explicit treatment of catch fleets (commercial, recreational)
Appendix 2	10	Assessment model starting point (1956, 1963)
Appendix 2	11	Selectivity blocks (number, location, selectivity form, etc.)
Appendix 2	12	Catch precision assumptions
Appendix 2	13	Terminal recruitment (handling of recruitment deviations)

Table A.80. Summary diagnostics and results from the base Gulf of Maine haddock ASAP model (ASAP_BASE).

Model		**ASAP_BASE**
Model description		ASAP base run (SHG/TOGA)
Maximum gradient (*conv. criteria* < *1e-4*)		1.10E-05
Number of parameters		125
Objective function		2526
Components of objective function	**Recruit devs**	110
	Suvey age comps	874
	Catch age comps	644
	Index fit	702
	Catch fit	196
RMSE	**Fleet 1**	0.34
	Index 1	2.10
	Index 2	2.00
	Index total	2.05
	Recruit devs	3.91
SSB_{1977} (mt)		9,470
SSB_{2013} (mt)		4,500
$F_{mult, 2013}$		0.31
Mohn's rho (7 year peel)	**SSB**	-0.03
	F_{mult}	0.05
	Age 1 N	0.18

Table A.81. ASAP_BASE model estimates of selectivity-at-age and corresponding coefficients of variation (CV) for the Gulf of Maine haddock fishery and NEFSC spring and fall surveys.

Block/Index		Selectivity	CV
Fleet block 1 (1977-1988)	1	0.00	0.61
	2	0.22	0.12
	3	0.39	0.11
	4	0.64	0.10
	5	0.65	0.11
	6	0.77	0.12
	7	1.00	
	8	1.00	0.00
	9^+	1.00	0.00
Fleet block 2 (1989-2004)	1	0.01	0.27
	2	0.06	0.16
	3	0.34	0.14
	4	0.55	0.14
	5	0.72	0.14
	6	0.96	0.14
	7	1.00	
	8	0.94	0.21
	9^+	0.67	0.30
Fleet block 3 (2005-2013)	1	0.01	0.42
	2	0.06	0.21
	3	0.22	0.17
	4	0.31	0.17
	5	0.56	0.15
	6	0.77	0.15
	7	1.00	
	8	1.00	0.00
	9^+	0.74	0.19
NEFSC spring	1	0.74	0.13
	2	0.62	0.14
	3	0.79	0.14
	4	1.00	0.00
	5	0.87	0.17
	6	1.00	
	7	0.82	0.25
	8	0.75	0.34
	9^+	0.76	0.30
NEFSC fall	1	0.28	0.14
	2	0.35	0.14
	3	0.60	0.13
	4	0.62	0.14
	5	0.76	0.15
	6	1.00	
	7	1.00	0.00
	8	1.00	0.00
	9^+	0.66	0.25

Table A.82. Comparison of the summary diagnostics from the ASAP base model (ASAP_BASE) which uses survey minimum area swept indices assumption a wing spread footprint of 0.012 nm^2 to a sensitivity run (ASAP_DOOR_SPREAD) assuming a door spread footprint of 0.023 nm^2.

Model		**ASAP_BASE**	**ASAP_DOOR_SPREAD**
Model description		Min area swept indices based on wing spread	Min area swept indices based on door spread
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05	8.20E-05
Number of parameters		125	125
Objective function		2526	2472
Components of objective function	**Recruit devs**	110	110
	Suvey age comps	874	873
	Catch age comps	644	644
	Index fit	702	648
	Catch fit	196	196
RMSE	**Fleet 1**	0.34	0.34
	Index 1	2.10	2.10
	Index 2	2.00	2.00
	Index total	2.05	2.05
	Recruit devs	3.91	3.91
SSB_{1977} (mt)		9,470	9,469
SSB_{2013} (mt)		4,500	4,500
$F_{mult, 2013}$		0.31	0.31
Mohn's rho (7 year peel)	**SSB**	-0.03	-0.03
	F_{mult}	0.05	0.05
	Age 1 N	0.18	0.18
Survey q	**NEFSC spring**	0.26	0.13
	NEFSC fall	0.99	0.48

Table A.83. Comparison of the summary diagnostics from the ASAP base model (ASAP_BASE) and the various intermediate models used to build the bridge to the ASAP_final_temp6 model.

Model		ASAP_BASE	ASAP_final_temp1	ASAP_final_temp2	ASAP_final_temp3	ASAP_final_temp4	ASAP_final_temp5	ASAP_final_temp6
Description			Modified RecDevs (lambda=1, CV=2.0)	Increased catch CVs (0.1-0.2)	Survey CV adjustments (spring+0.3, fall+0.2)	ESS adjustments to catch and surveys	Selectivity-at-age adjustments	Minor selectivity-at-age adjustments
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05	7.90E-05	2.90E-05	8.90E-05	5.20E-05	6.80E-05	4.50E-05
Number of parameters		125	125	125	125	125	125	123
Objective function		2526	2705	2728	2690	2389	2386	2386
Components of objective function	Recruit devs	110	332	332	328	328	327	327
	Suvey age comps	874	871	870	869	666	667	667
	Catch age comps	644	640	636	633	539	538	538
	Index fit	702	666	660	633	630	630	630
	Catch fit	196	196	231	227	225	225	225
RMSE	Fleet 1	0.34	0.33	0.65	0.48	0.36	0.36	0.36
	Index 1	2.10	1.91	1.87	1.21	1.18	1.18	1.18
	Index 2	2.00	1.68	1.63	1.17	1.13	1.12	1.12
	Index total	2.05	1.80	1.75	1.19	1.16	1.15	1.15
	Recruit devs	3.91	1.05	1.05	1.01	1.04	1.04	1.04
SSB_{1977} (mt)		9,470	9,476	9,452	9,033	9,634	9,566	9,566
SSB_{2013} (mt)		4,500	6,831	6,681	5,131	4,837	4,671	4,671
$F_{mult, 2013}$		0.31	0.23	0.24	0.31	0.31	0.35	0.35
1998 YC Age-1 (000s)		13,733	13,662	13,431	13,175	14,248	13,900	13,900
2010 YC Age-1 (000s)		5,942	11,150	10,990	8,383	7,465	7,358	7,358
2012 YC Age-1 (000s)		3,998	34,043	33,467	17,954	21,911	21,530	21,530
Mohn's rho (7 year peel)	SSB	-0.03	-0.14	-0.15	-0.21	-0.18	-0.19	-0.19
	F_{mult}	0.05	0.20	0.23	0.38	0.26	0.31	0.31
	Age 1 N	0.18	-0.19	-0.20	-0.24	-0.27	-0.27	-0.27

Table A.84. Comparison of the fleet and index selectivity parameters and the corresponding coefficients of variation (CV) from the Gulf of Maine haddock ASAP_BASE, ASAP_final_temp6, ASAP_temp10 and ASAP_temp11 models.

Block/index	Age	ASAP_BASE		ASAP_final_temp6		ASAP_final_temp10		ASAP_final_temp11	
		Selectivity	CV	Selectivity	CV	Selectivity	CV	Selectivity	CV
Fleet block 1 (1977-1988)	1	0.00	0.61	0.00	0.72	0.00	0.72	0.00	0.72
	2	0.22	0.12	0.20	0.18	0.20	0.18	0.20	0.18
	3	0.39	0.11	0.35	0.17	0.34	0.17	0.34	0.17
	4	0.64	0.10	0.57	0.17	0.56	0.16	0.56	0.16
	5	0.65	0.11	0.55	0.18	0.54	0.17	0.54	0.17
	6	0.77	0.12	0.63	0.19	0.62	0.18	0.62	0.18
	7	1.00		0.75	0.21	0.75	0.21	0.75	0.21
	8	1.00	0.00	1.00		1.00		1.00	
	9^+	1.00	0.00	1.00		1.00		1.00	
Fleet block 2 (1989-2004)	1	0.01	0.27	0.00	0.37	0.00	0.36	0.00	0.36
	2	0.06	0.16	0.04	0.23	0.04	0.22	0.04	0.22
	3	0.34	0.14	0.22	0.21	0.21	0.19	0.21	0.19
	4	0.55	0.14	0.39	0.20	0.37	0.19	0.37	0.19
	5	0.72	0.14	0.57	0.20	0.53	0.19	0.53	0.19
	6	0.96	0.14	0.81	0.19	0.77	0.18	0.77	0.18
	7	1.00		0.88	0.21	0.84	0.21	0.84	0.21
	8	0.94	0.21	1.00		1.00		1.00	
	9^+	0.67	0.30	1.00		1.00		1.00	
Fleet block 3 (2005-2013)	1	0.01	0.42	0.00	0.37	0.00	0.36	0.01	0.34
	2	0.06	0.21	0.05	0.19	0.05	0.19	0.05	0.19
	3	0.22	0.17	0.19	0.15	0.19	0.14	0.19	0.14
	4	0.31	0.17	0.30	0.15	0.30	0.15	0.30	0.15
	5	0.56	0.15	0.53	0.14	0.52	0.13	0.52	0.13
	6	0.77	0.15	0.71	0.13	0.69	0.13	0.69	0.13
	7	1.00		0.83	0.12	0.82	0.12	0.82	0.12
	8	1.00	0.00	1.00		1.00		1.00	
	9^+	0.74	0.19	0.75	0.18	0.83	0.16	0.83	0.16
NEFSC spring	1	0.74	0.13	0.67	0.15	0.74	0.13	0.78	0.13
	2	0.62	0.14	0.68	0.15	0.74	0.14	0.75	0.14
	3	0.79	0.14	0.89	0.15	0.96	0.14	0.95	0.14
	4	1.00	0.00	1.00		1.00		1.00	
	5	0.87	0.17	0.73	0.20	1.00		1.00	
	6	1.00		1.00	0.01	1.00		1.00	
	7	0.82	0.25	0.99	0.23	1.00		1.00	
	8	0.75	0.34	0.82	0.31	1.00		1.00	
	9^+	0.76	0.30	0.90	0.27	1.00		1.00	
NEFSC fall	1	0.28	0.14	0.23	0.25	0.29	0.15	0.29	0.15
	2	0.35	0.14	0.36	0.24	0.45	0.15	0.45	0.14
	3	0.60	0.13	0.51	0.24	0.64	0.14	0.64	0.14
	4	0.62	0.14	0.56	0.24	0.71	0.15	0.71	0.15
	5	0.76	0.15	0.75	0.24	0.94	0.15	0.94	0.15
	6	1.00		0.84	0.25	1.00		1.00	
	7	1.00	0.00	0.71	0.27	1.00		1.00	
	8	1.00	0.00	1.00		1.00		1.00	
	9^+	0.66	0.25	0.55	0.31	1.00		1.00	

Table A.85. Comparison of the summary diagnostics from the ASAP base model (ASAP_BASE) and the ASAP_final_temp6-11 models.

Model		ASAP_BASE	ASAP_final_temp6	ASAP_final_temp7	ASAP_final_temp8	ASAP_final_temp9	ASAP_final_temp10	ASAP_final_temp11
Description			CV_{TS}=2.0	CV_{TS}=1.0	$CV_{1977\text{-}2010}$=1.0, $CV_{2011\text{-}2013}$=0.5	$CV_{1977\text{-}2012}$=1.0, CV_{2013}=0.5	CV_{TS}=1.0, selectivity adjustments from 'temp7' run	$CV_{1977\text{-}2012}$=1.0, CV_{2013}=0.5, selectivity identical to 'temp10'
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05	4.50E-05	2.60E-05	0.000418	0.000261	3.60E-05	8.90E-05
Number of parameters		125	123	123	123	123	115	115
Objective function		2526	2386	2396	2406	2404	2400	2407
Components of objective function	Recruit devs	110	327	334	336	336	334	336
	Suvey age comps	874	667	667	667	667	670	671
	Catch age comps	644	538	538	538	538	539	539
	Index fit	702	630	632	639	637	632	636
	Catch fit	196	225	225	226	226	226	226
RMSE	Fleet 1	0.34	0.36	0.37	0.38	0.38	0.38	0.39
	Index 1	2.10	1.18	1.20	1.27	1.25	1.20	1.24
	Index 2	2.00	1.12	1.15	1.24	1.21	1.15	1.21
	Index total	2.05	1.15	1.18	1.26	1.23	1.17	1.23
	Recruit devs	3.91	1.04	1.53	1.63	1.60	1.53	1.60
SSB_{1977} (mt)		9,470	9,566	9,473	9,463	9,468	9,438	9,432
SSB_{2013} (mt)		4,500	4,671	4,245	3,021	3,597	4,153	3,517
$F_{mult, 2013}$		0.31	0.35	0.38	0.54	0.45	0.39	0.46
1998 YC Age-1 (000s)		13,733	13,900	13,681	13,432	13,546	13,429	13,304
2010 YC Age-1 (000s)		5,942	7,358	6,681	4,749	5,703	6,659	5,685
2012 YC Age-1 (000s)		3,998	21,530	16,698	8,159	9,057	16,565	8,978
Mohn's rho (7 year peel)	SSB	-0.03	-0.19	-0.15	-0.07	-0.10	-0.15	-0.10
	F_{mult}	0.05	0.31	0.27	0.18	0.21	0.30	0.24
	Age 1 N	0.18	-0.27	-0.11	0.04	-0.03	-0.10	-0.02

Table A.86. Gulf of Maine haddock January 1 numbers-at-age (000s) from 1977 to 2013 as estimated from the ASAP_final_temp10 model.

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
1977	5,997	13,897	1,450	2,025	850	463	0	0	46
1978	1,476	4,898	10,203	983	1,217	518	270	0	22
1979	6,048	1,205	3,546	6,748	567	714	289	139	10
1980	6,435	4,941	893	2,441	4,159	354	428	163	74
1981	4,612	5,252	3,505	570	1,331	2,309	186	205	94
1982	774	3,766	3,779	2,295	324	768	1,266	94	128
1983	2,445	632	2,679	2,426	1,261	181	406	611	89
1984	1,043	1,993	415	1,500	1,066	570	75	146	190
1985	282	851	1,369	251	747	543	271	32	114
1986	265	229	559	766	110	337	224	97	39
1987	134	215	138	269	263	39	107	58	24
1988	443	110	147	82	131	131	18	44	27
1989	187	361	77	93	44	72	68	9	27
1990	244	153	289	56	61	27	37	34	16
1991	267	199	119	187	30	28	9	12	13
1992	711	218	154	75	95	12	8	3	6
1993	1,318	579	168	96	38	38	4	2	2
1994	2,903	1,076	458	116	58	20	17	2	1
1995	2,540	2,373	864	341	81	37	11	9	2
1996	1,080	2,075	1,893	623	223	48	19	6	5
1997	2,179	883	1,678	1,458	458	156	31	12	6
1998	2,276	1,780	709	1,247	1,006	292	89	17	10
1999	13,429	1,861	1,436	540	899	685	184	55	16
2000	2,547	10,986	1,510	1,126	409	657	477	126	47
2001	1,121	2,083	8,888	1,167	832	289	435	309	107
2002	1,216	917	1,689	6,938	878	602	198	293	270
2003	219	995	745	1,327	5,283	647	424	137	379
2004	6,281	179	807	585	1,009	3,883	453	293	345
2005	386	5,139	146	636	447	747	2,753	317	433
2006	1,118	316	4,147	113	478	315	501	1,778	473
2007	1,218	915	255	3,243	86	345	218	336	1,457
2008	215	996	735	195	2,379	58	218	131	1,066
2009	301	176	802	565	144	1,634	38	135	736
2010	966	246	142	621	424	102	1,092	24	555
2011	6,659	790	198	109	457	288	65	664	350
2012	2,090	5,443	635	152	80	313	186	40	601
2013	16,565	1,708	4,348	474	108	51	183	102	348

Exclude final 2 years (1977-2011)	
Median recruitment	1,121
Mean recruitment	2,267
Geometric mean	1,137
All years (1977-2013)	
Median recruitment	1,216
Mean recruitment	2,648
Geometric mean	1,242

Table A.87. Gulf of Maine haddock January 1 biomass (mt) and spawning stock biomass (mt) and exploitable biomass from 1977 to 2013 as estimated from the ASAP_final_temp10 model.

Year	January 1 biomass (mt)	Spawning stock biomass (mt)	Exploitable biomass (mt)
1977	17,102	9,438	7,313
1978	18,168	13,392	9,686
1979	19,034	15,178	11,474
1980	19,834	14,400	11,955
1981	19,422	13,675	11,130
1982	17,639	13,068	10,830
1983	13,353	9,895	9,173
1984	8,774	6,618	6,124
1985	6,565	4,796	4,091
1986	3,909	2,735	2,582
1987	1,931	1,456	1,456
1988	1,366	1,049	923
1989	1,072	759	678
1990	1,124	793	621
1991	999	679	484
1992	987	600	329
1993	1,191	610	313
1994	2,026	1,003	483
1995	3,450	1,802	819
1996	4,267	2,962	1,338
1997	6,062	4,568	2,616
1998	7,386	5,646	3,283
1999	7,900	5,606	3,194
2000	11,021	6,607	4,080
2001	13,966	10,840	6,184
2002	15,007	13,206	7,367
2003	12,629	11,341	7,810
2004	11,302	9,641	8,219
2005	10,276	8,098	6,728
2006	8,909	7,443	5,787
2007	7,610	6,427	4,446
2008	6,435	5,464	4,035
2009	5,453	4,771	3,746
2010	4,617	3,904	3,214
2011	4,856	3,062	2,743
2012	5,396	2,961	2,012
2013	7,749	4,153	2,158
Min	987	600	313
Max	19,834	15,178	11,955
Average	8,346	6,180	4,579

Table A.88. Gulf of Maine haddock fishing mortality-at-age and the fully recruited fishing mortality (F_{full}) from 1977 to 2013 as estimated from the ASAP_final_temp10 model.

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9^{+}	F_{full}
1977	0.00	0.11	0.19	0.31	0.30	0.34	0.41	0.55	0.55	0.55
1978	0.00	0.12	0.21	0.35	0.33	0.38	0.46	0.62	0.62	0.62
1979	0.00	0.10	0.17	0.28	0.27	0.31	0.38	0.50	0.50	0.50
1980	0.00	0.14	0.25	0.41	0.39	0.44	0.54	0.72	0.72	0.72
1981	0.00	0.13	0.22	0.37	0.35	0.40	0.49	0.65	0.65	0.65
1982	0.00	0.14	0.24	0.40	0.38	0.44	0.53	0.71	0.71	0.71
1983	0.00	0.22	0.38	0.62	0.59	0.68	0.82	1.10	1.10	1.10
1984	0.00	0.18	0.30	0.50	0.47	0.54	0.66	0.88	0.88	0.88
1985	0.00	0.22	0.38	0.62	0.60	0.68	0.83	1.11	1.11	1.11
1986	0.01	0.31	0.53	0.87	0.83	0.95	1.15	1.54	1.54	1.54
1987	0.00	0.18	0.32	0.52	0.50	0.57	0.69	0.93	0.93	0.93
1988	0.00	0.15	0.26	0.42	0.40	0.46	0.55	0.74	0.74	0.74
1989	0.00	0.03	0.12	0.22	0.31	0.45	0.50	0.59	0.59	0.59
1990	0.00	0.05	0.23	0.41	0.60	0.86	0.94	1.12	1.12	1.12
1991	0.00	0.06	0.27	0.48	0.70	1.00	1.10	1.31	1.31	1.31
1992	0.00	0.06	0.27	0.49	0.70	1.02	1.11	1.32	1.32	1.32
1993	0.00	0.03	0.17	0.30	0.44	0.63	0.69	0.82	0.82	0.82
1994	0.00	0.02	0.09	0.17	0.24	0.35	0.38	0.46	0.46	0.46
1995	0.00	0.03	0.13	0.22	0.33	0.47	0.51	0.61	0.61	0.61
1996	0.00	0.01	0.06	0.11	0.16	0.23	0.25	0.29	0.29	0.29
1997	0.00	0.02	0.10	0.17	0.25	0.36	0.39	0.47	0.47	0.47
1998	0.00	0.01	0.07	0.13	0.18	0.27	0.29	0.35	0.35	0.35
1999	0.00	0.01	0.04	0.08	0.11	0.16	0.18	0.21	0.21	0.21
2000	0.00	0.01	0.06	0.10	0.15	0.21	0.23	0.28	0.28	0.28
2001	0.00	0.01	0.05	0.08	0.12	0.18	0.19	0.23	0.23	0.23
2002	0.00	0.01	0.04	0.07	0.11	0.15	0.17	0.20	0.20	0.20
2003	0.00	0.01	0.04	0.07	0.11	0.16	0.17	0.20	0.20	0.20
2004	0.00	0.01	0.04	0.07	0.10	0.14	0.16	0.19	0.19	0.19
2005	0.00	0.01	0.05	0.09	0.15	0.20	0.24	0.29	0.24	0.29
2006	0.00	0.01	0.05	0.07	0.13	0.17	0.20	0.24	0.20	0.24
2007	0.00	0.02	0.07	0.11	0.19	0.26	0.31	0.37	0.31	0.37
2008	0.00	0.02	0.06	0.10	0.18	0.23	0.28	0.34	0.28	0.34
2009	0.00	0.01	0.06	0.09	0.15	0.20	0.24	0.29	0.24	0.29
2010	0.00	0.02	0.07	0.11	0.19	0.25	0.30	0.36	0.30	0.36
2011	0.00	0.02	0.06	0.10	0.18	0.24	0.28	0.34	0.28	0.34
2012	0.00	0.02	0.09	0.14	0.25	0.34	0.40	0.49	0.41	0.49
2013	0.00	0.02	0.07	0.12	0.20	0.27	0.32	0.39	0.33	0.39

Table A.89. Summary of the Gulf of Maine haddock 2013 point estimates and their corresponding 90% probability intervals for the ASAP_final_temp10 model.

Metric	**ASAP point estimate**	**90% probability interval**
SSB_{2013} (mt)	4,153	(2,960 - 6,043)
B_{2013} (mt)	7,749	(5,470 - 11,039)
F_{full}	0.39	(0.24 - 0.60)

Table A.90. Inputs to the Gulf of Maine haddock yield per recruit analysis. Values in italics are the coefficients of variation (CV) used in the stochastic calculations (zero values were replaced with values of 0.001). Natural mortality CVs were assumed.

Age	Catch weights (kg)		Stock weights (kg)		Fishery selectivity		Maturity		Natural mortality	
1	0.30	*0.14*	0.15	*0.14*	0.00	*0.36*	0.04	*0.23*	0.20	*0.10*
2	0.60	*0.11*	0.40	*0.13*	0.05	*0.19*	0.28	*0.08*	0.20	*0.10*
3	0.89	*0.11*	0.71	*0.07*	0.19	*0.14*	0.81	*0.02*	0.20	*0.10*
4	1.17	*0.06*	1.00	*0.05*	0.30	*0.15*	0.98	*0.00*	0.20	*0.10*
5	1.40	*0.05*	1.24	*0.03*	0.52	*0.13*	1.00	*0.00*	0.20	*0.10*
6	1.55	*0.05*	1.43	*0.03*	0.69	*0.13*	1.00	*0.00*	0.20	*0.10*
7	1.70	*0.05*	1.59	*0.08*	0.82	*0.12*	1.00	*0.00*	0.20	*0.10*
8	1.96	*0.07*	1.82	*0.03*	1.00	*0.00*	1.00	*0.00*	0.20	*0.10*
9^+	2.04	*0.04*	2.04	*0.04*	0.83	*0.16*	1.00	*0.00*	0.20	*0.10*

Table A.91. Fishing mortality (F), yield per recruit (YPR) and mean age across a range of percent maximum spawning potential (% MSP) values.

F	YPR (kg)	SSB/R	% MSP	Mean Age	F	YPR (kg)	SSB/R	% MSP	Mean Age
0.00	0.00	5.05	100.0	5.52	0.50	0.48	1.93	38.3	3.10
0.01	0.03	4.86	96.3	5.36	0.51	0.48	1.91	37.9	3.09
0.02	0.06	4.69	92.9	5.22	0.52	0.48	1.89	37.5	3.07
0.03	0.09	4.53	89.7	5.09	0.53	0.49	1.87	37.2	3.06
0.04	0.12	4.38	86.8	4.96	0.54	0.49	1.86	36.8	3.05
0.05	0.14	4.24	84.0	4.85	0.55	0.49	1.84	36.4	3.03
0.06	0.16	4.11	81.5	4.75	0.56	0.49	1.82	36.1	3.02
0.07	0.18	3.99	79.1	4.65	0.57	0.49	1.80	35.8	3.01
0.08	0.20	3.88	76.9	4.56	0.58	0.50	1.79	35.4	3.00
0.09	0.22	3.77	74.8	4.48	0.59	0.50	1.77	35.1	2.99
0.10	0.24	3.68	72.9	4.40	0.60	0.50	1.76	34.8	2.97
0.11	0.25	3.58	71.0	4.33	0.61	0.50	1.74	34.5	2.96
0.12	0.26	3.49	69.3	4.26	0.62	0.50	1.72	34.2	2.95
0.13	0.28	3.41	67.6	4.19	0.63	0.50	1.71	33.9	2.94
0.14	0.29	3.33	66.1	4.13	0.64	0.50	1.70	33.6	2.93
0.15	0.30	3.26	64.6	4.08	0.65	0.51	1.68	33.3	2.92
0.16	0.31	3.19	63.2	4.02	0.66	0.51	1.67	33.0	2.91
0.17	0.32	3.12	61.9	3.97	0.67	0.51	1.65	32.8	2.90
0.18	0.33	3.06	60.6	3.92	0.68	0.51	1.64	32.5	2.89
0.19	0.34	3.00	59.4	3.88	0.69	0.51	1.63	32.3	2.88
0.20	0.35	2.94	58.2	3.83	0.70	0.51	1.61	32.0	2.88
0.21	0.36	2.88	57.2	3.79	0.71	0.51	1.60	31.7	2.87
0.22	0.37	2.83	56.1	3.75	0.72	0.51	1.59	31.5	2.86
0.23	0.37	2.78	55.1	3.71	0.73	0.51	1.58	31.3	2.85
0.24	0.38	2.73	54.1	3.68	0.74	0.52	1.57	31.0	2.84
0.25	0.39	2.69	53.2	3.64	0.75	0.52	1.55	30.8	2.83
0.26	0.39	2.64	52.3	3.61	0.76	0.52	1.54	30.6	2.82
0.27	0.40	2.60	51.5	3.58	0.77	0.52	1.53	30.3	2.82
0.28	0.40	2.56	50.7	3.55	0.78	0.52	1.52	30.1	2.81
0.29	0.41	2.52	49.9	3.52	0.79	0.52	1.51	29.9	2.80
0.30	0.41	2.48	49.1	3.49	0.80	0.52	1.50	29.7	2.79
0.31	0.42	2.44	48.4	3.46	0.81	0.52	1.49	29.5	2.79
0.32	0.42	2.41	47.7	3.44	0.82	0.52	1.48	29.3	2.78
0.33	0.43	2.37	47.0	3.41	0.83	0.52	1.47	29.1	2.77
0.34	0.43	2.34	46.4	3.39	0.84	0.52	1.46	28.9	2.77
0.35	0.44	2.31	45.7	3.37	0.85	0.52	1.45	28.7	2.76
0.36	0.44	2.28	45.1	3.35	0.86	0.52	1.44	28.5	2.75
0.37	0.44	2.25	44.5	3.32	0.87	0.53	1.43	28.3	2.74
0.38	0.45	2.22	44.0	3.30	0.88	0.53	1.42	28.1	2.74
0.39	0.45	2.19	43.4	3.28	0.89	0.53	1.41	28.0	2.73
0.40	0.45	2.16	42.9	3.26	0.90	0.53	1.40	27.8	2.73
0.41	0.46	2.14	42.4	3.24	0.91	0.53	1.39	27.6	2.72
0.42	0.46	2.11	41.9	3.23	0.92	0.53	1.38	27.4	2.71
0.43	0.46	2.09	41.4	3.21	0.93	0.53	1.38	27.3	2.71
0.44	0.47	2.06	40.9	3.19	0.94	0.53	1.37	27.1	2.70
0.45	0.47	2.04	40.4	3.18	0.95	0.53	1.36	26.9	2.69
0.46	0.47	2.02	40.0	3.16	0.96	0.53	1.35	26.8	2.69
0.47	0.47	2.00	39.5	3.14	0.97	0.53	1.34	26.6	2.68
0.48	0.48	1.97	39.1	3.13	0.98	0.53	1.33	26.4	2.68
0.49	0.48	1.95	38.7	3.11	0.99	0.53	1.33	26.3	2.67

Table A.92. Proxy reference points from the Gulf of Maine haddock yield per recruit (YPR) analysis and the corresponding fishing mortality, spawning stock biomass per recruit (SSB/R), biomass per recruit and mean age values. Italicized numbers in parentheses next to the F results indicate the corresponding CVs.

Ref. point	F		YPR (kg)	SSB/R (kg)	Biomass/ Recruit (kg)	Mean Age
F_0	0.00		0.00	5.05	5.79	5.52
$F_{0.1}$	0.38	*(0.10)*	0.45	2.22	2.93	3.31
F_{MAX}	2.31	*(0.23)*	0.55	0.78	1.43	2.27
$F_{40\%}$	0.46	*(0.10)*	0.47	2.02	2.73	3.16

Table A.93. Non-parametric proxy reference points for Gulf of Maine haddock based on the empirical cumulative density functions from two different age-1 recruitment time series: 1977-2011 and 1977-2013. **The SAW 59 WG recommended that the 1977-2011 time series be used for stock status determination.**

Recruitment series	F_{MSY} (proxy)	Fmsy	SSB_{MSY} (mt)	MSY (mt)	Median age1 recruitment
1977-2011	$F_{40\%}$	0.46 (0.36 - 0.54)	4,108 (1,774 - 7,861)	955 (421 - 1,807)	1,121
1977-2013*	$F_{40\%}$	0.46 (0.36 - 0.54)	4,613 (1,936 - 9,903)	1,079 (460 - 2,271)	1,207

* *Sensitivity only*

Table A.94. Short-term projections of total fishery yield and spawning stock biomass for Gulf of Maine haddock based on a harvest scenario of a) fishing at $F_{40\%}$ between 2014 and 2017 and b) an assumed catch of 500 mt in 2014 and fishing at $F_{40\%}$ between 2015 and 2017. Projections are shown based on two different population models to highlight the sensitivity of catch projections to the size of the 2012 year class. Projection results are shown for the base ASAP model (ASAP_final_temp10) and a sensitivity model that constrains the size of the terminal year class (ASAP_final_temp11). Confidence intervals in parentheses are 90% intervals.

		ASAP_final_temp10 (1977-2011 recruitment)						
Year	**Input**	**Catch (mt)**		**Spawning stock biomass (mt)**		**Harvest strategy**	**F_{full}**	
2013	Catch input/model result	692		4,153	(2,690 - 6,043)		0.39	(0.24 - 0.60)
2014	Projection	1,085	(713 - 1,605)	6,341	(4,272 - 9,237)	$F_{40\%}$	0.46	
2015	Projection	1,752	(1,140 - 2,633)	10,014	(6,556 - 15,250)	$F_{40\%}$	0.46	
2016	Projection	2,085	(1,367 - 3,181)	10,844	(7,036 - 16,645)	$F_{40\%}$	0.46	
2017	Projection	2,424	(1,567 - 3,755)	9,808	(6,355 - 14,914)	$F_{40\%}$	0.46	
2013	Catch input/model result	692		4,153	(2,690 - 6,043)		0.39	(0.24 - 0.60)
2014	Imputed catch	500		6,472	(4,328 - 9,473)		0.20	(0.13 - 0.31)
2015	Projection	1,871	(1,189 - 2,848)	10,507	(6,788 - 16,090)	$F_{40\%}$	0.46	
2016	Projection	2,189	(1,409 - 3,369)	11,223	(7,223 - 17,291)	$F_{40\%}$	0.46	
2017	Projection	2,512	(1,607 - 3,896)	10,078	(6,487 - 15,332)	$F_{40\%}$	0.46	

		ASAP_final_temp11 (1977-2011 recruitment)						
Year	**Input**	**Catch (mt)**		**Spawning stock biomass (mt)**		**Harvest strategy**	**F_{full}**	
2013	Catch input/model result	692		3,643	(2,500 - 5,089)		0.43	(0.28 - 0.67)
2014	Projection	870	(563 - 1,276)	4,961	(3,323 - 7,036)	$F_{40\%}$	0.46	
2015	Projection	1,271	(843 - 1,850)	6,833	(4,620 - 9,805)	$F_{40\%}$	0.46	
2016	Projection	1,456	(989 - 2,104)	7,148	(4,869 - 10,253)	$F_{40\%}$	0.46	
2017	Projection	1,620	(1,099 - 2,376)	6,568	(4,459 - 9,719)	$F_{40\%}$	0.46	
2013	Catch input/model result	692		3,643	(2,500 - 5,089)		0.43	(0.28 - 0.67)
2014	Imputed catch	500		5,050	(3,345 - 7,213)		0.25	(0.17 - 0.40)
2015	Projection	1,350	(863 - 2,011)	7,154	(4,698 - 10,401)	$F_{40\%}$	0.46	
2016	Projection	1,524	(1,004 - 2,239)	7,388	(4,947 - 10,679)	$F_{40\%}$	0.46	
2017	Projection	1,674	(1,113 - 2,473)	6,739	(4,525 - 9,986)	$F_{40\%}$	0.46	

Table A.95. Short-term projections of total fishery yield and spawning stock biomass for Gulf of Maine haddock based on a harvest scenario of a) fishing at 75% $F_{40\%}$ between 2014 and 2017 and b) an assumed catch of 500 mt in 2014 and fishing at 75% $F_{40\%}$ between 2015 and 2017. Projections are shown based on two different population models to highlight the sensitivity of catch projections to the size of the 2012 year class. Projection results are shown for the base ASAP model (ASAP_final_temp10) and a sensitivity model that constrains the size of the terminal year class (ASAP_final_temp11). Confidence intervals in parentheses are 90% intervals.

		ASAP_final_temp10 (1977-2011 recruitment)							
Year	**Input**	**Catch (mt)**		**Spawning stock biomass (mt)**		**Harvest strategy**	**F_{full}**	
2013	Catch input/model result	692		4,153	(2,690 - 6,043)		0.39	(0.24 - 0.60)
2014	Projection	844	(554 - 1,250)	6,396	(4,308 - 9,315)	75% of $F_{40\%}$	0.35	
2015	Projection	1,399	(911 - 2,102)	10,313	(6,768 - 15,681)	75% of $F_{40\%}$	0.35	
2016	Projection	1,722	(1,129 - 2,620)	11,463	(7,464 - 17,521)	75% of $F_{40\%}$	0.35	
2017	Projection	2,078	(1,348 - 3,202)	10,747	(6,982 - 16,226)	75% of $F_{40\%}$	0.35	
2013	Catch input/model result	692		4,153	(2,690 - 6,043)		0.39	(0.24 - 0.60)
2014	Imputed catch	500		6,472	(4,328 - 9,473)		0.20	(0.13 - 0.31)
2015	Projection	1,454	(924 - 2,214)	10,605	(6,854 - 16,241)	75% of $F_{40\%}$	0.35	
2016	Projection	1,772	(1,139 - 2,720)	11,709	(7,545 - 18,018)	75% of $F_{40\%}$	0.35	
2017	Projection	2,125	(1,360 - 3,288)	10,923	(7,056 - 16,574)	75% of $F_{40\%}$	0.35	

		ASAP_final_temp11 (1977-2011 recruitment)						
Year	**Input**	**Catch (mt)**		**Spawning stock biomass (mt)**		**Harvest strategy**	**F_{full}**	
2013	Catch input/model result	692		3,643	(2,500 - 5,089)		0.43	(0.28 - 0.67)
2014	Projection	677	(438 - 993)	5,008	(3,354 - 7,105)	75% of $F_{40\%}$	0.35	
2015	Projection	1,022	(677 - 1,487)	7,066	(4,781 - 10,116)	75% of $F_{40\%}$	0.35	
2016	Projection	1,213	(822 - 1,754)	7,604	(5,195 - 10,882)	75% of $F_{40\%}$	0.35	
2017	Projection	1,399	(948 - 2,048)	7,235	(4,928 - 10,596)	75% of $F_{40\%}$	0.35	
2013	Catch input/model result	692		3,643	(2,500 - 5,089)		0.43	(0.28 - 0.67)
2014	Imputed catch	500		5,050	(3,345 - 7,213)		0.25	(0.17 - 0.40)
2015	Projection	1,051	(671 - 1,565)	7,230	(4,749 - 10,502)	75% of $F_{40\%}$	0.35	
2016	Projection	1,241	(816 - 1,824)	7,732	(5,182 - 11,165)	75% of $F_{40\%}$	0.35	
2017	Projection	1,423	(944 - 2,102)	7,321	(4,931 - 10,781)	75% of $F_{40\%}$	0.35	

Figures

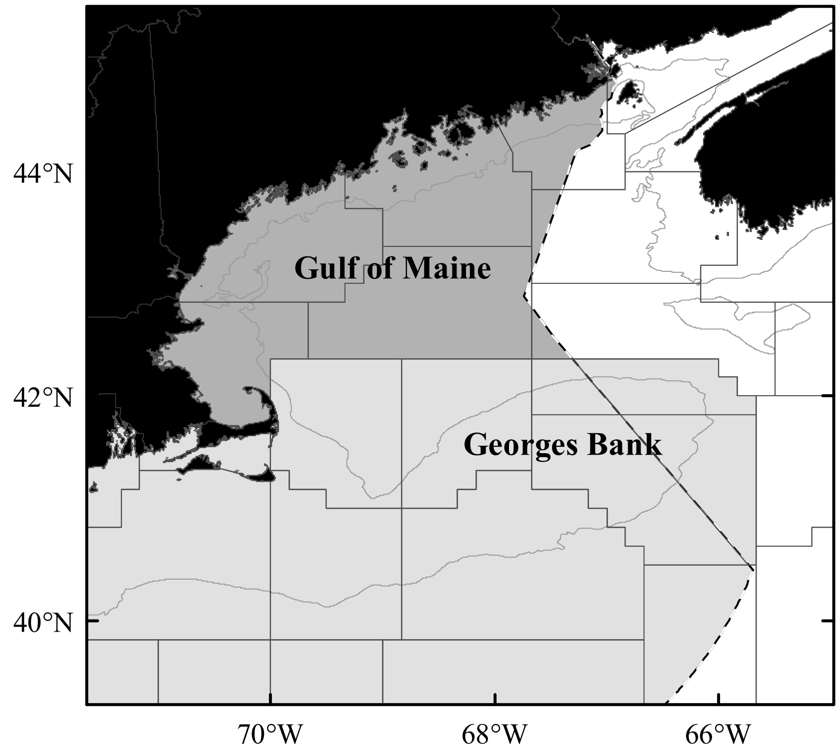


Figure A.1. Map showing the delineation of the Gulf of Maine and Georges Bank haddock (*Melanogrammus aeglefinus*) stocks. The United States exclusive economic zone (EEZ) is indicated by the dashed line.

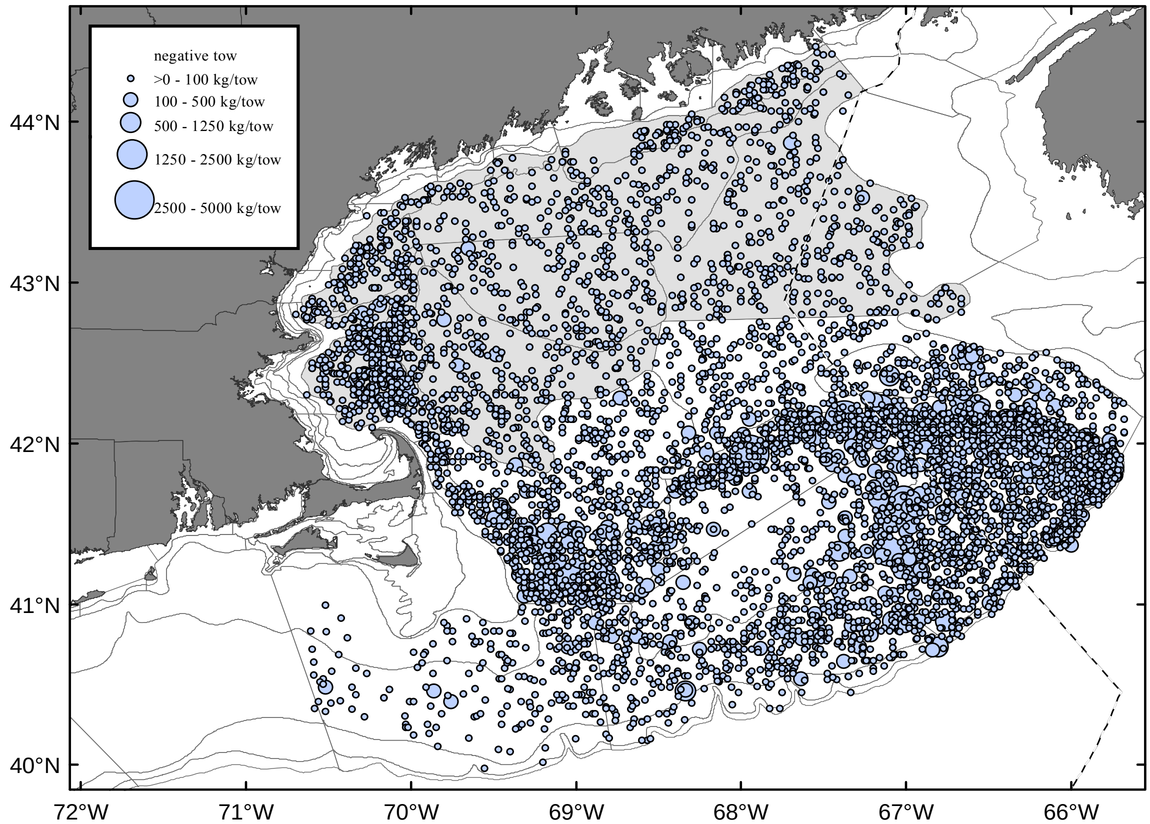


Figure A.2. Distributions of haddock catches from the Northeast Fisheries Science Center's (NEFSC) spring and fall bottom trawl surveys within the Gulf of Maine and Georges Bank regions.

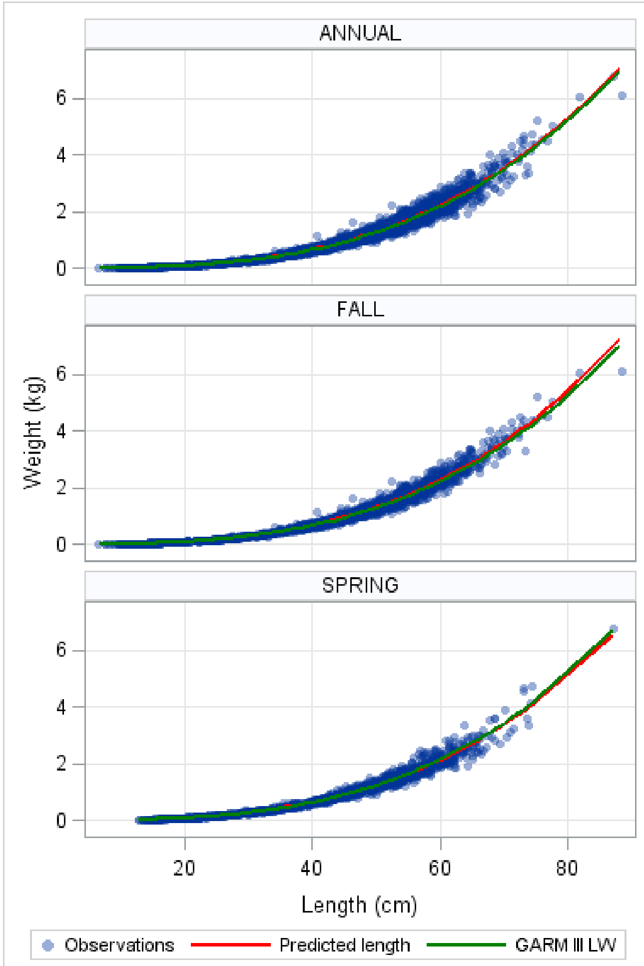


Figure A.3. Gulf of Maine haddock seasonal and annual length-weight (LW) relationships estimated from NEFSC bottom trawl survey data from 1992 to 2013. Re-estimated LW relationships are compared to the corresponding LW equations used for the GARM III assessment.

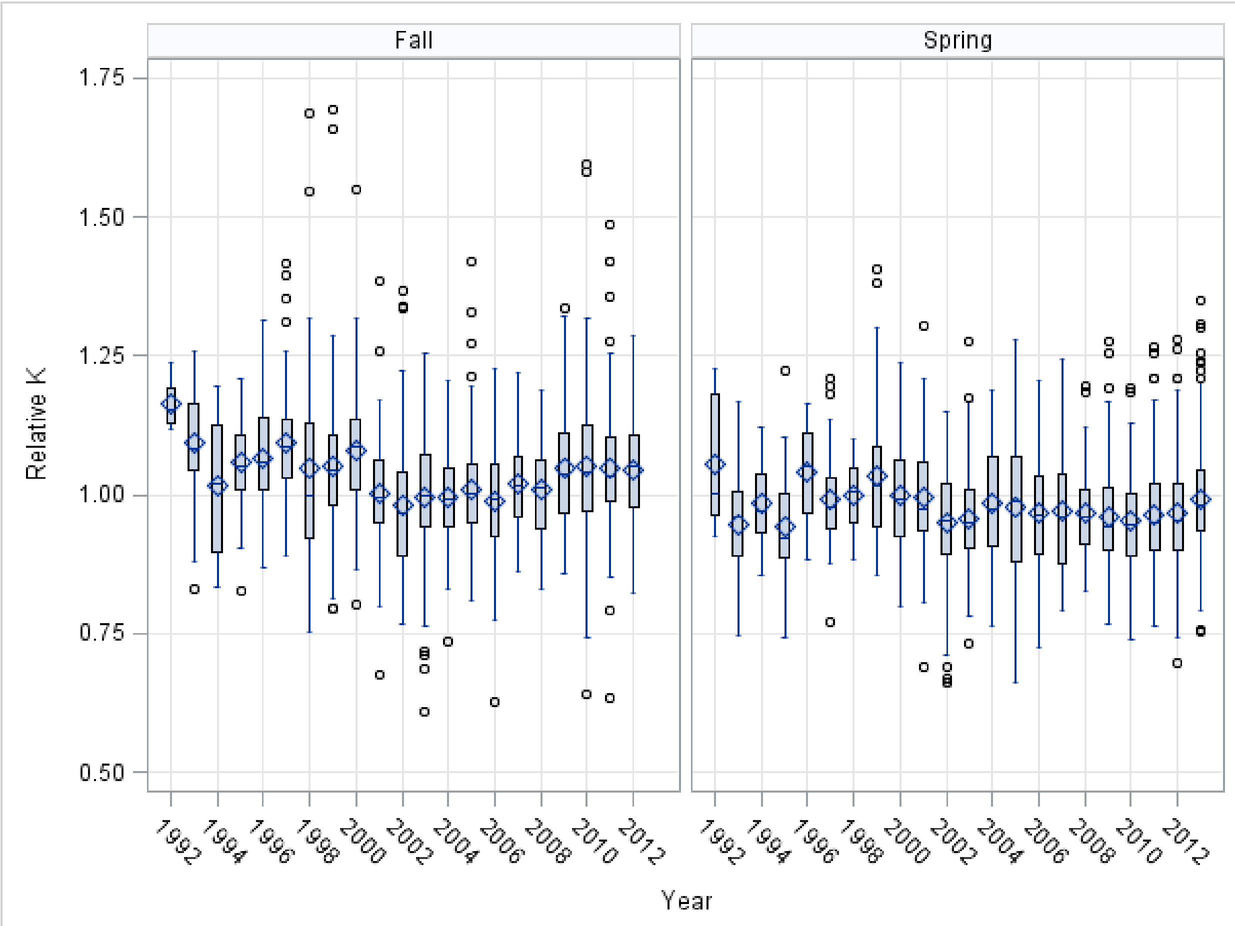


Figure A.4. Annual trends in the seasonal condition factor of Gulf of Maine haddock based on length and weight data collected from the NEFSC bottom trawl survey between 1992 and 2013.

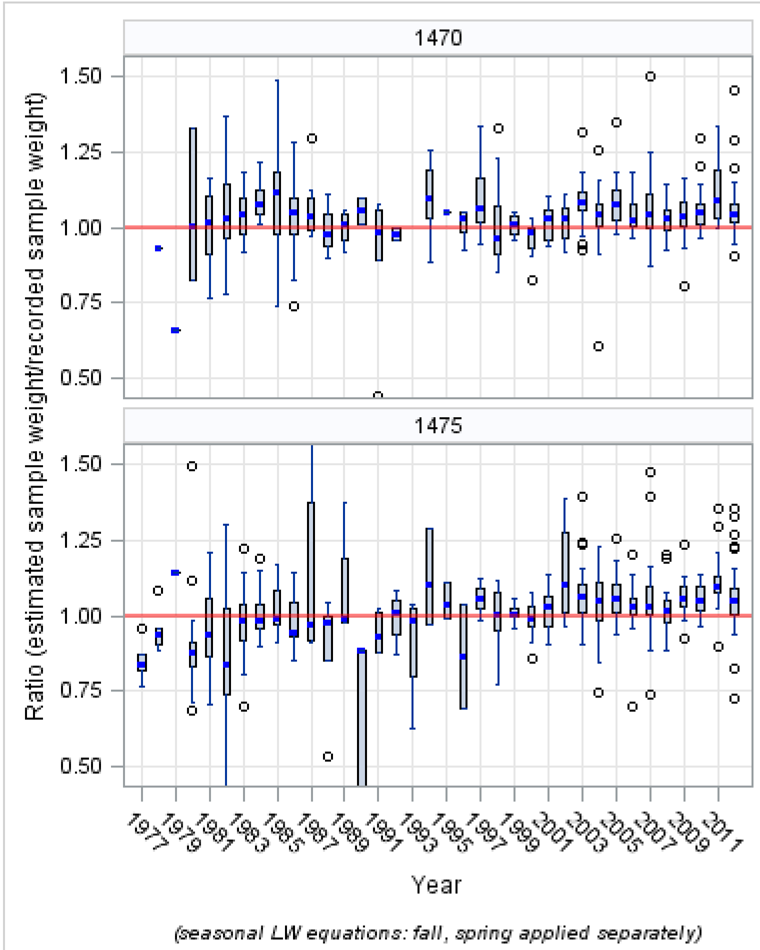


Figure A.5. Distribution of the ratios of estimated commercial biological sample weights to the recorded sample weight by market category (large=1470, scrod=1475) and year using the established gutted-to-live conversion factor of 1.14. Estimated sample weights were obtained by applying the seasonal (spring, fall) length weight equations to the recorded length distribution of the sample. The solid red line indicates the 1.0 equality line.

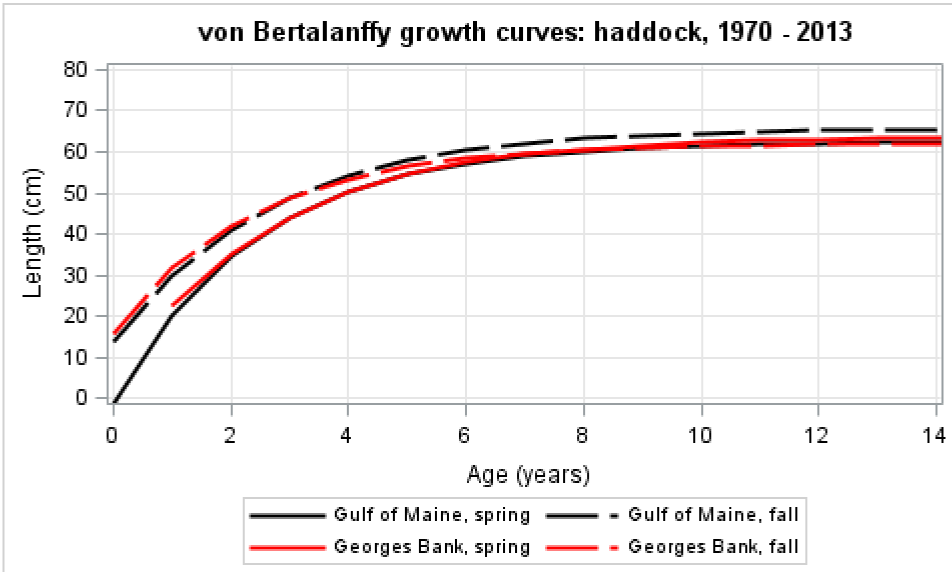


Figure A.6. Comparison of spring and fall von Bertalanffy growth curves for the Gulf of Maine and Georges Banks haddock stocks as estimated from data collected from the Northeast Fisheries Science Center bottom trawl survey between 1970 and 2013.

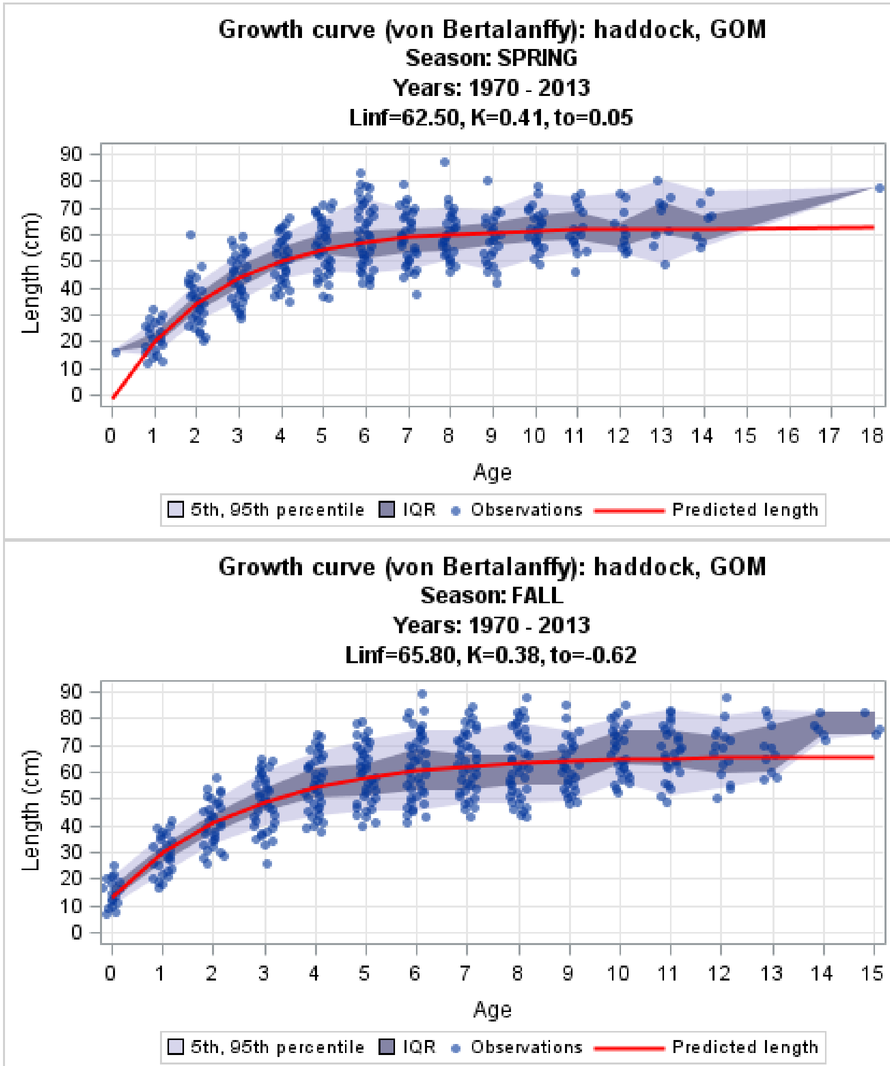


Figure A.7. Gulf of Maine haddock spring (top) and fall (bottom) von Bertalanffy growth curves estimated from data collected from the Northeast Fisheries Science Center bottom trawl survey between 1970 and 2013.

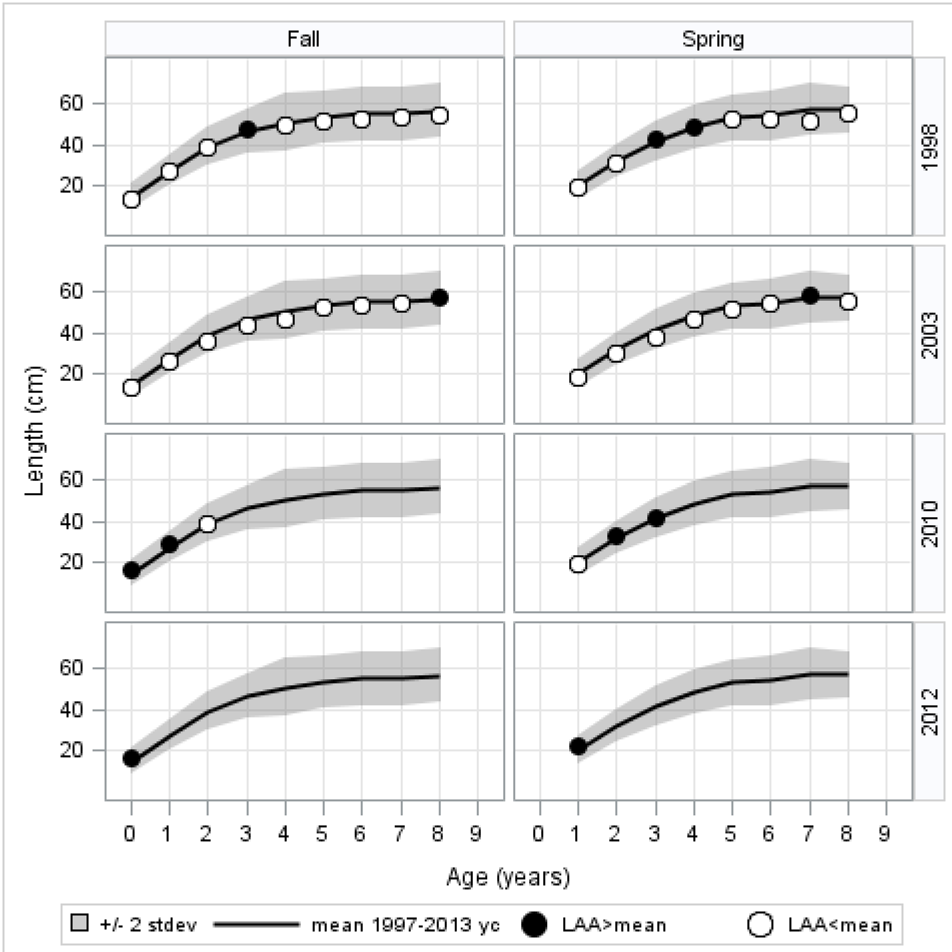


Figure A.8. Mean length-at-age (LAA) of the 1998, 2003, 2010 and 2012 year-classes as estimated from the Northeast Fisheries Science Center's (NEFSC) fall (left) and spring (right) bottom trawl surveys. The mean LAA for each cohort is compared to the 1997-2013 time series average (solid line) ± 2 standard deviations (grey band).

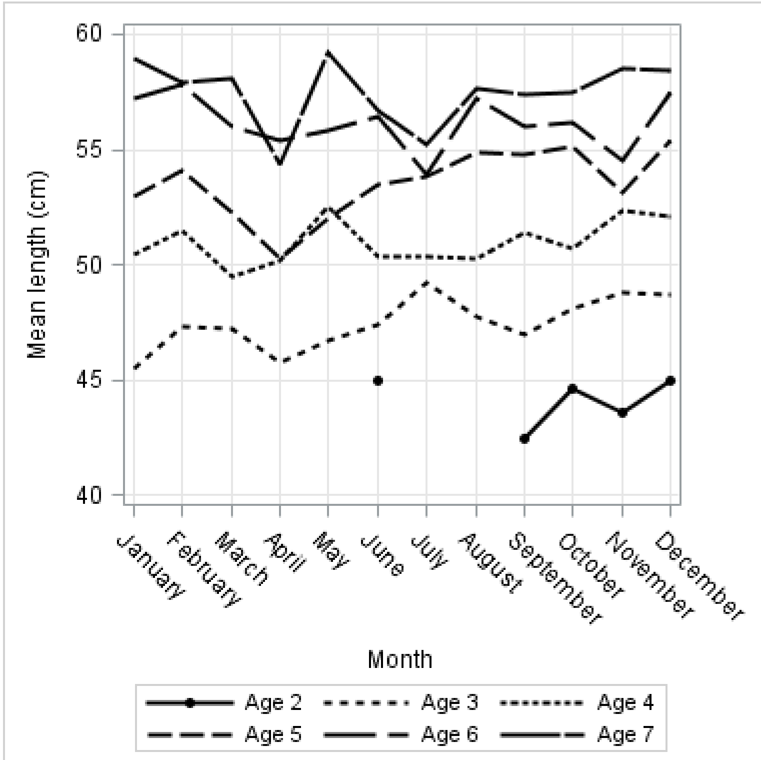


Figure A.9. Mean length-at-age of Gulf of Maine haddock by month as estimated from commercial port samples taken between 2004 and 2013.

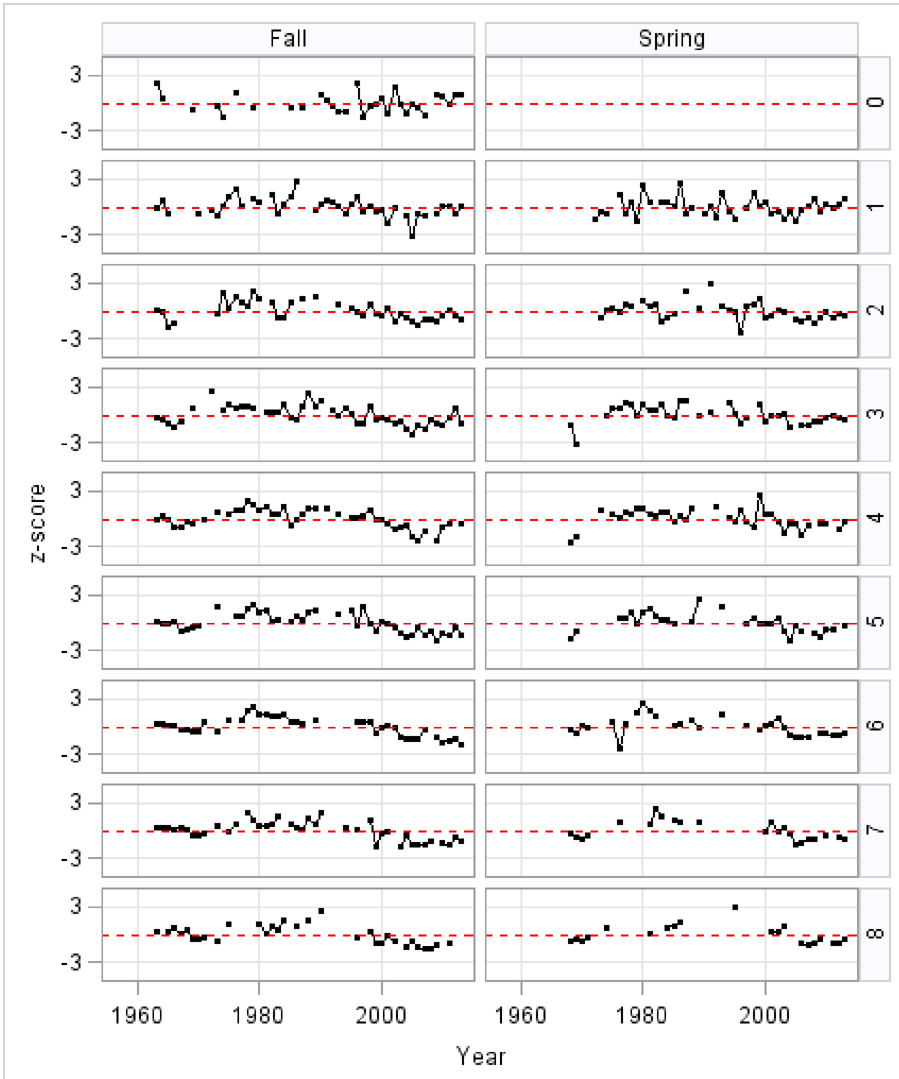


Figure A.10. Average lengths-at-age of Gulf of Maine haddock age 0 to 8 from 1963 to 2013. Survey lengths are based on the average lengths-at-age of haddock sampled from the Northeast Fisheries Science Center spring and fall bottom trawl survey. Average lengths are presented as z-scores ([*x*-μ]/σ).

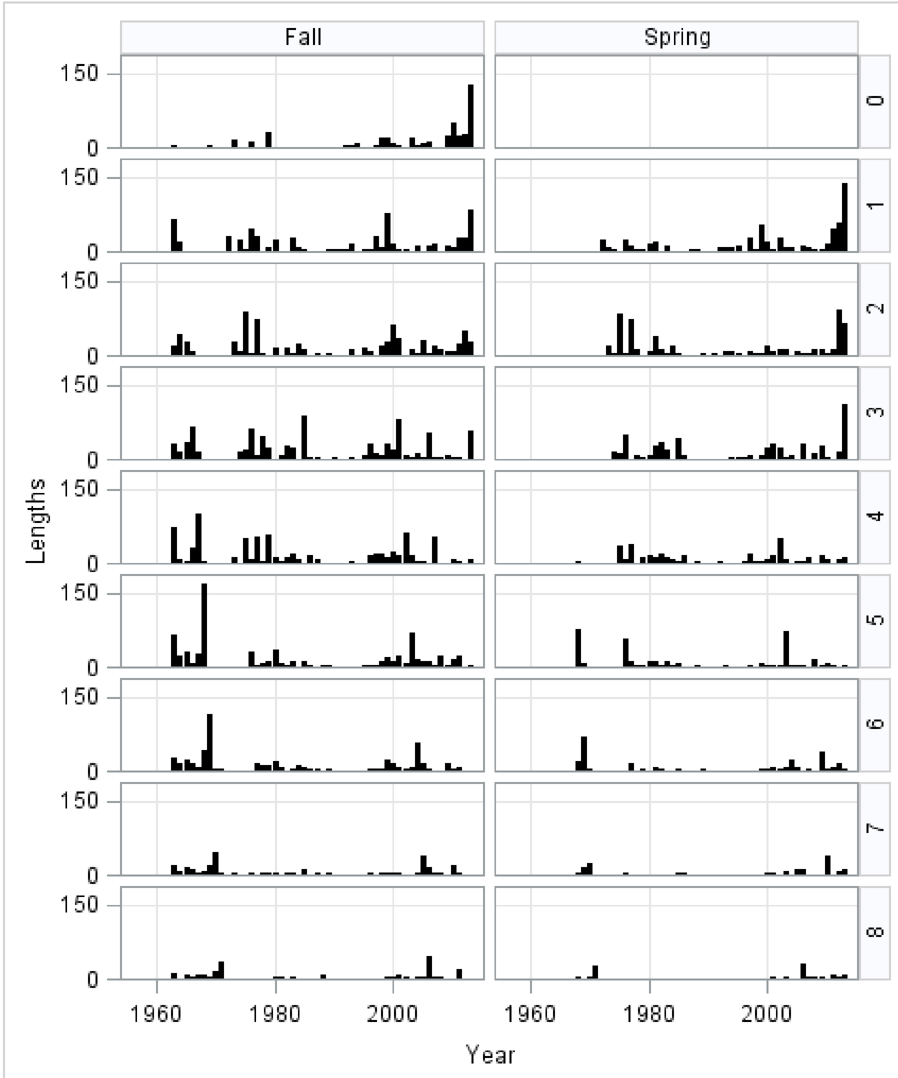


Figure A.11. Number of length measurements of age 0 to 8 Gulf of Maine haddock taken from the Northeast Fisheries Science Center spring and fall bottom trawl survey between 1963 to 2013.

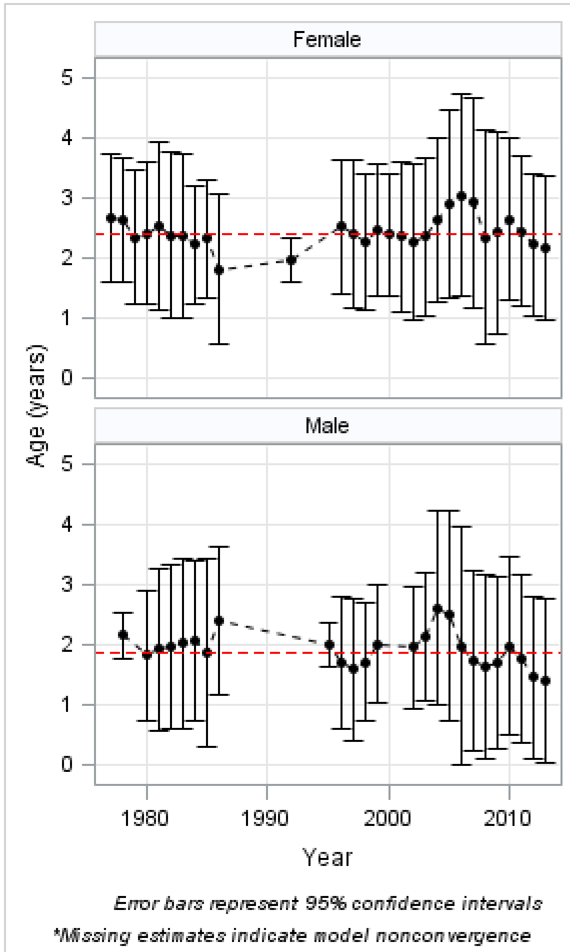


Figure A.12. Annual estimates of age-at-50% maturity ($A_{50\%}$) and the corresponding 95% confidence intervals for female and male Gulf of Maine haddock from 1970 to 2013. Average maturity has been estimated from data collected from the Northeast Fisheries Science Center (NEFSC) spring bottom trawl survey. Years in which the A*50%* could not be estimated are omitted from the plots.

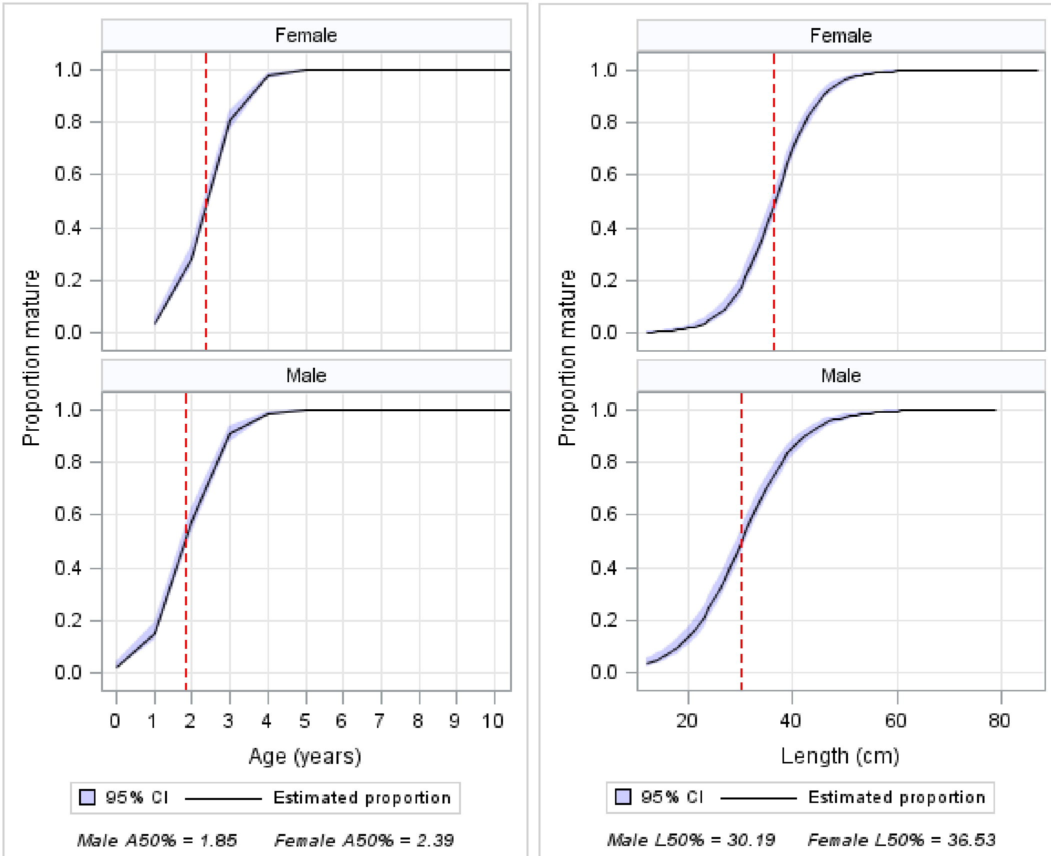


Figure A.13. Age- and length-based maturity ogives for female and male Gulf of Maine haddock based on time series averages of maturity and age information collected from the Northeast Fisheries Science Center (NEFSC) spring bottom trawl survey from 1977 to 2013. The dashed red line indicates the age/length at 50% maturity ($A_{50\%}$, $L_{50\%}$).

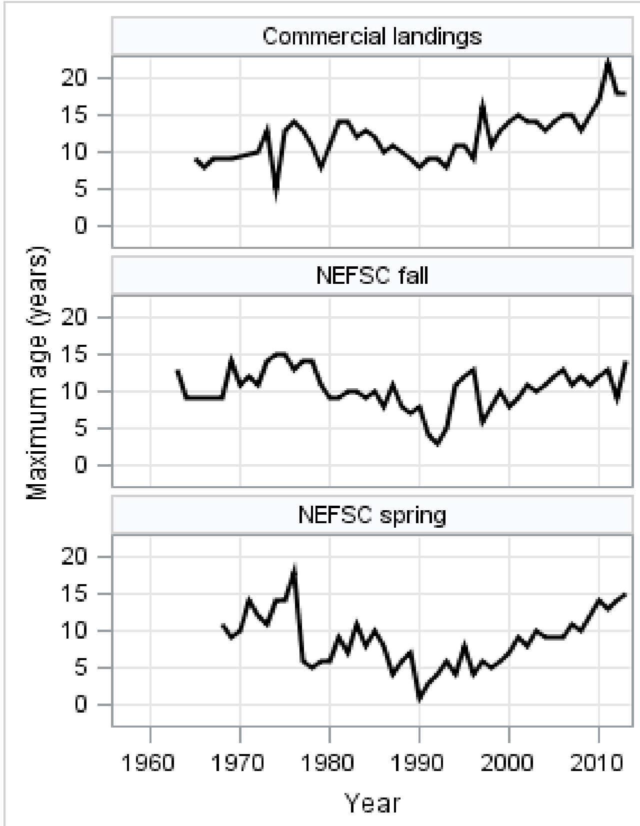


Figure A.14. Maximum age of Gulf of Maine haddock observed from the Northeast Fisheries Science Center (NEFSC) bottom trawl surveys and commercial landings between 1963 and 2013.

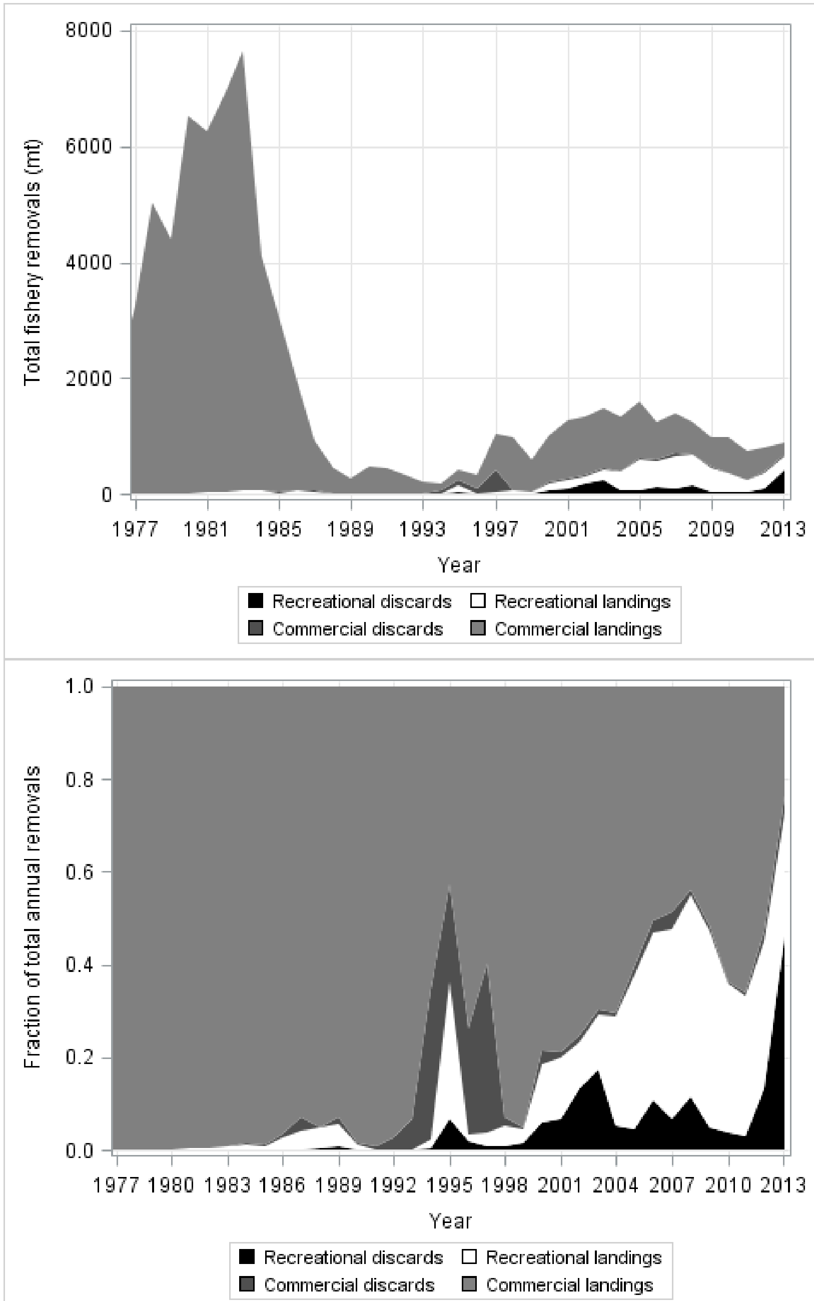


Figure A.15. Total (top) and fractional (as a fraction of the total, bottom) catch of Gulf of Maine haddock from 1977 to 2013 by fleet (commercial and recreational) and disposition (landed, discarded). **Recreational discard estimates shown do not account for post-release survival.**

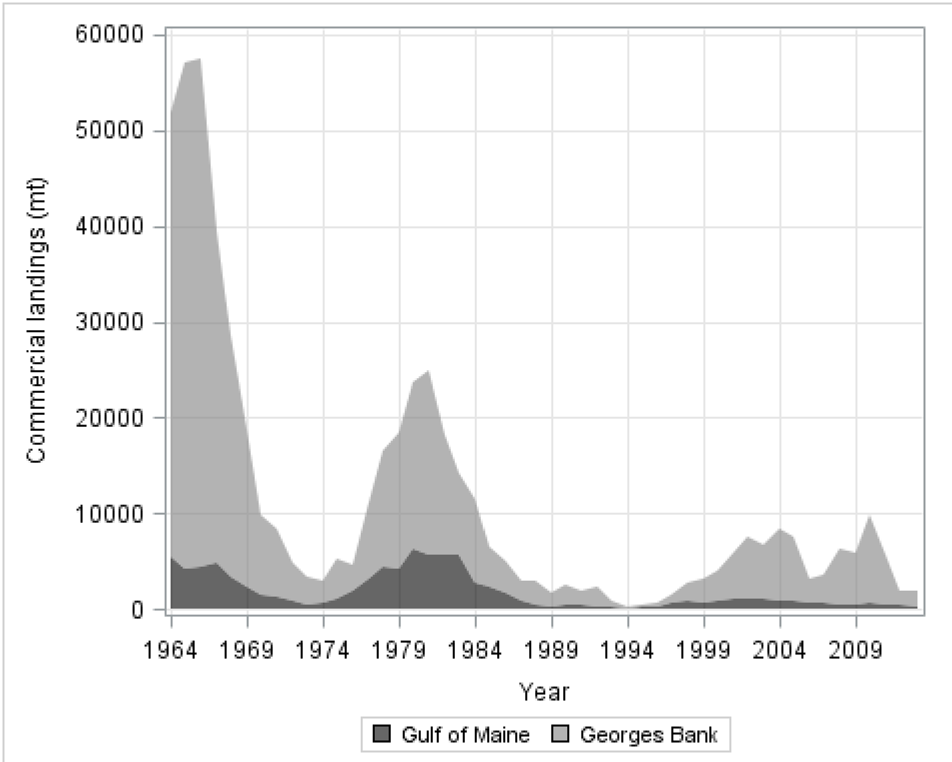


Figure A.16. Total United States commercial landings of Gulf of Maine and Georges Bank haddock from 1964 to 2013.

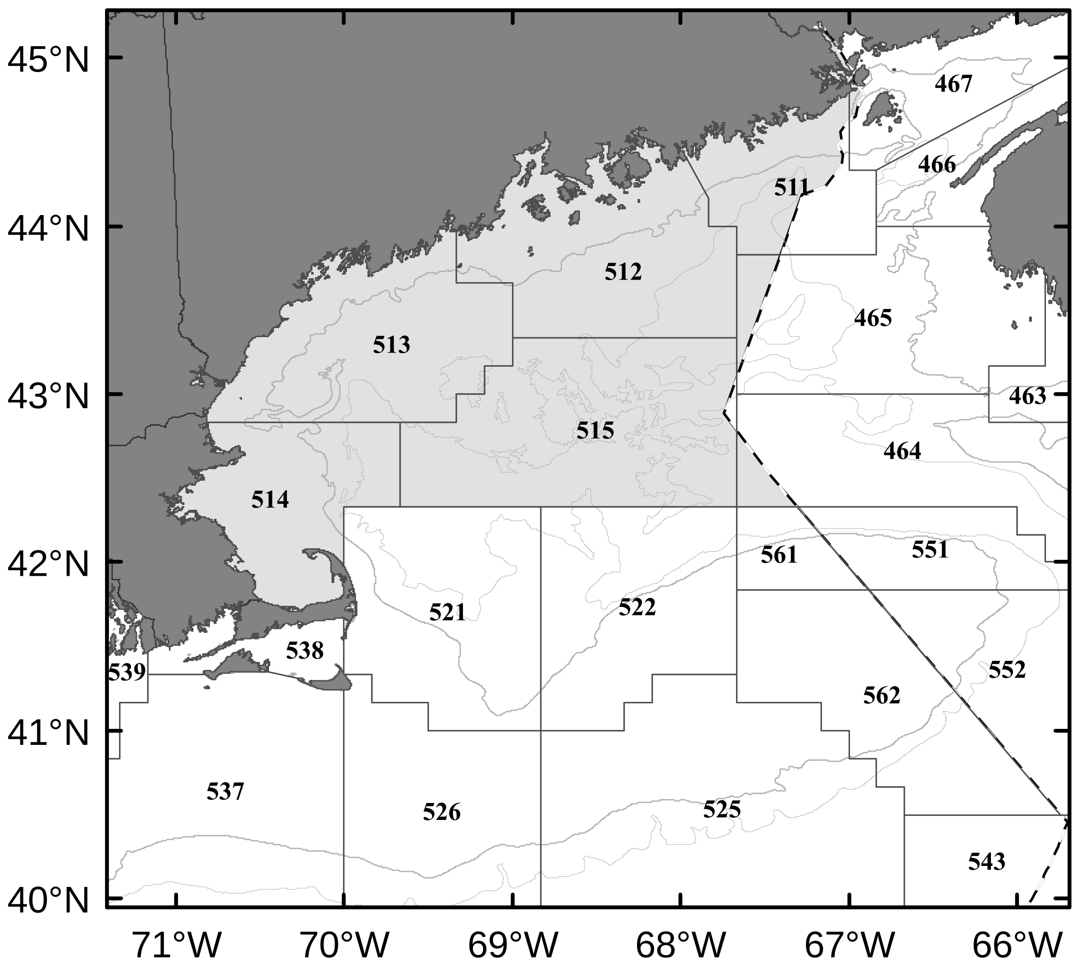


Figure A.17. Map of the Gulf of Maine haddock management area (shaded grey). The United States exclusive economic zone (EEZ) is indicated by the dashed line.

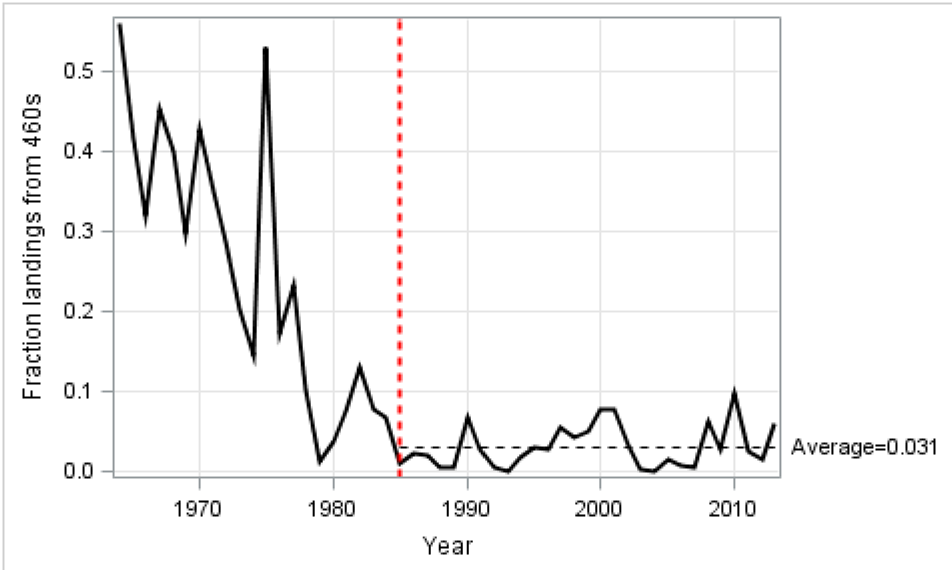


Figure A.18. Percentage of total commercial landings of Gulf of Maine haddock from statistical areas 464, 465 and 467 between 1964 and 2013. The Hague Line, which formally defined the Exclusive Econonimic Zones of the United States and Canada was adopted on October 12, 1984 (dashed red line).

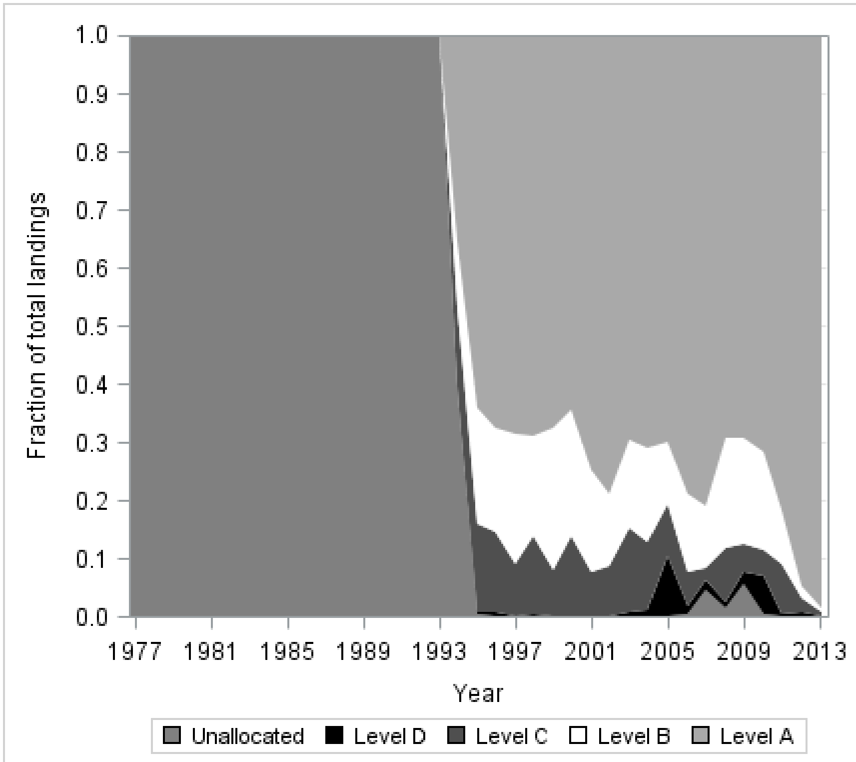


Figure A.19. Fraction of the Gulf of Maine haddock commercial landings by allocation level between 1977 and 2013. Prior to 1994 landings were allocated based on a port interview process. From 1994 onward landings were allocated to statistical area and gear type based on a standardized allocation scheme described in Wigley et al. (2008).

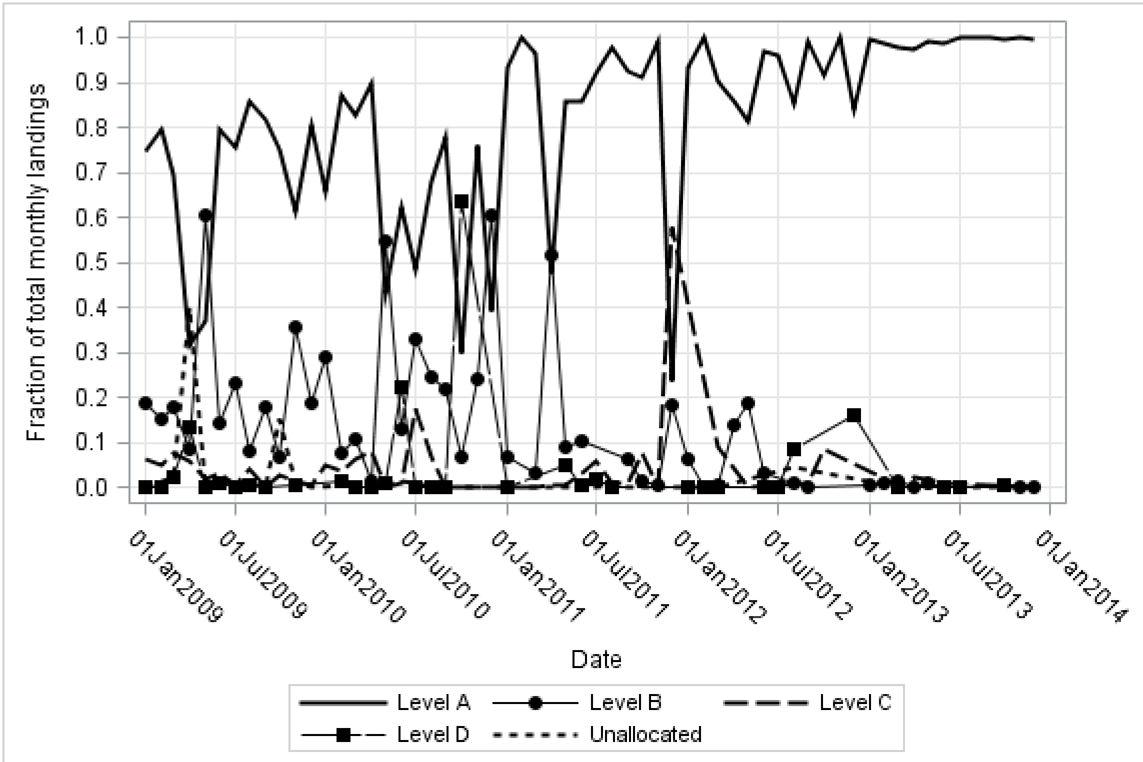


Figure A.20. Fraction of the Gulf of Maine haddock commercial landings by allocation level between 2009 and 2013 by month.

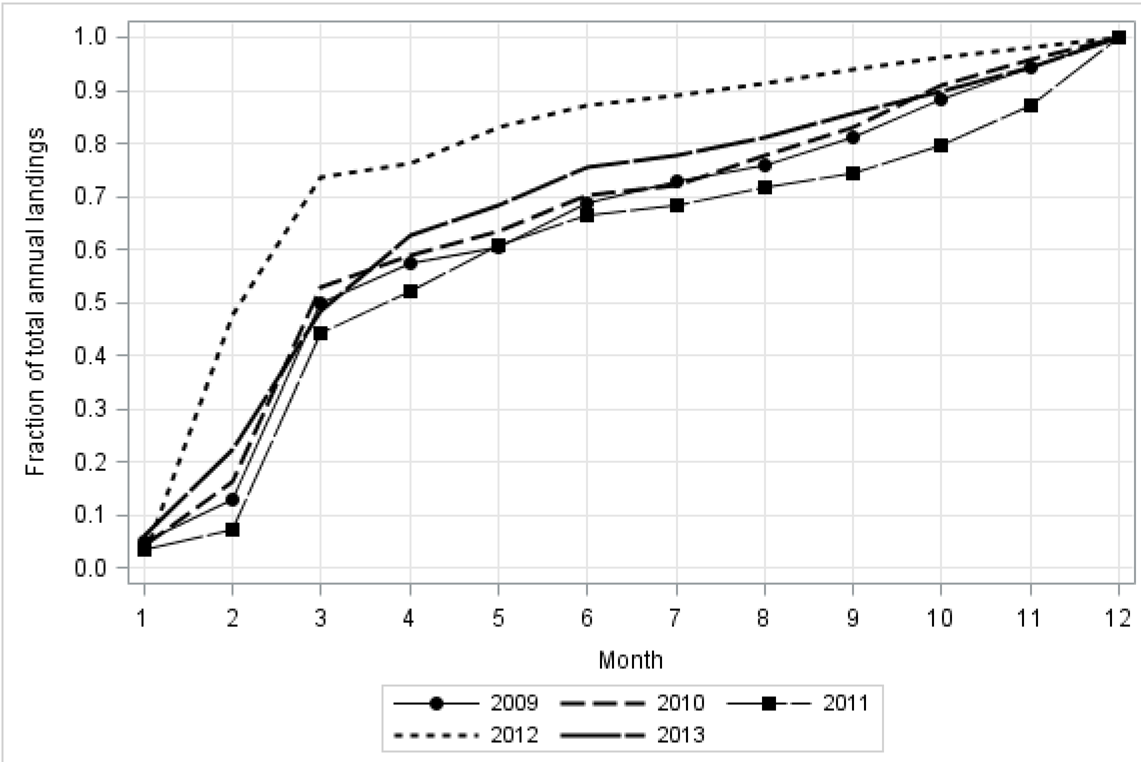


Figure A.21. Cumulative monthly commercial landings of Gulf of Maine haddock by year from 2009 to 2013.

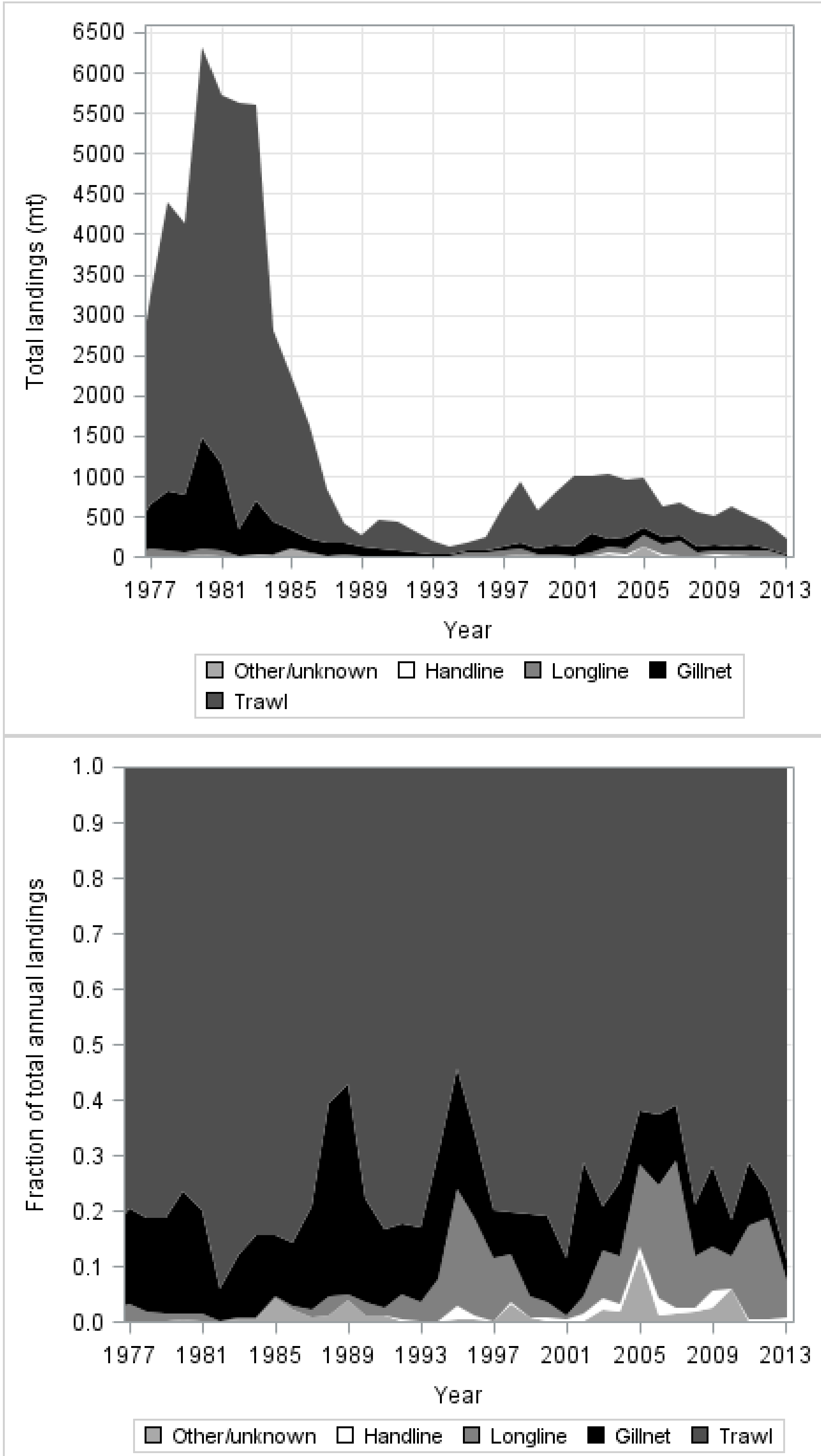


Figure A.22. Total (top) and fractional (as a fraction of the total, bottom) commercial landings of Gulf of Maine haddock by gear from 1977 to 2013.

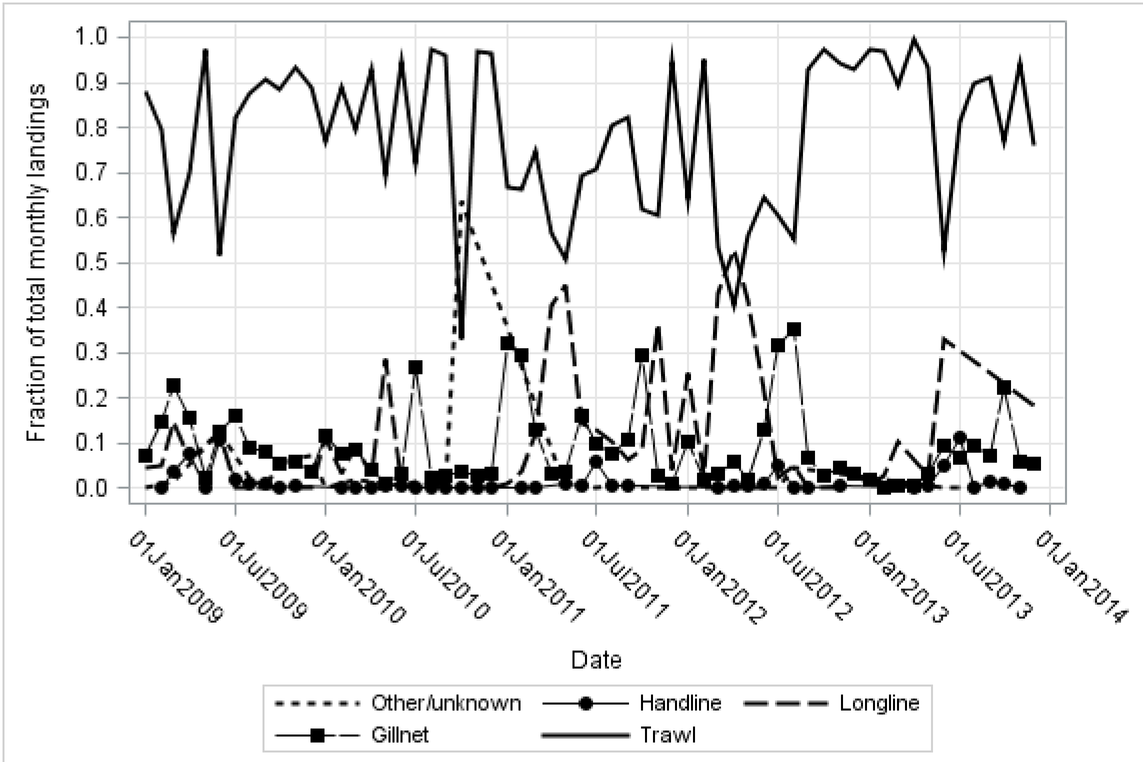


Figure A.23. Monthly commercial landings patterns (as a fraction of the total landings) of Gulf of Maine haddock by gear from 2009 to 2013.

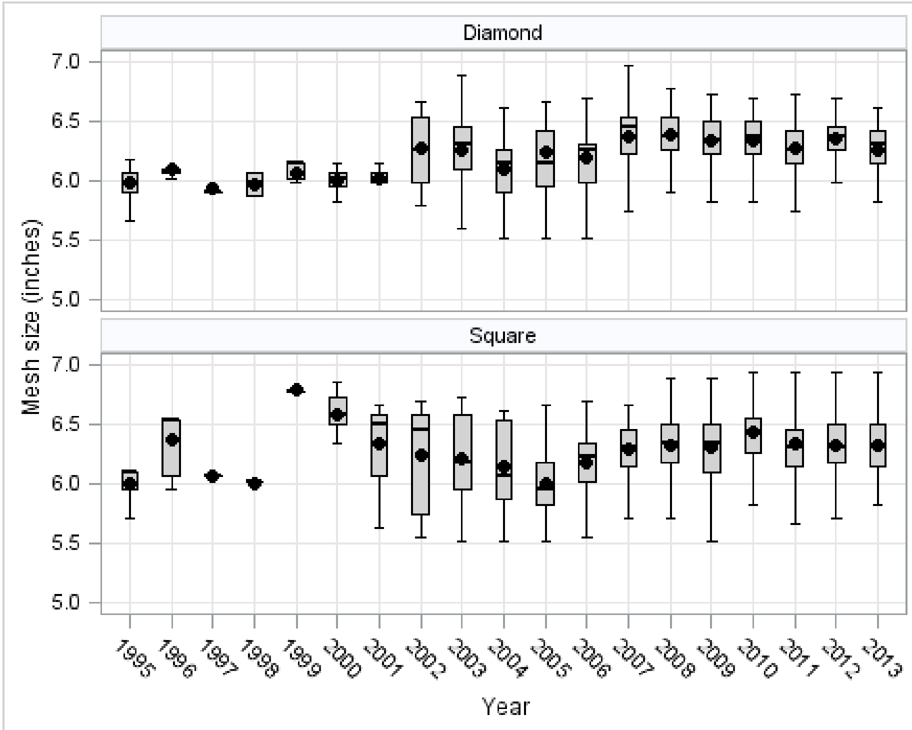


Figure A.24. Distribution of average mesh size for diamond and square hung mesh from observed otter trawl hauls in the Gulf of Maine from 1995 to 2013.

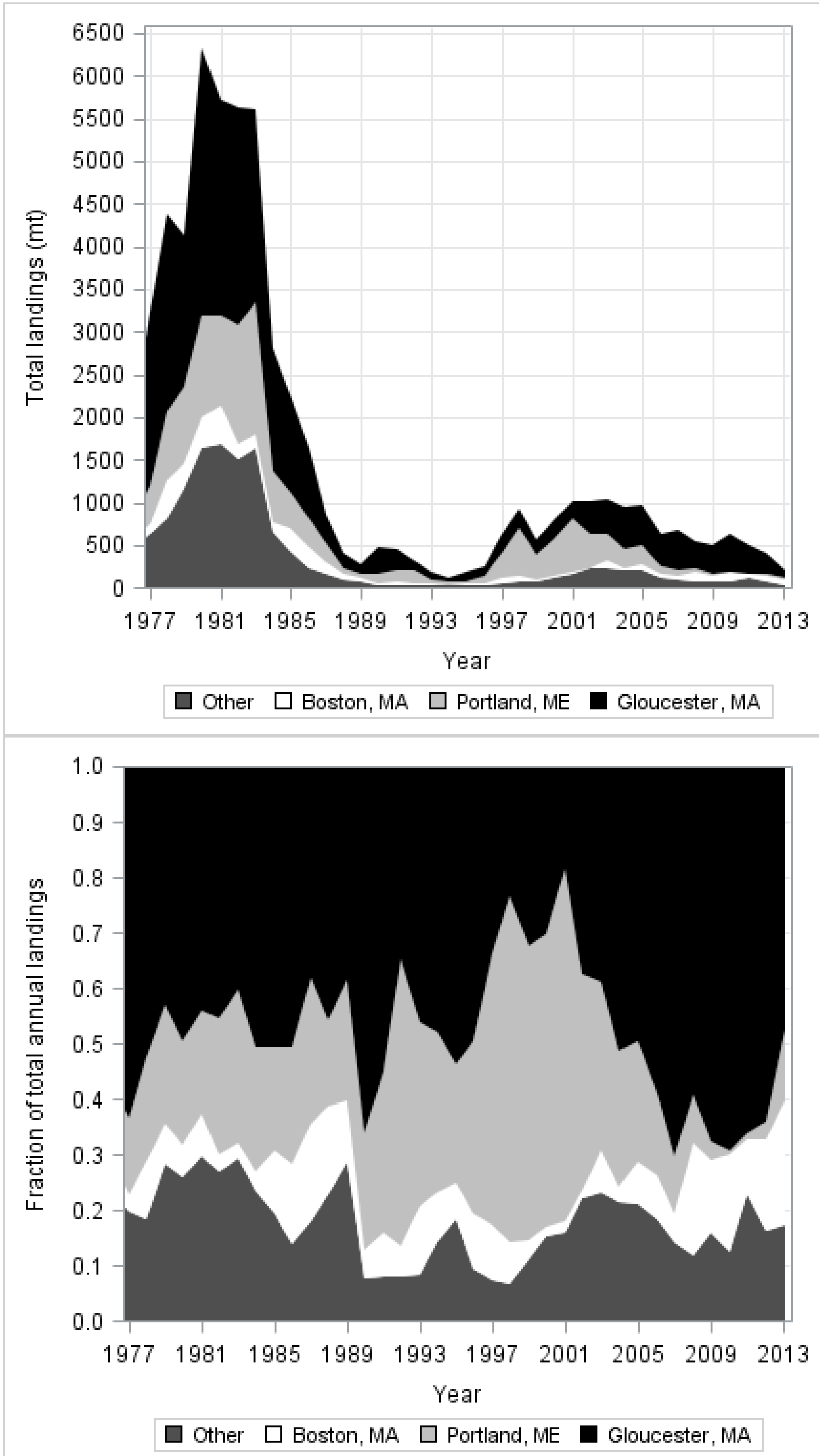


Figure A.25. Total (top) and fractional (as a fraction of the total, bottom) commercial landings of Gulf of Maine haddock by port from 1977 to 2013.

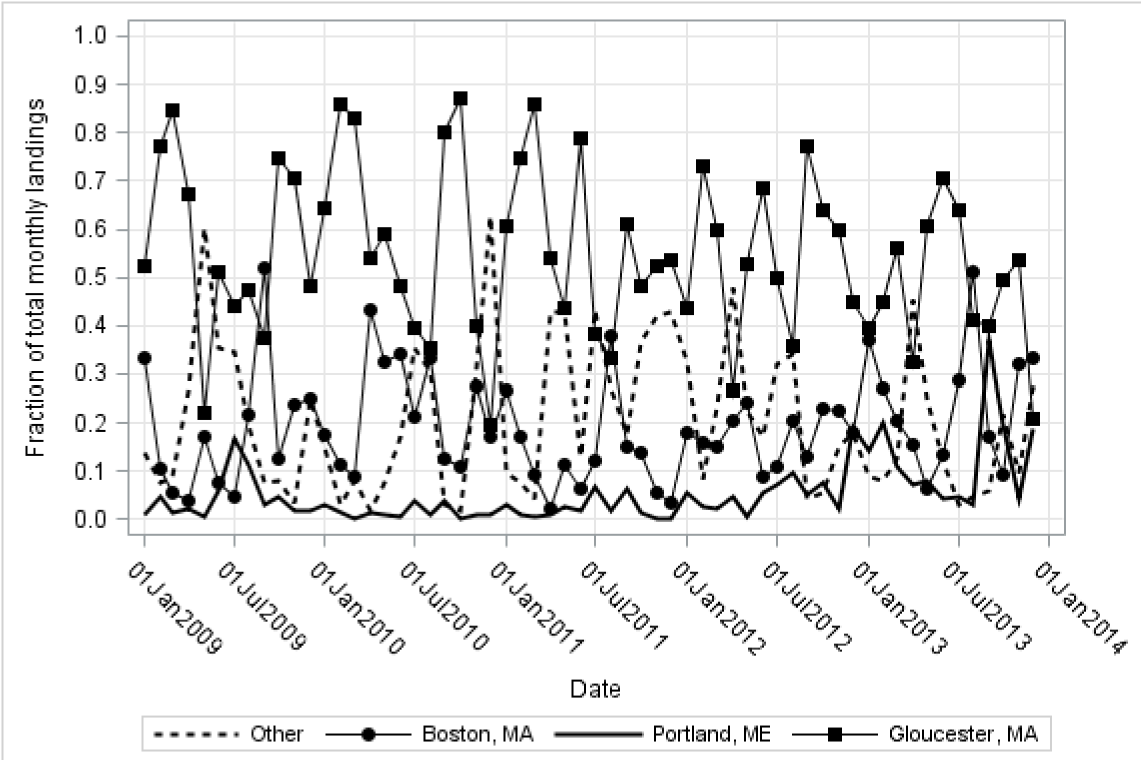


Figure A.26. Monthly commercial landings patterns (as a fraction of the total landings) of Gulf of Maine haddock by port from 2009 to 2013.

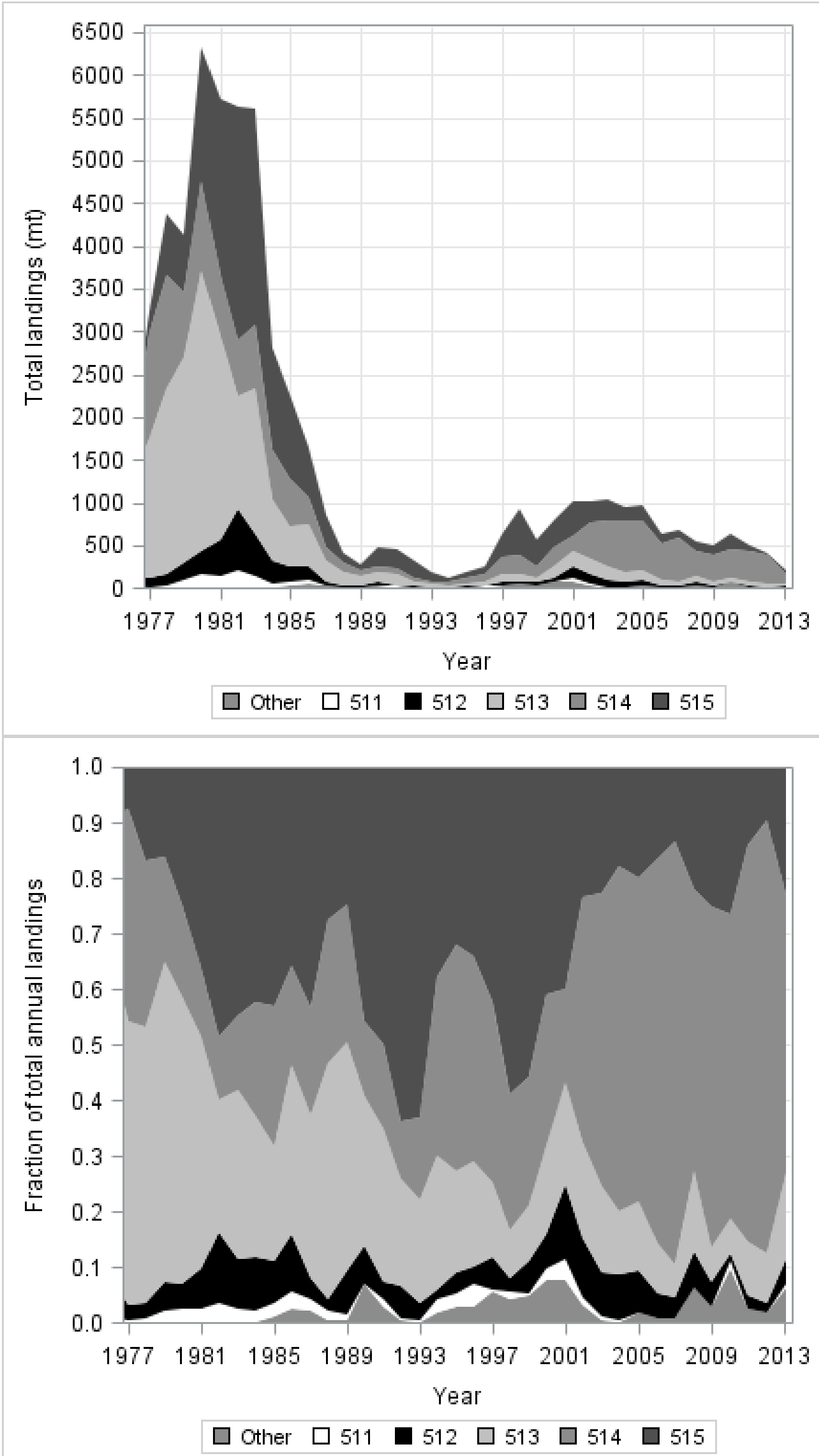


Figure A.27. Total (top) and fractional (as a fraction of the total, bottom) commercial landings of Gulf of Maine haddock by statistical area from 1977 to 2013.

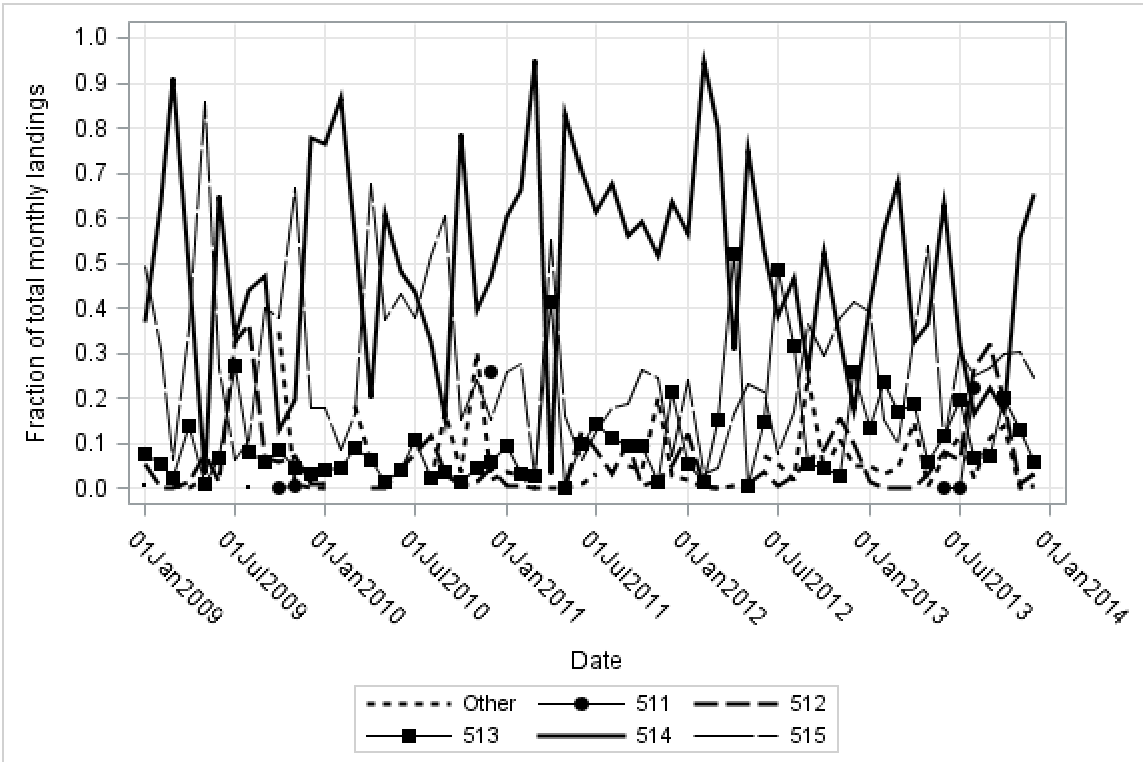


Figure A.28. Monthly commercial landings patterns (as a fraction of the total landings) of Gulf of Maine haddock by statistical area from 2009 to 2013.

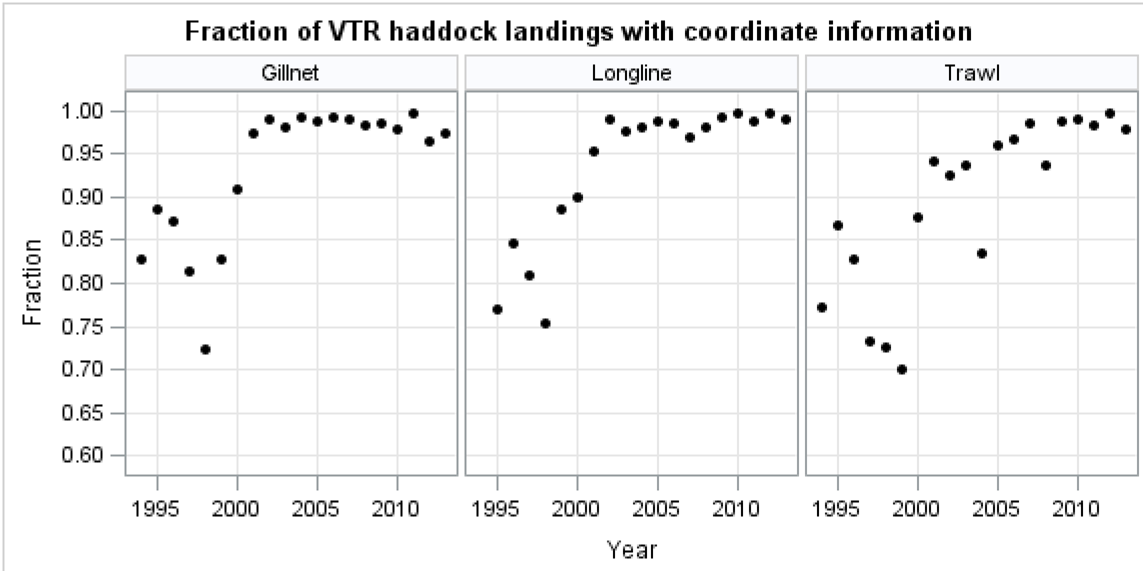


Figure A.29. Fraction of total Gulf of Maine haddock commercial landings reported on vessel trip reports (VTR) with latitude and longitude coordinate information, by gear type from 1994 to 2013.

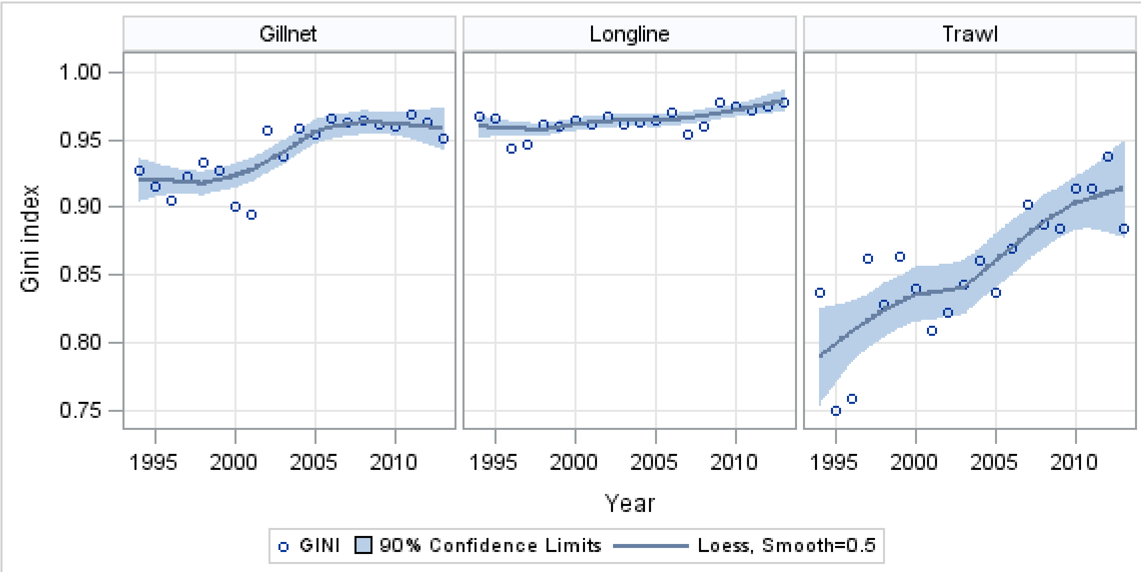


Figure A.30. Gini indices for the Gulf of Maine haddock commercial landings of the sink gillnet, longline and otter trawl fleets from 1994-2013. Indices are based on the spatial distribution of the retained catch reported on vessel trip reports.

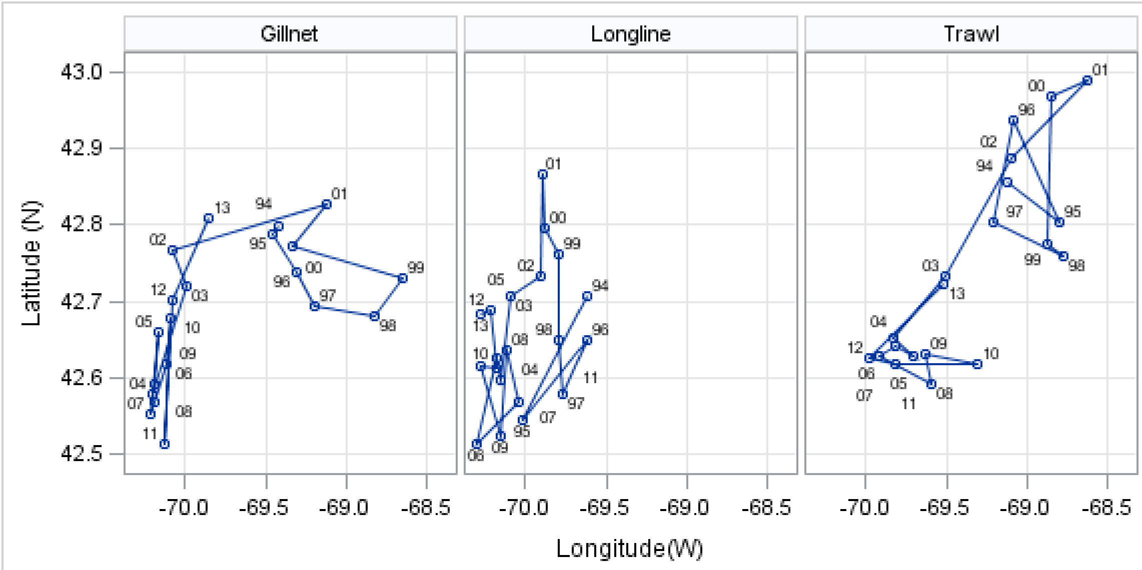


Figure A.31. Landings-weighted mean location (centroid) of Gulf of Maine haddock commercial landings of the sink gillnet, longline and otter trawl fleets from 1994-2013. Centroids are based on the spatial distribution of the retained catch reported on vessel trip reports.

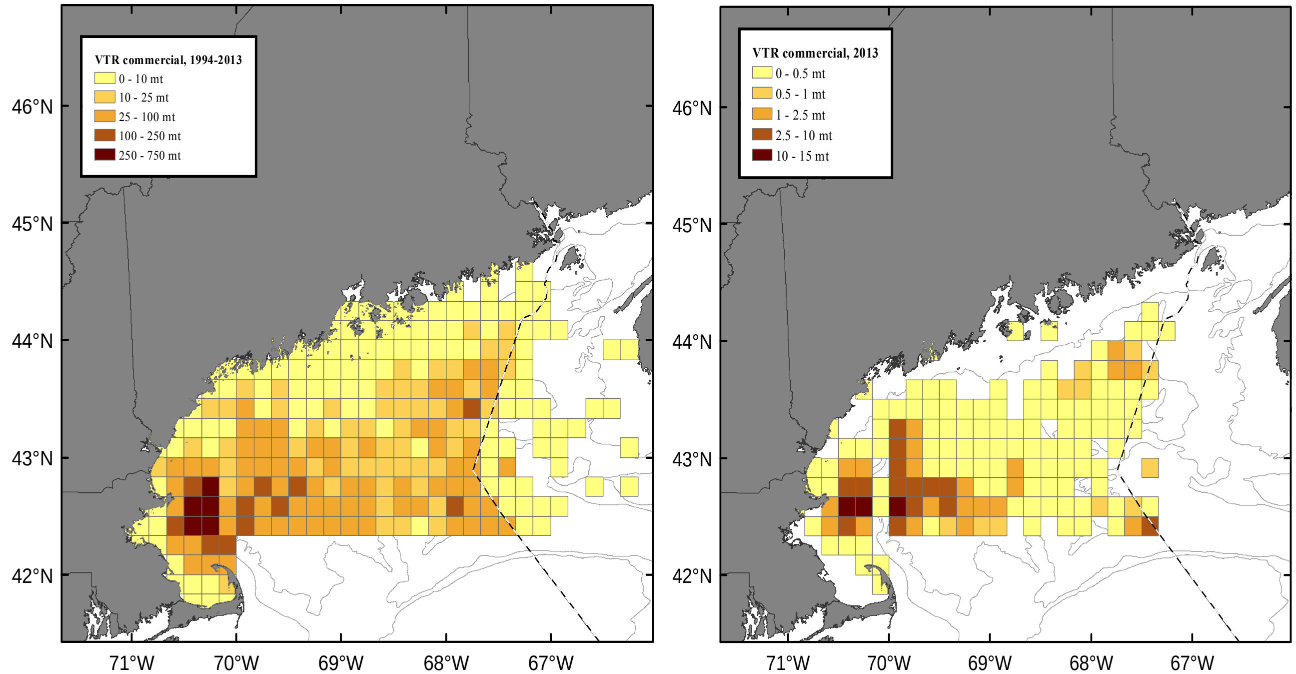


Figure A.32. Comparison of the distribution of Gulf of Maine haddock commercial landings per ten minute square in 2013 (right) to the aggregate distribution from 1994 to 2013 (right).

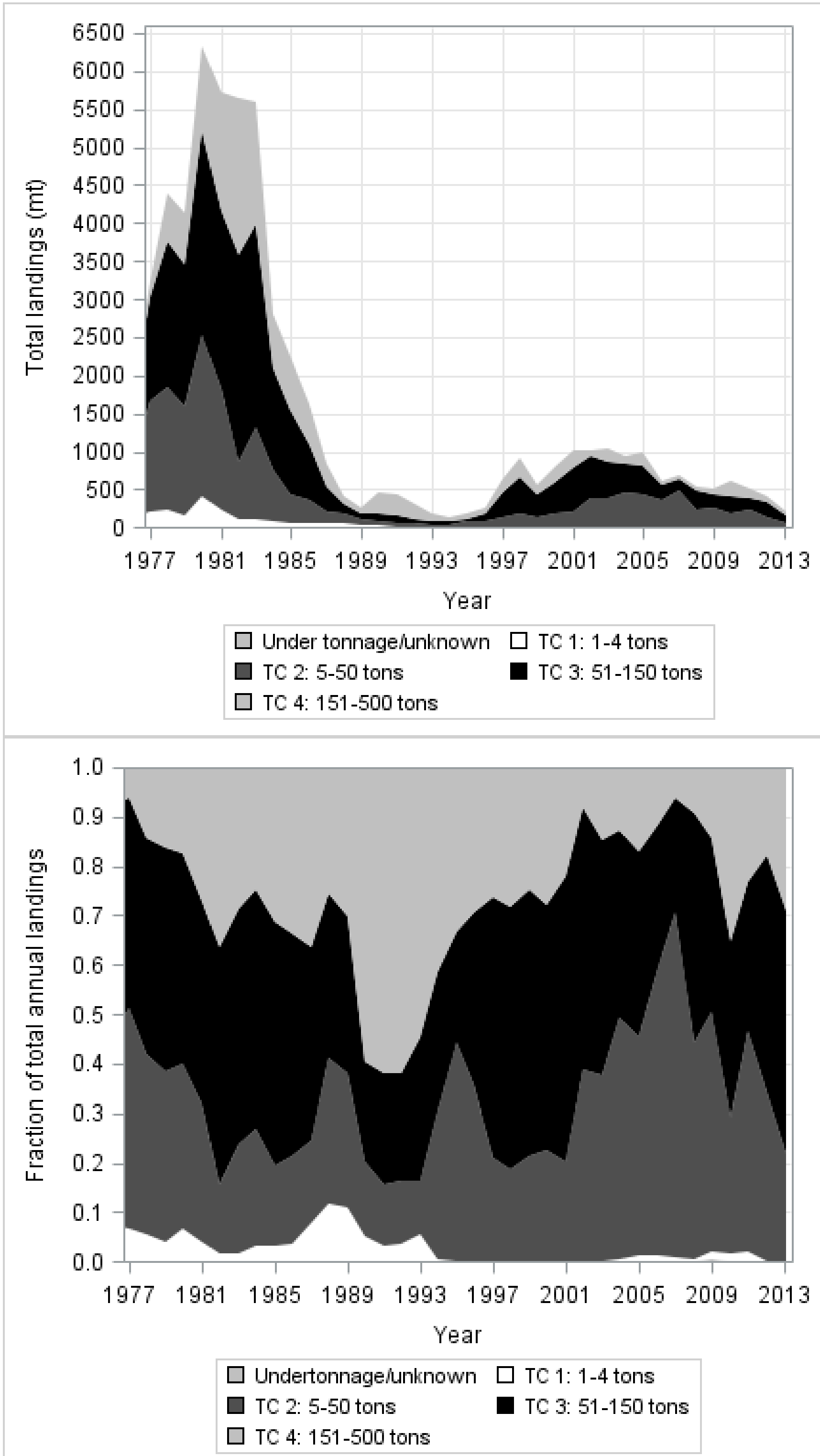


Figure A.33. Total (top) and fractional (as a fraction of the total, bottom) commercial landings of Gulf of Maine haddock by vessel ton class from 1977 to 2013.

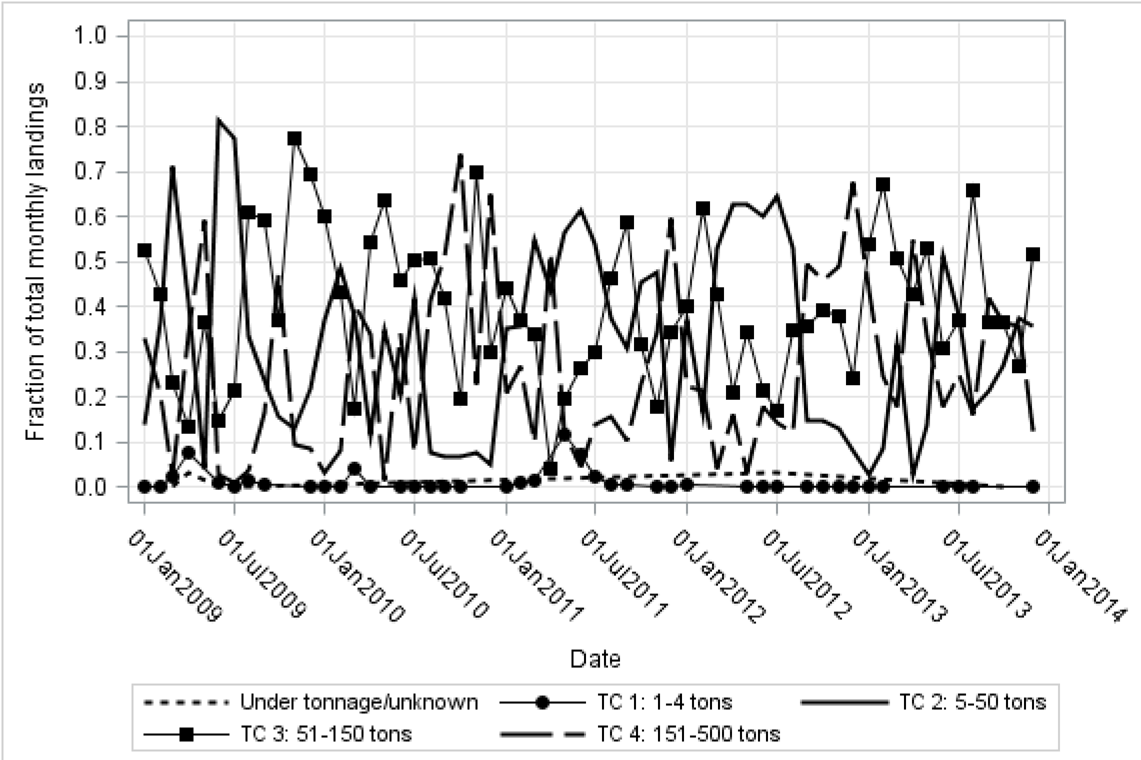


Figure A.34. Monthly commercial landing patterns (as a fraction of the total landings) of Gulf of Maine haddock by ton class from 2009 to 2013.

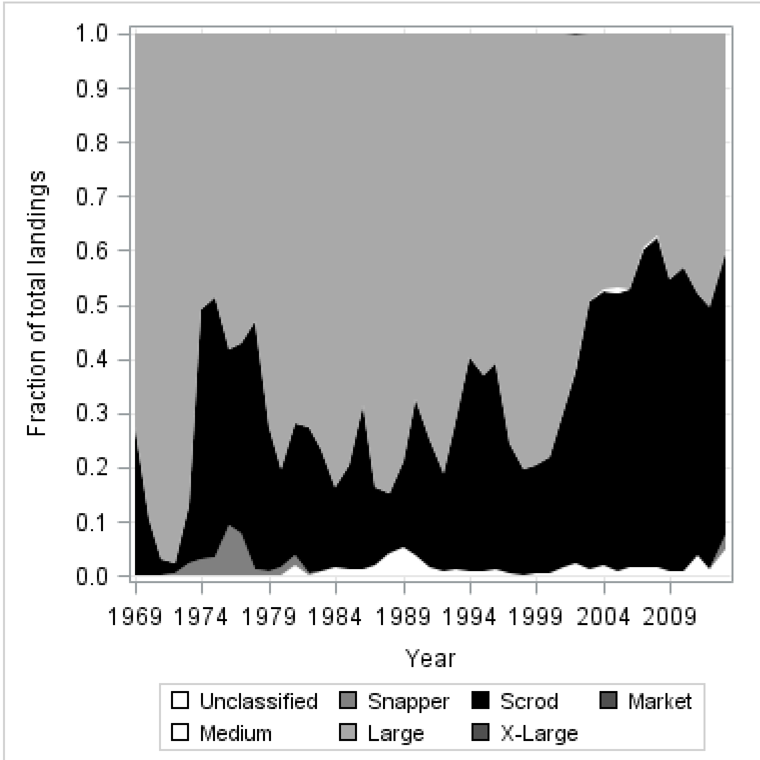


Figure A.35. Fraction of commercial landings of Gulf of Maine haddock by market category from 1969 to 2013.

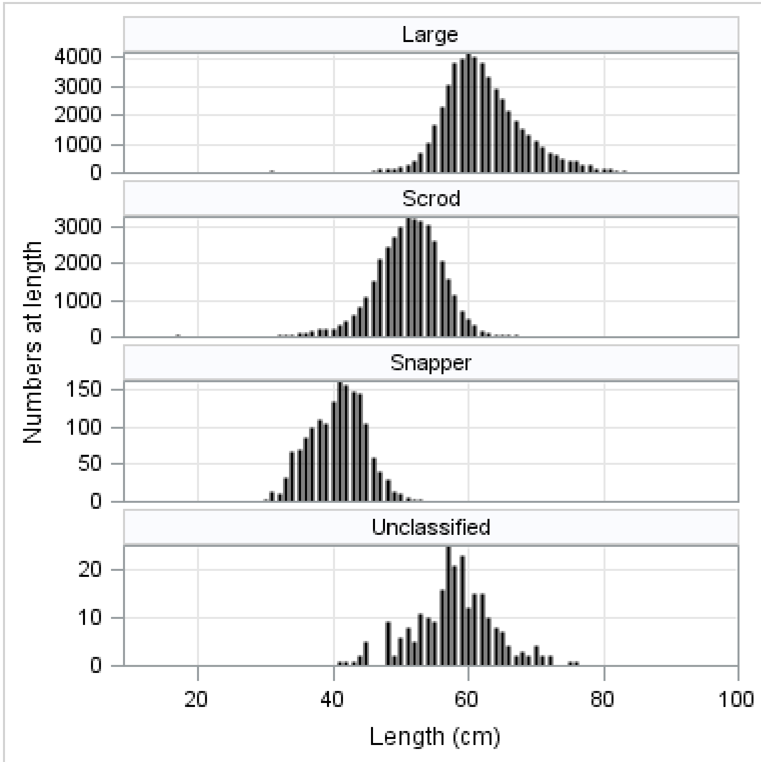


Figure A.36. Length frequency distributions of all reported market categories of Gulf of Maine haddock. Length frequency information has been binned across all years, 1969-2013. *Note that the scales of the y-axis vary by market category.*

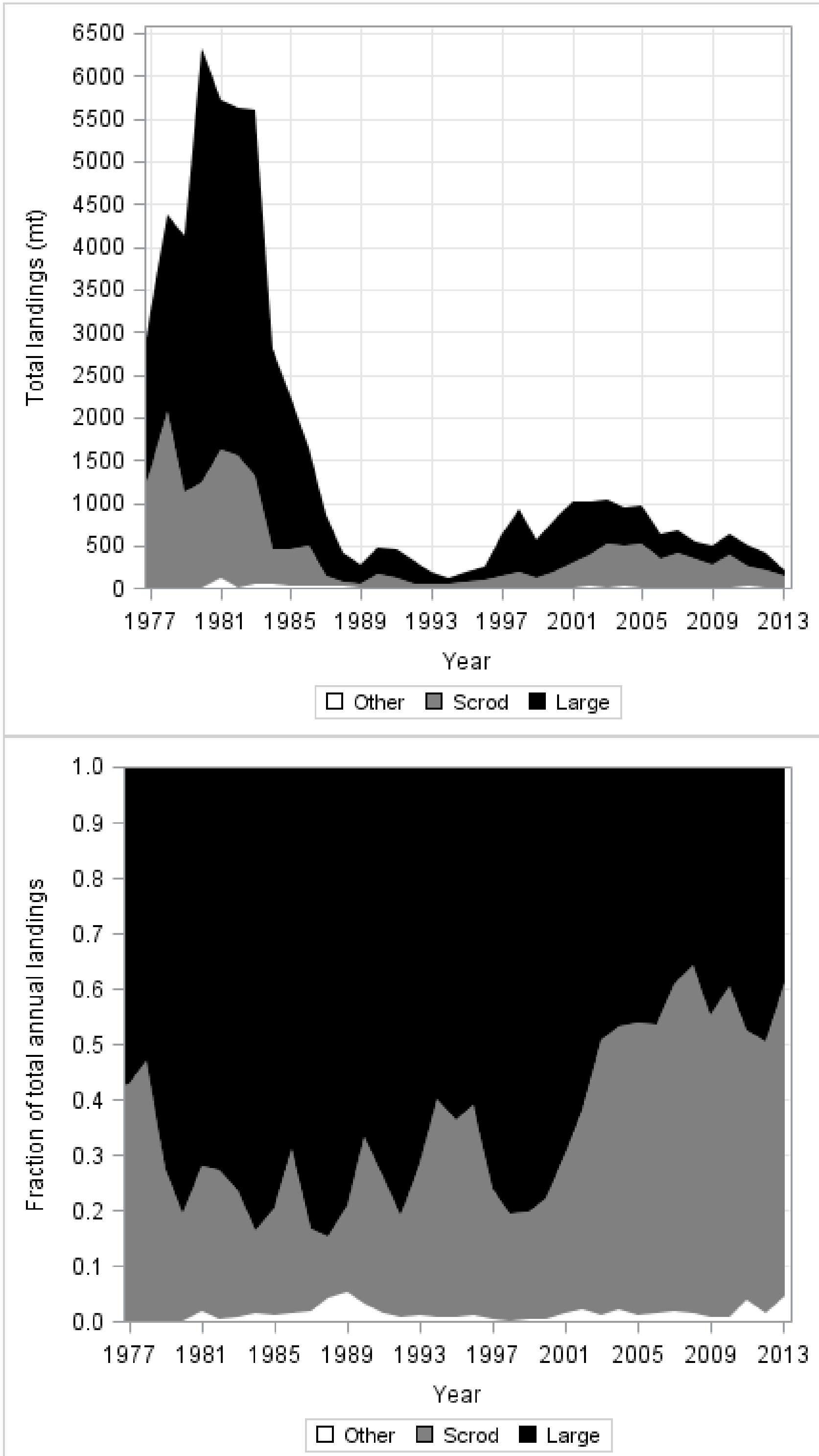


Figure A.37. Total (top) and fractional (as a fraction of the total, bottom) commercial landings of Gulf of Maine haddock by market category from 1977 to 2013. *Note that the snapper and extra-large market categories have been combined with the scrod and large market categories, respectively.*

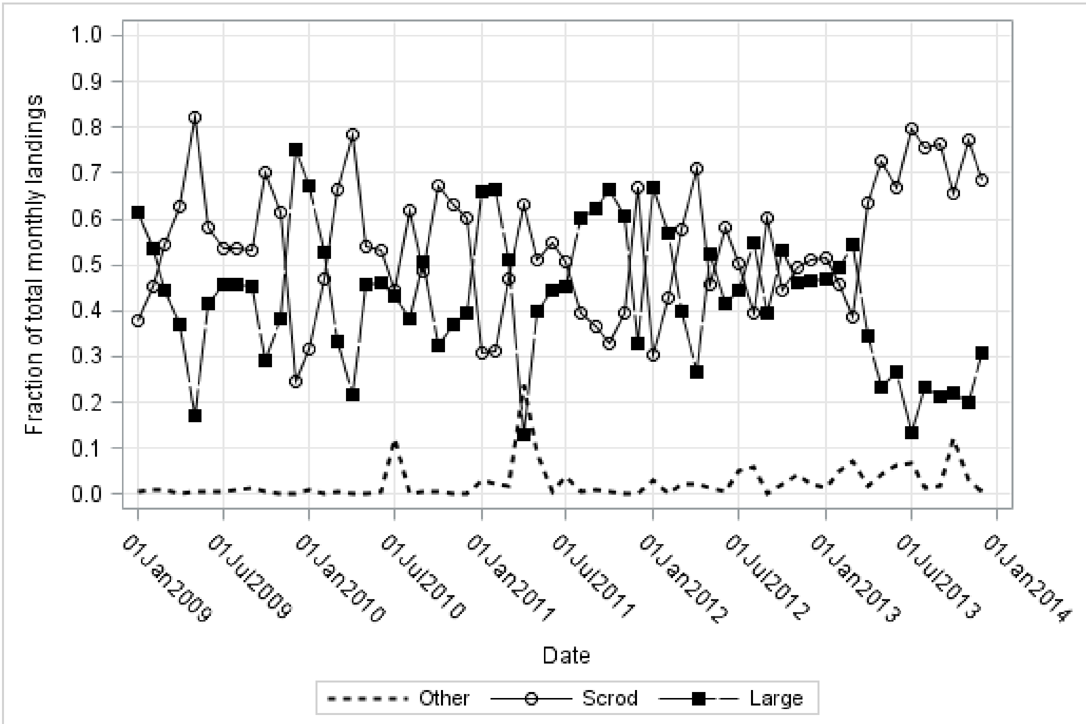


Figure A.38. Monthly commercial landing patterns (as a fraction of the total landings) of Gulf of Maine haddock by market category from 2009 to 2013. *Note that the snapper and extra-large market categories have been combined with the scrod and large market categories, respectively.*

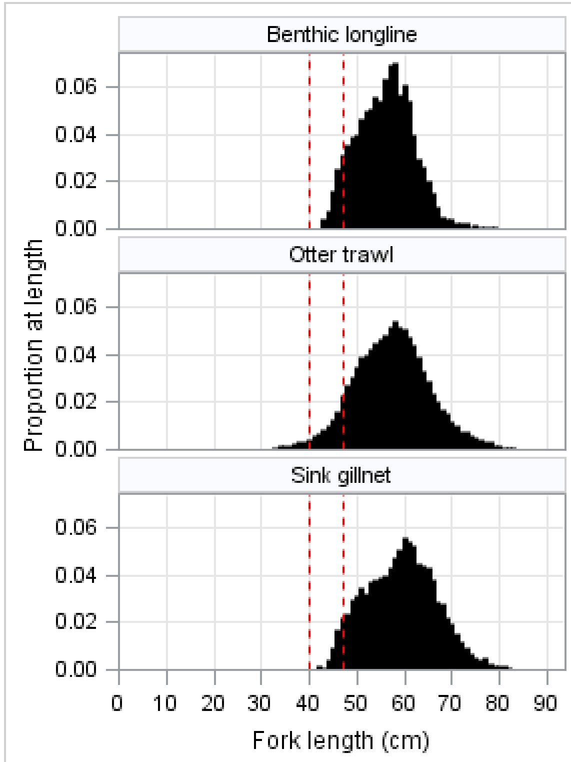


Figure A.39. Gear-specific frequency distributions of landed Gulf of Maine haddock lengths collected by port samplers between 1989 and 2013. The range of commercial minimum retention sizes over the time period is indicated by the dashed red lines.

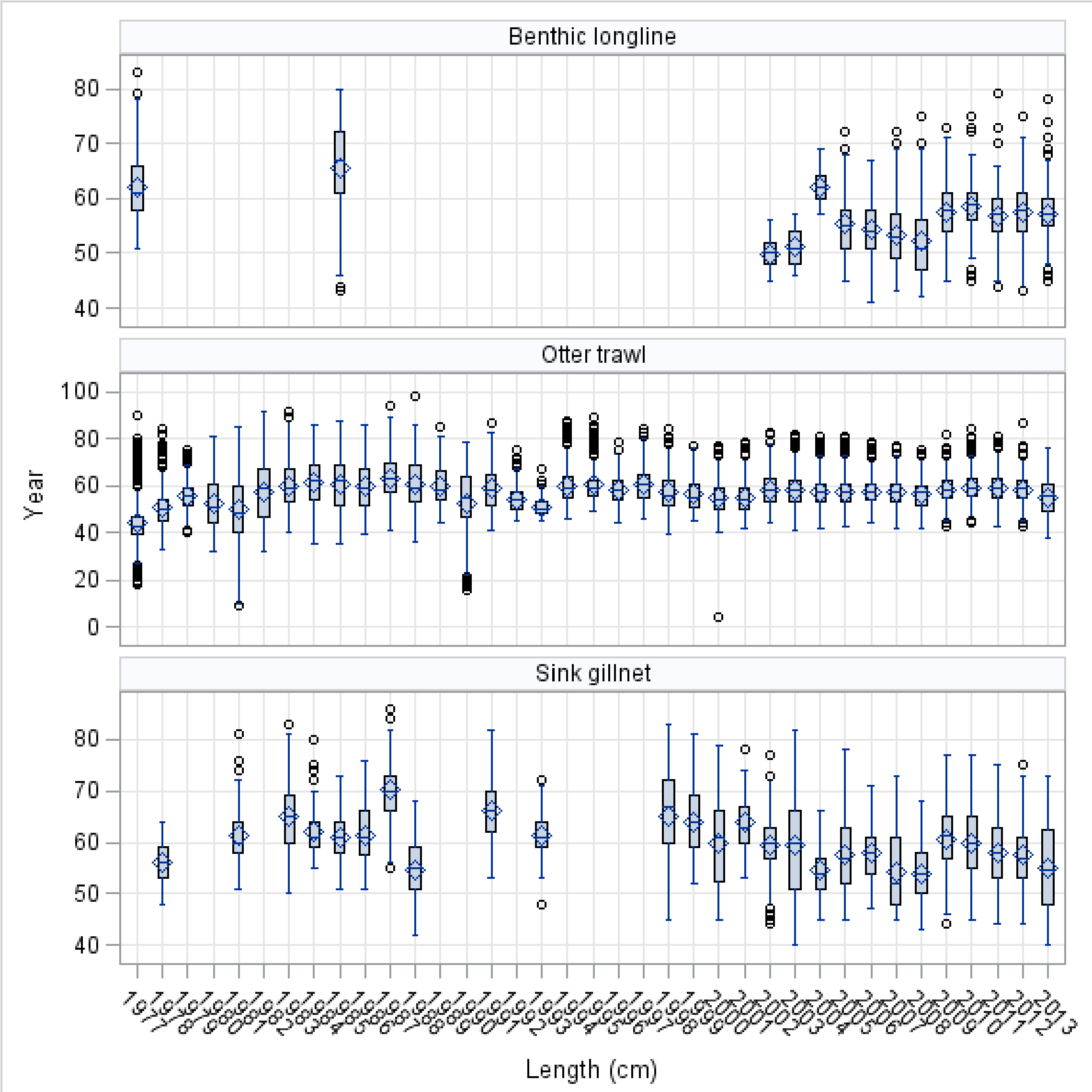


Figure A.40. Box plots showing the length distribution of Gulf of Maine haddock landed by the commercial fishery, by gear type, between 1989 and 2013. Missing years indicate that there were no sampled landings for that gear/year combination.

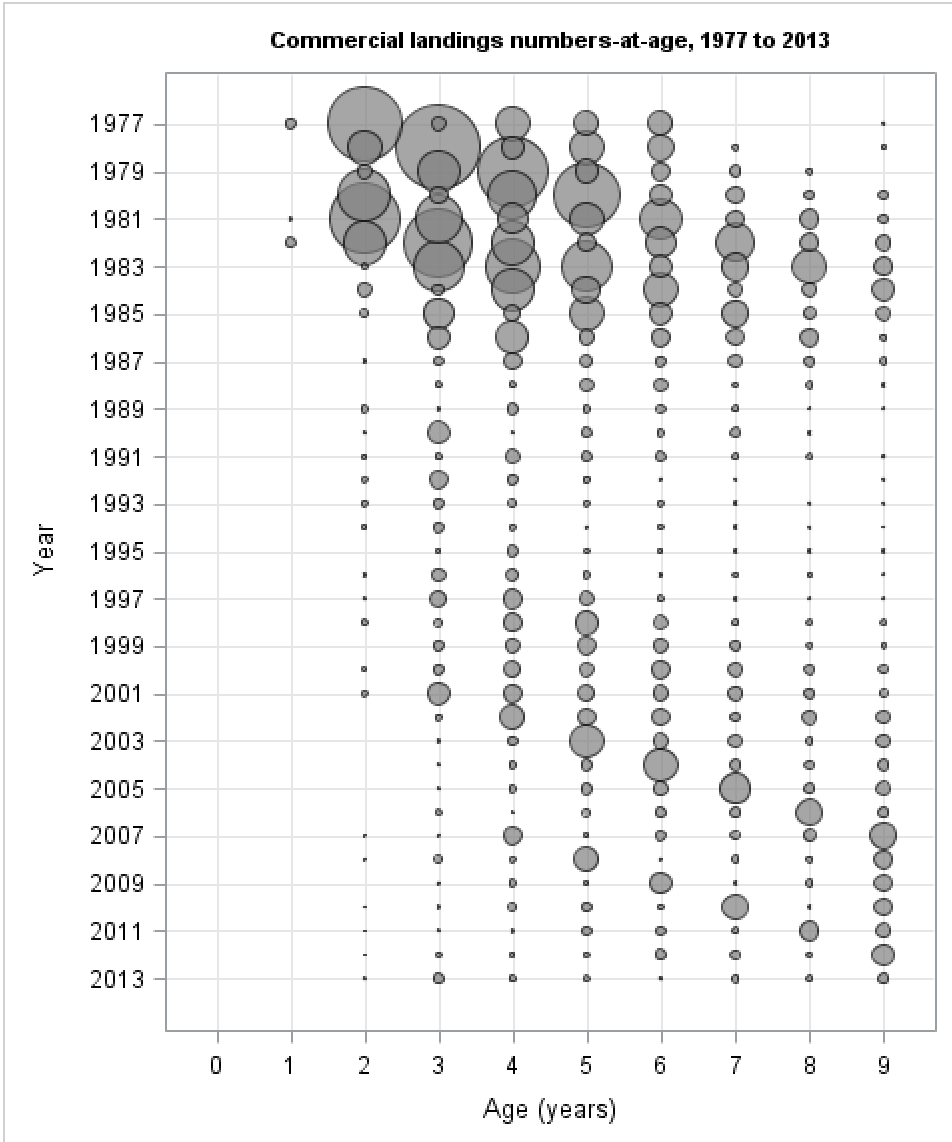


Figure A.41. Commercial landings-at-age of Gulf of Maine haddock from 1977 to 2013.

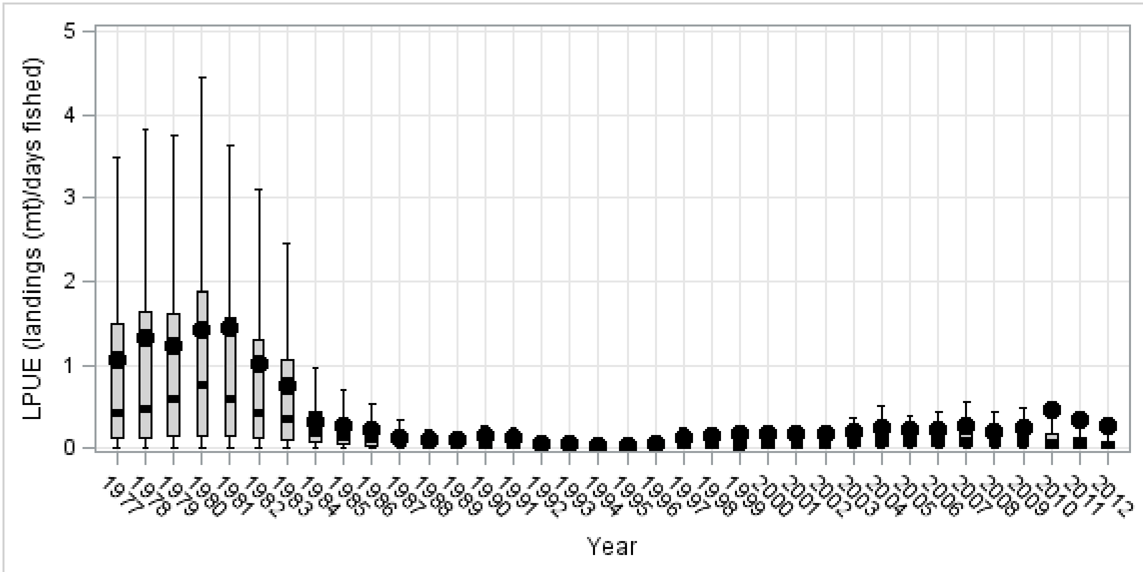


Figure A.42. Box-plot distributions of nominal Gulf of Maine haddock landings (mt) per days fished by the commercial trawl fishery from 1977 to 2012. Commercial dealer and vessel trip report data were used in this analysis and only includes trips that reported landing Gulf of Maine haddock. *Note that commercial dealer data were only available through 2012 at the time of the LPUE analysis.*

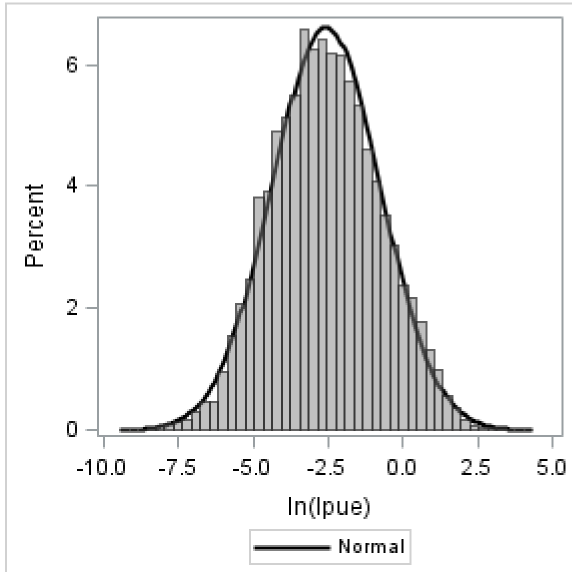


Figure A.43. Distribution of log transformed nominal commercial trawl LPUE.

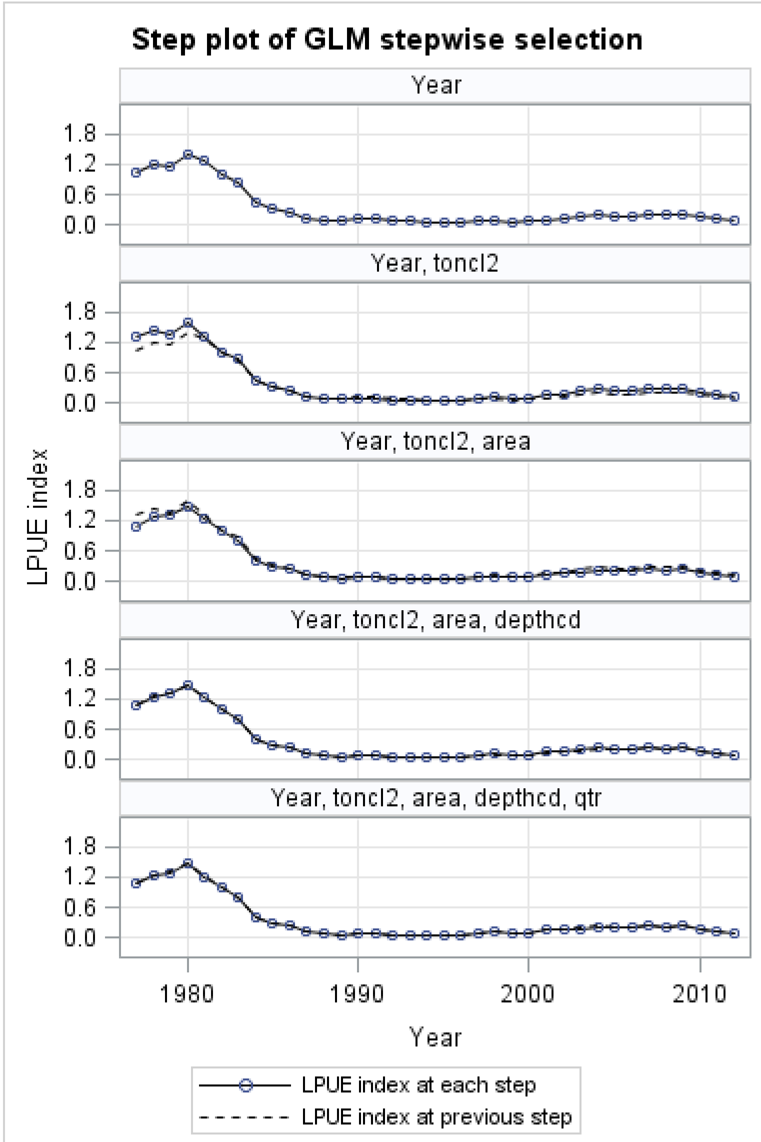


Figure A.44. Step plot of the commercial trawl landings-per-unit-effort (LPUE) standardization model. Each panel shows the standardized abundance index as explanatory variables are added to the model through stepwise selection. The index from previous step is indicated with a dashed line.

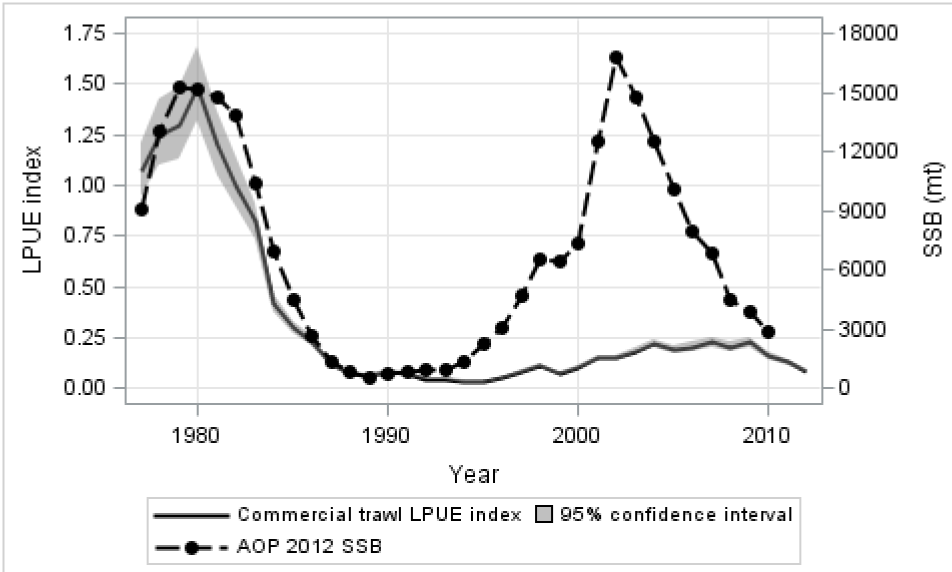


Figure A.45. Time series of the commercial trawl LPUE index overlaid on the spawning stock biomass (SSB) estimate from the AOP 2012 assessment from 1977 to 2012. *Note that the AOP 2012 SSB estimates only extends through 2010 and commercial data were only available through 2012 at the time of the LPUE analysis.*

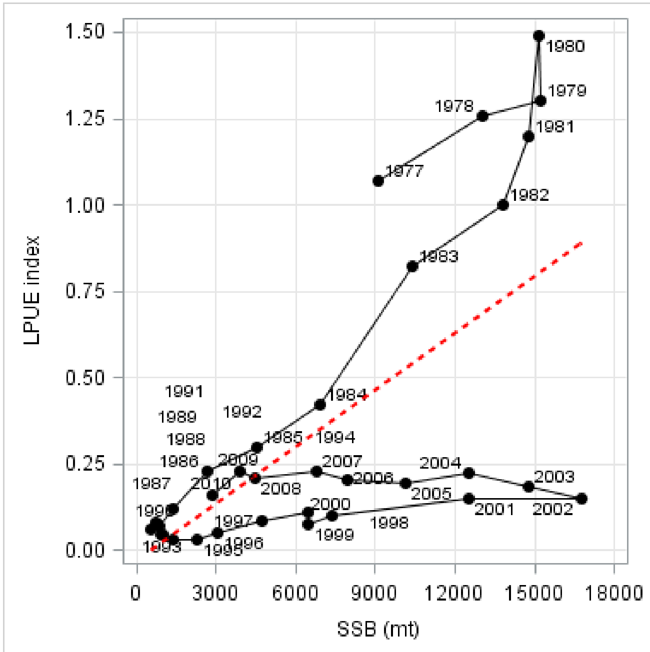


Figure A.46. Linear regression of the commercial trawl LPUE index on the spawning stock biomass (SSB) estimate from the AOP 2012 assessment.

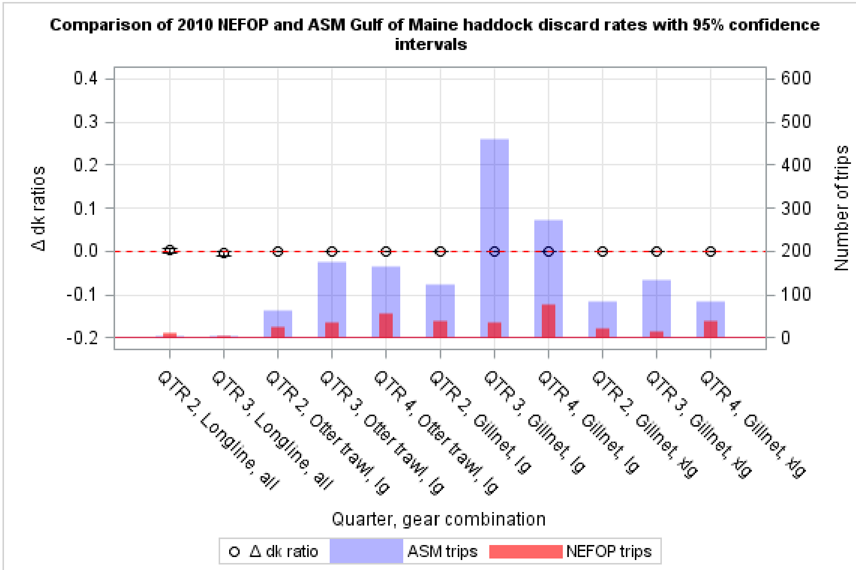


Figure A.47. Differences between the 2010 Gulf of Maine haddock discard rates estimated from data collected by groundfish at-sea monitors (ASMs) and certified observers showing 95% confidence intervals (dots) and the number of trips included in each analysis (bars) broken down by gear-mesh combination and quarter (adapted from Wigley et al. 2012).

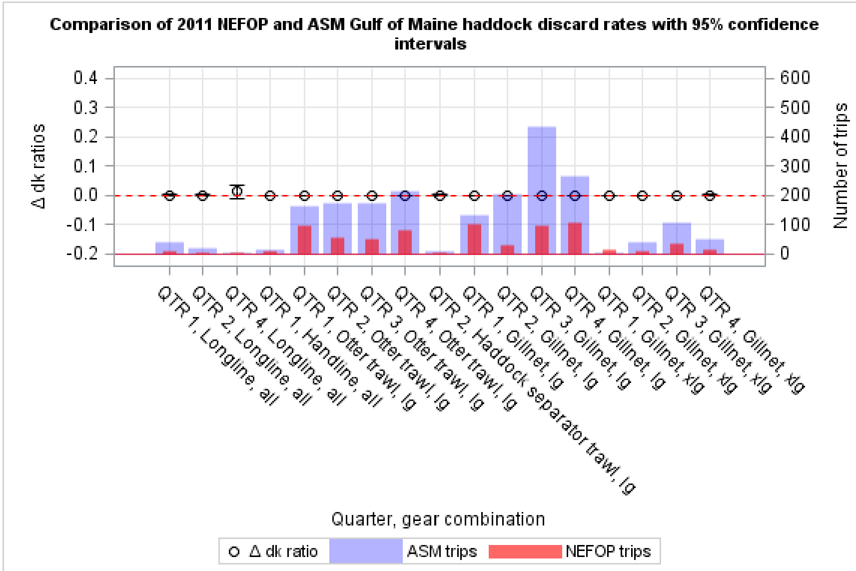


Figure A.48. Differences between the 2011 Gulf of Maine haddock discard rates estimated from data collected by groundfish at-sea monitors (ASMs) and certified observers showing 95% confidence intervals (dots) and the number of trips included in each analysis (bars) broken down by gear-mesh combination and quarter (adapted from Wigley et al. 2012).

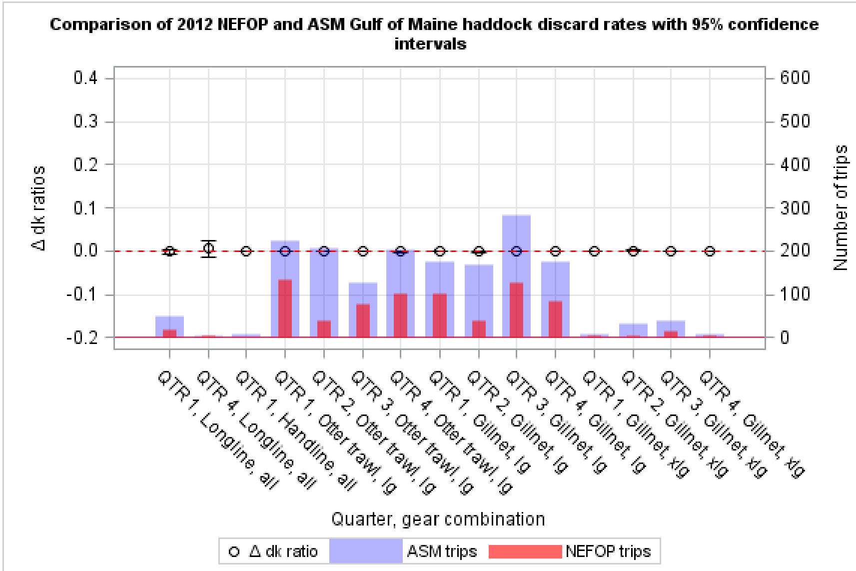


Figure A.49. Differences between the 2012 Gulf of Maine haddock discard rates estimated from data collected by groundfish at-sea monitors (ASMs) and certified observers showing 95% confidence intervals (dots) and the number of trips included in each analysis (bars) broken down by gear-mesh combination and quarter (adapted from Wigley et al. 2012).

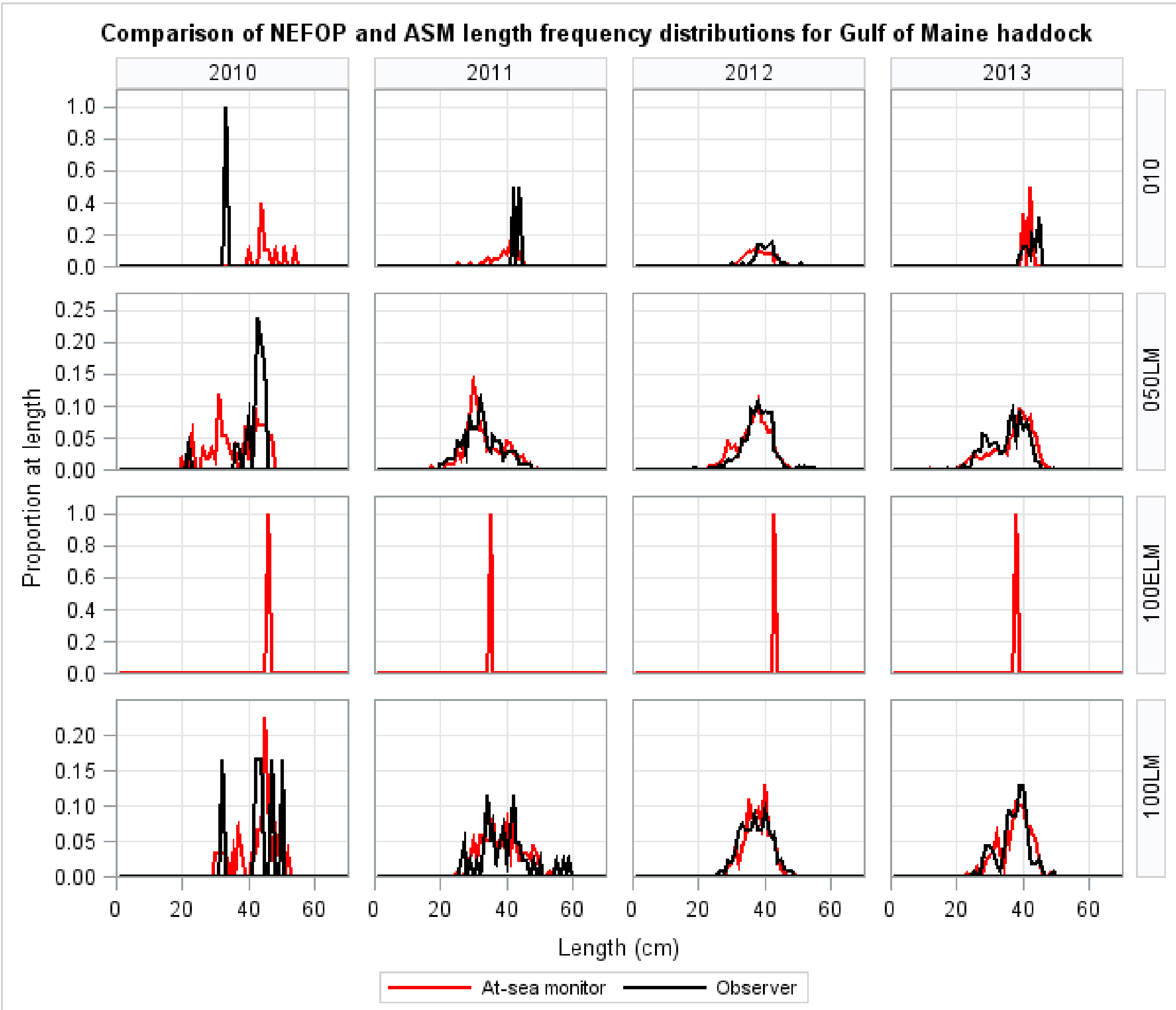


Figure A.50. Length frequency distributions of Gulf of Maine haddock commercials discards estimated from data collected by groundfish at-sea monitors (ASMs) and certified observers between 2010 and 2013. The gear codes displayed on the right hand axis are: longline (010), large mesh otter trawl (050LM), extra-large mesh sink gillnet (100ELM), large-mesh sink gillnet (100LM).

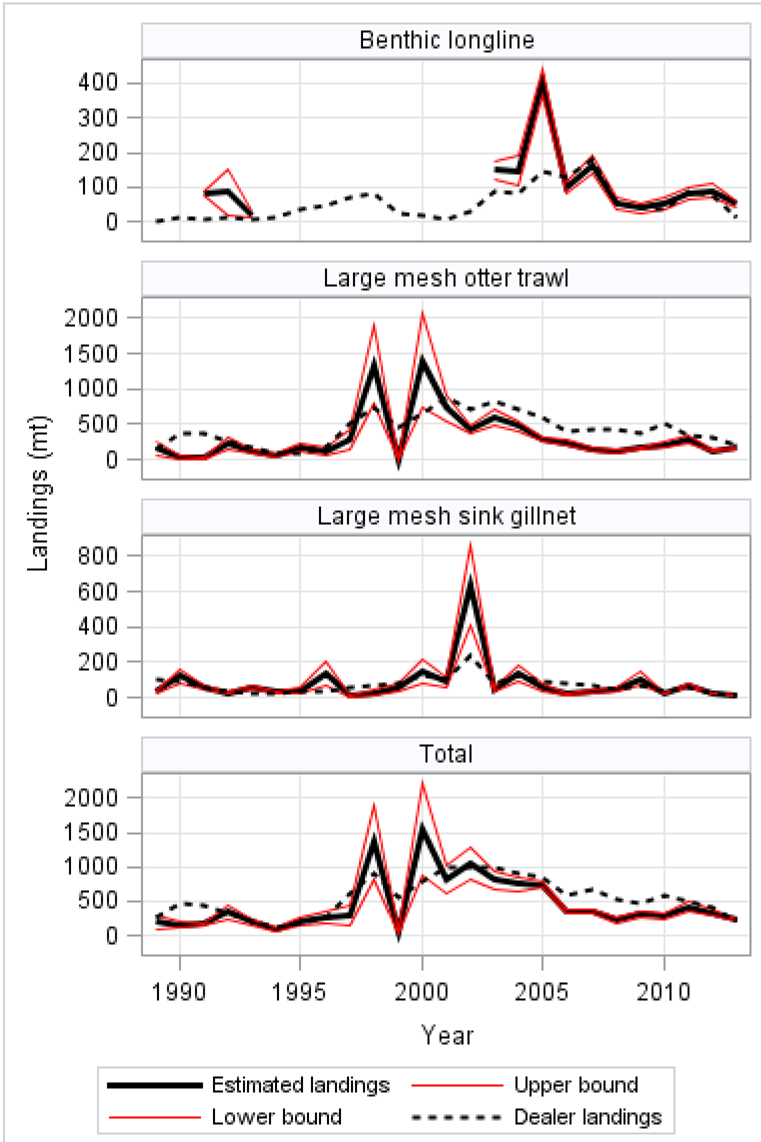


Figure A.51. Comparison of Gulf of Maine haddock landings estimates generated using the Standardized Bycatch Reporting Methodology (SBRM, Wigley et al. 2007) combined ratio approach to the stock landings from the Commercial Fisheries Database AA tables. Landings are shown only for longline, gillnet and otter trawl gears. The comparison provides a cross validation of both the discard estimation and landings allocation procedure.

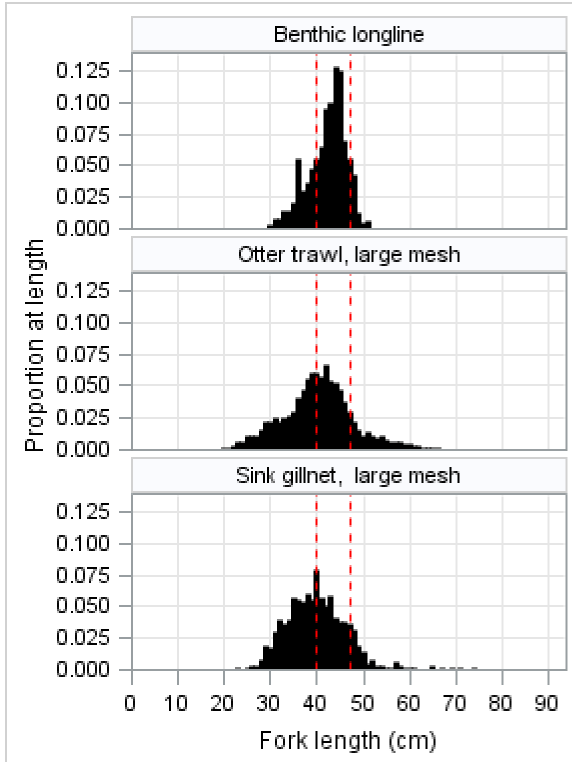


Figure A.52. Gear-specific frequency distribution of discarded Gulf of Maine haddock lengths collected by at-sea observers between 1989 and 2013. The range of commercial minimum retention sizes over the time period is indicated by the dashed red lines.

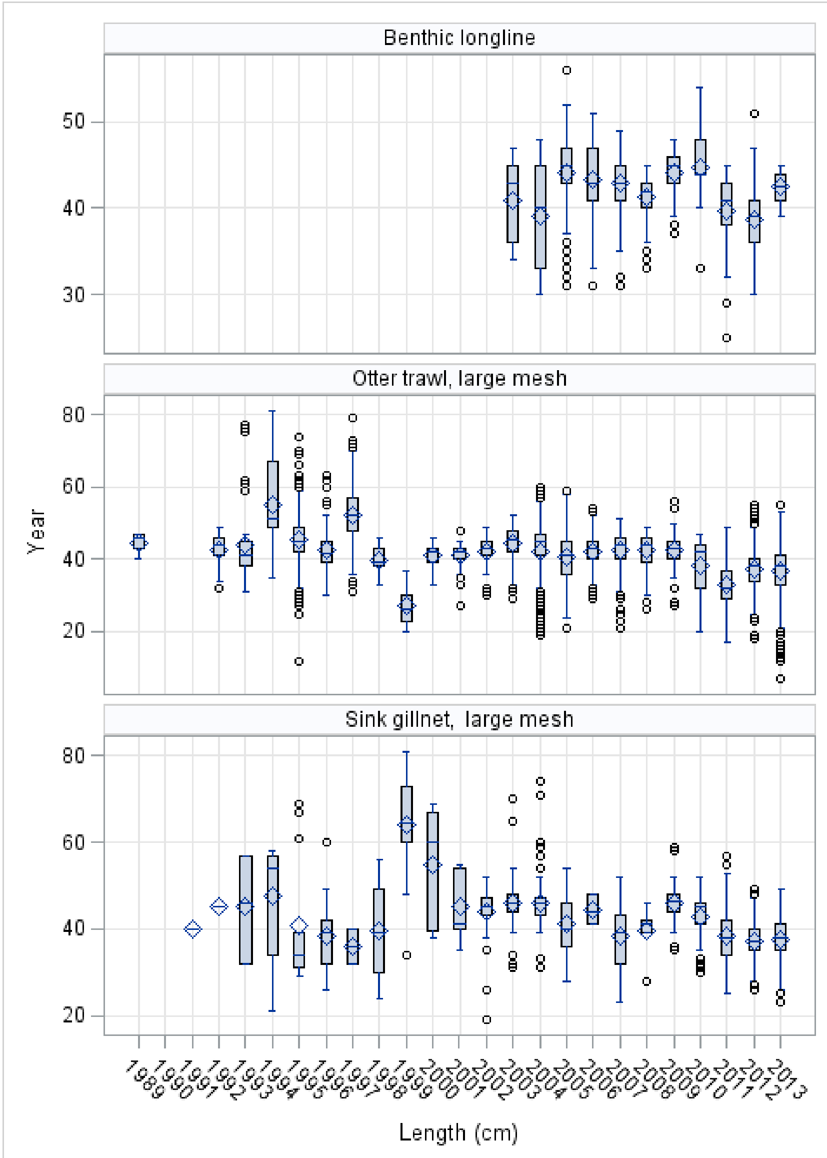


Figure A.53. Box plots showing the length distribution of Gulf of Maine haddock discarded by the commercial fishery by gear type between 1989 and 2013. Missing years indicate that there were either no observed trips for that gear in the Gulf of Maine or no haddock were observed to have been discarded.

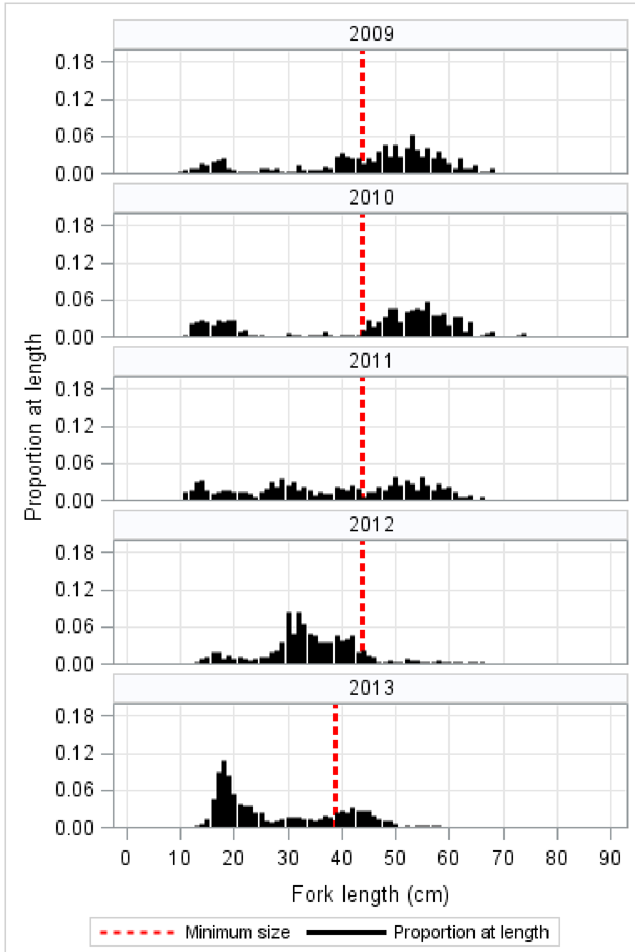


Figure A.54. Frequency distribution of Gulf of Maine haddock lengths collected from Northeast Fisheries Science Center (NEFSC) bottom trawl surveys between 2009 and 2013. The commercial minimum retention size for the specific year is indicated by the dashed red line.

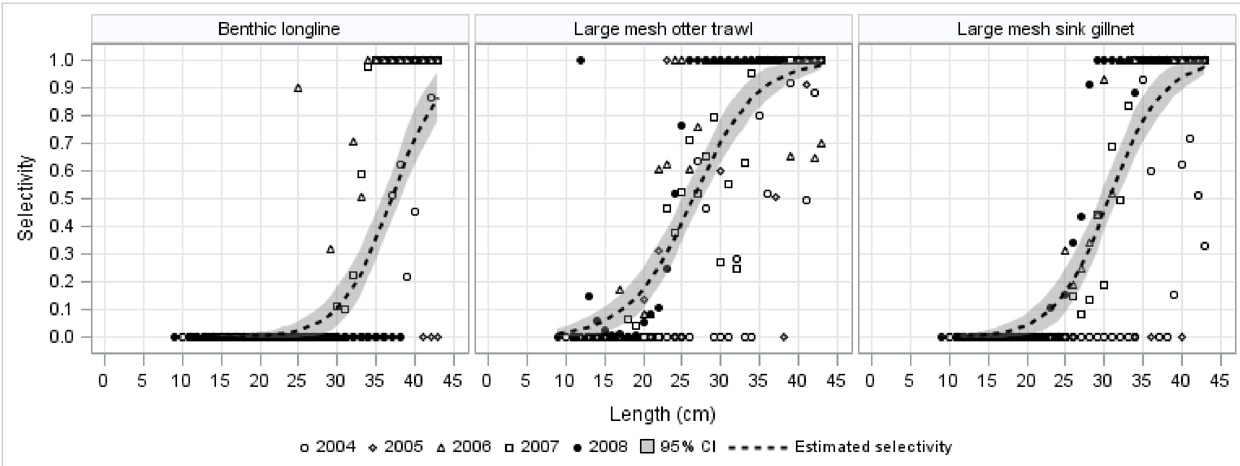


Figure A.55. Estimated selectivity ogives for benthic longline, large mesh (5.5" – 7.9") otter trawl and sink gillnet and the corresponding 95% confidence intervals (CI) for Gulf of Maine haddock. Selectivity ogives were estimated from the logistic fits to the aggregated annual estimates of selectivity-at-length.

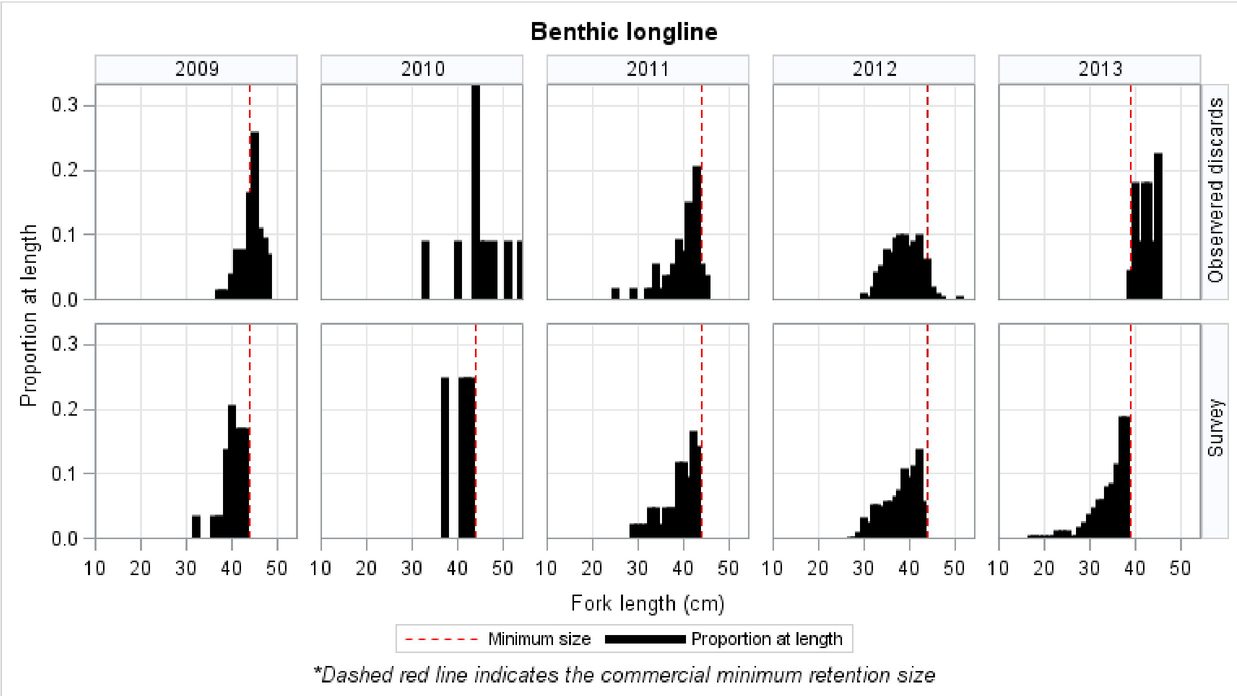


Figure A.56. Comparison of the survey filter-based estimates of discards-at-length for benthic longline gear to the direct observer observations from 2009 to 2013 for Gulf of Maine haddock. The dashed red line represents the commercial minimum retention size for the specific year.

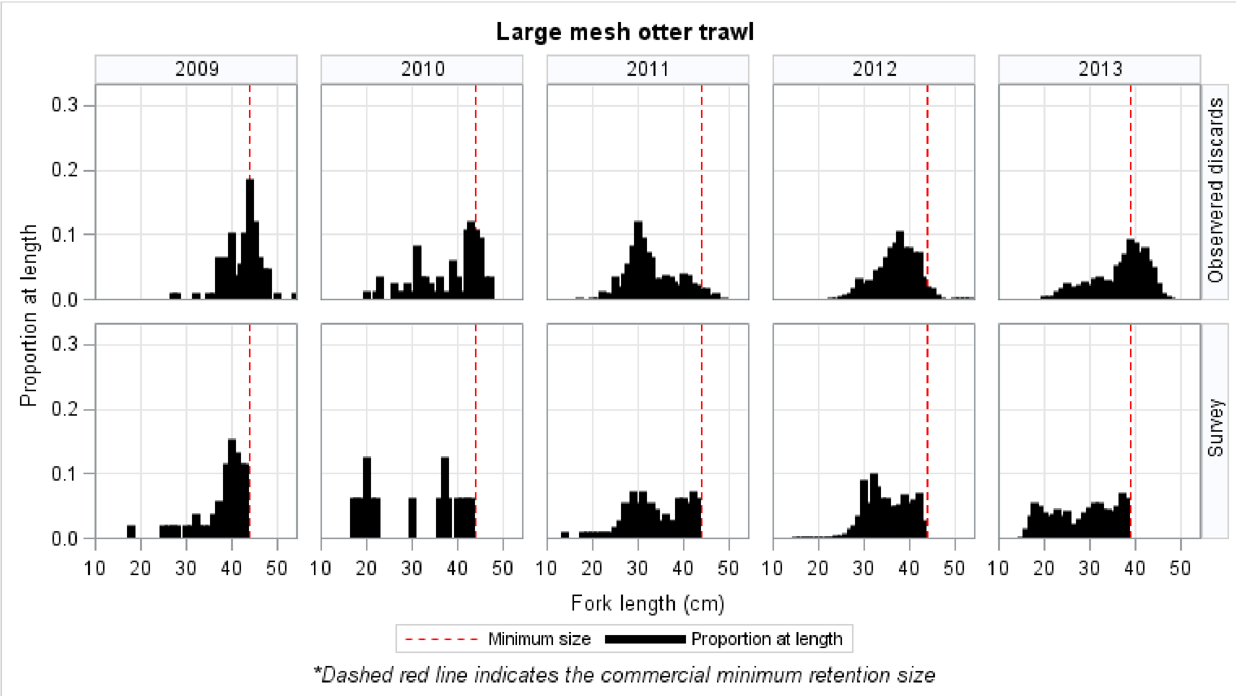


Figure A.57. Comparison of the survey filter-based estimates of discards-at-length for large mesh (5.5" – 7.9") otter trawl gear to the direct observer observations from 2009 to 2013 for Gulf of Maine haddock. The dashed red line represents the commercial minimum retention size for the specific year.

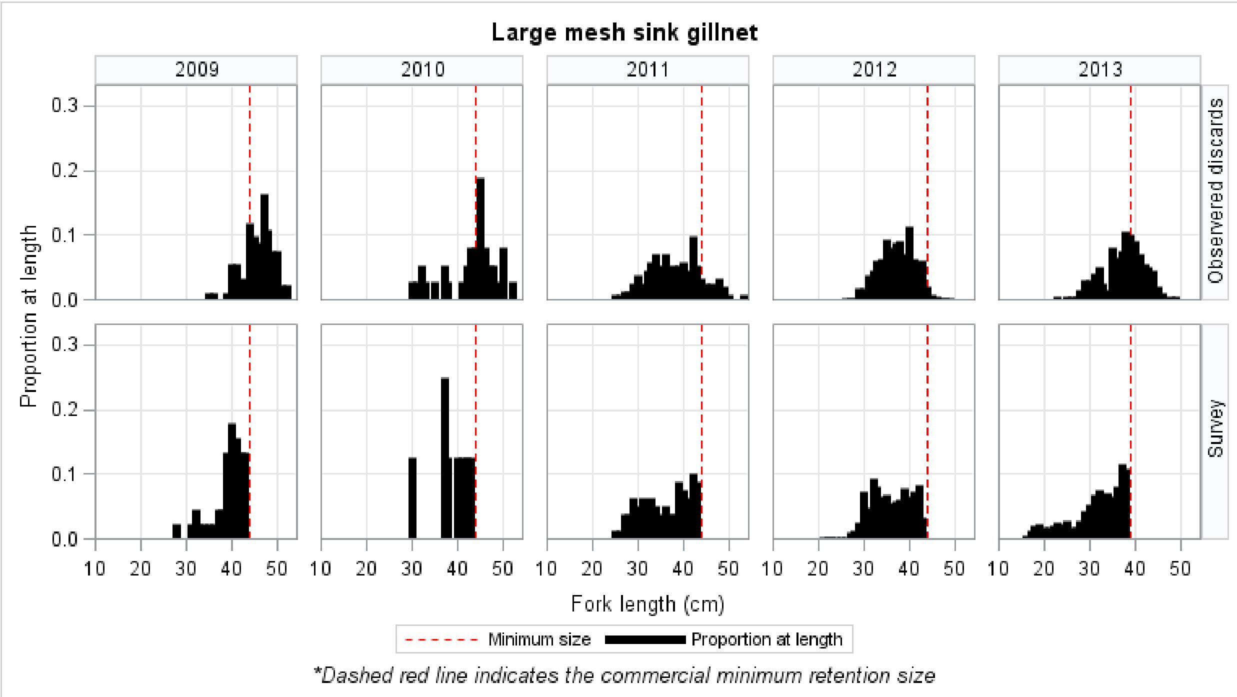


Figure A.58. Comparison of the survey filter-based estimates of discards-at-length for large (5.5" – 7.9") mesh sink gillnet gear to the direct observer observations from 2009 to 2013 for Gulf of Maine haddock. The dashed red line represents the commercial minimum retention size for the specific year.

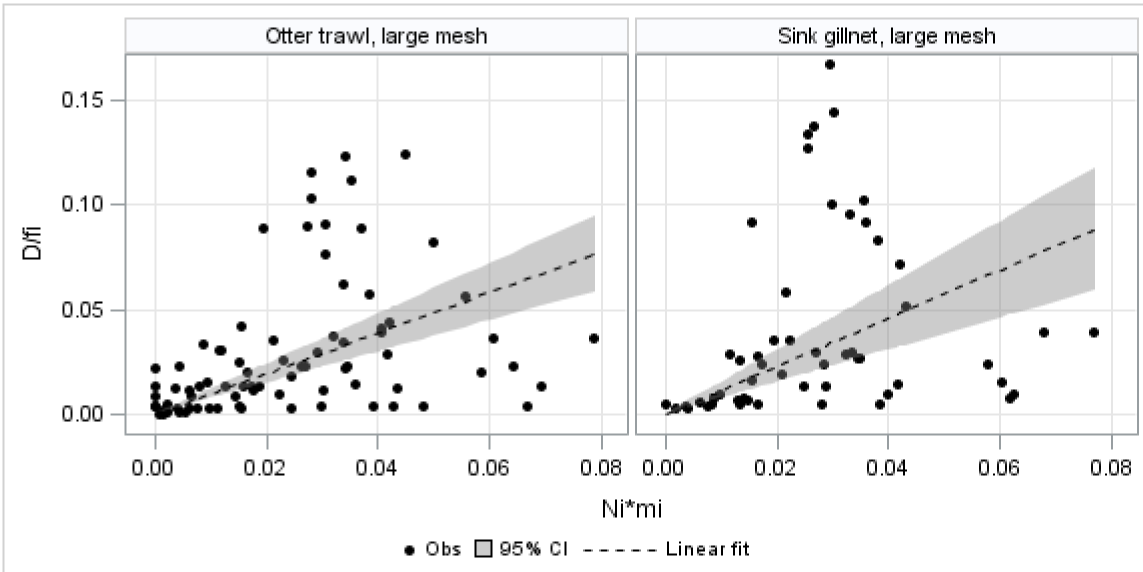


Figure A.59. Plots of the relationship by gear type between fraction of Gulf of Maine haddock observed discarded-at-length (D_i/f) and the estimated number at length from the survey-filter method (N_i•m_i) for large mesh (5.5" – 7.9") otter trawl and large mesh (5.5" – 7.9") sink gillnet gear. The slope of the relationship (q) is the proportionality constant required to expand the survey-filter estimates of numbers-at-length to estimates of total discards-at-length.

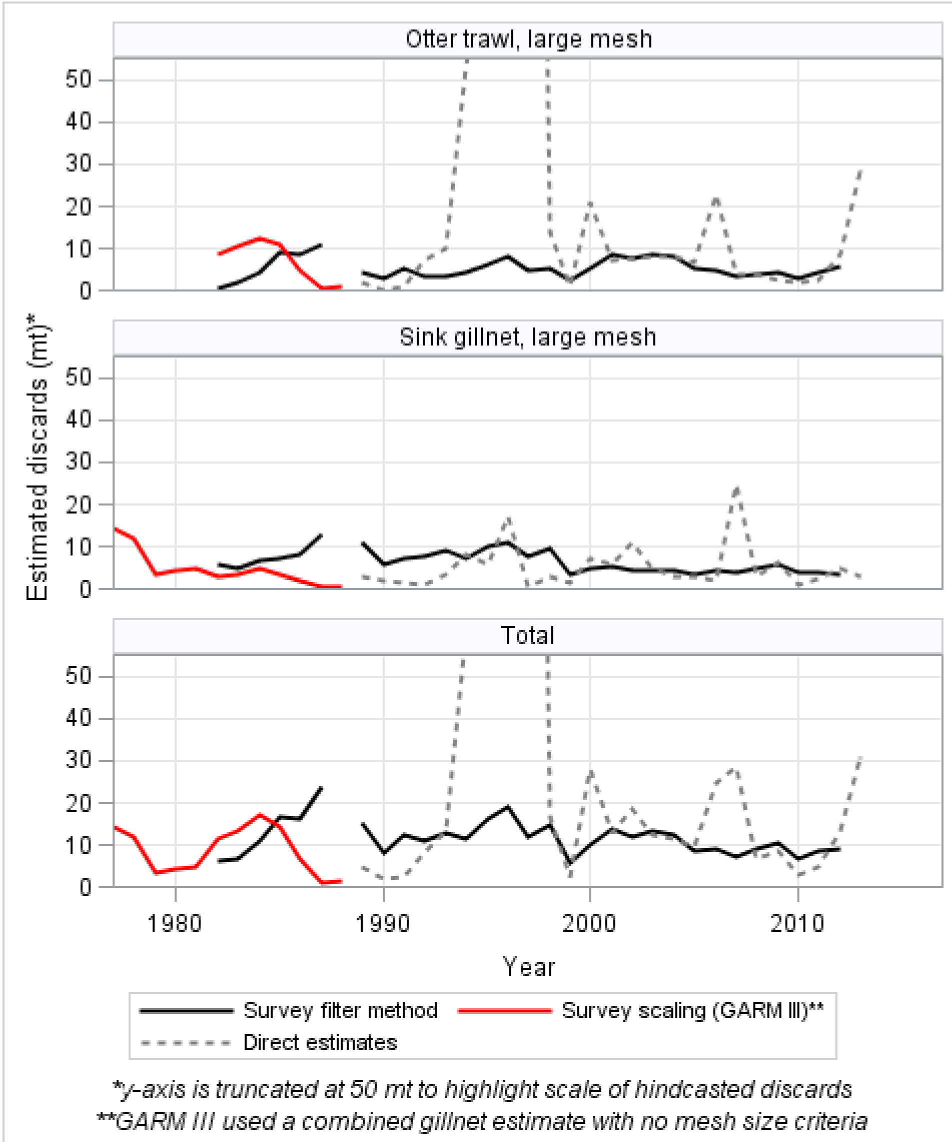


Figure A.60. Comparison of Gulf of Maine haddock discard estimates for large mesh (5.5” – 7.9”) otter trawl and sink gillnet gears calculated using the survey-filter hindcast method to the survey scaling method used in GARM III and the direct estimates obtained from 1989 to 2013. *Note that the y-axis has been truncated at 50 mt to preserve scale.*

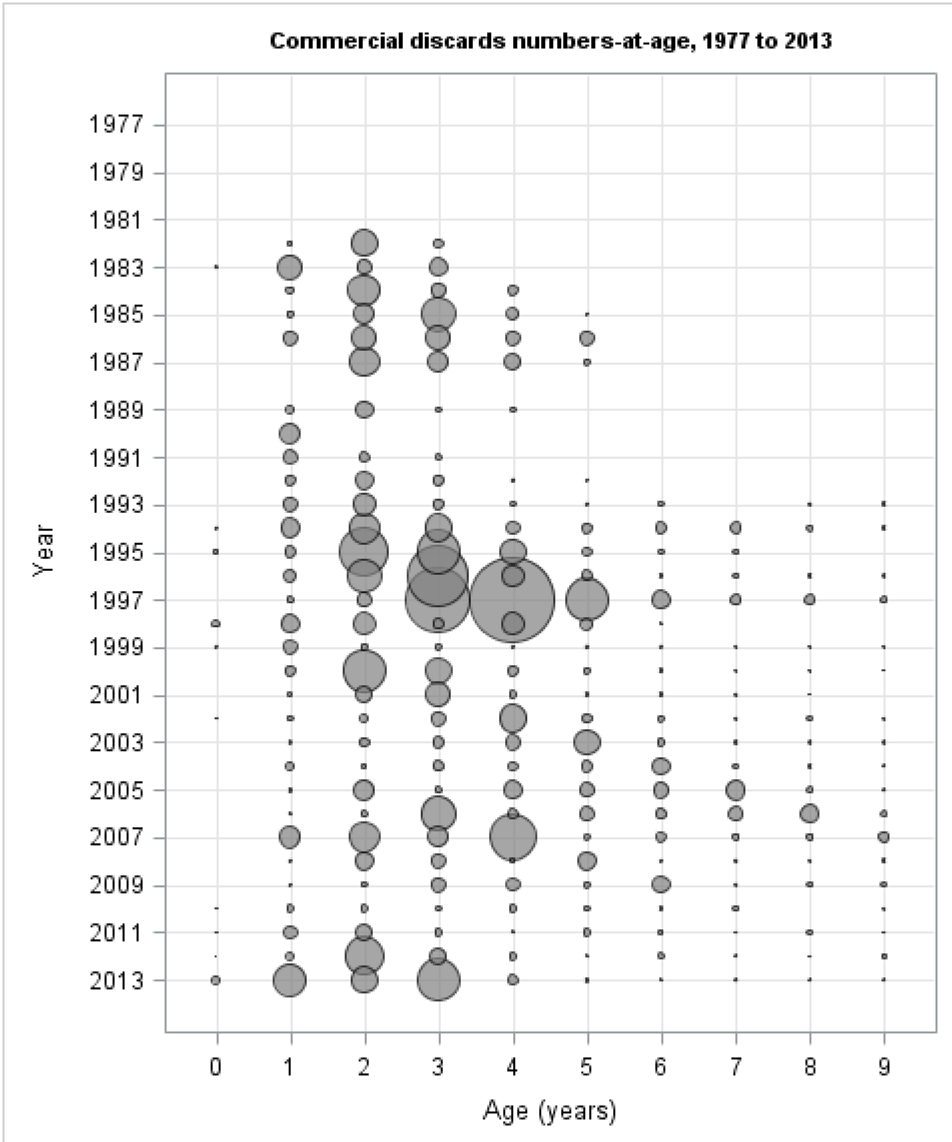


Figure A.61. Commercial discards-at-age of Gulf of Maine haddock from 1977 to 2013. *Note that commercial discards were not estimated pre-1982.*

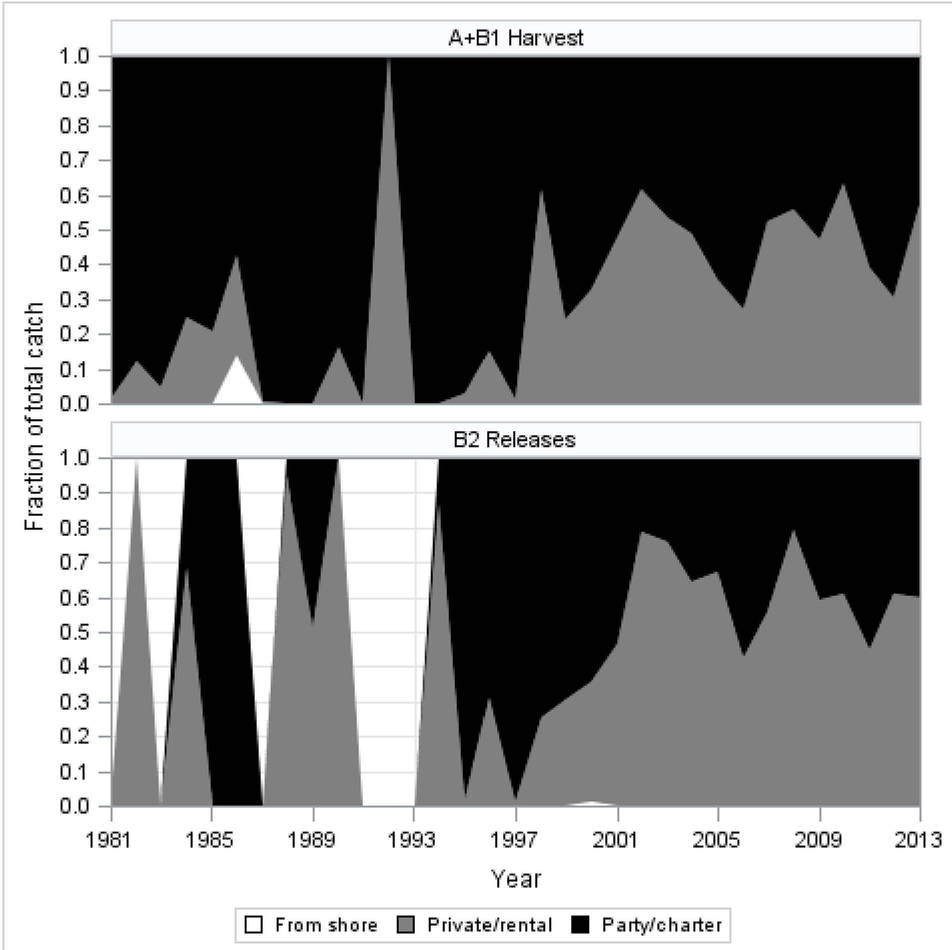


Figure A.62. Fractional distribution of recreational catch of Gulf of Maine haddock by fishing model based on MRFSS/MRIP data.

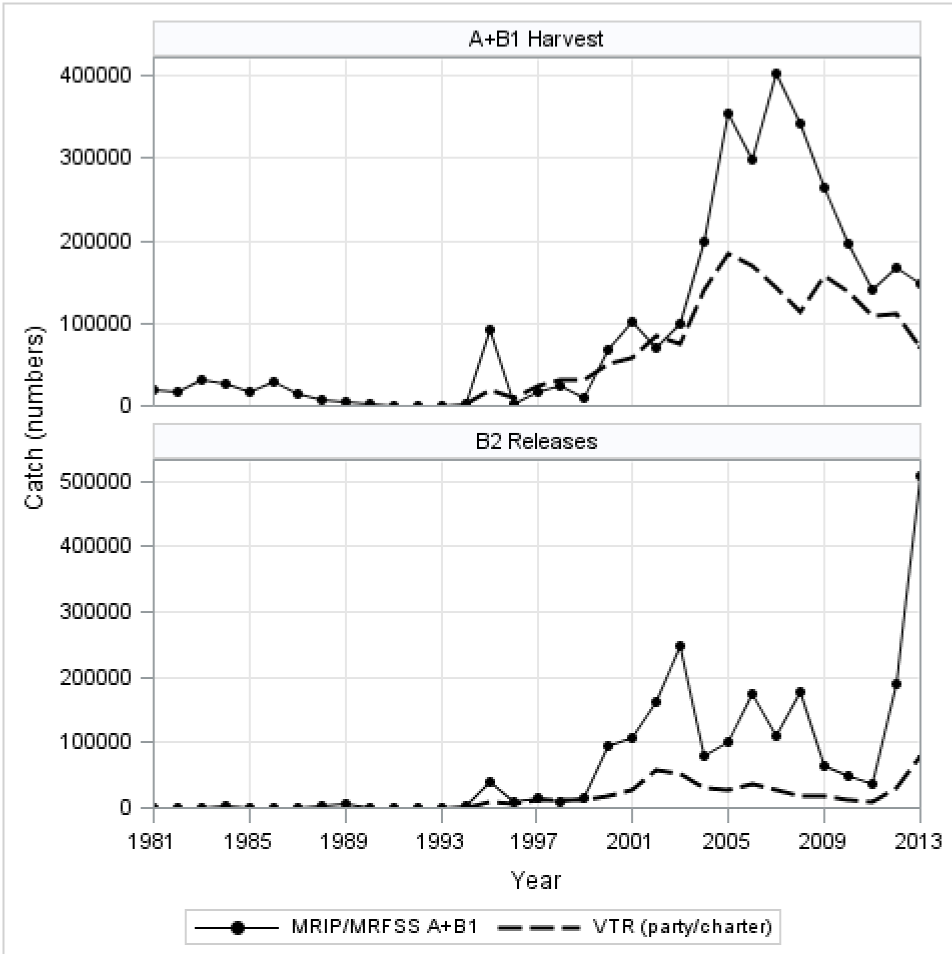


Figure A.63. Comparison of Gulf of Maine haddock recreational harvest (landings) and releases (discards) estimates derived through the Marine Recreational Fisheries Statistics Survey (MRFSS)/Marine Recreational Information Program (MRIP) to recreational landings reported on vessel trip reports (VTRs) between 1981 and 2013. **Note: MRFSS/MRIP data collection began in 1981 and VTR data collection began in 1994.*

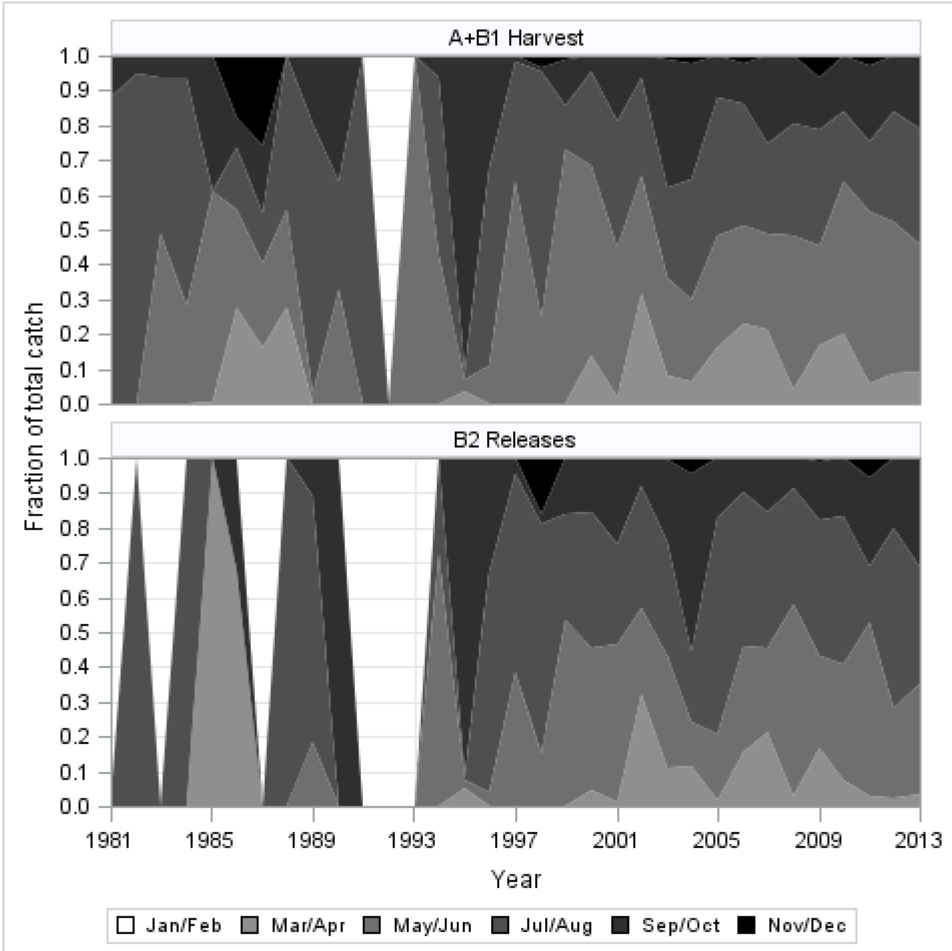


Figure A.64. Fractional distribution of recreational catch of Gulf of Maine haddock by wave (two month time blocks) based on vessel trip report (VTR) data.

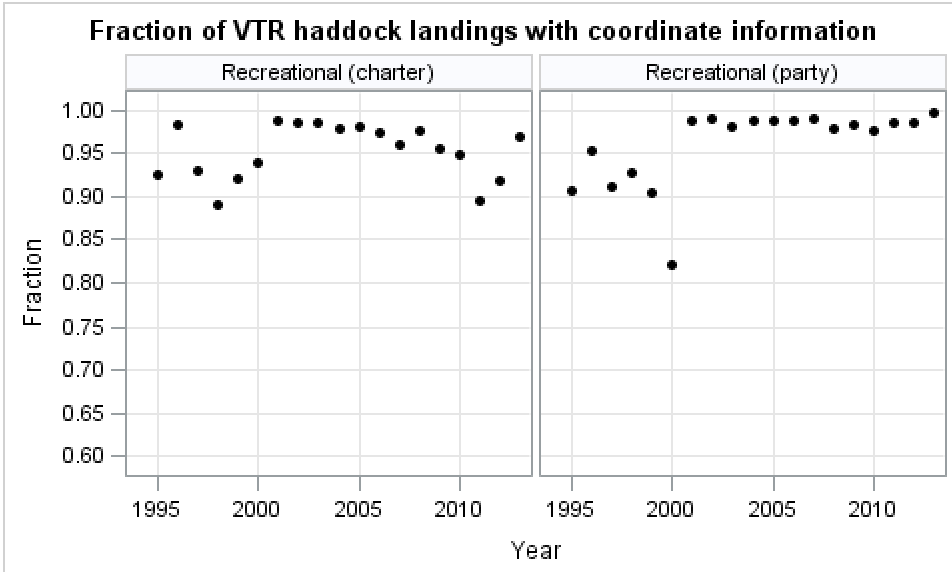


Figure A.65. Fraction of total Gulf of Maine haddock recreational landings reported on vessel trip reports (VTRs) with latitude and longitude coordinate information, by fleet from 1994 to 2013.

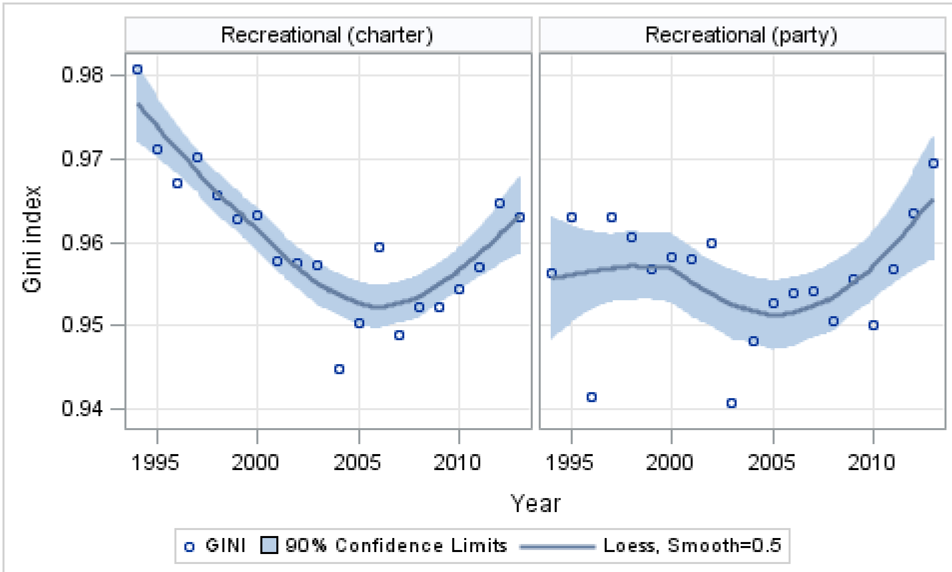


Figure A.66. Gini indices for the Gulf of Maine haddock for the recreational charter and party boat fleets from 1994 to 2013. Indices were based on the spatial distribution of the retained catch reported on vessel trip reports (VTRs).

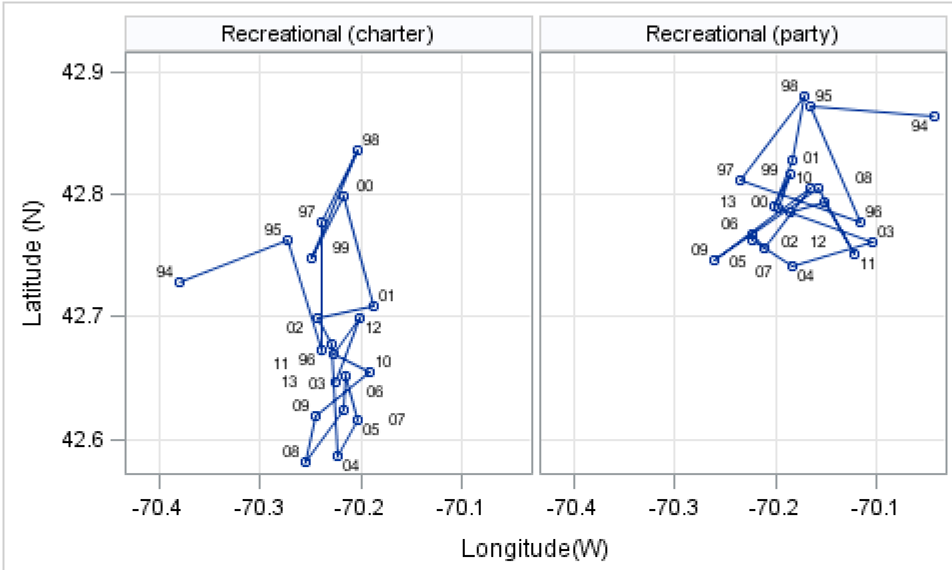


Figure A.67. Landings-weighted mean location (centroid) for the Gulf of Maine haddock recreational charter and party boat fleets from 1994 to 2013. Centroids were calculated using the spatial distribution of the retained catch reported on vessel trip reports (VTRs).

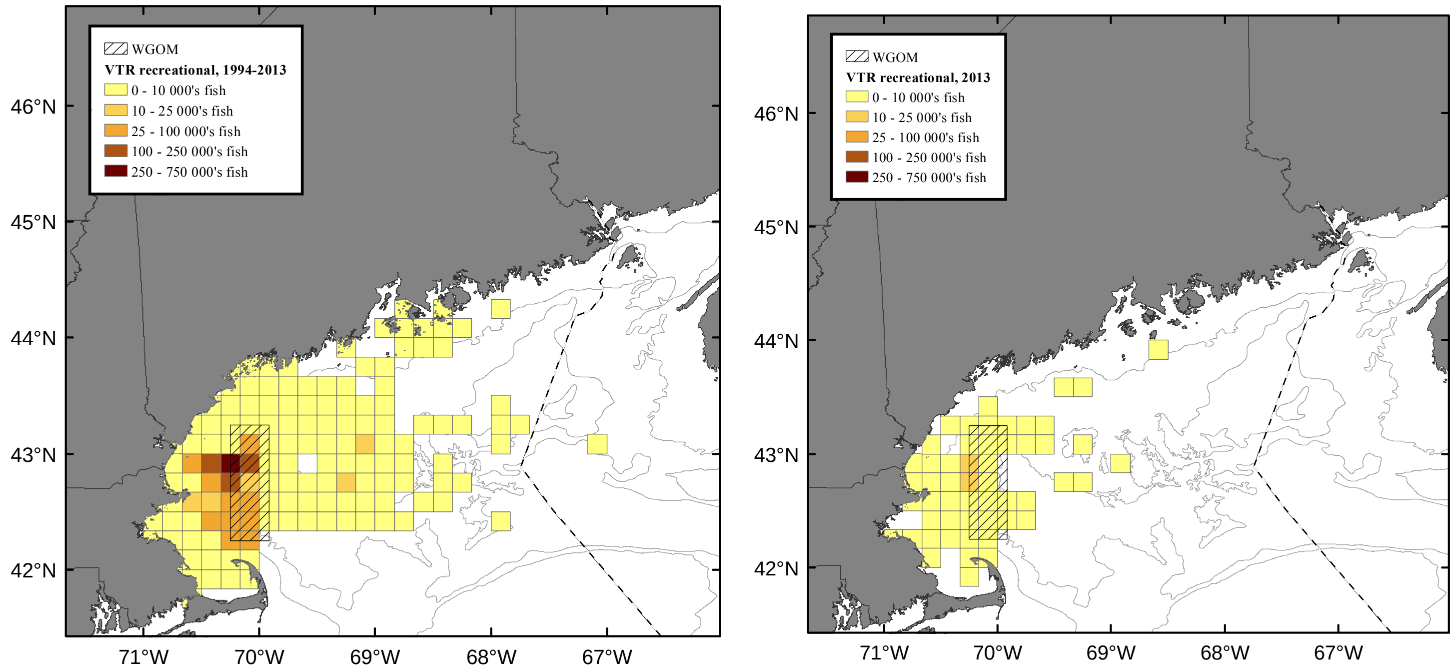


Figure A.68. Comparison of the distribution of Gulf of Maine haddock recreational landings per ten minute square in 2013 (right) to the aggregate distribution from 1994 to 2013 (left). The location of the Western Gulf of Maine (WGOM) closed area is indicated.

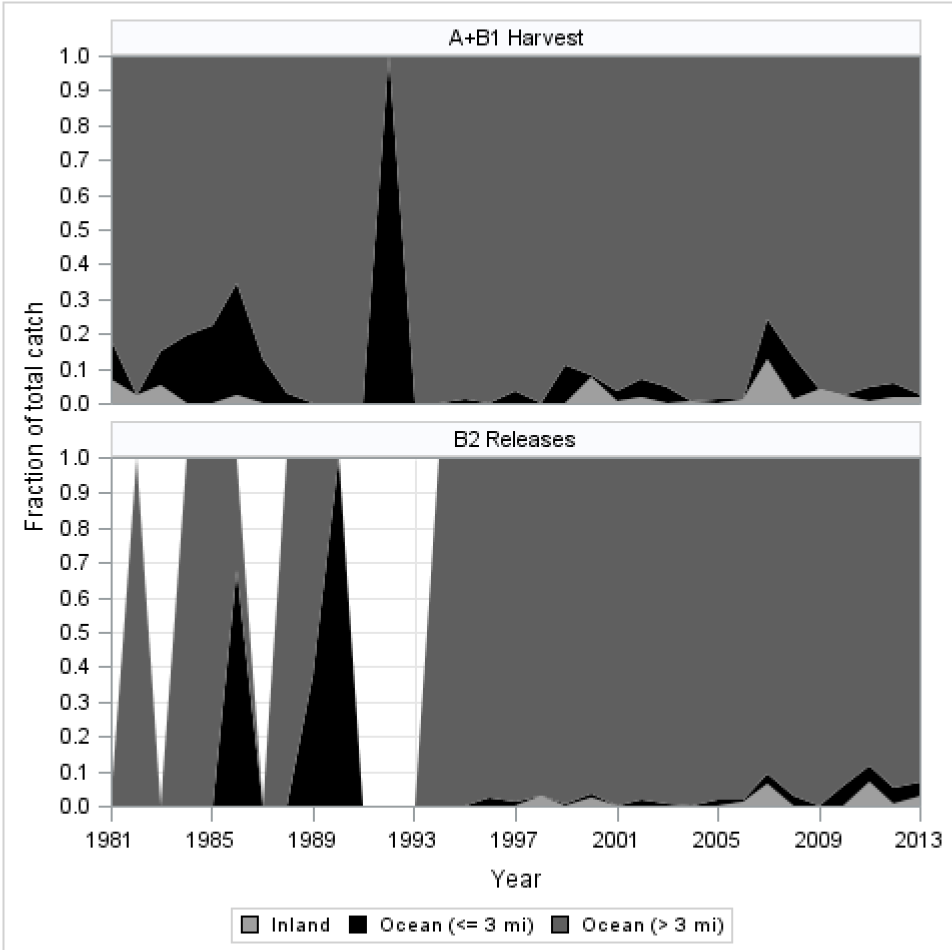


Figure A.69. Fractional distribution of recreational catch of Gulf of Maine haddock by fishing area. This summary uses MRFSS/MRIP data.

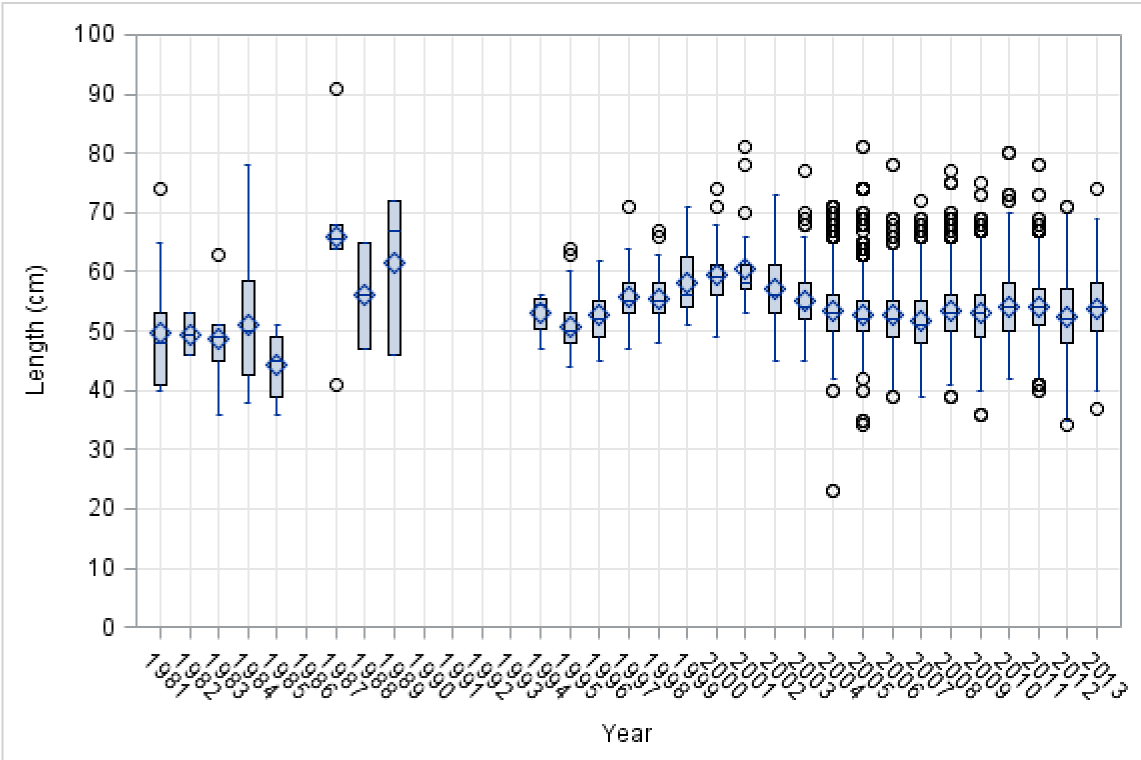


Figure A.70. Box plots showing the length distribution of Gulf of Maine haddock recreational harvest (AB1 catch) between 1981 and 2013.

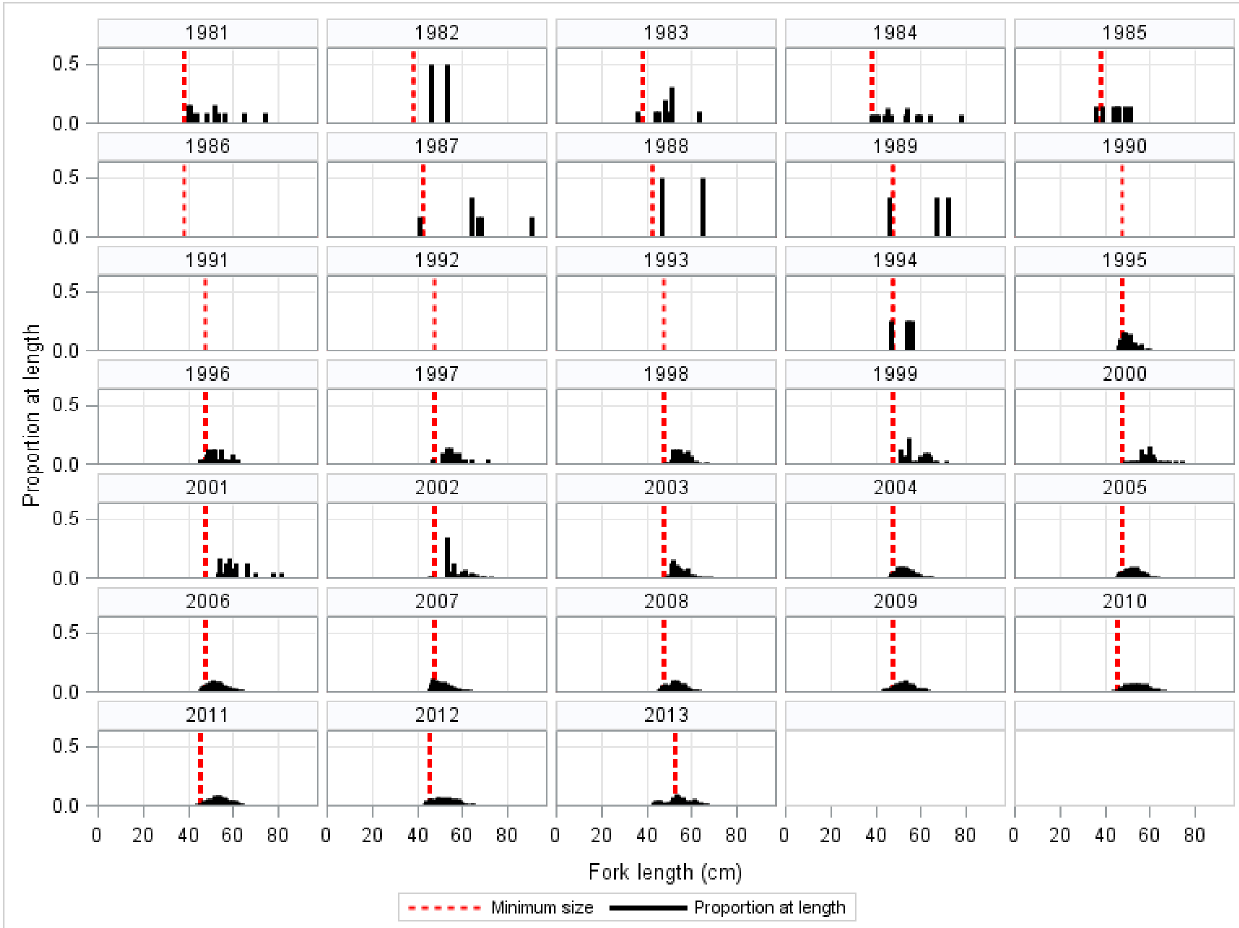


Figure A.71. Length frequency distribution of Gulf of Maine haddock recreational harvest (AB1 catch) between 1981 and 2013. Minimum retention sizes for the specific years are indicated by a dashed red line.

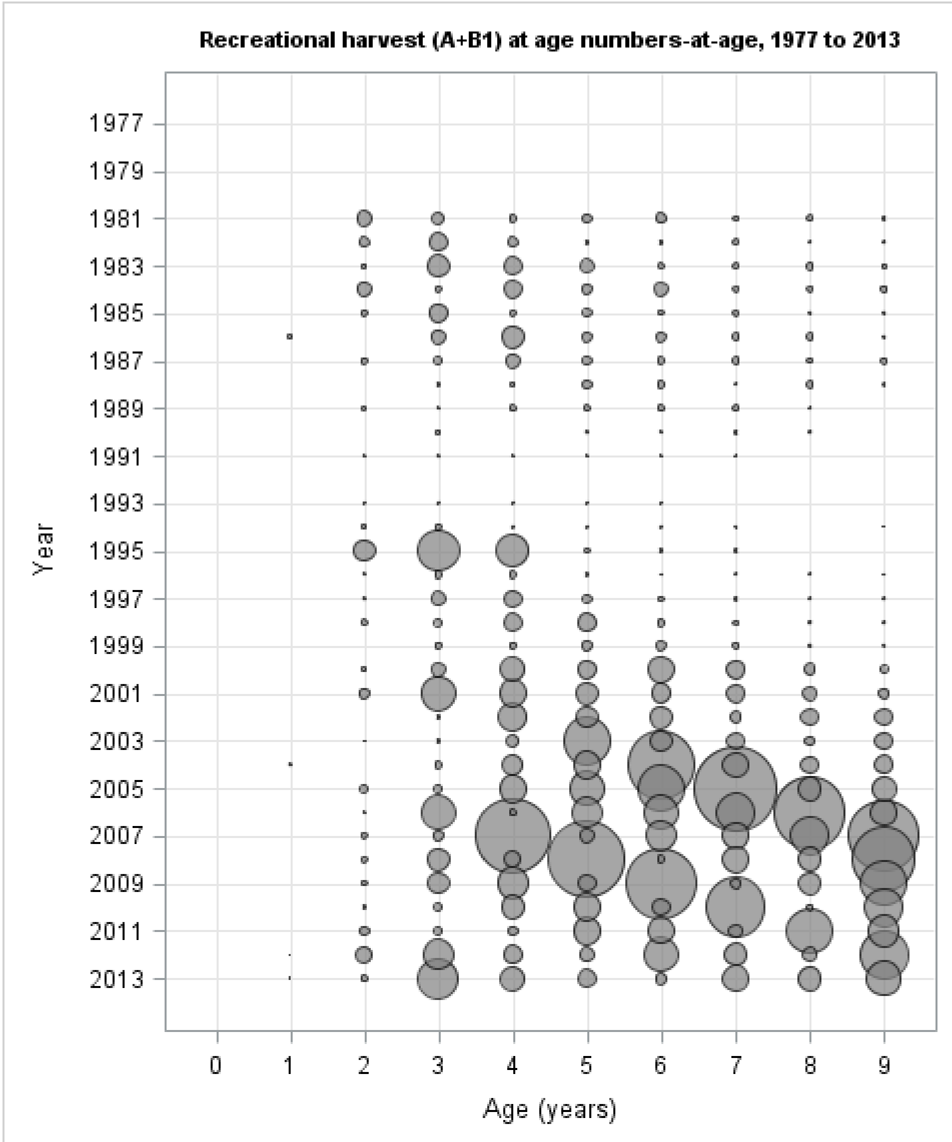


Figure A.72. Recreational harvest (landings)-at-age of Gulf of Maine haddock from 1977 to 2013. *Note that estimates of recreational harvest are not available prior to 1981.*

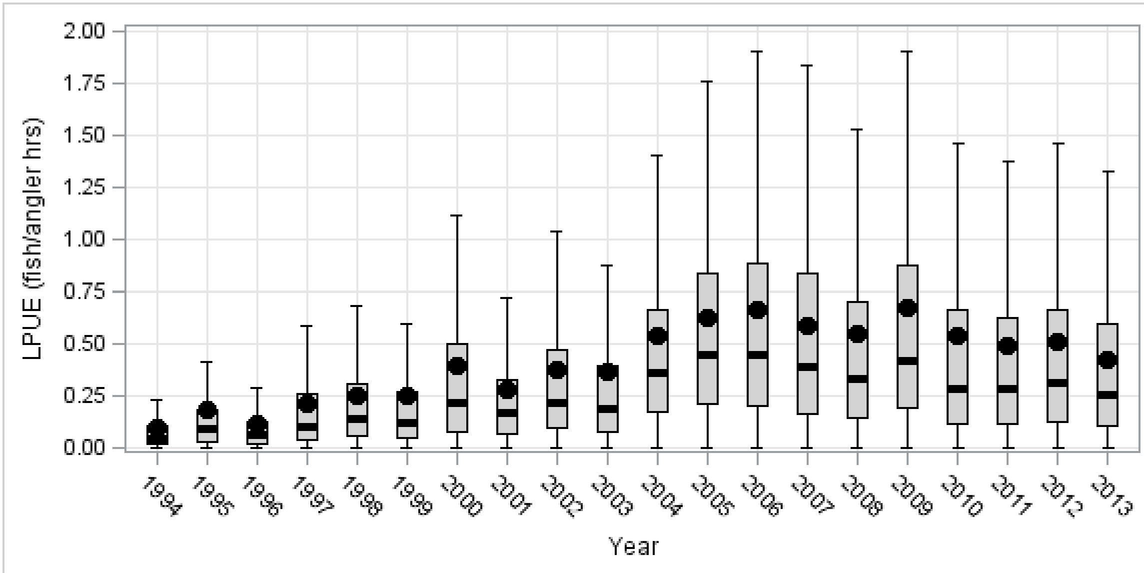


Figure A.73. Box-plot distributions of nominal Gulf of Maine haddock recreational landings (count) per angler hour from 1994 to 2013. Vessel trip report (VTR) data were used in this analysis and only includes trips that reported landing Gulf of Maine haddock.

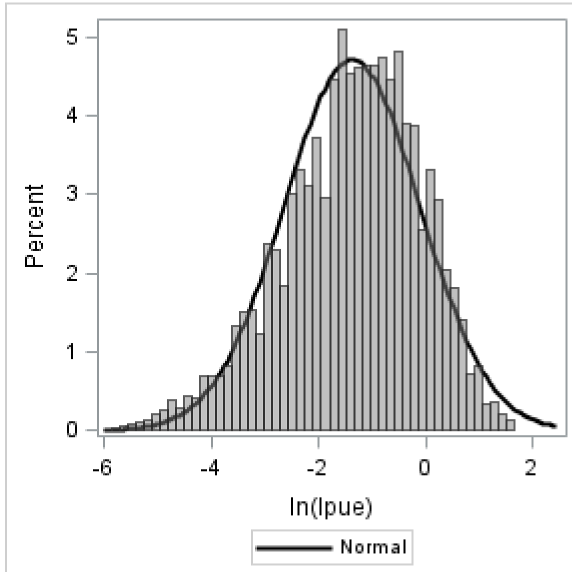


Figure A.74. Distribution of log transformed nominal recreational LPUE.

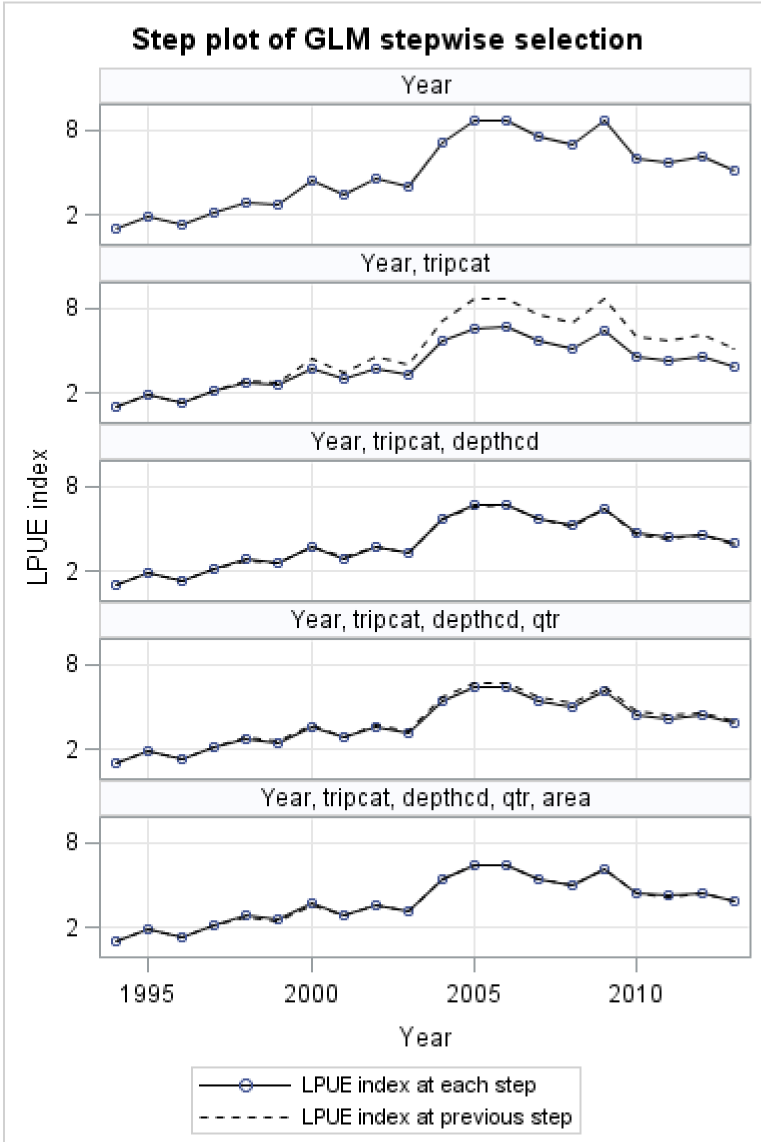


Figure A.75. Step plot of the recreational landings-per-unit-effort (LPUE) standardization model. Each panel shows the standardized abundance index as each explanatory variable is added to the model through stepwise selection. The index from previous step is indicated with a dashed line.

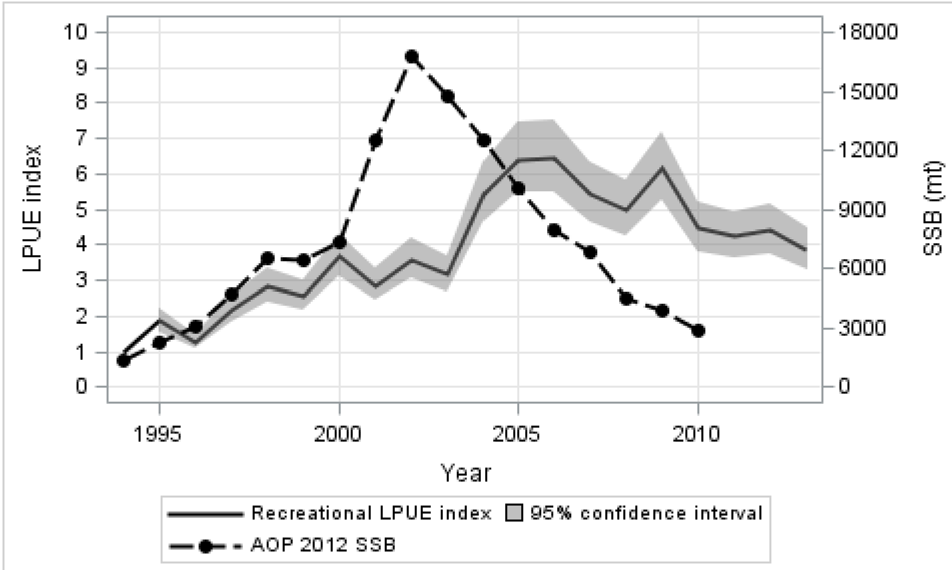


Figure A.76. Time series of the recreational LPUE index overlaid on the AOP 2012 assessment spawning stock biomass (SSB) estimate. *Note that the AOP 2012 SSB estimates only extends through 2010.*

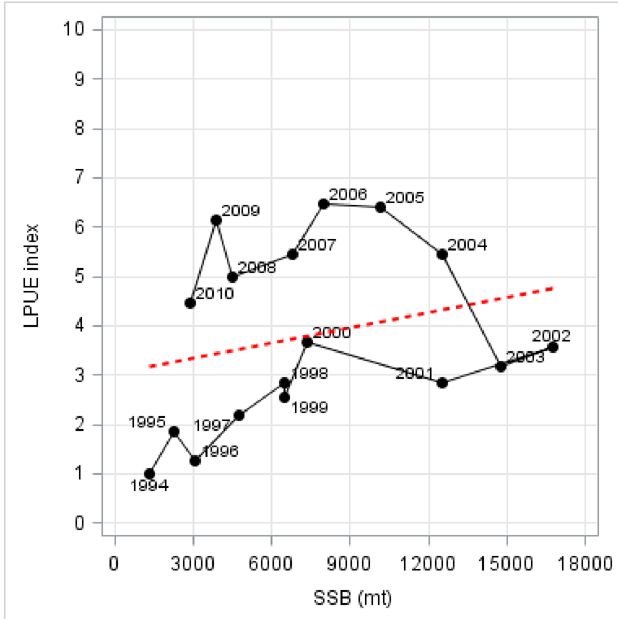


Figure A.77. Linear regression of the recreational LPUE index on spawning stock biomass (SSB) estimate from the AOP 2012 assessment.

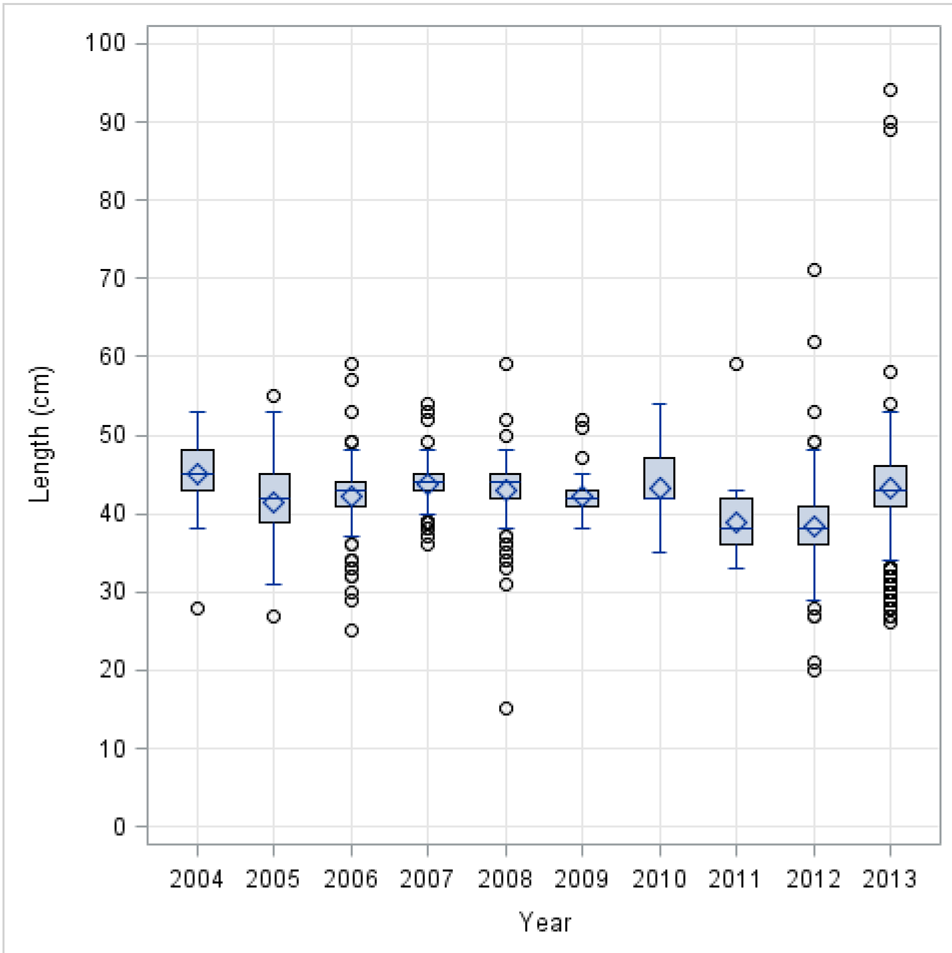


Figure A.78. Box plots showing the length distribution of Gulf of Maine haddock recreational releases (B2 catch) between 2004 and 2013.

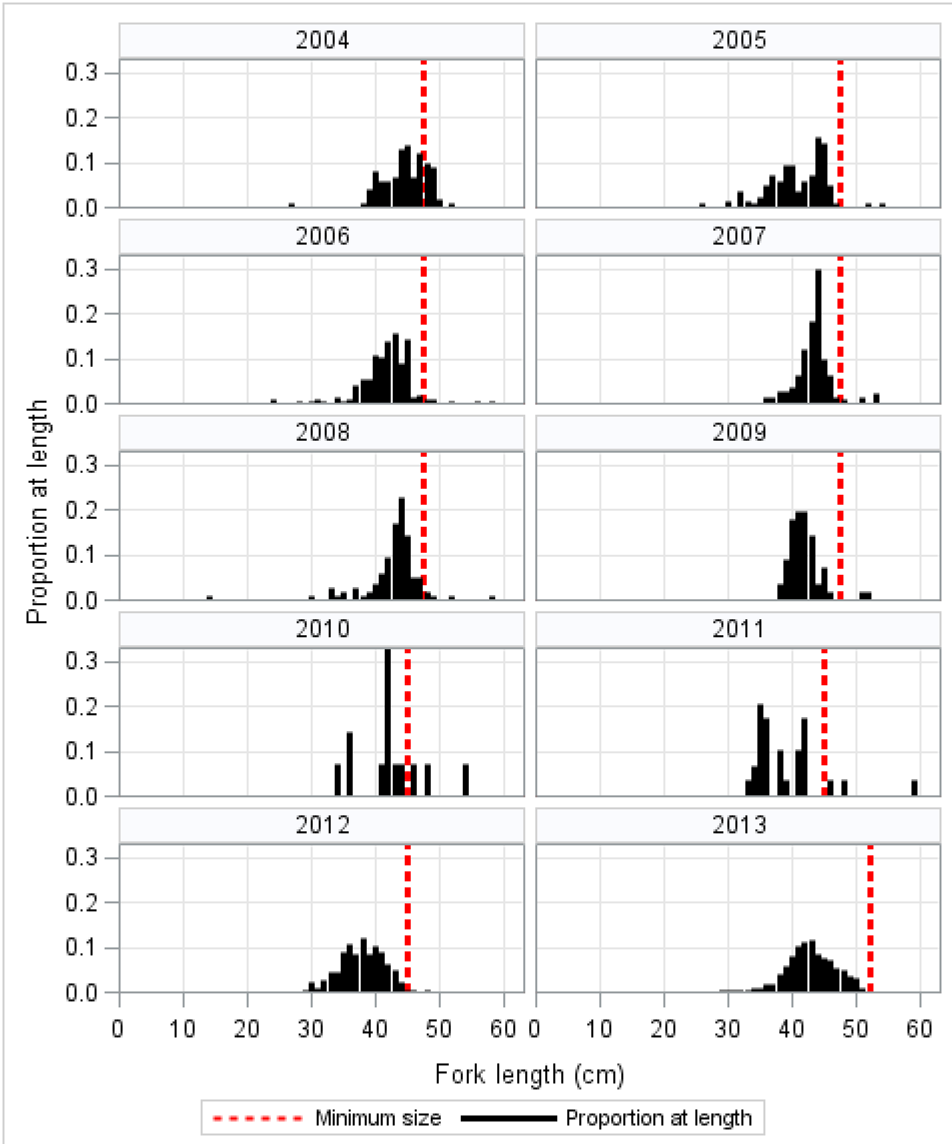


Figure A.79. Length frequency distribution of Gulf of Maine haddock recreational releases (B2 catch) between 2004 and 2013. Minimum retention sizes for the specific years are indicated by a dashed red line.

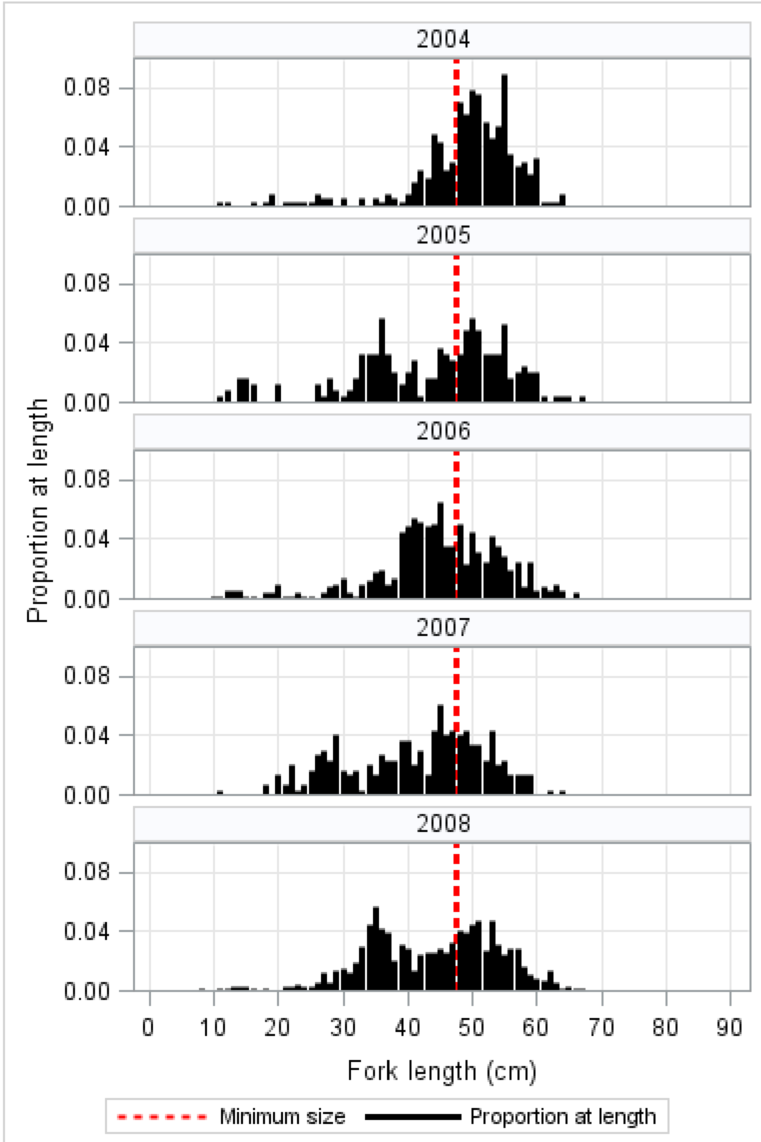


Figure A.80. Frequency distribution of Gulf of Maine haddock lengths collected from Northeast Fisheries Science Center (NEFSC) bottom trawl surveys between 2004 and 2008. The recreational minimum retention size for the specific year is shown by the dashed red line.

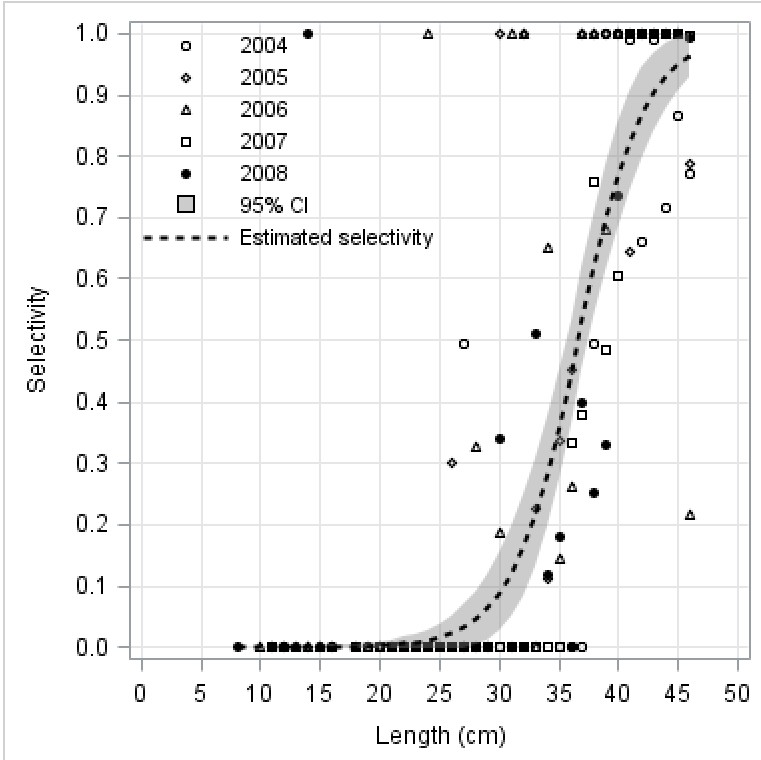


Figure A.81. Estimated selectivity ogives for the recreational fishery and the corresponding 95% confidence intervals (CI) for Gulf of Maine haddock. Selectivity ogives were estimated from the logistic fits to the aggregated annual estimates of selectivity-at-length.

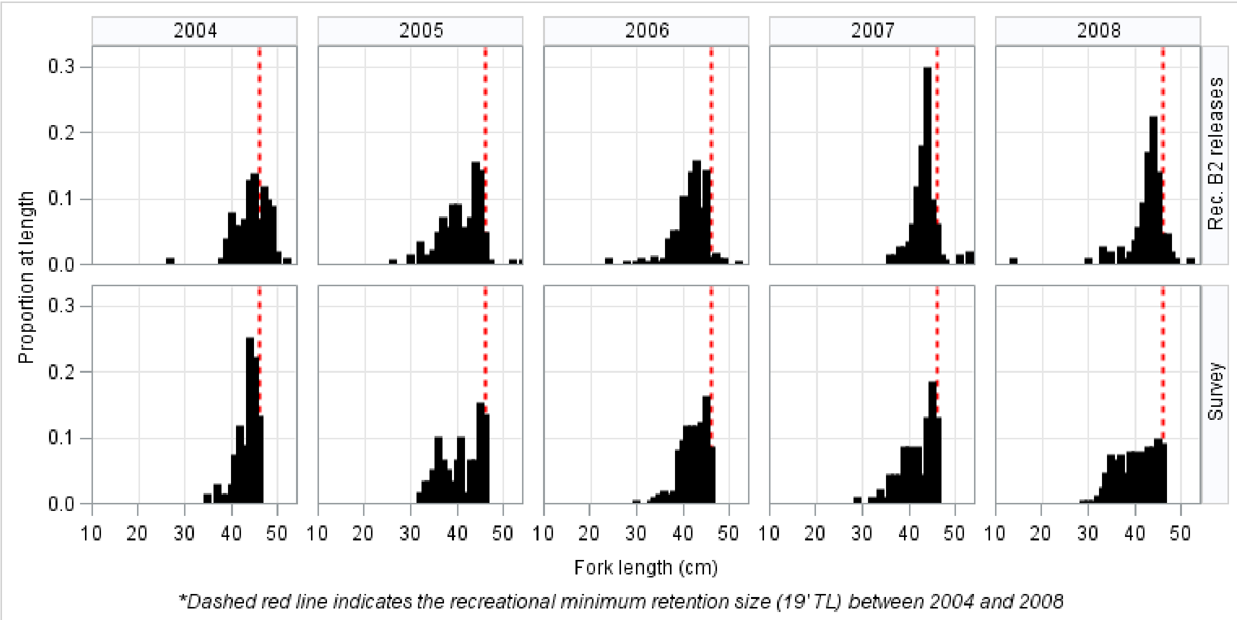


Figure A.82. Comparison of the survey filter-based estimates of discards-at-length for the recreational fishery to the direct observer observations from 2009 to 2013 for Gulf of Maine haddock. The dashed red line represents the commercial minimum retention size for the specific year.

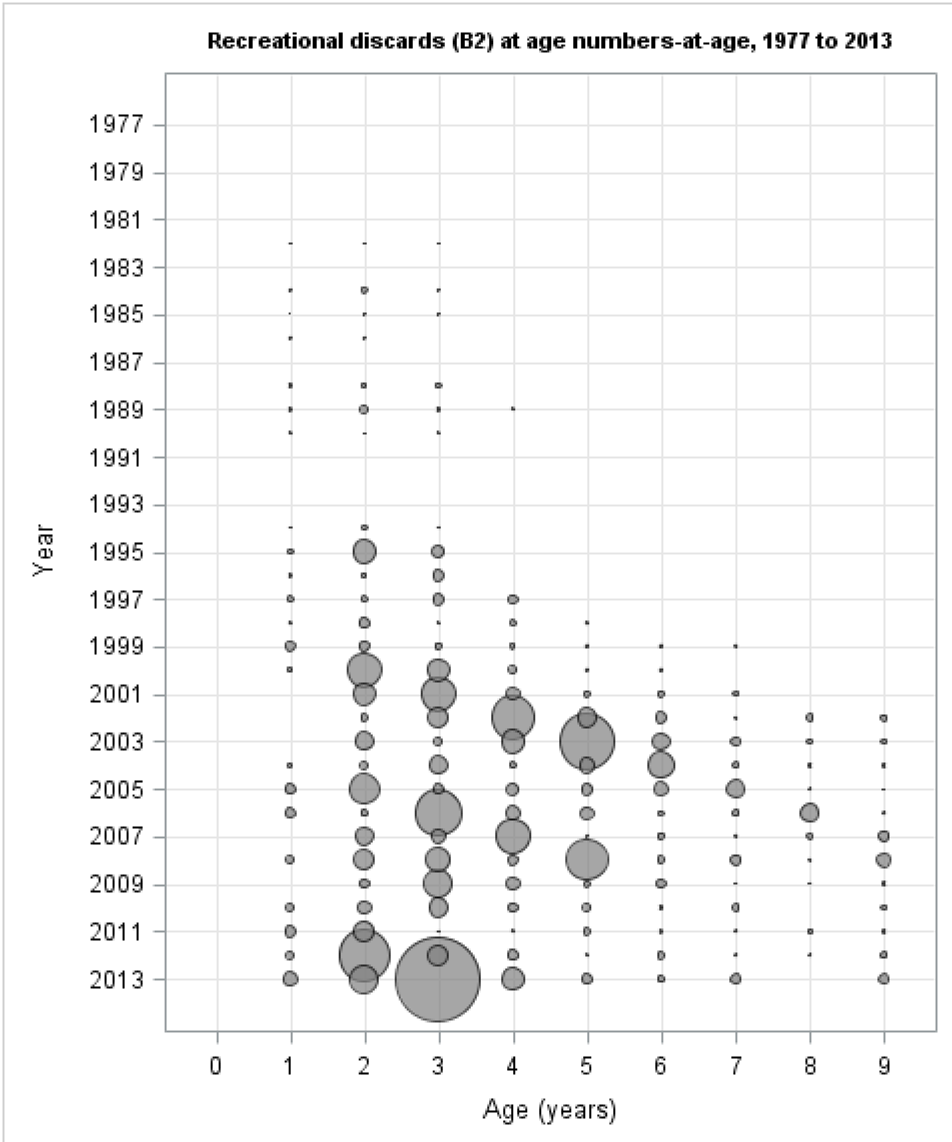


Figure A.83. Recreational discards (releases)-at-age of Gulf of Maine haddock from 1977 to 2013. *Note that estimates of recreational releases are not available prior to 1981.*

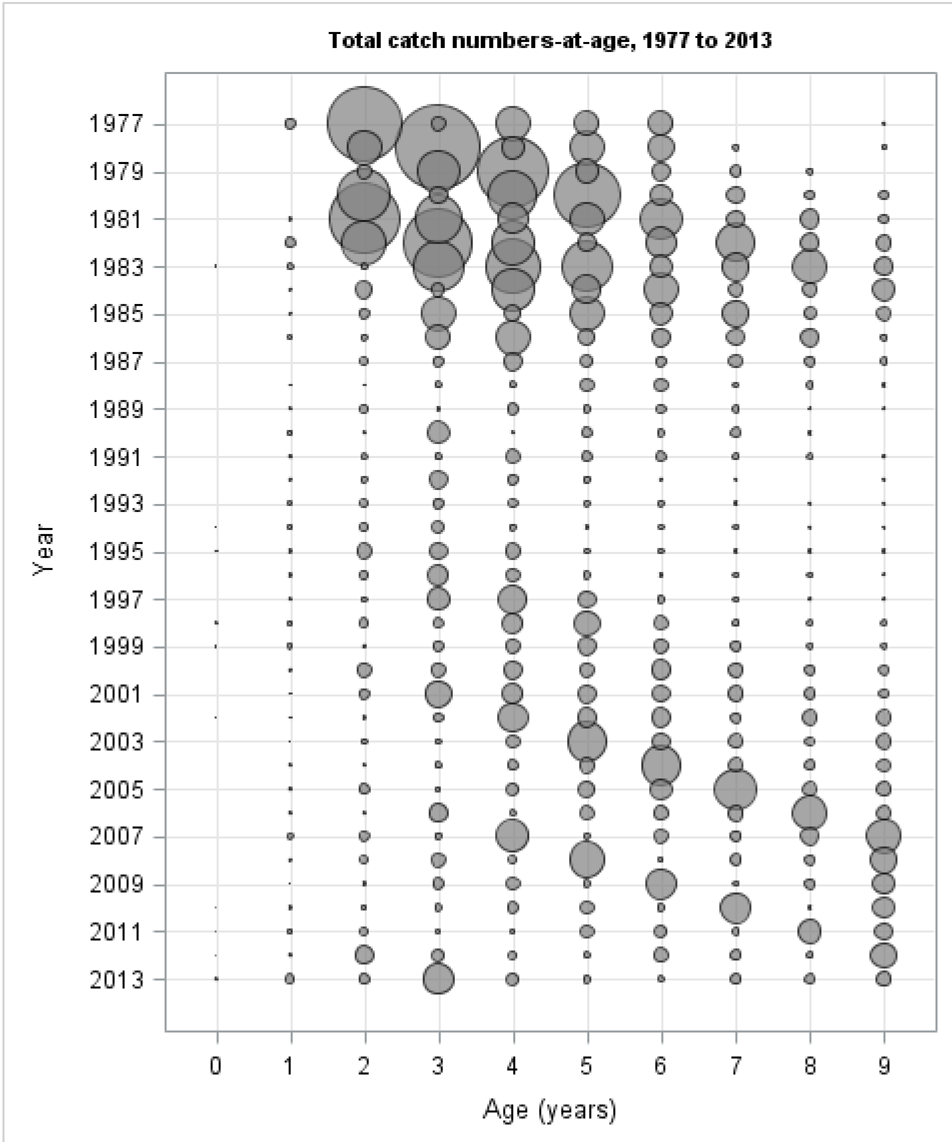


Figure A.84. Total commercial and recreational catch-at-age of Gulf of Maine haddock from 1977 to 2013.

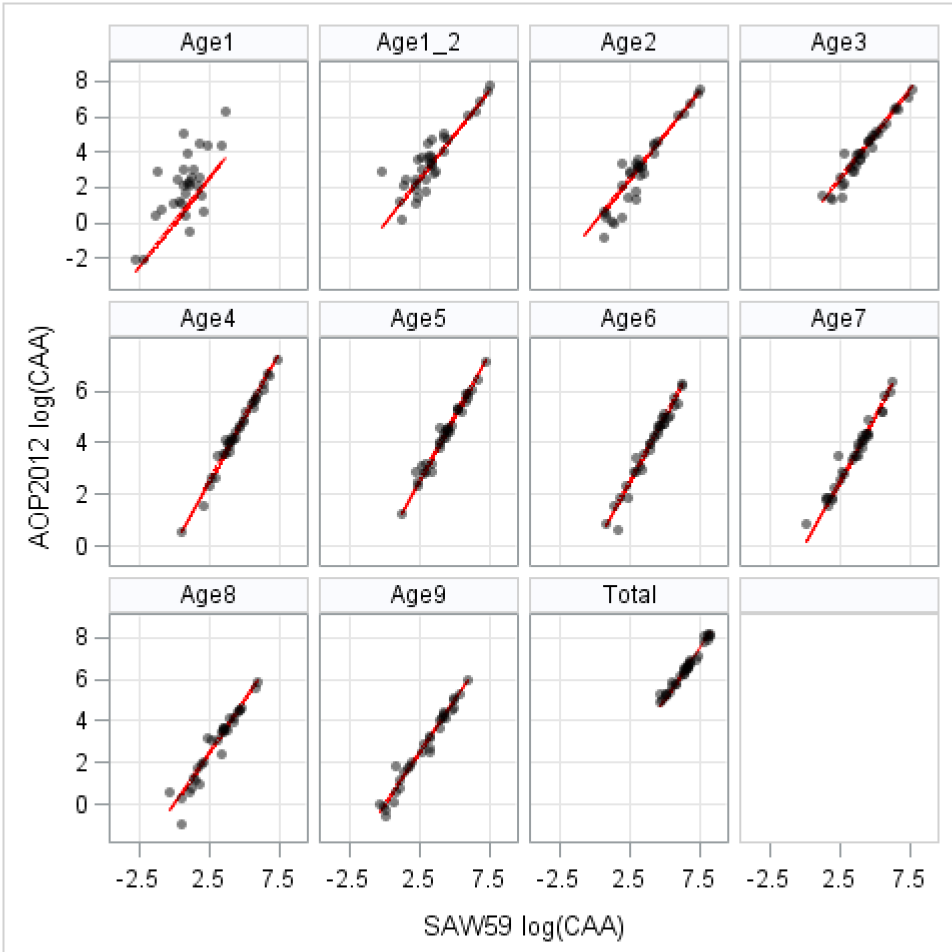
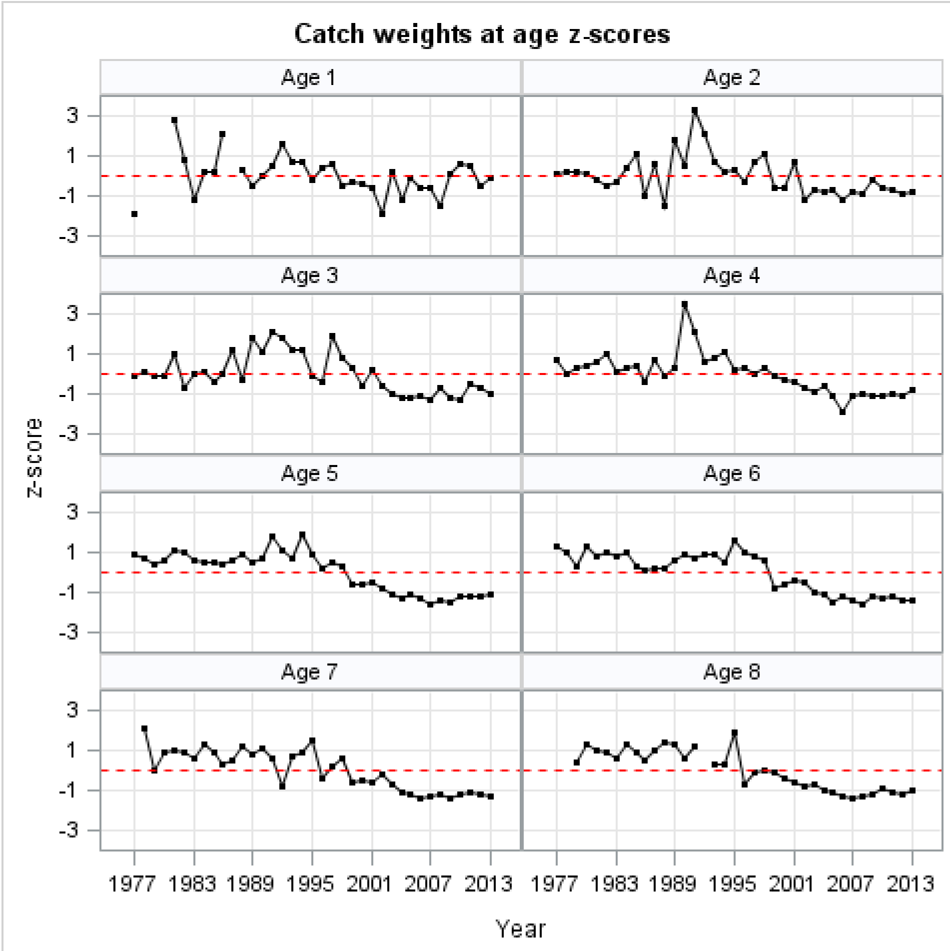


Figure A.85. Comparison of the 2012 AOP estimated catch-at-age to the SAW/SARC 59 updated catch-at-age. *Note that the second plot from the top left corner reflects a comparison of aggregated age-1 and age-2 catch.*



Figue A.86. Average catch weights-at-age of age-1 to age-8 Gulf of Maine haddock from 1977 to 2013. Weights-at-age were estimated using a number weighted average of commercial landing, commercial discard, recreational landings, and recreational discards weights-at-age. Average weights are presented as z-scores ([*x*-μ]/σ).

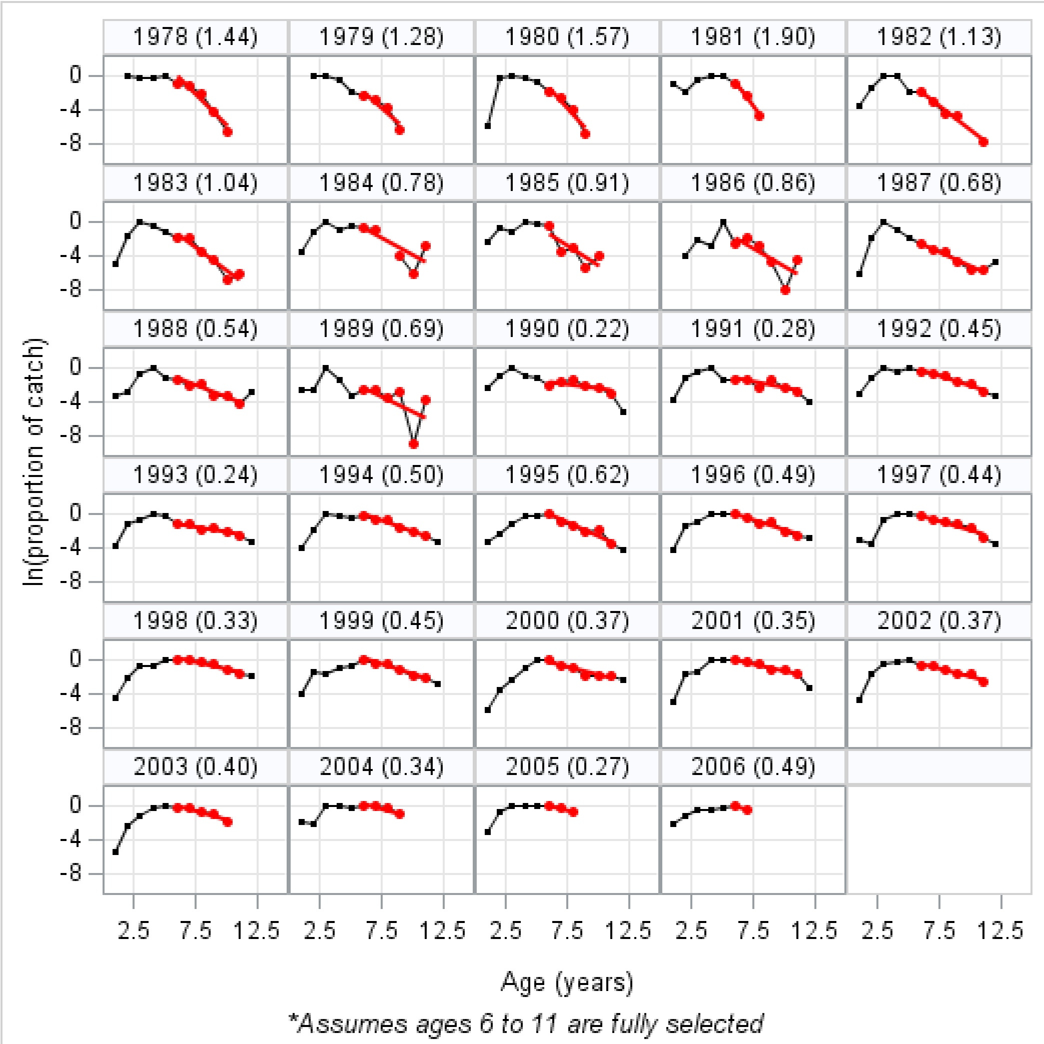


Figure A.87. Gulf of Maine haddock year class curves computed on ages 6-11 (red circles) log-transformed catch (commercial and recreational landings and discards). The corresponding slope of each regression line is shown next to the year class label above each plot.

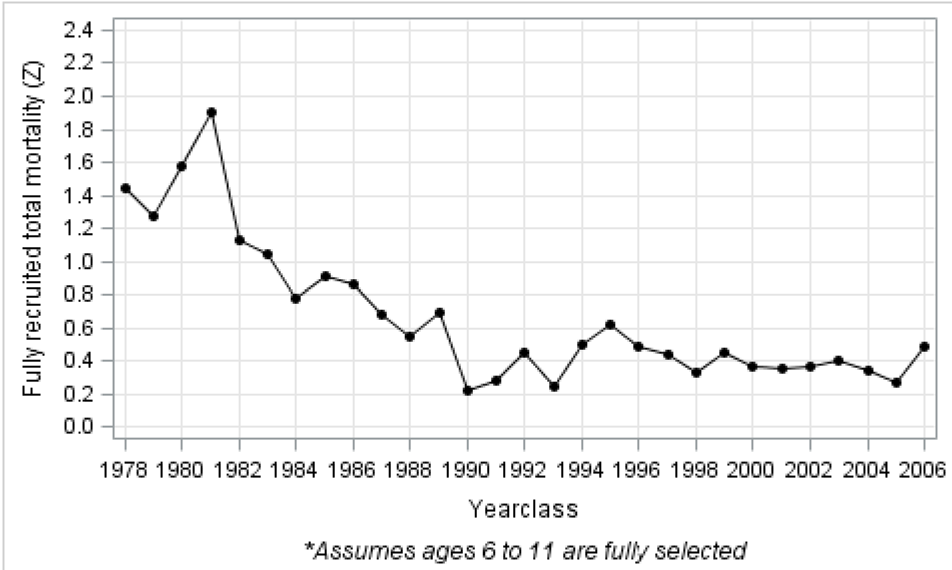


Figure A.88. Annual estimates of Gulf of Maine haddock total mortality (Z) as estimated from the year class curve analsyses for total catch for the 1978 to 2006 year classes.

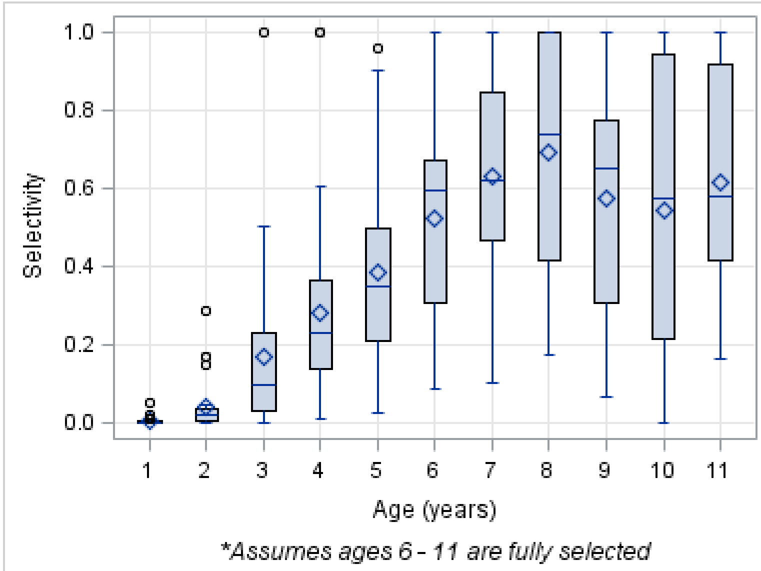
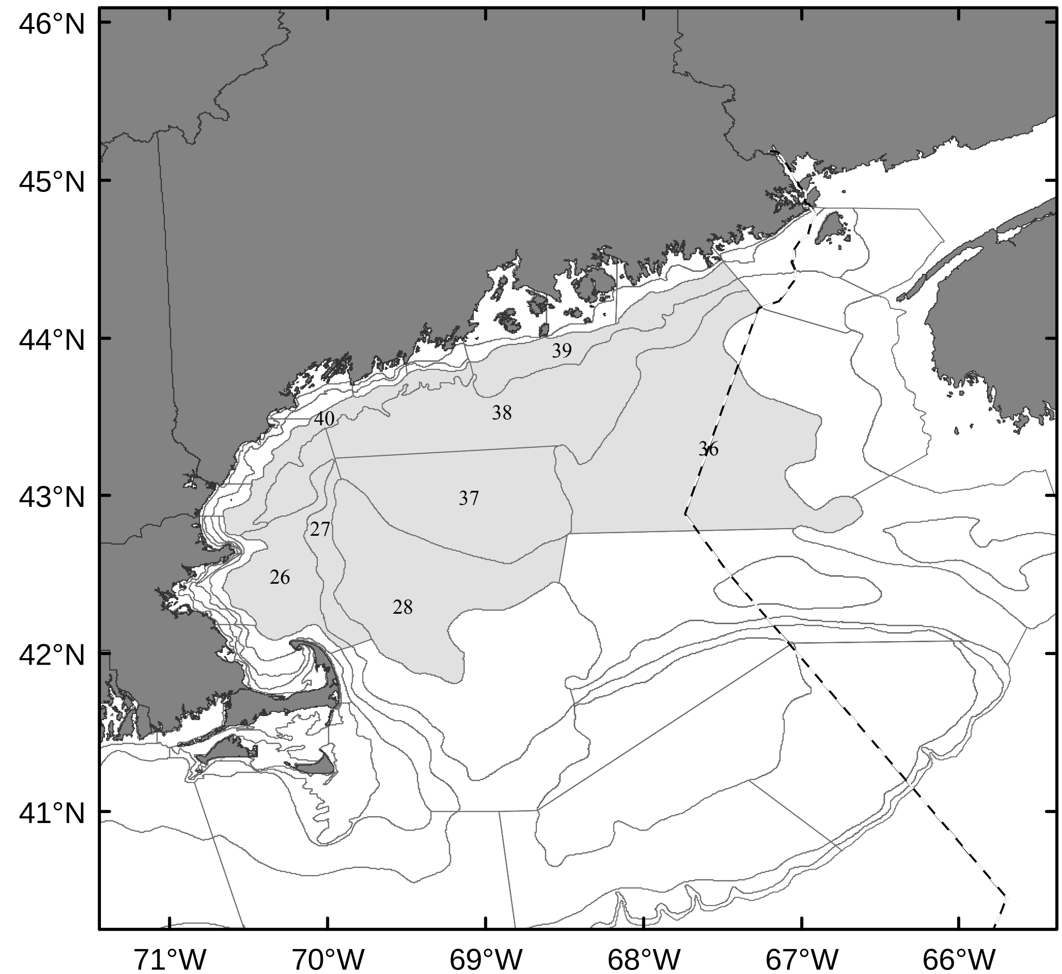
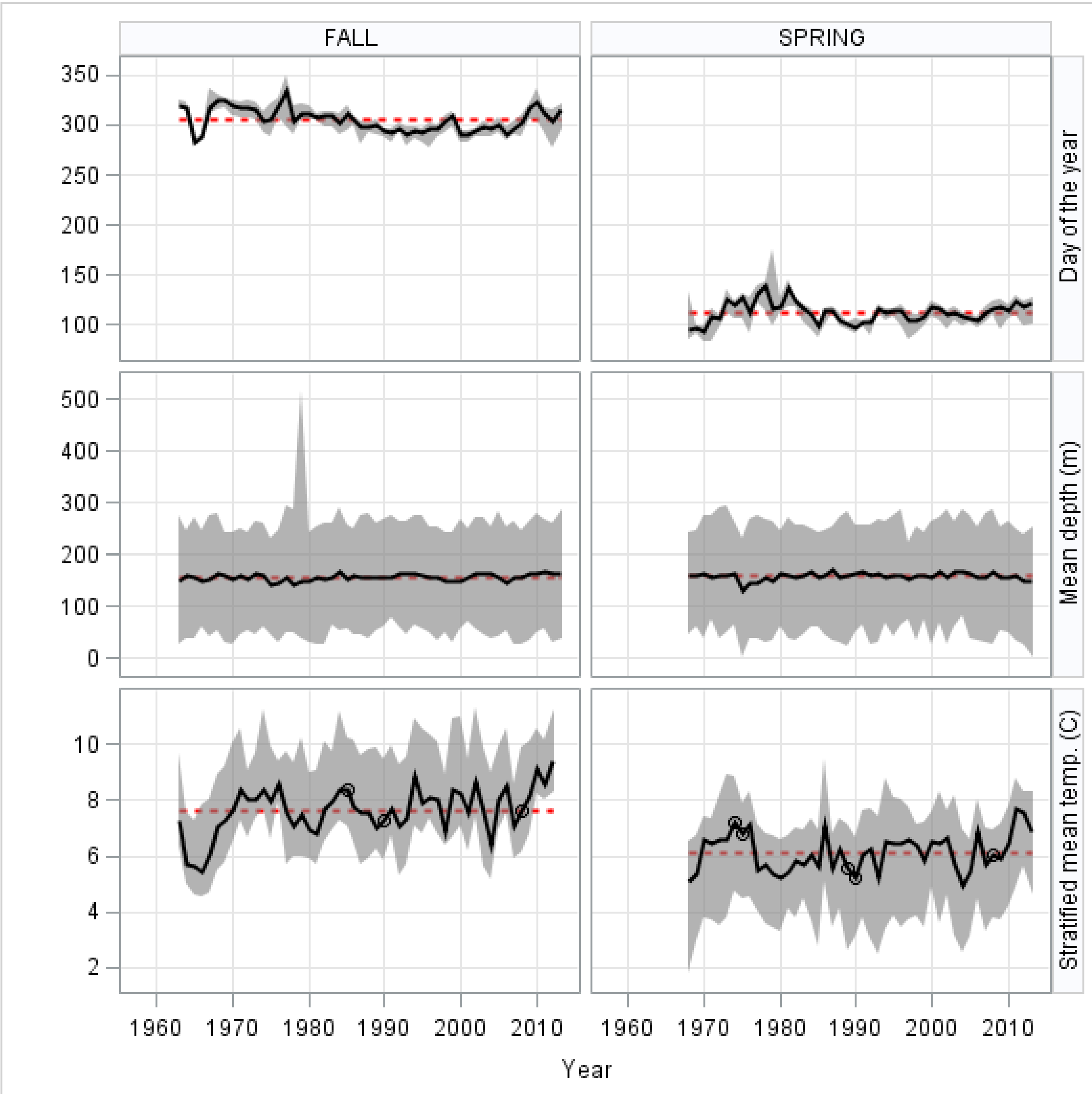


Figure A.89. Distribution of catch selectivity-at-age as estimated from the residuals fits to the Gulf of Maine total catch curve analysis.



Figue A.90. Map of the Notheast Fisheries Science Center (NEFSC) bottom trawl offshore survey strata used to construct NEFSC survey indices for Gulf of Maine haddock stock assessment (shaded grey).



Figue A.91. Mean day of the year, depth and bottom temperature for the Northeast Fisheries Science Center (NEFSC) spring and fall bottom trawl surveys in the Gulf of Maine. Shaded areas indicate the range between the minimum and maximum observation. *Day of the year is expressed as Julian days (e.g., January 1 is day 1 and December 31 is day 365/66). Years marked with circles in the mean temperature plot indicate years when not all survey stratum were sampled and therefore the mean temperature may not be representative of the entire survey area.*

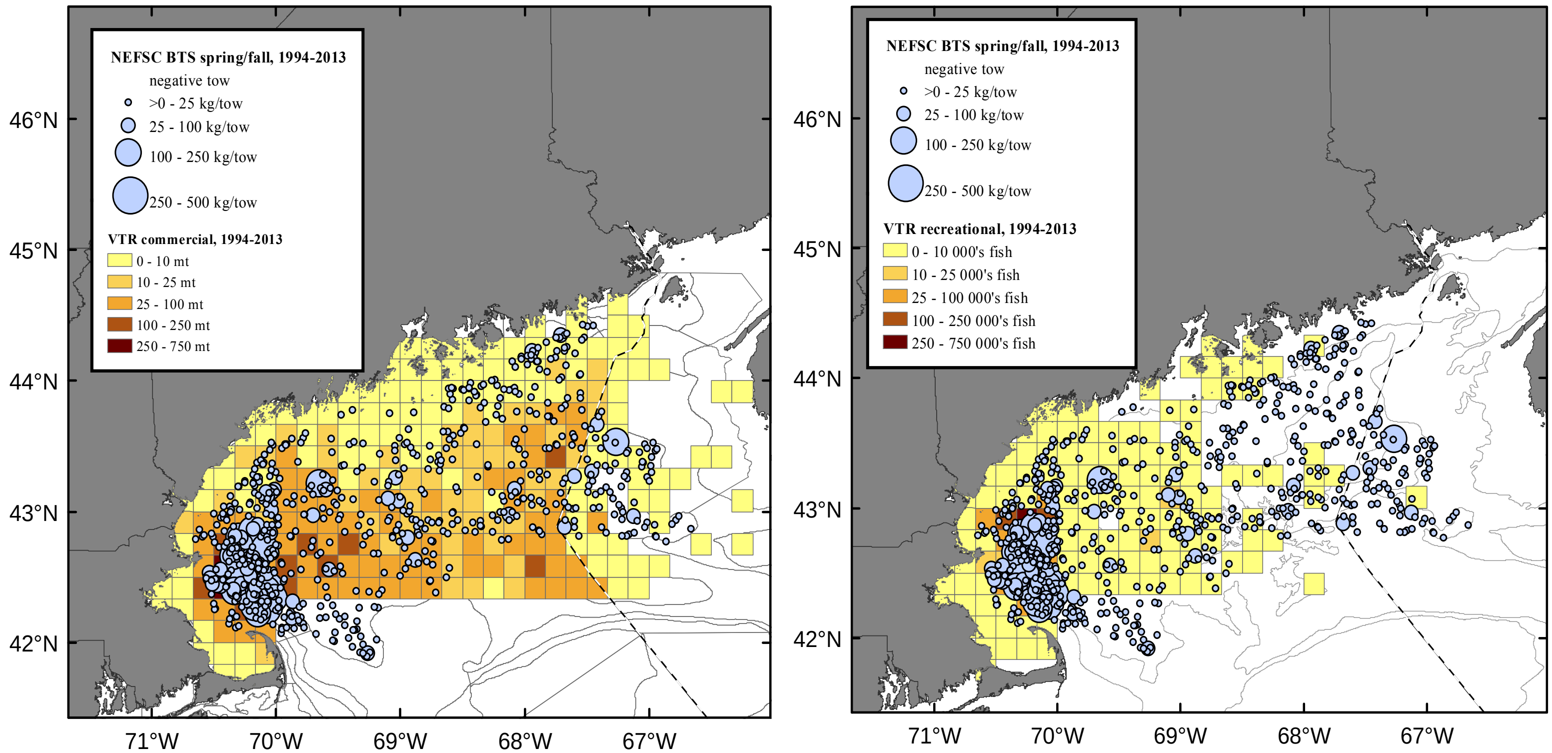


Figure A.92. Spatial overlap of survey catches (kg/tow) of Gulf of Maine haddock from the Northeast Fisheries Science Center (NEFSC) bottom trawl survey (spring and fall combined) and recreational and commercial landings. On the left, NEFSC survey catches from 1994 – 2013 are overlaid on the VTR-reported commercial landings binned to ten minute squares. On the right, NEFSC survey catches from 1994 – 2013 are overlaid on the VTR-reported recreational landings binned to ten minute squares. **Note the different units of measure between the commercial and recreational landings.*

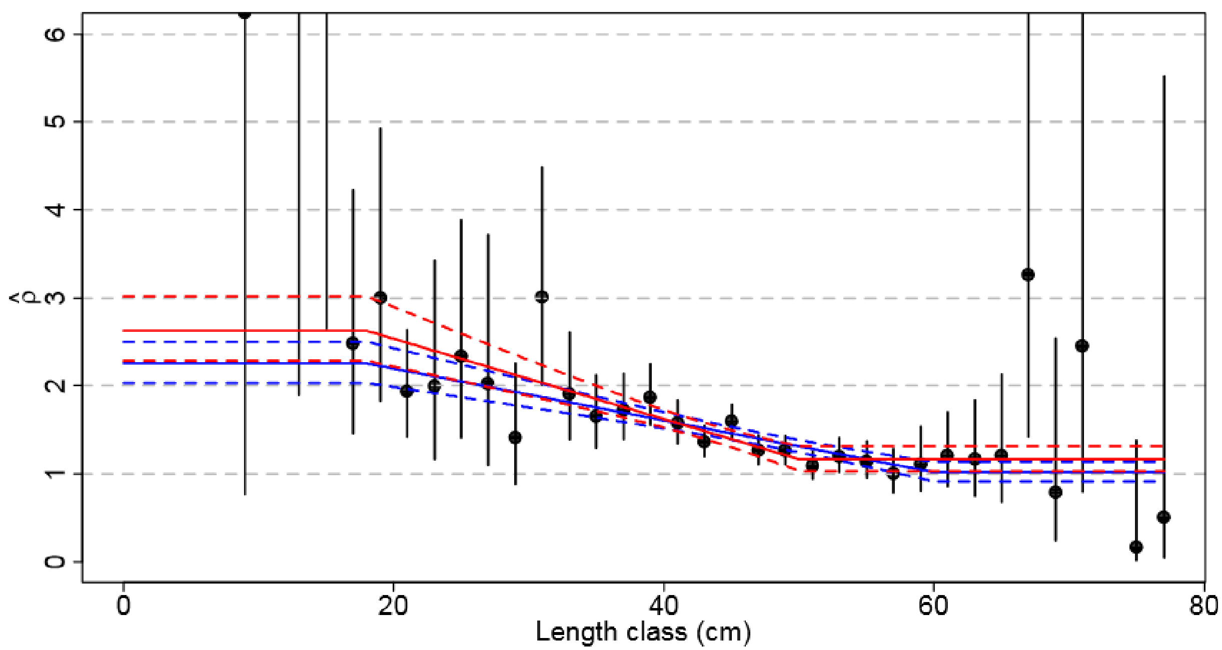


Figure A.93. Beta-binomial based estimates of calibration factors and corresponding 95% confidence intervals by length class (2 cm bins) for haddock. The black points and vertical bars represent results where different calibration factors are estimated for each length class.The blue lines represent results from a segmented regression model where the two points connecting the segments are known (18 and 60 cm) and the red lines represent results from a segmented regression model where the first point (18 cm) is known but the second is estimated. Segmented regression fits are based on data from fish ≥18 cm.

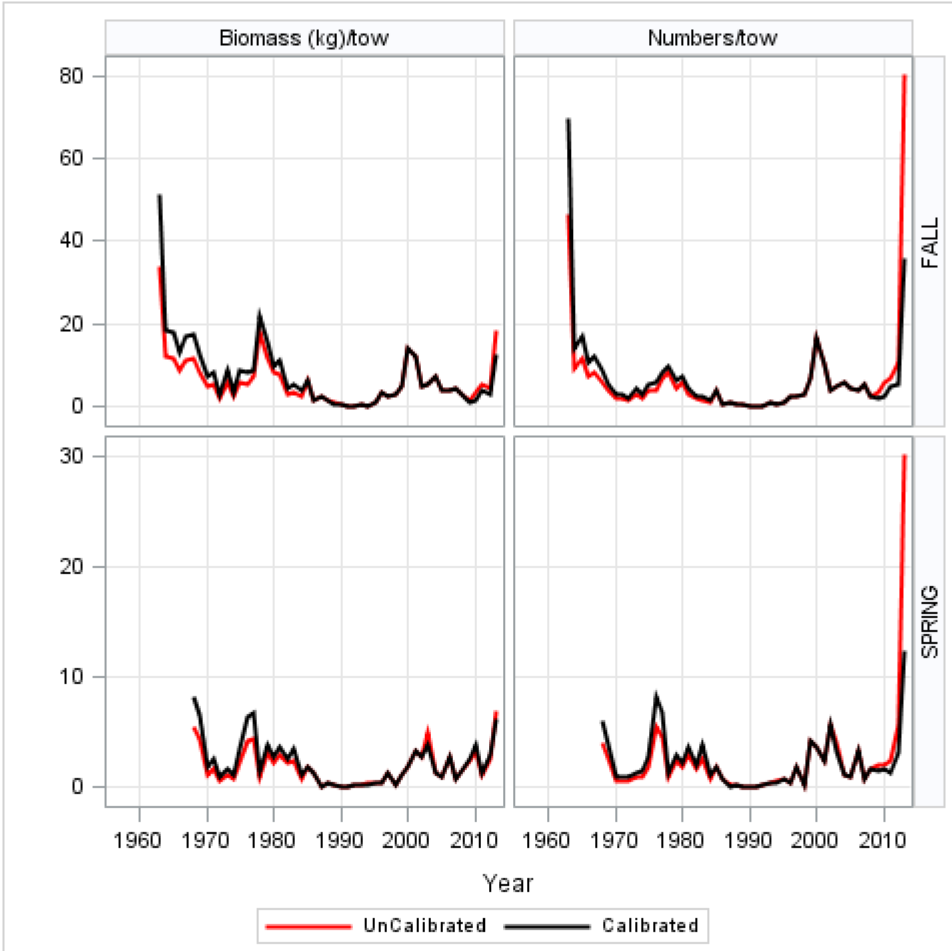


Figure A.94. Northeast Fisheries Science Center spring and fall survey indices of abundance (numbers/tow) and biomass (kg/tow) showing both raw (uncalibrated) and vessel, door and survey calibrated indices over time for Gulf of Maine haddock.

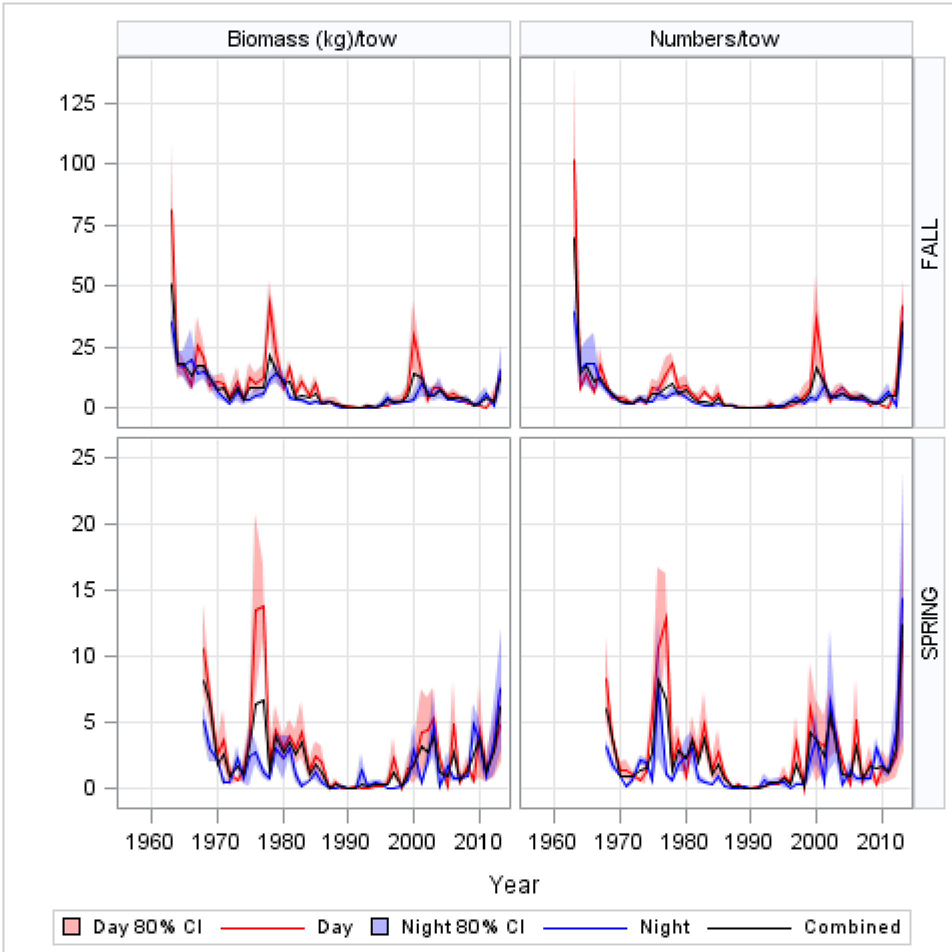


Figure A.95. Northeast Fisheries Science Center spring and fall survey indices of abundance (numbers/tow) and biomass (kg/tow) broken down by day- and night-only tows and the corresponding 80% confidence interval (CI) for Gulf of Maine haddock. The aggregate survey indices are shown in black.

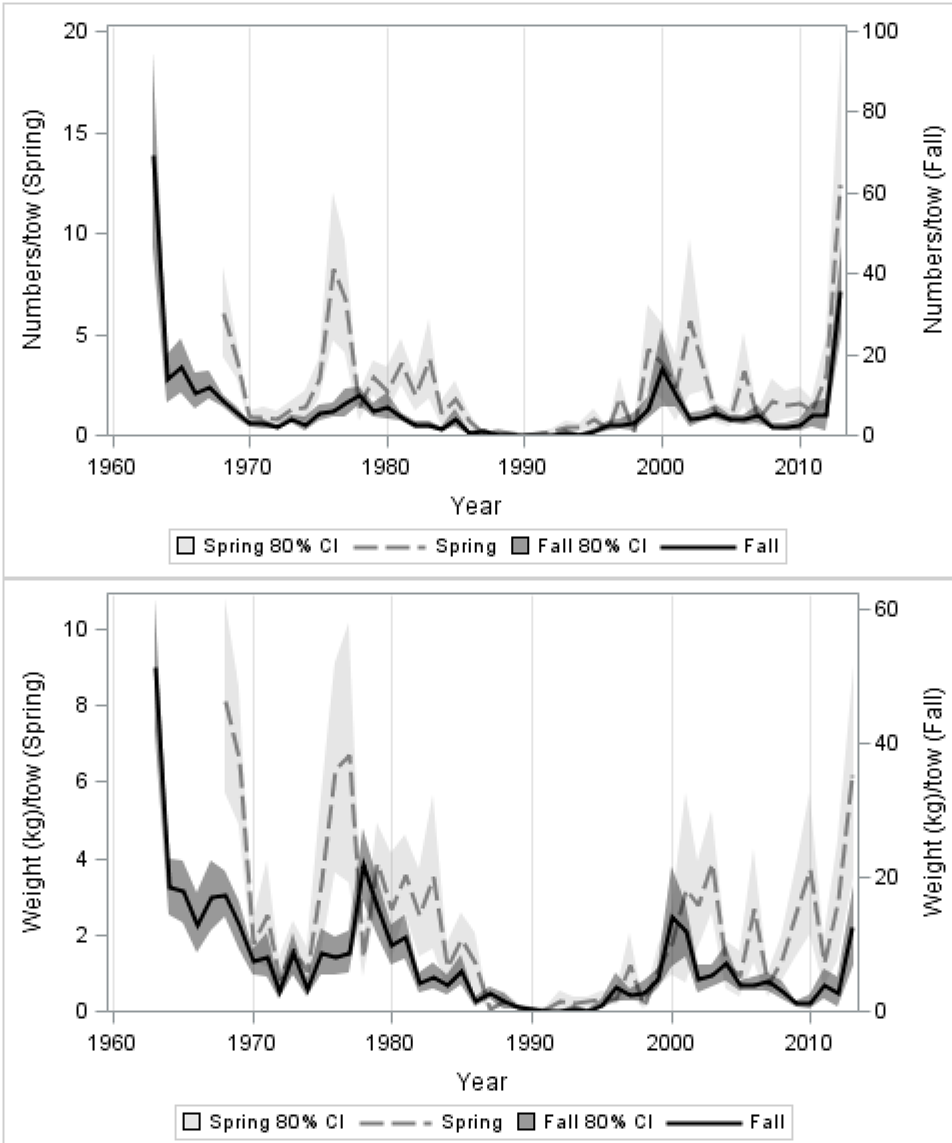


Figure A.96. Northeast Fisheries Science Center (NEFSC) spring and fall bottom trawl survey abundance (numbers/tow) and biomass (kg/tow) indices for Gulf of Maine haddock from 1963 to 2013. The TOGA tow criteria used for years 2009-2013. *Note that the spring survey did not begin until 1968.*

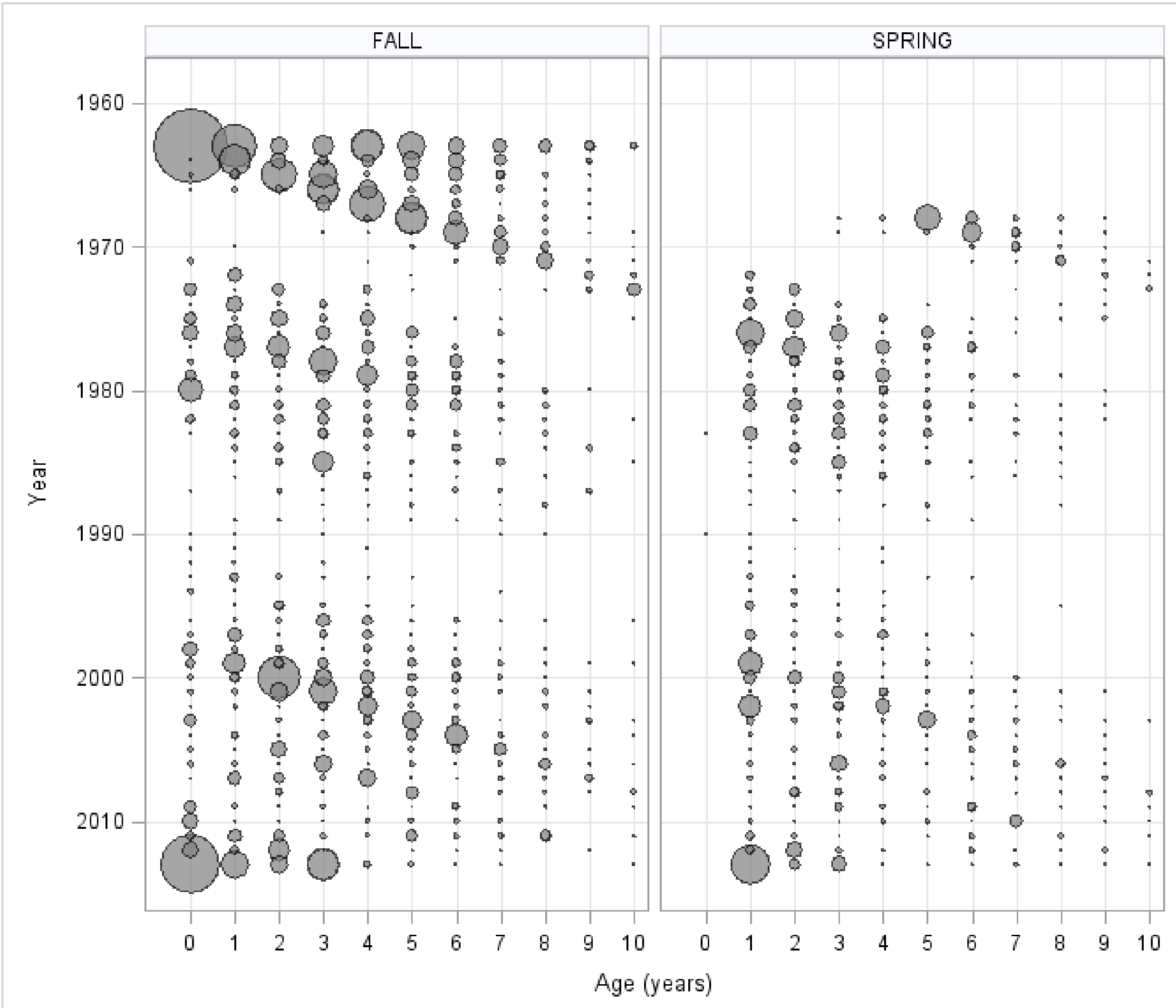


Figure A.97. NEFSC spring and fall bottom trawl survey Gulf of Maine haddock abundance indices-at-age from 1963 to 2013. *Note that the spring survey did not begin until 1968.*

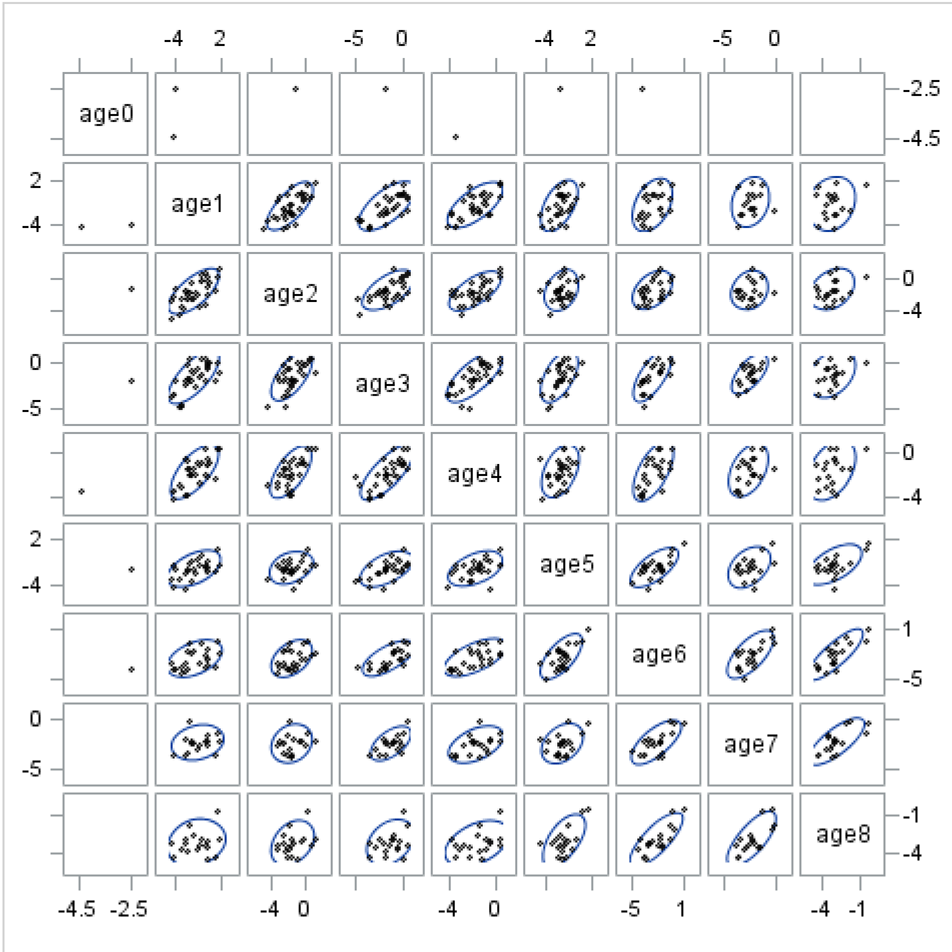


Figure A.98. Scatter plots showing the level of agreement between Northeast Fisheries Science Center (NEFSC) spring bottom trawl survey Gulf of Maine haddock indices-at-age (log transformed) on a cohort basis. The 80% confidence ellipses are shown.

Figure A.99. Scatter plots showing the level of agreement between Northeast Fisheries Science Center (NEFSC) fall bottom trawl survey Gulf of Maine haddock indices-at-age (log transformed) on a cohort basis. The 80% confidence ellipses are shown.

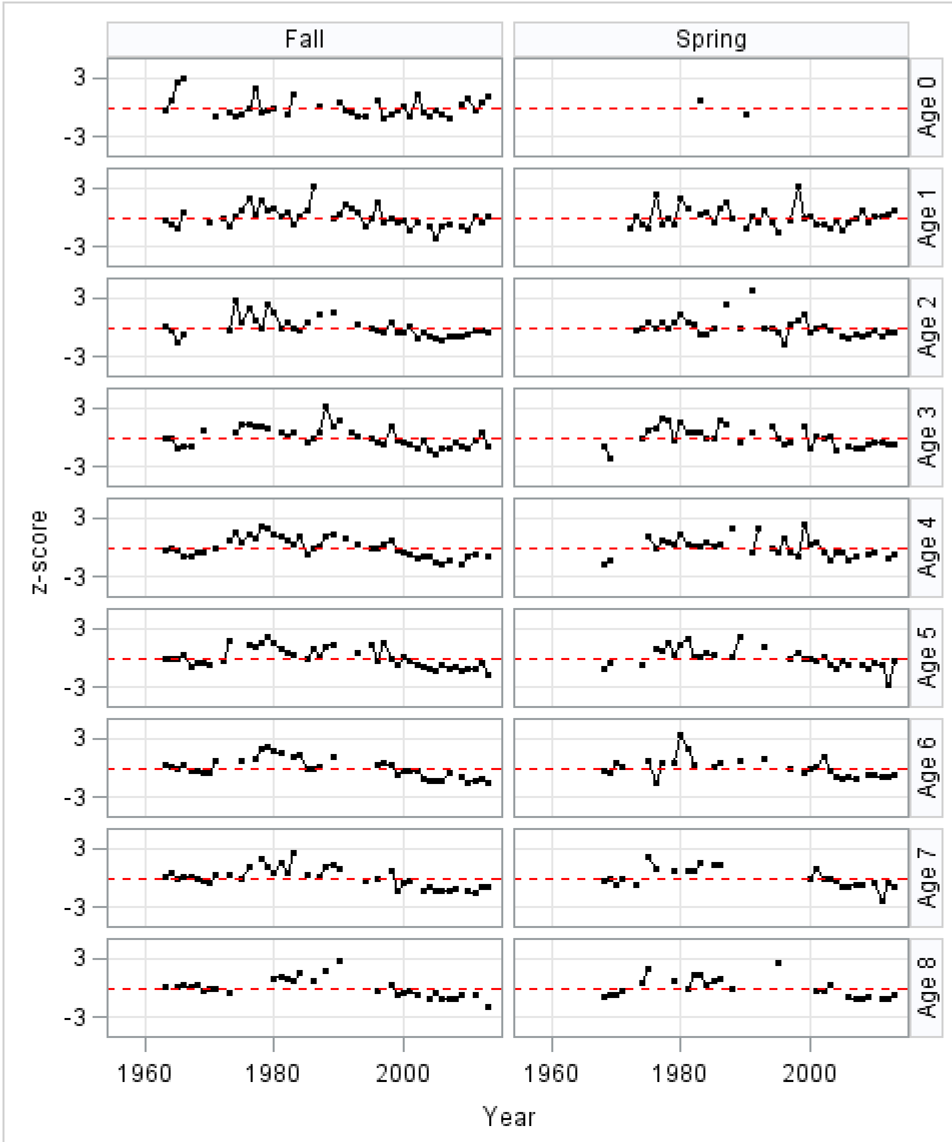


Figure A.100. Average weights-at-age of Gulf of Maine haddock age 0 to 8 from 1963 to 2013. Survey weights are based on the average weights-at-age of haddock sampled from the Northeast Fisheries Science Center spring and fall bottom trawl survey. Average weights are presented as z-scores ([*x*-μ]/σ).

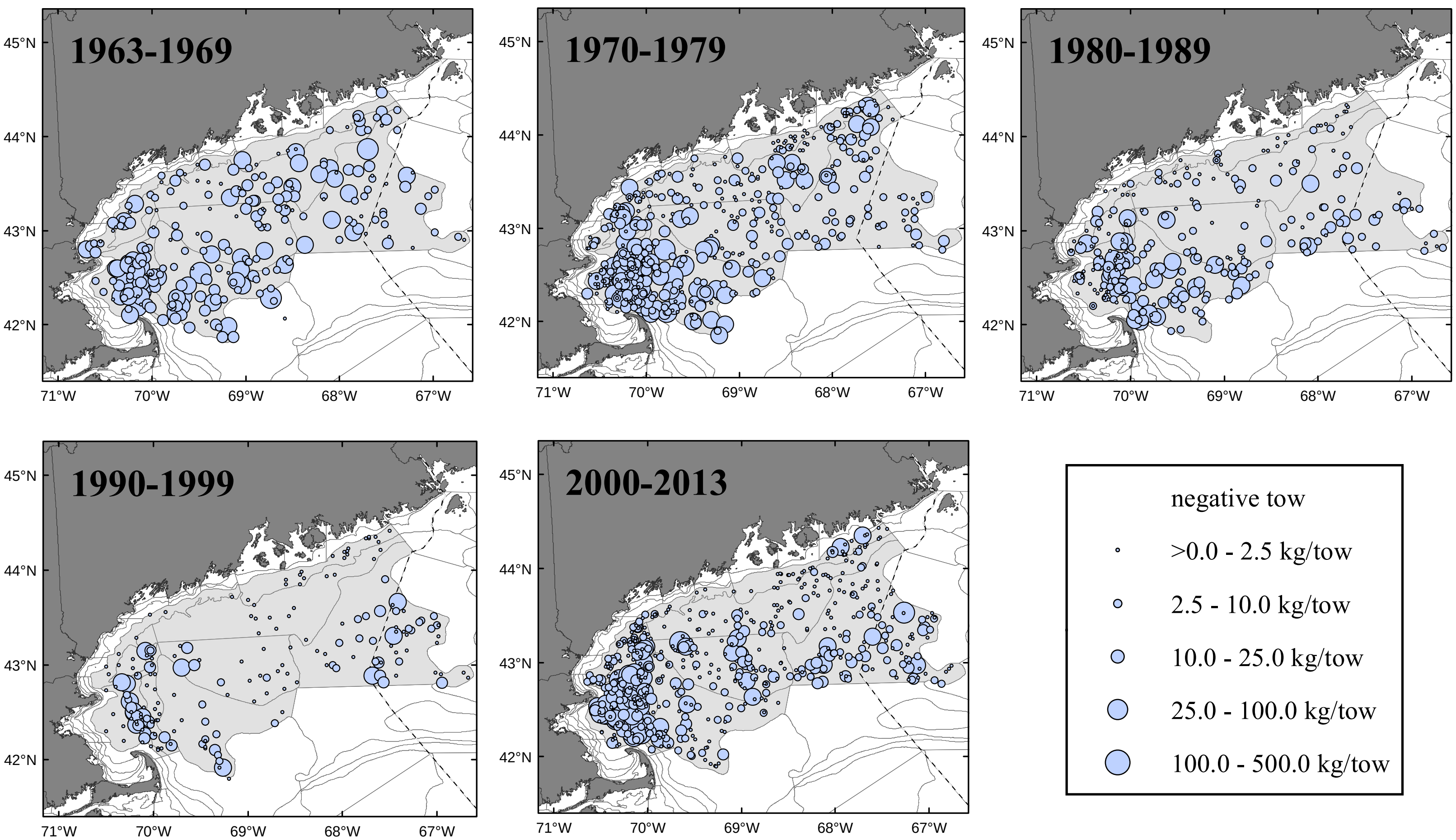


Figure A.101. Spatial distribution of Gulf of Maine haddock catches (numbers/tow) from the Northeast Fisheries Science Center spring bottom trawl survey from 1963 – 2013.

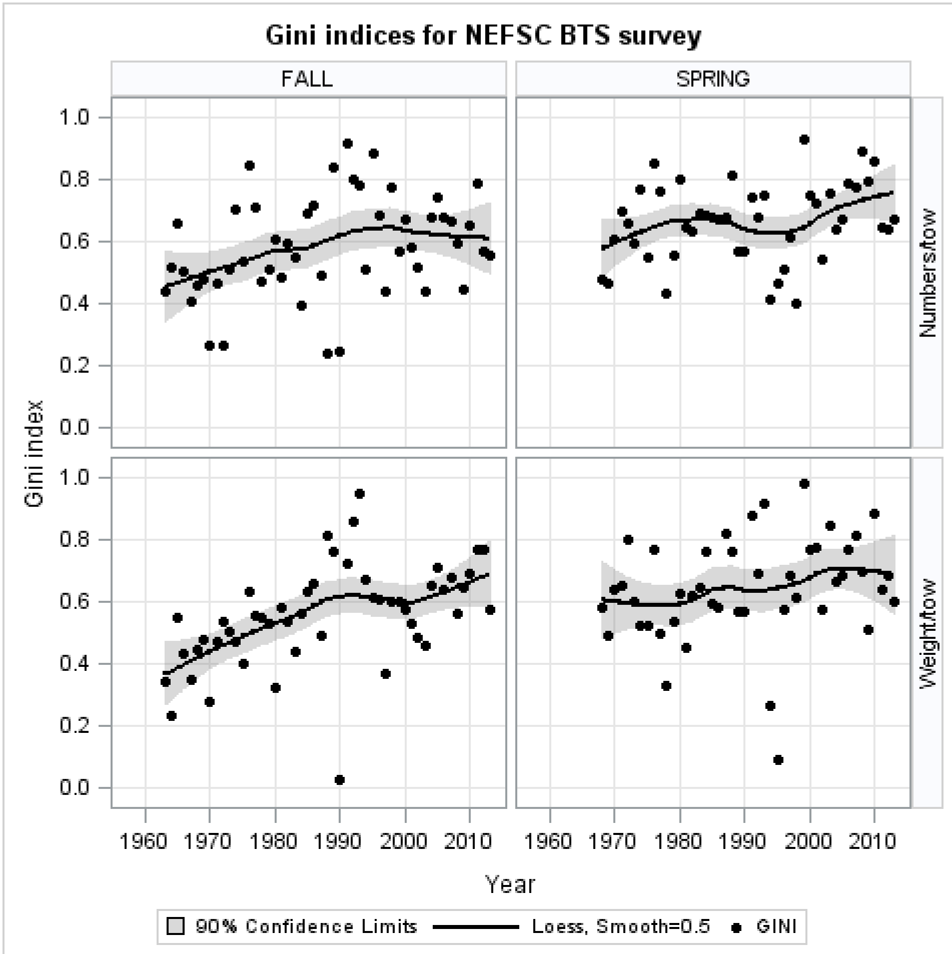


Figure A.102. Gini indices for Gulf of Maine haddock from the Northeast Fisheries Science Center (NEFSC) fall and spring bottom trawl surveys in terms of abundance (numbers/tow) and biomass (kg/tow). A LOESS smooth has been fit to the data with smoothing parameter of 0.5. The LOESS smooth is shown by the solid black line along with the corresponding 90% confidence interval.

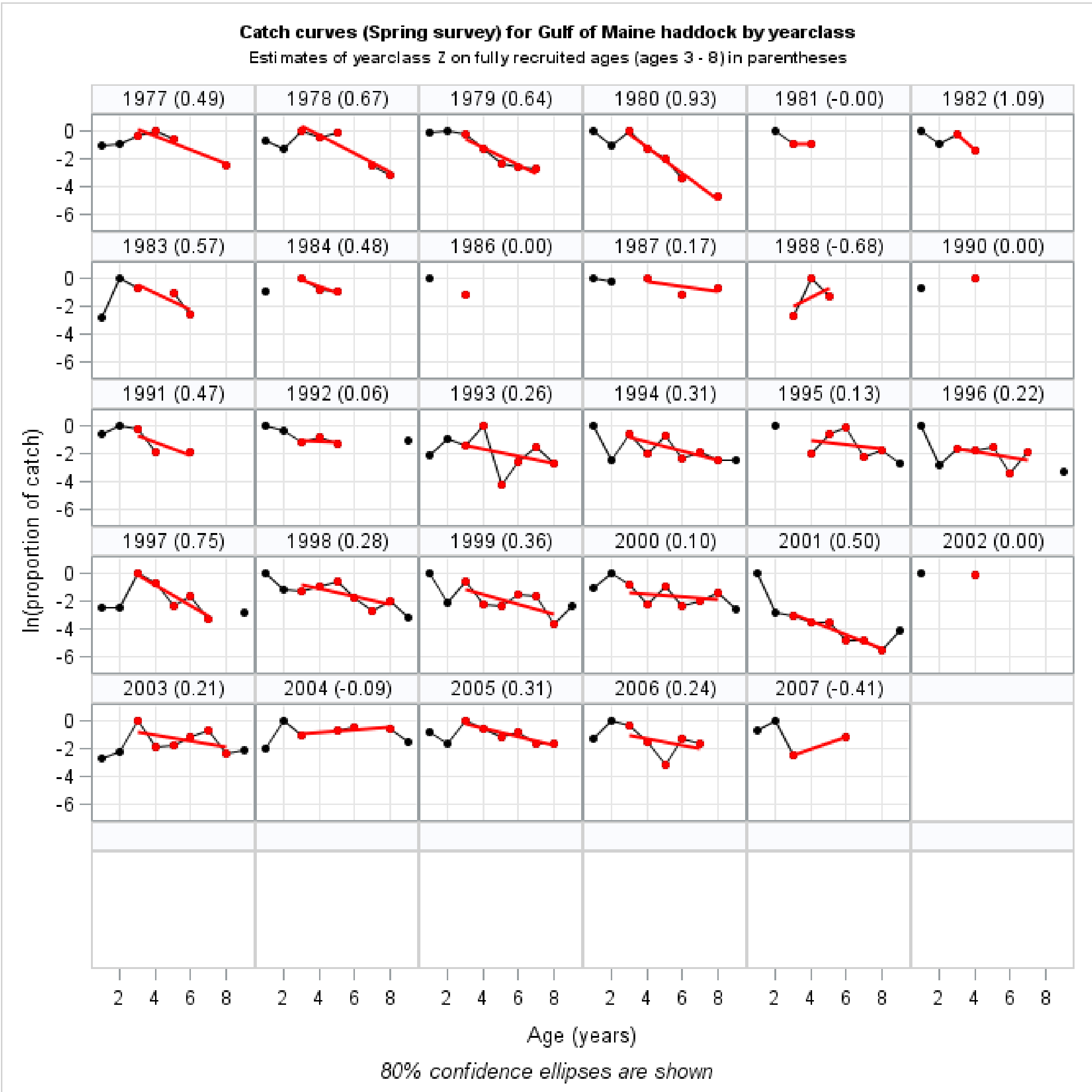


Figure A.103. Gulf of Maine haddock year class curves computed on ages 3-8 (red circles) log-transformed Northeast Fisheries Science Center (NEFSC) spring bottom trawl survey abundance (numbers/tow) indices. The corresponding slope of each regression line is shown next to the year class label above each plot.

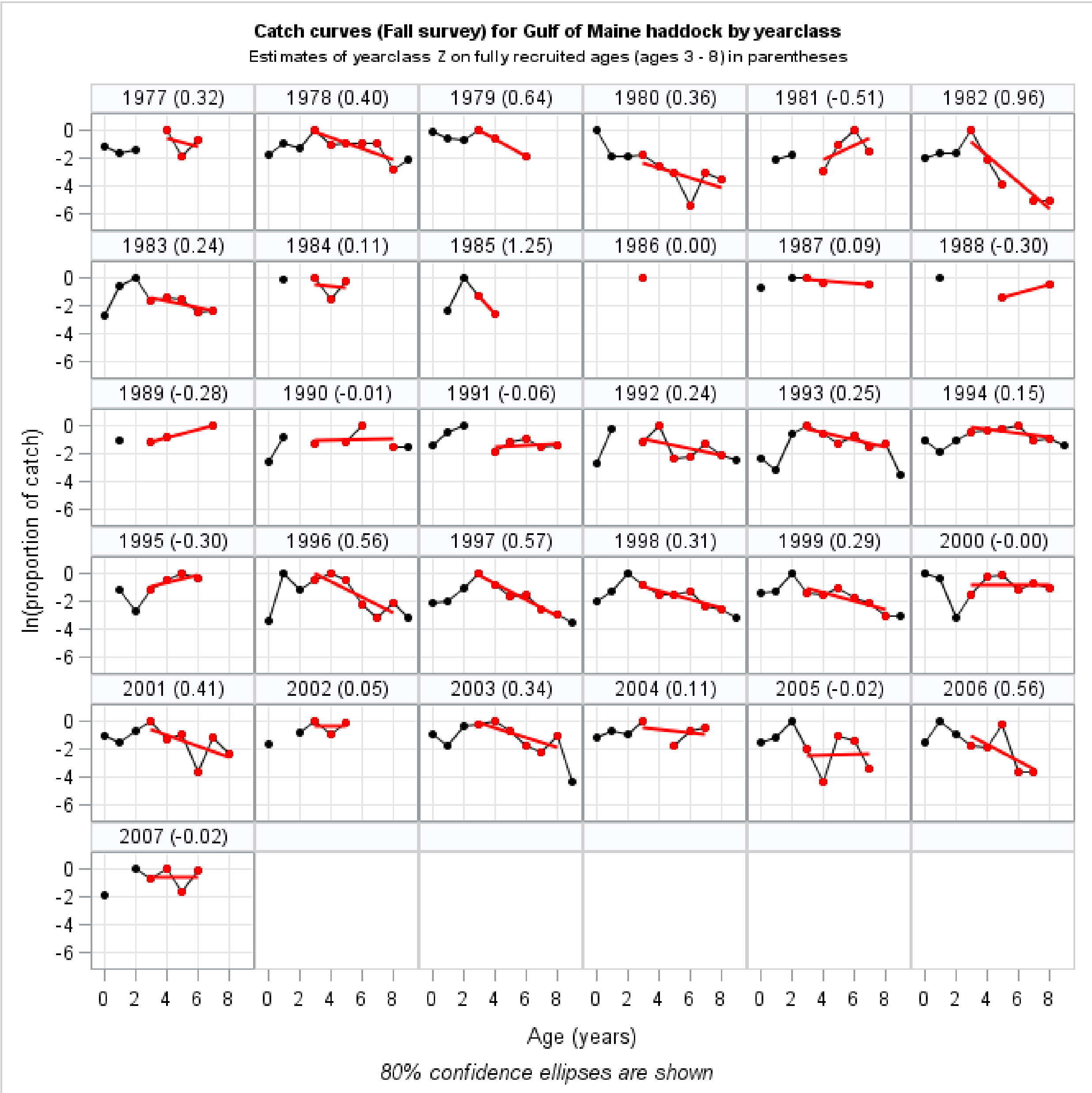


Figure A.104. Gulf of Maine haddock year class curves computed on ages 3-8 (red circles) log-transformed Northeast Fisheries Science Center (NEFSC) fall bottom trawl survey abundance (numbers/tow) indices. The corresponding slope of each regression line is shown next to the year class label above each plot.

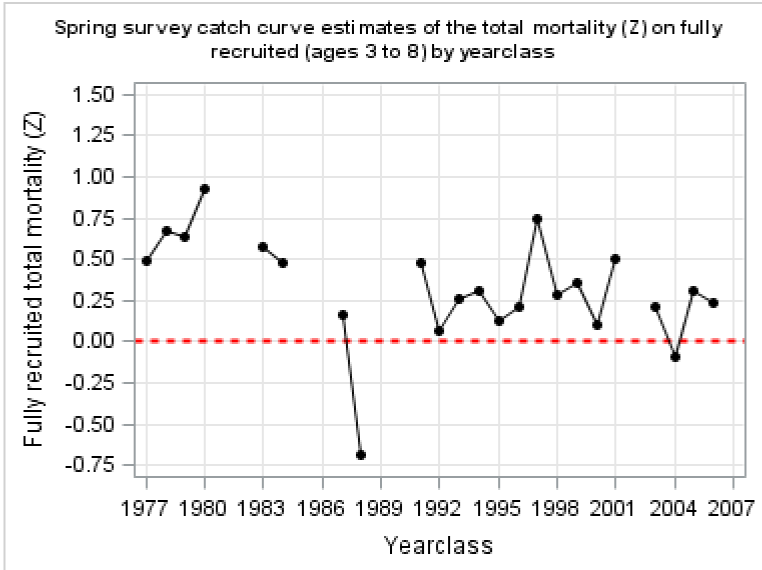


Figure A.105. Plots of the annual estimates of Gulf of Maine haddock total mortality (Z) as estimated from the year class curve analysis for total catch and Northeast Fisheries Science Center (NEFSC) spring bottom trawl surveys.

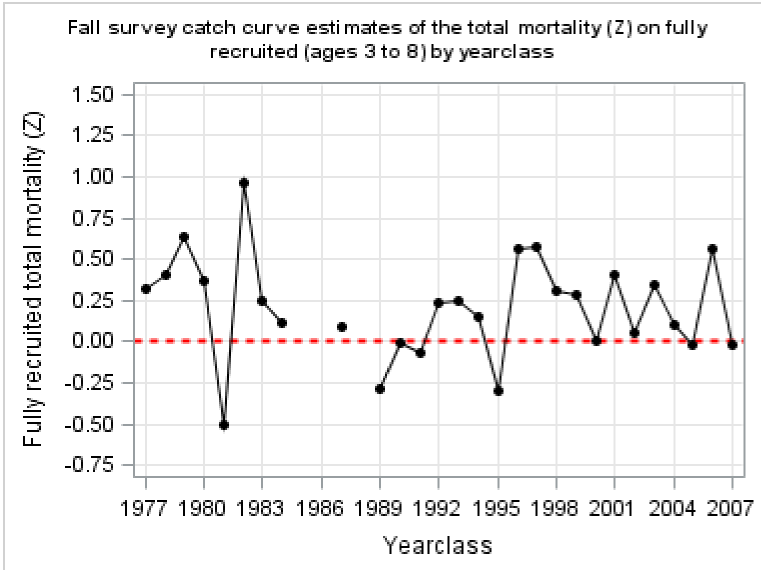


Figure A.106. Plots of the annual estimates of Gulf of Maine haddock total mortality (Z) as estimated from the year class curve analysis for total catch and Northeast Fisheries Science Center (NEFSC) fall bottom trawl surveys.

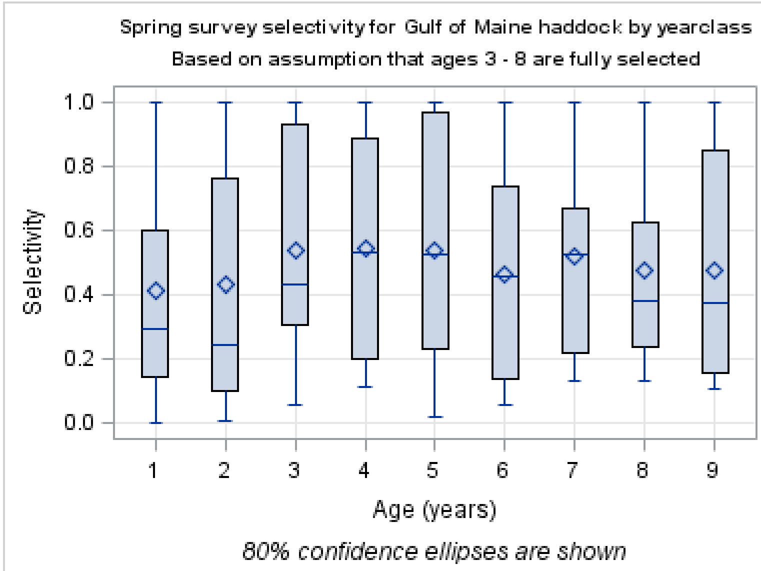


Figure A.107. Distribution of catch selectivity-at-age as estimated from the residuals fits to the Northeast Fisheries Science Center (NEFSC) spring survey indices catch curve analysis for Gulf of Maine haddock.

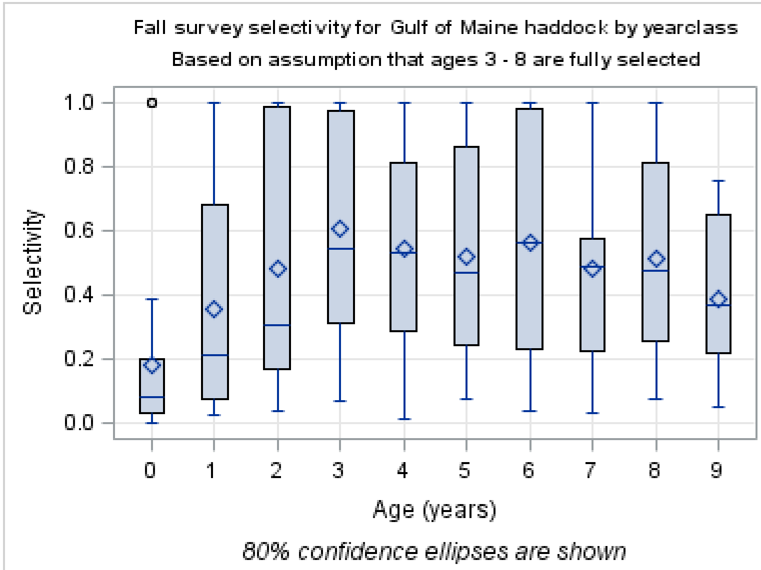


Figure A.108. Distribution of catch selectivity-at-age as estimated from the residuals fits to the Northeast Fisheries Science Center (NEFSC) fall survey indices catch curve analysis for Gulf of Maine haddock.

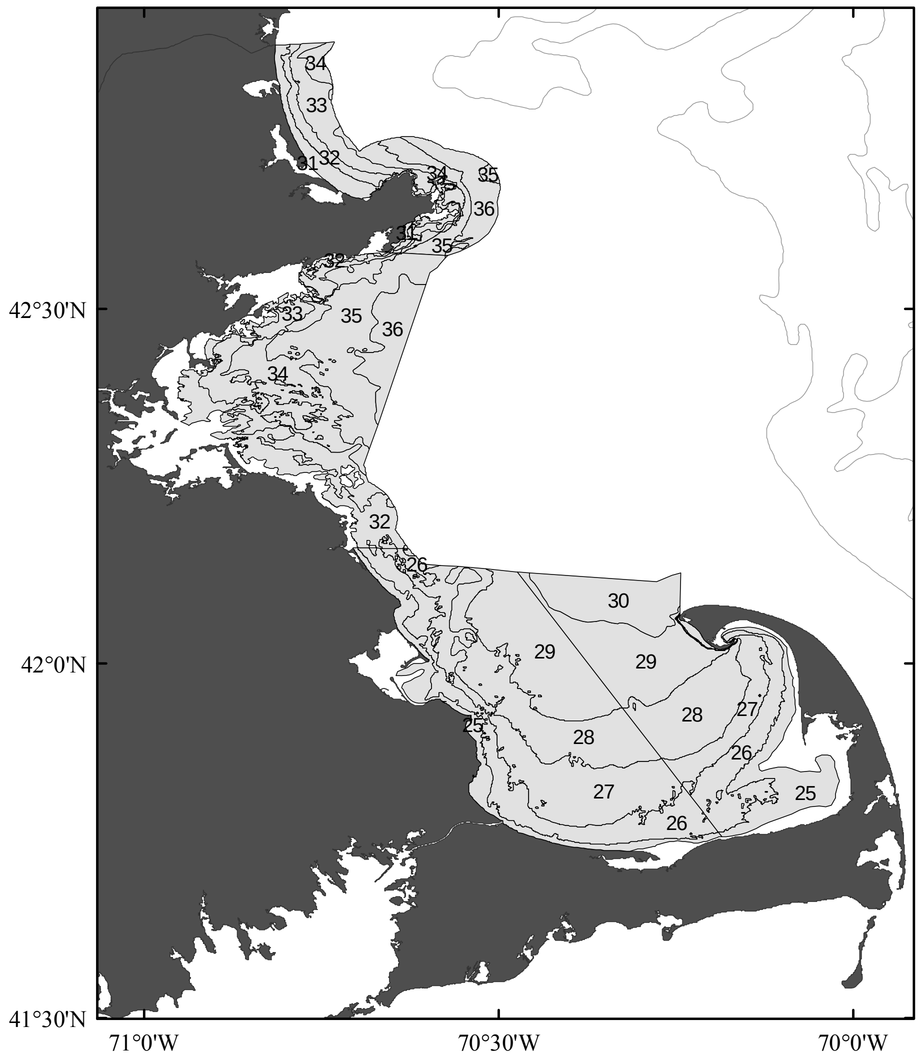


Figure A.109. Map of the Massachusetts Deparment of Marine Fisheries (MADMF) bottom trawl survey strata used to construct MADMF survey indices for Gulf of Maine haddock stock assessment (shaded grey).

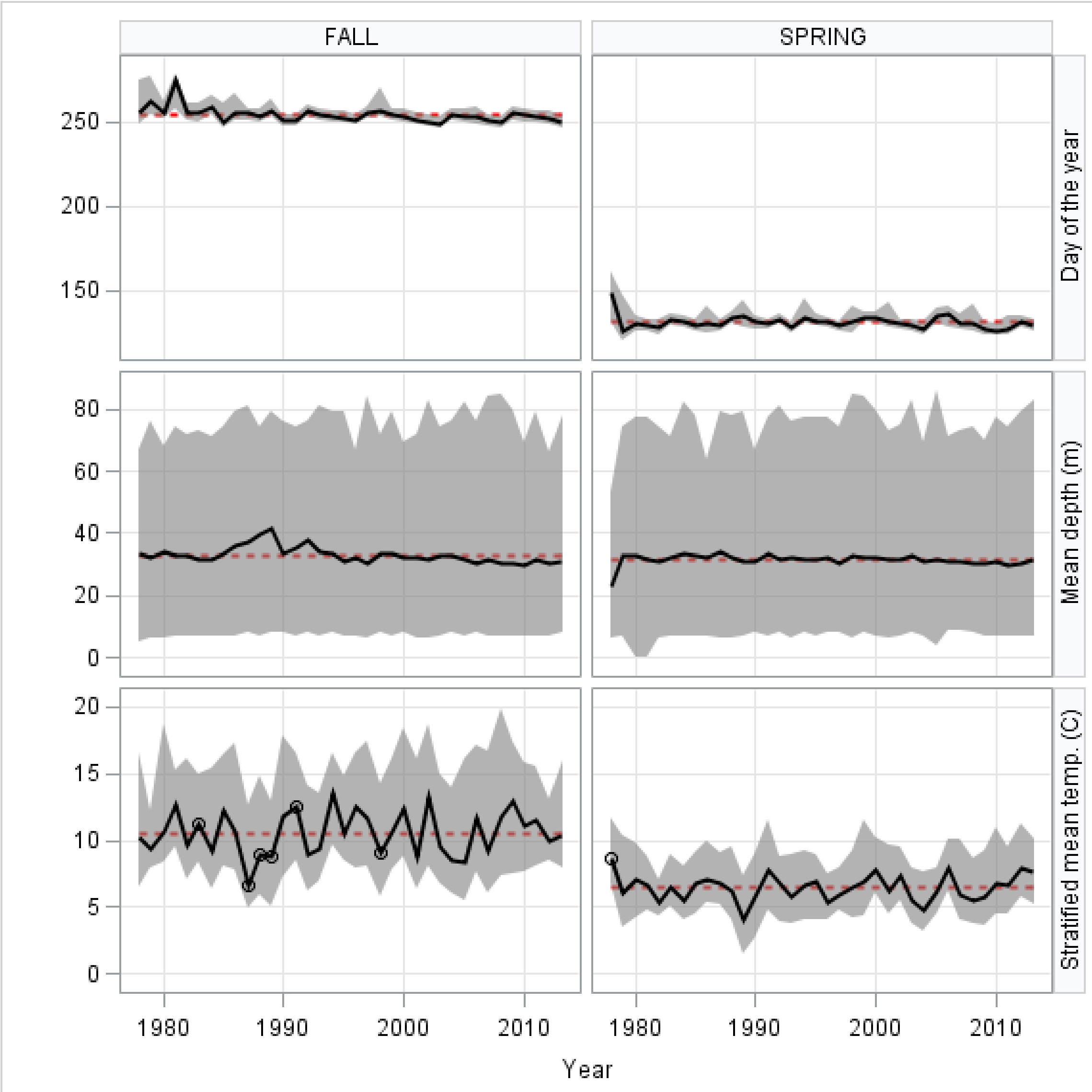


Figure A.110. Mean day of the year, depth and bottom temperature for the Massachusetts Department of Marine Fisheries (MADMF) spring and fall bottom trawl surveys in the Gulf of Maine. Shaded areas indicate the range between the minimum and maximum observation. *Day of the year is expressed as Julian days (e.g., January 1 is day 1 and December 31 is day 365/66). Years marked with circles in the mean temperature plot indicate years when not all survey stratum were sampled and therefore the mean temperature may not be representative of the entire survey area.*

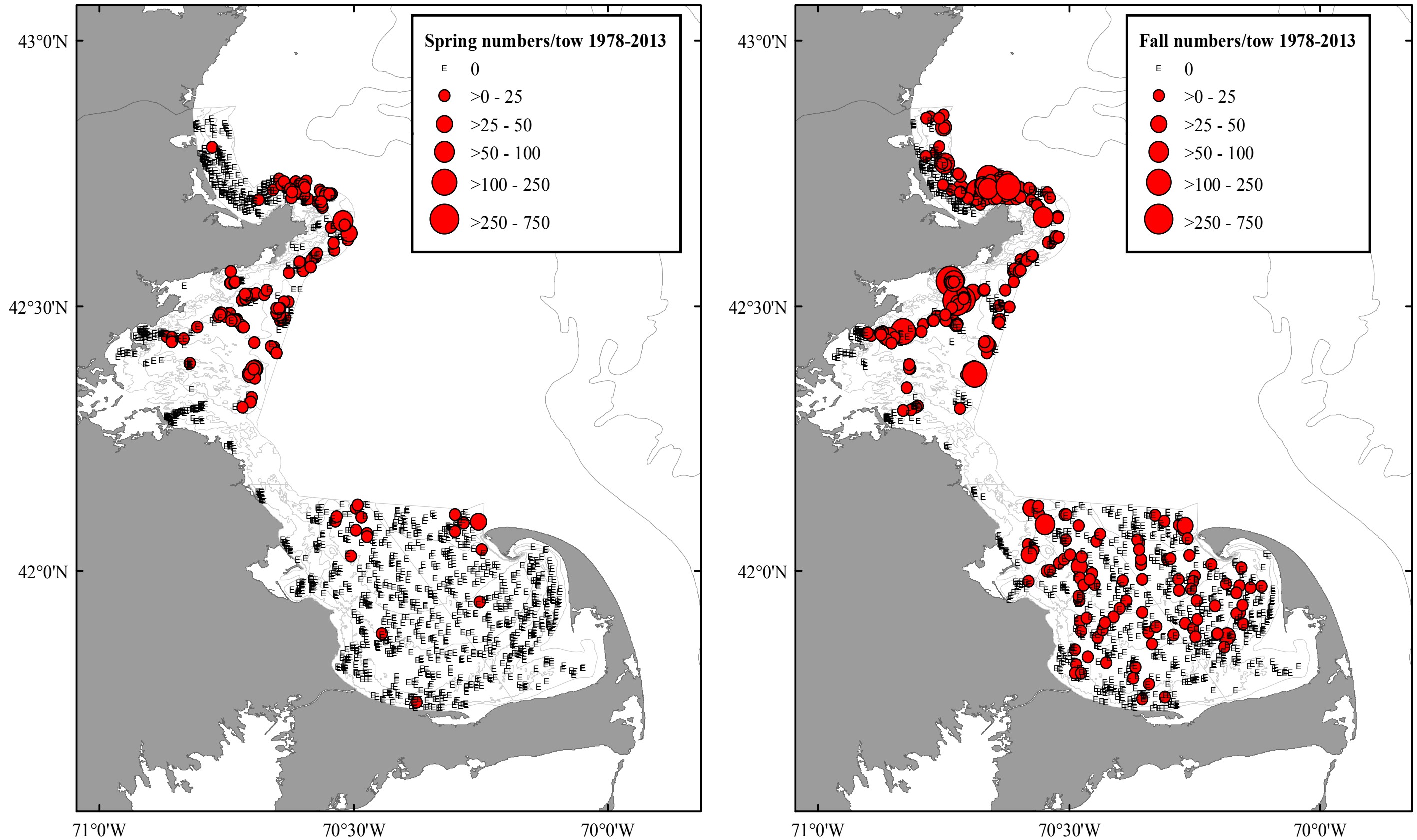


Figure A.111. Spatial distribution of Gulf of Maine haddock catches (numbers/tow) from the the Massachusetts Department of Marine Fisheries (MADMF) spring and fall bottom trawl surveys from 1978 – 2013.

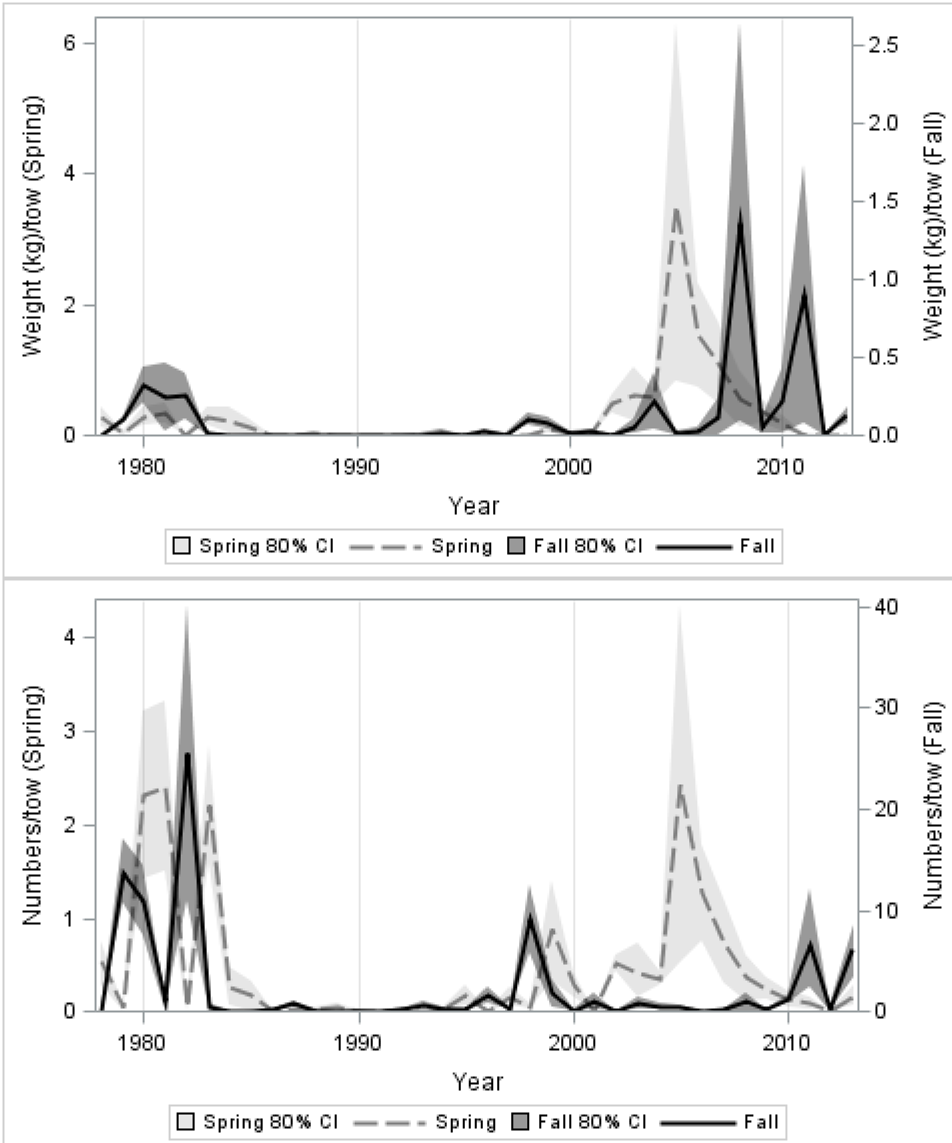


Figure A.112. Massachusetts Department of Marine Fisheries (MADMF) spring and fall bottom trawl survey Gulf of Maine haddock biomass (kg/tow) and abundance (numbers/tow) indices from 1978 to 2013.

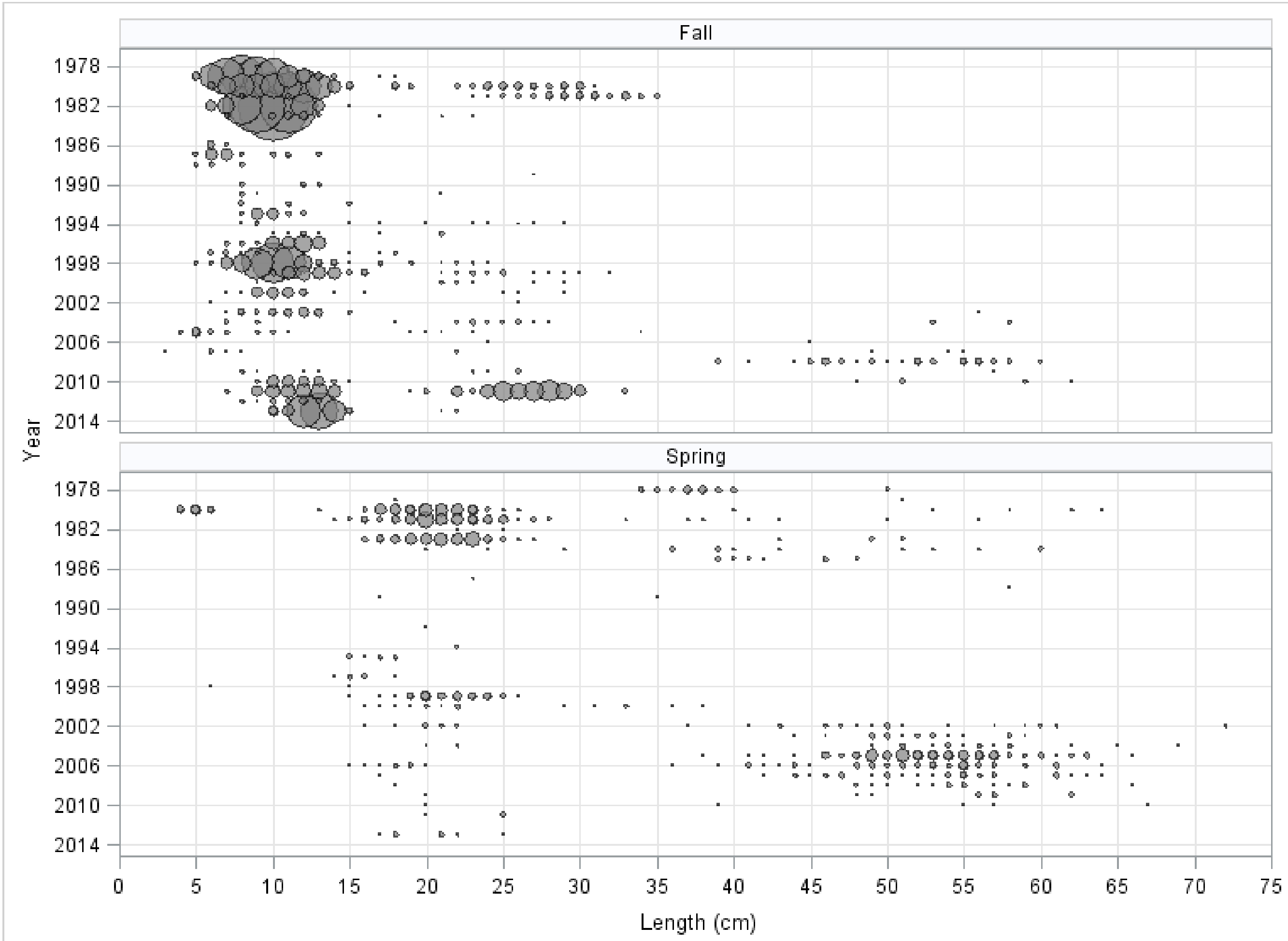


Figure A.113. Gulf of Maine haddock lengths-at-age from the Massachusetts Department of Marine Fisheries (MADMF) spring and fall bottom trawl surveys from 1978-2013.

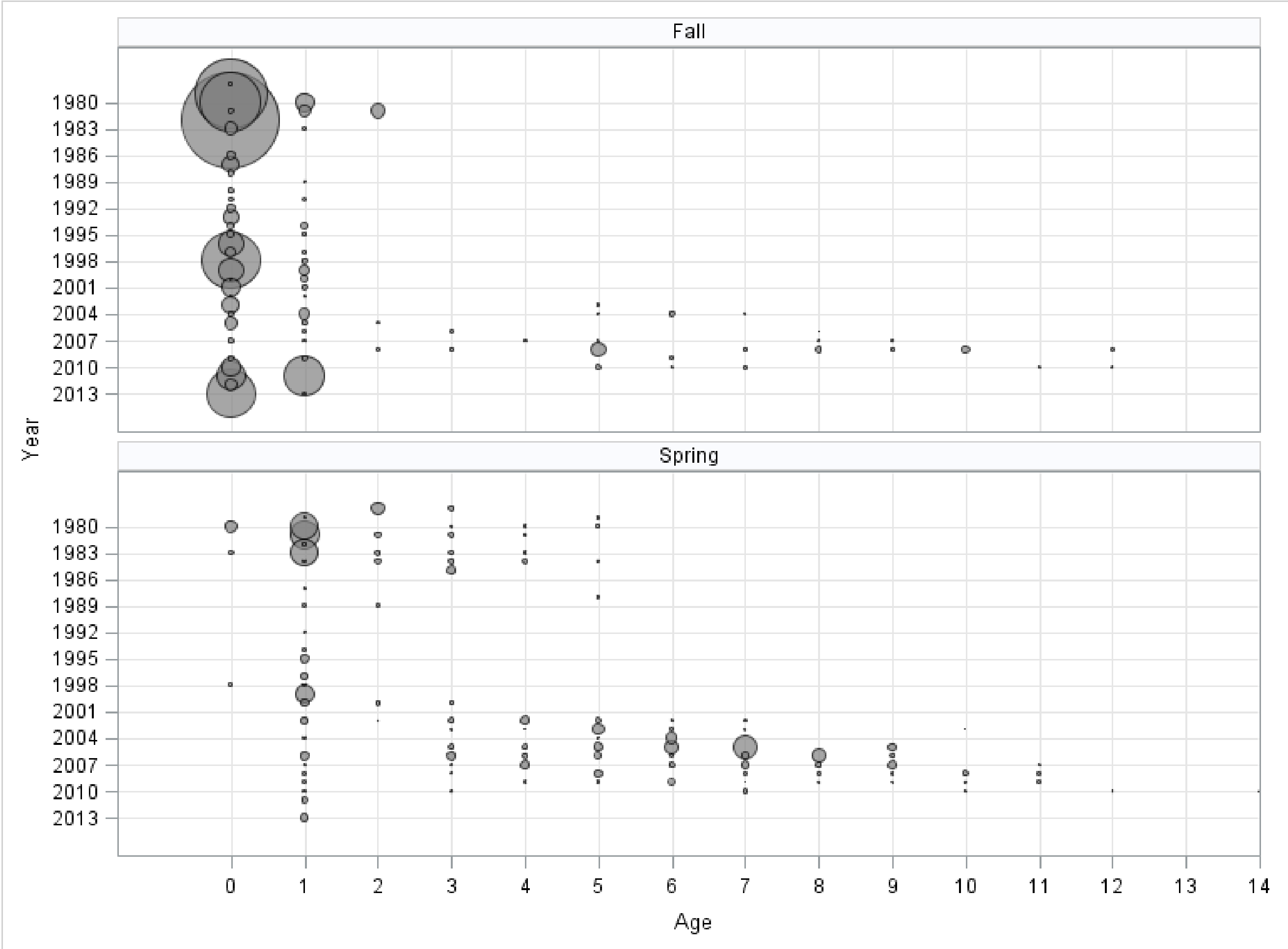


Figure A.114. Gulf of Maine haddock numbers-at-age from Massachusetts Department of Marine Fisheries (MADMF) spring and fall bottom trawl survey from 1978 to 2013. *Note that indices-at-age were constructed from age-length information borrowed from the Northeast Fisheries Science Center survey data.*

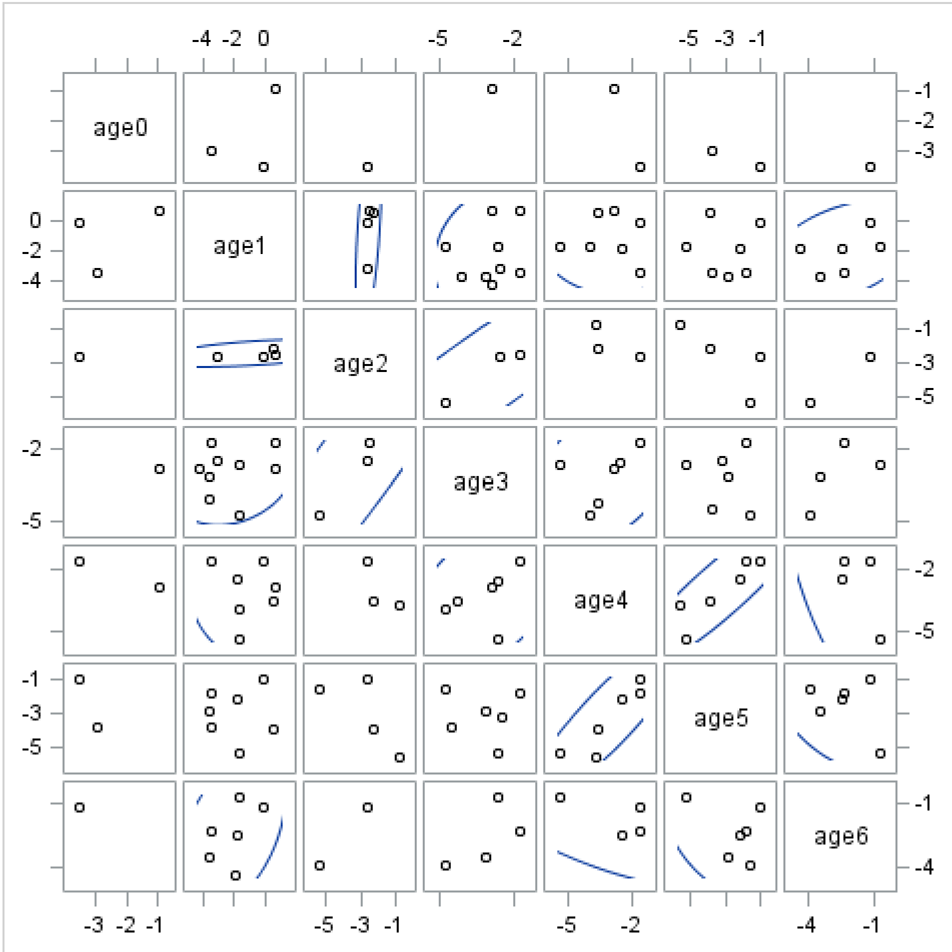


Figure A.115. Scatter plots showing the level of agreement between Massachusetts Department of Marine Fisheries (MADMF) spring bottom trawl survey Gulf of Maine haddock indices-at-age (log transformed) on a cohort basis. The 80% confidence ellipses are shown.

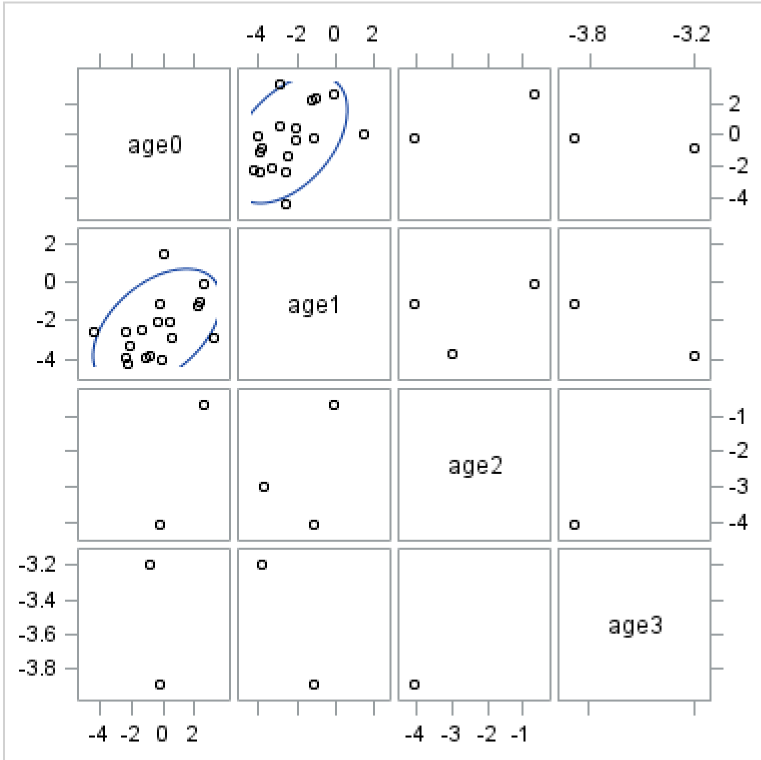


Figure A.116. Scatter plots showing the level of agreement between Massachusetts Department of Marine Fisheries (MADMF) fall bottom trawl survey Gulf of Maine haddock indices-at-age (log transformed) on a cohort basis. The 80% confidence ellipses are shown.

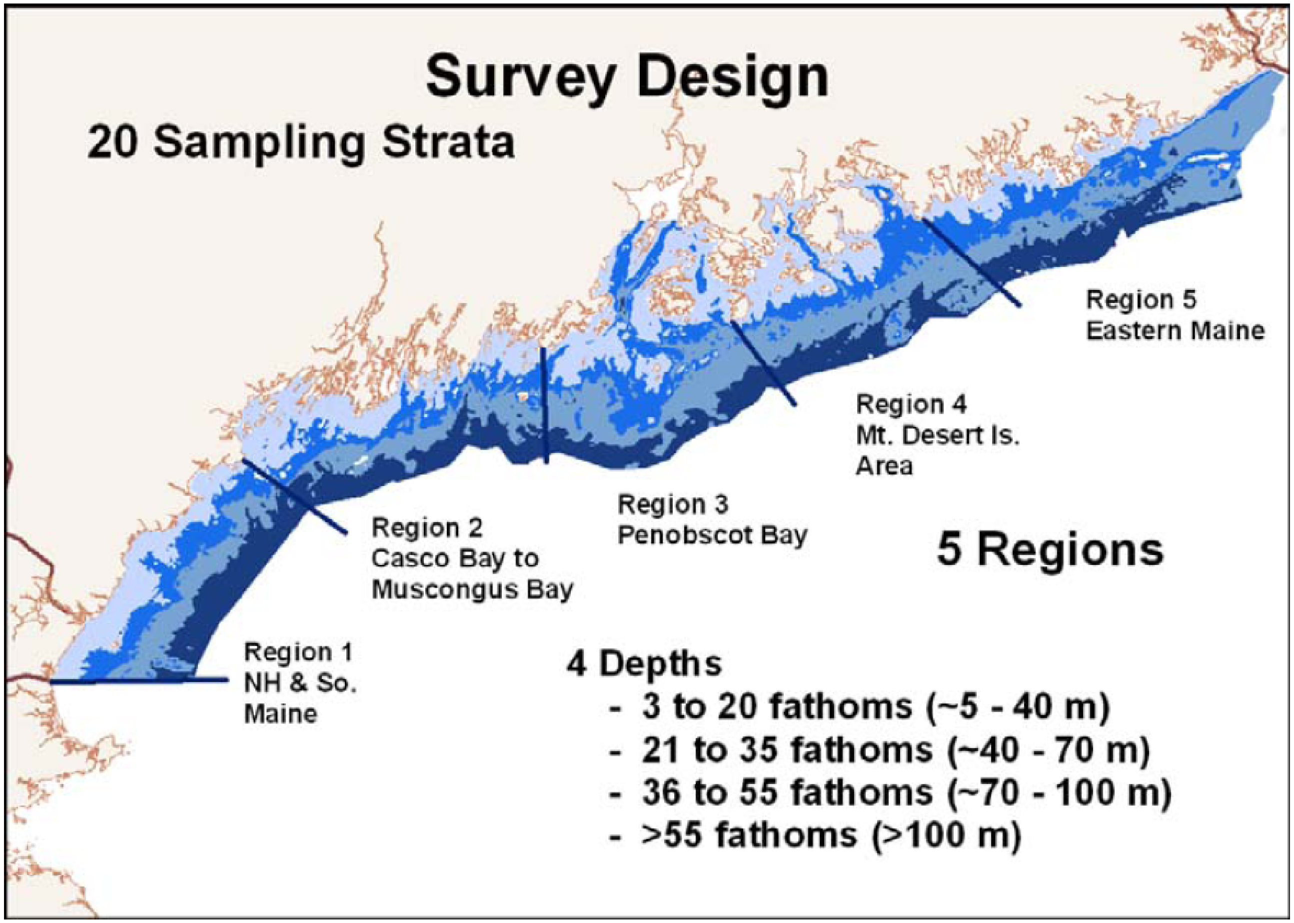


Figure A.117. Map of the Maine-New Hampshire inshore groundfish trawl survey strata set (map from Sherman et al. 2005).

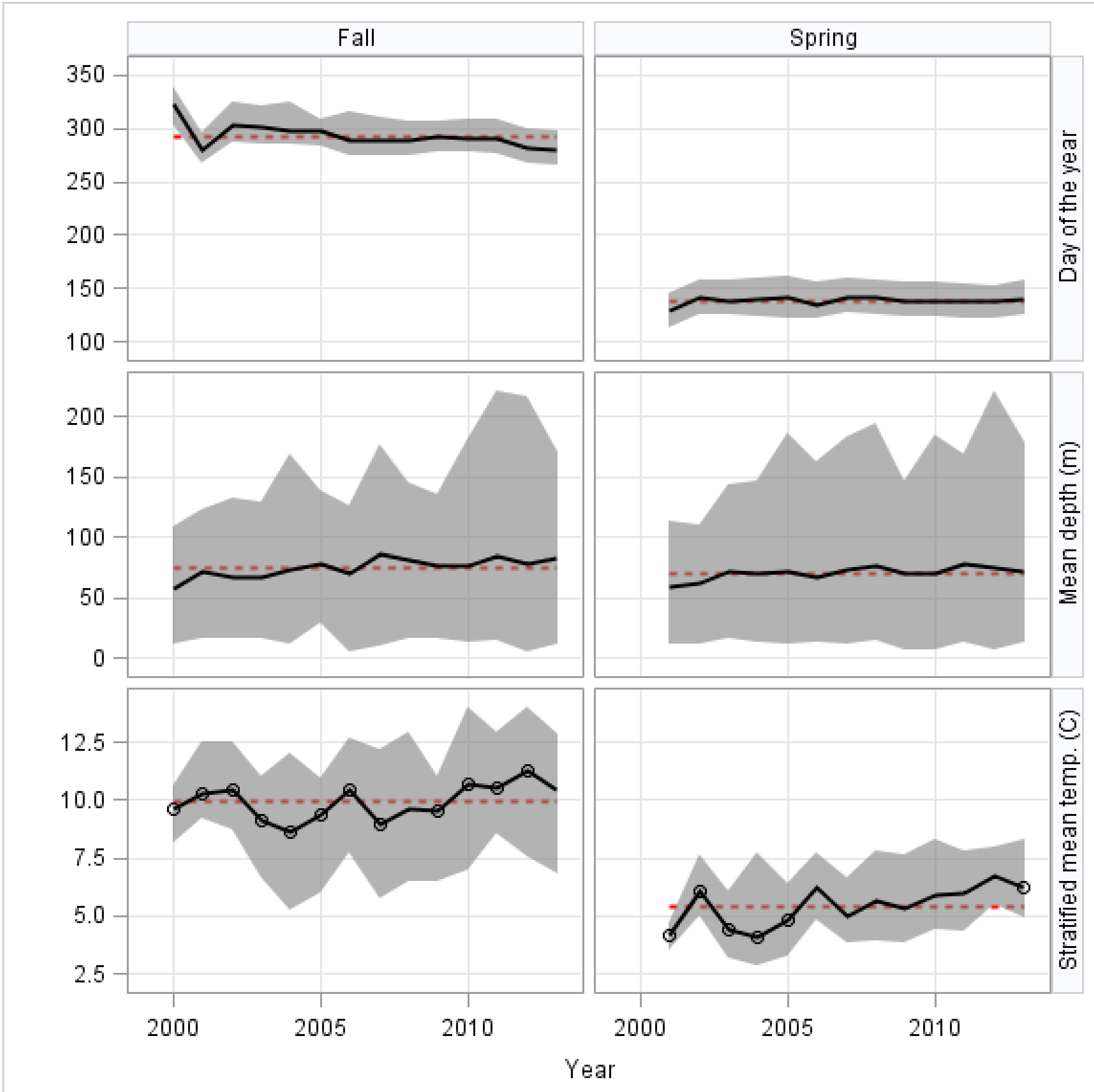


Figure A.118. Mean day of the year, depth and bottom temperature for the Maine-New Hamphire spring and fall inshore groundfish trawl survey. Shaded areas indicate the range between the minimum and maximum observation. *Day of the year is expressed as Julian days (e.g., January 1 is day 1 and December 31 is day 365/66). Years marked with circles in the mean temperature plot indicate years when not all survey stratum were sampled and therefore the mean temperature may not be representative of the entire survey area.*

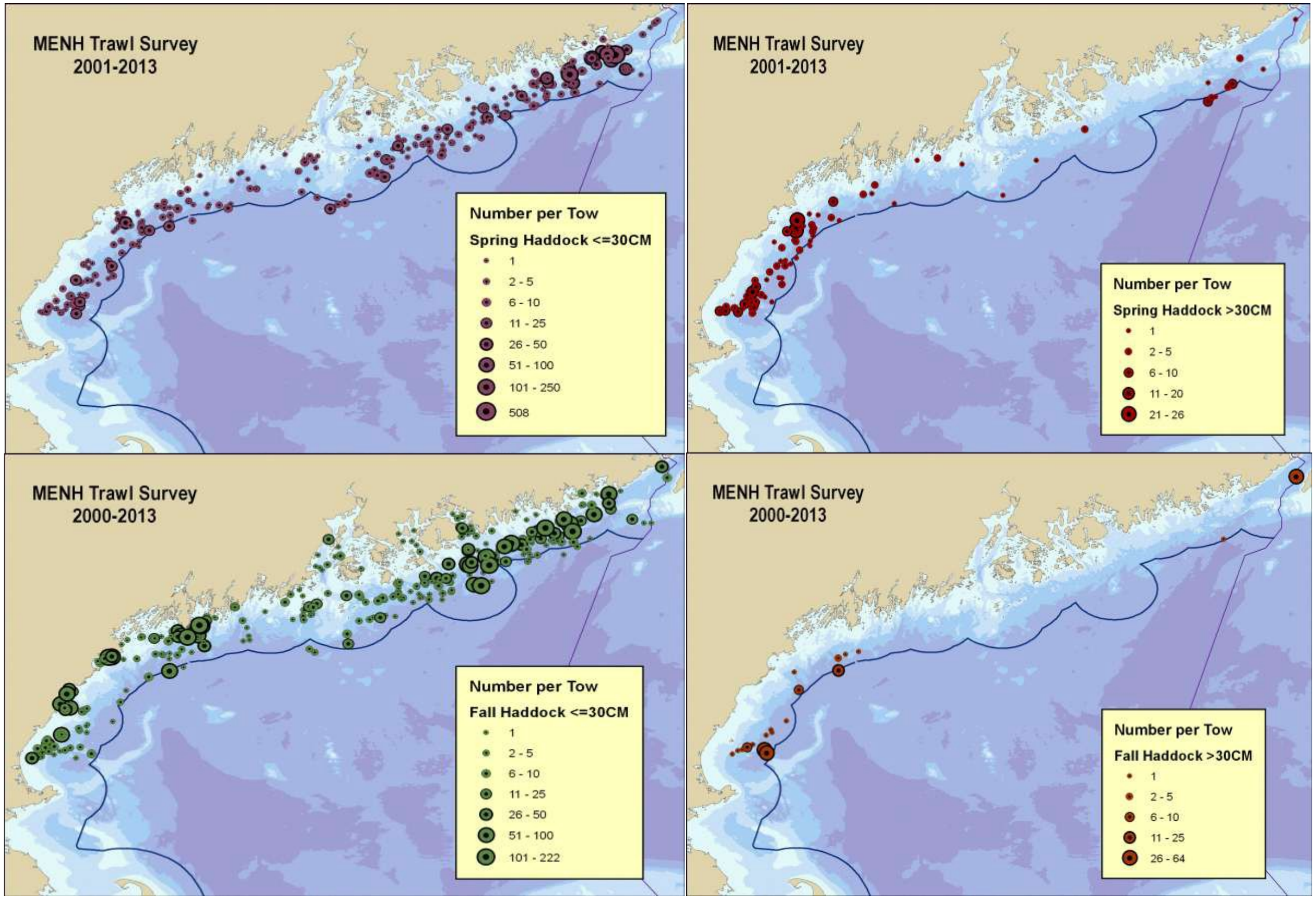


Figure A.119. Spatial distribution of Gulf of Maine haddock catches (numbers/tow) from the spring (top) and fall (bottom) Maine-New Hamphire (MENH) inshore groundfish trawl survey from 2000-2013 for fish ≤30 cm (left) and > 30 cm (right). Maps provided by S. Sherman (pers. comm.).

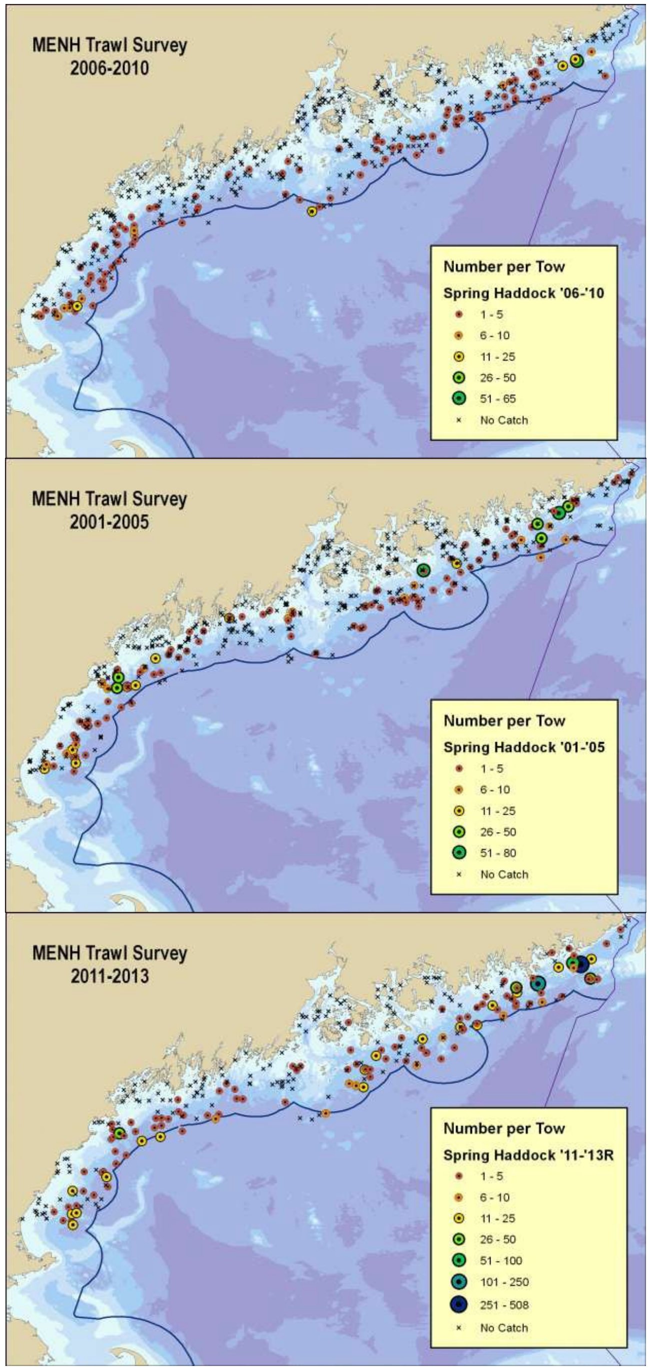


Figure A.120. Spatial distribution of Gulf of Maine haddock catches (numbers/tow) from the spring Maine-New Hamphire (MENH) inshore groundfish trawl survey from 2001-2015 (top), 2006-2010 (middle), and 2011-2013 (bottom). Maps provided by S. Sherman (pers. comm.).

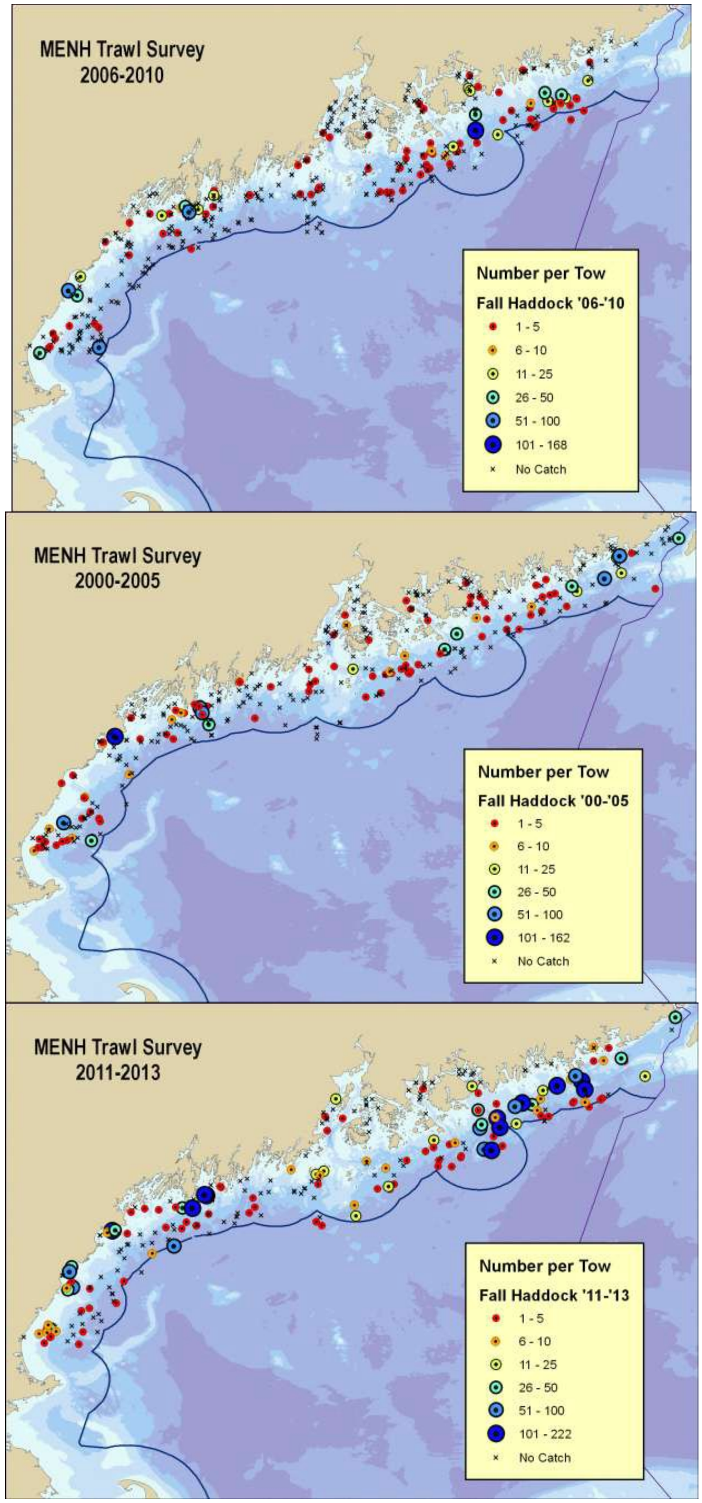


Figure A.121. Spatial distribution of Gulf of Maine haddock catches (numbers/tow) from the fall Maine-New Hamphire (MENH) inshore groundfish trawl survey from 2000-2015 (top), 2006-2010 (middle), and 2011-2013 (bottom). Maps provided by S. Sherman (pers. comm.).

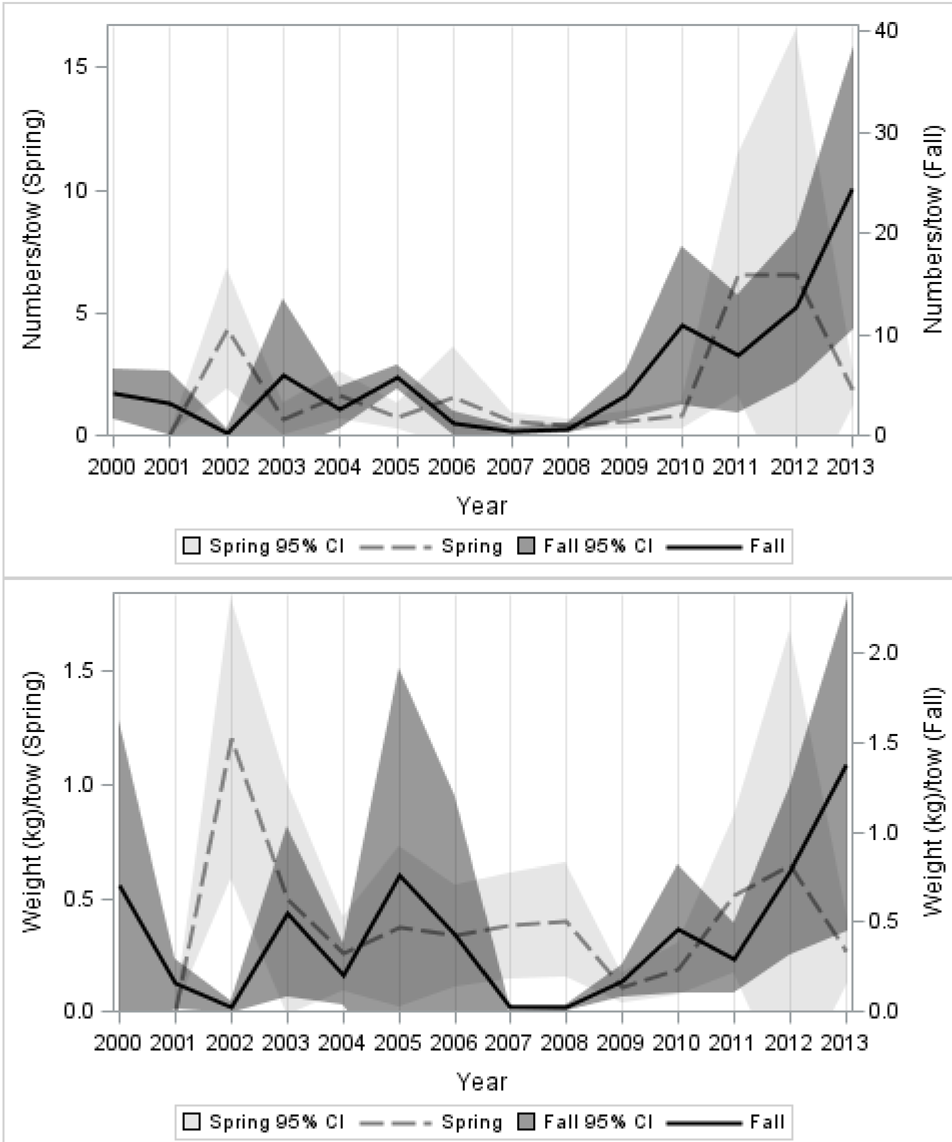


Figure A.122. Maine-New Hampshire inshore (MENH) groundfish trawl survey spring and fall Gulf of Maine haddock biomass (kg/tow) and abundance (numbers/tow) indices from 2000 to 2013. *Note that the spring survey did not begin until 2001.*

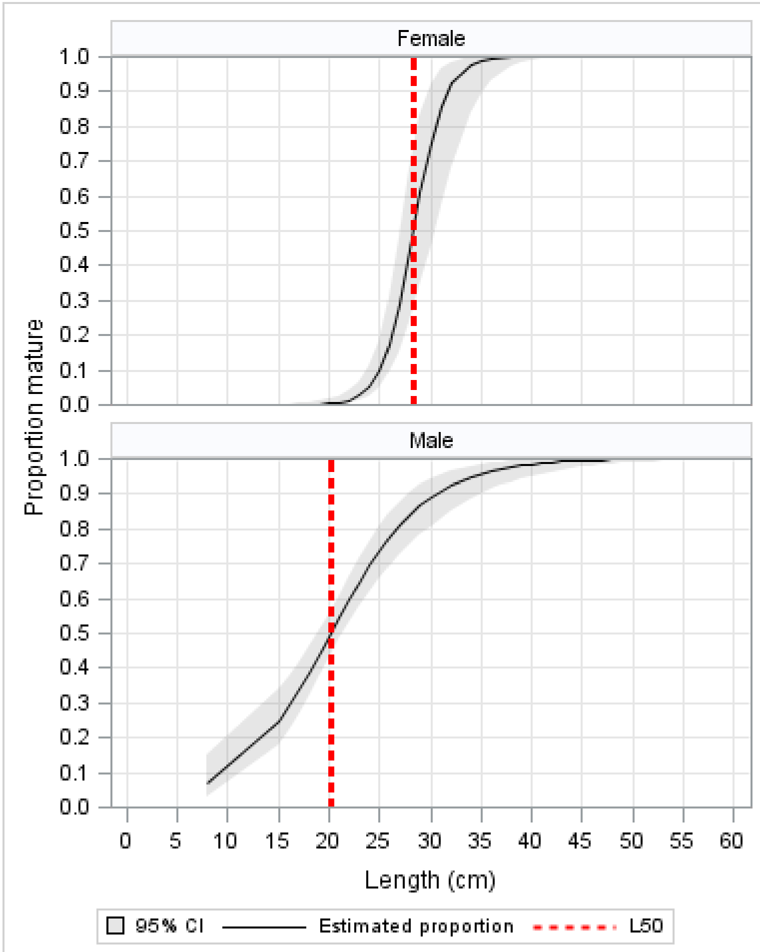


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Figure A.123. Length-based maturity ogives for female (left) and male (right) Gulf of Maine haddock based on time series averages of maturity and length information collected from the Maine-New Hampshire (MENH) spring inshore groundfish trawl survey between 2001 and 2011. The dashed red line indicates the length at 50% maturity.

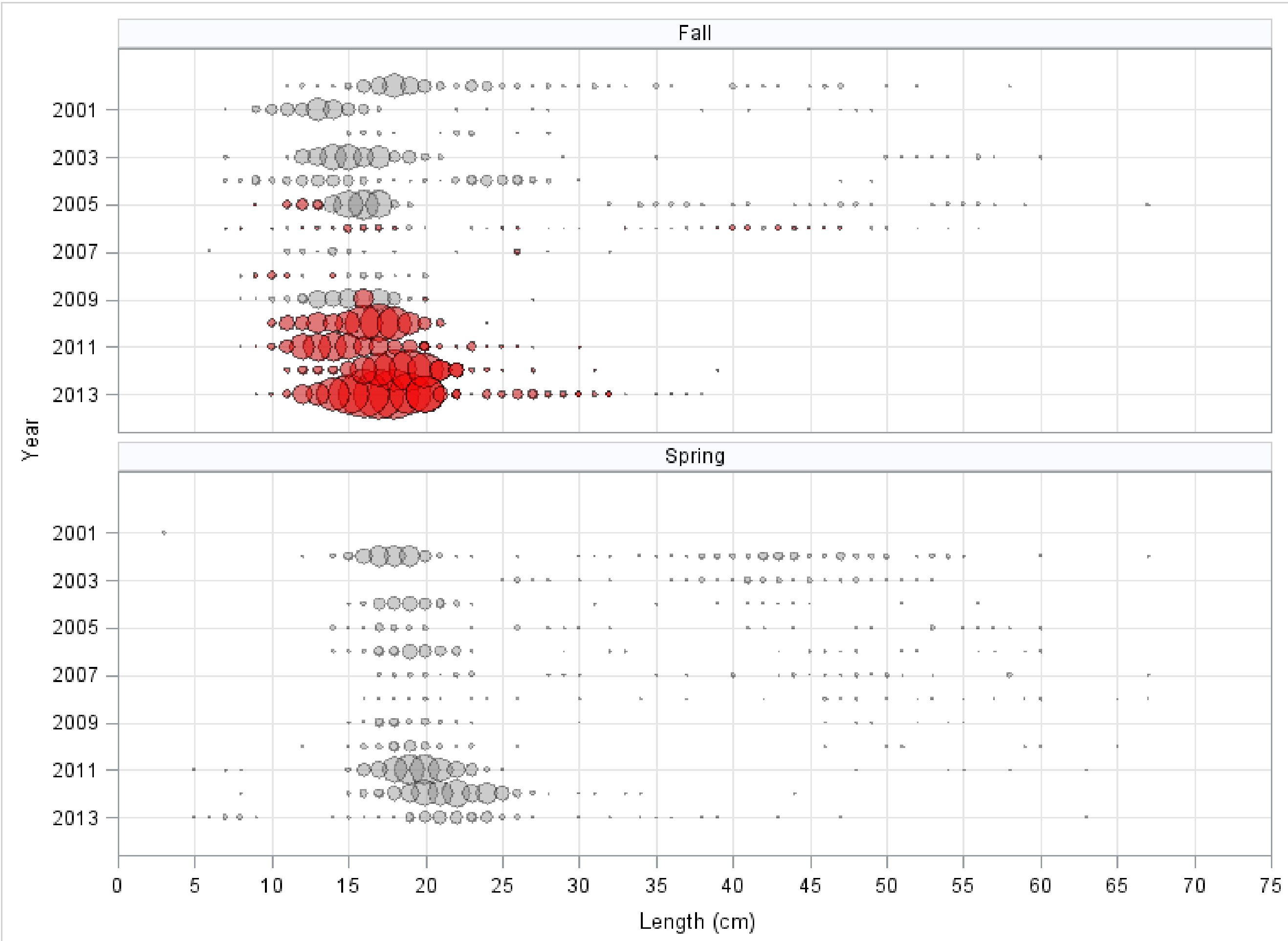


Figure A.124. Length distributions of Gulf of Maine haddock sampled in the Maine-New Hampshire (MENH) inshore groundfish trawl spring (top) and fall (bottom) surveys from 2001 to 2013. The red shaded bubbles indicate the length intervals for which there is age information available directly from the MENH survey.

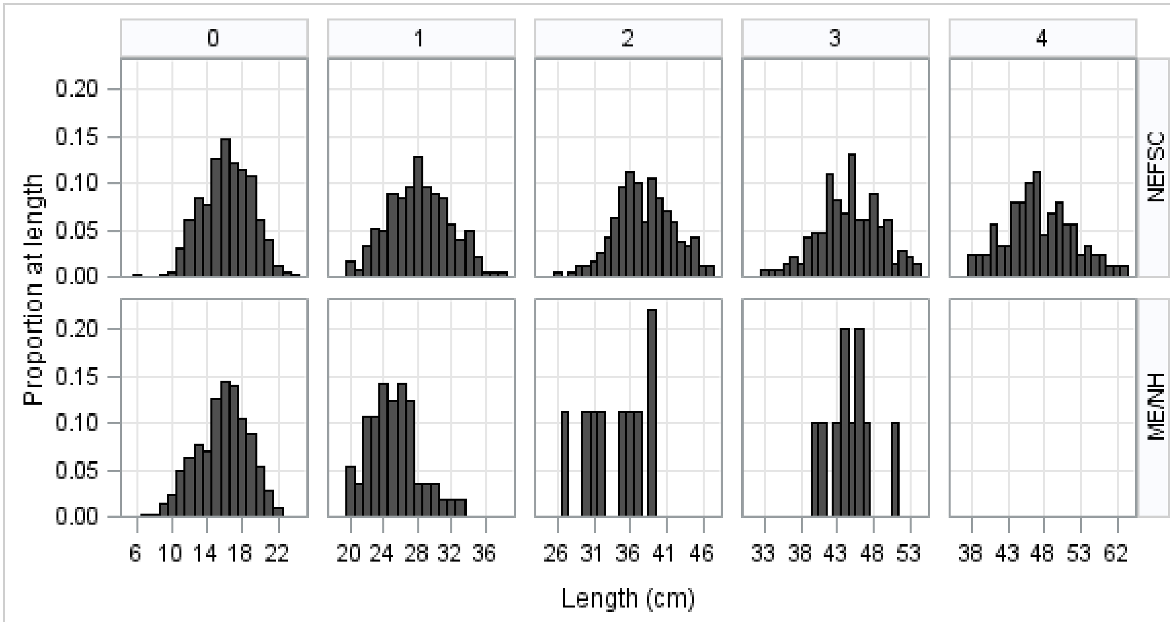


Figure A.125. Comparison of Gulf of Maine haddock length frequency distributions by age class from the Maine-New Hampshire (MENH) and Northeast Fisheries Science Center (NEFSC) fall bottom trawl survey. Data have been aggregated over the years 2005 to 2013.

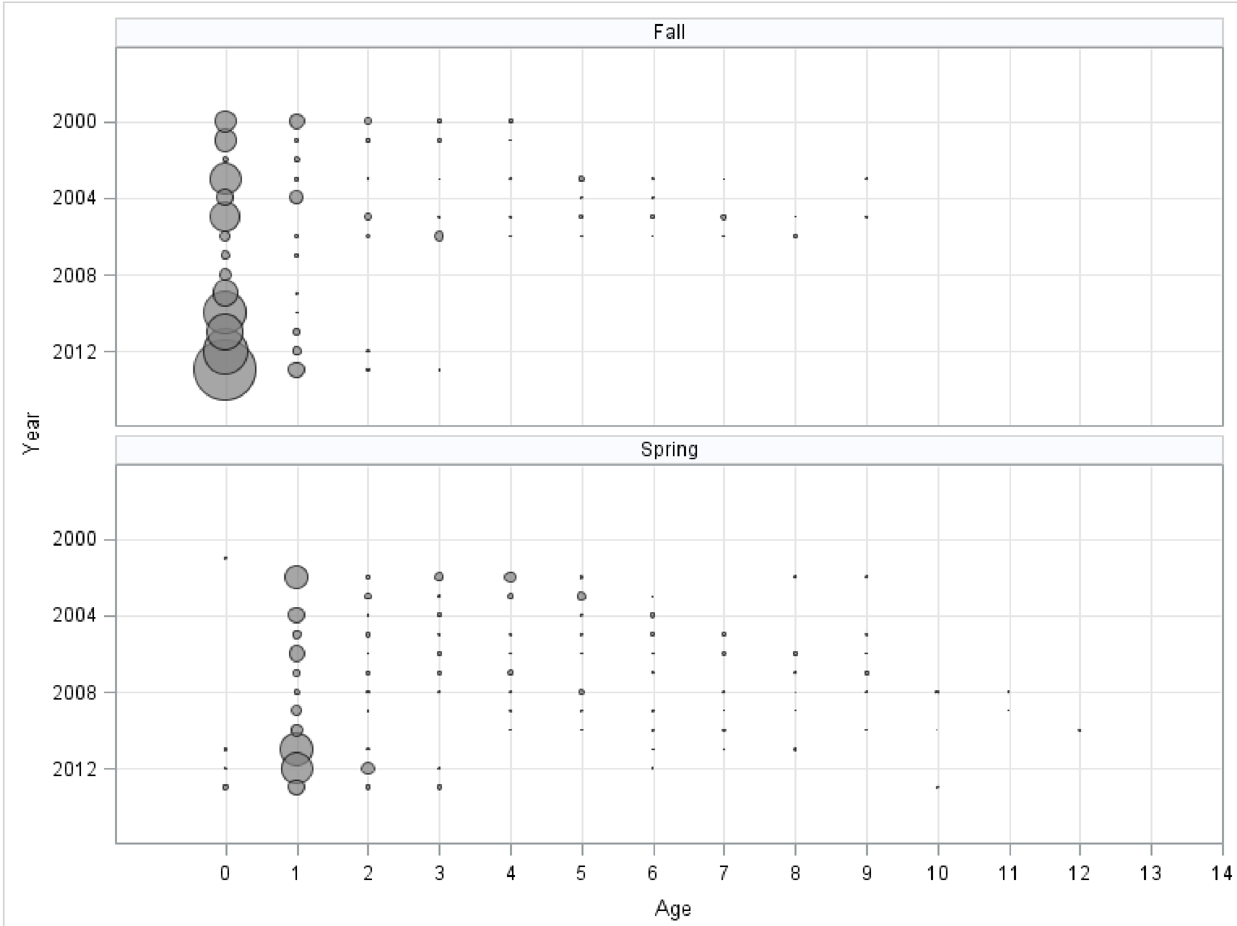


Figure A.126. Gulf of Maine haddock numbers-at-age from Maine-New Hampshire (MENH) spring and fall inshore groundfish trawl survey from 2000 to 2013. *Note: 1) the spring survey did not begin until 2001; 2)* indices-at-age were constructed from age-length information borrowed from the Northeast Fisheries Science Center survey data with the exception of the fall 2005-2013 indices were MENH age-length information was augmented with NEFSC data.

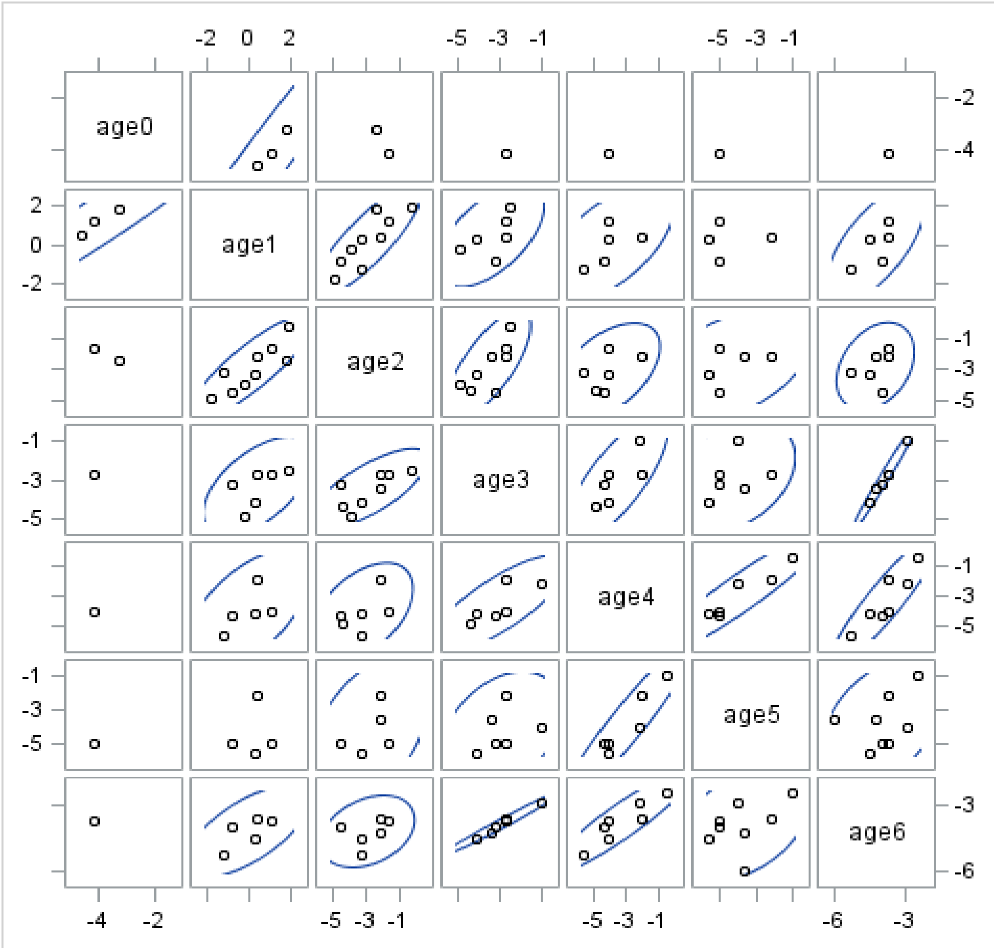


Figure A.127. Scatter plots showing the level of agreement between Massachusetts Maine – New Hampshire (MENH) spring inshore groundfish trawl survey Gulf of Maine haddock indices-at-age (log transformed) on a cohort basis. 80% confidence ellipses are shown.

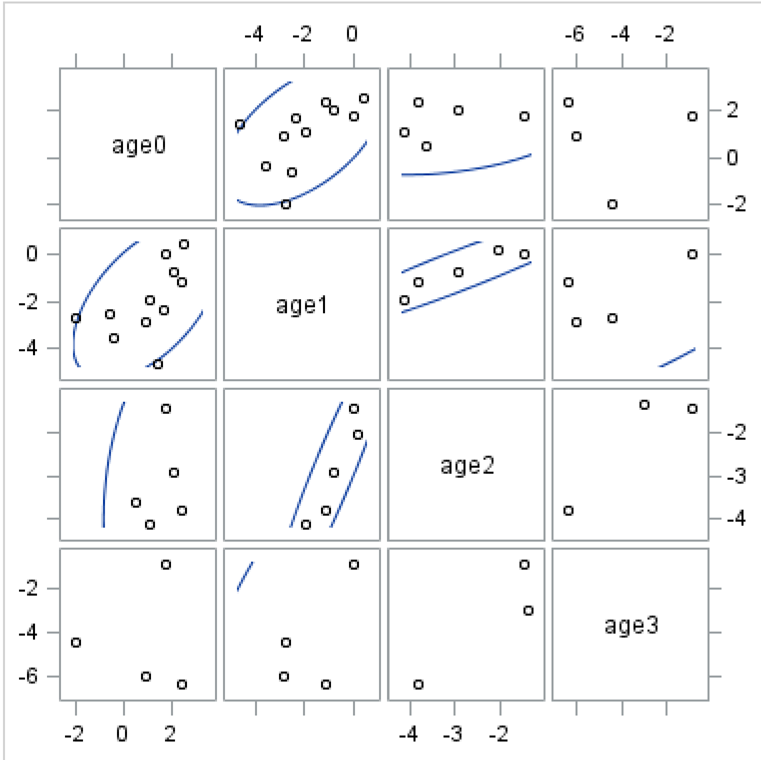


Figure A.128. Scatter plots showing the level of agreement between Maine – New Hampshire (MENH) fall inshore groundfish trawl survey Gulf of Maine haddock indices-at-age (log transformed) on a cohort basis. 80% confidence ellipses are shown.

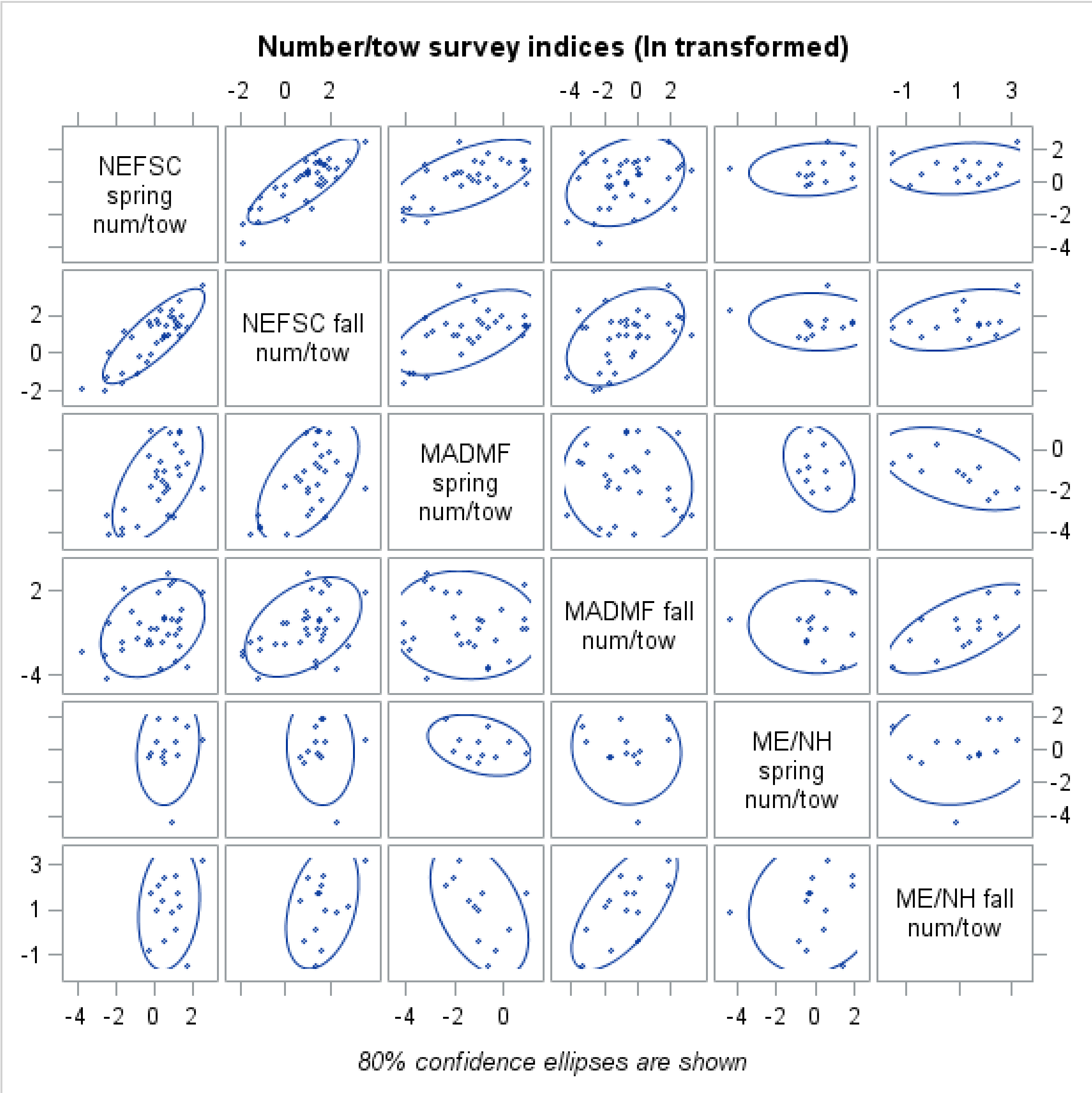


Figure A.129. Scatter plots showing the level of agreement between the Northeast Fisheries Science Center (NEFSC), Massachusetts Department of Marine Fisheries (MADMF) and the Maine – New Hampshire (ME/NH) inshore groundfish trawl survey Gulf of Maine haddock abundance (numbers/tow) indices (log transformed). 80% confidence ellipses are shown.

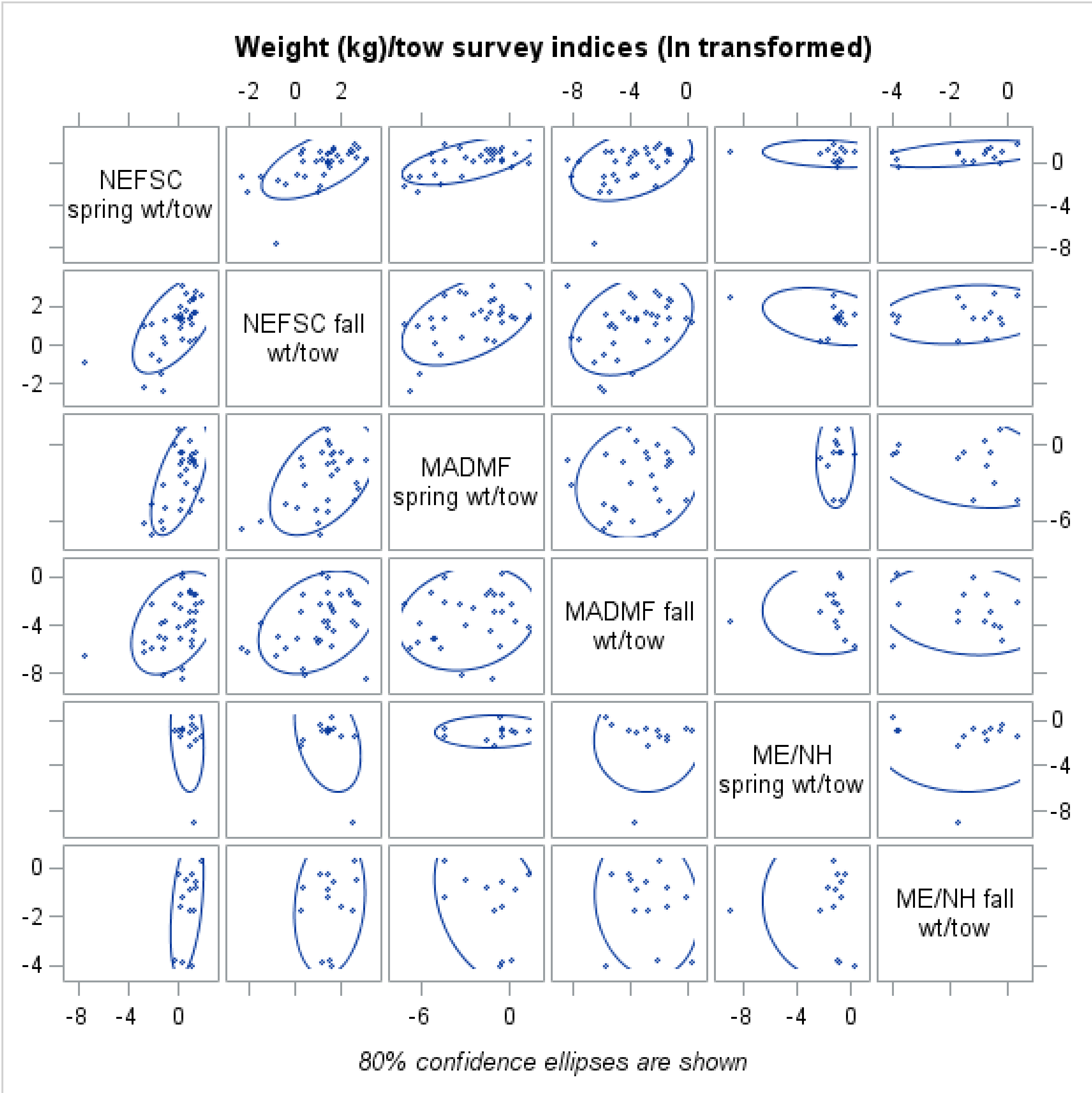


Figure A.130. Scatter plots showing the level of agreement between the Northeast Fisheries Science Center (NEFSC), Massachusetts Department of Marine Fisheries (MADMF) and the Maine – New Hampshire (ME/NH) inshore groundfish trawl survey Gulf of Maine haddock biomass (weight/tow) indices (log transformed). 80% confidence ellipses are shown.

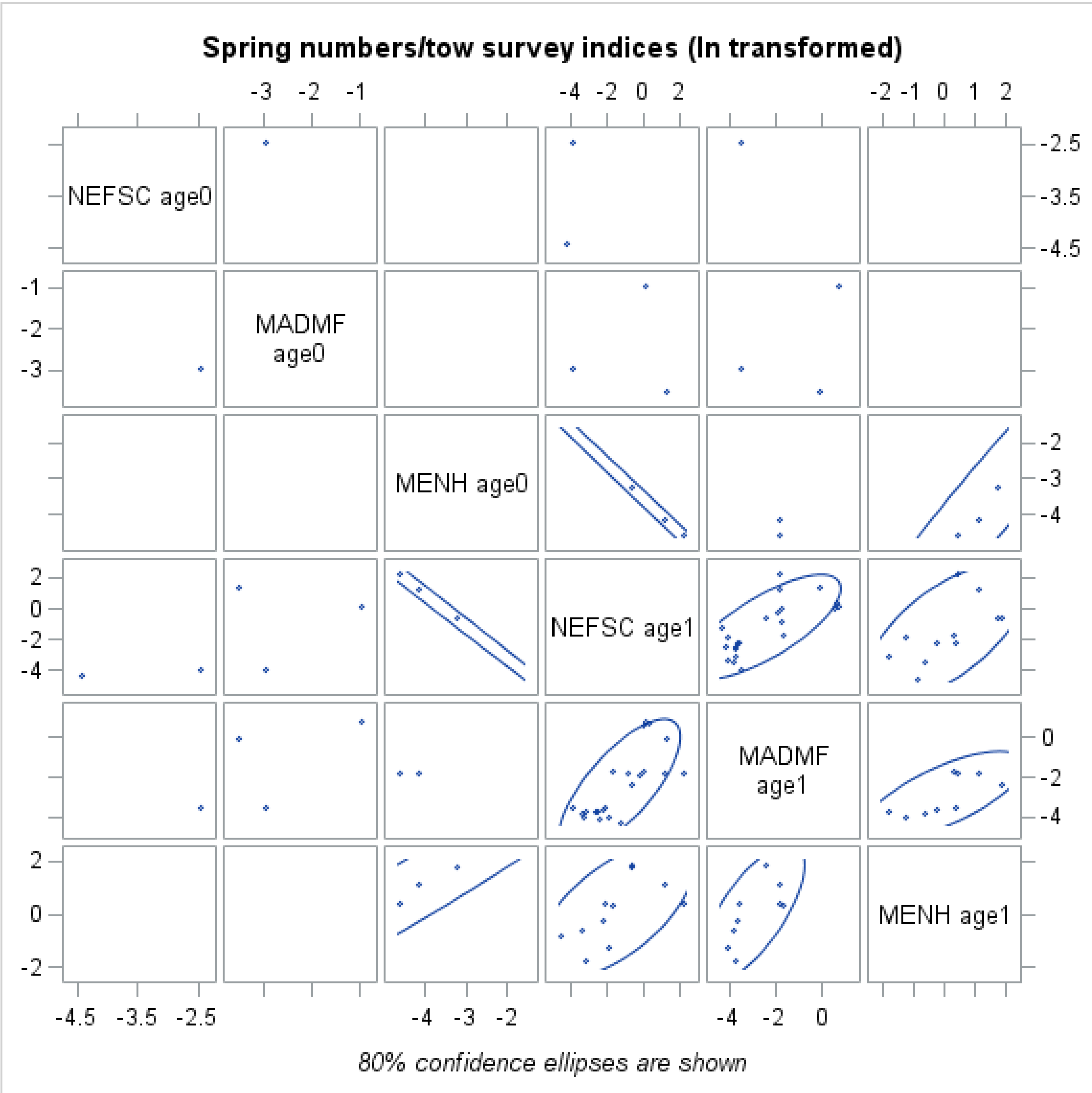


Figure A.131. Scatter plots showing the level of agreement between the Northeast Fisheries Science Center (NEFSC), Massachusetts Department of Marine Fisheries (MADMF) and the Maine – New Hampshire (ME/NH) inshore groundfish trawl survey Gulf of Maine haddock spring abundance (numbers/tow) indices (log transformed) for age-0 and age-1 fish on a cohort basis. 80% confidence ellipses are shown.

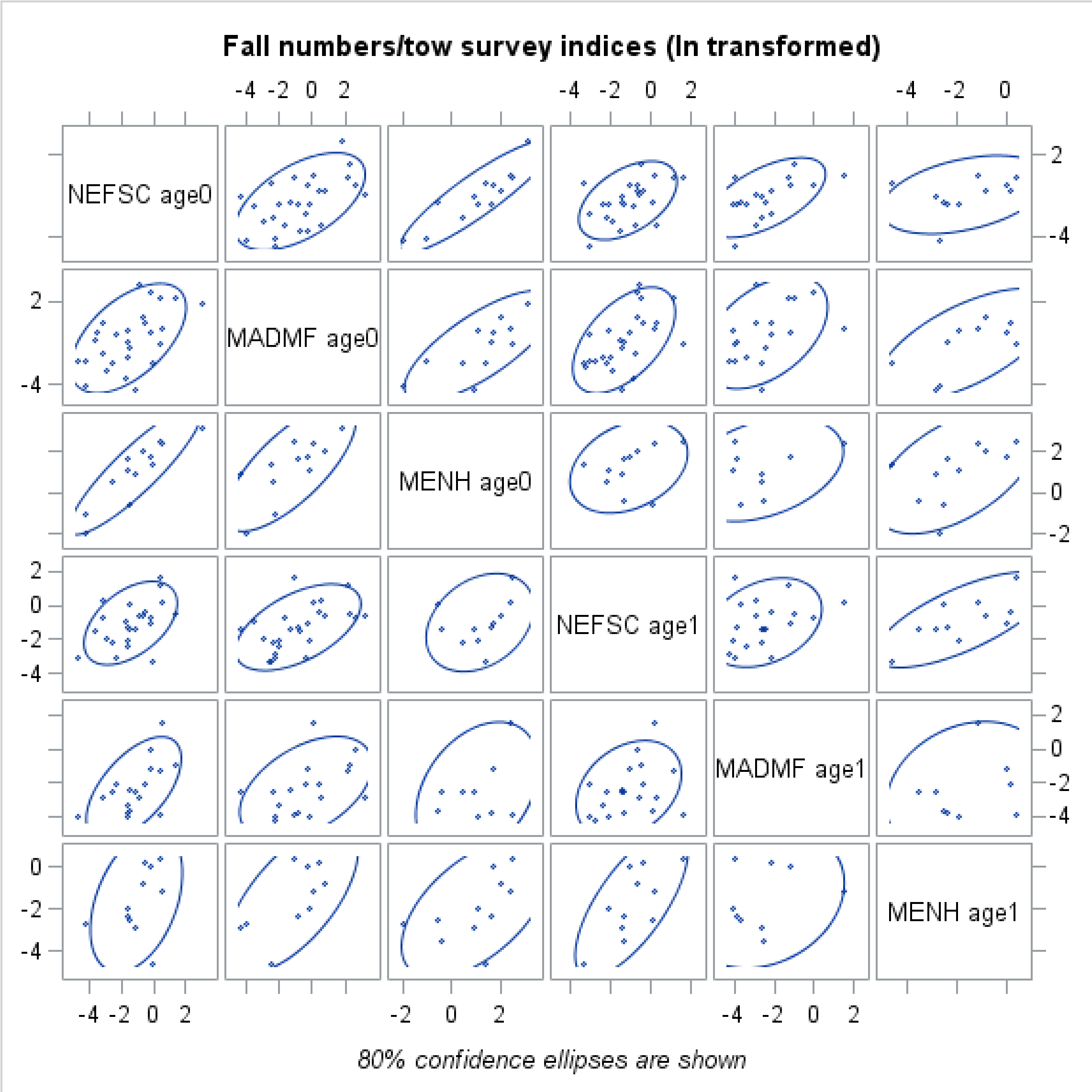


Figure A.132. Scatter plots showing the level of agreement between the Northeast Fisheries Science Center (NEFSC), Massachusetts Department of Marine Fisheries (MADMF) and the Maine – New Hampshire (ME/NH) inshore groundfish trawl survey Gulf of Maine haddock fall abundance (numbers/tow) indices (log transformed) for age-0 and age-1 fish on a cohort basis. 80% confidence ellipses are shown.

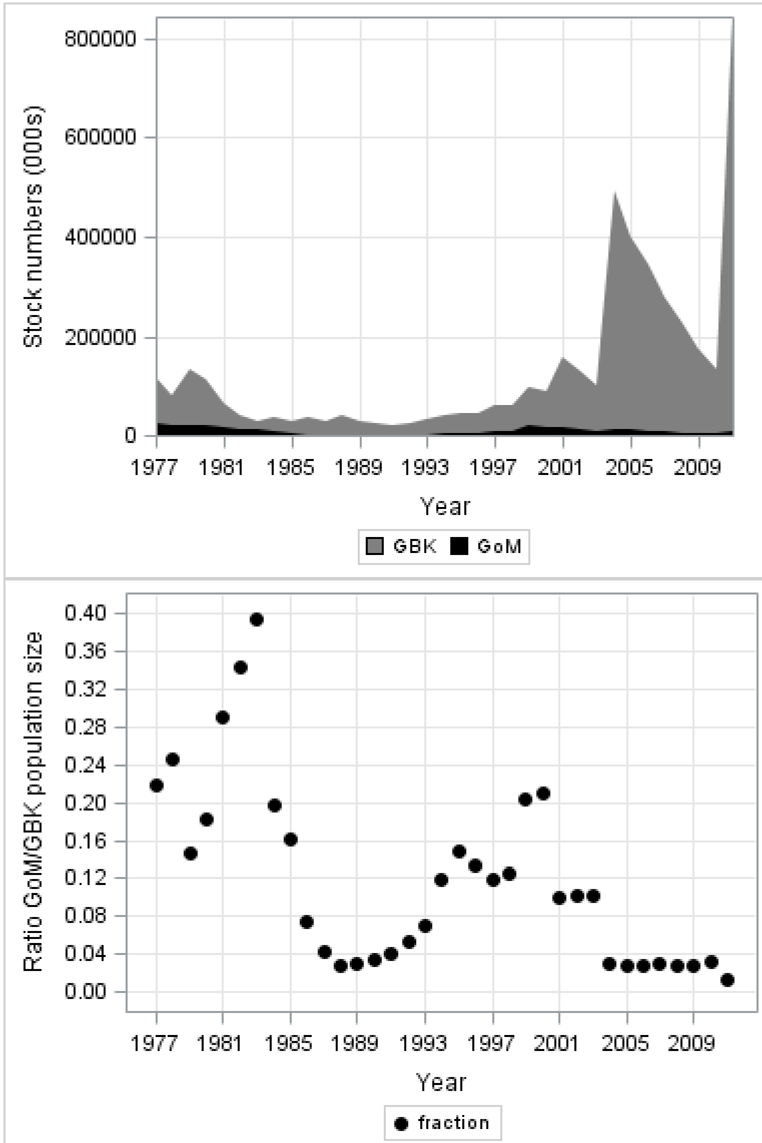


Figure A.133. Georges Bank (GBK) and Gulf of Maine (GoM) haddock population size between 1977 and 2011. The top plot shows the stock population in terms of numbers (000s fish) and the bottom expressed in terms of the ratio of GoM/GBK population in numbers.

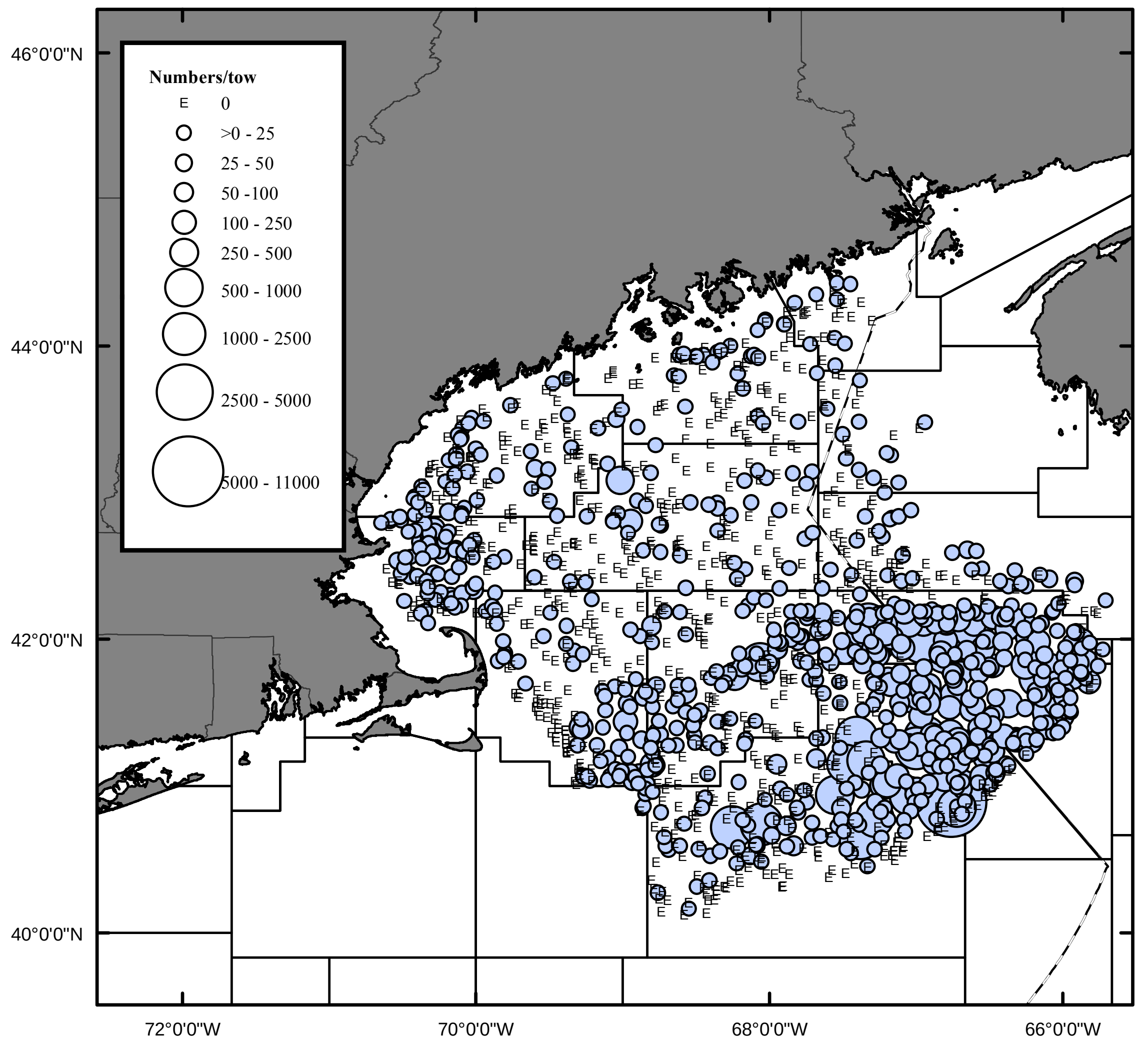


Figure A.134. Distribution plots of catches of haddock from the Northeast Fisheries Science Center spring bottom trawl survey between 2003 and 2013.

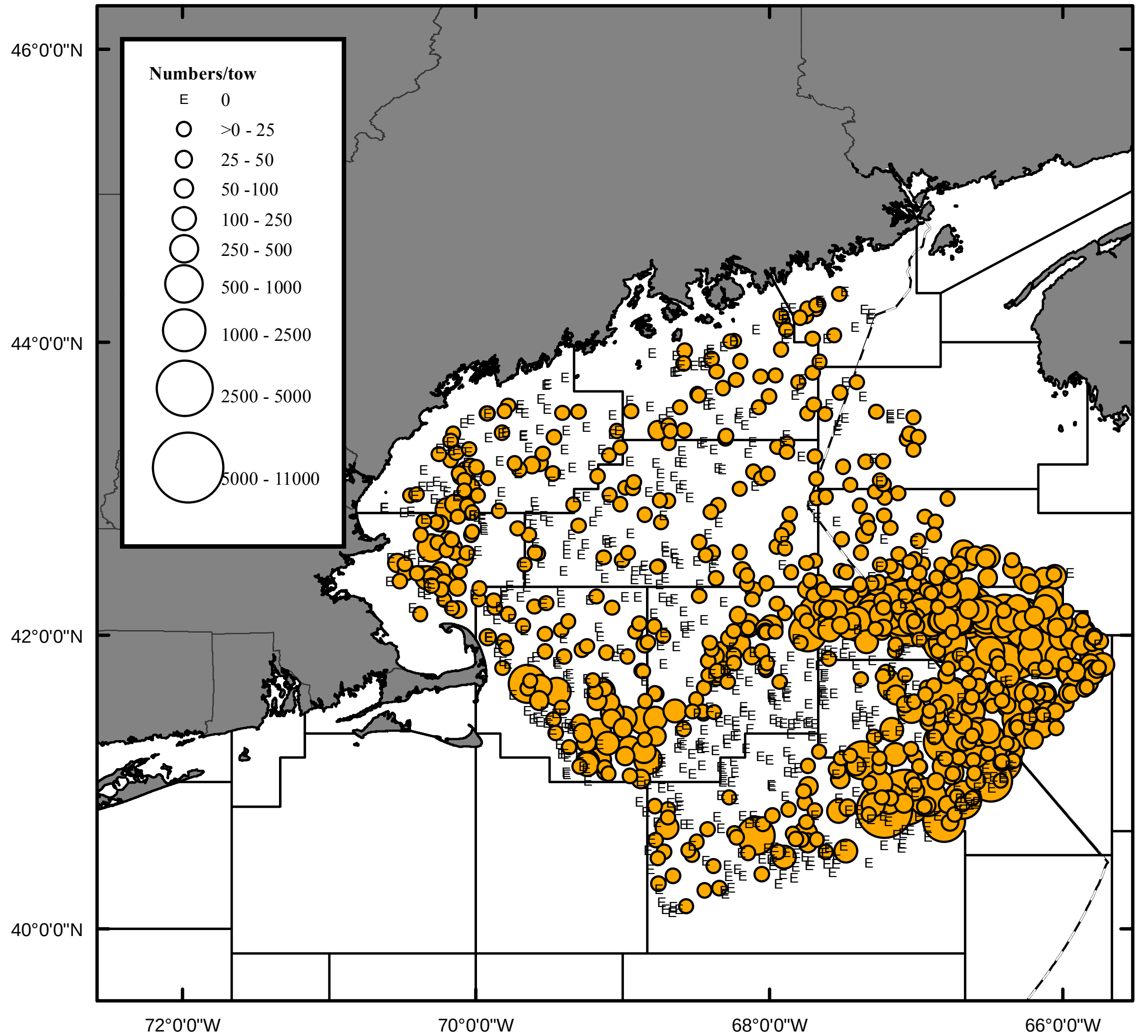


Figure A.135. Distribution plots of catches of haddock from the Northeast Fisheries Science Center fall bottom trawl survey between 2003 and 2012.

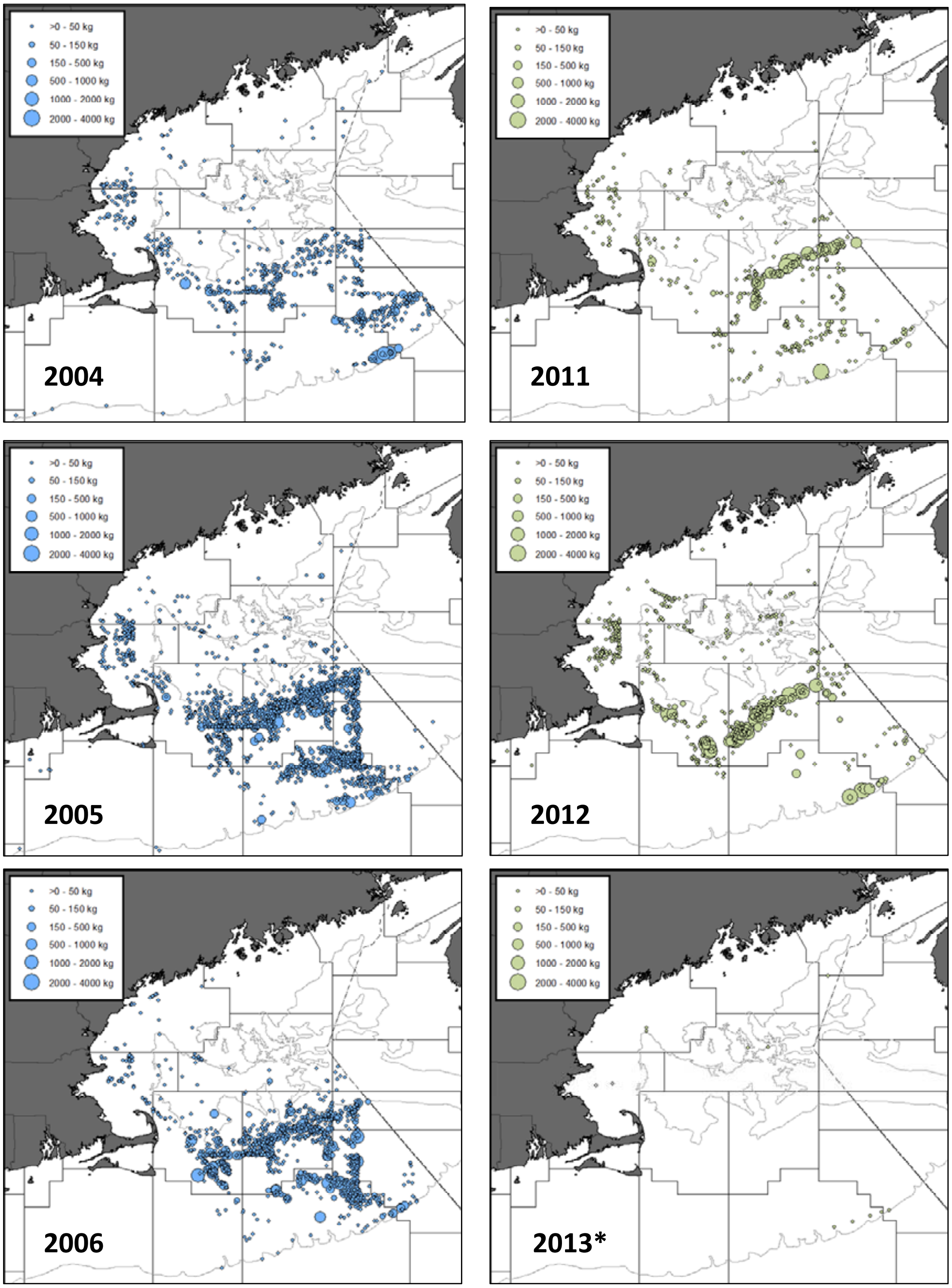


Figure A.136. Distribution plots of catches of haddock less than 46 cm as recorded by fishery observers following the spawning of the 2003 and 2010 year classes on Georges Bank. **Note that at the time of the analysis there was limited information available for 2013.*

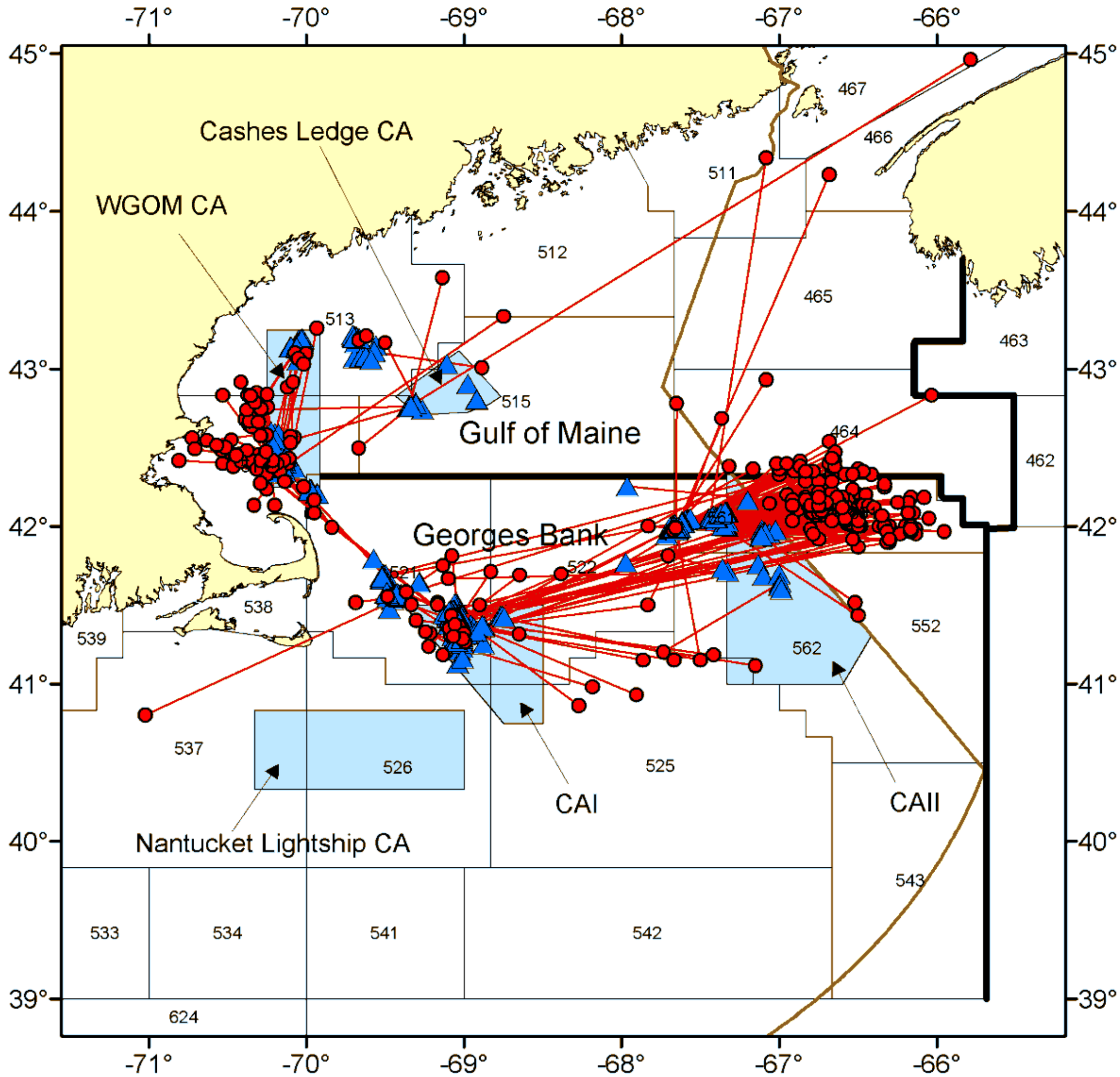


Figure A.137. Plot of release (red circle) and recapture (blue triangle) locations of haddock tagged during the Northeast Consortium Cooperative Haddock Tagging Project reported through September 2, 2008 in relation to Georges Bank and Gulf of Maine haddock stock boundary (solid black line) to the west of the Hague line (light tan line) (Brodziak et al. 2008).

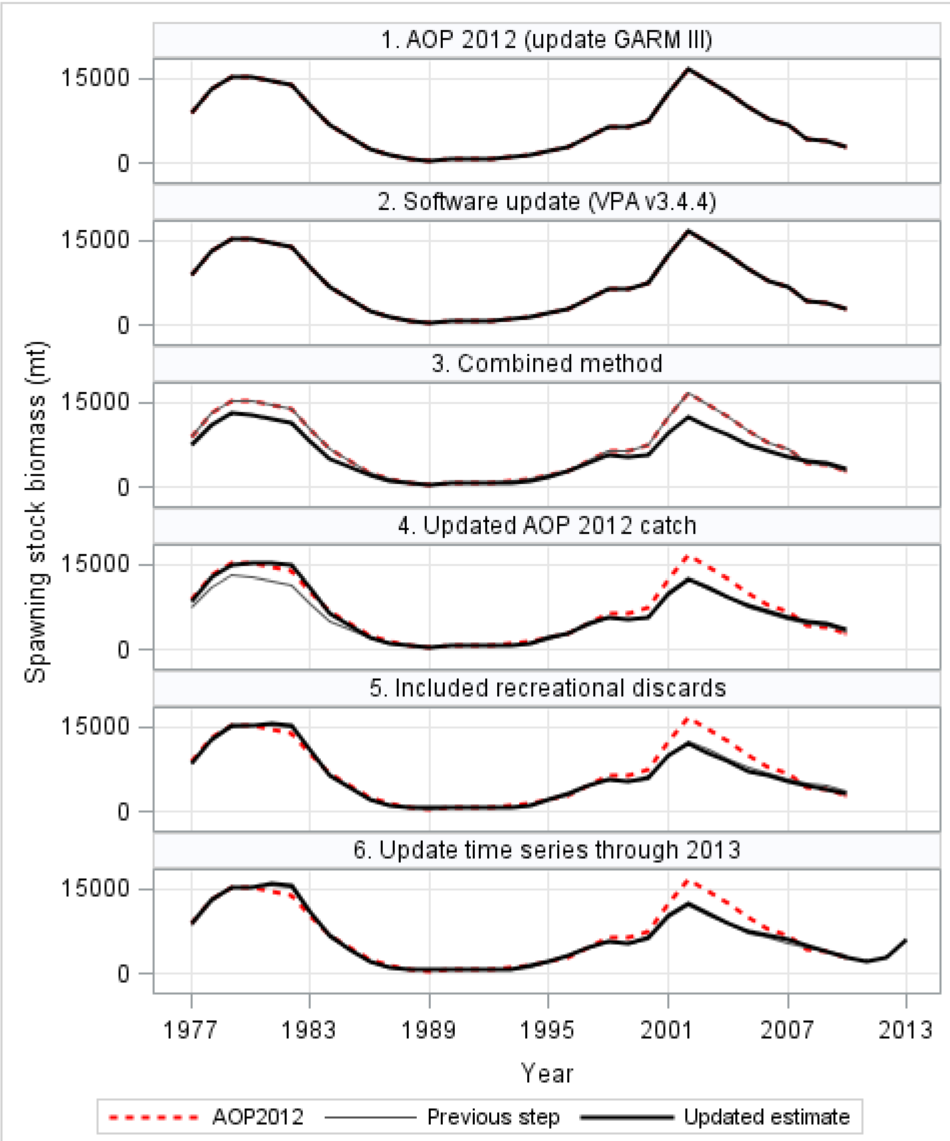


Figure A138. Summary of the impacts of the ADAPT-VPA model update process on the time series of Gulf of Maine haddock spawning stock biomass. The result from each model update is indicated by the solid black line and can be compared to the 2012 AOP result (dashed red line) and the result from the previous model run (grey line).

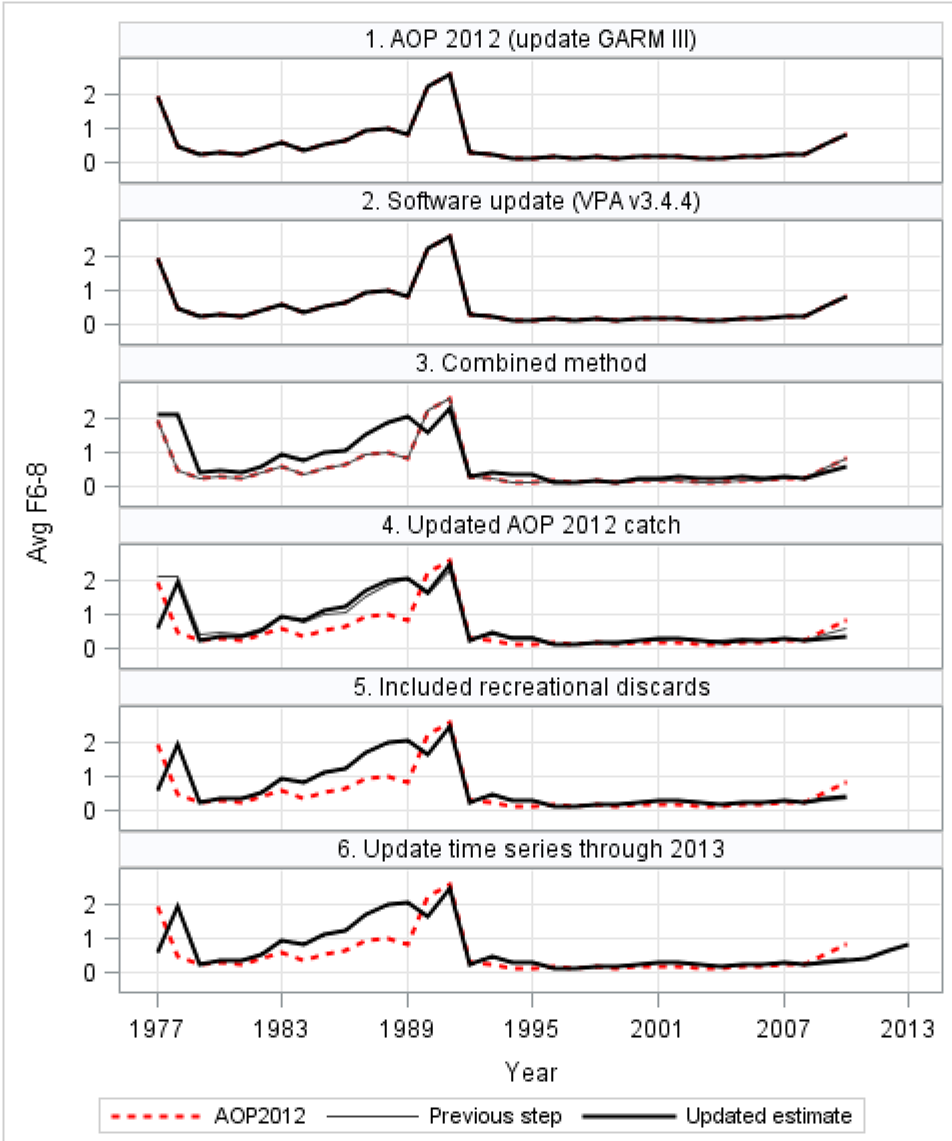


Figure A.139. Summary of the impacts of the ADAPT-VPA model update process on the time series of Gulf of Maine haddock average fishing mortality on ages 6-8. The result from each model update is indicated by the solid black line and can be compared to the 2012 AOP result (dashed red line) and the result from the previous model run (grey line).

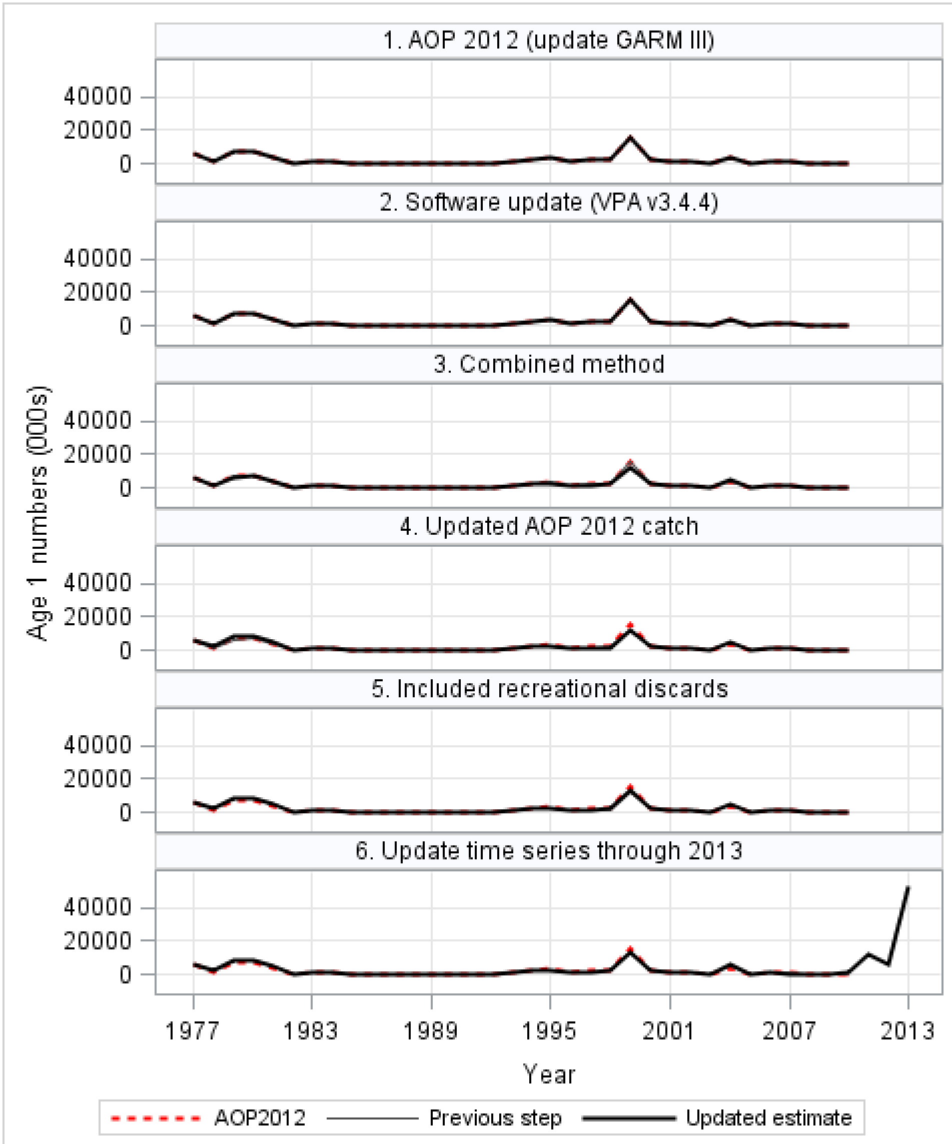


Figure A.140. Summary of the impacts of the ADAPT-VPA model update process on the time series of Gulf of Maine haddock age-1 recrutiment. The result from each model update is indicated by the solid black line and can be compared to the 2012 AOP result (dashed red line) and the result from the previous model run (grey line).

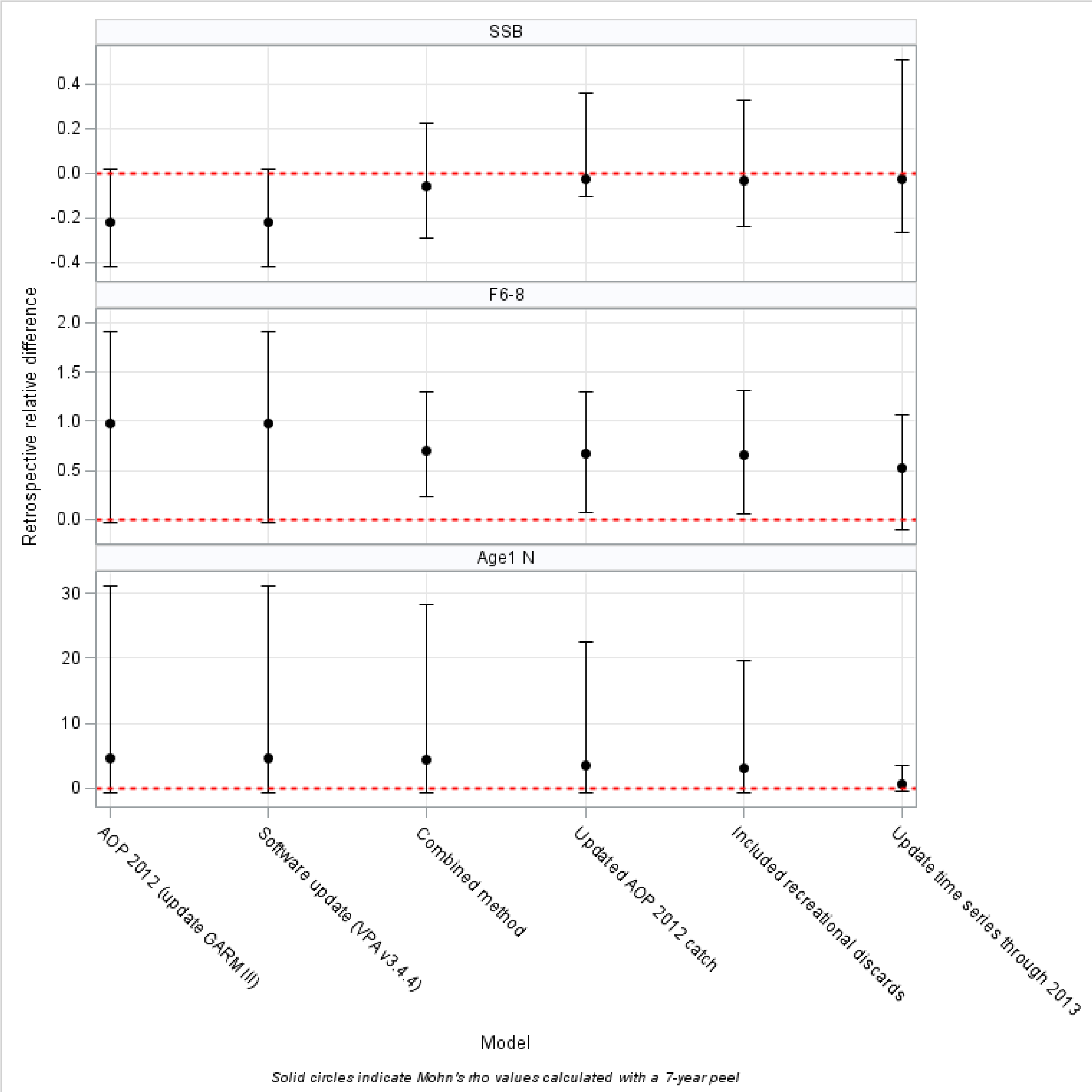


Figure A.141. Summary of the Mohn's rho values (dots) and minimum and maximum observed relative differences resulting from a seven year peel for the six Gulf of Maine haddock ADAPT-VPA model runs used to build the bridge between the 2012 AOP model to an updated SAW/SARC 59 VPA model.

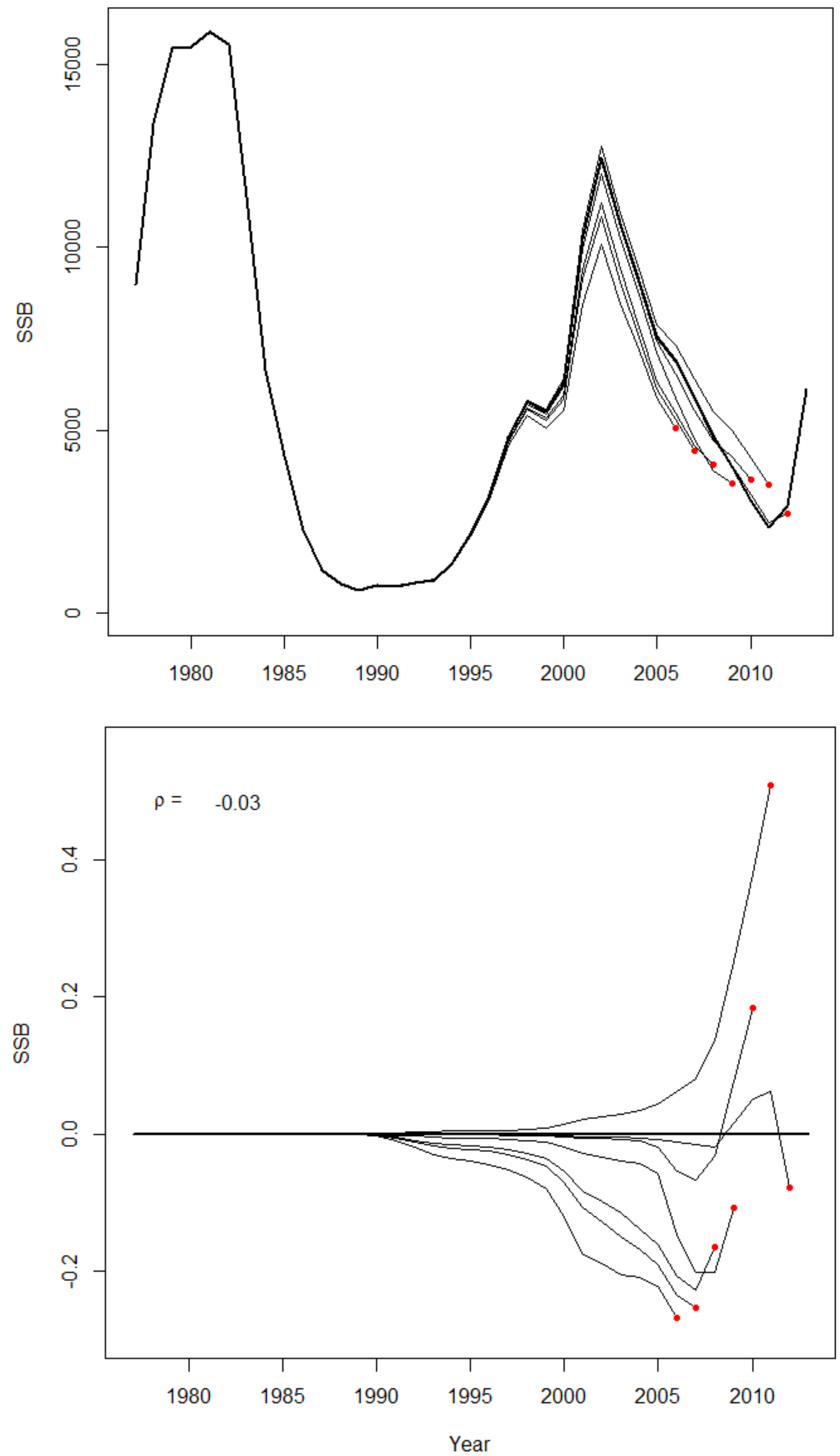


Figure A.142. Retrospective patterns in Gulf of Maine haddock spawning stock biomass (mt) in absolute (top) and relative (bottom) terms from the 2013 update of the ADAPT-VPA model (run 6). A 7-year retrospective peel is shown along with the corresponding Mohn's rho (ρ) value.

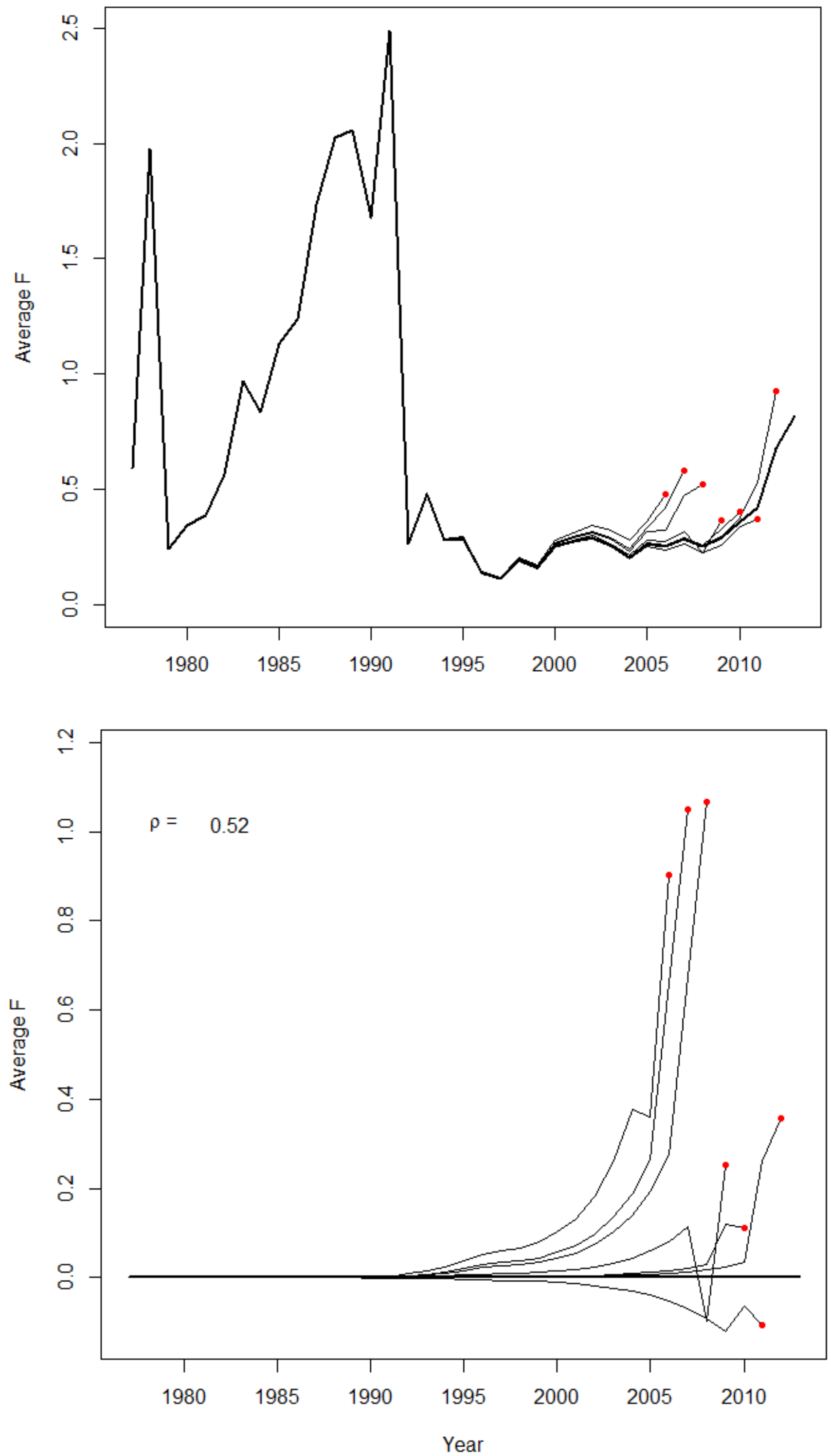
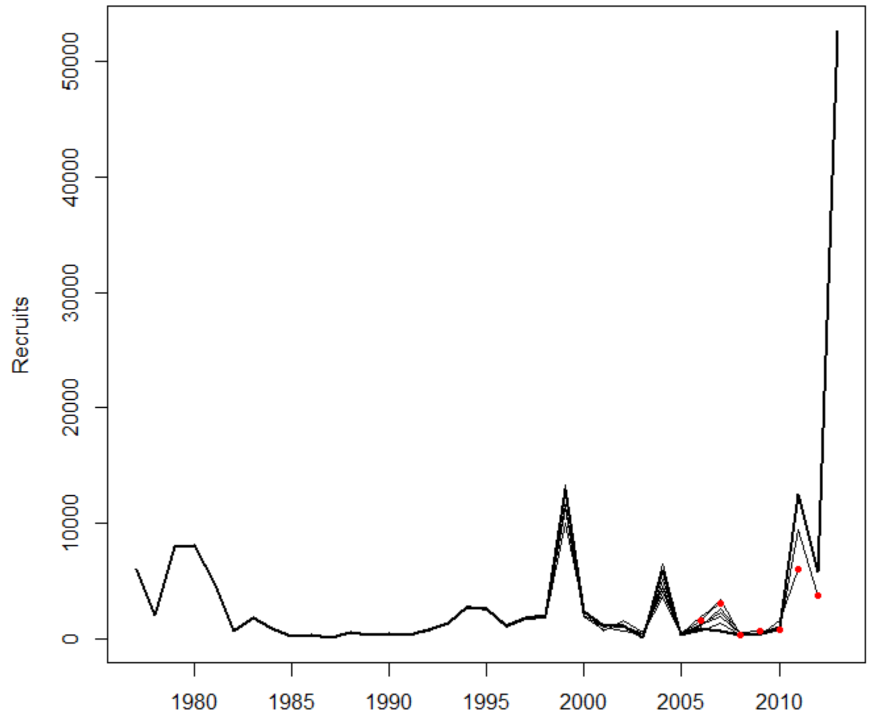


Figure A.143. Retrospective patterns in Gulf of Maine haddock fishing mortality (average ages 6-8) in absolute (top) and relative (bottom) terms from the 2013 update of the ADAPT-VPA model (run 6). A 7-year retrospective peel is shown along with the corresponding Mohn's rho (ρ) value.



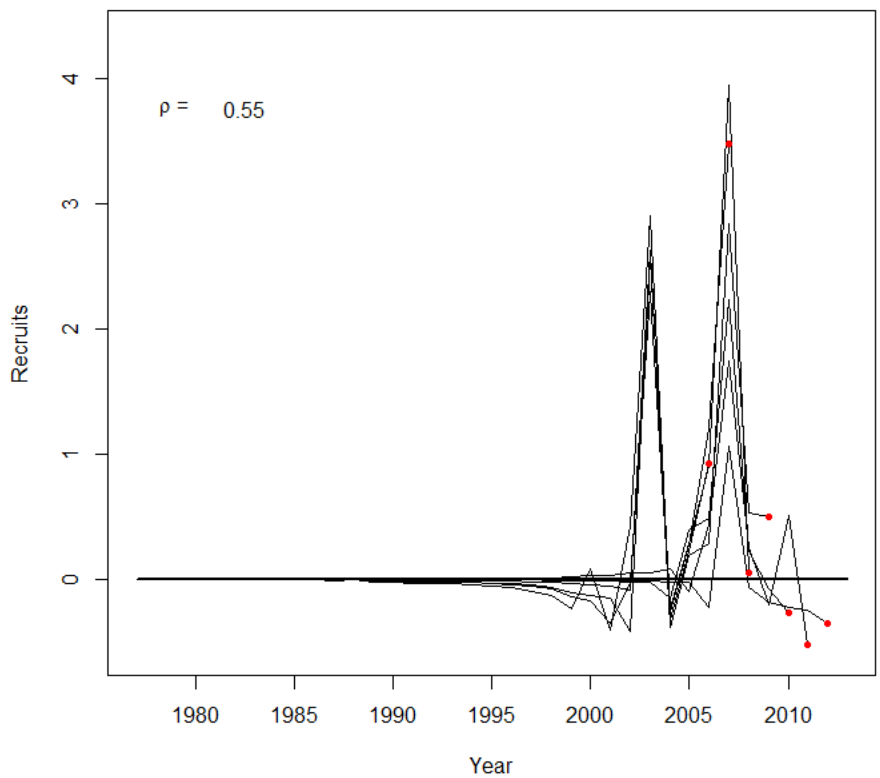


Figure A.144. Retrospective patterns in Gulf of Maine haddock age-1 recruitment (000s) in absolute (top) and relative (bottom) terms from the 2013 update of the ADAPT-VPA model (run 6). A 7-year retrospective peel is shown along with the corresponding Mohn's rho (ρ) value.

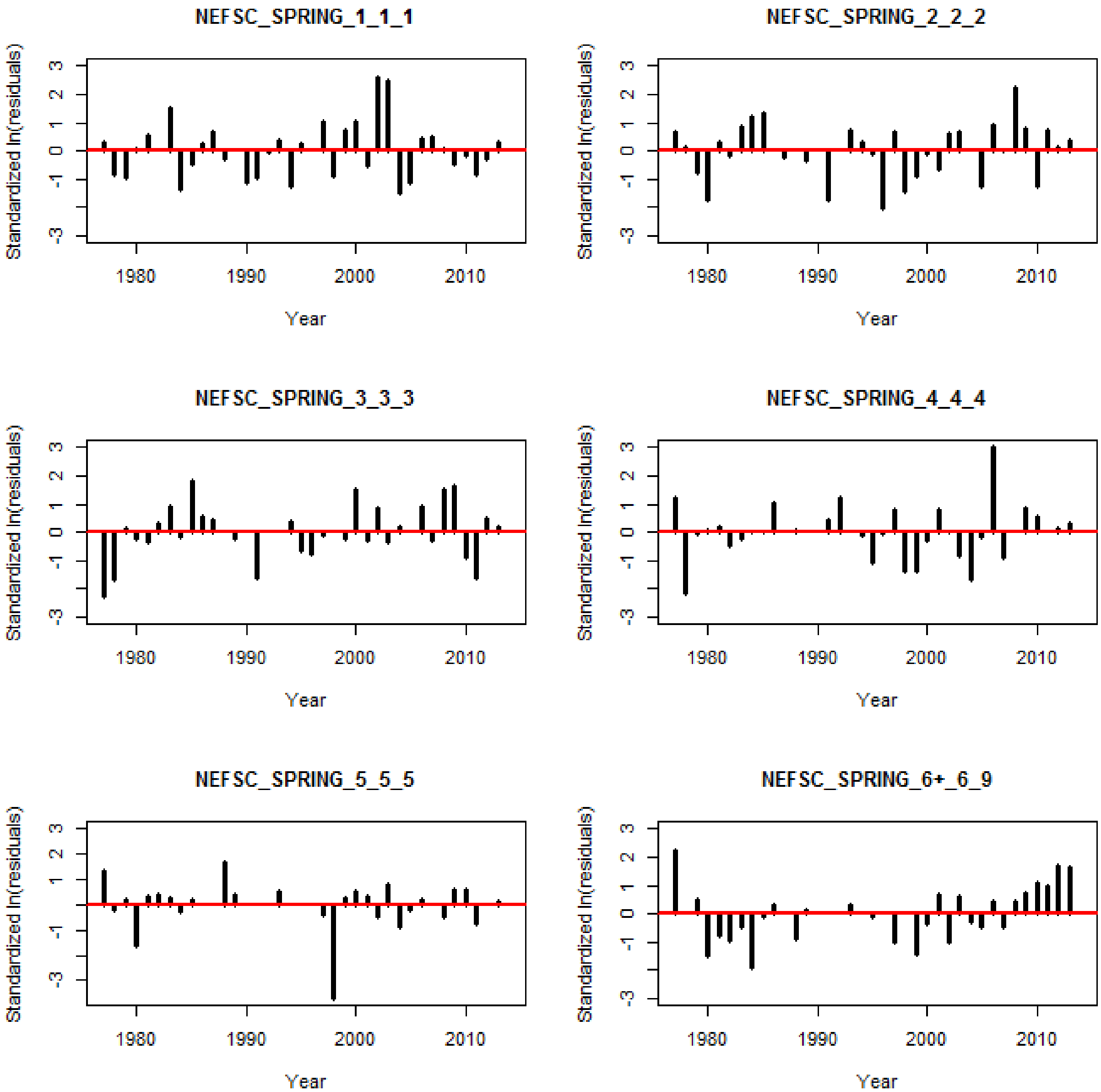


Figure A.145.a. Residuals of the NEFSC survey fits of the 2013 update of the ADAPT-VPA Gulf of Maine haddock assessment model (run 6). Residuals for the NEFSC spring survey age-1 (NEFSC_SPRING_1_1_1) to age-6^{+} (NEFSC_SPRING_6+_6_9) are shown.

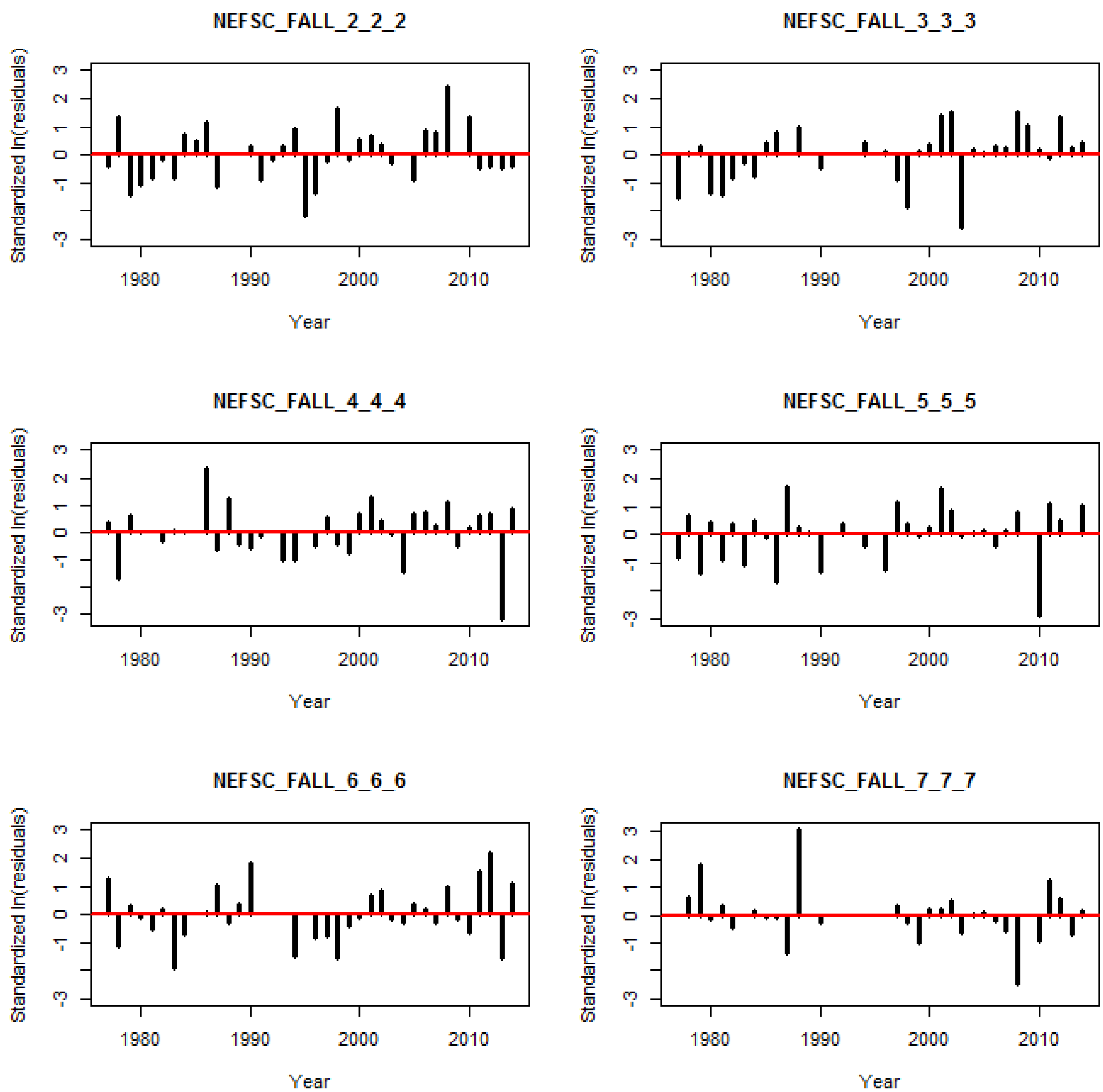
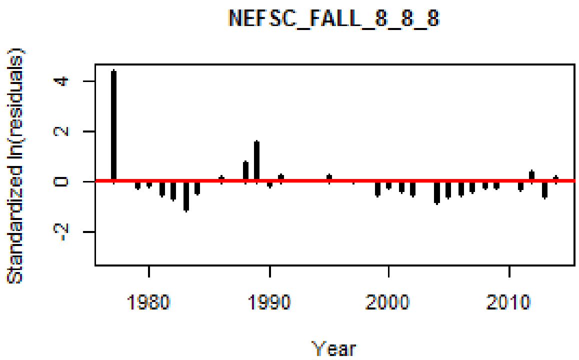


Figure A.145.b. Residuals of the NEFSC survey fits of the 2013 update of the ADAPT-VPA Gulf of Maine haddock assessment model (run 6). Residuals for the NEFSC fall survey age-1 (NEFSC_FALL_2_2_2) to age-6 (NEFSC_FALL_7_7_7) are shown. **Note: fall surveys have been lagged forward a year and an age.*



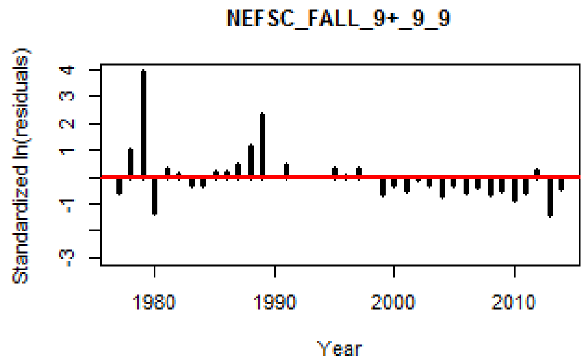


Figure A.145.c. Residuals of the NEFSC survey fits of the 2013 update of the ADAPT-VPA Gulf of Maine haddock assessment model (run 6). Residuals for the NEFSC fall survey ages 7 (NEFSC_FALL_8_8_8) to age 8^+ (NEFSC_FALL_9+_9_9) are shown. **Note: fall surveys have been lagged forward a year and an age.*

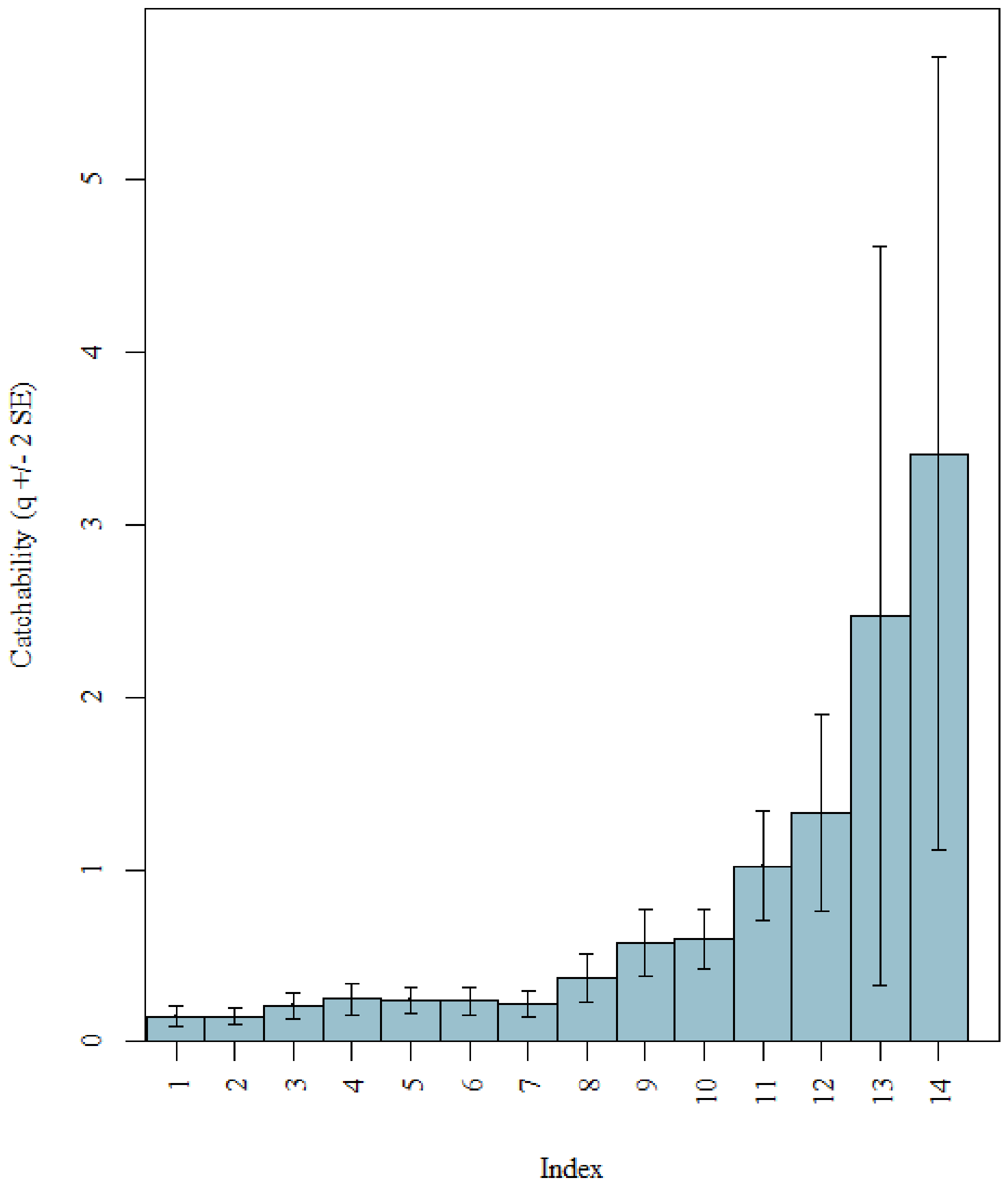


Figure A.146. Patterns in NEFSC survey catchability (q) for the 2013 update of the ADAPT-VPA model (run 6). Indices 1-6=NEFSC spring (ages 2-6^+), indices 7-14=NEFSC fall (ages 1-8^+).

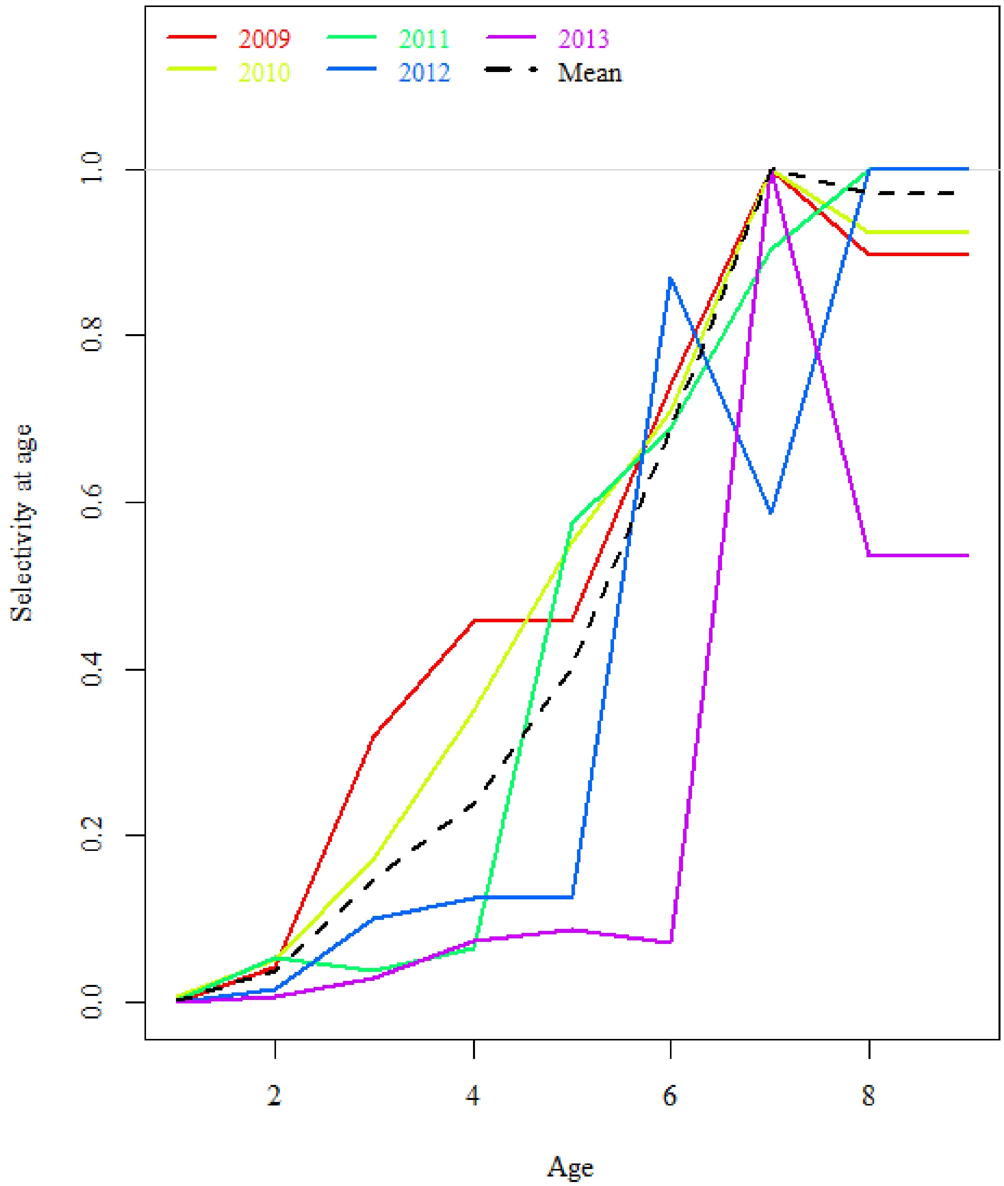


Figure A.147. Catch selectivity patterns for the 2013 update of the ADAPT-VPA Gulf of Maine haddock model (run 6) over the last five years of the model, 2009 through 2013.

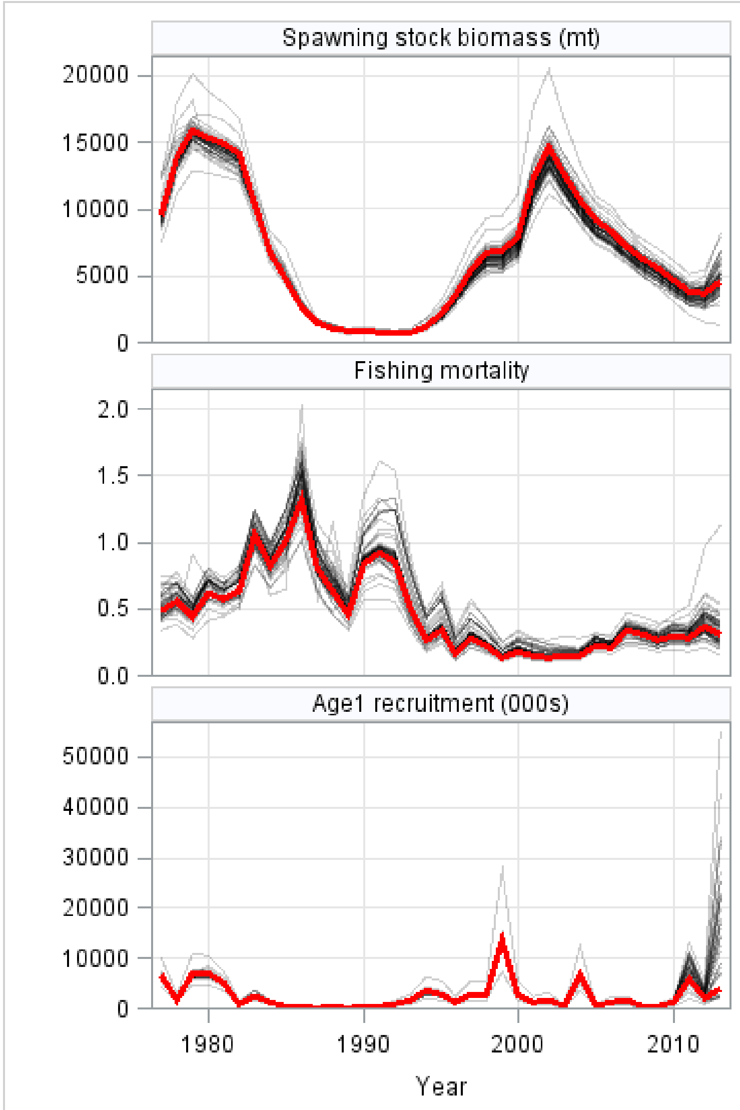


Figure A.148. Estimates of Gulf of Maine haddock spawning stock biomass (top), fishing mortality (middle) and age-1 recruitment (bottom) from the ASAP sensitivity runs conducted (each grey line represents a single sensitivity run). The results of the ASAP_BASE model are shown by a solid red line.

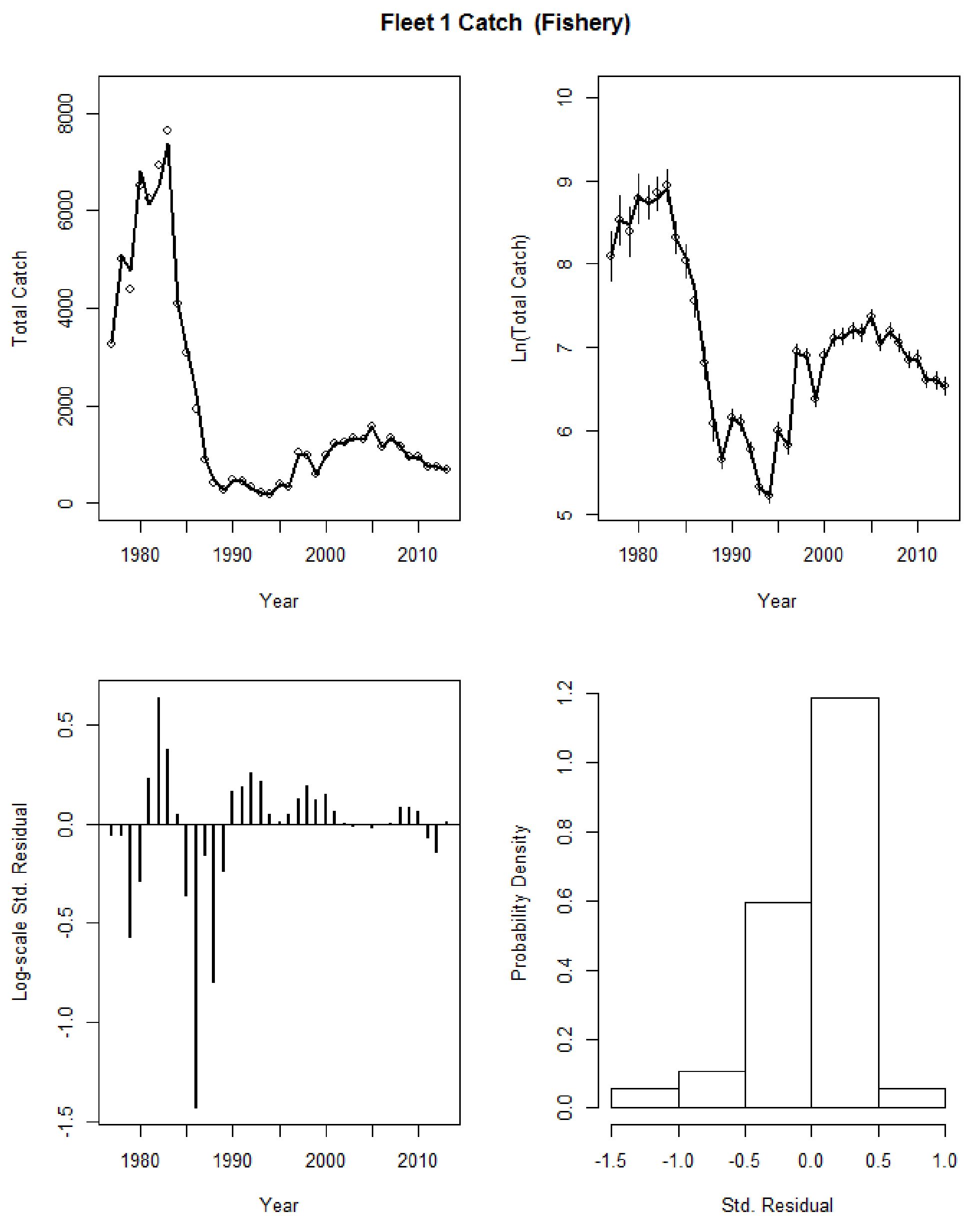


Figure A.149. ASAP_BASE model fit to the total Gulf of Maine haddock fishery catch (Fleet 1).

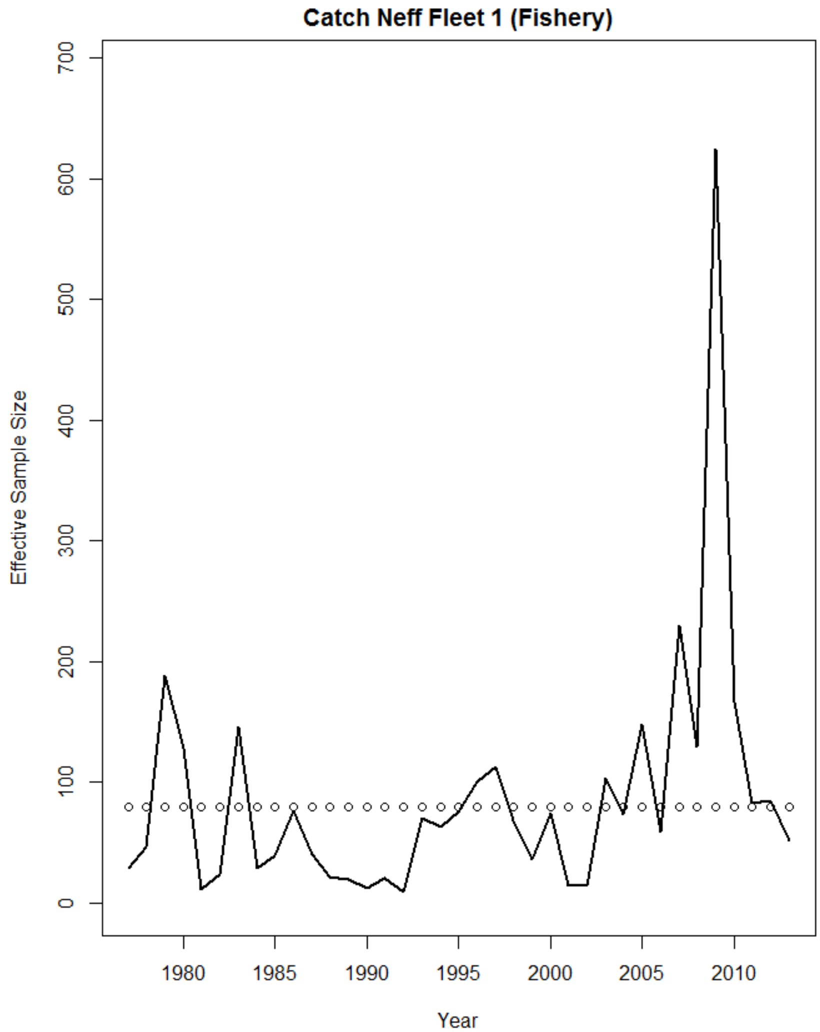


Figure A.150. ASAP_BASE model comparison of input effective sample size versus the model estimated effective sample size for the Gulf of Maine haddock fishery (Fleet 1) catch.

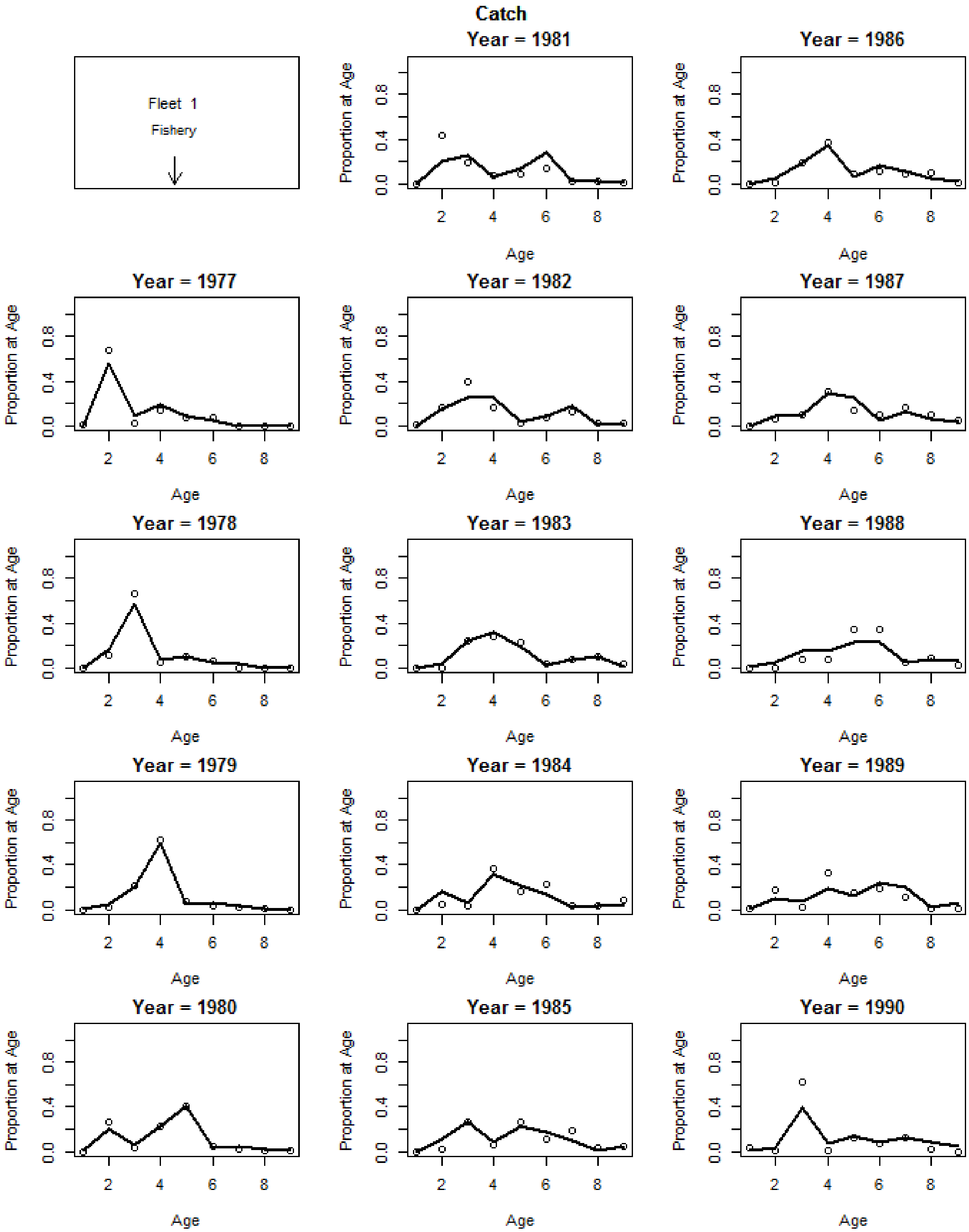


Figure A.151.a. Comparison of the ASAP_BASE estimates of Gulf of Maine haddock proportion-at-age in the fishery to the data estimates.

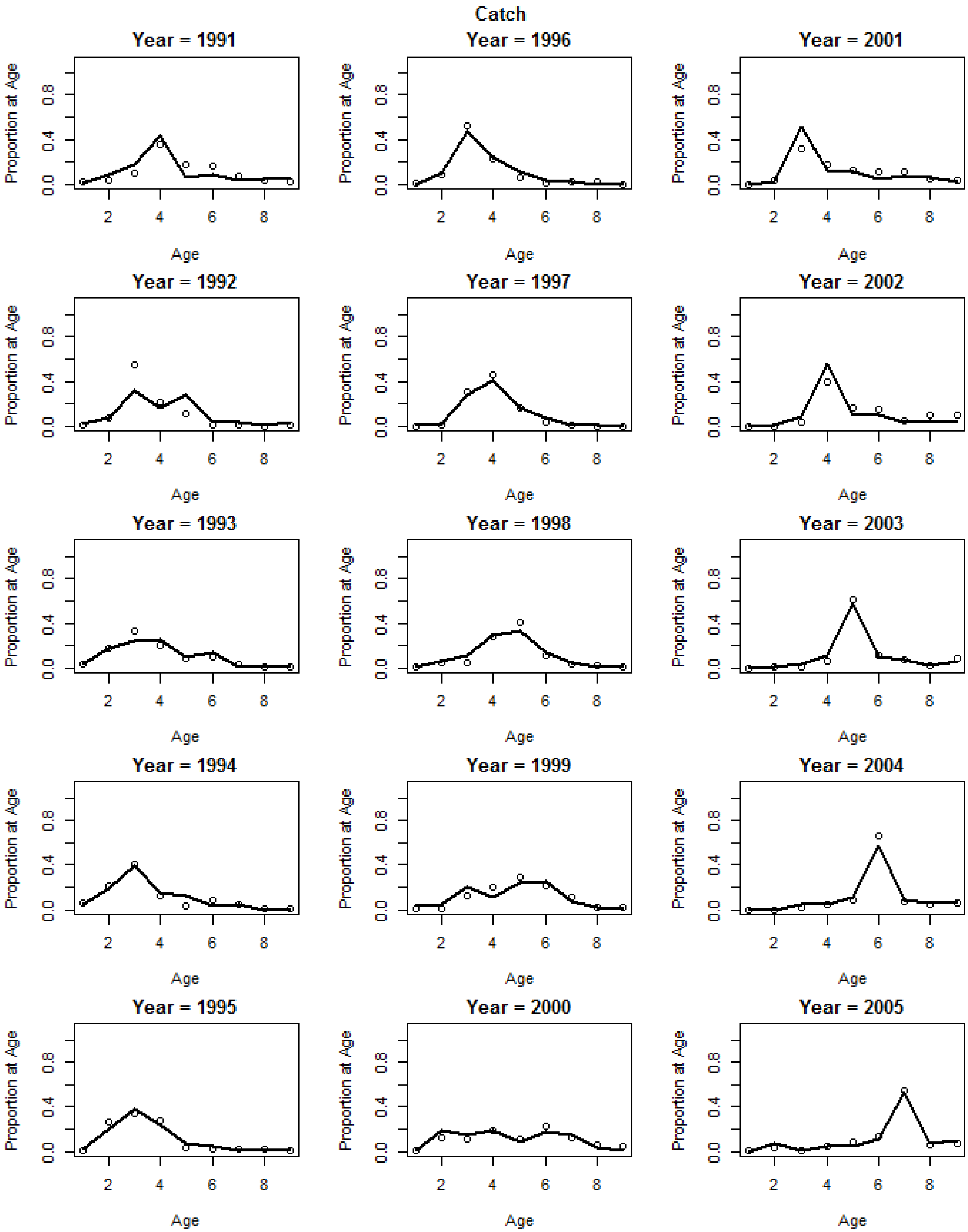


Figure A.151.b. Comparison of the ASAP_BASE estimates of Gulf of Maine haddock proportion-at-age in the fishery to the data estimates.

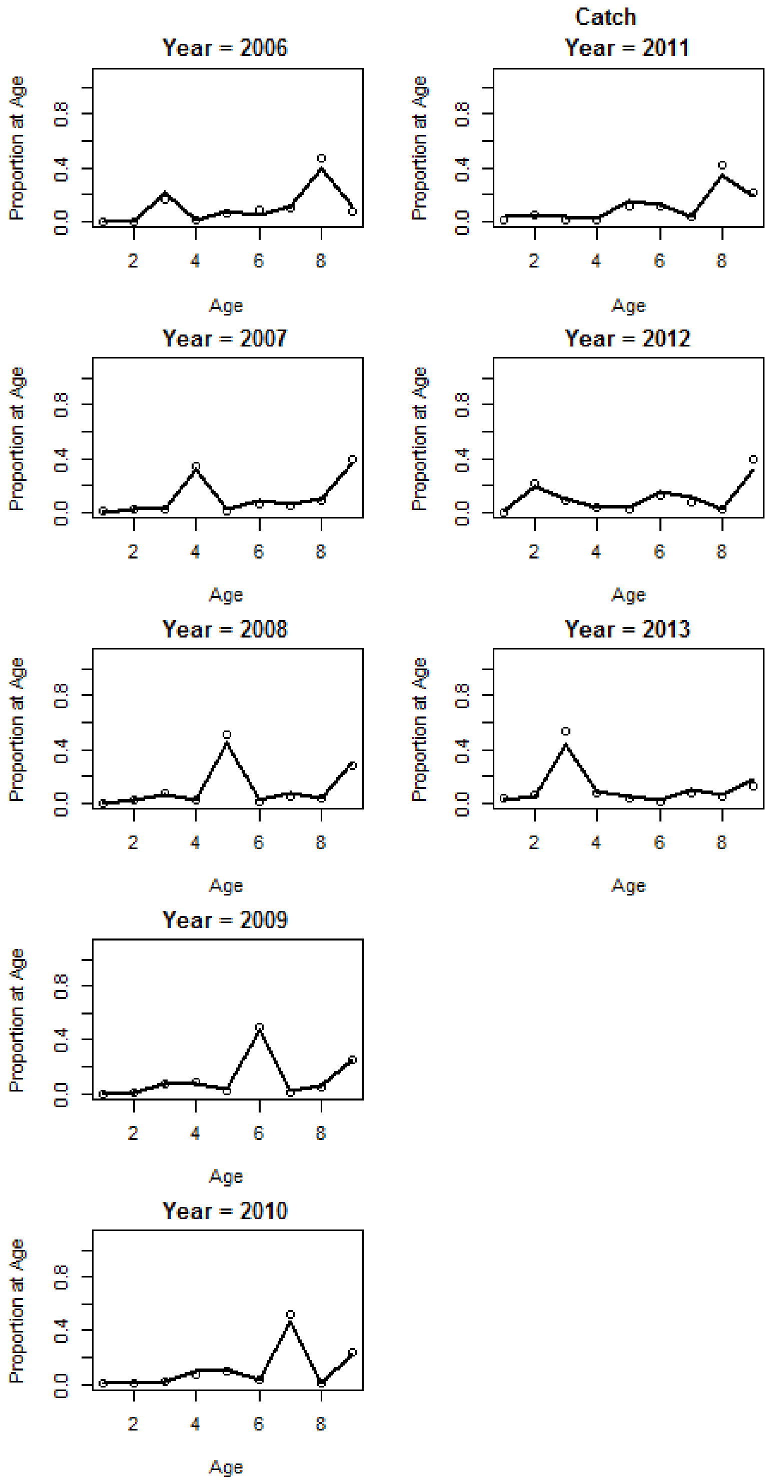


Figure A.151.c. Comparison of the ASAP_BASE estimates of Gulf of Maine haddock proportion-at-age in the fishery to the data estimates.

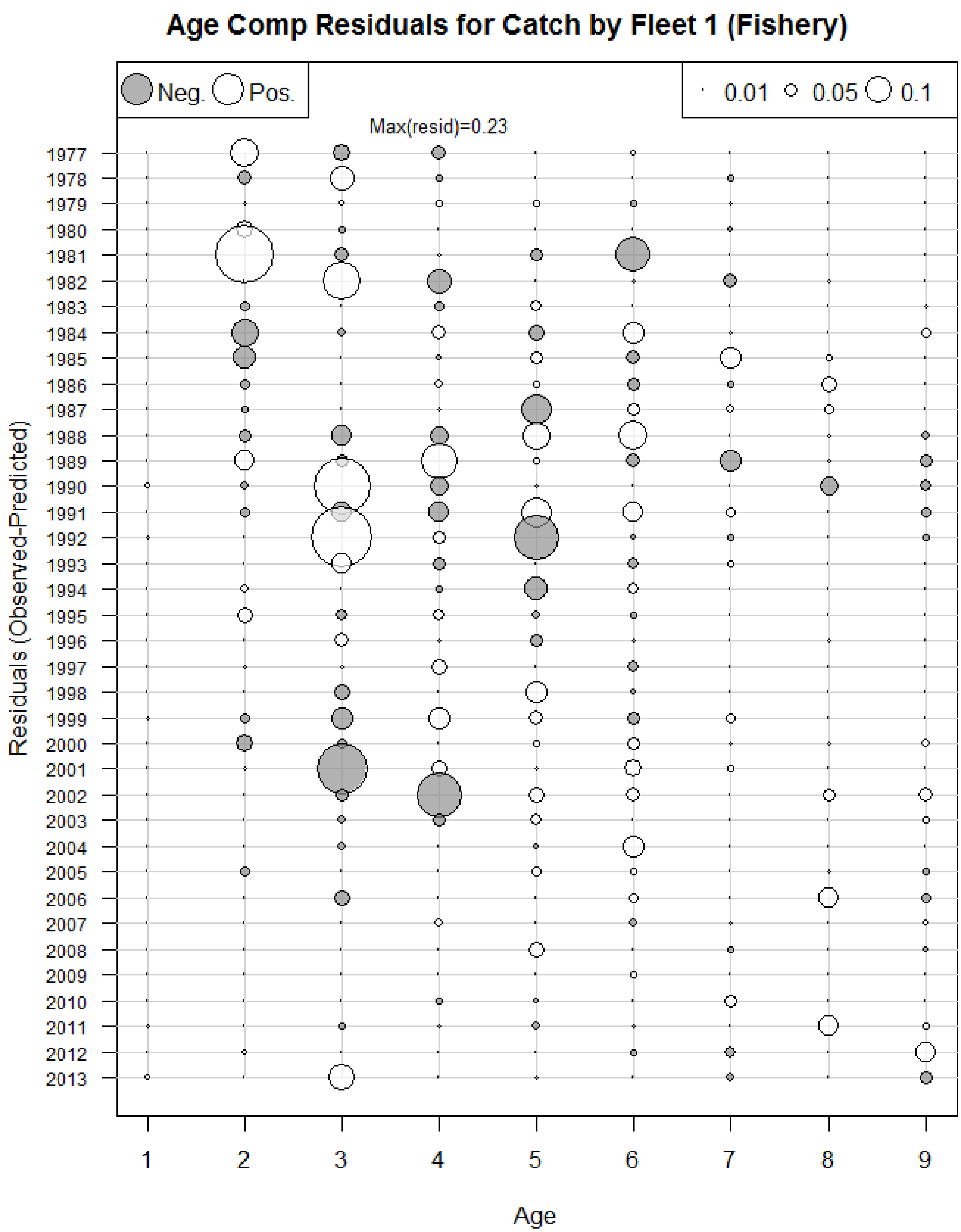


Figure A.152. ASAP_BASE model fit residuals for the Gulf of Maine haddock fishery (Fleet 1) catch-at-age.

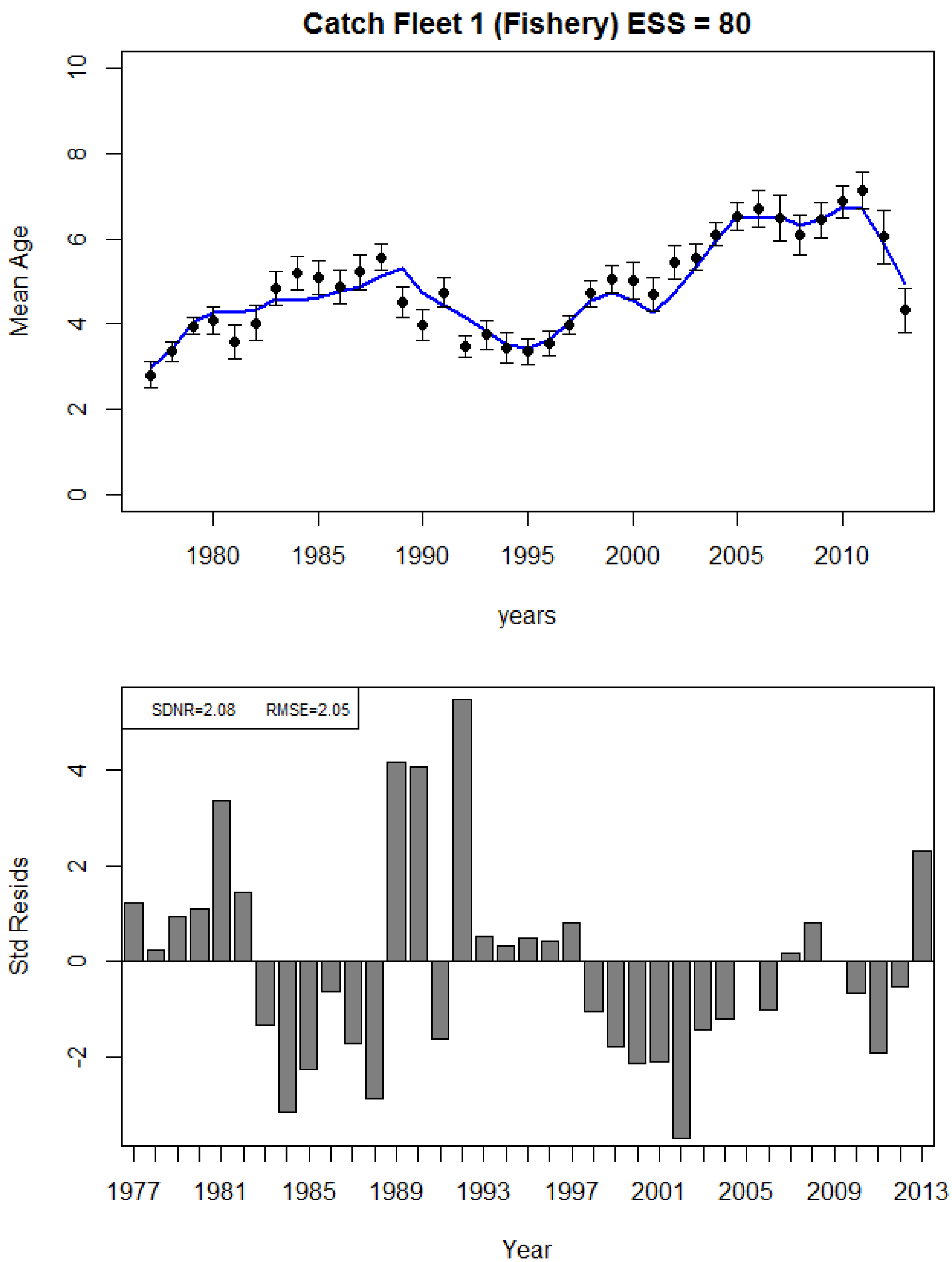


Figure A.153. ASAP_BASE predicted mean age of Gulf of Maine haddock in the fishery (Fleet 1) catch (blue line) compared to observed mean age (top plot) and the residuals about the mean (bottom plot).

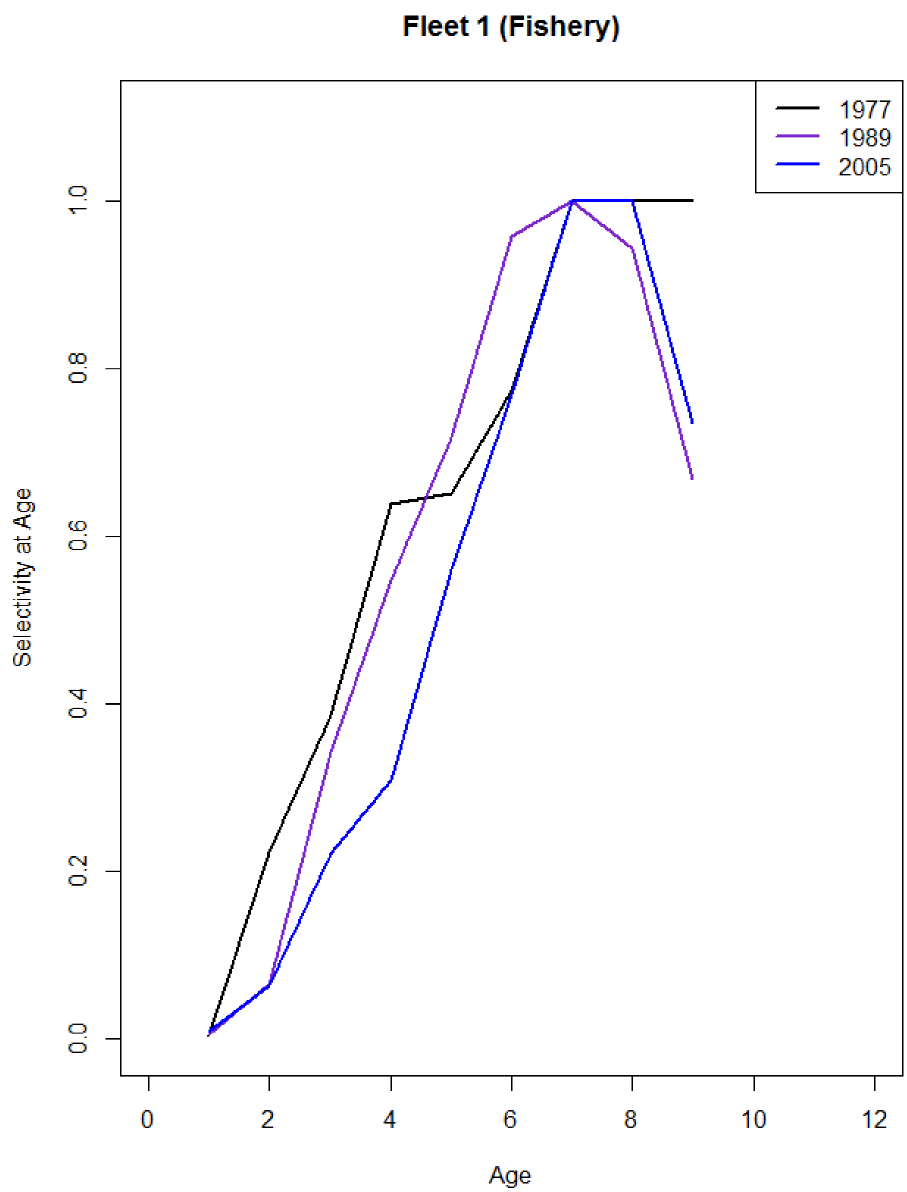


Figure A.154. Gulf of Maine haddock fishery (Fleet 1) selectivity-at-age for block 1 (1977-1988), block 2 (1989-2004) and block 3 (2005-2013) as estimated by the ASAP_BASE model.

Index 1 (NEFSC spring (SHG/TOGA))

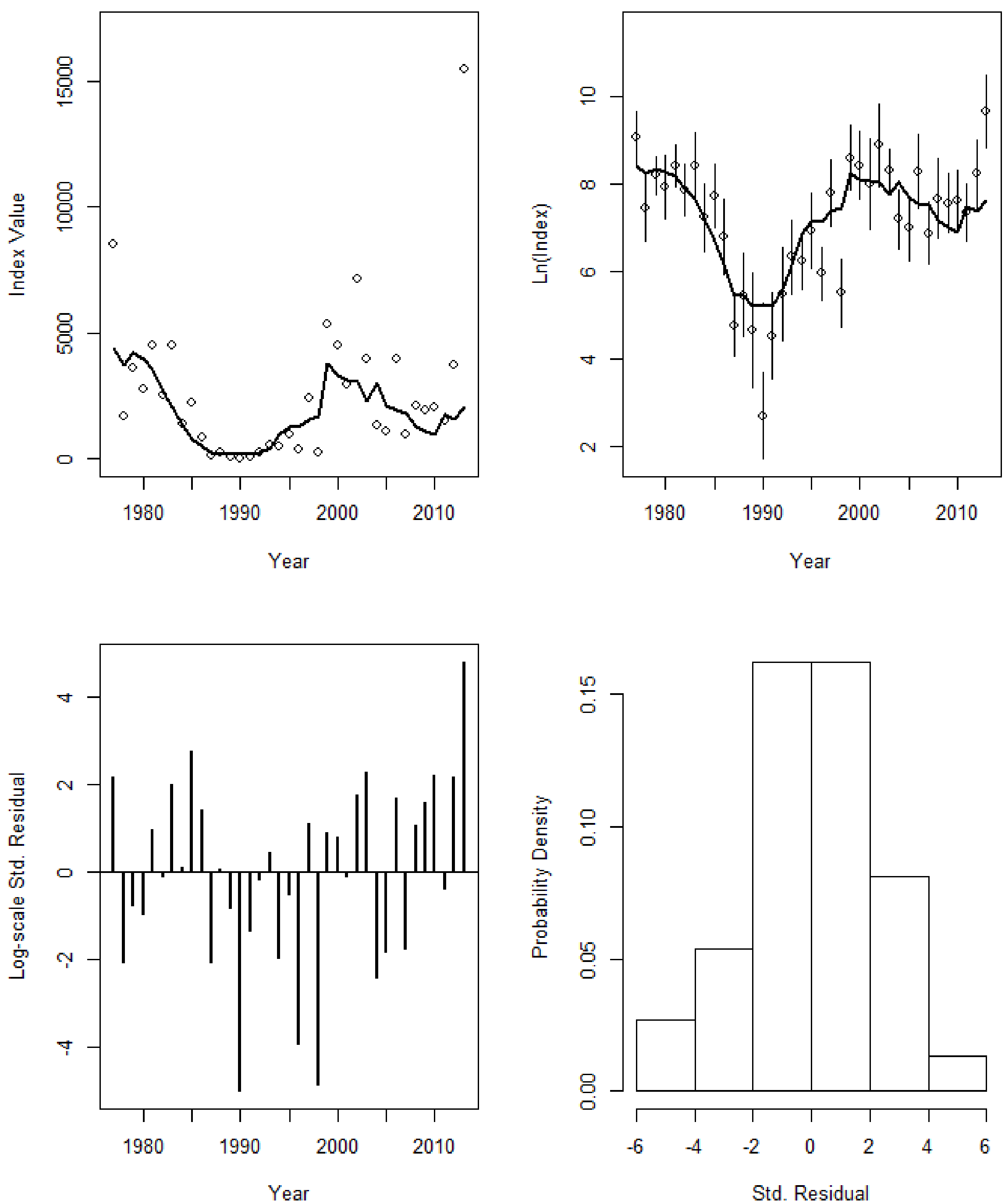


Figure A.155. ASAP_BASE model fit to the NEFSC spring survey Gulf of Maine haddock index (Index 1).

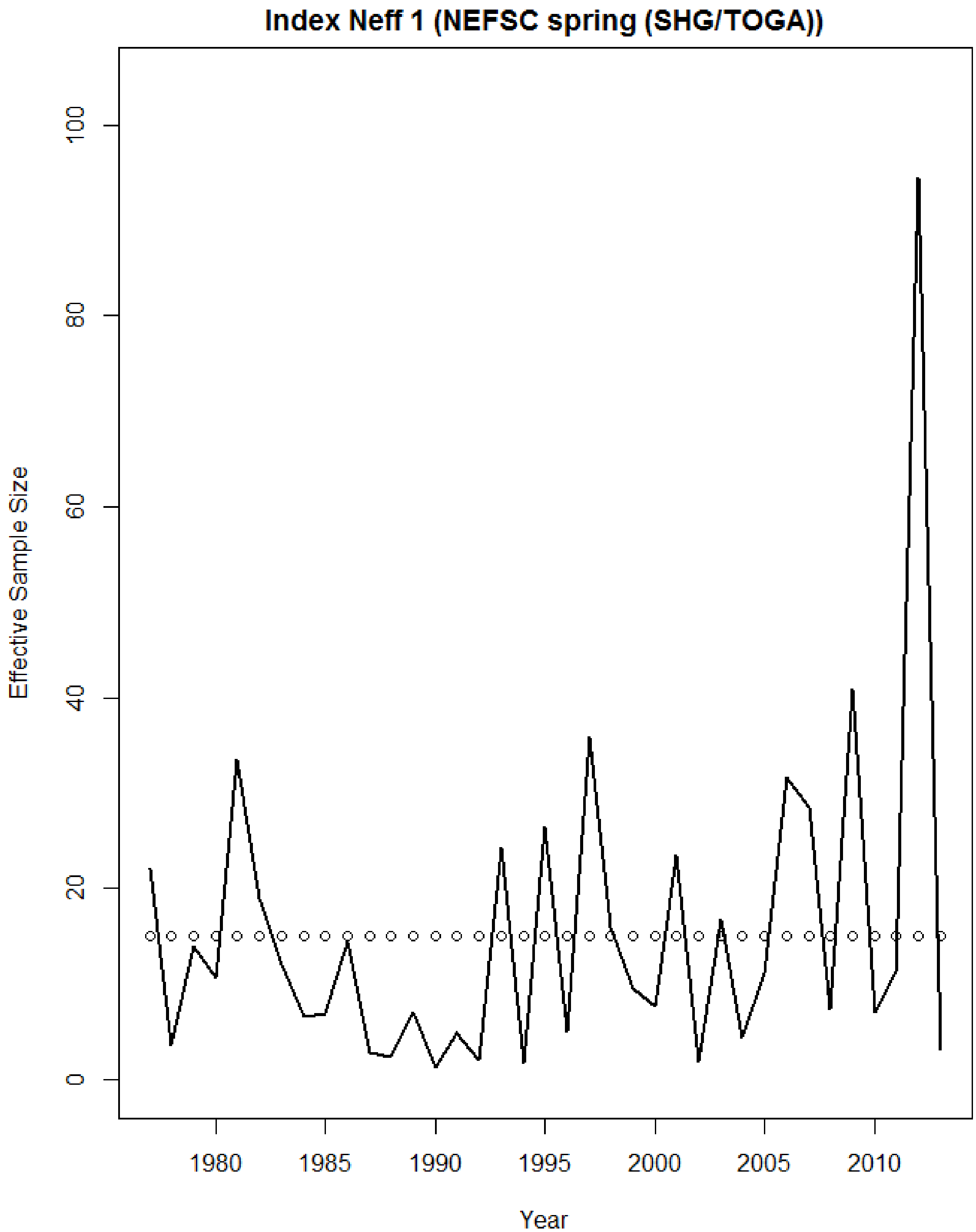


Figure A.156. ASAP_BASE model comparison of input effective sample size versus the model estimated effective sample size for the NEFSC spring survey Gulf of Maine haddock index (Index 1).

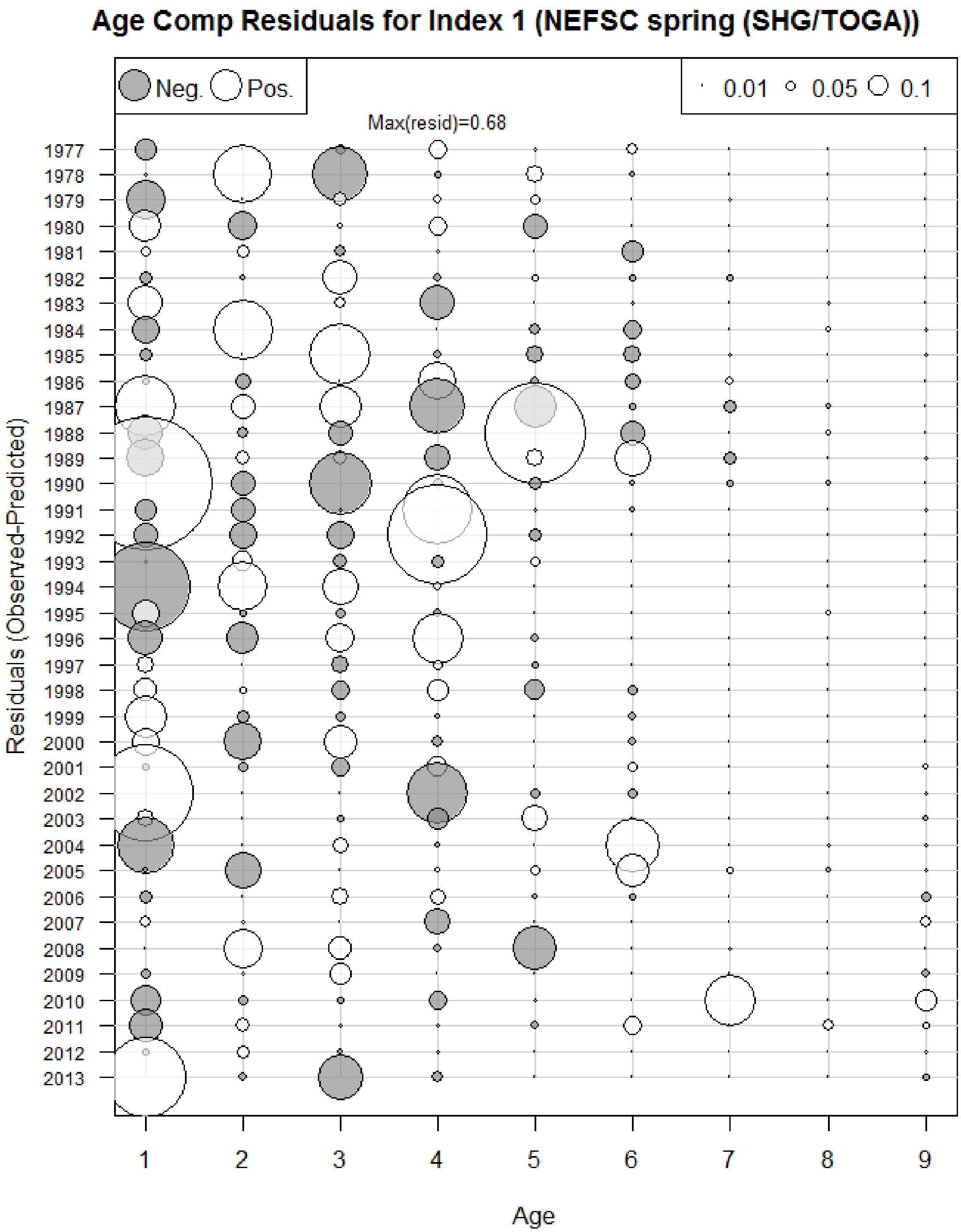


Figure A.157. ASAP_BASE model fit residuals of model fits to the NEFSC spring survey Gulf of Maine haddock indices-at-age (Index 1).

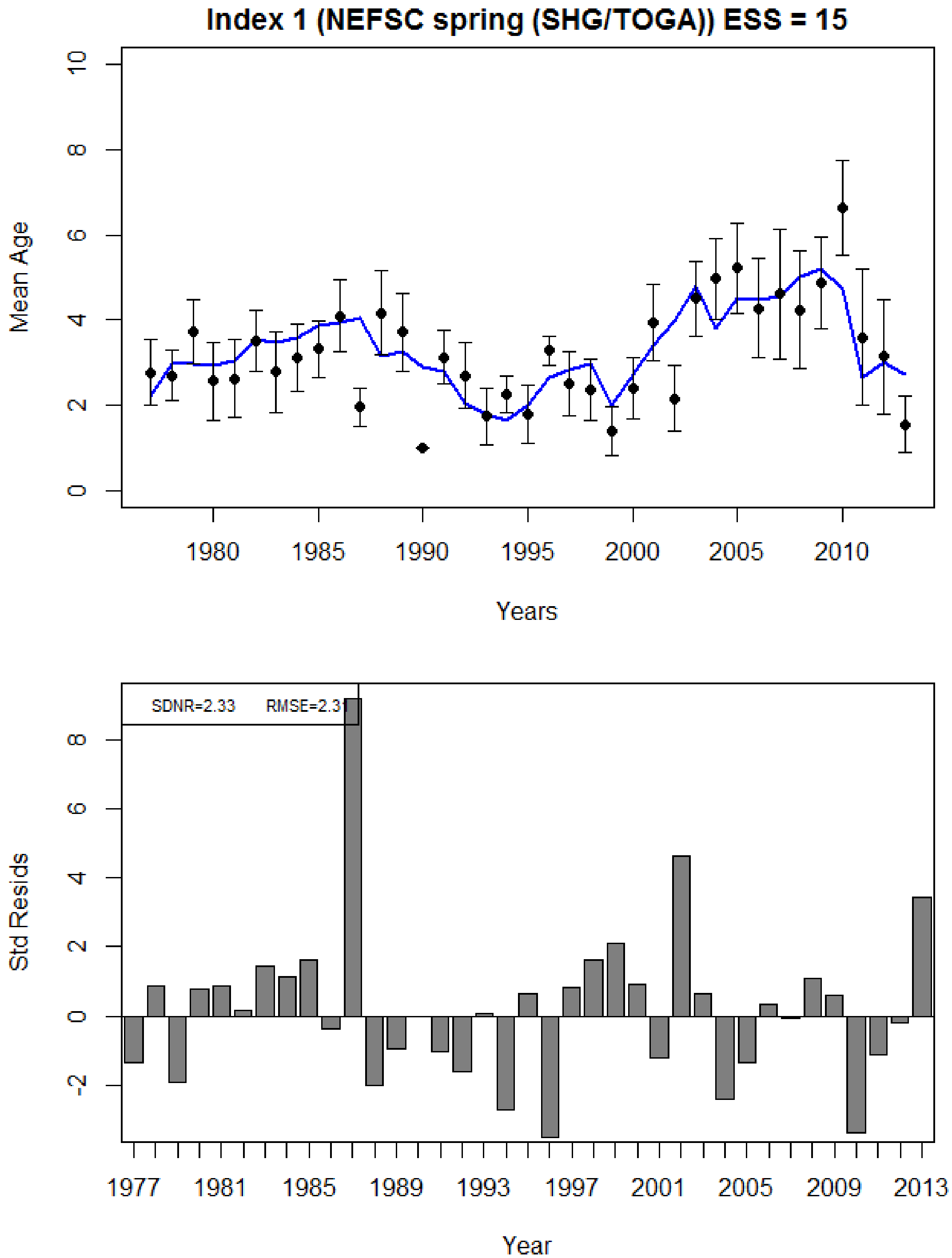


Figure A.158. ASAP_BASE predicted mean age of Gulf of Maine haddock in the NEFSC spring (Index 1) survey (blue line) compared to observed mean age (top plot) and the residuals about the mean (bottom plot).

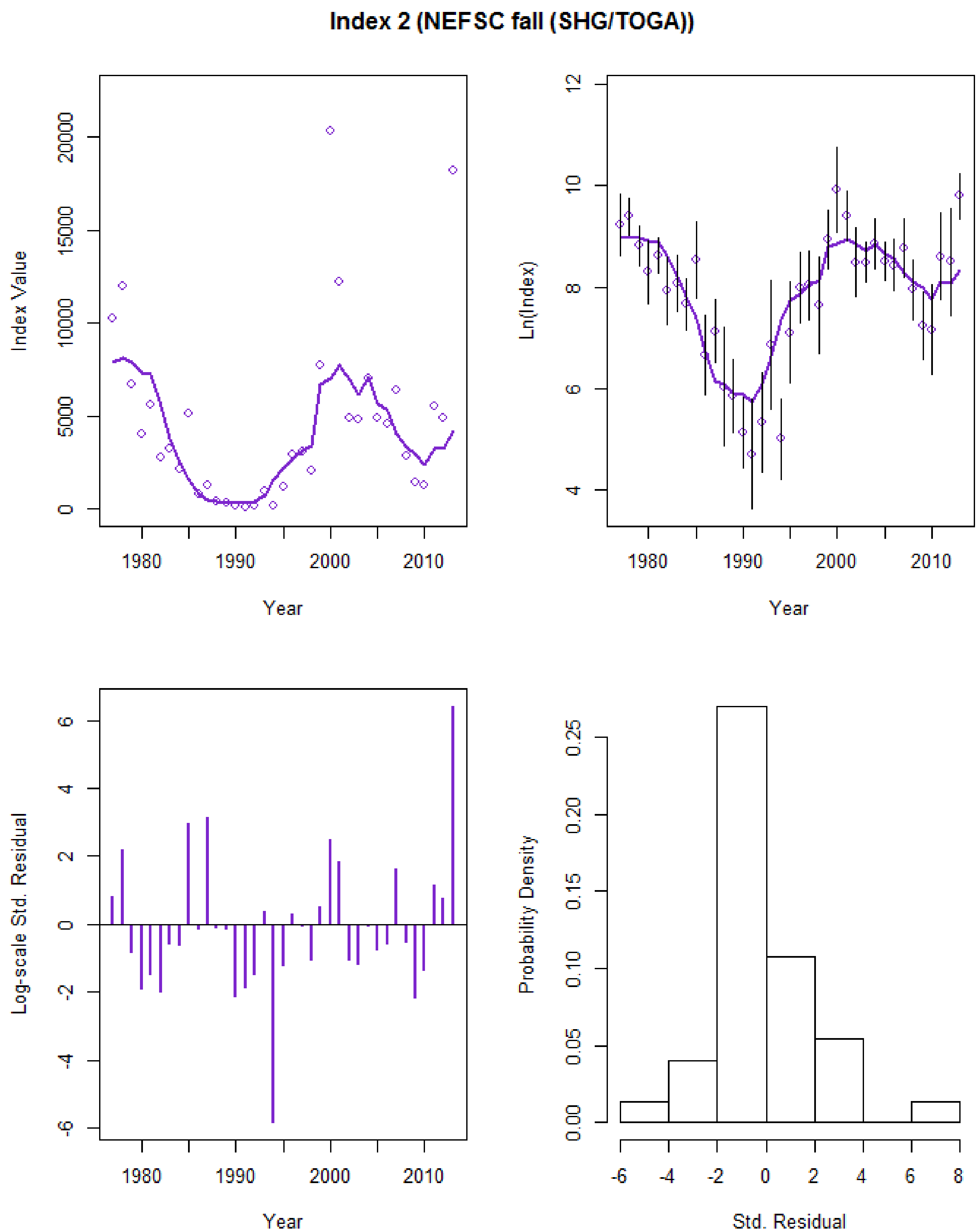


Figure A.159. ASAP_BASE model fit to the NEFSC fall survey Gulf of Maine haddock index (Index 2).

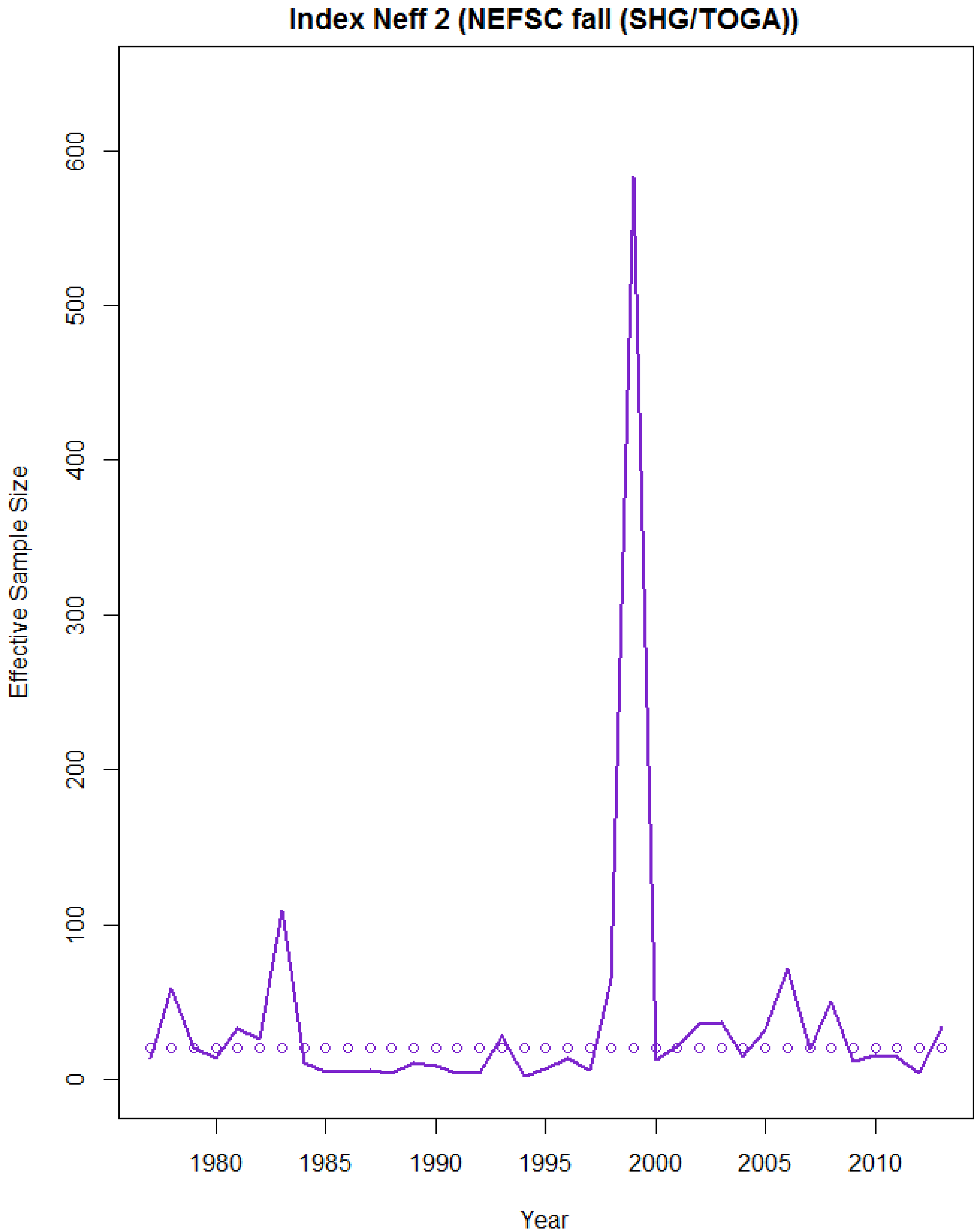


Figure A.160. ASAP_BASE model comparison of input effective sample size versus the model estimated effective sample size for the NEFSC fall survey Gulf of Maine haddock index (Index 2).

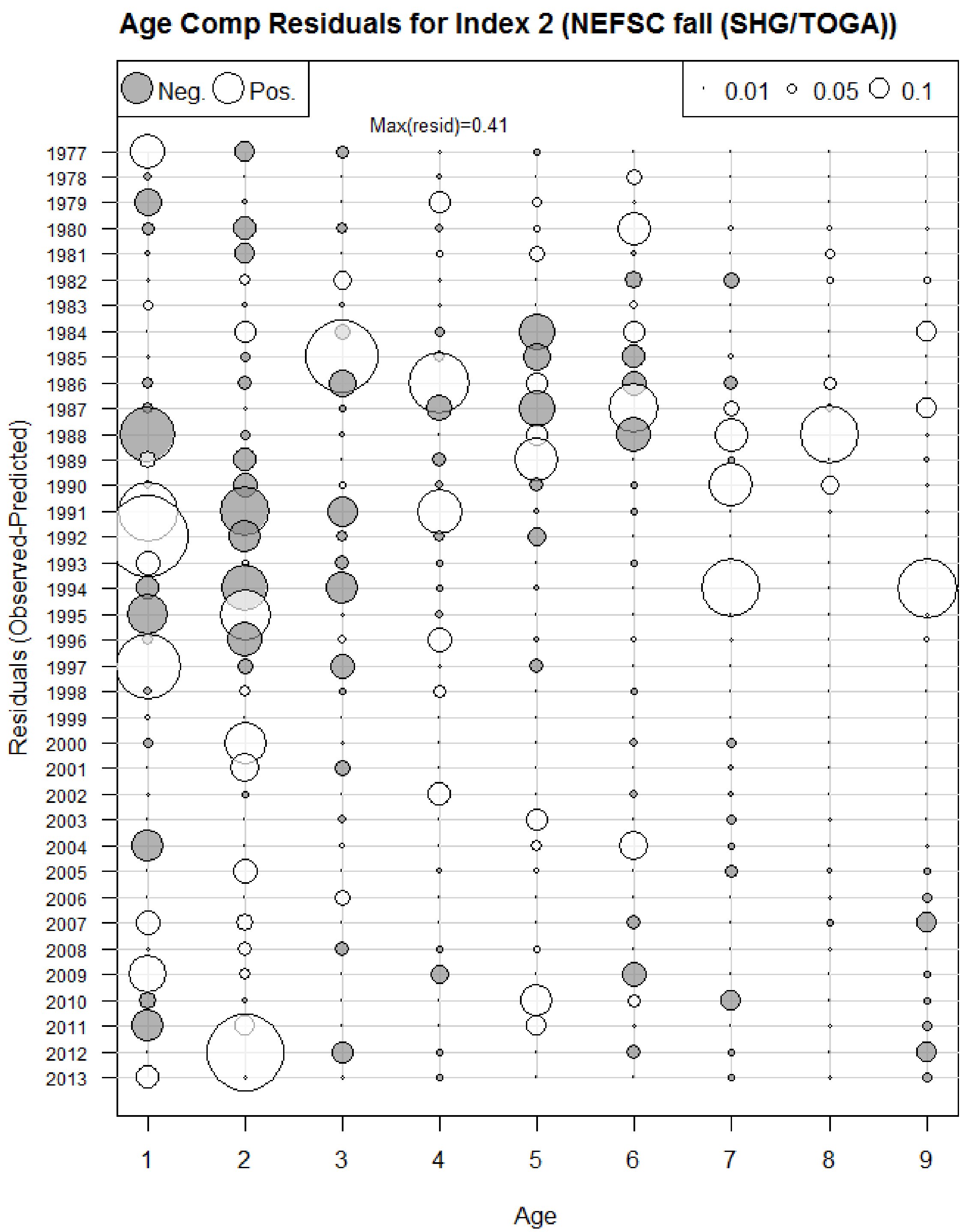


Figure A.161. ASAP_BASE model fit residuals of model fits to the NEFSC fall survey Gulf of Maine haddock indices-at-age (Index 2).

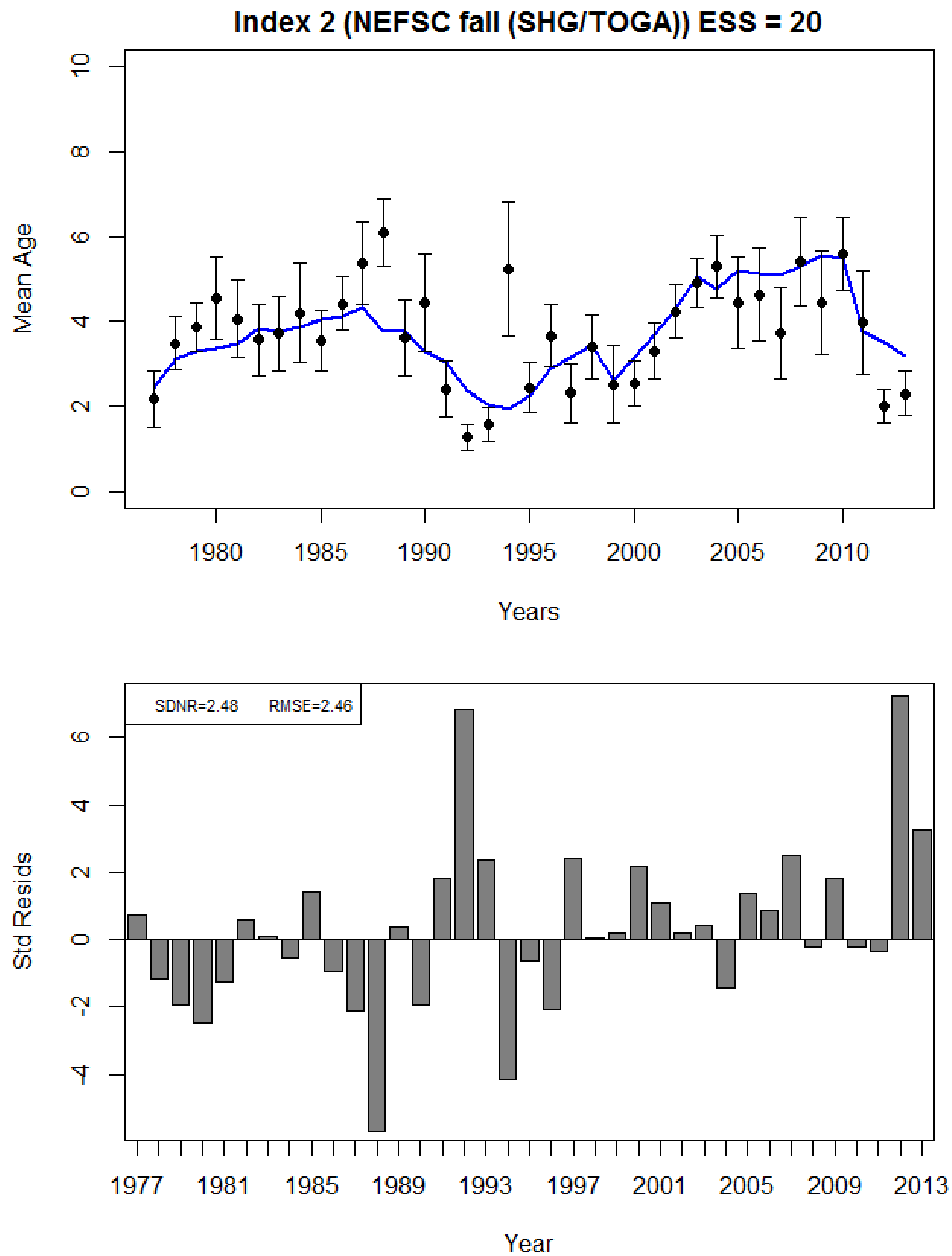


Figure A.162. ASAP_BASE predicted mean age of Gulf of Maine haddock in the NEFSC fall (Index 2) survey (blue line) compared to observed mean age (top plot) and the residuals about the mean (bottom plot).

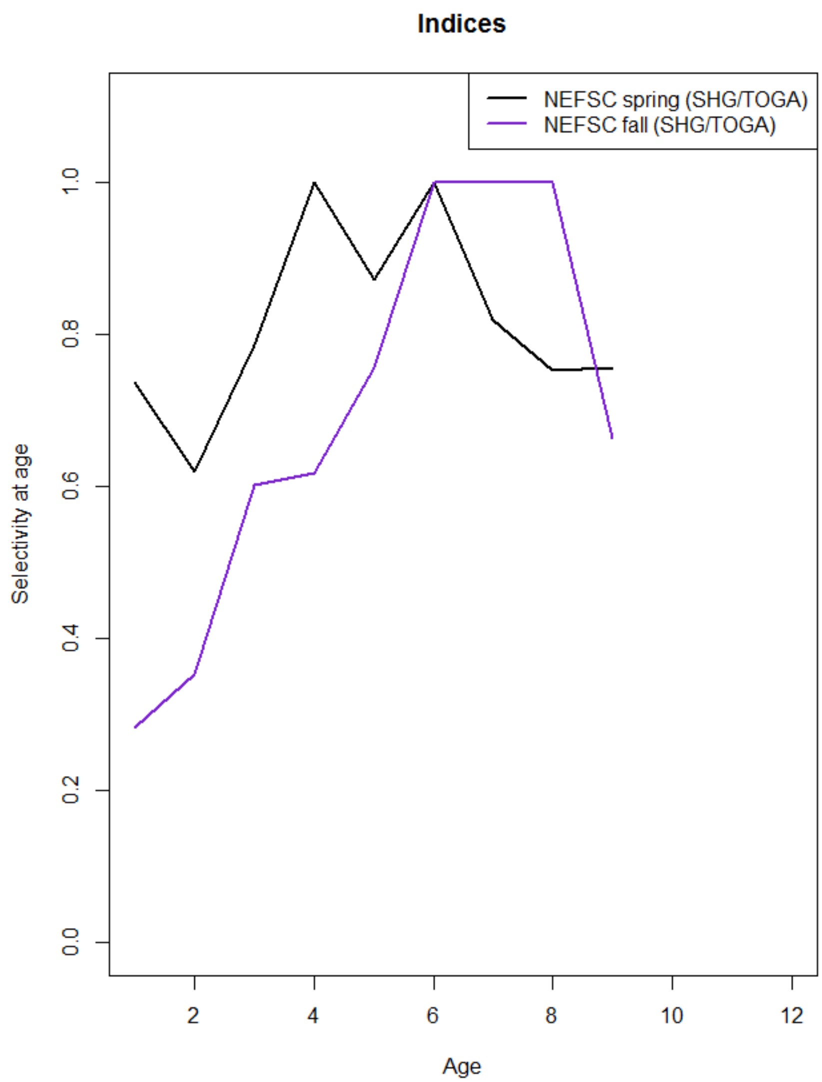


Figure A.163. Gulf of Maine haddock selectivity-at-age for the NEFSC spring (Index 1), and fall (Index 2) from the ASAP_BASE model.

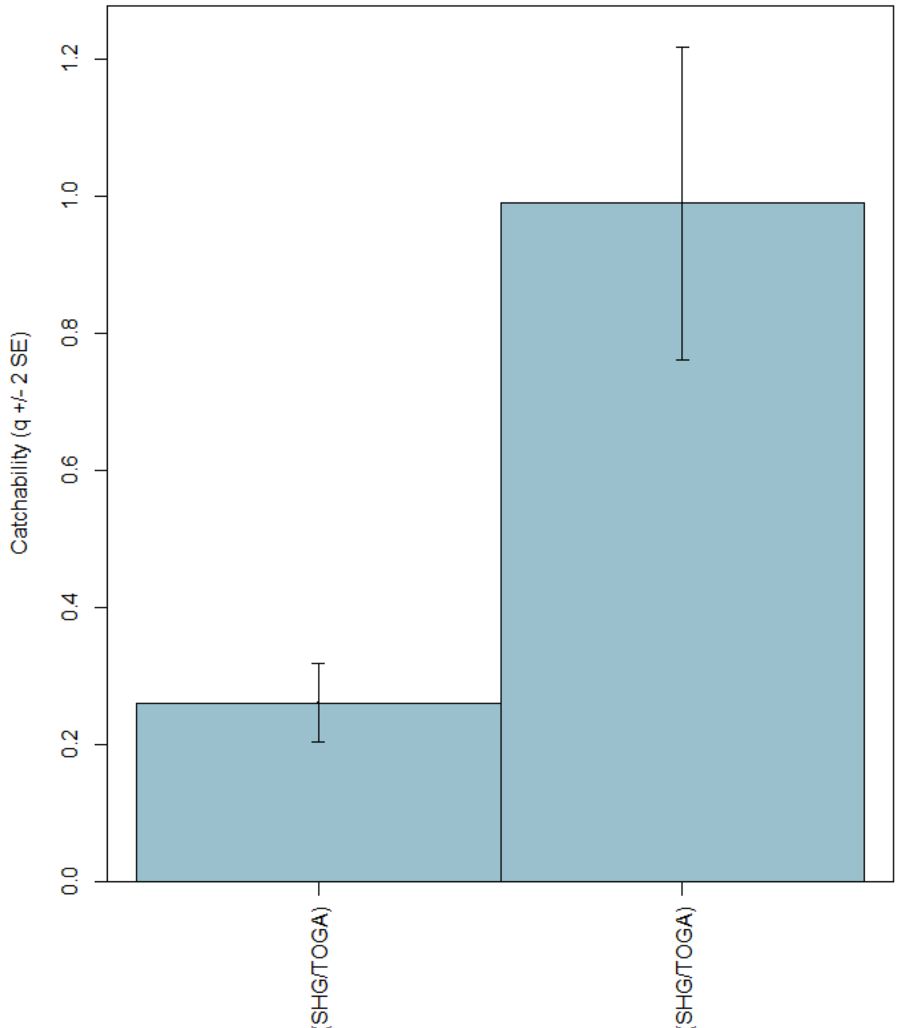


Figure A.164. Gulf of Maine haddock survey catchability, *q*, for the NEFSC spring (Index 1), and fall (Index 2) surveys from the ASAP_BASE model.

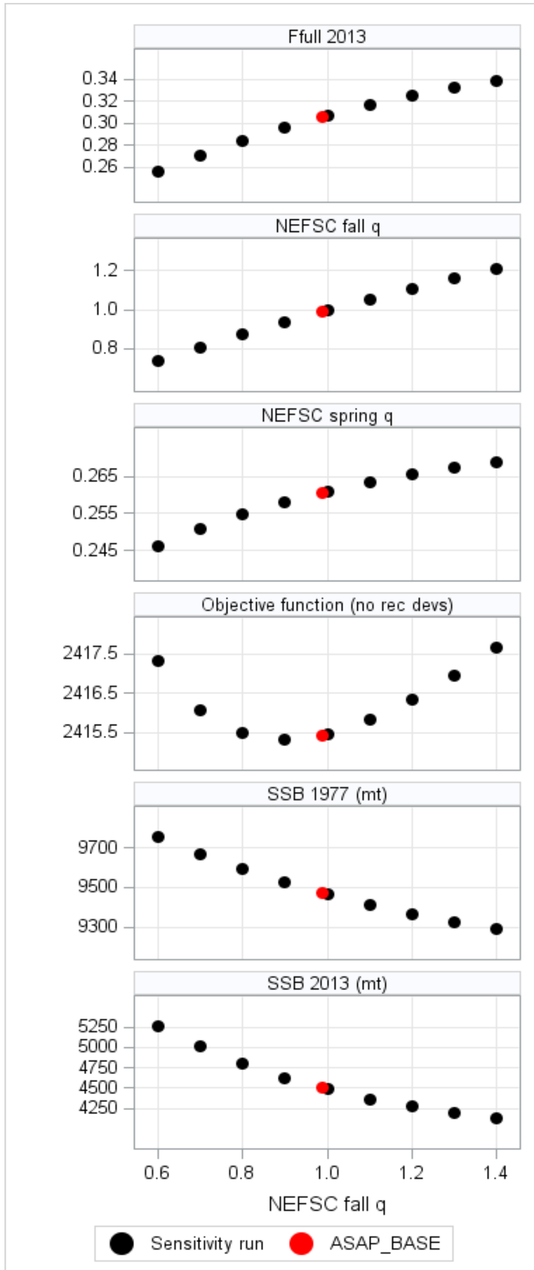


Figure A.165. Sensitivity analysis showing the response of the Gulf of Maine haddock ASAP_BASE model to different assumptions of survey catchability (*q*) of the Northeast Fisheries Science Center fall survey.

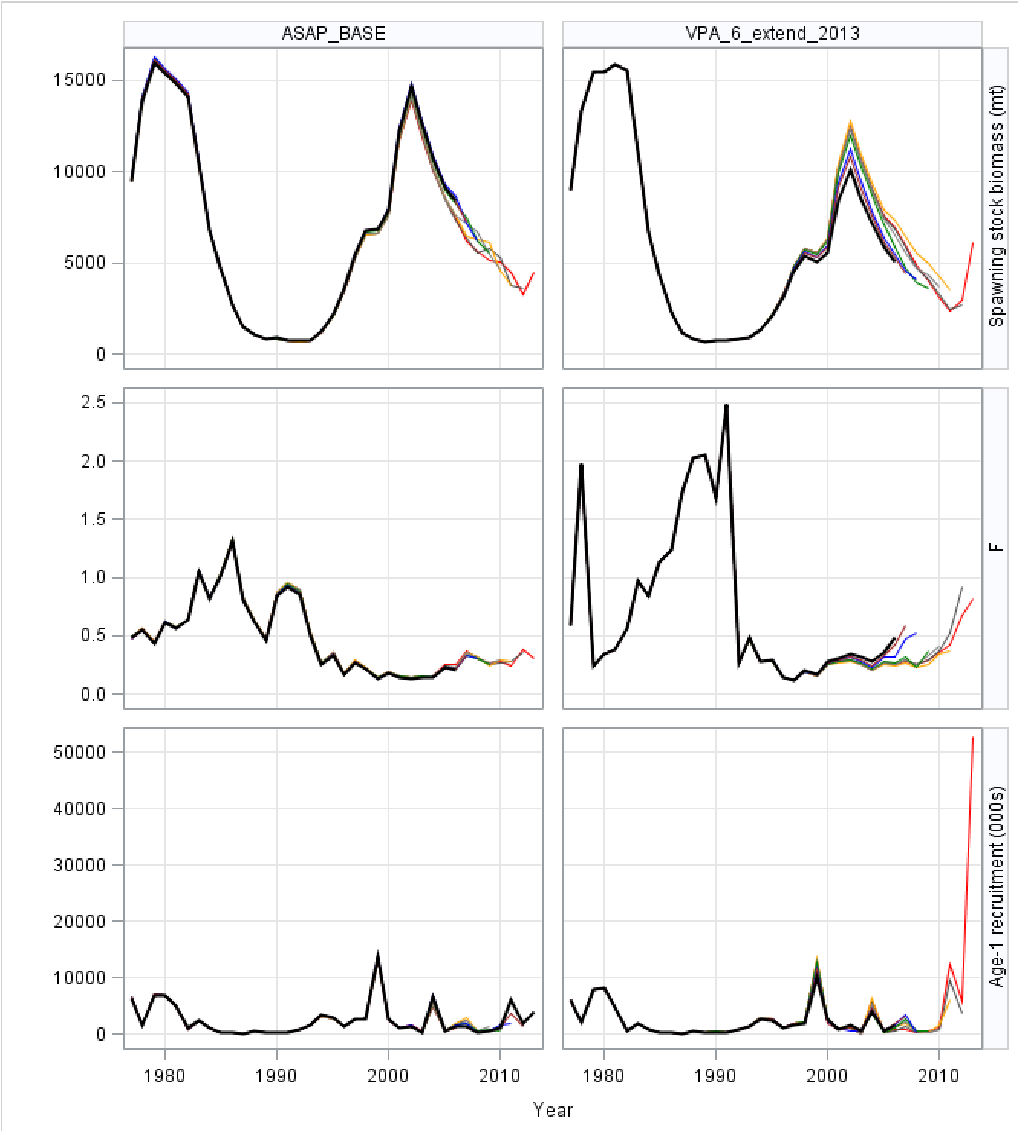


Figure A.166. Comparison of the time series of spawning stock biomass, fishing mortality and recruitment estimates from the 2013 update of the VPA model (VPA_6_extend_2013) to the base ASAP model (ASAP_BASE). Seven years of retrospective peels are shown for both model runs. *Note that the fishing mortality basis is different between the ASAP (F_{full}) and VPA (avg. $F_{6\text{-}8}$) runs.*

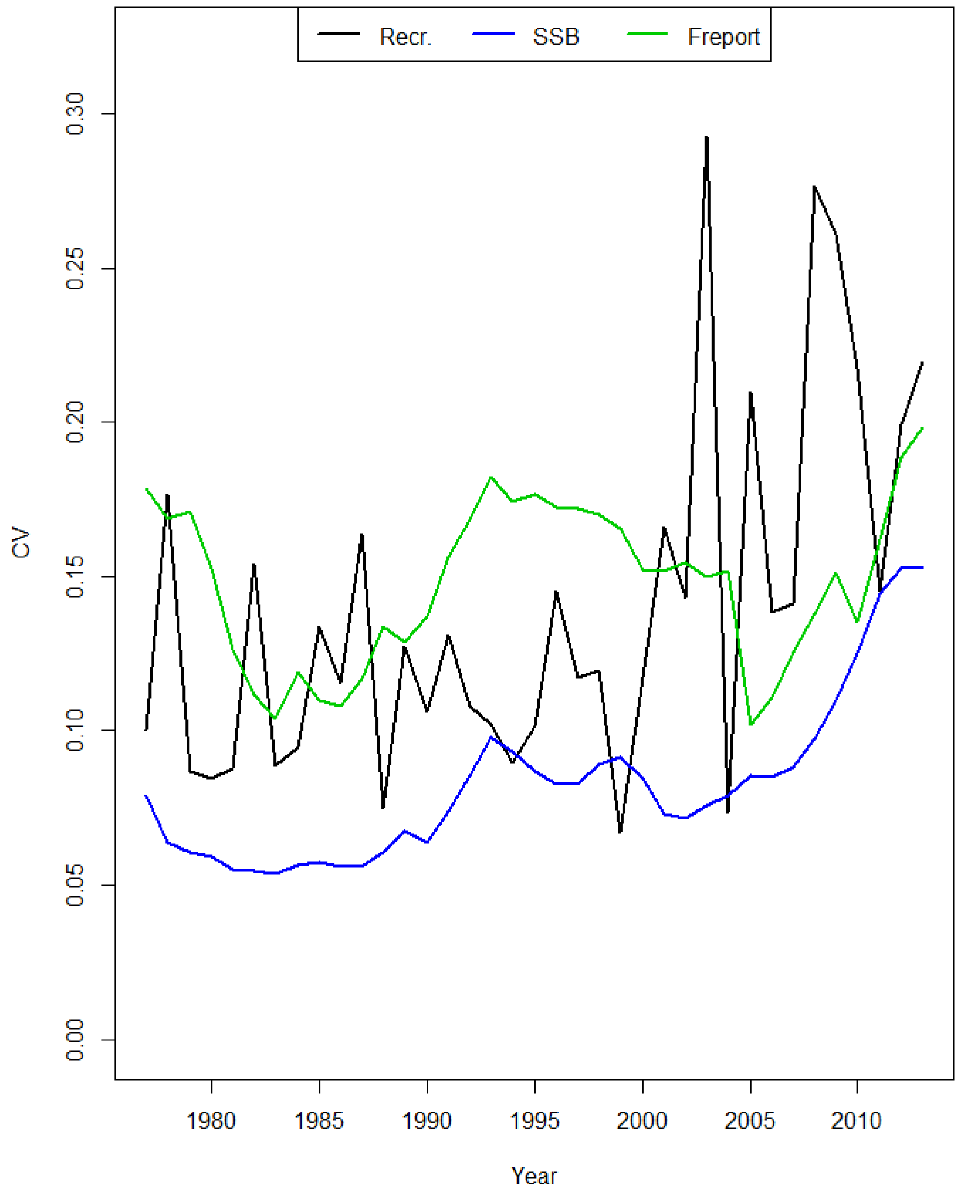


Figure A.167. Hessian-based coefficients of variation (CV) for the ASAP_BASE model estimates of Gulf of Maine haddock spawning stock biomass (SSB), fishing mortality (F_{full}) and age-1 recruitment.

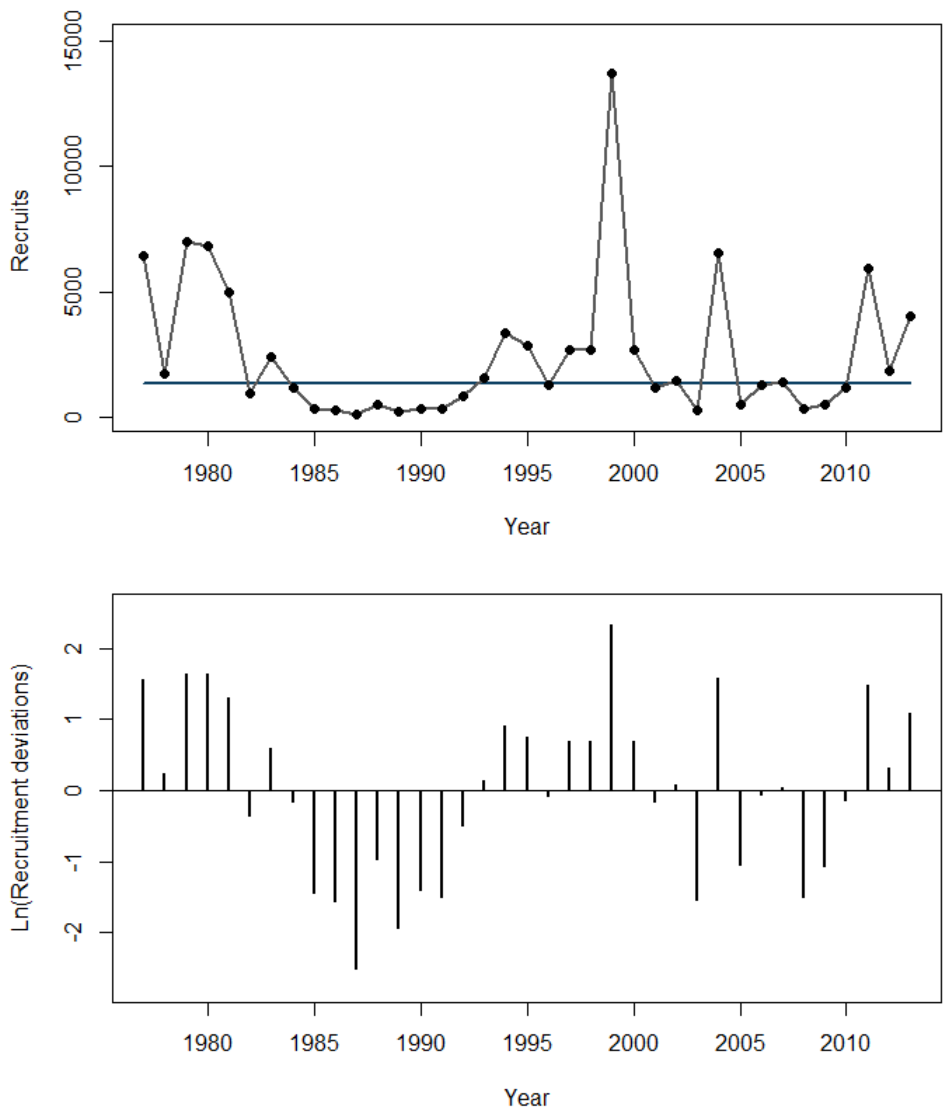


Figure A.168. ASAP_BASE estimated Gulf of Maine haddock age-1 recruitment and recruitment residuals from the geometric mean.

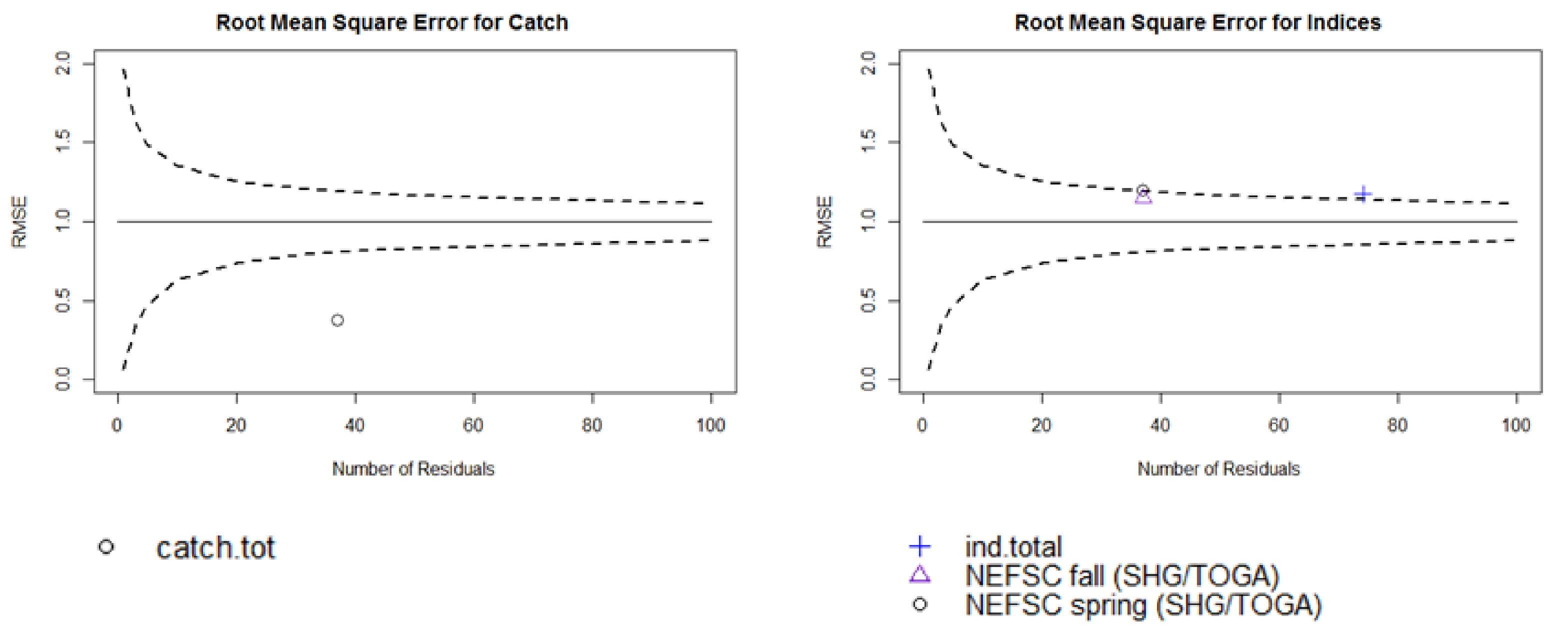


Figure A.169. ASAP_final_temp10 model fit root mean square error (RMSE) values for total catch (left) and survey indices (right) as a function of the number of residuals. The median and 80% confidence interval of the root mean square error from a normal distribution with mean zero and standard deviation one for a range of sample sizes is overlaid on the model RMSE values for reference.

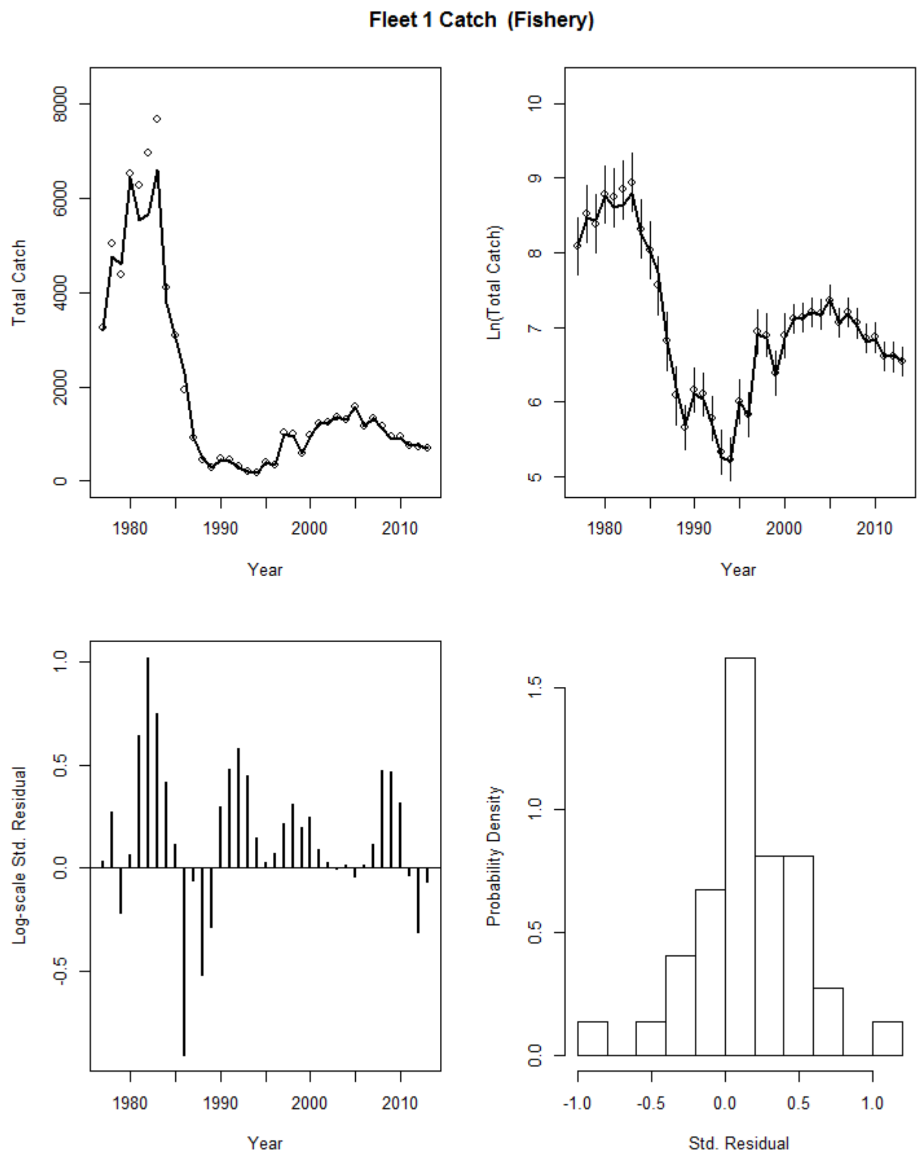


Figure A.170. ASAP_final_temp10 model fit to the total Gulf of Maine haddock fishery catch (Fleet 1).

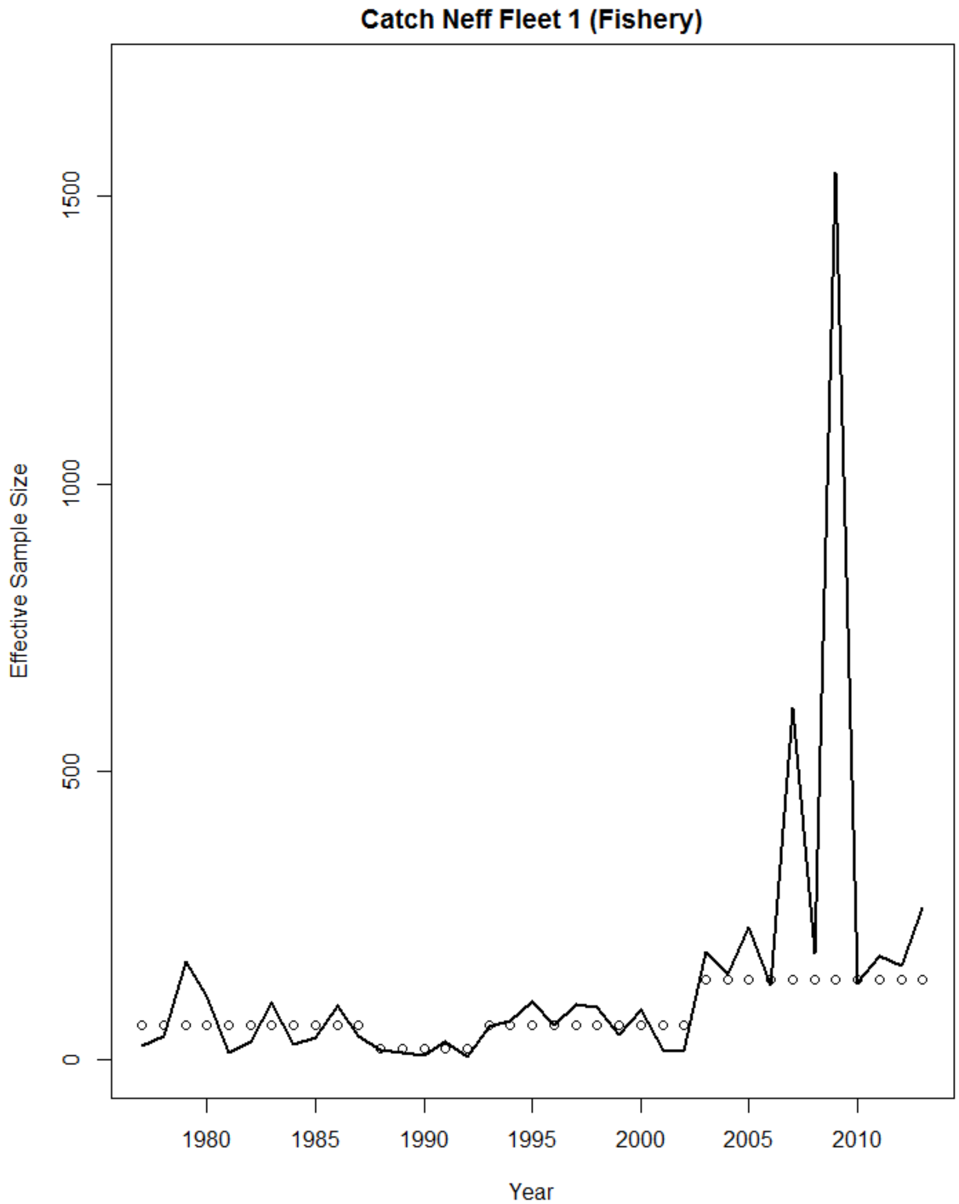


Figure A.171. ASAP_final_temp10 model comparison of input effective sample size versus the model estimated effective sample size for the Gulf of Maine haddock fishery catch (Fleet 1).

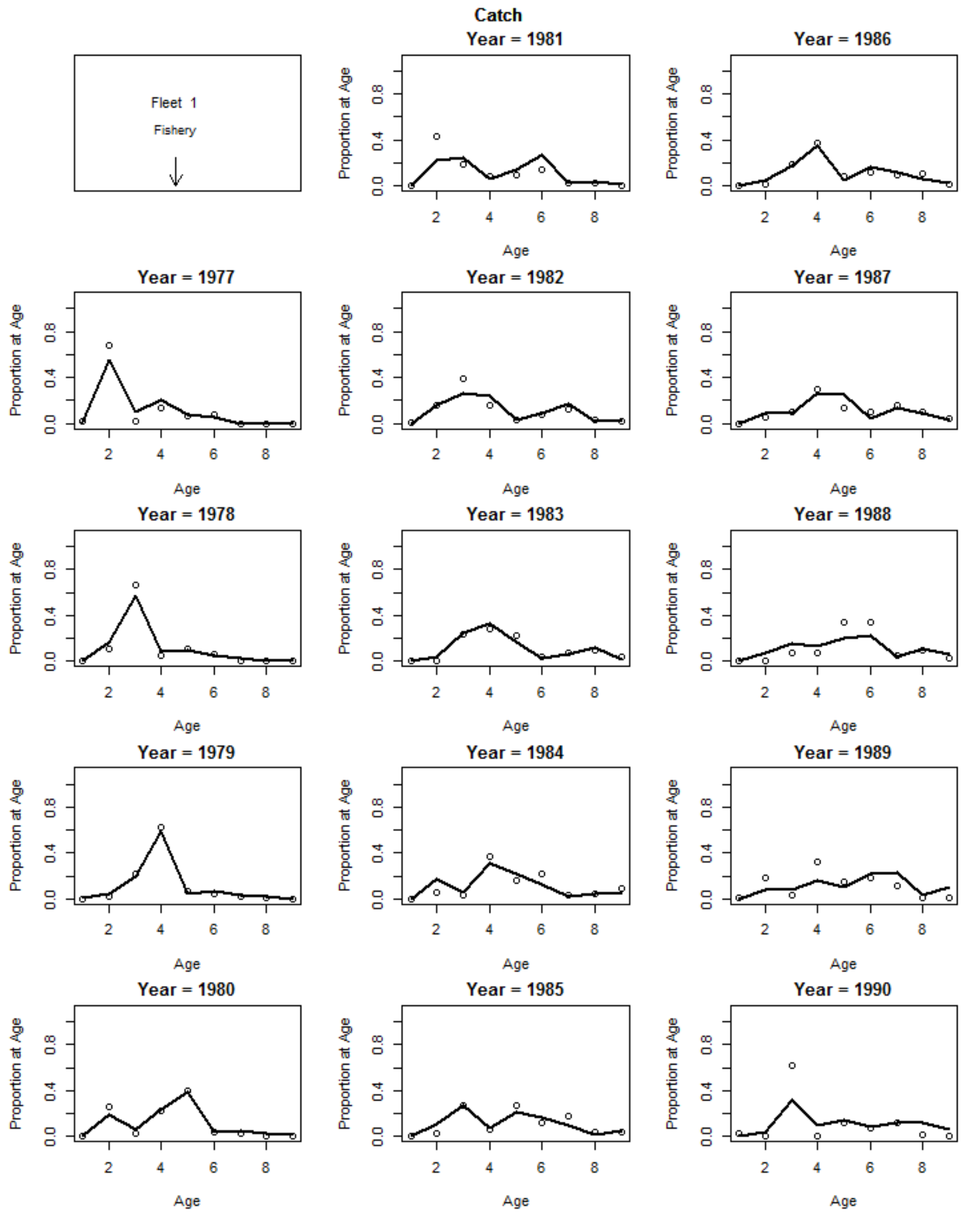


Figure A.172.a. Comparison of the ASAP_final_temp10 model estimates of Gulf of Maine haddock proportion-at-age in the fishery to the data estimates.

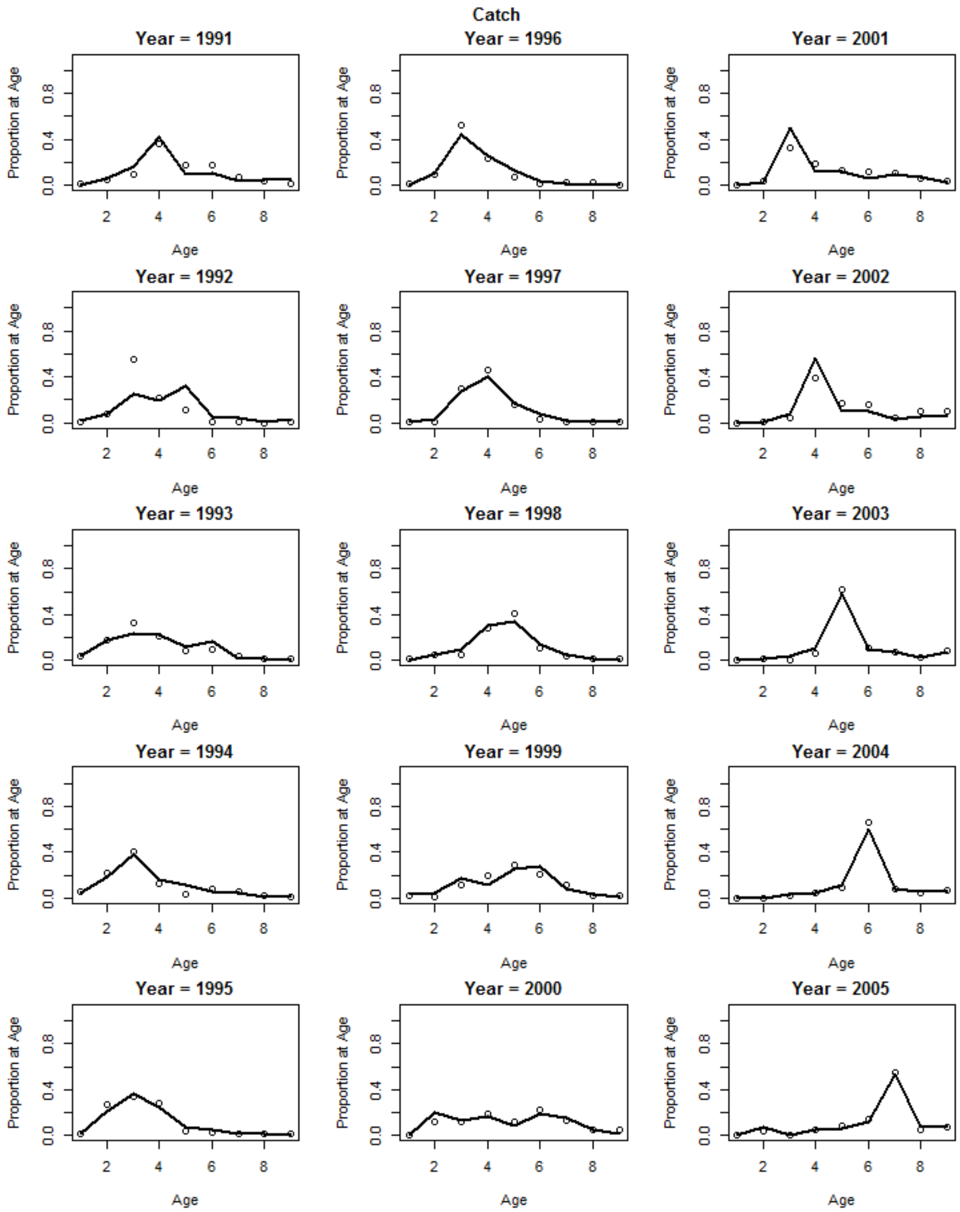


Figure A.172.b. Comparison of the ASAP_final_temp10 model estimates of Gulf of Maine haddock proportion-at-age in the fishery to the data estimates.

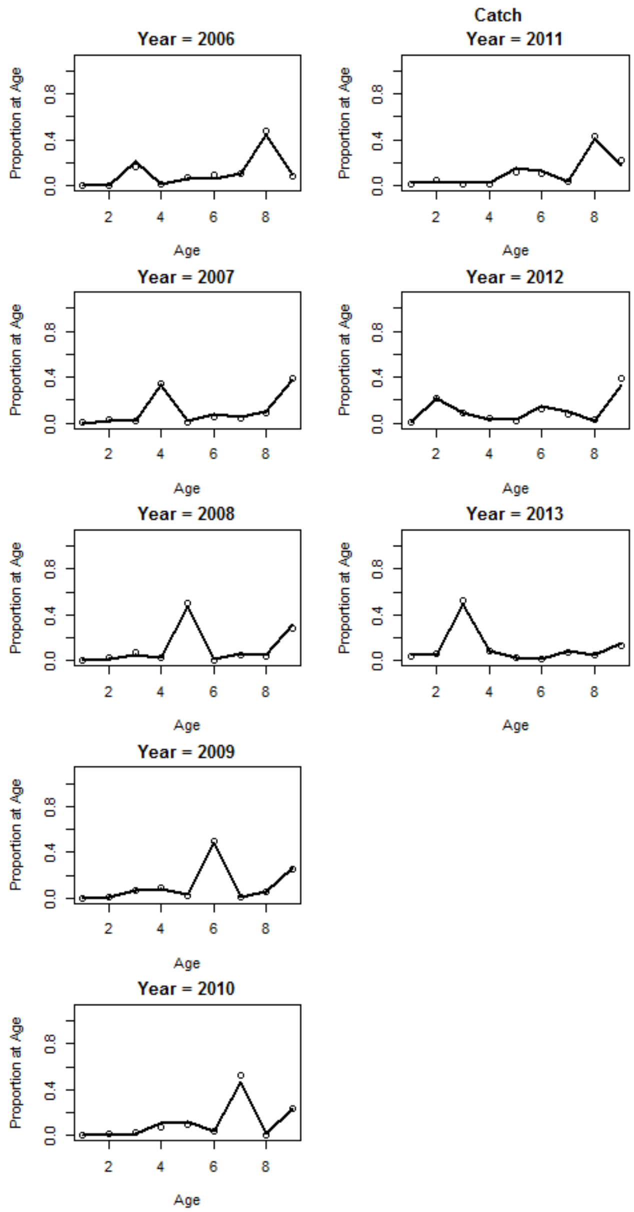


Figure A.172.c. Comparison of the ASAP_final_temp10 model estimates of Gulf of Maine haddock proportion-at-age in the fishery to the data estimates.

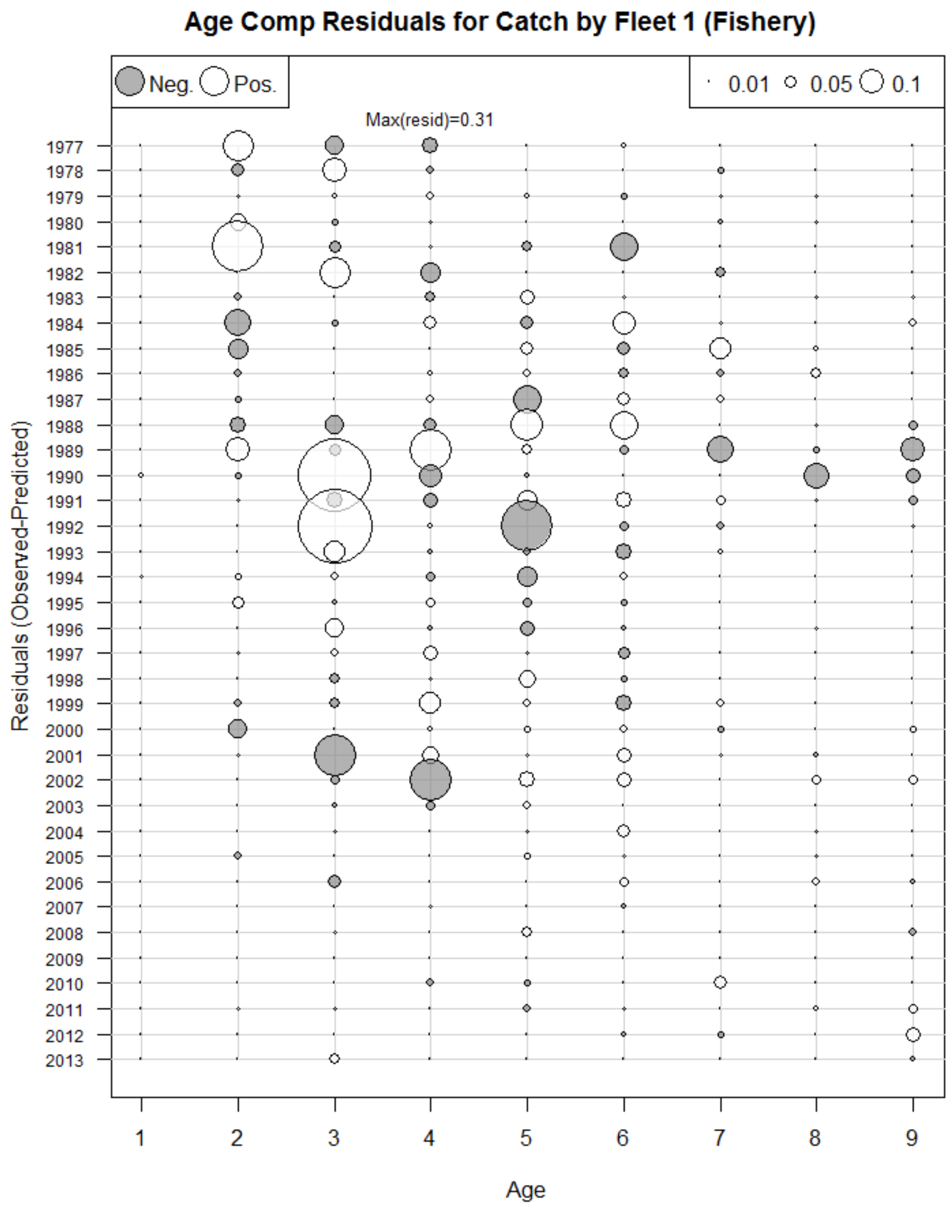


Figure A.173. ASAP_final_temp10 model fit residuals for the Gulf of Maine haddock fishery (Fleet 1) catch-at-age.

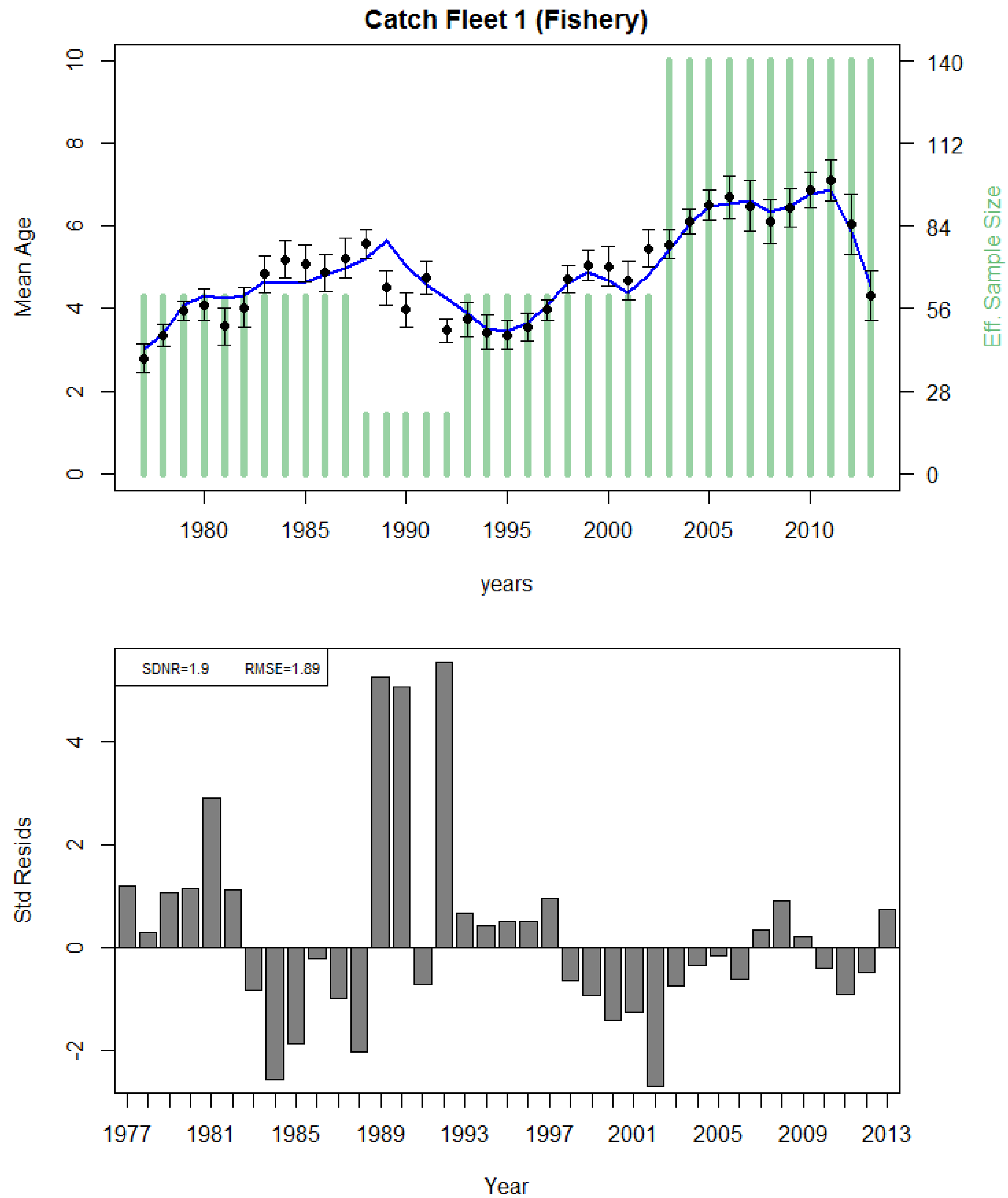


Figure A.174. ASAP_final_temp10 predicted mean age of Gulf of Maine haddock in the fishery (Fleet 1) catch (blue line) compared to observed mean age (top plot) and the residuals about the mean (bottom plot).

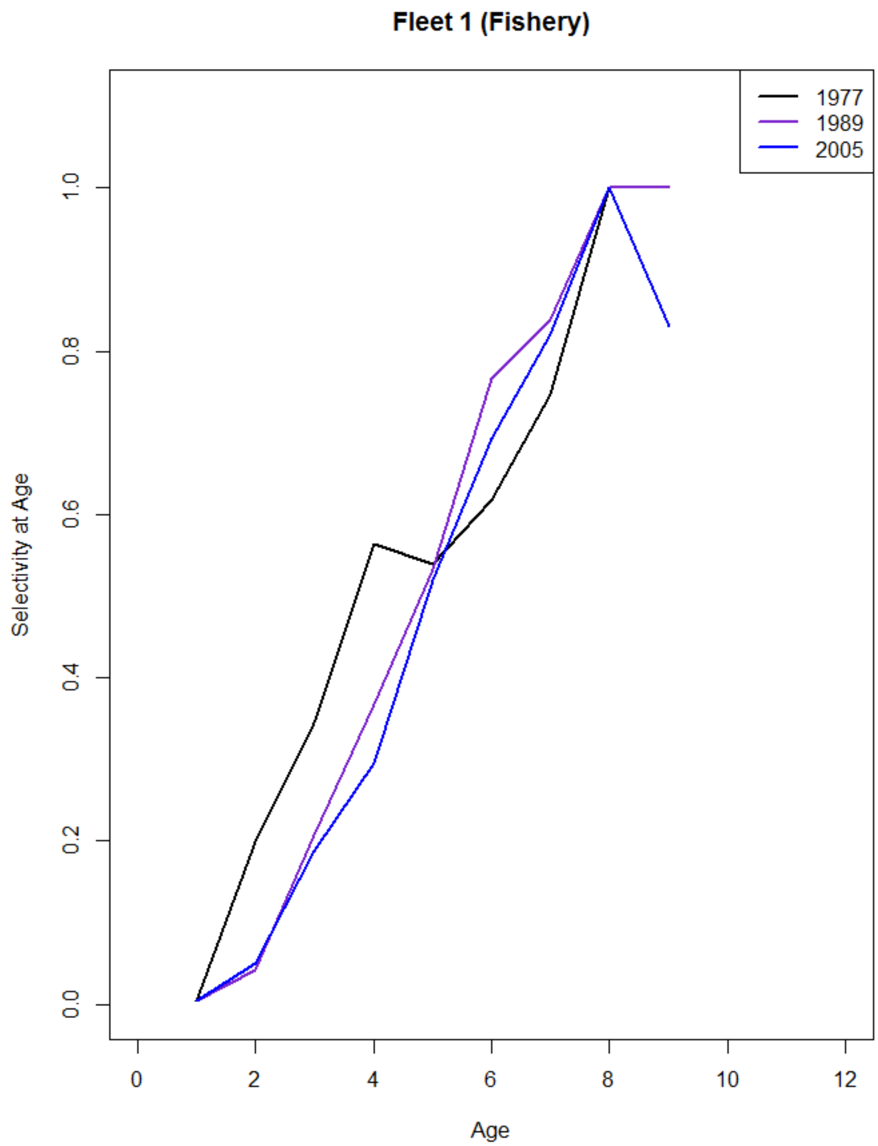


Figure A.175. Gulf of Maine haddock fishery (Fleet 1) selectivity blocks for block 1 (1977-1988), block 2 (1989-2004) and block 3 (2005-2013) as estimated by the ASAP_final_temp10 model.

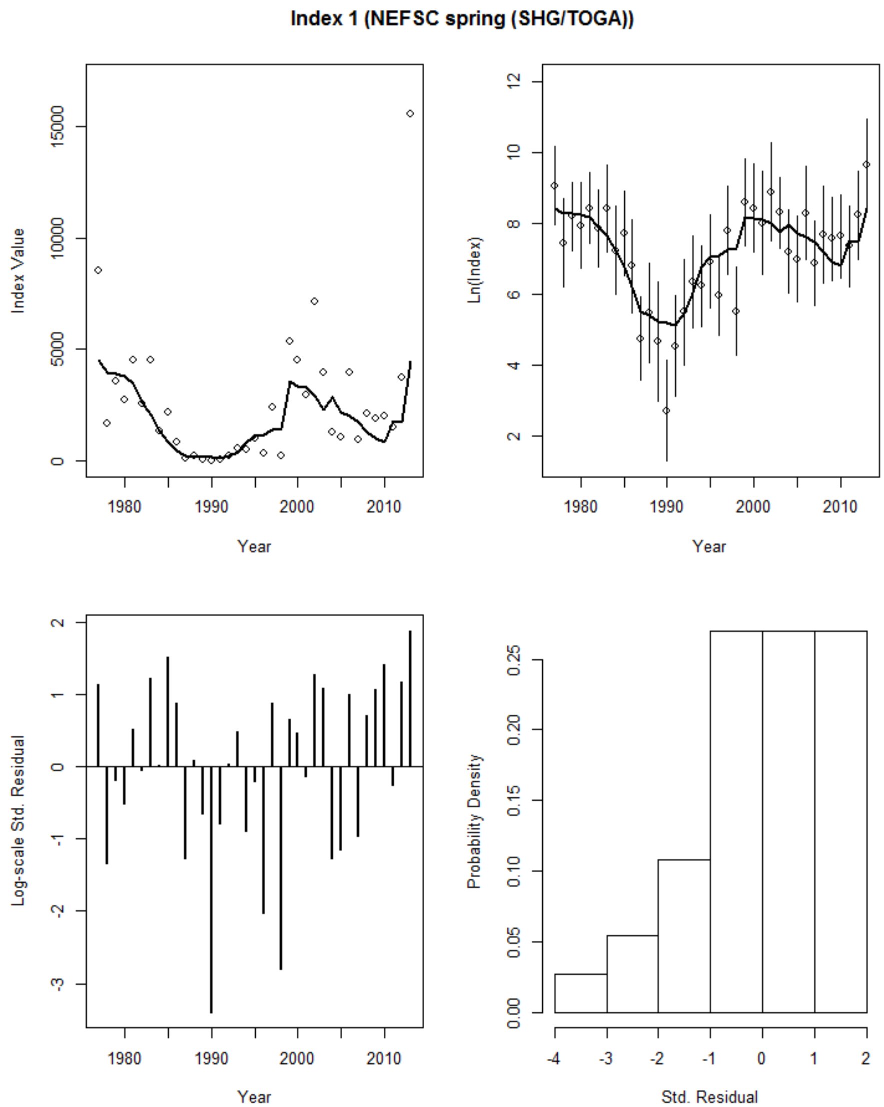


Figure A.176. ASAP_final_temp10 model fit to the NEFSC spring survey Gulf of Maine haddock index (Index 1).

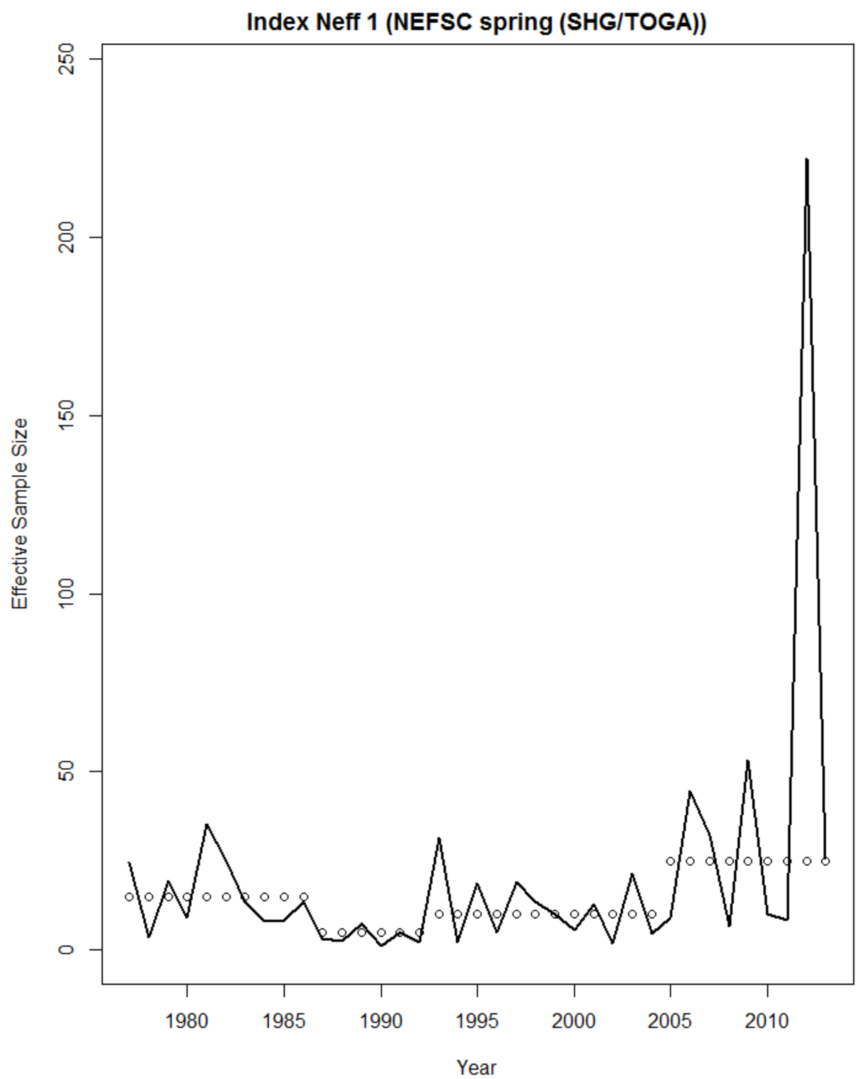


Figure A.177. ASAP_final_temp10 model comparison of input effective sample size versus the model estimated effective sample size for the NEFSC spring survey Gulf of Maine haddock index (Index 1).

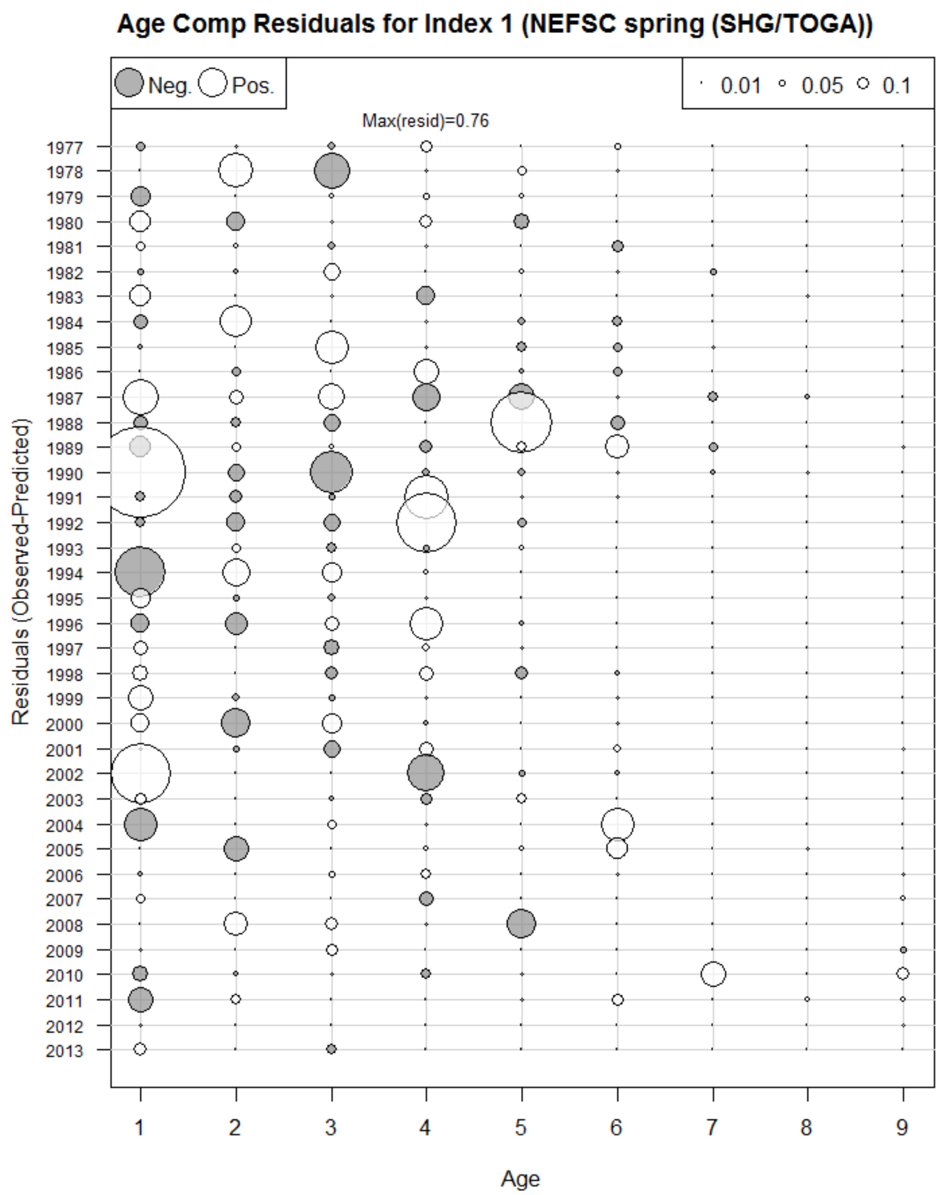
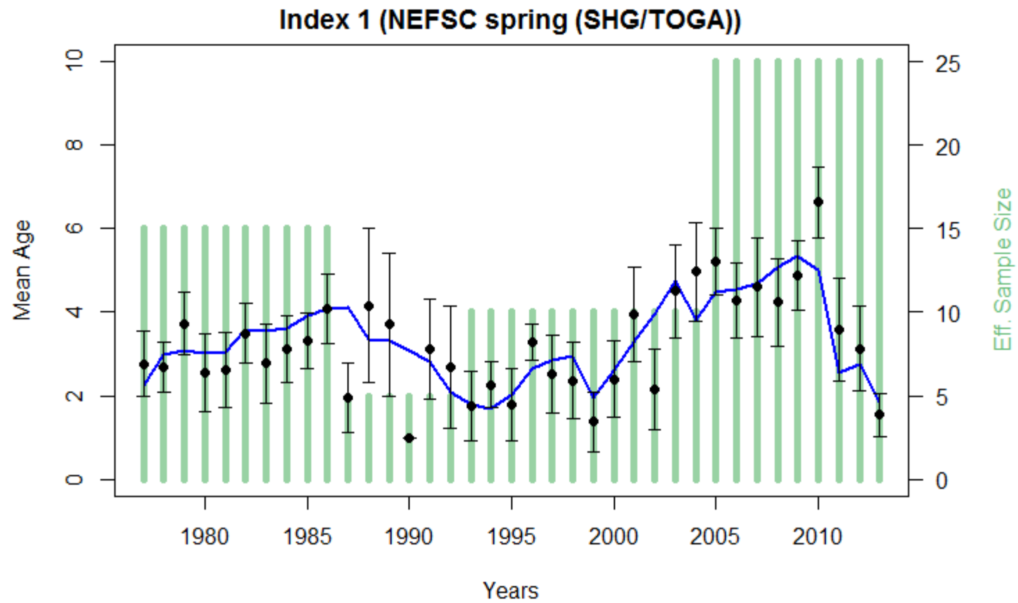


Figure A.178. ASAP_final_temp10 model fit residuals for the NEFSC spring survey Gulf of Maine haddock indices-at-age (Index 1).



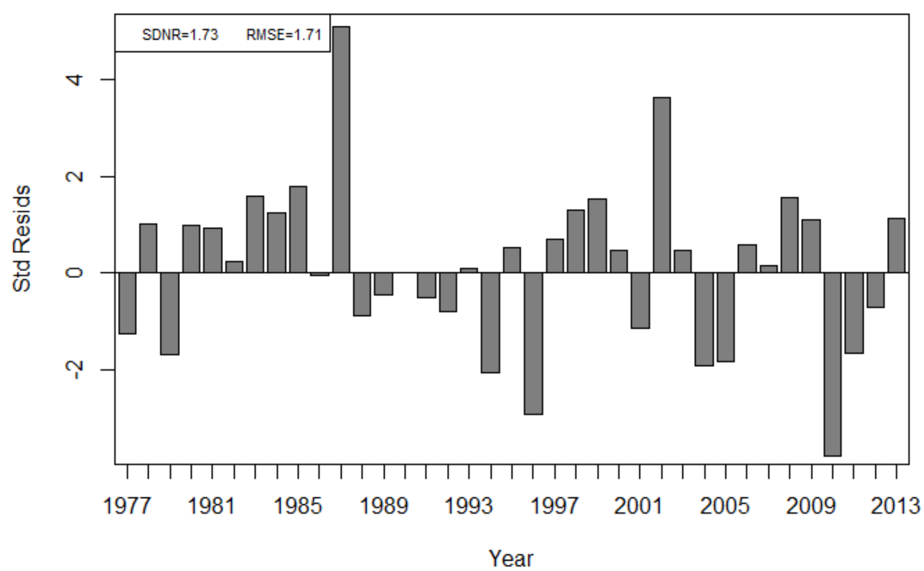


Figure A.179. ASAP_final_temp10 predicted mean age of Gulf of Maine haddock in the NEFSC spring (Index 1) survey (blue line) compared to observed mean age (top plot) and the residuals about the mean (bottom plot).

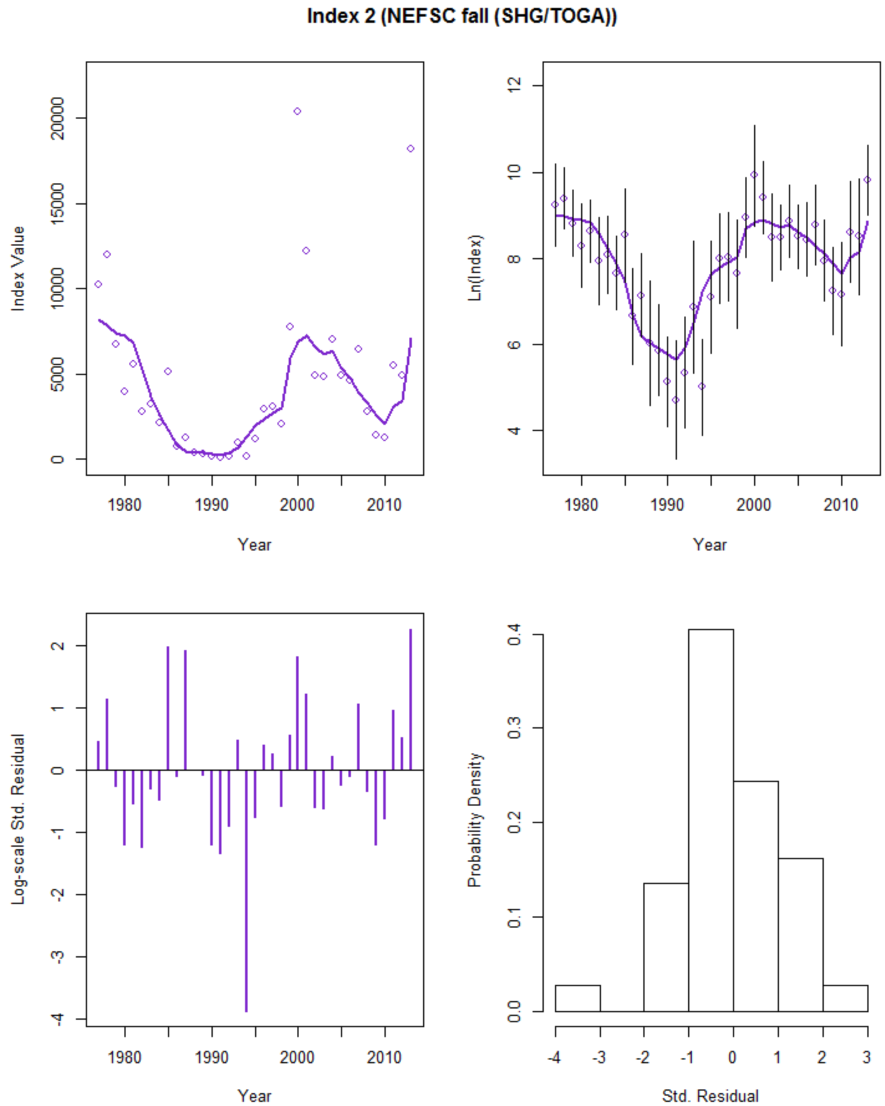


Figure A.180. ASAP_final_temp10 model fit to the NEFSC fall survey Gulf of Maine haddock index (Index 2).

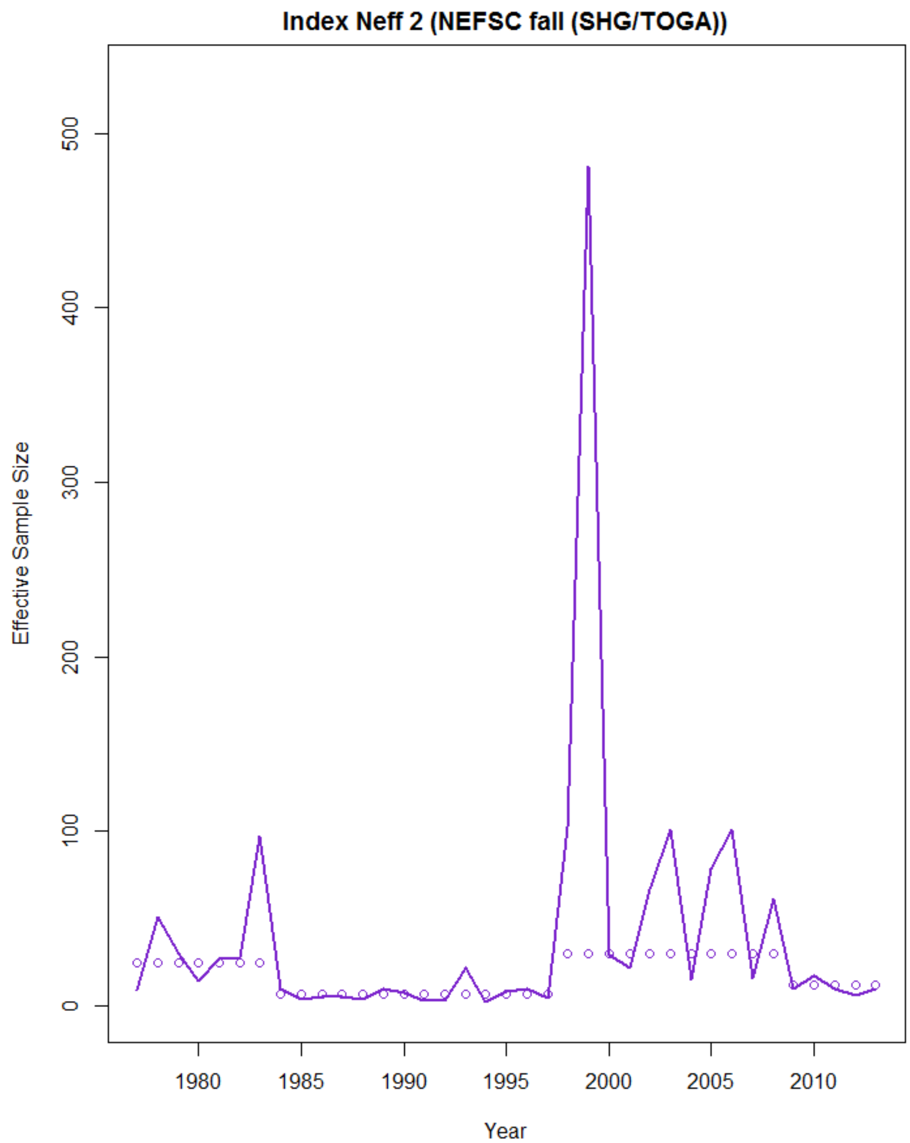


Figure A.181. ASAP_final_temp10 model comparison of input effective sample size versus the model estimated effective sample size for the NEFSC fall survey Gulf of Maine haddock index (Index 2).

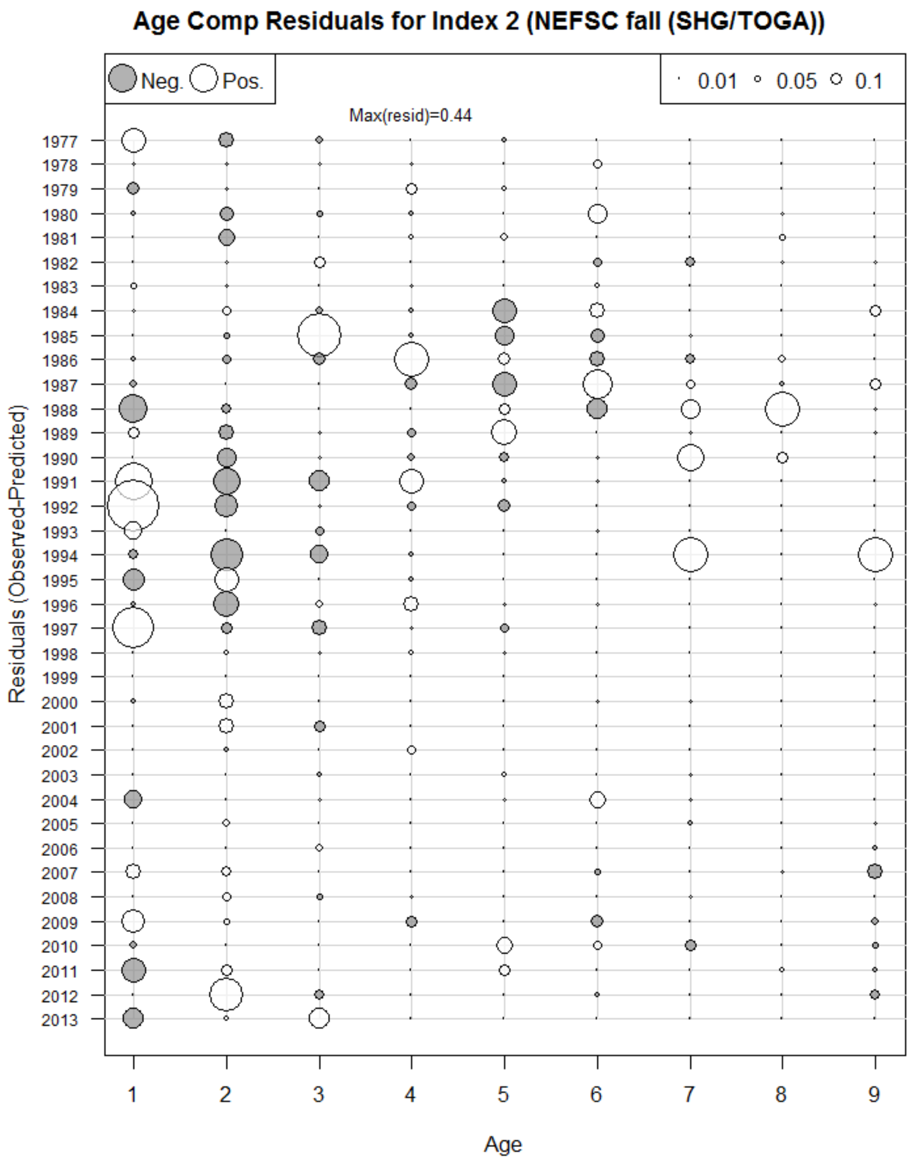


Figure A.182. ASAP_final_temp10 model fit residuals for the NEFSC fall survey Gulf of Maine haddock indices-at-age (Index 2).

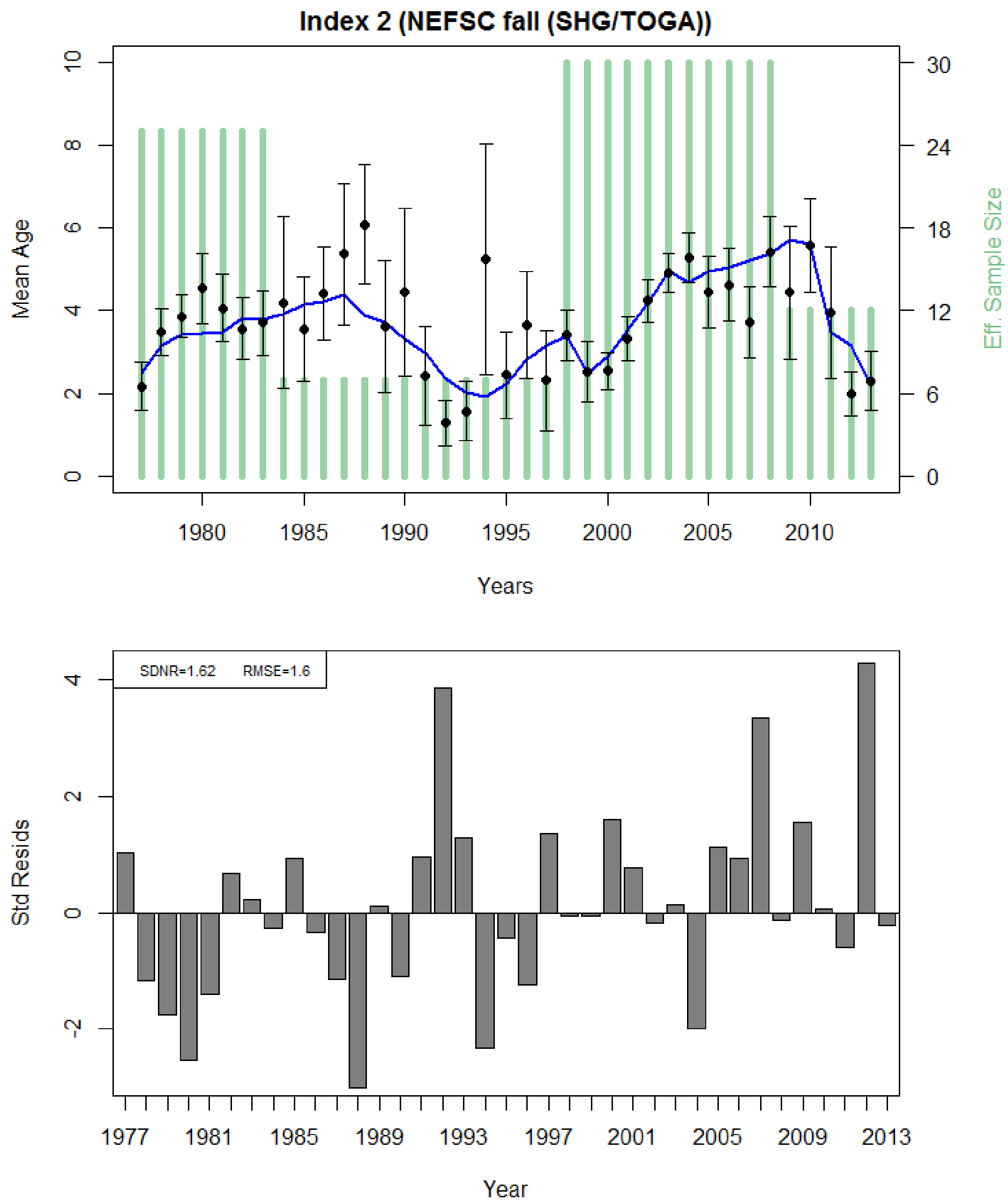


Figure A.183. ASAP_final_temp10 predicted mean age of Gulf of Maine haddock in the NEFSC fall (Index 2) survey (blue line) compared to observed mean age (top plot) and the residuals about the mean (bottom plot).

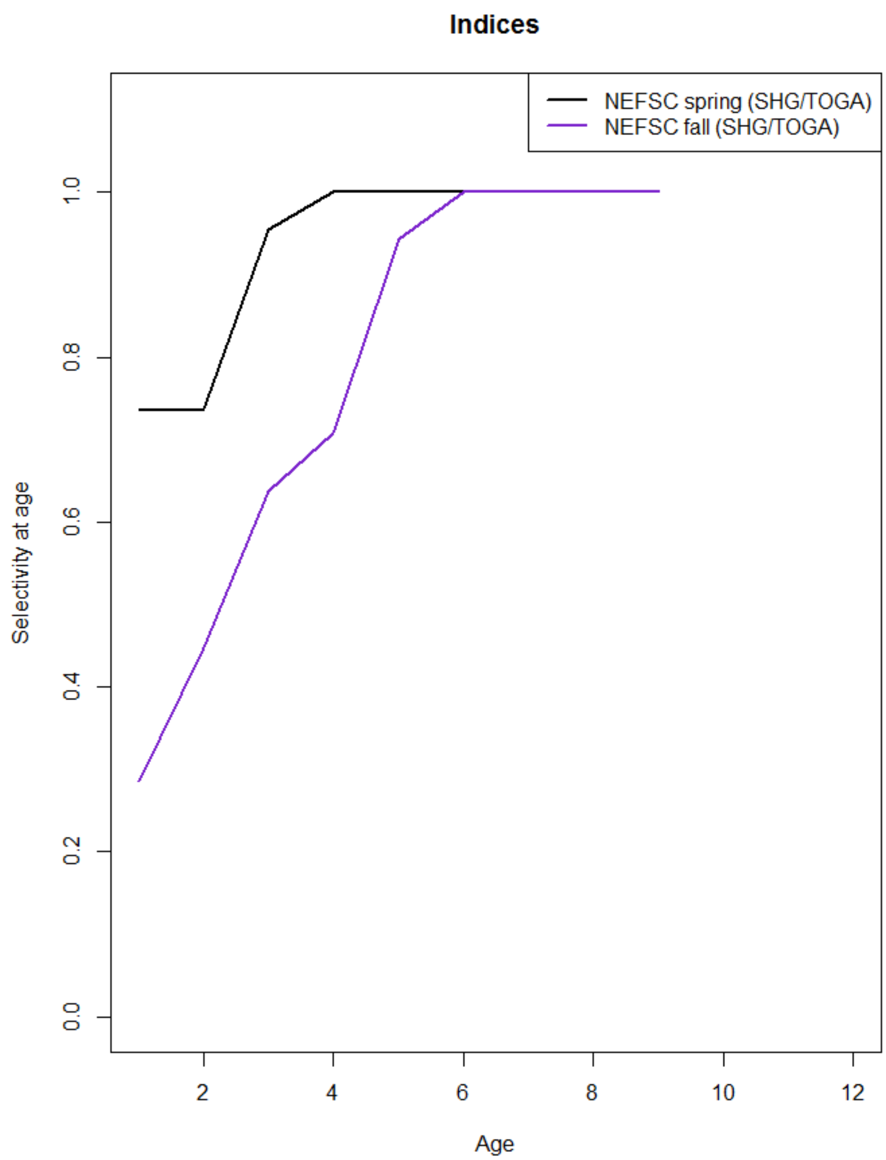


Figure A.184. Gulf of Maine haddock selectivity-at-age for the NEFSC spring (Index 1), and fall (Index 2) from the ASAP_final_temp10 model.

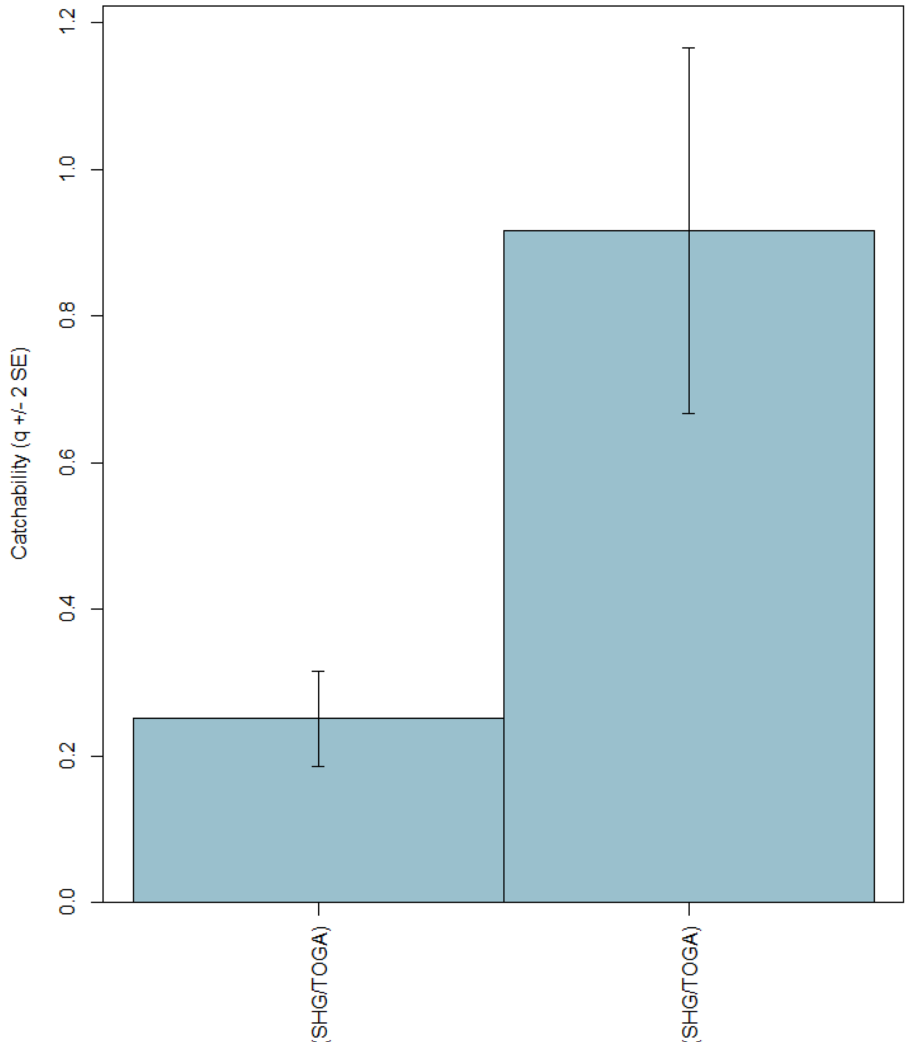


Figure A.185. Gulf of Maine haddock survey catchability, *q*, for the NEFSC spring (Index 1), and fall (Index 2) surveys from the ASAP_final_temp10 model.

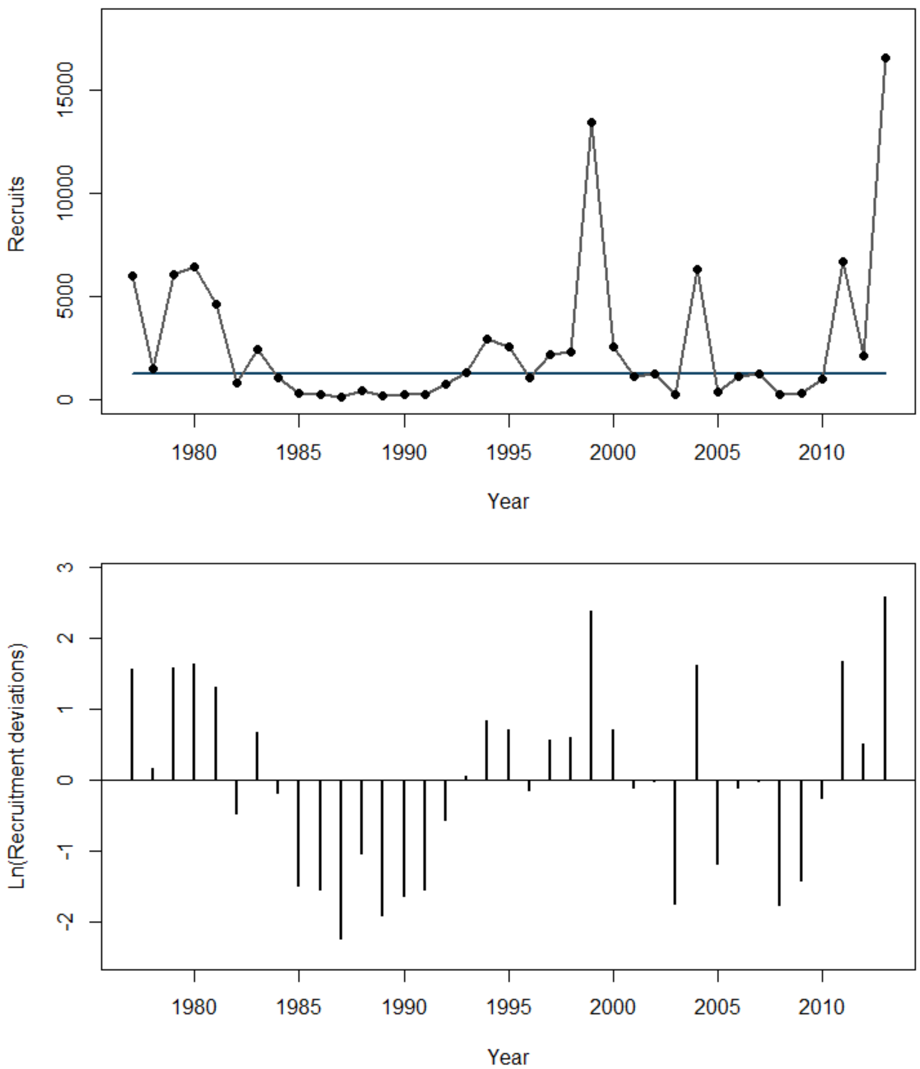


Figure A.186. ASAP_final_temp10 estimated Gulf of Maine haddock age-1 recruitment and recruitment residuals from the geometric mean.

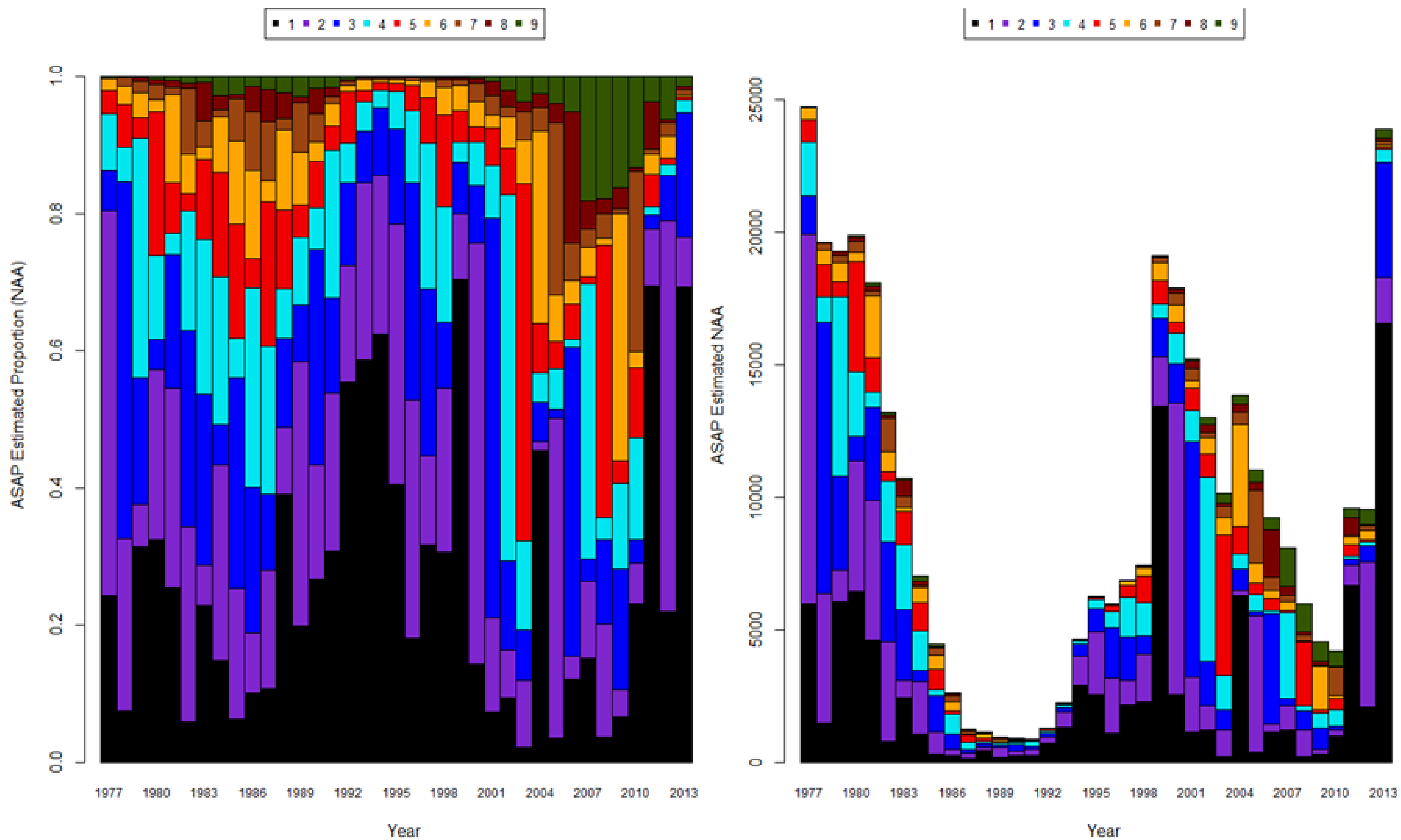


Figure A.187. ASAP_final_temp10 model estimates of Gulf of Maine haddock numbers-at-age in relative (left) terms and absolute (right) numbers (000s).

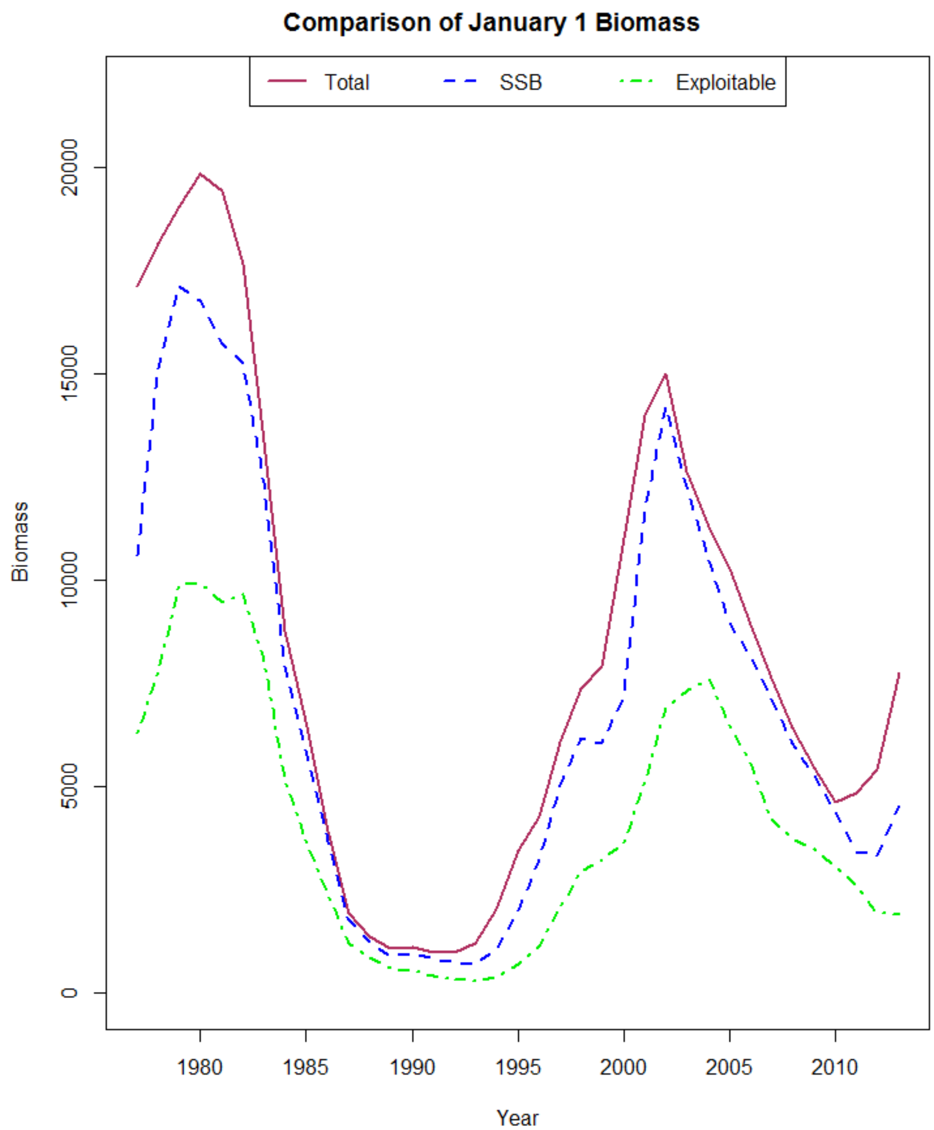


Figure A.188. Comparison of ASAP_final_temp10 model estimates of Gulf of Maine haddock January 1 biomass after application of maturity ogive (SSB) and fleet selectivity ogives (exploitable).

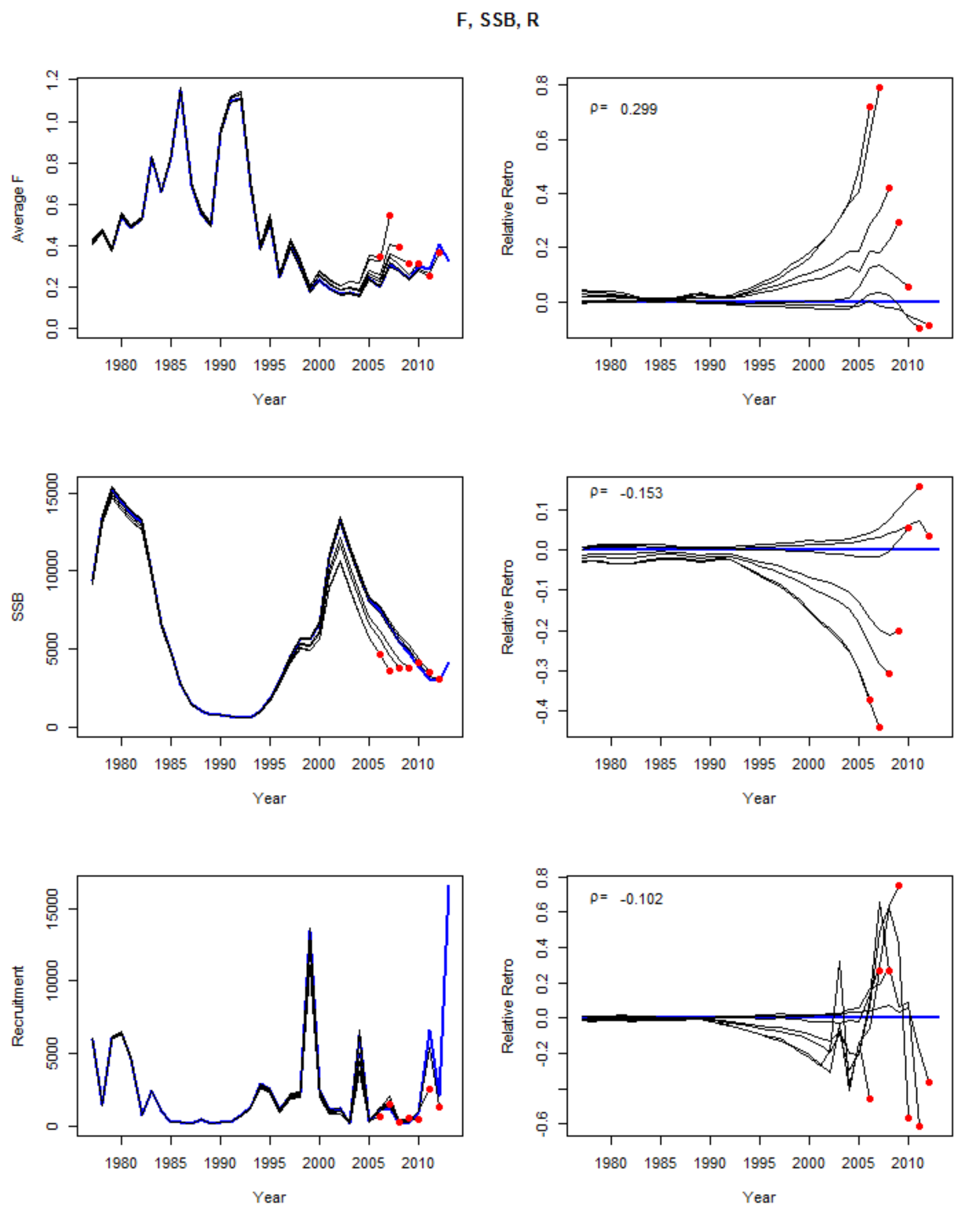


Figure A.189. Model retrospective error patterns for the Gulf of Maine Atlantic haddock ASAP_final_temp10 model. The retrospective error is shown both in absolute (left) and relative (right) terms. The Mohn's rho (ρ) value based on a seven year retrospective peel is indicated in the upper left hand corner of the relative plots.

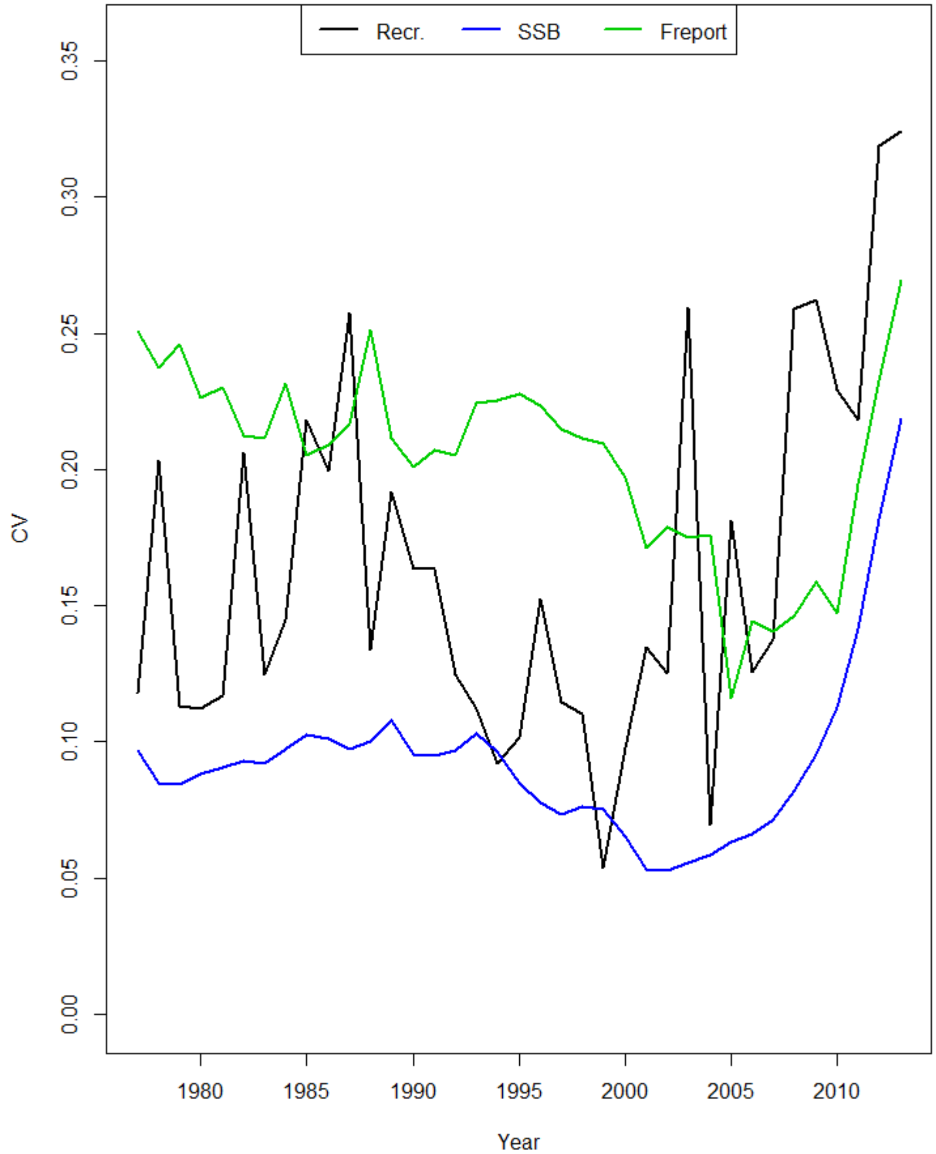
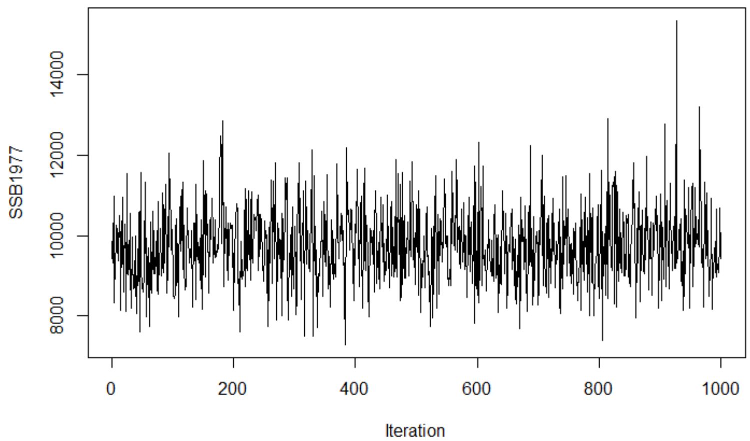


Figure A.190. Hessian-based coefficients of variation (CV) for the ASAP_final_temp10 model estimates of Gulf of Maine haddock spawning stock biomass (SSB), fishing mortality (F_{full}) and age-1 recruitment.



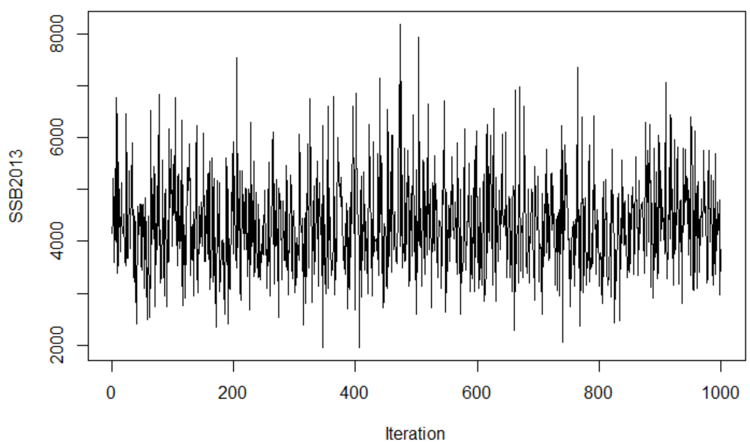


Figure A.191. Trace of MCMC chains for Gulf of Maine haddock 1977 and 2013 spawning stock biomass from the ASAP_final_temp10 model. Each chain had an initial length of 5,000,000 and was thinned at a rate of one out of every 5000th resulting in a final chain length of 1000.

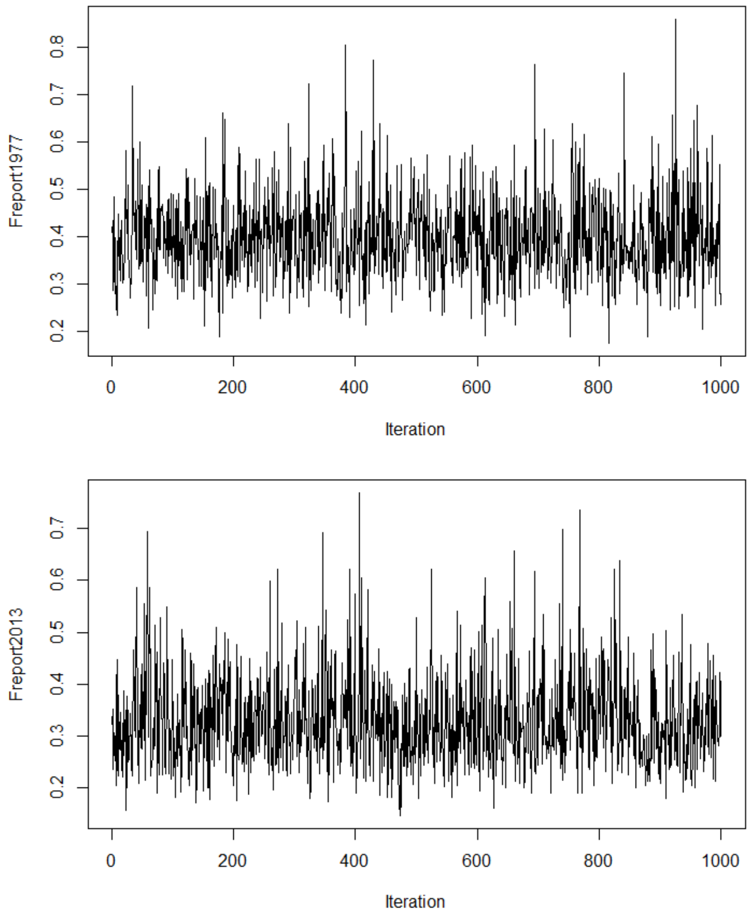


Figure A.192. Trace of MCMC chains for Gulf of Maine haddock 1977 and 2013 fishing mortality from the ASAP_final_temp10 model. Each chain had an initial length of 5,000,000 and was thinned at a rate of one out of every 5000th resulting in a final chain length of 1000.

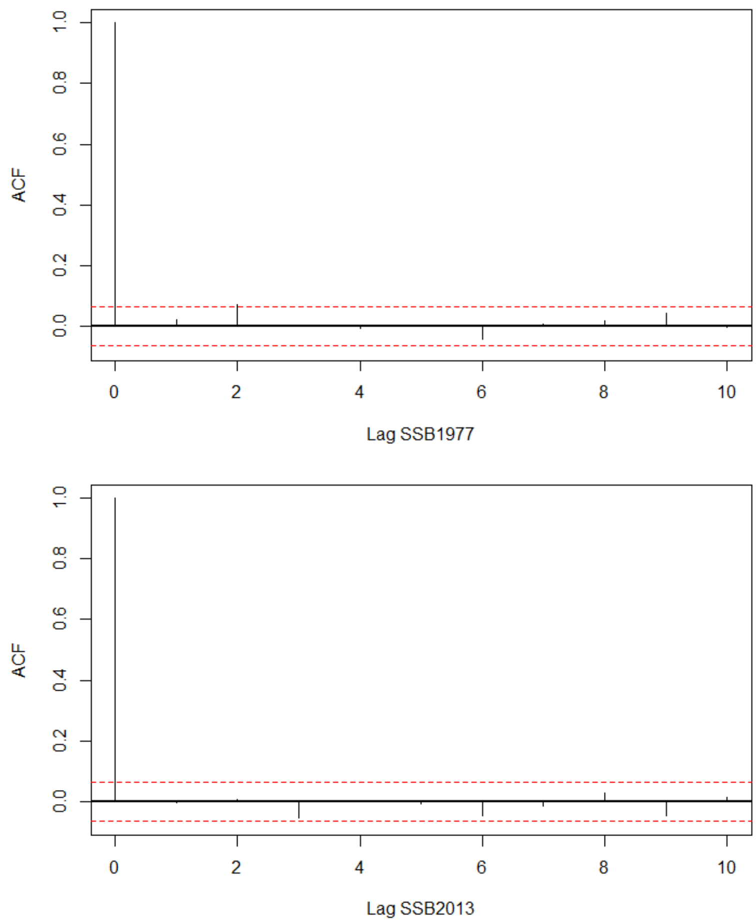


Figure A.193. Autocorrelation within the 1977 and 2013 Gulf of Maine haddock spawning stock biomass (SSB) MCMC chains from the ASAP_final_temp10 model.

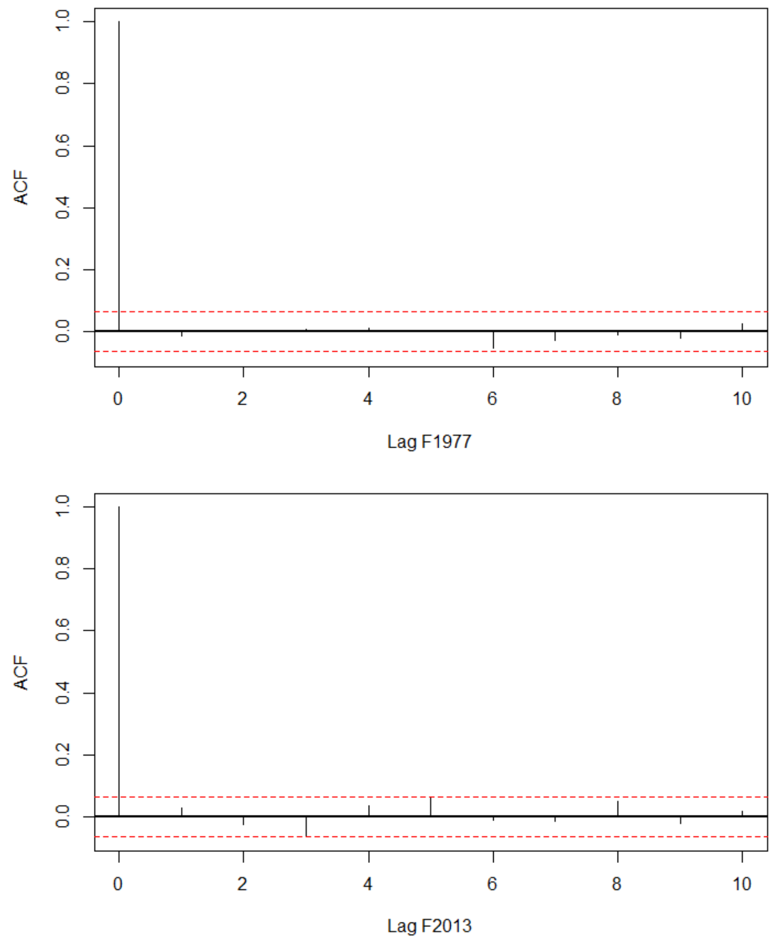


Figure A.194. Autocorrelation within the 1977 and 2013 Gulf of Maine haddock fishing mortality (F) MCMC chains from the ASAP_final_temp10 model.

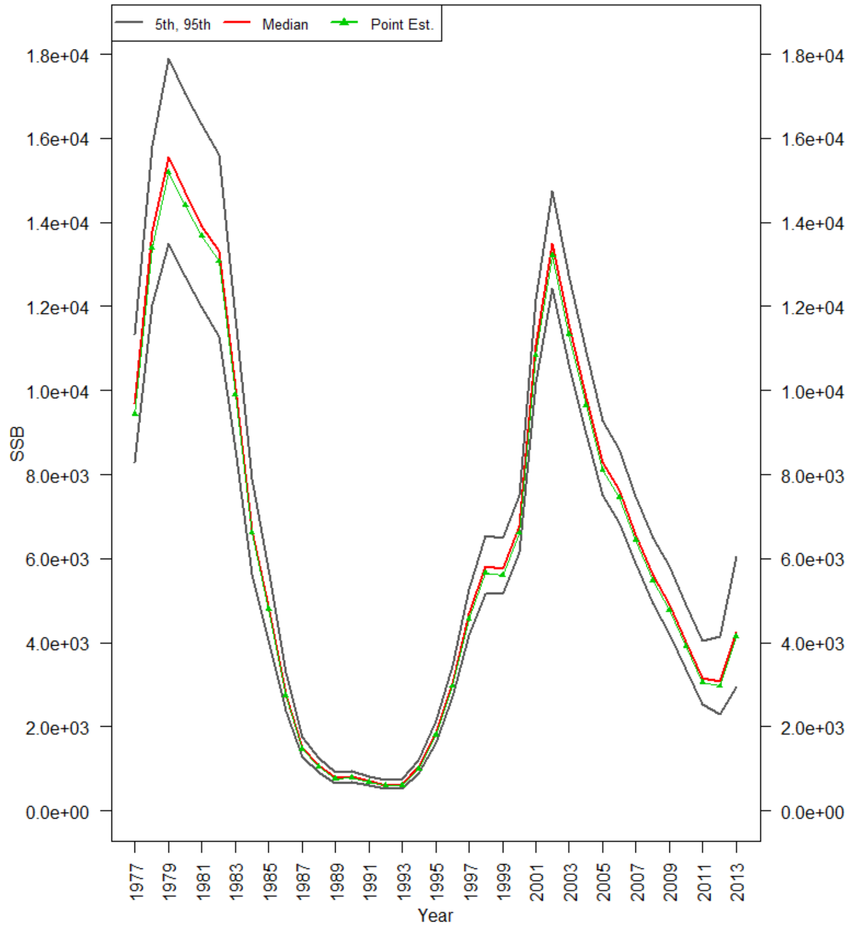


Figure A.195. 90% probability interval for Gulf of Maine haddock spawning stock biomass (SSB) from the ASAP_final_temp10 model. The median value is in red, while the 5th and 95th percentiles are in dark grey. The point estimate from the base model (joint posterior modes) is showin in the thin green line with filled triangles.

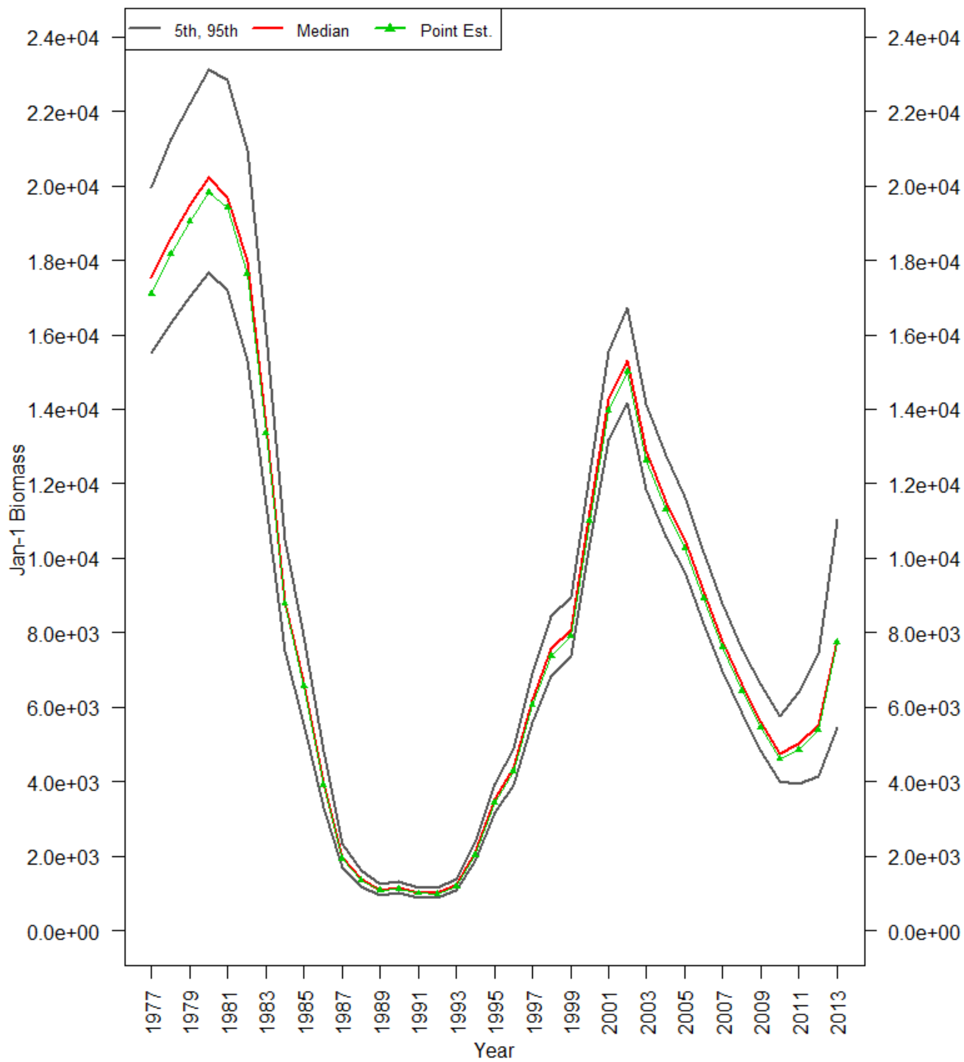


Figure A.196. 90% probability interval for Gulf of Maine haddock January 1 biomass from the ASAP_final_temp10 model. The median value is in red, while the 5th and 95th percentiles are in dark grey. The point estimate from the base model (joint posterior modes) is showin in the thin green line with filled triangles.

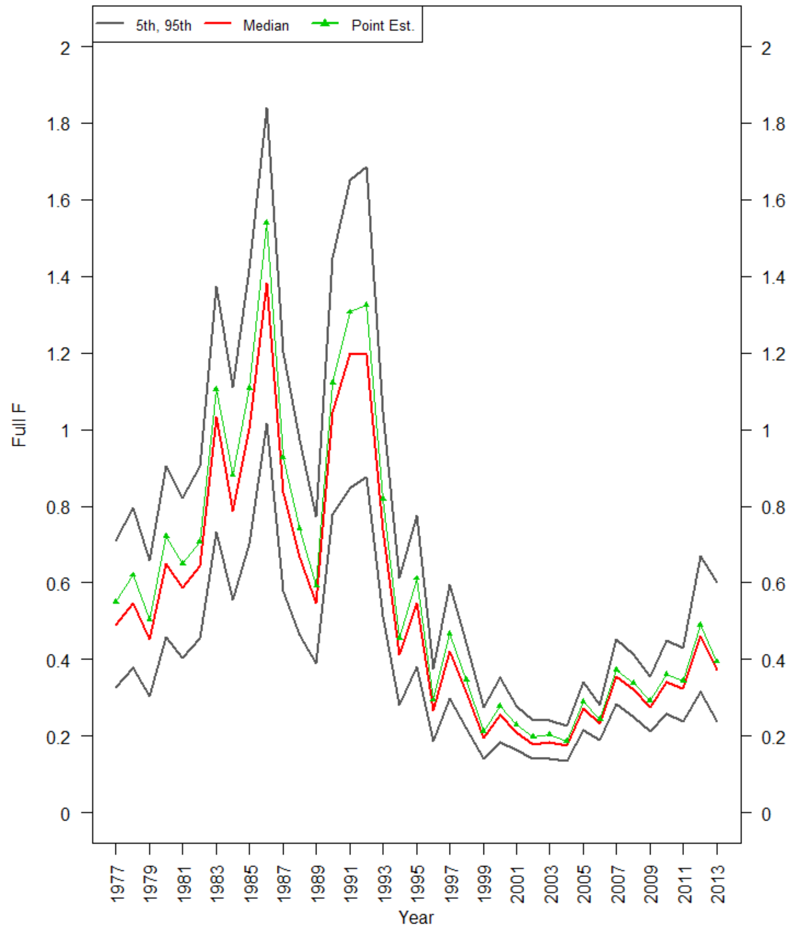


Figure A.197. 90% probability interval for Gulf of Maine haddock fully recruited fishing mortality (Full F or F_{full}) from the ASAP_final_temp10 model. The median value is in red, while the 5th and 95th percentiles are in dark grey. The point estimate from the base model (joint posterior modes) is showin in the thin green line with filled triangles.

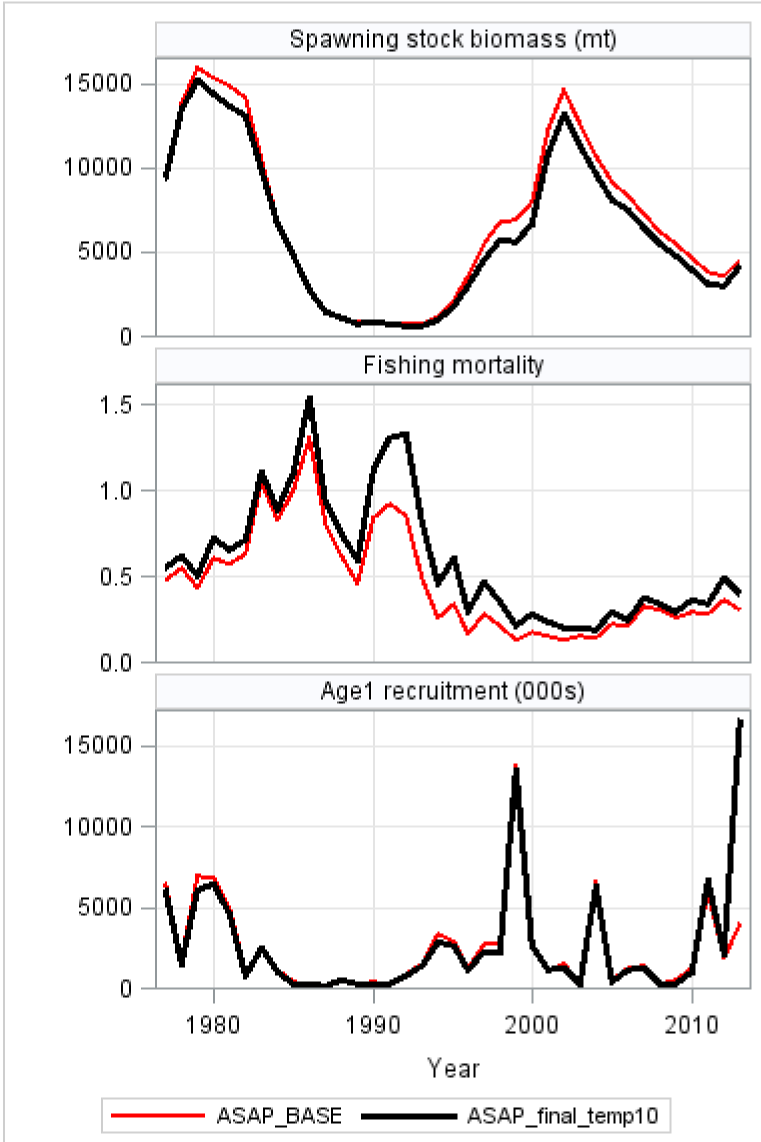


Figure A.198. Comparison of the spawning stock biomass, fully recruited fishing mortality and age-1 recruitment from the ASAP_BASE and ASAP_final_temp10 models.

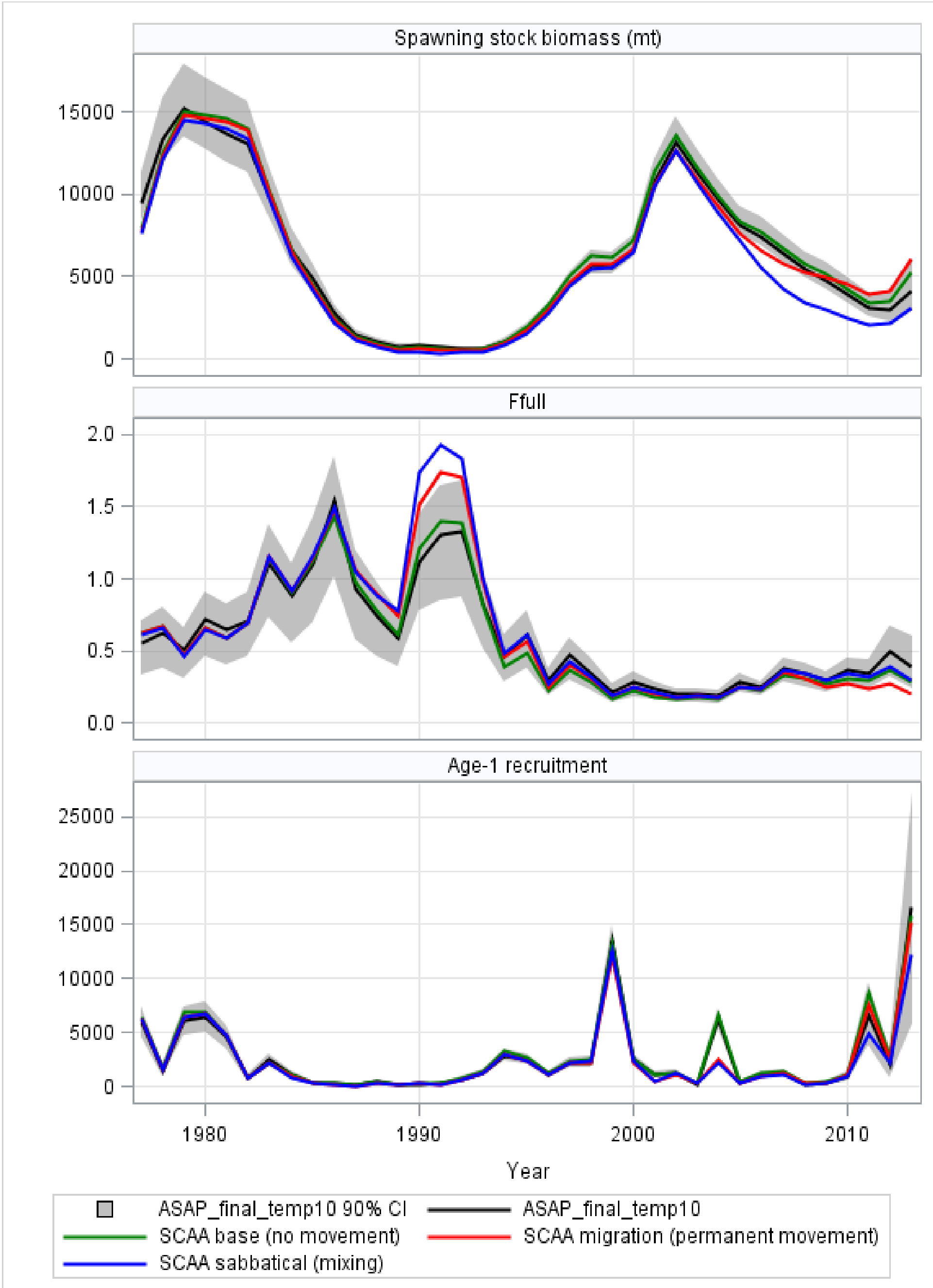
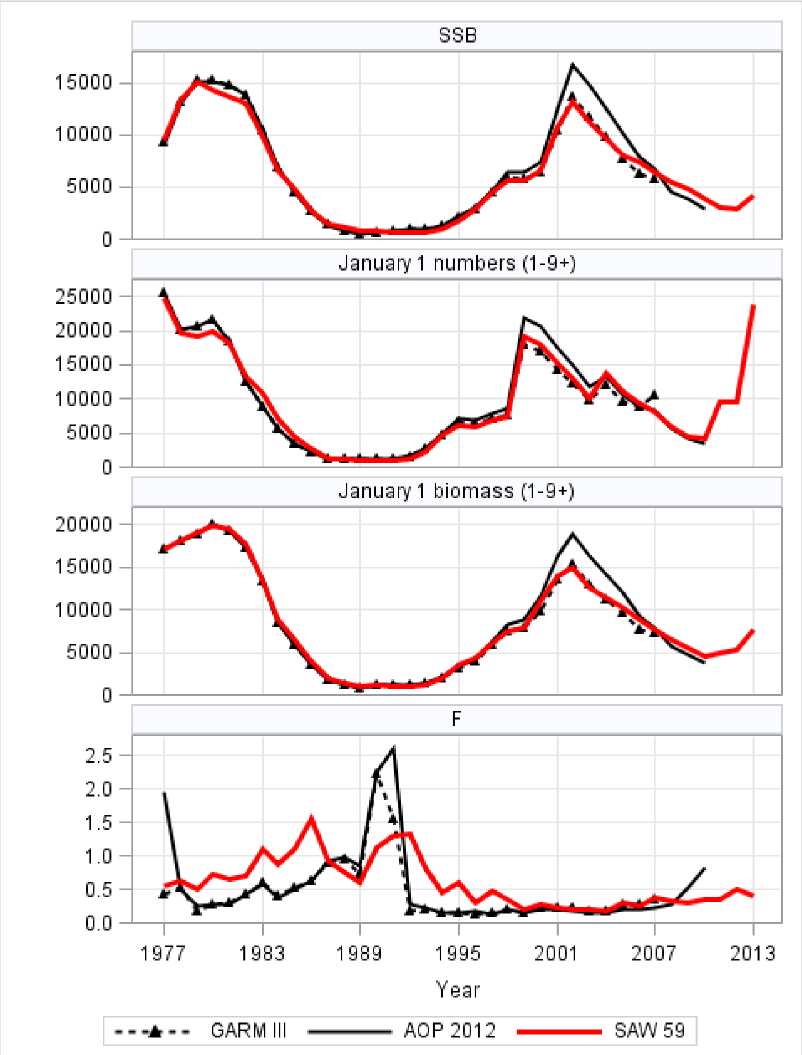


Figure A.199. Comparison of the spawning stock biomass, fully recruited fishing mortality (F_{full}) and age-1 recruitment from the ASAP_final_temp10 model and the SCAA models, two of which incorporate mixing between the Gulf of Maine and Georges Bank stocks. The 90% probability interval from the ASAP_final_temp10 model is indicated by the grey band.



F basis varies by assessment: Ages6-8, N-weighted (GARM III), Ages6-8 (AOP 2012), Full-F (SAW 59)

Figure A.200. Comparison of estimates of average spawning stock biomass (SSB), January 1 stock numbers, January 1 stock biomass, and fishing mortality (F) from previous age-based Gulf of Maine haddock stock assessments.**Note that the F basis varies by assessment (see figure footnote).*

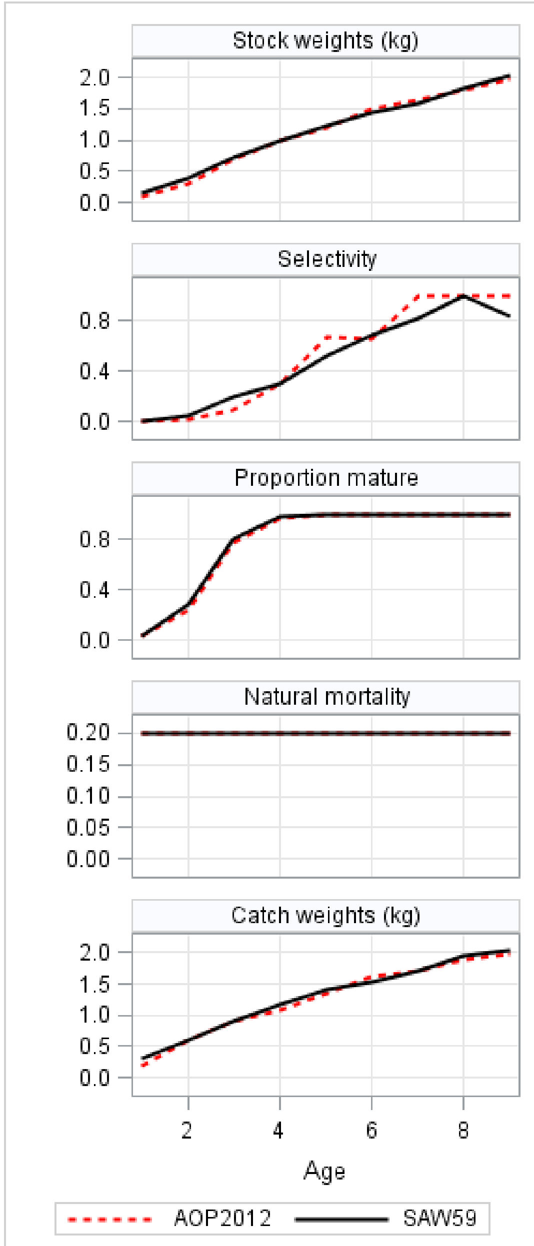


Figure A.201. Comparison of the yield-per-recruit/projection inputs used for the 2012 AOP Gulf of Maine haddock assessment to the current SAW/SARC 59 assessment.

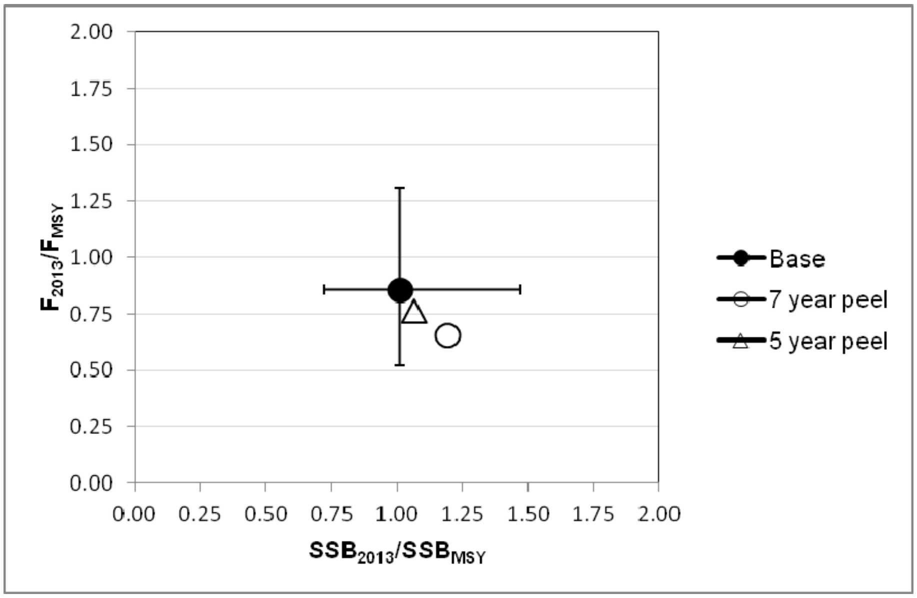


Figure A.202. Comparison of 2013 Gulf of Maine haddock fishing mortality (F_{full}) and spawning stock biomass (SSB) relative to the F_{MSY} ($F_{40\%}$) and SSB_{MSY} proxies both without (solid black circle) and with accounting for retrospective bias based on either a 5-year (open triangle) or 7-year (open circle) retrospective peel. The unadjusted point is shown with the corresponding 90% confidence intervals.

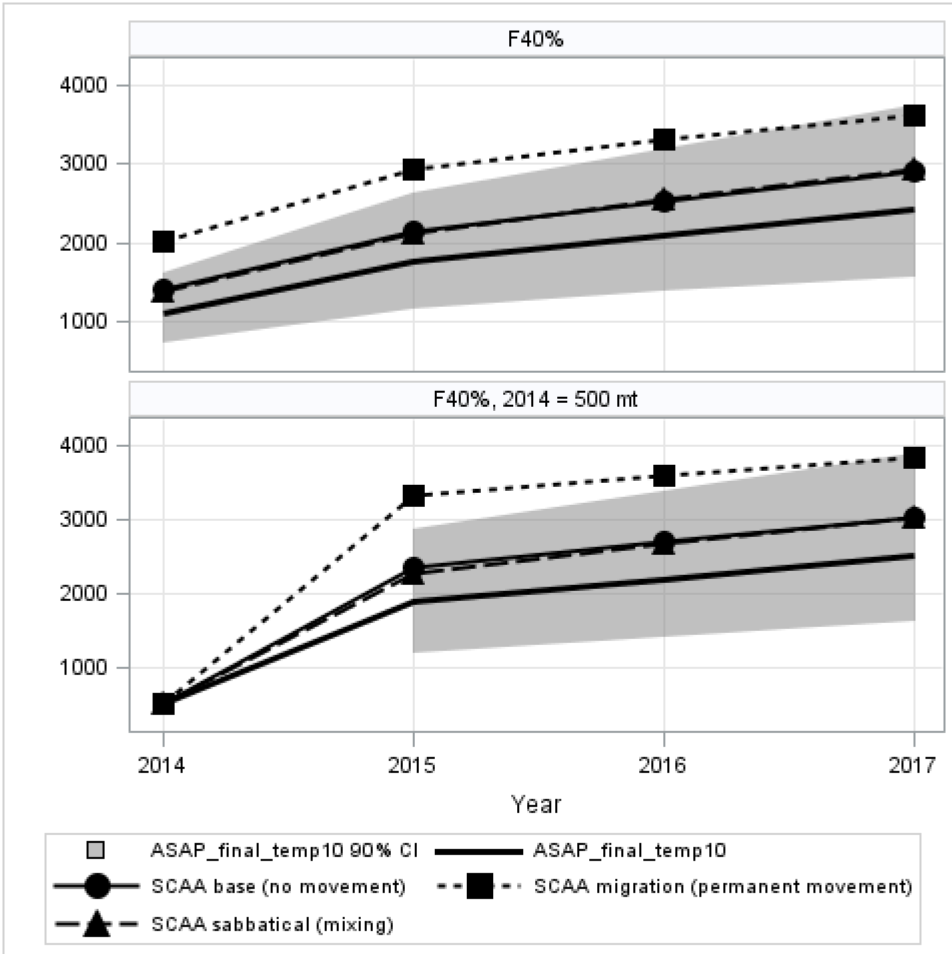


Figure A.203. Short term projections of total fishery yield and spawning stock biomass for Gulf of Maine haddock based on a harvest scenario of a) fishing at $F_{40\%}$ between 2014 and 2017 and b) an assumed catch of 500 mt in 2014 and fishing at $F_{40\%}$ between 2015 and 2017. Projections from the base ASAP model (ASAP_final_temp10) are compared to three alternate runs the from the SCAA model, two of which incorporate mixing between the Gulf of Maine and Georges Bank stocks.

Appendix A.1. List of SAW 59 Gulf of Maine Haddock SAW 59 Working Group participants

The following people participated in all or part of the Working Group meeting, June 2-6, 2014.

Name	Affiliation
Larry Alade	NEFSC Woods Hole
Terry Alexander	NEFMC member, Industry Advisor
Rich Bell	NEFSC Narragansett
Liz Brooks	NEFSC Woods Hole
Doug Butterworth	Univ. of Cape Town (Lead Industry Science Consultant)
Steve Correia	MA Div Marine Fisheries
Jamie Cournane	NEFMC staff
Jonathan Deroba	NEFSC Woods Hole
Dan Hennen	NEFSC Woods Hole
Fiona Hogan	NEFMC staff
Chris Legault	NEFSC Woods Hole, NEFMC SSC Member
Brian Linton	NEFSC Woods Hole
Tim Miller	NEFSC Woods Hole
Paul Nitschke	NEFSC Woods Hole
Loretta O'Brien	NEFSC Woods Hole
Jackie O'Dell	Northeast Seafood Coalition
Mike Palmer	NEFSC Woods Hole (Lead NEFSC Scientist)
Rebecca Rademeyer	Univ. of Cape Town
Paul Rago	NEFSC Woods Hole
Maggie Raymond	Associated Fisheries of Maine
David Richardson	NEFSC Narragansett
Fred Serchuk	NEFSC Woods Hole
Mark Terceiro	NEFSC Woods Hole (Meeting Chair)
Susan Wigley	NEFSC Woods Hole
Tony Wood	NEFSC Woods Hole
Chao Zou	Univ. of Rhode Island

Appendix A.2. Additional ASAP sensitivity runs

This appendix provides results from sensitivity runs that were conducted on the SAW/SARC 59 ASAP reference model (ASAP_BASE). The sensitivity runs fell into two categories: 1) determining whether an alternate model formulation offered improved fit to the data; and 2) evaluating the sensitivity of the model with respect to a range of assumptions.

A.2.1. Profiling across a range of natural mortality values

A sensitivity analysis explored the response of the ASAP base (ASAP_BASE) model to natural mortality (M) estimates ranging from 0.0 to 1.0. There was model preference for M around 0.1 (Fig. A.2.1), with a four point difference in the objective between the M=0.1 and M=0.2 runs. This does not indicate strong preference for a lower M, but is suggestive the M may be lower than the M=0.2 assumption. The occurrence of age-15 fish showing up in considerable numbers in recent years is indicative of an environment of low natural mortality, at least over the recent time period.

A.2.2. Evaluation of the impacts of using SHG or TOGA tow evaluation criteria

During the transition from the FSV Albatross IV survey vessel to the FSV Henry B. Bigelow there were several noted changes in survey protocols which are summarized in table A.51. One of these changes was a change in the protocol used to determine when survey tows could be considered 'representative' and of sufficient quality for inclusion in the calculation of stratified mean survey indices. The procedure from the Albatross survey years, known as the station-haul-gear (SHG) criteria, evaluates the station type (random vs. non-random, etc.), the haul type (good haul, bad haul due to being too short or too long, etc.) and gear condition (no gear damage, gear damage, etc.). The Bigelow survey protocol uses a revised tow-operations-gear-acquisition (TOGA) criteria. The primary difference between the SHG and TOGA criteria is that the TOGA criteria takes advantage of the extensive sensor information collected on the net performance (bottom contact, wing spread, door spread, head rope high, etc.) to determine when a survey tow should be considered 'representative' and included in survey indices. The differences in survey indices between the protocols is variable, though in general, reflect similar trends (Table A.53).

The ASAP_BASE model was constructed using the TOGA selection criteria for the Bigelow years (2009-2013), though the updated VPA model used the SHG indices for all years (1977-2013). The transition from the SHG to TOGA indices was not explicitly evaluated in a model bridge building step. To document the impacts on the results of the ASAP_BASE model resulting from the use of indices based on the TOGA criteria, a single sensitivity run was conducted using the SHG indices in the ASAP model (ASAP_SHG). All other data configurations were left identical to the ASAP_BASE model. The model results between the two runs are nearly identical (Fig. A.2.2). Interestingly, the only discernible differences between the two runs is at the beginning of the time series when survey data were identical.

A.2.3. Use of survey numbers vs. biomass indices

Analyses were undertaken to compare the use of either survey aggregate abundance (numbers/tow) or biomass (weight/tow) in the model fitting. The abundance indices at age are presented in Tables A.57 and A.63 for the NEFSC spring and NEFSC fall surveys respectively. Biomass indices at age are presented in Tables A.60 and A.66 for the NEFSC spring and NEFSC fall surveys respectively. To correctly convert indices-at-age to numbers (which are the units that the ASAP model is tuning to) the model requires input of survey weights-at-age (e.g., Fig. A.99). The survey weight-at-age matrices contained several holes for age/year combinations, particularly among the older ages. Rather than impute the various holes, this sensitivity run used the stock/January 1 weight matrix (Table A.48) as a surrogate for the spring survey and the catch weight-at-age matrix assuming 100% recreational discard mortality (Table A.47) for the fall survey. A sensitivity model, ASAP_BIOMASS was constructed using the survey biomass indices and weight-at-age matrices as described above.

Summary model diagnostics for the ASAP_BASE and ASAP_BIOMASS model are presented in Table A.2.2. The two indices provided similar results in terms of biomass and fishing mortality, though the ASAP_BASE model had a higher terminal (2013) biomass and lower fishing mortality compared to the biomass model. The CVs on the 2013 spawning stock biomass were 0.15 for both the abundance-based model and the biomass-based model indicating that the biomass model offered no improvements in the precision on the terminal estimates. Additionally, the retrospective error was worse for the biomass model.

A.2.4. Treatment of the Bigelow survey years as independent survey indices

A number of operational changes have been made to the NEFSC spring and fall surveys during over the assessment times series including a changes in vessel (Delaware/Albatross historically and introduction of the Bigelow in 2009), trawl doors (between 1984 and 85) and trawl net (Yankee 36/41 in spring survey). The changes are summarized in Table A.52. Trends in the calibrated and un-calibrated surveys indices were very similar, but vary in scale (Fig. A.95). The ASAP_BASE model is tuned to the combined Albatross/Bigelow-calibrated series. To evaluate the sensitivity of the base model to the Bigelow calibrations, two separate sensitivity runs were conducted. The first tuned the model to the Albatross/Bigelow-converted series and a separate un-calibrated Bigelow series (ASAP_BIGELOW_SPLIT). The second run tuned the model to separate non-overlapping Albatross (1977-2008) and Bigelow (2009-2013) survey series (ASAP_BIGELOW_NOOVERLAP).

Summary model diagnostics for the ASAP_BASE, ASAP_BIGELOW_SPLIT and ASAP_BIGELOW_NOOVERLAP model are presented in Table A.2.3. Survey catchabilities for the NEFSC spring and fall surveys were similar across all three model runs, but the Bigelow spring and fall catchabilities varied considerably between the ASAP_BIGELOW_SPLIT and ASAP_BIGELOW_NOOVERLAP runs, with an approximate doubling of catchability under the non-overlapping run. The retrospective error increased under the ASAP_BIGELOW_NOOVERLAP run. Estimated survey selectivity for the NEFSC spring and fall surveys was similar between the ASAP_BASE and ASAP_BIGELOW_SPLIT runs, though the spring survey became more domed and the fall survey, less domed, under the ASAP_BIGELOW_NOOVERLAP run. The estimated Bigelow selectivity patterns were irregular and poorly estimated in both of the Bigelow runs. It could be that the presence of at least two moderate-to-large years classes within the Bigelow series are confounding the selectivity estimates for this short survey series.

The large increase in estimated Bigelow catchability lead to lowered estimates of spawning stock biomass and increased estimates of fishing mortality at the end of the time series of the ASAP_BIGELOW_NOOVERLAP run (Fig. A.2.3). The model fits to the survey indices were similar across runs (Fig. A.2.4), though the incorporation of the Bigelow series in addition to the single Albatross/Bigelow calibrated series does result in slightly tighter fits to the terminal spring and fall Albatross/Bigelow calibrated series observations. This translates in slightly higher estimates of the 2012 year class size, but has little impact on the spawning biomass and fishing mortality estimates. The results of this sensitivity show little utility in applying non-overlapping Albatross and Bigelow series owing to the poor diagnostics in the way of retrospective error and imprecisely estimated survey selectivity. The combined run applying both the calibrated and un-calibrated series suffers from the same issues with respect to estimation of survey selectivity, but it is helpful for gaining a better understanding the relative catchability differences between the Albatross and the Bigelow surveys. While the survey catchabilities are not directly translatable to the length-based calibration factors applied to the Bigelow series, the scale in the catchability differences between the spring (0.87/0.26=3.35) and fall (1.90/1.00=1.90) survey series from the ASAP_BIGELOW_SPLIT is of similar scale to the range of the length-based calibration factors (1.164-2.626; Table A.50).

A.2.5. Inclusion/exclusion of survey indices

To better understand how the model results are being influenced by each of the survey indices, the ASAP_BASE model was run using only one index at a time. The two sensitivity runs were ASAP_NEFSC_spring (spring survey only) and ASAP_NEFSC_fall (fall survey only). In both sensitivity runs all other model configurations were left unchanged relative to the ASAP_BASE model.

Both of the single-survey model runs exhibited lower biomass and higher fishing mortality at the start and end of the time series compared to the base run; however, the trajectories throughout the time series were variable (Table A.2.4, Fig. A.2.5). Retrospective error increased in both single-survey models. The precision in the spawning stock biomass over the time series was improved by using both survey indices in the model (Fig. A.2.6). Inclusion of only the NEFSC spring survey in the model leads to a large decrease in precision at the end of the time series. Overall, the model performance is improved through the inclusion of both the NEFSC spring and fall surveys in the base model.

A.2.6. Inclusion of state surveys

Previous Gulf of Maine haddock assessments have only included the NEFSC spring and fall bottom trawl surveys as tuning indices. Both the Maine-New Hampshire (MENH) inshore bottom trawl survey and Massachusetts Department of Marine Fisheries (MADMF) bottom trawl survey encounter haddock in limited quantities. The MENH fall survey began in 2000 and the spring survey began in 2001. The MADMF survey began in 1978. Both surveys are characterized by high CVs (Tables A.69 and A.72) and catches are primarily comprised of smaller haddock (Figs. A.112 and A.123). Until recently, neither the MENH nor MADMF surveys had been aged. As of 2014, all of the otoliths collected from the MENH fall survey have been aged; however, otoliths collection did not begin until 2005 and was extremely limiting in some years (e.g., collected otoliths ranged from 2 to 117/year). The lack of age information, high imprecision and weak coherence with the NEFSC survey signal (Figs. A.128-129) precluded their use in previous assessments.

For this assessment, NEFSC age-length information has been applied to the MENH and MADMF indices-at-length to generate indices-at-age for the state surveys. Plots of indices-at-age show that both surveys primarily capture age-0 and 1 fish (Figs. A.113 and A.125), though there are sufficient number of some older fish captured such that some of the stronger Gulf of Maine haddock year classes track through the survey indices-at-age (e.g., 1998, 2003). The NEFSC spring age-1 index exhibits a moderate degree of correlation with both the MADMF and MENH age-1 indices (Fig. A.130), and the NEFSC fall age-1 index shows moderate degree of correlation with the MENH age-1 index (Fig. A.131).

Four different sensitivity runs of the base ASAP model were conducted which incorporated the state survey indices. The first two included the MADMF and MENH survey indices-at-age separately to evaluate the influence of each of the surveys on the base model and examine the model fits to both the aggregate survey indices and the indices-at-age (ASAP_MADMF and ASAP_MENH). The other two focused on fits to only the age-1 recruitment indices. The first of the models using state recruitment indices incorporated both the MADMF and MENH age-1 indices (ASAP_STATE_RECRUITMENT), the second used only the age-1 recruitment indices from the MENH survey (ASAP_MENH_RECRUITMENT).

A summary of model diagnostics from the four runs is presented in Table A.2.5. Model fits to the MADMF spring and fall survey are poor with large residuals in the fits to both spring and fall surveys (Fig. A.2.7). This is not unexpected given the large CVs on the MADMF indices (average 0.52 and 0.49 for spring and fall, respectively). The large number of zero survey observations in both spring and fall surveys are notable features of the MADMF survey. The model fit of the MENH spring survey is reasonable, though there are several large residuals in the fall survey, though no strong patterning in the fits to either survey (Fig. A.2.8). The fits to the indices-at-age show large residuals in the MADMF age-1 index, with several large positive and negative blocks throughout the survey time series (Fig. A.2.9).There are no strong residual patterns in the MENH spring survey, though the fall survey fits to the age-2 index are poor and the age-1 index contains several periods of large positive residuals. The trade-off between the age-1 and age-2 indices may be indicative of ageing issues complicated by the augmentation of the MENH fall survey with age-length information from the NEFSC survey. Index selectivity patterns for both surveys are characterized by low selectivity for older ages and high imprecision in estimates of selectivity-at-age (Table A.2.5). The combination of residual patterns and poorly estimated selectivity at older ages suggests that there is limited utility in incorporating the older age classes from the state surveys in the tuning of the ASAP model.

The model fits to the state recruitment indices exhibit similar patterns to the fits to the aggregate surveys (Fig. A.2.10), which is expected given that the majority of the signal in the aggregate 1-9^+ indices is coming from the age-1 index. Among the recruitment indices, the MENH spring survey appears to have the best model fit. Incorporating all state recruitment indices into a single model resulted in some diagnostics problems, notably, large residual errors in the estimation of SSB and fishing mortality (Table A.2.5), overall problems with the estimation of the 2003 year class in the retrospective analysis (Fig. A.2.11) and an overall underestimation of the year class size which manifested itself in a large cohort effect on the 2003 year class in the fits to catch-at-age (Fig. A.2.12). The run with only the MENH recruitment indices does not have these diagnostics issues. Additionally, the addition of the recruitment indices does offer marginally improved CVs on the estimation of age-1 recruitment (Fig. A.2.13). However, the model results between the ASAP_BASE and ASAP_MENH_RECRUITMENT are similar with respect to biomass scale, terminal SSB and F and retrospective patterns, suggesting that the indicating that model results are relatively insensitive to the inclusion of the additional recruitment indices.

A.2.7. Inclusion of catch-per-unit-effort (CPUE) indices

While several concerns were noted in the main report regarding the use of catch/landings-per-unit-effort (LPUE) indices in the base assessment model, sensitivity runs were conducted incorporating the commercial (ASAP_LPUE_COM) and recreational (ASAP_LPUE_REC) LPUE indices separately within the base ASAP model.

A summary of model diagnostics for the three runs is presented in Table A.2.6. Model fits to both the commercial and recreation LPUE indices exhibit a poor fit the index with strong residual patterning (Fig. A.2.14). These fits suggest that the LPUE indices are not reflective of stock abundance and should not be used for model tuning.

A.2.8. Sensitivity to recreational discard mortality assumptions

Previous assessments of Gulf of Maine haddock have not included estimates of recreational discards. In this assessment, the base model has assumed that fish discarded in the recreational fishery suffer 50% mortality, though there is little empirical information to evaluate this assumption. To understand the sensitivity of the assessment results to this assumption, the base ASAP model was run using two alternate discard mortality assumptions: 0% (ASAP_REC_0_MORT) and 100% (ASAP_REC_100_MORT). Catch weights-at-age were adjusted accordingly in each of the sensitivity runs (e.g., Tables A.46-47).

A summary of model diagnostics for the three runs is presented in Table A.2.7. The model diagnostics are similar for all three models. Decreasing the discard mortality assumption lowers the selectivity for ages 2-4 in the third selectivity block (2005-2013; Fig. A.2.15). This corresponds to the period of increasing recreational discard. The selectivity patterns in the other two blocks are similar across mortality assumptions. With the exception of the estimate of 2013 fishing mortality, which increases with increasing discard mortality, the model results are largely insensitive to the recreational discard mortality assumption (Fig. A.2.16),

A.2.9. Explicit treatment of catch fleets

A sensitivity run of the base ASAP model treated the commercial and recreational fleets separately, as opposed the ASAP_BASE model where the fishery catch is modeled in aggregate. The model configuration of the two fleet run (ASAP_2FLEET) was identical to that of the base run. The recreational fleet was modeled with only two selectivity blocks: 1977-1988, and 1989-2013. The 1988/89 period corresponds to an increase in minimum retention size from 17 to 19 inches (Table A.4). The CV on the recreational catch was set at 0.2 for all years, owing to the overall higher uncertainty in the recreational catch estimates.

A comparison of the summary diagnostics between the base and two fleet models is presented in Table A.2.8. Overall, the two fleet model has generally good diagnostics, with reasonable fits to both the commercial and recreational catches (Fig. A.2.17) and fits to the catch-at-age. There was a period of large residuals in the recreational catch-at-age in the pre-1995 period (Fig. A.2.18) when length sampling was poor (Table A.35). The residual patterns present in the base model are also reflected in the fits to the commercial catch-at-age. The results of the two models are nearly identical, with terminal SSB estimated at 4,500 mt in the base model and 4,242 mt in the two fleet model. The partitioning of fishing mortality mirrors the catch patterns, with the recreational fishing mortality increasing over the last decade, with the recreational and commercial fully recruited fishing mortality nearly equal in 2013 (Fig. A.2.19). Retrospective error was low in both the base and two fleet models. Selectivity was generally well estimated for the recreational fleet (Table A.2.8). Interestingly, the commercial fishery exhibits only weak evidence of doming, though there is evidence that the recreational fleet may be driving the fleet doming evident in the third selectivity block of the ASAP_BASE model. Domed selectivity in the recreational fishery could be explained by the spatial distribution of the recreational fishery – it's plausible that the older haddock may be in the deeper waters to the east of where the recreational fleet is operating in the western Gulf of Maine.

A.2.10. Assessment starting points (e.g., 1956, 1963 vs. 1977)

The ASAP_BASE model run begins in 1977, the year for which catch-at-age information is first available. Two alternate start points were explored within the framework of the ASAP_BASE model: 1956 (ASAP_hist_1956), the year when catch information are first available, and 1963 (ASAP_hist_1963), the year when the NEFSC fall bottom trawl survey began and survey indices-at-age first start. The NEFSC spring bottom trawl survey did not begin until 1968. The catch CV was increased to 0.2 in years before 1977. The uncertainty of catch pre-1964 which predates the modern commercial data collection program, is probably higher than the catch from 1964-1977, though no explicit adjustment is made in these sensitivity models.

A summary of model diagnostics is presented in Table A.2.9. In both runs, the model-estimated fishery catch exceeds the observed catch until 1980, with a strong negative residual pattern in these early years (Fig. A.2.20). The historical runs, ASAP_hist_1956 and ASAP_hist_1963, have nearly identical trends in spawning stock biomass, fishing mortality and age-1 recruitment during the overlapping years (Fig. A.2.21). With respect to evaluating the current condition of the stock, the choice in starting year has little impact, though both historical runs have slightly lower terminal SSB and higher fishing mortality. The 1956 run has a notable burn-in period from 1956 to 1960 – a large amount of fish are created in the 9^+ group at the start of the model and these fish leave the population before the availability of age information in 1963.

Extending the time series back in time establishes additional contrast in the spawner-recruit relationship, though given the large-burn in period in the 1956 run, the SR relationship of the pre-1964 year classes is questionable. The 1963 run shows evidence of small year classes spawned at high SSB, though the 1963 year class is the largest of the time series and was spawned during a year when SSB was near time series highs (Fig. A.2.22).

A.2.11. Placement of selectivity blocks

The base ASAP model included three fishery selectivity blocks: (1) 1977-1988, (2) 1989-2004, (3) 2005-2013. The model fits to the observed catch-at-age were generally good; however, there were some year class effects present, primarily associated with the 1998 and 2003 year classes. Several alternate model formulations explored different selectivity blocking to see if improvements could be made in the catch-at-age fits.

Two initial sensitivity runs exploring fits to a one- (ASAP_BASE_1BLOCK) and two-block (ASAP_BASE_2BLOCK) model did not yield appreciable gains in the catch-at-age fits. Summary diagnostics for these sensitivity runs are provided in Table A.83. The single block model exhibited strong residual runs in the fits to age-2 fish, did not mitigate the year class effects (Fig. A.2.23) and increased the retrospective error (Table A.83). The two-block model employed a split at 1993/1994. The choice of the split was informed by increases in the regulated mesh sizes (5.0 inches to 6.0 inches) and implementation of haddock trip limits (Table A.3). The post-1994 period also corresponds to a period of increasing recreational catch (Table A.8 and Fig. A.15). The two-block model exhibited nearly identical residual patterning to the three-block model and similar retrospective error. However, despite a decrease of eight parameters, the objective function increased by two points, primarily as results of a slight decrease in the fits to the catch-at-age. The two-block model is more parsimonious, but may not offer the flexibility to capture finer-scale dynamics of the fishery related to the increasing catch coming from the recreational fishery over time and changes in commercial minimum retention sizes from 2007 onward (Table A.4). The accurate estimation of selectivity in the most recent period has implications on reference point determination and setting catch advice.

Further refinement of the three-block model was explored by attempting different placements of the break between the second and third selectivity blocks. Several preliminary runs not described here were explored to find a year break point that offered improvements in the objective function and catch-at-age residual patterns. These runs resulted in an alternate three-block model with a split between the second and third blocks between 1999/2000 (ASAP_BASE_1989_2000). The first block timing remained unchanged.

Summary diagnostics for the three-block sensitivity runs are provided in Table A.84. The ASAP_BASE_1989_2000 model offered 35 point improvement in the objective function compared to the ASAP_BASE model with the improvements coming in the way of the fits to the catch-at-age and indices-at-age. The terminal spawning stock biomass and fishing mortality estimates were similar between the two models, though the 1999/2000 split model exhibited a minor increase in the retrospective error relative to the base model. Estimated selectivity was similar between the two runs for the first selectivity block, with both models estimating flat-topped selectivity in the 1977-1988 time period. There was severe doming in the second selectivity block, with the selectivity of the 9^+ group dropping to 0.11. The selectivity for ages 8 and 9^+ was poorly estimated, with CVs > 0.30. This doming effect is likely the result of the limited information on the older fish - the 1989-1999 period is characterized by a truncated population structure and very few old fish in the fishery catch. Compared to the ASAP_BASE model run, the selectivity in the third block of the ASAP_BASE_1989_2000 model was lower. The catch-at-age residual patterns were slightly improved in the ASAP_BASE_1989_2000 model, with reductions in the residuals on the 1998 and 2003 year classes (Fig. A.2.24). While the ASAP_BASE_1989_2000 model does exhibit some improved diagnostics, the severe doming in the second selectivity block is a concern. Additionally, some of the concerns with the two-block model (e.g., the increasing recreational fishery catch and changes in commercial minimum retention sizes in more recent period) also apply to the ASAP_BASE_1989_2000 model.

Given that the 1999/2000 run was suggestive of flat-topped selectivity in the first and third blocks, and poorly estimated selectivity on the older ages in the second block, an alternate formulation of the 1999/2000 run was explored where fishery selectivity was modeled using a single logistic function (ASAP_BASE_1989_2000_SL). The selectivity parameters were well estimated in the single logistic run. However, there was an overall degradation in model fit under this model, with 14 point increase in the objective function relative to the ASAP_BASE model and 49 point increase compared to the ASAP_BASE_1989_2000 model. The fits to the catch-at-age were the primary driver of the increase in the objective function (Table A.84). There is evidence for doming in the fishery, which is likely a function of the catchability of the recreational fishery. Use of a single logistic function to model fishery selectivity is probably not appropriate, particularly in the later part of the timer series when recreational catch increased.

Each of the three three-selectivity block models has advantages and disadvantages. However, the impacts on stock determination are minimal with the terminal spawning stock biomass and fishing mortality similar between all runs. While these investigations were useful, the decision was made to retain the ASAP_BASE formulation with three selectivity blocks and the 1988/1989 and 2004/2005 splits.

A.2.12. Catch precision assumptions

The CVs on the aggregate catch used in the base ASAP model varied from 0.15 early in the time series to 0.05 after 1989. The changes in assumed precision were reflective of the incorporation of direct commercial discard estimates in 1989 and imprecise recreational catch estimates early in the time period. Two different sensitivity runs were conducted to evaluate the impacts of assuming increased catch imprecision on model performance and results. The first run increased CVs across the time series by 100% (range = 0.10 - 0.30; ASAP_CATCH_CV_100) and the second increased CVs across the time series by 200% (range = 0.15 - 0.45; ASAP_CATCH_CV_200).

The model runs and summary diagnostics are presented in Table A.2.12. Increasing catch CVs lead to slight improvements in the model fits to the survey indices, survey age composition and catch age composition. The primary effect of the higher CVs was reduced fit to the aggregate catch, primarily in the early part of the time series, with very little overall change in the residual patterns, only in the magnitude of the standardized residuals (Fig. A.2.25). The 2013 estimates of spawning stock biomass ranged from 4,500 mt in the base model to 4,951 mt under the 200% increase model with only a minor change in terminal fishing mortality. Overall, increasing CVs on the aggregate catch had negligible impacts on the assessment results. The catch CVs assumed in the base model are likely too low, whereas the 200% model assumed CVs probably capture the upper limit of believable precision levels. The final ASAP model should utilize CVs somewhere in the mid-range of these sensitivities, though the choice is likely to have minimal impacts on the assessment results.

A.2.13. Terminal recruitment assumptions

In both the VPA and preliminary ASAP models, the most notable feature of the assessment results is the presence of occasionally large year classes, which in turn lead to subsequent increases in spawning stock biomass. As has been seen from the results of various sensitivities examined, the overall biomass scale and time series trends in biomass and fishing mortality are insensitive to the model assumptions; however, the estimates of terminal recruitment are sensitive to model configurations.

The ASAP model allows the deviations to be constrained by applying a penalty on the deviations. For the base run the penalty function (lambda) was set at 0.2 and the CVs on the recruitment deviations were set at 0.5 for all years except the final three years, which were set at 0.1. This was an attempt to apply ‘shrinkage’ to the mean of the terminal year cohorts were there are limited observations available from which to accurately estimate year class size. This decision was based on past experience with Gulf of Maine haddock in a VPA model framework (Palmer et al. 2014b). However, within the context of the ASAP, statistical catch-at-age model, the application of shrinkage to the mean had not been evaluated. Within ASAP, there are several ways to model recruitment deviations. One method is to not apply any penalty function to the recruitment deviations (set lambda=0) and allow recruitment to be freely estimated. Another is set lambda at 1, but apply some constraint on the recruitment deviations through the adjustment of the CV values. When applying shrinkage to the mean, the shrinkage can be applied over several terminal years (e.g., those years not fully recruited to the surveys or fishery) or just on the terminal year.

A sensitivity was conducted evaluating the retrospective performance of the ASAP model under the four different configurations described above. The performance of the model was evaluated back to 2000 to understand how well the model estimated the size of the 1998 year class with only three years of information, which is identical to the current scenario with respect to being able to estimate the size of the 2010 year class. For these sensitivities, a single-selectivity block variant of the ASAP_BASE model was created. Using a single selectivity block avoids complications that can occur when retrospective runs pass through selectivity blocks. For all runs, the recruitment deviation CV was set equal to 0.8, with the exception of the shrinkage runs. Where shrinkage was applied to the last four years, the CV was decreased by 30%/year (e.g., 2009=0.80, 2010=0.59, 2011=0.39, 2012=0.27, 2013=0.19). For the runs where shrinkage was only applied to the terminal year, the CV in the terminal year was set at 0.3.

The results of the retrospective analysis on a cohort basis are shown in Figure A.2.26. Results are expressed in terms of relative estimation error compared to the 2013 estimates of age-1 recruitment from the 'no shrinkage' model. Overall, the performance of the individual methods was variable. While the 'shrinkage on last four years' model performed poorly for some year classes (e.g., 2004, 2007), for some year classes it out-performed the other methods (e.g., 2000, 2001, 2009). For the three large year classes within the time series (1998, 2003, 2010), there were no clear 'preferred' method. Box plot distributions of the retrospective error by method were examined at yearly intervals in an attempt to summarize the performance of the methods when only one, two, three and four years of information were available with which to estimate the year class size. When only one year of information was available the 'no shrinkage' method had the lowest error distribution and exhibited the lowest degree of mean- and median-bias (Fig. A.2.27). For all other years, the 'no shrinkage', 'shrinkage in the terminal year' and 'lambda=0' methods appeared to perform similarly with only marginal differences between the methods. An interesting result from this analysis, is that at the three and four year horizons, all methods had a tendency to underestimate year class size. While not conclusive, this analysis does suggest that within a Gulf of Maine haddock ASAP model, the 'no shrinkage' method offers a lower degree of recruitment estimation error compared to the other methods evaluated.

The specification of the recruitment deviations CVs will affect the level of constraint the model places on recruitment estimates. The sensitivity of the model to recruitment deviation CVs was evaluated by profiling across CV values from 0.6 to 2.4. This sensitivity used the ASAP_BASE model, with the exception that the recruitment deviation lambda was set at 1.0 and the CV was held constant for all years. The results of the profiling exercise are shown in Figure A.2.28. Based on the profiles of the likelihoods, there is model preference for CVs on the order of 2.0; this is the point when the RMSE on the recruitment deviations approaches 1. Within this range, model results are relatively stable (e.g., 2013 SSB, 2010 and 2012 year class sizes and SSB and F retrospective patterns). One concern with the model runs at the high CVs are the fits to the survey indices - models with high recruitment deviation CVs tend to 'chase' the 2013 survey observation (Fig. A.2.29). Based on the model fits to large survey observations earlier in the time series, this degree of fit, seems unlikely.

Appendix A.2. Tables

Table A.2.1. Summary of model diagnostics from several ASAP runs exploring the sensitivity of the ASAP_BASE model to varying levels of natural mortality (M). For runs above M=0.4, only results from every other run are shown.

Model		ASAP_M_0	ASAP_M_1	ASAP_BASE	ASAP_M_3	ASAP_M_4	ASAP_M_6	ASAP_M_8	ASAP_M_10
Model description		M=0.0	M=0.1	M=0.2	M=0.3	M=0.4	M=0.6	M=0.8	M=1.0
Maximum gradient (*conv. criteria < 1e-4*)		1.62E-04	1.01E-03	1.10E-05	3.30E-05	4.28E-04	1.97E-04	1.26E-04	1.45E-03
Number of parameters		125	125	125	125	125	125	125	125
Objective function		2525	2520	2526	2542	2572	2671	2761	2840
Components of objective function	Recruit devs	105	107	110	114	119	129	144	160
	Suvey age comps	886	877	874	873	873	883	891	914
	Catch age comps	644	642	644	649	660	706	754	794
	Index fit	694	697	702	709	723	753	777	777
	Catch fit	197	196	196	197	198	199	194	194
RMSE	Fleet 1	0.37	0.34	0.34	0.38	0.44	0.52	0.07	0.03
	Index 1	2.03	2.06	2.10	2.16	2.26	2.47	2.65	2.65
	Index 2	1.96	1.98	2.00	2.04	2.12	2.26	2.34	2.34
	Index total	2.00	2.02	2.05	2.10	2.19	2.37	2.50	2.50
	Recruit devs	3.98	3.93	3.91	3.91	3.93	3.95	3.71	3.73
SSB_{1977} (mt)		6,057	7,491	9,470	16,907	24,006	62,059	1,013,640	2,196,730
SSB_{2013} (mt)		6,861	4,958	4,500	6,355	8,476	21,049	326,461	683,818
$F_{mult,\ 2013}$		0.24	0.28	0.31	0.35	0.38	0.34	0.05	0.04
Mohn's rho (7 year peel)	SSB	0.25	0.10	-0.03	-0.15	-0.25	-0.34	-0.64	-0.52
	F_{mult}	-0.26	-0.13	0.05	0.24	0.47	0.82	4.85	4.08
	Age 1 N	0.32	0.24	0.18	0.11	0.05	-0.03	-0.65	-0.52
Notes		M set to 0.0001 to get model to run							

Table A.2.2. Summary of model diagnostics from a sensitivity analysis of the Gulf of Maine haddock ASAP_BASE model to the use of survey abundance (numbers) or biomass (weight) indices.

Model		**ASAP_BASE**	**ASAP_BIOMASS**
Model description		Fit to NEFSC abundance indices	Fit to NEFSC biomass indices
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05	2.03E-04
Number of parameters		125	125
Objective function		2526	2670
Components of objective function	**Recruit devs**	110	100
	Suvey age comps	874	980
	Catch age comps	644	647
	Index fit	702	745
	Catch fit	196	197
RMSE	**Fleet 1**	0.34	0.40
	Index 1	2.10	2.59
	Index 2	2.00	1.90
	Index total	2.05	2.27
	Recruit devs	3.91	3.58
SSB_{1977} (mt)		9,470	10,586
SSB_{2013} (mt)		4,500	3,801
SSB_{2013} CV		0.15	0.15
$F_{mult, 2013}$		0.31	0.38
Mohn's rho (7 year peel)	**SSB**	-0.03	-0.14
	F_{mult}	0.05	0.32
	Age 1 N	0.18	0.16

Table A.2.3. Summary of model diagnostics for variants of the ASAP_BASE model fit to (1) a single Albatross/Bigelow-calibrated series (fall and spring; ASAP_BASE), (2) both the Albatross/Bigelow-converted series and a separate un-calibrated Bigelow series (ASAP_BIGELOW_SPLIT), and (3) separate Albatross (1977-2008) and Bigelow (2009-2013) non-overlapping series (ASAP_BIGELOW_NOOVERLAP).

Model		ASAP_BASE		ASAP_BIGELOW_SPLIT		ASAP_BIGELOW_NOOVERLAP	
Model description		Fit to NEFSC abundance indices		Treat Bigelow years as separate survey		Treat Bigelow years as separate survey	
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05		6.54E-04		9.10E-05	
Number of parameters		125		143		143	
Objective function		2526		2721		2470	
Components of objective function	Recruit devs	110		123		86	
	Suvey age comps	874		957		860	
	Catch age comps	644		643		640	
	Index fit	702		801		688	
	Catch fit	196		196		196	
RMSE	Fleet 1	0.34		0.34		0.34	
	Index 1	2.10		2.09		1.98	
	Index 2	2.00		1.98		1.75	
	Index 3			1.75		2.09	
	Index 4			1.84		2.41	
	Index total	2.05		2.01		1.92	
	Recruit devs	3.91		4.34		3.04	
Survey q	NEFSC spring	0.26	*(0.11)*	0.26	*(0.11)*	0.25	*(0.11)*
	NEFSC fall	0.99	*(0.11)*	1.00	*(0.11)*	1.10	*(0.11)*
	Bigelow spring			0.87	*(0.31)*	1.67	*(0.20)*
	Bigelow fall			1.90	*(0.23)*	4.04	*(0.21)*
SSB_{1977} (mt)		9,470		9,445		9,469	
SSB_{2013} (mt)		4,500		3,850		1,357	
$F_{mult,2013}$		0.31		0.36		1.13	
Mohn's rho (7 year peel)	SSB	-0.03		0.04		0.36	
	F_{mult}	0.05		-0.02		-0.24	
	Age 1 N	0.18		0.31		1.36	
Block/Index		Selectivity	CV	Selectivity	CV	Selectivity	CV
NEFSC spring	1	0.74	0.13	0.74	0.13	0.74	0.14
	2	0.62	0.14	0.62	0.14	0.61	0.15
	3	0.79	0.14	0.79	0.14	0.79	0.15
	4	1.00	0.00	1.00	0.00	1.00	0.00
	5	0.87	0.17	0.87	0.17	0.93	0.18
	6	1.00				1.00	
	7	0.82	0.25	0.82	0.25	0.57	0.31
	8	0.75	0.34	0.76	0.34	0.68	0.39
	9^+	0.76	0.30	0.76	0.30	0.51	0.38
NEFSC fall	1	0.28	0.14	0.29	0.14	0.23	0.15
	2	0.35	0.14	0.35	0.14	0.28	0.15
	3	0.60	0.13	0.60	0.13	0.50	0.14
	4	0.62	0.14	0.61	0.14	0.58	0.15
	5	0.76	0.15	0.75	0.14	0.65	0.15
	6	1.00				1.00	
	7	1.00	0.00	1.00	*0.00*	1.00	0.00
	8	1.00	0.00	1.00	0.00	1.00	0.00
	9^+	0.66	0.25	0.66	0.25	0.80	0.26
Bigelow spring	1			0.89	0.36	1.00	0.00
	2			0.96	0.42	1.00	0.00
	3			0.84	0.47	0.87	0.41
	4			0.49	0.61	0.43	0.58
	5			0.37	0.83	0.31	0.80
	6					1.00	
	7			1.00	0.00	1.00	0.00
	8			0.68	0.80	0.56	0.76
	9^+			0.71	0.50	0.49	0.45
Bigelow fall	1			0.61	0.27	0.74	0.26
	2			1.00	0.00	1.00	0.00
	3			0.97	0.32	1.00	0.00
	4			0.34	0.53	0.30	0.52
	5			1.00	0.00	1.00	0.00
	6					1.00	
	7			0.46	0.54	0.38	0.54
	8			0.99	0.58	0.89	0.59
	9^+			0.25	0.57	0.18	0.57

Table A.2.4. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock ASAP_BASE model to inclusion of only a single NEFSC survey index at one time.

Model		ASAP_BASE	ASAP_NEFSC_spring	ASAP_NEFSC_fall
Model description		Fit to NEFSC abundance indices	Run _BASE w/ spring only	Run _BASE w/ fall only
Maximum gradient (*conv. criteria* < *1e-4*)		1.10E-05	6.97E-04	1.26E-04
Number of parameters		125	116	116
Objective function		2526	1652	1785
Components of objective function	**Recruit devs**	110	89	99
	Suvey age comps	874	381	489
	Catch age comps	644	629	634
	Index fit	702	381	368
	Catch fit	196	196	195
RMSE	**Fleet 1**	0.34	0.29	0.24
	Index 1	2.10		
	Index 2	2.00		
	Index total	2.05	2.25	2.14
	Recruit devs	3.91	3.13	3.52
SSB_{1977} (mt)		9,470	8,347	9,314
SSB_{2013} (mt)		4,500	2,828	3,825
$F_{mult, 2013}$		0.31	0.47	0.40
Mohn's rho (7 year peel)	**SSB**	-0.03	0.19	-0.05
	F_{mult}	0.05	-0.13	0.15
	Age 1 N	0.18	0.29	0.75
Survey q	**NEFSC spring**	0.26	0.27	
	NEFSC fall	0.99		0.96

Table A.2.5. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model to the incorporation of state survey indices.

Model		ASAP_BASE	ASAP_MADMF	ASAP_MENH	ASAP_STATE_RECRUITMENT	ASAP_MENH_RECRUITMENT
Model description		Fit to NEFSC abundance indices	Include MADMF inshore surveys	Include MADMF inshore surveys	Include state surveys as recruitment indices	Include MENH survey as recruitment indices
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05	9.70E-05	2.90E-05	8.00E-05	2.60E-05
Number of parameters		125	143	143	129	127
Objective function		2526	3111	2761	3132	2670
Components of objective function	Recruit devs	110	102	110	102	112
	Suvey age comps	874	1082	940	875	872
	Catch age comps	644	658	643	681	644
	Index fit	702	1070	871	1272	846
	Catch fit	196	198	196	202	196
RMSE	Fleet 1	0.34	0.48	0.34	0.64	0.38
	Index 1	2.10	2.05	2.12	2.09	2.11
	Index 2	2.00	2.03	2.03	2.07	2.01
	Index 3		2.83	0.88	4.37	1.19
	Index 4		2.50	1.88	2.60	1.63
	Index 5				0.88	
	Index 6				1.83	
	Index total	2.05	2.33	1.94	2.61	1.93
	Recruit devs	3.91	3.64	3.91	3.64	3.98
Survey q	NEFSC spring	0.26	0.27	0.26	0.28	0.26
	NEFSC fall	0.99	1.10	1.00	1.12	1.00
	MADMF spring		0.02		0.01	
	MADMF fall		0.01		0.01	
	MENH spring			0.17	0.19	0.17
	MENH fall			0.05	0.04	0.03
SSB_{1977} (mt)		9,470	9,224	9,441	9,320	9,452
SSB_{2013} (mt)		4,500	4,240	3,923	3,512	4,082
$F_{mult,2013}$		0.31	0.32	0.35	0.43	0.34
Mohn's rho (7 year peel)	SSB	-0.03	0.00	-0.04	-0.30	-0.04
	F_{mult}	0.05	0.00	0.04	0.51	0.05
	Age 1 N	0.18	0.19	0.08	-0.20	-0.07

	Block/Index	Selectivity	CV	Selectivity	CV	Selectivity	CV
State spring	1			1.00		1.00	
	2			0.18	0.34	0.08	0.54
	3			0.33	0.32	0.05	0.76
	4			0.26	0.36	0.08	0.59
	5			0.93	0.28	0.08	0.57
	6			0.43	0.36	0.05	0.95
	7			1.00	0.00	0.05	0.96
	8			1.00	0.00	0.05	1.35
	9^+			0.88	0.47	0.05	0.85
State fall	1			1.00		1.00	
	2			0.05	0.50	0.29	0.35
	3			0.02	0.65	0.09	0.50
	4			0.04	0.68	0.02	1.14
	5			0.28	0.37	0.12	0.60
	6			0.22	0.58	0.04	0.98
	7			0.17	0.69	0.16	0.84
	8			0.15	0.85	0.04	1.61
	9^+			0.46	0.53	0.01	2.01

Table A.2.6. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model to the incorporation of commercial (COM_LPUE), recreational (REC_LPUE) landings-per-unit-effort indices.

Model		**ASAP_BASE**		**ASAP_LPUE_COM**		**ASAP_LPUE_REC**	
Model description		Fit to NEFSC abundance indices		Commercial LPUE		Recreational LPUE	
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05		2.96E-04		5.90E-05	
Number of parameters		125		126		126	
Objective function		2526		2535		2575	
Components of objective function	**Recruit devs**	110		109		112	
	Suvey age comps	874		871		874	
	Catch age comps	644		672		654	
	Index fit	702		684		739	
	Catch fit	196		198		196	
RMSE	**Fleet 1**	0.34		0.45		0.35	
	Index 1	2.10		2.12		2.07	
	Index 2	2.00		2.07		1.98	
	Index 3			1.59		1.67	
	Index total	2.05		1.94		1.96	
	Recruit devs	3.91		3.89		3.96	
SSB_{1977} (mt)		9,470		9,520		9,513	
SSB_{2013} (mt)		4,500		3,441		5,437	
$F_{mult,2013}$		0.31		0.53		0.21	
Mohn's rho (7 year peel)	**SSB**	-0.03		-0.15		-0.05	
	F_{mult}	0.05		0.34		0.07	
	Age 1 N	0.18		0.17		0.05	
	Notes					*Note that 0.2 were added to the CVs	
Block/Index		**Selectivity**	**CV**	**Selectivity**	**CV**	**Selectivity**	**CV**
Fleet block 1 (1977-1988)	**1**	0.00	0.61	0.01	0.60	0.00	0.61
	2	0.22	0.12	0.32	0.09	0.22	0.12
	3	0.39	0.11	0.51	0.07	0.39	0.11
	4	0.64	0.10	0.87	0.07	0.64	0.10
	5	0.65	0.11	1.00	0.00	0.65	0.11
	6	0.77	0.12	1.00	0.00	0.78	0.12
	7	1.00					
	8	1.00	0.00	1.00	0.00	1.00	0.00
	9^+	1.00	0.00	1.00	0.00	1.00	0.00
Fleet block 2 (1989-2004)	**1**	0.01	0.27	0.00	0.26	0.01	0.25
	2	0.06	0.16	0.04	0.13	0.05	0.13
	3	0.34	0.14	0.20	0.11	0.27	0.10
	4	0.55	0.14	0.32	0.11	0.41	0.11
	5	0.72	0.14	0.42	0.11	0.53	0.11
	6	0.96	0.14	0.57	0.11	0.73	0.11
	7	1.00					
	8	0.94	0.21	0.63	0.19	0.70	0.19
	9^+	0.67	0.30	0.34	0.28	0.47	0.27
Fleet block 3 (2005-2013)	**1**	0.01	0.42	0.01	0.41	0.01	0.41
	2	0.06	0.21	0.04	0.20	0.08	0.20
	3	0.22	0.17	0.15	0.16	0.26	0.15
	4	0.31	0.17	0.24	0.16	0.35	0.16
	5	0.56	0.15	0.43	0.15	0.63	0.15
	6	0.77	0.15	0.60	0.14	0.91	0.14
	7	1.00					
	8	1.00	0.00	0.88	0.14	1.00	0.00
	9^+	0.74	0.19	0.80	0.18	1.00	0.00

Table A.2.7. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model to different assumptions on the mortality of fish discarded in the recreational fishery.

Model		**ASAP_REC_0_MORT**	**ASAP_BASE**	**ASAP_REC_100_MORT**
Model description		ASAP base run (0% rec disc mort)	ASAP base run (50% rec disc mort)	ASAP base run (100% rec disc mort)
Maximum gradient (*conv. criteria < 1e-4*)		1.31E-03	1.10E-05	3.00E-05
Number of parameters		125	125	125
Objective function		2534	2526	2523
Components of objective function	**Recruit devs**	108	110	111
	Suvey age comps	878	874	871
	Catch age comps	650	644	643
	Index fit	702	702	701
	Catch fit	195	196	197
RMSE	**Fleet 1**	0.34	0.34	0.34
	Index 1	2.10	2.10	2.09
	Index 2	2.01	2.00	2.00
	Index total	2.06	2.05	2.05
	Recruit devs	3.83	3.91	3.94
SSB_{1977} (mt)		9,507	9,470	9,461
SSB_{2013} (mt)		4,764	4,500	4,419
$F_{mult,2013}$		0.21	0.31	0.38
Mohn's rho (7 year peel)	**SSB**	-0.06	-0.03	-0.04
	F_{mult}	0.11	0.05	0.05
	Age 1 N	0.16	0.18	0.14

Table A.2.8. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model with explicit treatment of the commercial and recreational fleets.

Model		ASAP_BASE		ASAP_2_FLEETS	
Model description		Fit to NEFSC abundance indices		Commercial and recreational catch	
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05		1.60E-04	
Number of parameters		125		178	
Objective function		2526		2905	
Components of objective function	Recruit devs	110		111	
	Suvey age comps	874		878	
	Catch age comps	644		967	
	Index fit	702		702	
	Catch fit	196		247	
RMSE	Fleet 1	0.34		0.35	
	Fleet 2			0.12	
	Catch total			0.26	
	Index 1	2.10		2.11	
	Index 2	2.00		2.00	
	Index total	2.05		2.06	
	Recruit devs	3.91		3.96	
SSB_{1977} (mt)		9,470		9,167	
SSB_{2013} (mt)		4,500		4,242	
$F_{mult, 2013}$		0.31		0.45	
Mohn's rho (7 year peel)	SSB	-0.03		0.04	
	F_{mult}	0.05		-0.02	
	Age 1 N	0.18		0.30	
Block/Index		Selectivity	CV	Selectivity	CV
Fleet block 1 (1977-1988)	1	0.00	0.61	0.00	0.64
	2	0.22	0.12	0.19	0.17
	3	0.39	0.11	0.33	0.16
	4	0.64	0.10	0.56	0.16
	5	0.65	0.11	0.57	0.16
	6	0.77	0.12	0.69	0.17
	7	1.00		0.81	0.20
	8	1.00	0.00	1.00	
	9^+	1.00	0.00	0.97	0.36
Fleet block 2 (1989-2004)	1	0.01	0.27	0.01	0.26
	2	0.06	0.16	0.06	0.15
	3	0.34	0.14	0.31	0.12
	4	0.55	0.14	0.51	0.12
	5	0.72	0.14	0.68	0.12
	6	0.96	0.14	0.94	0.12
	7	1.00		1.00	
	8	0.94	0.21	1.00	0.00
	9^+	0.67	0.30	0.87	0.29
Fleet block 3 (2005-2013)	1	0.01	0.42	0.01	0.36
	2	0.06	0.21	0.03	0.27
	3	0.22	0.17	0.11	0.21
	4	0.31	0.17	0.20	0.20
	5	0.56	0.15	0.46	0.18
	6	0.77	0.15	0.63	0.17
	7	1.00		0.81	0.16
	8	1.00	0.00	1.00	
	9^+	0.74	0.19	0.71	0.19
Recreational block 1 (1977-1999)	1			0.01	0.42
	2			0.10	0.21
	3			0.41	0.19
	4			0.37	0.20
	5			0.41	0.21
	6			0.35	0.25
	7			0.45	0.31
	8			1.00	
	9^+			0.36	0.55
Recreational block 2 (2000-2013)	1			0.01	0.86
	2			0.12	0.33
	3			0.33	0.29
	4			0.46	0.30
	5			0.72	0.29
	6			0.96	0.28
	7			0.91	0.30
	8			1.00	
	9^+			0.68	0.31

Table A.2.9. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model to the assessment starting year.

Model		**ASAP_BASE**	**ASAP_HIST_1956**	**ASAP_HIST_1963**
Model description		Starts in 1977 w/start of CAA	Starts in 1956 w/ start of catch series	Starts in 1963 w/ start of NEFSC survey
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05	4.90E-05	3.80E-05
Number of parameters		125	167	153
Objective function		2526	3153	3088
Components of objective function	**Recruit devs**	110	160	146
	Suvey age comps	874	1039	1039
	Catch age comps	644	662	663
	Index fit	702	936	938
	Catch fit	196	357	301
RMSE	**Fleet 1**	0.34	0.56	0.58
	Index 1	2.10	2.08	2.09
	Index 2	2.00	2.01	2.03
	Index total	2.05	2.04	2.06
	Recruit devs	3.91	3.58	3.77
SSB_{start} (mt)			58,610	19,131
SSB_{1977} (mt)		9,470	12,248	12,582
SSB_{2013} (mt)		4,500	3,595	3,493
$F_{mult, 2013}$		0.31	0.40	0.41
Mohn's rho (7 year peel)	**SSB**	-0.03	-0.16	-0.16
	F_{mult}	0.05	0.28	0.27
	Age 1 N	0.18	0.17	0.14
	Notes		Some retro iterations did not converge	

Table A.2.10. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model exploring one- and two-selectivity block model formulations.

Model		ASAP_BASE_1BLOCK		ASAP_BASE_2BLOCK		ASAP_BASE	
Model description		1 catch selectivity block		2 catch selectivity blocks (1994)		3 catch selectivity blocks (1989,2005)	
Maximum gradient (*conv. criteria < 1e-4*)		3.45E-04		2.20E-05		1.10E-05	
Number of parameters		109		117		125	
Objective function		2575		2528		2526	
Components of objective function	Recruit devs	109		113		110	
	Suvey age comps	880		869		874	
	Catch age comps	684		654		644	
	Index fit	706		697		702	
	Catch fit	197		196		196	
RMSE	Fleet 1	0.36		0.32		0.34	
	Index 1	2.12		2.07		2.10	
	Index 2	2.04		1.97		2.00	
	Index total	2.08		2.02		2.05	
	Recruit devs	3.86		4.01		3.91	
SSB_{1977} (mt)		8,708		9,027		9,470	
SSB_{2013} (mt)		4,309		3,821		4,500	
$F_{mult, 2013}$		0.27		0.41		0.31	
Mohn's rho (7 year peel)	SSB	-0.13		-0.04		-0.03	
	F_{mult}	0.15		0.07		0.05	
	Age 1 N	0.19		0.25		0.18	
Block/Index		Selectivity	CV	Selectivity	CV	Selectivity	CV
Fleet block 1	A50%						
	Slope						
Fleet block 2	A50%						
	Slope						
Fleet block 3	A50%						
	Slope						
Fleet block 1	1	0.01	0.21	0.00	0.61	0.00	0.61
	2	0.09	0.11	0.19	0.15	0.22	0.12
	3	0.30	0.10	0.33	0.13	0.39	0.11
	4	0.48	0.10	0.55	0.13	0.64	0.10
	5	0.57	0.10	0.56	0.14	0.65	0.11
	6	0.74	0.10	0.68	0.14	0.77	0.12
	7	0.84	0.10	0.76	0.17	1.00	
	8	1.00		1.00		1.00	0.00
	9^{+}	0.79	0.18	1.00	0.00	1.00	0.00
Fleet block 2	1			0.01	0.23	0.01	0.27
	2			0.05	0.14	0.06	0.16
	3			0.26	0.13	0.34	0.14
	4			0.41	0.13	0.55	0.14
	5			0.57	0.12	0.72	0.14
	6			0.78	0.12	0.96	0.14
	7			0.87	0.13	1.00	
	8			1.00		0.94	0.21
	9^{+}			0.76	0.19	0.67	0.30
Fleet block 3	1					0.01	0.42
	2					0.06	0.21
	3					0.22	0.17
	4					0.31	0.17
	5					0.56	0.15
	6					0.77	0.15
	7					1.00	
	8					1.00	0.00
	9^{+}					0.74	0.19
NEFSC spring	1	0.75	0.13	0.74	0.13	0.74	0.13
	2	0.63	0.14	0.62	0.14	0.62	0.14
	3	0.79	0.14	0.79	0.14	0.79	0.14
	4	1.00	0.00	1.00	0.00	1.00	0.00
	5	0.86	0.17	0.87	0.17	0.87	0.17
	6	1.00		1.00		1.00	
	7	0.84	0.25	0.82	0.25	0.82	0.25
	8	0.76	0.34	0.75	0.34	0.75	0.34
	9^{+}	0.78	0.30	0.81	0.31	0.76	0.30
NEFSC fall	1	0.29	0.13	0.28	0.14	0.28	0.14
	2	0.35	0.14	0.35	0.14	0.35	0.14
	3	0.60	0.13	0.60	0.13	0.60	0.13
	4	0.60	0.14	0.61	0.14	0.62	0.14
	5	0.74	0.14	0.75	0.15	0.76	0.15
	6	1.00		1.00		1.00	
	7	1.00	0.00	1.00	0.00	1.00	0.00
	8	1.00	0.00	1.00	0.00	1.00	0.00
	9^{+}	0.67	0.26	0.71	0.26	0.66	0.25

Table A.2.11. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model exploring alternate three-block models.

Model		**ASAP_BASE**		**ASAP_BASE_1989_2000**		**ASAP_BASE_1989_2000_SL**	
Model description		3 catch selectivity blocks (1989,2005)		3 catch selectivity blocks (1989,2000)		3 catch selectivity blocks (1989,2000), catch selectivity modelled as single logistic function	
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05		1.85E-04		2.35E-04	
Number of parameters		125		125		107	
Objective function		2526		2491		2540	
Components of objective function	**Recruit devs**	110		112		112	
	Suvey age comps	874		858		861	
	Catch age comps	644		623		672	
	Index fit	702		701		700	
	Catch fit	196		196		196	
RMSE	**Fleet 1**	0.34		0.29		0.31	
	Index 1	2.10		2.10		0.00	
	Index 2	2.00		1.99		2.08	
	Index total	2.05		2.05		1.99	
	Recruit devs	3.91		3.99		2.04	
SSB_{1977} (mt)		9,470		9,529		9,096	
SSB_{2013} (mt)		4,500		4,391		4,050	
$F_{mult, 2013}$		0.31		0.33		0.41	
Mohn's rho (7 year peel)	**SSB**	-0.03		0.11		0.10	
	F_{mult}	0.05		-0.09		-0.03	
	Age 1 N	0.18		0.29		0.30	
Block/Index		**Selectivity**	**CV**	**Selectivity**	**CV**	**Selectivity**	**CV**
Fleet block 1	A50%					2.90	0.05
	Slope					0.62	0.11
Fleet block 2	A50%					2.85	0.03
	Slope					0.40	0.06
Fleet block 3	A50%					4.62	0.04
	Slope					0.84	0.07
Fleet block 1	**1**	0.00	0.61	0.00	0.61		
	2	0.22	0.12	0.24	0.13		
	3	0.39	0.11	0.42	0.11		
	4	0.64	0.10	0.68	0.11		
	5	0.65	0.11	0.69	0.11		
	6	0.77	0.12	0.80	0.13		
	7	1.00		1.00			
	8	1.00	0.00	1.00	0.00		
	9^+	1.00	0.00	1.00	0.00		
Fleet block 2	**1**	0.01	0.27	0.01	0.31		
	2	0.06	0.16	0.09	0.22		
	3	0.34	0.14	0.51	0.19		
	4	0.55	0.14	0.77	0.19		
	5	0.72	0.14	0.85	0.19		
	6	0.96	0.14	0.91	0.20		
	7	1.00		1.00			
	8	0.94	0.21	0.57	0.33		
	9^+	0.67	0.30	0.11	0.43		
Fleet block 3	**1**	0.01	0.42	0.01	0.39		
	2	0.06	0.21	0.05	0.18		
	3	0.22	0.17	0.17	0.14		
	4	0.31	0.17	0.32	0.13		
	5	0.56	0.15	0.60	0.12		
	6	0.77	0.15	0.91	0.11		
	7	1.00		1.00			
	8	1.00	0.00	1.00	0.00		
	9^+	0.74	0.19	0.90	0.18		
NEFSC spring	**1**	0.74	0.13	0.75	0.13	0.72	0.14
	2	0.62	0.14	0.63	0.14	0.61	0.15
	3	0.79	0.14	0.79	0.14	0.76	0.16
	4	1.00	0.00	1.00	0.00	1.00	
	5	0.87	0.17	0.87	0.17	0.88	0.18
	6	1.00		1.00		0.99	0.21
	7	0.82	0.25	0.79	0.25	0.84	0.26
	8	0.75	0.34	0.70	0.34	0.75	0.34
	9^+	0.76	0.30	0.71	0.30	0.86	0.28
NEFSC fall	**1**	0.28	0.14	0.30	0.14	0.21	0.21
	2	0.35	0.14	0.37	0.14	0.26	0.22
	3	0.60	0.13	0.64	0.13	0.45	0.21
	4	0.62	0.14	0.65	0.14	0.47	0.22
	5	0.76	0.15	0.78	0.15	0.59	0.22
	6	1.00		1.00		0.68	0.23
	7	1.00	0.00	1.00	0.00	0.80	0.24
	8	1.00	0.00	1.00	0.00	1.00	
	9^+	0.66	0.25	0.63	0.25	0.61	0.27

Table A.2.12. Summary model diagnostics from a sensitivity analysis of the Gulf of Maine haddock base ASAP model exploring varying levels of assumed precision in fishery catches.

Model		**ASAP_BASE**	**ASAP_CATCH_CV_100**	**ASAP_CATCH_CV_200**
Model description		Fleet CVs ranged 0.05-0.15	Inflate CVs by 100%	Inflate CVs by 200%
Maximum gradient (*conv. criteria < 1e-4*)		1.10E-05	9.90E-05	8.20E-05
Number of parameters		125	125	125
Objective function		2526	2545	2552
Components of objective function	**Recruit devs**	110	110	111
	Suvey age comps	874	873	871
	Catch age comps	644	640	635
	Index fit	702	695	688
	Catch fit	196	227	247
RMSE	**Fleet 1**	0.34	0.63	0.84
	Index 1	2.10	2.07	2.03
	Index 2	2.00	1.95	1.89
	Index total	2.05	2.00	1.96
	Recruit devs	3.91	3.91	3.92
SSB_{1977} (mt)		9,470	9,927	10,854
SSB_{2013} (mt)		4,500	4,651	4,951
$F_{mult, 2013}$		0.31	0.29	0.27
Mohn's rho (7 year peel)	**SSB**	-0.03	0.00	0.05
	F_{mult}	0.05	0.01	-0.05
	Age 1 N	0.18	0.19	0.22
Survey q	**NEFSC spring**	0.26	0.25	0.24
	NEFSC fall	0.99	0.97	0.93

Appendix A.2. Figures

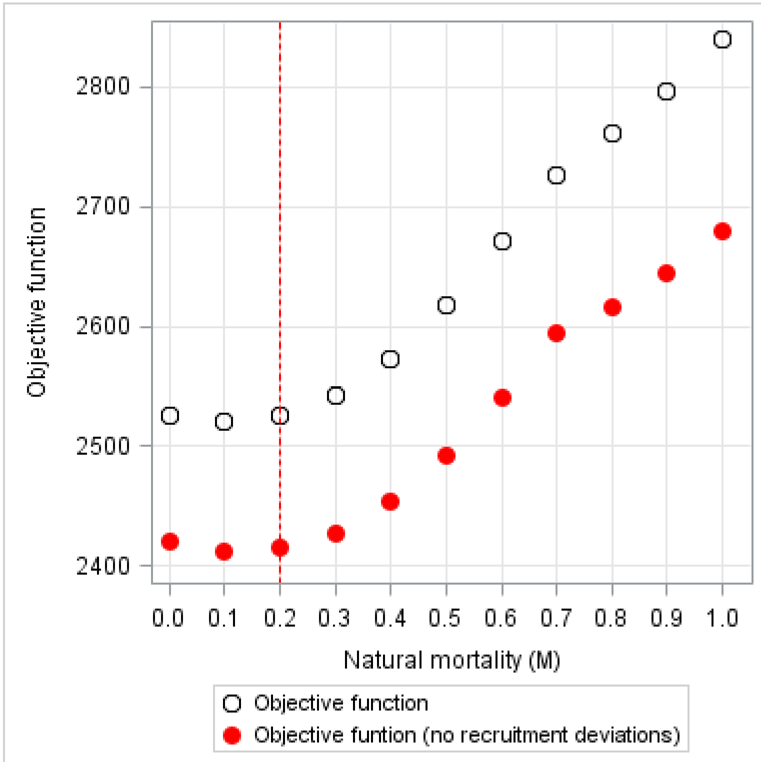


Figure A.2.1. Response of the model objective function to profiling over a range of Gulf of Maine haddock natural mortality values. The dashed red line indicates the M=0.2 assumption applied in the ASAP_BASE model.

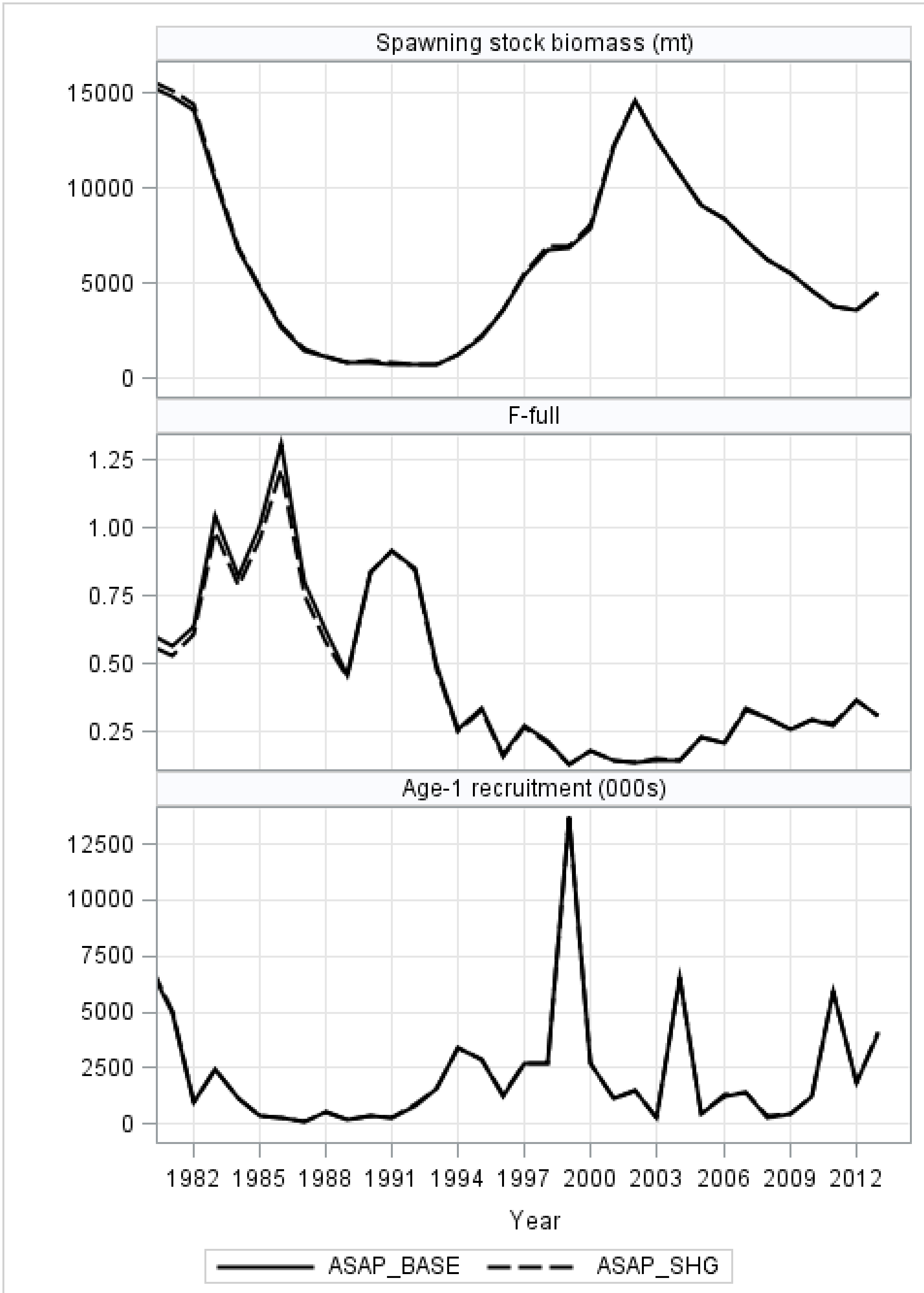


Figure A.2.2. Sensitivity of the Gulf of Maine haddock ASAP_BASE assessment model to use of either the SHG or TOGA tow evaluation criteria.

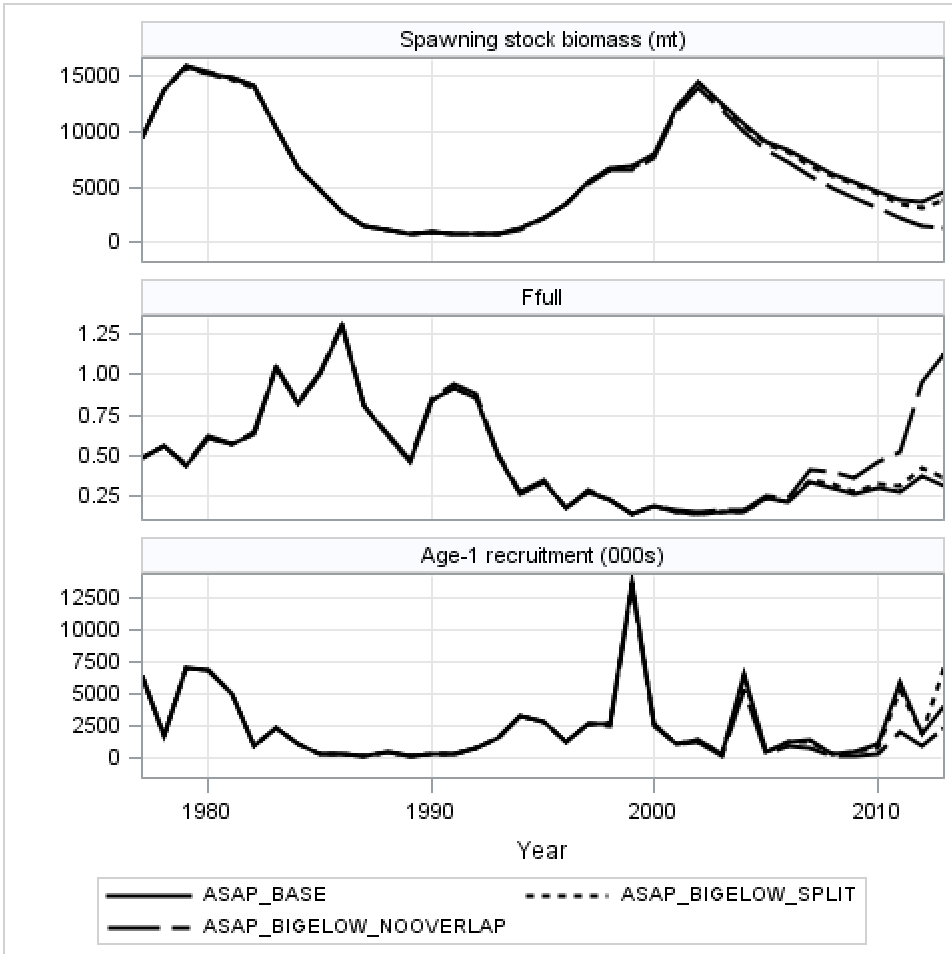


Figure A.2.3. Comparison of model results for two model variants of the Gulf of Maine haddock ASAP_BASE model. Sensitivity models were fit to both the Albatross/Bigelow-calibrated series and a separate un-calibrated Bigelow series (ASAP_BIGELOW_SPLIT), and separate Albatross (1977-2008) and Bigelow (2009-2013) non-overlapping series (ASAP_BIGELOW_NOOVERLAP).

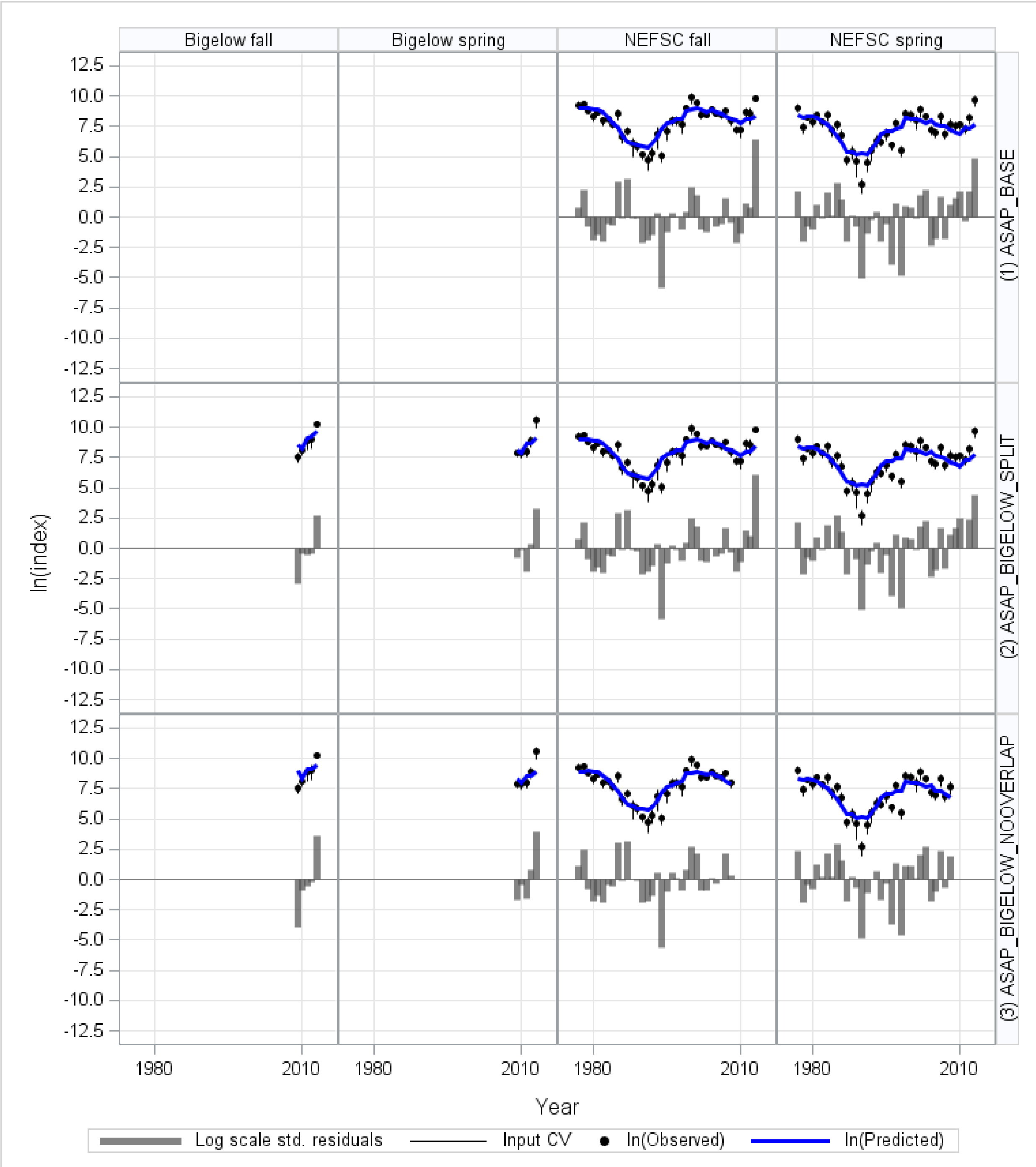


Figure A.2.4. Comparison of model fits to survey indices for variants of the ASAP_BASE model fit to (1) a single Albatross/Bigelow-calibrated series (fall and spring), (2) both the Albatross/Bigelow-converted series and a separate un-calibrated Bigelow series, and (3) separate Albatross (1977-2008) and Bigelow (2009-2013) non-overlapping series.

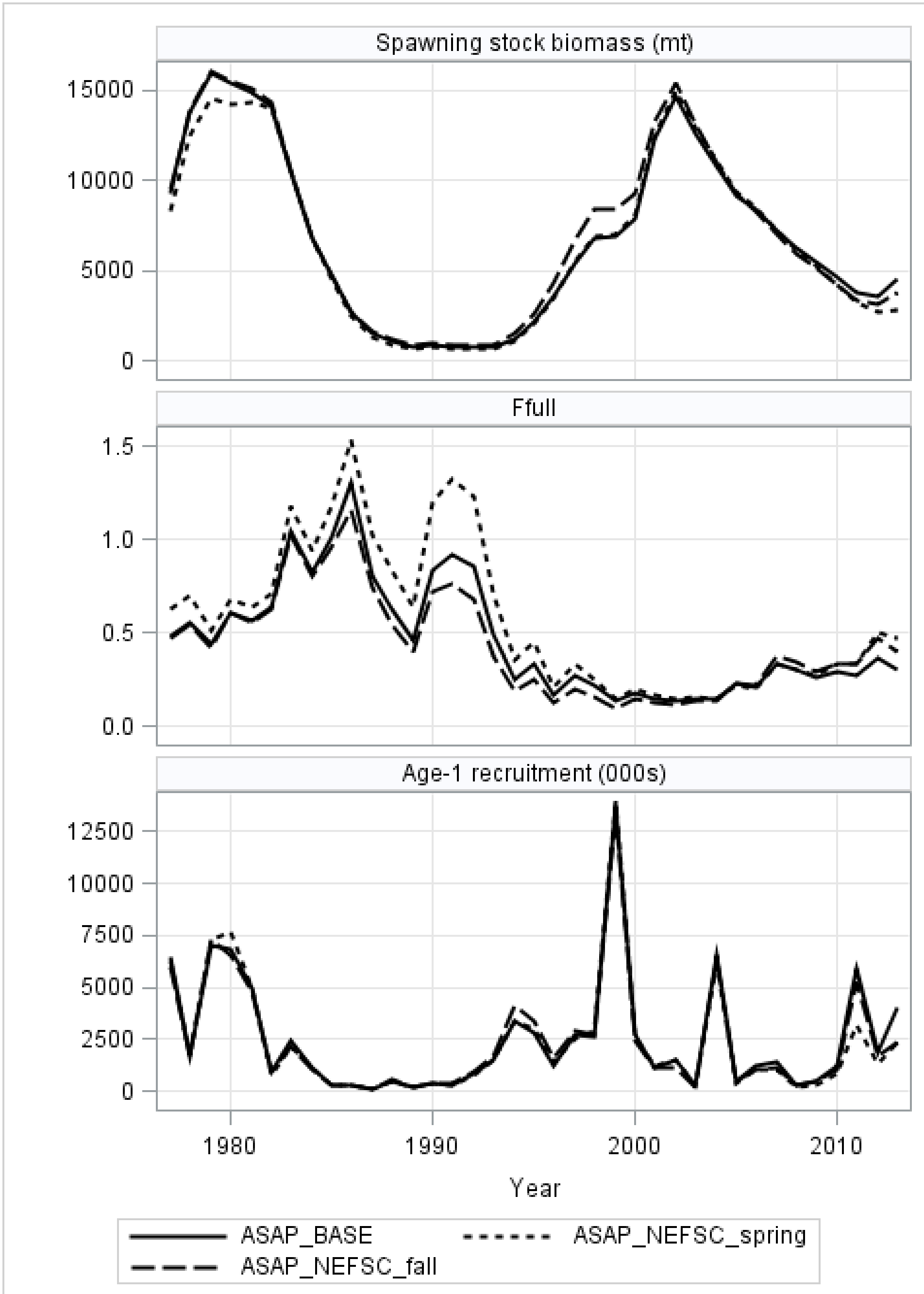


Figure A.2.5. Model results from the ASAP_BASE model compared to the results when run using only the NEFSC spring or NEFSC fall survey.

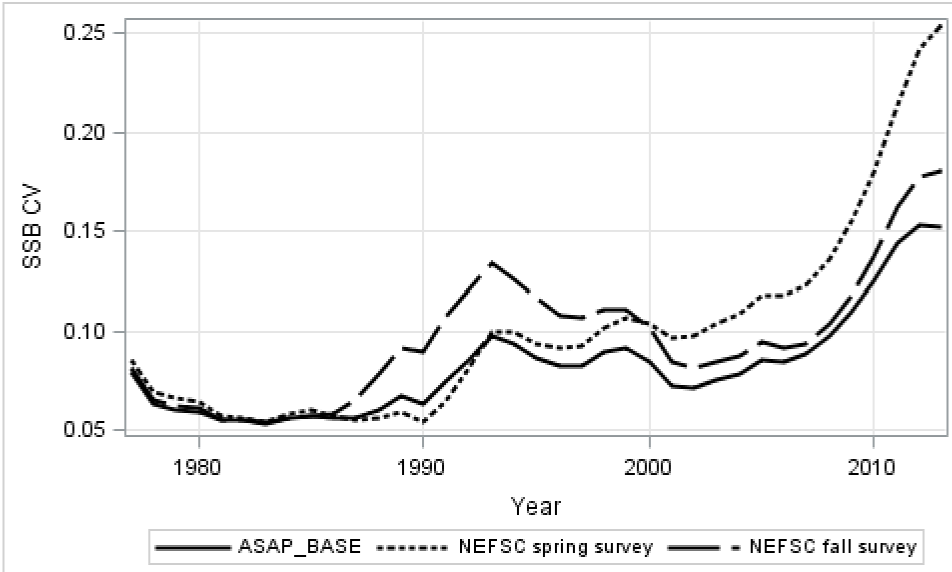


Figure A.2.6. Comparison of the coefficient of variation (CV) in spawning stock biomass (SSB) estimates between the ASAP_BASE model and the model run using only the NEFSC spring or NEFSC fall survey.

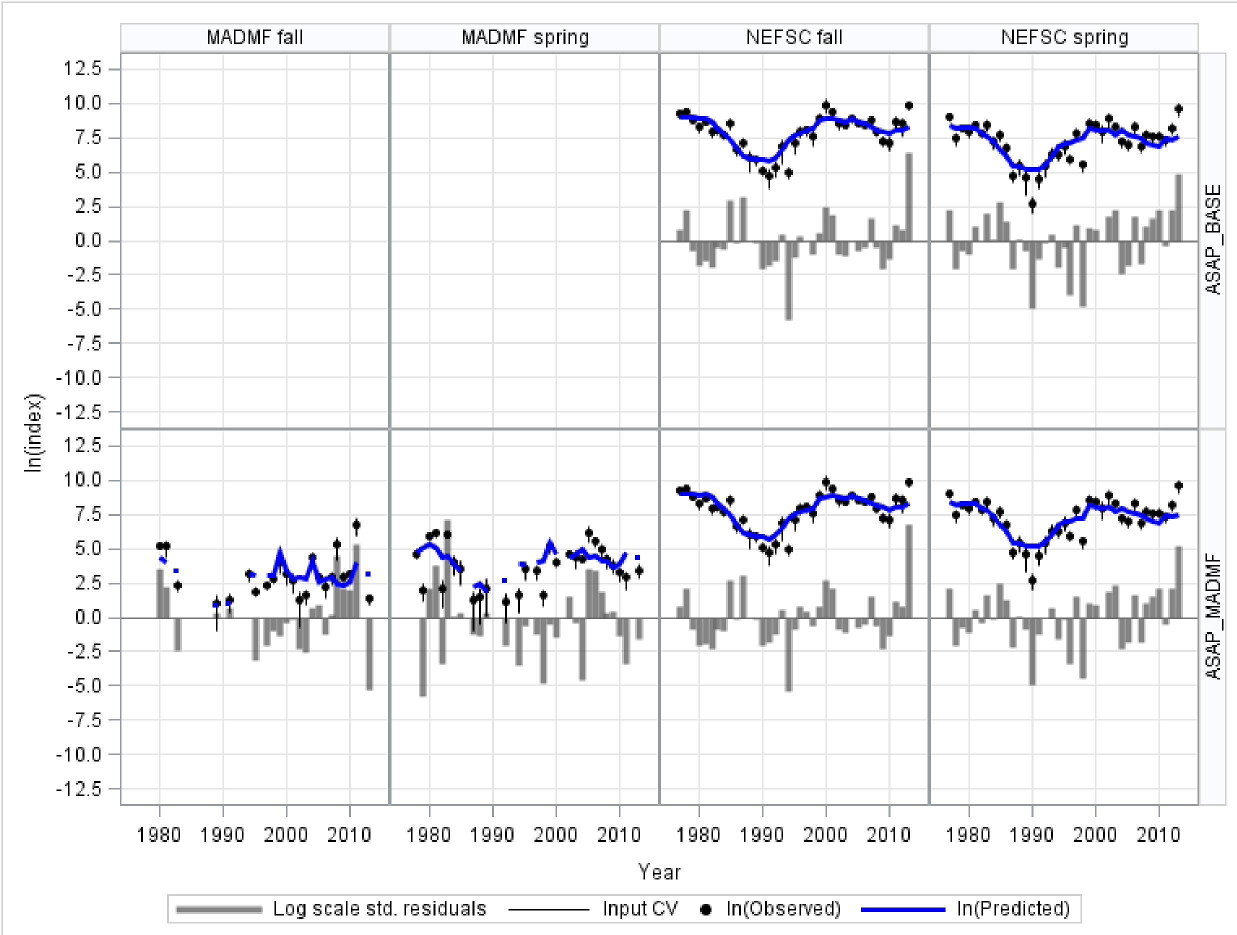


Figure A.2.7. Comparison of model fits to survey indices for both the base ASAP model (ASAP_BASE) model and the base ASAP model with Massachusetts Department of Marine Fisheries (MADMF) spring and fall survey indices.

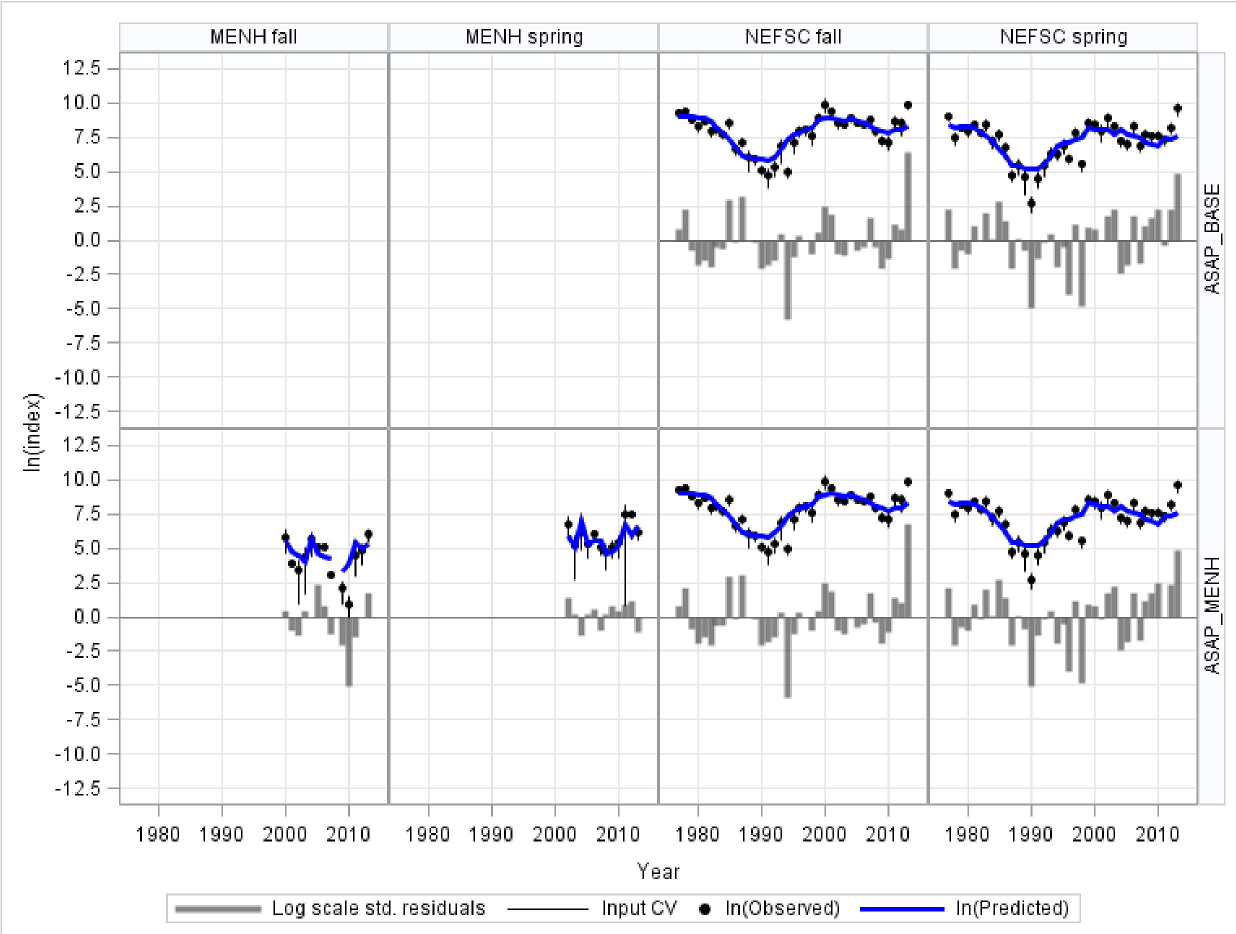


Figure A.2.8. Comparison of model fits to survey indices for both the base ASAP model (ASAP_BASE) model and the base ASAP model with Maine-New Hampshire (MENH) inshore spring and fall survey indices.

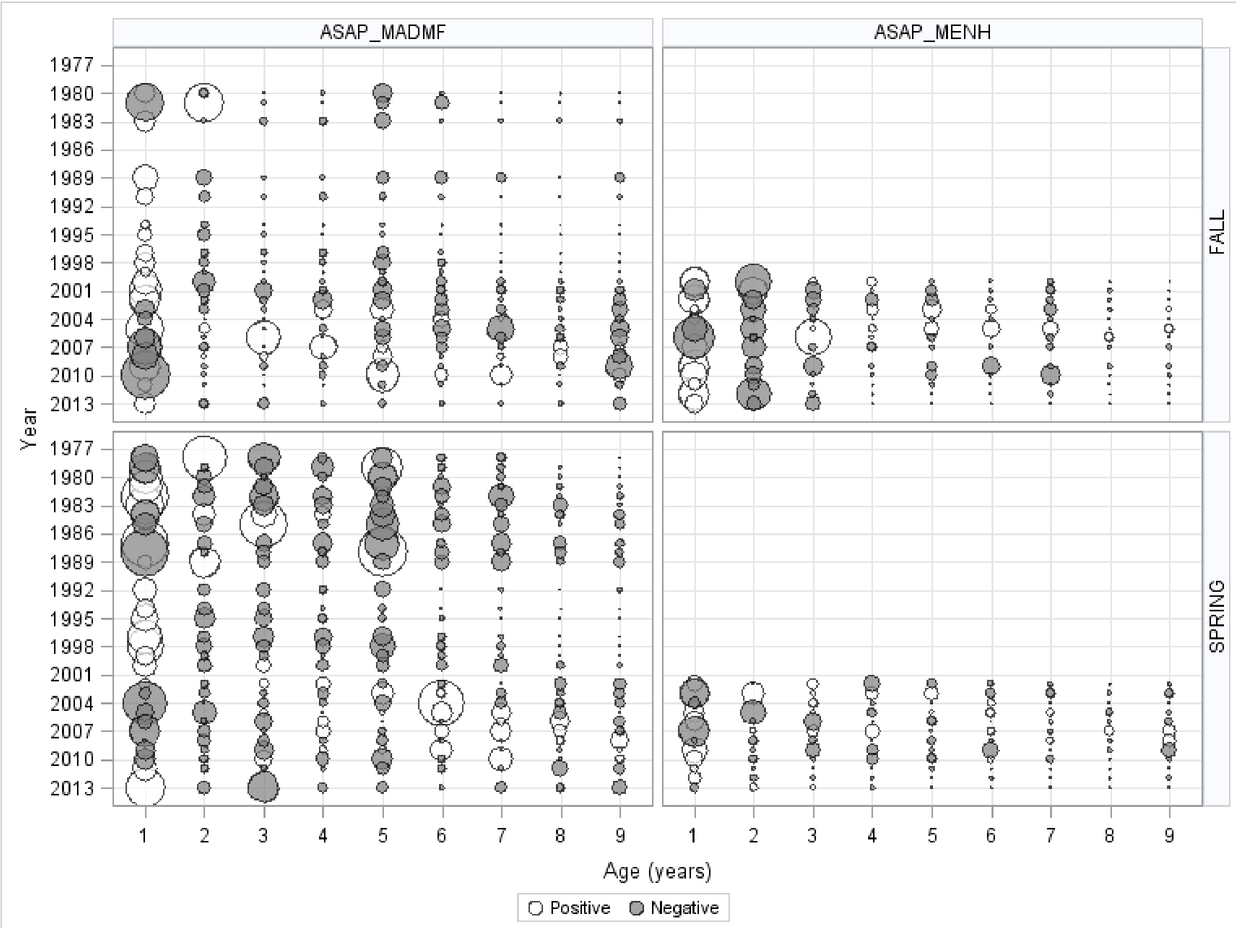


Figure A.2.9. Residual plots of the Gulf of Maine haddock ASAP model fits to the Massachusetts Department of Marine Fisheries (ASAP_MADMF) and Maine-New Hampshire inshore (ASAP_MENH) indices-at-age.

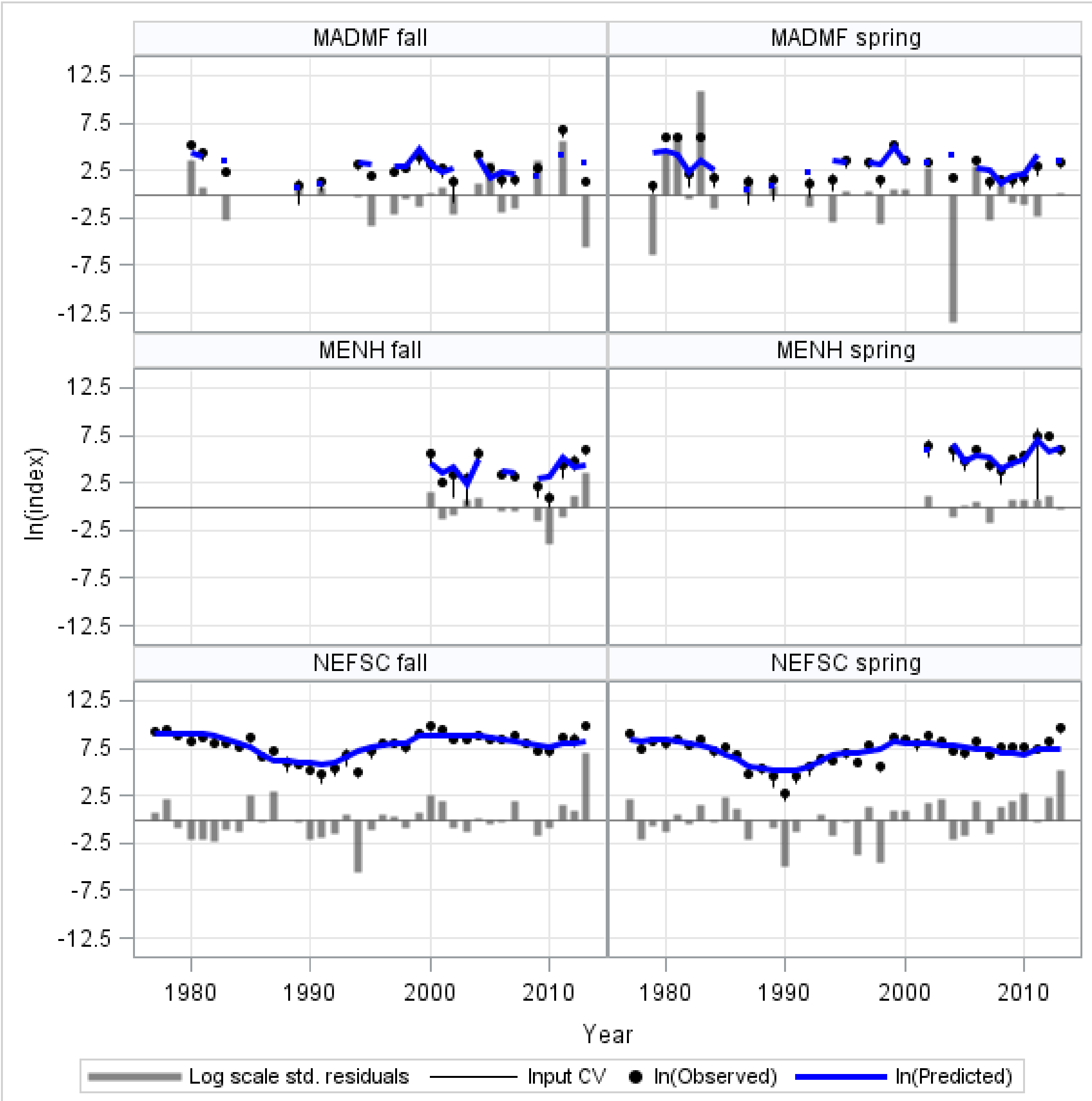


Figure A.2.10. Model fits to survey indices from a run of the base ASAP model which included both the Massachusetts Department of Marine Fisheries (MADMF) and Maine-New Hampshire (MENH) inshore spring and fall survey indices.

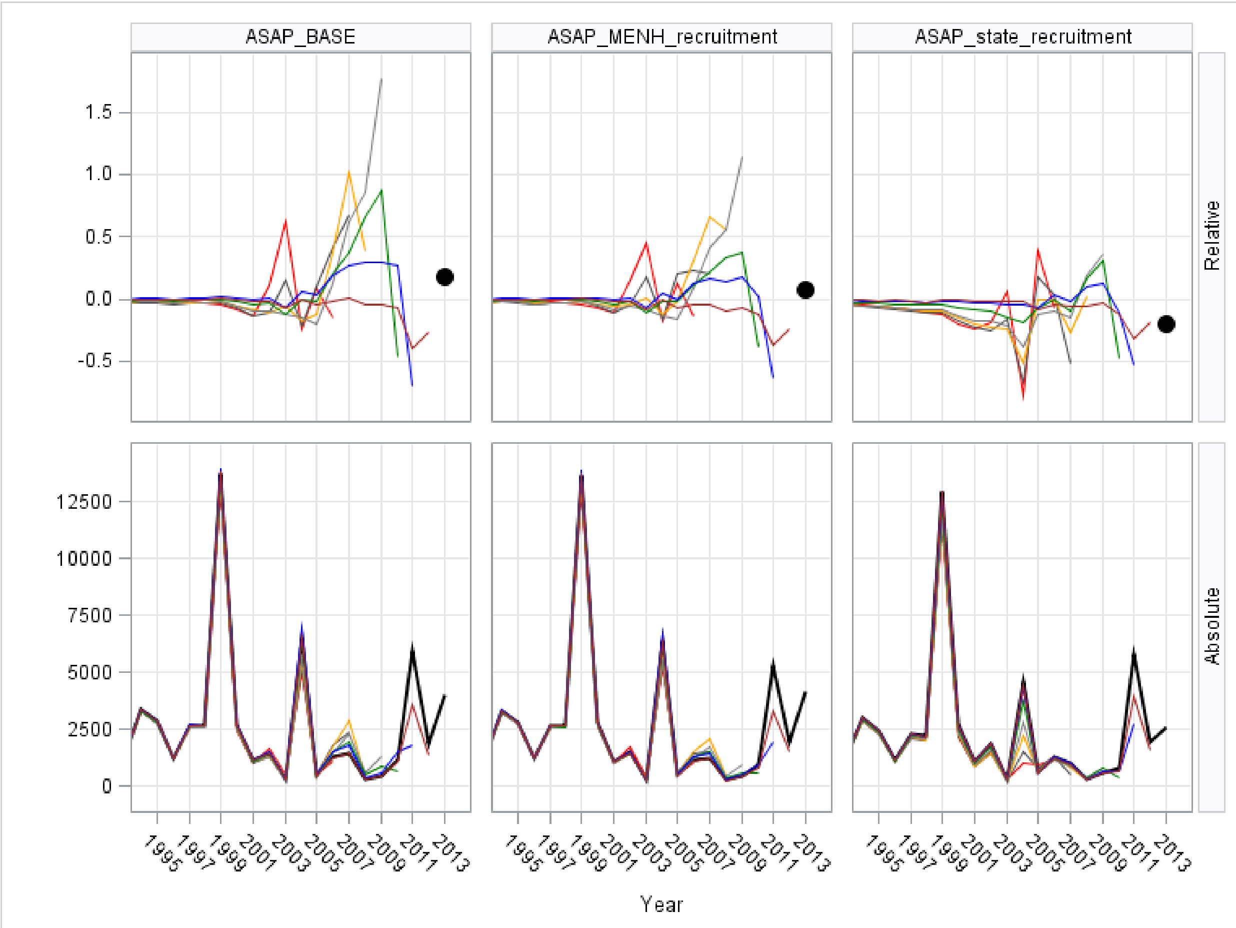


Figure A.2.11. Retrospective plots for age-1 recruitment from the ASAP_BASE, ASAP_MENH_recruitment and ASAP_state_recruitment models. Plots are shown on both the relative difference (top) and absolute (bottom) scales. The average relative difference over a 7-year peel (Mohn's rho) is indicated by the black circle in the top plot.

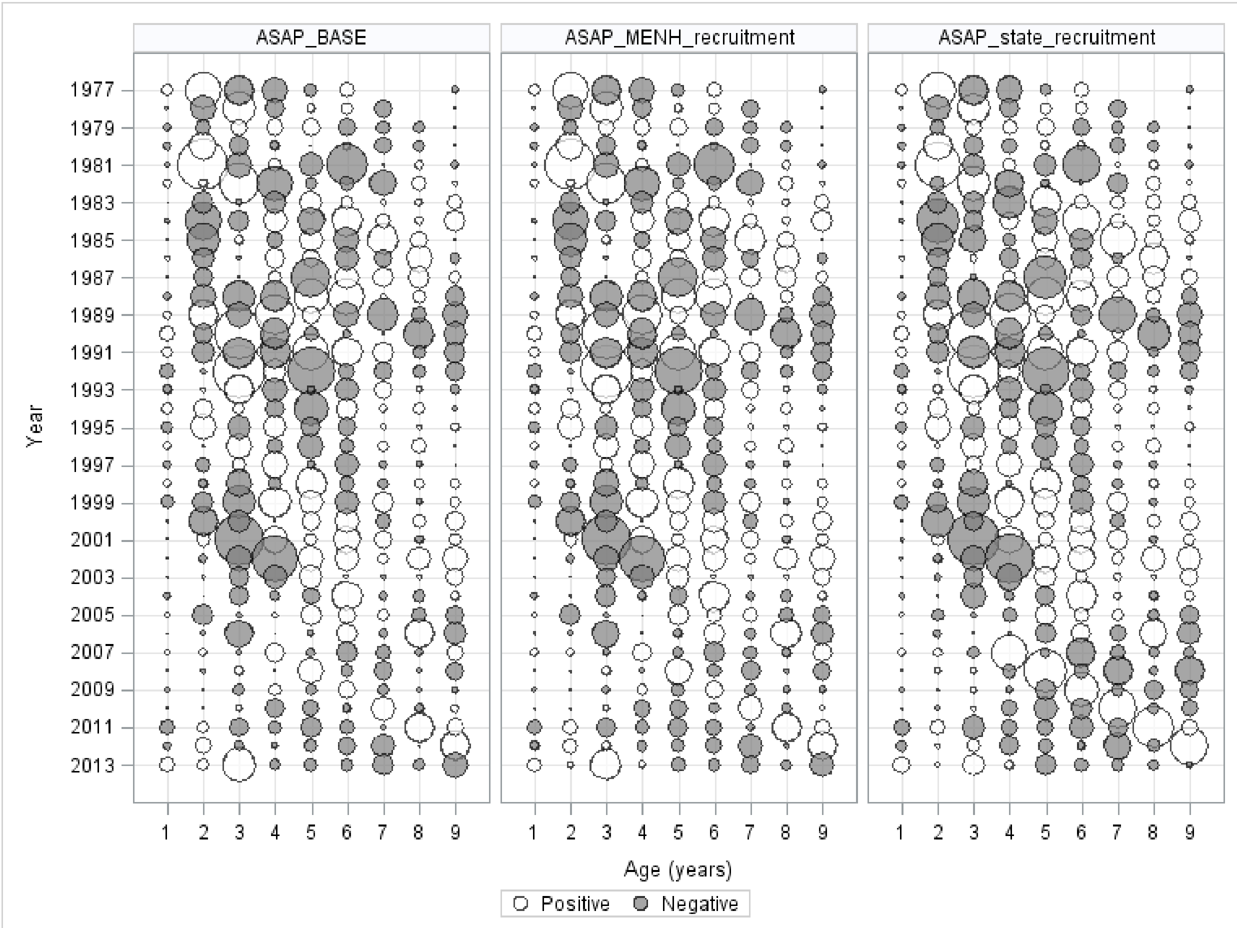


Figure A.2.12. Residual plots of the Gulf of Maine haddock catch-at-age fits compared between the ASAP_BASE model and the model tuned with age-1 spring and fall survey indices from the Maine-New Hampshire inshore survey (ASAP_MENH_recruitment) and both Maine-New Hampshire and Massachusetts Department of Marine Fisheries (ASAP_state_recruitment).

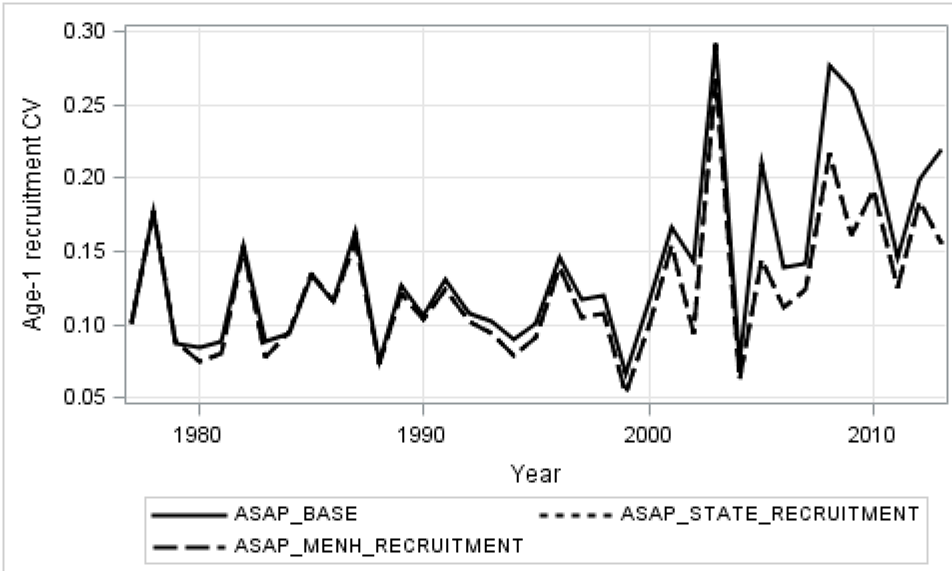


Figure A.2.13. Comparison of the coefficient of variation (CV) age-1 recruitment estimates between the ASAP_BASE model and the model runs incorporating age-1 spring and fall survey indices from the Maine-New Hampshire inshore survey (ASAP_MENH_recruitment) and both Maine-New Hampshire and Massachusetts Department of Marine Fisheries (ASAP_state_recruitment).

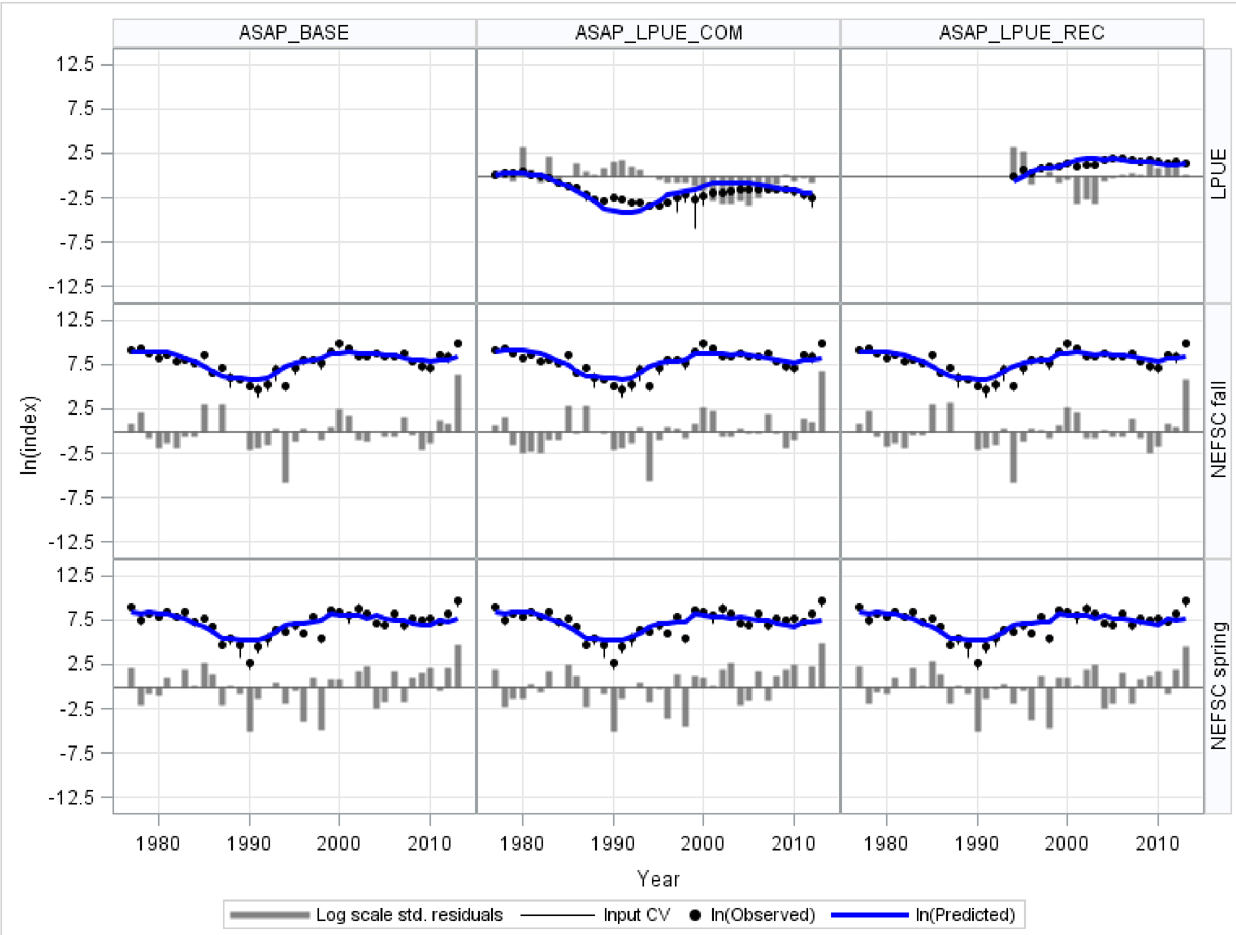


Figure A.2.14. Comparison of model fits to the NEFSC spring and fall survey indices and landings-per-unit-effort (LPUE) indices from both the commercial and recreational fishery. The ASAP_BASE model is compared to an equivalent model run incorporating either the commercial (ASAP_LPUE_COM) or recreational (ASAP_LPUE_REC) LPUE index.

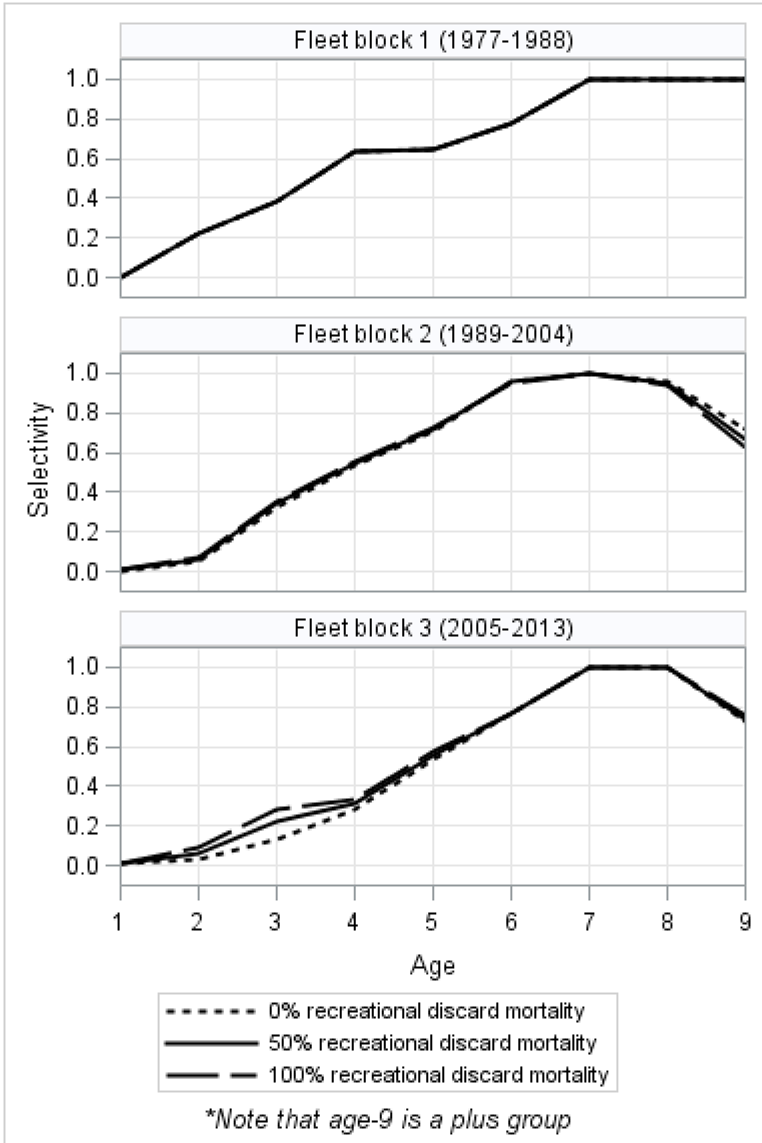


Figure A.2.15. Estimated fleet selectivities under a range of discard mortality sections for each of the fleet selectivity blocks.

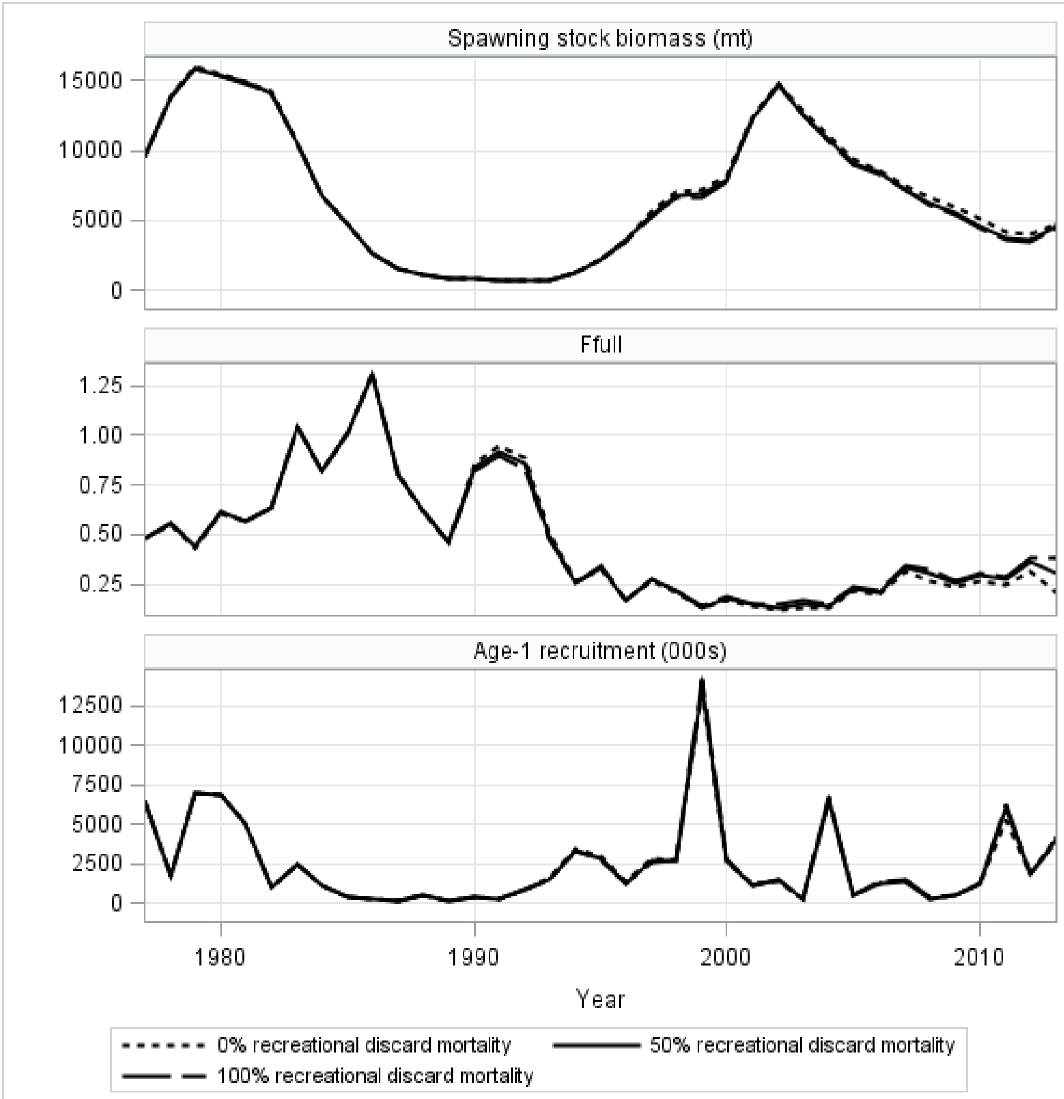


Figure A.2.16. Model results from the ASAP_BASE model under a range of discard mortality sections.

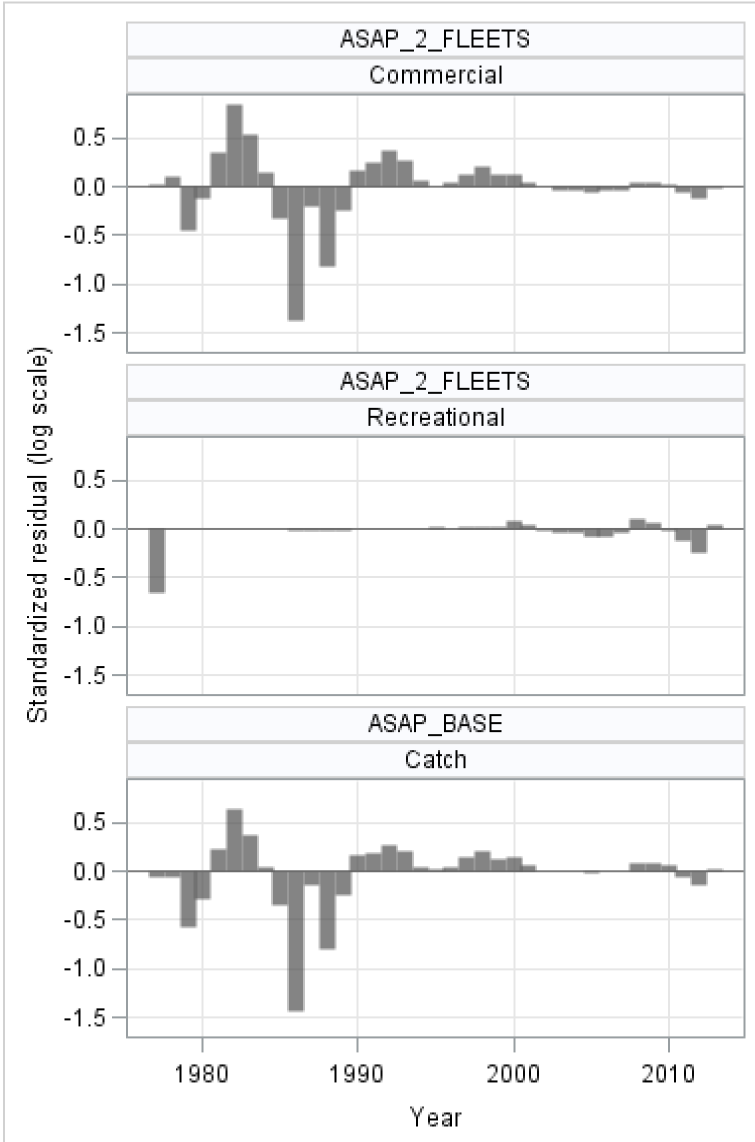


Figure A.2.17. Comparison of model fits to catch time series for both the ASAP_BASE model which includes an aggregated commercial and recreational catch and the ASAP_2_FLEET model which treats commercial and recreational catch separately.

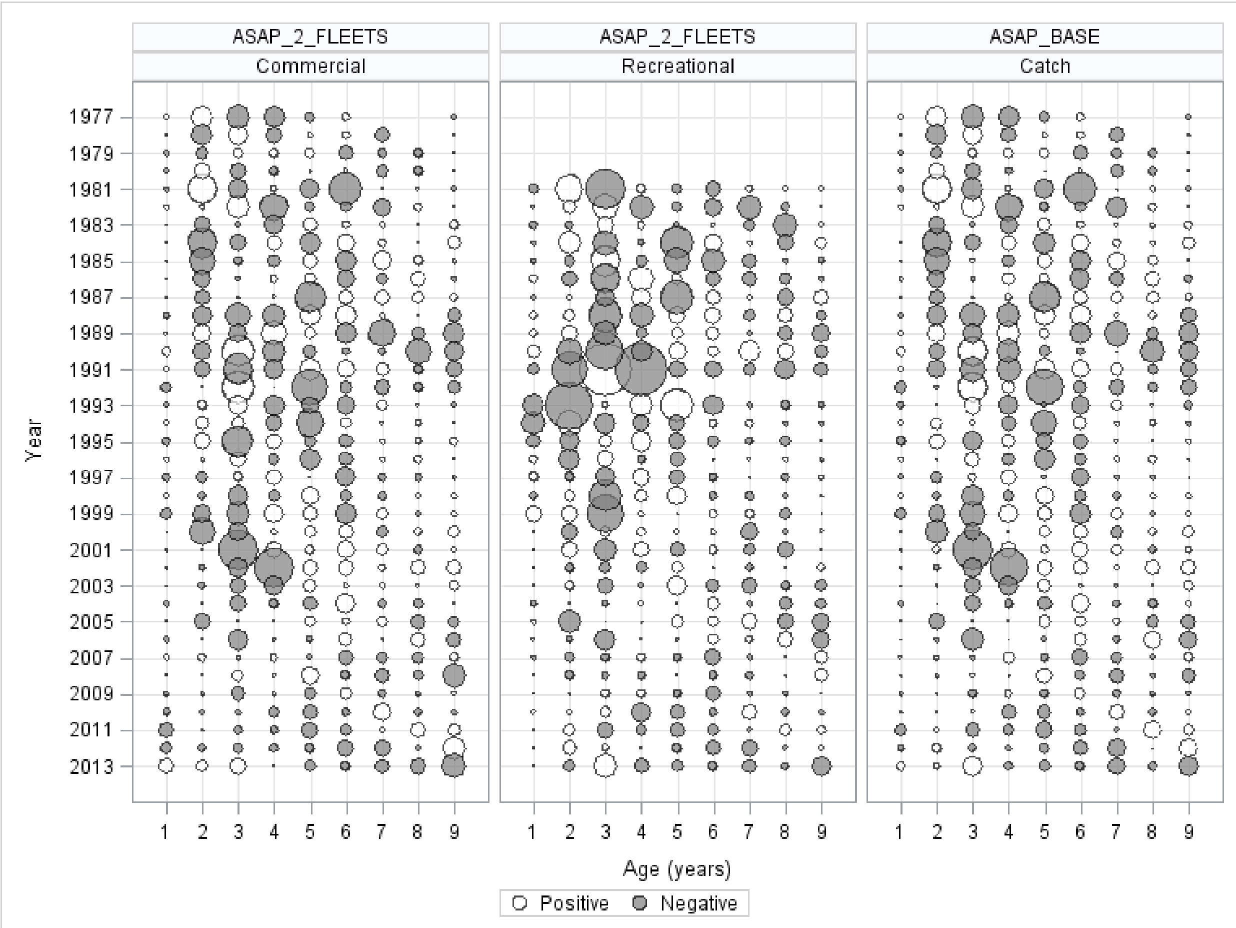


Figure A.2.18. Residual plots of the Gulf of Maine Atlantic cod catch-at-age fits compared between the ASAP_BASE model and the ASAP_2_FLEET model.

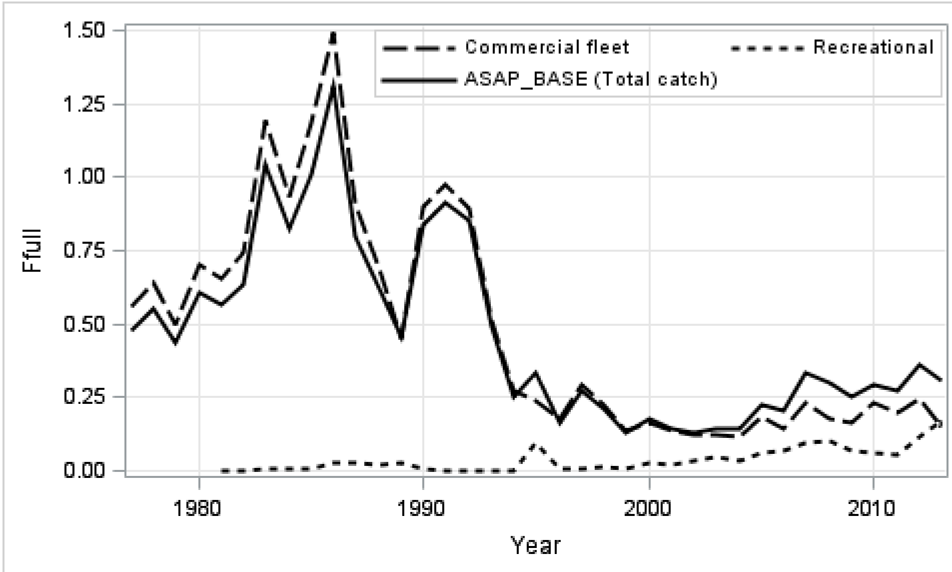


Figure A.2.19. Plot of the fully recruited fishing mortality from ASAP_BASE model and the dis-aggregated commercial and recreational fishing mortality from the ASAP_2_FLEET model.

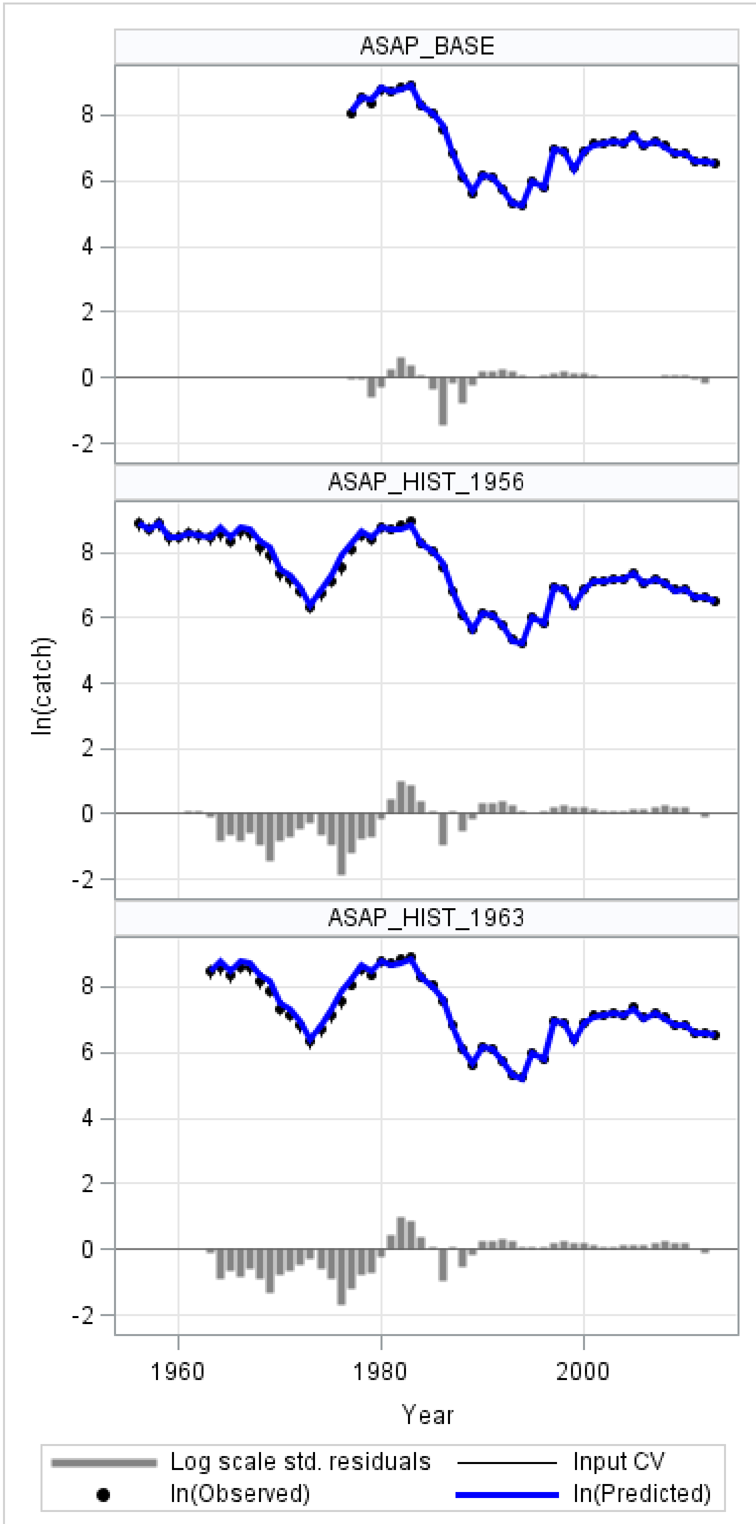


Figure A.2.20. Comparison of model fits to fishery catches from three variants of the base ASAP model with different starting years: ASAP_BASE (1977), ASAP_HIST_1956 and ASAP_HIST_1963.

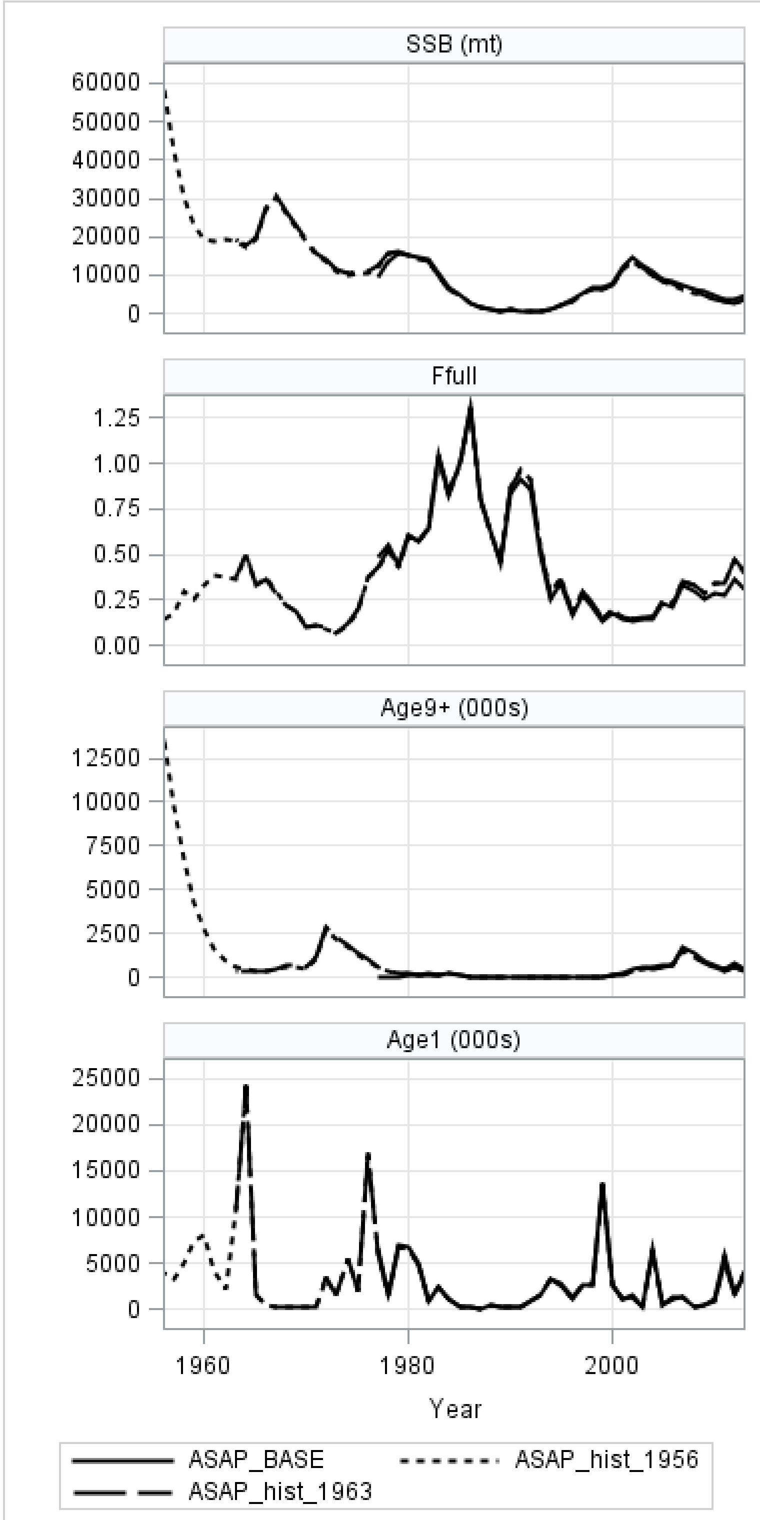


Figure A.2.21. Comparison of the Gulf of Maine haddock assessment results from models using different starting years. All models are based on the ASAP_BASE model which starts in 1977. The ASAP_HIST_1956 and ASAP_HIST_1963 models started in 1956 and 1963, respectively.

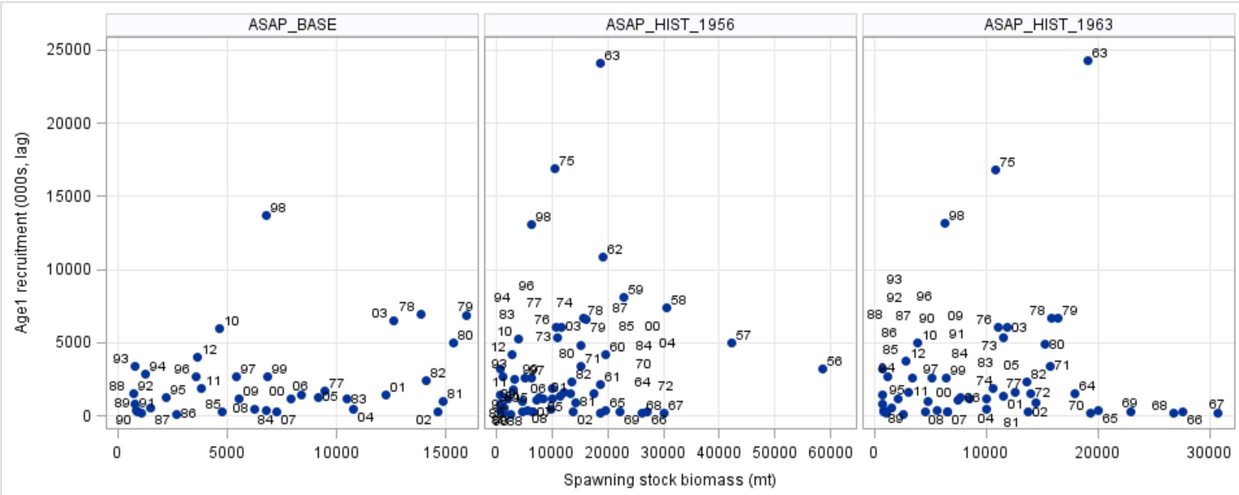


Figure A.2.22. Scatter plots of Gulf of Maine haddock age-1 recruits vs. spawning stock biomass from the ASAP_BASE, ASAP _HIST_1956, and ASAP _HIST_1963 ASAP models. The starting year for each of the models was 1977, 1956 and 1963 respectively. The data labels indicate the spawning year of the individual year classes.

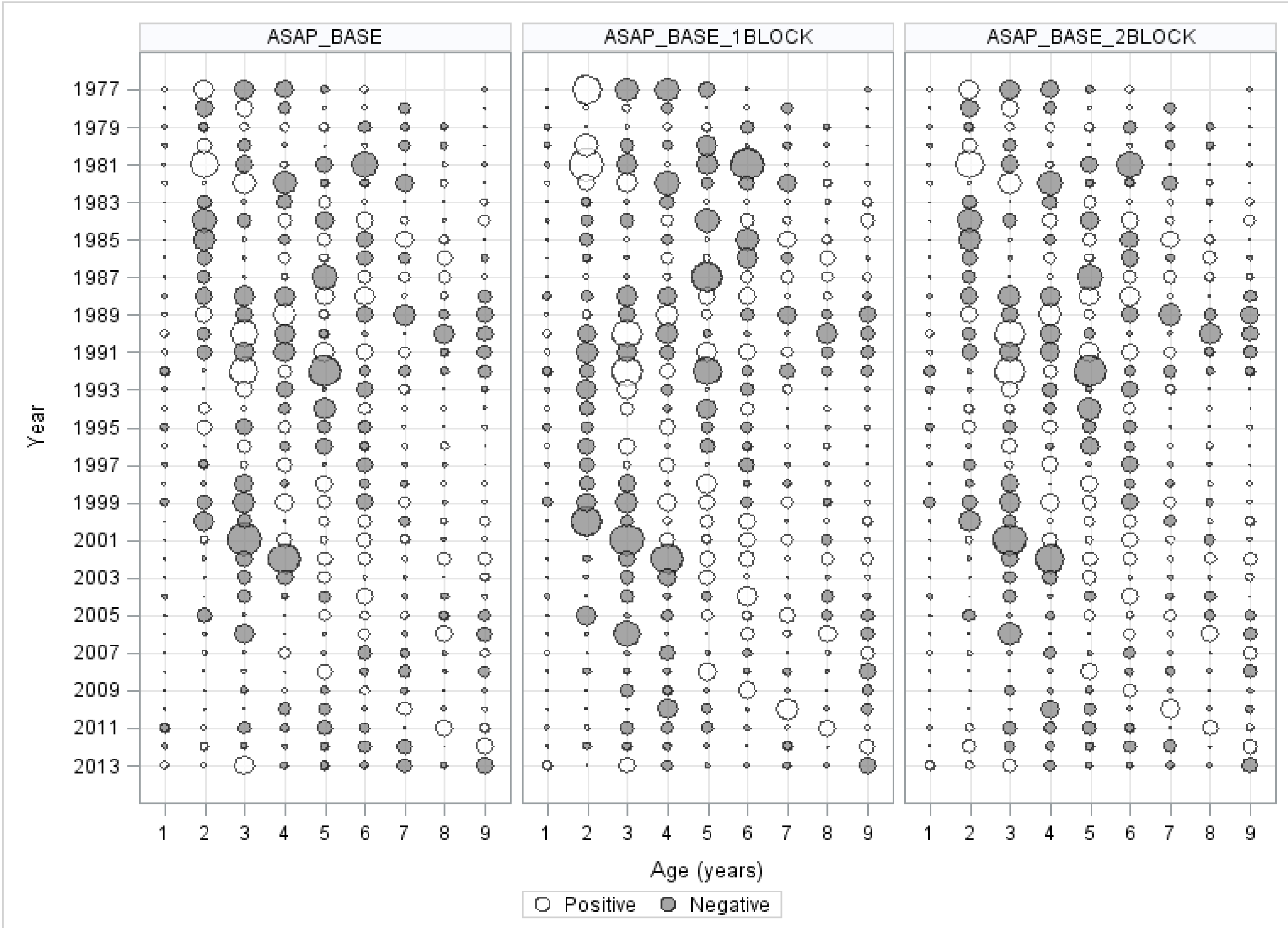


Figure A.2.23. Residual plots of the Gulf of Maine haddock catch-at-age fits compared between the ASAP_BASE model and one- and two-selectivity block sensitivity runs.

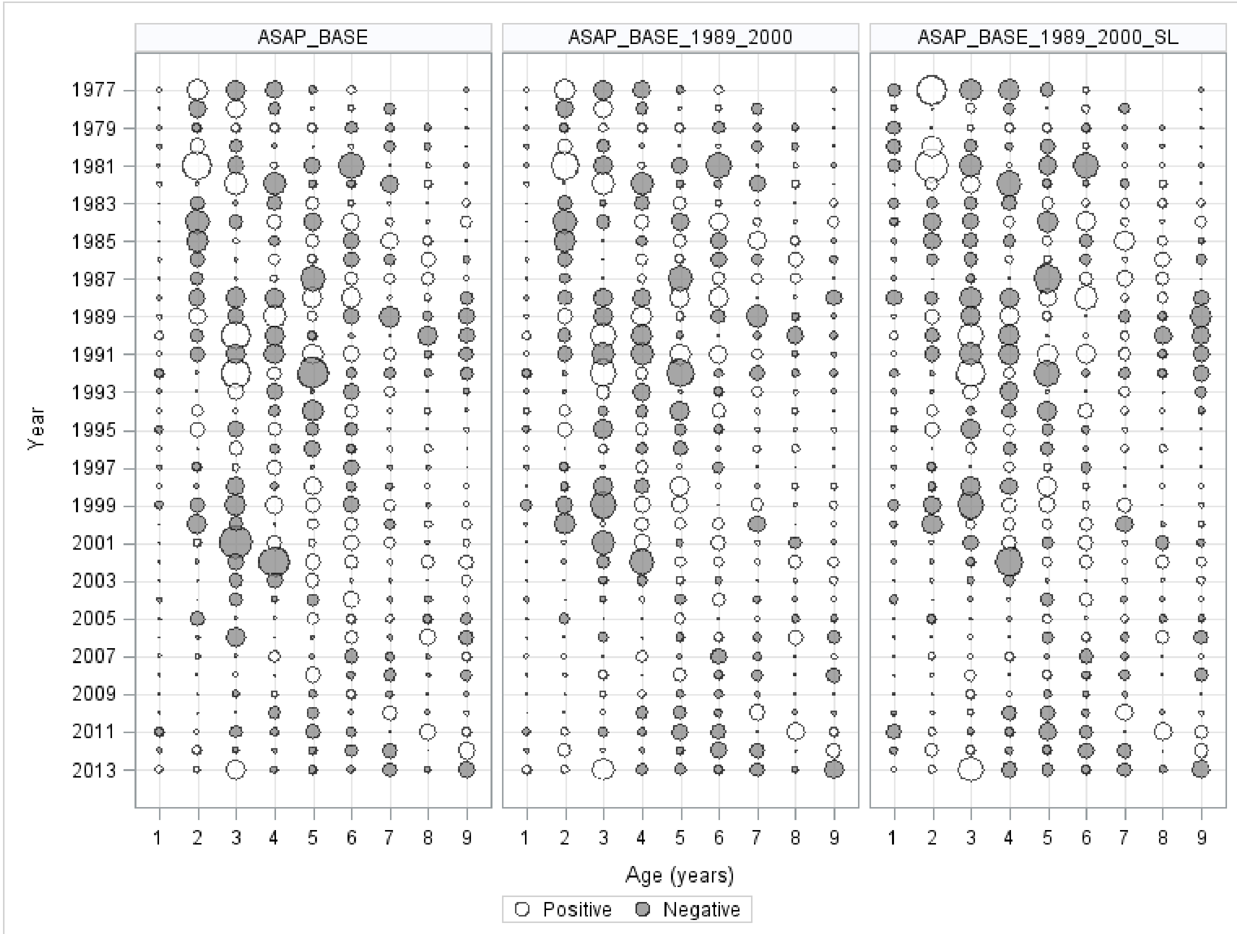


Figure A.2.24. Residual plots of the Gulf of Maine haddock catch-at-age fits compared between the ASAP_BASE model and variants of a similar three-block model exploring a 1999/2000 split between the second and third selectivity block (ASAP_BASE_1989_2000). The ASAP_BASE model employs a 2004/2005 split.

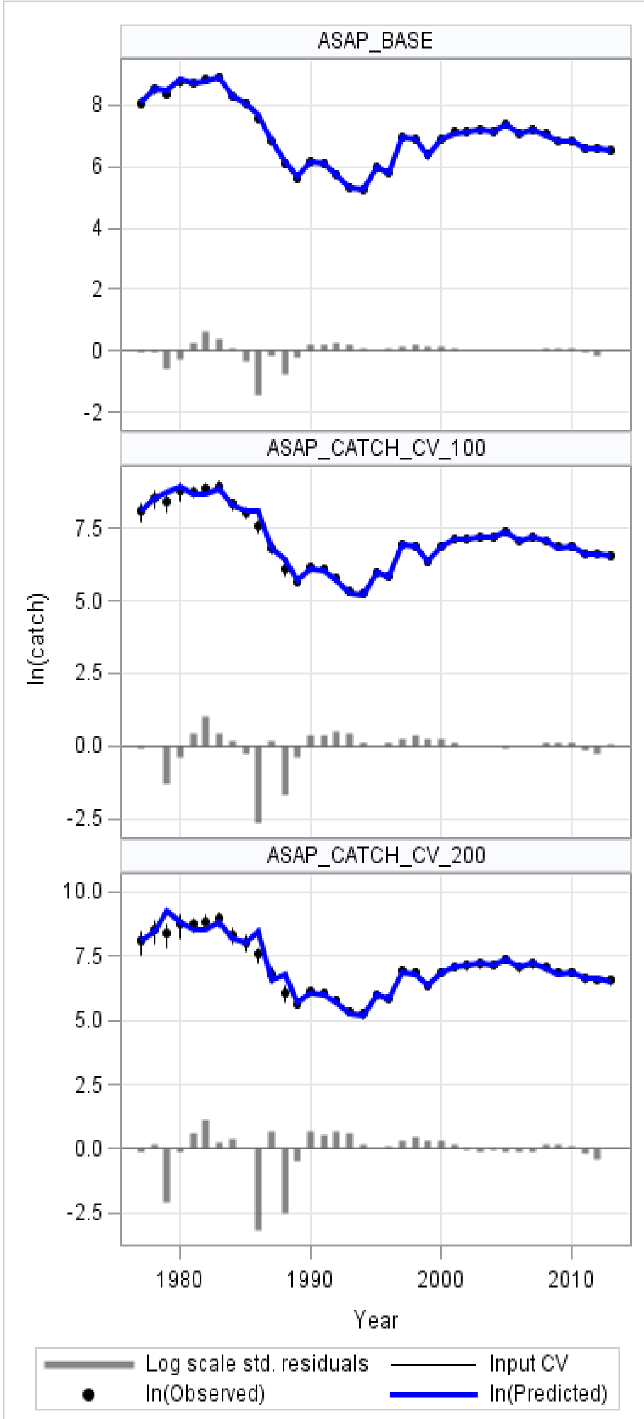


Figure A.2.25. Comparison of model fits to fishery catches from three variants of the base ASAP model with varying levels of assumed precision on fishery catches.

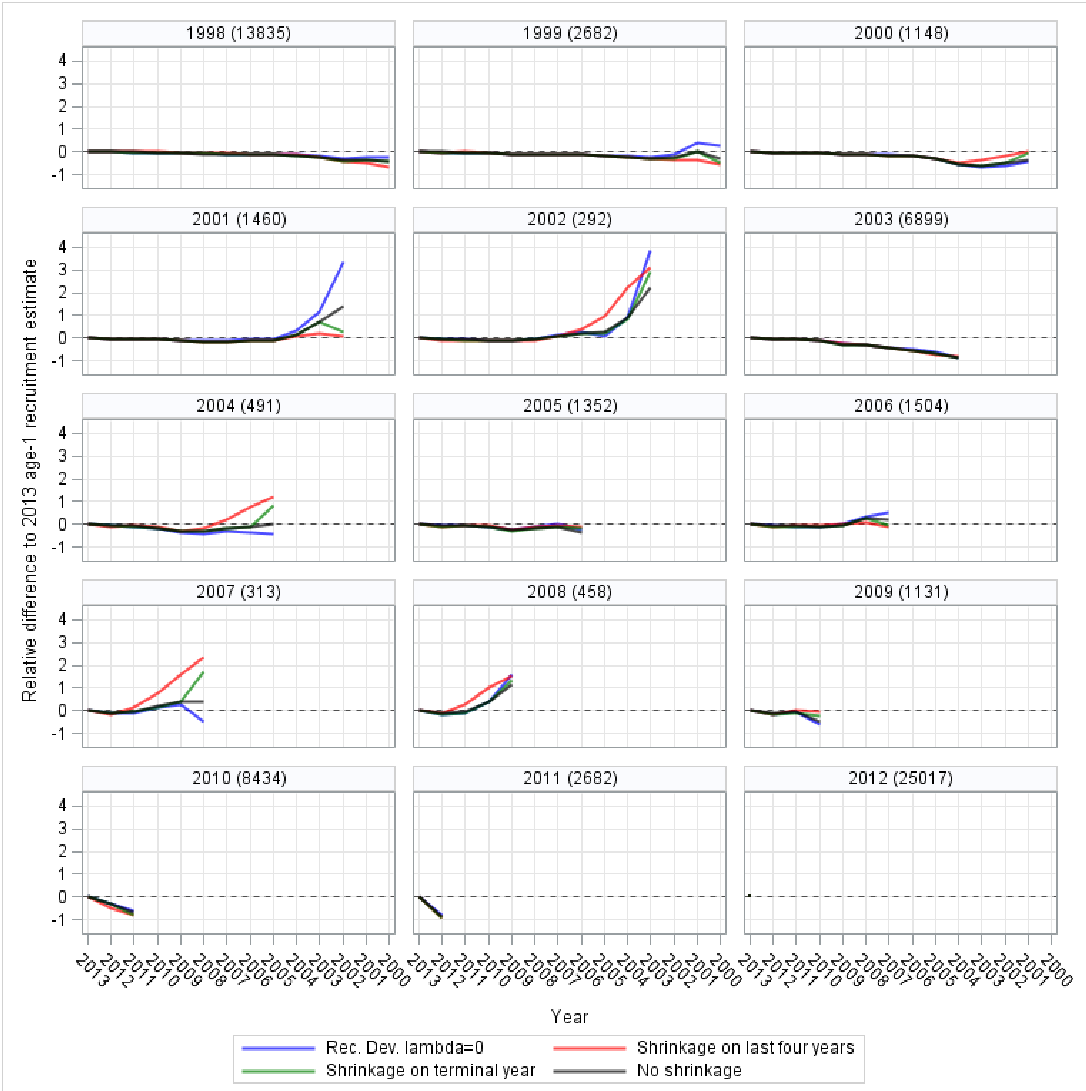


Figure A.2.26. Retrospective estimates of year class size using four different methods to model recruitment deviations. The 2013 estimate of year class size based on the 'No shrinkage' model are indicated in parentheses next to the year class identifier.

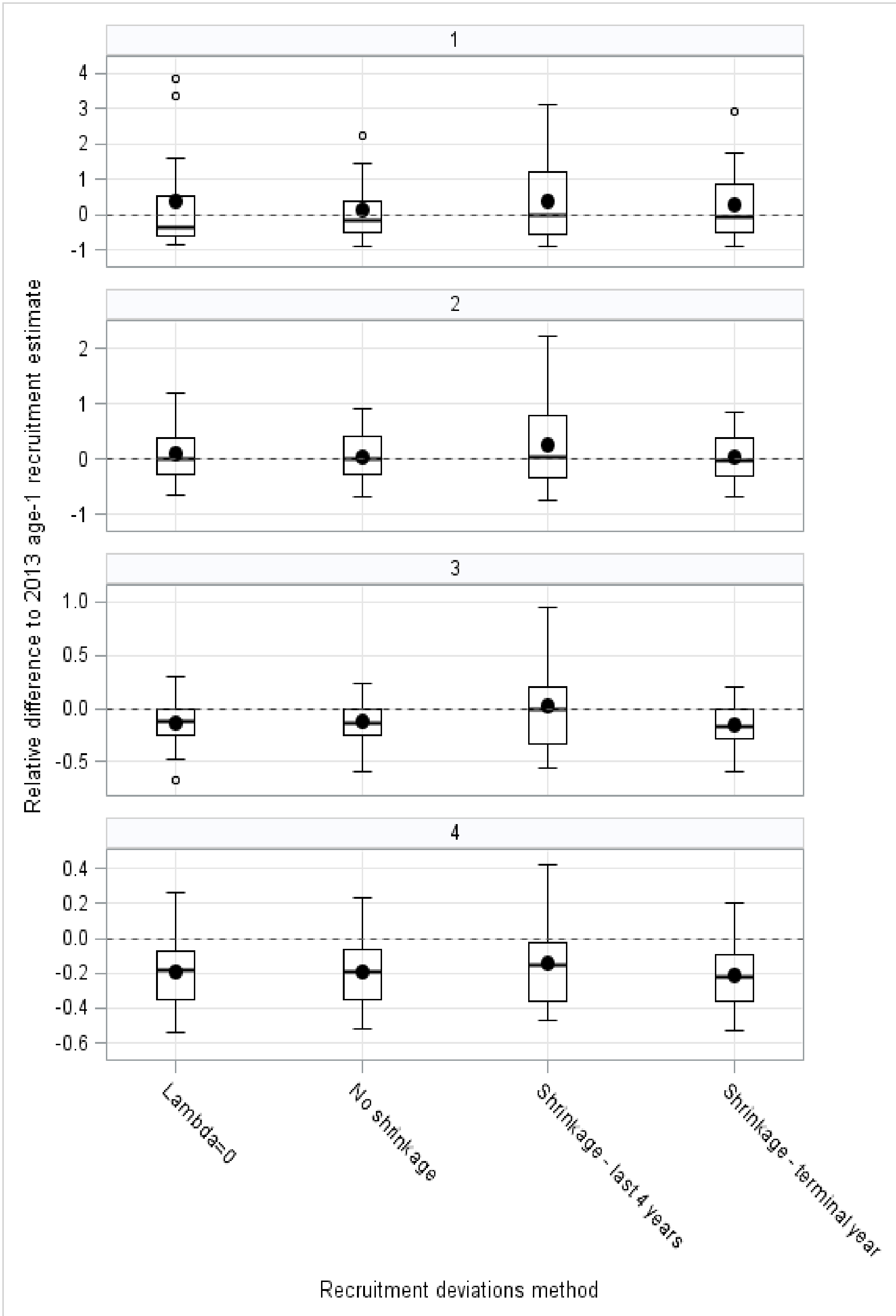


Figure A.2.27. Distribution of the retrospective errors from four different methods to model recruitment deviations. Panels represent the number of years from the initial spawning event (e.g., the ‘4’ panel reflects the distribution of errors four years from the spawning event). Note that the scale varies between panels.

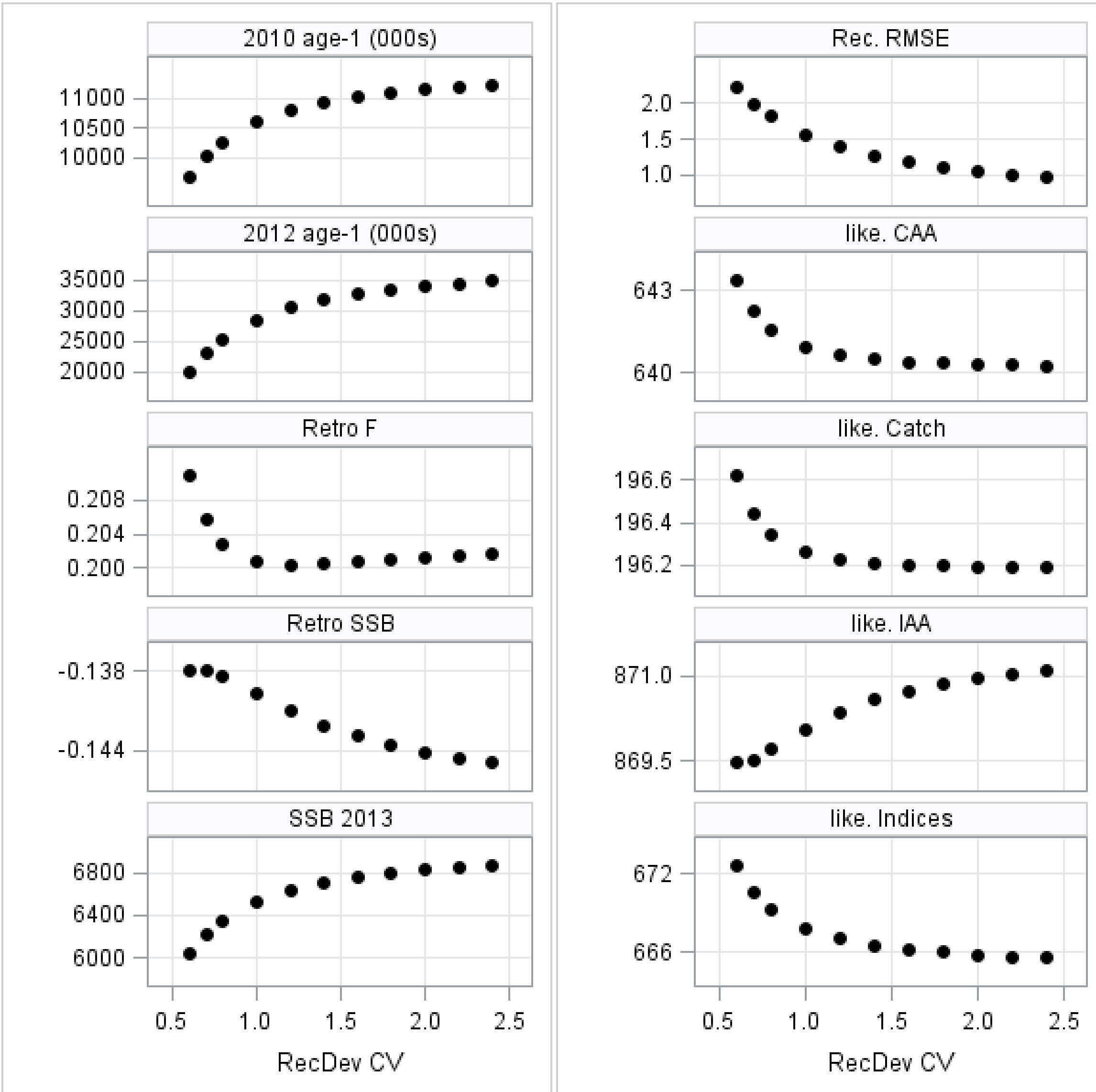


Figure A.2.28. Model response to profiling over a range of recruitment deviation coefficient of variation (CV) values.

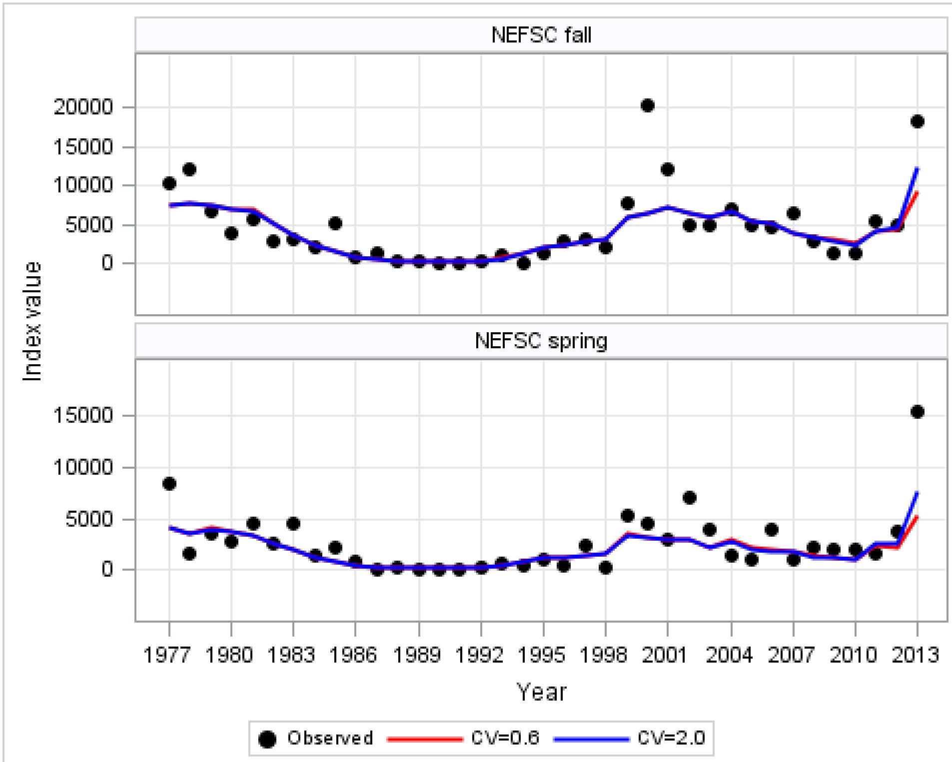


Figure A.2.29. Example of model fits to the NEFSC spring and fall bottom trawl surveys when the recruitment deviation coefficient of variation (CV) is set at 0.6 and 2.0.

Appendix A.3. Gulf of Maine haddock SCAA exploratory model runs

Assessments of the Gulf of Maine haddock[1]

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June 2014

Summary

The Gulf of Maine haddock population is assessed in three ways. The first considers the haddock in this area to be an isolated stock. The other two incorporate movement into that area, either permanent or temporary, by haddock from Georges Bank. The evidence for such movement from these analyses alone is sufficient to point to scenarios involving limited movement being of similar plausibility to that of an isolated stock. Catch projections under constant fishing mortality are found to be sensitive, in particular, to this possibility of limited movement for the case where the movement is permanent. Assessment results for the most recent recruitment are sensitive to the procedure used to shrink this estimate to the mean.

Introduction

This paper presents results for three approaches to the assessment of the Gulf of Maine (GoM) haddock stock, all of which use SCAA methodology (see, e.g., Butterworth and Rademeyer, 2011). The first approach explores assessment options when the stock is treated as isolated. The second allows for interchanges in the form of permanent migration from (and to) the neighbouring Georges Bank (GB) haddock population. The third approach (known in the IWC Scientific Committee as the "sabbatical model") also allows for interchanges, but these are not of a permanent nature. Some GB haddock may visit the GoM area during a year, and perhaps be caught there; however if not suffering mortality of some form, they return later that same year to the GB area.

The paper first summarises the data used, and then details the methodologies applied for the isolated stock and interchange models, followed by the assumptions made for calculating four-year catch projections. The results of applying these methodologies, together with some sensitivity tests, are then discussed, followed by some concluding remarks.

1 This paper is a revision of an earlier version presented to the SAW meeting held at the NEFSC, Woods Hole over 2-5 June, 2014. Here Base Case run assumptions have been made to maximize comparability with the preferred ASAP model described in the main text of the report.

Data

The catch and survey based data together with some biological data for the GoM haddock population were kindly provided by Michael Palmer, and are listed in Tables in Annex A.

The second and third assessment approaches, which take interchange (movement) into account, utilise estimates of annual numbers-at-age from the most recent GB haddock assessment for the period from 1977 to 2011 (NEFSC, 2012). These values are listed in Table A8 of Annex A. This Table includes projections to 2017 kindly provided by Liz Brooks; the basis for the computation of these projections is detailed in Table A8's caption.

Methodology

The details of the basic SCAA assessment methodology are provided in Annex B.

Isolated stock

In the interests of maximal comparability with preferred ASAP model of the main text of the report, the following methodological options were chosen/implemented for this Base Case SCAA run (SCAA BC1).

- The stock recruitment curve was assumed to be constant with log-normally distributed residuals. The contribution to the negative log-likelihood from these residuals was calculated assuming a residual CV of 1 (this correspond to a $\sigma_{\mathrm{R,y}}$ value of 0.833, which is roughly comparable, though slightly below, the level of variation shown in assessment outputs).
- Selectivities-at-age for the fishery and survey series were estimated separately for each age, though the survey selectivities were set flat above certain ages (see Annex B, section B.4.1, for further details). These decisions were AIC-justified.

Some other choices amongst the standard SCAA options that were made were as follows.

- The multinomial-mimicking "*sqrt(p)*" formulation of the proportions-at-age contribution to the overall negative log-likelihood (Butterworth and Rademeyer, 2012) was used, rather than the "adjusted log-normal", as the former deals more naturally with the relatively large numbers of zeros in the catch-at-age matrices for this stock.
- These proportion-at-age contributions to the negative log likelihood were fully weighted (W_{CAA}=1 – see equation B14), as is broadly comparable to the approach used to set effective sample sizes for the preferred ASAP model. The variance of the associated residuals was estimated assuming age-independence.

The authors' base case choices for implementing the SCAA, differ from those of the preferred ASAP model in one respect.

- The numbers at age vector for the starting year was estimated only to age 3, and thereafter the procedure of equations B9 and B10 of Annex B used (AIC justified).

In addition certain sensitivity runs were pursued:

- An alternative lower value of 0.5 for the recruitment CV for 2013, corresponding to setting $\sigma_{\mathrm{R,y}}$ for 2013 to 0.472, was considered to stabilise this estimation to a greater extent. Note that the rightmost term in equation B18 of Annex B includes years to 2010 only, so that changing "weights" in this way on the last year's recruitment does not directly impact the estimate of the geometric mean recruitment R_{gm}.

- The standard deviation of the (transformed) proportion-at-age residuals (σ_{CAA} – see equation B16 of Annex B) for each series was estimated separately for each age rather than for all ages combined.
- The contribution of the proportion-at-age data to the negative log likelihood was down weighted (W_{CAA}=0.5 – see equation B14), to show the effect of possible non-independence of these data.
- The fishing "fleet" was disaggregated into commercial landings, commercial discards, and recreational landings together with recreational discards.

Migration model

There is evidence of interchange between the GoM and GB haddock stocks (e.g. Begg, 1998), but unfortunately the tagging exercises conducted to date have not been designed in a way that allows annual interchange proportions to be estimated reliably. However, since (for example) survey results would have included GB haddock which had moved into the GoM area, it is possible to extend the assessment to take this into account. Normally this would require assessing both stocks simultaneously, but an advantage in this case is that the GB stock is assessed to be so much larger than the GoM stock. This enables the results from the GB stock assessment (NEFSC, 2012) (kindly projected into the future by Liz Brooks, see Table A8 of Annex A) to be used directly, since unlike for the GoM haddock, those GB results would hardly change in such a joint assessment.

In the case of permanent interchange (i.e. migration) between the GoM and GB haddock stocks, equations B1 and B2 of Annex B are replaced by the following equations:

$$N_{y+1,a+1}^{GoM} = \left(N_{y,a}^{GoM} + \mu\rho_y N_{y,a}^{GB} - \lambda N_{y,a}^{GoM}\right)e^{-Z_{y,a}} \qquad \text{for } 1 \le a \le m-2 \qquad (1)$$

$$N_{y+1,m} = \left(N_{y,m-1}^{GoM} + \mu\rho_y N_{y,m-1}^{GB} - \lambda N_{y,m-1}^{GoM}\right)e^{-Z_{y,m-1}} + \left(N_{y,m}^{GoM} + \mu\rho_y N_{y,m}^{GB} - \lambda N_{y,m}^{GoM}\right)e^{-Z_{y,m}} \qquad (2)$$

where

μ is the proportion of the GB haddock (above a critical level) migrating annually and permanently to the Gulf of Maine, with a value estimated when fitting the model,

λ is the proportion of GoM haddock migrating annually and permanently to Georges Bank,

$$\rho_y = \frac{\sum_a N_{y,a}^{GB} - GB_{crit}}{\sum_a N_{y,a}^{GB}} \qquad (3)$$

and

GB_{crit} is the level of 2+ GB haddock numbers below which no GB haddock are assumed to immigrate into the GoM (i.e. the GB stock has to be "large" for any such migration to take place). For all the runs except one sensitivity, $GB_{crit} = 0$. For this sensitivity, $GB_{crit} = 47559$, which is half of the 1977-2013 average of the numbers of 2+ fish, so that movement occurs about 50% of the time over the this period.

The lower bound for age a in equation (1) is adjusted to correspond to the lowest age at which interchange takes place. This is taken to be a=2 for the Base Case implementation, based on indications to this effect provided in NEFC (1986).

Sabbatical model

Under the sabbatical model for movement, each year a proportion (θ) of the GB haddock "visit" the GoM area each year and mix with the GoM haddock (and hence are assumed to be available for capture in this area, and to be amongst the haddock monitored by the two NEFSC surveys each year). The GoM catches of haddock are taken from GoM and GB haddock in proportion to their relative abundances by age in the GoM area. Hence the fishing mortality F_y applies to both the GoM haddock stock and to the GB haddock "visitors". The total predicted catch C_y^* is computed as:

$$C_y^* = \sum_{a=1}^{m} w_{y,a}^{\text{mid}} N_{y,a}^* S_{y,a} F_y \left(1 - e^{-Z_{y,a}}\right) / Z_{y,a} \qquad (4)$$

where

$$N_{y,a}^* = \left(N_{y,a}^{GoM} + \theta N_{y,a}^{GB}\right) \qquad (5)$$

$$Z_{y,a} = F_y S_{y,a} + M_a \qquad (6)$$

and the θ term in equation (5) (where the value of θ is estimated when fitting the model) applies only to ages for which movement is assumed to occur (a = 2+ for the Base Case, as for the migration model).

Spawning biomass (equation B5) is computed using the GoM haddock numbers only ($N_{y,a}$), while predicted survey indices (equation B7) and catches-at-age (B17) are computed with the GoM + GB visitors numbers ($N_{y,a}^*$), i.e. equations B7 and B17 are replaced by:

$$N_y^{\text{surv}} = \sum_{a=1}^{m} S_a^{\text{surv}} N_{y,a}^* e^{-Z_{y,a} T^{surv}/12} \qquad (7)$$

$$\hat{p}_{y,a}^{surv} = S_a^{\text{surv}} N_{y,a}^* e^{-Z_{y,a} T^{surv}/12} \Big/ \sum_{a'=0}^{m} S_{a'}^{\text{surv}} N_{y,a}^* e^{-Z_{y,a} T^{surv}/12} \qquad (8)$$

Projections

Four-year projections have been run under constant fishing mortalities of F_{MSY}, where F_{MSY} is taken to be $F_{40\%}$, as estimated in this paper or as estimated for the preferred ASAP model (see Annex B, section B.4.3). For these projections, the following assumptions have been made:

- the weight-at-age and commercial selectivity vectors are taken as the 2009-2013 average, as assumed for the $F_{40\%}$ computations for the preferred ASAP model - see Annex B section B.4.3;
- future recruitments are taken to be constant at their arithmetic mean level over the period chosen for the preferred ASAP model, i.e. 1977 to 2011 (to avoid inclusion of the recruitments for 2012 and 2013 for which the estimates have high variance); and
- in the cases with interchange (permanent migration) between the GoM and GB haddock stocks, the future GB haddock stock and age-structure is projected over the 2012 to 2017 period on the basis detailed in the caption to Table A8 of Annex A.

Results and Discussion

<u>Isolated stock</u>
Comparisons of the results from the preferred ASAP model SCAA Base Case without movement (SCAA-BC1) are provided in Table 1 and Figure 1, and evidence little difference. This SCAA-Base Case exhibits a reasonable fit to the survey indices of abundance and proportion-at-age data for both the fishery and the surveys, and indicates a slightly higher current spawning biomass than the preferred ASAP model does.

Table 1 and Figure 2 show the consequences of reducing the value assumed for the variability ($\sigma_{R,y}$) of the recruitment for the most recent years (2013) to a CV of 0.5 compared to the SCAA-Base Case choice of 1.0. This has a major impact on the estimate of recruitment for the last year, which drops by more than 50%, but the estimate of spawning biomass for 2013 falls only slightly. Formally the choice of 1.0 (corresponding, roughly, to the variability shown by past recruitment) is the most appropriate statistically for the shrinkage to the mean of the estimates that would otherwise result. However this leads to a high variance associated with the 2013 recruitment estimate. A case could be made that a lower choice than 1.0 is appropriate in the interests of providing more robust estimates, but the difficulty is in choosing what value it would be best to set in any such down-weighting parameter. Results are also shown for downweighting the contribution of the proportions-at-age data to the negative log likelihood. This has little impact on estimates, though the confidence intervals shown in Table 1 widen slightly, and those for parameters such as selectivity-at-age somewhat more so.

Allowing for the variance parameter associated with the proportions-at-age residuals (σ_{CAA}) to be estimated separately by age improves the fit to the data, and to an extent which is AIC justified (Table 1 and Figure 3). However, as the impact of allowing for this effect on key results (such as those for spawning biomass) is minimal, this adjustment was not incorporated into the SCAA Base Case to maintain greater comparability with the preferred ASAP model. Similarly, initial attempts to disaggregate the fishing "fleet" into commercial landing, discards and recreational components also led to little difference in such results, and hence was not explored further.

<u>Migration model</u>
Results for the permanent migration model are shown in Table 4 for the best estimate of 0.2% (SCAA BC2) for the annual proportion μ moving from the GB to the GoM area. Results for a range of μ values are shown in Figure 4. These indicate that the estimate of μ = 0.2% is significantly different from zero at the 10% (and 5%) levels, while the diagnostics shown in Figure 5 evidence a satisfactory fit to the data. For μ = 0.2%, the recent spawning biomass estimates are not greatly affected; they do become appreciably larger for higher values of μ, but those results are not compatible with the data. If movement is allowed in the reverse direction as well (i.e. the λ parameter is set to be different from zero), results are hardly affected (see Figure 6), so that λ has been kept at zero for all subsequent results for this model.

Table 4 and Figures 7 to 9 provide results for some sensitivities to SCAA BC2. Changing the age at which fish can move from the GB to the GoM area from 2+ to either 1+ or 3+ impacts results, but only to small extents. The consequences of allowing random annual variation (with a CV = 1.0) about a mean proportional movement of 0.2%, and of precluding movement below an abundance threshold for GB haddock, are also relatively small.

Sabbatical model

Results for the sabbatical model (non-permanent interchange) are given in Table 3 and Figure 10, again indicating reasonable fits to the data. The best estimate of the proportion of GB haddock moving temporarily each year to the GoM area, θ, is 0.75% (SCAA BC3). This is shown to be statistically different from zero at the 10% (and 5%) levels in Figure 11.

Figure 11 also shows spawning biomass trajectories for various values of θ for the component of the haddock in the GoM area belonging to the true GoM stock. Unsurprisingly, this is less for larger values of θ, as those reflect greater proportions of the catch from the GoM area being comprised of GB haddock. In addition, the Figure shows how this proportion has changed over time for the different values of θ. Table 3 and Figure 12 show that changing the age at which fish can move from the GB to the GoM area from 2+ to either 1+ or 3+ has some though not a substantial impact on results.

Overall comparison and Retrospectives

Figure 13 compares the results for the Base Case for no movement model (SCAA BC1) with those for the two models which allow for movement (SCAA-BC2 for permanent and SCAA-BC3 for temporary migration). The first two sets of results are fairly similar, but the sabbatical model (SCAA-BC3) unsurprisingly shows lower spawning biomass and recruitment values as these plots do not include the haddock "visiting" the GoM area from the GB stock, even though those haddock contribute to catches made in the GoM area.

Figure 14 shows retrospective plots for all three models. None reflect serious systematic trends. The estimates of the movement parameters μ and θ are stable and consistently significantly different from zero. Examination of the negative log likelihood contributions in Tables 2 and 3 shows that it is the proportions at age data that provide the key information to allow the values of these parameters to be estimated. These negative log likelihoods also indicate a preference for the migration over the sabbatical model, but not to any substantial extent; indeed from a biological perspective, one might tend to consider the sabbatical model as the more plausible of the two.

Catch projections

Four year catch projections under F_{MSY} are shown in Table 4. For the sabbatical model scenarios, results given reflect the total catch allowed, and this will include a component of GB haddock. The figures in parentheses in Table 4 show the part of this that comes from the "true" GoM haddock stock only. The F_{MSY} values are provided by the $F_{40\%}$ proxy, though this is calculated in two ways: first for the SCAA model estimates (and specific to the model in question with or without movement), and then for the preferred ASAP model (see Annex B, section B.4.3 for details).

These projection results are quite similar for the no movement and sabbatical models, but give values some 20-40% higher for the permanent migration model.

Concluding Remarks

The results above for catch projections (in particular) from these assessment model variants for the GoM haddock stock point to two key factors to which model outputs are particularly sensitive. These are:

1) the extent to which the estimate of recruitment for the most recent year is shrunk to the mean; and

2) how the possibility of exchanges with GB haddock is best to be taken into account; the estimates of annual movement proportions, although small in percentage terms, are statistically significant at the 5% level so that the associated exchange hypotheses

are plausible; furthermore in the case of permanent exchange, catch projections under F_{MSY} proxies are increased by amounts in roughly the 20-40% range.

Acknowledgements

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Table 1: Estimates of abundance and related quantities for the Gulf of Maine haddock for the preferred ASAP model and SCAA runs for isolated stock (with no movement) assessments. Values in parentheses are Hessian based 90% CIs. Biomass units in this and all following tables are mt unless otherwise indicated. The fishing mortality F applies to the commercially fully selected 7+ fish.

	No movement models								
	ASAP	SCAA BC1		CV_{SR}=0.5 in 2013		Age-specific σ_{CAA}		W_{CAA}=0.5	
'-lnL:overall		-1438.7		-1431.8		-1474.5		-692.8	
'-lnL:Index		11.0		14.8		12.5		10.7	
'-lnL:comCAA		-743.9		-743.4		-765.2		-369.1	
'-lnL:indexCAA		-749.8		-748.2		-763.6		-375.3	
'-lnL:catch		0.9		0.8		0.9		0.4	
-lnL:epsμ		0.0		0.0		0.0		0.0	
'-lnL:RecRes		43.1		44.1		41.0		40.5	
B^{sp}_{1977}	9438	7871	(6693; 9049)	7875	(6695; 9055)	7878	(6724; 9032)	8032	(6496; 9568)
B^{sp}_{2013}	4153	5206	(2889; 7522)	3943	(1999; 5887)	4861	(2611; 7111)	5523	(3039; 8008)
$B^{sp}_{2013}/B^{sp}_{1977}$	0.44	0.66	(0.35; 0.97)	0.50	(0.24; 0.76)	0.62	(0.32; 0.92)	0.69	(0.35; 1.02)
$N_{1,1977}$	5997	6441	(5198; 7683)	6428	(5186; 7670)	6821	(5472; 8170)	6359	(4682; 8036)
$N_{1,2011}$	6659	8667	(4771; 12563)	6578	(3422; 9735)	7704	(4135; 11273)	8998	(4637; 13359)
$N_{1,2013}$	16565	15824	(5309; 26339)	6280	(2462; 10098)	14889	(5036; 24742)	14030	(2758; 25302)
$F_{30\%}$		0.74	-	0.73	-	0.69	-	0.74	-
$B_{MSY}(F_{30\%})$		2652	(2299; 3005)	2553	(2211; 2895)	2667	(2324; 3010)	2724	(2262; 3186)
$MSY(F_{30\%})$		908	(828; 988)	873	(797; 948)	910	(826; 994)	932	(839; 1025)
$F_{40\%}$	0.46	0.43	-	0.42	-	0.41	-	0.43	-
$B_{MSY}(F_{40\%})$		3536	(3109; 3964)	3404	(2990; 3817)	3556	(3136; 3977)	3632	(3087; 4177)
$MSY(F_{40\%})$		825	(751; 899)	793	(722; 863)	825	(748; 903)	848	(760; 935)
Spring q	0.25	0.26	(0.22; 0.30)	0.27	(0.23; 0.31)	0.25	(0.22; 0.29)	0.27	(0.22; 0.32)
Fall q	0.92	0.94	(0.71; 1.16)	0.97	(0.74; 1.20)	0.98	(0.76; 1.21)	0.96	(0.65; 1.26)
Spring AddVar		0.10	(0.00*; 0.33)	0.12	(0.00*; 0.37)	0.10	(0.00*; 0.34)	0.10	(0.00*; 0.33)
Fall AddVar		0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)
σ_{Rout}		1.27	(1.21; 1.33)	1.22	(1.17; 1.27)	1.24	(1.18; 1.30)	1.23	(1.16; 1.30)

Table 2: Estimates of abundance and related quantities for the Gulf of Maine haddock for the SCAA migration model (i.e. with movement of 2+ year old haddock for the Base Case BC2). Values in parentheses are Hessian based 90% CIs. The value of μ is the proportion of (here 2+ year old for BC2) GB haddock which permanently migrate to the GoM each year, while λ specifies the proportion of such migration in the reverse direction. The text explains the role of the GBcrit constraint.

	No movement		Migration models												
	SCAA BC1		SCAA BC2		$\lambda=\mu$		1+ moving		3+ moving		GBcrit=50% of 1977-2013 average		Random effects on μ	
'-lnL:overall	-1438.7		-1447.7		-1447.6		-1446.5		-1447.1		-1444.8		-1452.4	
'-lnL:Index	11.0		9.9		9.9		10.2		9.5		10.7		9.7	
'-lnL:comCAA	-743.9		-750.9		-750.9		-750.1		-751.7		-746.0		-757.0	
'-lnL:indexCAA	-749.8		-753.7		-753.8		-753.4		-751.9		-752.6		-756.4	
'-lnL:catch	0.9		0.9		0.9		0.9		0.9		0.9		0.9	
-lnL:epsμ	0.0		0.0		0.0		0.0		0.0		0.0		3.7	
'-lnL:RecRes	43.1		46.1		46.2		45.9		46.1		42.3		46.8	
μ (%)	0		0.20	(0.13; 0.28)	0.20	(0.13; 0.28)	0.18	(0.11; 0.24)	0.22	(0.13; 0.31)	0.28	(0.15; 0.41)	0.20	(0.20; 0.20)
B^{sp}_{1977}	7871	(6693; 9049)	7742	(6570; 8913)	7779	(6602; 8955)	7755	(6584; 8926)	7718	(6549; 8888)	7842	(6663; 9022)	7710	(6541; 8879)
B^{sp}_{2013}	5206	(2889; 7522)	6061	(3690; 8431)	6079	(3703; 8454)	5690	(3428; 7952)	6394	(3873; 8915)	6705	(3819; 9591)	5376	(3050; 7702)
$B^{sp}_{2013}/B^{sp}_{1977}$	0.66	(0.35; 0.97)	0.78	(0.45; 1.11)	0.78	(0.45; 1.11)	0.73	(0.42; 1.05)	0.83	(0.48; 1.18)	0.85	(0.47; 1.24)	0.70	(0.38; 1.02)
$N_{1,1977}$	6441	(5198; 7683)	6259	(5046; 7472)	6309	(5087; 7531)	6274	(5059; 7490)	6286	(5073; 7499)	6406	(5172; 7641)	6206	(5009; 7402)
$N_{1,2011}$	8667	(4771; 12563)	7493	(3983; 11003)	7561	(4019; 11103)	7311	(3857; 10766)	8607	(4875; 12339)	8124	(4176; 12073)	6928	(3388; 10469)
$N_{1,2013}$	15824	(5309; 26339)	15270	(5533; 25007)	15405	(5579; 25231)	14594	(5151; 24037)	16544	(6036; 27052)	16960	(5964; 27956)	13776	(4745; 22807)
$F_{30\%}$	0.74	-	0.62	-	0.62	-	0.63	-	0.61	-	0.63	-	0.64	-
$B_{MSY}(F_{30\%})$	2652	(2299; 3005)	2226	(1841; 2611)	2249	(1864; 2634)	2275	(1898; 2651)	2322	(1936; 2709)	2467	(2072; 2863)	2085	(1741; 2429)
$MSY(F_{30\%})$	908	(828; 988)	755	(676; 835)	763	(683; 843)	773	(694; 851)	788	(711; 864)	837	(753; 921)	708	(623; 793)
$F_{40\%}$	0.43	-	0.37	-	0.37	-	0.38	-	0.37	-	0.38	-	0.38	-
$B_{MSY}(F_{40\%})$	3536	(3109; 3964)	2968	(2509; 3427)	2999	(2540; 3458)	3033	(2584; 3482)	3097	(2640; 3554)	3290	(2819; 3760)	2780	(2359; 3201)
$MSY(F_{40\%})$	825	(751; 899)	684	(612; 756)	691	(619; 764)	700	(629; 771)	713	(644; 783)	758	(681; 836)	642	(564; 720)
Spring q	0.26	(0.22; 0.30)	0.27	(0.23; 0.31)	0.27	(0.23; 0.31)	0.27	(0.23; 0.31)	0.27	(0.23; 0.31)	0.25	(0.21; 0.29)	0.28	(0.24; 0.32)
Fall q	0.94	(0.71; 1.16)	0.92	(0.69; 1.14)	0.91	(0.69; 1.14)	0.93	(0.71; 1.16)	0.91	(0.69; 1.14)	0.85	(0.63; 1.07)	0.98	(0.73; 1.22)
Spring AddVar	0.10	(0.00*; 0.33)	0.08	(0.00*; 0.30)	0.08	(0.00*; 0.30)	0.09	(0.00*; 0.31)	0.08	(0.00*; 0.30)	0.09	(0.00*; 0.31)	0.09	(0.00*; 0.30)
Fall AddVar	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)
σ_{Rout}	1.27	(1.21; 1.33)	1.31	(1.24; 1.39)	1.32	(1.24; 1.39)	1.31	(1.24; 1.38)	1.31	(1.25; 1.38)	1.26	(1.19; 1.32)	1.32	(1.25; 1.40)

Table 3: Estimates of abundance and related quantities for the Gulf of Maine haddock stock for the SCAA sabbatical model. The value of is the proportion of GB haddock (aged 2+ for the Base Case BC3) which move temporarily to the GoM area each year; the values shown in the Table do not include those GB fish, and refer to haddock from the GoM stock only). Values in parentheses are Hessian based 90% CIs.

	No movement		Sabbatical models					
	SCAA BC1		SCAA BC3		1+ moving		3+ moving	
'-lnL:overall	-1438.7		-1446.3		-1441.2		-1445.8	
'-lnL:Index	11.0		11.0		10.2		10.0	
'-lnL:comCAA	-743.9		-747.3		-747.6		-749.7	
'-lnL:indexCAA	-749.8		-757.4		-750.1		-753.5	
'-lnL:catch	0.9		0.9		0.9		0.9	
-lnL:epsμ	0.0		0.0		0.0		0.0	
'-lnL:RecRes	43.1		46.4		45.3		46.5	
θ (%)	0		0.75	(0.49; 1.02)	0.53	(0.20; 0.85)	0.64	(0.40; 0.89)
B^{sp}_{1977}	7871	(6693; 9049)	7602	(6419; 8786)	7632	(6453; 8811)	7698	(6514; 8881)
B^{sp}_{2013}	5206	(2889; 7522)	3131	(1580; 4682)	3707	(1671; 5744)	4185	(2427; 5944)
$B^{sp}_{2013}/B^{sp}_{1977}$	0.66	(0.35; 0.97)	0.41	(0.20; 0.63)	0.49	(0.21; 0.76)	0.54	(0.30; 0.79)
$N_{1,1977}$	6441	(5198; 7683)	6255	(5035; 7475)	6304	(5075; 7533)	6284	(5066; 7501)
$N_{1,2011}$	8667	(4771; 12563)	4813	(1933; 7693)	5895	(2177; 9613)	6929	(3792; 10066)
$N_{1,2013}$	15824	(5309; 26339)	12277	(4394; 20160)	12195	(2580; 21810)	14611	(5443; 23777)
$F_{30\%}$	0.74	-	0.69	-	0.71	-	0.69	-
$B_{MSY}(F_{30\%})$	2652	(2299; 3005)	2123	(1761; 2484)	2280	(1861; 2700)	2280	(1926; 2634)
$MSY(F_{30\%})$	908	(828; 988)	725	(640; 810)	780	(675; 885)	777	(697; 856)
$F_{40\%}$	0.43	-	0.41	-	0.41	-	0.41	-
$B_{MSY}(F_{40\%})$	3536	(3109; 3964)	2830	(2390; 3270)	3041	(2524; 3558)	3040	(2614; 3465)
$MSY(F_{40\%})$	825	(751; 899)	658	(580; 736)	708	(612; 804)	706	(633; 778)
Spring q	0.26	(0.22; 0.30)	0.28	(0.24; 0.32)	0.28	(0.24; 0.32)	0.27	(0.23; 0.31)
Fall q	0.94	(0.71; 1.16)	1.01	(0.77; 1.24)	1.02	(0.78; 1.26)	0.98	(0.75; 1.21)
Spring AddVar	0.10	(0.00*; 0.33)	0.09	(0.00*; 0.32)	0.10	(0.00*; 0.33)	0.09	(0.00*; 0.30)
Fall AddVar	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)	0.00	(0.00; 0.00)
σ_{Rout}	1.27	(1.21; 1.33)	1.32	(1.25; 1.39)	1.30	(1.23; 1.38)	1.32	(1.25; 1.39)

Table 4: Catch (mt) projections from 2014 for the three SCAA Base Cases under $F_{40\%}$ as estimated by the SCAA models, and F=0.46 (the value of estimated for $F_{40\%}$ for the preferred ASAP model - see Annex B, section B.4.3). The lowest section of the Table shows results for F=0.46 from 2015 with 500mt for the 2014 catch for these three Base Cases. For the sabbatical model, the values in parentheses refer to the catch arising from the GoM haddock stock only.

	1	2	3	
	SCAA BC1 "no movement"	SCAA BC2 "migration model"	SCAA BC3 "sabbatical model"	
$F_{40\%}$ (SCAA)	(0.43)	(0.37)	(0.41)	
C_{2014}	1318	1661	1226	(848)
C_{2015}	2018	2503	1918	(1373)
C_{2016}	2414	2909	2326	(1709)
C_{2017}	2793	3294	2711	(2021)
$F_{40\%}$ (ASAP)=0.46 (applied to all three cases)				
C_{2014}	1409	2011	1374	(951)
C_{2015}	2139	2942	2127	(1517)
C_{2016}	2531	3313	2545	(1857)
C_{2017}	2894	3627	2920	(2153)
500mt, $F_{40\%}$ (ASAP)=0.46 (applied to all three cases)				
C_{2014}	500	500	500	(345)
C_{2015}	2341	3315	2264	(1654)
C_{2016}	2702	3604	2661	(1973)
C_{2017}	3030	3842	3013	(2245)

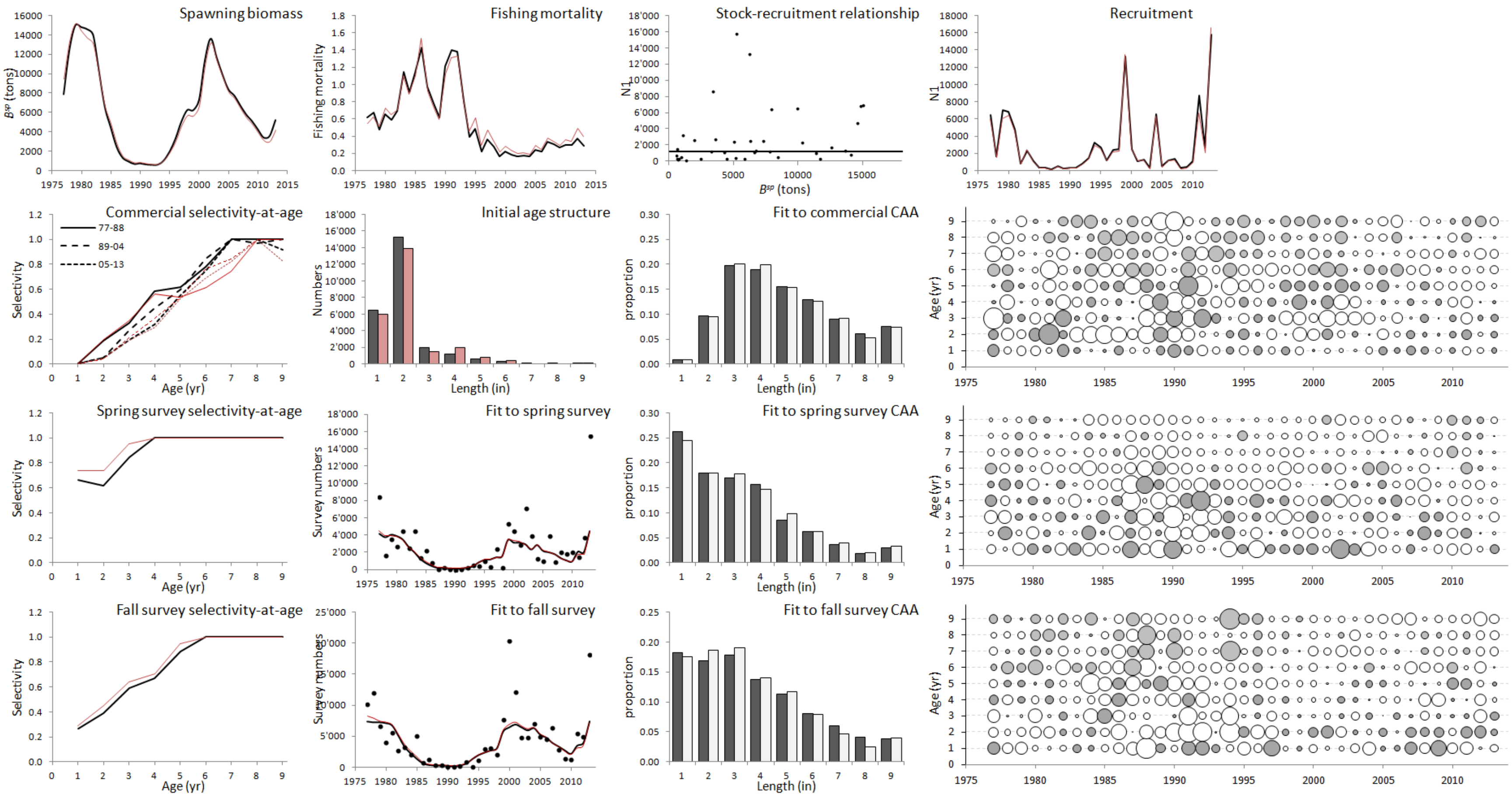

Figure 1: Comparison of the **SCAA-BC1 (isolated stock so no movement)** (in black) results with the **preferred ASAP-model** (in red).The fits to the CAA data are first shown as the averages over all years for each age, and then as bubble plots of standardised residuals. The area of the bubble is proportional to the magnitude of the corresponding residuals. For positive residuals the bubbles are grey, whereas for negative residuals the bubbles are white.

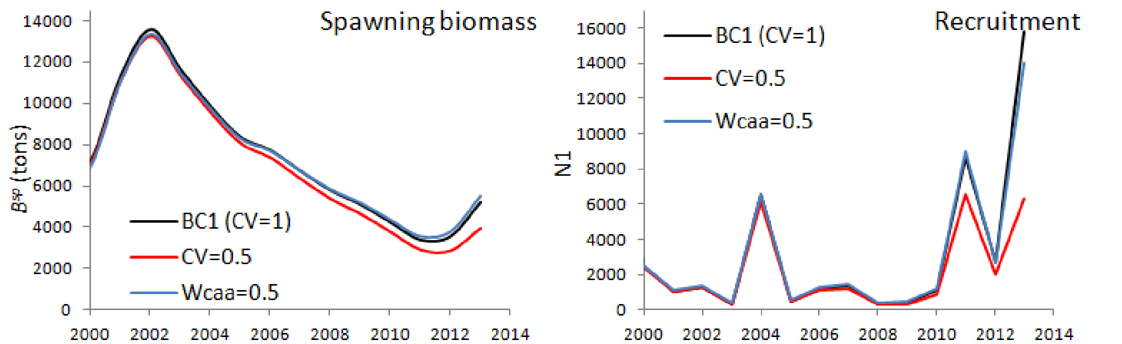

Figure 2: Comparison of spawning biomass and recruitment trajectories for the **SCAA-BC1 (isolated stock so no movement)** with a different stock-recruitment residual CV for 2013 (red lines), and for the proportions-at-age contributions to the negative log likelihood downweighted by a multiplicative factor of 0.5 (blue lines). The SCAA-Base Case assessment uses a CV of 1.0 for recruitment residuals for all years and is shown in black in the plots.

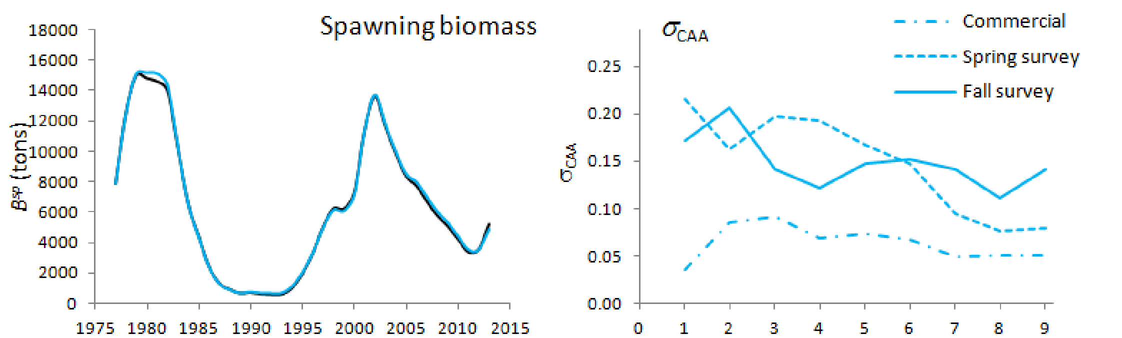

Figure 3: Comparison of spawning biomass trajectories for the **SCAA-Base Case (isolated stock so no movement)** (black line) and the sensitivity using age-specific σ_{CAA} values for the commercial and survey CAA data (blue line). The estimated σ_{CAA} values are also shown.

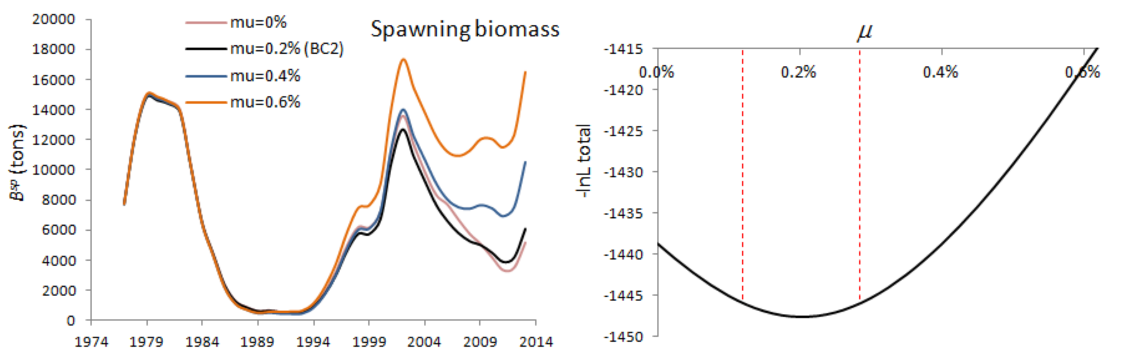

Figure 4: Comparison of spawning biomass trajectories for the **SCAA-BC2 (with movement)** with a series of fixed alternative movement proportions. Note that the μ=0% trajectory corresponds to **SCAA-BC1 (with no movement).** The right side plot shows the likelihood profile for the movement proportion μ (the vertical dashed lines correspond to the 90% confidence limits).

Figure 5: Comparison of the **SCAA-BC1 (isolated stock so no movement)** (in black) and **SCAA-BC2 (with movement)** (in blue) results. The fits to the CAA data are shown for SCAA-BC2 only, first as the averages over all years for each age, and then as bubble plots of standardised residuals. The area of the bubble is proportional to the magnitude of the corresponding residuals. For positive residuals the bubbles are light blue, whereas for negative residuals the bubbles are white.

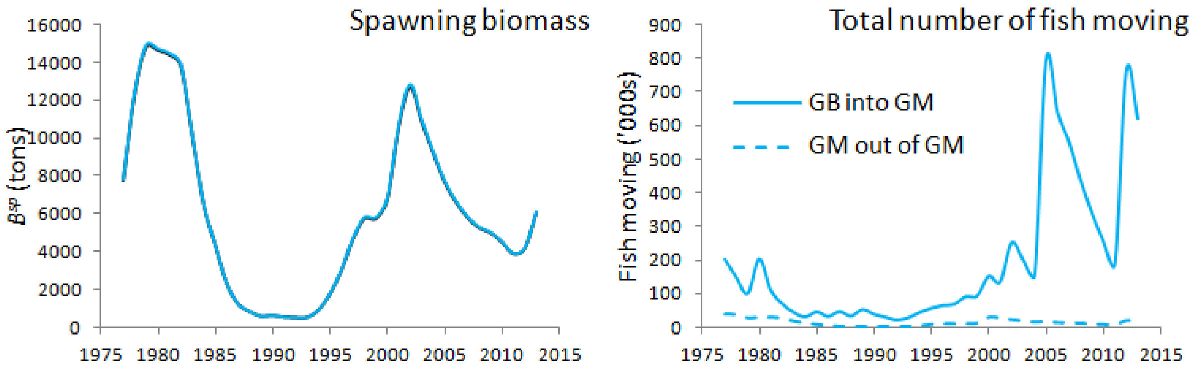

Figure 6: Comparison of spawning biomass trajectories for the **SCAA-BC2 (with movement)** (black line) and the sensitivity that also includes Gulf of Maine haddock emigrating out of the Gulf of Maine ($\lambda = \mu$) (blue line, which nearly always covers the black line). The right side-plot shows the total number of fish estimated to move in and out of the Gulf of Maine each year for this sensitivity.

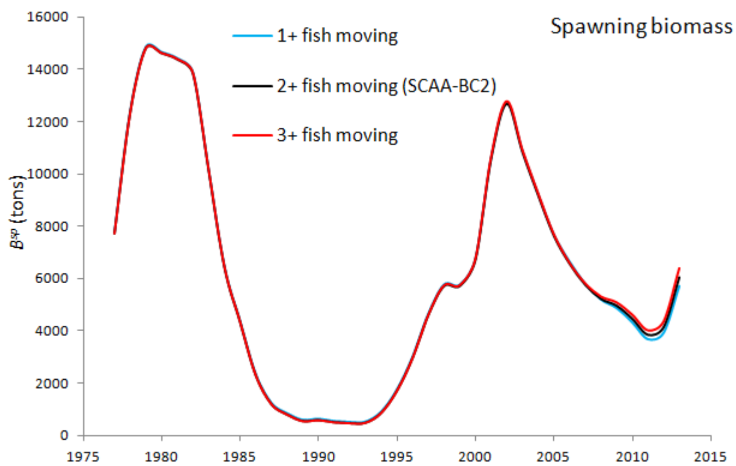

Figure 7: Comparison of spawning biomass trajectories for the **SCAA-BC2 (with movement)** with sensitivities to the choice of the age at which fish start to move (note that μ is estimated separately for each of these runs).

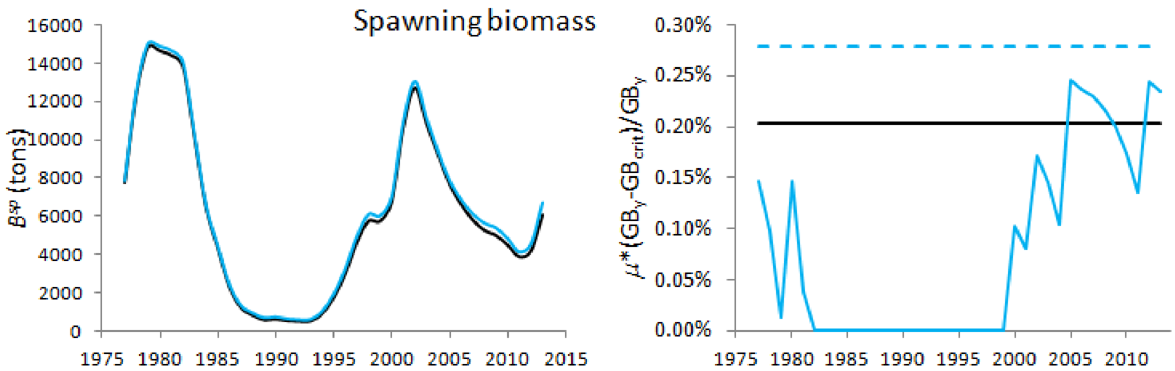

Figure 8: Comparison of spawning biomass trajectories for the **SCAA-BC2 (with movement)** (estimated $\mu = 0.20\%$, black line) and the sensitivity in which George's Bank fish move into Gulf of Maine only if the total number of fish of age 2+ is greater than GB_{crit} (see the text for details of how this threshold is defined). The horizontal dashed blue line is the maximum value which the proportion moving can attain in this sensitivity ($\mu = 0.28\%$).

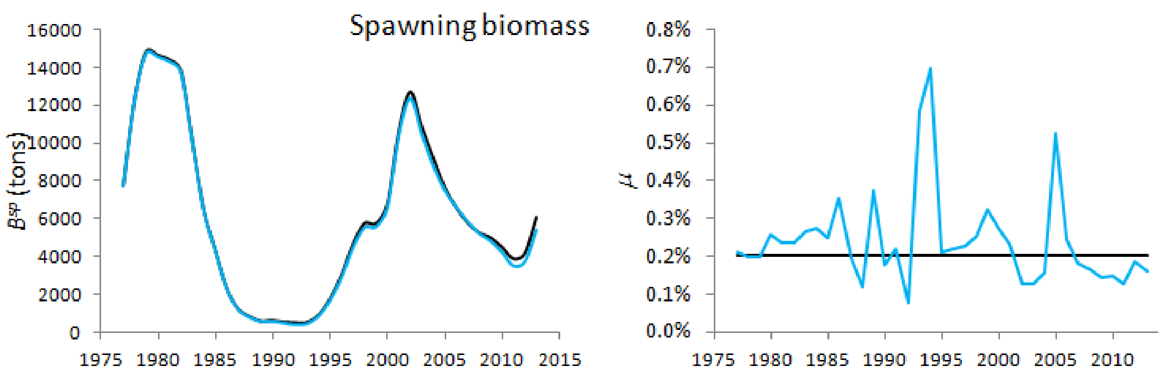

Figure 9: Comparison of spawning biomass trajectories for the **SCAA-BC2 (with movement)** (estimated $\mu = 0.20\%$, black line) and the sensitivity with random effects about μ = 0.20% (fixed) (blue line).The right side plot shows the fixed μ value together with the annual values estimated under the random effects approach.

Figure 10: Comparison of the **SCAA-BC1 (isolated stock so no movement)** (in black) and **SCAA-BC3 (sabbatical model)** (in blue) results. The fits to the CAA data are shown for SCAA-BC3 only, first as the averages over all years for each age, and then as bubble plots of standardised residuals. The area of the bubble is proportional to the magnitude of the corresponding residuals. For positive residuals the bubbles are light blue, whereas for negative residuals the bubbles are white.

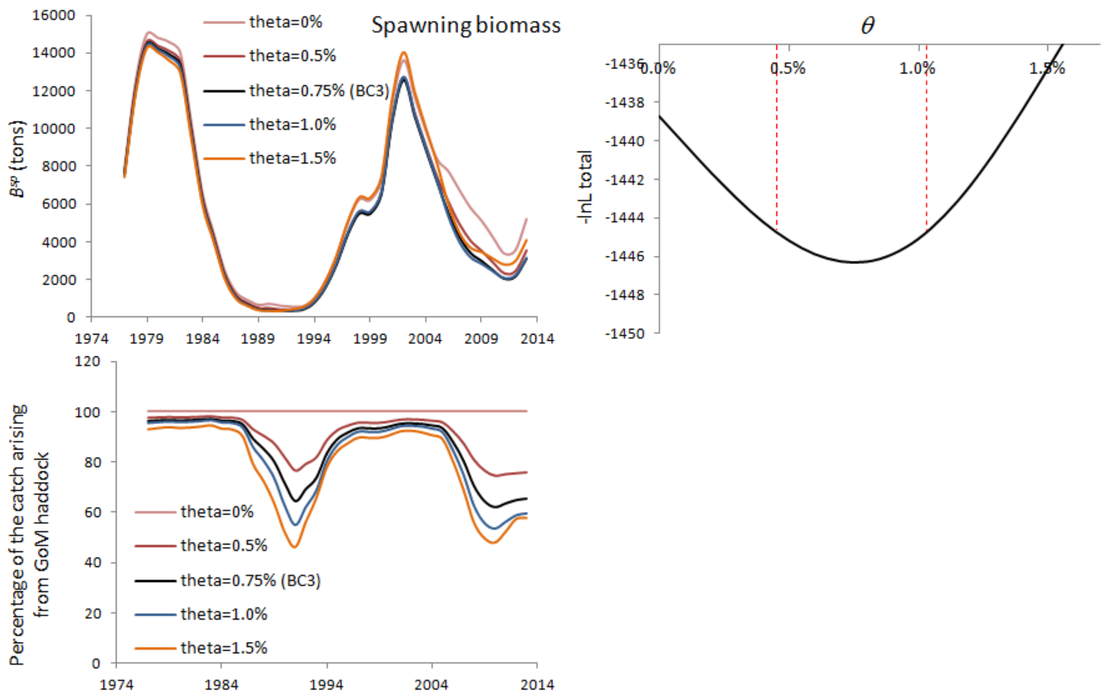

Figure 11: Comparison of spawning biomass trajectories for the **SCAA-BC3 (sabbatical model)** with a series of fixed alternative movement proportions (top left plot). Note that the θ=0% trajectory corresponds to the **SCAA-Base Case with no movement.** The top right side plot shows the likelihood profile for the movement proportion θ (the vertical dashed lines correspond to the 90% confidence limits). The bottom plot shows the percentage of the total haddock catch in the GoM area arising from the "true" GoM haddock stock for a series of θ values.

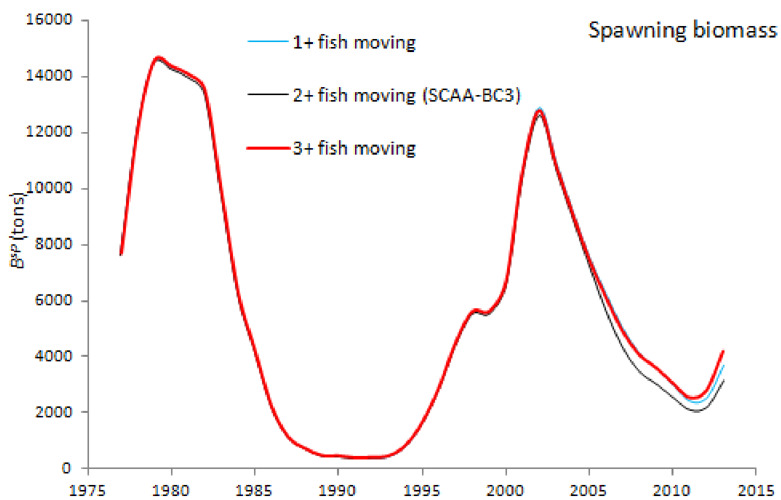

Figure 12: Comparison of spawning biomass trajectories for the **SCAA-BC3 (sabbatical model)** with sensitivities to the choice of the age at which fish start to move (note that θ is estimated separately for each of these runs).

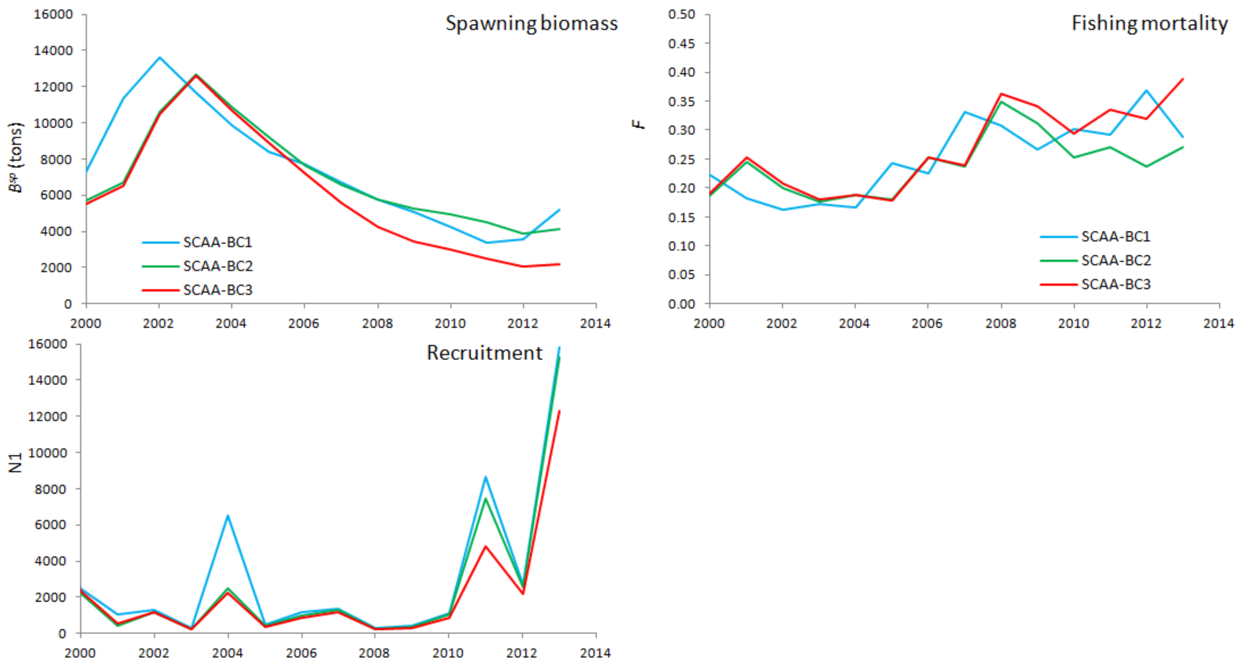

Figure 13: Comparison of spawning biomass, fishing mortality and recruitment trajectories for the **three SCAA Base Cases**. Note that the results shown for SCAA-BC3 (sabbatical model) exclude fish from the GB stock present in the GoM.

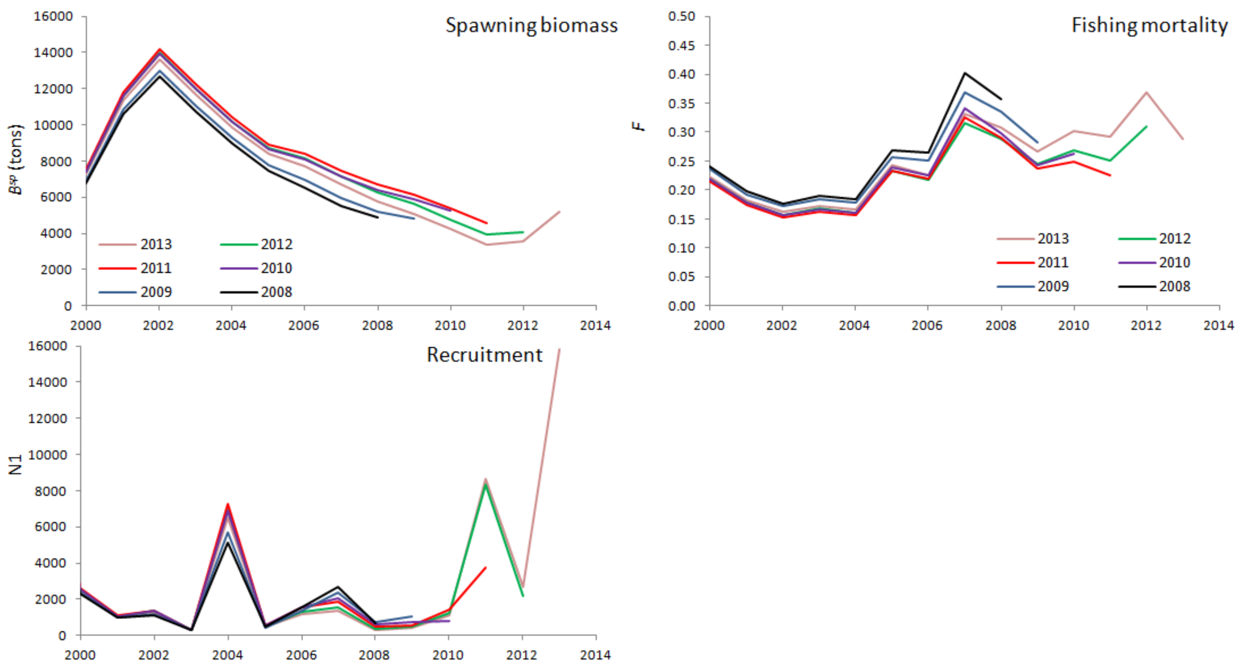

Figure 14a: Retrospective plots for **SCAA-BC1 (no movement).**

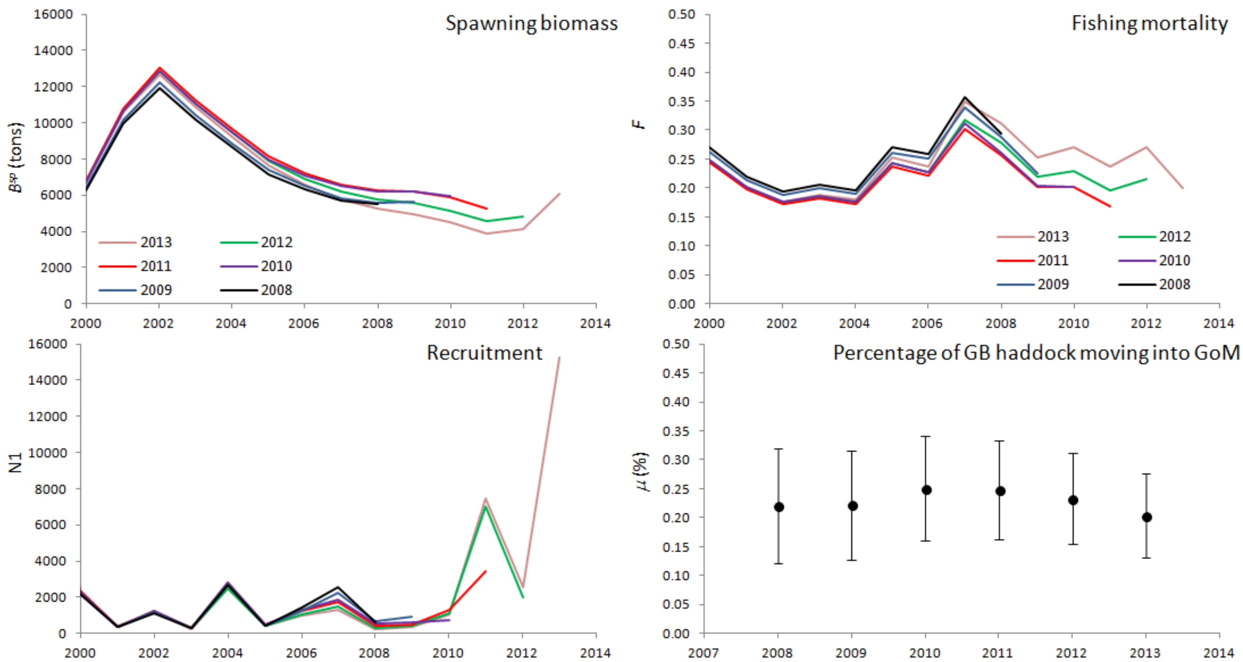

Figure 14b: Retrospective plots for **SCAA-BC2 migration model (permanent movement)**. The error bars for μ show the 90% Hessian-base CIs.

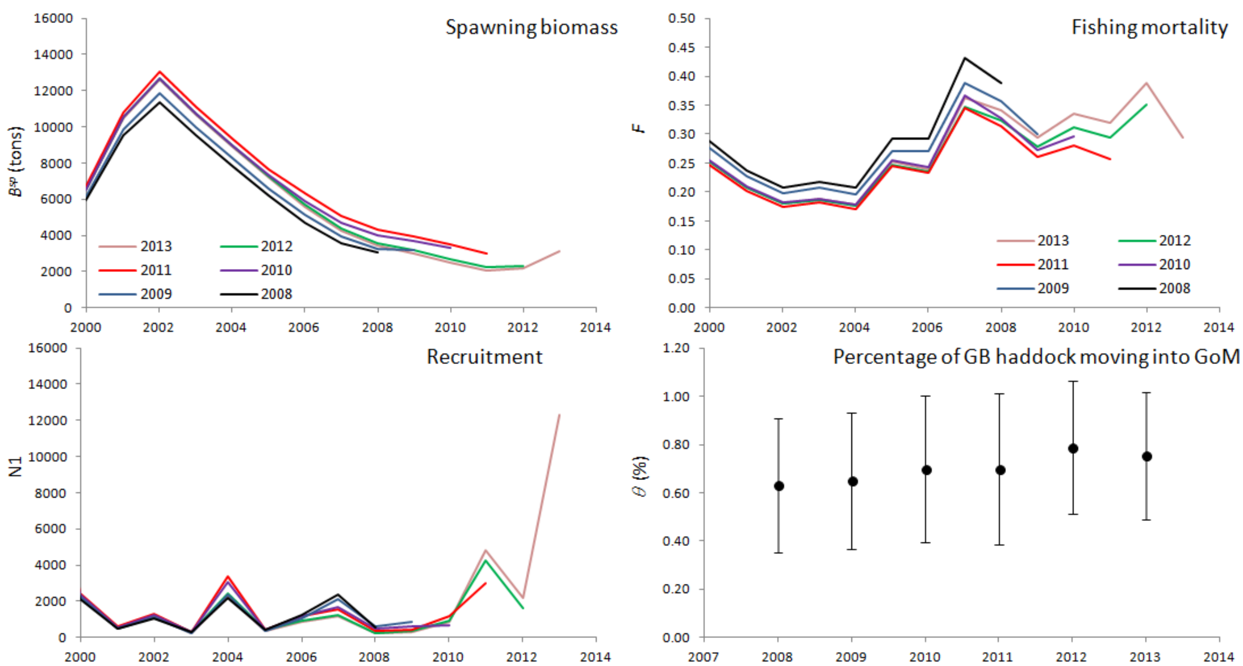

Figure 14c: Retrospective plots for **SCAA-BC3 sabbatical model (temporary movement)**. The error bars for θ show the 90% Hessian-base CIs.

ANNEX A – Data

Table A1: Total catch (metric tons) of haddock from the Gulf of Maine, 1977-2013 (Michael Palmer, pers. commn).

Year	Total catches (mt)	Year	Total catches (mt)	Year	Total catches (mt)
1977	3256.1	1990	472.4	2003	1346.7
1978	5023.5	1991	446.6	2004	1307.9
1979	4387.6	1992	321.4	2005	1576.7
1980	6520.6	1993	206.9	2006	1166.9
1981	6264.5	1994	186.7	2007	1343.2
1982	6941.7	1995	403.7	2008	1161.6
1983	7655.6	1996	341.0	2009	945.6
1984	4101.4	1997	1037.9	2010	958.1
1985	3088.2	1998	988.4	2011	744.2
1986	1922.2	1999	594.1	2012	739.1
1987	909.4	2000	985.5	2013	692.4
1988	438.8	2001	1232.4		
1989	284.6	2002	1251.8		

Table A2: Mean weight-at-age (kg) at the beginning of the year for the Gulf of Maine haddock stock (Michael Palmer, pers. commn).

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
1977	0.043	0.593	0.967	1.822	2.321	2.856	3.539	3.648	4.686
1978	0.083	0.296	0.967	1.400	2.213	2.820	3.948	3.888	6.088
1979	0.143	0.296	0.947	1.492	1.951	2.546	2.937	4.081	4.724
1980	0.169	0.506	0.951	1.463	2.077	2.754	3.095	3.558	4.204
1981	0.185	0.566	1.073	1.520	2.212	2.689	3.502	3.882	3.917
1982	0.151	0.589	0.826	1.800	2.267	2.864	3.322	3.886	4.293
1983	0.105	0.501	0.863	1.311	2.253	2.777	3.237	3.682	4.215
1984	0.123	0.384	0.907	1.471	2.003	2.743	3.413	3.897	4.073
1985	0.139	0.557	0.932	1.513	2.054	2.483	3.370	4.006	4.153
1986	0.174	0.400	1.081	1.247	2.052	2.416	2.850	3.612	4.592
1987	0.146	0.653	0.898	1.547	1.870	2.431	2.857	3.602	5.279
1988	0.142	0.380	0.958	1.607	2.268	2.490	3.178	3.902	5.180
1989	0.126	0.598	0.835	1.398	1.950	2.708	3.025	4.195	4.244
1990	0.135	0.464	1.292	2.375	2.113	2.669	3.275	3.651	4.189
1991	0.146	0.658	1.249	2.025	3.175	2.692	3.192	4.075	3.881
1992	0.165	0.643	1.604	1.926	2.656	3.047	2.480	3.548	3.450
1993	0.149	0.630	1.378	1.898	2.192	2.838	3.226	2.667	3.657
1994	0.148	0.533	1.186	1.866	2.500	2.606	3.315	3.402	3.721
1995	0.162	0.527	0.944	1.678	2.349	3.286	3.395	4.342	5.665
1996	0.076	0.439	0.906	1.436	1.974	2.819	2.953	3.141	3.164
1997	0.147	0.539	1.095	1.329	2.050	2.557	3.065	2.752	3.607
1998	0.124	0.579	1.108	1.800	1.914	2.574	3.170	3.067	2.988
1999	0.077	0.397	1.106	1.523	1.792	2.061	2.543	3.200	3.295
2000	0.137	0.411	0.778	1.402	1.685	1.882	2.143	2.477	3.101
2001	0.122	0.469	0.852	1.190	1.657	1.971	2.120	2.436	2.532
2002	0.086	0.346	0.908	1.252	1.532	1.955	2.385	2.258	2.624
2003	0.147	0.267	0.668	1.076	1.354	1.654	2.112	2.480	2.502
2004	0.107	0.428	0.681	1.109	1.249	1.528	1.761	2.060	2.202
2005	0.133	0.333	0.656	0.940	1.401	1.372	1.663	1.880	2.297
2006	0.121	0.375	0.690	0.722	1.215	1.537	1.461	1.668	2.006
2007	0.122	0.374	0.614	0.938	0.916	1.404	1.632	1.536	1.707
2008	0.096	0.362	0.743	0.954	1.190	1.213	1.565	1.681	1.744
2009	0.136	0.329	0.645	1.016	1.217	1.447	1.382	1.754	1.946
2010	0.150	0.421	0.741	0.928	1.238	1.399	1.674	1.825	2.067
2011	0.177	0.462	0.775	0.959	1.246	1.493	1.671	1.820	2.113
2012	0.125	0.441	0.724	1.041	1.292	1.414	1.670	1.807	1.975
2013	0.138	0.370	0.673	1.048	1.210	1.421	1.532	1.890	2.106

Table A3: Mean weight-at-age (kg) of landings for the Gulf of Maine haddock stock (Michael Palmer, pers. commn).

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
1977	0.113	0.757	1.163	2.008	2.558	3.358	3.709	3.587	4.686
1978	0.113	0.777	1.234	1.684	2.438	3.108	4.642	4.075	6.088
1979	0.337	0.774	1.155	1.805	2.261	2.659	2.775	3.587	4.724
1980	0.468	0.760	1.168	1.852	2.389	3.354	3.602	4.562	4.204
1981	0.560	0.685	1.516	1.978	2.641	3.026	3.657	4.184	3.917
1982	0.376	0.620	0.995	2.137	2.598	3.106	3.646	4.129	4.293
1983	0.181	0.667	1.200	1.727	2.376	2.969	3.373	3.719	4.215
1984	0.313	0.816	1.233	1.803	2.324	3.166	3.923	4.502	4.073
1985	0.315	0.980	1.068	1.859	2.339	2.652	3.588	4.090	4.153
1986	0.503	0.507	1.192	1.456	2.265	2.495	3.062	3.636	4.592
1987	0.350	0.856	1.592	2.008	2.402	2.609	3.272	4.236	5.279
1988	0.331	0.412	1.100	1.623	2.561	2.582	3.871	4.652	5.180
1989	0.251	1.126	1.779	1.824	2.343	2.864	3.543	4.545	4.244
1990	0.296	0.831	1.543	3.331	2.450	3.041	3.745	3.762	4.189
1991	0.347	1.459	1.880	2.657	3.027	2.958	3.350	4.433	3.881
1992	0.448	1.192	1.764	1.973	2.654	3.067	2.079	3.757	3.450
1993	0.364	0.885	1.592	2.041	2.436	3.035	3.393	3.422	3.657
1994	0.362	0.787	1.589	2.186	3.062	2.788	3.620	3.410	3.721
1995	0.275	0.802	1.156	1.774	2.525	3.526	4.133	5.209	5.665
1996	0.337	0.674	1.073	1.803	2.196	3.148	2.473	2.387	3.164
1997	0.354	0.891	1.802	1.662	2.330	2.977	2.985	3.063	3.607
1998	0.250	0.975	1.448	1.827	2.212	2.843	3.376	3.152	2.988
1999	0.266	0.611	1.309	1.608	1.765	1.926	2.281	3.033	3.295
2000	0.260	0.607	1.022	1.535	1.773	2.013	2.390	2.696	3.101
2001	0.242	0.889	1.260	1.490	1.811	2.210	2.243	2.483	2.532
2002	0.121	0.473	1.025	1.340	1.631	2.143	2.598	2.303	2.644
2003	0.318	0.583	0.887	1.230	1.468	1.770	2.134	2.425	2.513
2004	0.185	0.560	0.809	1.373	1.358	1.681	1.820	2.027	2.208
2005	0.286	0.583	0.815	1.139	1.464	1.443	1.684	1.954	2.297
2006	0.238	0.474	0.840	0.745	1.359	1.644	1.507	1.683	2.008
2007	0.243	0.560	0.777	1.121	1.203	1.510	1.625	1.578	1.714
2008	0.156	0.544	0.995	1.207	1.341	1.339	1.700	1.740	1.758
2009	0.304	0.699	0.809	1.135	1.282	1.625	1.563	1.877	1.947
2010	0.350	0.609	0.785	1.129	1.406	1.563	1.731	2.131	2.069
2011	0.341	0.588	1.029	1.191	1.401	1.602	1.801	1.915	2.113
2012	0.246	0.538	0.954	1.106	1.406	1.451	1.742	1.815	1.979
2013	0.283	0.550	0.870	1.267	1.498	1.486	1.658	2.051	2.104

Table A4: Maturity-at-age for Gulf of Maine haddock (Michael Palmer, pers. commn).

Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
0.036	0.284	0.809	0.978	0.998	1.000	1.000	1.000	1.000

Table A5: Total (commercial and recreational landings and discards) catches-at-age for the Gulf of Maine haddock stock (Michael Palmer, pers. commn).

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
1977	39'755	1'762'962	53'167	366'967	184'629	189'299	0	0	2'411
1978	0	374'650	2'291'417	172'388	363'003	208'654	10'580	0	5'290
1979	0	67'315	559'608	1'576'962	183'133	99'093	45'294	10'898	0
1980	0	884'750	104'084	755'832	1'366'770	143'816	95'570	27'794	25'756
1981	2'068	1'604'702	721'620	293'675	342'978	545'064	92'209	117'389	27'084
1982	30'427	620'596	1'519'414	620'677	100'582	300'972	477'524	107'352	75'881
1983	10'807	12'387	836'523	976'308	791'273	148'624	252'954	348'053	115'652
1984	1'202	88'981	49'873	597'996	256'658	364'974	62'198	64'813	147'568
1985	889	30'219	349'627	85'945	356'193	152'044	242'038	47'396	54'557
1986	4'278	10'819	183'531	358'782	81'336	114'028	86'352	102'482	14'690
1987	0	20'569	34'669	106'129	48'809	34'435	56'925	33'835	16'451
1988	305	471	12'442	12'340	54'752	55'564	7'635	15'049	4'149
1989	1'390	23'187	3'477	42'428	19'308	23'964	15'004	764	943
1990	7'007	2'033	143'062	1'686	28'820	17'607	27'466	4'050	0
1991	3'130	7'153	16'338	58'599	28'398	27'861	12'632	5'811	3'140
1992	1'819	13'092	94'371	36'543	19'112	2'246	1'134	0	1'895
1993	3'654	20'094	36'293	22'965	9'918	10'957	4'586	1'713	1'158
1994	6'455	23'681	44'531	13'600	3'419	9'230	5'675	1'711	705
1995	2'722	71'268	90'548	75'684	10'164	6'273	4'656	4'345	3'038
1996	2'789	23'528	129'505	56'458	16'363	4'055	7'112	5'599	1'162
1997	1'673	7'336	166'754	256'770	90'137	18'896	6'878	2'788	2'280
1998	5'833	23'752	25'097	132'738	192'766	52'737	17'433	8'611	7'557
1999	5'325	3'788	39'470	65'848	96'775	69'185	38'452	7'149	5'879
2000	2'355	68'641	66'083	106'777	65'090	128'456	72'058	31'811	25'699
2001	250	29'532	235'124	133'562	96'770	87'348	80'744	40'447	24'091
2002	420	2'372	27'821	275'333	117'143	110'413	32'129	70'430	68'048
2003	112	10'816	6'947	54'106	506'905	90'486	62'967	21'551	70'262
2004	1'834	1'903	14'147	32'957	71'982	512'704	59'670	34'048	51'126
2005	3'102	33'576	6'330	49'301	84'813	138'500	534'892	53'682	71'810
2006	2'461	2'174	123'574	8'459	52'663	71'713	83'463	366'977	61'040
2007	7'838	24'934	17'335	332'664	11'399	54'415	43'212	87'858	371'105
2008	1'779	18'476	55'817	19'091	407'625	5'033	42'176	29'576	225'281
2009	62	3'367	40'230	51'070	15'069	294'907	5'334	31'991	146'769
2010	2'412	6'725	13'762	39'606	52'249	19'113	294'288	3'477	134'175
2011	6'319	19'160	4'549	4'734	50'912	47'368	16'338	181'006	93'273
2012	2'642	110'559	48'841	19'988	12'290	67'218	37'764	13'802	204'436
2013	23'980	36'977	317'467	48'103	18'637	9'159	41'078	27'677	76'878

Table A6: Catch-at-age of haddock in the NEFSC offshore spring research vessel bottom trawl surveys in the Gulf of Maine, 1977-2013 (Michael Palmer, pers. commn).

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+	Cumulative 1-9+	CV
1977	1306.1	4237.2	40.7	1731.1	499.9	700.8	0.0	0.0	0.0	8515.7	(0.31)
1978	107.0	854.5	444.4	37.7	255.0	0.0	0.0	0.0	0.0	1698.5	(0.40)
1979	365.2	121.2	879.4	1587.7	292.3	168.7	114.5	65.3	0.0	3594.3	(0.22)
1980	1319.1	191.3	223.4	684.5	274.2	31.6	0.0	0.0	46.0	2770.0	(0.38)
1981	1396.3	1370.0	688.0	299.7	437.0	208.9	31.7	42.2	50.9	4524.7	(0.25)
1982	0.0	505.5	1168.5	466.4	177.4	86.3	128.5	15.8	15.8	2564.1	(0.31)
1983	1746.6	167.1	1361.8	384.0	608.6	0.0	136.0	67.6	28.1	4499.9	(0.40)
1984	24.3	713.9	68.1	374.5	134.9	0.0	0.0	56.1	0.0	1371.9	(0.41)
1985	16.0	401.1	1349.6	68.3	194.3	103.8	62.4	24.8	0.0	2220.2	(0.38)
1986	63.9	0.0	203.0	453.3	0.0	45.5	91.2	28.7	0.0	885.5	(0.46)
1987	45.0	30.8	38.8	0.0	0.0	0.0	0.0	0.0	0.0	114.6	(0.36)
1988	54.4	0.0	0.0	18.2	149.2	0.0	0.0	13.0	0.0	234.7	(0.52)
1989	0.0	44.6	14.9	0.0	14.9	29.7	0.0	0.0	0.0	104.1	(0.75)
1990	14.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	14.9	(0.54)
1991	17.8	9.3	9.3	55.9	0.0	0.0	0.0	0.0	0.0	92.2	(0.53)
1992	106.0	0.0	0.0	136.3	0.0	0.0	0.0	0.0	0.0	242.2	(0.59)
1993	326.3	183.4	0.0	0.0	36.2	18.2	0.0	0.0	0.0	564.0	(0.45)
1994	92.2	227.7	145.8	37.2	0.0	0.0	0.0	0.0	0.0	502.9	(0.34)
1995	552.6	300.0	99.1	28.7	0.0	0.0	0.0	28.7	0.0	1009.0	(0.46)
1996	0.0	45.7	183.4	153.4	0.0	0.0	0.0	0.0	0.0	382.5	(0.31)
1997	970.2	262.6	322.1	753.1	87.5	27.9	0.0	0.0	0.0	2423.6	(0.40)
1998	100.6	57.7	0.0	77.4	11.3	0.0	0.0	0.0	0.0	247.0	(0.41)
1999	4663.8	108.8	199.9	36.2	280.9	55.0	0.0	0.0	0.0	5344.7	(0.39)
2000	1293.0	1465.8	1218.3	171.1	155.4	54.4	163.1	0.0	0.0	4521.0	(0.41)
2001	91.4	158.1	1369.4	616.0	218.3	241.1	90.6	53.4	122.5	2960.7	(0.56)
2002	4132.1	257.9	751.4	1772.7	122.7	34.2	27.9	45.5	0.0	7144.4	(0.51)
2003	449.8	260.6	113.6	134.6	2525.9	254.4	151.9	44.6	61.4	3996.9	(0.25)
2004	144.0	0.0	193.1	27.2	119.2	782.2	45.5	0.0	18.2	1329.4	(0.35)
2005	12.9	214.8	0.0	124.0	101.1	274.3	316.4	0.0	36.2	1079.7	(0.39)
2006	224.1	95.7	2068.1	397.9	130.4	24.0	251.9	682.5	72.4	3947.0	(0.45)
2007	194.8	106.2	34.9	303.5	0.0	35.3	36.2	35.3	218.8	965.1	(0.37)
2008	54.4	706.8	508.9	0.0	379.5	0.0	33.4	64.8	368.2	2116.0	(0.49)
2009	40.2	111.5	527.1	274.5	49.8	685.7	0.0	16.9	211.5	1917.3	(0.36)
2010	129.0	15.8	9.9	163.8	161.7	63.3	985.2	0.0	512.8	2041.4	(0.35)
2011	694.6	248.6	12.1	0.0	31.1	218.3	0.0	203.0	136.4	1544.2	(0.34)
2012	666.2	1991.0	241.2	61.4	0.0	203.4	105.6	54.4	406.3	3729.4	(0.41)
2013	11846.9	1011.1	2037.6	168.6	45.3	33.6	145.3	100.2	116.9	15505.4	(0.44)

Table A7: Catch-at-age of haddock in the NEFSC offshore autumn research vessel bottom trawl surveys in the Gulf of Maine, 1977-2013 (Michael Palmer, pers. commn).

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+	Cumulative 1-9+	
1977	3863.8	4289.8	158.9	1309.2	149.9	360.0	0.0	0.0	128.6	10260.4	(0.32)
1978	86.2	1941.1	7098.5	184.7	954.8	1450.9	156.2	0.0	145.2	12017.6	(0.18)
1979	508.5	109.6	1389.5	3425.1	597.4	535.9	153.1	0.0	16.4	6735.6	(0.20)
1980	637.1	365.7	0.0	373.5	1330.3	796.2	226.8	161.1	116.1	4006.9	(0.33)
1981	749.0	588.0	1276.7	421.7	1002.1	1004.6	152.2	386.6	0.0	5581.0	(0.18)
1982	46.3	733.6	1134.3	470.1	61.9	0.0	120.2	120.2	120.2	2806.9	(0.35)
1983	667.8	63.3	845.3	629.9	502.6	222.2	85.5	205.5	0.0	3222.2	(0.28)
1984	263.3	668.2	0.0	365.9	0.0	517.2	0.0	42.7	267.7	2124.9	(0.26)
1985	111.5	495.9	3499.9	21.3	240.6	167.5	487.2	0.0	84.8	5108.6	(0.40)
1986	18.2	0.0	94.7	443.3	127.0	22.7	0.0	74.1	0.0	779.9	(0.41)
1987	0.0	190.5	127.3	118.2	76.2	377.3	222.3	0.0	148.2	1259.9	(0.32)
1988	0.0	0.0	51.9	28.7	114.7	0.0	80.5	143.4	0.0	419.2	(0.65)
1989	74.3	74.3	23.5	14.9	102.5	41.5	23.5	0.0	0.0	354.5	(0.38)
1990	29.7	0.0	70.5	0.0	0.0	0.0	47.1	23.5	0.0	170.8	(0.37)
1991	59.4	0.0	0.0	52.1	0.0	0.0	0.0	0.0	0.0	111.5	(0.60)
1992	181.1	0.0	28.7	0.0	0.0	0.0	0.0	0.0	0.0	209.8	(0.53)
1993	585.0	283.3	37.1	37.1	18.5	0.0	0.0	0.0	0.0	961.0	(0.72)
1994	59.1	0.0	0.0	0.0	0.0	0.0	45.5	0.0	45.5	150.0	(0.42)
1995	117.9	756.6	230.6	45.5	44.7	0.0	0.0	0.0	28.7	1223.9	(0.54)
1996	158.6	244.6	1330.7	774.4	85.0	142.3	88.9	45.5	90.9	2960.9	(0.37)
1997	1663.6	37.7	482.3	724.1	76.9	113.2	0.0	0.0	0.0	3097.8	(0.36)
1998	301.7	521.5	162.7	539.8	379.0	87.9	61.7	30.8	0.0	2085.3	(0.51)
1999	4043.7	744.5	1038.1	316.6	598.6	643.0	212.0	73.8	72.3	7742.6	(0.30)
2000	828.4	14160.3	2112.0	1632.1	531.8	726.5	276.6	92.6	0.0	20360.1	(0.45)
2001	301.1	2876.5	6089.8	899.9	1012.5	376.6	241.5	366.9	64.3	12229.0	(0.25)
2002	150.9	17.2	677.0	3074.0	428.1	180.7	0.0	269.4	95.7	4893.0	(0.35)
2003	0.0	334.7	89.9	630.8	3089.2	439.3	66.6	0.0	209.3	4859.7	(0.20)
2004	435.2	36.2	683.9	313.1	1037.4	4050.5	154.9	195.8	146.5	7053.6	(0.25)
2005	137.1	1995.2	83.9	184.5	375.2	509.4	1431.4	110.7	112.7	4940.2	(0.19)
2006	330.8	104.5	2230.7	33.3	257.3	134.8	363.5	1062.1	91.7	4608.6	(0.26)
2007	1333.5	1061.6	276.6	2665.4	75.9	17.7	204.3	143.0	657.8	6435.8	(0.30)
2008	0.0	505.4	138.7	0.0	1308.7	0.0	201.9	142.7	541.1	2838.4	(0.31)
2009	323.5	115.0	235.2	14.4	50.6	435.4	0.0	64.6	174.6	1413.3	(0.35)
2010	43.1	26.6	62.2	210.5	363.9	139.7	301.1	0.0	151.4	1298.5	(0.47)
2011	1491.4	1045.8	245.9	121.1	1064.0	276.4	169.7	936.0	148.4	5498.7	(0.46)
2012	658.7	4122.1	13.0	0.0	24.7	34.7	34.7	0.0	34.7	4922.6	(0.58)
2013	6333.5	2273.5	8469.4	519.5	325.5	112.6	34.7	0.0	100.5	18169.3	(0.23)

Table A8: Estimated numbers at age for Georges Bank haddock for ages 1-9+ for 1977 to 2011 from NEFSC (2012, Table B17). The projected numbers (in *italics*) for 2012 to 2017 were kindly provided by Liz Brooks, based on the following assumptions (Liz Brooks, pers. commn):

1. the fully selected *F* is 0.15 in 2011 to2016;
2. the recruitment in 2012 does not appear large based on surveys, and hence is possibly similar to recent recruitment (excluding 2010);
3. at first glimpse of 2013 recruitment seems VERY large; here it is arbitrarily assumed to be the same size as 2013 year-class; and
4. recruitment in years 2014-2017 is assumed to be time series median (from Table B17 in GB haddock report: NEFSC, 2012)

Year	Age1	Age2	Age3	Age4	Age5	Age6	Age7	Age8	Age9+
1977	13'983	86'117	4'726	4'473	2'540	1'132	73	196	558
1978	6'125	11'447	52'655	3'698	3'041	1'610	602	56	270
1979	83'888	5'014	8'680	30'082	2'751	1'975	851	367	176
1980	10'934	68'674	4'081	5'539	18'124	1'775	1'245	412	218
1981	7'364	8'945	28'384	3'027	3'653	9'382	918	529	315
1982	2'581	6'028	5'744	13'325	1'727	2'143	5'355	453	470
1983	3'284	2'112	3'879	3'226	7'533	1'060	1'240	3'370	320
1984	18'080	2'688	1'534	2'438	2'015	4'138	621	846	1'720
1985	2'518	14'801	2'116	986	1'335	1'288	2'044	296	577
1986	16'786	2'061	9'900	1'227	628	670	844	1'190	268
1987	2'614	13'738	1'638	5'549	801	381	391	555	706
1988	19'995	2'140	9'414	1'223	3'066	544	245	239	395
1989	1'364	16'366	1'704	5'517	877	1'656	308	150	230
1990	3'406	1'115	12'081	1'285	3'600	523	927	197	174
1991	2'716	2'732	902	8'362	810	1'939	290	563	213
1992	10'741	2'217	1'799	628	4'719	533	1'123	123	263
1993	15'568	8'718	1'576	1'107	337	2'110	274	538	261
1994	15'420	12'716	6'810	896	601	169	1'063	168	334
1995	12'687	12'601	9'926	4'503	517	364	72	589	156
1996	11'778	10'372	10'232	7'572	3'262	370	271	52	586
1997	23'451	9'637	8'441	7'866	5'372	2'265	244	202	537
1998	14'637	19'187	7'760	6'664	5'768	3'892	1'658	184	382
1999	49'156	11'979	15'501	5'929	4'953	4'032	2'673	1'205	352
2000	11'668	40'242	9'768	11'874	4'366	3'510	2'791	1'843	703
2001	90'866	9'551	32'580	7'433	8'306	3'045	2'398	1'946	1'422
2002	5'551	74'382	7'689	24'515	5'188	5'647	1'903	1'570	2'824
2003	2'870	4'542	60'540	5'983	17'209	3'414	3'629	1'197	2'192
2004	412'375	2'345	3'703	47'812	4'483	11'670	2'251	2'324	1'685
2005	7'985	337'041	1'890	2'922	34'534	3'014	6'917	1'227	1'336
2006	28'833	6'520	275'392	1'510	2'086	20'631	1'769	4'008	1'084
2007	7'123	23'458	5'322	222'615	1'172	1'371	12'024	1'154	2'296
2008	9'365	5'814	19'042	4'148	173'187	801	927	8'192	2'065
2009	4'773	7'660	4'723	15'197	3'122	130'269	552	661	4'480
2010	7'605	3'891	6'152	3'687	11'644	2'298	94'968	376	2'206
2011	374'008	6'195	3'132	4'658	2'672	8'234	1'481	65'649	300
2012	*8'000*	*305'513*	*5'022*	*2'477*	*3'603*	*1'967*	*5'802*	*1'061*	*47'393*
2013	*374'008*	*6'535*	*247'686*	*3'972*	*1'916*	*2'653*	*1'386*	*4'157*	*34'820*
2014	*11'301*	*305'513*	*5'298*	*195'868*	*3'072*	*1'411*	*1'869*	*993*	*28'010*
2015	*11'301*	*9'231*	*247'686*	*4'190*	*151'496*	*2'262*	*994*	*1'339*	*20'842*
2016	*11'301*	*9'231*	*7'484*	*195'868*	*3'240*	*111'547*	*1'594*	*712*	*15'940*
2017	*11'301*	*9'231*	*7'484*	*5'918*	*151'496*	*2'386*	*78'606*	*1'142*	*11'967*

ANNEX B - The Statistical Catch-at-Age Model

The text following sets out the equations and other general specifications of the SCAA followed by details of the contributions to the (penalised) log-likelihood function from the different sources of data available and assumptions concerning the stock-recruitment relationship. Quasi-Newton minimization is then applied to minimize the total negative log-likelihood function to estimate parameter values (the package AD Model Builder[TM], Otter Research, Ltd is used for this purpose).

B.1. Population dynamics

B.1.1 Numbers-at-age

The resource dynamics are modelled by the following set of population dynamics equations:

$N_1 = R_{y+1}$ (B1)

$N_{y+1,a+1} = N_{y,a} e^{-Z_{y,a}}$ for $1 \le a \le m-2$ (B2)

$N_{y+1,m} = N_{y,m-1} e^{-Z_{y,m-1}} + N_{y,m} e^{-Z_{y,m}}$ (B3)

where

$N_{y,a}$ is the number of fish of age *a* at the start of year *y*,

R_y is the recruitment (number of 1-year-old fish) at the start of year *y*,

m is the maximum age considered (taken to be a plus-group, and set here to be 9).

$Z_{y,a} = F_y S_{y,a} + M_a$ is the total mortality in year *y* on fish of age *a*, where

M_a denotes the natural mortality rate for fish of age *a*,

F_y is the fishing mortality of a fully selected age class in year *y*, and

$S_{y,a}$ is the commercial selectivity at age *a* for year *y*.

B.1.2. Recruitment

The number of recruits (i.e. new 1-year old) at the start of year *y* is taken as an average recruitment, allowing for annual fluctuation about the deterministic relationship.

$R_y = R_{gm} e^{\varsigma_y}$ (B4)

R_{gm} is the geometric mean (median under a log-normality assumption) recruitment over the period considered (see equation B18 below),

ς_y reflects fluctuation about the expected recruitment for year *y*, which is assumed to be normally distributed with standard deviation $\sigma_{R,y}$ (which is input in the applications considered here); these residuals are treated as estimable parameters in the model fitting process.

The spawning biomass at the start of year *y*, is computed as:

$$B_y^{\text{sp}} = \sum_{a=1}^{m} f_a w_{y,a}^{\text{strt}} N_{y,a} e^{-Z_{y,a}/4} \qquad \text{(B5)}$$

because spawning for the haddock stock under consideration is taken to occur three months after the start of the year and some mortality has therefore occurred,

where

$w_{y,a}^{\text{strt}}$ is the mass of fish of age *a* during spawning (Table A2), and

f_a is the proportion of fish of age *a* that are mature (Table A4).

B.1.3. Total catch and catches-at-age

The total catch by mass in year *y* is given by:

$$C_y = \sum_{a=1}^{m} w_{y,a}^{\text{mid}} C_{y,a} = \sum_{a=1}^{m} w_{y,a}^{\text{mid}} N_{y,a} \; S_{y,a} F_y \left(1 - e^{-Z_{y,a}}\right) / Z_{y,a} \qquad \text{(B6)}$$

where

$w_{y,a}^{\text{mid}}$ denotes the mass of fish of age *a* landed in year *y* (Table A3),

$C_{y,a}$ is the catch-at-age, i.e. the number of fish of age *a*, caught in year *y*,

The model estimate of survey index is computed as:

$$N_y^{\text{surv}} = \sum_{a=1}^{m} S_a^{\text{surv}} N_{y,a} e^{-Z_{y,a} T^{surv}/12} \qquad \text{(B7)}$$

where

S_a^{surv} is the survey selectivity for age *a*, which is taken to be year-independent, and

T^{surv} is the month in which the survey is taking place (T^{surv} =4 for spring surveys and T^{surv} =10 for fall surveys)

B.1.4. Initial conditions

For the first year (y_0) considered in the model, the numbers-at-age are estimated directly for ages 1 to a^{est}, with a parameter ϕ mimicking recent average fishing mortality for ages above a^{est}, i.e.

$$N_{y_0,a} = N_{\text{start},a} \qquad \text{for } 1 \le a \le a^{est} \qquad \text{(B8)}$$

and

$$N_{\text{start},a} = N_{\text{start},a-1} e^{-M_{a-1}} (1-\phi S_{a-1}) \qquad \text{for } a^{est} < a \le m-1 \qquad \text{(B9)}$$

$$N_{\text{start},m} = N_{\text{start},m-1} e^{-M_{m-1}} (1-\phi S_{m-1}) \big/ (1-e^{-M_m}(1-\phi S_m)) \qquad \text{(B10)}$$

B.2. The (penalised) likelihood function

The model can be fit to (a subset of) survey abundance indices, and commercial and survey catch-at-age data to estimate model parameters (which may include residuals about the stock-recruitment function, facilitated through the incorporation of a penalty function described below). Contributions by each of these to the negative of the (penalised) log-likelihood (- $\ell n L$) are as follows.

B2.1. Survey abundance data

The likelihood is calculated assuming that a survey index is lognormally distributed about its expected value:

$$I_y^{surv} = \hat{I}_y^{surv} \exp\left(\varepsilon_y^{surv}\right) \quad \text{or} \quad \varepsilon_y^{surv} = \ell n\left(I_y^{surv}\right) - \ell n\left(\hat{I}_y^{surv}\right) \qquad \text{(B11)}$$

where

I_y^{surv} is the survey biomass index for survey *surv* in year *y*,

$\hat{I}_y^{surv} = \hat{q}^{surv} \widehat{N}_y^{surv}$ is the corresponding model estimate, where

$\hat{q}^{surv}$ is the constant of proportionality (catchability) for the survey series *surv*, and

ε_y^{surv} from $N\left(0, \left(\sigma_y^{surv}\right)^2\right)$.

The contribution of the survey biomass data to the negative of the log-likelihood function (after removal of constants) is then given by:

$$-\ell n L^{\text{survey}} = \sum_{surv} \sum_{y} \left\{ \ell n \left(\sqrt{\left(\sigma_y^{surv}\right)^2 + \left(\sigma_{Add}^{surv}\right)^2} \right) + \left(\varepsilon_y^{surv}\right)^2 / \left[2\left(\left(\sigma_y^{surv}\right)^2 + \left(\sigma_{Add}^{surv}\right)^2 \right) \right] \right\} \qquad \text{(B12)}$$

where

σ_y^{surv} is the standard deviation of the residuals for the logarithm of index *i* in year *y* (which is input), and

σ_{Add}^{surv} is the square root of the additional variance for survey biomass series *surv*, which is estimated in the model fitting procedure, with an upper bound of 0.5.

The catchability coefficient q^{surv} for survey biomass index *surv* is estimated by its maximum likelihood value:

$$\ell n\, \hat{q}^{surv} = 1/n_{surv} \sum_{y} \left(\ln I_y^{surv} - \ln \widehat{N}_y^{surv} \right) \qquad \text{(B13)}$$

B.2.3. Commercial catches-at-age

The contribution of the catch-at-age data to the negative of the log-likelihood function is given by:

$$-\ell\mathrm{n}\, L^{\mathrm{CAA}} = W_{CAA}\sum_{y}\sum_{a}\left[\ell n\left(\sigma_a^{com}\right)+\left(\sqrt{p_{y,a}}-\sqrt{\hat{p}_{y,a}}\right)^2/2\left(\sigma_a^{\mathrm{com}}\right)^2\right] \quad \text{(B14)}$$

where

$p_{y,a} = C_{y,a}/\sum_{a'} C_{y,a'}$ is the observed proportion of fish caught in year *y* that are of age *a*,

$\hat{p}_{y,a} = \hat{C}_{y,a}/\sum_{a'} \hat{C}_{y,a'}$ is the model-predicted proportion of fish caught in year *y* that are of age *a*,

where

$$\hat{C}_{y,a} = N_{y,a}\ S_{y,a} F_y \left(1-e^{-Z_{y,a}}\right)\Big/Z_{y,a} \quad \text{(B15)}$$

W_{CAA} is a relative weighting accorded to these data in the negative log-likelihood, which is set equal to 1 for the Base Case runs in these analyses,

and

σ_{CAA}^{com} is the standard deviation associated with the catch-at-age data, which is estimated in the fitting procedure by:

$$\hat{\sigma}_{CAA}^{com} = \sqrt{\sum_{y}\left(\sqrt{p_{y,a}}-\sqrt{\hat{p}_{y,a}}\right)^2/\sum_{y}1} \quad \text{(B16)}$$

This formulation mimics a multinomial form for the error distribution by forcing a near-equivalent variance-mean relationship for the error distributions.

Commercial catches-at-age are incorporated in the likelihood function using equation (B14), for which the summation over age *a* is taken from age a_{minus} (considered as a minus group) to a_{plus} (a plus group), taken here as 1 and 9 respectively.

B.2.4. Survey catches-at-age

The survey catches-at-age are incorporated into the negative of the log-likelihood in an analogous manner to the commercial catches-at-age (equation (B14)) where:

$p_{y,a}^{surv} = C_{y,a}^{surv}/\sum_{a'} C_{y,a'}^{surv}$ is the observed proportion of fish of age *a* in year *y* for survey *surv*,

$\hat{p}_{y,a}^{surv}$ is the expected proportion of fish of age *a* in year *y* in the survey *surv*, given by:

$$\hat{p}_{y,a}^{surv} = S_a^{\mathrm{surv}} N_{y,a} e^{-Z_{y,a}T^{surv}/12} \Big/ \sum_{a'=0}^{m} S_{a'}^{\mathrm{surv}} N_{y,a'} e^{-Z_{y,a'}T^{surv}/12} \quad \text{(B17)}$$

As for the commercial data, the minus and plus groups for both surveys are taken here as 1 and 9 respectively.

B.2.5. Stock-recruitment function residuals

The stock-recruitment residuals are assumed to be lognormally distributed and serially correlated. Thus, the contribution of the recruitment residuals to the negative of the (now penalised) log-likelihood function is given by:

$$-\ell nL^{\text{pen}} = W_{SRpen}\left[\sum_{y=1977}^{2013}\left[\varepsilon_y^2/2\sigma_{\text{R},y}^2\right]+10000\left(\sum_{y=1977}^{2010}\varepsilon_y\right)^2\right] \quad \text{(B18)}$$

where

ε_y from $N\left(0,\left(\sigma_{R,y}\right)^2\right)$,

$\sigma_{R,y} = \sqrt{\ln\left(CV_y^2+1\right)}$ is the standard deviation of the log-residuals, which is input. For the SCAA-Base assessment, $CV_y = 1$ for all years.

$W_{SRpen} = 1$.

Note that the purpose of the second term on the right hand side of equation B.18 is to ensure that R_{gm} corresponds to the geometric mean (likely to closely approximate the median) of the pre-2011 recruitments.

B.2.5. Catches

$$-\ell nL^{\text{Catch}} = \sum_u\sum_y\left[\frac{\ell n\, C_y - \ell n\, \hat{C}_y}{2\sigma_{\text{C,y}}^2}\right] \quad \text{(B19)}$$

where

C_y is the observed catch in year y,

$\hat{C}_y$ is the predicted catch in year y, and

$\sigma_{\text{C,y}}$ is the input CV in year y. It is taken to be 0.15 over 1977-1981, 0.1 over 1982-1988 and 0.05 thereafter, as for the preferred ASAP model..

B.3. Estimation of precision

Where quoted, CV's or 90% probability interval estimates are based on the Hessian.

B.4. Model parameters

B.4.1. Fishing selectivity-at-age:

The commercial and survey fishing selectivities are estimated separately for each age. For the NEFSC offshore surveys, the fishing selectivities are assumed to be flat from age 4 and 6 onwards for the spring and fall surveys respectively.

The commercial selectivity is taken to differ over three blocks, as for the preferred ASAP model: 1977-1988, 1989-2004 and 2005-2013. These selectivities are set to 1 for age 7, and may not increase for greater ages.

B.4.2. Natural mortality

This was set to 0.2, independent of year and age.

B.4.3. Biological reference points

In the computation of the biological reference points, the weight-at-age, maturity-at-age and commercial selectivity vectors are taken as the average over the 2009-2013 period.

B. STOCK ASSESSMENT FOR ATLANTIC SEA SCALLOPS IN 2014, UPDATED THROUGH 2013

Invertebrate Subcommittee[1]

B1. TERMS OF REFERENCE

1) Estimate removals from all sources including landings, discards, incidental mortality, and natural mortality. Describe the spatial and temporal distribution of landings, discards, and fishing effort. Characterize the uncertainty in these assumptions and sources of data. If possible using sensitivity analyses, consider the potential effects that changes in fishing gear, fishing behavior, and management may have on the assumptions.
2) Present the survey data being used in the assessment (e.g., regional indices of relative or absolute abundance, recruitment, size data, etc.). Characterize the uncertainty and any bias in these sources of data.
3) Investigate the role of environmental and ecological factors in determining recruitment success. If possible, integrate the results into the stock assessment.
4) Estimate annual fishing mortality, recruitment and stock biomass for the time series, and estimate their uncertainty. Report these elements for both the combined resource and by sub-region. Include a historical retrospective analysis to allow a comparison with previous assessment results and previous projections.
5) State the existing stock status definitions for "overfished" and "overfishing". Then update or redefine biological reference points (BRPs; point estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, F_{MSY} and MSY) and provide estimates of their uncertainty. Comment on the scientific adequacy of existing BRPs and the "new" (i.e., updated, redefined, or alternative) BRPs.
6) Evaluate stock status with respect to the existing model (from previous peer reviewed accepted assessment) and with respect to a new model or model formulation developed for this peer review.
    a. Update the existing model with new data and evaluate stock status (overfished and overfishing) with respect to the existing BRP estimates.
    b. Then use the newly proposed model and evaluate stock status with respect to "new" BRPs and their estimates (from TOR-5).
7) Evaluate the realism of stock and catch projections and compute the statistical distribution (e.g., probability density function) of the OFL (overfishing level).
    a. Provide numerical annual projections (through 2016). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. Use a sensitivity analysis approach in which a range of assumptions about the most important uncertainties in the assessment are considered (e.g., terminal year abundance, variability in recruitment).
    b. Comment on the realism of the projections. Consider the major uncertainties in the assessment as well as sensitivity of the projections to various assumptions.

    Describe this stock's vulnerability (see "Appendix to the SAW TORs") to becoming overfished, and how this could affect the choice of ABC.

[1] See Appendix B1 for meetings and members of the Invertebrate Subcommittee who helped prepare this assessment.

8) Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in most recent SARC reviewed assessment and review panel reports. Identify new research recommendations.

B2. EXECUTIVE SUMMARY

TOR-1 (Estimate removals from landings, discards, incidental mortality, and natural mortality…) U.S. sea scallop landings were high and stable during 2003-2012, averaging about 25,000 mt meats, almost three times higher than the long-term 1950-1999 mean. Landings in 2013 declined to 18,641 mt meats, the lowest since 2000, but still over twice the long-term mean. About 65% of landings during 2003-2012 were from the Mid-Atlantic region, 32% from Georges Bank, 2% from Southern New England and under 1% from the Gulf of Maine; the proportion from the Mid-Atlantic was higher than in earlier periods. A shift in the fishery towards Georges Bank occurred in 2013, when 64% of the landings were from Georges Bank, 32% from the Mid-Atlantic, 2% from Southern New England and 3% from the Gulf of Maine. Discards were highly variable with year and region. Maximum discards were 2553 mt meats in 2003. Discards have decreased since 2004, likely due to changes in gear regulations; estimated discards in 2013 were 437 mt meats. Incidental fishing mortality (mortality of scallops that interact with the gear but are not caught) is highly uncertain; based on two studies from the 1970s and 1980s, incidental fishing mortality on small scallops was estimated as 0.2 times fully recruited fishing mortality on Georges Bank, and 0.1 times fully recruited fishing mortality in the Mid-Atlantic. Natural mortality for all but the largest size group was estimated at 0.16 for Georges Bank and 0.2 for the Mid-Atlantic, an increase from 0.12 and 0.15, respectively, in the last assessment. Plus group natural mortality was estimated as 1.5 times that of smaller scallops.

TOR-2 (Survey data). A scallop survey using a lined scallop dredge and a random-stratified design has been conducted every year since 1979 on Georges Bank and the Mid-Atlantic Bight. Based on this survey, biomass and abundance remained relatively low from 1979-1995 on Georges Bank and 1979-1998 in the Mid-Atlantic. The indices rose dramatically starting in 1995 on Georges Bank and 1998 in the Mid-Atlantic, and were fairly stable from 2003-2009. Decreases have been observed in both regions in recent years, although the indices are still well above levels observed previous to 1995. Paired tows experiments that compared dredge catches to densities observed using the HabCam towed camera system estimated the efficiency of the dredge as 0.41 on sand and 0.27 on gravel/cobble habitat (Appendix B4).

A video drop camera survey was conducted between 2003 and 2012 on Georges Bank and the Mid-Atlantic, using a systematic grid design. This survey generally shows declining trends, with biomass and abundance somewhat less than the expanded dredge survey.

A towed camera (“HabCam”) survey was used for the first time in this assessment (Appendix B6). The survey was conducted during 2011-2013 on Georges Bank and 2012-2013 in the Mid-Atlantic. HabCam is towed behind a vessel, taking rapid-fire photographs of the sea bottom. Estimates from HabCam were obtained using a model-based approach, using a zero-inflated generalized additive model combined with kriging of the residuals. Biomass and abundance estimates from HabCam were similar to those from the dredge.

TOR-3 (Environmental effects on recruitment). Two putative environmental factors were explored as predictors of recruitment in the Mid-Atlantic Bight (Appendix B8). A tentative relationship was found between food supply (phytoplankton) and recruitment. Additionally, the spatio-temporal distribution of the sea star *Astropecten americanus*, a predator of small invertebrates, including juvenile sea scallops, appear to correlate to the spatio-temporal patterns of scallop recruitment in the southern Mid-Atlantic Bight.

TOR-4 (Estimation of F, Biomass, Recruitment). A forward projecting size-structured estimation model (CASA) was used for estimation of biomass, fishing mortality and recruitment. Growth in the model was based on growth increment data from shell growth ring analysis. Three models were used, one each for the open and closed portions of Georges Bank, and a model for the Mid-Atlantic. The models appeared to give good estimation for some years, but in the Georges Bank Closed and Mid-Atlantic models, estimates of abundance and biomass had poor diagnostics in years associated with very strong year classes. Model estimated biomass and abundance generally declined, and fishing mortality increased, during 1975-1995. The biomass in the Georges Bank closed areas increased rapidly after these areas were closed to fishing in 1994. Estimated biomass in Georges Bank open and the Mid-Atlantic increased more gradually as fishing mortality was slowly reduced starting around 1998. Estimated overall fully recruited fishing mortality in 2013 was 0.32, and biomass was estimated at 132,561 mt meats. This was slightly higher than direct expanded estimates from the dredge survey (129,113 mt meats) and HabCam (111,157 mt meats). Explorations were made in incorporating density-dependent mortality on juvenile scallops into the CASA model in order to better model the population dynamics of large year classes, and initial results appear to be promising.

TOR-5 (Stock status definition). The SYM (Stochastic Yield Model) was used to estimate reference points. This model explicitly takes into account parameter uncertainty, including key uncertainties in natural mortality and stock-recruit relationships, when estimating maximal sustainable yield (MSY) and the associated biomass and fishing mortality reference points B_{MSY} and F_{MSY}. Estimated whole stock MSY, F_{MSY} and B_{MSY} were 23,798 mt meats, 0.48 and 96,480 mt meats, respectively.

TOR-6 (Evaluate stock status). The estimated fishing mortality in 2013 was 0.32, which was below both the previous and new F_{MSY} estimates (0.38 and 0.48, respectively). The estimated biomass in 2013 is 132,561 mt meats. The stock is considered overfished if the biomass is less than half of B_{MSY}. B_{MSY} was estimated as 125,358 in the previous assessment and 96,480 mt meats in this assessment. Thus, the 2013 stock biomass was above both B_{MSY} estimates. Therefore, it can be concluded that the sea scallop stock was neither overfished nor was overfishing occurring in 2013, regardless of whether the previous or new reference points are used.

TOR-7 (Projections) Projections were conducted using the SAMS (Scallop Area Management Simulator), which models scallops on a relatively fine spatial scale in order to model effects such as closures and reopenings of areas. Example simulations, based on expected management during 2014-2016, predicts gradual increases in biomass and landings.

B3. INTRODUCTION AND LIFE HISTORY

The Atlantic sea scallop, *Placopecten magellanicus,* is a bivalve mollusk that occurs on the eastern North American continental shelf from Cape Hatteras to the Gulf of St. Lawrence and Newfoundland. Major aggregations in US waters occur in the Mid-Atlantic from Virginia to Long Island, on Georges Bank, in the Great South Channel, and, to a lesser extent, in the Gulf of Maine (Hart and Chute 2004). In Georges Bank and the Mid-Atlantic, sea scallops are harvested primarily at depths of 30 to 100 m, whereas the bulk of landings from the Gulf of Maine are from near-shore waters. This assessment focuses on the two main portions of the sea scallop stock and fishery, Georges Bank in the north and the Mid-Atlantic in the south (Figure B3.1). Results for Georges Bank and the Mid-Atlantic are combined to evaluate the stock as a whole.

US landings during 2003-2012 exceeded 24,000 mt each year, roughly twice the long-term mean, but declined to 18,641 mt in 2013.2 US ex-vessel sea scallop revenues were over $500 million in 2011-2012 and $465 million in 2013, making the sea scallop fishery the most valuable fishery in the US during these years. Unusually strong recruitment in the Mid-Atlantic Bight area and increased yield per recruit due to effort reduction, area rotation, and gear restrictions were the key contributors to high landings during the most recent period. The drop off in 2013 reflects weaker recruitment in the Mid-Atlantic during 2009-2011 (2007-2009 year classes). The mean meat weight of landed scallops was over 25 g after 2005 (when the Amendment 10 management plan went into effect), compared to less than 14 g during the early to mid 1990s.

Access area closures and openings used for rotational fishery management have had a strong influence on sea scallop population dynamics (Figure B3.1). Roughly 40% of the productive scallop grounds on Georges Bank and Nantucket Shoals were closed to both groundfish and scallop gear during most of the time since December 1994. Portions of the closed areas have been reopened to limited fishing during 1999-2000 and since 2004. In the Mid-Atlantic, there have been four rotational scallop areas. These areas are generally closed for two to three years, and then reopened to allow harvesting. The areas are closed again after observations of strong recruitment until the small scallops grow to fishable size.

Sea scallops in U.S. waters have been assessed using forward projecting size-structured models since 2007. Fishing mortality, biomass and recruitment are estimated using a version of the CASA (Catch-At-Size Analysis) model based loosely on Sullivan et al. (1990). Forecasts are done using the SAMS (Scallop Area Management Simulator) model, which models the scallop fishery and population on a relatively fine regional scale, in order to help understand the effects of area management such as closing and reopening areas to fishing. Reference points are calculated using the SYM model (Stochastic Yield Model, Hart 2013). All of these models were specifically developed for use with sea scallops.

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2 In this assessment, landings and biomass are reported in metric tons (mt) of scallop meats, unless otherwise indicated.

Life History and Distribution

Sea scallops are found in the Northwest Atlantic Ocean from North Carolina to Newfoundland along the continental shelf typically on firm sand and gravel bottoms (Hart and Chute 2004). Sea scallops feed by filtering phytoplankton, microzooplankton, and detritus particles. Sea scallops are broadcast spawners with separate sexes. Sea scallops mature at about age 2 (~40-75 mm SH[3]), but gamete production is limited until age 4. Larvae are planktonic for 5-8 weeks before settling to the bottom. Scallops fully recruit to the NEFSC lined dredge survey at 40 mm SH, and to the current commercial fishery at around 90-105 mm SH, although sea scallops between 70-90 mm were common in landings prior to 2000.

According to Amendment 10 of the Atlantic Sea Scallop Fishery Management Plan, all sea scallops in the US EEZ belong to a single stock but there are two principal stock assessment regions (Mid-Atlantic and Georges Bank). The US sea scallop stock can be divided into Georges Bank, Mid-Atlantic, Southern New England, and Gulf of Maine regional components based on survey data, fishery patterns, and other information (NEFSC 2004, Figure B3.1). However, Southern New England is considered to be part of the Georges Bank region for assessment modeling purposes. Most of the scallops in the Gulf of Maine lie in state waters, and are managed by the states of Maine and Massachusetts. See Appendix B7 for an assessment of sea scallops in the Northern Gulf of Maine federal management area.

Growth

Sea scallop growth can be inferred using visible "rings" laid down on the shell. These rings have been confirmed as annual marks, although the year one ring is typically missing (Stephenson and Dickie 1954, Merrill et al. 1966, Hart and Chute 2009a, Chute et al. 2012). Studies in Canadian waters indicated that the rings are laid down during the winter (Stephenson and Dickie 1954, Tan et al. 1988) but a recent stable isotope study showed that the rings from scallops in US waters are laid down near the temperature maximum, likely coinciding with the fall spawn (Chute et al. 2012).

Obtaining absolute age from shell rings can be problematic for some scallops because the first few rings may be missing or obscure, especially on older scallops (Claereboudt and Himmelman, 1996). For this reason, Hart and Chute (2009b) treated the distance between rings as annual growth increments, with age unknown. They introduced a method to estimate von Bertalanffy growth parameters from such data which includes random effects on both L_∞ and K to take into account variation in growth among individuals. This method gives estimates of mean von Bertalanffy coefficients and the variance of these parameters among individuals in the population. These parameters and variances are used to estimate growth transition matrices for CASA. The von Bertalanffy parameter t_0 cannot be estimated using growth increments, but estimates of this parameter are not required in a size-structured assessment.

The growth estimates in Hart and Chute (2009b) were based on scallops collected between 2001 and 2007. NEFSC (2010) added additional data from shells collected in 2008. New data from

3 Scallop body size is measured as shell height (SH), the maximum distance between the umbo and shell margin.

shells collected during 1988, 1993 and 2009-2012 are used in this assessment (Table B3-1). Growth on Georges Bank showed little temporal variability during the 2001-2012 time period, but the shells collected in 2010-2012 in the Mid-Atlantic appear to growth slightly faster than those from 2001-2009 (Figure B3.2).

Scallop growth during 1988 and 1993 was substantially slower than in recent years (Figure B3.3). A comparison of the growth increments from these years to 2001-2012 indicate little difference between these periods for scallops less than 76 mm, the ring size for commercial dredge gear before 1994 (Figure B3.3). However, there appears to have been less and less fast growing scallops as shell height increased. This pattern is consistent with preferential removal of faster growing scallops by the fishery. In part, this may be due to a "Lee's effect", where the faster growing scallops recruit earlier to the fishery and die sooner. However, spatial fishery patterns likely also play a role because areas containing faster growing scallops were likely fished harder. Similarly, commercial-sized scallops in the Georges Bank closed areas grow faster and have a greater asymptotic size than in the areas opened to fishing (Table B3-1; Figure B3.2; Hart and Chute 2009b).

Maturity and fecundity

Scallops reach sexual maturity at about age 2. Sea scallops > 40 mm SH are reliably detected in surveys used in this assessment and are all considered mature individuals. Thus biomass estimates for scallops 40+ mm in this assessment are effectively spawning biomass estimates. However, individuals younger than 4 years may contribute little to total egg production because fecundity increases rapidly with age (MacDonald and Thompson 1985; NEFSC 1993).

Sea scallop spawning generally occurs in late summer or early autumn throughout their range. Spring spawns and minor "dribble" spawns may also occur at other times. The spring spawn is often strong in the Mid-Atlantic Bight (DuPaul et al. 1989). Spring spawns on Georges Bank are less substantial but may be increasing in strength with warmer winter water temperatures (Almeida et al. 1994, Dibacco et al. 1995, Thompson et al. 2014). Out of 14 scallops (6 from Georges Bank and 8 from the Mid-Atlantic) analyzed by stable isotopes, only one, from Delmarva in the southern Mid-Atlantic, was found to be spring-spawned, while the others were fall spawned (Chute et al. 2012). No assumption regarding timing of spawning is made in this assessment, as it is not required for size-structured models.

Shell height/meat weight relationships

Shell height-meat weight relationships allow conversion from numbers of scallops at a given size to meat weights. For sea scallops $W=\exp(\alpha+\beta\ln(H))$, where W is meat weight in grams and H is shell height in mm (Appendix B3). Meat weights depend on factors which affect feeding and metabolic rates, including depth and location. Meat weights decrease with depth, probably because of reduced food (phytoplankton) supply.

Shell height/meat weight data were collected during annual NEFSC sea scallop surveys during 2001-2013. Unlike previous studies, where meats were either frozen or brought in live and then weighed on land, meats were weighted at sea just after they were shucked (Hennen and Hart

2012). These data have been used in scallop assessments since 2007, and were updated for this assessment (Appendix B3).

Depth and subarea had a significant effect on the shell height/meat weight relationships (Appendix B3). In this assessment, covariate-adjusted shell height/meat weight relationships were used to calculate survey biomass, while simple relationships (depth omitted) were used in modeling (CASA, SAMS and SYM) where depth is not explicit (Table B3.2).

Meat weights for scallops in the commercial fishery may differ from those predicted from research survey data for a number of reasons. First, the shell height-meat weight relationship varies seasonally, in part due to the reproductive cycle, so that meat weights collected during the NEFSC survey in July and August may differ from those in the rest of year (Hennen and Hart 2012). Additionally, commercial fishers concentrate on speed, and often leave some meat on the shell during shucking (Naidu 1987, Kirkley and DuPaul 1989). On the other hand, meats in fishery catches may gain weight due to water uptake during storage on ice (DuPaul et al. 1990). Finally, fishers may target areas with relatively large meat weight at shell height, and thus may increase commercial meat weights compared to that collected on the research vessel.

Observer data were used to adjust predicted meat weights based on survey data for seasonal variation and for commercial fishing practices. Annual commercial meat weight anomalies were computed based on the seasonal patterns of landings together with the mean monthly commercial meat weight at shell height. The average annual meat weight anomalies are used in assessment modeling to calculate fishery meat weights.

Shell height/Meat weight relationships

	Mid-Atlantic		Georges Bank	
	a	B	a	b
NEFSC (2014)	-9.33	2.66	-8.79	2.55
NEFSC(2014) , open areas			-9.37	2.65
NEFSC (2014), closed areas			-8.26	2.45
Hennen Hart (2012)/NEFSC 2010	-10.8	2.97	10.25	2.85
Lai and Helser (2004)	-12.34	3.28	11.44	3.07
Serchuk and Rak (1983)	-12.16	3.25	-11.77	3.17
Haynes (1966)	-11.09	3.04	-10.84	2.95

Natural mortality

Assessments prior to 2010 assumed a natural mortality rate of $M = 0.1$ based on Merrill and Posgay (1964). A reanalysis of the Merrill and Posgay study indicated that an unbiased estimate for M was approximately 0.12 (NEFSC 2010), with a corresponding estimate in the Mid-Atlantic of 0.15. Hart et al. (2013) estimated M within the CASA stock assessment model as 0.16 in the Georges Bank closed areas.

No direct estimate of M is available for Mid-Atlantic sea scallops. The ratio of the growth coefficient K to M is generally regarded as a life history invariant that should be approximately constant for similar organisms (Beverton and Holt 1959, Chernov 1993). Applying this idea and using updated growth parameter estimates indicates that sea scallop natural mortality in the Mid-Atlantic should be about 0.53/0.44 that of Georges Bank (see the estimates of growth coefficients above). Using $M = 0.16$ in Georges Bank, M is about 0.2 in the Mid-Atlantic. These are the estimates used in this assessment for all but the largest size group (plus group).

MacDonald and Thompson (1986) directly observed sea scallop natural mortality in a near-shore population off of Newfoundland. They found that mortality was low from 60-130 mm SH, but increased substantially for scallops larger than 130 mm. A large cohort of 2 year old scallops (1997 cohort) was observed in 1999 at a station in the Nantucket Lightship Closed Area in an area where recruitment is rare and sporadic and which has been closed to scallop fishing since 1994. A second, smaller cohort of 2 year olds was observed there in the 2000 survey, but almost no recruitment has been observed at this site since. This station has been sampled using the NEFSC survey dredge every year since 2003. The catches at this station indicate low mortality until the dominant 1997 cohort reached 11 years old, after which numbers caught declined substantially. Both these studies thus suggest that natural mortality of very old scallops may be higher than younger ones. Likelihood profiles from the Georges Bank closed CASA model, discussed in section 6, suggest the mortality of the plus group is most likely about 1.5 times that of smaller scallops. Therefore, for this assessment, the plus group natural mortality was assumed to be 1.5 times that of smaller sizes, (0.24 on Georges Bank and 0.3 in the Mid-Atlantic).

MacDonald and Thompson (1986) observed scallops as old as 19 years. The oldest observed in the NEFSC age and growth program are at least 18 years old on Georges Bank and 15 years old in the Mid-Atlantic. These oldest ages are consistent with the natural mortality assumptions given above.

Table B3.1. Regional von Bertalanffy growth parameter estimates from mixed-effects models for sea scallops. SD L_∞ and SD K are the estimated standard deviation of these parameters among individuals in the population.

Source	Region	Years	L_∞	SE	K	SE	SD L_∞	SD K
NEFSC (2014)	Mid-Atlantic	2010-2012	138.0	0.5	0.522	0.005	12.7	0.05
	Mid-Atlantic	2001-2009	131.7	0.3	0.535	0.003	13.6	0.13
	Mid-Atlantic	1988,1993	118.9	2	0.551	0.02	20.8	0.15
	Georges Bank (All)	2001-2012	144.0	0.2	0.44	0.002	13.9	0.11
	Georges Bank (All)	1988,1993	133.4	1.4	0.498	0.013	9.2	0.09
	Georges Bank (Closed)	2001-2012	147.6	0.3	0.426	0.002	12.8	0.11
	Georges Bank (Open)	2001-2012	137.4	0.3	0.442	0.002	11.4	0.11
NEFSC (2010)	Mid-Atlantic	2001-2008	132.1	0.3	0.527	0.004	13.3	0.14
	Georges Bank	2001-2008	144.0	0.3	0.429	0.002	14.5	0.11
Hart & Chute (2009)	Mid-Atlantic	2001-2007	133.3	0.4	0.508	0.004	13.4	0.13
	Georges Bank (All)	2001-2007	143.9	0.3	0.427	0.002	14.8	0.11
	Georges Bank (Open)	2001-2007	136.3	0.5	0.457	0.004	15.1	0.12
	Georges Bank (Closed)	2001-2007	147.8	0.3	0.413	0.003	13.2	0.1
Serchuk et al (1979)	Mid-Atlantic	?	151.8		0.300			
	Georges Bank	?	152.5		0.337			

Table B3.2. Simple shell height-meat weight relationships $W=\exp(\alpha+\beta\ln(H))$ for sea scallops. W is meat weight in grams and H is shell height in mm.

	Mid-Atlantic		Georges Bank	
	a	b	a	b
NEFSC (2014)	-9.33	2.66	-7.46	2.61
NEFSC(2014) , open areas			-9.37	2.65
NEFSC (2014), closed areas			-8.26	2.45
Hennen & Hart (2012)/NEFSC 2010	-10.8	2.97	10.25	2.85
Lai and Helser (2004)	-12.34	3.28	11.44	3.07
Serchuk and Rak (1983)	-12.16	3.25	-11.77	3.17
Haynes (1966)	-11.09	3.04	-10.84	2.95

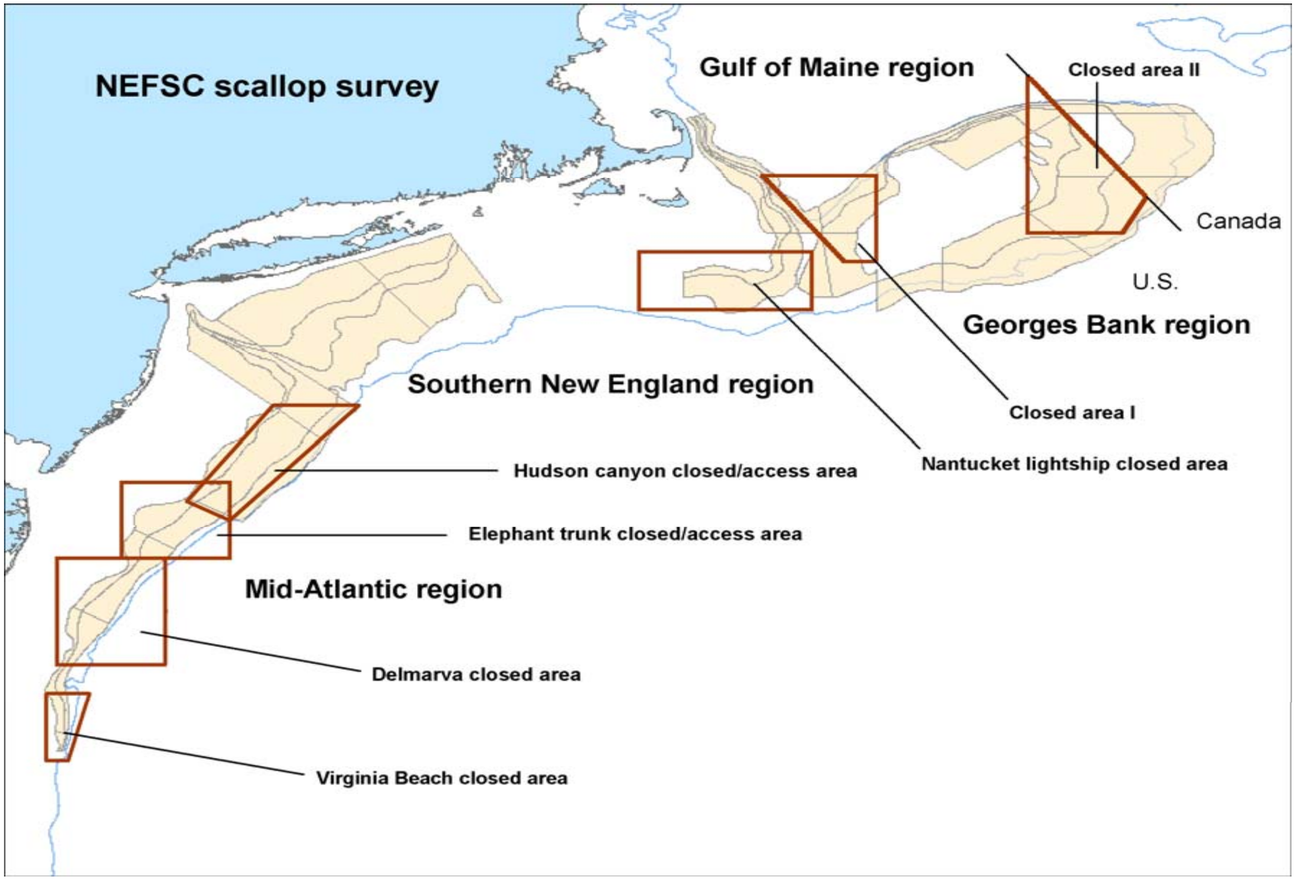


Figure B3.1 Stock assessment and management areas for sea scallops in US waters. The NEFSC scallop survey strata shown in yellow are the areas that are regularly surveyed by the NEFSC dredge survey, which have with appreciable scallop densities.

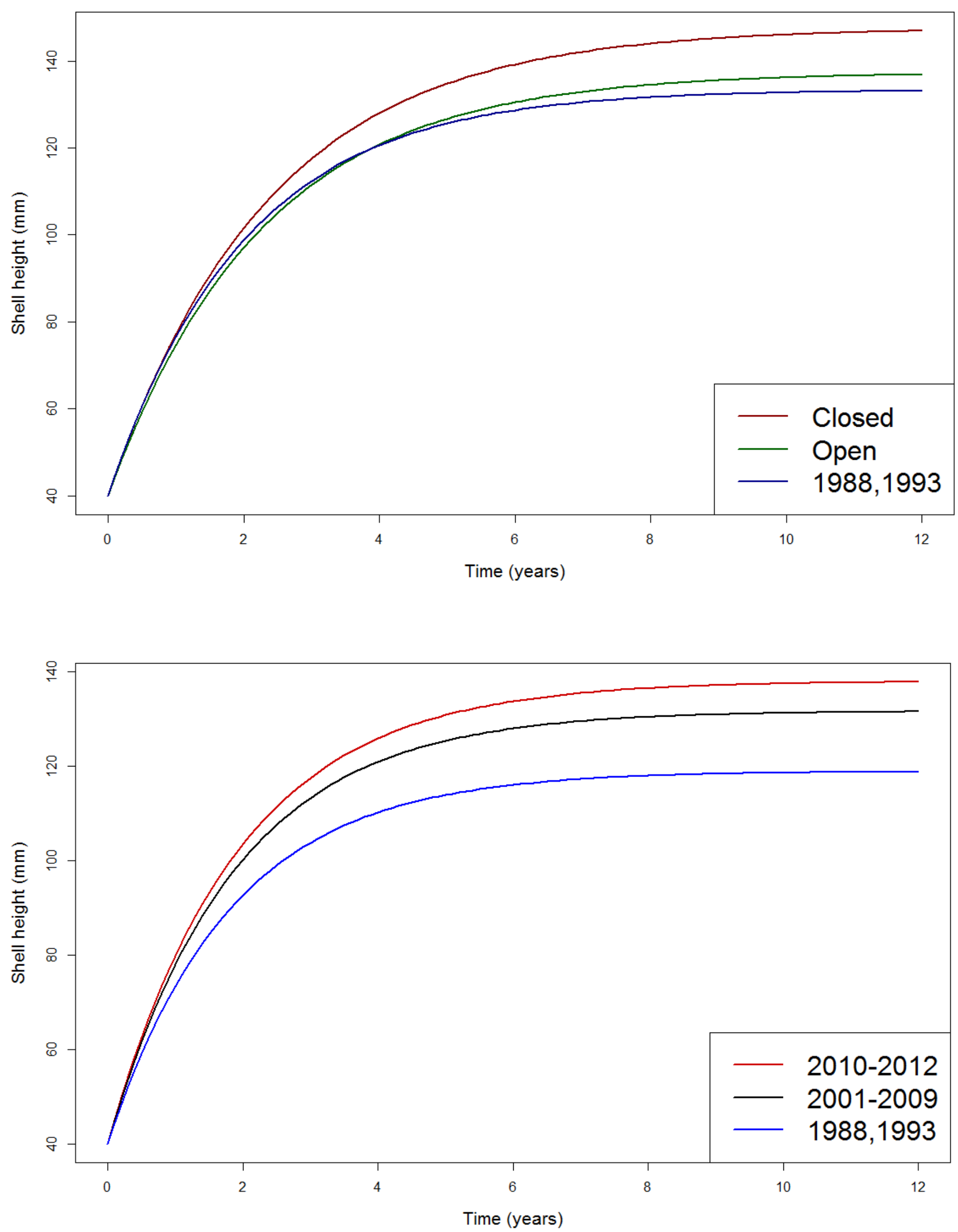


Figure B3.2. Growth curves for sea scallops in the Georges Bank (top) and Mid-Atlantic regions (bottom) for various areas and time periods. The Georges Bank open and closed area growth curves were based on shells collected between 2001-2012.

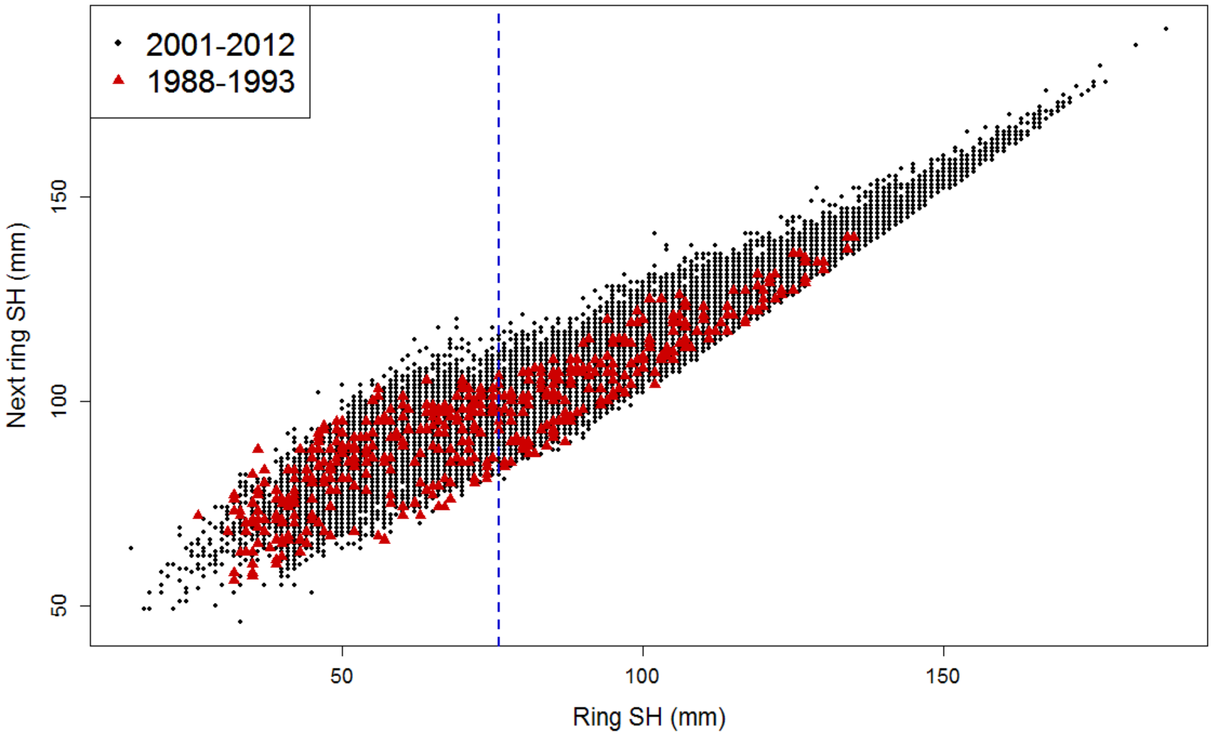


Figure B3.3. Comparison of growth increments from shells collected on Georges Bank between 2001 and 2012 and those collected which fishing effort was much higher (1988,1993). The dashed blue line is at 76 mm, the diameter of most commercial dredge rings prior to 1994.

B4. COMMERCIAL AND RECREATIONAL CATCH (TOR-1)

The US sea scallop fishery is conducted mainly by about 350 vessels with limited access permits. Two types of allocations are given to each limited access vessel. The first is a number of trips to rotational access areas that had been closed to scallop fishing in the past (with a trip limit, typically 12,000-18,000 lbs or 5,443-8,165 kg meats). The second is days at sea (DAS), which can be used in areas outside the closed and access areas. Vessels fishing under days at sea allocations are restricted to a 7 man crew and must shuck their scallops at sea in order to limit their processing power.

The remainder of landings come from vessels operating under "General Category" permits that are currently restricted to 272 kg meats (600 lbs) per trip, with a maximum of one trip per day. Landings from these vessels were less than 1% of total landings in the late 1990s, but increased to about 10% of landings during 2007-2009, and currently constitute about 6-7% of total landings. This type of permit had been open access, but was converted to an individual transferable quota (ITQ) fishery in March 2010.

Principal ports in the sea scallop fishery are New Bedford, MA, Cape May, NJ, and Hampton Roads, VA, but lesser amounts of scallops are landed in many ports from North Carolina to Maine. Toothless offshore (New Bedford style) scallop dredges are the main gear type in all regions, although some scallop fishing is done with otter trawls in the Mid-Atlantic, and a small fraction of the catch in the Gulf of Maine comes from divers. A typical limited access vessel tows two 4-4.6 m dredges, but some limited access vessels are restricted to a single 3.2 m dredge, and most general category vessels also use a single smaller dredge. Recreational catch is negligible.

Management history

The sea scallop fishery in the US EEZ is managed under the Atlantic Sea Scallop Fishery Management Plan (FMP) which was implemented on May 15, 1982. From 1982 to 1994, the primary management control was a minimum average meat weight requirement for landings. In 1984, Georges Bank was divided into US and Canadian EEZs; prior to this time, US and Canadian vessels fished on both sides of the current boundary.

FMP Amendment 4 (NEFMC 1993), implemented in 1994, changed the management strategy from meat count regulation to limited access combined with effort control and gear regulations. Limited access permits were issued to vessels with a history in the fishery; no new permits have been issued since. Incremental restrictions were made on days-at-sea (DAS), minimum ring size, and crew limits; DAS has been reduced from over 200 in 1994 to 31 in open areas in 2014. The minimum size of the rings in the dredge bag was gradually been increased from 76 mm in 1994 to 102 mm since December 2004. The minimum size of the twine top mesh has also been gradually increased from 6” to 10” since December 2004; while this measure was intended mainly to allow better escapement of finfish, it also likely improves the escapement of small scallops.

In addition to these measures, three large areas on Georges Bank and Nantucket Shoals were closed to groundfish and scallop fishing in December 1994 (Figure B3-1). Scallop biomass rapidly increased in these areas between 1994-2004 (Hart and Rago 2006). Two areas in the Mid-Atlantic were closed to scallop fishing in April 1998 for three years in order to similarly increase scallop biomass and mean weight.

Sea scallops were formally declared overfished in 1997, and Amendment 7 was implemented during 1998 with more stringent days-at-sea limitations and a mortality schedule intended to rebuild the stocks within ten years. Subsequent analyses considering effects of closed areas indicated that the stocks would rebuild with less severe effort reductions than called for in Amendment 7, so the days at sea limitations were relaxed. A combination of the closures, effort reduction, gear and crew restrictions led to a rapid increase in biomass (Hart and Rago 2006), and sea scallops were rebuilt by 2001.

Prior to 2004, there were a number of ad hoc area management measures, including the Georges Bank and Mid-Atlantic closures in 1994 and 1998, limited reopenings of portions of the Georges Bank closed areas between June 1999 and January 2001, and reopening of the first Mid-Atlantic rotational areas in 2001. A new set of regulations was implemented as Amendment 10 during 2004. This amendment formalized an area based management system, with provisions and criteria for new rotational closures, and separate allocations (DAS or TACs) for reopening closed areas (rotational areas) and general open areas. The three Georges Bank closed areas have been divided into access areas, where fishing is periodically permitted, and long-term closures, where no scallop fishing is permitted (Figure B3.1). In most years, one or two of the three Georges Bank access areas are open to fishing, limited by a fixed number of trips and a trip limit.

Unlike the Georges Bank closed areas, which are generally closed to all scallop and groundfish fishing, the Mid-Atlantic rotational areas are specific to the scallop fishery (Figure B3.1). Two areas (Hudson Canyon South and Virginia Beach) were closed in 1998 and then reopened in 2001. Although the small Virginia Beach closure in the far south of the scallops' range was unsuccessful, scallop biomass built up in Hudson Canyon Closed Area while it was closed, and substantial landings were obtained from Hudson Canyon during 2001-2007. This area was again closed in 2008, reopened in 2011 and closed for a third time in 2014. A third rotational closure, the Elephant Trunk area east of Delaware Bay, was closed in 2004 after extremely high densities of small scallops were observed in surveys during 2002 and 2003. About 30,000 mt of scallops worth about $500 million were landed from that area after it was reopened in 2007. It was closed again in December 2012 after high numbers of small scallops were again observed in surveys. A fourth closed area, Delmarva, directly south of the Elephant Trunk area, was closed in 2007, reopened in 2009, closed in 2012 and reopened in 2014.

Landings

Sea scallop landings in the US increased substantially after the mid-1940's (Figure B4.1), with peaks occurring around 1960, 1978, 1990, and 2004. Maximum landings were 29,109 mt meats in 2004. Landings during 2001-2012 were all over 20,000 mt, whereas the maximum in the 20th century was 17,107 mt in 1990. Landings in 2013 were 18,641 mt, their lowest since 2000, but still higher than any year prior to 2001.

Landings from the Georges Bank and the Mid-Atlantic regions have dominated the fishery since 1964 (Table B4-1; Figure B4.2). Proration of total commercial sea scallop landings into Georges Bank, Mid-Atlantic, Southern New England, and Gulf of Maine used standard allocation procedures (Wigley et al. 2008).

US Georges Bank landings had peaks during the early 1960's, around 1980 and 1990, but declined precipitously during 1993 and remained low through 1998 (Figure B4-2). Landings in Georges Bank during 1999-2004 were fairly steady, averaging almost 5000 mt annually, and then increased in 2005-2006, primarily due to reopening of portions of the groundfish closed areas to scallop fishing. Georges Bank landings increased again in 2012-2013, mainly due to shift of "open" effort from the Mid-Atlantic to Georges Bank

Prior to the mid-1980s, Mid-Atlantic landings were generally lower than those on Georges Bank. Mid-Atlantic landings during 1962-1982 averaged less than 1800 mt per year (Figure B4.2). An upward trend in both recruitment and landings has been evident in the Mid-Atlantic since the mid-eighties. Landings peaked in 2004 at 24,494 mt. Mid-Atlantic landings declined after 2011, reflecting the poor 2007-2009 year classes there and concomitant effort shifts onto Georges Bank.

Landings from other areas (Gulf of Maine and Southern New England) are minor in comparison (Table B4-1). Most of the Gulf of Maine stock is assessed and managed by the State of Maine because it is primarily in state waters. However, the Nothern Gulf of Maine management area is managed by the New England Fishery Management Council with separate regulations (see Appendix B7 for an updated assessment). Gulf of Maine landings were less than 1% of the total US sea scallop landings in most recent years. Maximum landings in the Gulf of Maine were 1,614 mt during 1980.

Fishing effort and LPUE

Prior to 1994, landings and effort data were collected during port interviews by port agents which was combined with dealer data. Since 1994, commercial data are available in dealer reports (DR) and in vessel trip report (VTR) logbooks. DR give landings, but not area fished, and have reported landings by market category since 1998. VTR data contain information about area fished, fishing effort, and retained catches of sea scallops. Ability to link DR and VTR reports in data processing is reduced by incomplete data reports and other problems, although there have been significant improvements recently. A standardized method (Wigley et al. 2008) for matching DR to VTRs and assigning landings to fishing areas was used in this assessment for 1994-2013.

Landings per unit effort (LPUE, Figure B4.3) was computed as landings per day fished (days fished represent the time in days that gear was fishing). This was obtained from the port interview records for larger vessels prior to 1994 and from at-sea observers on limited access vessels afterwards. LPUE shows a general downward trend from the beginning of the time series to around 1998, with occasional spikes upward due to strong recruitment events. LPUE increased considerably since then as the stock recovered. Note the close correspondence in most years between the LPUE in the Mid-Atlantic and Georges Bank, probably reflecting the mobility of the scallop fishing fleet; if one area has higher catch rates, it is fished harder until the rates are equalized. Although comparisons of LPUE before and after the change in data collection procedures need to be made cautiously, there is no clear break in the LPUE trend in 1994.

Fishing effort (days fished) was computed as the product of LPUE and landings (Figure B4.4). This effort metric reflects the days fished that would have been required to obtain the reported total landings with limited access vessels. General category vessels, which usually fish with one small dredge would likely fish for several days to account for a single "day fished" of effort. Effort in the US sea scallop fishery generally increased from the mid-1970s to about 1991, and then decreased during the 1990s, first because of low catch rates, and later as a result of effort reduction measures. Effort increased in the Mid-Atlantic during 2000-2005, initially due to reactivation of latent effort among limited access vessels, and then to increases in general category effort. Total effort since 2005 has remained fairly stable, although there have been shifts between regions.

Discards and discard mortality

Sea scallops are sometimes discarded on directed scallop trips because they are too small to be economically profitable to shuck, or because of high-grading, particularly during access area trips (Figure B4.5). Ratios of discard to total catch (by weight) were recorded by sea samplers aboard commercial vessels since 1992 and used to estimate discarded scallops (Appendix B2). Sampling intensity on non-access area trips was low until 2003.

Discarded sea scallops may suffer mortality on deck due to crushing, high temperatures, or desiccation. There may also be mortality after they are thrown back into the water from physiological stress and shock, or from increased predation due to shock and inability to swim or shell damage (Veale et al. 2000, Jenkins and Brand 2001). Murawski and Serchuk (1989) estimated that about 90% of tagged scallops were still living several days after being tagged and placed back in the water. Total discard mortality of discarded scallops (including mortality on deck) is uncertain but has been estimated as 20% in previous assessments (e.g., NEFSC 2010). However, discard mortality may be higher during the Mid-Atlantic during the summer due to high water and deck temperatures, and likely strongly depends in both regions on fishing practices. Scallops returned to the water promptly have much higher chances of survival than ones left on deck for longer periods.

Incidental mortality

Scallop dredges likely kill and injure some scallops that are contacted by the gear but not caught, primarily due to damage (e.g., crushing) to the shells by the dredge. Caddy (1973) estimated that

15-20% of the scallops remaining in the track of a dredge were killed. Murawski and Serchuk (1989) estimated that less than 5% of the scallops remaining in the track of a dredge suffered non-landed mortality. Caddy's study was done in a relatively hard bottom area in Canada, while the Murawski and Serchuk study was in sandy bottom off the coast of New Jersey. It is possible that the difference in indirect mortality estimated in these two studies was due to different bottom types (Murawski and Serchuk 1989).

In order to use these studies to relate landed and non-landed fishing mortality in stock assessment calculations, it is necessary to know the efficiency e of the dredge (the probability that a fully recruited scallop in the path of a dredge is captured). Denote by c the fraction of scallops that suffer mortality among sea scallops in the path of the dredge but not caught. The best available information indicates that $c = 0.15$-0.2 (Caddy 1973), and $c < 0.05$ (Murawski and Serchuk 1989). The ratio R of scallops in the path of the dredge that were caught, to those killed but not caught is:

$$R = e/[c(1-e)] \qquad (4.1)$$

If scallops suffer direct (i.e., landed) fishing mortality at rate F_L, then the rate of indirect (non-landed) fishing mortality will be (Hart 2003):

$$F_I = F_L / R = F_L \, c \, (1-e)/e. \qquad (4.2)$$

If, for example, the commercial dredge efficiency e is 50%, then $F_I = F_L \, c$, where F_L is the fully recruited fishing mortality rate for sea scallops. Assuming $c = 0.15$ to 0.2 (Caddy 1973) gives $F_I = 0.15 \, F_L$ to $0.2 \, F_L$. With $c < 0.05$ (Murawski and Serchuk 1989) $F_I < 0.05 \, F_L$. For this assessment, incidental mortality was assumed to be $0.2 \, F_L$ in Georges Bank and $0.1 \, F_L$ in the Mid-Atlantic.

Prior assessments applied the incidental mortality F_I from equation (4.2) to all sizes of scallops. However, the observations of Caddy (1973) and Murawski and Serchuk (1989) were in terms of mortality of scallops remaining after a pass of a dredge. Thus, the incidental fishing mortality as a function of shell height h should be:

$$F_{Ih}(h) = F_I \, (1 - q(h)) \qquad (4.3)$$

where q(h) is the catchability of commercial gear on a scallop of shell height h. We took q(h) to be:

$$q(h) = q_0 \, s(h) \qquad (4.4)$$

where q_0 is 0.5 on Georges Bank and 0.6 in the Mid-Atlantic (commercial gear is more efficient on large scallops than the survey dredge, see e.g., Yochum and DuPaul 2008), and s(h) is commercial size selectivity estimated by the CASA model. All of these calculations take place in the assessment model itself.

Commercial shell height data

Since most sea scallops are shucked at sea, it has sometimes been difficult to obtain reliable commercial size compositions. Port samples of shells brought in by scallopers have been collected, but there are questions about whether the samples were representative of the landings and catch. Port samples taken during the meat count era often appear to be selected for their size rather than being randomly sampled, and the size composition of port samples from 1992-1994 differed considerably from those collected by at-sea observers during this same period. For this reason, commercial size compositions from port samples after 1984 when meat count regulations were in force are not used in this assessment.

Sea samplers (observers) have collected shell heights of kept scallops from commercial vessels since 1992, and discarded scallops since 1994. Although these data are likely more reliable than that from port sampling, they still must be interpreted cautiously for years prior to 2003 due to limited observer coverage (except for the access area fisheries, which always have had good observer coverage). Except for 2006, observer coverage rates have been over 5% since 2003, and were over 10% during 2012-2013.

Shell heights from port and sea sampling data indicate that sea scallops between 70-90 mm often made up a considerable portion of the landings during 1975-1998, but sizes selected by the fishery have increased since then, so that scallops less than 90 mm were rarely taken since 2002 (Figure B4.6).

Dealer data (landings) have been reported by market categories (under 10 meats per pound, 10-20 meats per pound, 20-30 meats per pound etc) since 1998 (Figure B4.7). These data also indicate a trend towards larger sea scallops in landings in recent years. While nearly half the landings in 1998 were in the smaller market categories (more than 30 meats per pound), 75% or more of recent landings were below 20 count and about 99% were below 30 count.

Table B4.1. US scallop landings 1964-2013 (mt meats), by region and gear type. Dredge gear was recorded as "other" prior to 1978.

	Gulf of Maine				Georges Bank				S. New England				Mid Atlantic Bight				Total			
Year	dredge	trawl	other	sum	dredge	trawl	other	sum	dredge	trawl	other	sum	dredge	trawl	other	sum	dredge	trawl	other	sum
1964		0	208	208		0	6,241	6,241		52	3	55		0	137	137		52	6,590	6,642
1965		0	117	117		3	1,478	1,481		2	24	26		0	3,974	3,974		5	5,592	5,598
1966		0	102	102		0	883	884		0	8	8		0	4,061	4,061		1	5,055	5,056
1967		0	80	80		4	1,217	1,221		0	8	8		0	1,873	1,873		4	3,178	3,182
1968		0	113	113		0	993	994		0	56	56		0	2,437	2,437		0	3,599	3,599
1969		1	122	123		8	1,316	1,324		0	18	19		5	846	851		14	2,302	2,317
1970		0	132	132		5	1,410	1,415		0	6	6		14	459	473		19	2,006	2,026
1971		4	358	362		18	1,311	1,329		0	7	7		0	274	274		22	1,949	1,971
1972		1	524	525		5	816	821		0	2	2		5	653	658		11	1,995	2,006
1973		0	460	460		15	1,065	1,080		0	3	3		4	245	249		19	1,773	1,792
1974		0	223	223		15	911	926		0	4	5		0	937	938		16	2,076	2,091
1975		6	741	746		13	844	857		8	42	50		52	1,506	1,558		80	3,132	3,212
1976		3	364	366		38	1,723	1,761		4	3	7		819	2,972	3,791		361	5,061	5,422
1977		4	254	258		27	4,709	4,736		1	10	11		255	2,564	2,819		58	7,536	7,595
1978	242	1	0	243	5,532	37	0	5,569	25	2	0	27	4,435	207	0	4,642	10,234	247	0	10,481
1979	401	5	1	407	6,253	25	7	6,285	61	5	0	66	2,857	29	1	2,888	9,572	64	9	9,645
1980	1,489	122	3	1,614	5,382	34	2	5,419	130	3	0	133	2,202	85	79	2,366	9,204	245	83	9,532
1981	1,225	73	7	1,305	7,787	56	0	7,843	68	1	0	69	772	14	2	788	9,852	144	9	10,005
1982	631	28	5	664	6,204	119	0	6,322	126	0	0	126	1,602	6	2	1,610	8,562	153	7	8,723
1983	815	72	7	895	4,247	32	4	4,284	243	1	0	243	3,092	19	10	3,121	8,398	124	21	8,542
1984	651	18	10	678	3,011	29	3	3,043	161	3	0	164	3,695	53	2	3,750	7,518	103	14	7,635
1985	408	3	10	421	2,860	34	0	2,894	77	4	0	82	3,230	49	2	3,281	6,575	90	12	6,677
1986	308	2	6	316	4,428	10	0	4,438	76	2	0	78	3,407	386	6	3,799	8,218	400	12	8,631
1987	373	0	9	382	4,821	30	0	4,851	67	1	0	68	7,639	1,168	1	8,808	12,900	1,199	10	14,109
1988	506	7	13	526	6,036	18	0	6,054	65	4	0	68	6,071	938	8	7,017	12,678	966	21	13,666
1989	600	0	44	644	5,637	25	0	5,661	127	11	0	138	7,894	534	5	8,433	14,258	570	49	14,876
1990	545	0	28	574	9,972	10	0	9,982	110	6	0	116	6,364	541	10	6,915	16,991	558	38	17,587
1991	527	3	75	605	9,235	77	0	9,311	55	16	0	71	6,408	878	14	7,300	16,225	973	89	17,288
1992	676	2	45	722	8,230	7	0	8,238	119	5	0	124	4,562	570	5	5,137	13,587	584	50	14,221
1993	763	2	32	797	3,637	18	0	3,655	65	1	0	66	2,412	393	3	2,808	6,878	413	36	7,327
1994	410	6	9	425	1,182	7	0	1,189	29	1	0	30	5,211	754	0	5,965	6,832	768	9	7,609
1995	342	6	13	361	992	4	1	997	41	2	0	43	5,786	798	7	6,591	7,161	810	21	7,992
1996	544	5	12	561	2,126	7	4	2,137	59	5	0	64	4,467	653	4	5,124	7,196	670	20	7,886
1997	673	5	21	699	2,347	9	1	2,357	81	11	3	95	2,703	378	1	3,082	5,804	403	26	6,233
1998	392	5	15	412	2,045	19	1	2,065	103	3	0	106	2,411	564	6	2,981	4,951	591	22	5,564
1999	267	2	2	271	5,172	6	1	5,179	78	1	0	79	3,629	959	1	4,589	9,146	968	4	10,118
2000	162	21	43	226	4,910	40	5	4,955	85	3	1	89	8,139	1,210	2	9,351	13,296	1,274	51	14,621
2001	335	7	1	343	4,879	58	6	4,943	28	37	0	65	14,144	1,543	16	15,703	19,386	1,645	23	21,054
2002	386	18	1	405	5,967	33	11	6,011	20	12	0	32	15,981	1,426	36	17,443	22,354	1,489	48	23,891
2003	197	3	1	201	4,859	22	2	4,883	53	4	0	57	19,040	1,226	10	20,276	24,149	1,255	13	25,417
2004	165	12	0	177	4,249	146	11	4,406	830	151	11	992	22,313	1,194	26	23,533	27,557	1,503	48	29,108
2005	163	12	12	187	8,958	69	15	9,042	845	13	40	898	14,361	1,096	109	15,566	24,327	1,190	176	25,693
2006	147	3	5	155	15,688	51	21	15,760	2,029	10	8	2,047	7,944	782	46	8,772	25,808	846	80	26,734
2007	97	8	12	117	9,419	45	18	9,482	335	18	7	360	16,234	345	55	16,634	26,085	416	92	26,593
2008	103	12	5	120	6,405	24	11	6,440	303	6	16	325	16,819	556	13	17,388	23,630	598	45	24,273
2009	81	0	3	84	6,451	8	16	6,475	216	1	3	220	17,487	12	1,851	19,350	24,235	21	1,873	26,129
2010	148	13	6	168	5,826	18	47	5,890	254	9	26	290	19,172	281	97	19,550	25,400	321	177	25,898
2011	193	17	2	212	8,159	14	135	8,309	338	24	24	386	17,224	318	205	17,747	25,914	373	366	26,653
2012	392	22	3	417	13,671	37	16	13,724	118	4	32	154	11,172	272	176	11,620	25,353	334	228	25,915
2013	449	43	6	498	11,823	27	25	11,875	308	13	5	326	5,683	229	54	5,966	18,263	311	89	18,664

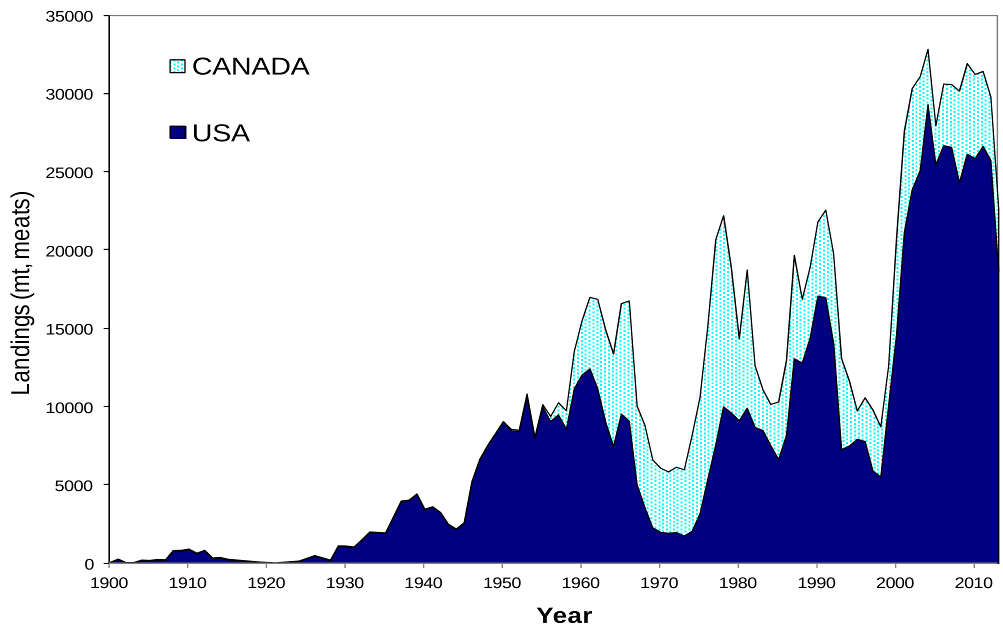


Figure B4.1. Sea scallop landings in NAFO areas 5-6 (North Carolina to Georges Bank).

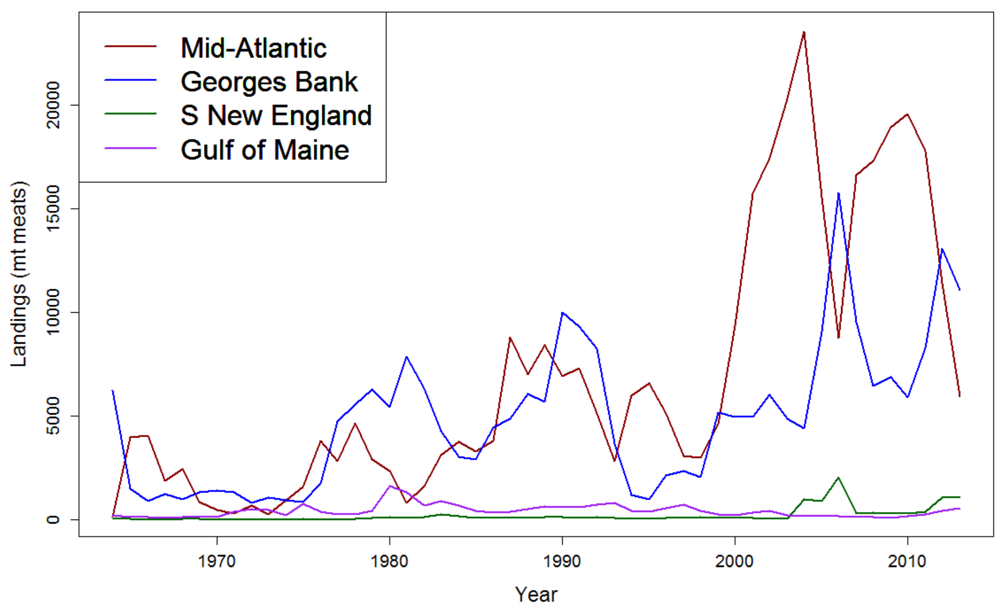


Figure B4.2. US sea scallop landings during 1964-2013, by region.

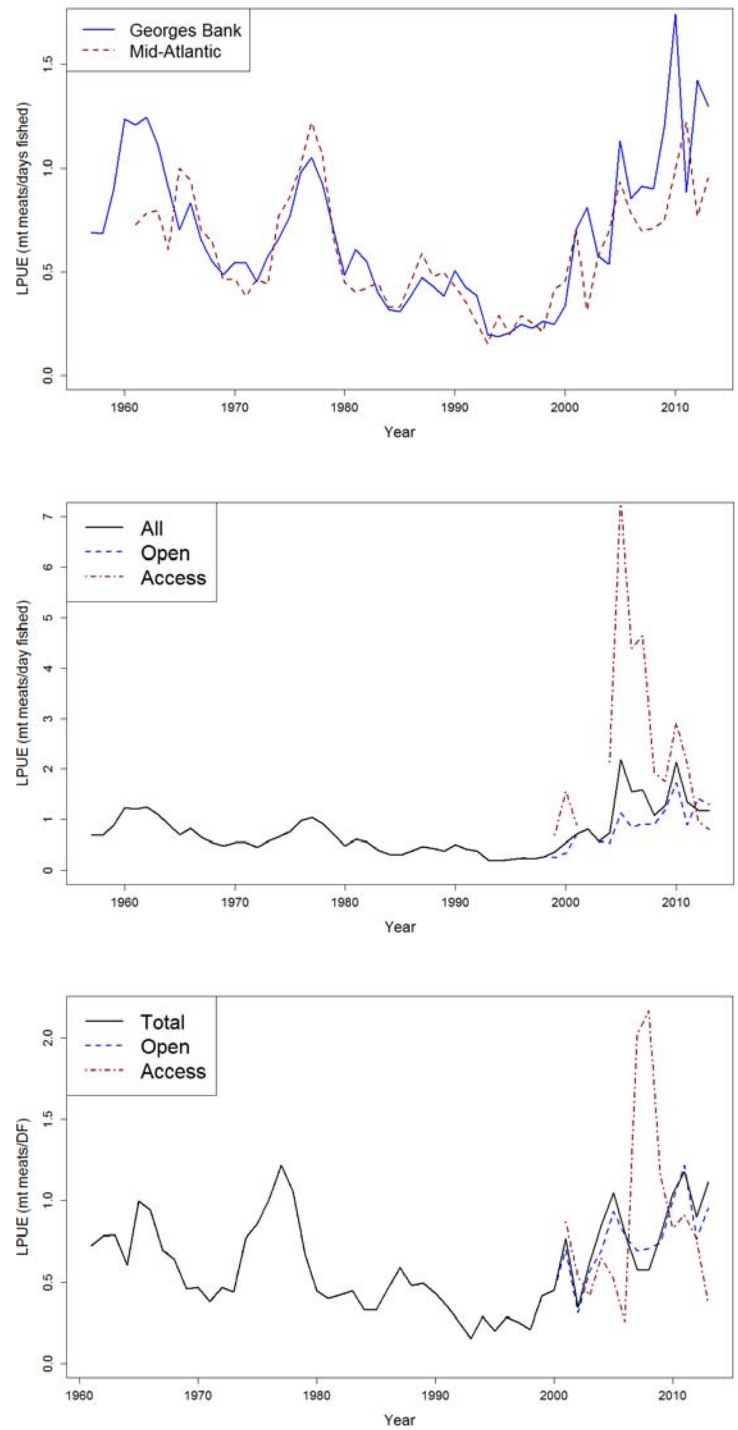


Figure B4.3 *Top*: landings per unit effort (LPUE) on Georges Bank and the Mid-Atlantic, excluding access area trips. *Middle:* LPUE on Georges Bank, separated into access and open areas and combined. *Bottom:* LPUE in the Mid-Atlantic, separated into access and open areas and combined.

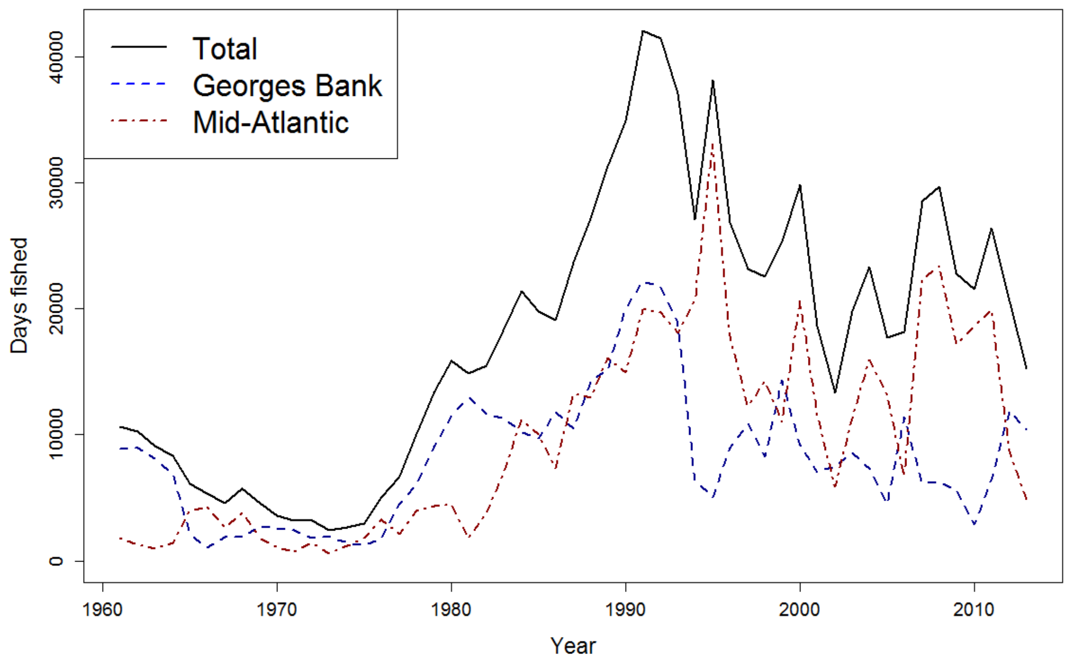


Figure B4.4 Sea scallop fishing effort in the US, 1961-2013.

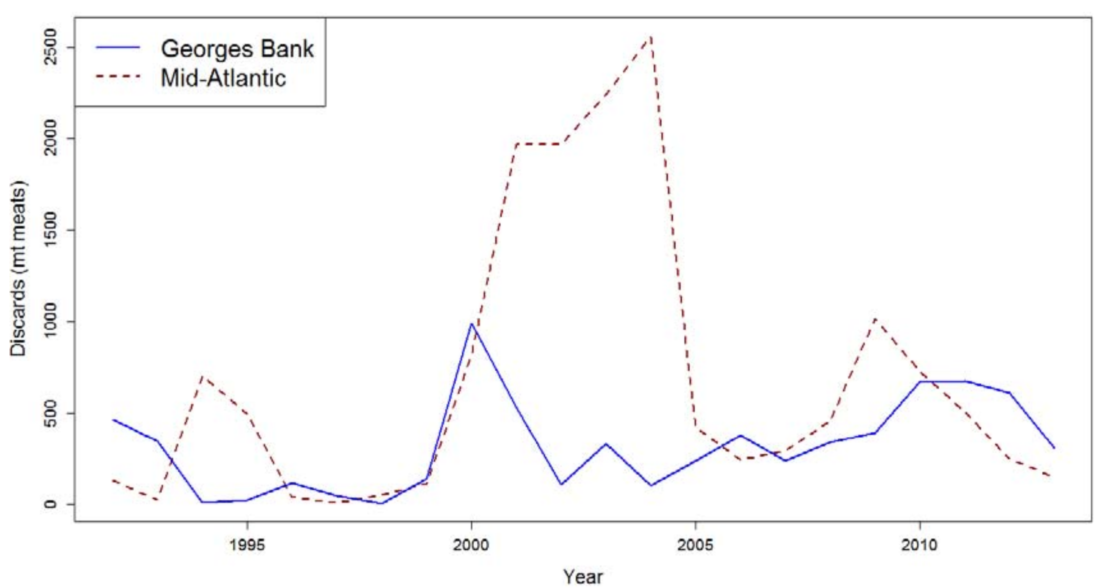


Figure B4.5. Estimated discards in the US scallop fishery, 1992-2013.

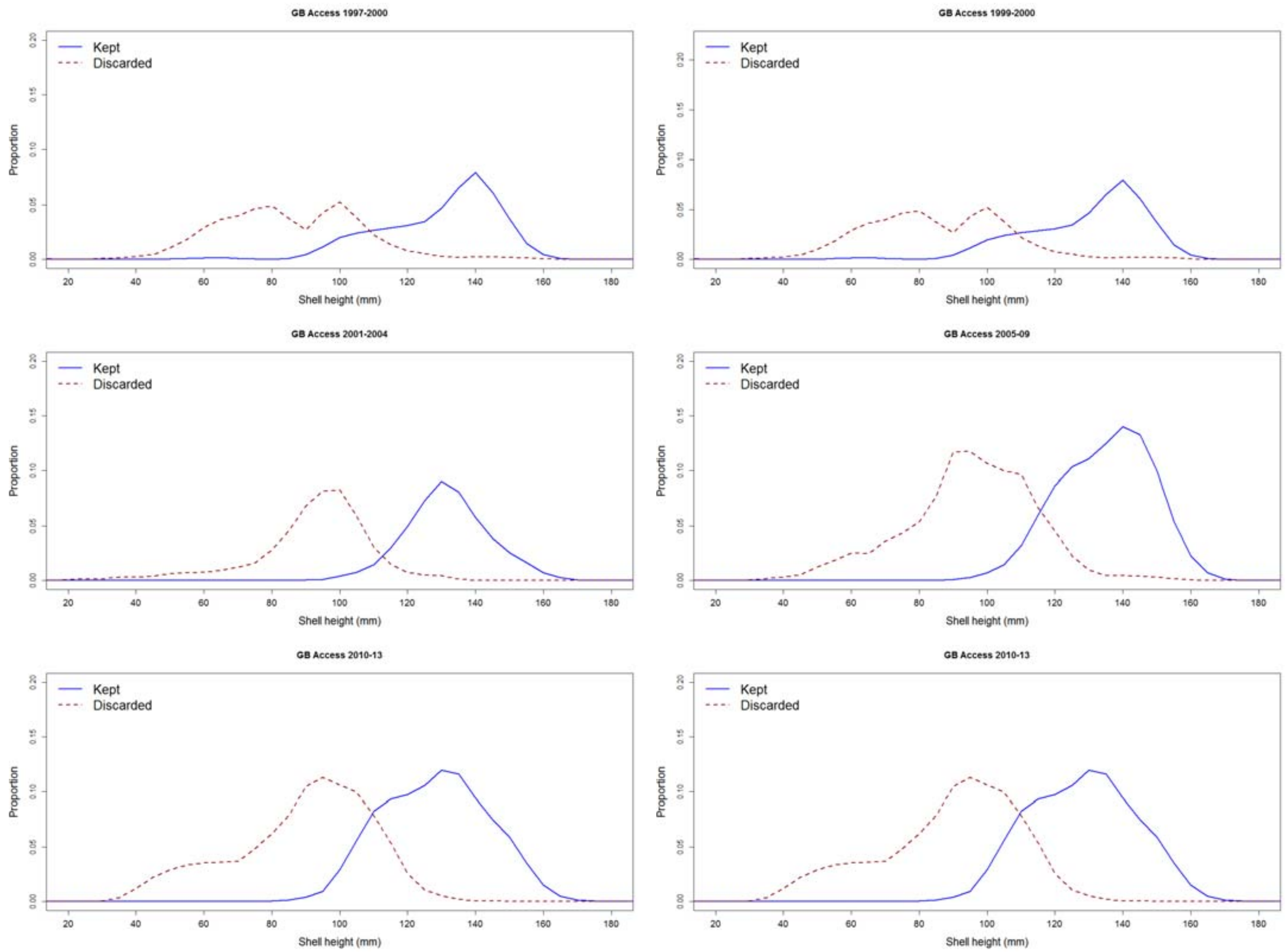


Figure B4.6. Shell heights of commercial kept (solid line) and discarded (dashed line) sea scallops from Georges Bank access areas, based on data from sea samplers.

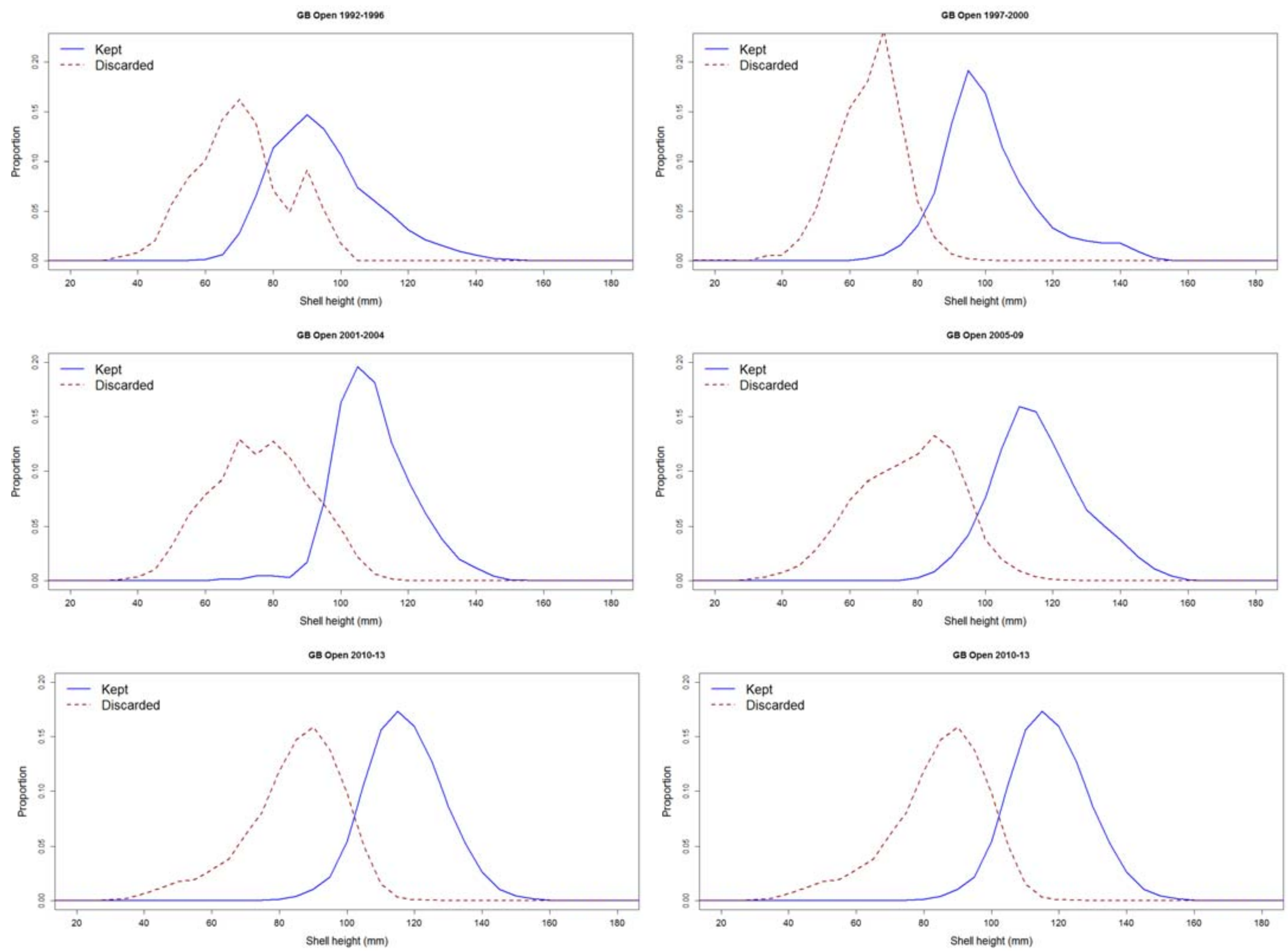


Figure B4.6 (cont). Shell heights of commercial kept (solid line) and discarded (dashed line) sea scallops from Georges Bank open areas.

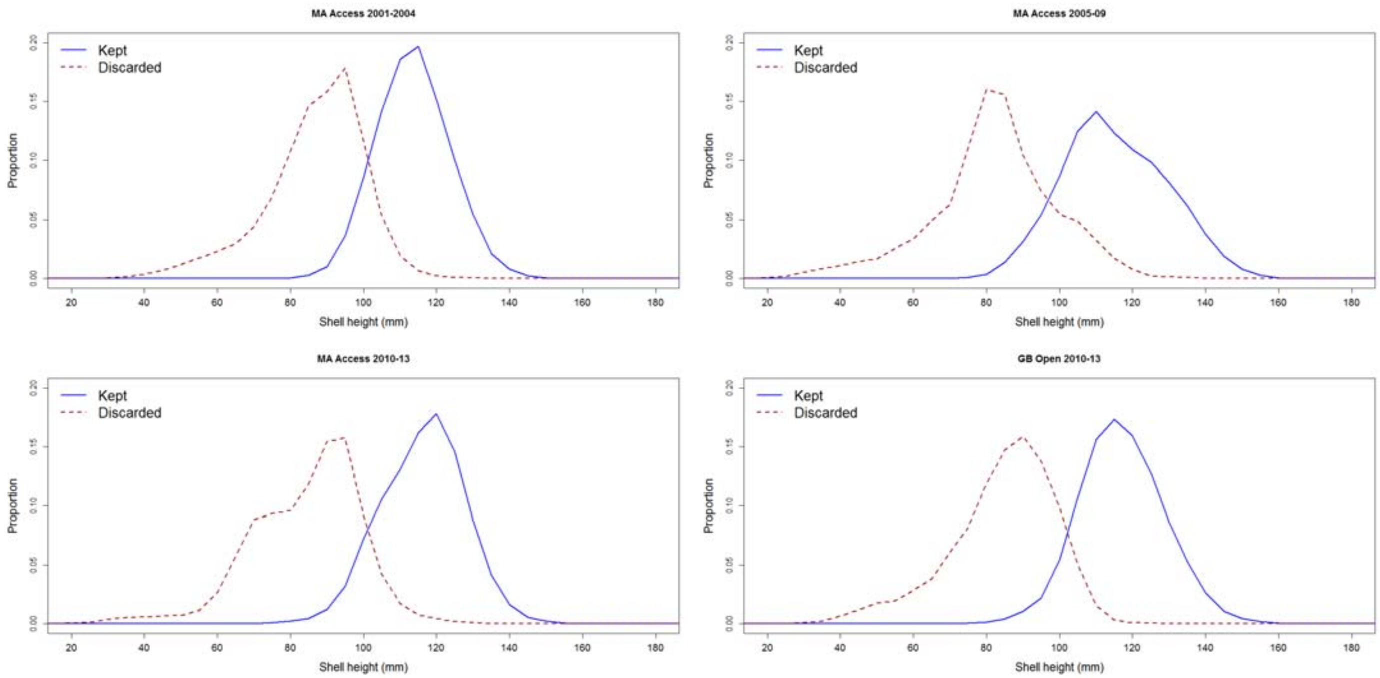


Figure B4.6 (cont.). Shell heights of commercial kept (solid line) and discarded (dashed line) sea scallops from Mid-Atlantic Bight access areas.

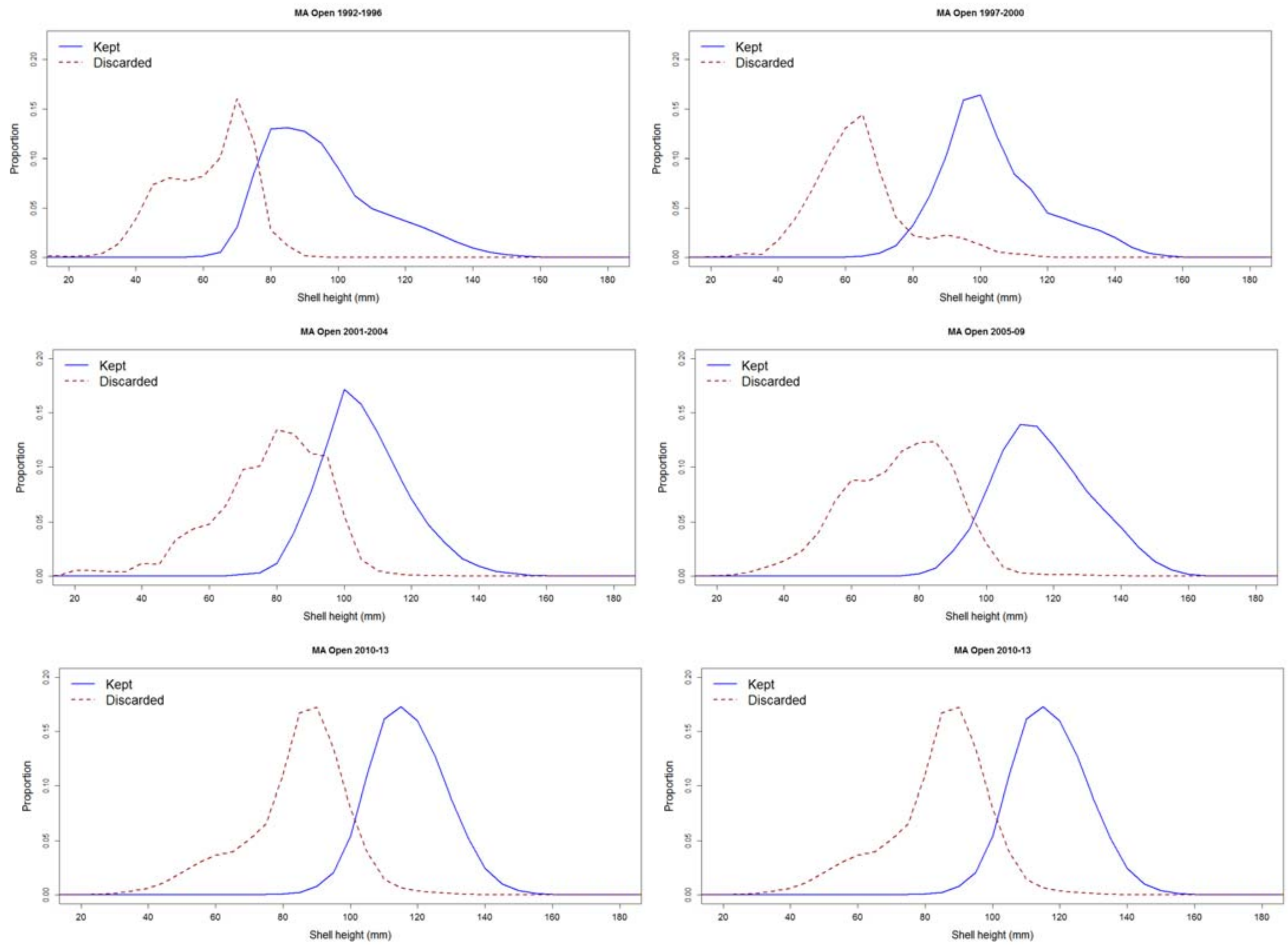


Figure B4.6 (cont.). Shell heights of commercial kept (solid line) and discarded (dashed line) sea scallops from Mid-Atlantic Bight open areas.

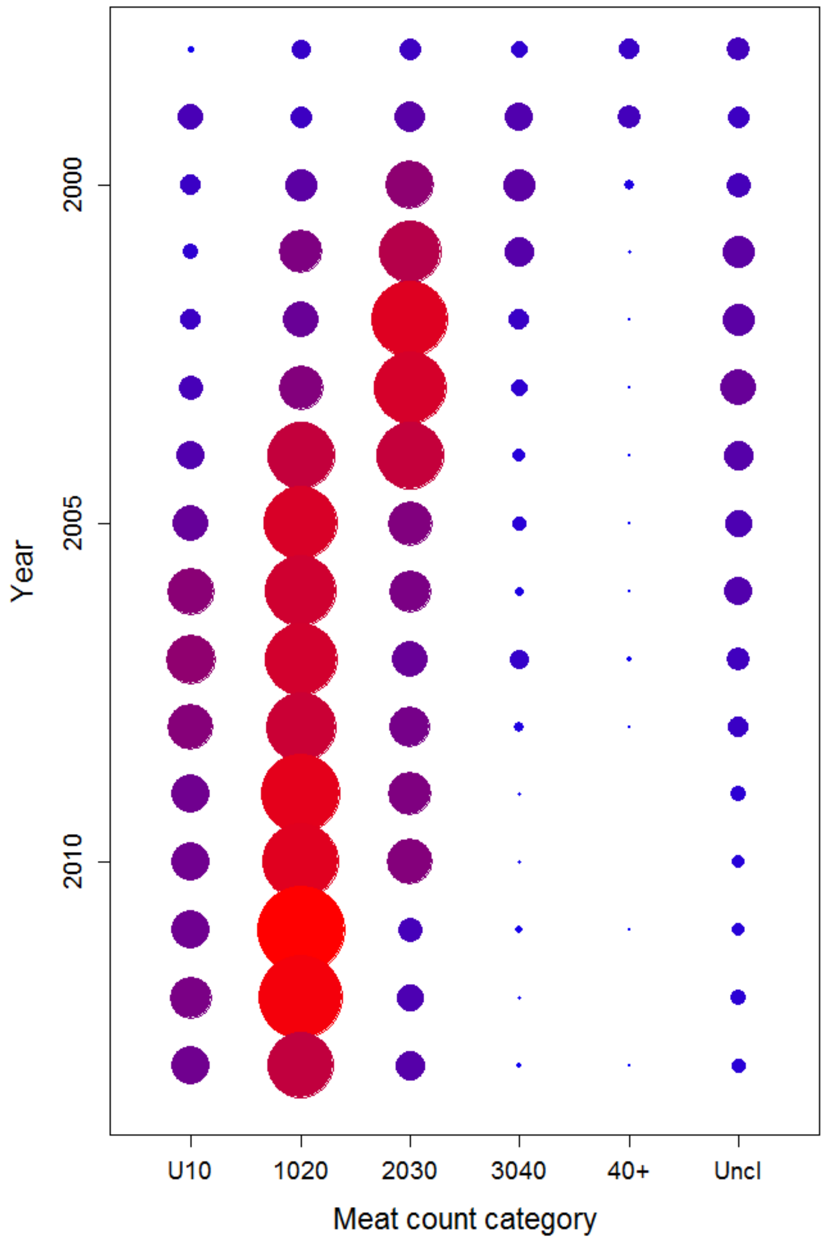


Figure B4.7. Landings by commercial meat count category (U10 = less than 10 meats per lb, 1020 = between 10-20 meats per pound, 2030 = between 20-30 meats per pound, 40+ = over 40 meats per pound, and Uncl = unclassified). The areas of the bubbles are proportional to landings.

B5. SURVEY DATA (TOR-2)

Dredge surveys

Sea scallop dredge surveys were conducted by NEFSC in 1975 and annually after 1977 to measure abundance and size composition of sea scallops in the Georges Bank and Mid-Atlantic regions (Figures B3-1 and B5-1). Means and standard errors were calculated using standard methods for stratified random surveys (Cochran 1977, Serchuk and Wigley 1989; Wigley and Serchuk 1996; Smith 1997).

The 1975-1978 surveys used a 3.08 m (10') unlined New Bedford scallop dredge with 54 mm rings. A 2.44 m New Bedford survey dredge with 54 mm rings and a 38 mm plastic liner has been used since 1979. Based on comparisons between camera and dredge data, scallops greater than 40 mm are considered fully selected by the lined survey dredge gear (NEFSC 2007). The survey covers Georges Bank and the Mid-Atlantic, using a random-stratified design. At each station, the dredge is deployed for 15 minutes. Caught scallops are counted and measured, and subsamples are weighed (meat weight, gonad weight, whole weight, see Hennen and Hart 2012). The shells from the subsamples are brought to shore for growth analysis.

The *R/V Albatross IV* was used for all NEFSC scallop surveys from 1975-2007, except during 1990-1993, when the *R/V Oregon* was used instead. Surveys by the *R/V Albatross IV* during 1989 and 1999 were incomplete on Georges Bank. In 1989, the *R/V Oregon* and *R/V Chapman* were used to sample the South Channel and a section of the Southeast Part of Georges Bank. Serchuk and Wigley (1989) did not find significant differences in catch rates between the *R/V Albatross IV*, *R/V Oregon* and *R/V Chapman.* The *F/V Tradition* was used to complete the 1999 survey on Georges Bank. NEFSC (2001) found no statistically significant differences in catch rates between the *F/V Tradition and R/V Albatross IV* from 21 comparison stations after adjustments were made for tow path length. Therefore, survey dredge tows from these other vessels were used without adjustment except for normalizing for tow distance as discussed below. The northern edge of Georges Bank was not covered by the NEFSC survey until 1982. Data from the Canadian scallop survey during 1979-1981, which used the same gear as the NEFSC survey, was used to cover the northern edge in those years (NEFSC 2010).

In 2008-2013, the NEFSC scallop survey was conducted on the *R/V Hugh Sharp*. Direct and indirect comparisons between the catches by the *R/V Hugh Sharp*, *R/V Albatross IV* and commercial vessels towing the lined survey dredge were not significantly different (NEFSC 2010). However, average catches were slightly greater (~5%) on the *R/V Hugh Sharp.* Comparison of tow distance data from dredge sensor data indicate that tow lengths from the *R/V Hugh Sharp* were about 8% longer on average than those on the *R/V Albatross IV* or commercial vessels (Figure B5.2).

In NEFSC (2010), tows on the *R/V Hugh Sharp* were reduced by 5% to compensate for the apparent differences among survey vessels. For this assessment, each tow was normalized to a tow length of 1 nm. Because dredge sensor data is only available for a subset of the tows, regression equations were developed based on tows where the sensor data is available to predict tow distance using nominal tow distance and depth as predictors. Nominal tow distance is the nominal tow time (i.e., the time elapsed after the winch is locked at the beginning of the tow to the time when haul back begins) times the mean vessel speed between these times. Separate relationships were developed for the *R/V Albatross IV* (which was assumed to also apply to the other vessels used from 1989-1999), and the *R/V Hugh Sharp:*

Tow length = -0.0388 + 0.001484*Depth + 1.061*Nominal length (*R/V Hugh Sharp)*
Tow length = 0.0864 – 0.000444*Depth + 0.972*Nominal length (*R/V Albatross IV)*

where tow length is in nautical miles and depth is in meters.
Rock excluder chains have been used on NEFSC sea scallop survey dredge since 2004 in certain hard bottom strata to enhance safety at sea and increase reliability (NEFSC 2004). Based on paired tow trials with and without excluders, the best overall estimate was that rock chains increased survey catches on hard grounds by a factor of 1.31 (CV = 0.2). To accommodate rock chain effects in hard bottom areas, survey data collected prior to 2004 from strata 49-52 and in the portions of strata 651, 661, 71 and 74 within Closed Area II were multiplied by 1.31 prior to calculating stratified random means for larger areas. Variance calculations in these strata include a term to account for the uncertainty in the adjustment factor (NEFSC 2007).

The survey area on Georges Bank used in conducting the survey and to tabulate survey data for assessment purposes was modified in this assessment to eliminate marginal scallop habitat. The modified survey area was used to calculate stratified mean catch per tow for the dredge in all years in this assessment. Stratum 72 comprises a shallow area on the northern portion of Georges Bank (Figure B5-3). Most of this stratum has few scallops, but there is a small deep portion where larger catches are often observed. Using the entire stratum induces high variability in the mean number in this stratum, depending primarily on how many tows were in the productive portion. For this reason, stratum 72 was reduced to contain the productive portion of the stratum only (Figure B5-3). Similarly, scallops are more abundant in the northern portion of stratum 74 than in the southern portion. Therefore, only the northern portion of Stratum 74 was used in the survey index. Finally, stratum 631, where the density of scallops is very low, was eliminated from the Georges Bank survey index completely. These changes resulted in a reduction in the total surveyed area on Georges Bank from 7,281 nm^2 to 6,416 nm^2.

Relatively high abundance of sea scallops in closed areas makes it necessary to further post-stratify survey data by splitting NEFSC shellfish strata that cross open/closed area boundaries. After re-stratification, the original and new strata were combined into open, closed or other areas as required for assessment and management purposes (NEFSC 1999, Figures B3-1 and B5-1).

The Virginia Institute of Marine Science (VIMS) has conducted intensive dredge surveys of selected regions on commercial vessels since 2005, using partially randomized grid designs (Figure B5.1). These surveys use two dredges fished side-by-side; the NEFSC lined survey dredge is deployed on one side while a commercial dredge is used on the other side. Comparisons between commercial vessels and the *R/V Albatross IV* indicate suggest that the survey dredge has the same fishing power on these vessels (NEFSC 2010). In the last several years, VIMS has conducted several hundred tows per year.

All VIMS data for fully covered strata (original or post-stratified) were treated in the same way as NEFSC tows. The partially randomized grid design was treated as random when calculating variances. This likely slightly overstates the true sample variance.

A relatively small number of unsurveyed strata were filled by imputation. Imputation procedures were similar to those in NEFSC (2010). In brief, GAM models were fit to estimate trends in average catch rates over time for individual survey strata with strata nested within subregions. Length composition data for such strata was estimated by the stratified mean length composition for other strata in the same region.

Capture efficiency of the survey dredge was estimated by comparing dredge catches to densities observed by the HabCam system towed at the same location (Appendix B4). The best estimates of dredge efficiency were 0.41 on sand substrates, and 0.27 on rougher gravel/cobble substrates. These, together with estimates of tow path length and stock area (see above) were used to expand mean catch per tow and estimate stock size in absolute terms. For these purposes, the South Channel and northern portion of Closed Area II are considered to have gravel/cobble bottom while the northern edge of Georges Bank, west of Closed Area II are considered mixed sand with gravel/cobble, where dredge efficiency average 0.34. All other areas, including all of the Mid-Atlantic are assumed to be predominately sand and are expanded assuming a survey dredge efficiency of 0.41.

Dredge survey stock size was increased by 10% in the Mid-Atlantic and 4% on Georges Bank to account for scallops at low densities outside the survey strata set used to calculate mean catch per tow. NEFSC (2010) estimated that about 10% of the scallops in the Mid-Atlantic and 3% of the scallops on Georges Bank lie outside the regular dredge survey strata. The new adjustment for Georges Bank was increased from 3% to 4% to also account for scallops in the areas that were dropped from the survey strata set.

<u>Dredge survey results</u>
Biomass and abundance trends for the dredge survey are presented in Table B5-1 and Figure B5-4. Based on dredge survey estimates, biomass and abundance on Georges Bank were generally low until around 1995. Very large increases were observed during 1995-2000 after implementation of closures and effort reduction measures. Biomass has remained high since, although some decreases have occurred during the last several years.

In the Mid-Atlantic Bight, dredge abundance and biomass indices were at low levels during 1979-1997, and then increased rapidly during 1998-2003 due to area closures, reduced fishing mortality, changes in fishery selectivity, and strong recruitment. Biomass was relatively stable during 2003-2008, but then declined, in part due to poor recruitment and fishing down of rotational areas. A slight increase was observed in 2013 due to growth of the large 2010 year class. Survey shell height frequencies show a trend to larger shell heights in both regions since 1995.

SMAST Video Survey

Video survey data was collected by the School for Marine Sciences and Technology (SMAST), University of Massachusetts, Dartmouth between 2003 and 2012 (Table B5-2, Stokesbury et al. 2004). This survey is conducted using drop video cameras; each station consists of clusters of four drops, and stations are placed on a grid generally 3 nm apart. Although there are several cameras on the camera pyramid, the survey index is based on the "large" camera, a standard definition video camera which was mounted 1.575 m above the bottom in the center of the sampling frame. Each drop quadrat covers about 2.8 m^2.

The precision of measurements must be considered in interpreting shell height data from video. Based on tank experiments, Jacobson et al. (2010) estimated the error associated with shell height measurements from the large video camera had a standard deviation of 6.1 mm. Field measurements are likely less precise than in a tank. For this reason, measurement error was estimated in this assessment by fitting SMAST shell heights to dredge shell heights from the same year and region that were convolved with a Gaussian kernel with mean 0 and standard deviation σ. The standard deviation that best fit the SMAST shell heights over all years and regions was 11 mm. This is the value used in modeling for this assessment.

Video survey data are expressed as densities (number m^{-2}). Variances for estimated densities are approximated using the estimator for a simple random survey applied to station means. There was some variability in the areas covered during each year.

HabCam Towed Camera Survey

HabCam is an underwater towed digital camera system (Appendix B6). The camera(s) take rapid-fire still photos of the sea floor (typically 6/sec) as it is towed at typical speeds between 5-7 knots at roughly 2 m above the bottom. Camera output is sent to the vessel using a fiber optic cable, where it is recorded on hard disk together with related metadata.

Two HabCam vehicles are in operation (Figure B5-5). The first, known as "v2", carries a single camera, and has been in operation since 2005. The second, known as "v4" carries two cameras to allow 3D viewing and more precise measurements, as well as a side-scan sonar and a full array of oceanographic sensors (e.g., CTD, chlorophyll, dissolved oxygen, pH, CDOM, water spectrometer, etc.), and was first deployed in 2012. "v1" and "v3" were prototypes that have not seen routine use.

Region-scale HabCam surveys were conducted on Georges Bank in 2011 using the v2 system, and on both Georges Bank and the Mid-Atlantic in 2012 and 2013 using the v4 system. All broadscale HabCam survey were conducted on the *R/V Hugh Sharp*. The broadscale survey was supplemented in all three years by intensive surveys of selected areas using the v2 system deployed on the *F/V Kathy Marie*. Because of the large number of images collected, only subsets were examined for sea scallop measurements and counts; typically between 1/50 to 1/200 photographs were analyzed, corresponding to about one every 25 to 100 meters. These were expanded to large scales using a zero-inflated generalized additive model followed by ordinary kriging of the residuals (Table B5-3; Figure B5-6; Appendix B6). An alternative method, taking stratified means of the main transects, gave similar results. More details on the HabCam survey and the associated geostatistical methodologies can be found in Appendix B6.

Measurement error was estimated for HabCam by comparing the shell heights to dredge data, as was done for the SMAST survey. Best fit occurred at a standard deviation of 12.7 mm, which is what was used in the modeling.

The expanded dredge survey time series together with the two optical surveys are shown in Figures B5-7 and B5-8.

Table B5.1. Dredge survey data for sea scallops on Georges Bank (below), in the Mid-Atlantic (next page) and whole stock (3rd page).

Year	Abundance (mean N/tow)	CV	Biomass index (kg/tow meats)	CV	Number of tows	Proportion positive tows	Mean meat weight (g)	Expanded abundance (millions)	Expanded biomass (mt meats)
1979	87.4	0.41	1.697	0.34	108	0.89	19.4	1,269	24,628
1980	75.8	0.24	0.920	0.16	118	0.81	12.1	1,031	12,498
1981	61.2	0.13	1.079	0.13	82	0.83	17.6	753	13,272
1982	132.9	0.46	1.080	0.32	118	0.83	8.1	2,076	16,876
1983	61.2	0.22	0.810	0.21	126	0.88	13.2	890	11,785
1984	39.3	0.11	0.577	0.10	128	0.85	14.7	536	7,887
1985	61.8	0.15	0.731	0.16	154	0.90	11.8	830	9,816
1986	116.8	0.13	1.070	0.10	153	0.90	9.2	1,445	13,237
1987	120.1	0.17	1.173	0.16	170	0.86	9.8	1,619	15,815
1988	98.7	0.16	0.993	0.14	175	0.80	10.1	1,289	12,967
1989	63.6	0.11	0.631	0.08	120	0.78	9.9	806	7,999
1990	184.1	0.24	1.511	0.22	175	0.81	8.2	2,415	19,823
1991	257.9	0.37	1.633	0.25	176	0.89	6.3	3,678	23,292
1992	232.0	0.44	2.020	0.43	171	0.89	8.7	3,300	28,737
1993	61.8	0.24	0.577	0.16	164	0.87	9.3	753	7,027
1994	46.7	0.20	0.518	0.16	177	0.84	11.1	561	6,217
1995	111.8	0.20	0.873	0.16	176	0.88	7.8	1,637	12,774
1996	133.6	0.20	1.617	0.19	171	0.90	12.1	1,855	22,458
1997	89.4	0.15	1.606	0.17	190	0.88	18.0	1,292	23,212
1998	283.0	0.26	4.003	0.32	195	0.87	14.1	3,646	51,566
1999	193.5	0.15	3.391	0.16	173	0.98	17.5	2,663	46,663
2000	766.7	0.29	8.198	0.22	164	0.91	10.7	9,996	106,882
2001	408.9	0.13	6.761	0.13	208	0.95	16.5	5,560	91,938
2002	334.5	0.14	7.195	0.14	214	0.93	21.5	4,498	96,764
2003	277.9	0.12	6.749	0.13	207	0.94	24.3	3,839	93,236
2004	291.5	0.11	8.301	0.12	218	0.94	28.5	3,959	112,749
2005	265.6	0.12	6.792	0.09	343	0.95	25.6	3,888	99,436
2006	221.3	0.13	6.123	0.13	236	0.94	27.7	3,258	90,145
2007	224.8	0.10	4.722	0.07	363	0.97	21.0	3,453	72,533
2008	321.8	0.10	6.460	0.08	239	0.97	20.1	4,805	96,444
2009	362.7	0.15	6.151	0.11	214	0.97	17.0	5,497	93,229
2010	413.1	0.21	7.652	0.09	268	0.97	18.5	6,407	118,682
2011	279.4	0.12	6.971	0.08	225	0.96	25.0	3,946	98,469
2012	225.3	0.13	5.034	0.08	224	0.97	22.3	3,488	77,936
2013	336.5	0.23	4.856	0.14	213	0.94	14.4	4,416	63,723

Table B5.1. (continued – dredge survey data for the Mid Atlantic region)

Year	Abundance (mean N/tow)	CV	Biomass index (kg/tow meats)	CV	Number of tows	Proportion positive tows	Mean meat weight (g)	Expanded abundance (millions)	Expanded biomass (mt meats)
1979	34.7	0.10	0.665	0.10	166	0.92	19.2	590	11,329
1980	42.8	0.12	0.577	0.08	167	0.94	13.5	755	9,829
1981	32.1	0.16	0.457	0.13	167	0.91	14.3	565	7,791
1982	33.5	0.11	0.497	0.08	185	0.91	14.8	591	8,458
1983	32.3	0.10	0.458	0.08	193	0.89	14.2	569	7,794
1984	32.2	0.11	0.444	0.09	204	0.91	13.8	567	7,560
1985	74.1	0.12	0.739	0.09	201	0.94	10.0	1,307	12,582
1986	129.6	0.09	1.295	0.08	226	0.93	10.0	2,285	22,057
1987	131.9	0.08	1.177	0.07	226	0.93	8.9	2,326	20,054
1988	147.8	0.10	1.738	0.08	227	0.91	11.8	2,606	29,610
1989	172.8	0.09	1.553	0.07	244	0.93	9.0	3,047	26,452
1990	215.2	0.22	1.789	0.18	216	0.89	8.3	3,794	30,463
1991	81.0	0.10	0.945	0.10	228	0.92	11.7	1,428	16,100
1992	43.5	0.11	0.526	0.07	229	0.87	12.1	767	8,956
1993	135.6	0.10	0.852	0.08	214	0.96	6.3	2,391	14,513
1994	145.1	0.13	1.141	0.09	227	0.94	7.9	2,558	19,430
1995	173.4	0.13	1.605	0.11	227	0.96	9.3	3,057	27,333
1996	58.8	0.08	0.747	0.07	211	0.89	12.7	1,037	12,718
1997	43.2	0.13	0.504	0.06	225	0.93	11.7	762	8,590
1998	168.4	0.15	1.343	0.12	215	0.92	8.0	2,969	22,872
1999	238.3	0.24	2.239	0.20	226	0.92	9.4	4,202	38,143
2000	292.1	0.14	3.719	0.13	229	0.88	12.7	5,152	63,348
2001	308.4	0.11	4.124	0.12	227	0.90	13.4	5,438	70,236
2002	284.0	0.10	4.224	0.11	206	0.89	14.9	5,009	71,952
2003	654.5	0.16	7.007	0.10	201	0.90	10.7	11,541	119,339
2004	471.0	0.12	6.093	0.08	248	0.89	12.9	8,305	103,772
2005	344.6	0.08	6.048	0.07	278	0.94	17.5	6,077	103,005
2006	386.6	0.09	6.917	0.07	302	0.95	17.9	6,818	117,810
2007	314.6	0.06	6.097	0.06	304	0.94	19.4	5,549	103,852
2008	373.7	0.09	6.258	0.08	259	0.97	16.7	6,591	106,586
2009	370.5	0.12	7.007	0.10	196	0.92	18.9	6,533	119,343
2010	250.3	0.08	5.115	0.07	281	0.94	20.4	4,414	87,126
2011	172.7	0.10	3.840	0.10	298	0.96	22.2	3,045	65,396
2012	260.2	0.12	3.194	0.06	269	0.94	12.3	4,589	54,407
2013	256.1	0.10	3.746	0.08	309	0.98	14.6	4,517	63,796

Table B5.1. (continued – dredge survey data for the whole stock)

Year	Abundance (mean N/tow)	CV	Biomass index (kg/tow meats)	CV	Number of tows	Proportion positive tows	Mean meat weight (g)	Expanded abundance (millions)	Expanded biomass (mt meats)
1979	57.6	0.27	1.113	0.23	274	0.91	19.3	1,859	35,957
1980	57.2	0.15	0.726	0.09	285	0.89	12.6	1,786	22,327
1981	44.7	0.10	0.727	0.09	249	0.88	16.4	1,318	21,063
1982	76.7	0.35	0.750	0.20	303	0.88	9.6	2,667	25,334
1983	44.8	0.14	0.611	0.13	319	0.88	13.6	1,459	19,579
1984	35.3	0.08	0.502	0.07	332	0.89	14.3	1,103	15,447
1985	68.8	0.09	0.735	0.08	355	0.92	10.8	2,137	22,398
1986	124.0	0.08	1.197	0.06	379	0.92	9.6	3,730	35,294
1987	126.8	0.09	1.176	0.08	396	0.90	9.3	3,945	35,869
1988	126.5	0.08	1.415	0.07	402	0.86	11.1	3,895	42,577
1989	125.3	0.07	1.153	0.06	364	0.88	9.2	3,853	34,451
1990	201.7	0.16	1.668	0.14	391	0.85	8.3	6,209	50,286
1991	157.8	0.27	1.244	0.15	404	0.91	7.7	5,106	39,392
1992	125.4	0.35	1.175	0.32	400	0.88	9.3	4,067	37,693
1993	103.6	0.10	0.733	0.08	378	0.92	7.2	3,144	21,540
1994	102.4	0.11	0.870	0.08	404	0.90	8.6	3,119	25,647
1995	146.6	0.11	1.287	0.09	403	0.92	8.7	4,694	40,107
1996	91.3	0.13	1.125	0.12	382	0.90	12.3	2,892	35,176
1997	63.3	0.10	0.983	0.12	415	0.91	15.8	2,054	31,802
1998	218.2	0.16	2.498	0.22	410	0.90	11.7	6,615	74,438
1999	218.8	0.16	2.739	0.13	399	0.95	12.8	6,865	84,806
2000	498.2	0.20	5.664	0.15	393	0.89	11.3	15,148	170,230
2001	352.0	0.09	5.269	0.09	435	0.93	15.1	10,998	162,174
2002	305.9	0.08	5.514	0.09	420	0.91	18.3	9,507	168,716
2003	490.9	0.12	6.895	0.08	408	0.92	14.5	15,380	212,575
2004	393.0	0.09	7.051	0.07	466	0.91	18.5	12,264	216,521
2005	310.3	0.07	6.371	0.05	621	0.95	20.8	9,965	202,441
2006	314.8	0.08	6.572	0.06	538	0.95	21.2	10,076	207,955
2007	275.6	0.05	5.500	0.04	667	0.95	20.0	9,002	176,385
2008	351.2	0.07	6.346	0.06	498	0.97	18.2	11,396	203,030
2009	367.1	0.09	6.635	0.08	410	0.95	18.0	12,030	212,572
2010	321.0	0.12	6.217	0.06	549	0.95	19.3	10,821	205,808
2011	219.0	0.08	5.199	0.06	523	0.96	23.8	6,991	163,865
2012	245.0	0.09	3.993	0.05	493	0.96	16.7	8,077	132,343
2013	291.0	0.12	4.228	0.08	522	0.96	14.5	8,933	127,519

Table B5.2. SMAST Large Camera survey data for sea scallops on Georges Bank, the Mid-Atlantic and combined.

Year	Density (N/m^2)	CV	N stations	Survey area (km^2)	Number (millions)	Mean wt (g)	Biomass (mt)
Georges Bank							
2003	0.147	0.08	929	28,677	4,213	21.9	92,343
2004	0.122	0.12	935	28,863	3,513	26.6	93,341
2005	0.116	0.11	902	27,844	3,235	24.5	79,370
2006	0.110	0.11	939	28,986	3,177	20.9	66,527
2007	0.142	0.11	912	28,153	3,989	17.8	70,858
2008	0.098	0.09	910	28,091	2,744	13.9	38,113
2009	0.157	0.11	899	27,751	4,351	12.1	52,779
2010	0.116	0.10	939	27,937	3,241	18.1	58,682
2011	0.147	0.12	918	28,338	4,169	15.6	64,885
2012	0.129	0.14	892	27,535	3,555	14.7	52,184
Mid-Atlantic							
2003	0.483	0.17	804	24,819	11,995	8.7	103,889
2004	0.224	0.10	840	25,930	5,801	12.9	75,032
2005	0.210	0.12	864	26,671	5,598	14.0	78,141
2006	0.191	0.10	897	27,690	5,292	13.7	72,312
2007	0.179	0.09	941	29,048	5,202	14.5	75,227
2008	0.184	0.10	931	28,739	5,288	14.3	75,356
2009	0.134	0.06	928	28,647	3,844	15.1	57,904
2010	0.109	0.08	988	30,499	3,324	20.6	68,363
2011	0.066	0.06	1,359	41,951	2,756	23.3	64,305
2012	0.111	0.08	1,168	35,999	3,996	9.3	37,187
Whole stock							
2003	0.303	0.12	1,733	53,496	16,208	12.1	196,232
2004	0.170	0.08	1,775	54,793	9,313	18.1	168,374
2005	0.162	0.08	1,766	54,515	8,834	17.8	157,512
2006	0.149	0.07	1,836	56,676	8,468	16.4	138,839
2007	0.161	0.07	1,853	57,201	9,192	15.9	146,085
2008	0.141	0.07	1,841	56,830	8,032	14.1	113,469
2009	0.145	0.06	1,827	56,398	8,196	13.5	110,683
2010	0.112	0.07	1,927	58,436	6,565	19.4	127,045
2011	0.099	0.08	2,277	70,289	6,925	18.7	129,189
2012	0.119	0.08	2,060	63,534	7,551	11.8	89,372

Table B5.3. Summary of HabCam abundance and biomass data for sea scallops used in this assessment. "Images" is the number of images annotated. "Images w/scallops" is the number of images in which scallops were observed. "GAM + kriging" results were estimated using the preferred approach (zero-inflated GAM models with ordinary kriging of residuals). Alternative stratified mean estimates are also shown. See Appendix B6 for further details.

Stock	Year	Annotated Images	Images w/scallops	Number (millions)				Biomass (mt)			
				Stratified random	CV	GAM + kriging	CV	Stratified random	CV	GAM + kriging	CV
GB	2011	202,257	21,428	3,992	0.02	3,832	0.31	110,204	0.02	102,819	0.12
	2012	36,304	7,189	4,003	0.03	4,642	0.14	94,025	0.03	94,040	0.08
	2013	33,864	4,671	3,562	0.03	4,049	0.09	54,683	0.03	49,671	0.29
MAB	2012	20,969	2,095	4,166	0.03	4,902	0.13	50,574	0.04	49,196	0.12
	2013	42,213	3,627	5,064	0.05	4,611	0.07	62,315	0.04	61,485	0.13
Total	2012	57,273	9,284	8,169	0.02	9,545	0.10	144,598	0.02	143,236	0.07
	2013	76,077	8,298	8,627	0.03	8,659	0.06	116,998	0.03	111,157	0.15

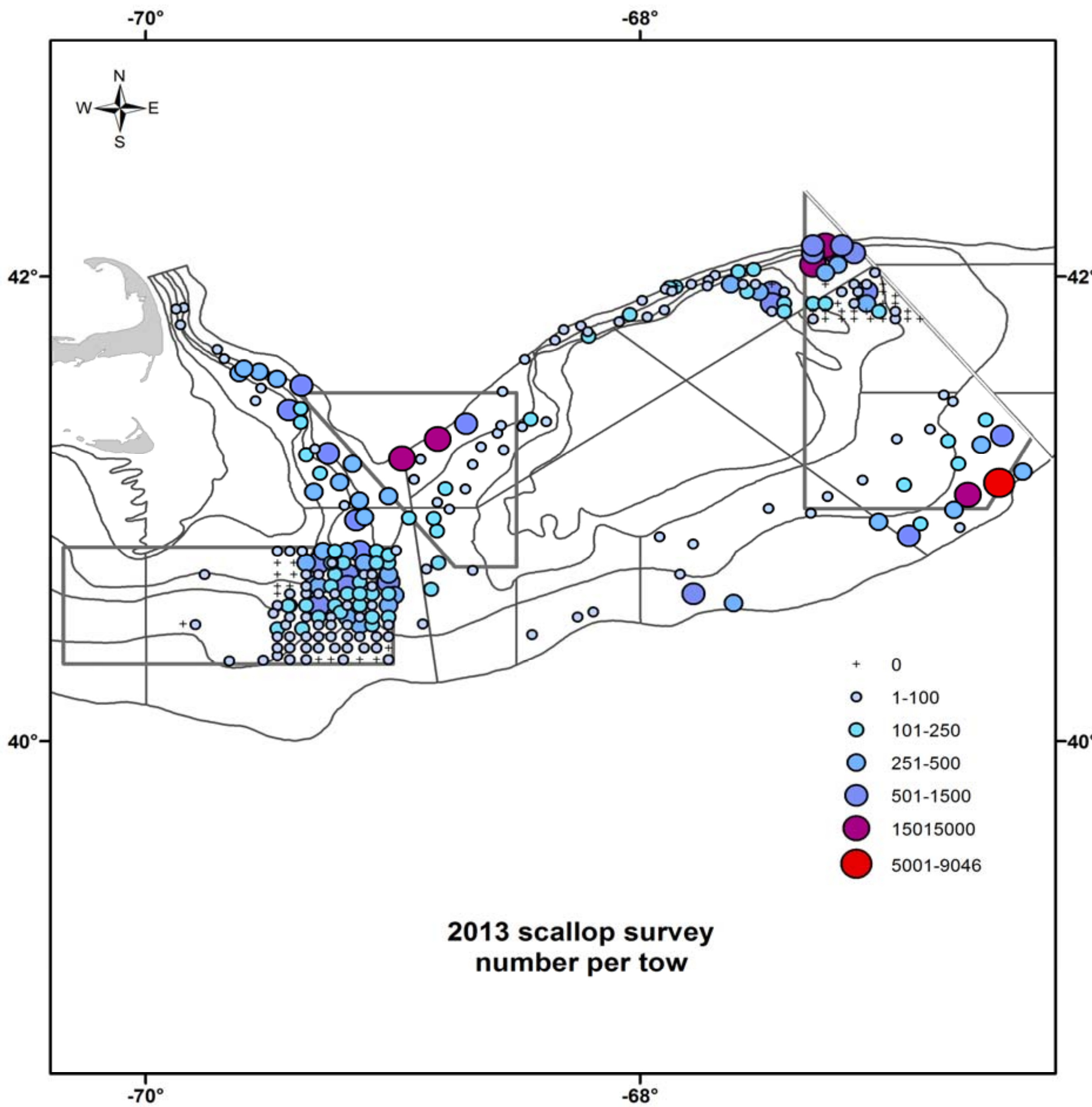


Figure B5.1(a). Dredge survey (NEFSC and VIMS) scallop dredge survey catch number in numbers per tow for Georges Bank.

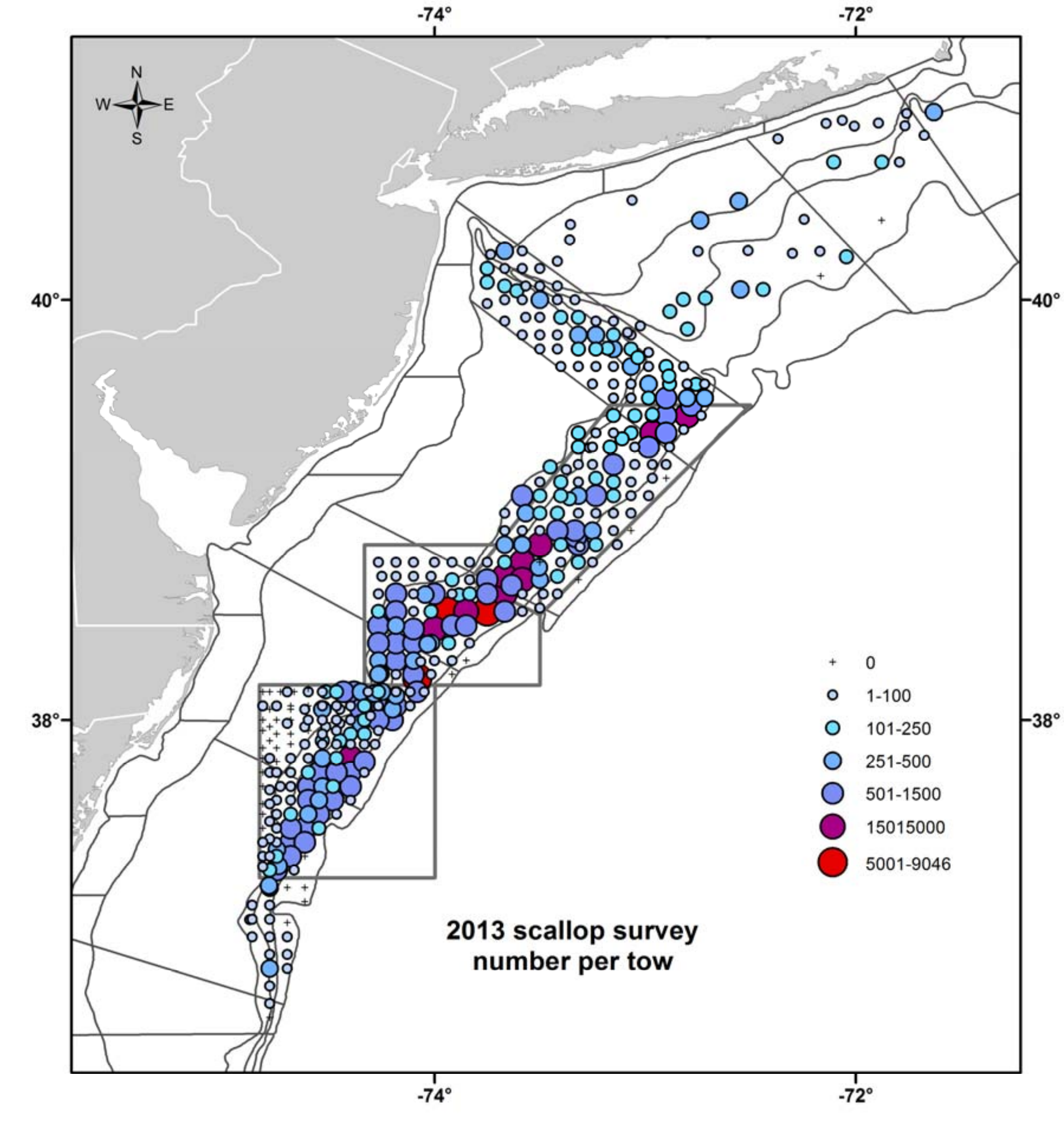


Figure B5.1(b). Dredge survey (NEFSC and VIMS) scallop catch number in numbers per tow for the Mid-Atlantic Bight.

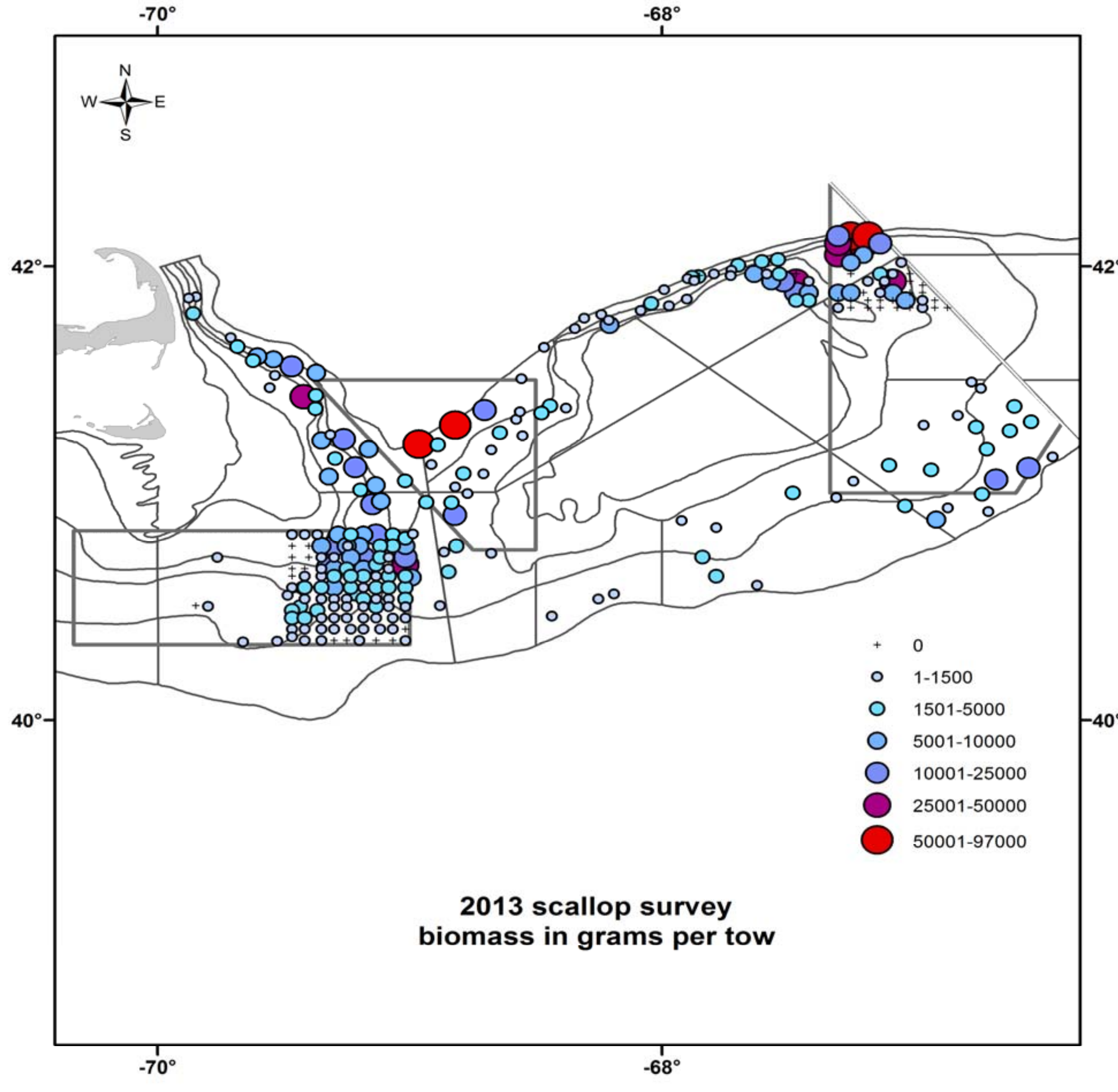


Figure B5.1(c). . Dredge survey (NEFSC and VIMS) scallop catch biomass in grams meats per tow for biomass in grams meat per tow, Georges Bank.

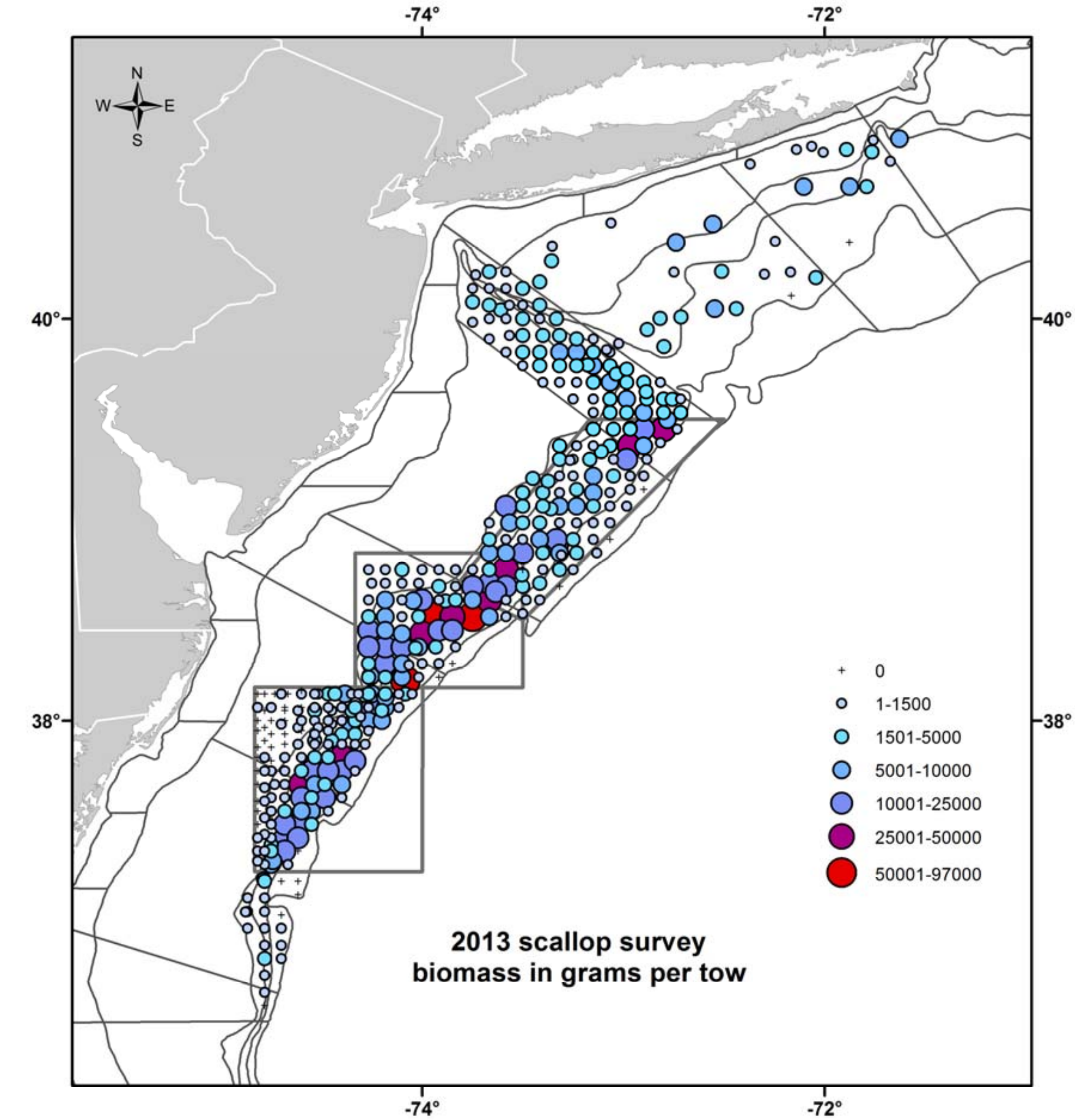


Figure B5.1 (d). Dredge survey (NEFSC and VIMS) scallop catch biomass in grams meats per tow for the Mid-Atlantic Bight.

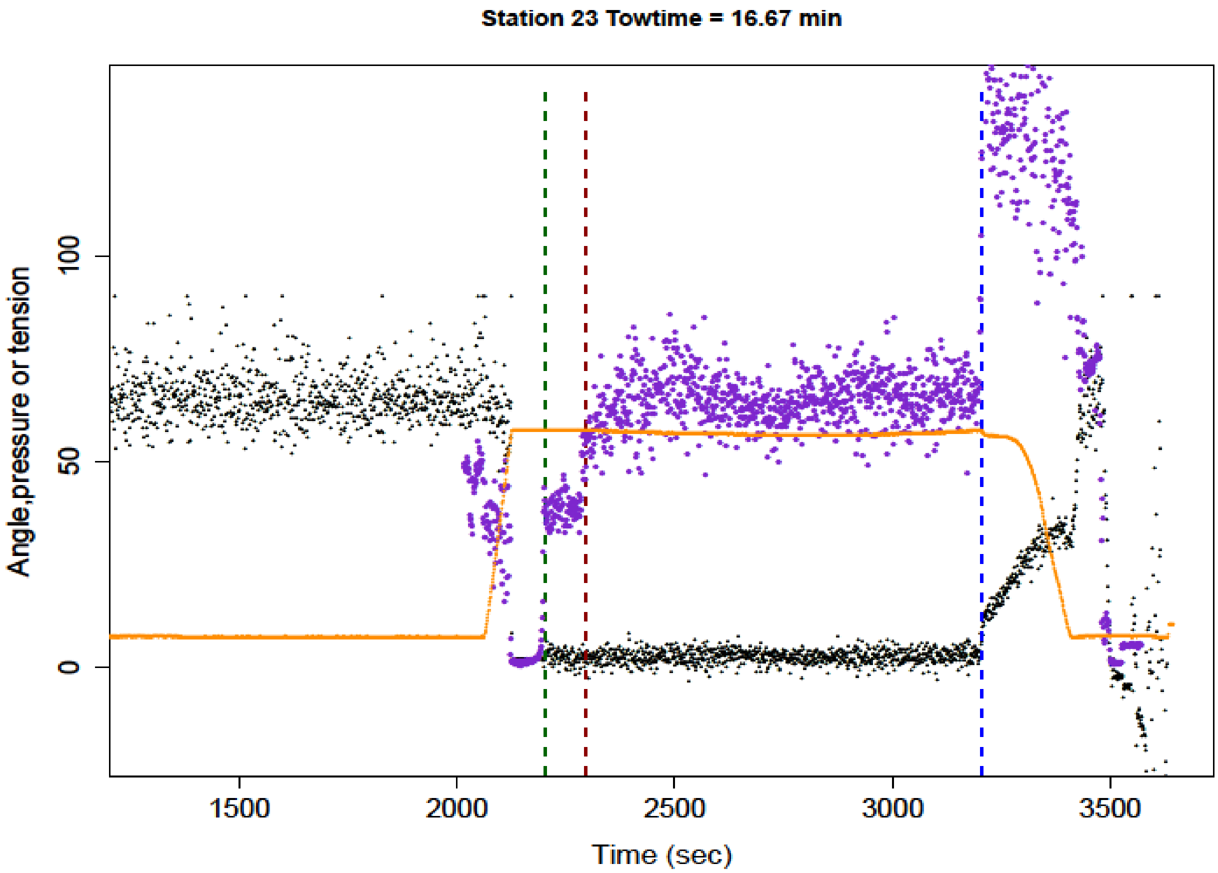


Figure B5.2. Dredge sensor data for an example tow on the *R/V Hugh Sharp* in 2013. The small black dots represent dredge angle, the orange line is pressure (a surrogate for water depth), and the purple dots are cable tension. When the dredge first hits the bottom, cable tension is zero, indicating that the dredge is not moving. The sudden increase in cable tension occurs when the tow has begun (green line), which typically is before the winch is locked (nominal tow start, red line). At tow end (blue line), sudden changes are seen in dredge angle, cable tension, and pressure.

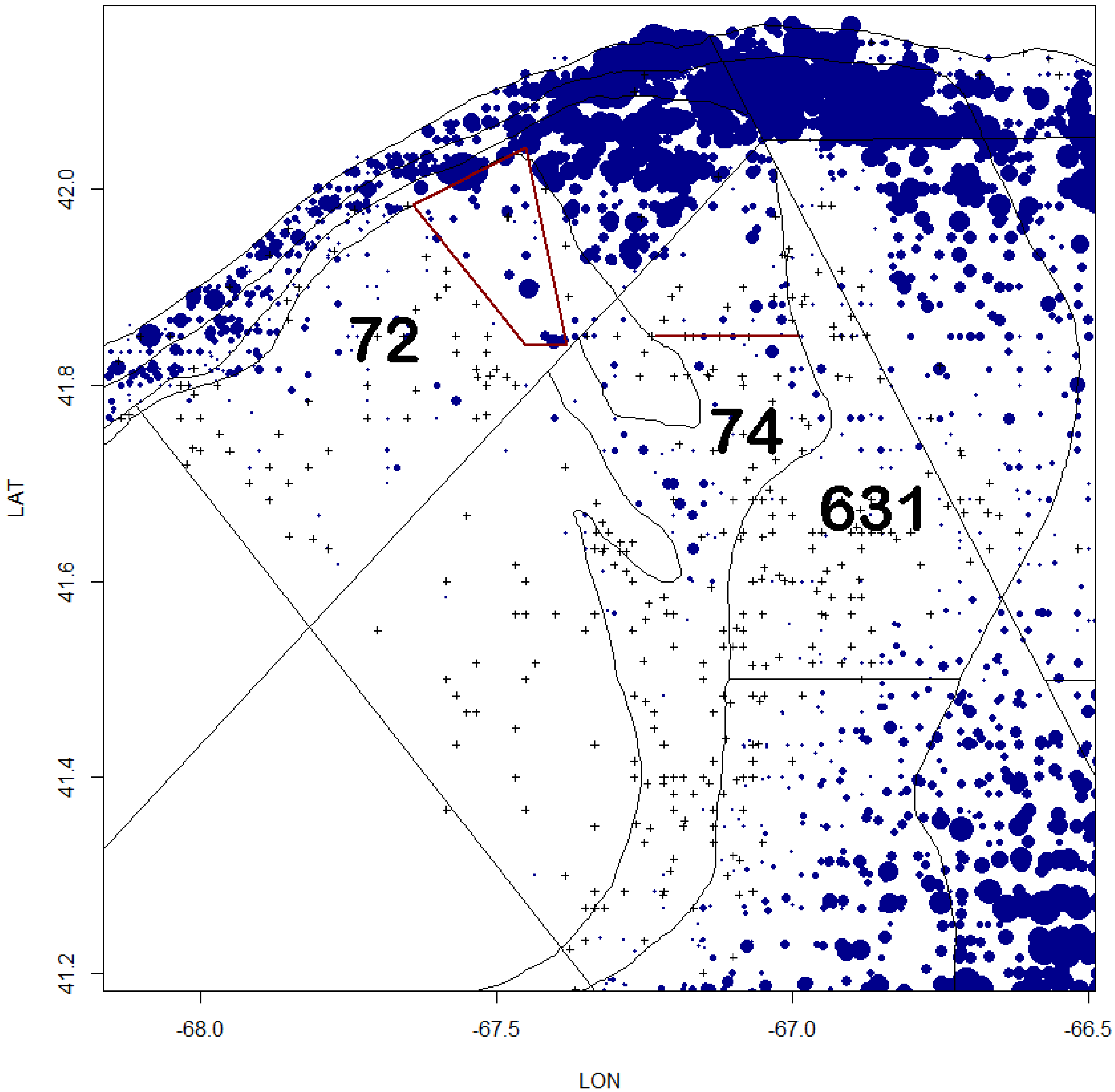


Figure B5.3 Scallop catches (in weight per tow) for all NEFSC dredge tows 1979-2013 in the northeast portion of Georges Bank, showing the two strata (72 and 74) whose areas were modified and the stratum (631) that was dropped. The red polygon in stratum 72 shows the portion of the stratum that is retained in the survey index. The portion of stratum 74 retained in the survey index is the area north of the red line. Catches with zero scallops are shown by plus marks.

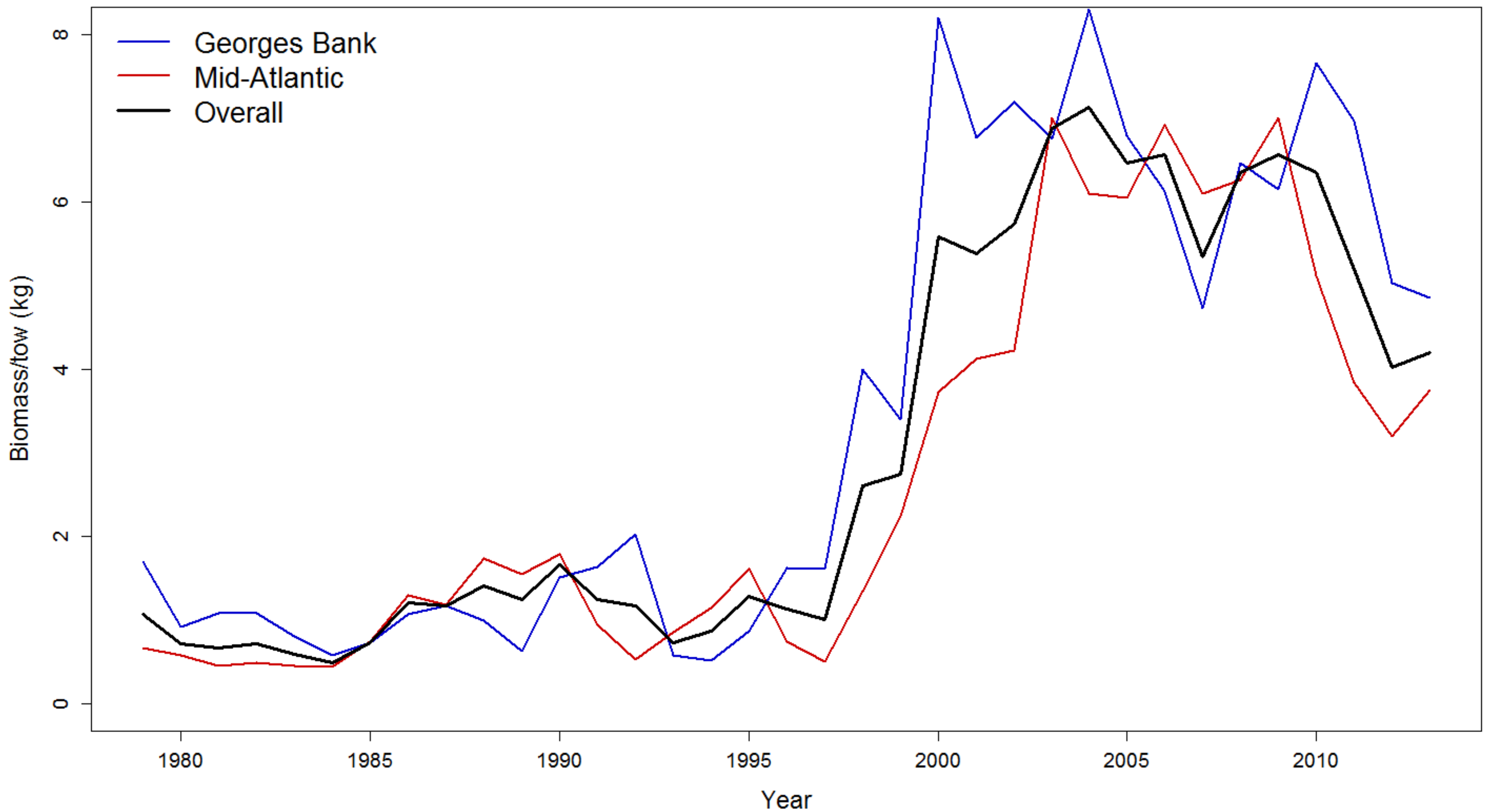


Figure B5.4. Mean stratified biomass from dredge surveys on Georges Bank, the Mid-Atlantic, and overall, 1979-2013.

Figure B5.5. The “v2” (top left) and “v4” (top right) HabCam systems, with an example image taken by v4 in the Elephant Trunk area of the Mid-Atlantic in 2013.

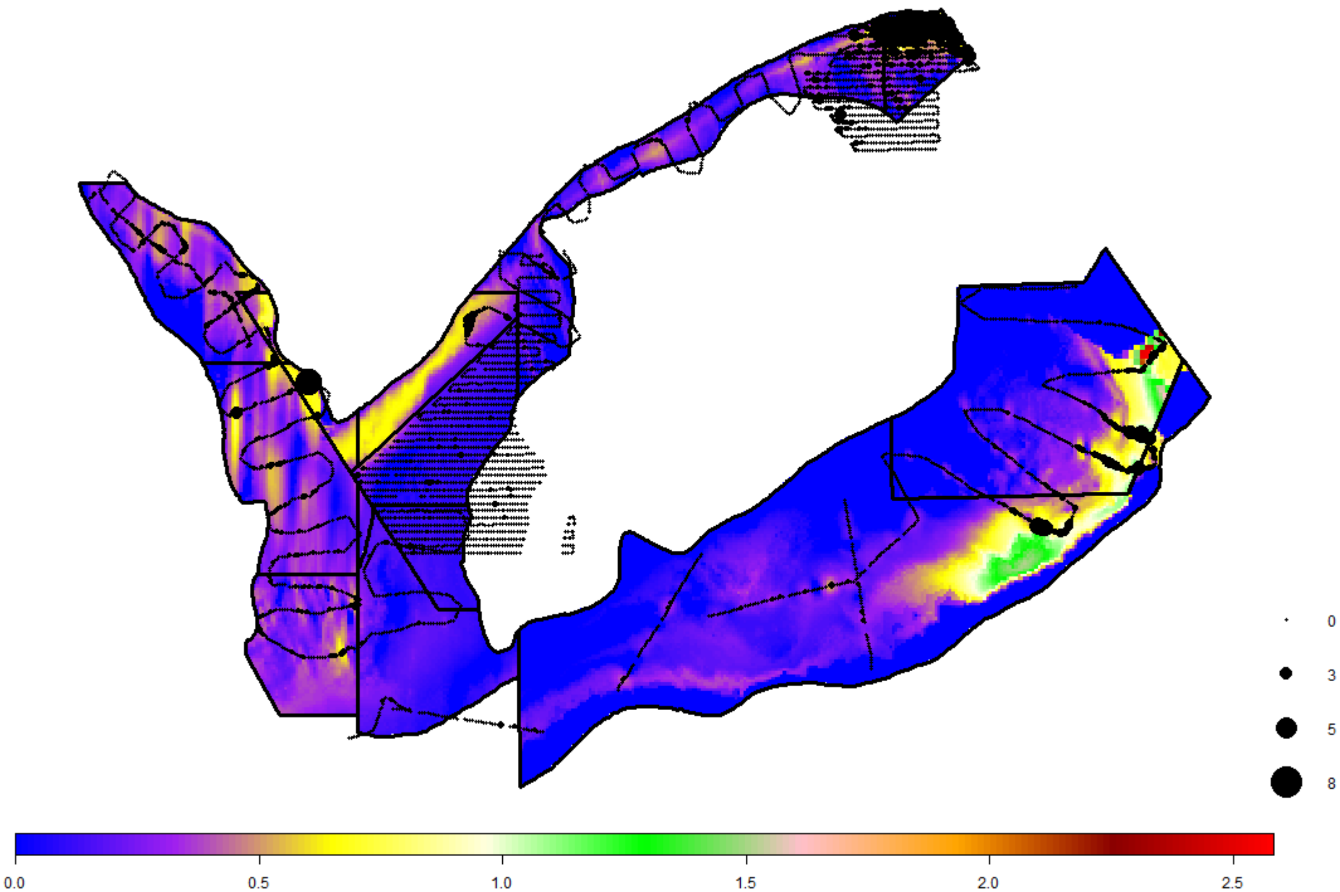


Figure B5.6(a). Estimated scallop densities (# m^2) on Georges Bank in 2013 based on HabCam data using the GAM plus ordinary kriging method. The survey trackline (black line) together with observations of scallops (black dots) are also shown.

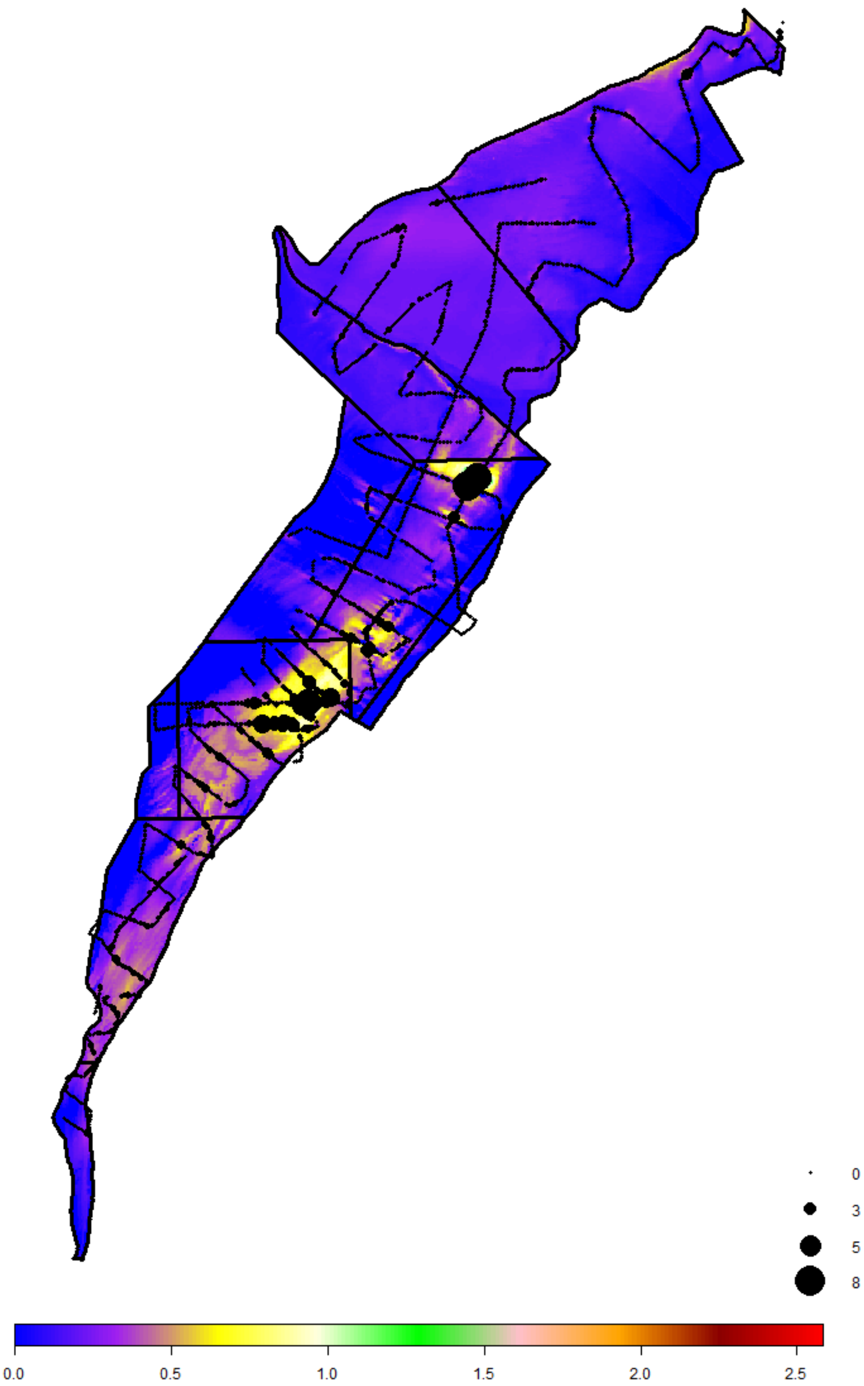


Figure B5.6(b). Estimated scallop densities (# m^{2}) in the Mid-Atlantic in 2013 based on HabCam data using the GAM plus ordinary kriging method. The survey trackline (black line) together with observations of scallops (black dots) are also shown.

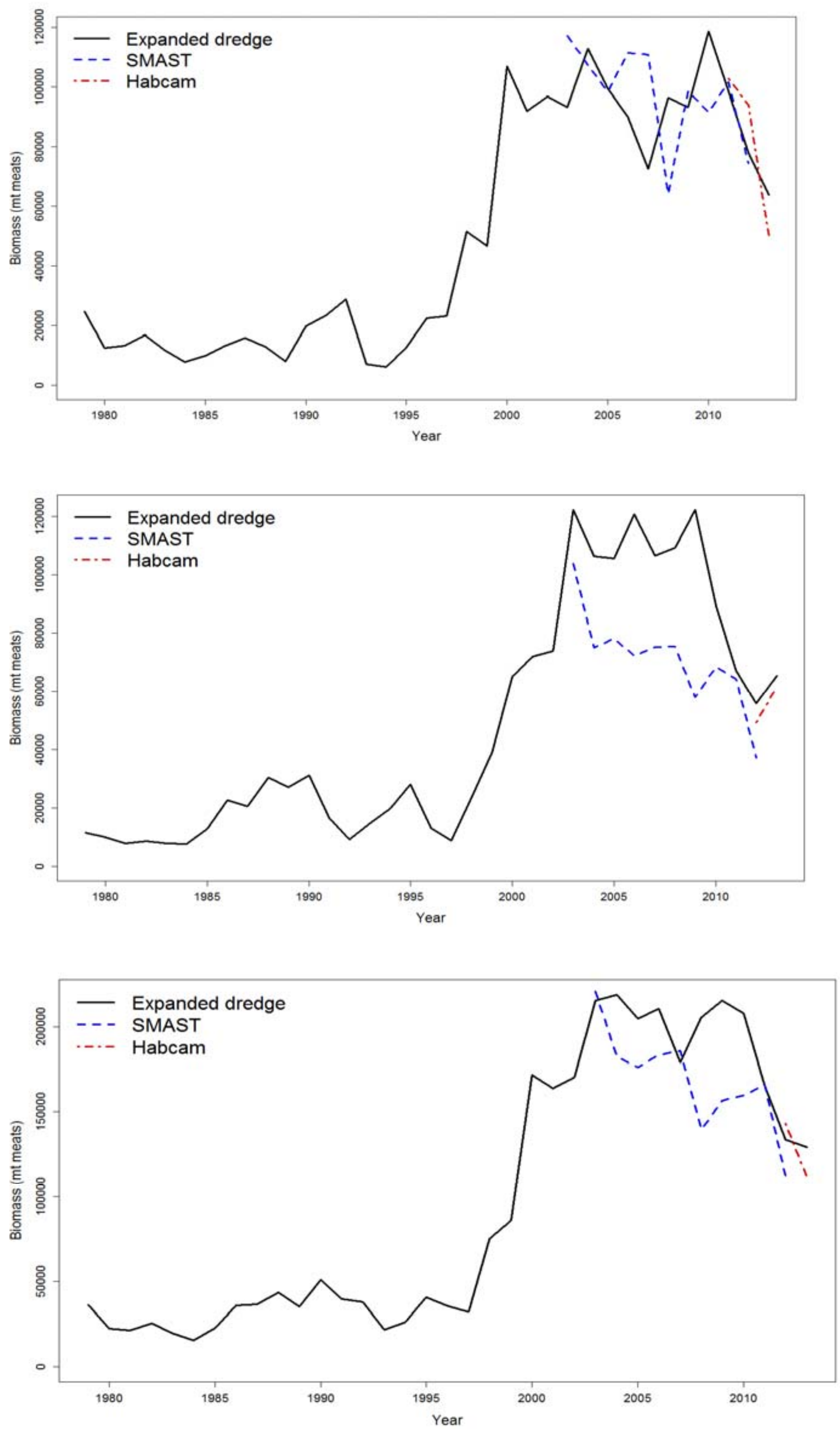


Figure B5.7. Comparison of dredge, SMAST video and HabCam survey biomass estimates for Georges Bank (top), Mid-Atlantic (middle), and combined stock (bottom).

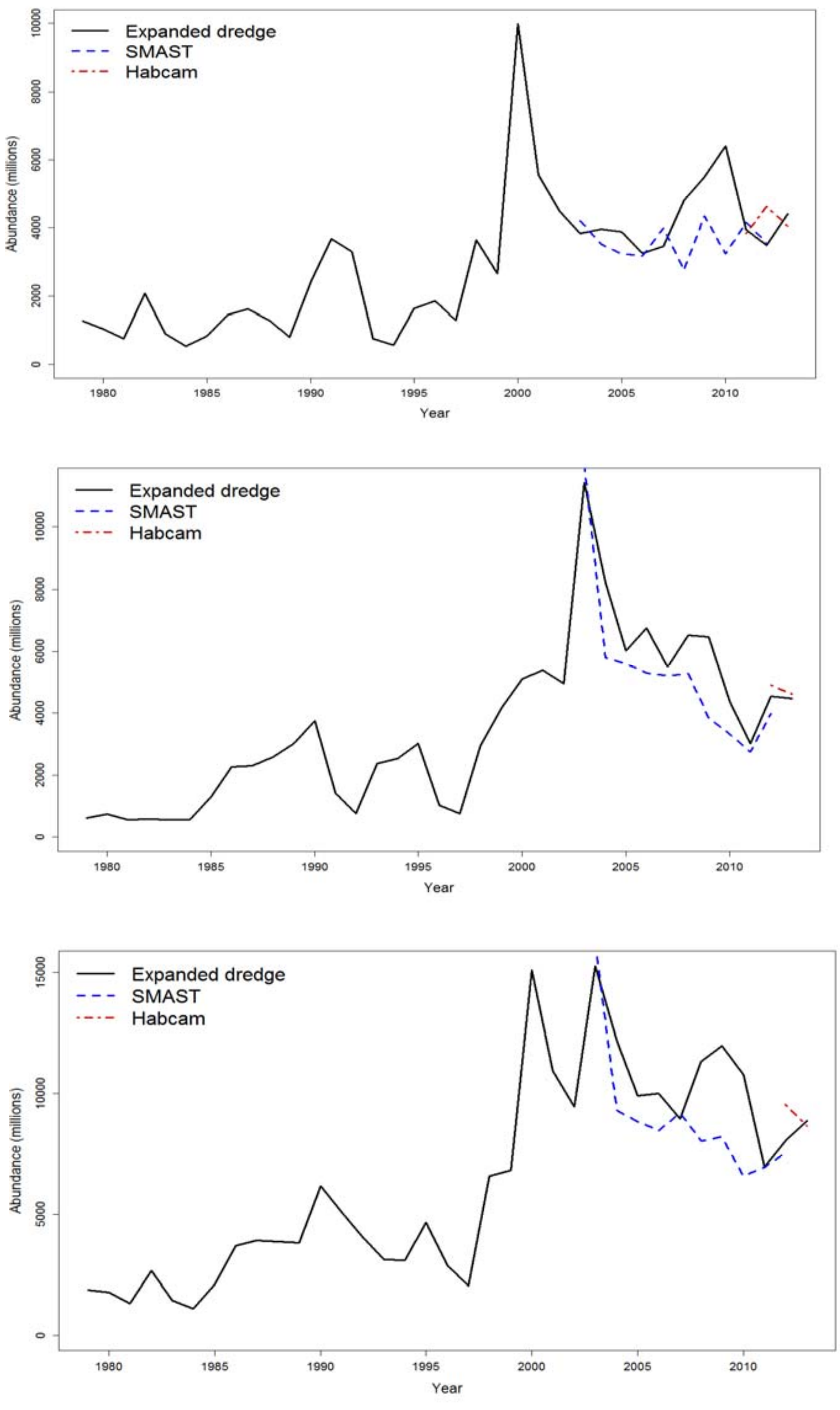


Figure B5.8. Comparison of dredge, SMAST video and HabCam survey abundance estimates for Georges Bank (top), Mid-Atlantic (middle), and combined stock (bottom).

B6. FISHING MORTALITY, BIOMASS, AND RECRUITMENT ESTIMATES (TOR 4)

A catch-at-size-analysis (CASA, Sullivan et al 1990) was used as the primary assessment estimation model. This model has been used for US sea scallop assessments since 2007 (NEFSC 2007, 2010). It performed well in simulation testing using the SAMS model as the operating model (NEFSC 2007; Hart et al. 2013). An additional and simpler "empirical" modeling approach was used for comparison to CASA results (see below and Appendix B9).

For the first time in this assessment, Georges Bank sea scallops were assessed using separate CASA models for open and closed areas. In previous sea scallop assessments, Georges Bank was modeled as a single region containing open, closed and rotational areas. Domed fishery selectivity patterns were used for the Georges Bank stock when there was no fishing in closed areas because large scallops are most common in the closed areas and thus experience less fishing mortality on average than smaller commercial-sized scallops. Using simulated and real data, Hart et al. (2013) concluded that splitting Georges Bank into open and closed areas gave more stable and likely more precise results, probably due to problems modeling complicated and ephemeral domed selectivity patterns. Separating the open and closed areas allows the use of simple logistic selectivity models for fishery size data, rather than domes. As in previous assessments, scallops in the Mid-Atlantic were assessed using single CASA model.

All three CASA models (Georges Bank open, Georges Bank closed and Mid-Atlantic Bight) were run from 1975-2013. Shell heights were modeled with 5mm shell height bins starting at 20mm, but only scallops larger than 40mm were used in tuning because smaller scallops are not fully selected in any of the surveys. The lined dredge and HabCam surveys were assumed to have flat selectivity for scallops 40+ mm. Selectivity of the SMAST large camera and unlined dredge surveys was fixed at experimentally determined values (NEFSC 2007). Selectivity of the winter trawl survey, used in the Mid-Atlantic model only, was assumed logistic with parameters estimated by the model.

Population shell height/meat weight conversions used parameters estimated from 2001-2013 research vessel survey data. Fishery meat weights were adjusted based on estimated seasonal anomalies and the seasonal distribution of landings in that year (see Appendix B3). Commercial shell height (size composition) data for 1975-1984 was from port samples, and 1992-2013 data were from sea samples (observers). The final (plus) group included L_∞. The meat weights for the plus group bin in a given year were the mean observed weight of scallops in the plus group in the dredge survey (for the population) or in port or sea samples (for the fishery, Figure B6.1).

CASA models growth using stochastic growth transition matrices that describe the probabilities for each starting size group of reaching new size groups after one year of growth. In previous assessments, transition matrices were derived directly from shell increment data, and a single transition matrix was used for the entire time series (Hart and Chute 2009a). Several growth transition matrices were used in this assessment to represent growth in different time periods because of new evidence indicating that apparent growth has changed over time. The fishery tends to select large, fast growing individuals so that smaller and relatively slow growing individuals are over-represented in the residual population; the extent of the reduced growth depends on the level of fishing effort.

The growth matrices were based on von Bertalanfy growth parameters and their variances (that measure variability among individual scallops) estimated from growth increment data using mixed-effects models (Hart and Chute 2009b, see life history section). The matrix was constructed by drawing L_∞ and K values from independent normal distributions with means and variances among individuals estimated by the mixed-effect model. One thousand parameters were drawn for each 0.05 mm interval within each 5 mm starting size bin and used to simulate one year of growth. The resulting binned scallop shell heights were converted to proportions that estimate the desired transition probabilities. Transition matrices constructed in this way were smoother, but similar, to matrices derived directly from growth increments in past assessments.

Prior probabilities (also known as likelihood constraints) are used to incorporate knowledge regarding absolute scale from the surveys. Priors on survey catchability were used for the lined dredge, the SMAST large camera, and HabCam surveys. Priors were calculated assuming that catchability parameters for these surveys have a beta distribution with specified mean and coefficient of variation (CV). The assumed CVs for catchability priors were 0.15 for SMAST and the dredge survey and 0.1 for HabCam. The CV for HabCam is smaller because it is expected to give the most accurate scaling.

For use with priors, the dredge survey was expanded to an absolute scale assuming flat selectivity, experimentally derived estimates of capture efficiency and best estimates of stock area and areas swept by the dredge tows (Appendix B4). SMAST large camera data were expanded after using the experimentally derived selectivity curve to adjust for reduced selectivity of small scallops. After this adjustment, SMAST abundance and size data were expanded assuming flat selectivity and 100% capture efficiency. Expansion of the HabCam survey assumed 100% detectability of scallops > 40mm by the camera.

The estimated catchability parameters from CASA are useful diagnostics when compared to their priors. In the CASA model, I=qN where I is a survey abundance observation, N is abundance available to the survey and q (with expected value 1.0) is the catchability parameter. Relatively high estimates of q indicate relatively low estimated abundance and *vice-versa* because abundance N=I/q.

The catchability parameters estimates described above could, in principal, be larger or smaller than one but beta distributions in CASA do not allow values larger than one. Moreover, we wanted to use a symmetrical beta distribution so that the probability of being slightly larger or smaller than the expected value was the same. We met these objectives in a convenient fashion and without additional programming by multiplying the survey abundance data in the model by 0.5 so that the mean of the prior distributions and expected catchability values were 0.5. This rescaling is simply for convenience; it replaces the target 1.0 for catchability by 0.5 with no other effect on model estimates.

CVs for survey data and effective samples sizes for length data were tuned in preliminary model runs so that the median of assumed values used in tuning were similar to expected values based on goodness of fit. Asymptotic delta method variances calculated in CASA with AD-Model Builder software were used to compute variances and CVs. Sensitivity and profile analyses were also used to describe uncertainty.

CASA model for Georges Bank Open

The model was tuned to the lined dredge survey (1979-2013), the SMAST large video camera survey (2003-2012), the HabCam survey (2011-2013) and the unlined dredge survey (1975 and 1977). The commercial fishery selectivity periods were 1975-1998, 1999-2004, and 2005-2013 so that there was separate fishery selectivity curve during each period.

Two growth matrices were used: one derived from shells collected from the Georges Bank open areas from 2001-2012, and the other from shells collected from all of Georges Bank during 1988 and 1993. The first growth matrix is from a period of moderate fishing pressure while the second is from a period of high fishing pressure. The first transition matrix was used during 1975-1985 and 1999-2013 when fishing effort was moderate and the second matrix was used from 1986-1998, when fishing effort was the highest. Natural mortality was set at $M = 0.16$ ($M = 0.24$ on the plus group) and incidental fishing mortality was set at 0.2 times fully recruited fishing mortality for the smallest size group as described elsewhere in this report. Results are shown in Figures B6.2 to B6.15.

The resulting basecase model fit survey abundance, trends and size data reasonably well (Figures B6.2 to B6.5). Mean estimated posterior efficiencies for the dredge, SMAST and HabCam surveys ranged from 0.53-0.66 (compared to the prior mean 0.5), indicating that CASA abundance estimates were slightly lower than the survey abundance data on average (Figure B6.7). Model estimates of fishing mortality were consistent in scale with the Beverton-Holt (1956) length-based equilibrium estimator (Figure B6.13).

Fishery selectivity strongly shifted over time toward larger shell heights, reflecting changes in gear and targeting practices (Figure B6.8). The size at 50% selectivity moved from about 75 mm before 1999, to 90 mm during 1999-2004, and 100 mm since 2005.

Biomass and abundance generally declined and fishing mortality increased during 1975-1995, with these trends reversing themselves after 1995. As a result of the changes in selectivity and fully recruited fishing mortality, survival to large shell heights has increased substantially in recent years (Figures B6.10-11).

The Georges Bank Open runs show very little retrospective pattern with a seven year peel (Figure B6.15). However, over the last three years, there has been a tendency for the model to overestimate biomass and underestimate fishing mortality.

CASA model for Georges Bank Closed

The model was tuned to the same surveys as used for Georges Bank open areas. There were three growth periods in the model. The first, from 1975-1986, used data from shells collected in the open areas during 2001-2012 that reflected moderate fishing pressure. The second 1987-1995 used data from shells collected from all of Georges Bank during 1988 and 1993 when fishing pressure was high. The third period 1996-2013 is based on shells from the Georges Bank Closed Areas during 2001-2012 when fishing was low or zero. Natural and incidental fishing mortality assumptions were the same as the open area model (i.e., $M = 0.16$ and $M = 0.24$ on the plus group). Incidental fishing mortality was set at 0.2 times fully recruited fishing mortality for the smallest size group. Results are shown in Figures B6-16 to B6-25.

Model abundance estimates generally track dredge survey abundance data well during 1979-1997, but are below survey abundance for 1998-2010. Mean posterior efficiencies for catchability were 0.68 for the dredge, 0.74 for SMAST and 0.39 for HabCam so that the dredge and SMAST surveys were above the prior mean of 0.5 while HabCam was below

(Figure B6.21). The discrepancy between the surveys is likely due to the fact that the HabCam survey was only conducted in 2011-2013, when estimated abundance tended to be above the surveys, whereas there were years that the model was well below expanded estimates from both the other surveys. The model estimated abundance and biomass for 2013 above both the dredge and HabCam surveys.

The model generally fit shell height data and survey data, except for years with very strong recruitment events, when the model tended to be below the survey data (Figures B6.17 to B6.19). CASA model estimates of fishing mortality about the same scale as Beverton-Holt estimates (Figure B6.27).

Estimated fishing mortality increased from 1975-1993 (Figures B6.23 and B6.25) and were low or zero afterward. This resulted in a dramatic increase in biomass during 1994-2004, and a build-up of large scallops (Figure B6.23 and B6.24). Fishery selectivity since 1999 shifted strongly to large scallops (Figure B6.22), even more so than in the open areas, because scallop fishermen tend to select the largest market category (U-10s, i.e., over 45 g meat weight) which usually commands a premium price.

The model for Georges Bank closed areas has a moderate retrospective pattern (Figure B6.29, Mohn's $\rho = 0.33$), where estimates of biomass decrease, and fishing mortality increase, as more years of data are added.

When 6 or 7 years of data are removed, the model fits the survey data well (Figure B6.28). However, the declines in biomass observed in surveys in recent years cannot be fully explained by fishery removals and the assumed natural mortality, so that the model lowers the biomass for previous years as more years of data are added.

CASA model for combined Georges Bank open and closed areas

Biomass and fishing mortality estimates for Georges Bank open and closed combined (Figure B6.30) show generally decreasing biomass and increasing fishing mortality from 1975-1992, with peak fishing mortality of 1.69 in 1992, and minimum biomass of 5,903 mt in 1993. Fishing mortality since 1995 has generally been between 0.2 and 0.4, and biomass increased substantially between 1994 and 2003. Estimated 2013 biomass and fishing mortality for Georges Bank combined is 86,460 mt and F = 0.30, respectively. Retrospective scores for the entire Georges Bank region fell between the scores for the open and closed portions only (Figure B6.30b).

CASA Model for the Mid-Atlantic

The Mid-Atlantic CASA model uses the surveys also used for Georges Bank plus the NEFSC winter bottom trawl survey which was conducted between 1992 and 2007. The winter survey used flatfish trawl gear similar to commercial scallop trawls and should have caught scallops fairly reliably. Preliminary runs with potentially domed selectivity for the winter trawl survey did not indicate that selectivity was reduced for large scallops, so selectivity was modeled using a logistic curve with parameters estimated by the model. Survey efficiency priors and selectivity assumptions for the other three surveys were the same as for Georges Bank. The fishery selectivity periods were 1975-1979, 1980-1997, 1998-2001, 2002-2004 and 2005-2013. The first period was modeled as domed (double logistic) selectivity, due to indications in the data of higher mortality on intermediate sized scallops. This was likely caused by fishing effort that was concentrated in only a portion of the stock, so that most large scallops were in areas outside the intensively fished area where densities were lower. All the other periods were assumed to have logistic selectivity. Natural mortality was set at $M = 0.2$ with $M = 0.3$ for the plus group, and incidental fishing mortality was set at 0.1 times fully recruited fishing mortality for the smallest size group.

Three growth periods were used: the 1975-1985 and 1998-2008 periods were modeled based on shells collected during 2001-2009 when fishing pressure was moderate. Growth during 1986-1997 was based on shells collected in 1988 and 1993 when fishing effort was high. Growth during 2009-2013 was based on shells collected during 2010-2012 when growth was apparently somewhat faster than during 2001-2009.

Preliminary runs using the effective sample size tuned to match model fits for the dredge survey gave unrealistic results with the model estimating lower fishing mortality in the early 1990s, when fishing effort was the highest, than ten years later. In addition, the model predicted a build-up of scallops in the plus group during the early 1990s contrary to dredge survey shell heights. Estimated fishing mortalities conflicted with those from the Beverton-Holt equilibrium estimator.

For these reasons, the effective sample size of the dredge shell heights was increased to an average of about 800 so that the dredge size data fit the model more closely (Figures B6-31 to B6-43). This resulted in much more realistic fishing mortality and shell height estimates (Figure B6.40). The increased effective sample size is ad-hoc but corresponds to an effective sample size of about 4 scallops per tow which is not unreasonable. Results are shown in Figures B6.33 to B6-44.

The final model fit survey abundance data well for some years, but was often below survey estimates during and after strong recruitment events (Figures B6.31). This was especially apparent starting in 2003, when a very strong year class was observed in both the dredge and SMAST surveys. Because of this conflict, posterior efficiencies were high and near the upper bounds of their priors (over 0.8 for the dredge and SMAST surveys and over 0.6 for HabCam relative to the prior target 0.5, Figure B6.37). Model estimates of shell heights generally fit the data well, except the model estimates of some strong year classes were below those of surveys (Figures B6.32 to B6.35).

Fishery selectivity was strongly domed during 1975-1979 but shifted to a logistic shape and moved father to the right during subsequent periods as would be expected based on management and fishery changes (Figure B6.38). By 2005-2013, only the plus group was fully selected. Model estimated fishing mortality on larger scallops generally increased during 1975-1995, reaching a maximum fully recruited fishing mortality of about 1.5 in 1995, and then declined (Figure B6.39 and B6.41). This decline was much greater for small scallops, which were affected by the shifting selectivity as well as the decline in fully recruited fishing mortality. Abundance and biomass were relatively low during 1975-1998, and then rapidly increased from 1998-2003 (Figures B6.39). Biomass and abundance declined during 2009-2012, primarily as a result of poor recruitment. Recruitment appears to have been substantially stronger since 1998 (Figure B6.39).

The model for sea scallops in the Mid-Atlantic Bight showed a fairly strong retrospective pattern for the earliest three years, with biomass decreasing and fishing mortality increasing as more years of data were added (Figure B6.44). However, this pattern has disappeared during the last several years and has reversed directions slightly.

Whole stock biomass, abundance and mortality

Biomass, egg production, abundance, recruitment and fishable mean abundance were estimated for the whole stock and for Georges Bank as a whole by adding estimates for the Mid-Atlantic Bight and Georges Bank Open and Closed (Table B6.1). For example, whole stock fishing mortality rates for each year were calculated: $F = \left(C_M + C_{Go} + C_{Gc}\right) / \left(\overline{N}_M + \overline{N}_{Go} + \overline{N}_{Gc}\right)$ where C_M, C_{Go}, C_{Gc} are catch numbers for the Mid-Atlantic Bight, Georges Open and Georges Closed areas. Terms in the denominator are average fishable abundances during each year calculated in the CASA model as $\overline{N} = \sum_L \frac{N_L\left(1-e^{-Z_L}\right)}{Z_L}$. The simple ratio formula used to calculate whole stock F is an "exact" solution because the catch equation can be written $C = F\overline{N}$.

Whole stock variances were calculated assuming that estimation errors for Georges Bank open and closed, and the Mid-Atlantic Bight were independent. In particular, variances for biomass, abundance and catch estimates were the sum of the variances for Georges Bank open and closed and the Mid-Atlantic Bight. CVs for the ratios estimating whole stock F were approximated $CV_F = \sqrt{CV_C^2 + CV_{\overline{N}}^2}$, which is exact if catch number C_N and average abundance $\overline{N}$ are independent and lognormally distributed (Deming 1960). The CV for measurement errors in catch for each region CV_C=0.05 is the same as assumed in fitting the CASA model. Variances for the stock as a whole depend on the assumption that model errors in Georges Bank and the Mid-Atlantic are independent. These variances would be higher if a positive correlation between model errors exists, and lower if they are negatively correlated.

Like the trends for smaller areas, whole-stock fishing mortality generally increased from 1975-1992 and then declined (Table B6.1 and Figure B6.45). Whole stock biomass, abundance and fishing mortality in 2013 were respectively 132,561 mt meats, 8014 million and 0.32. The biomass and abundance in 2013 were the highest in the 1975-2013 time series. Retrospective scores for the entire sea scallops stock were in the same range as scores for individual regions (Figure B.45b).

The standard errors estimated by the CASA model in this assessment are too small and do not capture all of the underlying uncertainties. The long time series of relatively precise dredge survey data and recent optical survey data, assumptions that survey selectivity is known and prior information on survey efficiencies likely contribute to the underestimation of uncertainty. It is also possible that the survey catchability estimates near the bounds of their priors artificially reduce variance. Comparisons with expanded survey data, retrospective and sensitivity analyses as well as likelihood profiles shown below better describe the uncertainties in the assessment.

Historical retrospective analysis

The current CASA model estimates can be compared to those from the last two benchmark assessments (SARC-45/NEFSC 2007 and SARC-50/NEFSC 2010), and also updates of the SARC-50 model configurations through 2011 and 2012 (Figures B6.46). While the estimates have been fairly stable, there has been a tendency for biomass and recruitment to be revised downward, and fishing mortality upward over time.

It is also of interest to compare the SARC-50 configuration updated through 2013 to the present model. There is a more substantial difference in the Georges Bank models, where the stock was assessed as a whole in the SARC-50 and using separate models for open and closed areas in the current assessment (Hart et al. 2013). The biomass plots indicate modest differences between the two configurations (Figure B6.47). Fishing mortality estimates for the two models are not completely comparable because of differences in estimated selectivity between the models.

Likelihood profile analysis

Likelihood profiles were constructed for natural mortality (Figure B6.48) with plus group natural mortality was fixed at 1.5x that of smaller scallops. For both Georges Bank open and closed, total -log likelihood was minimized at about M = 0.22. For the open areas, the survey trend component of the likelihood (sum over all surveys) was smallest at lower M values, whereas the likelihood for the size data (sum of fishery and all surveys) and Q priors were minimized at larger M values. There was a similar pattern for Georges Bank closed, although the survey trend likelihood component was minimized at about M = 0.18. For Mid-Atlantic sea scallops, the total –log likelihood was minimized near the assumed M = 0.2. The likelihood component for size composition was minimized at a lower natural mortality, whereas the component for the Q prior was minimized at higher M. Effects on stock estimates were evaluated by sensitivity analysis (see below).

Another likelihood profile analysis was constructed for natural mortality of the plus group. Because of the limited number of scallops in the plus groups in the other two models, this was conducted for the Georges Bank closed area model only. Natural mortality for the smaller size groups was fixed at M = 0.16 as in the basecase model. The size composition data component of the likelihood was minimized at low plus group mortality, whereas the –log likelihood of the survey trends and q priors decreased and fit improved as plus group mortality increased (Figure B6.49). Total –log likelihood was minimized at a plus group M of about 0.24, or 1.5 times that of smaller size groups. The latter is the assumption of natural mortality on the plus group made in all the models.

Profiles over dredge survey catchability
A final set of likelihood profile analyses were used to explore differences between CASA model abundance estimates and survey swept-area abundance data as well as the tendency for dredge, SMAST and HabCam survey catchability estimates to fall near the upper bound of their prior distributions (Tables B6-2 to B6-4).

Models for the Mid-Atlantic Bight, Georges Bank closed and Georges Bank open areas were run with the catchability parameter (Q) for the dredge survey fixed at a range of values between 0.4 and 1.2. Goodness of fit (unweighted negative log likelihood) for each type of data as well as measures of stock biomass and fishing mortality were recorded after each run. The profiles were run with catchability priors turned off so that they would not interfere with fit to any of the data in the model.

If the survey swept-area abundance data and model agree about stock size, then the CASA model's catchability estimates for the dredge, SMAST and HabCam data should be in the lower end of the range (Q=0.4-0.6) because of the way the survey data in CASA are scaled. At higher values of Q, the model estimates stock sizes lower than the swept-area abundance data and *vice-versa*.

Results indicate that the most of the data for all three areas fit best when dredge survey Q is higher than its expected value and estimated abundance is lower on average than indicated by the survey swept-area abundance data (Tables B6-2 to B6-4). This tendency is most pronounced in the Mid-Atlantic Bight area. The cause of these discrepancies is not clear.

Sensitivity analysis

To test the sensitivity of the model outputs to key assumptions, CASA model runs were conducted with alternative assumptions regarding natural mortality, survey priors and incidental mortality. Alternative assumptions about natural mortality on Georges Bank were M = 0.12 (as in SARC-50) and M = 0.20, and M = 0.15 (SARC-50) and M = 0.25 in the Mid-Atlantic. Runs were conducted with the survey priors turned off, at twice the assumed CVs ("loose priors": 0.3 for dredge and SMAST, and 0.2 for HabCam) and at half the assumed CVs ("tight priors": 0.075 for the dredge and SMAST, and 0.05 for HabCam). Alternative assumptions for incidental mortality were either zero or twice the assumed value (0.4 for Georges Bank and 0.2 for the Mid-Atlantic).

Variations in the assumed natural mortality had little effect on Georges Bank Open runs. Assumptions about survey priors had modest effects only in the last several years (Figure B6.50). The assumed value of natural mortality had a stronger effect on Georges Bank Closed runs, especially in the first 15 years after the closures. The higher natural mortality rate allowed the model to estimate a biomass closer to that estimated by the surveys during the 1998-2008 period. However, the value of natural mortality had little influence on the 2013 estimated biomass. Tighter survey priors induced higher biomass estimates, mainly from 2002-2013, whereas loose or no priors induced lower estimates.

The assumed natural mortality rate also had limited effects in the Mid-Atlantic Bight runs, and primarily affected the estimated biomass during 2000-2010. Loose or no survey priors decreased biomass estimates in the Mid-Atlantic, mainly in the last 5 years of the time series. Effects on fishing mortality were generally modest and in the reverse direction of effects on biomass (Figure B6.51). The assumed level of incidental mortality had little effect on model estimates of biomass (Figure B6.52).

Experimental runs with density-dependent natural mortality on juvenile scallops

Scallop abundance estimates from the CASA model were typically below those of the surveys when strong recruitment was observed in the surveys. This suggests that natural mortality of juveniles may increase at high density. If this is the case, CASA models would be below the surveys for those years because observations of the strong year class in subsequent years would indicate less scallops than would be expected based on the initial survey observations and assumptions regarding natural and incidental mortality. High natural mortality on large year classes of juveniles ignored in modeling would induce retrospective patterns like that observed, where estimates of strong year classes and abundance would decline as more years of data were added.

There is also experimental evidence of density-dependent natural mortality on juvenile sea scallops. Wong et al. (2005) seeded juvenile scallops in experimental plots at densities of 1, 6 or 69 m^{-2}. Scallop density in the high-density sites declined markedly due to both predation (and in particular predation by *Cancer* spp. crabs) and dispersal, resulting in final densities of about 1 m^{-2} regardless of treatment. Predation rates of *Cancer* crabs on juvenile sea scallops appear to be greater when scallops are more common than alternative prey species, and increase with increasing scallop density (Barbeau et al. 1998, Wong and Barbeau 2005). Thus, *Cancer* crabs are potential agents of density dependence in juvenile sea scallops; they primarily consume scallops less than 70 mm, and almost all less than 90 mm (Elner and Jamieson 1979, Lake et al. 1987).

In order to model density-dependent juvenile mortality, we defined the number of juveniles as the J = L(H), where H is scallop numbers at shell height and L is a declining logistic function. For this initial exploration, the inflection point L_{50} of the logistic function was set at 80 mm, and the slope of the logistic function was also fixed (Figure B6.53). Natural mortality of juveniles of shell height H was assumed to be M_0 (H) + kL(H)J, where M_0 is a fixed constant and k is a parameter estimated by the model. For this preliminary work, M_0 was set at half of the adult natural mortality (i.e., 0.08 for Georges Bank and 0.1 for the Mid-Atlantic) at small sizes, and increases to full adult natural mortality at large sizes (i.e., $M_0(H) = M[2-L(H)]/2$, where M is the natural mortality on adults).

Example runs are shown here for Georges Bank Open and Closed; density-dependence in the Mid-Atlantic model was difficult to estimate. Both Georges Bank models showed improved fits to the survey data, especially Gerorges Bank Closed (Figures B6.53 and B6.54). Estimated natural mortality of juveniles ranged between about 0.15 and 1. The working group thought these preliminary model runs were promising and recommended further development of this approach.

Empirical Assessment

The empirical assessment used simple techniques to estimate sea scallop stock abundance, biomass and fishing mortality in the Mid-Atlantic, Georges Bank and combined stock areas without using a stock assessment model (Appendix B5). The purpose was to evaluate the accuracy of CASA estimates as independently as possible by taking advantage of the three surveys (dredge, SMAST and HabCam) that can be used to estimate stock size directly. However, empirical results could be used in place of CASA model estimates if the later were unavailable. The data and various parameters used in the empirical analysis are a subset of those also used in the CASA model and were all obtained independently in field studies or other analyses rather than from a stock assessment model.

Empirical and CASA model estimates of abundance and fishing mortality show similar trends in all regions (Tables 3-4 and Figure 7 all in Appendix B5). However, empirical abundance estimates were usually higher reflecting the tension in CASA models between matching the scale of the abundance data (matching the prior on Q) versus fitting the survey and fishery data which was evident in likelihood profile analysis over a ranges of dredge survey catchability (Tables B6-2 to B6-4). As expected, fishing mortality estimates show the inverse pattern with empirical generally lower than CASA estimates.

Table B6.1. CASA model estimates and standard errors for July 1 abundance and biomass (40+mm SH), and fully recruited fishing mortality for George Bank open, closed, and total. (See following pages).

	Georges Bank Open						Georges Bank Closed						Georges Bank Total					
Year	Abund (millions)	SE	Biomass (mt)	SE	F	SE	Abund (millions)	SE	Biomass (mt)	SE	F	SE	Abund (millions)	SE	Biomass (mt)	SE	F	SE
1975	969	37	16322	622	0.08	0.01	537	23	10625	461	0.09	0.01	1507	623	26946	622	0.09	0.01
1976	1023	35	17449	666	0.19	0.01	601	23	11952	478	0.14	0.01	1624	667	29401	666	0.17	0.01
1977	859	32	16389	634	0.30	0.02	502	20	11651	464	0.28	0.02	1361	635	28040	634	0.29	0.02
1978	752	27	14047	567	0.34	0.02	460	18	10155	412	0.34	0.02	1212	568	24202	567	0.34	0.03
1979	602	24	11299	482	0.45	0.03	312	15	7504	353	0.58	0.04	914	483	18803	482	0.50	0.04
1980	678	25	9484	394	0.43	0.03	359	17	5948	291	0.49	0.04	1037	395	15432	394	0.45	0.03
1981	575	22	8118	313	0.63	0.04	299	15	5160	265	0.58	0.04	875	314	13279	313	0.61	0.05
1982	500	19	6080	249	0.87	0.06	241	15	4371	276	0.49	0.04	741	250	10451	249	0.73	0.06
1983	358	17	4632	230	0.74	0.05	206	18	3667	314	0.56	0.04	565	231	8298	230	0.67	0.05
1984	314	18	3978	244	0.54	0.03	230	21	3682	352	0.26	0.02	543	245	7660	244	0.43	0.04
1985	334	21	3792	257	0.61	0.04	265	26	4034	408	0.47	0.03	598	258	7827	257	0.54	0.05
1986	490	26	3676	239	1.19	0.08	392	35	4551	433	0.72	0.05	883	240	8227	239	0.95	0.09
1987	524	25	4389	239	0.84	0.05	440	45	5005	541	0.89	0.06	964	240	9394	239	0.86	0.08
1988	393	23	4233	270	0.95	0.06	804	62	7335	605	0.87	0.06	1197	271	11568	270	0.91	0.14
1989	451	26	3803	266	0.98	0.06	816	57	10092	728	0.52	0.04	1268	267	13895	266	0.65	0.09
1990	535	26	4033	229	1.21	0.08	674	44	9074	570	1.10	0.08	1209	230	13108	229	1.13	0.13
1991	634	26	4293	188	1.49	0.10	583	30	6445	313	1.44	0.10	1217	190	10738	188	1.46	0.14
1992	376	15	3366	135	1.69	0.11	352	24	4070	269	1.70	0.12	728	136	7435	135	1.69	0.16
1993	222	11	2270	119	1.13	0.07	343	34	3633	368	0.92	0.07	564	120	5903	119	1.02	0.13
1994	220	14	2200	143	0.53	0.03	351	37	4890	546	0.13	0.01	571	143	7090	143	0.26	0.04
1995	440	19	3278	166	0.55	0.04	522	44	7743	726	0.00	0.00	962	167	11022	166	0.17	0.04
1996	466	20	4369	196	0.77	0.05	629	48	11235	905	0.00	0.00	1095	197	15603	196	0.26	0.05
1997	451	22	4456	225	0.81	0.05	691	52	15342	1142	0.00	0.00	1142	226	19798	225	0.24	0.05
1998	637	33	5260	259	0.67	0.04	1014	64	20416	1347	0.00	0.00	1651	261	25676	259	0.30	0.04
1999	1015	44	7770	325	0.90	0.06	988	65	23875	1552	0.20	0.01	2003	328	31645	325	0.44	0.06
2000	1306	45	11600	404	0.60	0.04	1687	86	29443	1689	0.15	0.01	2993	406	41043	404	0.35	0.04
2001	1328	42	14741	468	0.59	0.04	1900	84	38707	1881	0.03	0.002	3229	469	53448	468	0.31	0.04
2002	1174	39	15006	478	0.65	0.04	1918	80	47889	2063	0.00	0.00	3092	480	62895	478	0.29	0.04
2003	1210	37	14775	481	0.53	0.03	2058	79	55666	2216	0.00	0.00	3268	482	70441	481	0.19	0.03
2004	1149	37	16192	521	0.27	0.02	1860	72	58707	2292	0.07	0.005	3008	523	74899	521	0.14	0.02
2005	1257	43	18019	576	0.34	0.02	1676	70	55653	2303	0.15	0.01	2933	577	73672	576	0.21	0.03
2006	1213	47	16459	558	0.85	0.05	1380	66	47466	2251	0.25	0.02	2593	560	63925	558	0.44	0.06
2007	1562	61	16564	605	0.60	0.04	1359	72	41169	2219	0.16	0.01	2921	608	57733	605	0.30	0.04
2008	1694	73	19653	800	0.57	0.04	1376	77	39837	2245	0.07	0.005	3070	803	59489	800	0.25	0.04
2009	1838	91	22826	1101	0.48	0.03	1565	89	41774	2358	0.05	0.004	3403	1105	64600	1101	0.24	0.03
2010	1862	105	26747	1485	0.24	0.01	1689	101	44361	2558	0.09	0.01	3551	1488	71109	1485	0.16	0.02
2011	1994	127	31320	1924	0.17	0.01	1928	127	46717	2908	0.18	0.01	3923	1928	78037	1924	0.17	0.02
2012	1871	140	32374	2400	0.36	0.02	2077	154	48792	3423	0.21	0.02	3948	2404	81166	2400	0.29	0.03
2013	2006	211	29533	2834	0.54	0.03	2756	251	56926	4275	0.06	0.00	4762	2842	86460	2834	0.30	0.04

Figure B6.1 continued. CASA model estimates and standard errors for July 1 abundance and biomass (40+mm SH), and fully recruited fishing mortality for Mid-Atlantic and Total (GB and MA combined).

			Mid-Atlantic						**Total**			
Year	Abund	SE	Biomass	SE	F	SE	Abund	SE	Biomass	SE	F	SE
	(millions)		(mt)				(millions)		(mt)			
1975	516	26	5890	305	0.56	0.05	2023	50	32837	832	0.17	0.02
1976	632	22	6709	355	1.02	0.10	2256	47	36110	893	0.31	0.03
1977	644	21	8372	307	0.53	0.05	2004	43	36412	844	0.35	0.03
1978	496	15	7821	246	1.07	0.10	1708	36	32023	743	0.49	0.04
1979	328	10	6108	194	0.97	0.09	1241	30	24911	628	0.59	0.04
1980	318	10	4820	172	0.46	0.04	1355	32	20252	519	0.45	0.03
1981	417	12	5601	192	0.17	0.02	1292	30	18880	453	0.50	0.04
1982	473	14	6912	226	0.29	0.03	1215	28	17363	435	0.56	0.04
1983	528	15	7093	236	0.56	0.05	1092	29	15391	455	0.62	0.05
1984	573	18	7021	249	0.68	0.07	1116	33	14681	496	0.54	0.05
1985	799	24	8002	286	0.61	0.06	1397	41	15829	561	0.58	0.05
1986	1087	32	11482	382	0.44	0.04	1969	54	19708	625	0.65	0.05
1987	1270	37	12113	393	0.93	0.09	2234	63	21506	711	0.90	0.08
1988	1230	40	12613	445	0.77	0.07	2427	77	24181	798	0.84	0.07
1989	1212	35	11149	368	1.20	0.12	2480	72	25044	858	0.87	0.08
1990	1097	30	10541	326	1.06	0.10	2306	60	23649	695	1.10	0.09
1991	735	21	8520	263	1.10	0.11	1952	45	19258	450	1.30	0.10
1992	515	18	5733	213	1.12	0.11	1242	34	13168	369	1.47	0.11
1993	941	35	6381	257	0.90	0.09	1505	50	12284	464	0.97	0.08
1994	1405	59	9885	465	1.38	0.13	1976	71	16975	731	0.78	0.10
1995	1044	30	10031	306	1.51	0.15	2007	57	21052	805	0.81	0.11
1996	583	18	7737	246	0.81	0.08	1678	55	23340	958	0.46	0.05
1997	649	25	6606	257	0.61	0.06	1790	62	26404	1191	0.33	0.03
1998	1484	49	9934	364	1.08	0.10	3135	87	35610	1419	0.46	0.04
1999	2655	74	22092	691	0.80	0.08	4658	108	53736	1730	0.57	0.05
2000	3275	84	36301	1025	0.66	0.06	6268	128	77344	2016	0.51	0.06
2001	3355	80	43631	1155	0.69	0.07	6583	123	97079	2257	0.51	0.06
2002	3076	73	44862	1165	0.68	0.07	6168	115	107757	2417	0.47	0.05
2003	3991	87	45517	1109	0.75	0.07	7259	124	115958	2524	0.43	0.05
2004	3801	88	50849	1198	0.93	0.09	6809	120	125748	2638	0.43	0.06
2005	3790	92	52694	1334	0.80	0.08	6723	123	126366	2723	0.41	0.04
2006	3856	99	61284	1650	0.35	0.03	6449	128	125209	2846	0.40	0.03
2007	3681	92	62298	1673	0.62	0.06	6602	132	120031	2844	0.46	0.05
2008	3879	88	58561	1504	0.70	0.07	6948	138	118050	2818	0.47	0.06
2009	3209	74	54706	1272	0.82	0.08	6612	147	119306	2897	0.49	0.06
2010	2343	61	44283	1215	0.85	0.08	5894	158	115392	3197	0.43	0.05
2011	1675	57	33973	1159	0.87	0.08	5598	188	112010	3674	0.39	0.05
2012	2808	134	30516	1468	0.74	0.07	6756	248	111682	4431	0.40	0.03
2013	3253	182	46101	2649	0.39	0.04	8014	375	132561	5772	0.32	0.03

Table B6.2. CASA model likelihood profile analysis over a range of values for dredge survey catchability (Q) in the MAB region. Catchability priors were turned off in profile runs. The basecase run (with priors turned on) is colored yellow, runs with Q in the 0.4-0.6 expected range based on swept-area abundance are blue, and the run with the best fit to the data are salmon in color. The best fit occurs where the likelihood is zero (bold face).

Dredge survey Q	0.40	0.50	0.60	0.70	0.80	0.87	1.00	1.09	1.19
Total unweighted	618.500	480.800	264.000	137.200	48.800	166.600	**0.000**	200.100	336.100
Catch weight	208.349	194.825	183.041	153.940	44.222	2.902	**0.000**	37.589	124.802
Recruitment deviations	15.023	10.032	6.819	3.128	**0.000**	0.054	0.145	0.275	0.981
Survey trends-all	133.689	73.315	34.576	14.634	**0.000**	0.036	4.412	8.122	13.144
Length data-all	295.900	237.100	74.000	**0.000**	39.100	43.600	29.900	188.600	231.600
Survey trends									
Dredge	116.668	61.413	27.035	10.436	0.238	**0.000**	3.760	6.556	10.062
SMAST.LrgCam	3.447	2.008	1.269	1.253	**0.000**	0.135	2.860	4.160	5.381
Winter.BTS	11.039	7.668	5.160	3.130	0.366	0.372	0.124	0.070	**0.000**
Unlined.Dredge	**0.000**	0.001	0.216	0.265	0.260	0.286	0.314	0.118	0.176
HabCam	5.317	5.007	3.677	2.333	1.918	2.025	0.137	**0.000**	0.307
Length data									
Commercial.Fishery	152.900	152.930	35.200	**0.000**	11.880	24.840	29.840	190.150	211.040
Dredge	144.700	87.300	42.700	4.700	37.600	28.600	3.600	**0.000**	15.000
SMAST.LrgCam	10.061	9.551	8.364	6.129	0.110	**0.000**	6.562	11.919	16.906
Winter.BTS	**0.000**	0.080	0.750	1.200	1.960	1.860	2.150	2.030	2.280
Unlined.Dredge	0.297	0.299	2.289	5.140	5.039	5.258	5.197	0.093	**0.000**
HabCam	5.450	4.350	2.190	0.290	**0.000**	0.420	0.020	1.860	3.870
Mean 2011-2014 biomass (mt)	434,402	218,487	105,529	60,996	54,785	49,710	28,262	23,758	20,395
Mean 2011-2014 abun. wtd. F	0.01	0.02	0.06	0.12	0.14	0.15	0.31	0.38	0.45

Table B6.3 CASA model likelihood profile analysis over a range of values for dredge survey catchability (Q) in the GBK-open region. Catchability priors were turned off in profile runs. The basecase run (with priors turned on) is colored yellow, runs with Q in the 0.4-0.6 expected range based on swept-area abundance are blue, and the run with the best fit to the data are salmon in color. The best fit occurs where the likelihood is zero (bold face).

Dredge survey Q	0.40	0.50	0.60	0.66	0.75	0.84	0.98	1.09	1.09
Total unweighted	146.13	56.34	6.89	122.74	**0.00**	37.65	140.49	195.09	236.24
Catch weight	9.32	5.35	1.20	**0.00**	1.78	11.45	16.86	10.53	11.29
Recruitment deviations	20.90	13.53	6.54	3.36	**0.00**	0.08	4.44	7.02	8.19
Survey trends-all	116.68	49.64	11.75	1.74	**0.00**	6.88	3.63	0.92	3.91
Length data-all	11.83	0.41	**0.00**	2.43	10.82	31.84	128.17	189.19	224.95
Survey trends									
Dredge	89.48	35.27	7.60	0.94	0.25	4.54	1.07	**0.00**	2.89
SMAST.LrgCam	28.16	15.34	5.11	1.66	**0.00**	0.57	0.45	0.24	0.27
Unl.10ft.Dredge.40+mm	0.88	0.89	0.90	0.91	0.92	0.89	0.47	0.14	**0.00**
HabCam	0.02	**0.00**	0.01	0.09	0.70	2.75	3.51	2.42	2.61
Length data									
Commercial.Fishery	1.59	**0.00**	3.40	7.49	13.81	27.06	48.45	62.18	72.65
Dredge	15.04	8.82	4.18	1.25	**0.00**	4.28	64.41	99.38	118.23
SMAST.LrgCam	3.04	**0.00**	1.16	2.75	6.27	9.52	22.83	33.86	39.59
Unl.10ft.Dredge.40+mm	0.05	0.07	0.09	0.09	0.08	**0.00**	0.29	1.06	1.36
HabCam	1.44	0.86	0.51	0.18	**0.00**	0.31	1.52	2.03	2.44
Mean 2011-2014 biomass (mt)	181,251	97,471	53,744	38,653	25,051	19,297	18,282	18,897	18,377
Mean 2011-2014 abun. wtd. F	0.03	0.05	0.10	0.15	0.27	0.46	0.53	0.49	0.52

Table B6.4. CASA model likelihood profile analysis over a range of values for dredge survey catchability (Q) in the GBK-closed region. Catchability priors were turned off in profile runs. The basecase run (with priors turned on) is colored yellow, runs with Q in the 0.4-0.6 expected range based on swept-area abundance are blue, and the run with the best fit to the data are salmon in color. The best fit occurs where the likelihood is zero (bold face).

Dredge survey Q	**0.40**	**0.50**	**0.60**	**0.68**	**0.75**	**0.85**	**1.00**	**1.10**	**1.20**
Total unweighted	36.04	22.05	11.81	142.32	2.40	**0.00**	2.50	8.54	18.41
Catch weight	0.01	0.02	0.02	0.10	**0.00**	0.05	0.47	1.15	2.32
Recruitment deviations	14.95	11.32	8.32	5.56	4.77	2.96	1.04	0.30	**0.00**
Survey trends-all	14.33	7.22	3.06	**0.00**	1.19	2.59	8.51	14.94	23.17
Length data-all	14.60	11.33	8.26	8.96	4.30	2.26	0.33	**0.00**	0.77
Survey trends									
Dredge	8.53	3.90	1.53	**0.00**	0.58	1.13	3.28	5.48	8.25
SMAST.LrgCam	5.35	2.93	1.20	**0.00**	0.41	1.37	5.35	9.73	15.33
Unl.10ft.Dredge.40+mm	**0.00**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HabCam	0.85	0.80	0.73	0.40	0.60	0.49	0.28	0.13	**0.00**
Length data									
Commercial.Fishery	1.06	1.22	1.18	2.51	0.89	0.61	0.17	**0.00**	0.21
Dredge	10.72	8.59	6.68	4.52	4.22	2.87	1.31	0.55	**0.00**
SMAST.LrgCam	5.82	4.37	3.05	4.95	1.44	0.66	0.02	**0.00**	0.42
Unl.10ft.Dredge.40+mm	0.08	0.10	0.12	0.14	0.13	0.12	0.09	0.05	**0.00**
HabCam	0.08	0.22	0.40	**0.00**	0.79	1.16	1.91	2.57	3.31
Mean 2011-2014 biomass (mt)	125,498	88,137	64,526	50,812	42,317	32,537	22,304	17,508	13,888
Mean 2011-2014 abun. wtd. F	0.06	0.08	0.12	0.15	0.19	0.25	0.37	0.49	0.62

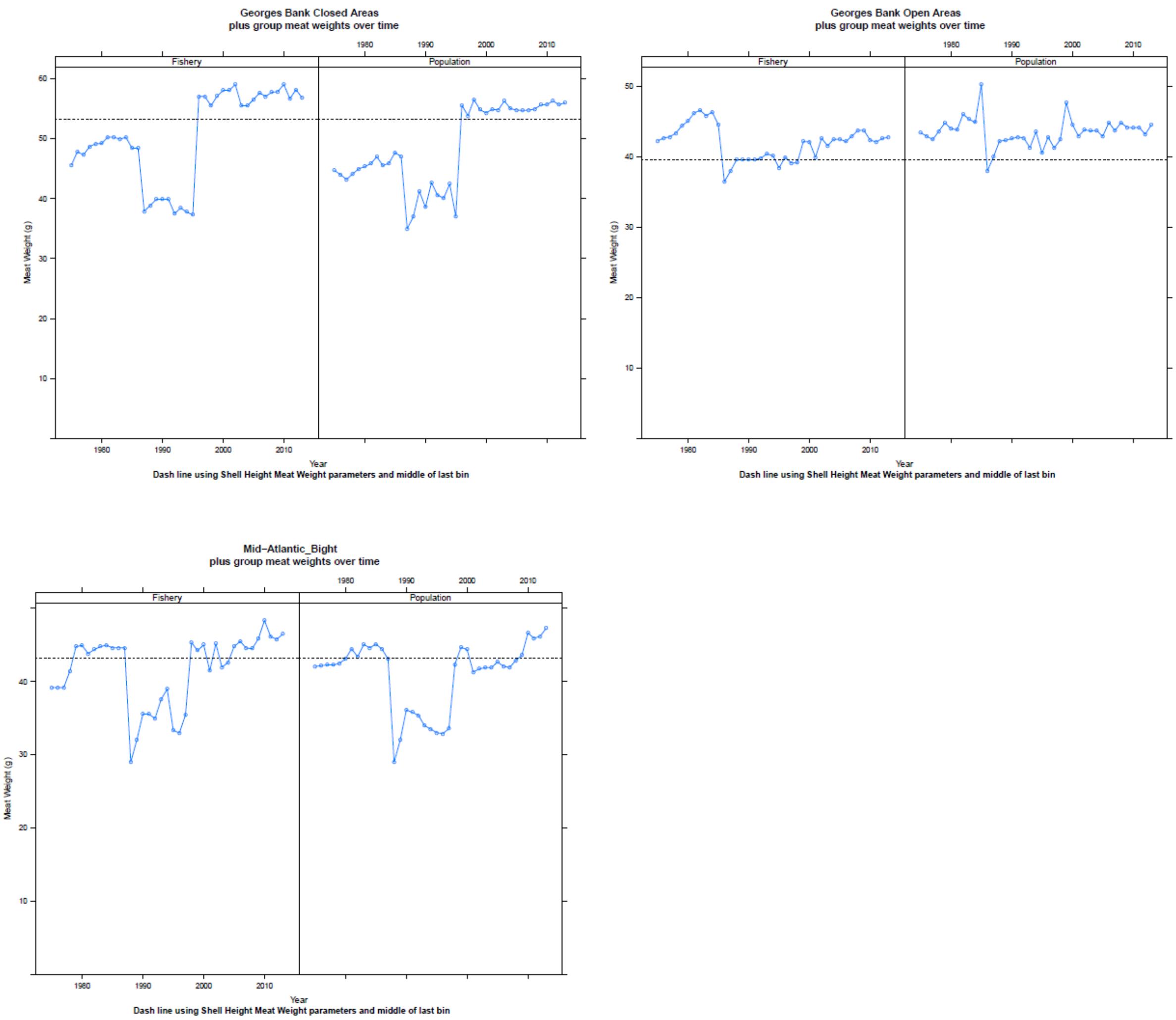
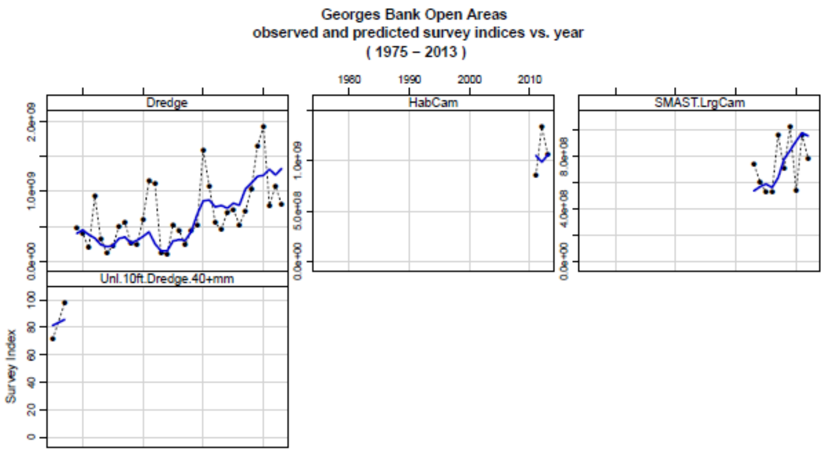


Figure B6.1. Estimated plus group meat weights for the population and the fishery in the open and closed portions of Georges Bank, and in the Mid-Atlantic Bight. The plus group represents scallops in the largest bin which contained L_∞.

(A)



(B)

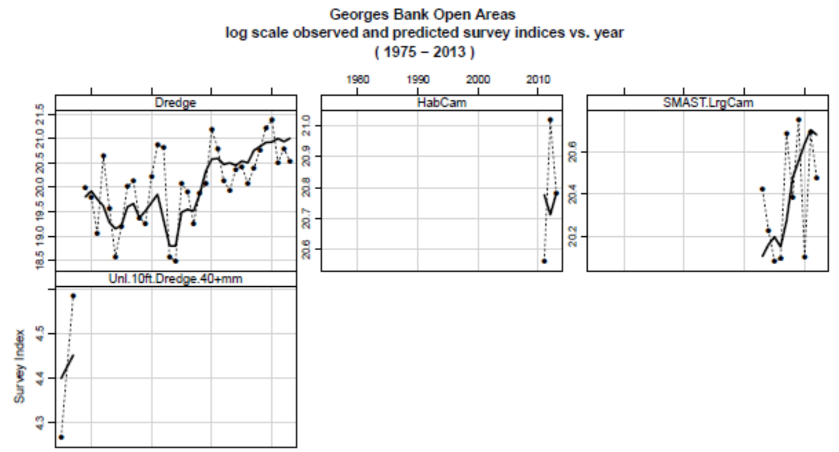


Figure B6.2. Observed survey trend (solid circles) and corresponding model estimates (lines) for the NEFSC lined dredge survey, the HabCam survey, the SMAST large camera survey and the NEFSC unlined dredge survey on Georges Bank open areas. Results are shown on a linear scale (A) and a log scale (B).

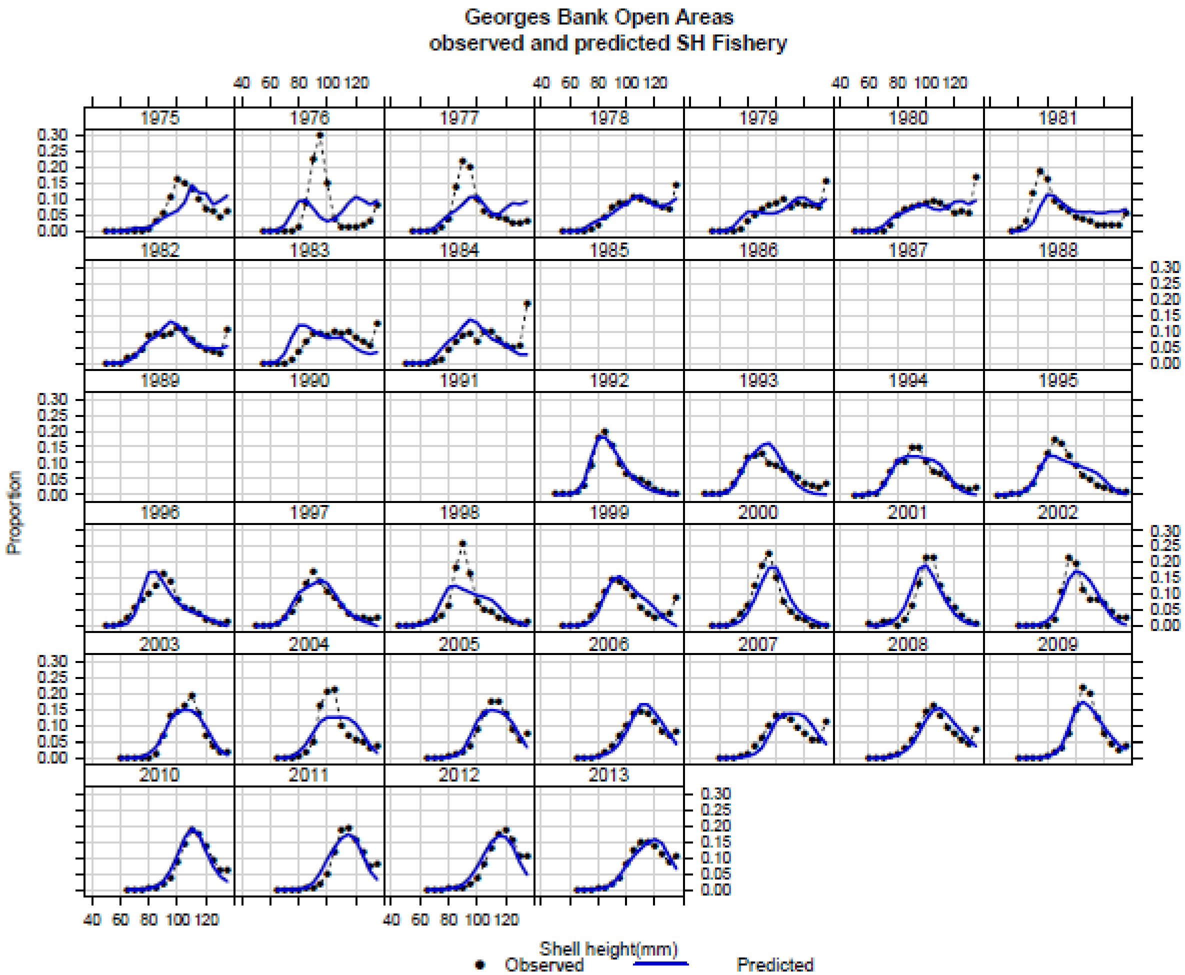


Figure B6.3. Comparison of observed fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for Georges Bank open areas.

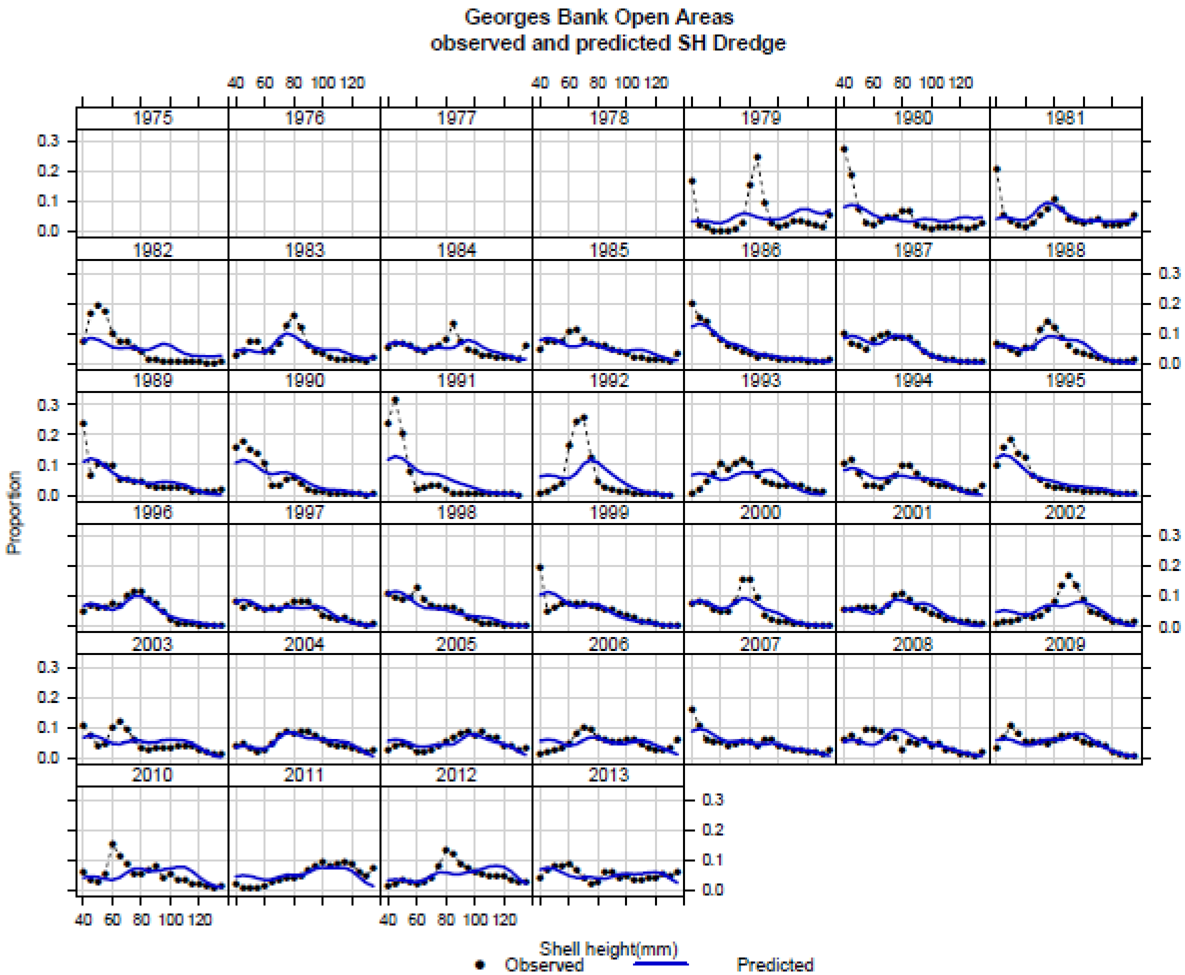


Figure B6.4. NEFSC lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank open areas.

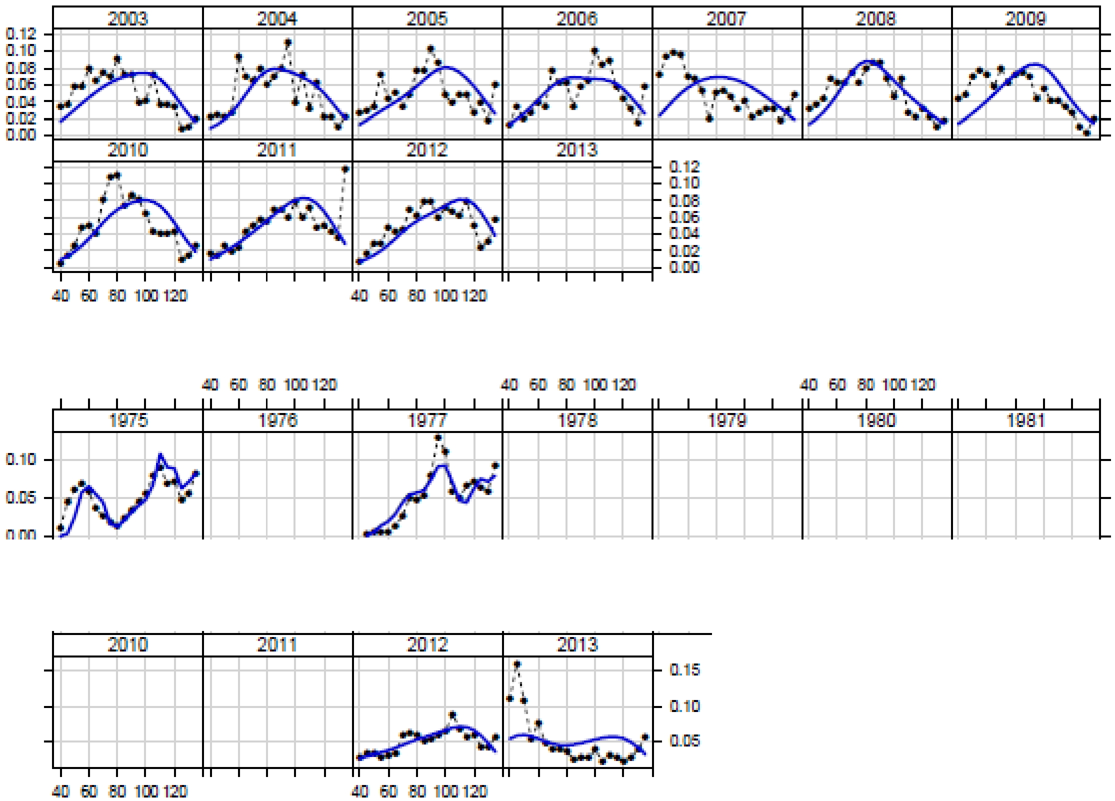


Figure B6.5. Shell height proportions for the SMAST large camera survey (top), the NEFSC unlined dredge survey (middle) and the HabCam survey (bottom) with model predicted proportions (lines) for Georges Bank.

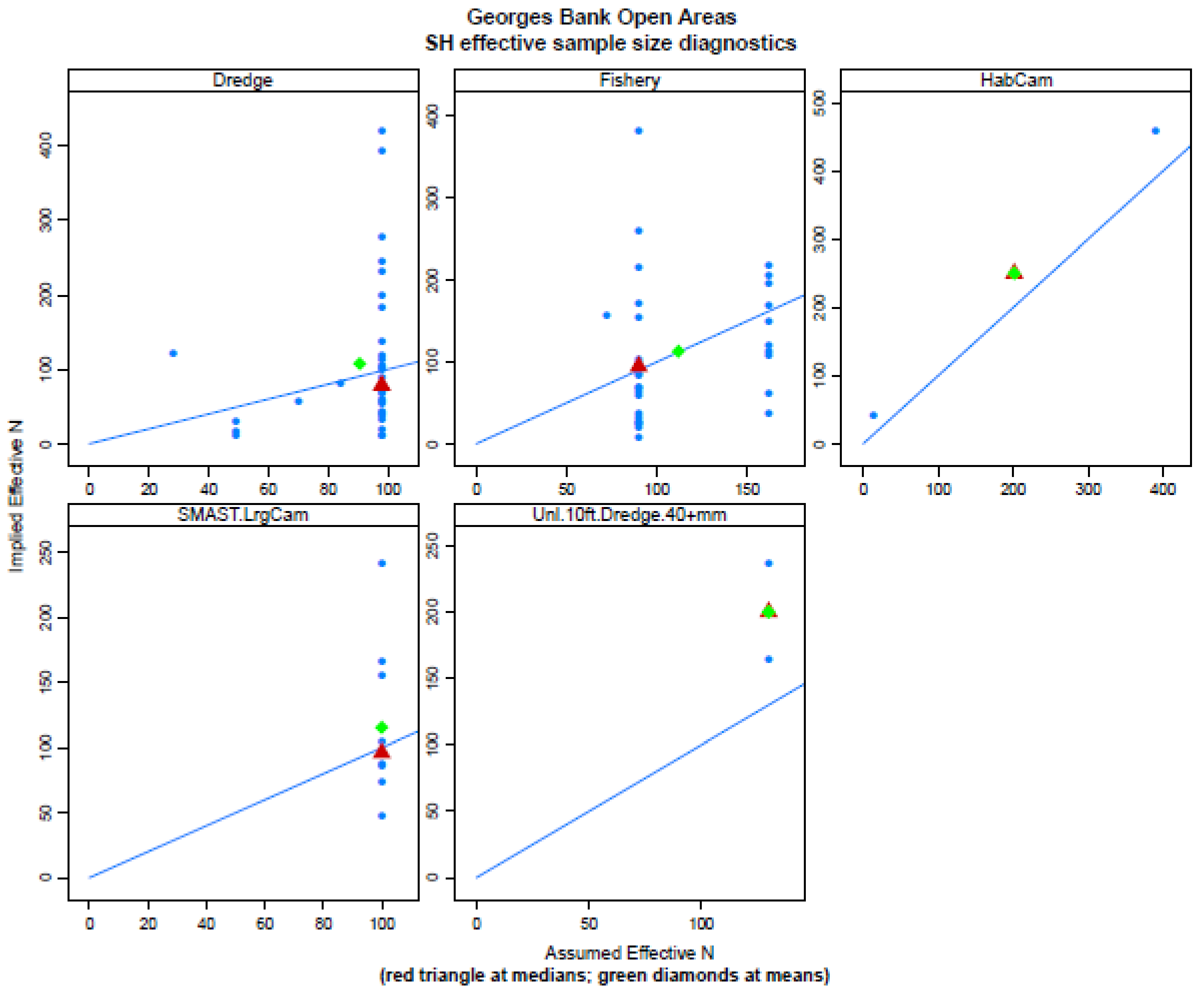


Figure B6.6. Assumed and model implied effective sample sizes for the four surveys (NEFSC unlined dredge, HabCam, SMAST large camera, NEFSC unlined dredge) and the fishery shell height compositions for Georges Bank open areas. The triangle is the median and the diamond is the mean.

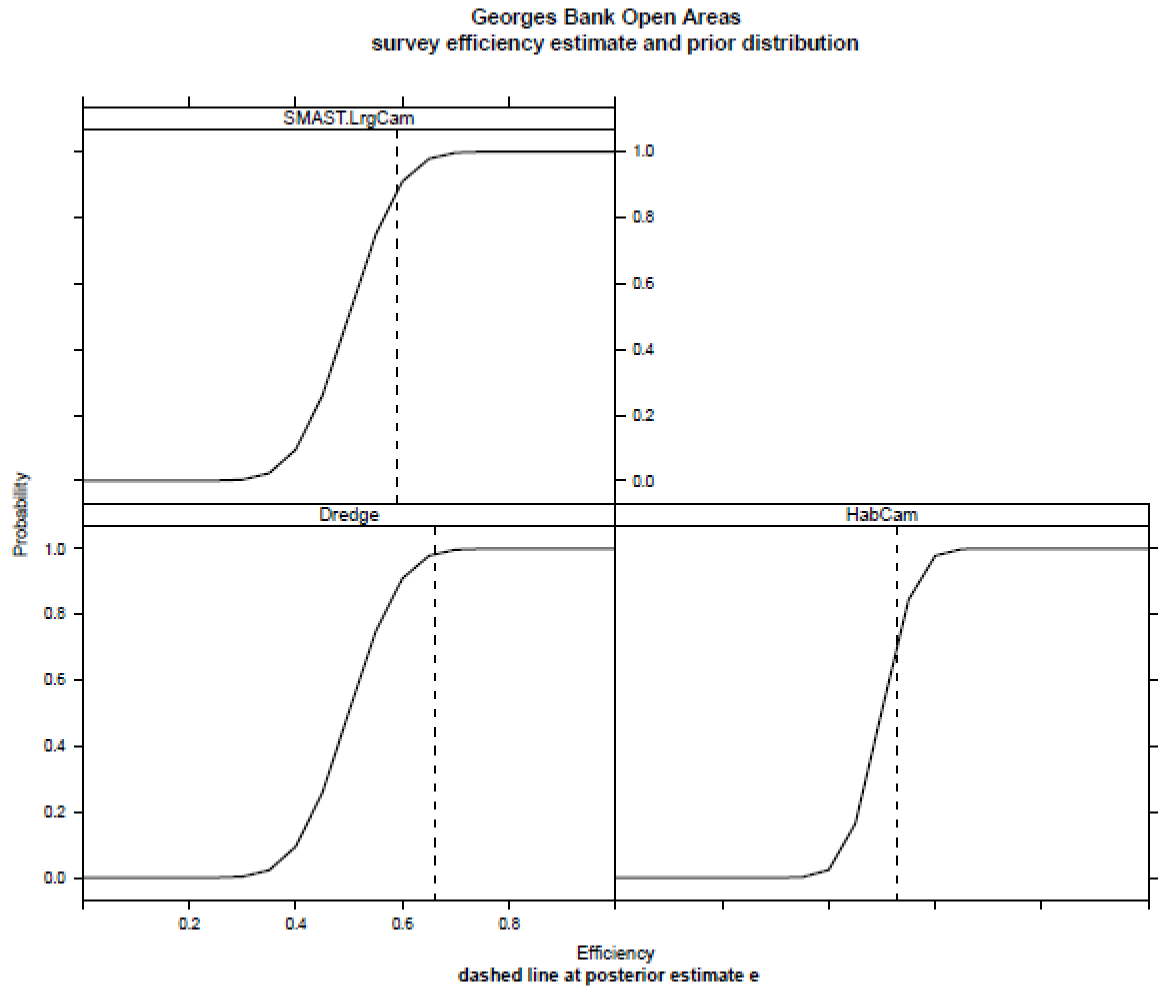
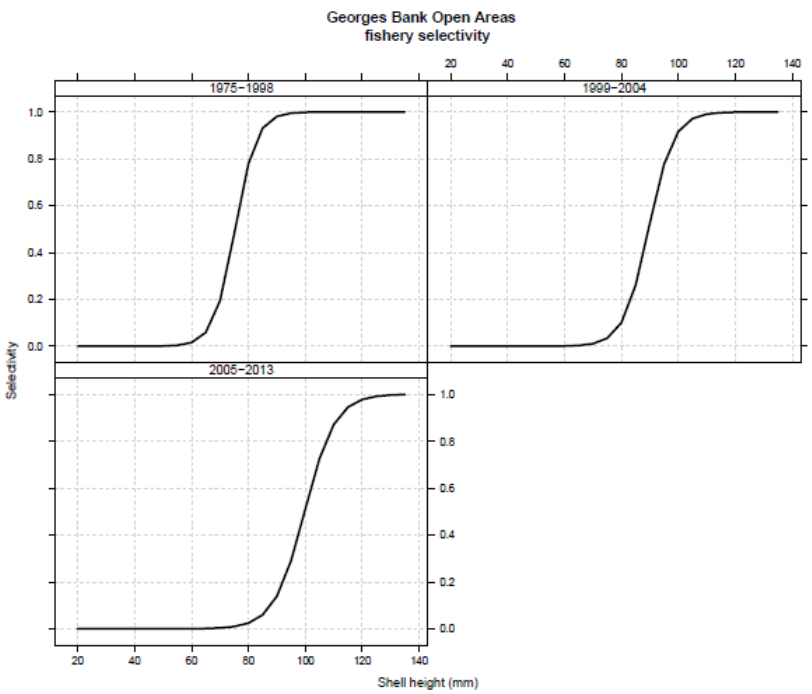


Figure B6.7. Prior cumulative distributions for catchability of the large camera video survey (top) lined dredge survey (bottom left) and HabCam survey (bottom right) for Georges Bank open areas. The dashed lines are the mean posterior estimate for survey catchability. For the purposes of this plot, the surveys were adjusted to have a mean prior catchability of 0.5

(A)



(B)

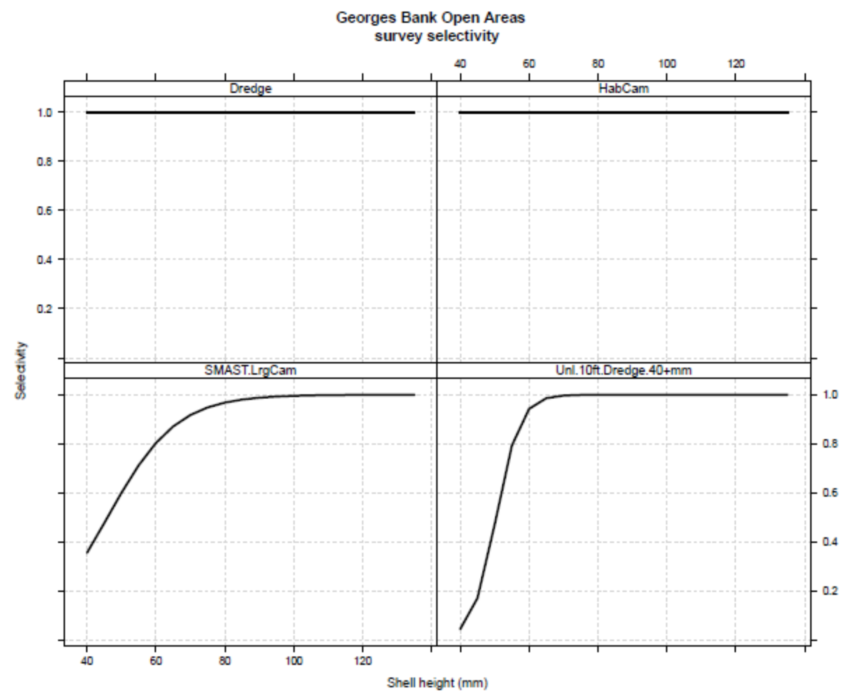


Figure B6.8. (A) Estimated fishery selectivity curves and (B) assumed survey selectivity curves (lined dredge top left, HabCam top right, large camera bottom left, and unlined dredge bottom right) for Georges Bank open areas.

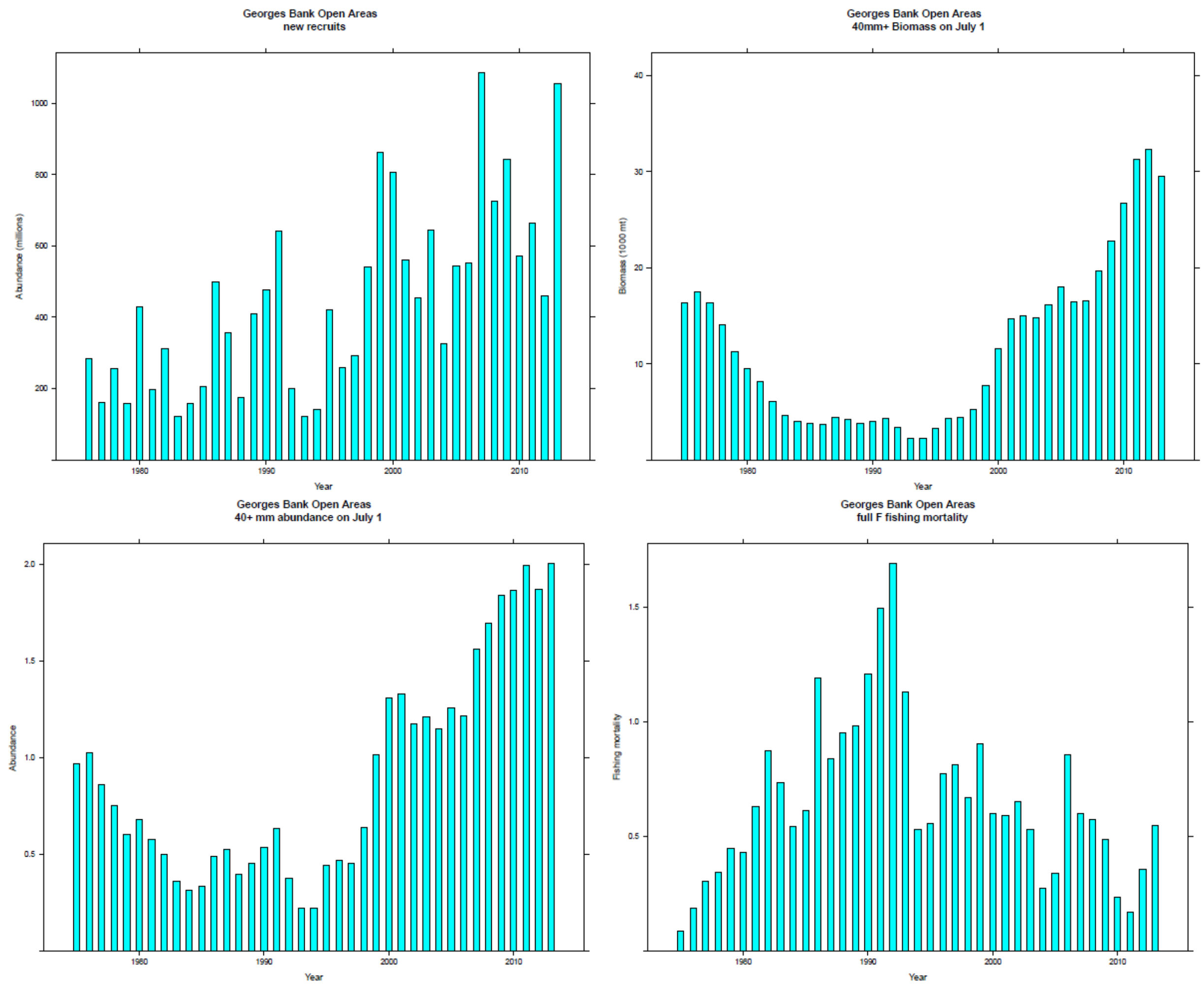


Figure B6.9. CASA model estimated recruitment (top left), July 1 biomass (top right), July 1 abundance (bottom left) and fully recruited fishing mortality (bottom right) for Georges Bank open areas.

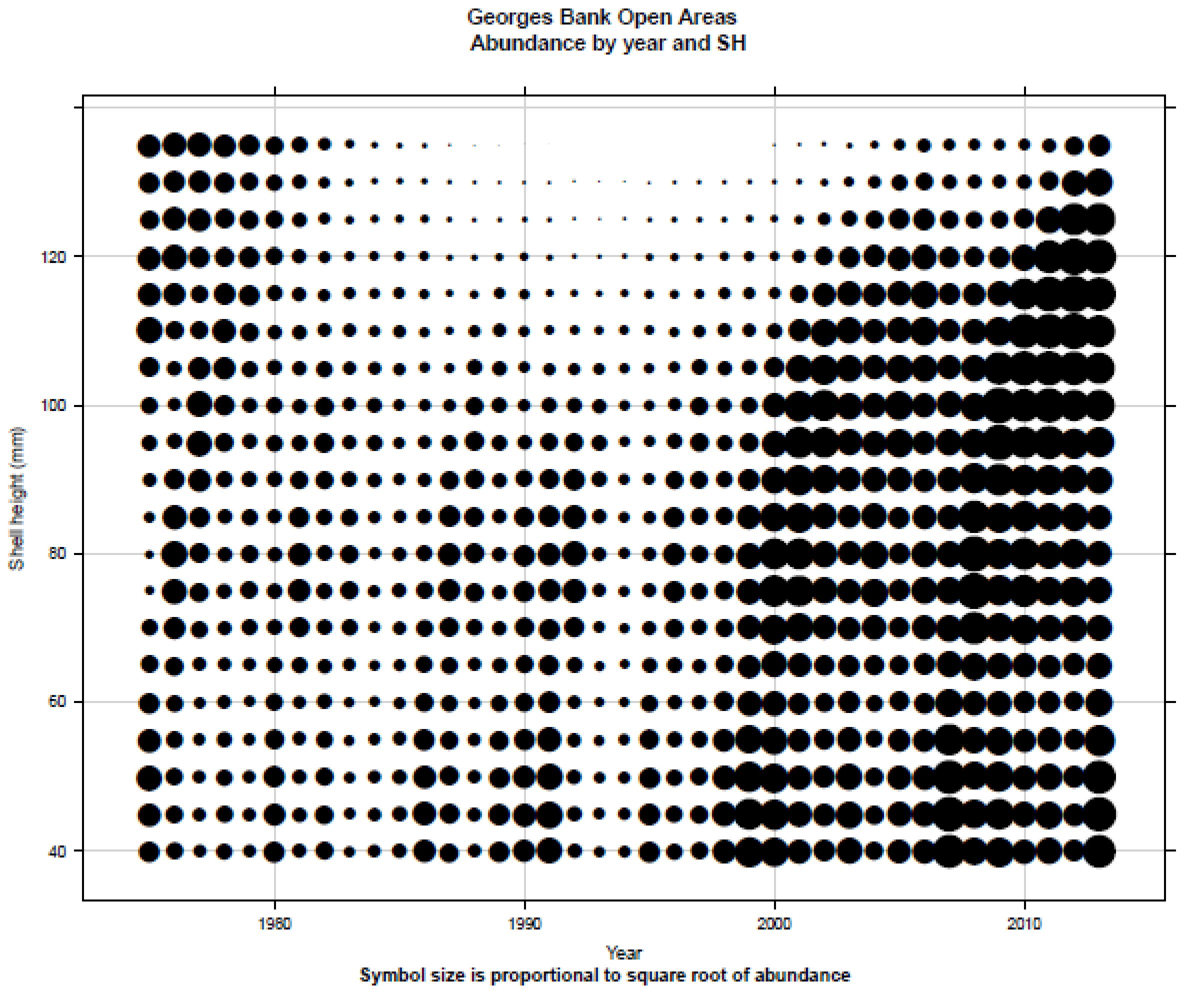


Figure B6.10. Model estimated abundances at shell height for Georges Bank open areas. Symbol areas are proportional to abundance.

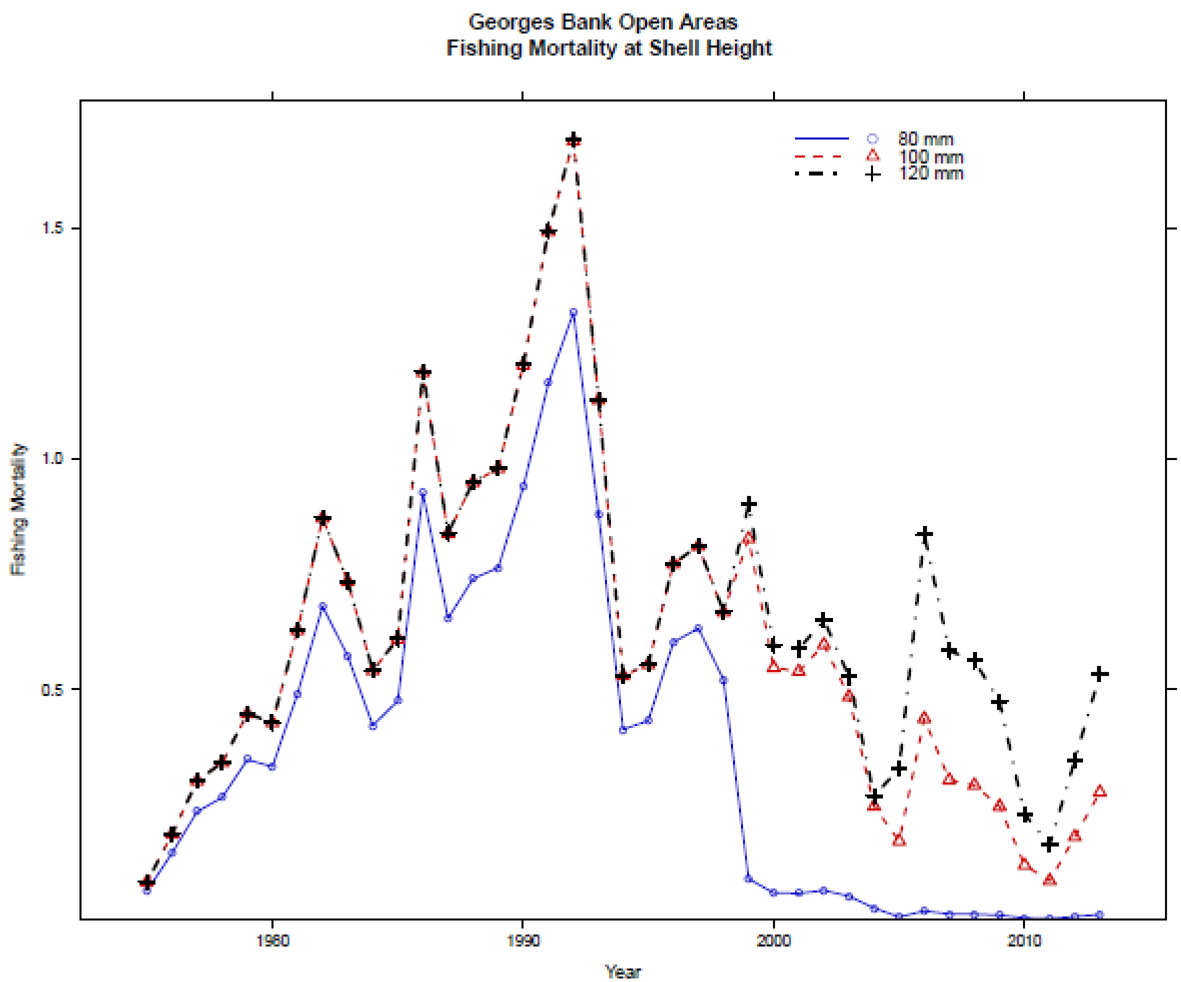


Figure B6.11. CASA model estimated fishing mortality at 80 mm (solid line with circles), 100 mm (dashed line with triangles) and 120 mm SH (dashed line with crosses) for Georges Bank open areas.

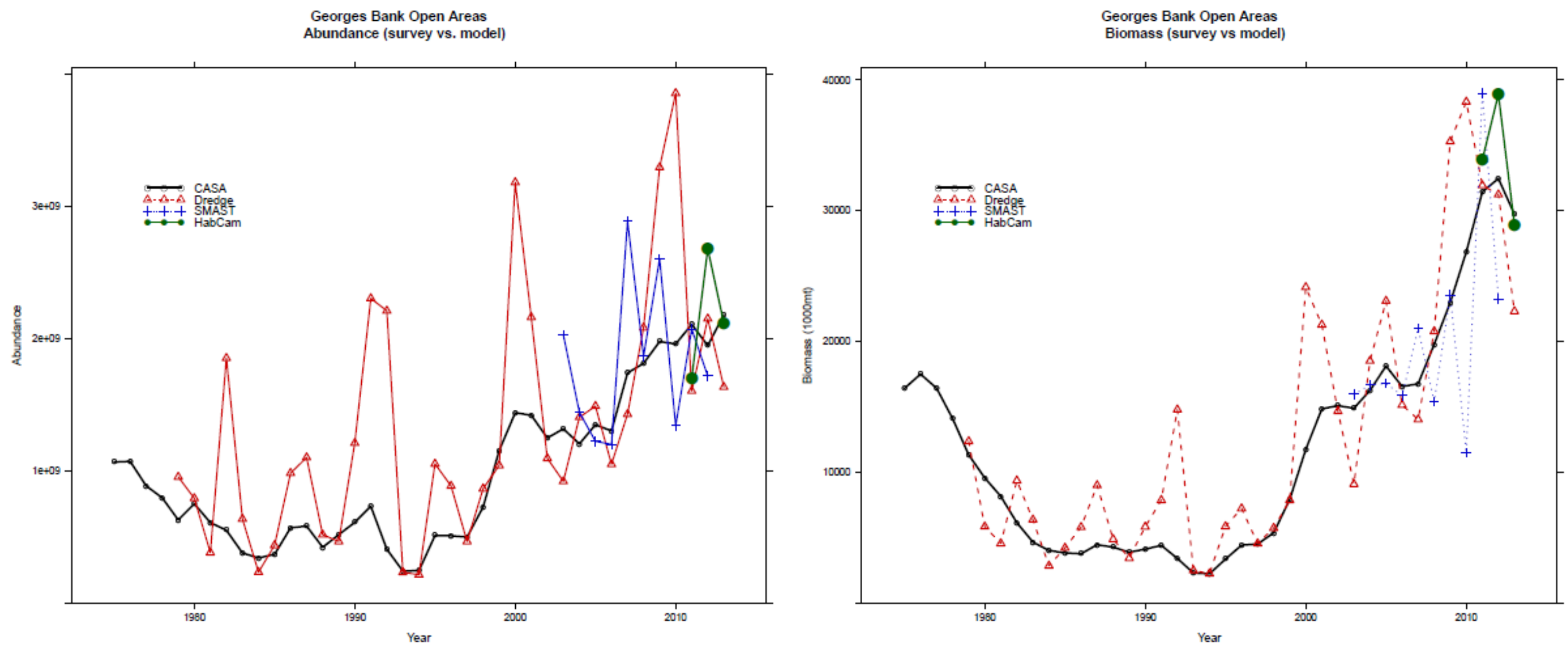


Figure B6.12. Comparison of CASA model estimated abundance (left) and biomass (right) with expanded estimates from the lined dredge survey (dashed red line with triangles), SMAST large camera survey (dotted blue line with crosses) and HabCam (solid line with circles) for Georges Bank open areas.

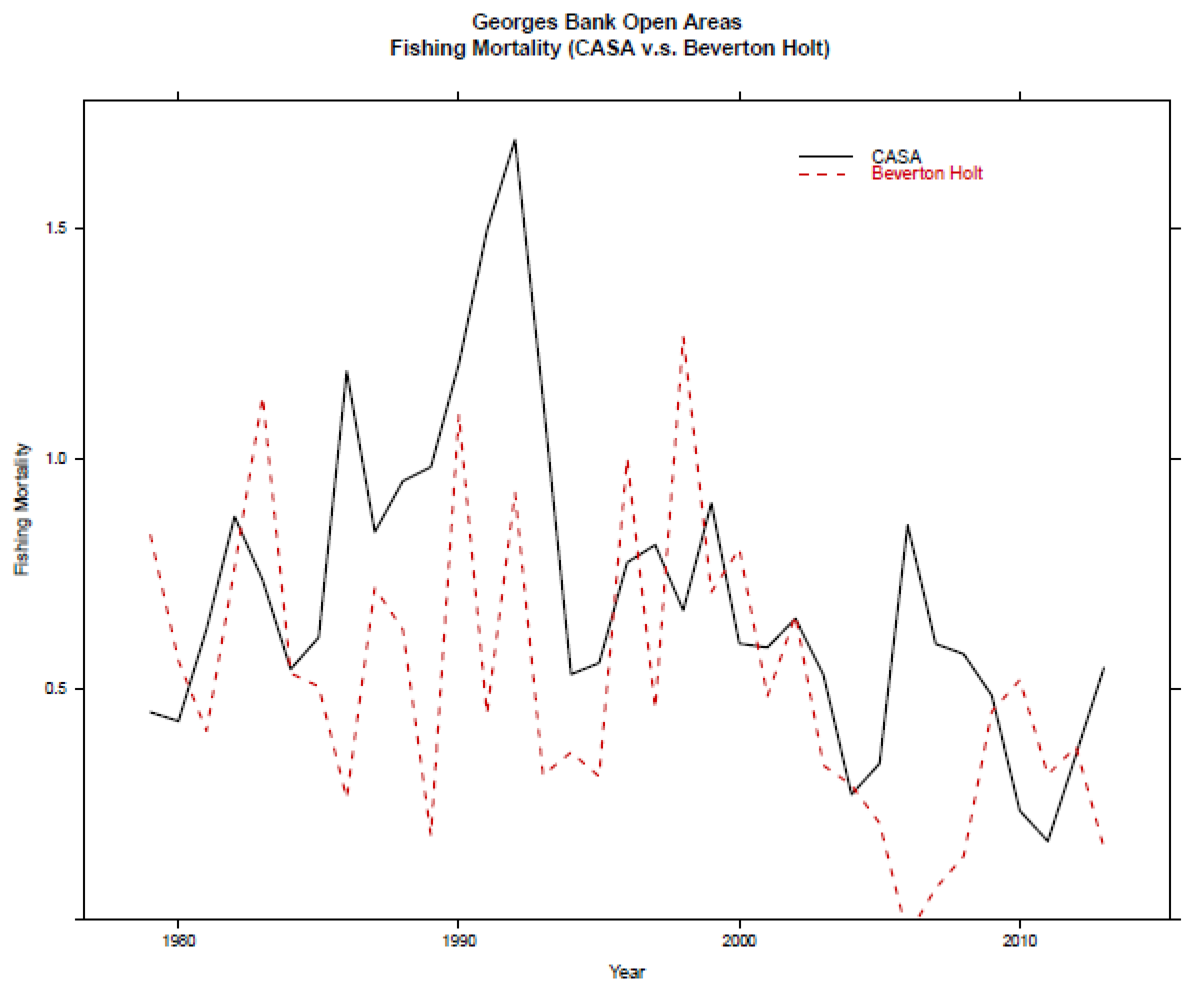


Figure B6.13. Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium length-based estimator for Georges Bank open areas.

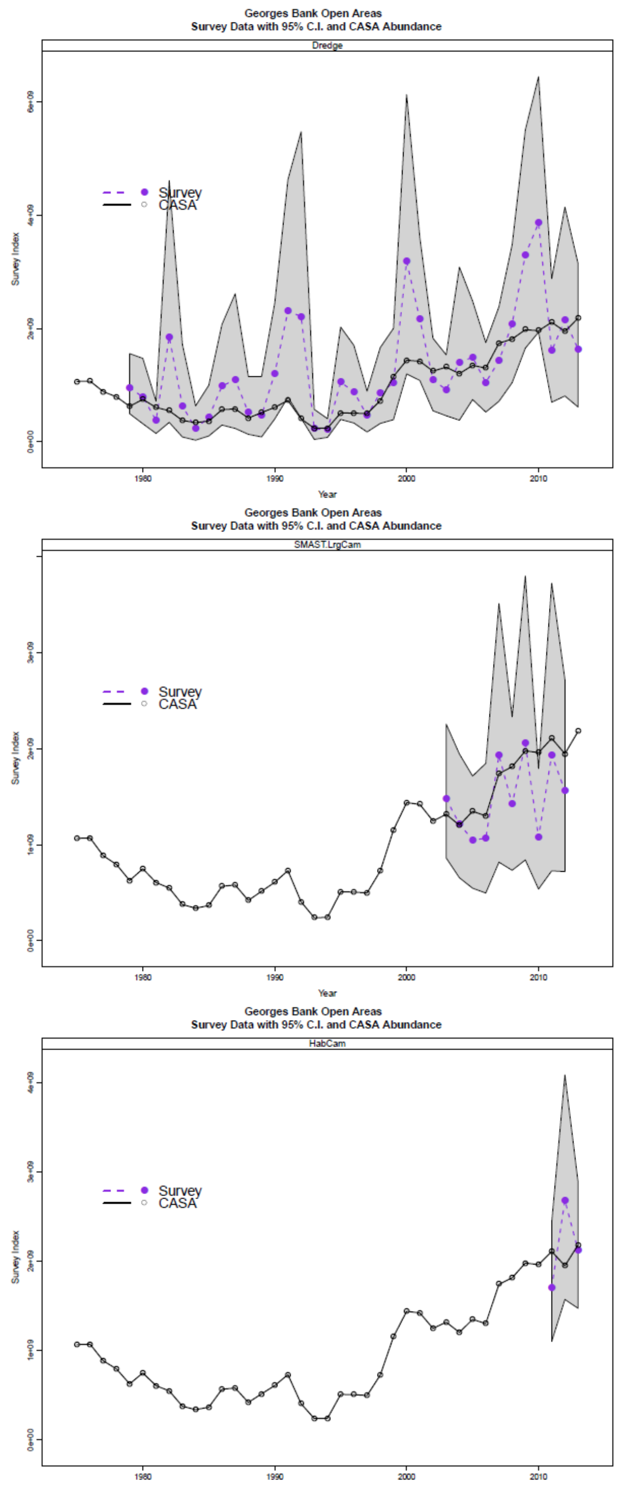


Figure B6.14. CASA model (black line with solid circles) for Georges Bank open areas compared to expanded survey estimates with their 95% C.I.s: dredge (top), SMAST (middle), and HabCam (lower)

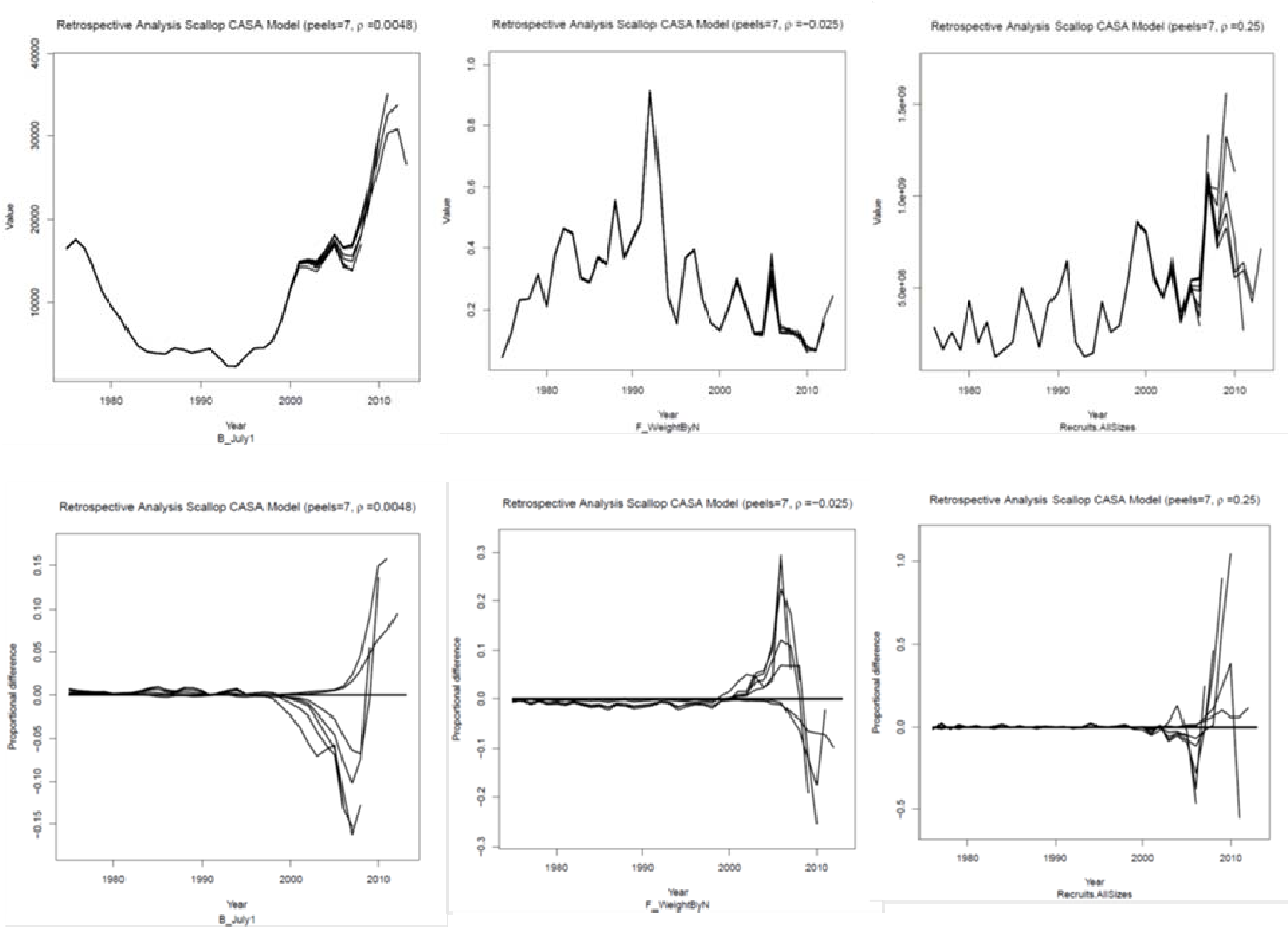
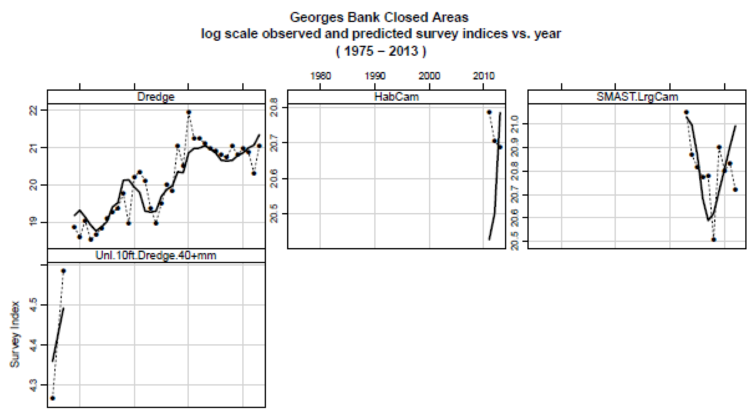


Figure B6.15. Retrospective plots for biomass, fishing mortality and recruitment, shown both on absolute and relative scales for Georges Bank open areas.

(A)



(B)

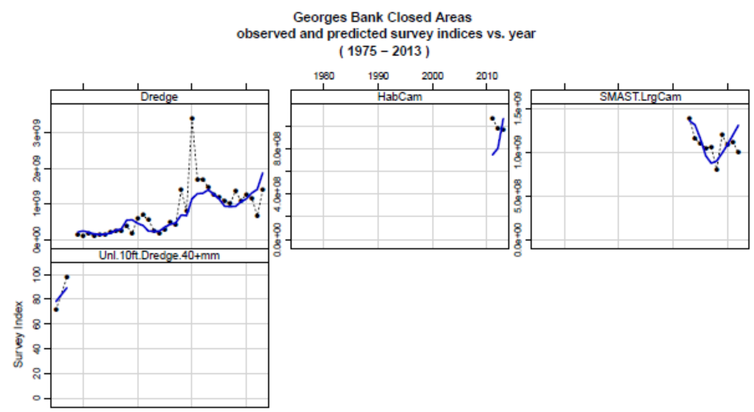


Figure B6.16. Comparison between survey trend (solid circles) and corresponding model estimates (lines) for the NEFSC lined dredge survey, the HabCam survey, The SMAST large camera survey and the NEFSC unlined dredge survey in the Georges Bank closed areas. Results are shown on a linear scale (A) and a log scale (B).

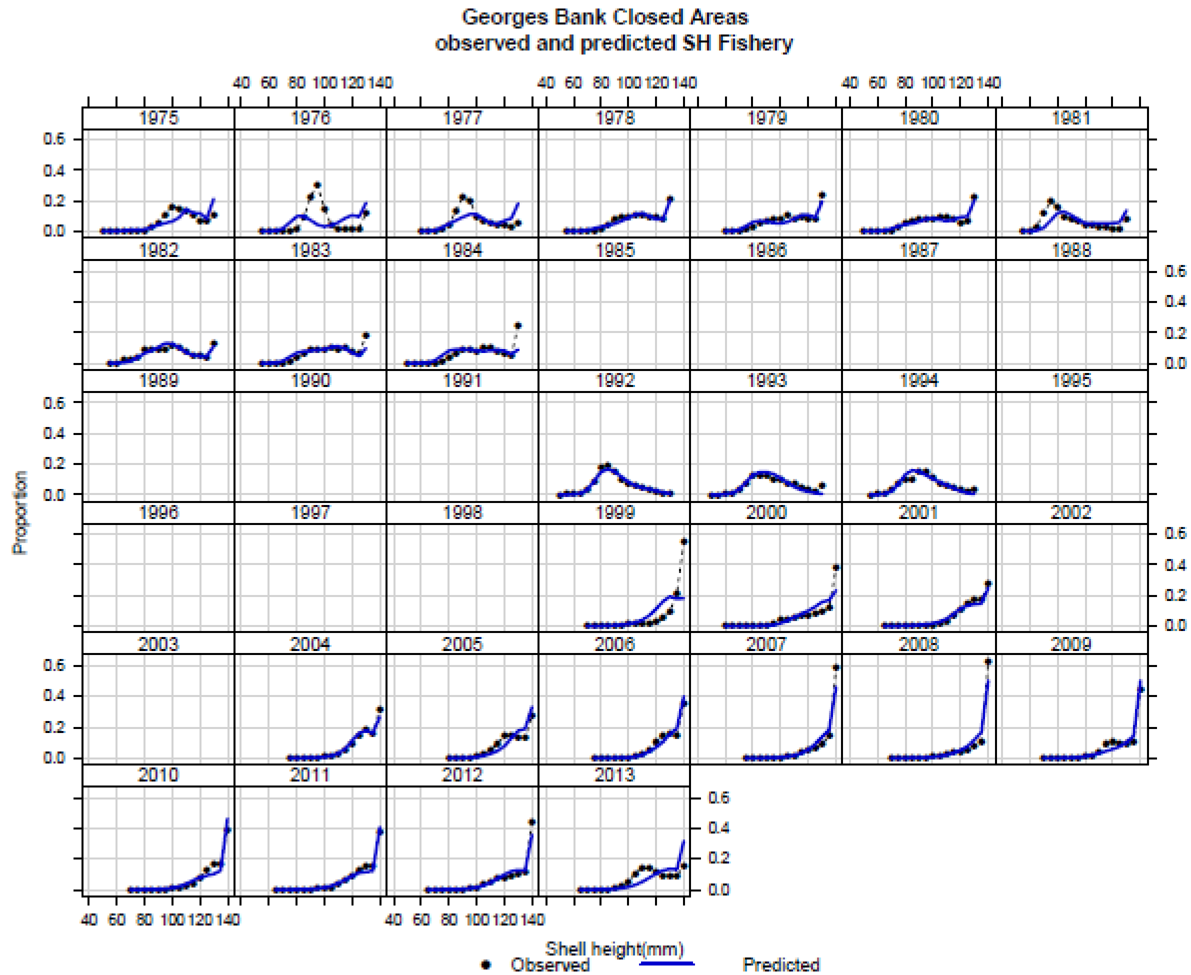


Figure B6.17. Comparison of fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for Georges Bank closed areas.

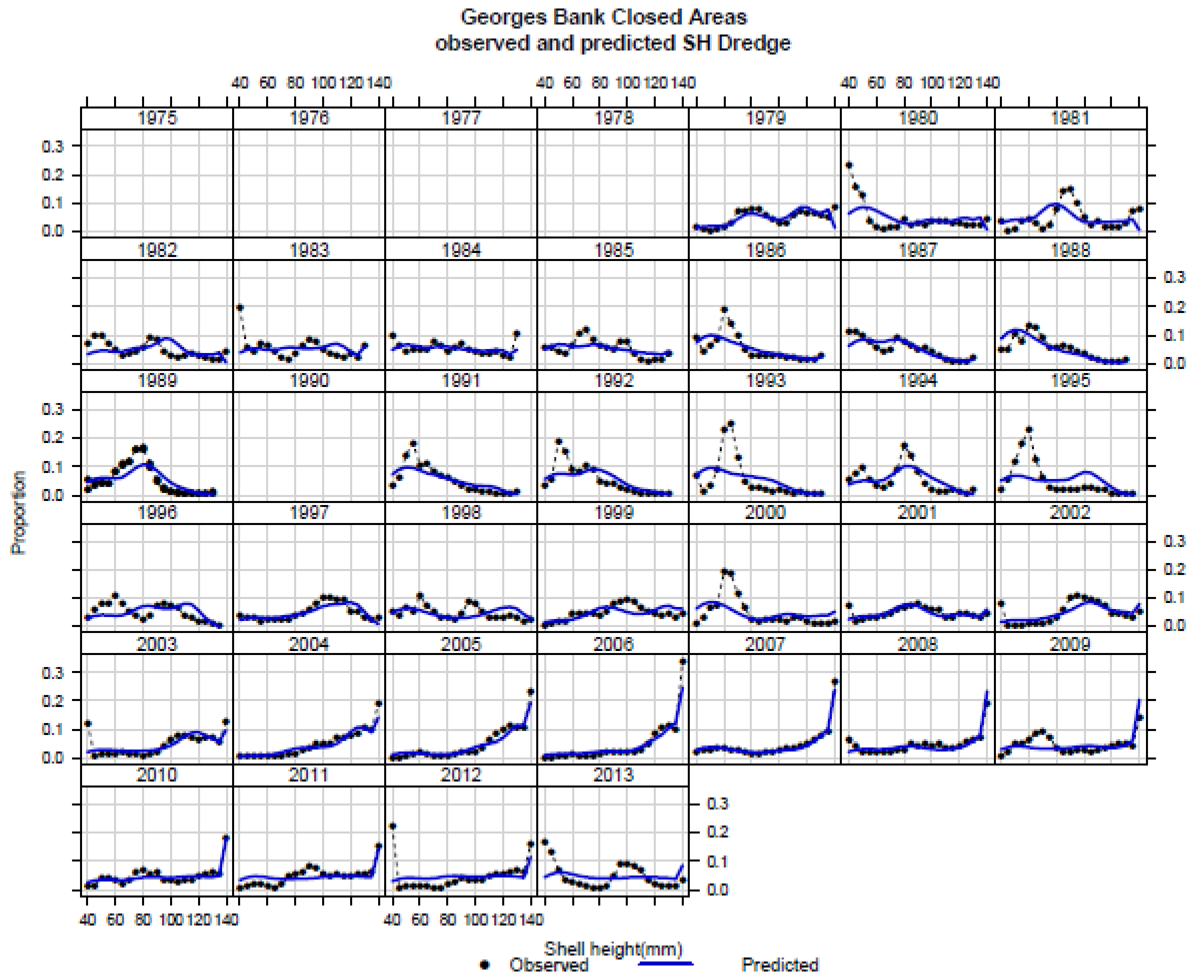


Figure B6.18. NEFSC lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for Georges Bank closed areas.

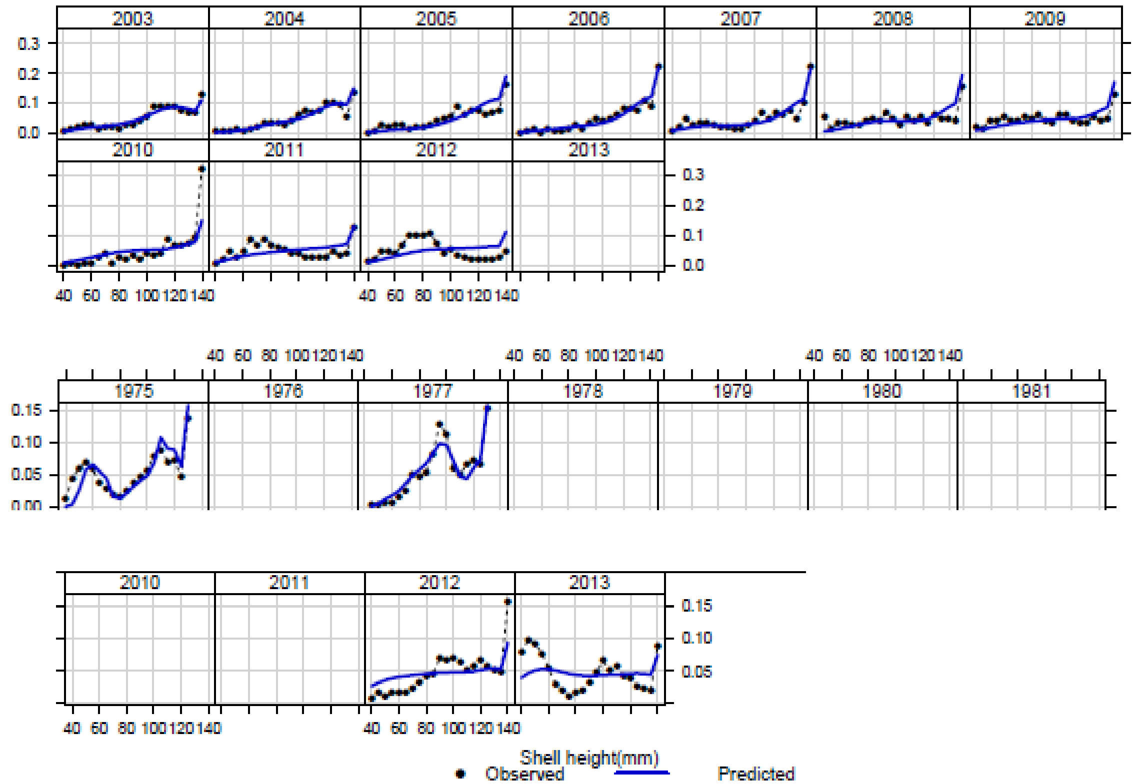


Figure B6.19. Shell height proportions for the SMAST large camera survey (top), the NEFSC unlined dredge survey (middle) and the HabCam survey (bottom) with model predicted proportions (lines) for Georges Bank closed areas.

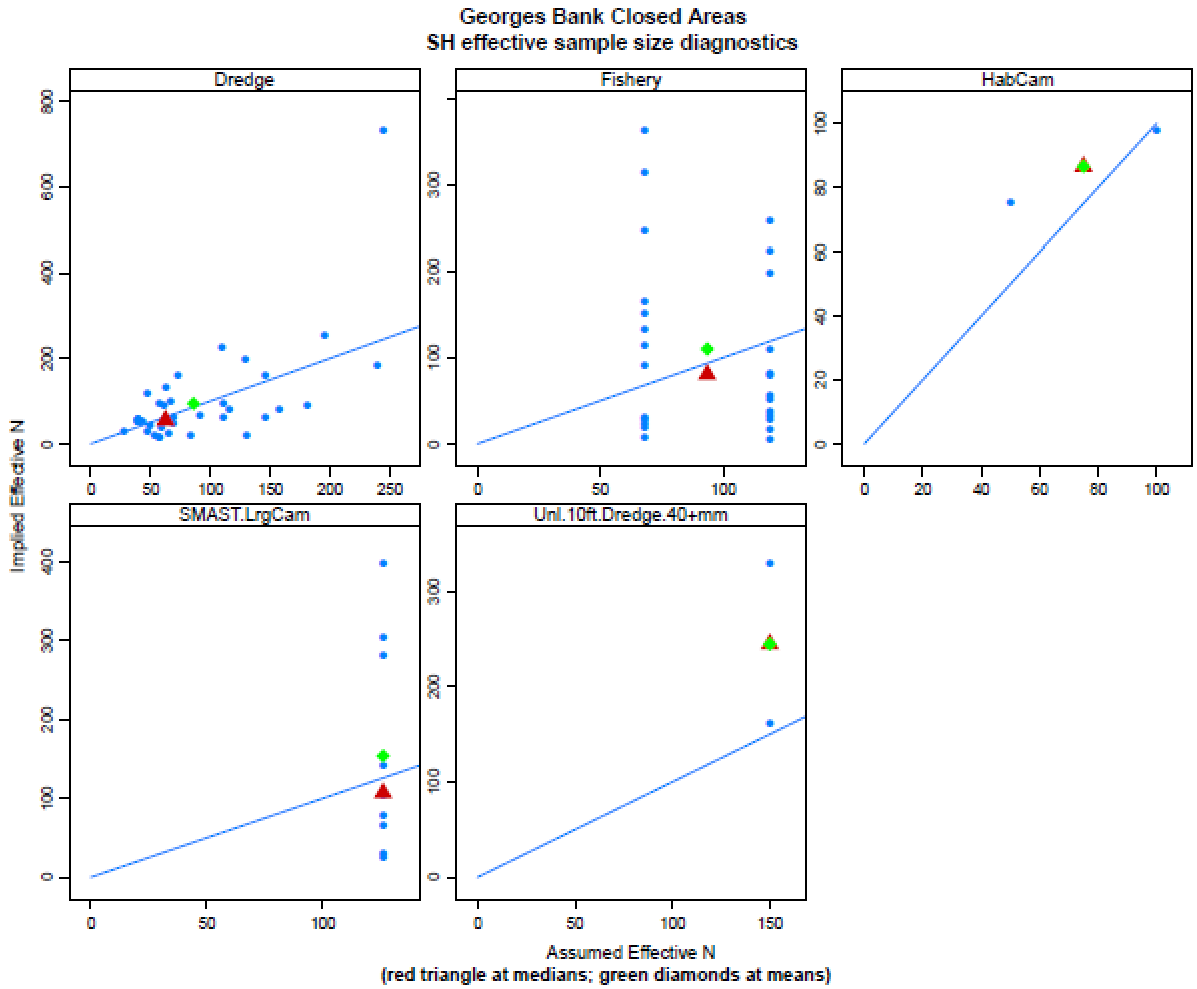


Figure B6.20. Assumed and model implied effective sample sizes for the four surveys (NEFSC unlined dredge, HabCam, SMAST large camera, NEFSC unlined dredge) and the fishery shell height compositions for Georges Bank closed areas. The triangle is the median and the diamond is the mean.

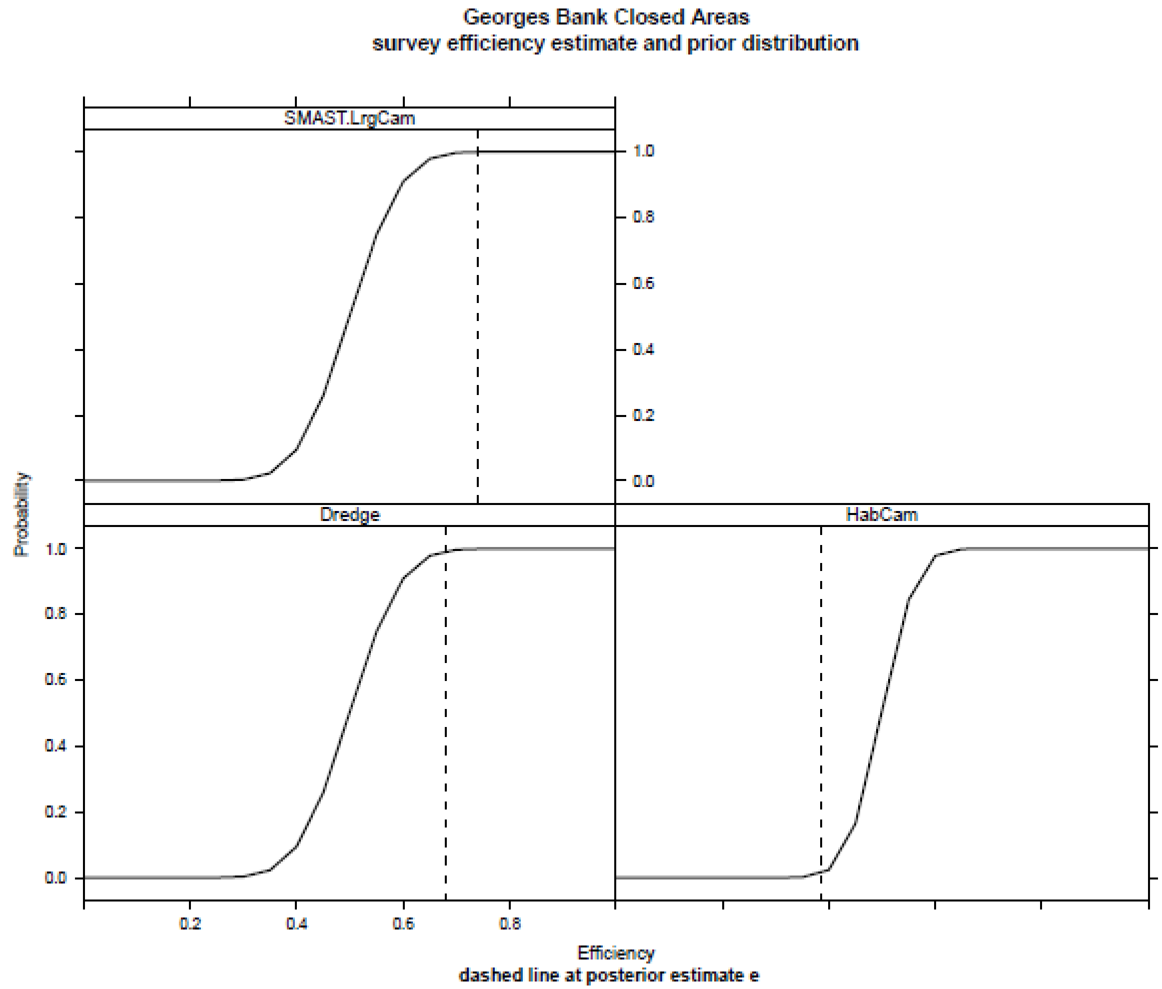


Figure B6.21. Prior cumulative distributions for catchability of the large camera video survey (top) lined dredge survey (bottom left) and HabCam survey (bottom right) for Georges Bank closed areas. The dashed lines are the mean posterior estimate for survey catchability. For the purposes of this plot, the surveys were adjusted to have a mean prior catchability of 0.5

(A) (B)

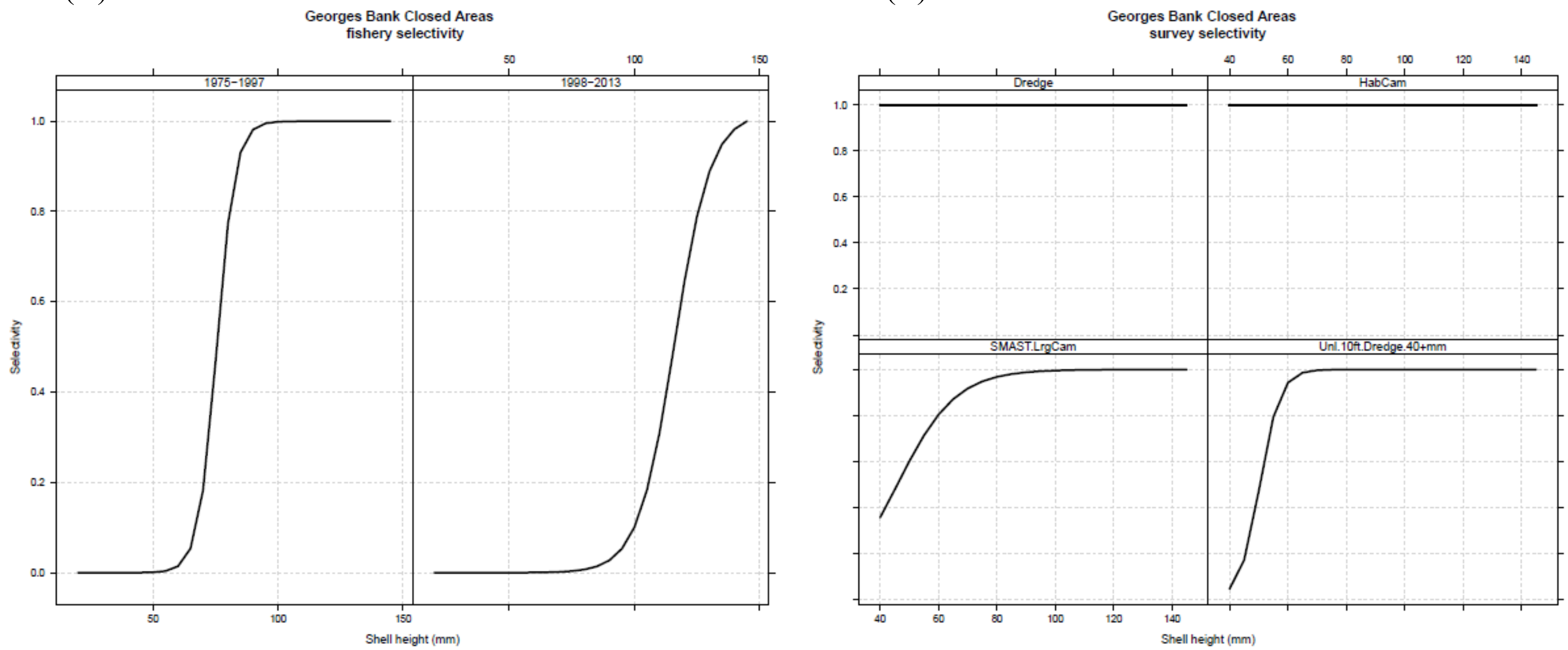


Figure B6.22. (A) Estimated fishery selectivity curves and (B) assumed survey selectivity curves (lined dredge top left, HabCam top right, large camera bottom left, and unlined dredge bottom right) for Georges Bank closed areas.

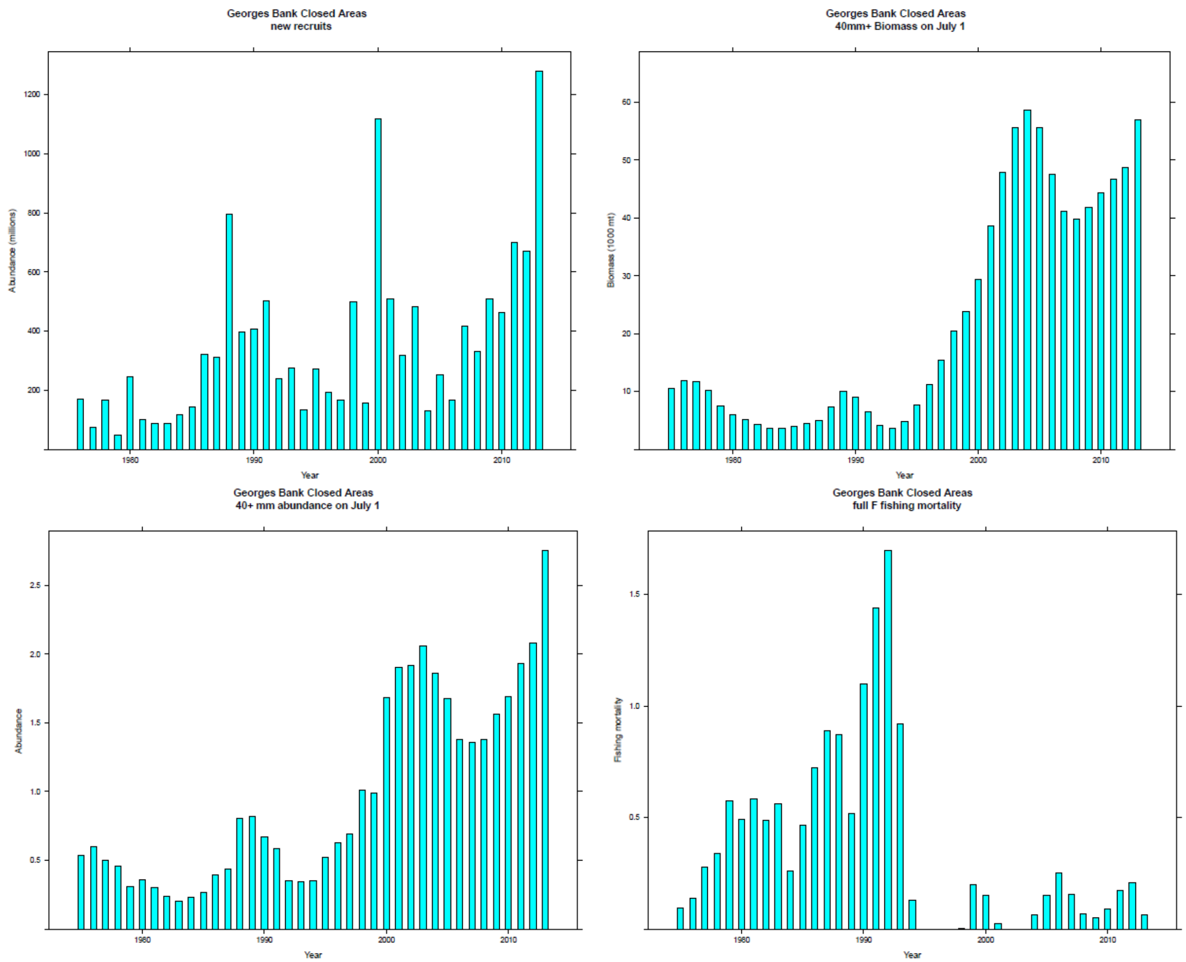


Figure B6.23. CASA model estimated recruitment (top left), July 1 biomass (top right), July 1 abundance (bottom left) and fully recruited fishing mortality (bottom right) for Georges Bank closed areas.

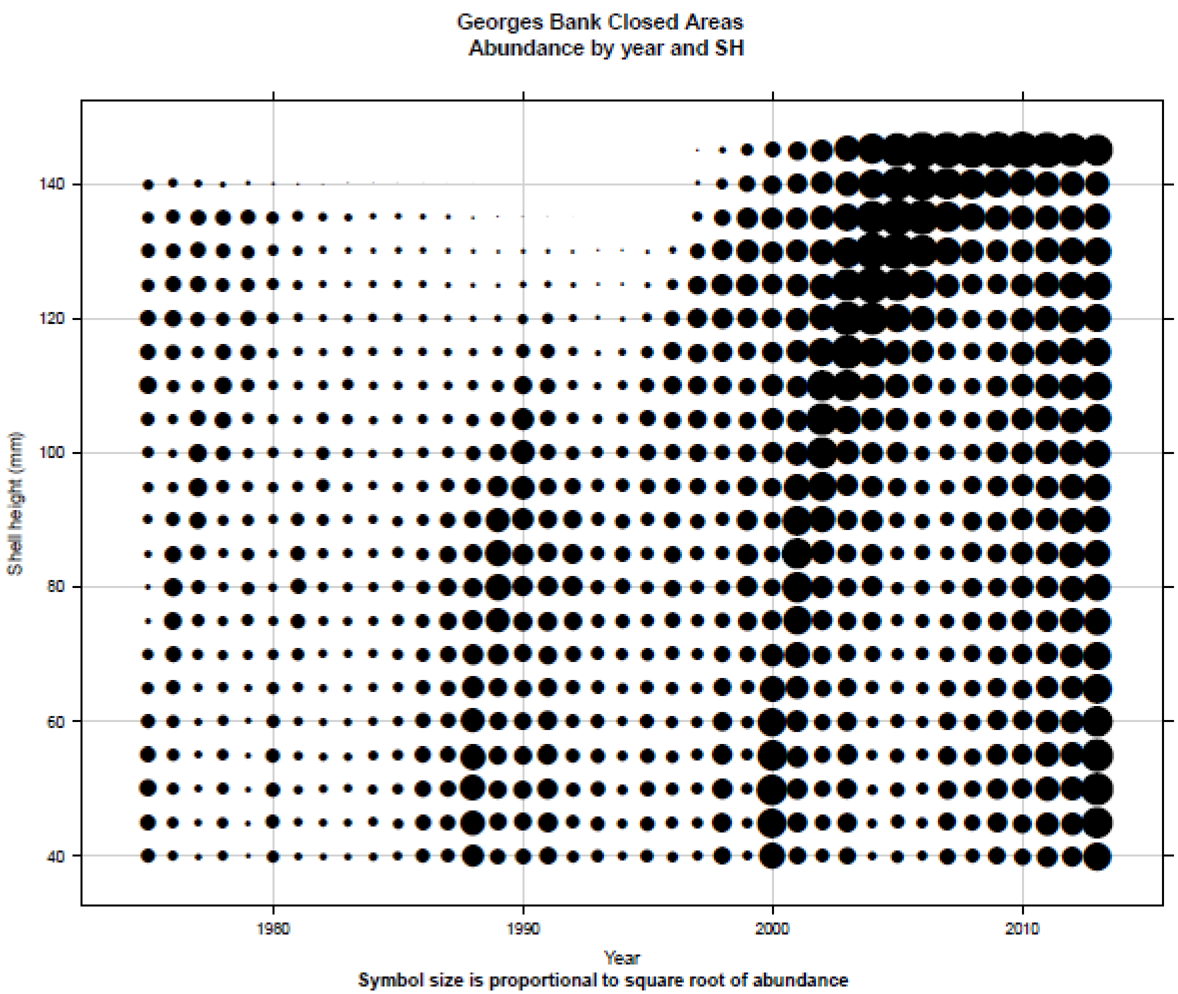


Figure B6.24. Model estimated abundances at shell height for Georges Bank closed areas. Symbol areas are proportional abundance.

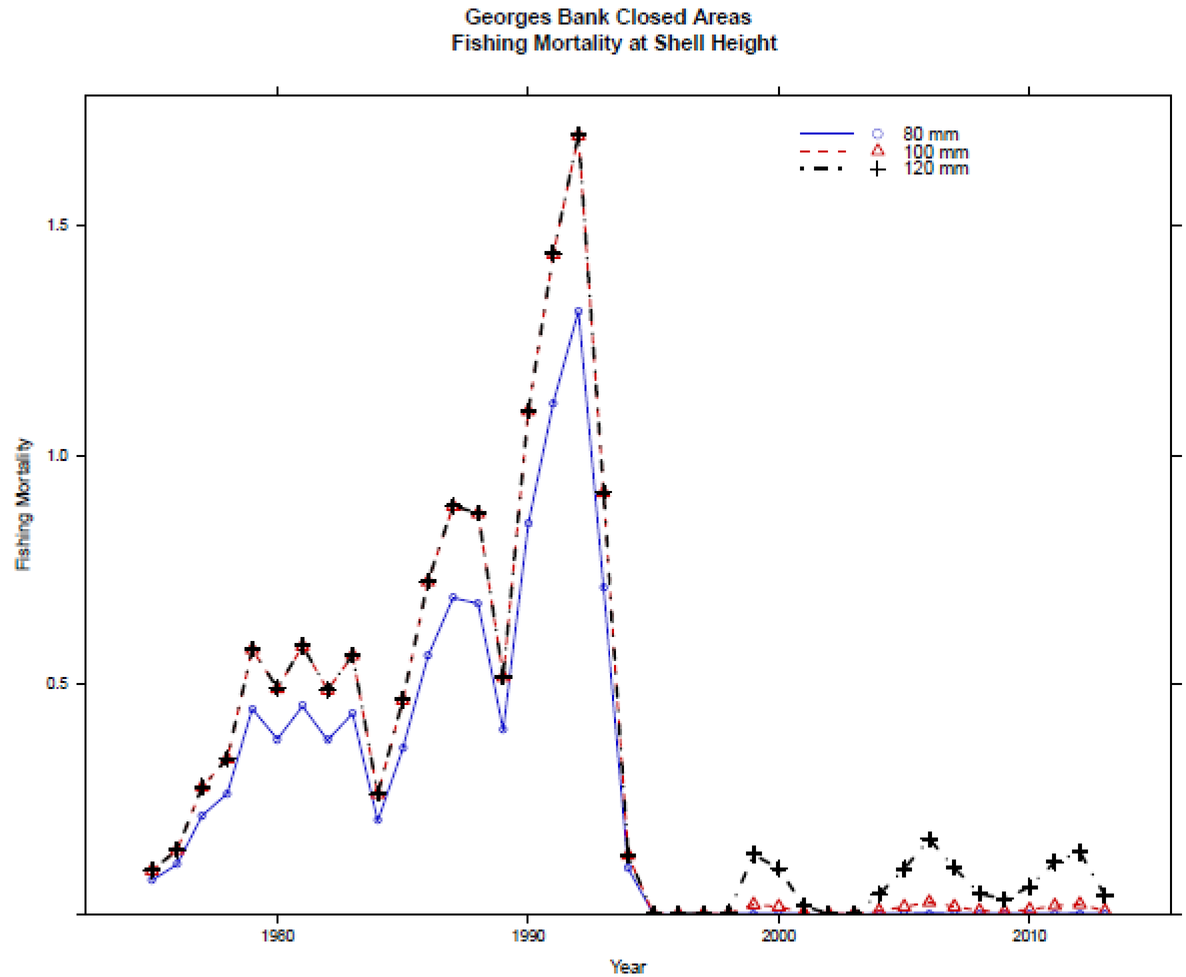


Figure B6.25. CASA model estimated fishing mortality at 80 mm (solid line with circles), 100 mm (dashed line with triangles) and 120 mm SH (dashed line with crosses) for Georges Bank closed areas.

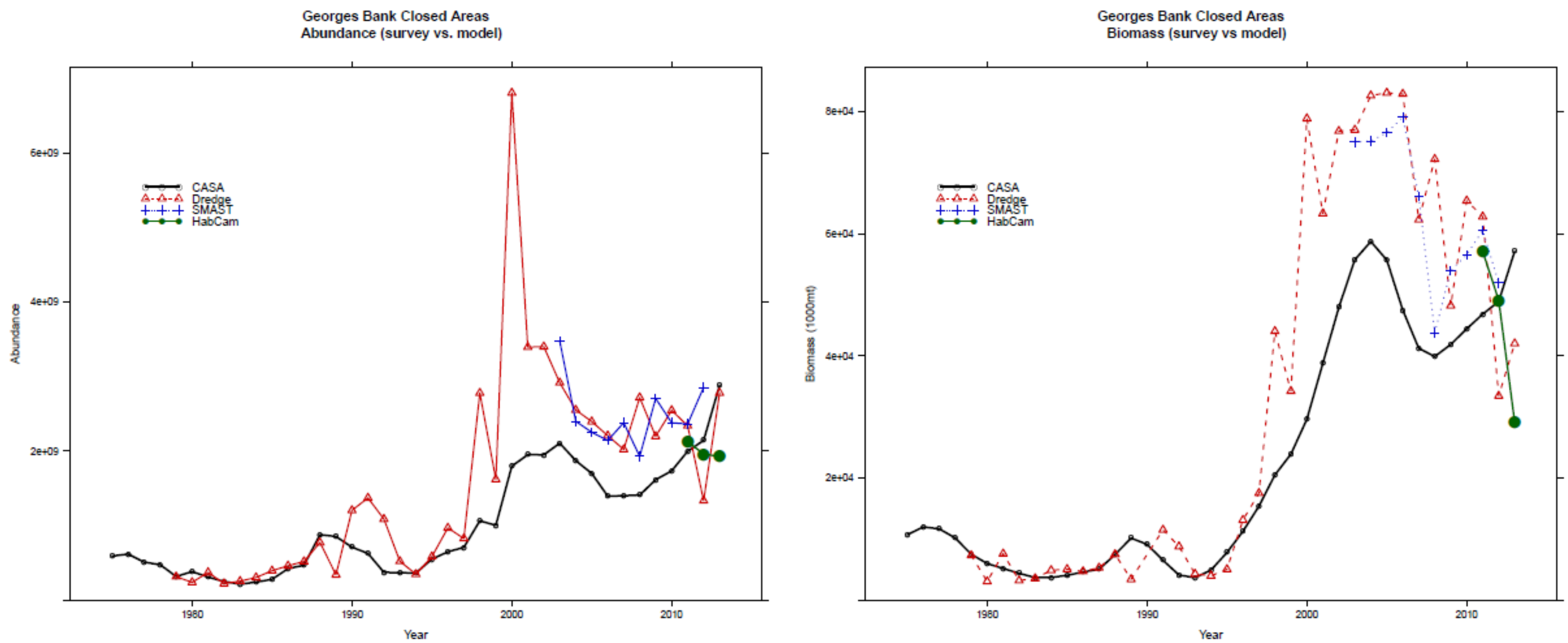


Figure B6.26. Comparison of CASA model estimated abundance (left) and biomass (right) with estimates from the lined dredge survey (dashed line with triangles), SMAST large camera survey (dotted line with crosses) and HabCam (solid line with circles) for Georges Bank closed areas. The dredge survey was expanded assuming an efficiency of 0.41 on sand and 0.27 on gravel/cobble.

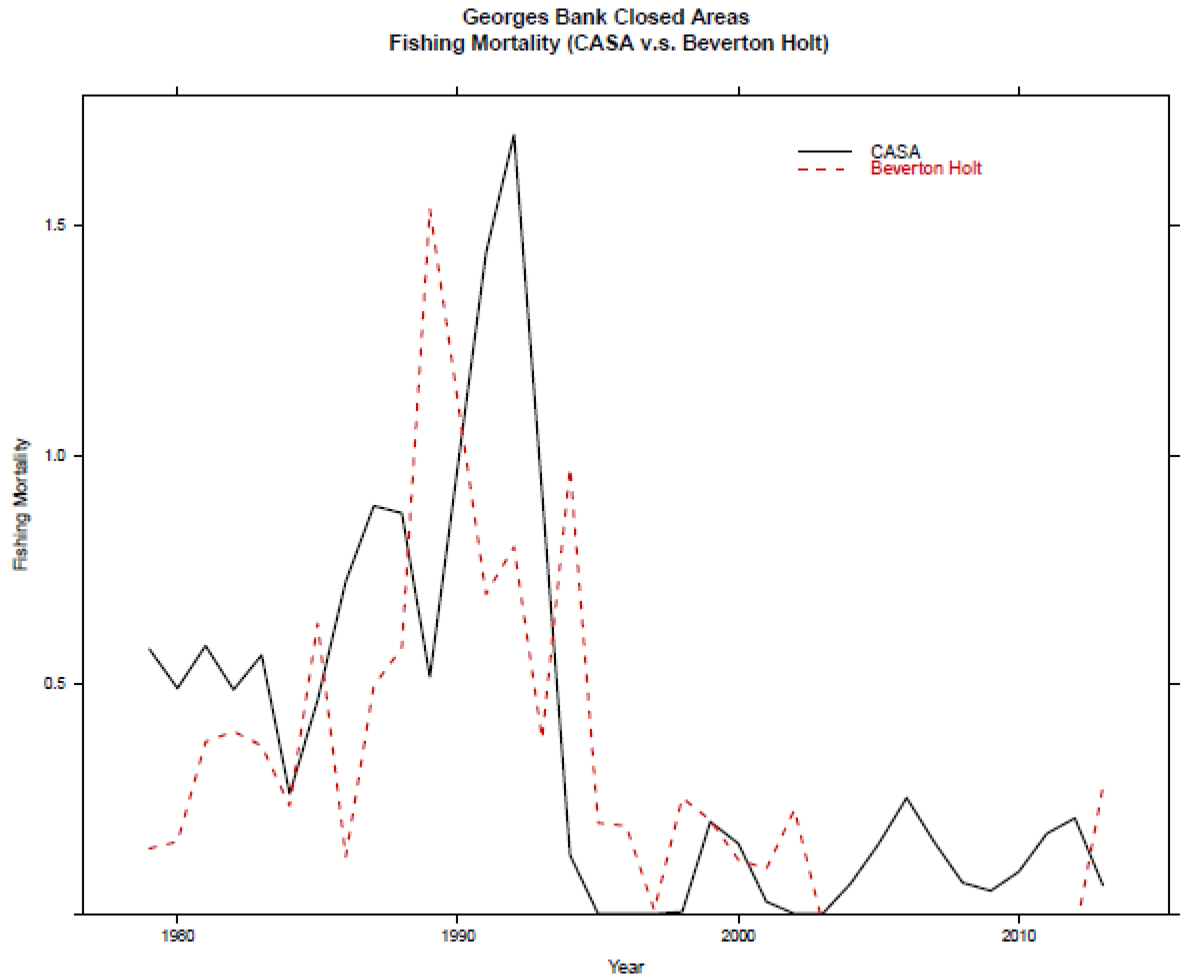


Figure B6.27. Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium estimator for the Georges Bank closed areas.

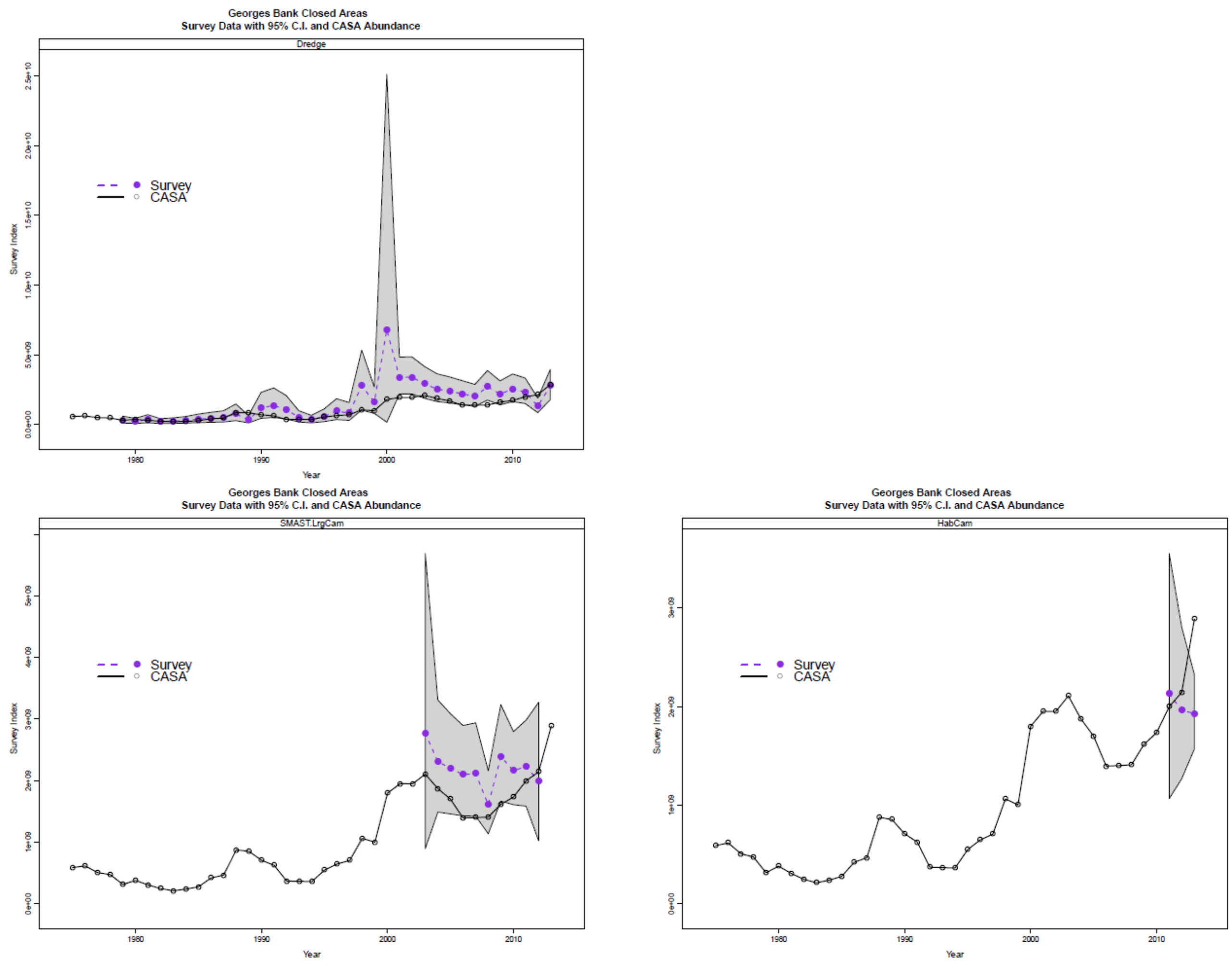


Figure B6.28. CASA estimated abundance compared to that from the dredge survey (top), the SMAST survey (left bottom), and the HabCam survey (right bottom), for Georges Bank closed areas.

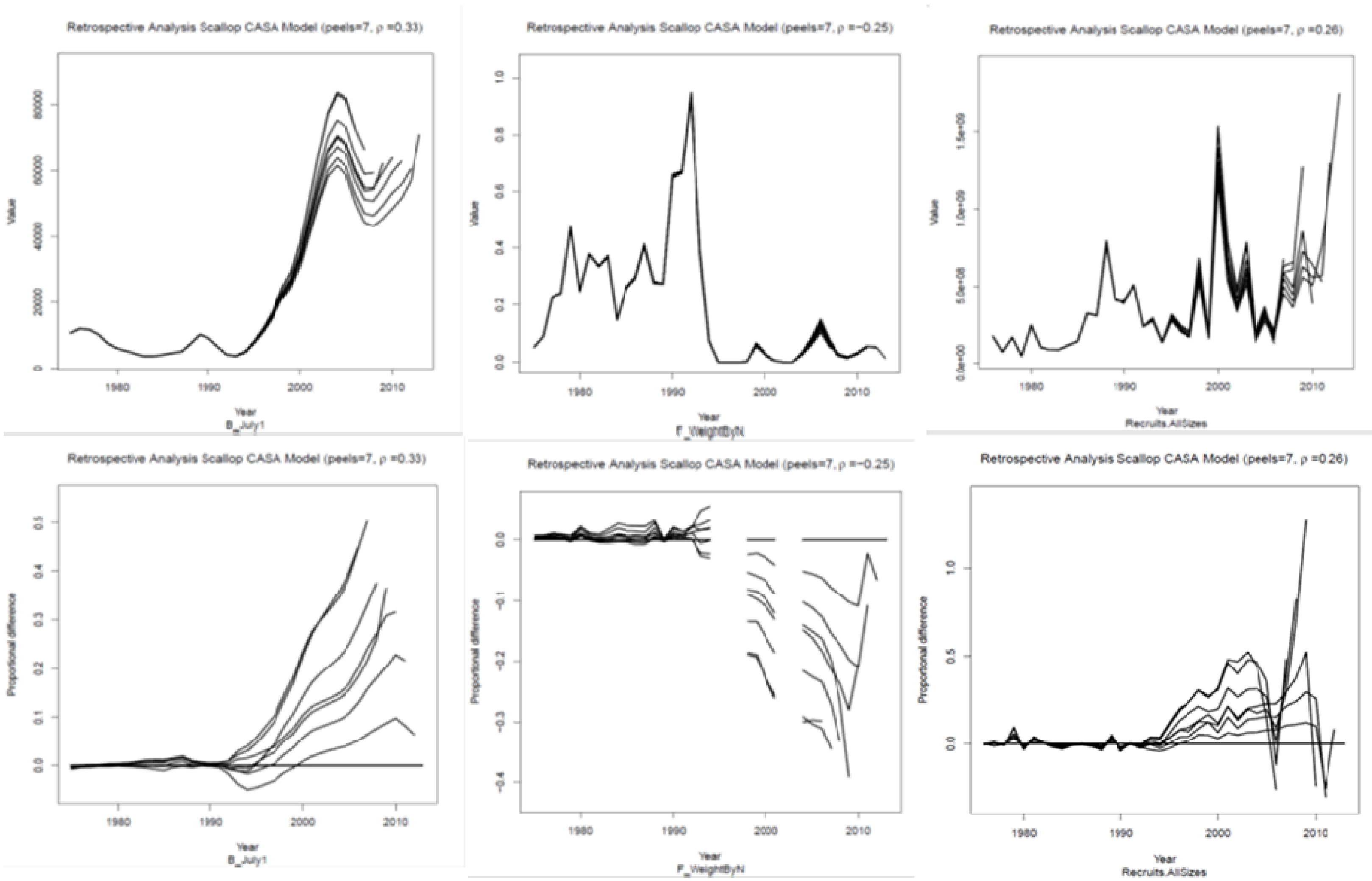


Figure B6.29. Retrospective plots for biomass, fishing mortality and recruitment for Georges Bank closed areas. Retrospectives are shown on both absolute and relative scales.

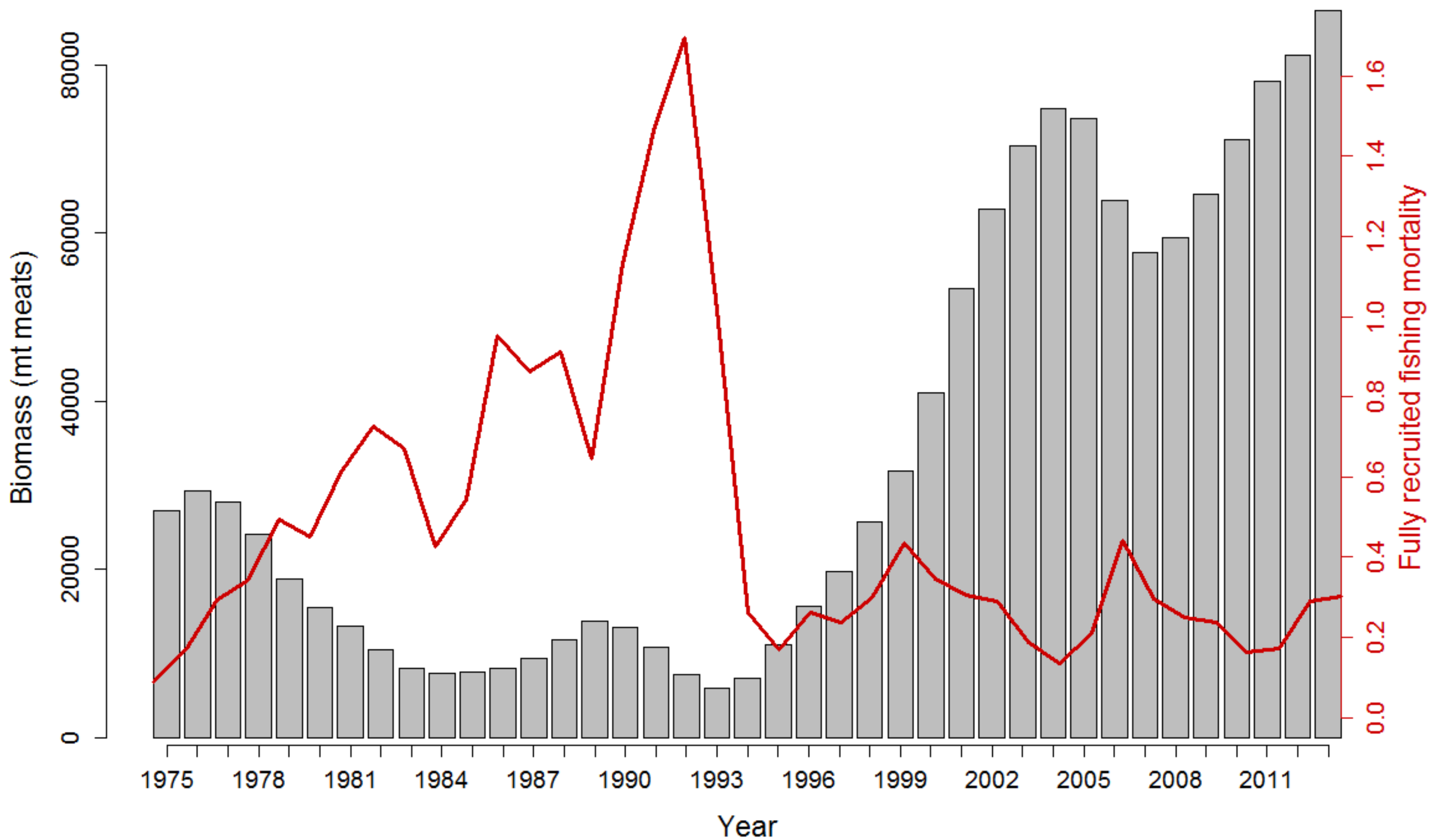


Figure B6.30. Estimated biomass and fully recruited fishing mortality for Georges Bank sea scallops (open and closed combined).

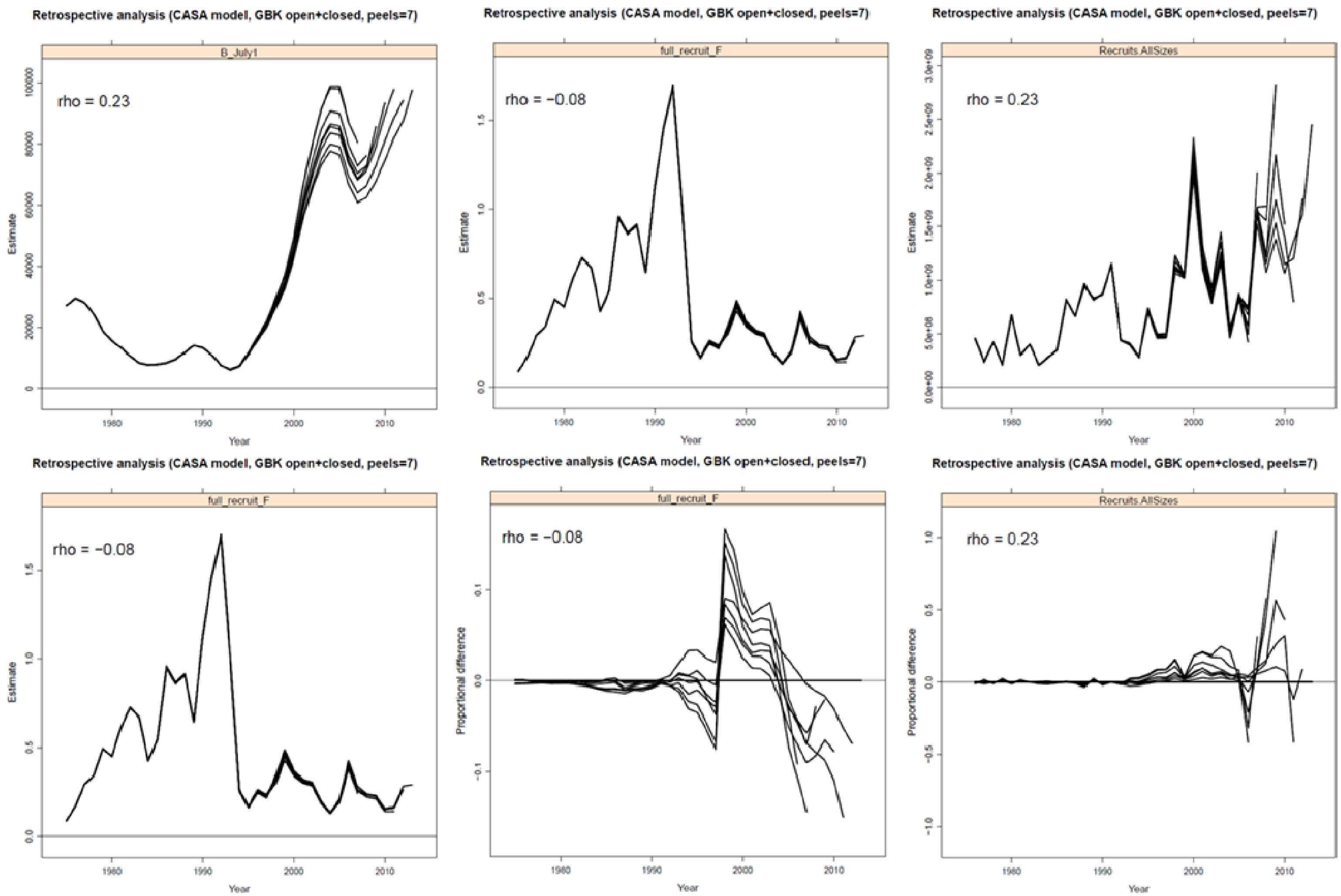
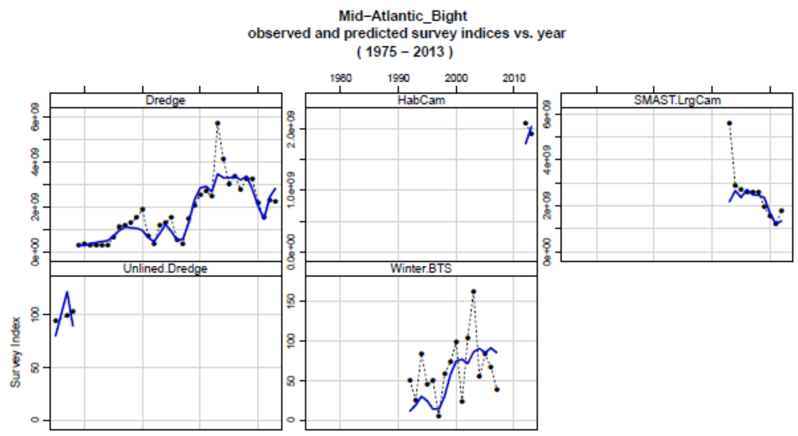


Figure B6.30b. Retrospective plots for the combined Georges Bank open and closed areas.

(A)



(B)

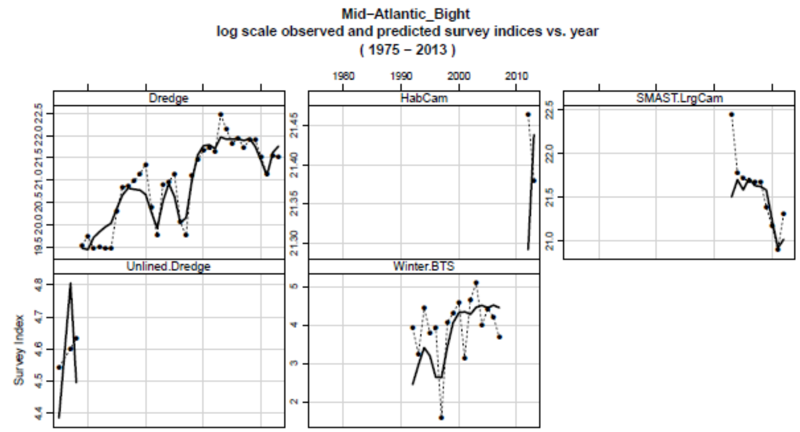


Figure B6.31. Survey trend (solid circles) and corresponding model estimates (lines) for the NEFSC lined dredge survey, the HabCam survey, The SMAST large camera survey, the NEFSC unlined dredge survey, and the NEFSC winter bottom trawl survey in the Mid-Atlantic Bight. Results are shown on a linear scale (A) and a log scale (B).

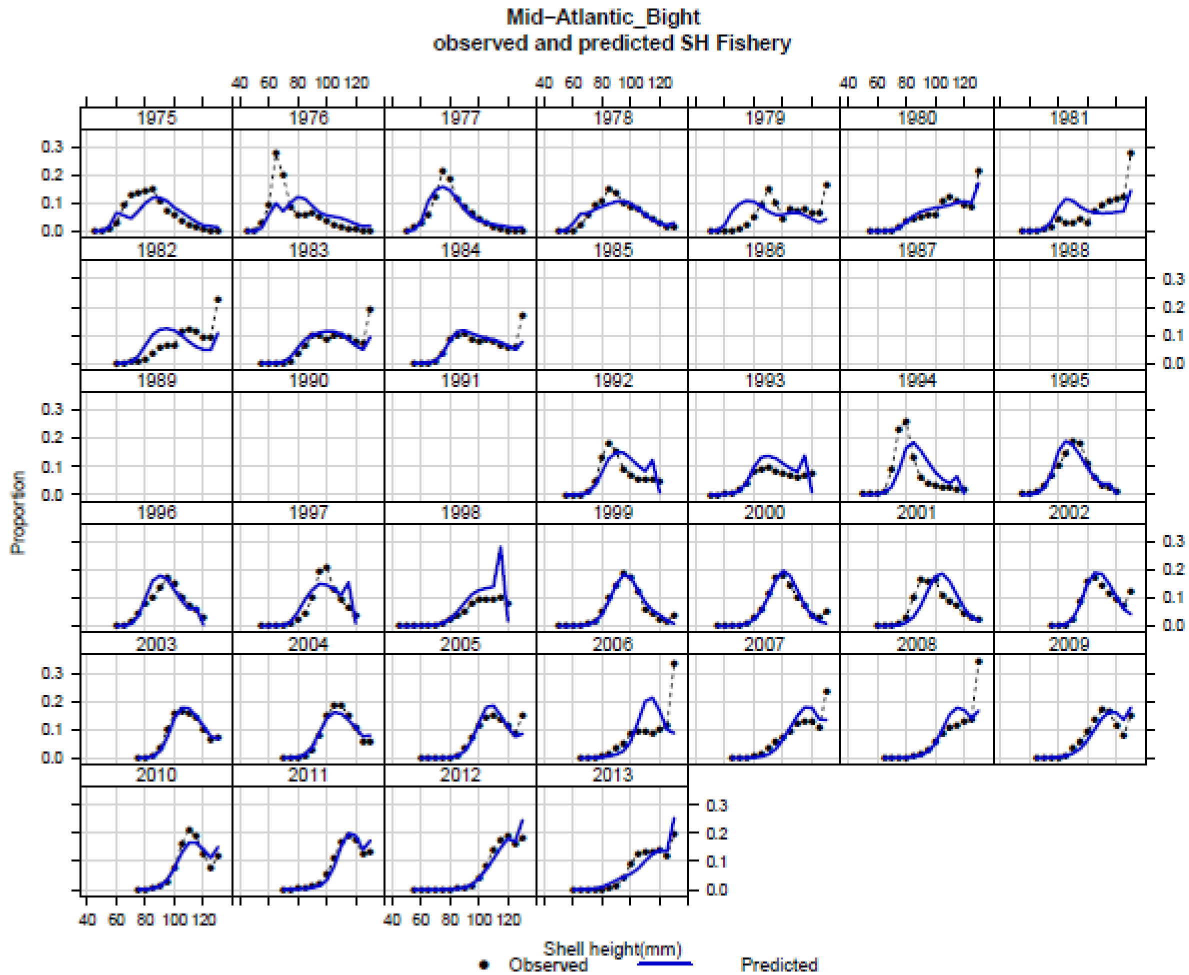


Figure B6.32. Comparison of fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for the Mid-Atlantic Bight.

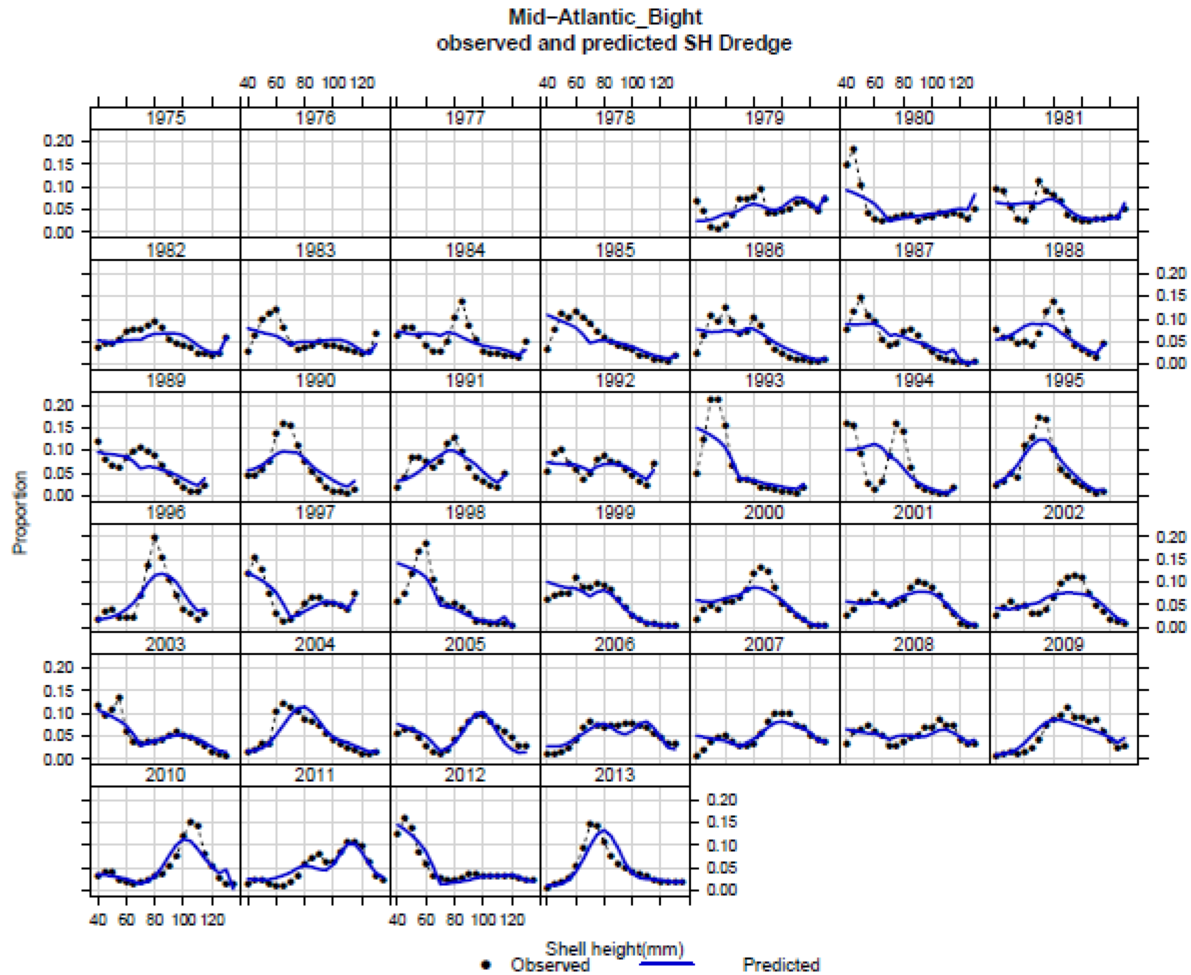


Figure B6.33. NEFSC lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (lines) for the Mid-Atlantic Bight.

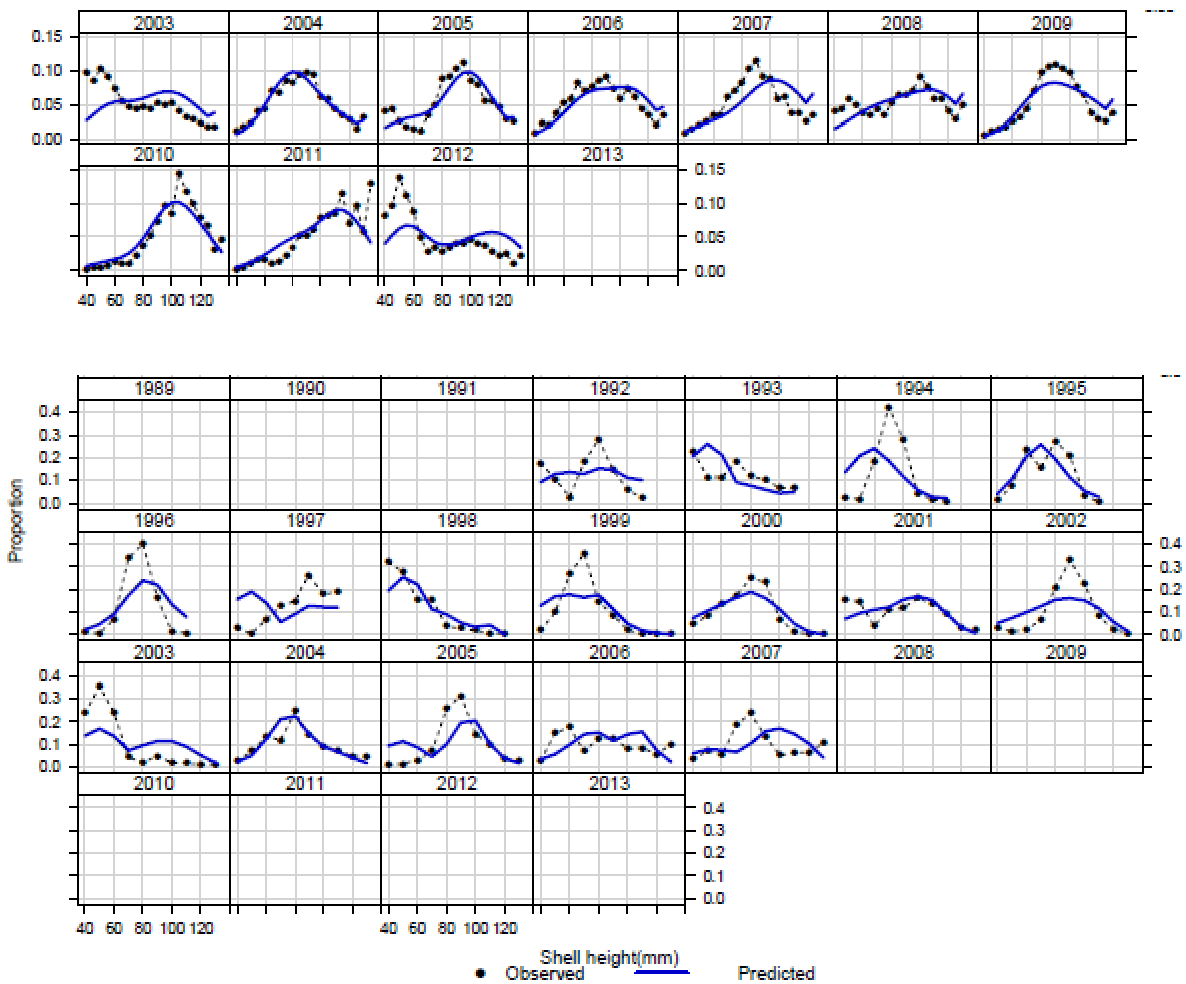


Figure B6.34. Shell height proportions for the SMAST large camera survey (top), and the NEFSC winter bottom trawl survey (bottom) with model predicted proportions (lines) for the Mid-Atlantic Bight.

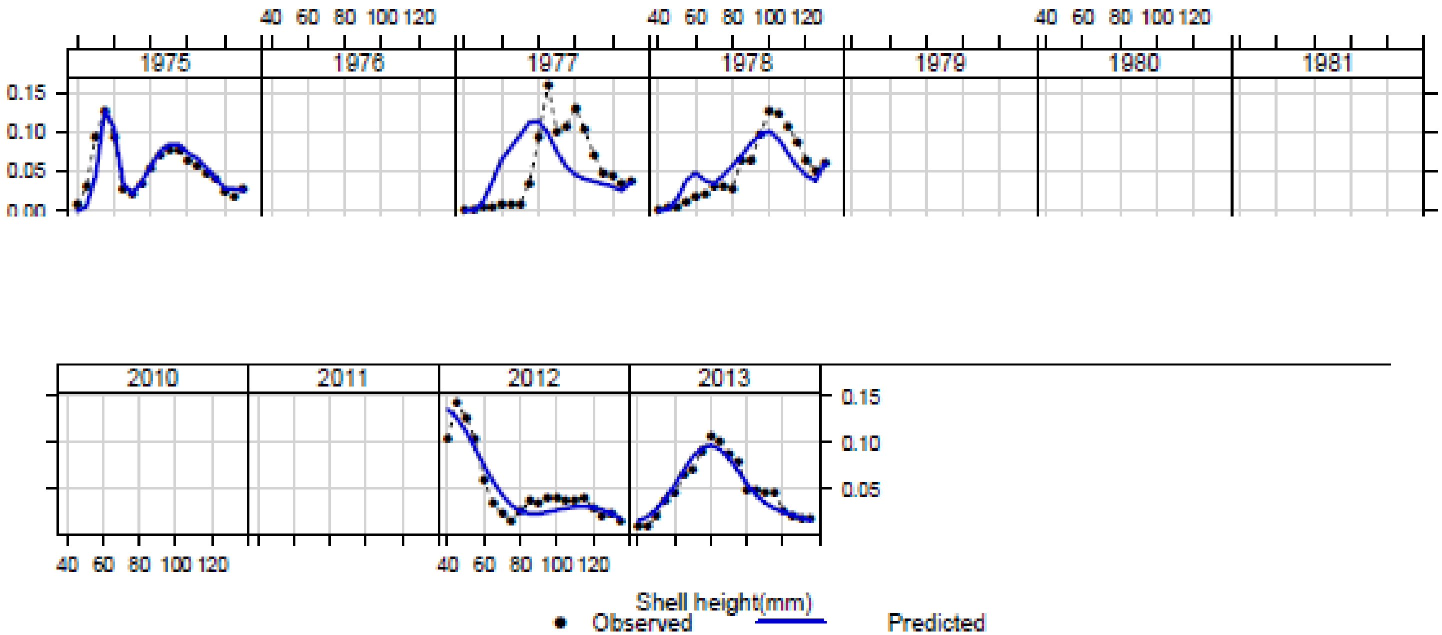


Figure B6.35. Shell height proportions for the NEFSC unlined dredge survey (top) and the HabCam survey (bottom) with model predicted proportions (lines) for the Mid-Atlantic Bight.

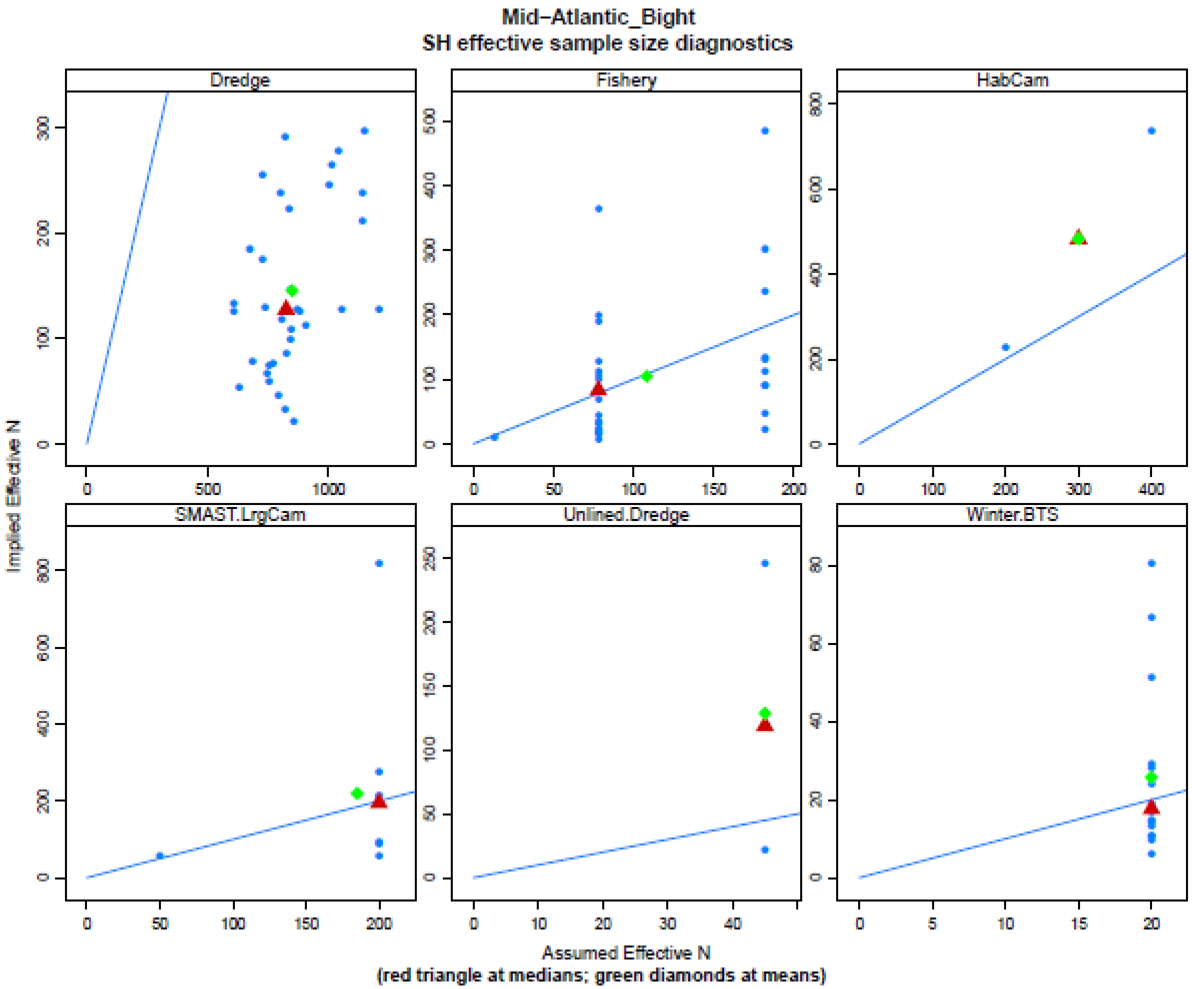


Figure B6.36 Assumed and implied effective sample sizes for the five surveys (NEFSC unlined dredge, HabCam, SMAST large camera, NEFSC unlined dredge, winter bottom trawl survey) and the fishery shell height compositions for the Mid-Atlantic Bight. The triangle is the median and the diamond is the mean.

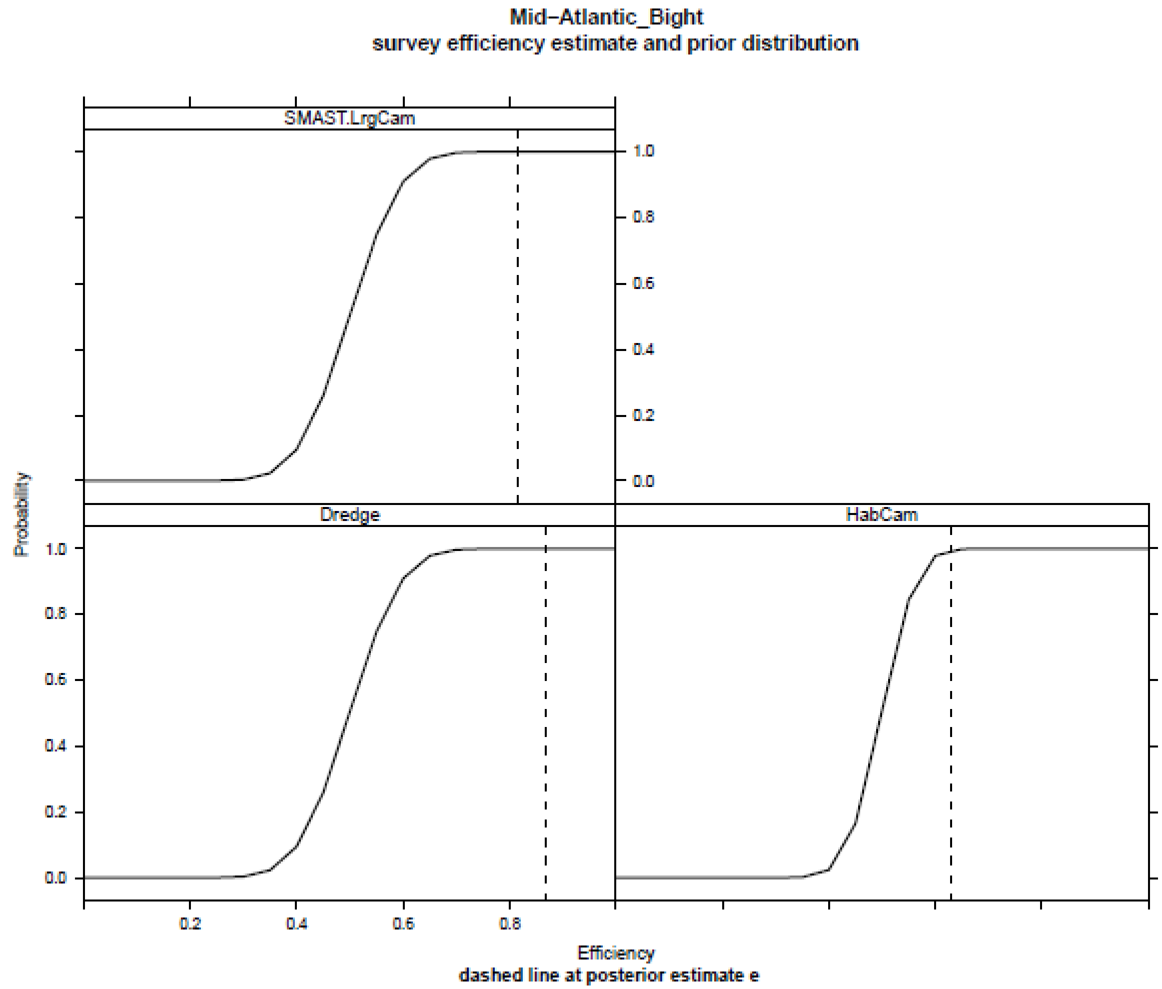
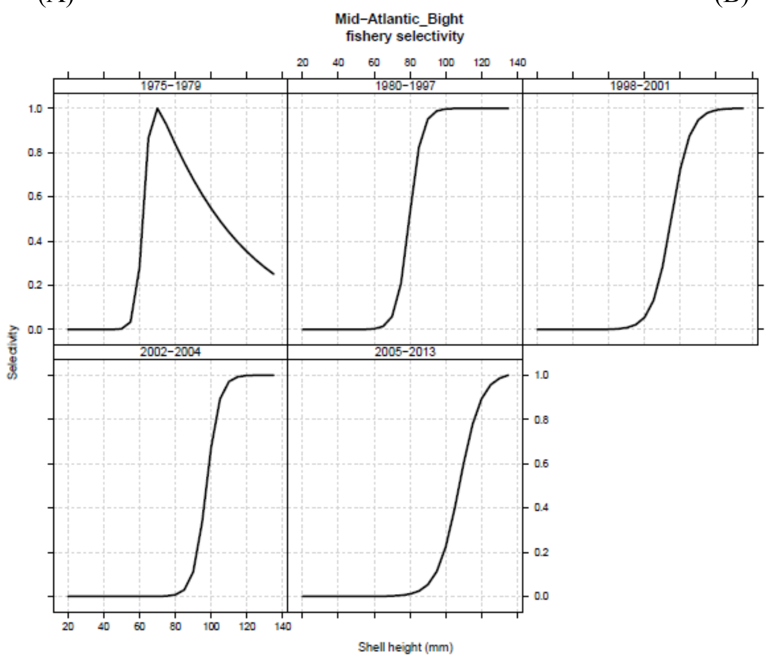


Figure B6.37. Prior cumulative distributions for catchability of the large camera video survey (top) lined dredge survey (bottom left) and HabCam survey (bottom right) for the Mid-Atlantic Bight. The dashed lines are the mean posterior estimate for survey efficiency. For the purposes of this plot, the surveys were adjusted to have a mean prior catchability of 0.5

(A)



(B)

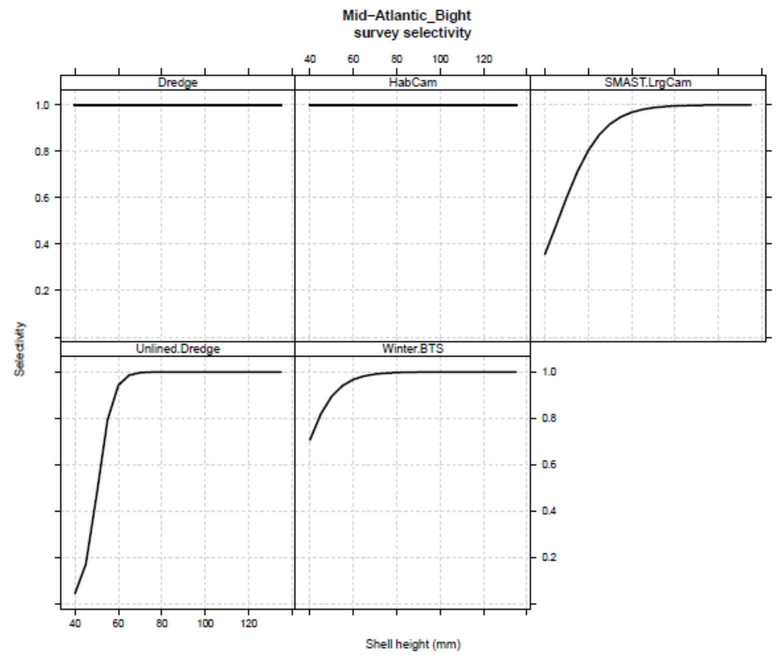


Figure B6.38. (A) Estimated fishery selectivity curves and (B) survey selectivity curves (lined dredge top left, HabCam top middle, large camera top right, unlined dredge bottom left, and winter bottom trawl bottom middle) for the Mid-Atlantic Bight.

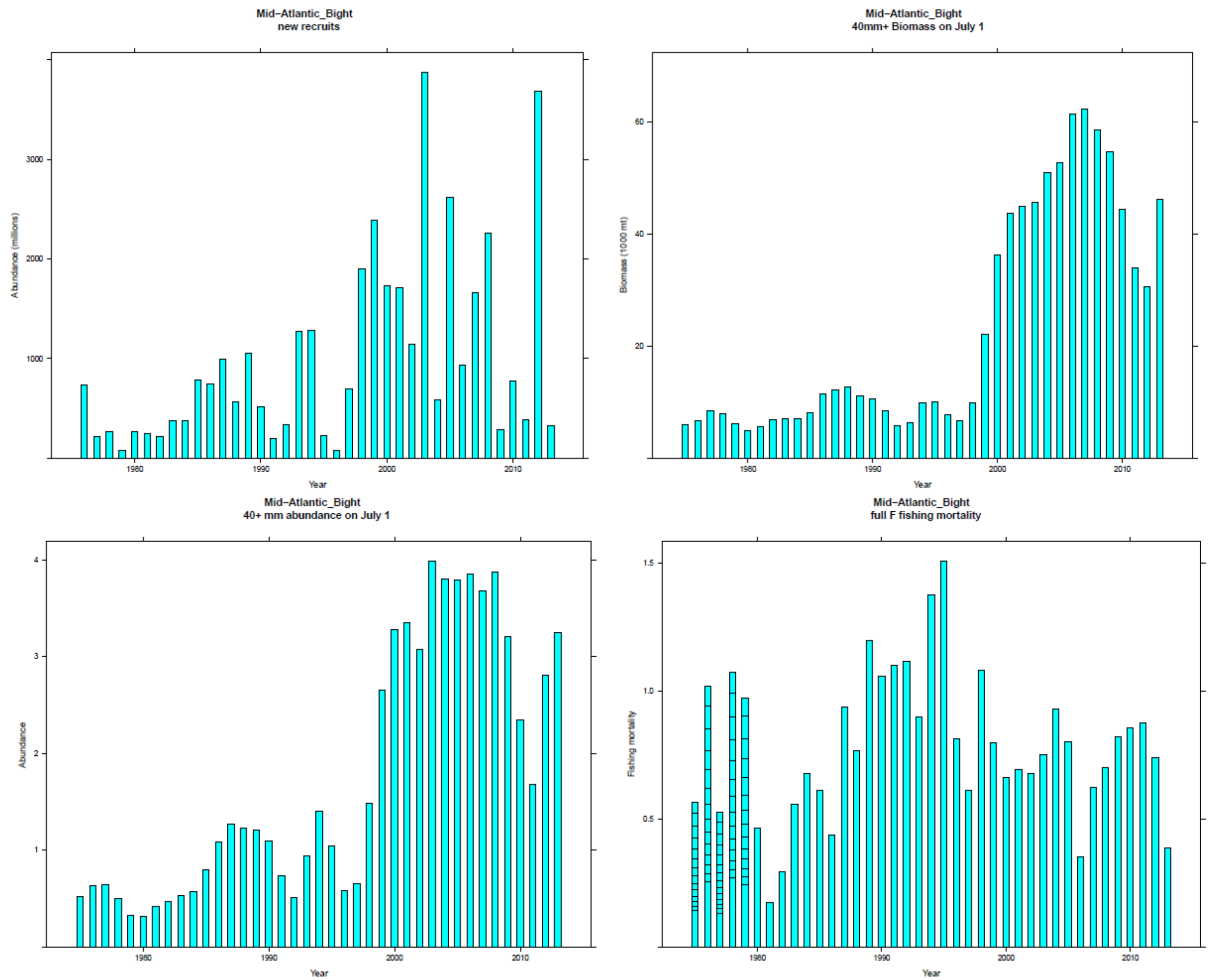


Figure B6.39. CASA model estimated recruitment (top left), July 1 biomass (top right), July 1 abundance (bottom left), and fully recruited fishing mortality (bottom right) for the Mid-Atlantic Bight.

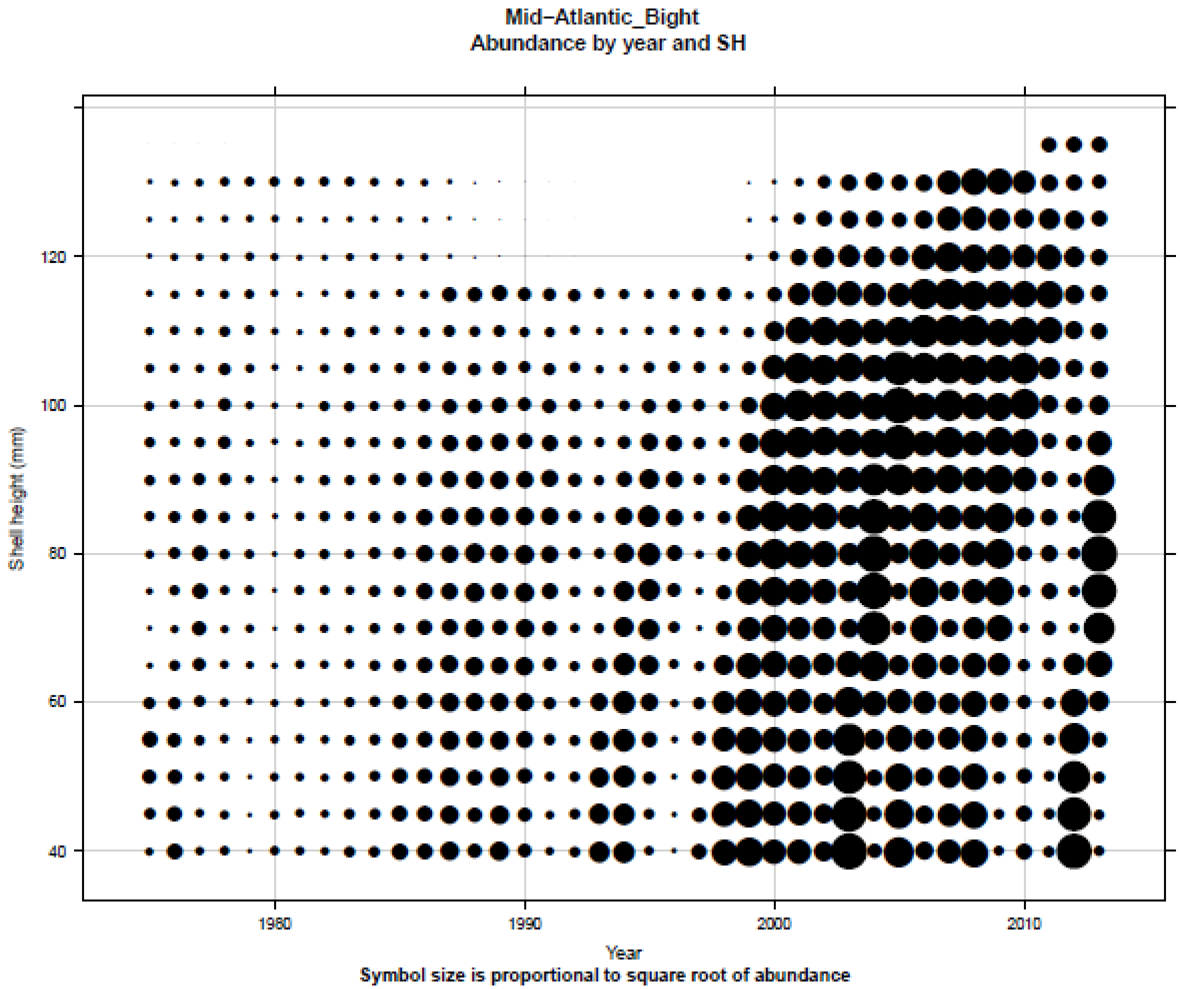


Figure B6.40. Model estimated abundances at shell height for the Mid-Atlantic Bight. Symbol areas are proportional to abundance.

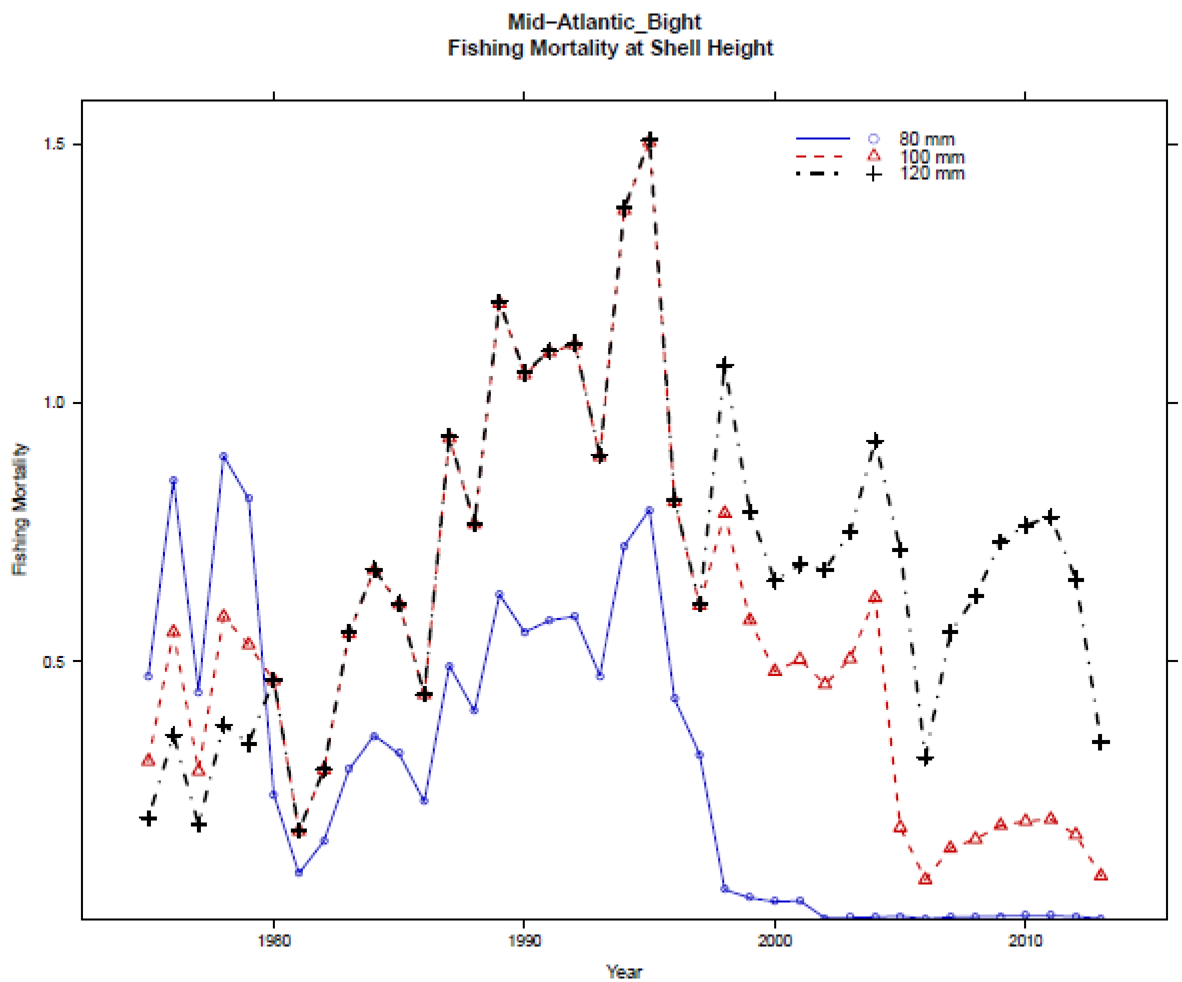


Figure B6.41. CASA model estimated fishing mortality at 80 mm (solid line with circles), 100 mm (dashed line with triangles) and 120 mm SH (dashed line with crosses) for the Mid-Atlantic Bight.

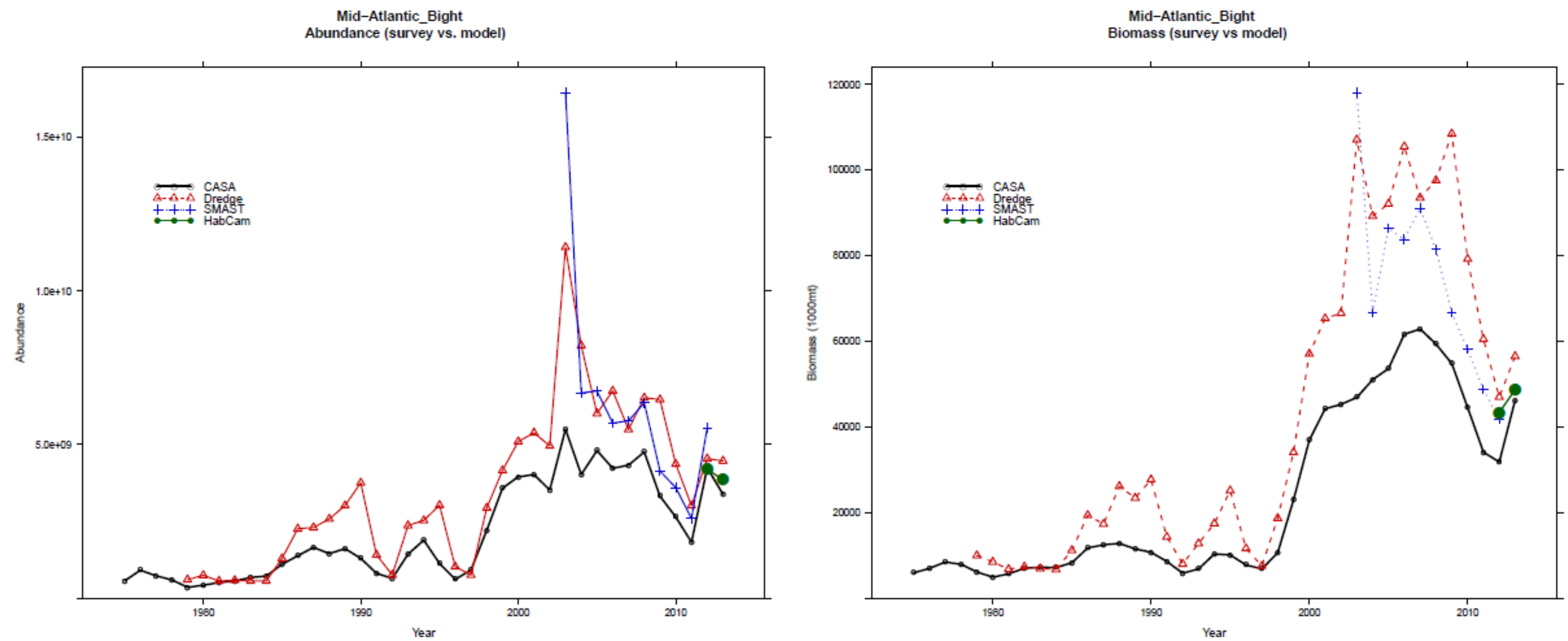


Figure B6.42. Comparison of CASA model estimated abundance (left) and biomass (right) with estimates from the lined dredge survey (dashed line with triangles), SMAST large camera survey (dotted line with crosses) and HabCam (solid line with circles) for the Mid-Atlantic Bight. The dredge survey was expanded assuming an efficiency of 0.41.

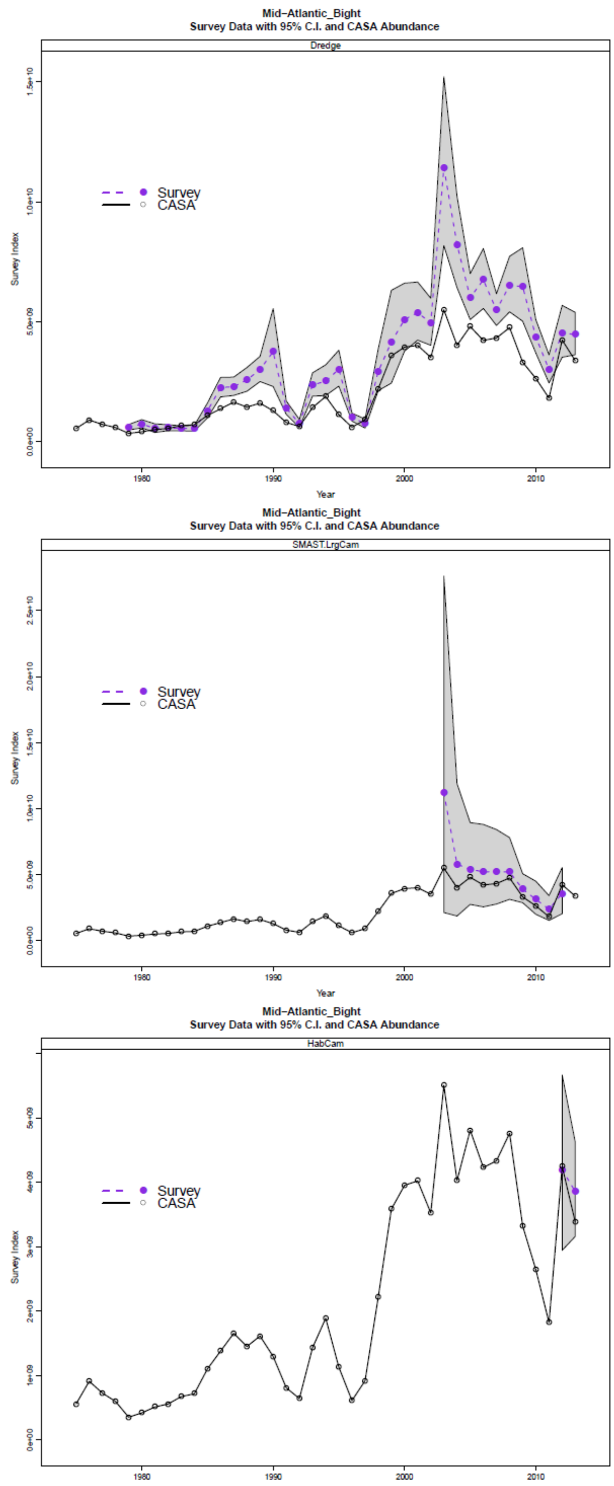


Figure B6.42b. CASA estimated abundance compared to that from the dredge survey (top), the SMAST survey (middle), and the HabCam survey (bottom), for the mid-Atlantic bight.

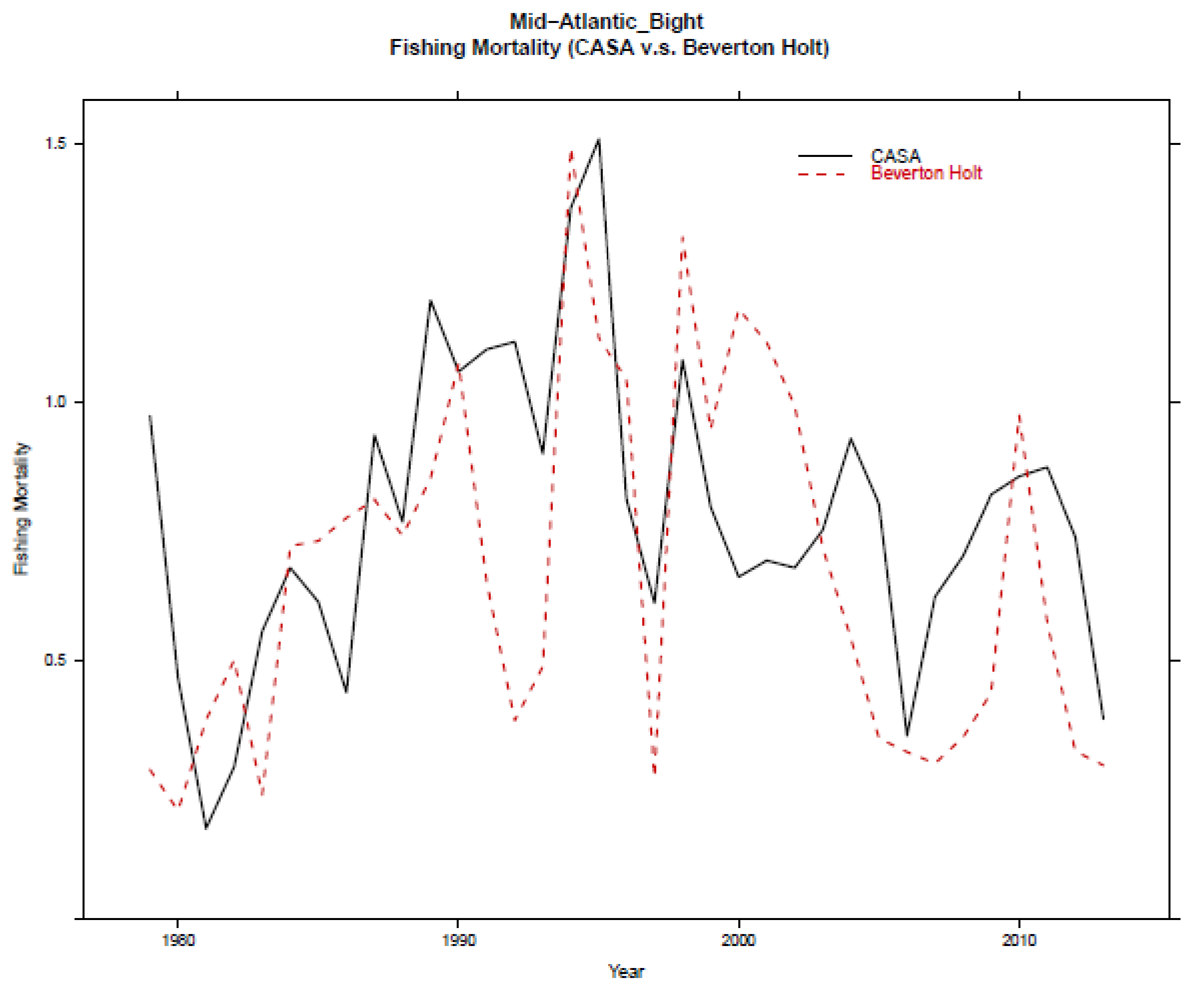


Figure B6.43. Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium estimator for the Mid-Atlantic Bight.

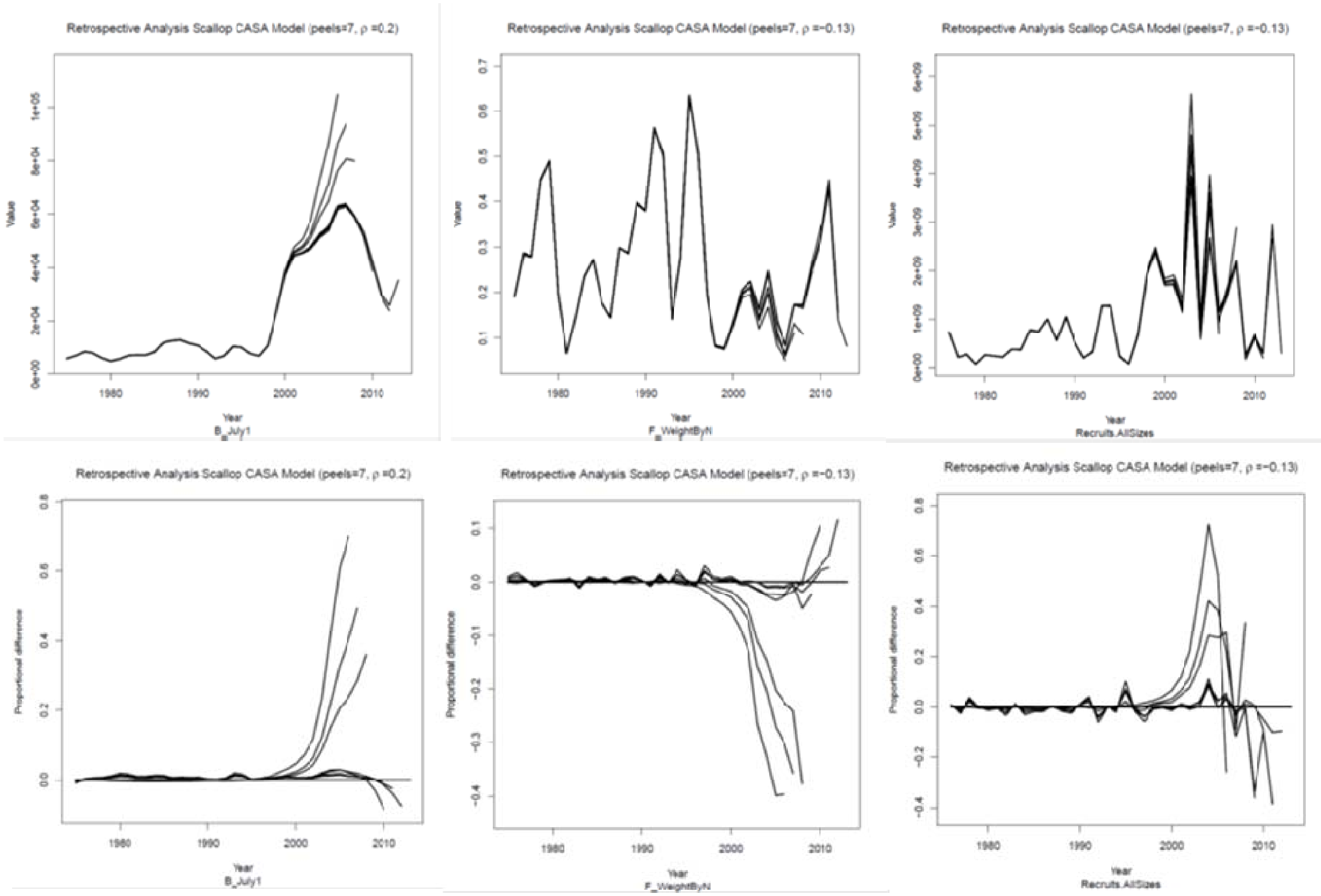


Figure B6.44. Retrospective plots for biomass, fishing mortality and recruitment for the Mid-Atlantic Bight. Retrospective patterns are shown on both absolute and relative scales.

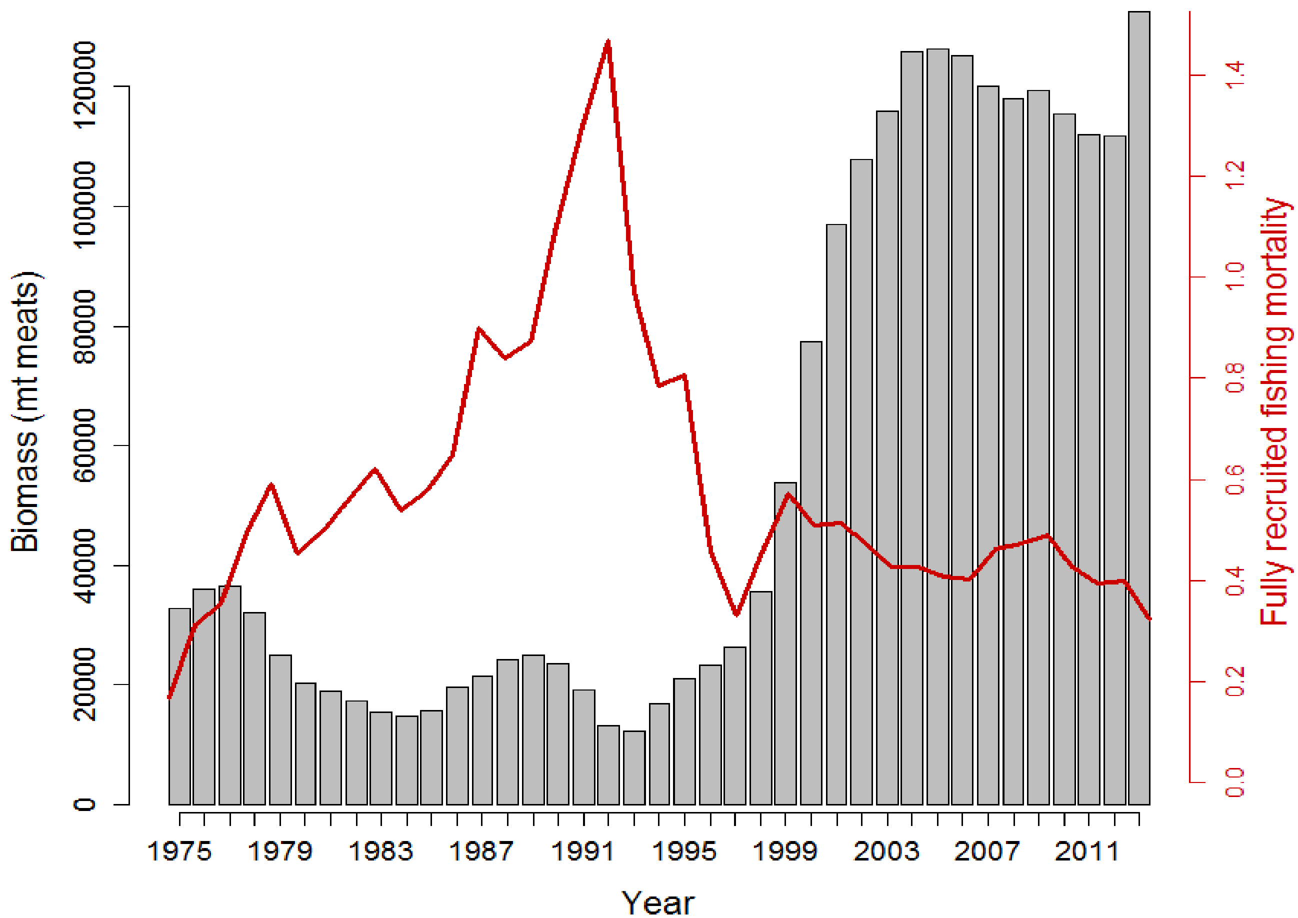


Figure B6.45. Total estimated biomass and fully recruited fishing mortality for Georges Bank and Mid-Atlantic combined.

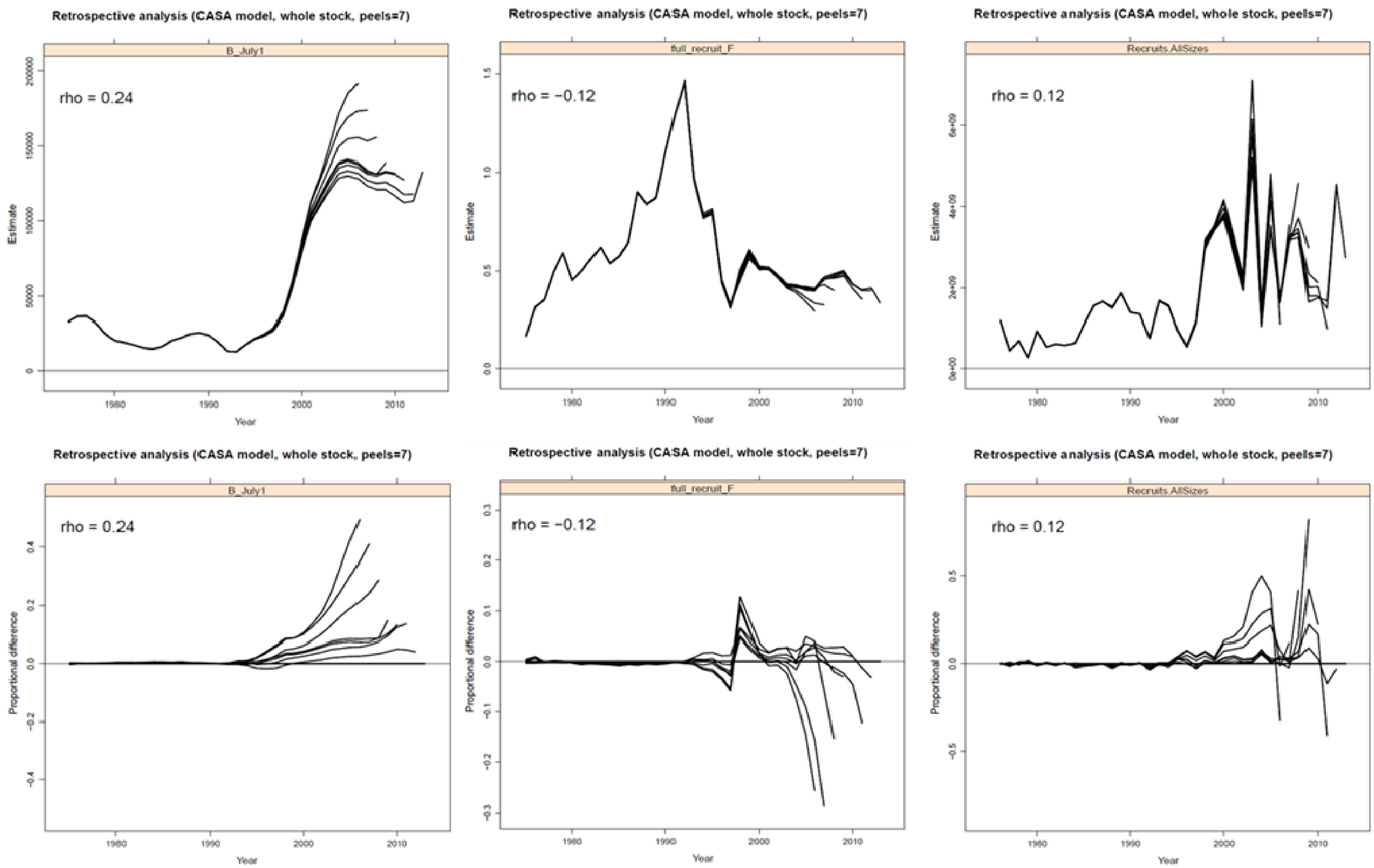


Figure B6.45b. Retrospective plots for the entire sea scallop stock.

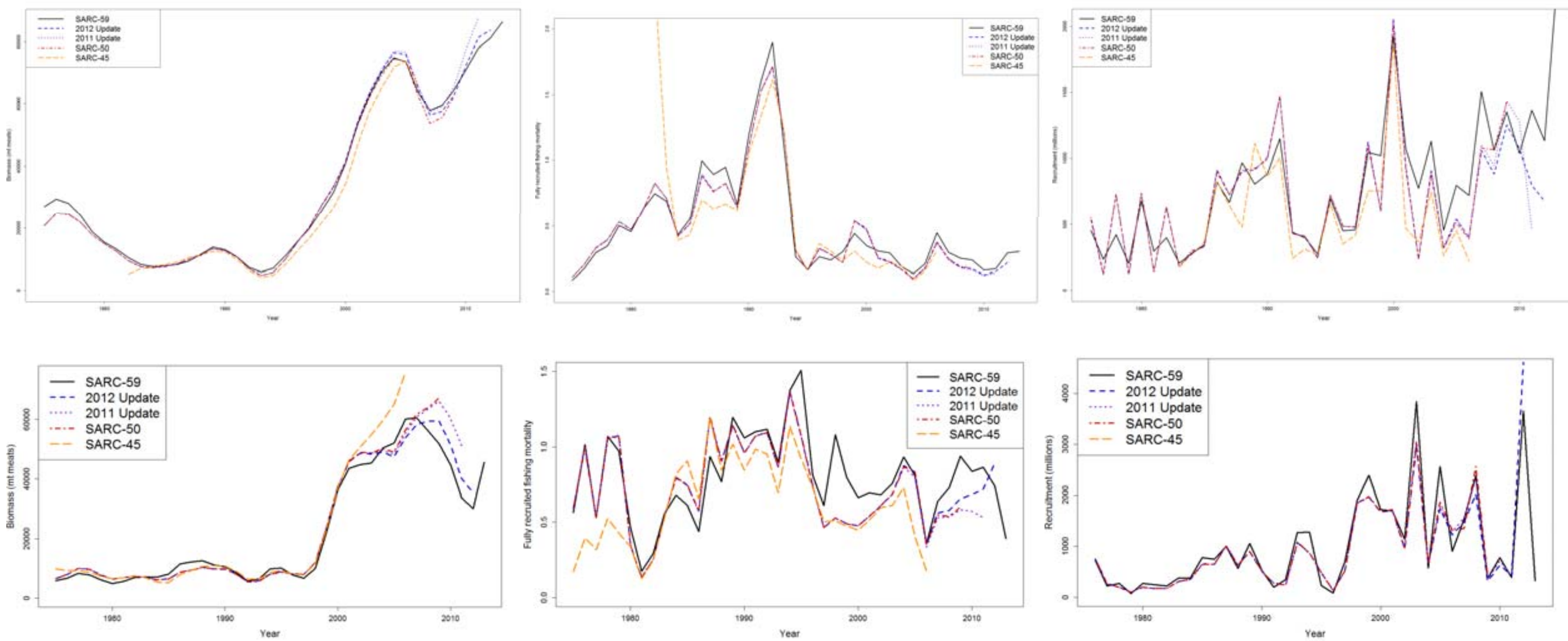


Figure B6.46. Comparison of current CASA model estimates of biomass (left), fishing mortality (middle), and recruitment (right) to previous CASA model estimates for Georges Bank (top) and the Mid-Atlantic (bottom) sea scallops.

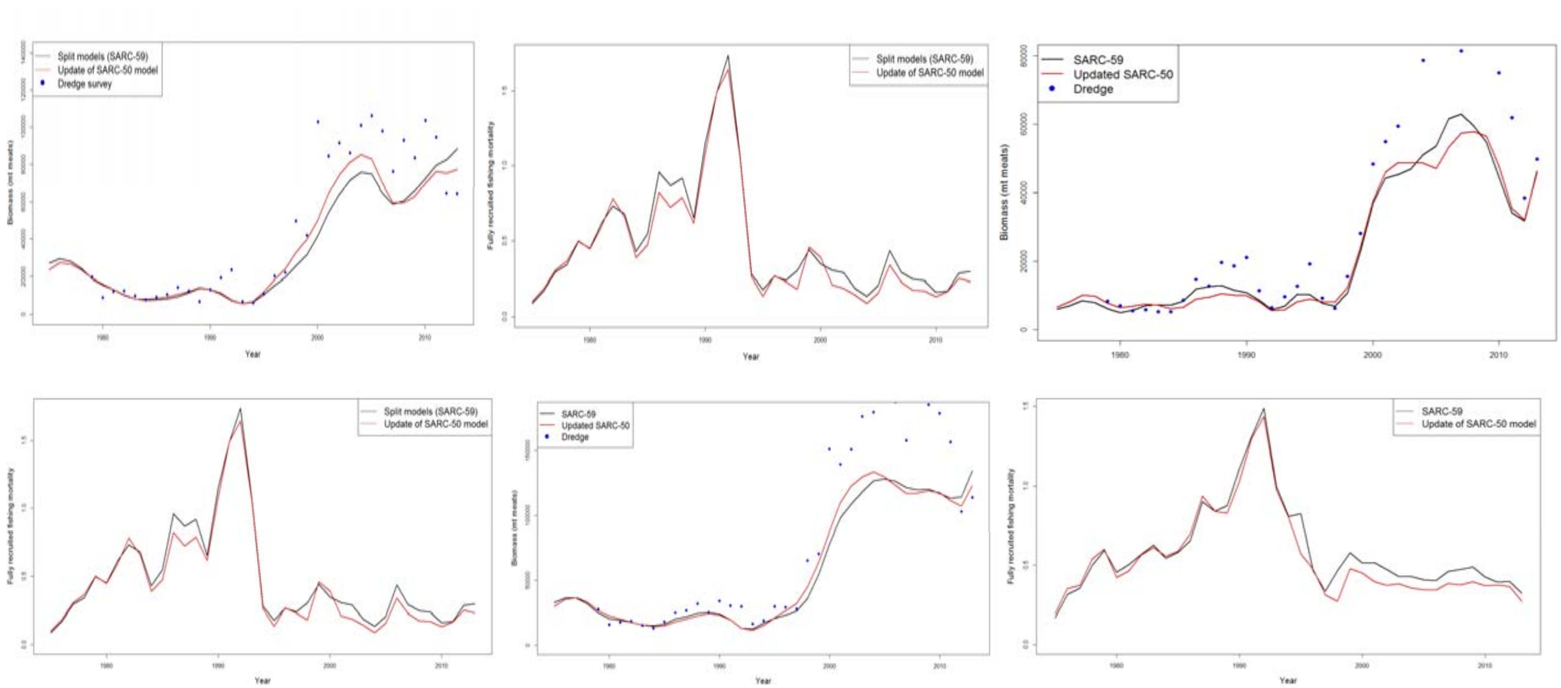


Figure B6.47. Comparisons of biomass and F estimates for the current configurations of the CASA model with the SARC-50 configurations, updated through 2013, for Georges Bank (top left and middle), Mid-Atlantic (top right and bottom left) and total (bottom middle and right). Expanded dredge survey estimates are also given for the biomass plots.

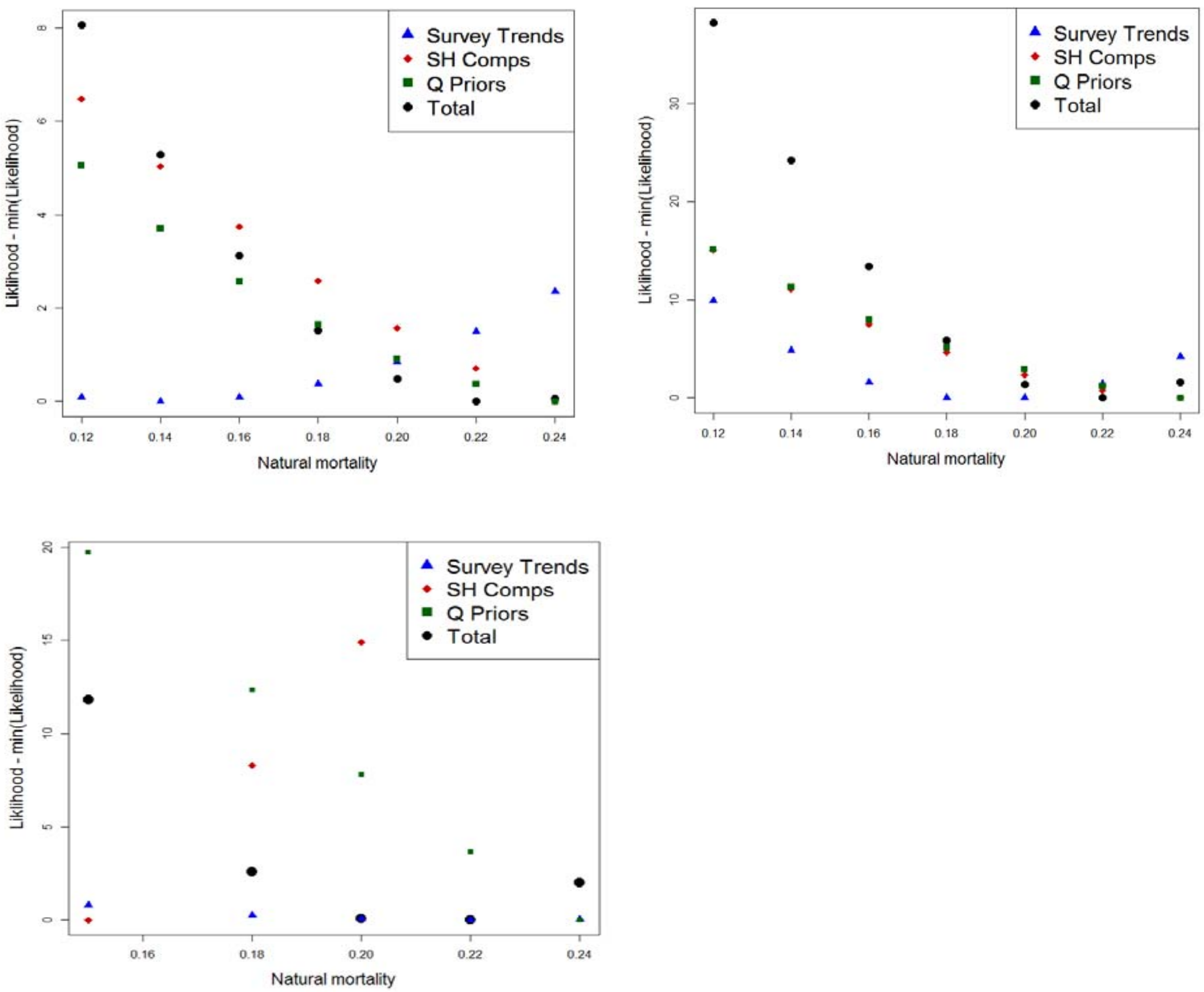


Figure B6.48. Likelihood profiles over the assumed natural mortality for all but the largest size bin (plus group mortality is 1.5x smaller sizes) for (top left) Georges Bank Open, (top right) Georges Bank Closed, (bottom) Mid-Atlantic sea scallops.

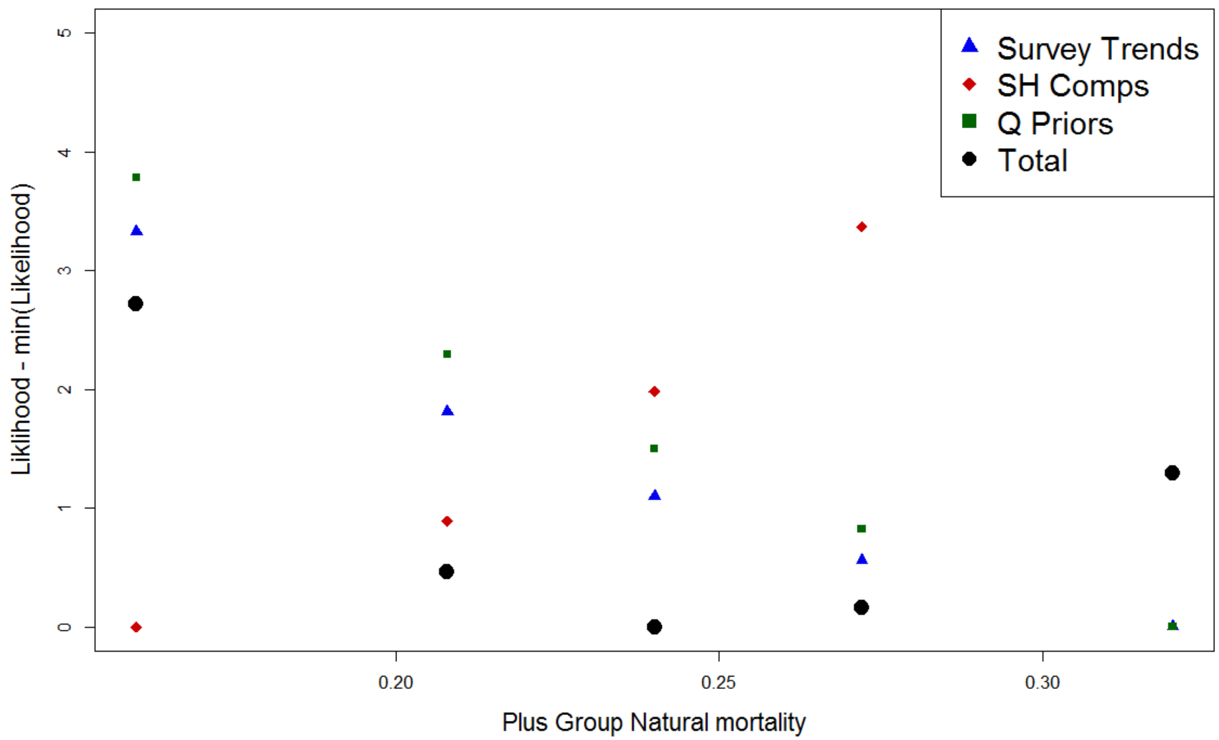


Figure B6.49. Likelihood profile analysis for the assumed plus-group natural mortality in the CASA model for sea scallops in Georges Bank closed areas. Natural mortality on the smaller size classes was fixed at 0.16.

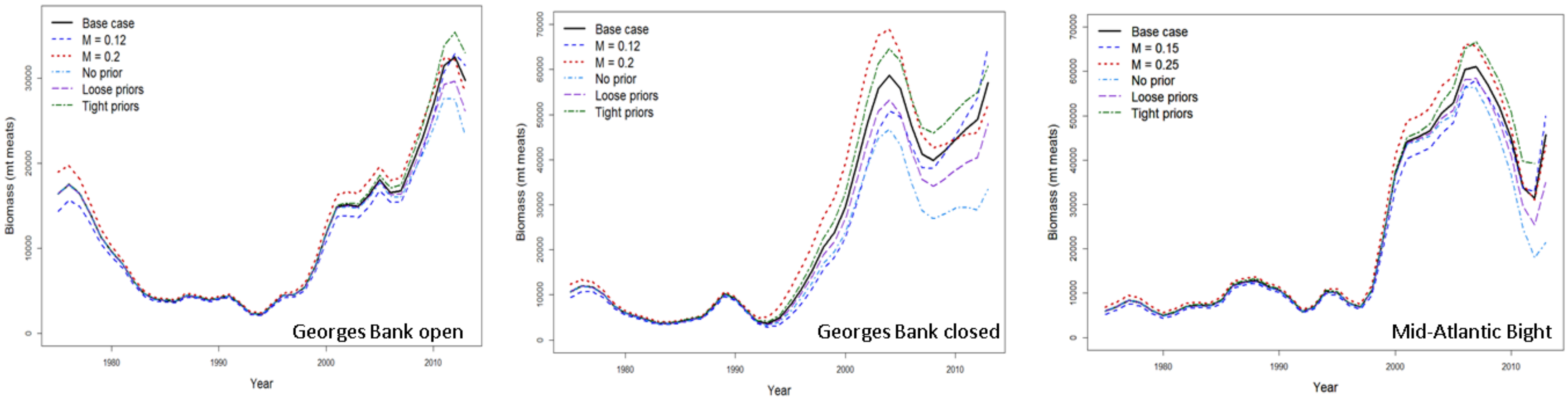


Figure B6.50. Sensitivity of estimated biomass to assumptions about natural mortality and survey efficiency priors in CASA models for Georges Bank open (left), Georges Bank closed (middle), and the Mid-Atlantic Bight (right).

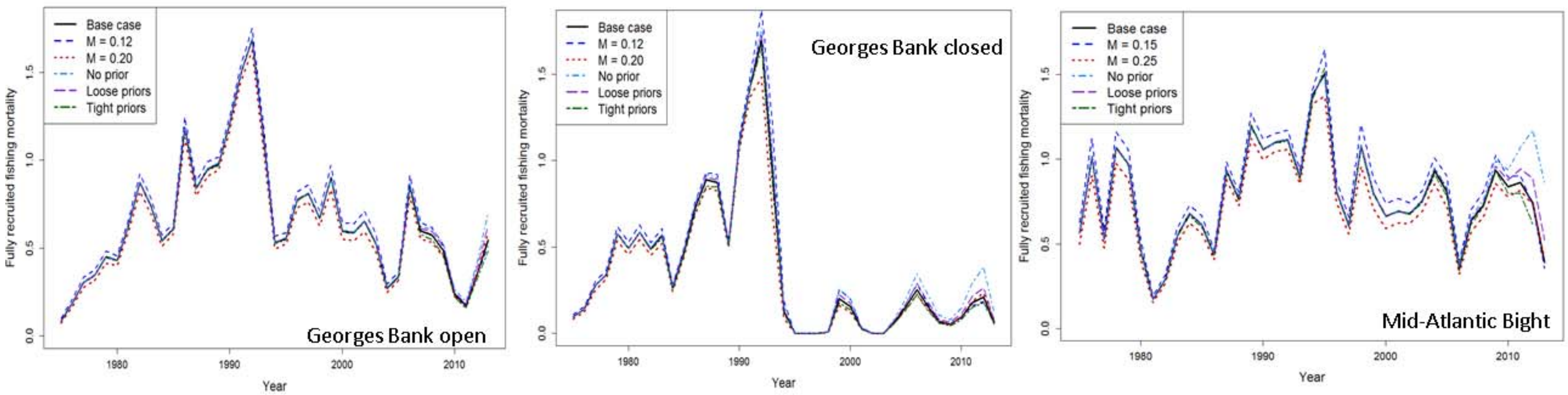


Figure B6.51. Sensitivity of estimated fishing mortality to assumptions regarding natural mortality and survey efficiency priors in CASA models for Georges Bank open (left), Georges Bank closed (middle), and the Mid-Atlantic Bight (right).

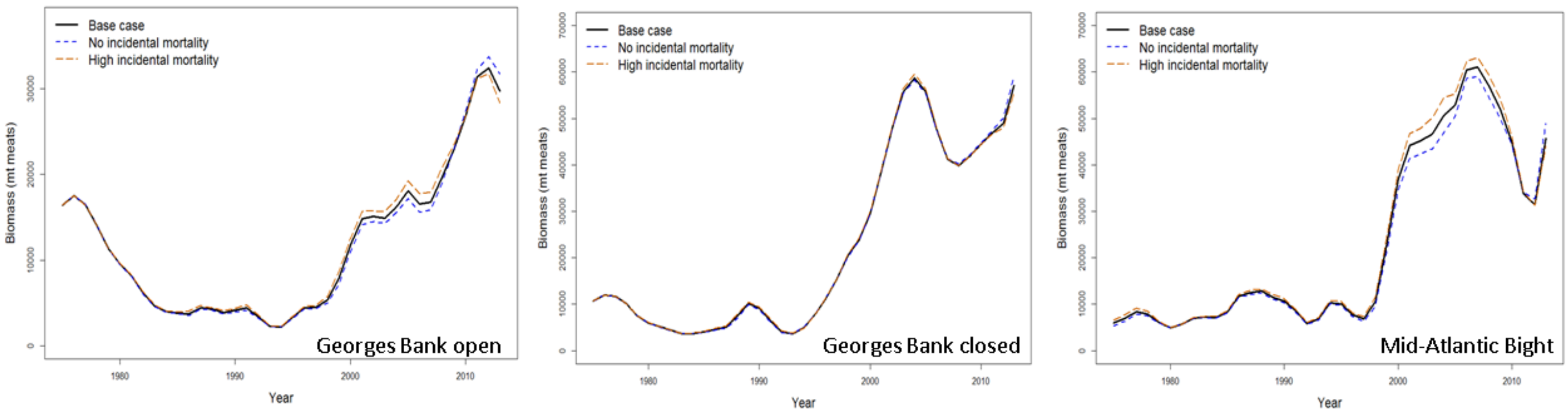
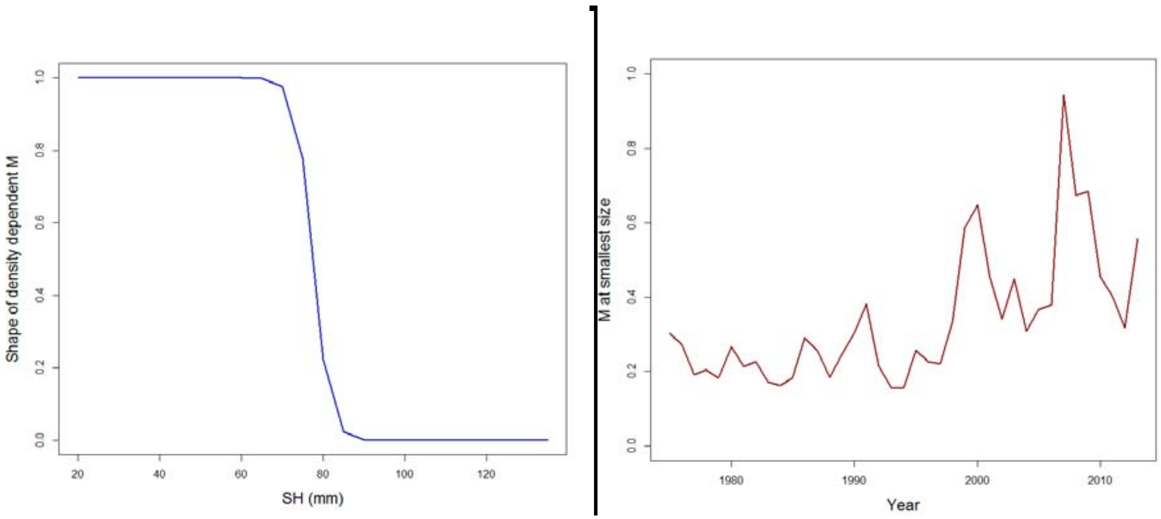


Figure B6.52. Sensitivity of estimated biomass to assumptions regarding incidental fishing mortality in CASA models for Georges Bank open (left), Georges Bank closed (middle), and the Mid-Atlantic Bight (right).



B6.53. (continued). Form of logistic curve used to define juveniles in an experimental model for density-dependent natural mortality in the Georges Bank open area.

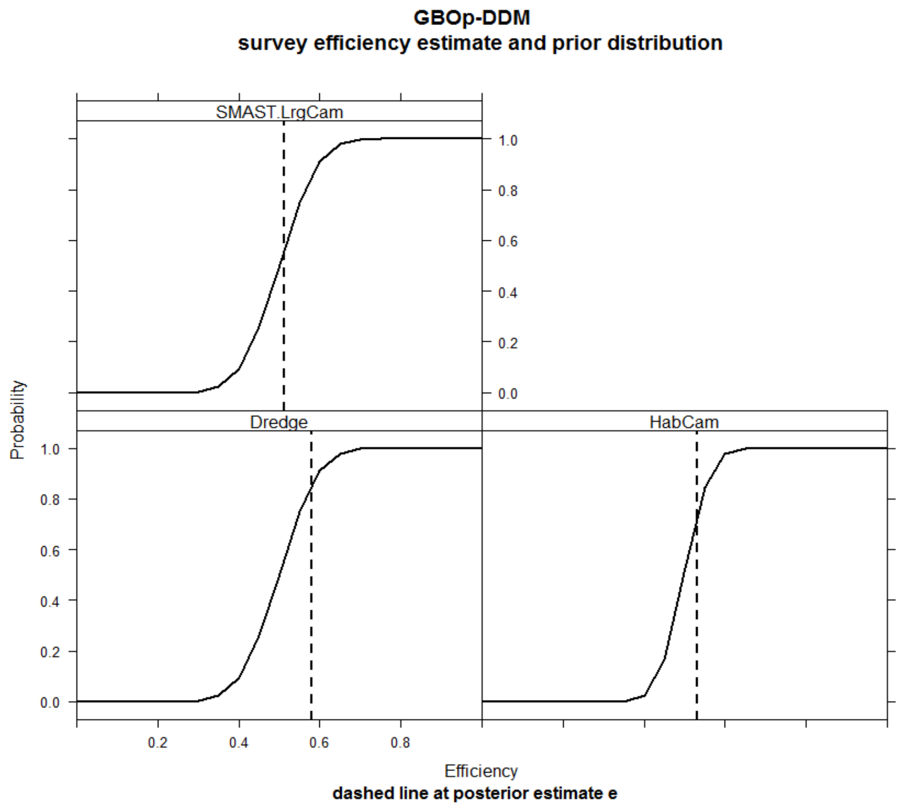


Figure B6.53. Output from experimental density-dependent natural mortality model for Georges Bank open. *Above*: Efficiency priors for three main surveys in an experimental model for density-dependent natural mortality in the Georges Bank open area.

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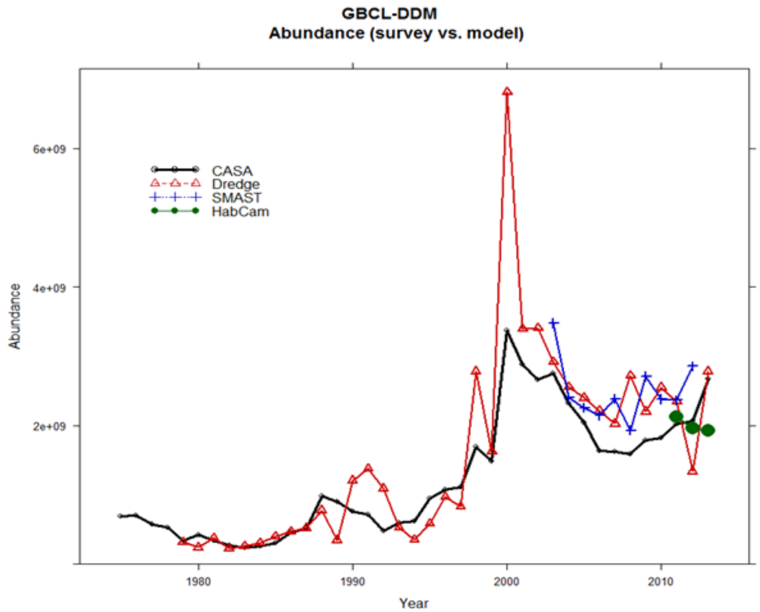


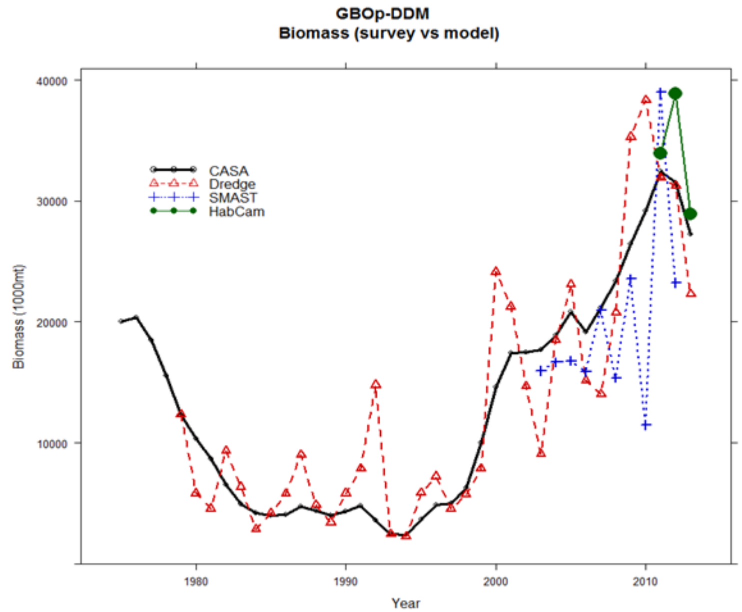


Figure B6.53. (continued). Model estimates of abundance (above) and biomass (below), together with survey stock size estimates from the dredge, SMAST and HabCam surveys in an experimental model for density-dependent natural mortality in the Georges Bank open area.

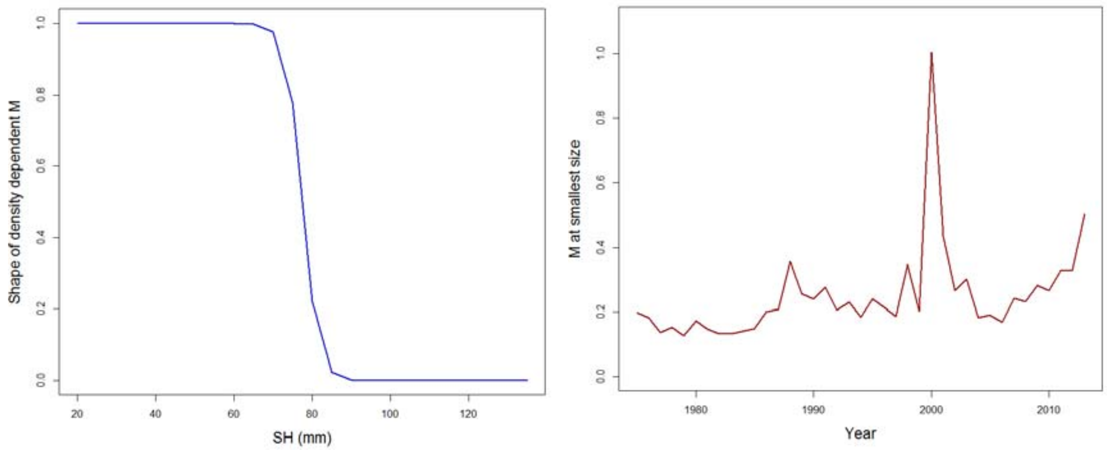


Figure B6.54. Output from experimental density-dependent natural mortality model for Georges Bank open. *Above:* Form of logistic curve used to define juveniles, and estimated natural mortality in the smallest size bin in an experimental model for density-dependent natural mortality in the Georges Bank closed area.

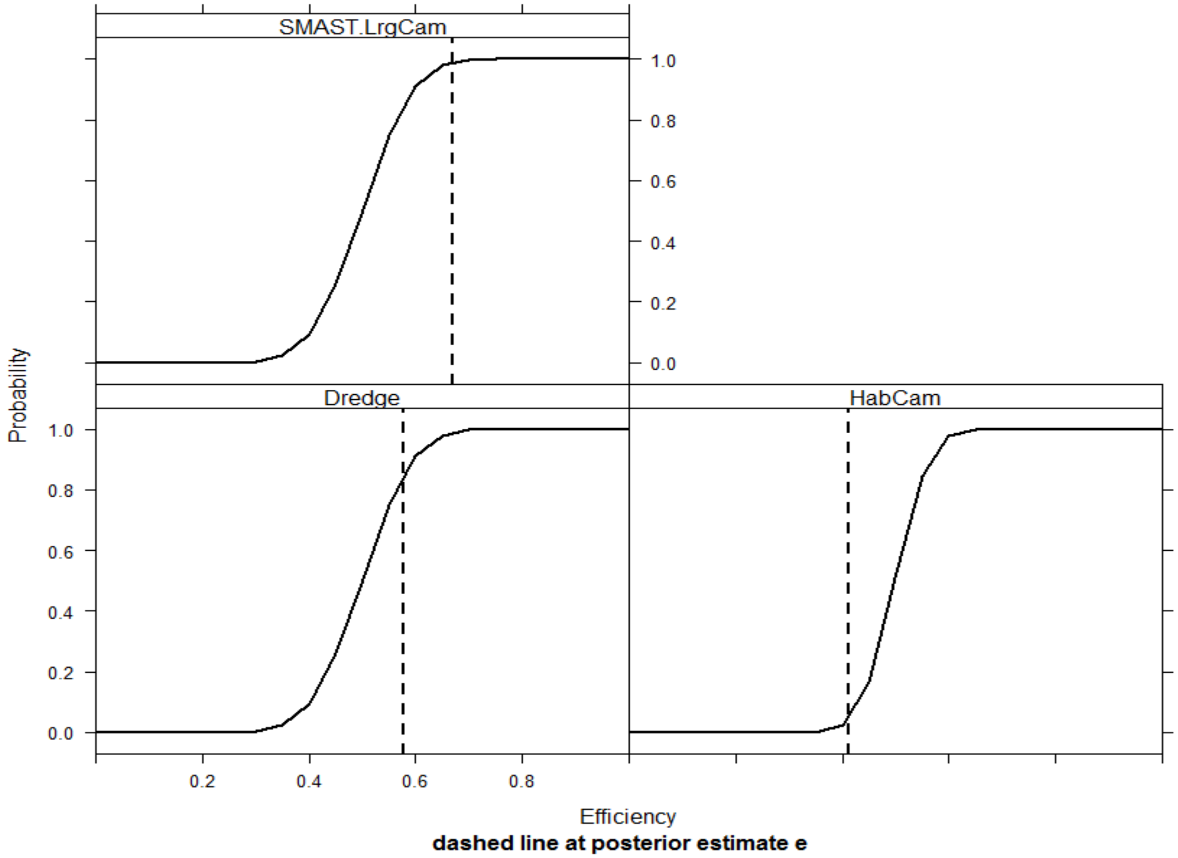


Figure B6.54. (continued). Efficiency priors for three main surveys in an experimental model for density-dependent natural mortality in the Georges Bank open area.

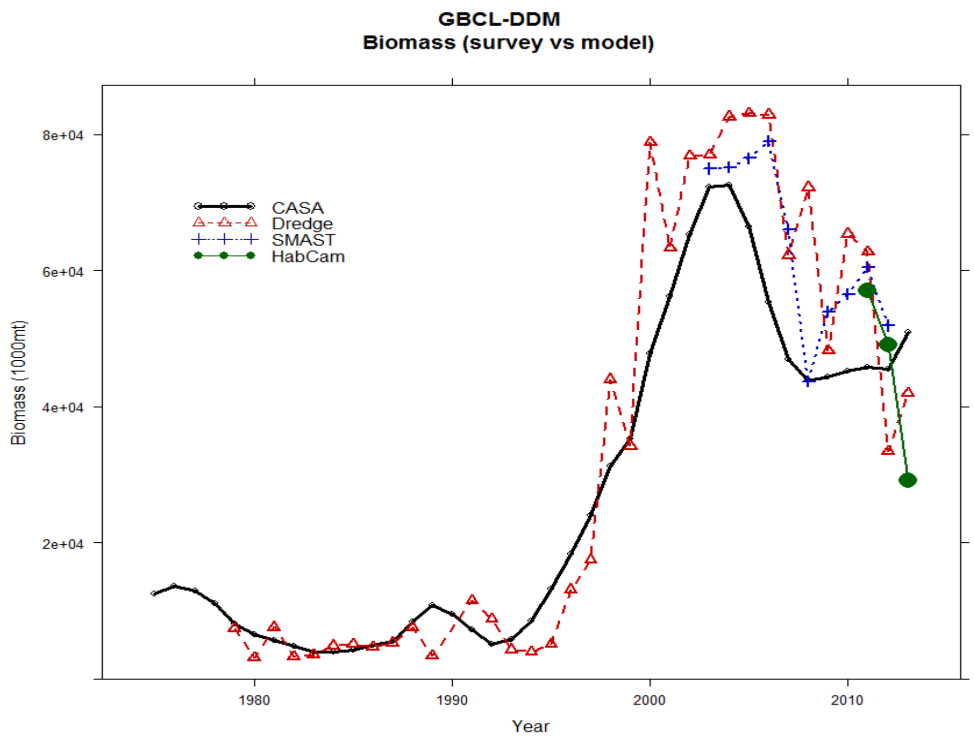


Figure B6.54. (continued). Model estimates of biomass, together with survey stock size estimates from the dredge, SMAST and HabCam surveys in an experimental model for density-dependent natural mortality in the Georges Bank open area.

B7. REFERENCE POINTS (TOR 5)

Per recruit reference points F_{MAX} and B_{MAX} were used as proxies for F_{MSY} and B_{MSY} in assessments prior to 2010 (SARC-50). F_{MAX} is the fishing mortality rate for fully recruited scallops that generates maximum yield-per-recruit. B_{MAX} was defined as the product of BPR_{MAX} (biomass per recruit at $F= F_{\text{MAX}}$ from yield-per-recruit analysis) and median numbers of recruits. As selectivity has shifted to larger scallops, yield per recruit curves have become increasingly flat, particularly in the Mid-Atlantic, making per-recruit reference points unstable. Additionally, recruitment has been stronger during the recent period when biomass has been high, suggesting that spawner-recruit relationships should be included. Finally, risk-based reference points are needed to calculate Acceptable Catch Levels/Allowable Biological Catch (ACLs/ABCs) and target fishing mortalities.

To address these issues, the SARC-50 assessment introduced a stochastic model (SYM – Stochastic Yield Model; Hart 2013) for calculating reference points and their uncertainty. It uses Monte-Carlo simulations to propagate the uncertainty in per recruit and stock-recruit calculations while calculating yield curves. B_{MSY} and F_{MSY} reference points are estimated at points where the (trimmed mean) yield curve peaks.

<u>Stochastic yield model</u>

The SYM model combines per-recruit calculations with stock-recruit relationships in order to estimate yield curves, as discussed in Beverton and Holt (1957) and Shepherd (1982). However, the SYM approach treats both the per-recruit and the stock-recruit relationships as being uncertain, and takes this uncertainty into account.

Although the SYM model is separate from CASA, efforts were made to make the two models as compatible as possible. Recruits are initially spread out over 10 size bins (20-70 mm). Growth was modelled using the same stochastic growth matrices used in the CASA model for the most recent period.

Per recruit calculations depend on a number of parameters that each carry a level of uncertainty:

1) Shell height/meat weight parameters a and b
2) Natural mortality rate M
3) Fishery selectivity parameters α and β
4) The cull size of the catch and the fraction of discards that survive
5) The level of incidental fishing mortality, i.e., non-catch mortality caused by fishing.

Details for each of these parameters are given below.

Shell height/meat weight relationships - Meat weight W at shell height H is calculated using:

$$W = \exp(a + b\ln(H))$$

The means, variances and covariance of parameters a and b were taken from Appendix B3. Similar to the growth parameters, the estimates of a and b have a strong negative correlation. This means that the predicted meat weight at a given shell height carries less uncertainty than it would appear from the variances of the individual parameters. Meat weights vary seasonally, with the greatest meat weights during the late spring and early summer (Appendix B3; Hennen and Hart 2012). However, Haynes (1966) constructed a number of monthly shell height/meat weight relationships, and did not find any significant trend in the slopes indicating that seasonality should not affect the F_{MAX} or F_{MSY} reference point. For this reason, seasonal variability was not considered a source of uncertainty for this analysis.

Natural mortality M - Natural mortality for sea scallops was estimated by Merrill and Posgay (1964) as

$$M = \frac{1}{S}\frac{C}{L} \qquad (1)$$

where L is the number of live scallops, S is the mean clapper separation time and C is the number of clappers. Probably the greatest uncertainty in this calculation is the mean separation time S. For example, Dickie (1955) estimated S to be 100 days (14.3 weeks), less than half that estimated by Merrill and Posgay (33 weeks). Reflecting this uncertainty, it was assumed S was distributed as a gamma random variable, with mean set to match the assumed mean natural mortality for each region (S=20.625 weeks on Georges Bank and 16.5 weeks in the Mid-Atlantic) and standard deviation 12 weeks. The resulting distribution of M has the desirable characteristic of being skewed to the right. The skew is reasonable because, for example, a natural mortality of $M = 0.3$ is possible, but an $M = 0$, or even close to zero, is not. Note that because S appears in the denominator of the formula above, the expected value of M is not equal to applying equation (1) with the mean value of S.

Fishery selectivity - Fishery selectivity s was estimated using an ascending logistic curve of the form:

$$s = \frac{1}{1+\exp(\alpha - \beta H)}$$

where H is shell height. The means and covariances of the α and β parameters were taken as estimated by the CASA stock assessment model during the most recent selectivity period. For Georges Bank, we used the open area selectivity in the most recent period, since reference points are calculated under the assumption that all areas are fished. Note that fishery selectivity reflects targeting and discarding as well as gear selectivity.

Cull size and discard mortality - Sea scallops that are caught but are less than 90 mm are assumed to be discarded, based on observer data. Sea scallops likely tolerate discarding fairly well, provided they are returned to the water relatively promptly and they are not damaged by the capture process or their time on deck. Here, discard mortality was simulated as a gamma distribution, with a mean of 0.2 and a standard deviation of 0.15, reflecting the high uncertainty in this parameter. This feature is also included in the SAMS projection model but not in the CASA model.

Incidental fishing mortality - Incidental fishing mortality occurs when scallops are killed but not captured by the gear. Consistent with the assumptions of the CASA model, incidental mortality F_I was estimated as 0.2 on Georges Bank and 0.1 in the Mid-Atlantic for the smallest size group. Because of the considerable uncertainty in these numbers, incidental mortality was simulated here with a gamma distribution with these means and coefficients of variation of 0.75.

Stock-recruit relationships - Beverton-Holt stock-recruit curves were fitted to spawning stock and recruitment estimates from basecase CASA model runs:

$$R = \frac{sB}{\gamma + B},$$

assuming square-root-normal errors (Figure B7.1). Here R is recruitment, B is spawning stock biomass (or egg production), and s and γ are parameters, representing the asymptotic recruitment when B is large, and the spawning stock biomass where the expected recruitment is half its asymptotic value, respectively. Standard errors of the stock-recruit parameters and their correlation were estimated using the delta method.

Calculation of equilibrium yield per recruit and yield

At each iteration of the simulation model, parameter values were drawn from their corresponding distribution and per recruit and yield curves were calculated. This was repeated $n = 100{,}000$ times and the results of each iteration were stored. The stock-recruit parameters were simulated as correlated square-root normals (chi-squared with 1 df).

For each run, equilibrium recruitment at fishing mortality F is given by

$$R = s - \gamma / b(F)$$

where b is biomass per recruit. Total yield is therefore

$$Y(F) = y(F)R = y(F)[(s-\gamma)/b(F)]$$

where y is yield per recruit.

Although simulation results in this assessment were stable, mean yield curves calculated by this method can be disproportionately influenced by outliers (Hart 2013). For this reason, a 10% trimmed mean was used to obtain the central tendency of per recruit and yield curves as a function of fishing mortality. The probabilistic F_{MSY} (and F_{MAX}) were taken as the fishing mortality that maximizes the trimmed mean yield curve (yield per recruit curve). The probabilistic MSY and B_{MSY} are the trimmed mean yield and biomass at F_{MSY} over all runs.

Results

Stock-recruit curves were better defined on Georges Bank than in the Mid-Atlantic (Figure B7.1). While Y_{MAX} and B_{MAX} values were generally well defined, F_{MAX} was highly uncertain in both regions, and hit the $F = 1$ bound in a majority of the simulations in the Mid-Atlantic (Figures B7.2 to B7.4). MSY based reference points were better defined, as potential stock-recruit relationships tend to reduce F_{MSY} to well below F_{MAX} (Figures B7.5 and B7.7).

MSY estimates for the combined Georges Bank and Mid-Atlantic areas range from 10,000 mt to 40,000 mt meats, and B_{MSY} between about 40,000 to 150,000 mt (Figures B7.8 to B7.10). F_{MSY} values for the combined stock are highly uncertain.

Trimmed mean yield curves have a maximum at $F_{MSY} = 0.3$ on Georges Bank, and $F_{MSY} = 0.74$ in the Mid-Atlantic, with corresponding MSY values of 9,148 and 15,737 mt meats, respectively (Table B7.1, Figure B7.11). Trimmed mean estimates for the combined stock are F = 0.48, MSY =23,798 mt, and B_{MSY} = 96,480 mt. The entire distribution of yield for the combined stock is shown in Figure B7-12).

Special considerations for sedentary resources under area management

The above reference point calculations are based on the assumption that fishing mortality risk does not vary among individuals. For sedentary organisms such as sea scallops, these assumptions are never even approximately true. With closed and rotational area management, the assumption of uniform fishing mortality is strongly violated (Hart 2001, 2003; Smith and Rago 2004). In such situations, mean yield-per-recruit, averaged over all recruits, may be different than yield-per-recruit obtained by a conventional per-recruit calculation performed on a recruit that suffers the mean fishing mortality risk (Hart 2001). In these types of situations, estimates of fishing mortality may be biased low, because individuals with low mortality risk are overrepresented in the population (Hart 2001, 2003).

Reference point	SARC-50	SARC-59
F_{MSY}	0.38	0.48
B_{TARGET}=B_{MSY} (mt, meats)	125,358	96,480
$B_{THRESHOLD}$=1/2 B_{MSY} (mt, meats)	62,679	48,240
MSY (mt, meats)	24,975	23,798

Table B7-1. Previous (SARC-50) and revised (SARC-59) reference points for sea scallops.

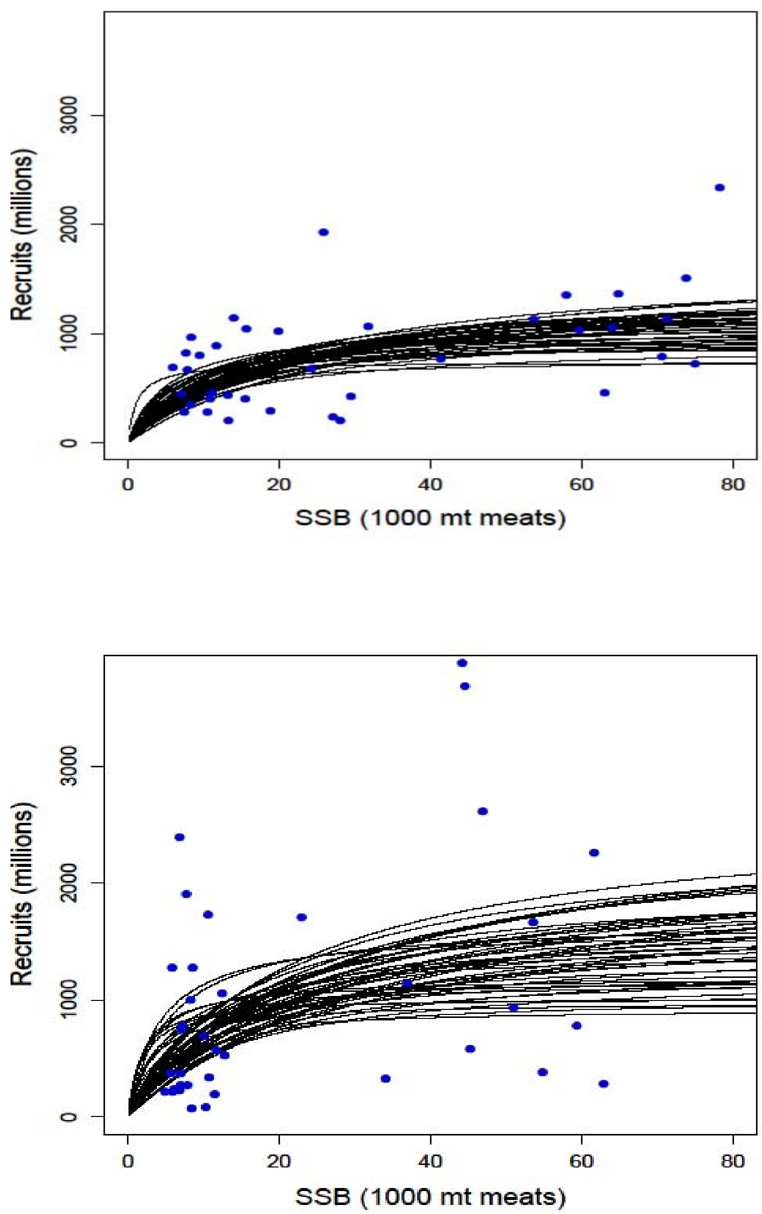


Figure B7.1 Stock-recruit relationships for Georges Bank (top) and the Mid-Atlantic (bottom) showing spawner-recruit estimates from the CASA model (blue dots) and 50 example fitted Beverton-Holt curves.

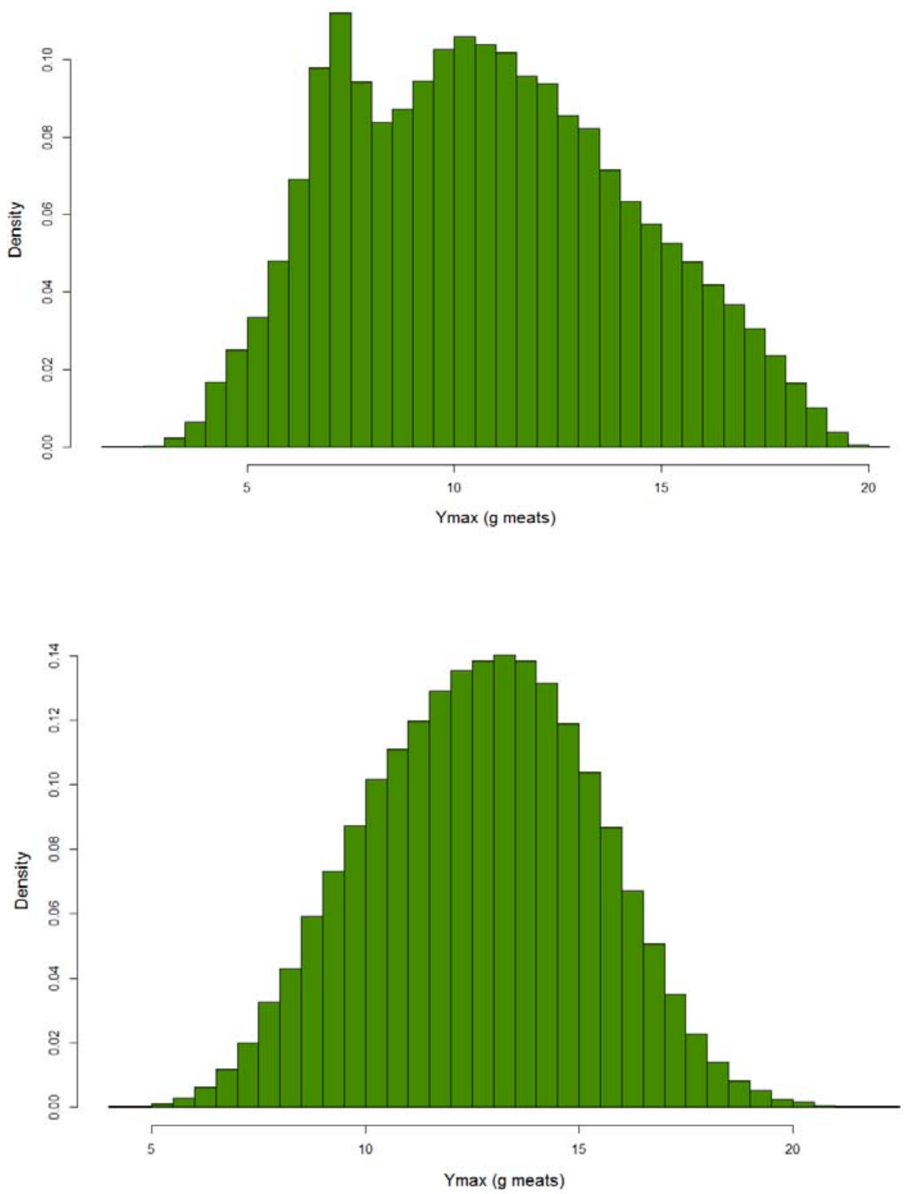
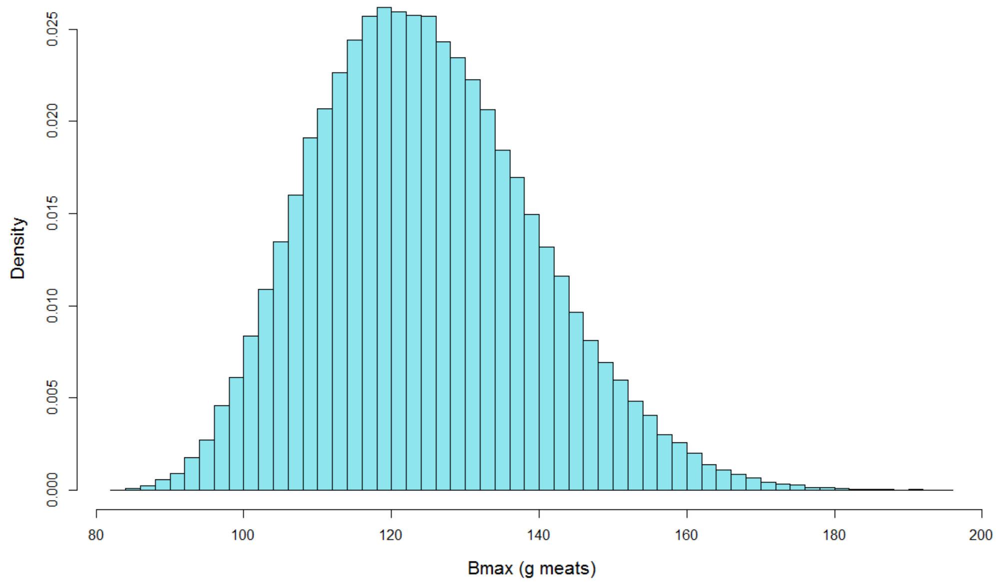


Figure B7.2. Probability distributions for maximum yield per recruit $Y_{\max}$ in the Georges Bank (top) and Mid-Atlantic (bottom) regions.



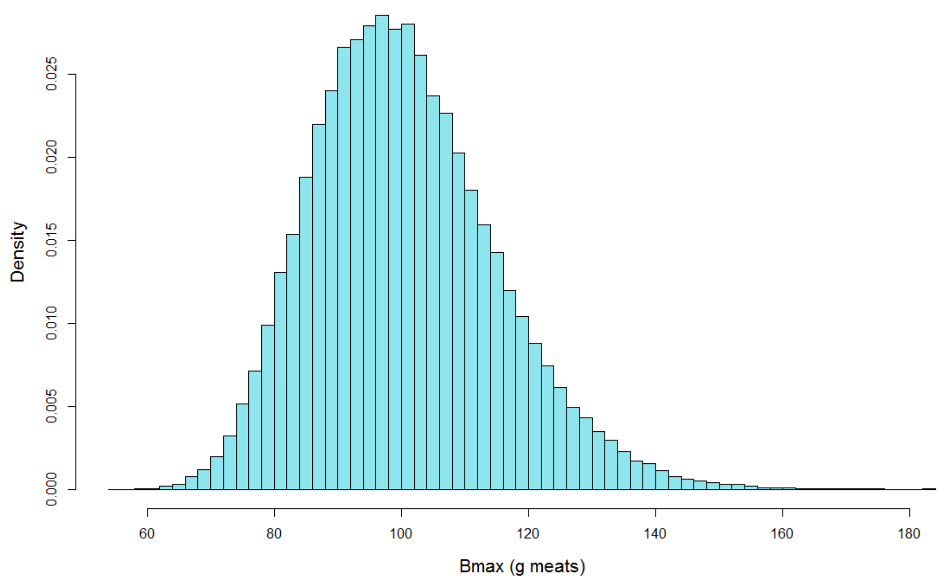


Figure B7.3. Probability distributions for biomass per recruit at B_{max} in the Georges Bank (top) and the Mid-Atlantic (bottom) regions.

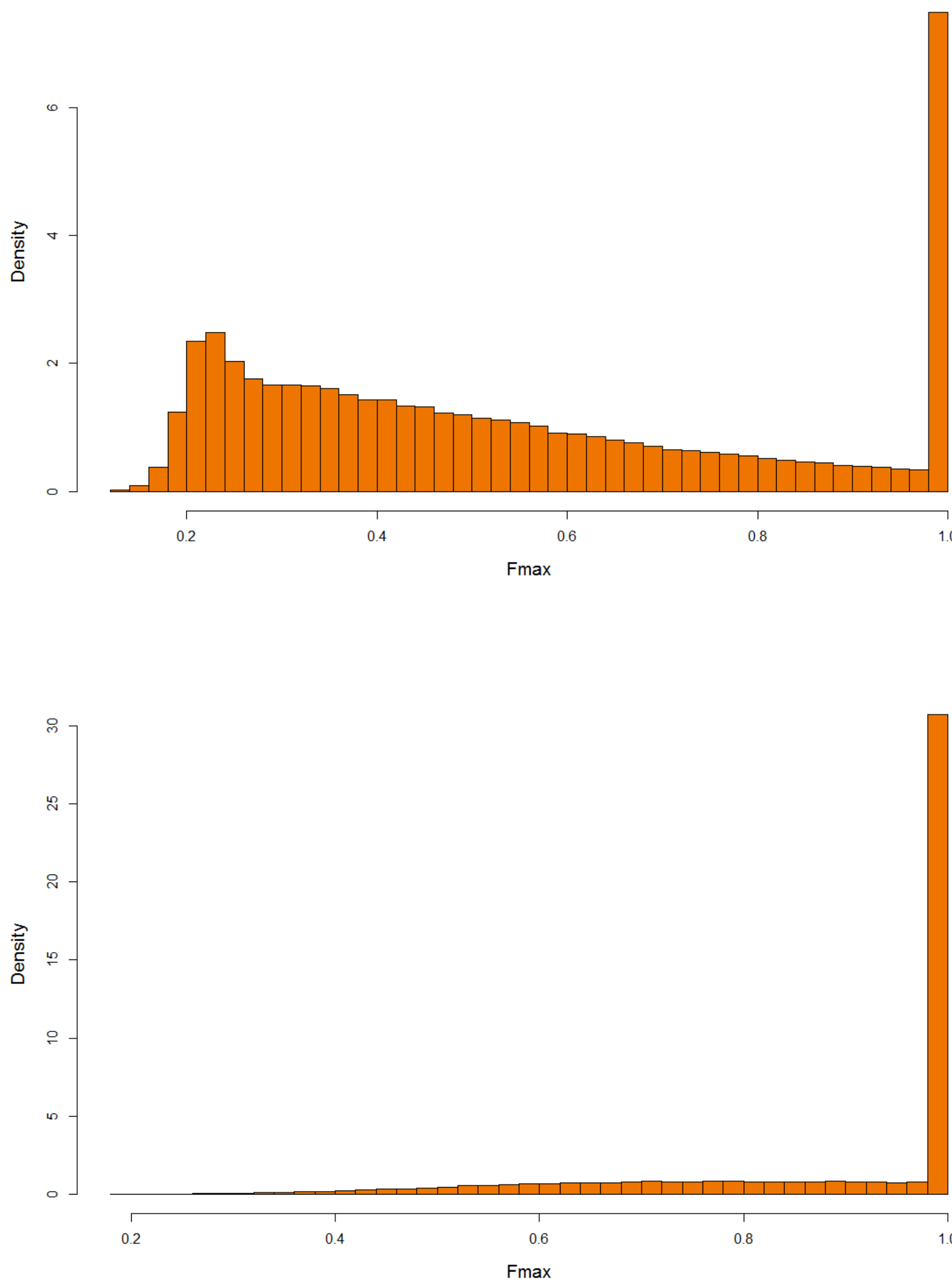
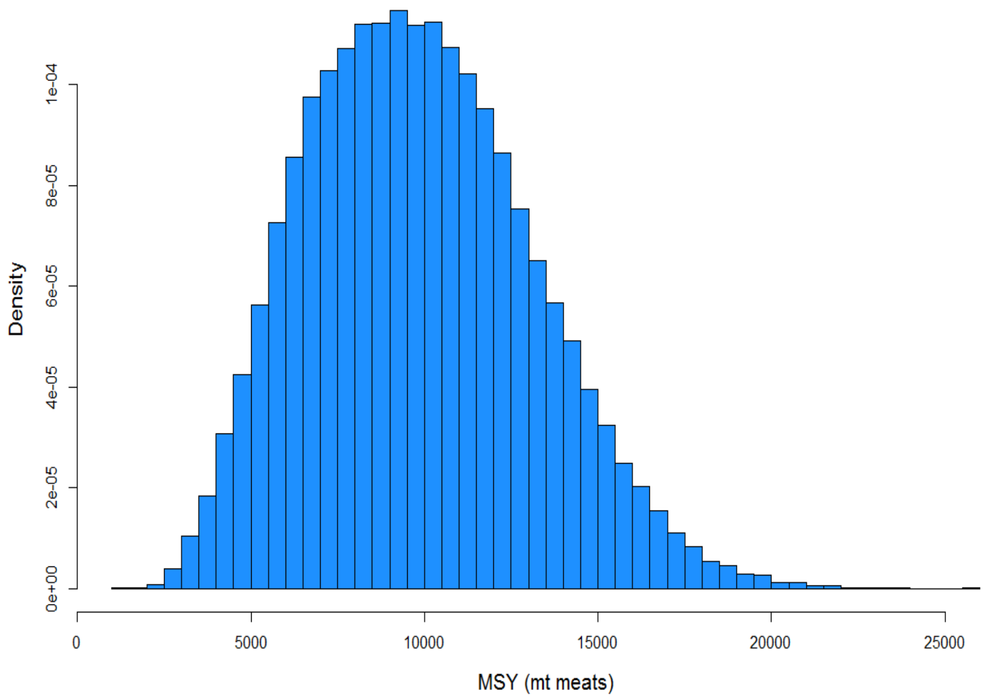


Figure B7.4. Probability distributions for the fishing mortality that gives maximum yield per recruit (F_{max}) in the Georges Bank (top) and Mid-Atlantic Bight (bottom) regions.



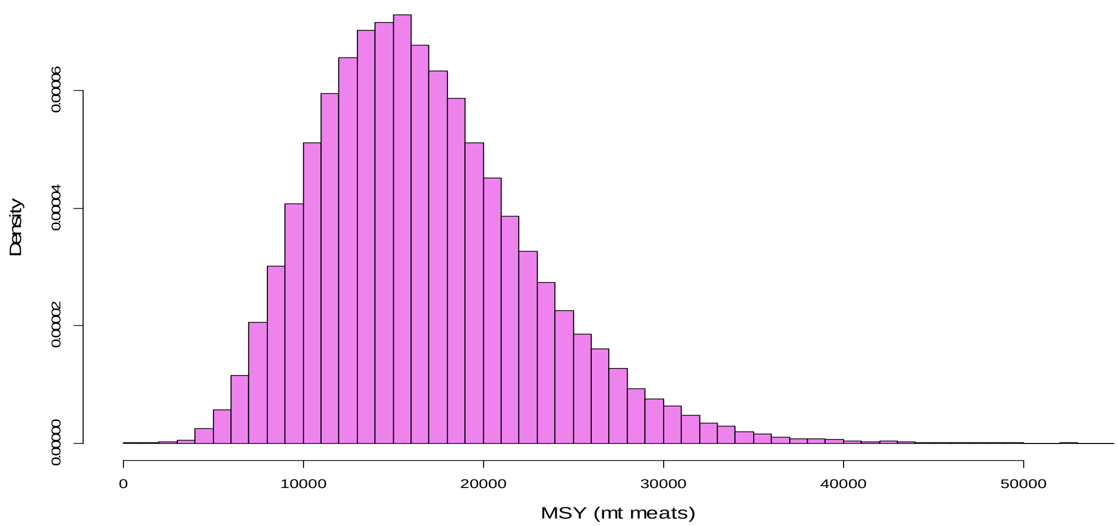
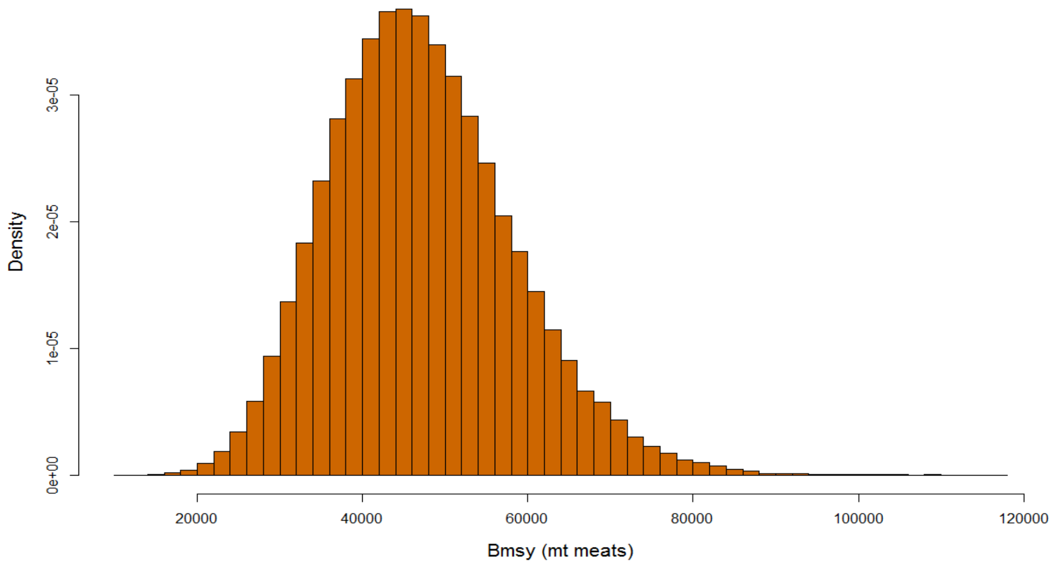


Figure B7.5. Probability distributions for MSY in the Georges Bank (top) and Mid-Atlantic (bottom) regions.



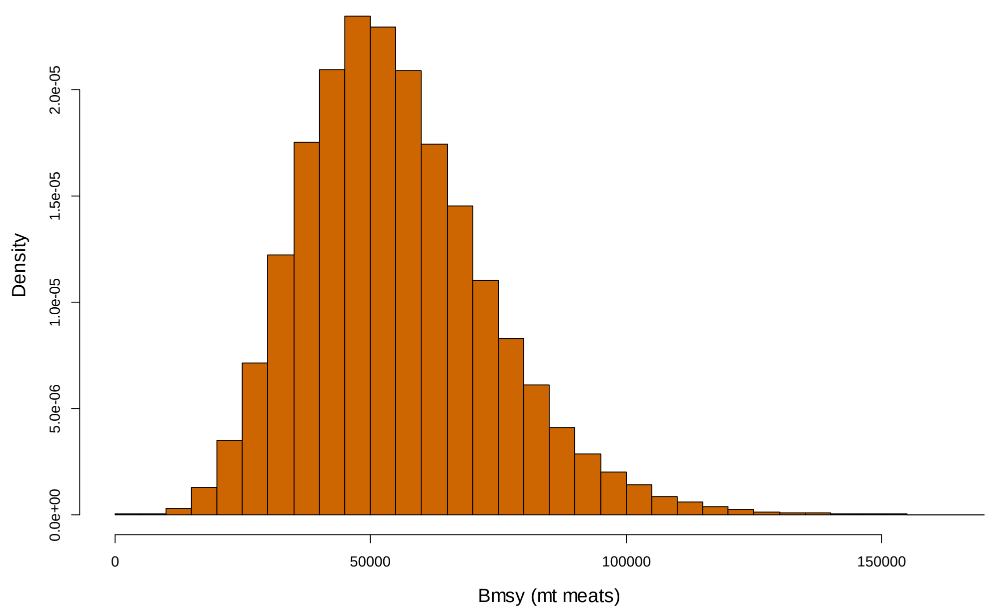


Figure B7.6. Probability distributions for B_{MSY} in the Georges Bank (top) and Mid-Atlantic (bottom) regions.

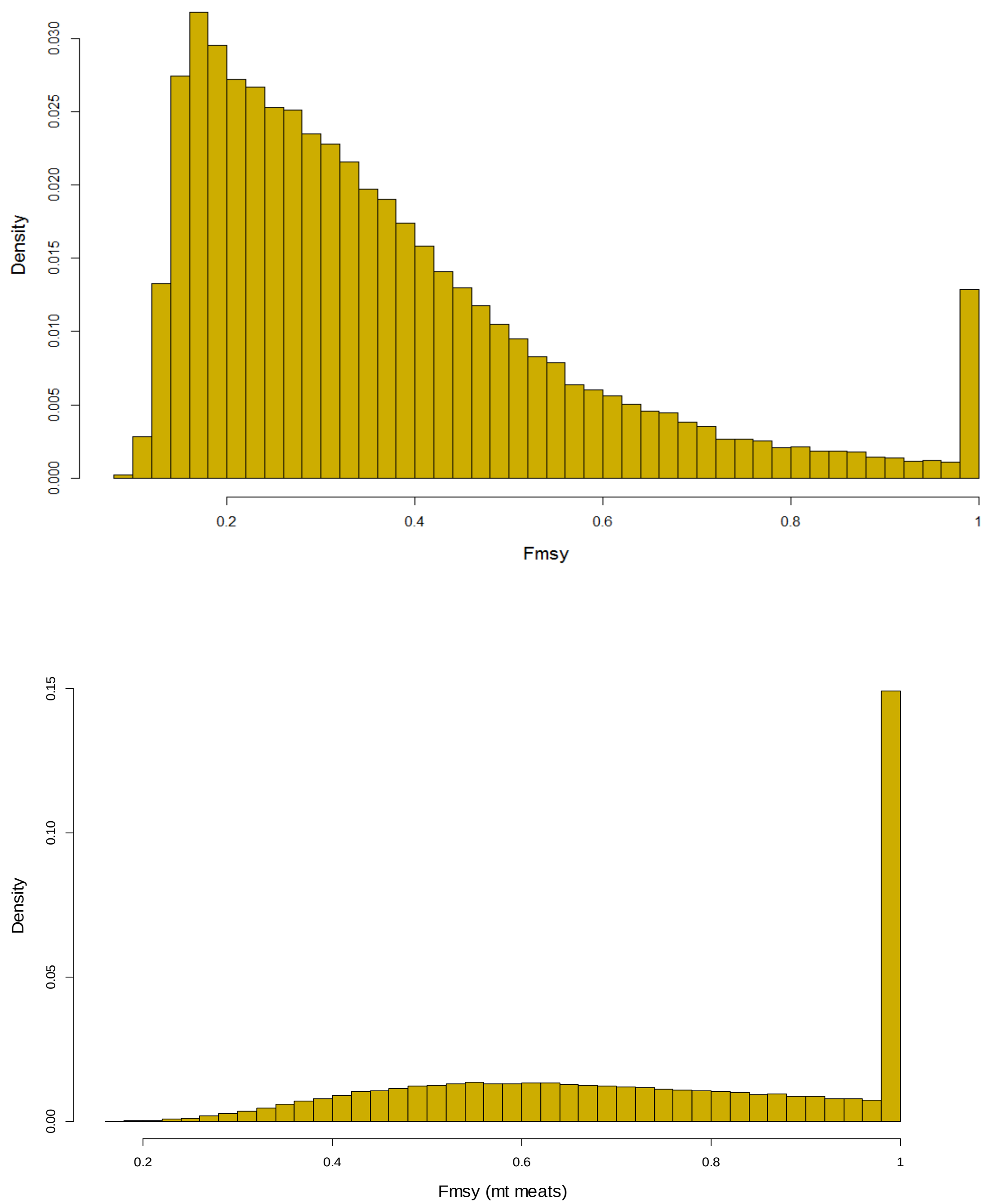


Figure B7.7 Probability distributions for F_{msy} in the Georges Bank (top) and Mid-Atlantic (bottom) regions.

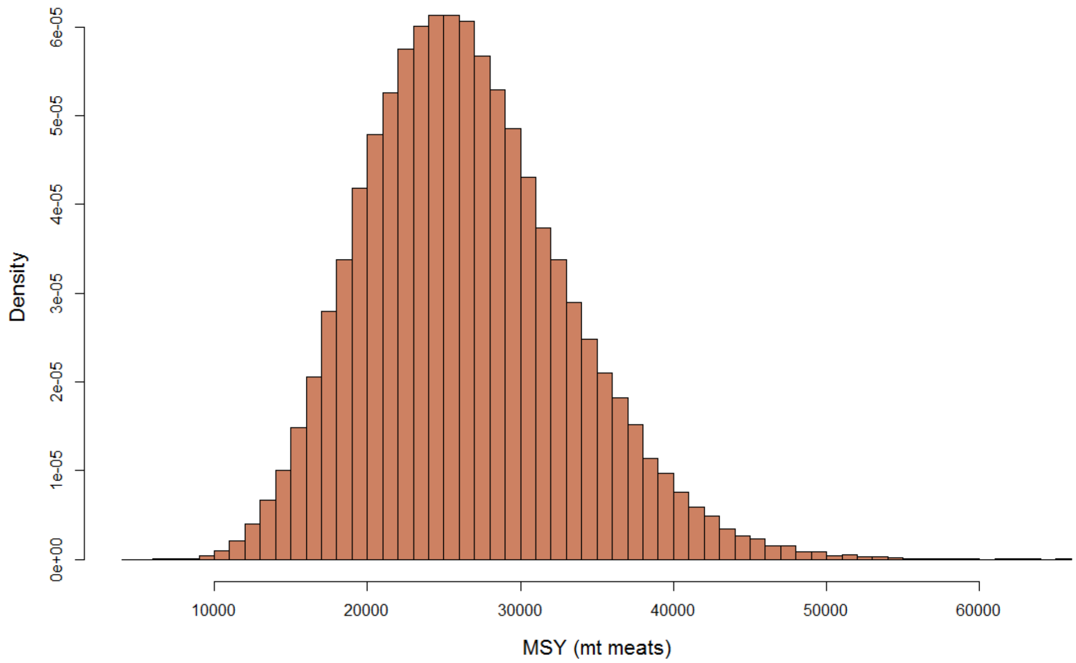


Figure B7.8. Probability distribution for MSY in the combined Georges Bank and Mid-Atlantic region.

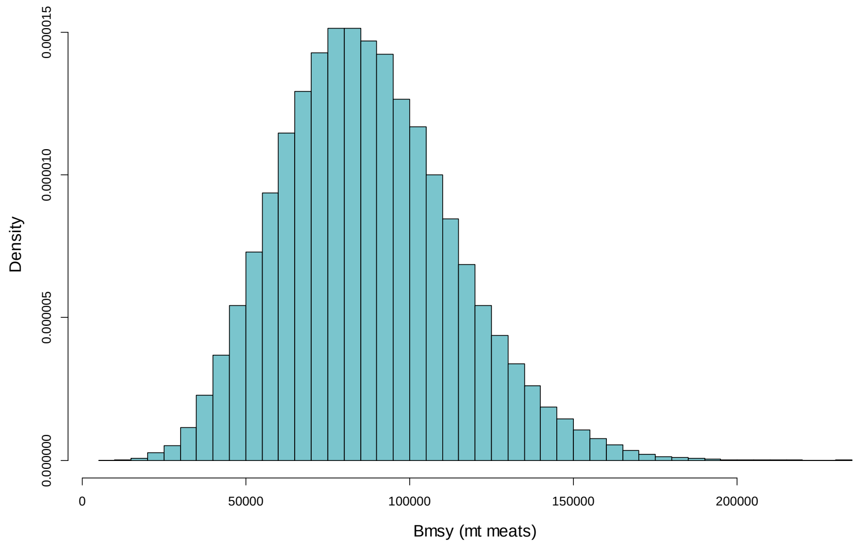


Figure B7.9. Probability distribution for B_{MSY} in the combined Georges Bank and the Mid-Atlantic region.

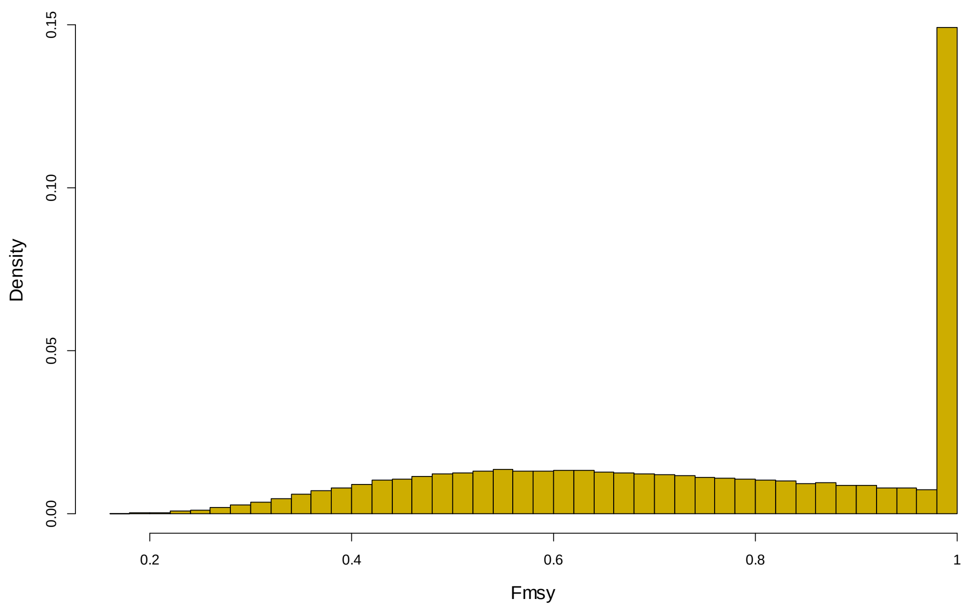


Figure B7.10. Probability distribution for F_{MSY} in the combined Georges Bank and the Mid-Atlantic region.

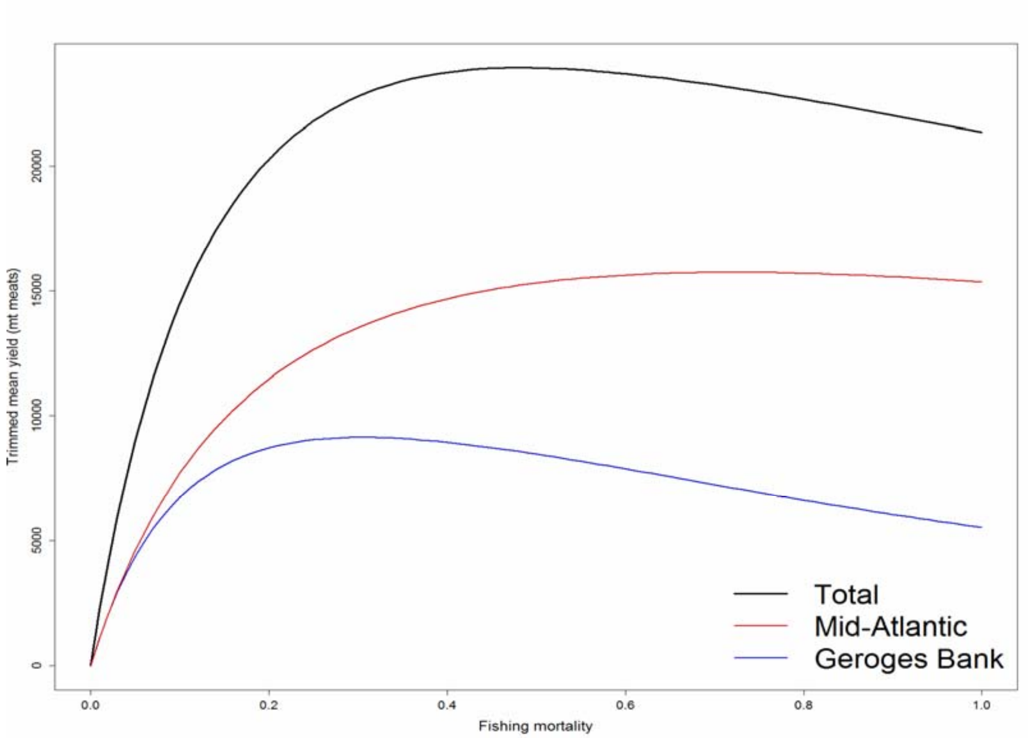


Figure B7.11. Trimmed mean yield as a function of fishing mortality for Georges Bank, the Mid-Atlantic, and combined areas.

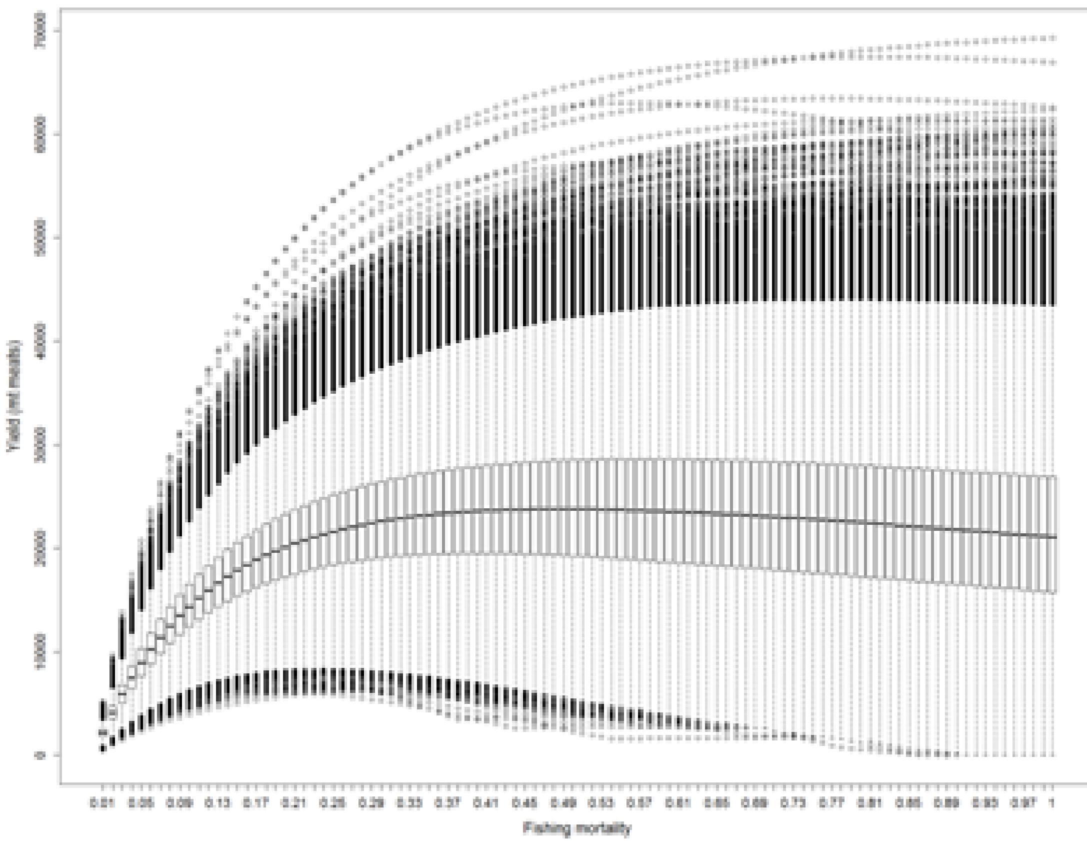


Figure B7.12. Boxplots for yield in the combined Georges Bank and Mid-Atlantic region as a function of fishing mortality.

B8 - Status Determination (TOR 6)

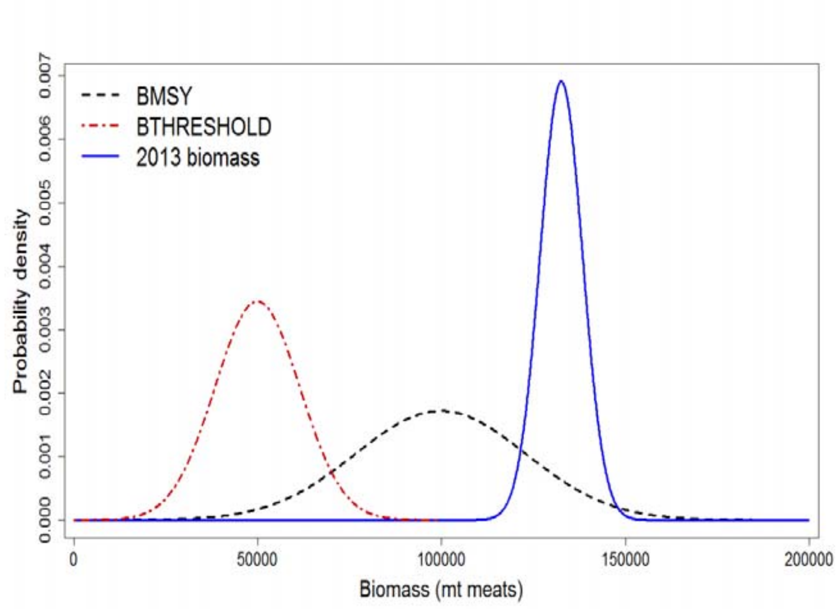
According to the Amendment 10 overfishing definition (NEFMC 2003), sea scallops are overfished when the survey biomass index for the whole stock falls below 1/2 B_{TARGET}, with B_{TARGET} set equal to B_{MSY} or its proxy (see table below). The current $B_{THRESHOLD}$ is 62,679 mt (NEFSC 2010) and the recommended value in this assessment is 48,240 mt. The estimated combined stock biomass in 2013 was 132,561 mt, which is above both $B_{THRESHOLD}$ reference point values. Thus, the stock is not overfished based on either criterion.

None of the 100,000 simulations done for the SYM model estimated a B_{MSY} that was greater than twice the CASA estimated 2013 biomass. The standard error in the 2013 CASA biomass was estimated at 5772 mt, which is likely underestimates the uncertainty. However, given that both surveys estimated biomasses over 110,000 mt in 2013, it is highly likely that the actual biomass in 2013 was above 100,000 mt. Because less than 1% of the SYM runs estimated a B_{MSY} greater than 200,000 mt, it can be concluded that the chances that the stock is overfished is very small, probably less than 1% (Figure B8.1).

The current $F_{\mathrm{MSY}} = 0.38$ (NEFSC 2010) and the recommended F_{MSY} in this assessment is 0.48. The estimated fishing mortality for the whole stock in 2013 was 0.32, which is below both F_{MSY} reference points. Therefore, overfishing was not occurring in 2013 based on either criterion.

Based on SYM model results, there is about a 12% chance that F_{MSY} is below 0.32. The standard error for fishing mortality in 2013 was 0.03 from the CASA model. Combining these results indicate that the probability of overfishing in 2013 was about 13% (Figure B8.1). This probability of overfishing is likely understated because CASA is probably underestimating uncertainty.

Type	**2013 stock estimate**	**Reference point**	**NEFSC (2010)**		**Recommended this assessment**	
			BRP	**Overfished, overfishing?**	**BRP**	**Overfished, overfishing?**
Biomass (mt)	132,561	$B_{target}=B_{MSY}$ $B_{Threshold}=B_{Target}/2$	125,358 62,679	No	96,480 48,240	No
Fishing mortality	0.32	F_{MSY}	0.38	No	0.48	No



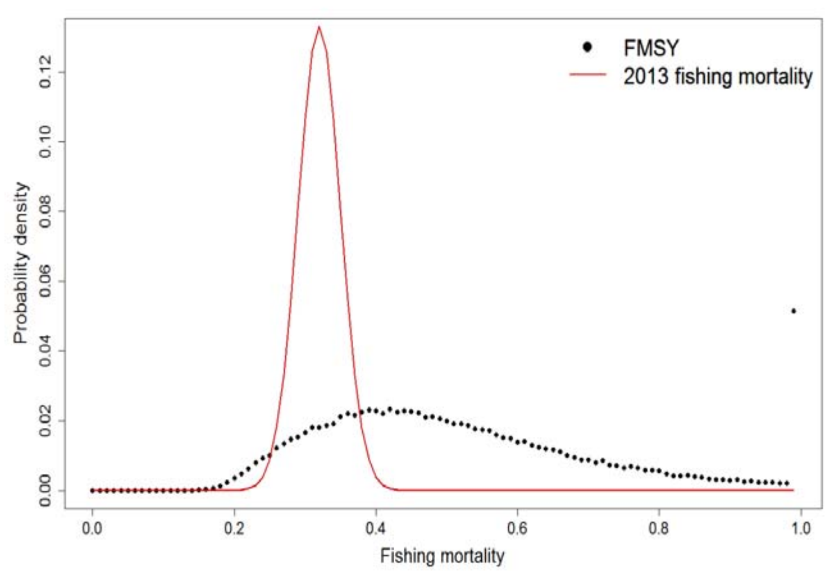


Figure B8.1. *Top*: Probability distributions for B_{MSY}, $B_{THRSHOLD}$ and 2013 biomass. *Bottom*: Probability distributions of F_{MSY} and 2013 fishing mortality.

B9 STOCK PROJECTIONS (TOR 7)

Because of the sedentary nature of sea scallops, fishing mortality can vary considerably in space even in the absence of area specific management (Hart 2001). Rotational management and long-term closures exacerbate this heterogeneity. Projections that ignore spatial variation can be unrealistic and misleading. For example, suppose 80% of the stock biomass is in areas closed to fishing (as occurred in some years in Georges Bank). A stock projection that ignored the closure and assumed an overall *F* of 0.2 would forecast landings nearly equal to the entire stock biomass in the areas open to fishing. Thus, using a non-spatial forecasting model could lead to unsustainable harvest levels under area management. For these reasons, a spatial forecasting model (the Scallop Area Management Simulator, SAMS) was developed for use in sea scallop management (Appendix B10). Various versions of SAMS have been used since 1999.

Growth is modeled in SAMS and CASA in a similar manner, except that each subarea of Georges Bank and the Mid-Atlantic in SAMS has its own stochastic growth transition matrix derived from the shell increments collected in that area. Mortality and recruitment are also area-specific. Fishing mortality can either be explicitly specified in each area, or calculated using a simple fleet dynamics model that assumes fishing effort is proportional to estimated LPUE.

Projected recruitment is modeled stochastically with the log-transformed mean and covariance for recruitment in each area matching that observed in NEFSC dredge survey time series. In the example projection shown here, initial conditions are based on regional shell height data from the 2013 dredge surveys, with mean regional biomass (Georges Bank open and closed, and Mid-Atlantic) set to match CASA estimates for 2013. Initial values in each subarea are varied according to specified uncertainties. Natural mortality for each run is selected from the same distributions used in the SYM reference point model. Further details regarding the SAMS model are given in Appendix B10.

One set of example runs are used in this assessment to demonstrate of the utility of the SAMS model. Projections used to manage the fishery are carried out by the Scallop Plan Development Team while evaluating potential management measures. For example, SAMS runs for management in 2015-2016 will be updated with 2014 survey data in the fall of 2014 after this assessment is complete.

Example SAMS runs

For the example simulations, the stock area was split into 16 subareas (Figure B3-1), seven in the Mid-Atlantic (Virginia Beach, Delmarva, Elephant Trunk, Hudson Canyon South, New York Bight, Long Island, and New York Bight inshore) and ten on Georges Bank (Closed Area I, II and Nantucket Lightship EFH closures, Closed Area I, II and Nantucket Lightship access areas, Great South Channel proposed closure and the remainder of the Great South Channel, Northern Edge and Peak, and Southeast Part).

The EFH (Essential Fish Habitat) closures on Georges Bank were assumed to be closed for the duration of the simulations. The Georges Bank access areas were assumed to be fished on a rotating basis corresponding to actual management in 2013-2014, and probable management in

2015 (Closed Area I is fished in 2013, 2016, 2017, Closed Area II in 2013, 2014, 2016 and 2017, Nantucket Lightship in 2013, 2014, 2016, and 2017). The Hudson Canyon South rotational closure area was assumed to be fished in 2013, closed in 2014-2015, and fished 2016-2017. The Elephant Trunk rotational area was assumed closed in 2013-2014, and fished in 2015-2017. Delmarva was closed in 2013 and fished in 2014-2017. All other areas (Virginia Beach, New York Bight, Long Island, South Channel, Northern Edge and Peak, Southeast Part) were part of the open areas, where scallop fishermen may chose where to fish, subject to a day at sea limit. These days at sea limits were set at 33 days in 2013, 31 days in 2014, and the number of days that will result in an open area $F=F_{MSY} = 0.48$ in 2015-2017. The effort distribution in the open areas was assumed proportional to projected catch rates.

A total of n=1000 projection runs were performed in this example with stochastic initial conditions, recruitment, and natural mortality. Example result indicate that projected mean biomass in both regions would increase modestly from 2013-2016 (Figure B9.1). Fishing mortality is projected to increase in the Mid-Atlantic, primarily due to reopening of the Elephant Trunk and Hudson Canyon South rotational areas. Fishing mortality is expected to be fairly steady and low on Georges Bank. Landings are expected to rise from about 17,000 mt in 2014 to 23,000 mt in 2017, due to reopening rotational areas. While there is some uncertainty in projected biomass, fishing mortality and landings (Figure B9.2), the example projections indicate almost no chance of either overfishing or the stock becoming overfished in the near future under the assumed management conditions. Results from the SAMS model include projected biomass for each management area as well as for the Georges Bank and Mid-Atlantic and combined areas (Figure B9.3).

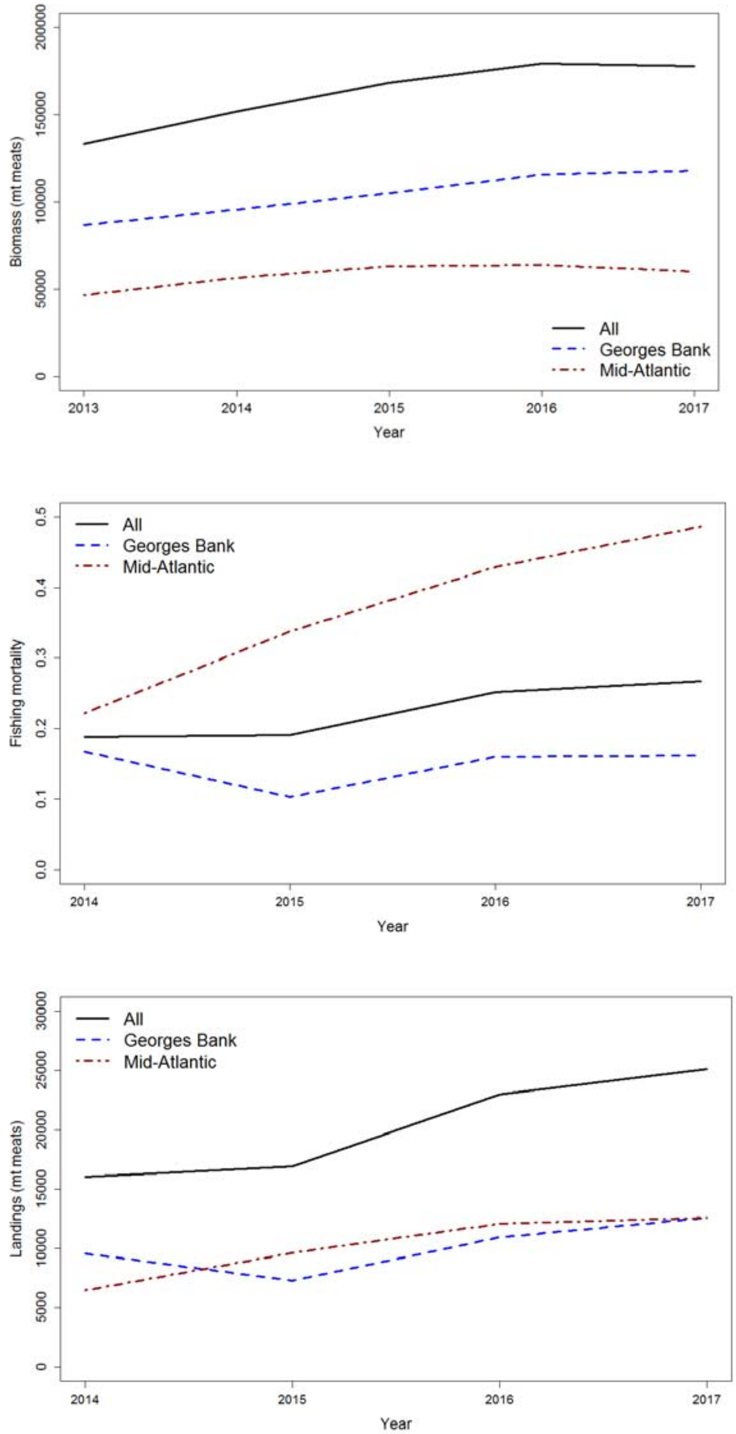


Figure B9.1. Mean projected biomass (top), fishing mortality (middle), and landings (bottom) for sea scallops in the Georges Bank, Mid-Atlantic and combined regions based on an example projection analysis with the SAMS model.

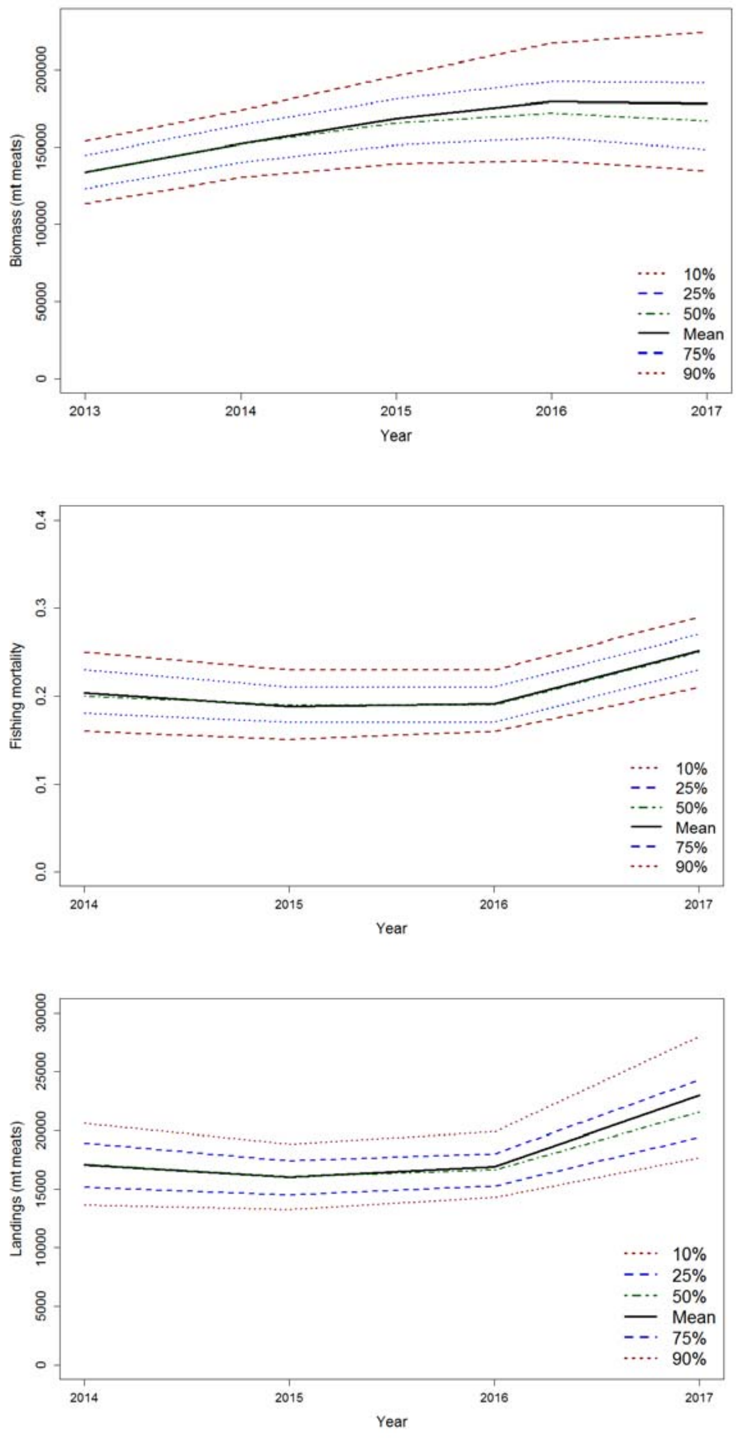


Figure B9.2. Mean and 10th, 25th, 50th, 75th, and 90th percentiles of projected total biomass (top), fishing mortality (middle) and landings (bottom).

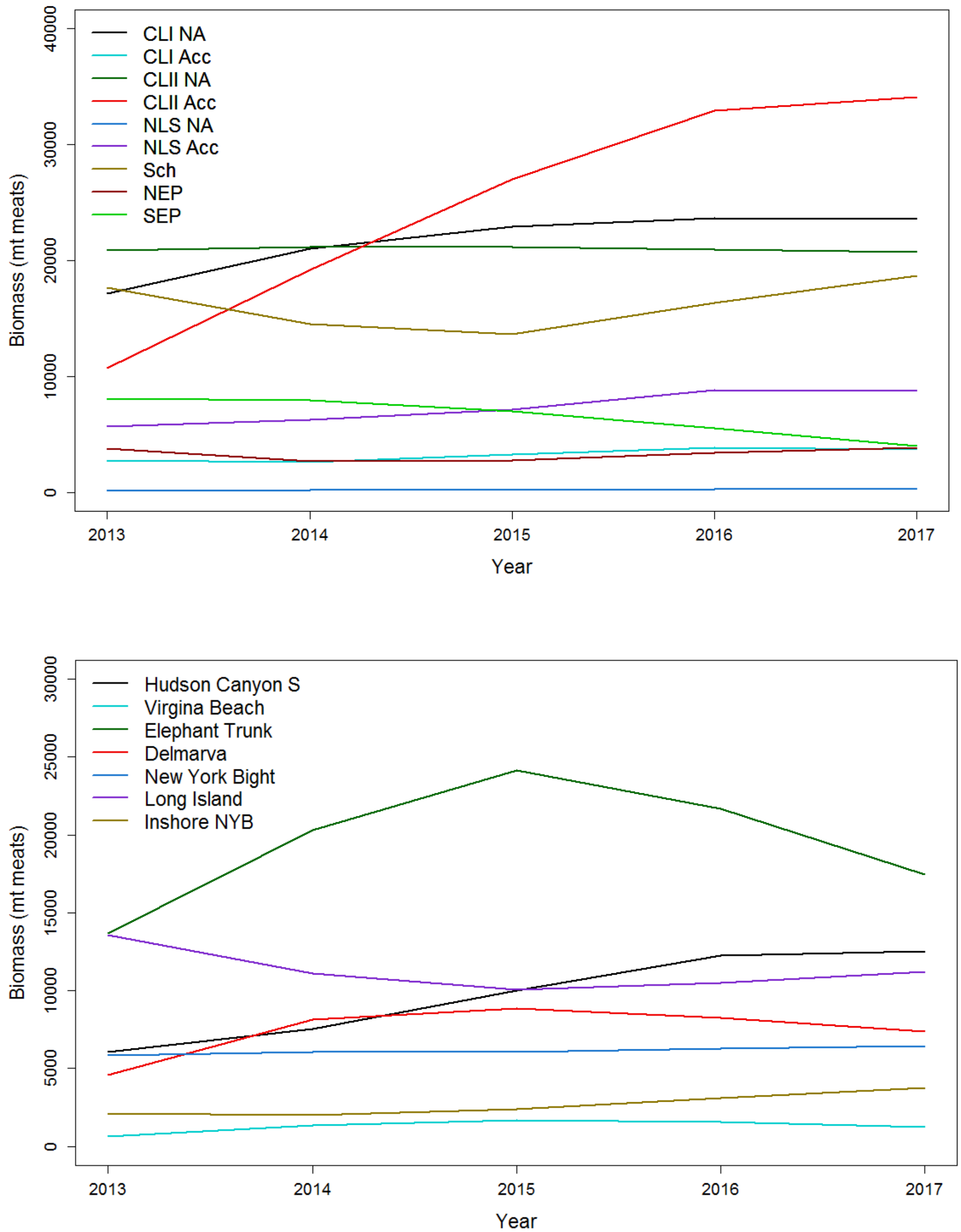


Figure B9.3 Mean projected biomass by subarea in the Georges Bank (top) and Mid-Atlantic (bottom) regions.

B10 – ENVIRONMENTAL EFFECTS ON RECRUITMENT (TOR 3)

Two potential environmental drivers of recruitment were explored: food supply (phytoplankton), and the abundance of a major predator of small scallops, the sea star *Astropecten americanus*. A tentative relationship was found between chlorophyll and scallop recruitment in the Mid-Atlantic Bight. Negative relationships were found between the spatio-temporal abundance of *A. americanus* and scallop recruitment. Both these topics are discussed in Appendix B8.

B11 - RESEARCH RECOMMENDATIONS (TOR 8)

<u>Progress on recommendations from SARC-50 (NEFSC 2010)</u>

1. Look into a way to fit discarded scallops, which have a different length frequency from the rest of the population, into the model. *No progress.*
2. Evaluate the effect of the four-inch rings on incidental mortality. Now that a larger fraction of small scallops are traveling through the mesh, has incidental mortality increased or are the scallops relatively unscathed? *Incidental mortality calculations were improved for this assessment to account for fishery selectivity. Several field projects were funded in 2014 to investigate the extent of incidental mortality from the currently configured fishing gear.*
3. Consider finding a better way to express the variation in the HabCam abundance data (the data were kriged for this assessment, and the variance was calculated by summing the variance of each of the kriged grids). *Two-stage GAM/Kriging models and stratified mean methods were introduced in this assessment, and several methods for calculating variance were investigated and compared in this assessment by simulation and analysis of actual data.*
4. Look at the historical patterns of the "whole stock"; how the spatial patterns of scallops and the fishery have changed over time. *These topics are handled in the description of survey and fishery data to the extent they are relevant.*
5. Estimate incidental mortality by running HabCam or an AUV along dredge tracks. *Several projects were funded this year to do work along these lines.*
6. Effort should be made to make sure the survey dredge is fitted with a camera at some point during the survey to record the movements of the dredge. This will help answer some questions about when the dredge starts and stops fishing, and the determination of tow times. *Five survey dredge tows were conducted with a camera mounted to the dredge that allowed improved interpretation of dredge sensor data.*
7. Seasonal patterns in scallop shell growth need to be analyzed and this data incorporated into the model. *No progress; the assessment team did not feel this is a high priority.*
8. Stock-recruit relationships should be calculated for various sub-sections of the stock, smaller areas than just MAB and GBK to look for possible patterns or relationships.
Appendix B8 examined the relationship between recruitment in the southern Mid-Atlantic and biomass in the entire stock.
9. Further refine the estimate of the extent of scallop habitat relative to that of the survey. *New VIMS dredge and HabCam and SMAST optical surveys were used to identify stock boundaries and improve understanding of the relationship between the dredge survey and stock areas.*
10. Age archived scallop shells from the 1980s and 1990s. *Archived shells from 1988 and 1993 were used to estimate growth matrices to represent growth when fishing mortality was high in the CASA models. However, additional years should be analyzed as described in a new research recommendation.*
11. Continue to look at patterns of seasonality in weight of the meats and gonads, and timing of spawning. *Annual meat weight anomalies used to adjust mean body weight of individual scallops in the fishery and to compute catch numbers were substantially improved. Shell height-meat weight relationships based on survey data were updated.*

New recommendations

The Invertebrate Subcommittee identified the following research topics while preparing this assessment. The topics listed below are all considered worthwhile and are not listed in order of priority.

1. Investigate methods for better survey coordination between the various survey programs.
2. Evaluate effects of uncertainty in identifying dead scallops in optical surveys and improve procedures for identifying dead scallops.
3. Collect data to refine estimates of incidental mortality. Analytical procedures were improved this assessment but further progress awaits collection of more data.
4. Improve training of annotators used in optical surveys to identify and count specimens. For example, develop and consistently apply criteria for identifying inexact shell height measurements. Formalize QA/QC procedures including revaluation of annotator accuracy. Develop and maintain reference images for training and testing.
5. Continue work to improve and simplify survey design and analytical procedures for HabCam. Ideally, procedures might be automated to the extent possible and integrated into routine survey operations.
6. Quantify and improve accuracy of SAMS projection models used to specify harvest levels. Recent projections appear to overestimate stock size to some extent.
7. Reduce uncertainty about stock size estimates from surveys and the CASA model. In particular, continue work on density dependent natural mortality for small scallops in stock assessment, reference point and projection models.
8. Collect additional biological data on a regional basis including growth increments from shells collected during historical dredge surveys, seasonality of spawning based on observer data, natural mortality on large scallops due to disease and senescence, and size-specific reproductive output.
9. Refine models that predict scallop recruitment based on chlorophyll and predator data in order to improve estimates from stock assessment and projection models. Investigate statistical approaches to estimating year class strength directly from survey data.
10. Investigate and quantify the utility of multiple scallop surveys.

B. Sea Scallop Assessment Report Appendixes

Appendix B1 - Invertebrate Subcommittee meetings and participants

Appendix B2 - Sea Scallop Discard Estimates

Appendix B3 - Shell Height Meat Weight Relationships

Appendix B4 - Estimation of Dredge efficiency from paired dredge HabCam observations

Appendix B5 - Empirical Assessment

Appendix B6 - NEFSC HabCam survey for sea scallops: survey design, implementation, and data analysis

Appendix B7 - Assessment of the sea scallop resource in the Northern Gulf of Maine management area

Appendix B8 - Relationships between chlorophyll and scallop recruitment potentially useful for stock projections and assessment modeling

Appendix B9 - Technical documentation for the CASA length structured stock assessment model used in the SARC 59 sea scallop stock assessment

Appendix B10 – Forecasting methodology (SAMS Model)

Appendix B1. Invertebrate Subcommittee meetings and participants

The Invertebrate Subcommittee met March 17-21, April 21-25, May 27-30, June 6, June 18 and June 23 during 2014 while preparing the SARC-59 stock assessment for Atlantic sea scallops. Meetings during March-May were held in the Stephen H. Clark Conference Room at the Northeast Fisheries Science Center in Woods Hole, MA with some participation by video conference. Meetings in June were exclusively by video conference. The following members participated in one or more meetings.

Larry Jacobson, NEFSC, chair
Dvora Hart, NEFSC, Assessment Team Lead
Burton Shank, NEFSC
Jia-Han Chang, NEFSC
Jiashen Tang, NEFSC
Toni Chute, NEFSC
Vic Nordahl, NEFSC
Chris Legault, NEFSC
Dan Hennen, NEFSC
Mark Terciero, NEFSC
Kevin Friedland, NEFSC
Paul Rago, NEFSC
Stephen Smith, DFO, Canada
Mary Beth Tooley, NEFMC
Dierdre Boelke, NEFMC
David Rudders, VIMS
Bill DuPaul, VIMS
Carl Huntsberger, Coonamesset Farm Foundation
Ron Smolowitz, Coonamesset Farm Foundation
Katherine Thompson, Coonamesset Farm Foundation
Daphne Munroe, Rutgers U.
Kevin Stokesbury, SMAST
Gregory DeCelles, SMAST
Susan Inglis, SMAST
Karen Bolles, HabCam Group
Richard Taylor, HabCam Group
Trish DeGraaf, Maine DMR
Kevin Kelly, Maine DMR
Matt Camisa, Massachusetts DMR
Sam Truesdell, University of Maine

Appendix B2. Sea Scallop Discard Estimates

Jessica Blaylock (NEFSC, Woods Hole, MA)

This paper presents discard estimates for Atlantic sea scallop (*Placopecten magellanicus*) for scallop dredge, scallop trawl and otter trawl fleets, calculated using the Standardized Bycatch Reporting Methodology (Wigley et al. 2007). This approach was also used in the previous assessment for this stock; however discard estimates were not included as input in the assessment model (NEFSC 2010).

Methods

Estimates of Atlantic sea scallop discards (mt meats) were derived for seven fleets using Northeast Fishery Observer Program (NEFOP) and Northeast Fishery Science Center (NEFSC) commercial landings (i.e., dealer) data for the 1989 to 2013 time period: Georges Bank and Mid-Atlantic Bight scallop dredge, Mid-Atlantic Bight scallop trawl, Georges Bank and Mid-Atlantic Bight small-mesh otter trawl, and Georges Bank and Mid-Atlantic Bight large-mesh otter trawl. Additionally, sea scallop discard estimates were also derived for scallop dredge fleets at a finer stratification level using NEFOP and Vessel Trip Report (VTR) data for the 1994 to 2013 time period. This analysis considered the two scallop dredge fleets above as four fleets: Georges Bank open and closed scallop dredge, and Mid-Atlantic Bight open and closed scallop dredge,

A broad stratification scheme was used with trips partitioned into fleets using the following four classification variables: calendar quarter, gear type, area fished, and mesh. Trips were not partitioned by trip category ('limited' versus 'general', for scallop dredge and scallop trawl) due to small sample size over the time series. Calendar quarter was based on landed date and used to capture seasonal variations in fishing activity. Gear type was based on Northeast gear codes (scallop dredge: negear 132; scallop trawl: negear 052; otter trawl: negear 050). Trips for which gear was unknown were excluded. Two broad geographical regions are defined for area fished based on statistical area: areas 520-562 constituted the Georges Bank (GBK) area, and areas 600 and above constituted the Mid-Atlantic Bight (MAB) area. Two mesh size groups were formed for otter trawl: small (mesh less than 5.5 inches) and large (5.5 inch mesh and greater). The additional analysis considering scallop dredge at a finer scale included access area as another classification variable. Here, two access area categories were used: ‘open’ and ‘closed’, where ‘closed’ includes all trips fishing in one of the scallop access areas (Closed Area I, Closed Area II and Nantucket Lightship in the GBK region; Hudson Canyon, Virginia Beach, Elephant Trunk, and Delmarva in the MAB region). Observer trips were assigned to the access area category based on program code, and VTR trips were assigned based on latitude and longitude.

Discards were estimated using a combined d/k_{all} ratio estimator (Cochran 1963), where d is discarded pounds of sea scallops and k_{all} is kept pounds of all species, calculated from NEFOP data. Discard weight was derived by multiplying the d/ k_{all} ratio of each fleet by the corresponding dealer or VTR landings (Wigley et al. 2007). Coefficients of variation (CV) were calculated as the ratio of the standard error of the discards divided by the discards.

In cases where limited observer data were available (i.e. two or less observed trips in a calendar quarter), an imputation approach was used to 'fill in' the missing (or incomplete) information using data from adjoining strata. In this imputation procedure, the temporal stratification (i.e., calendar quarter) was relaxed to entire year, recognizing that seasonal variations may occur that will thus not be accounted for. Numbers of annual observed trips by fleet are summarized in Tables 1 and 2.

To evaluate the proportion of estimated sea scallop discards to landings, the sum of the current discard estimates for scallop dredge was compared to the sum of estimated landings from Georges Bank, Southern New England, and Mid-Atlantic Bight for the 1992 to 2013 time period.

Results and Discussion

Annual Atlantic sea scallop discard estimates by fleet are presented in Tables 1, 2, and 3. Tables 1A-1D show estimates for the seven fleets without access area classification: Georges Bank and Mid-Atlantic Bight scallop dredge, Mid-Atlantic Bight scallop trawl, Georges Bank and Mid-Atlantic Bight small-mesh otter trawl, and Georges Bank and Mid-Atlantic Bight large-mesh otter trawl. Tables 2A-2B present discard estimates for the scallop dredge fleets at a finer scale that includes access area as a classification variable.

This analysis indicates that during the 1989 to 2013 time period, sea scallops were primarily discarded in the scallop dredge fleets (Tables 1A-1D, Table 3, Figure 1). For 2013, estimated discards from the Georges Bank and Mid-Atlantic Bight scallop dredge were 299 and 128 mt meats, respectively. Discard estimates for the other five fleets for the same year ranged from less than 1 mt meats (Georges Bank small-mesh otter trawl) to 10 mt meats (Mid-Atlantic Bight scallop trawl).

Discard estimates for scallop dredge at the access area classification level (Tables 2A-2B) suggest a higher discarding rate in the 'open' category fleets. For 2013, estimated discards from the Georges Bank open and closed scallop dredge fleets were 370 and 8 mt meats, respectively. Estimated discards from the Mid-Atlantic Bight open scallop dredge fleet were 46 mt meats; discards could not be estimated for 2013 for the Mid-Atlantic Bight closed scallop dredge fleet due to VTR trip misclassification.

The discard estimation presented here used a broad stratification approach. In addition, there are inherent limitations in the use of VTR data for trip assignment to the 'access area' category because of missing or inaccurate position data. Consequently, the discard estimates from scallop dredge at the access area classification level should be considered as preliminary.

Current estimates of discards and landings from scallop dredge fleets for 1994 to 2013 are presented in Figure 2. Total catch (discards plus landings) averaged 6,814 mt meats between 1993 and 1998. Catch increased in the following six years to peak at 31,435 mt meats in 2004, and averaged 26,560 mt meats from 2005 to 2012. Total catch in 2013 was 18,516 mt meats. Discards generally represent a small portion of total catch, with discard-to-landing ratios ranging from 0.010 in 1997 and 1998 to 0.1233 in 2000.

These results represent estimated sea scallop discards and landings in weight (mt meats). It is likely that discard-to-landing ratios of numbers would be higher because of the different size distribution of discarded scallops compared to that of landed scallops.

Acknowledgements

I wish to thank all the NEFOP observers for their diligent efforts to collect the discard information used in this analysis. Additionally, I would like to thank Toni Chute for her assistance with the classification of VTR trips to access area categories.

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Wigley, S.E., P.J. Rago, K.A. Sosebee, and D.L. Palka. 2007. The Analytic Component to the Standardized Bycatch Reporting Methodology Omnibus Amendment: Sampling Design, and Estimation of Precision and Accuracy (2nd Edition). US Dep. Commer., Northeast Fish. Sci. Cent. Ref. Doc. 07-09; 156 p. Available online: http://www.nefsc.noaa.gov/nefsc/publications/crd/crd0709/index.htm

Table 1A. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) for the Georges Bank (GBK) scallop dredge and Mid-Atlantic Bight (MAB) scallop dredge fleets, 1989-2013. Discards were not estimated prior to 1992 due to small sample size.

GBK scallop dredge				MAB scallop dredge			
YEAR	Trips	Discards (mt meats)	CV	YEAR	Trips	Discards (mt meats)	CV
1989				1989			
1990				1990			
1991	1			1991	1		
1992*	11	464	0.48	1992*	7	121	0.00
1993*	12	345	0.32	1993*	10	12	0.80
1994*	7	3	0.89	1994	16	576	0.54
1995*	6	22	0.62	1995*	20	322	0.28
1996	15	116	0.36	1996	23	24	0.71
1997*	11	46	0.73	1997*	18	8	1.14
1998*	9	4	0.57	1998*	16	48	0.66
1999*	63	141	0.28	1999*	8	8	0.56
2000*	228	989	0.09	2000	28	779	0.33
2001*	18	529	0.17	2001*	88	1,955	0.11
2002*	11	105	0.58	2002	87	1,894	0.13
2003*	14	328	0.58	2003	108	2,225	0.10
2004*	46	58	0.20	2004	235	2,446	0.09
2005	107	228	0.27	2005	220	357	0.19
2006	135	347	0.20	2006*	93	78	0.49
2007	180	231	0.21	2007	177	260	0.20
2008	216	334	0.14	2008	425	414	0.15
2009	81	380	0.26	2009	408	923	0.12
2010	98	668	0.18	2010	238	688	0.21
2011	141	668	0.18	2011	251	482	0.14
2012	222	603	0.11	2012	201	237	0.12
2013	269	299	0.14	2013	182	128	0.22

** Imputed data were used for discard estimation for these years.*

Table 1B. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) for the Mid-Atlantic Bight (MAB) scallop trawl fleet, 1989-2013. Discards were not estimated prior to 2004 due to small sample size.

MAB scallop trawl			
YEAR	Trips	Discards (mt meats)	CV
1989			
1990			
1991			
1992			
1993			
1994			
1995			
1996			
1997			
1998			
1999			
2000			
2001	4		
2002	1		
2003			
2004*	44	99	0.25
2005	137	61	0.13
2006*	30	150	0.33
2007	34	17	0.59
2008*	38	6	0.58
2009*	8	49	1.59
2010*	29	12	0.33
2011*	10	12	0.78
2012*	19	<1	0.75
2013*	20	10	0.35

** Imputed data were used for discard estimation for these years.*

Table 1C. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) for the Georges Bank (GBK) small-mesh otter trawl, and Mid-Atlantic Bight (MAB) small-mesh otter trawl fleets, 1989-2013.

GBK small-mesh otter trawl				MAB small-mesh otter trawl			
YEAR	Trips	Discards (mt meats)	CV	YEAR	Trips	Discards (mt meats)	CV
1989	65	2	0.53	1989	34	213	0.39
1990	31	<1	1.22	1990	47	8	0.44
1991	68	<1	0.80	1991	78	11	2.05
1992	42	<1	0.68	1992	47	6	0.53
1993	25	<1	0.57	1993*	16	8	0.81
1994*	18	7	1.88	1994*	15	29	0.78
1995*	11	<1	1.26	1995	63	71	0.23
1996*	10	0	0.00	1996	80	14	1.70
1997*	20	<1	0.87	1997*	48	1	2.76
1998*	6	<1	1.39	1998*	32	4	1.35
1999*	8	<1	2.62	1999	35	12	1.65
2000*	17	<1	0.49	2000	39	2	0.94
2001*	15	<1	0.64	2001	55	<1	8.75
2002*	33	<1	0.82	2002	32	68	0.34
2003	55	<1	1.11	2003	74	17	0.80
2004	109	2	0.96	2004	257	5	0.42
2005	194	<1	0.47	2005	172	4	0.32
2006	62	<1	0.56	2006	151	13	2.63
2007	60	<1	1.44	2007	218	5	0.56
2008	50	<1	0.49	2008	152	8	0.42
2009	199	<1	0.50	2009	286	23	0.52
2010	217	<1	0.54	2010	361	16	0.48
2011	168	<1	0.49	2011	365	5	0.33
2012	130	<1	0.83	2012	226	3	0.61
2013	186	<1	0.45	2013	395	5	0.35

** Imputed data were used for discard estimation for these years.*

Table 1D. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) for the Georges Bank (GBK) large-mesh otter trawl, and Mid-Atlantic Bight (MAB) large-mesh otter trawl fleets, 1989-2013. Discards were not estimated for MAB large-mesh otter trawl prior to 1992 due to small sample size.

GBK large-mesh otter trawl				MAB large-mesh otter trawl			
		Discards				Discards	
YEAR	Trips	(mt meats)	CV	YEAR	Trips	(mt meats)	CV
1989	27	1	0.88	1989	4		
1990	33	1	0.72	1990			
1991	34	4	0.54	1991	4		
1992	35	<1	1.10	1992*	14	4	0.40
1993	35	<1	1.30	1993*	12	3	1.54
1994	36	<1	1.21	1994*	21	99	0.53
1995	61	<1	0.36	1995	55	102	0.83
1996	38	<1	0.69	1996*	18	<1	0.62
1997	26	<1	1.00	1997*	9	1	0.62
1998*	10	<1	0.89	1998*	13	1	0.69
1999	20	<1	2.48	1999*	8	94	1.16
2000	30	2	0.66	2000*	26	32	0.57
2001	52	1	0.82	2001*	50	13	0.48
2002	83	2	0.61	2002*	39	8	2.36
2003	163	3	0.77	2003*	16	<1	2.26
2004	316	42	0.35	2004	109	9	0.43
2005	959	9	0.18	2005	93	1	0.94
2006	462	30	0.37	2006	71	3	2.39
2007	465	5	0.25	2007	160	12	0.59
2008	563	6	0.21	2008	132	29	0.88
2009	536	9	0.22	2009	167	19	0.22
2010	526	4	0.23	2010	274	9	0.73
2011	782	6	0.17	2011	253	9	1.00
2012	599	6	0.32	2012	169	4	0.78
2013	593	6	0.20	2013	251	7	0.53

** Imputed data were used for discard estimation for these years.*

Table 2A. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) by the Georges Bank (GBK) open scallop dredge and GBK closed scallop dredge fleets, 1994-2013. Discards were not estimated for the GBK open scallop dredge fleet in 2000 and 2001 due to small sample size.

GBK open scallop dredge				GBK closed scallop dredge			
YEAR	Trips	Discards (mt meats)	CV	YEAR	Trips	Discards (mt meats)	CV
1994*	7	2	0.82	1994	n/a		
1995*	6	23	0.63	1995	n/a		
1996	15	103	0.37	1996	n/a		
1997*	11	41	0.70	1997	n/a		
1998*	9	4	0.57	1998	n/a		
1999*	48	97	0.39	1999*	15	53	0.26
2000	2			2000	226	246	0.03
2001	2			2001	16	26	0.15
2002*	11	99	0.57	2002	n/a		
2003*	14	324	0.58	2003	n/a		
2004*	16	39	0.29	2004	30	25	0.19
2005	41	371	0.36	2005	66	40	0.27
2006*	56	783	0.25	2006	79	41	0.26
2007	53	194	0.30	2007	127	40	0.26
2008	73	202	0.23	2008	140	53	0.12
2009	58	295	0.33	2009*	23	24	0.30
2010	44	576	0.36	2010*	54	117	0.18
2011*	68	603	0.24	2011	71	84	0.20
2012	101	981	0.15	2012	119	48	0.11
2013	202	370	0.16	2013	30	8	0.07

** Imputed data were used for discard estimation for these years.*
n/a: not applicable

Table 2B. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) by the Mid-Atlantic Bight (MAB) open scallop dredge and MAB closed scallop dredge fleets, 1994-2013. Discards were not estimated for the MAB open scallop dredge fleet in 2001 due to small sample size.

MAB open scallop dredge				MAB closed scallop dredge			
		Discards				Discards	
YEAR	Trips	(mt meats)	CV	YEAR	Trips	(mt meats)	CV
1994	16	276	0.59	1994	n/a		
1995*	20	341	0.28	1995	n/a		
1996	23	22	0.72	1996	n/a		
1997*	18	8	1.15	1997	n/a		
1998*	16	42	0.66	1998	n/a		
1999*	8	7	0.56	1999	n/a		
2000	28	749	0.33	2000	n/a		
2001	3			2001	85	301	0.09
2002*	13	1,446	0.19	2002	74	151	0.11
2003	62	2,253	0.14	2003	46	120	0.12
2004	143	1,869	0.13	2004	92	510	0.10
2005	166	368	0.29	2005	54	39	0.21
2006*	87	71	0.39	2006*	6	3	0.49
2007	84	65	0.41	2007	93	63	0.22
2008	89	215	0.54	2008	336	97	0.14
2009	118	597	0.15	2009	290	219	0.13
2010	130	583	0.30	2010	108	94	0.20
2011	145	489	0.20	2011	45	22	0.22
2012	100	143	0.20	2012^			
2013	137	46	0.25	2013^			

** Imputed data were used for discard estimation for these years.*
^ no discard estimation because of VTR missclassification
n/a: not applicable

Table 3. Summary of sea scallop discard estimates (mt meats) from Table 1 by region, 1989-2013.

Georges Bank (GBK)					Mid-Atlantic Bight (MAB)					
YEAR	scallop dredge	small-mesh otter trawl	large-mesh otter trawl	Total	YEAR	scallop dredge	scallop trawl	small-mesh otter trawl	large-mesh otter trawl	Total
1989	*	2	1	4	1989	*	*	213	*	213
1990	*	<1	1	1	1990	*	*	8	*	8
1991	*	<1	4	5	1991	*	*	11	*	11
1992	464	<1	<1	465	1992	121	*	6	4	131
1993	345	<1	<1	346	1993	12	*	8	3	22
1994	3	7	<1	10	1994	576	*	29	99	703
1995	22	<1	<1	23	1995	322	*	71	102	495
1996	116	0	<1	116	1996	24	*	14	<1	38
1997	46	<1	<1	46	1997	8	*	1	1	11
1998	4	<1	<1	4	1998	48	*	4	1	53
1999	141	<1	<1	142	1999	8	*	12	94	114
2000	989	<1	2	991	2000	779	*	2	32	813
2001	529	<1	1	531	2001	1,955	*	<1	13	1,969
2002	105	<1	2	107	2002	1,894	*	68	8	1,970
2003	328	<1	3	332	2003	2,225	*	17	<1	2,244
2004	58	2	42	102	2004	2,446	99	5	9	2,559
2005	228	<1	9	238	2005	357	61	4	1	424
2006	347	<1	30	378	2006	78	150	13	3	244
2007	231	<1	5	236	2007	260	17	5	12	294
2008	334	<1	6	341	2008	414	6	8	29	457
2009	380	<1	9	389	2009	923	49	23	19	1,013
2010	668	<1	4	672	2010	688	12	16	9	724
2011	668	<1	6	675	2011	482	12	5	9	508
2012	603	<1	6	610	2012	237	<1	3	4	245
2013	299	<1	6	306	2013	128	10	5	7	150

** No discard estimate due to small sample size.*

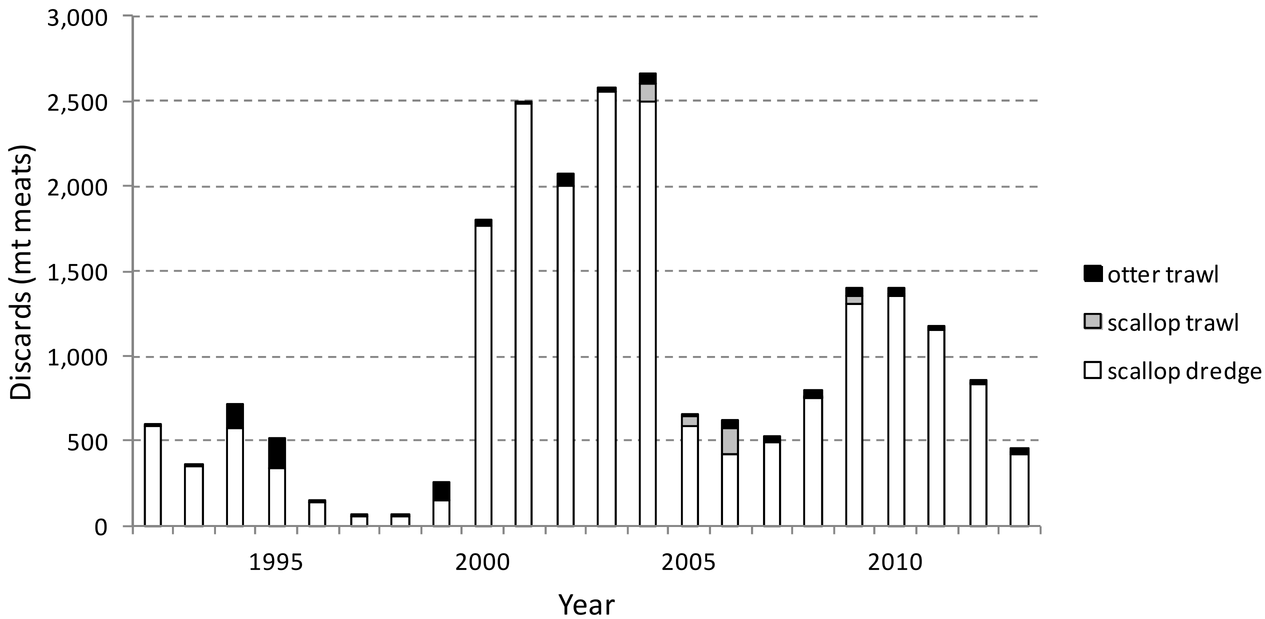

Figure 1. Sea scallop discard estimates (mt meats) from trips using scallop dredge, scallop trawl, and otter trawl gear presented in Table 1, 1992-2013. Discards from scallop trawl were not estimated prior to 2004 due to small sample size.

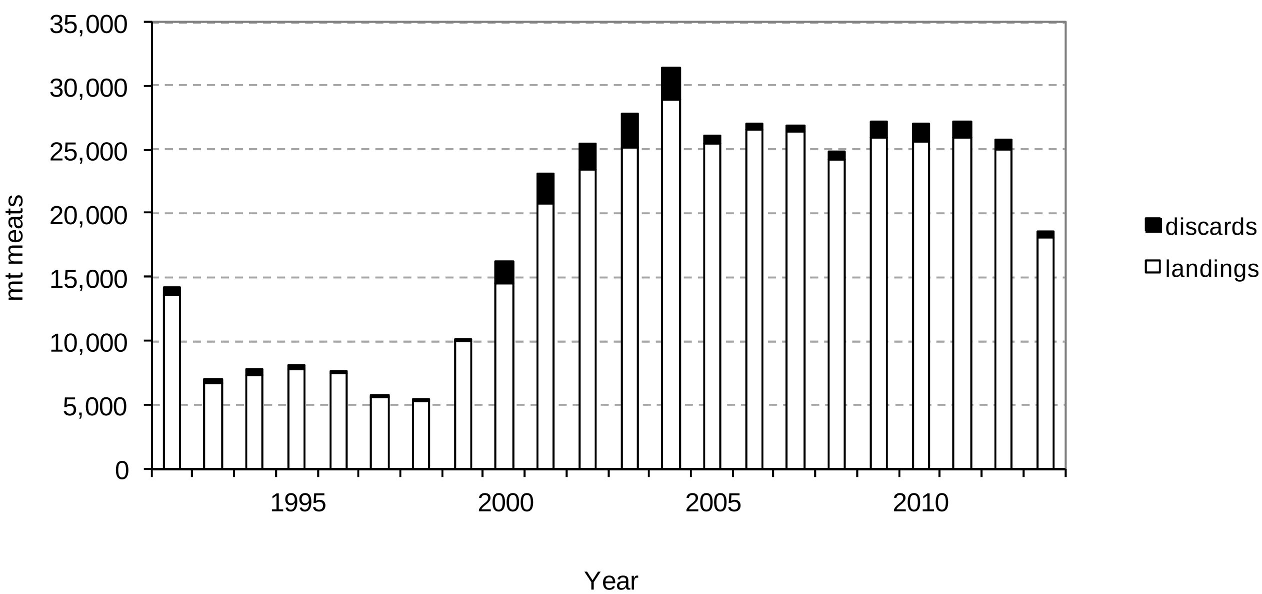

Figure 2. Estimated scallop landings and current estimated sea scallop discards from scallop dredge fleets (mt meats), 1992-2013.

Appendix B3. Shell Height Meat Weight Relationships

Dan Hennen, NEFSC, Woods Hole, MA

1 Methods

Sea scallops (averaging about 6 per station) were selected for analysis on roughly half of all NEFSC survey stations from 2004 to 2013. The scallops were measured to the nearest millimeter, carefully shucked, excess water was removed from the meat, and the meat was weighed to the nearest gram.

Preliminary analysis indicated a residual pattern for those scallops with shell heights less than 70 mm. The small weights of these scallops (1-3 g) combined with the fact that meat weight could only be measured to the nearest gram resulted substantial measurement error. For this reason, the analysis was restricted to scallops that are at least 70 mm shell height (Figure A1).

A generalized linear mixed model (GLMM) with a log link was used to predict meat weight using shell height, depth, density, latitude, and subarea (a finer scale regional division within each broad region). The GLMM used the gamma likelihood with a log link which is appropriate for data (such as these) with ”constant CV” error (McCullagh and Nelder [1989]). This method avoids log-transforming the response variable (meat weight) which can lead to biased estimates when the results are back-transformed. The best model was chosen by AIC (Tables 1-5; Burnham and Anderson [2002]). The grouping variable for the random effects was a combination of survey station number and the year in which the survey took place. Survey stations are chosen randomly (though stratified to fit NEFSC survey design specifications) and survey stations numbers are assigned sequentially so that a survey station number in one year does not have any particular relationship to the same station number in the next year. Thus, a grouping variable based on a combination of survey station number and year incorporates random variation in the data that is due to both time (year) and fine scale spatial differences (station number).

Several analyses using simplified versions of the best model were employed to explore the effects of year, subarea, and fishing regulations.

All data analysis was conducted using the R statistical program (v2.13.2).

1.1 Seasonal variation and commercial meat anomalies

The NMFS Observer program provided meat weight estimates from commercial catches that occurred throughout the year. These meat weights are based on meats that are shucked by fishermen. Meats from the observer program are not weighed individually. They are packed into a graduated cylinder and a volume for a sample (typically 100 scallops) is recorded. The meat weight for a sample was calculated using a density estimate of 1.05 $\frac{g}{ml^3}$ (Caddy and Radley-Walters [1972]; Smolowitz et al. [1989]). These “observed” meat weights are therefore an average weight for all the meats in the cylinder, not a direct observation of the weight of a meat. The observer program does generate approximate shell heights for individual scallops, though they are binned by 5 mm increments. Therefore predicted meat weights can be generated for each shell height represented in the sample. Predicted meat weights were calculated using the best model (by AIC) from the analysis of survey meat weights described above.

It was noted this year that in many cases the number of shells measured was > 100. Because there were only 100 scallop meats packed into the cylinder and there is no way to determine which of the shells were associated with the meats in the cylinder, all observations in which the shell heights exceeded 100 in number were excluded from this analysis. This correction reduced the sample size by approximately 52%, but reduced the error in predicted meat weights considerably (compare Figure 9 to Figure 10).

The best model was applied to predict meat weights for observer samples based on shell heights, latitude and longitude recorded for each sample during 2001 2013. Depth outliers were excluded by restricting maximum depths in the observed hauls from each subarea to the maximum depths observed in the survey for that subarea.

Predicted meat weights for each month were compared to the (observed) density derived meat weights for each month by $\frac{\text{pred.}-\text{obs.}}{\text{pred.}}$ (Figure 10). The median of these these ratios by month are referred to as the monthly meat weight "anomaly". A positive anomaly indicates that the observed meat weight was greater than the expected meat weight, while a negative anomaly indicates the opposite is true. Annuual meat weight anomalies for use in the CASA stock assessment model were computed by average the monthly values within a year using the landings during each month as weights.

2 Results and Discussion

In general, the observed meat weights (from observed volumes) should be less than the survey-based, predicted meat weights (a negative anomaly) because the commercially shucked scallops leave some meat on the shell, and because the surveys occur in late spring or summer (depending on the year), a time of typically high meat weight. The pattern in the anomaly calculated for MAB roughly follows this pattern in that the anomaly is negative in all months excluding April through July, a period that overlaps the survey (Figure 12). On Georges Bank, however, there were months of the year where the observed scallop meats were almost 15% heavier than the predicted meats, resulting in a positive anomaly (Figure 13). The positive anomaly appears in February through July. It is clear from examination of Figure 13 that either observed meat weights were heavier than expected and/or predicted meat weights lighter between January and May since 2009. In 2009, the timing of the survey was shifted to earlier in the year. Predicted meat weights have increased for scallops greater than about 130 mm since the last assessment (Figure 8). Therefore observed meat weights must have increased. In fact, observed meat weights have both increased and stabilized dramatically in the years since 2009 (Figure 14). It is possible that this reflects an increase in efficiency among fishers by selecting areas and time periods when meat weights were high. The early months of the year were not as well sampled by observers relative to the summer months and smaller sample sizes may be influencing this pattern as well (Table 6). There is also some indication of a systemic increase in meat weight for the region generally, based on the shell height to meat weight model estimates reflected in Figure 8, but this result is confounded with the shift in the timing of the survey.

The anomalies refine assessment model estimates of the total annual weight of meats removed by the fishing fleet, based on the lengths recorded by port-side samplers. To make the conversion from port-side shell height to meat weight, the median monthly meat weight anomalies were smoothed by a second order polynomial loess function with a span of 0.25 (months). This short smoothing span provided a modest smooth that allowed the data to strongly influence the model fit (Figures A15). The smooth was applied to a duplicated annual cycle (i.e. 24 months were fit, using identical data in each 12 month period) and the middle 12 months were selected and reordered so that January was the first month in the resulting model fit. This manipulation guaranteed that December and January produced linking estimates and minimized edge effects. The smoothed monthly anomalies were then weighted by the landings in each month in each year for which we have landings data (1975 − 2012) and annual median values were calculated.

The annual values were somewhat different from similar values calculated for the last assessment (Figures A16 -A17). The anomalies are generally lower (~ 2%) in the MAB and higher (~ 15%) in the GBK. The difference in the GBK region is due to the large shift in the monthly anomalies between the last assessment and the current one, based primarily on the increase in observed meat weight (Figure 14). The shift in the MAB is relatively minor and is likely attributable to a combination of the various manipulations to the observer data and small changes in the shell height to meat weight model.

3 Literature Cited

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J.F. Caddy and C. Radley-Walters. Estimating count per pound of scallop meats by volumetric measurement. Technical Report 1202, Fish. Res. Brd. Can. Man. Rep., 1972.

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R.J. Smolowitz, F.M. Serchuk, and R.J. Reidman. The use of a volumetric measure for determining sea scallop meat count. NOAA Tech. Mem. F/NER-1, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543-1026, 1989.

Table 1: AIC results from model fits to predict meat weight.

Formula	AIC	BIC	logLik	deviance
sh+d+sh*d+area+(sh+1)	101114.57	101267.34	-50537.28	101074.57
sh+d+lat+clop+area+(sh+1)	101123.48	101283.90	-50540.74	101081.48
sh+d+area+(sh+1)	101129.14	101274.28	-50545.57	101091.14
sh+d+lat+area+(sh+1)	101130.13	101282.90	-50545.06	101090.13
sh+d+clop+sh*d+(sh+1)	101166.05	101234.80	-50574.02	101148.05
sh+d+lat+clop+(sh+1)	101175.50	101244.25	-50578.75	101157.50
sh+d+clop+(sh+1)	101180.69	101241.80	-50582.35	101164.69
sh+d+sh*d+(sh+1)	101187.51	101248.63	-50585.76	101171.51
sh+d+lat+sh*d+(sh+1)	101188.53	101257.28	-50585.26	101170.53
sh+d+(sh+1)	101202.36	101255.83	-50594.18	101188.36
sh+area+(sh+1)	101288.53	101426.03	-50626.26	101252.53
sh+clop+(sh+1)	101359.04	101412.51	-50672.52	101345.04
sh+lat+(sh+1)	101363.62	101417.09	-50674.81	101349.62
d+(sh+1)	103485.29	103531.13	-51736.65	103473.29
sh+d+sh*d+(1)	105482.86	105528.69	-52735.43	105470.86
sh+d+area+(1)	105660.31	105790.17	-52813.16	105626.31
sh+d+clop+(1)	105750.75	105796.58	-52869.37	105738.75
sh+d+lat+(1)	105769.06	105814.89	-52878.53	105757.06
sh+d+(1)	105773.59	105811.78	-52881.79	105763.59
sh+area+(1)	105824.38	105946.60	-52896.19	105792.38
sh+clop+(1)	105915.93	105954.12	-52952.96	105905.93
sh+(1)	105923.56	105954.12	-52957.78	105915.56
sh+lat+(1)	105925.11	105963.31	-52957.56	105915.11
d+(1)	119777.65	119808.20	-59884.82	119769.65

Table 2: Results from model fits to predict meat weight. The coefficients estimated are: the intercept(int), ln(shell height) (sh), ln(depth) (d), latitude (lat), an interaction between ln(shell height) and ln(depth)(shXd) and an Identifier which is either a marker for a model with subarea coefficients (see Tables 3 and4) or a coefficient for closed vs. open (clop). Random effects are either on the shell height coefficient and intercept (sh+1) or intercept alone (1). The models are listed in order of increasing AIC (lowest AIC model is in the top row).

formula	int	sh	d	lat	shXd	Identifier
sh+d+sh*d+area+(sh+1)	-16.98(0.013)	4.6(0.021)	1.93(0.018)		-0.48(0.087)	1
sh+d+lat+clop+area+(sh+1)	-6.43(0.016)	2.61(0.022)	-0.38(0.019)	-0.02(0.012)		0.09(0.019)
sh+d+area+(sh+1)	-7.45(0.013)	2.61(0.021)	-0.38(0.018)			2
sh+d+lat+area+(sh+1)	-6.55(0.016)	2.61(0.022)	-0.39(0.019)	-0.02(0.012)		3
sh+d+clop+sh*d+(sh+1)	-17.08(0.006)	4.59(0.021)	1.94(0.016)		-0.48(0.087)	-0.06(0.008)
sh+d+lat+clop+(sh+1)	-8.02(0.006)	2.61(0.021)	-0.38(0.016)	0.01(0.003)		-0.07(0.008)
sh+d+clop+(sh+1)	-7.56(0.006)	2.61(0.021)	-0.36(0.016)			-0.06(0.008)
sh+d+sh*d+(sh+1)	-17.38(0.004)	4.64(0.021)	2.01(0.016)		-0.49(0.087)	
sh+d+lat+sh*d+(sh+1)	-17.56(0.004)	4.64(0.021)	2.01(0.016)	0.005(0.003)	-0.49(0.087)	
sh+d+(sh+1)	-9.09(0.004)	2.61(0.021)	-0.34(0.016)			
sh+area+(sh+1)	-9.07(0.013)	2.61(0.022)				4
sh+clop+(sh+1)	-9.04(0.006)	2.61(0.022)				-0.04(0.008)
sh+lat+(sh+1)	-8.63(0.004)	2.61(0.022)		-0.01(0.003)		
d+(sh+1)	4.96(0.005)		-0.36(0.019)			
sh+d+sh*d+(1)	-28.64(0.004)	6.98(0.015)	4.94(0.017)		-1.1(0.064)	
sh+d+area+(1)	-6.38(0.014)	2.38(0.016)	-0.38(0.019)			5
sh+d+clop+(1)	-6.64(0.006)	2.4(0.016)	-0.34(0.017)			-0.06(0.008)
sh+d+lat+(1)	-7.18(0.004)	2.4(0.016)	-0.34(0.017)	0.01(0.003)		
sh+d+(1)	-6.76(0.004)	2.4(0.016)	-0.32(0.017)			
sh+area+(1)	-7.99(0.014)	2.38(0.016)				6
sh+clop+(1)	-8.02(0.006)	2.39(0.016)				-0.04(0.009)
sh+(1)	-8.05(0.004)	2.39(0.016)				
sh+lat+(1)	-7.91(0.004)	2.39(0.016)		-0.003(0.003)		
d+(1)	4.69(0.007)		-0.31(0.028)			

Table 3: Results from model fits to predict meat weight in MAB subareas.

Identifier	VB	DMV	DMV.VB	ET	HC	NYB
1	-0.13(0.023)	-0.06(0.018)	-0.14(0.028)	-0.17(0.022)	-0.08(0.019)	-0.07(0.019)
2	-0.14(0.023)	-0.06(0.018)	-0.15(0.028)	-0.17(0.022)	-0.08(0.019)	-0.07(0.019)
3	-0.14(0.023)	-0.12(0.041)	-0.22(0.05)	-0.23(0.039)	-0.12(0.031)	-0.11(0.028)
4	-0.06(0.024)	0.04(0.018)	-0.03(0.028)	-0.07(0.022)	0.002(0.02)	0.04(0.019)
5	-0.14(0.024)	-0.05(0.019)	-0.2(0.029)	-0.24(0.023)	-0.11(0.02)	-0.07(0.02)
6	-0.07(0.025)	0.05(0.019)	-0.08(0.029)	-0.13(0.023)	-0.03(0.021)	0.04(0.02)

Table 4: Results from model fits to predict meat weight in GBK subareas.

Identifier	NLS	SCH	CA1	SEP	NEP	CA2
1	0.07(0.021)	-0.13(0.018)	0	-0.07(0.023)	-0.13(0.017)	0.004(0.017)
2	0.07(0.021)	-0.13(0.018)	0	-0.07(0.023)	-0.13(0.017)	0.005(0.017)
3	0.06(0.022)	-0.13(0.018)	0	-0.08(0.024)	-0.12(0.018)	0.008(0.017)
4	0.14(0.021)	-0.07(0.019)	0	-0.08(0.024)	-0.12(0.017)	0.05(0.018)
5	0.08(0.021)	-0.12(0.019)	0	-0.06(0.024)	-0.14(0.018)	0.001(0.018)
6	0.14(0.022)	-0.06(0.02)	0	-0.07(0.025)	-0.12(0.018)	0.04(0.018)

Table 5: Results from model fits to predict meat weight. Predictors are ln(shell height) (sh) ln(depth) (d), region (reg) and open vs. closed to fishing (clop). MAB and open coefficients are shown. GBK and closed are assumed to have coefficients equal to 0.

formula	int	sh	d	reg	clop	AIC	BIC
sh+d+reg+clop+(sh+1)	-7.35(0.012)	2.61(0.03)	-0.4(0.028)	-0.05(0.014)	-0.06(0.013)	101171	101240
sh+d+reg+(sh+1)	-7.46(0.009)	2.61(0.03)	-0.38(0.029)	-0.04(0.014)		101195	101256
sh+reg+clop+(sh+1)	-9.07(0.012)	2.61(0.03)		0.04(0.014)	-0.04(0.014)	101353	101414
sh+reg+(sh+1)	-9.09(3e-04)	2.61(4e-04)		0.04(0.01)		101361	101414

Table 6: Sample sizes for observed meat weights by month in GBK.

month	pre2010	post2009	Total
1	142	82	224
2	86	38	124
3	18	62	80
4	32	88	120
5	84	149	233
6	431	333	764
7	433	404	837
8	356	404	760
9	269	174	443
10	201	151	352
11	249	138	387
12	167	58	225
Total	2468	2081	4549

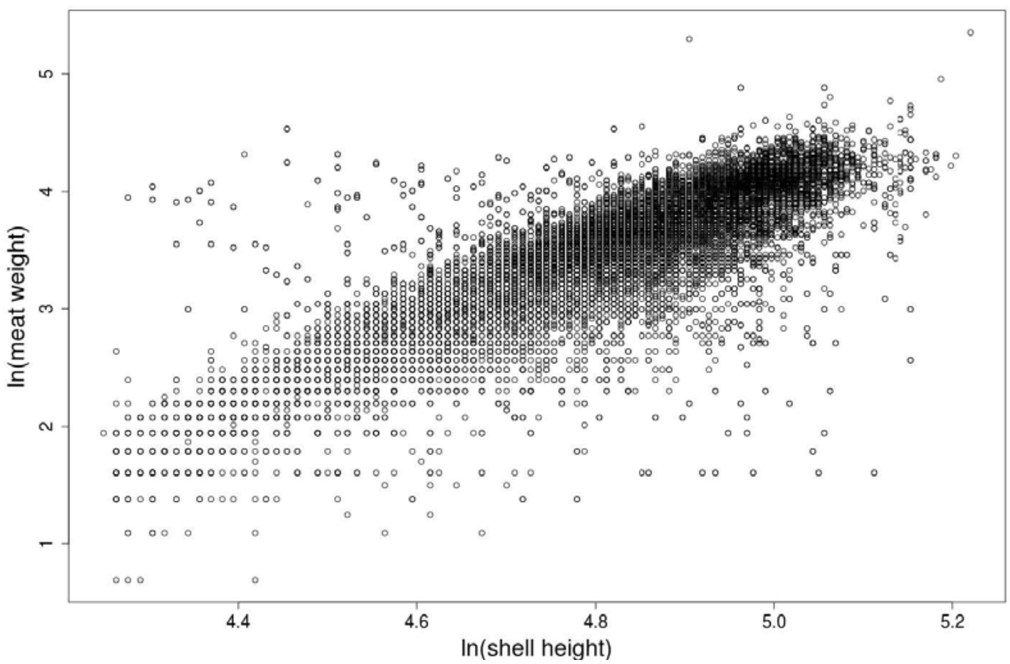

Figure 1: Natural log of shell height against the natural log of meat weights measured on NEFSC scallop surveys between 2003 and 2013.

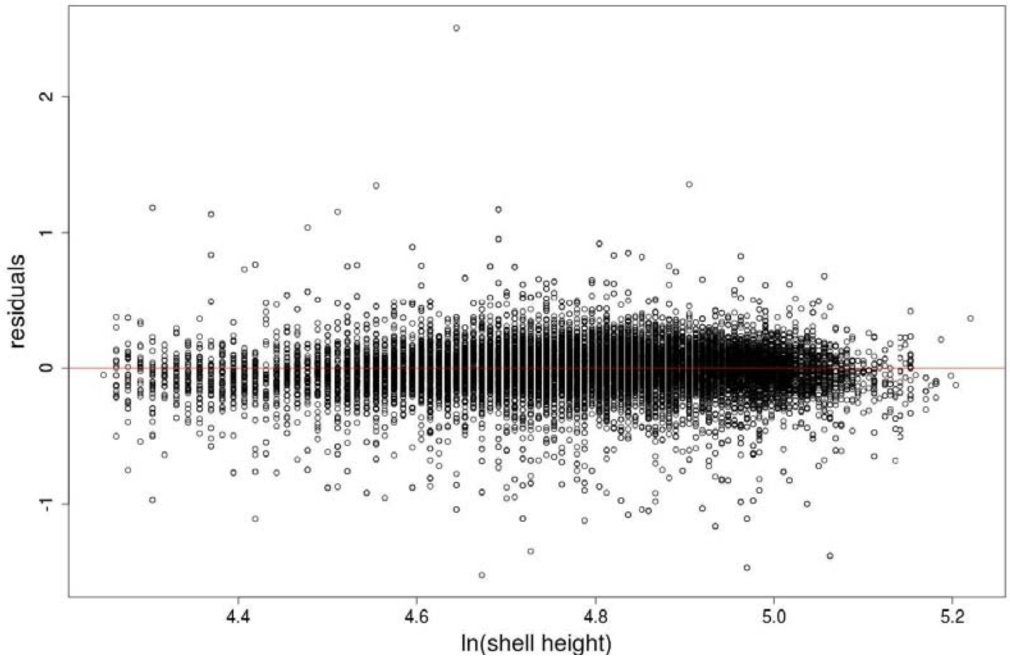

Figure 2: Residuals from the fit of best model predicting meat weight by the natural log of shell height.

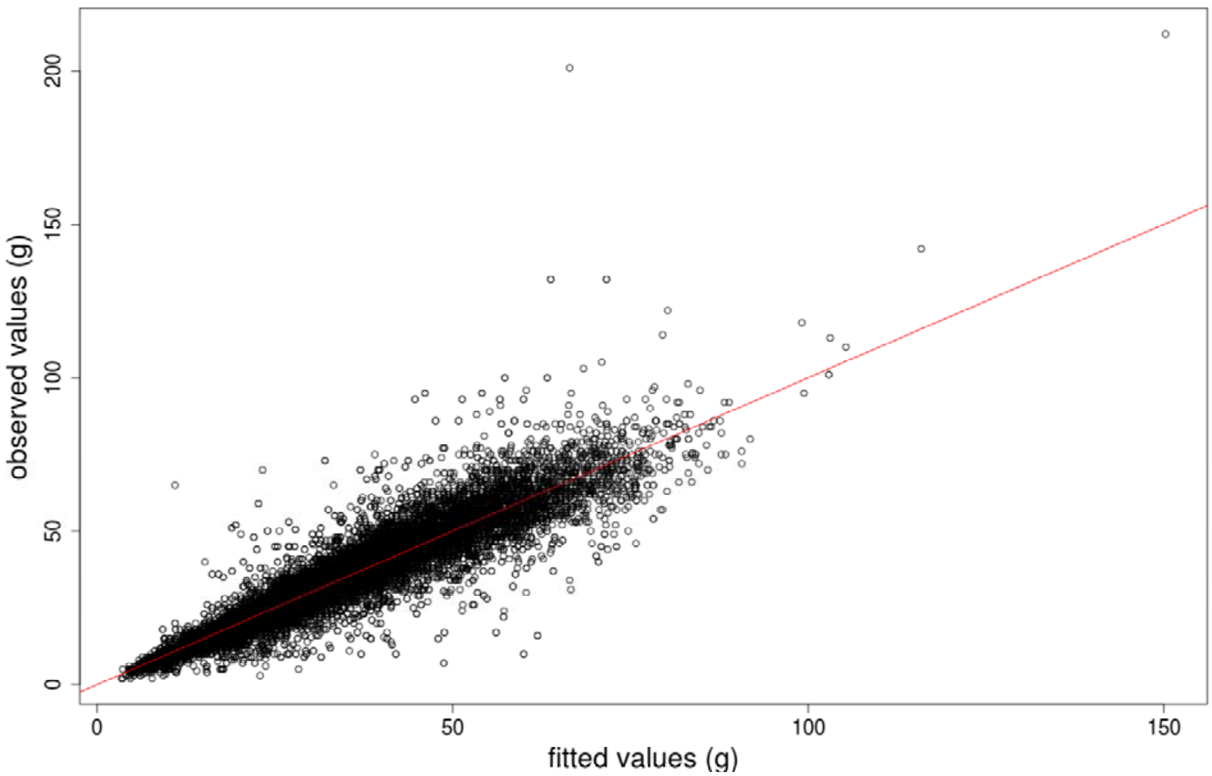

Figure 3: Observed vs. predicted meat weight using the best model by AIC.

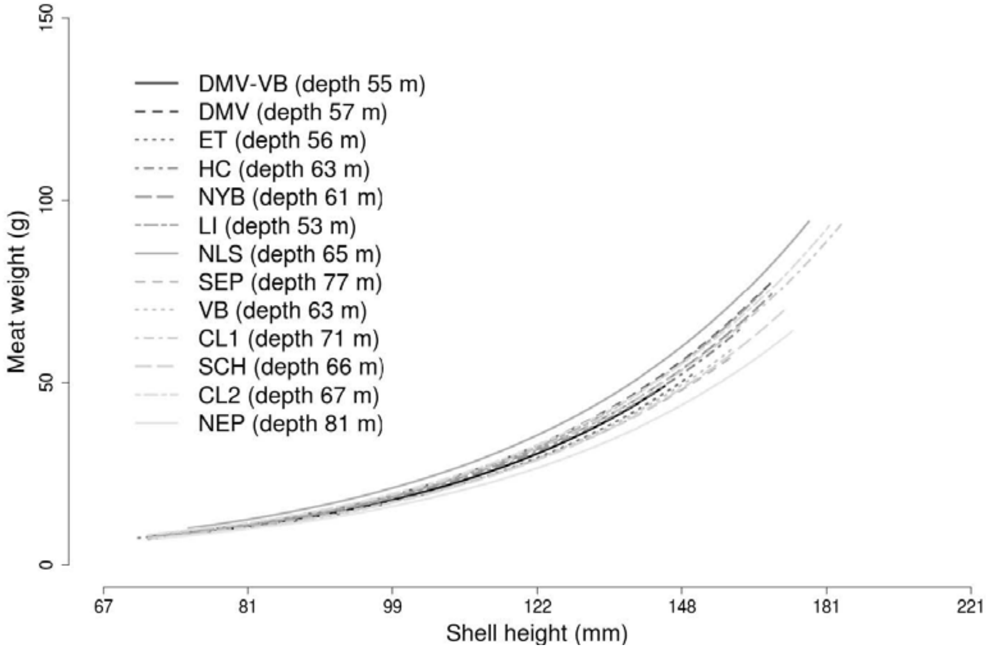

Figure 4: Meat weight curves by subarea. The depths used are the median depths observed in each subarea during all available years of the survey.

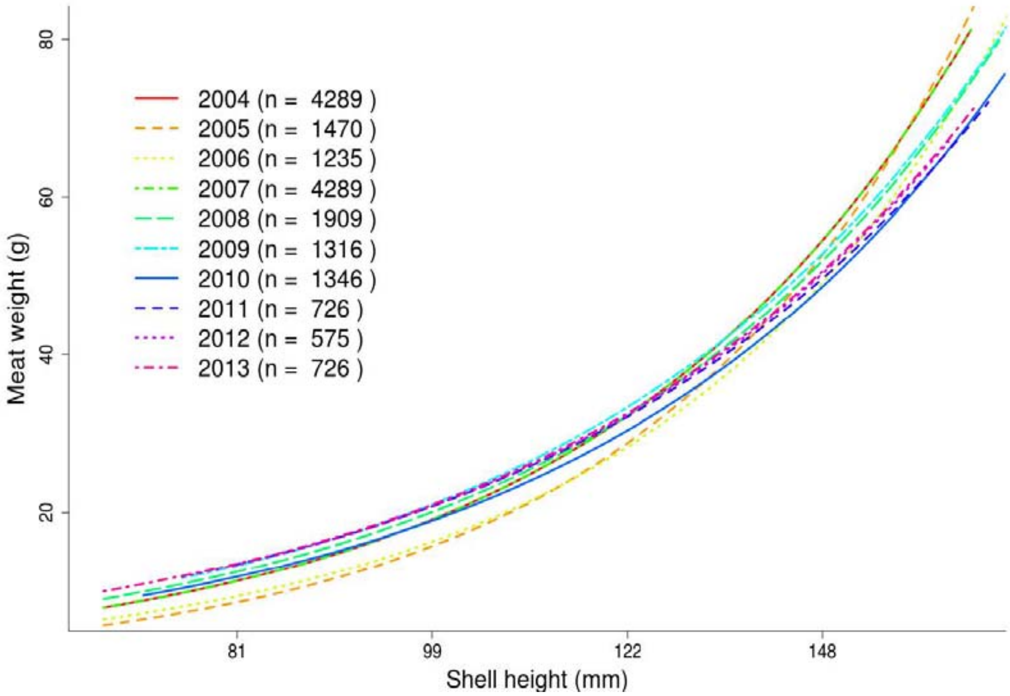

Figure 5: Meat weight curves by year. The curves are fits of the best model to annual subsets of the data. The sample size of each subset are shown in the legend.

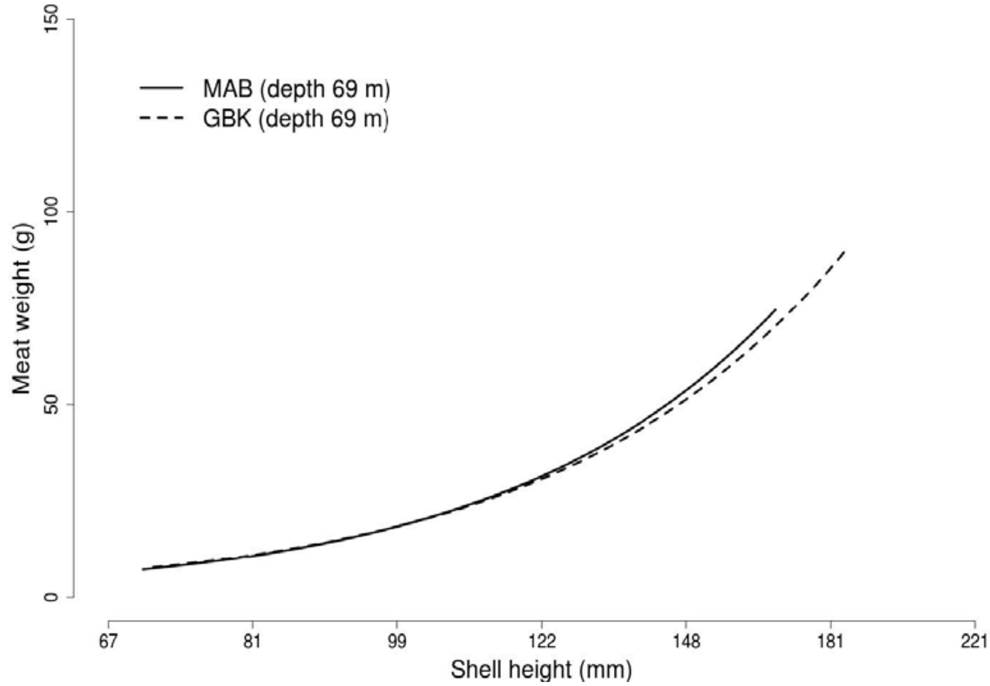

Figure 6: Shell height to meat weight relationship for each region based NEFSC survey data from 2003 -2013. The length of the curves represents the range of shell heights observed in each region.

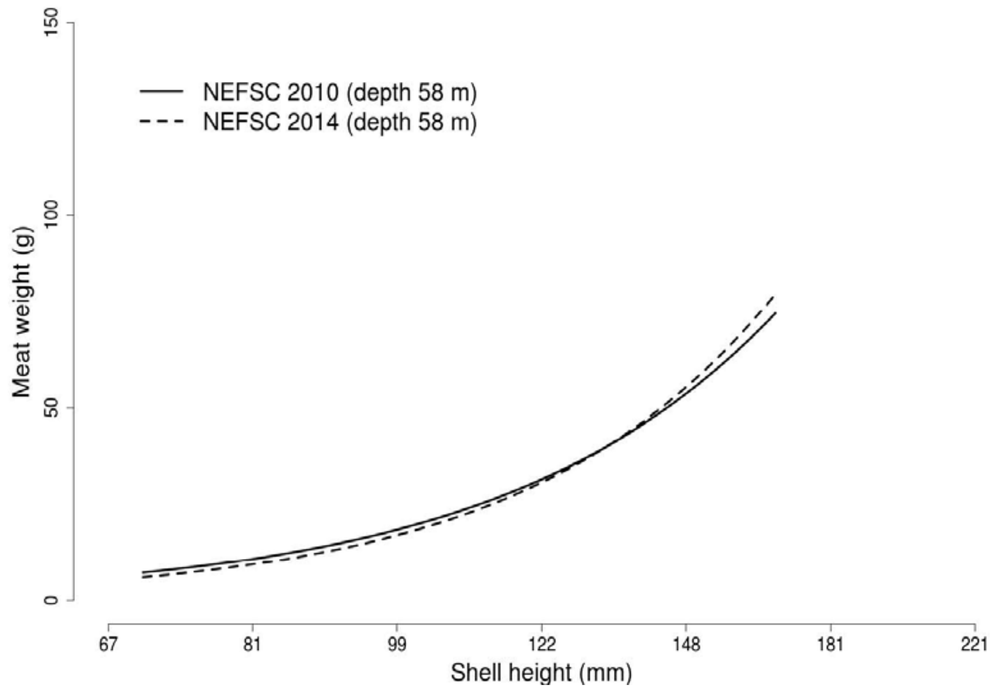

Figure 7: Shell height to meat weight relationship for two time periods in MAB. The length of the curves represents the range of shell heights observed in each period.

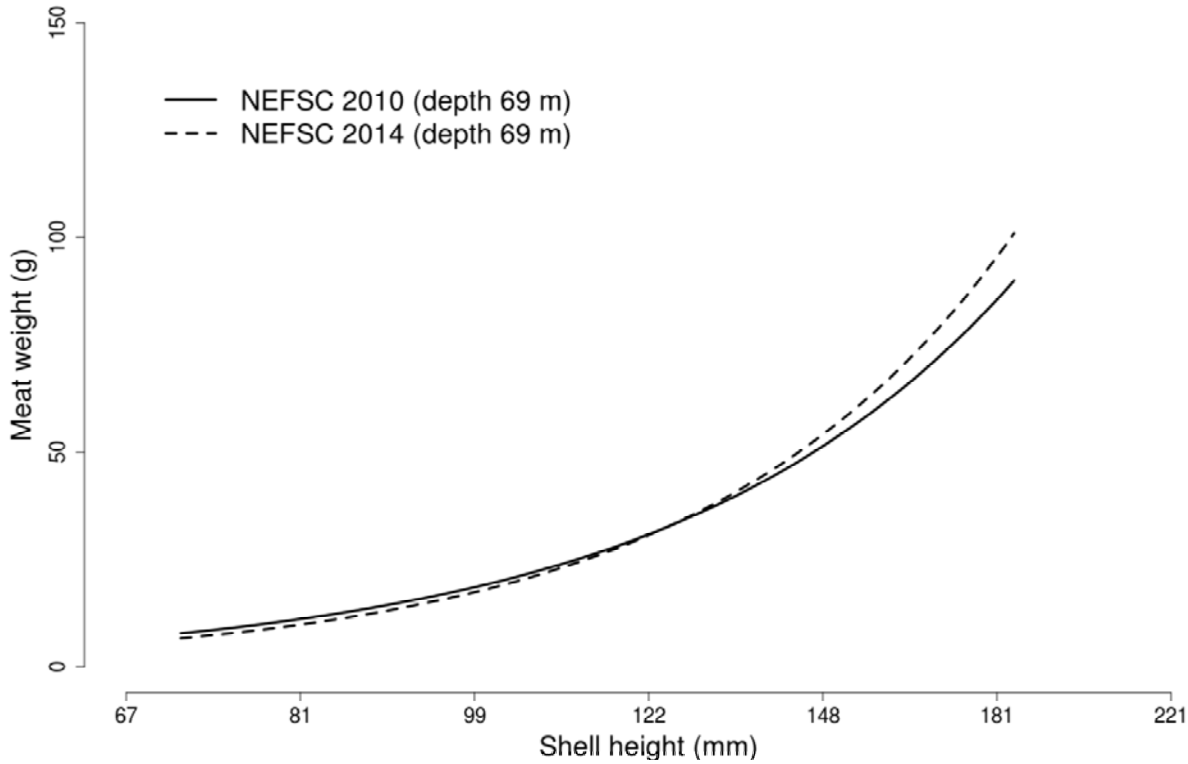

Figure 8: Shell height to meat weight relationship for two time periods in GBK. The length of the curves represents the range of shell heights observed in each period.

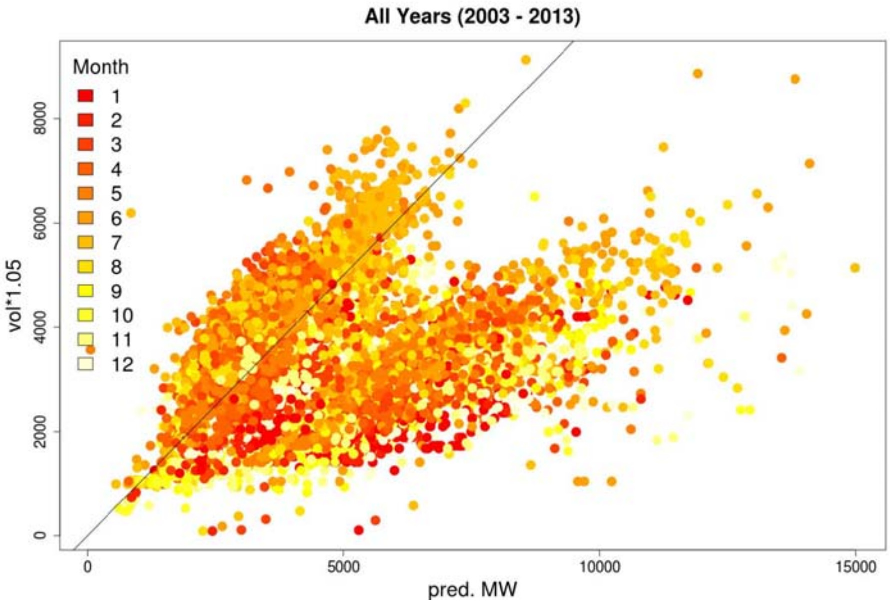

Figure 9: Meat weights estimated using data from the observer program compared to those expected based on NEFSC survey data. The solid line shows one to one correspondence and is for illustrative purposes only. The large cluster of points below the one to one line is an artifact of many more shells being measured for height than were packed into the cylinder for volume determination.

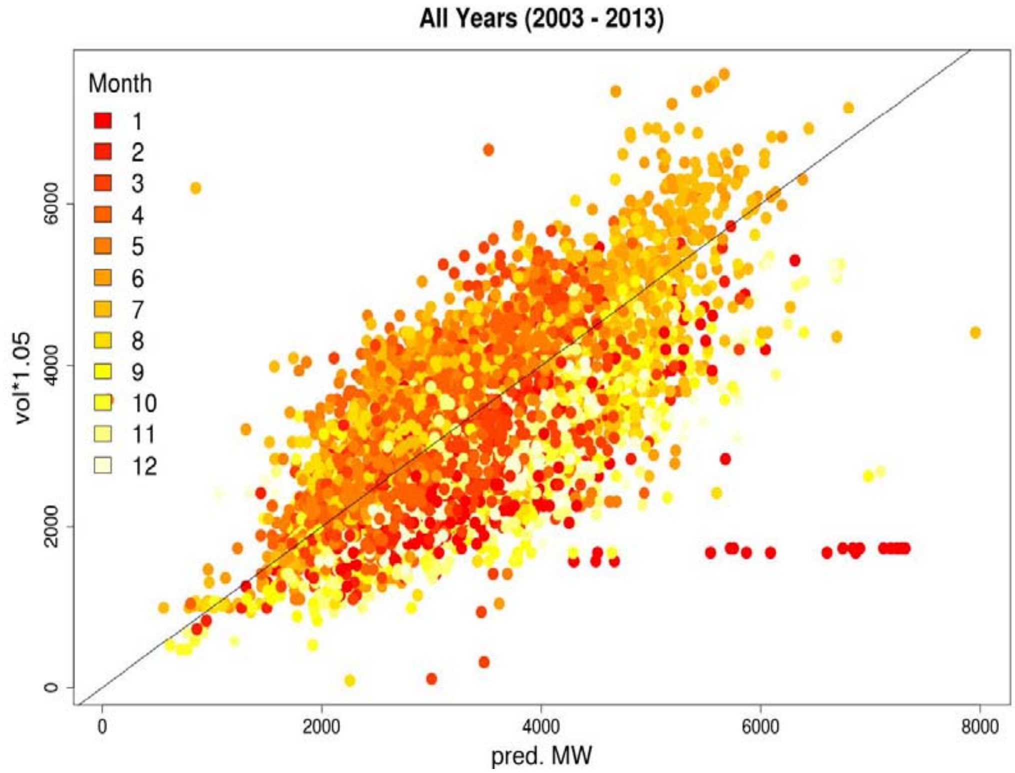

Figure 10: Meat weights estimated using data from the observer program compared to those expected based on NEFSC survey data. The solid line shows one to one correspondence and is for illustrative purposes only. Observations including more than 100 measured shells were excluded.

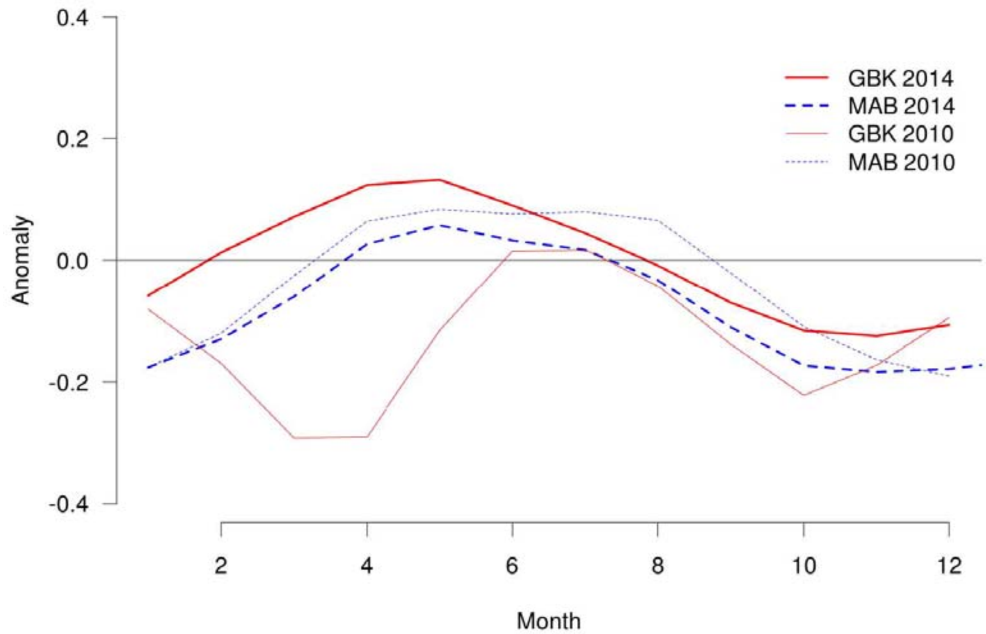

Figure 11: The anomalies estimated in the last assessment compared to the current anomalies.

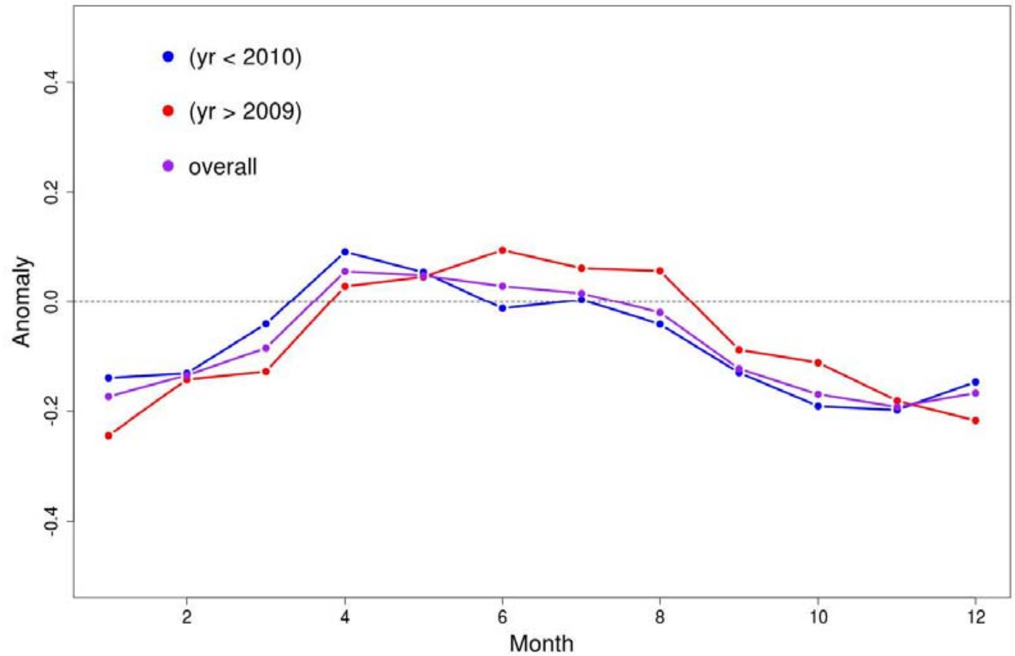

Figure 12: Monthly meat weight anomalies for the period prior to 2010, the period after 2010 and overall in the MAB.

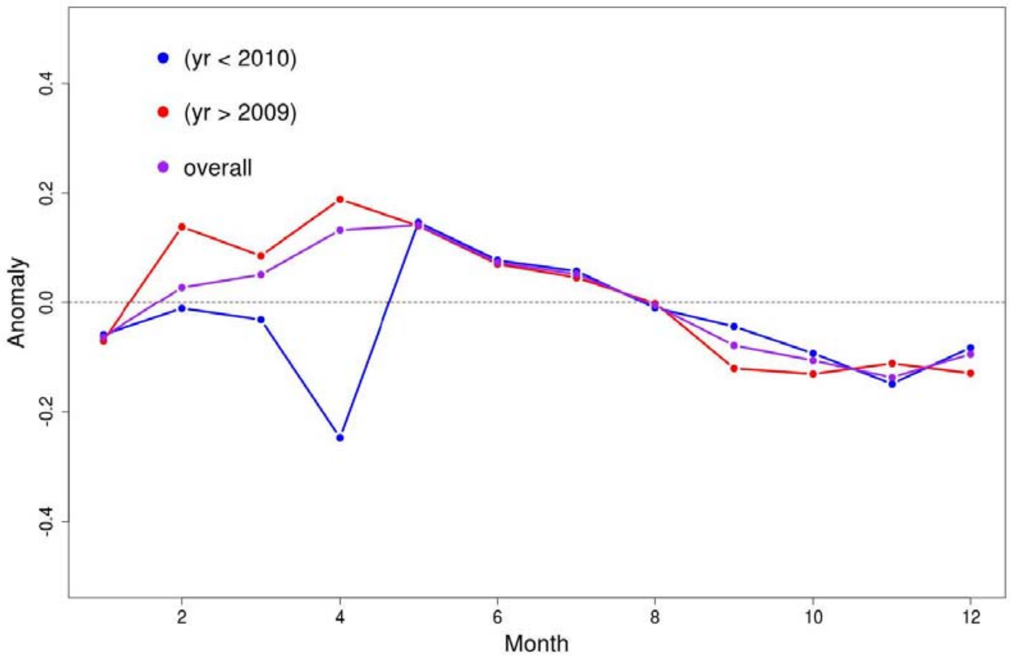

Figure 13: Monthly meat weight anomalies for the period prior to 2010, the period after 2010 and overall on GBK.

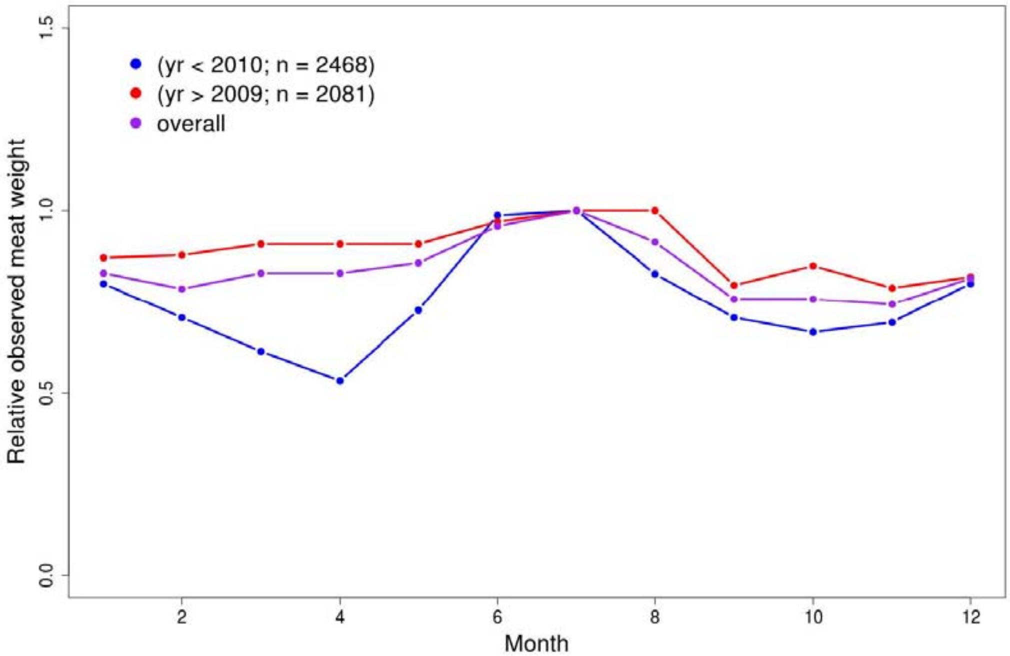

Figure 14: Relative monthly meat weight in observed commercial catches for the period prior to 2010, the period after 2010 and overall on GBK.

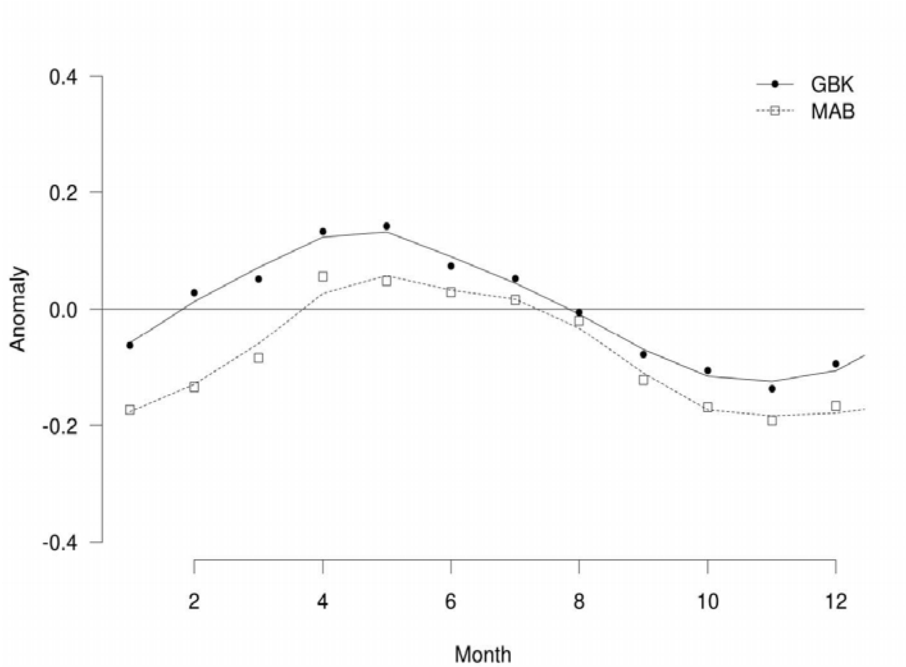

Figure 15: Smoothed anomalies for MAB and GBK.

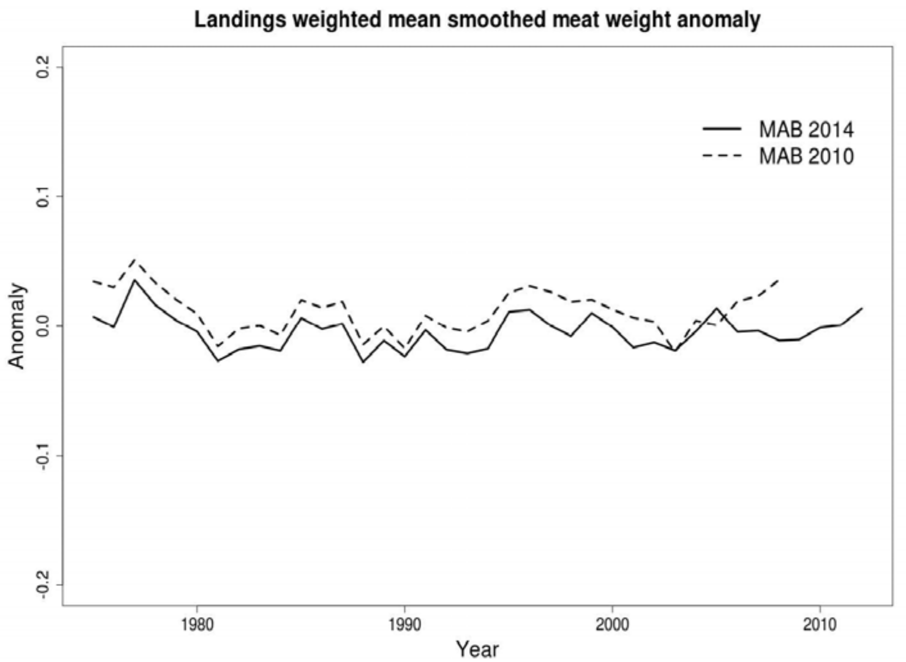

Figure 16: Landings weighted annual anomaly for MAB.

Figure 17: Landings weighted annual anomaly for GBK.

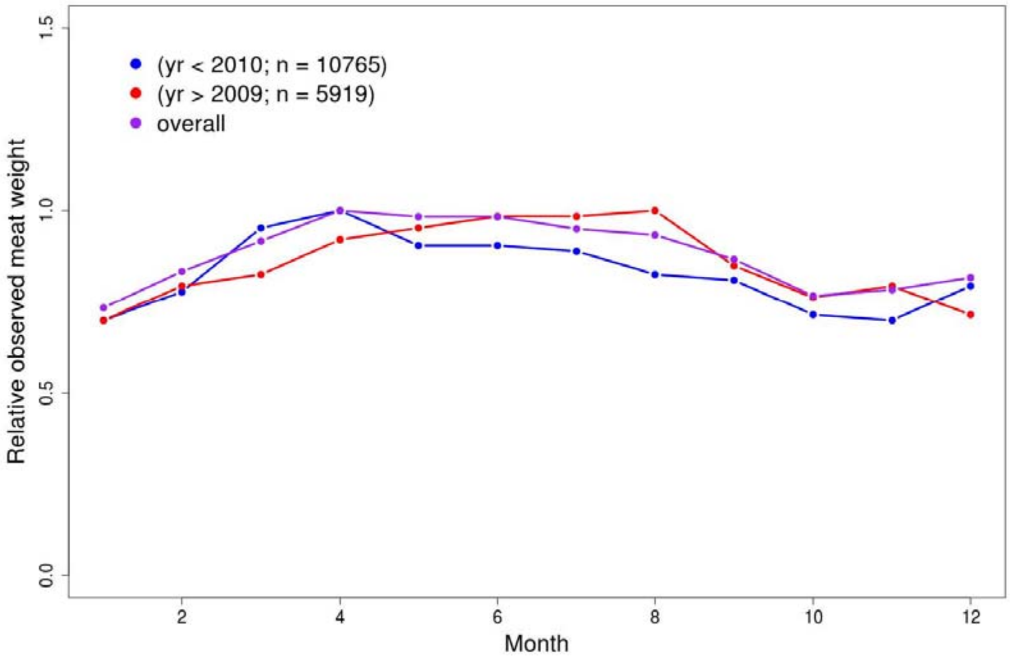

Figure 18: Relative monthly meat weight in observed commercial catches for the period prior to 2010, the period after 2010 and overall for MAB.

Appendix B4. Estimation of Dredge efficiency from paired dredge-HabCam observations

Timothy J. Miller, NEFSC, Woods Hole, MA

We use HabCam optical survey data to estimate capture efficiency of the NEFSC scallop survey dredge where capture efficiency is the probability of capture for a scallop in the path of the dredge. The literature on methods for analysis of comparative gear studies is extensive, but an alternative observation model is used here because HabCam provides hundreds or thousands of observation for each dredge tow. We develop a general hierarchical model for the dredge and HabCam observations, compare relative performance of a set of specific models, assess the statistical behavior of the estimators to determine the best model, and provide relatively precise estimates of the efficiency of the scallop survey dredge on sand and gravel/cobble substrates.

Materials and Methods

A dredge survey is conducted annually by the Northeast Fisheries Science Center to obtain relative abundance indices and other data for sea scallops. The dredge tows are conducted at stations according to a stratified random design. At a subset of these stations in 2008 and 2009, the HabCam optical survey device was also deployed. The HabCam captures images continuously along its track, but a thinned set were used in our analyses to make correlation between successive images within a station analyzed negligible. In all, we had 110 dredge stations where the number of sea scallops and swept area were recorded and where HabCam data including area searched, shell heights and number of scallops observed was recorded. There were 95-1,669 HabCam images used for each station.

The density of scallops differs by substrate type as based on HabCam as may the efficiency of the dredge. Sea scallop density is generally higher in sand than gravel substrates. We observe the substrate in each HabCam image, but the dredge track may cover various substrates which are not directly observed. The lack of these observations for the dredge makes estimation of relative efficiency for specific substrates impossible. However, sand and gravel/cobble substrates are more prevalent in particular survey strata. Sandy bottom is predominant in the Mid-Atlantic strata 6130, 6140, 6150, 6180, and 6190 and Georges Bank strata 6460, 6470, 6530, 6540, 6550, 6610, 6621, and 6670. Rock and gravel substrates are more common in Georges Bank strata 6490, 6500, 6510, 6520, 6651, 6652, 6661, 6662, and 6710. We therefore used stratum to establish proxies for substrate type when estimating dredge efficiency. In all there were 22 stations classified gravel (G) and 88 classified as sand (S).

Observation model

At station i out of n total stations, we have the numbers captured by the dredge NDi and the total number of sea scallops counted in associated HabCam images ni. For HabCam, we assume all scallops are observed in each image and that the surface area Aij of the substrate in the field of view is known. We also assume that the area swept by the dredge (determined using inclinometer sensors) is known. Conditional on the density of scallops in the image j at station i

δ_{Hij} , we assume the number of scallops observed in the image is Poisson distributed with mean

$$E(N_{Hij}|\delta_{Hij}, A_{Hij}) = \delta_{Hij}A_{Hij}. \tag{1}$$

Conditional on the density of scallops δ_{Di} and the known area swept by the dredge at station A_{Di}, we assume the number of captured scallops is Poisson distributed with mean

$$E(N_{Hij}|\delta_{Di}, A_{Di}) = q\delta_{Di}A_{Di} \tag{2}$$

where q is the efficiency of the dredge (cf. Paloheimo and Dickie 1964). Note that HabCam images is assumed to be 100% efficient at detecting scallops. More generally, q in Eq. 2 can be viewed as a relative efficiency when the HabCam is less than fully efficient.

We consider two different models for densities in each HabCam image δ_{Hij}. The first simply assumes that the densities within a station are equal $\delta_{Hij} = \delta_{Hi}$ and the second assumes that the densities are gamma distributed with station-specific mean δ_{Hi} and shape σ_{Hi}parameters,

$$f(\delta_{Hij}|\delta_{Hi}, \sigma_{Hi}) = \frac{\delta_{Hij}^{\sigma_{Hi}-1}\exp\left(-\delta_{Hij}\frac{\sigma_{Hi}}{\delta_{Hi}}\right)}{\Gamma(\sigma_{Hi})(\frac{\delta_{Hi}}{\sigma_{Hi}})^{\sigma_{Hi}}}.$$

In the former model the counts in the HabCam images $N_{Hij}|\delta_{Hi}$, are still conditionally Poisson distributed. In the latter model, they are negative binomial distributed in the with mean

$$E(N_{Hij}|\delta_{Hi}) = \delta_{Hi}A_{Hij}$$

and variance

$$V(N_{Hij}|\delta_{Hi}) = E(N_{Hij}|\delta_{Hi})\left[1 + \frac{E(N_{Hij}|\delta_{Hi})}{\sigma_{Hi}}\right].$$

For models where we assume the HabCam densities are gamma distributed we also consider variants where the shape parameter is constant across stations $\sigma_{Hi} = \sigma_H$ and where the shape parameter is itself gamma distributed with mean σ_H and shape parameterσ_{σ_H}. The former corresponds to an assumption that the variability of the densities observed in each image is constant across stations and the latter allows the variability to change from station to station. For stations where σ_{Hi} is large, the distribution of HabCam image observations is closer to Poisson.

The dredge efficiency q and densities δ_{Di} resulting in the dredge observations and the average densities δ_{Hi} for HabCam observations at a given station are not all estimable as fixed parameters. Estimation of dredge efficiency requires some assumption about the relationship of dredge and HabCam densities both within and across stations. We use a bivariate gamma distribution described by Moran (1969) to relate the densities producing the HabCam and dredge observations at each station (see Attachment B4-1). The distribution is a function of the mean and shape parameters for the marginal gamma distributions and a correlation parameter ($-1 <$

$\rho_\delta < 1$) that defines the relationship of dredge and HabCam densities within a station. The densities are independent when $\rho_\delta = 0$ and identical when $\rho_\delta = 1$. We assume the means of the dredge and HabCam densities are the same, but that these means are a function of the substrate type at a given station. The details for the different components of five plausible models we consider are provided in Table 1.

The general likelihood that we maximize for parameter estimation is

(3)

$$L = \prod_{i=1}^{n} \left\{ \int_0^\infty \int_0^\infty f(N_{Di}|\delta_{Di}) f(\delta_{Di}, \delta_{Hi}) \prod_{j=0}^{n_i} \left[\int_0^\infty f(N_{Hij}|\delta_{Hij}) f(\delta_{Hij}|\delta_{Hi}) d\delta_{Hij} \right] d\delta_{Di} d\delta_{Hi} \right\}.$$

Unobserved densities are treated as random effects and integrated out to obtain the marginal model likelihood. Models such as M_1 where HabCam densities within stations are assumed constant do not require the corresponding integration in Eq. 3. When densities within stations are gamma distributed, the numbers in the HabCam images conditional on δ_{Hi} are negative binomial distributed. The closed form for this marginal sub-model is computationally more efficient. Because the densities are marginally gamma distributed and the dredge counts are Poisson distributed conditional on the realized densities at each station, dredge observations $N_{Di}|\delta$ are marginally negative binomial distributed. The HabCam observations are also marginally negative binomial when the densities within a station are constant. In all models, the correlation of HabCam and dredge observations is defined by ρ.

We used AD Model Builder (Fournier et al. 2012) and the random effects library (Skaug and Fournier 2006) to maximize the marginal likelihood for all models. Parameters θ were estimated on log scale except ρ_δ which was defined as $\rho_\delta = -1 + 2/(1 + e^{-\theta})$. Standard errors were approximated using the delta method and asymmetric 95% confidence intervals were calculated by making the appropriate transformation of $\hat{\theta} \pm z_{1-\frac{\alpha}{2}} SE(\hat{\theta})$ where $\alpha = 0.05$ and $z_{1-\frac{\alpha}{2}}$ is the quantile of the standard normal distribution with cumulative probability $1 - \frac{\alpha}{2}$.

Simulation study

Because the methods were new, we used simulation to evaluate the reliability of the parameter estimates in the best model chosen by AIC. Using the parameter estimates from the best model, we simulated 1000 data sets and fit the same model to each data set. We calculated bias of parameter and standard error estimators and 95% confidence interval coverage.

Results

The best performing model M5 demonstrated that the efficiency of the dredge differed substantially in gravel (0.24) and sandy (0.40) substrates (Table 2). There were dramatic reductions in AIC between M_1 and M_2 and between M_2 and M_3. The reduction for M_2 implies strong evidence of variability in densities among HabCam observations within stations. The reduction in AIC for model M_3 implies strong evidence of variation among stations in the variance of HabCam observations. The very small difference in AIC values for M_3 and M_4 implies, implies that there is little evidence for differences in variability in mean densities among stations for both HabCam and dredge observations.

Mean densities were much greater in gravel substrates ($> 0.5\ m^2$) than sand substrates ($< 0.5\ m^2$) for all models. Because there were fewer stations in the gravel substrate than sand, the relative precision of mean density estimates for gravel was lower for all models (CV about 0.3 for gravel vs. about 0.1 for sand). The precision of the dredge efficiency estimate was lower in gravel also (CV about 0.14 for gravel vs. about 0.06 for sand) for the best performing model M_5. The correlation of mean densities for dredge and HabCam observations was high ($\rho_\delta > 0.9$) in all models.

Statistical behavior

Seventy file out of 1000 simulations with model M5 did not converge. However, average parameter estimates for the unconverged fits were similar to averages for simulations where the model did converge. The relative bias for estimates from converged model fits was negligible for most parameters except that the shape parameter σ_{σ_H}which determines the variability of HabCam densities at each station was biased high by about 12% (Table 3). Standard error estimates were negligible for most parameters except σ_{σ_H} (SE approximately -15%) and the efficiency of the dredge in gravel substrates (SE approximately 6%). Bias of coverage for 95% confidence intervals was also small with the exception of the parameter σ_{σ_H} (bias about -9%).

References

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Table 1. Details of the fixed effects and random effects sub-models in the hierarchical models we fitted to paired HABCAM and dredge data.

Model	$E(N_{Di}\|\delta_{Di}, q)$	$E(N_{Hij}\|\delta_{Hij})$	δ_{Hij}	σ_{Hi}	δ_{Di}, δ_{Hi}	n_p
M_1	$q\delta_{Di}A_{Di}$	$\delta_{Hij}A_{Hij}$	δ_{Hi}	—	$\mathrm{BGamma}(\delta(G,S),\sigma_\delta,\rho_\delta)$	5
M_2	$q\delta_{Di}A_{Di}$	$\delta_{Hij}A_{Hij}$	$\mathrm{Gamma}(\delta_{Hi},\sigma_{Hi})$	σ_H	$\mathrm{BGamma}(\delta(G,S),\sigma_\delta,\rho_\delta)$	6
M_3	$q\delta_{Di}A_{Di}$	$\delta_{Hij}A_{Hij}$	$\mathrm{Gamma}(\delta_{Hi},\sigma_{Hi})$	$\mathrm{Gamma}(\sigma_H,\sigma_{\sigma_H})$	$\mathrm{BGamma}(\delta(G,S),\sigma_\delta,\rho_\delta)$	7
M_4	$q\delta_{Di}A_{Di}$	$\delta_{Hij}A_{Hij}$	$\mathrm{Gamma}(\delta_{Hi},\sigma_{Hi})$	$\mathrm{Gamma}(\sigma_H,\sigma_{\sigma_H})$	$\mathrm{BGamma}(\delta(G,S),\sigma_{D\delta},\sigma_{H\delta},\rho_\delta)$	8
M_5	$q(G,S)\delta_{Di}A_{Di}$	$\delta_{Hij}A_{Hij}$	$\mathrm{Gamma}(\delta_{Hi},\sigma_{Hi})$	$\mathrm{Gamma}(\sigma_H,\sigma_{\sigma_H})$	$\mathrm{BGamma}(\delta(G,S),\sigma_\delta,\rho_\delta)$	8

Table 2. AIC and parameter estimates for each fitted model. Parameters denoted with (G) and (S) are specific to observations from gravel and sand substrates, respectively and (D) and (H) denote parameters specific to dredge and HABCAM observations, respectively.

Model	$\Delta(AIC)$	q	δ	σ_δ	ρ_δ	σ_H	σ_{σ_H}
M_1	6724.0	0.376 (0.020)	5.048 (1.476) (G) 0.470 (0.056) (S)	0.621 (0.084)	0.905 (0.022)	—	—
M_2	986.6	0.376 (0.020)	5.041 (1.480) (G) 0.469 (0.056) (S)	0.622 (0.084)	0.905 (0.022)	1.576 (0.044)	—
M_3	8.20	0.376 (0.020)	5.085 (1.505) (G) 0.469 (0.056) (S)	0.620 (0.084)	0.906 (0.022)	3.419 (0.625)	0.880 (0.207)
M_4	8.60	0.383 (0.021)	5.448 (1.653) (G) 0.461 (0.054) (S)	0.586 (0.084) (D) 0.647 (0.091) (H)	0.910 (0.021)	3.418 (0.624)	0.880 (0.207)
M_5	—	0.243 (0.034) (G) 0.400 (0.022) (S)	5.771 (1.708) (G) 0.458 (0.054) (S)	0.630 (0.085)	0.912 (0.020)	0.630 (0.624)	0.880 (0.207)

Table 3. Relative bias of parameter and standard error estimators and coverage probability of approximate 95% confidence interval for 925 simulated data sets with parameters specified from the best performing model M_5.

Parameter	Value	Relative Bias	SE	Relative Bias of SE	95% CI coverage
q (G)	0.24	0.01	0.03	−0.06	0.93
q (S)	0.40	0.00	0.02	−0.01	0.94
δ (G)	5.77	−0.01	1.33	0.00	0.94
δ (S)	0.46	0.00	0.06	−0.03	0.93
σ_δ	0.63	0.03	0.09	−0.03	0.94
ρ_δ	0.91	0.00	0.02	−0.04	0.94
σ_H	3.42	−0.03	0.53	−0.02	0.91
σ_{σ_H}	0.88	0.12	0.19	−0.15	0.86

Attachment B4-1. Bivariate gamma distribution.

This is the same formulation described by Moran (1969). Let Y_1 and Y_2 be bivariate standard normal distributed with correlation parameter ρ,

$$f(Y_1, Y_2) = \frac{1}{2\pi(1-\rho^2)^{\frac{1}{2}}} \exp\left[-\frac{1}{2(1-\rho^2)}(y_1^2 - 2\rho y_1 y_2 + y_2^2)\right].$$

Then letting the marginal distributions $F(X_1) = F(Y_1)$ and $F(X_2) = F(Y_2)$, where

$$F(X_i) = \int_0^{X_i} \frac{1}{\Gamma(\sigma_i)\beta_i^{\sigma_i}} w_i^{\sigma_i - 1} \exp\left(-w_i \beta_i^{-1}\right) dw_i,$$

X_1 and X_2 have a bivariate gamma distribution with means $\sigma_i \beta_i$ and marginal variances $\sigma_i \beta_i^2$, but correlation defined by ρ. When $\rho = 0$, X_1 and X_2 are independent and when $\rho = 1$, X_1 and X_2 are identically distributed.

Appendix B5. Empirical assessment

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Introduction

The empirical assessment used simple techniques to estimate sea scallop stock abundance, biomass and fishing mortality in the MAB, GBK and combined stock areas. The purpose was to evaluate the accuracy of CASA estimates as independently as possible. However, empirical results could be used in place of CASA model estimates if the later were unavailable. The data and various parameters used in the empirical analysis are a subset of those also used in the CASA model and were all obtained independently in field studies or other analyses rather than from a stock assessment model.

Materials and methods

Survey swept-area abundance data used in the empirical analysis were the best available estimates of total 40+ mm stock abundance and considered reliable. Abundance from the dredge and optical surveys (HabCam and SMAST large camera) were the same as used in CASA except that SMAST data were adjusted for logistic size selectivity using externally estimated selectivity curves (Appendix B7 in NEFSC 2007). In CASA, the same selectivity curves are applied in the model after data input. In addition abundance estimates were not rescaled for comparison to a prior distribution as in CASA although this had no impact on results. Size selectivity was assumed to be flat in the dredge and HabCam surveys.

Updated capture efficiency estimates were used in expansion of the dredge survey to calculate swept-area abundance prior to their use in this analysis (Appendix B4). Additional variance due to uncertainty about dredge efficiency was included (see below). Capture efficiency was assumed to be 100% in the dredge and HabCam surveys for scallops 40+ mm SH in calculating swept-area abundance for this analysis. Thus, capture efficiency was factored in to all of the survey abundance data prior to use here.

As in the CASA model analysis, dredge survey abundance estimates were adjusted to account for scallops in deep or shallow water areas not sampled by the dredge but no adjustments were made for areas of poor habitat within the survey area. Survey abundance at length data were not adjusted for errors in measuring shell height as in the CASA model although such errors are appreciable in the optical surveys because the adjustment requires information available in a simulation based stock assessment model. These type of errors smooth size composition estimates making modes lower, valleys higher and proportions in the largest and smallest length groups larger (Jacobson et al. ????).

Five mm length groups (40-45, 45-50 …) were used and the last length group was always a plus group. Intermediate calculations included all of the size groups in the original data but results are summarized using a 140+ mm size group, which is roughly the same as von Bertalanfy L_{max} (asymptotic mean size) estimates. Only years 2003-2013 where included because at least two surveys (dredge+SMAST, dredge+HabCam, or dredge+SMAST+HabCam) were conducted each year. Using multiple independent surveys helps smooth estimates without using a population dynamics model like CASA.

Total abundance in each year and for each size group ($N_{y,L}$) was estimated by averaging swept-area abundance estimates from each survey:

$$N_{y,L} = \frac{\sum_s N_{s,y,L}}{n_{s,y}}$$

where $N_{s,y,L}$ was swept area abundance data for year y and survey s while $n_{s,y}$=2 or 3 was the number of surveys. Total survey stock abundance was $N_y = \sum_L N_{y,L}$. Stratified random CVs for mean total number per tow and the number of positive tows by year in the dredge survey provide some information about precision of abundance data (Table 1 and Figure 1).

Variances for $N_{y,L}$ were calculated from length specific average CVs for mean number per tow in the dredge survey. Length specific variances were not easily available for the SMAST and HabCam surveys. In particular:

$$Var(N_{s,y,L}) = (CV_L N_{s,y,L})^2$$

where CV_L is the average CV at length in the dredge survey for either Georges Bank or the Mid-Atlantic (Figure 2). CVs for total abundance N_y were from the CVs for total catch per tow in each survey (Figure 1):

$$Var(N_{s,y}) = CV_{s,y} N_{s,y}$$

and:

$$Var(N_y) = \sum_s Var(N_{s,y})/n_s^2.$$

Dredge survey abundance CVs were increased to account for uncertainty in capture efficiency. CVs for dredge survey capture efficiency were 0.034/0.243=0.14 (gravel/cobble) and 0.022/0.4=0.05 (sand, Appendix B4). Therefore, the adjusted CV for a dredge survey abundance estimate was $\sqrt{CV_{s,y}^2 + 0.1^2}$ where 0.1 is close to the average CV for gravel/cobble and sand.

Uncertainty about stock area, area sampled, and other factors were ignored in calculating survey abundance. However, variance from these factors was probably modest relative to the variance in mean catch per tow and capture efficiency for the dredge survey. Uncertainty about stock area is relatively small because scallops are sessile with a static spatial distribution that is well defined by the optical surveys and covered effectively by each survey after the dredge data are adjusted for area not surveyed. Uncertainty about size selectivity in the experimentally derived size selectivity curve for the SMAST survey was ignored for lack of time but could have been included.

For plotting, mean abundance at length estimates were smoothed with GAM models fit assuming gamma errors using the mgcv library in the R programming language (Wood 2006):

gam(y~s(x),family=Gamma(link=log),weights=wts)

The variances used for weights were, for example, $Var(N_{y,L}) = [N_{y,L}\ CV(N_{y,L})]^2$. Assuming predicted values were gamma distributed, 95% percent confidence intervals were calculated for means equal to the fitted values and variances $Var(\hat{N}_{y,L}) = [\hat{N}_{y,L}\ CV(F_{y,L})]^2$. The variance of the fitted values calculated in the GAM was not used because it grossly underestimated uncertainty. Better confidence intervals might have been obtained by combining the CV above with the CV for uncertainty in the smooth trend calculated by the GAM software.

Fishing mortality rates by year and length ($F_{y,L}$) were approximated by dividing catch numbers by estimated abundance:

$$F_{y,L} = \frac{C_{y,L}}{N_{y,L}}$$

Where $C_{y,L}$ is catch number at length. This approximation is reasonable because the instantaneous rate of fishing mortality is exactly $F = C/\bar{N}$ (Ricker 1975) and because scallop surveys tend to occur near the middle of the year when abundance may be similar to average abundance (Table 2).

Catch numbers at length in each year ($C_{y,L}$) were calculated:

$$C_{y,L} = \frac{W_y}{m_y} p_{y,L}$$

where W_y is total meat weight for landings, m_y is mean weight of scallops in the catch and $p_{y,L}$ is a size-specific proportion of the total commercial catch. The mean weight (m_y) was calculated from commercial size composition data, survey shell height-meat weight parameters and annual commercial meat weight anomalies as in the CASA model.

Variances for fishing mortality were approximated based on CVs for average survey abundance and an assumed CV=10% for catch to give $CV(F_{y,L}) = \sqrt{CV(N_{y,L}) + 0.1^2}$. Abundance weighted fishing mortality (all sizes combined) was approximated $F_y = C_y/N_y$ with $(F_y) = \sqrt{CV(N_y) + 0.1^2}$.

CASA models include a correction for incidental mortality which is highest on the smallest size groups. This adjustment was not made in the empirical analysis because it requires an a-priori estimate of fishing mortality and fishery selectivity not available in the empirical analysis. Therefore, fishing mortality $F_{y,L}$ and F_y are underestimated relative to total fishery mortality. Fishing mortality attributable to landings and fully recruited fishing mortality are unaffected.

GAM models were used to smooth fishing mortality at size estimates and confidence intervals were estimated in a manner similar to abundance at size. The variances used for weights were $Var(F_{y,L}) = [F_{y,L}\ CV(F_{y,L})]^2$ and the variances used to calculate confidence intervals were $Var(\hat{F}_{y,L}) = [\hat{F}_{y,L}\ CV(F_{y,L})]^2$. Fully recruited fishing mortality was estimated using the gam model to predict $F_{y,L}$ over a wide range of narrowly spaced shell height values and selecting the largest value of predicted $F_{y,L}$.

Commercial size selectivity estimates are useful although not required in the empirical assessment or in projections which are handled independently in the SAMS model. However, for illustration, size selectivity by year and size $s_{y,L}$ was estimated by rescaling fishing mortality at size:

$$s_{y,L} = \frac{F_{y,L}}{max(F_{y,L})}$$

and then smoothing the rescaled estimates using a model for proportions:

gam(y~s(x),family=quasibinomial,weights=wts)

The weights were one when estimating selectivity at size in individual years. Weights equal n_s were used when selectivity estimates for multiple years were combined to estimate average fishery selectivity. After the GAM model was fit, predicted selectivity were rescaled again to a maximum value of one. Fishable abundance (available to the fishery) in each year A_y can be calculated using abundance at size and a fishery selectivity estimate although the estimates are not required for this empirical assessment. For example:

$$A_y = \sum_L s_L\, N_{y,L}$$

<u>Results</u>

Empirical abundance at size estimates appear reasonably precise and smooth although the smoothness is due partly to measurement errors is survey size data (Figure 3). The progression of two large year classes is clear during 2003-2006 in the Mid-Atlantic and during 2012-2013 in both regions. There are clear differences between the two regions in population size composition (e.g. the 140+ mm size group) seem clear. Important aspects of the fishery (relatively low exploitation rates and targeting large animals) are evident in comparing abundance andt catch numbers at size (Figure 4).

Empirical fishing mortality at length data show that fishing pressure is higher in the Mid-Atlantic than on Georges Bank (Figure 5). The working group concluded that the variation over time in fishery selectivity between domed and ascending patterns could be explained in terms of management measures that: 1) increased the minimum ring size on commercial vessels and decreased selectivity of small scallops during 1994-1995, 2) recruitment events, and 3) management measures that opened and closed rotational harvest areas where large scallops were common. Average fishery selectivity curves for 2003-2013 illustrate how selectivity for particular time periods can be estimated as needed for management related or other analyses (Figure 6).

Empirical abundance and fishing mortality for the combined Mid-Atlantic and Georges Bank regions were calculated by summing catch numbers and abundance for the Mid-Atlantic and Georges Bank regions and them computing approximate fishing mortality rates from the ratio of the sums. CVs and were calculated using standard formulas for sums of random variables.

Empirical and CASA model estimates of abundance and fishing mortality show similar trends in all regions (Tables 3-4 and Figure 7). However, empirical abundance estimates were generally higher reflecting the tension in CASA models between matching the scale of the abundance data (matching the prior on Q) versus fitting the survey and fishery data. As expected, fishing mortality show the inverse pattern with empirical generally lower than CASA estimates.

Fully recruited fishing mortality estimates from empirical calculations were usually lower than from CASA the CASA model as well (Figure 8). However, the comparison may not be very useful because of fully recruited F depends on fishery selectivity assumptions which differed in the two assessment approaches.

Status determination and catch advice

No special provisions are necessary for providing catch advice to the scallop fishery using the empirical methods. Catch advice is generated using a simulation models (SAMS) which is initialized using best estimates of abundance at length from surveys (i.e. using the empirical method).

Reference points used to determine if the scallop stock is overfished or if overfishing is occurring are more difficult. For this assessment, it would be reasonable to compare empirical fishing mortality estimates to reference points calculated in terms of landings divided by 1 July abundance from the SYM reference point model. The CASA model may be problematic due to the tension between scale of the model estimates and general fit to the data. However, the current condition of the stock (not overfished and overfishing not occurring) is clear based on both sets of models and common sense. Empirical and CASA results are broadly similar. If the trend in B/B_{MSY} estimates from the CASA and SYM models are roughly correct, then the ratio for 2013 should be sufficient to determine if the stock is overfished despite uncertainty about scale.

Advantages and disadvantages

It was advantageous to use both empirical and the complex CASA modeling approach for CASA, if only for comparison and to determine if the CASA model results were plausible. Empirical estimates depend almost entirely on data while the CASA model depends on data, biological assumptions (e.g. about growth and natural mortality) and modeling techniques. The empirical approach requires fewer assumptions about growth, natural mortality, size selectivity, etc. and uses most of the data also used in CASA. However, the empirical approach is sensitive to survey measurement errors which are relatively high in the Georges Bank area. It is therefore necessary to have multiple surveys each year for empirical estimation. The empirical approach cannot be applied in all years and the CASA model may give a clearer long term perspective on stock size and productivity.

In theory, the CASA model should do a better job of balancing goodness of fit to survey, catch and size composition data to estimate realistically smooth population trends. However, experience with many real stocks and models indicates that stock assessment models often have pathological problems that may be difficult to resolve due to many potential causes including inaccurate catch data, changes in natural mortality, etc..

An assessment model like CASA makes it easier to calculate reference points. Empirical reference point methods were not evaluated in this assessment but there are a number of methods that could be applied.

Empirical estimates do not suffer from retrospective patterns, which are usually blamed on model structure or assumptions about the data which may remain hidden in empirical analyses. CASA model results did not show retrospective error in this assessment but this was probably due to the proximity of the estimates to priors for survey capture efficiency with tension in the model pulling abundance estimates low enough so that implied capture efficiency estimates were trapped near the upper prior bound. The empirical estimates in this assessment for 2003-2013 are less sensitive to errors in historical catch which are often suspected when modeling problems occur.

Reference:

NEFSC. 2007. 45th Northeast Regional Stock Assessment Workshop (45th SAW): 45th SAW assessment report. US Dep Commer, Northeast Fish Sci Cent Ref Doc 07-16.

Table 1. Numbers of tows in which at least one scallop was caught in the MAB and GBK areas during dredge surveys during 2003-2013 by size group. For example, the 40 mm size group is 40-44.9 mm SH. The last size bin (140+ mm SH) is a plus group. The number of positive tows is a lower bound estimate for the effective sample size in each year/size group category..

Year	Size group (mm)																				
	40	45	50	55	60	65	70	75	80	85	90	95	100	105	110	115	120	125	130	135	140+
MAB																					
2003	110	113	120	127	145	146	145	151	147	145	152	158	160	159	156	157	135	122	91	56	39
2004	124	132	145	137	150	146	154	170	187	192	191	188	187	186	192	187	169	150	120	84	41
2005	157	160	170	161	147	152	142	168	188	205	215	217	220	224	224	223	216	210	194	164	127
2006	111	139	160	176	196	232	222	231	242	235	239	240	246	250	248	252	246	234	211	163	117
2007	70	97	130	148	150	172	186	204	209	218	237	249	250	257	250	249	244	237	203	168	131
2008	168	183	179	178	176	158	154	159	172	180	199	214	217	222	215	217	207	202	175	149	125
2009	77	88	104	97	114	108	121	147	152	151	160	152	157	153	162	157	156	150	130	103	86
2010	141	156	156	135	131	117	122	134	171	199	219	227	240	236	234	241	233	227	196	132	100
2011	119	149	151	146	123	111	96	117	165	191	223	214	219	225	232	230	238	238	225	187	163
2012	155	165	158	141	131	120	126	149	156	174	185	187	192	211	208	201	213	217	204	171	119
2013	99	129	164	167	213	216	229	227	222	232	231	238	224	220	220	216	213	214	203	161	140
GBK																					
2003	64	72	76	84	99	92	95	99	96	110	115	116	124	137	137	131	131	139	128	114	122
2004	83	94	96	105	102	92	95	108	120	141	140	145	148	156	153	164	166	169	163	141	140
2005	46	57	98	94	108	101	106	109	133	142	164	177	205	229	245	254	267	277	276	256	248
2006	67	74	88	103	108	96	103	96	93	112	127	138	138	144	154	154	170	172	173	165	172
2007	153	181	217	215	240	222	204	189	190	185	199	202	210	212	208	246	271	276	277	274	284
2008	111	114	129	146	156	145	131	141	138	148	158	174	178	183	168	170	159	176	169	180	196
2009	95	107	135	132	128	126	119	117	130	145	158	160	156	162	164	162	160	161	152	148	168
2010	81	77	92	88	111	108	117	130	152	150	170	161	185	193	214	215	219	223	224	206	216
2011	44	44	43	50	68	72	85	92	119	132	146	138	154	148	155	176	177	184	180	176	184
2012	61	86	100	105	100	94	107	107	125	133	144	155	151	157	168	174	176	181	181	178	177
2013	81	106	115	123	138	139	118	108	112	116	122	126	134	133	141	142	155	153	163	156	161

Table 2. Dates (Julian) for sea scallop surveys during 2003-2013 in the MAB and GBK regions.

Survey	Mid-Atlantic			Georges Bank			Comment
	Min	Max	Mid	Min	Max	Mid	
Dredge	130	215	173	163	230	197	1979-2013
SMAST	130	194	162	165	233	199	2003-2009
HabCam	153	201	177	159	210	184	2011-2012 for the Mid-Atlantic and 2011-2013 for Georges Bank

Table 3. Abundance and fishing mortality (estimates from the empirical approach and CASA model for the Georges Bank (top) and Mid-Atlantic (bottom) regions.

Year	Empirical					CASA	
	Abundance (Mid-year, 40+ mm, 10^6)	CV	Landings	Aprox. F	CV	Abundance (1 July, 40+ mm, 10^6)	Landings/ Abundance
Georges Bank							
2003	4,145	0.10	173	0.04	0.14	3,517	0.05
2004	3,788	0.12	133	0.04	0.15	3,159	0.04
2005	3,660	0.11	267	0.07	0.15	3,132	0.09
2006	3,216	0.11	448	0.14	0.15	2,769	0.16
2007	3,979	0.11	249	0.06	0.15	3,219	0.08
2008	3,941	0.10	179	0.05	0.14	3,300	0.05
2009	5,332	0.12	221	0.04	0.15	3,690	0.06
2010	4,883	0.17	170	0.03	0.19	3,801	0.04
2011	4,169	0.12	217	0.05	0.15	4,194	0.05
2012	3,498	0.08	316	0.09	0.13	4,607	0.07
2013	4,073	0.14	365	0.09	0.17	5,620	0.06
Mid-Atlantic							
2003	13,601	0.31	807	0.06	0.33	5,511	0.15
2004	7,324	0.21	918	0.13	0.23	4,036	0.23
2005	6,154	0.15	545	0.09	0.18	4,811	0.11
2006	6,261	0.15	272	0.04	0.18	4,226	0.06
2007	5,521	0.15	503	0.09	0.18	4,310	0.12
2008	6,340	0.13	463	0.07	0.16	4,647	0.10
2009	5,312	0.11	664	0.13	0.15	3,202	0.21
2010	3,794	0.11	687	0.18	0.15	2,458	0.28
2011	2,747	0.10	598	0.22	0.14	1,606	0.37
2012	4,617	0.10	365	0.08	0.14	3,387	0.11
2013	4,163	0.14	219	0.05	0.17	2,648	0.08

Table 4. Abundance and fishing mortality (estimates from the empirical approach and CASA model to the combined Georges Bank plus Mid-Atlantic regions (whole stock).

Year	Empirical					CASA	
	Abundance (Mid-year, 40+ mm, 10^6)	CV	Landings	Aprox. F	CV	Abundance (1 July, 40+ mm, 10^6)	Landings/ Abundance
				Whole stock			
2003	17,746	0.24	980	0.06	0.26	9,028	0.11
2004	11,112	0.14	1,051	0.09	0.17	7,195	0.15
2005	9,814	0.11	812	0.08	0.15	7,942	0.10
2006	9,477	0.11	720	0.08	0.15	6,994	0.10
2007	9,500	0.10	752	0.08	0.14	7,529	0.10
2008	10,281	0.09	643	0.06	0.13	7,946	0.08
2009	10,644	0.08	885	0.08	0.13	6,891	0.13
2010	8,677	0.11	857	0.10	0.15	6,259	0.14
2011	6,915	0.08	815	0.12	0.13	5,799	0.14
2012	8,115	0.07	681	0.08	0.12	7,995	0.09
2013	8,237	0.10	584	0.07	0.14	8,269	0.07

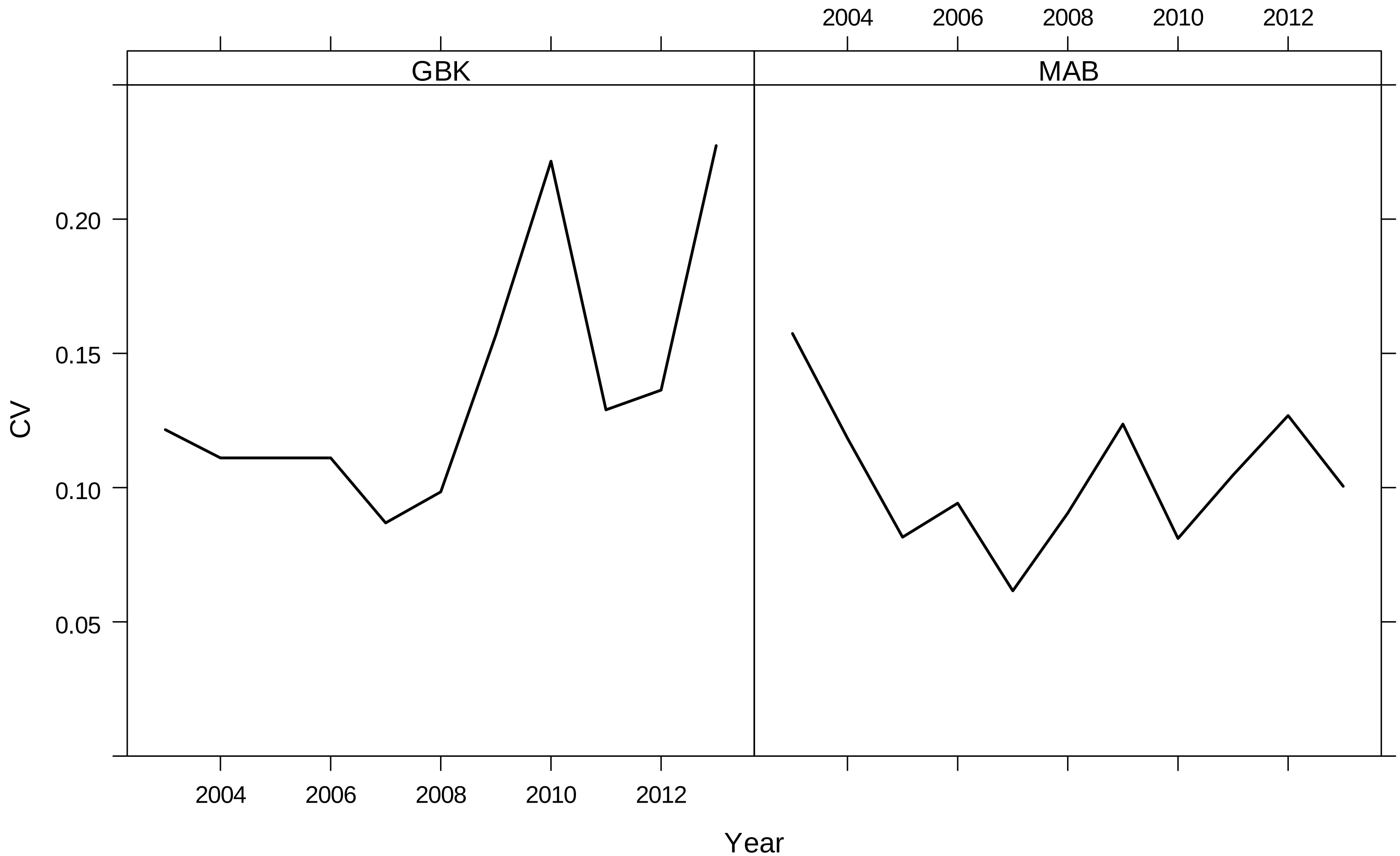


Figure 1. CVs for total mean catch per tow (all sizes) in the dredge survey during 2003-2013.

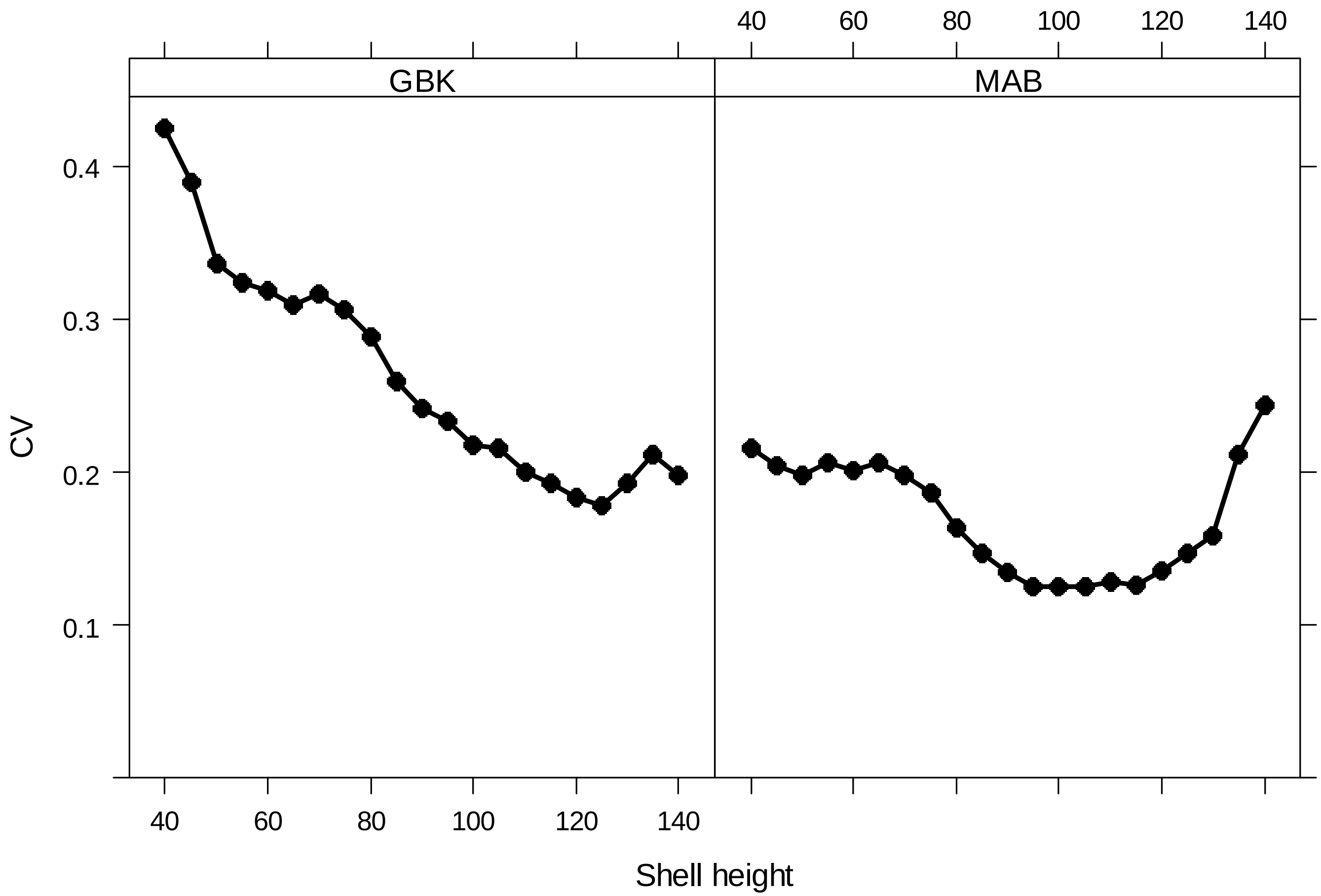


Figure 2. Average CVs for mean scallop catch per tow in the dredge survey during 1978-2013 by shell height size group and stock area.

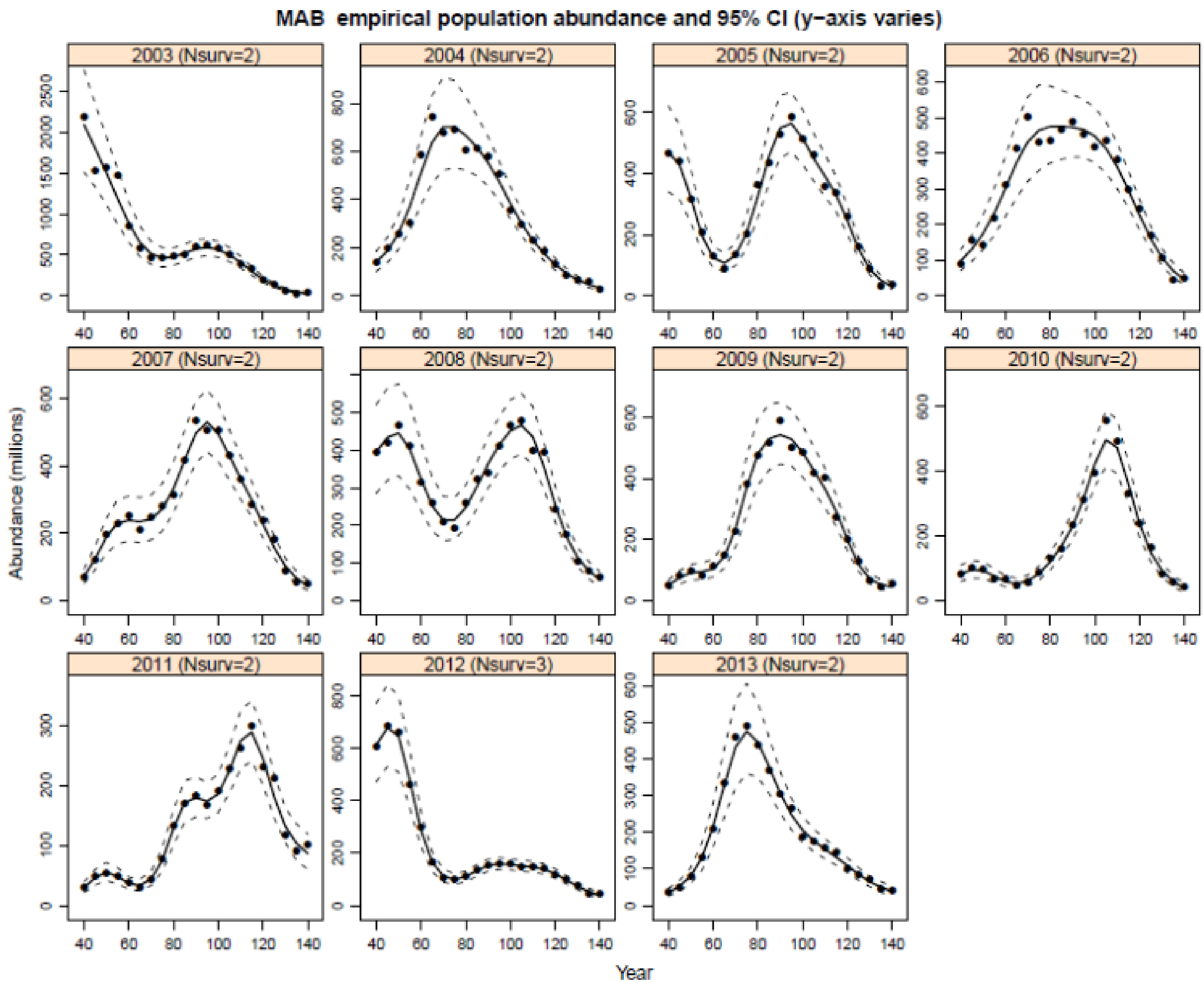


Figure 3a. Empirical abundance at length during 2003-2013 in the Mid-Atlantic region with approximate 95% confidence intervals. Note that the scales on the y-axis vary.

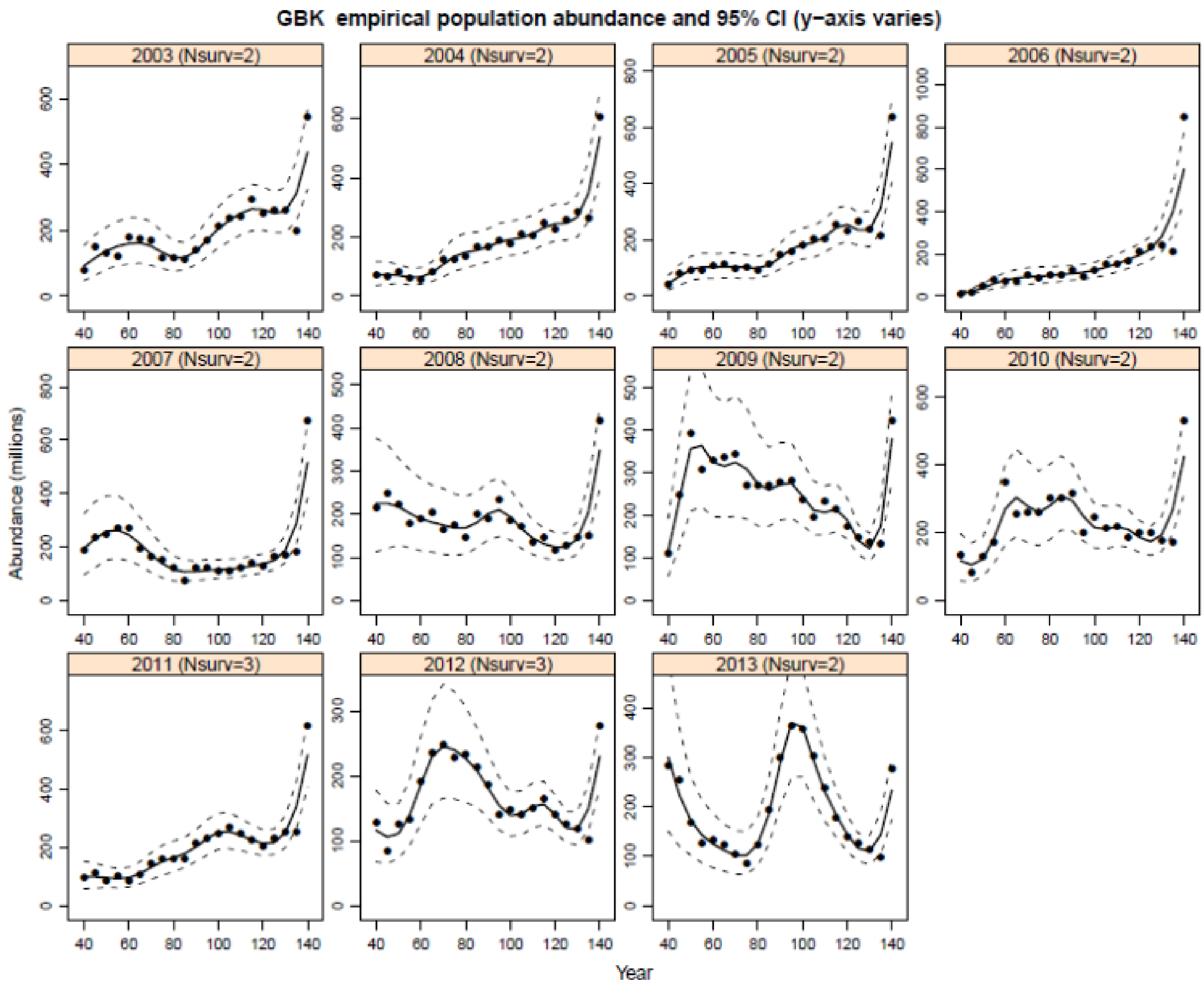


Figure 3b. Empirical abundance at length during 2003-2013 in the Georges Bank region with approximate 95% confidence intervals. Note that the scales on the y-axis vary.

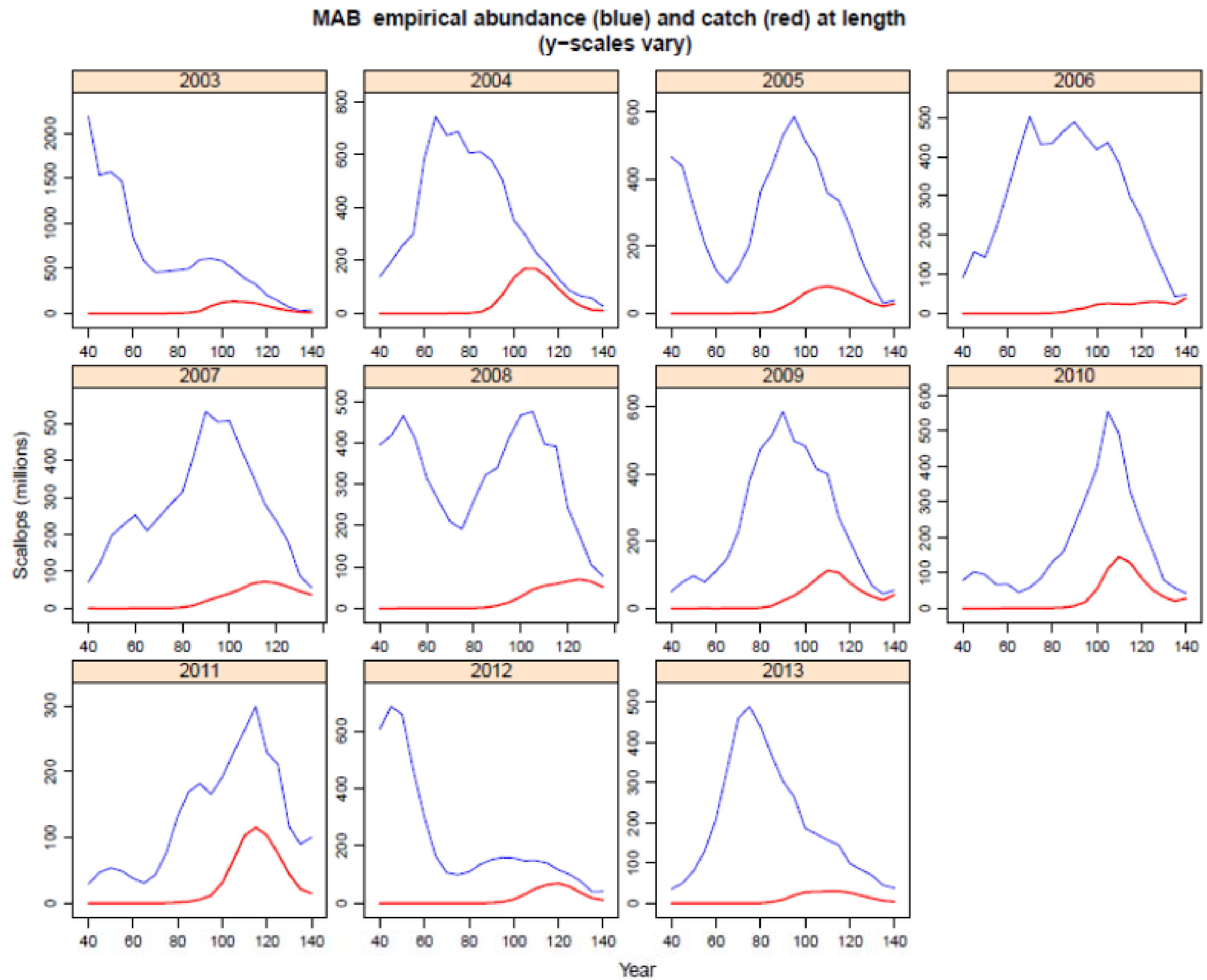


Figure 4a. Empirical abundance and catch at length during 2003-2013 in the Mid-Atlantic region. Note that the scales on the y-axis vary.

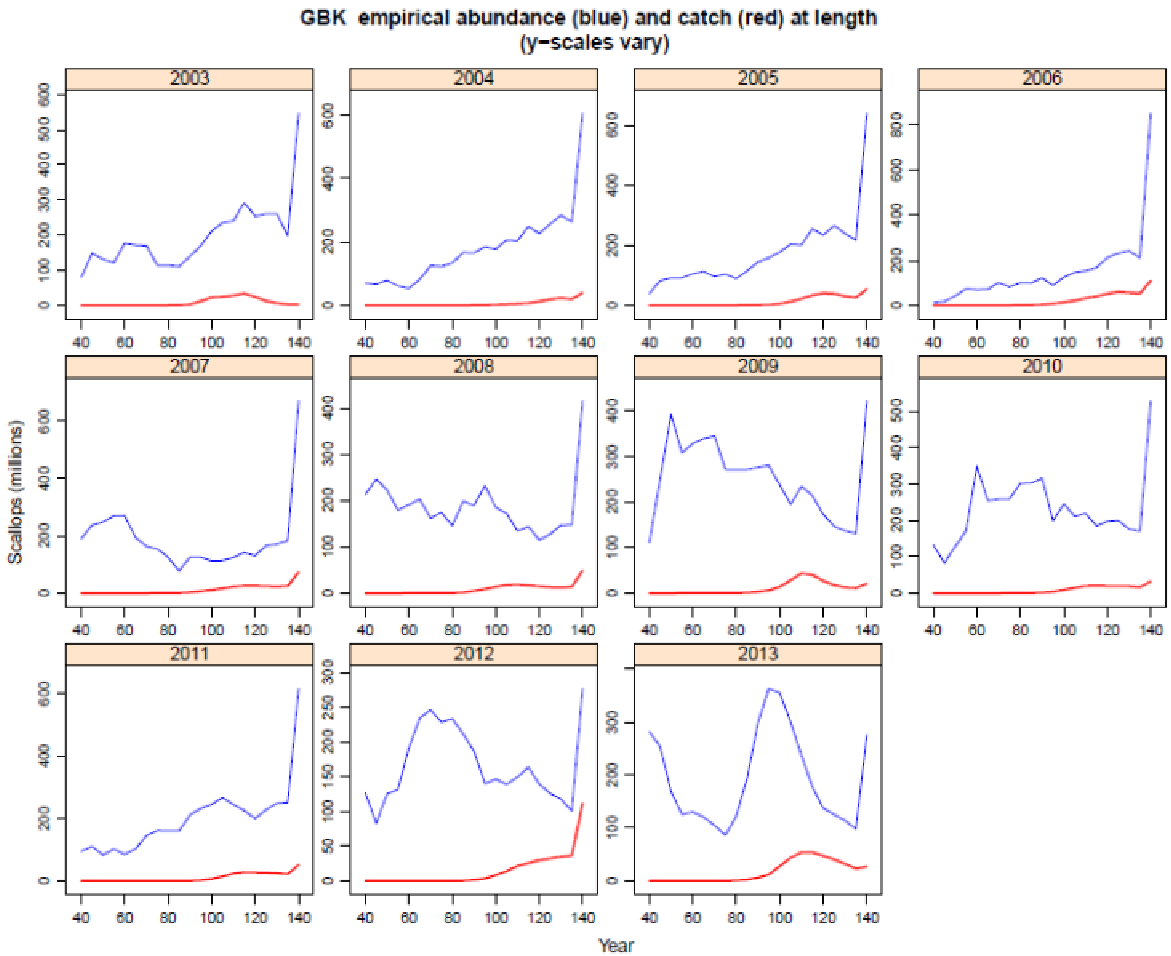


Figure 4b. Empirical abundance and catch at length during 2003-2013 in the Georges Bank region. Note that the scales on the y-axis vary.

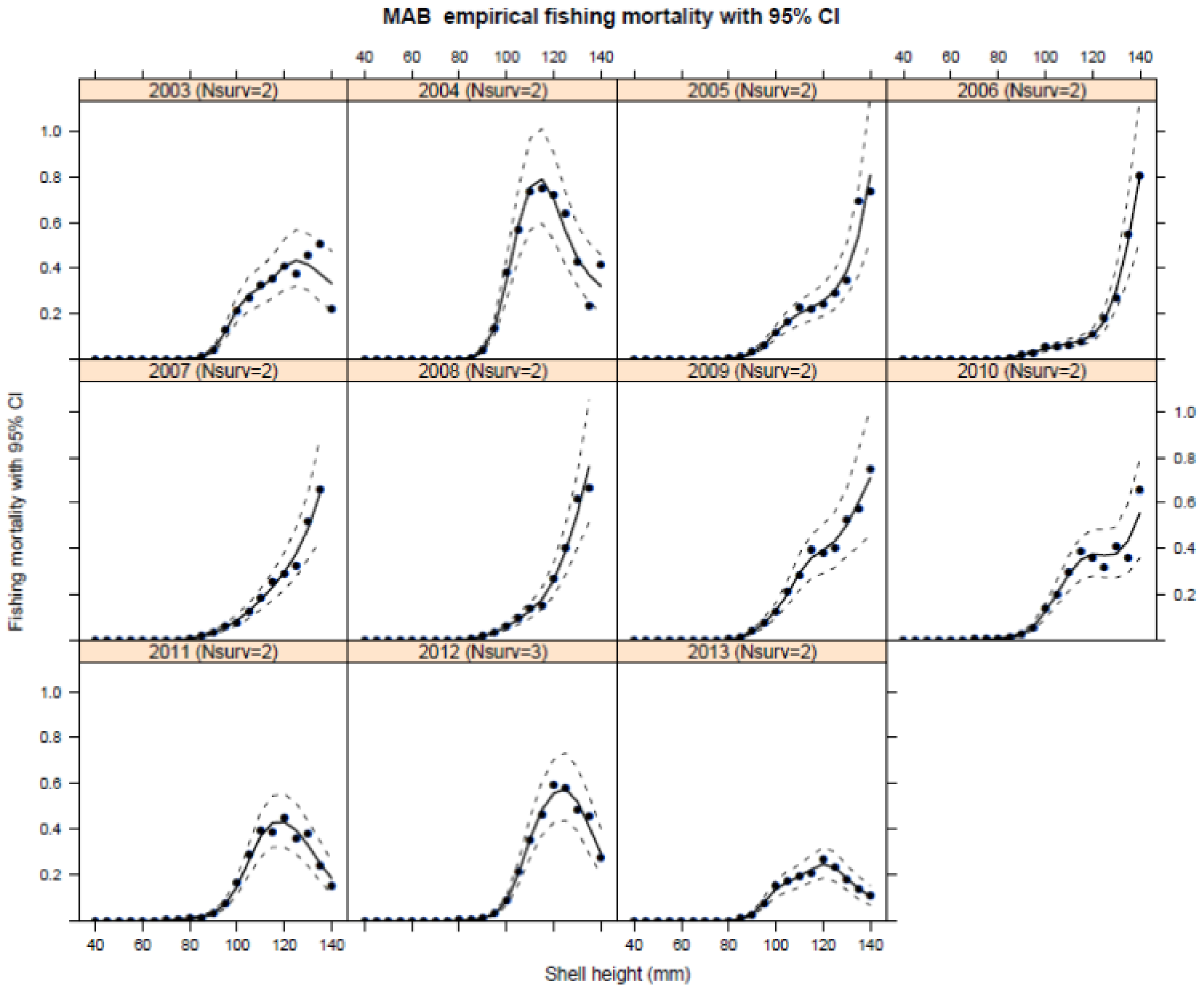


Figure 5a. Empirical fishing mortality at length during 2003-2013 in the Mid-Atlantic region with approximate 95% confidence intervals. Note that the scales on the y-axis differ (fishing mortality was typically higher in the Mid-Atlantic region).

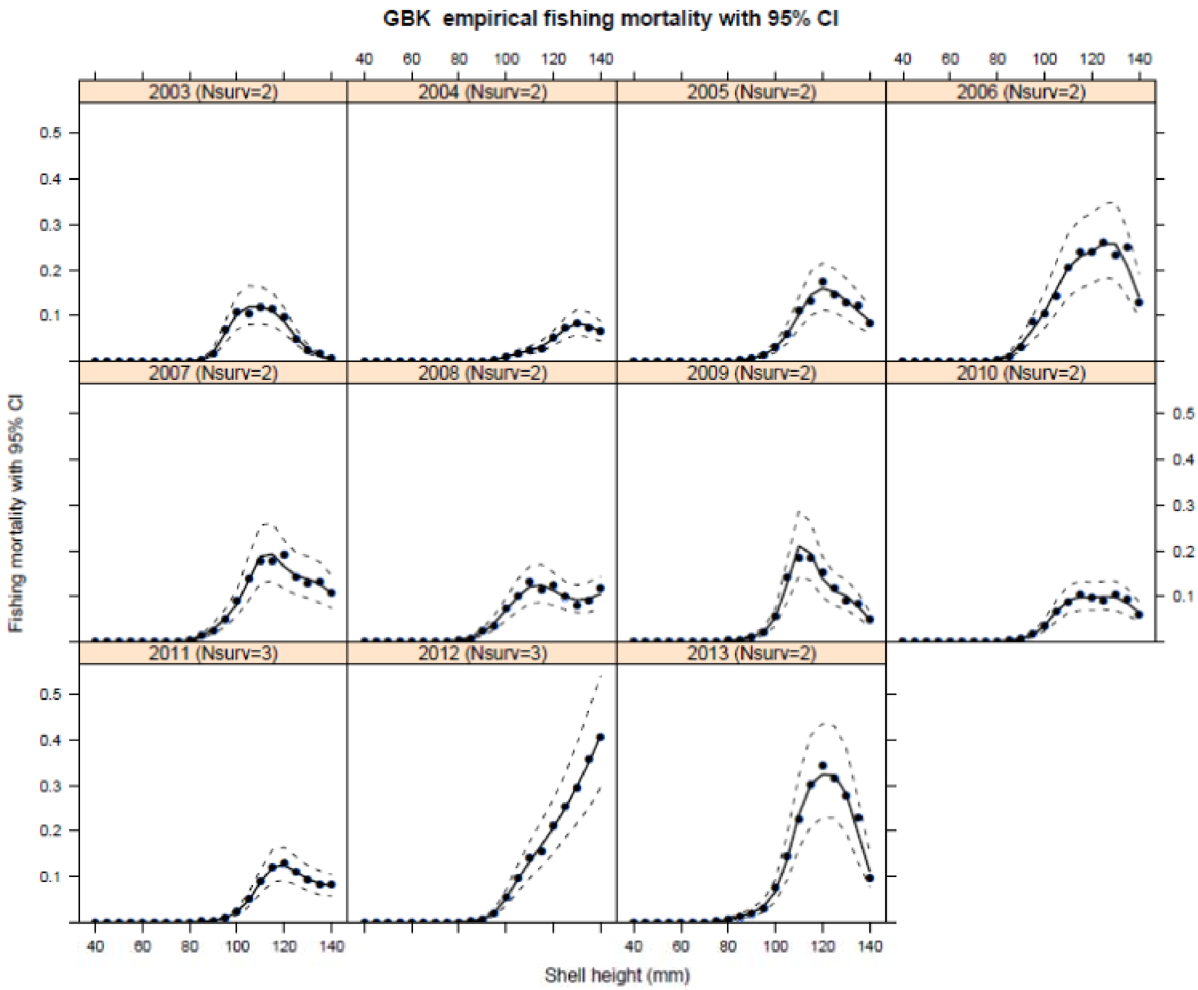


Figure 5b. Empirical fishing mortality at length during 2003-2013 in the Georges Bank region with approximate 95% confidence intervals. Note that the scales on the y-axis differ (fishing mortality was typically higher in the Mid-Atlantic region).

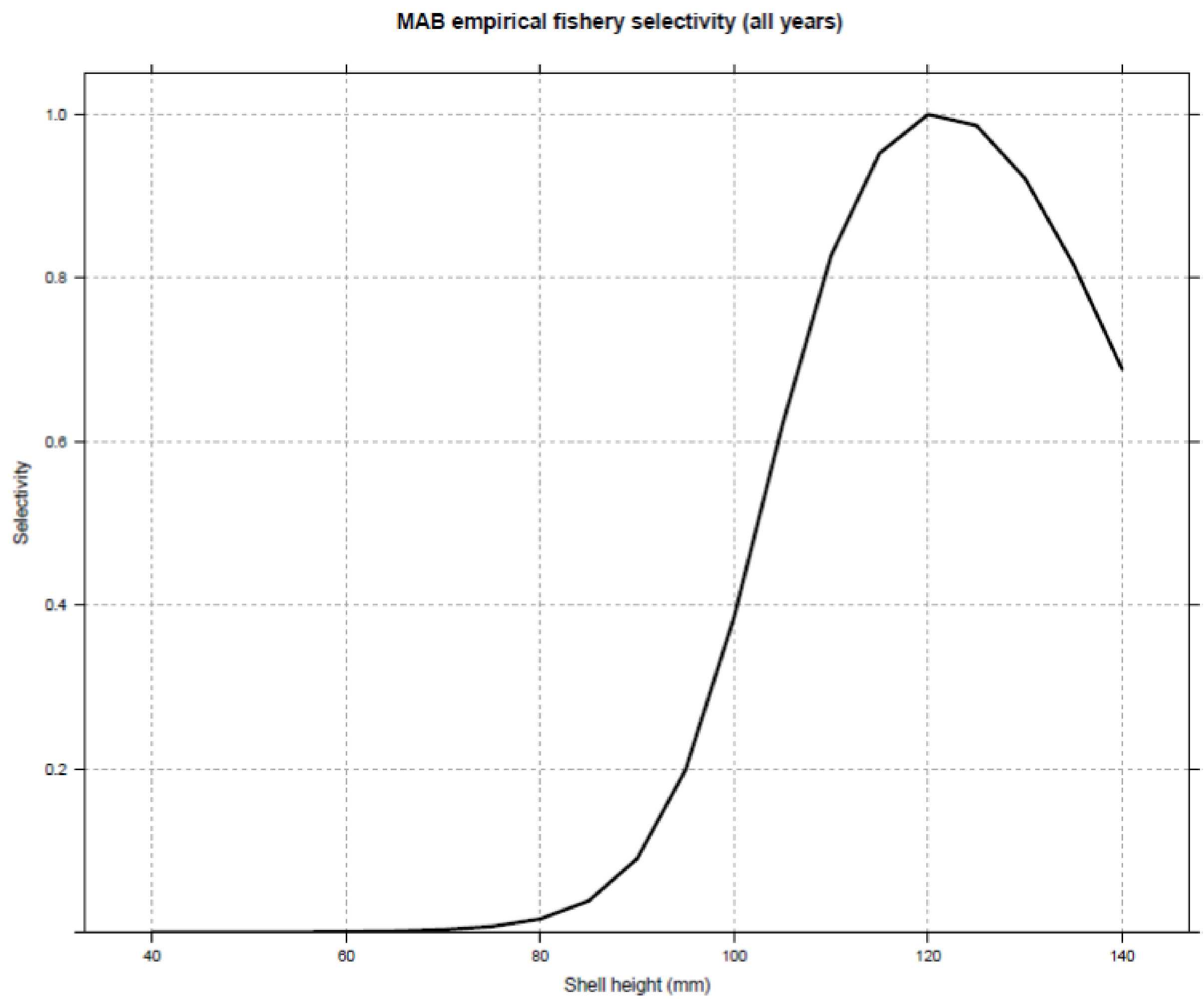


Figure 6a. Empirical estimates of average size selectivity for the scallop fishery during 2003-2013 in the Mid-Atlantic region. This curve was calculated by pooling data for different years and fitting a single line to show the trend. Another approach is to average the fitted selectivity curves for each year.

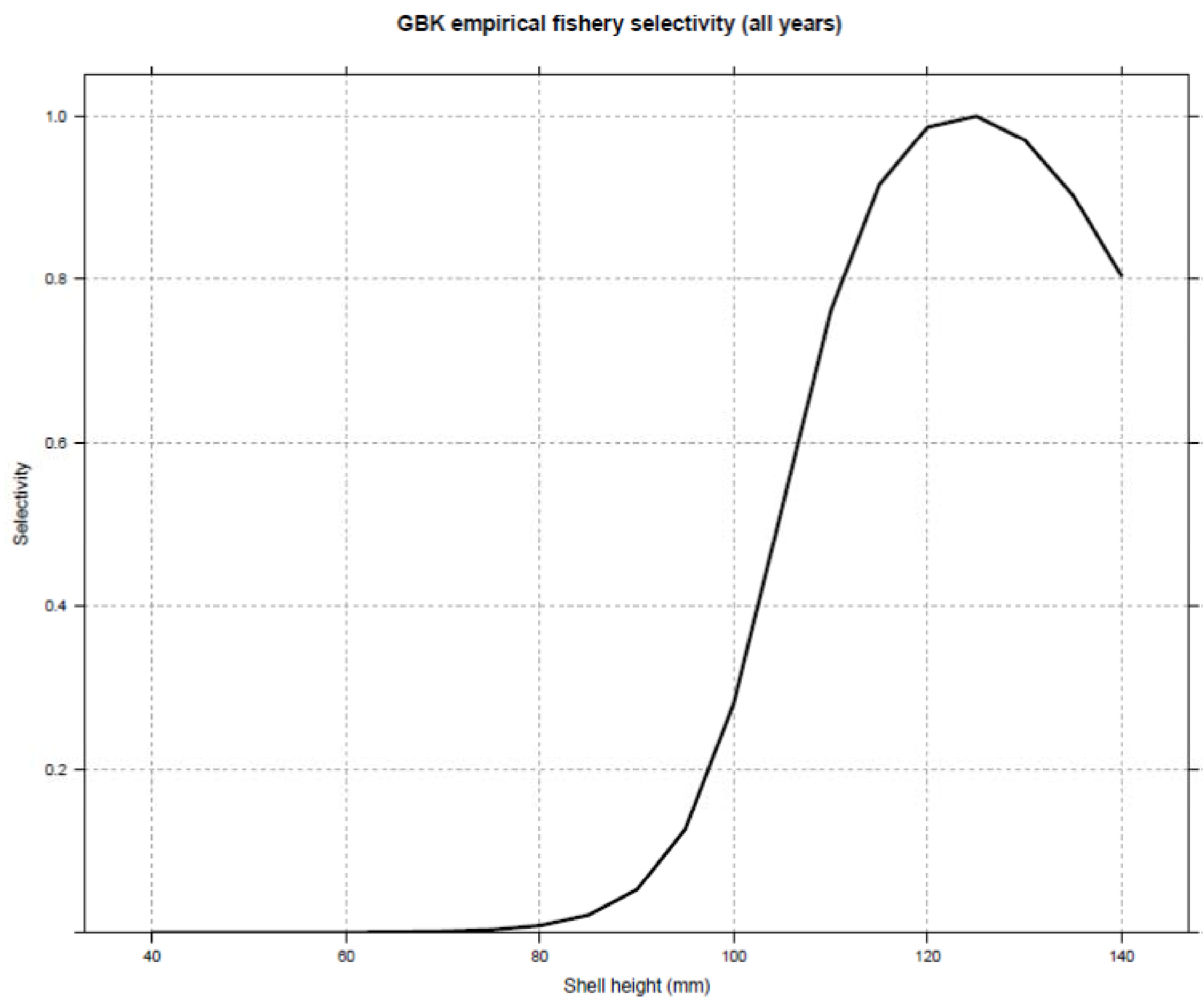


Figure 6b. Empirical estimates of average size selectivity for the scallop fishery during 2003-2013 in the Georges Bank region. This curve was calculated by pooling data for different years and fitting a single line to show the trend. Another approach is to average the fitted selectivity curves for each year.

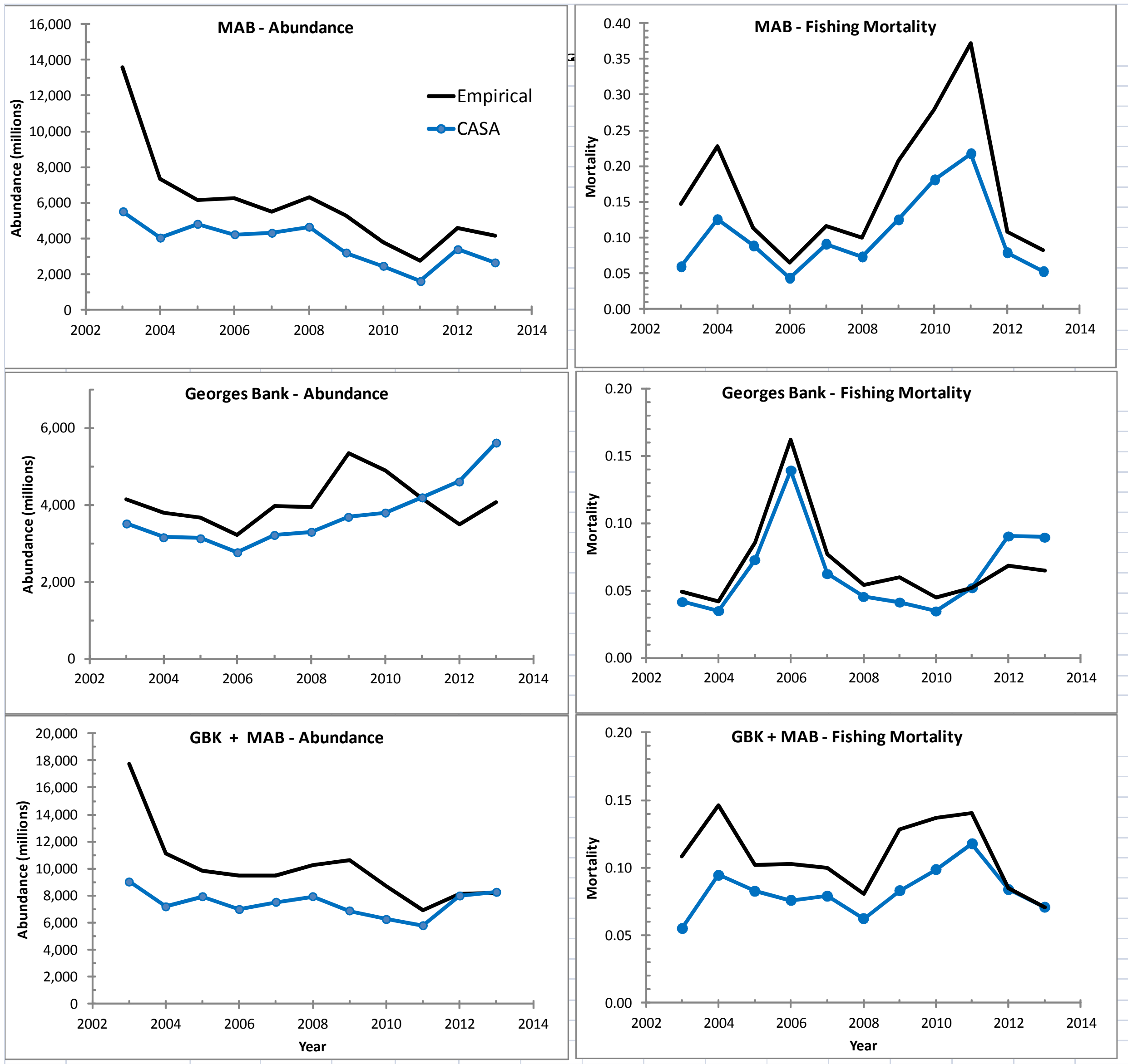
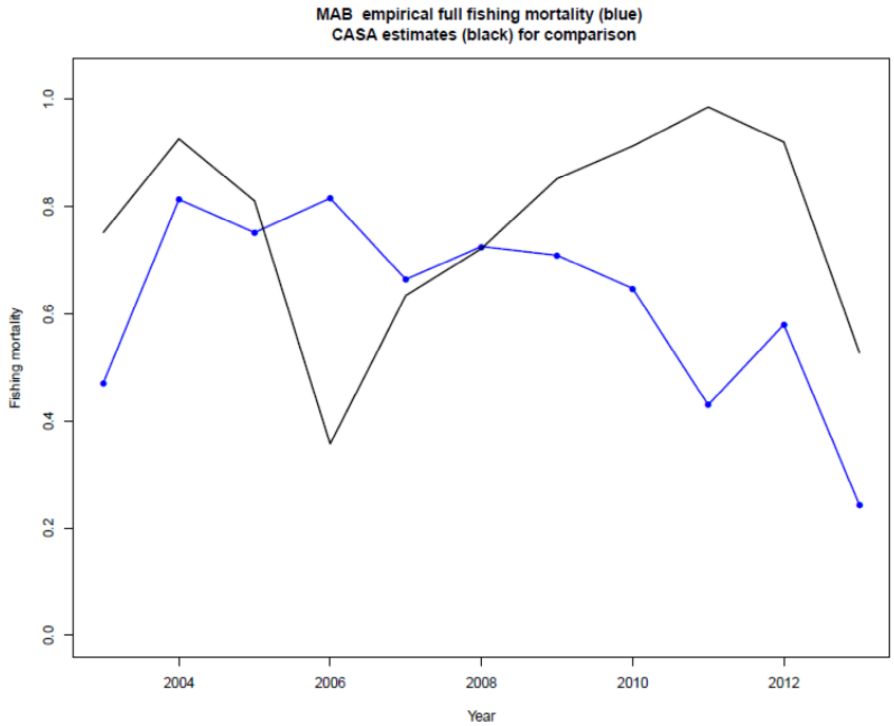


Figure 7. Abundance (left) and fishing mortality estimates (right) from the empirical method and the CASA model during 2003-2013 for the Georges Bank (top), Mid-Atlantic (middle) and combined (bottom) regions.



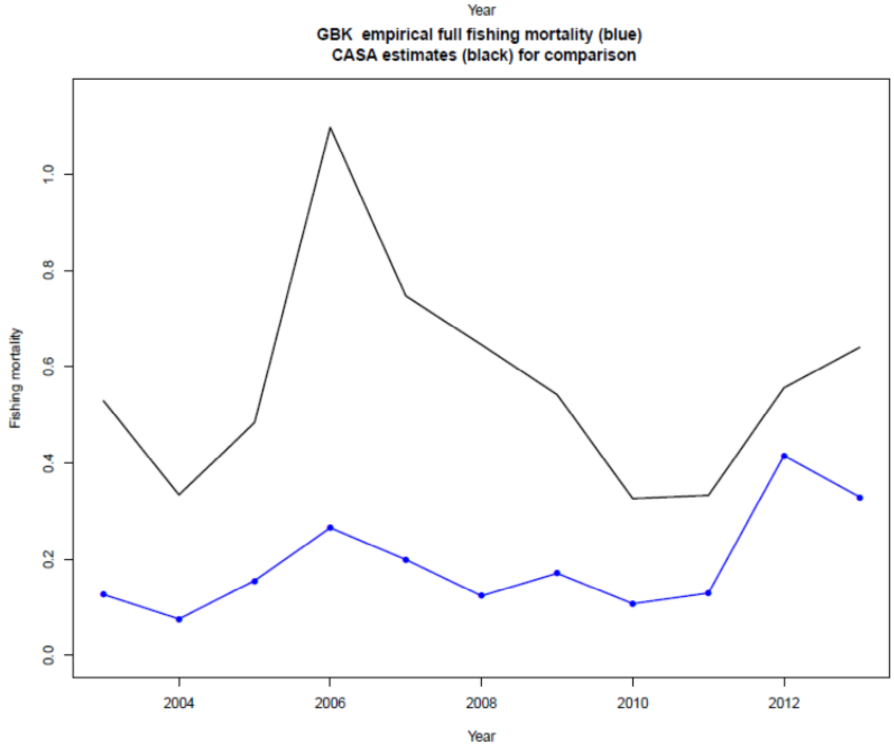


Figure 8. Fully recruited fishing mortality estimates for the Mid-Atlantic (top) and Georges Bank (bottom) regions. The empirical estimates are in blue, CASA estimates are black.

Appendix B6. NEFSC HabCam survey for sea scallops: survey design, implementation, and data analysis.

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This report contains five stand-alone sections that together describe HabCam gear and operations, simulation work used to develop and test survey designs, how the actual surveys during 2011-2013 were carried out, and how abundance estimates and size composition for 2011-2013 used in assessment models were made.

1. Introduction to the HabCam survey

HabCam is an underwater vehicle that was originally developed through collaboration of commercial fishermen, independent scientists, and staff at the Woods Hole Oceanographic Institute as a vehicle for documenting the size and abundance of benthic / demersal organisms and mapping sea floor habitats. The vehicle is towed behind a vessel, while actively "flown" ~2m off the bottom by a pilot. It collects overlapping, downward facing digital still imagery. Between 2005 and 2010, the HabCam group developed and improved this technology and successfully performed a number of surveys on the Mid-Atlantic continental shelf, Georges Bank and in the Gulf of Maine, primarily using the HabCam_V2 vehicle which preceded the current design. The development of the vehicle and many of these surveys were supported by the Sea Scallop Research Set-Aside program and the vehicle proved to be appropriate technology for assessing sea scallops (NEFSC 2010). In 2009 a paired HabCam / dredge experiment was conducted to determine the capture efficiency of the NEFSC survey dredge (probability of catching a scallop in the path of the dredge) and in 2011 the HabCam_V2 was used in the NEFSC scallop survey to get an estimate of the entire scallop resource on Georges Bank.

With an interest in making a HabCam-type survey a standard part of the sea scallop assessment survey, NEFSC secured funds from NOAA Office of Science and Technology and contracted WHOI to build a vehicle for NEFSC's use. This vehicle, (HabCam_V4 or NOAA HabCam), completed resource-wide surveys in 2012 and 2013, beginning a new assessment time series for sea scallops that is used for the first time in this assessment. The HabCam_V4 vehicle is equipped with stereo digital still cameras, altimeters, and a compliment of oceanographic sensors including temperature, salinity, water spectrometer, 3D side-scan sonar, and optical sensors for dissolved oxygen, cdom, and turbidity.

2. Survey Design

Because the HabCam vehicle collects a constant track of images, data derived from the images are autocorrelated and not appropriate for analysis as a random or stratified survey. Resource assessments from such data are typically use spatial models including Generalized Linear Models (GLMs) and Generalized Additive Models (GAMs) or geostatistical methods

4 First and second coauthors alphabetical.

such as kriging (Rivoird et al. 2008). Literature on sampling designs for this type of survey comes primarily from literature on acoustic surveys. With geostatistical methods, the uncertainty in the estimate at any given location increases with distance from the survey track. As a result, evenly spaced grids are optimal for acoustic surveys as the distance from the survey track is minimized with even spacing. A second common survey design is a two-stage approach where a low resolution survey is first performed to determine the location of high-density aggregations and a second high-resolution survey is conducted on the aggregations. The results from two-stage surveys are post-stratified to account for spatial heterogeneity in survey effort. In both cases, geostatistical methods assume that the mean and variance is homogeneous throughout a survey stratum.

The HabCam sea scallop survey differs from these situations because adult sea scallops are relatively easy to detect and intensively surveyed, do not move long distances, and because spatial heterogeneity is primarily driven by management measures and known habitat affinities. While geostatistical methods assume a landscape with a stationary mean (Figure 1a), a landscape with a higher mean along the center of the landscape (Figure 1b) is more realistic for sea scallops because densities typically decrease in habitats deeper and shallower than the optimal habitat for a region (Figure 1c). In this case, it may be advantageous to increase sampling effort in the core habitats along the center of the survey area. Given a survey track of evenly-spaced transects of equal length (Figure 2a) and assuming an underlying variogram model, we can derive a map of kriging variances for the survey at each location in the landscape (Figure 2b). If the mean density is higher in the center of the landscape instead of stationary, we may assume that the standard deviation of the mean is proportional to the mean (similar to a Gamma distribution) and calculate an adjusted kriging variance for each location as:

$$AdjVar_{x,y} = KrVar_{x,y} * [e^{(CE)}]^2 \quad (2.1)$$

Where $AdjVar_{x,y}$ is the adjusted variance of the estimate at a given location, $KrVar_{x,y}$ is the unadjusted variance at the location and $e^{(CE)}$ is the magnitude of the center effect from Figure 1c.

As a proof of concept, we used geostatistical simulation to examine the effect of allowing the mean (and variance) to vary across the shelf and longitudinally along the shelf. We first simulated varying the mean across the shelf and examined how the survey variances were affected by varying (1) the proportion of the effort concentrated along the center of the survey area and (2) the length of the survey track. We modeled the cross-shelf gradient as a double-logistic with higher densities along the center of the study area and the amplitude of the center effect varying from 0 (no effect) to 1 (variance is e^2 or 7.38 times higher along the center of the study area (Figure 3). To assess the effect of increasing sampling intensity along the center of the study area, we decreased the length of alternating transects (range from 0 – 100% of the total width of the study area) and increased the total number of transects to keep the total survey track length constant (Figure 4). We then varied the total survey track length from 1,000 to 4,000 pixels. For each simulation, we examined the resulting variance maps (i.e. Figure 5) and used the sum of the adjusted kriging variance (eq. 2.1) as a relative proxy for the variance of the survey. While this is not the true variance of the survey, as the variances are correlated across the landscape, we are not aware of established methods for calculating a kriging variance for survey areas with non-stationary variances and this should be an effective relative measure for comparison purposes.

The adjusted kriging variances varied across center effects and transect lengths (Figure 6). Optimal short transect lengths decreased as center effects increased and increased as total track length increased. The center effect and total track length interacted to produce an optimal short transect length. With a track length of 1,000 pixels, increasing the center effects from 0 and 1 decreased optimal short transect length from 67% to 30%. However, for track lengths of 4,000 pixels, varying the center effect from 0 to 1 only decreased optimal short transect length from 92% to 85%.

For a second simulation, we examined the effect of the mean and variance varying longitudinally along the survey area (i.e. zonal anisotropy, Figure 7). The zonal effect was implemented by dividing the landscape into two zones (upper and lower) and adding an additional, longitudinally-oriented logistic trend to the landscape. We then varied the amplitude of the longitudinal effect (Zone effect) the spacing of adjacent transects between the two zones, and total track length (Figure 8).

The optimal solutions for landscapes with Zone effects placed more transects in the zone with higher underlying means and variances (Figure 9). The effect was most notable for shorter total track lengths, increasing transect density in the higher mean zone by as much as 300% over the lower mean zone.

3. Survey Area and Design for Actual HabCam surveys

The above simulations indicate that the variance of a survey can be decreased by alternating the length of survey transects and increasing transect density in areas with known higher abundances. These simulation results are used informally in the design of each year's survey but actual survey design is based on researchers' knowledge of where the current stock biomass and incoming cohorts are.

The two stock areas (MAB and GB) are each divided into multiple subregions, based on changes in habitat type, habitat orientation (anisotropy), and management boundaries (Figure 10 and 11). These subregions are used both for designing the survey and for abundance estimation from the resulting survey data.

The extent of the survey area is based on an updated analysis of biomass patterns from the NMFS dredge and RSA surveys, Vessel Trip Reports, sea scallop observer trips, and Vessel Monitoring System data. In general, the current extent of the dredge survey was found to be very adequate for covering the scallop resource, though small areas were added to the extent of the HabCam survey to capture areas where there was evidence of adequate scallop densities or commercial activity.

The survey tracks are constructed in one long track for the MAB and three separate tracks for GB. Each track is bounded by a set of subregions. A midline, drawn along the center of biomass, runs through each set of subregions. Survey transects are centered around and oriented orthogonal to the midline.

3.1 Software and procedures used in designing HabCam surveys

In designing actual surveys, specialized software prompts a user to enter the total effort (survey days) to allocate to a track, the relative lengths of the short transects on the track, and the transect density offset for each subregion along the track. The software varies the relative transect densities and provides a number of alternative tracks of similar lengths for the user to choose among, based on appropriate allocation of effort across the subregions, how well each track works around complex bathymetric structures, and other logistical considerations.

4. Image Acquisition, Processing and Annotation.

The HabCam vehicle is towed along the survey track at speeds from 6 – 7 knots while a pilot maintains the unit at an altitude of ~2m off the bottom. Digital still images are generally collected at a sufficiently high frequency that ~35% of adjacent images overlap. Collected images are initially stored as raw TIFF-formatted images. The raw TIFFs are then light-field and color corrected to improve image quality and saved in processed PNG format. Each image is named with a unique identifier and metadata for each image is recorded including longitude, latitude, time, vehicle depth, bottom depth, and vehicle altitude, roll, and pitch as well as the data from the oceanographic sensors. The altitude of each image is critical for determining the field of view of the image and measuring objects in the images. As altitude can be measured in multiple ways, the value used for a particular image is based on the following list ordered by expected accuracy:

1. Altitude as measured via disparity mapping (parallax) from the stereo images
2. Altitude as measured by the altimeters on the vehicle
3. Altitude inferred from the side-scan sonar

The metadata associated with each image is then stored in a PostgreSQL database and used for selecting images for annotation.

We select blocks of images for annotation, termed "assignments", based on the spatial extent of the image set and a target image density. Based on the desired density of images to be annotated, we break the survey track into equal length segments and select one image from each segment. Individual image selection is biased towards preferred vehicle heights (Gaussian-weighted, based on known issues with water turbidity or other factors that affect image quality) but image selection is otherwise random within each segment.

The selected image list is uploaded to the Postgres database for direct observation and annotation using a web-based annotation tool. Additional assignments may be created once an assignment is completed if additional images are desired from the same region. In such cases, we first remove all images from a buffered region around each image that has already been annotated from the pool of available images before the next random subset of images is selected. The goal of this is to keep the density of annotated images consistent within subregions along the track.

Data on the abundance, size and behavior of scallops are extracted from each image using an online annotation tool developed by collaborators at WHOI (Figure 12). Only scallops where the center of the scallop is judged to be inside the image are enumerated. Scallops larger than about 35mm (age 2+) are measured by drawing a line over the shell while smaller scallops are only marked with a point and counted. Additional data are recorded including confidence in identification, swimming, dead, clappers, etc. Image quality may be poor due to turbid waters, extremely high or low altitudes, image corruption, or other objects obscuring the bottom. In this case, the image can be noted as poor quality and data from this image excluded from derived data sets. All annotations, as well as comments on image quality and sediment types, are recorded directly to the Postgres database by the annotation tool.

Because scallops are not always oriented normal to the camera or may be partially obscured, scallops measurements are either shell heights (umbo to opposite margin) or widths (lateral margins), whichever is judged to be more accurate. Shell widths are converted to shell heights using a statistical model derived from paired measurements of scallops that were well oriented to the camera:

$$\text{Shell\_height} = 3.538 + 1.034*(\text{Shell\_width}) - 0.0003502*(\text{Shell\_width})^2 \quad (4.1)$$

Shell height is calculated in pixels based on the start and end coordinates of the annotated line. The size of each pixel in an image is calculated from the altitude of the associated image, based on tank calibration experiments, and this pixel size is used to convert the shell height to actual millimeters. The altitude is also used to calculate the field of view for each image for density calculations.

For estimating size frequency distributions and abundance for each year, we constructed standardized data sets from the database and posted them to a common location on a network drive. The annual data sets include data from both the NEFSC HabCam surveys and from the HabCam group RSA surveys, which have to be drawn from multiple databases and corrected individually for problems in altitude measurements or other issues. The data sets include the metadata from all annotated images of acceptable quality, plus the classification of all scallops observed in each image and calculated lengths of for any scallop measured with a line segment.

5. Model-based estimation of sea scallop abundance and biomass

5.1 Introduction and summary

The goal of this section is to assess different model-based methods for estimating total abundance and biomass from HabCam and then apply these methods to HabCam data for 2011 - 2013 data to estimate abundance, biomass and size composition of sea scallops in the Georges Bank (GB) and Mid-Atlantic Bight (MAB) assessment regions (Figures 14 and 15). We also present design-based method (stratified mean) for this data set as an alternative to model-based methods and use it to validate the model-based estimates and CV's.

Scallop abundance or biomass data from HabCam are highly spatially autocorrelated and zero inflated, reflecting the patchiness of scallop distributions and the continuous nature of the observations. Thus, model-based estimation methods might be required to extrapolate observations along the observed track to larger areas. We used 2013 HabCam biomass data to test 3 geostatistical models: (1) ordinary kriging on spatially averaged data (OK), (2) zero-inflated Generalized Additive Models on spatially averaged data with kriged model residuals (GAM+OK), and (3) zero-inflated Generalized Additive Mixed Models where small scale variations are treated as random effects, combined with kriged model residuals (GAMM+OK). Effects of scale (neighborhood) size to average the data or scale of random effects was also evaluated. Co-located survey data from other gear types (dredge surveys from NEFSC and VIMS and video surveys from SMAST) were used for model validation. No single modeling approach and scale was consistently superior but GAM+OK performed better than OK and GAMM+OK in general.

We then conducted a simulation to evaluate performance of the 3 model-based methods along with a design-based method (stratified mean method, SM) and effects of scale size for data averaging and random effects. The GAM+OK method with small scale size outperformed the other 2 model-based methods and scale sizes in the simulation in terms of accuracy and precision of estimating mean and CV in most cases. SM estimates were more accurate and precise than the model-based estimates but only when the study region was stratified more correctly than might be expected in practice.

Based on the results of 2013 HabCam biomass data analysis and simulations, we selected the GAM+OK method to estimate scallop abundance and biomass for the GB and MAB stock for 2011 to 2013. SM estimates estimated with careful stratifications are also provided to back up the model-based estimates. Following are detailed descriptions of the simulation design, model- and design-based methods, simulation results, and procedures to estimate GB and MAB scallop abundance and biomass for 2011 to 2013.

5.2 Simulation Design

The area covered (domain) of simulated scallop populations was 50 km longitude and 100 km latitude (roughly the size of Hudson Canyon subregion, Figure 2) with a 100 m grid size. The scallop spatial distributions are non-stationary due to the influences of physical and biological environment including current, depth, and predator distributions (Brand, 1991). The simulated scallop population is therefore assumed to be heterogeneous in global trend (first-order effect), combined with stationary second-order effects. We simulated different first-order and second-order effects in order to test whether the abundance and biomass estimation methods are robust to the type of spatial distributions of the underlying population.

Variations in global mean quantity were simulated using a double logistic function

$$p_{i,j} = \frac{1}{1+\exp\left(-a\,(i-b)\right)} + \frac{1}{1+\exp\left(a\left(i-b+\frac{\max(i)}{2}\right)\right)}, \qquad (5.1)$$

where a and b parameters determine the shape of the logistic curve, and i and j are the longitude and latitude, respectively. The simulated first-order effects are high in the middle and decrease logistically toward the left and right edge of the simulation domain (Figure 16). Two types of first-order effects were simulated, one narrow but highly dense and the other wide and less dense (Figure 16).

Second-order effects were simulated as stationary Gaussian random fields with a spherical isotropic covariance structure (Cressie 1993)

$$\gamma(h) = \begin{cases} 0 & h = 0 \\ c_0 + c_1\{\frac{3}{2}\frac{h}{r} - \frac{1}{2}(\frac{h}{r})^3\} & 0 < h \leqq r, \\ c_0 + c_1 & h \geqq r \end{cases} \quad (5.2)$$

where c_0, c_1, and r are the nugget, partial sill, and range parameter, respectively. The nugget/sill ratio ($\frac{c_0}{c_0+c_1}$) determines randomness and r determines aggregations size of the second-order effects. We simulated combinations of 2 levels of nugget/sill ration and 2 levels of the range parameter resulting in 4 types of second-order effects: small aggregation, large aggregation, small aggregation with a large random noise, and large aggregation with a large random noise (Figure 17). We chose the parameter values based estimates from actual HabCam data.

Scallop distributions are patchy, resulting in HabCam data being highly zero-inflated (Table 1). To reflect the patchiness of scallop distribution, for each second-order realization, densities smaller than 90th percentile were set to zero. The zero-inflated second-order effects were combined with first-order effects to produce realistic simulated scallop distributions (Figure 18).

We simulated combinations of 2 first-order and 4 second-order effects resulting in 8 types of simulated population distributions. Thirty realizations were generated for each population type. Total abundance and biomass of each realization was scaled to equality across realizations. Each realization was surveyed using 30 different tracks. Shape and direction of tracks was designed to mimic the actual HabCam survey design.

Model-based and designed-based methods were used to estimate total biomass and abundance for the simulated populations. These estimation methods were evaluated using percent bias and percent root mean square error (RMSE)

$$\%\ \text{Bias} = \frac{\frac{\sum_{i=1}^{n}(\hat{T}_i - \mu)}{n}}{\mu} \quad (5.3)$$

$$\%\ \text{RMSE} = \frac{\sqrt{\frac{\sum_{i=1}^{n}(\hat{T}_i - \mu)^2}{n}}}{\mu}, \quad (5.4)$$

where $\hat{T}_i$ is the estimated total biomass or abundance for sample set i, μ is the true population size, and n is the total number of sample sets analyzed. Percent bias and percent RMSE of CVs for the precision of model estimates were also evaluated. The method that produced the least biased and most precise estimates was selected to analyze the actual HabCam data.

5.3 Model-Based Estimation

Kriging is one of the most widely used geostatistical method for spatial interpolation (Webster and Oliver 2001). We tested performance of 3 different kringing methods including

OK, GAM+OK, and GAMM+OK on the simulated scallop populations. OK is a standard version of the kriging models with the assumption of a constant mean and consideration of variation and distance between sample points (Hengl 2009, Webster and Oliver 2001). Although the constant mean assumption might not be reasonable for scallops, the simulation tests are necessary to determine whether the observed non-stationary pattern can be modeled as an autocorrelation among errors with a constant mean or a trend with mean changing with variance.

Isotropy and anisotropy is the variation of scallop abundance or biomass being identical or directionally dependent. It is not clear whether the samples are isotropic or anisotropic although actual observations indicate that first-order effects the simulated populations should have the largest variations along the horizontal axis. Therefore, we built both the isotropic and anisotropic models and selected the final OK model using RMSE

$$\mathrm{RMSE} = \sqrt{\frac{\sum_{i=1}^{n}(\hat{z}_i - z_i)^2}{n}} \qquad (5.5)$$

Total abundance or biomass (T) and its variance were estimated as

$$\hat{T} = \mathrm{A}\sum_{i=1}^{n} \hat{z}_i \qquad (5.6)$$

$$\mathrm{Var}(\hat{T}) = \mathrm{A}^2 \sum_{i=1}^{n} \sum_{j=1}^{n} \mathrm{Cov}(\hat{z}_i, \hat{z}_j), \qquad (5.7)$$

where $\hat{z}_i$ is the kriging estimates at location i and A is the grid size.

Regression kriging (RK) extends the OK to account for a global trend, which can be estimated by apply a regression model (e.g. GAM or GLM) to a series of ancillary variables (e.g. depth, latitude or longitude) then applying OK to the residuals of the regression model (Hengl 2009, Odeh et al. 1995). The final predictions of RK are obtained by summing the regression predicted values and the kriged residuals. This approach was criticized by Cressie (1993) and Lark et al. (2006) because the variogram estimates of the random component of spatial variation are theoretically biased. Generalized least squares and residual maximum likelihood-empirical best linear unbiased predictor are two potential solutions (Lark et al. 2006). However, Kitanidis (1993) and Minasny and McBratney (2007) showed that although these methods are theoretically preferable to RK, they did not substantially improve model predictions. We therefore used the RK approach.

Scallop data from the HabCam survey are highly spatially autocorrelated and zero inflated, reflecting the patchiness of scallop distributions. Therefore, we estimated the first order effects (over relatively large geographic areas) using a two-stage hurdle model which models the probability that scallops are found in a sample (presence/absence) separately from the density given that at least one scallop was found (Barry and Welsh, 2002). Predictions from the two models are combined to make the complete estimates of abundance and biomass. Hurdle model results were usually modified further to account for second order effects over smaller geographic areas as described below. We tested a hurdle GAM on data averaged within segments along the tracks (to reduce the autocorrelation and zero-inflation) and a hurdle GAMM where the fine-scale variations within track segments were treated as random effects. A quasi-binomial distribution was assumed for the presence/absence model and a quasi-Poisson distribution for the positive model. The first-order effects were estimated using an interaction term of latitude and longitude for both GAM and GAMM. OK was performed on the residuals using the same algorithm described above. Total abundance and biomass of GAM+OK and GAMM+OK model estimates were estimated using

$$\hat{T} = \mathrm{A}\sum_{i=1}^{n} \hat{x}_i\hat{y}_i + \hat{z}_i, \qquad (5.8)$$

where $\hat{x}_i$ is the probability of presense estimate, y_i is the positive estimate, $\hat{z}_i$ is the kriged residual at location i. By assuming that $\hat{x}$ and $\hat{y}$ are independent, the variance of the $\hat{T}$ was calculated using

$$\mathrm{Var}(\hat{T}) = \mathrm{A}^2\left(\sum_{i=1}^{n} \mathrm{E}^2\hat{x}_i\mathrm{Var}(\hat{y}_i) + \mathrm{E}^2\hat{y}_i\mathrm{Var}(\hat{x}_i) + \mathrm{Var}(\hat{x}_i)\mathrm{Var}(\hat{y}_i) + \sum_{i=1}^{n}\sum_{j=1}^{n} \mathrm{Cov}(\hat{x}_i, \hat{y}_i)\right) \qquad (5.9)$$

Effects of segment length to average the data or determine random effects along the tracks was evaluated. The dense scallop aggregations occurred at approximately 400 to 900 m (NESFC 2010) and therefore we tested 3 segment lengths, 750, 1500, and 2,250 m. These segment lengths were also used to define the grid size A.

5.4 Design-Based Estimation

We tested a SM method to estimate total abundance and biomass from the simulated data. Only horizontal transects were used in the SM estimation because variance of these transects were different from the vertical transects. Horizontal transects were post-stratified into 2 strata based on high and low first-order effects (Figure 19). Mean and its variance of the simulated scallops (t) by segment (j) and stratum (i) were calculated by

$$\bar{t}_{i,j} = \frac{\sum_{k=1}^{n_{i,j}} t_{i,j,k}}{n_{i,j}} \qquad (5.10)$$

$$\mathrm{Var}(\bar{t}_{i,j}) = \frac{\mathrm{Var}(t_{i,j,k})}{n_{i,j}}, \qquad (5.11)$$

where $n_{i,j}$ is the number of images by segment and stratum. Total abundance and biomass estimates ($\hat{T}$) and variance were estimated as

$$\hat{T} = \mathrm{A}\sum_{i=1}^{2} \mathrm{S}_i \frac{\sum_{j=1}^{n_i} \bar{t}_{i,j}}{n_i} \qquad (5.12)$$

$$\mathrm{Var}(\hat{T}) = \mathrm{A}^2 \sum_{i=1}^{2} \mathrm{S}_i^2 \sum_{j=1}^{n_i} \frac{\mathrm{Var}(\bar{t}_{i,j})}{n_i^2}, \qquad (5.13)$$

where n_i is the number of segments by stratum i, and S_i is the size of stratum i.

The simulation domain was well-stratified based on the first-order trend; however, we do not have the same information when dealing with the real data which tend to complicated as shown below using real data. We tested whether the SM estimates are sensitive to the stratification by enlarging (Stratified Mean Wide, SMW) and shrinking (Stratified Mean Narrow, SMN) the central high density stratum by 20% (Figure 19) and estimated total abundance and biomass under the original (incorrect) assumptions about stratum size.

5.5 Simulation Results

Proportion of converged model runs was 99% for GAM+OK and OK but 80-93% for GAMM+OK (Table 2). Percent bias and percent RMSE showed that GAM+OK with data averaged by 750 m (scale) is the best way to estimate the scallop biomass among all the model-based methods. For abundance, the method that produce the least biased estimates is GAM+OK with 1500 m scale, though it only outperformed the GAM+OK with 750 m scale by 0.006%. When both the bias and precision of the estimates are taking into account, GAM+OK with 750 m is the best way to estimate the scallop abundance (lowest percent RMSE, Table **Error! Reference source not found.**). The GAM+OK with 750 m segments also produce the least biased CV estimates for both biomass and abundance estimates (Table 2).

Percent bias and percent RMSE of the SM estimates are smaller than all the model-based estimates (except for the percent RMSE of the abundances estimated using GAM+OK with 750 m) but the CVs were highly underestimated. Beside the problems of estimating CVs, SM estimates were sensitive to the quality of post stratification. SMW and SMN estimates were biased and worse than all the model-based estimates.

Based on the simulation results, we concluded that GAM+OK method with data averaged over 750 m segments was the best way to estimate total abundance and biomass using HabCam data. SM estimates with careful stratifications were also provided in order to validate the model-based estimates although variances for the SM method are probably understated.

5.6 Analysis of actual HabCam data for 2011-2013

The HabCam data were collected during 2011-2013 in GB and durng 2012-2013 in MAB. We divided the GB and MAB stock region into 14 subregions based on geographic characteristics and management areas and analyzed them separately because their topology, orientation and covariance structures differ (Figures 14 and 15).

Images with altitudes higher than 4 m and scallops with measured shell heights smaller than 40 mm were excluded for estimating scallop abundance and biomass. The shell height (SH) measures were converted to meat weights (g) (MW) based Hennen and Hart (2012)

$$\text{MAB: } MW = -16.88 + 4.64\log(SH) + 1.57\log(D) - 0.43\log(SH)\log(D) \quad (5.14)$$

$$GB: MW = 14.38 + 2.826\log(SH) - 0.529\log(D) - 5.98\log(L), \quad (5.15)$$

where D is depth and L is latitude. The counts and weight data were summed by image and standardized into abundance and biomass per m^2 by field of view of the image. A summary of the HabCam data used by subregion for 2011-2013 is listed in Table 1.

As described above and based on the simulation results, the GAM+OK method with 750 m segments was used to estimate total abundance and biomass for each subregion. For estimation purposes, we constructed a 1-km buffer zone around each subregion and used the data within the buffered region to build the subregional models. An average of weight or count (t) by image (j) and distance group (i) weighted by field of view (f) was calculated for every 750 m segment along the tracks

$$\bar{t}_i = \sum_{j=1}^{n_i} \frac{f_{i,j} t_{i,j}}{\sum_{j=1}^{n_i} f_{i,j}}, \qquad (5.16)$$

The $\bar{t}_i$ was weighted by both variation (s) and number of images (n) in the hurdle GAM using

$$w_i = \frac{s_i - s_{(1)}}{2(s_{(n_i)} - s_{(1)})} + \frac{n_i - n_{(1)}}{2(n_{(n_i)} - n_{(1)})} \quad (5.17)$$

A hurdle GAM with a quasi-binomial distribution for the presence/absence model and quasi-Poisson distribution for the positive model was used to estimate the first-order trend with respect to latitude, longitude, and depth. Depth is correlated with latitude and / or longitude in some of the subregions. To prevent potential problems cause by collinearity, latitude and longitude were transformed into composite variables: latitude plus longitude and half of the latitude or longitude plus longitude/latitude. A list of models with the different combination of covariates is supplied in Table 3. Depth is included in all of the candidate models because it is one of the most important variables that affecting scallop distributions. The maximum amount of knots for interactions between covariates in GAM models was limited to 15 (reduced to 10 for some of the subregions) and 10 for the single terms to prevent over-fitting. We selected the final first-order model using the RMSE from a 10-fold cross validation.

OK were performed on the GAM residuals. We tested isotropic and a series of anisotropic (from 0 to 180 by 20 degrees) residual OK models and selected the final OK model using the median standard error (MedSE).

$$\text{MedSE} = \sum_{i=1}^{n} \text{Median}(\hat{\bar{t}}_i - \bar{t}_i) \qquad (5.18)$$

GAM and OK final models by subregion and year are listed in Table 4.

For the SM analysis we used only the data within the subregion. The transects were separated into segments based on the following criteria: parallel or perpendicular to depth contour, distance between points (2 km), depth strata, and distance along the transect (10 km). We first separated transects into segments at locations where the direction of the transects changed between parallel and perpendicular to the depth contour. These segments were further separated into smaller ones by depth strata or any location where the distance of any two points in the segment was greater than 2 km. The resulting segments were again broken into smaller ones if length where segments were longer than 10 km. An example of segmentations of the HabCam data (abundance data for 2013) is in Figure 20.

Thresholds for the depth strata were estimated using a maximum likelihood based change point analysis (Killick et al. 2010). A GAM with a quasi-Poisson error distribution was built for each subregion. The depth partial residuals from the GAM were used in the change point analysis to estimate the depth thresholds. The thresholds were detected based on changes in mean or variance or both mean and variance of the partial residuals. Each subregion is post-stratified into a maximum of 3 depth strata. The depth stratification was done for each year by subregion and separately for abundance and biomass data.

The mean count or weight and its variance was estimated by segment and stratum using equations 10 and 11 and weighted by total field of view (f) and length of the segment (d) to estimate the total abundance or biomass and its variance

$$\hat{T} = \mathrm{A} \sum_{i=1}^{3} \mathrm{S}_i \sum_{j=1}^{n_i} \mathrm{w}_{i,j} \bar{t}_{i,j} \qquad (5.19)$$

$$\mathrm{Var}(\hat{T}) = \mathrm{A}^2 \sum_{i=1}^{3} \mathrm{S}_i^2 \sum_{j=1}^{n_i} \mathrm{w}_{i,j}^2 \mathrm{Var}(\bar{t}_{i,j}), \qquad (5.20)$$

where n_i is number of segments within depth stratum i, S_i is the size of depth stratum i, and $w_{i,j}$ is the weighting factor

$$w_{i,j} = \frac{d_{i,j} - d_{i,(1)}}{2(d_{i,(n_i)}) - d_{i,(1)})} + \frac{f_{i,j} - f_{i,(1)}}{2(f_{i,(n_i)} - f_{i,(1)})} \qquad (5.21)$$

The resulting GAM+OK and SM abundance and biomass estimates and CV's by subregion are listed in Table 5 and by stock in Table 6 for 2011-2013.

5.7 Size composition data for assessment modeling

Calculating scallop size frequency distributions from HabCam data for use in this assessment required re-stratifying Georges Bank for each year for appropriate spatial expansions because inclusion of the RSA surveys resulted in very high densities of annotated images in localized areas (Figure 13). A simple union of the sea scallop strata and HabCam estimation areas was sufficient for the Mid Atlantic in 2012 and 2013 as there were no RSA surveys in this region. Based on these stratifications, we derived stratified size frequency distributions by calculating the density of scallops within each strata and size class, weighted these densities by strata area, and averaging across the region. No adjustments for measurement errors were made although such measurement errors in the two optical surveys for sea scallops (HabCam and SMAST) may have standard deviations on the order of 1 cm. Instead, this type of error is accommodated in the CASA stock assessment model as predicted population length distributions are transformed into predicted length composition observations (Jacobson et al. 2010).

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Table 1: Sample size, percent zero, mean weight and count per m^2 of for HabCam data by regions during 2011-2013.

Stock	Year	Subregion	Sample Size	% Zero	Meat Wt (g/m^2)	Meat Ct (m^2)
GB	2011	CA1	1942	0.86	15.09	0.39
GB	2011	CA2_N	213	0.91	26.93	1.11
GB	2011	CA2_S	614	0.96	3.22	0.1
GB	2011	GSC_NW	1022	0.83	21.31	0.56
GB	2011	GSC_SE	677	0.97	24.67	0.42
GB	2011	NF	797	0.96	7.77	0.24
GB	2011	NLS	349	0.94	8.48	0.25
GB	2011	SF	554	0.99	2.34	0.08
GB	2012	CA1	660	0.91	6.18	0.35
GB	2012	CA2_N	1382	0.52	27.95	0.91
GB	2012	CA2_S	1415	0.93	3.34	0.12
GB	2012	GSC_NW	735	0.77	8.5	0.47
GB	2012	GSC_SE	276	0.94	5.42	0.23
GB	2012	NF	1486	0.82	22.84	0.75
GB	2012	NLS	298	0.87	5.85	0.26
GB	2012	SF	982	0.96	3.83	0.14
GB	2013	CA1	2054	0.95	1.54	0.07
GB	2013	CA2_N	1015	0.61	21.83	0.51
GB	2013	CA2_S	476	0.86	2.14	0.29
GB	2013	GSC_NW	953	0.86	2.99	0.15
GB	2013	GSC_SE	676	0.95	1.77	0.07
GB	2013	NF	1818	0.93	11.34	0.28
GB	2013	NLS	322	0.85	2.8	0.13
GB	2013	SF	491	0.84	2.05	0.3
MAB	2012	DMV_VB	753	0.9	0.84	0.11
MAB	2012	ET	665	0.85	1.28	0.19
MAB	2012	HC	1159	0.9	1.66	0.15
MAB	2012	HCnr	732	0.93	1.45	0.1
MAB	2012	HCsr	619	0.92	1.86	0.14
MAB	2012	LI	486	0.95	1.24	0.07
MAB	2013	DMV_VB	561	0.91	1.93	0.17
MAB	2013	ET	922	0.87	4.25	0.35
MAB	2013	HC	1114	0.96	2.02	0.18
MAB	2013	HCnr	657	0.95	1.55	0.08
MAB	2013	HCsr	585	0.96	1.7	0.14
MAB	2013	LI	608	0.96	1.55	0.08

Table 2: Percent bias, CV, percent RMSE, estimated CV and number of converged sample runs for biomass and abundance estimates by segment sizes and estimation methods.

Model Type	Scale	% Bias	CV	% RMSE	Estimated CV	# Runs	% Bias	CV	% RMSE	Estimated CV	# Runs
GAM	750	0.048	0.194	0.209	0.191	7194	0.038	0.167	0.177	0.16	7187
GAMM	750	0.088	0.19	0.225	0.308	6699	0.069	0.165	0.189	0.249	6106
OK	750	0.136	0.195	0.26	0.289	7196	0.098	0.173	0.214	0.241	7182
GAM	1500	0.052	0.276	0.295	0.173	7198	0.033	0.188	0.197	0.154	7195
GAMM	1500	0.088	0.192	0.227	0.465	6305	0.066	0.167	0.19	0.507	5774
OK	1500	0.173	0.385	0.484	0.272	7184	0.113	0.288	0.34	0.225	7194
GAM	2250	0.056	0.227	0.246	0.16	7199	0.036	0.206	0.198	0.156	7199
GAMM	2250	0.09	0.193	0.228	0.559	6342	0.063	0.206	0.19	0.651	5953
OK	2250	0.178	0.339	0.438	0.259	7199	0.126	0.206	0.415	0.213	7199
SM		-0.002	0.193	0.193	0.09	7200	0.001	0.206	0.181	0.064	7200
SMN		0.219	0.233	0.359	0.091	7200	0.168	0.206	0.294	0.068	7200
SMW		0.13	0.201	0.262	0.094	7200	0.085	0.206	0.216	0.067	7200

Table 3: List of GAMs tested in the 10-fold cross validation.

GAM Models
s(Longitude,Latitude,k=15)+s(Depth)
s(Latitude,Depth,k=15)
s(Longitude,Depth,k=15)
s(LatPlusHalfLong,Depth,k=15)
s(HalfLatPlusLong,Depth,k=15)
s(LatPlusLong,Depth,k=15)
s(Latitude)+s(Depth)
s(Longitude)+s(Depth)
s(LatPlusHalfLong)+s(Depth)
s(HalfLatPlusLong)+s(Depth)
s(LatPlusLong)+s(Depth)
s(Latitude)+Depth

Table 4: List of first-order and second-order models for the biomass and abundance estimates of GB and MAB subregions for 2011 to 2013.

Stock	Year	Subregion	GAM (Biomass)	GAM (Abundance)	OK (Biomass)	OK (Abundance)
GB	2011	CA1	s(HalfLatPlusLong) + s(Depth)	s(HalfLatPlusLong) + s(Depth)	No angle	Angle: 100
GB	2011	CA2_N	s(LatPlusLong) + s(Depth)	s(LatPlusLong) + s(Depth)	Angle: 0	No angle
GB	2011	CA2_S	s(Longitude, Depth, k = 15)	s(Longitude, Depth, k = 15)	No angle	Angle: 160
GB	2011	GSC_NW	s(Longitude, Latitude, k = 15) + s(Depth)	s(LatPlusLong) + s(Depth)	No angle	No angle
GB	2011	GSC_SE	s(Latitude, Depth, k = 10)	s(Latitude, Depth, k = 15)	No angle	Angle: 140
GB	2011	NF	s(LatPlusLong) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 60	No angle
GB	2011	NLS	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 120	Angle: 160
GB	2011	SF	s(Latitude) + Depth	s(LatPlusHalfLong, Depth, k = 15)	Angle: 160	No angle
GB	2012	CA1	s(HalfLatPlusLong) + s(Depth)	s(HalfLatPlusLong) + s(Depth)	Angle: 160	No angle
GB	2012	CA2_N	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 100	Angle: 60
GB	2012	CA2_S	s(Latitude, Depth, k = 15)	s(Latitude, Depth, k = 15)	Angle: 40	No angle
GB	2012	GSC_NW	s(Latitude) + s(Depth)	s(LatPlusLong) + s(Depth)	No angle	Angle: 0
GB	2012	GSC_SE	s(HalfLatPlusLong) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 60	Angle: 20
GB	2012	NF	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	No angle	No angle
GB	2012	NLS	s(HalfLatPlusLong) + s(Depth)	s(HalfLatPlusLong) + s(Depth)	Angle: 120	Angle: 80
GB	2012	SF	s(LatPlusLong, Depth, k = 15)	s(Longitude, Latitude, k = 15) + s(Depth)	No angle	Angle: 40
GB	2013	CA1	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Depth, k = 15)	Angle: 120	No angle
GB	2013	CA2_N	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	No angle	No angle
GB	2013	CA2_S	s(Longitude, Latitude, k = 15) + s(Depth)	s(Latitude, Depth, k = 10)	Angle: 0	Angle: 160
GB	2013	GSC_NW	s(Latitude) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 0	Angle: 0
GB	2013	GSC_SE	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 160	Angle: 20
GB	2013	NF	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 160	Angle: 160
GB	2013	NLS	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 0	Angle: 160
GB	2013	SF	s(LatPlusLong, Depth, k = 15)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 20	No angle
MAB	2012	DMV_VB	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 60	Angle: 60
MAB	2012	ET	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 80	No angle
MAB	2012	HC	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 160	No angle
MAB	2012	HCnr	s(LatPlusHalfLong, Depth, k = 15)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 140	Angle: 100
MAB	2012	HCsr	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 0	Angle: 60
MAB	2012	LI	s(Latitude) + s(Depth)	s(Latitude) + s(Depth)	Angle: 100	Angle: 0
MAB	2013	DMV_VB	s(LatPlusLong) + s(Depth)	s(LatPlusLong) + s(Depth)	No angle	Angle: 20
MAB	2013	ET	s(Longitude, Latitude, k = 15) + s(Depth)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 60	No angle
MAB	2013	HC	s(Longitude, Latitude, k = 15) + s(Depth)	s(LatPlusHalfLong) + s(Depth)	No angle	Angle: 120
MAB	2013	HCnr	s(Latitude) + s(Depth)	s(Latitude) + s(Depth)	Angle: 100	Angle: 40
MAB	2013	HCsr	s(Longitude, Latitude, k = 15) + s(Depth)	s(LatPlusHalfLong, Depth, k = 10)	No angle	No angle
MAB	2013	LI	s(LatPlusLong, Depth, k = 15)	s(Longitude, Latitude, k = 15) + s(Depth)	Angle: 0	Angle: 40

Table 5: Abundance and biomass and its CVs estimated using GAM+OK and SM methods by subregions for 2011 to 2013.

			Number (million)				Weight (mt)			
Stock	Year	Subregion	SM	GAM+OK	SM CV	GAM+OK CV	SM	GAM+OK	SM CV	GAM+OK CV
GB	2011	CA1	1151.70	1220.70	0.02	0.92	41772.14	42648.48	0.01	0.05
GB	2011	CA2_N	406.92	409.21	0.05	0.07	8325.85	12797.17	0.05	0.06
GB	2011	CA2_S	215.35	338.48	0.08	0.35	8882.94	10237.32	0.07	0.29
GB	2011	GSC_NW	1480.93	1289.01	0.04	0.17	32578.17	21675.43	0.04	0.15
GB	2011	GSC_SE	79.21	75.00	0.12	0.77	3578.14	2051.35	0.14	1.50
GB	2011	NF	336.35	201.78	0.10	0.09	5002.90	4631.38	0.08	1.70
GB	2011	NLS	218.19	159.66	0.07	0.06	7285.42	6224.57	0.07	0.17
GB	2011	SF	103.55	138.22	0.12	1.86	2778.85	2553.07	0.20	3.05
GB	2012	CA1	489.46	763.04	0.08	0.13	10102.43	11744.98	0.08	0.29
GB	2012	CA2_N	659.49	568.81	0.02	0.09	19660.00	21527.78	0.02	0.02
GB	2012	CA2_S	257.40	372.81	0.09	0.07	9803.77	9590.06	0.08	0.16
GB	2012	GSC_NW	1401.52	1721.65	0.05	0.04	25584.05	26266.07	0.05	0.22
GB	2012	GSC_SE	97.12	65.23	0.30	0.23	2390.65	4359.93	0.60	0.30
GB	2012	NF	375.65	259.75	0.05	0.09	8809.68	5919.12	0.05	0.23
GB	2012	NLS	275.23	256.81	0.14	0.44	8139.02	7111.74	0.16	0.14
GB	2012	SF	447.59	634.37	0.11	1.00	9534.90	7519.81	0.12	0.17
GB	2013	CA1	223.26	434.47	0.07	0.05	4479.75	6313.61	0.09	1.09
GB	2013	CA2_N	358.69	279.35	0.03	0.03	15818.66	12027.82	0.03	0.04
GB	2013	CA2_S	545.50	1026.61	0.04	0.09	5594.88	5445.98	0.10	0.05
GB	2013	GSC_NW	471.15	501.50	0.05	0.47	8518.39	8875.60	0.06	0.31
GB	2013	GSC_SE	78.82	57.64	0.16	0.90	1934.28	2281.77	0.21	0.08
GB	2013	NF	135.35	175.20	0.06	1.40	3413.10	4206.02	0.09	2.87
GB	2013	NLS	227.46	188.51	0.12	0.07	4519.21	4039.83	0.11	0.03
GB	2013	SF	1521.91	1385.35	0.06	0.05	10405.12	6480.77	0.09	0.18
MAB	2012	DMV_VB	487.11	340.30	0.06	0.09	3563.73	2657.57	0.08	0.08
MAB	2012	ET	1069.29	1431.26	0.06	0.02	7872.55	7455.85	0.06	0.68
MAB	2012	HC	1056.73	1417.64	0.05	0.02	12865.32	13196.17	0.07	0.10
MAB	2012	HCnr	497.09	616.72	0.11	0.99	8320.79	8607.06	0.12	0.03
MAB	2012	HCsr	418.46	435.87	0.13	0.15	6398.27	6531.35	0.12	0.03
MAB	2012	LI	637.03	660.37	0.11	0.04	11553.18	10748.32	0.11	0.25
MAB	2013	DMV_VB	594.70	529.23	0.07	0.09	5928.37	5742.01	0.05	0.05
MAB	2013	ET	1607.36	1555.18	0.04	0.04	20500.36	19429.08	0.04	0.05
MAB	2013	HC	1324.67	1091.30	0.08	0.16	9953.54	10758.67	0.09	0.05
MAB	2013	HCnr	644.33	502.73	0.26	0.48	8899.89	9953.83	0.14	0.78
MAB	2013	HCsr	262.77	266.72	0.27	0.40	5107.50	4946.65	0.26	0.11
MAB	2013	LI	630.57	665.43	0.09	0.10	11925.31	10655.17	0.10	0.06

Table 6: Abundance and biomass and its CVs estimated using GAM+OK and SM methods by stocks for 2011 to 2013.

			Number (million)				Weight (mt)			
Stock	Management Area	Year	SM	GAM+OK	SM CV	GAM+OK CV	SM	GAM+OK	SM CV	GAM+OK CV
GB	Close	2011	1992.16	2128.05	0.02	0.53	66266.35	71907.54	0.02	0.06
GB	Close	2012	1681.58	1961.46	0.04	0.08	47705.22	49974.57	0.04	0.08
GB	Close	2013	1354.91	1928.94	0.03	0.05	30412.50	27827.24	0.03	0.25
GB	Open	2011	2000.04	1704.01	0.04	0.20	43938.06	30911.23	0.03	0.39
GB	Open	2012	2321.88	2681.00	0.04	0.24	46319.28	44064.93	0.05	0.14
GB	Open	2013	2207.23	2119.68	0.04	0.17	24270.90	21844.16	0.05	0.57
GB	Total	2011	3992.20	3832.06	0.02	0.31	110204.42	102818.77	0.02	0.12
GB	Total	2012	4003.46	4642.46	0.03	0.14	94024.50	94039.50	0.03	0.08
GB	Total	2013	3562.13	4048.62	0.03	0.09	54683.40	49671.39	0.03	0.29
MAB	Total	2012	4165.70	4902.15	0.03	0.13	50573.84	49196.34	0.04	0.12
MAB	Total	2013	5064.39	4610.57	0.05	0.07	62314.98	61485.41	0.04	0.13

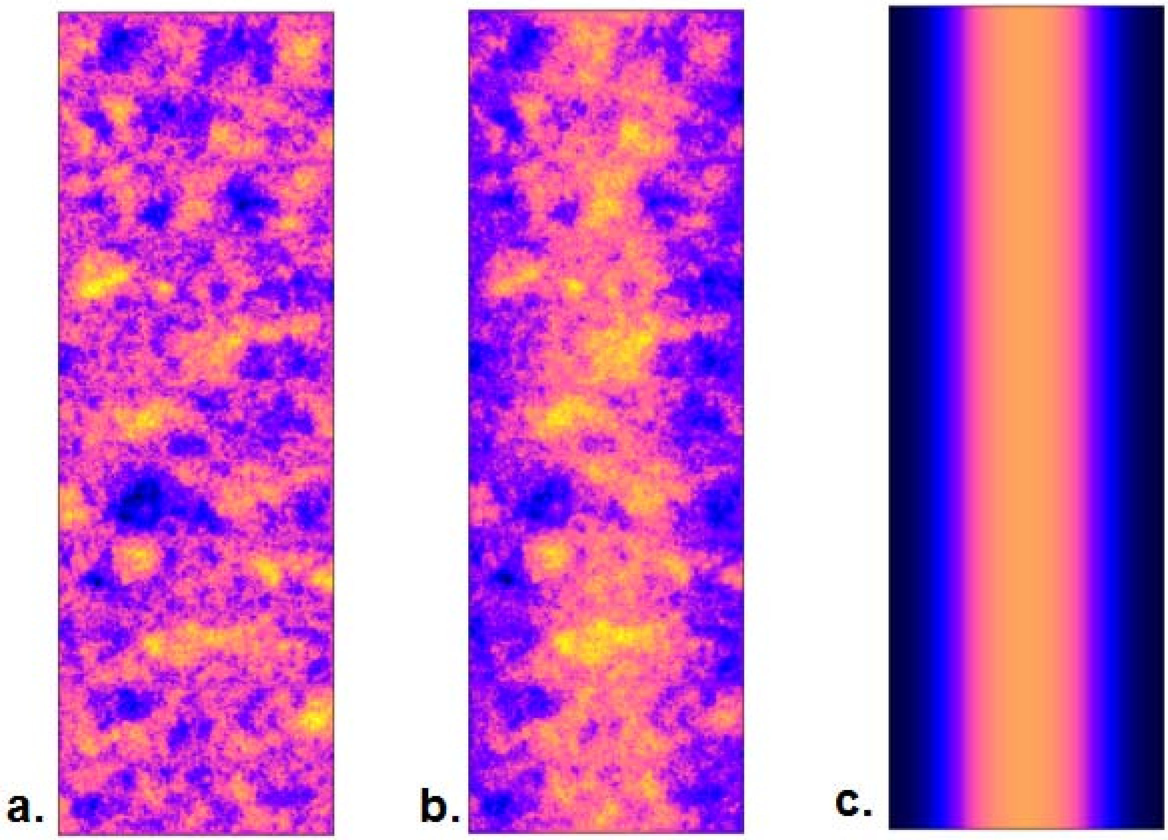

Figure 1. Hypothetical landscapes for (a) a landscape with a stationary mean, (b) a biased landscape with a higher mean along the center, and (c) the bias applied to landscape a to produce landscape b. In all plots, densities are higher in lighter-colored pixels.

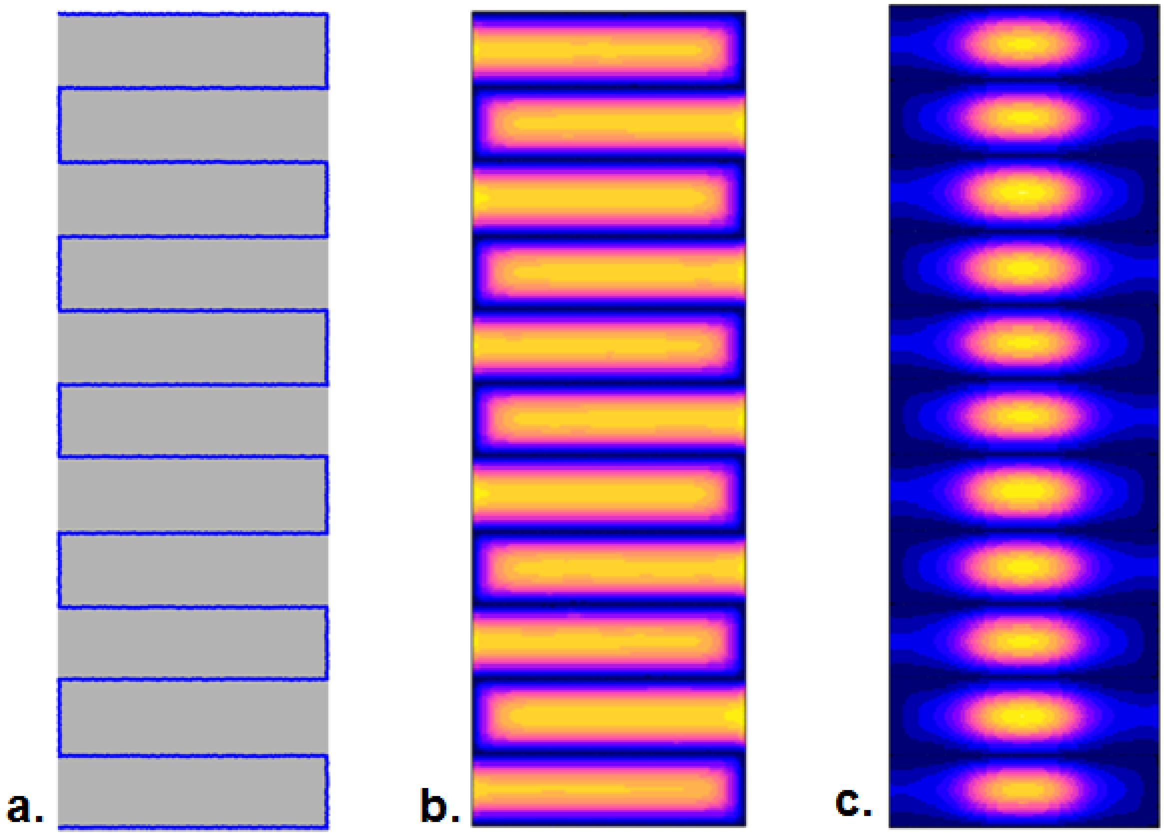

Figure 2 (a) A regularly spaced survey track across a rectangular survey area, (b) map of kriging variance derived from the survey track and an assumed underlying variogram model describing the data, (c) adjusted kriging variances resulting from applying the underlying trend from Figure 1c to (b).

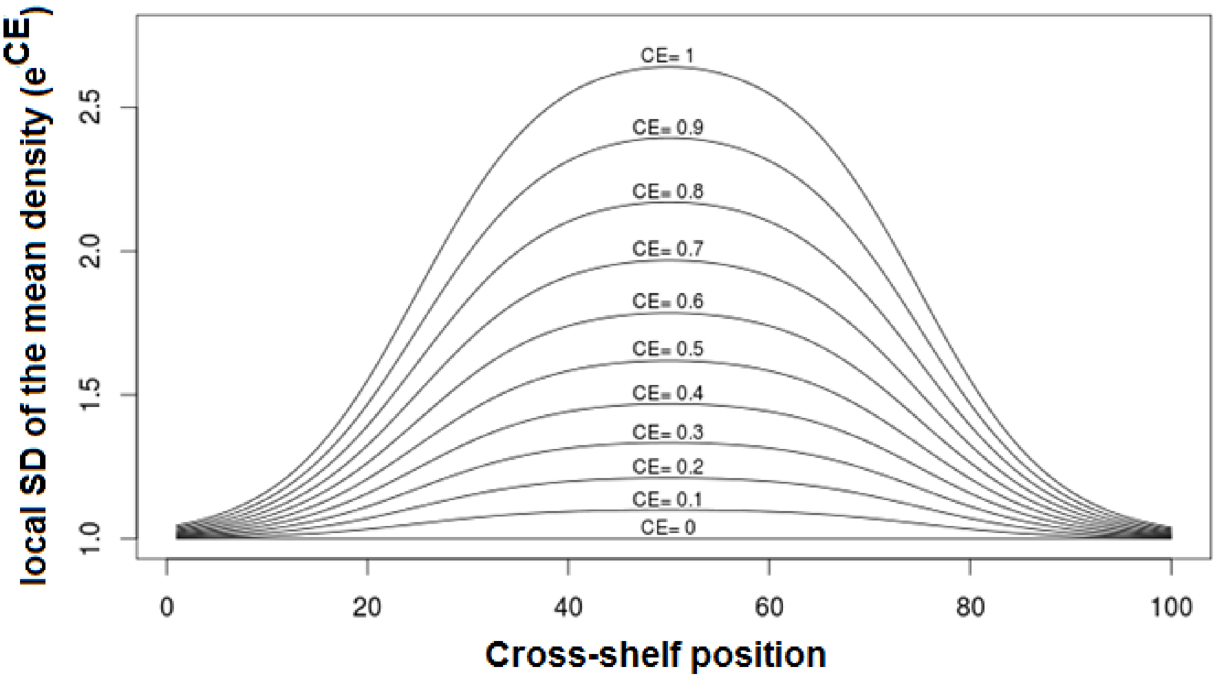

Figure 3. Levels of center effect used in simulations and resulting effects on the local standard deviation of the mean.

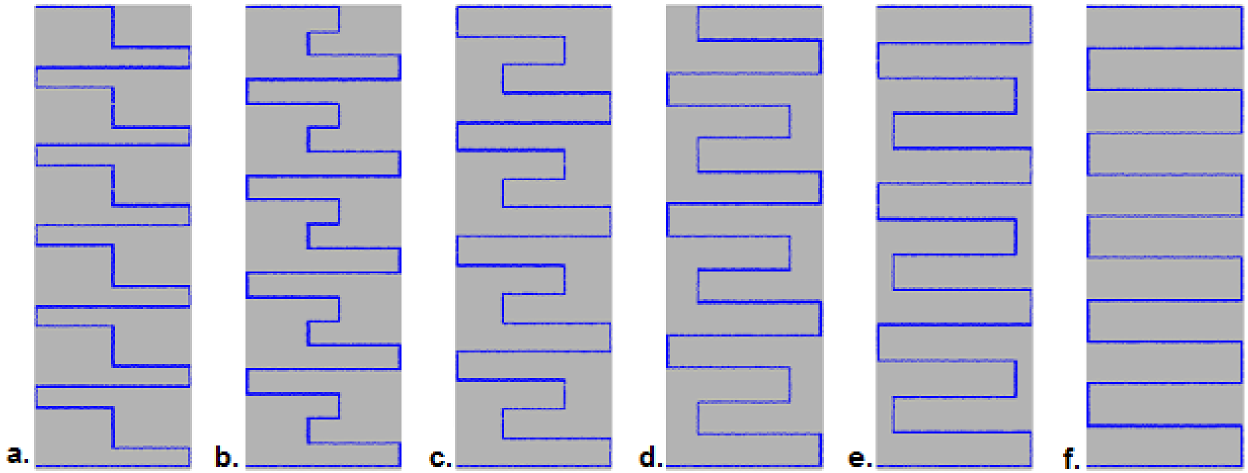

Figure 4a. Alternative survey configurations, alternating the length of transects along the track and keeping total survey length constant. Short transect lengths are (a) 0%, (b) 20%, (c) 40%, (d) 60%, (e) 80%, and (f) 100% of the length of the long transects.

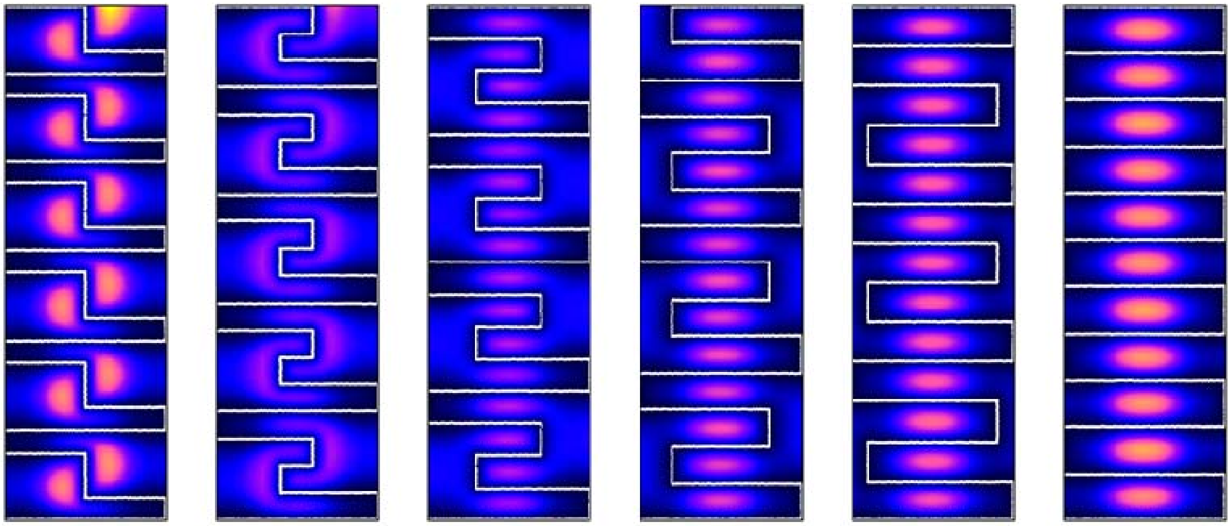

Figure 5. Variance maps for the survey tracks in Figure 4 with an applied center effect. Lighter colors indicate higher areas of variance.

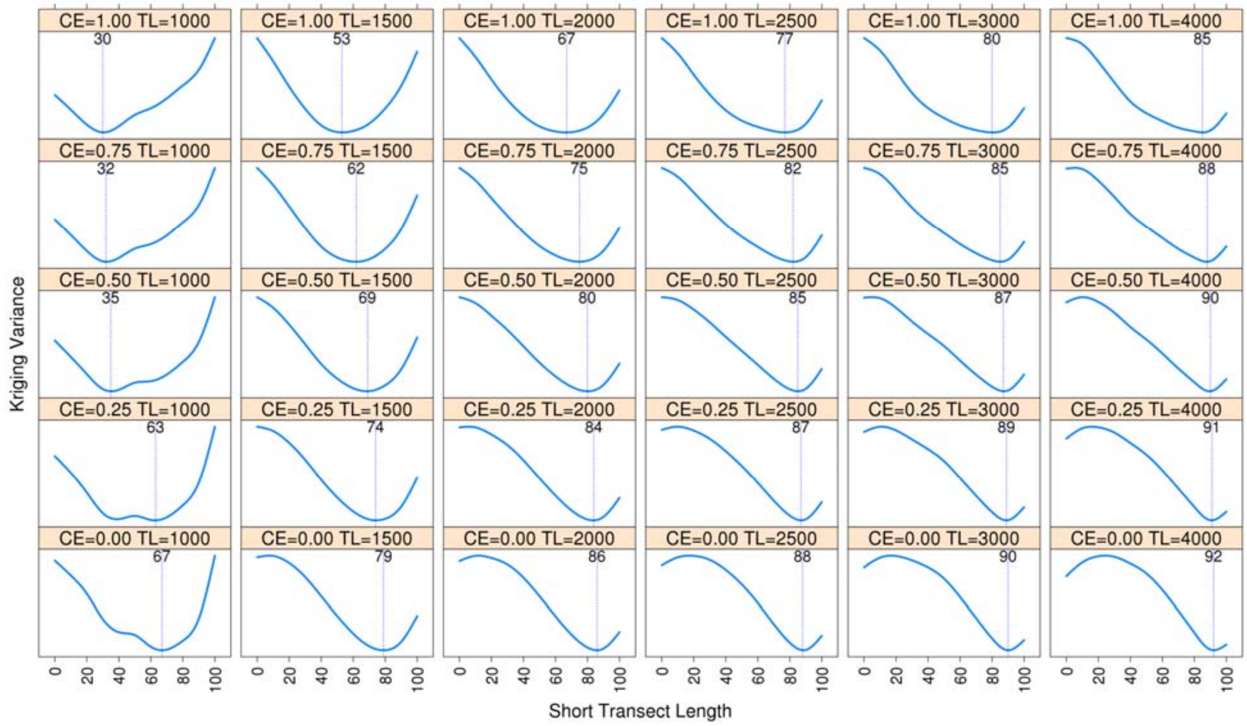

Figure 6. Adjusted kriging variances for different center effects (CE), short transect lengths and total survey track lengths (TL). Optimal solutions for each combination are marked with a dotted vertical line and labeled.

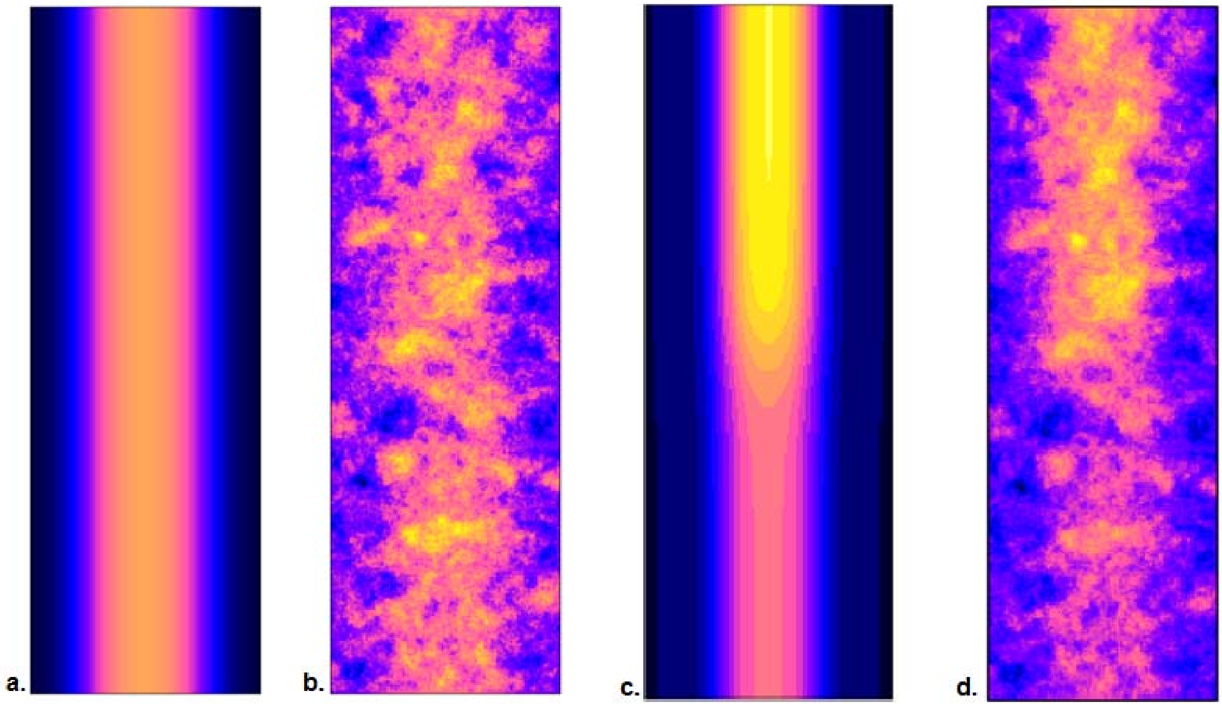

Figure 7. Comparison of survey landscapes without (a and b) and with (c and d) zonal anisotropy effects. Figures a and c represent the underlying trend in the mean while b and d represent the resulting simulated landscapes.

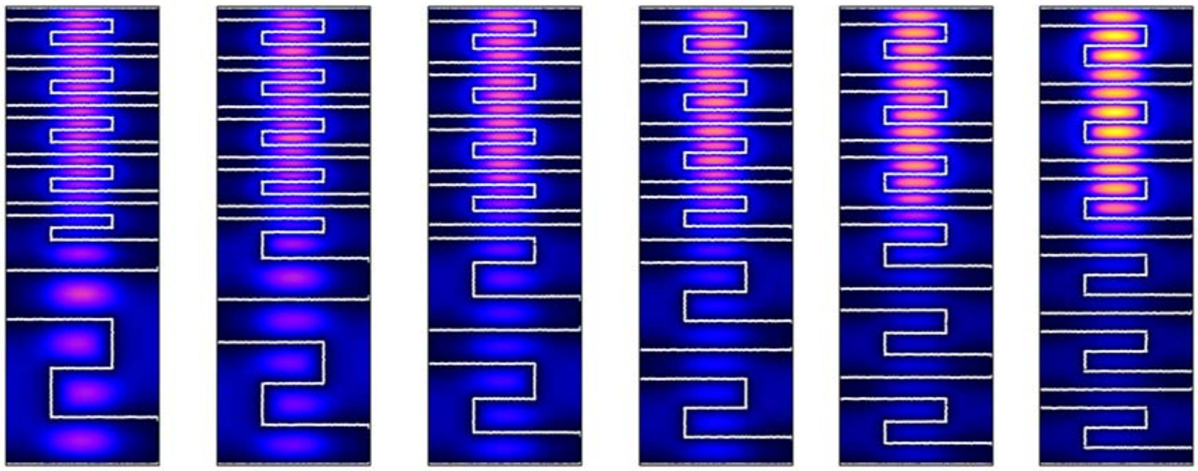

Figure 8. Example of varying transect density between zones and resulting variance maps for a simulated landscape with an underlying trend similar to C-7c. The survey track is represented in white. Lighter colors indicate higher variances.

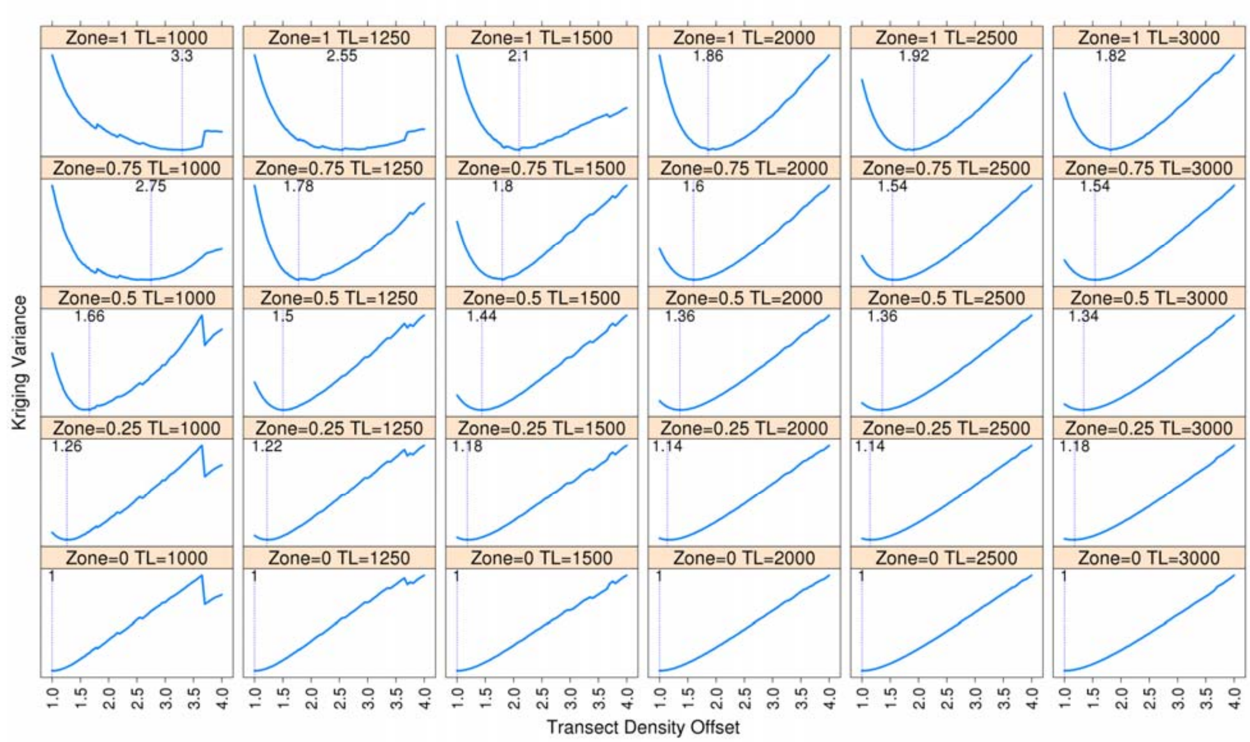

Figure 9. Zonal effects on transect density allocation. Higher “Transect Density Offests” represent the placement of proportionally more transects in the high density zones. Optimal solutions for each simulation set are labeled and marked with a dotted line.

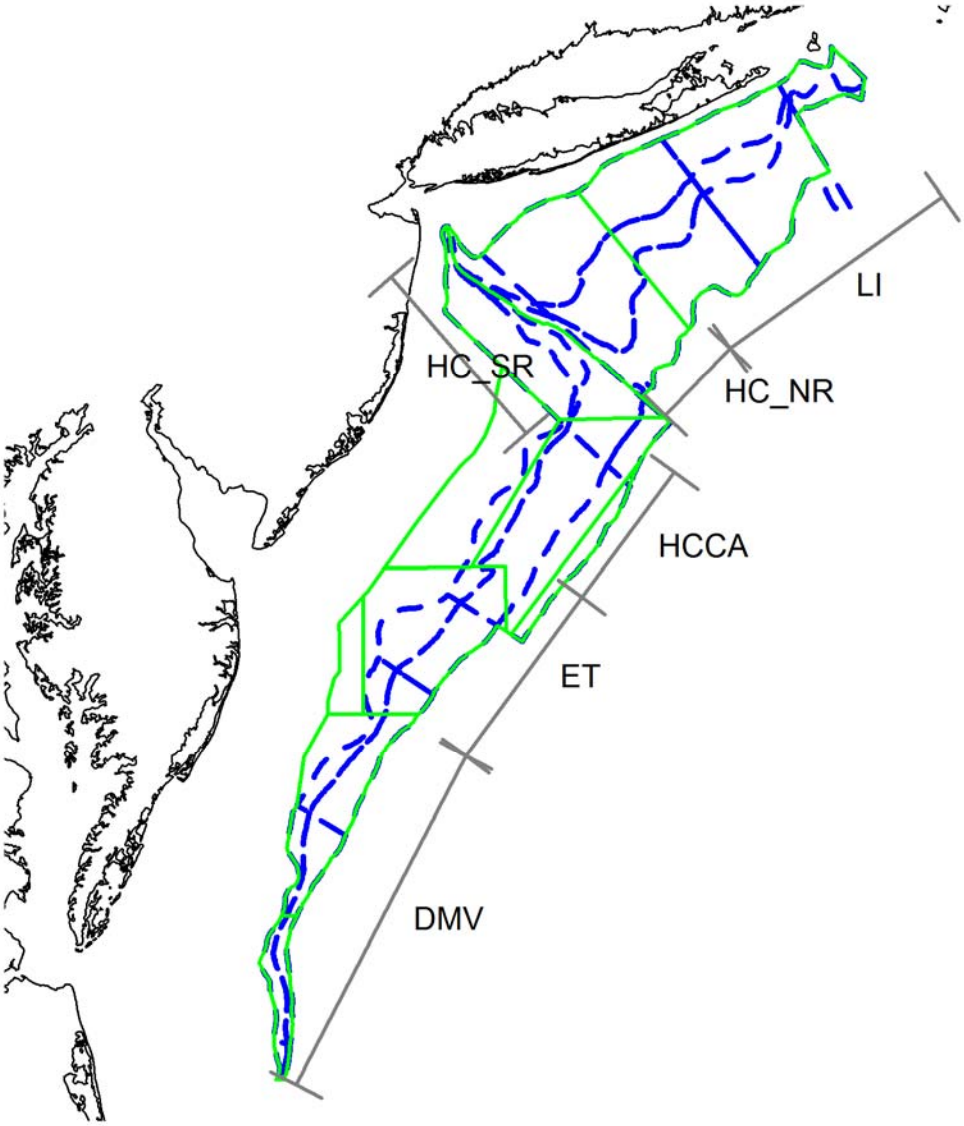

Figure 10. HabCam survey area (solid green line) compared to NEFSC scallop survey core strata (dashed blue line) in the MAB region. Subregions used for allocating survey effort and abundance estimation are: LI – Long Island, HC_NR – Hudson Canyon North Rim, HC_SR Hudson Canyon South Rim, HCCA – Hudson Canyon Closed Area, ET – Elephant Trunk, and DMV – DelMarVa.

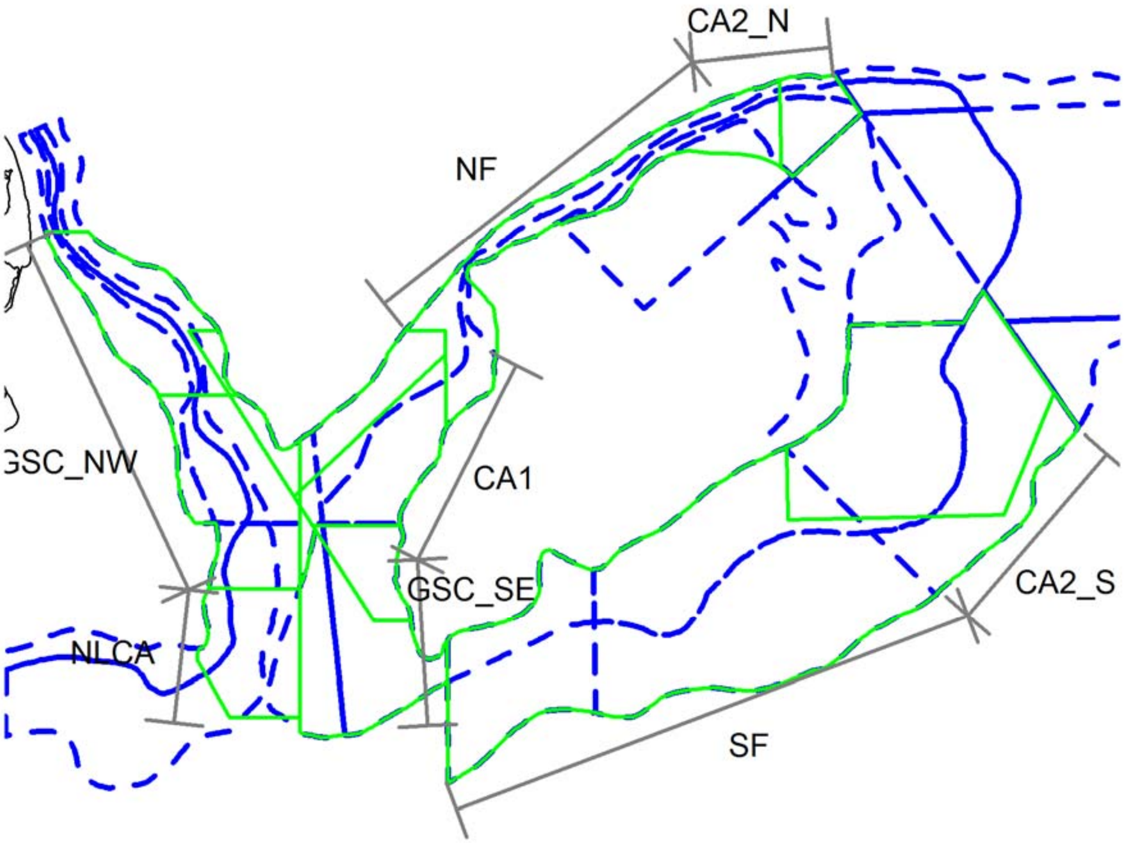

Figure 11. HabCam survey area (solid green line) compared to NEFSC scallop survey core strata (dashed blue line) for Georges Bank. Subregions used for allocating survey effort and abundance estimation are: GSC_NW – Great South Channel Northwest, NLCA – Nantucket Lightship Closed Area, GSC_SE – Great South Channel Southeast, CA1 – Closed Area 1, NF – Northern Flank, CA2_N – Closed Area 2 North, CA2_S – Closed Area 2 South, and SF – Southern Flank.

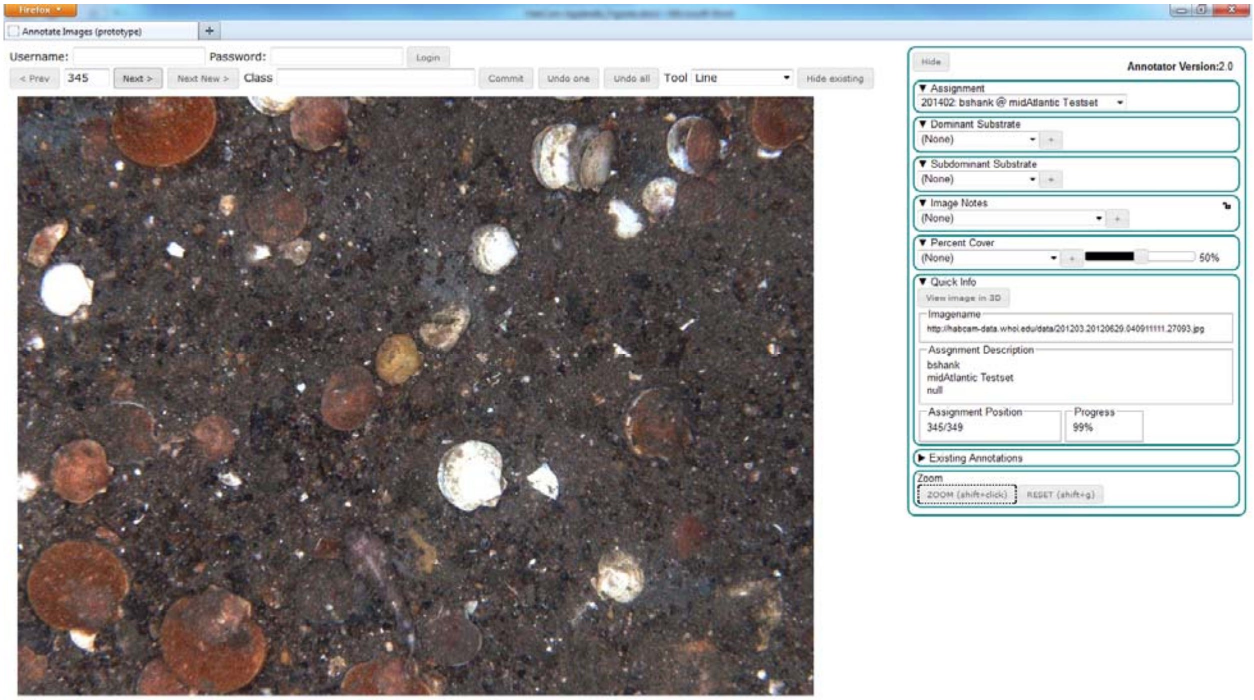

Figure 12. Screen image showing the web-based annotation tool for counting and measuring scallops from HabCam images.

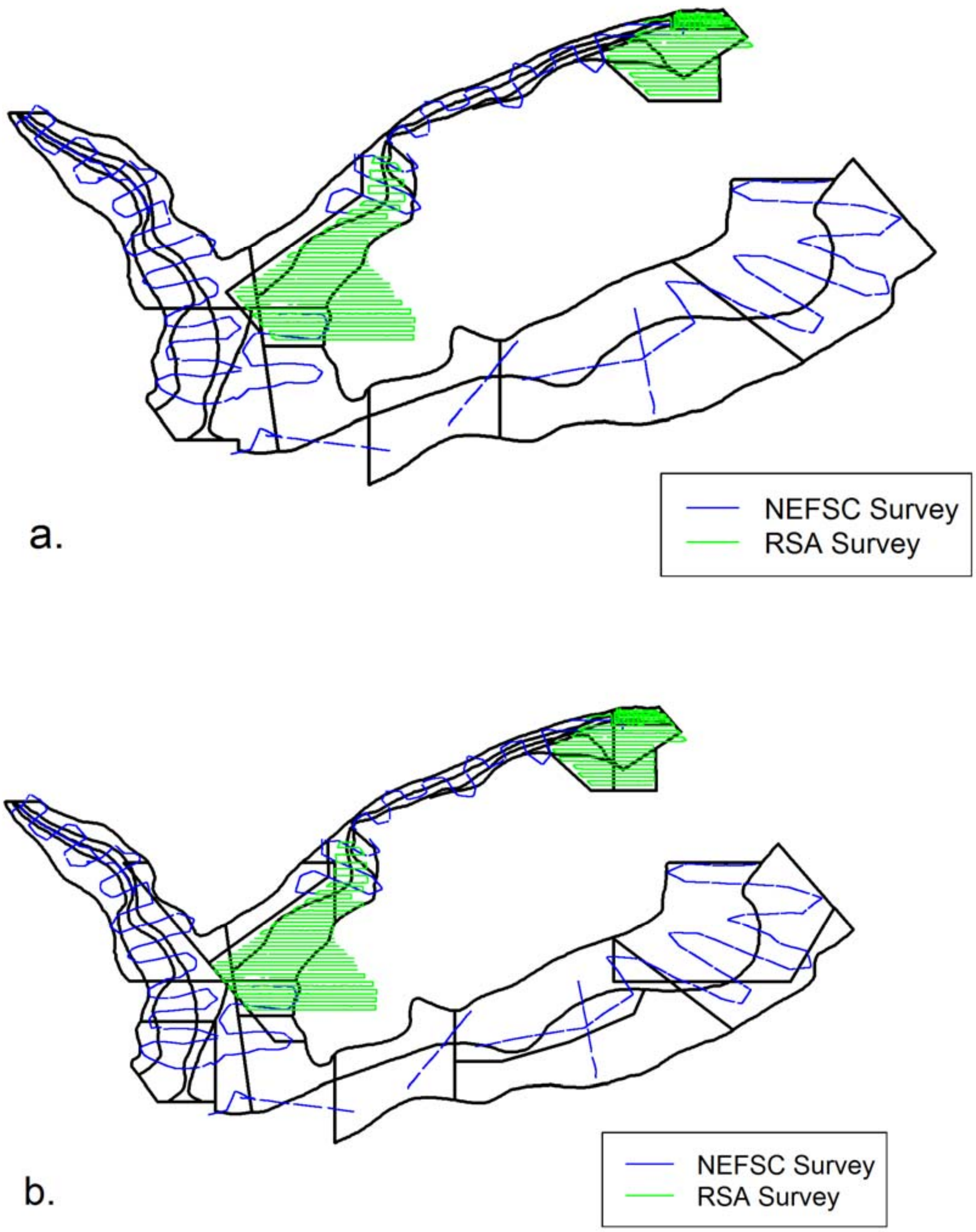

Figure 13 Example re-stratification of Georges Bank used for 2013 size frequencies: (a) open and closed areas combined and (b) open and closed areas separate.

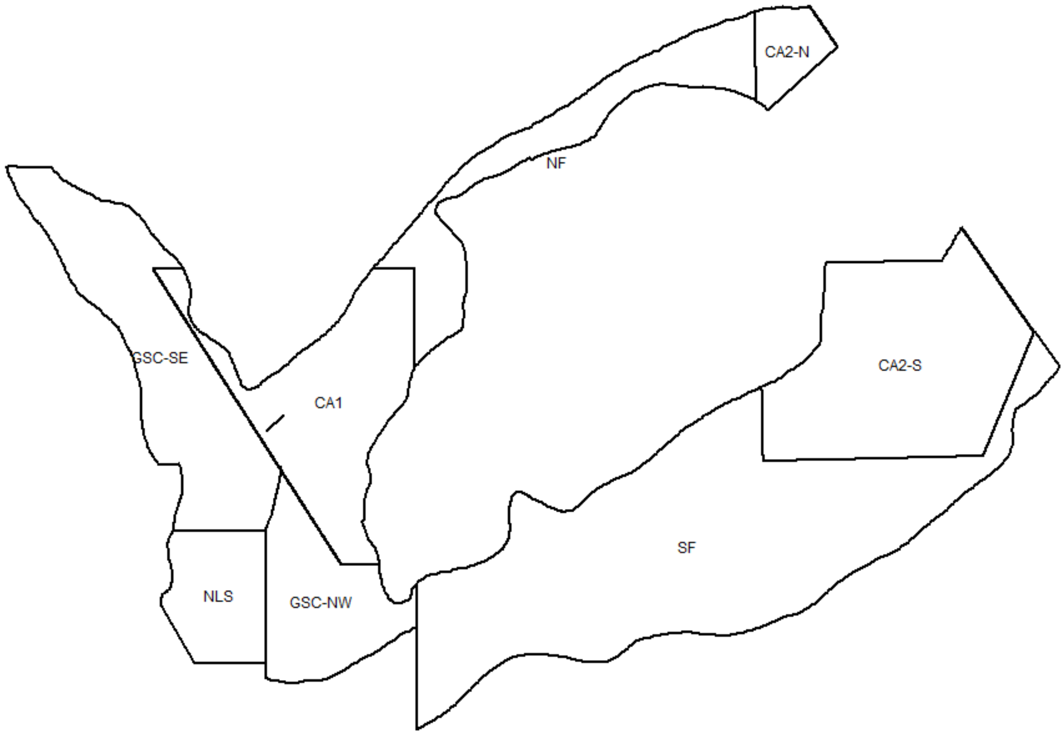

Figure 14: Subregions of the GB scallop stock area used in the HabCam survey.

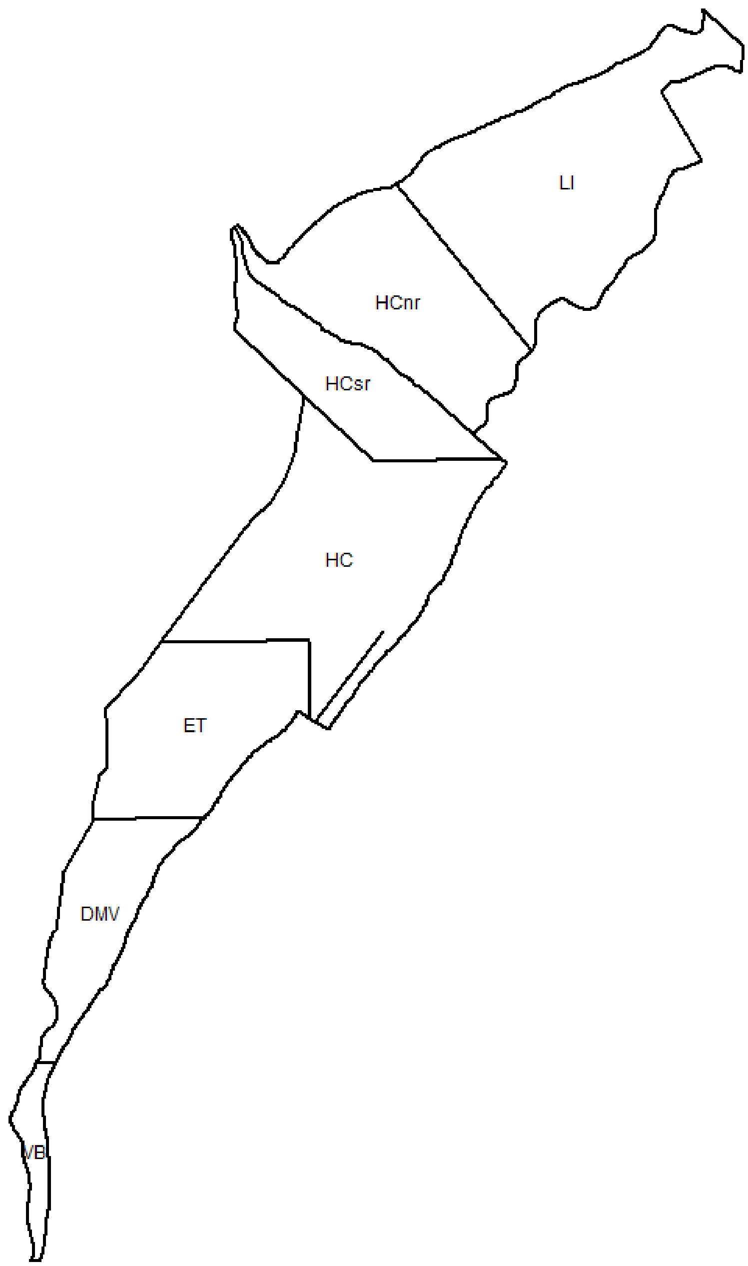

Figure 15: Subregions of the MAB scallop stock area used for the HabCam survey.

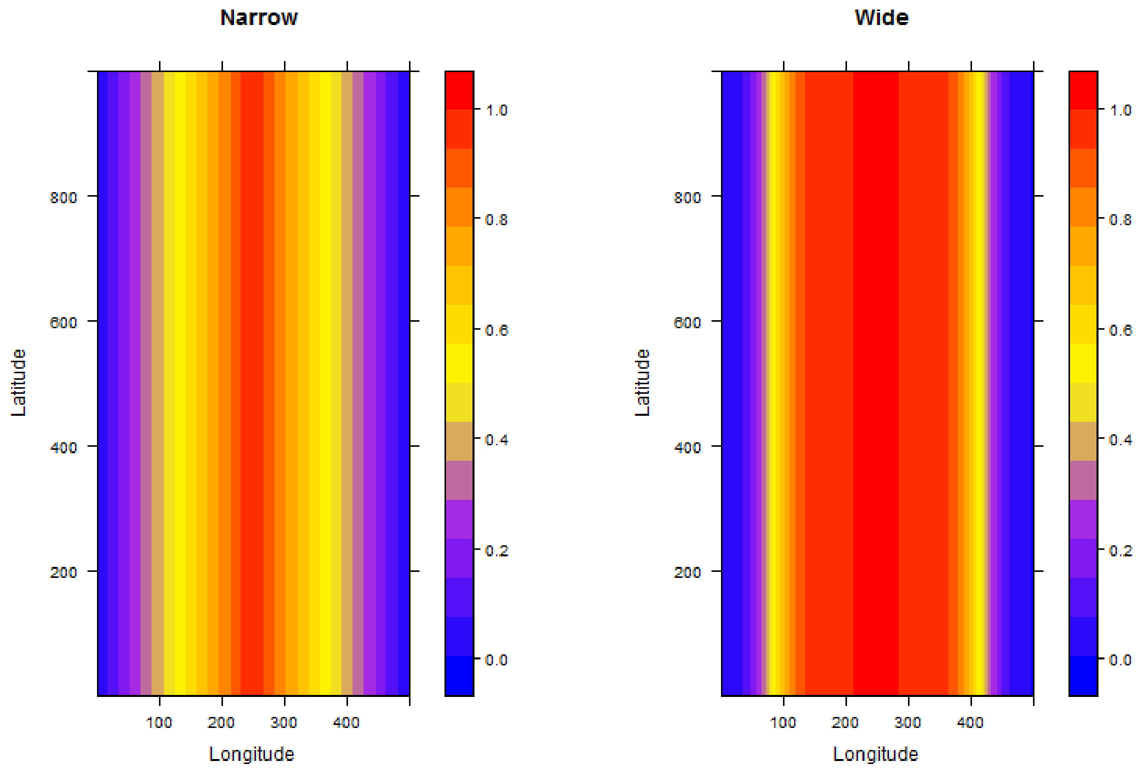

Figure 16: The two types of first-order effects used to simulate scallop populations: a narrow but highly dense first-order effect (left) and a wide but relatively less dense first-order effect (right).

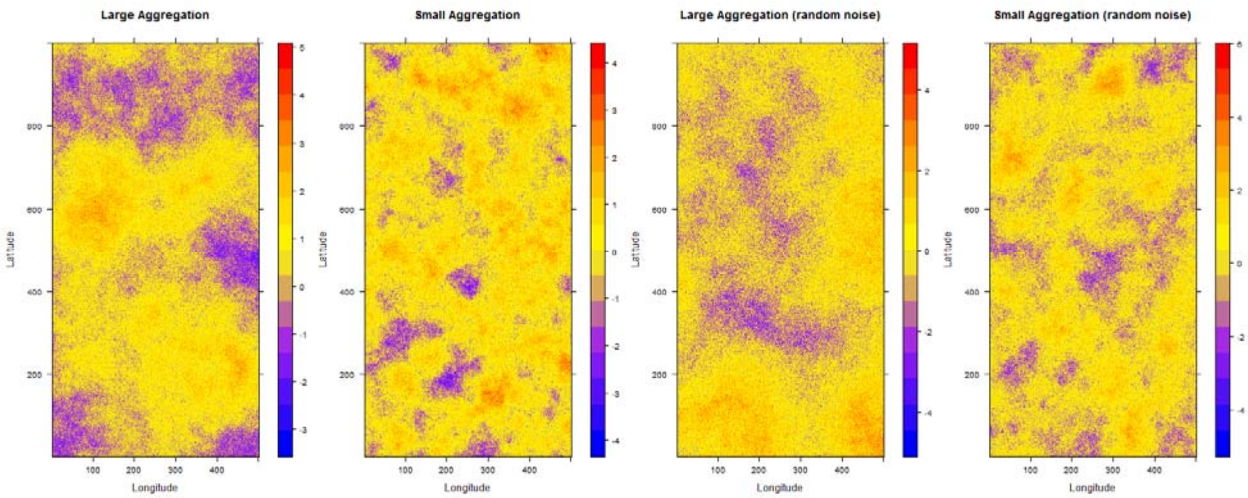

Figure 17: The four types of second-order trends tested to simulate scallop populations: large aggregations, small aggregations, large aggregations with a high random noise, and small aggregations with a high random noise (from left to right).

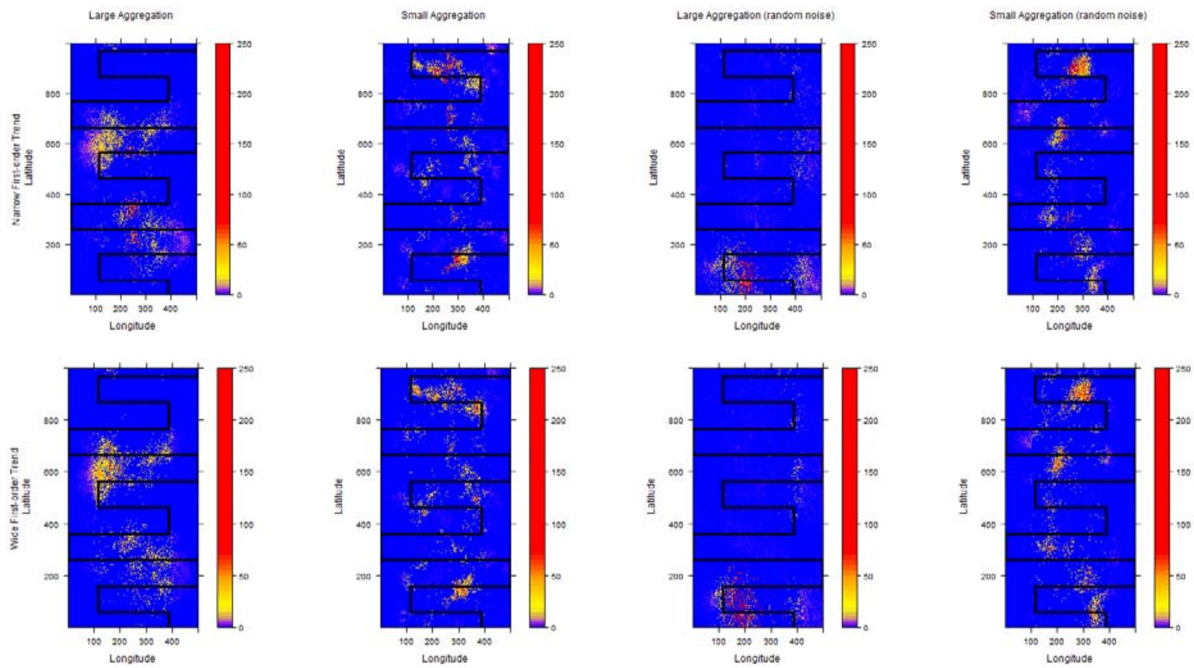

Figure 18: Example simulated scallop population distributions with an over-layed sampling track.

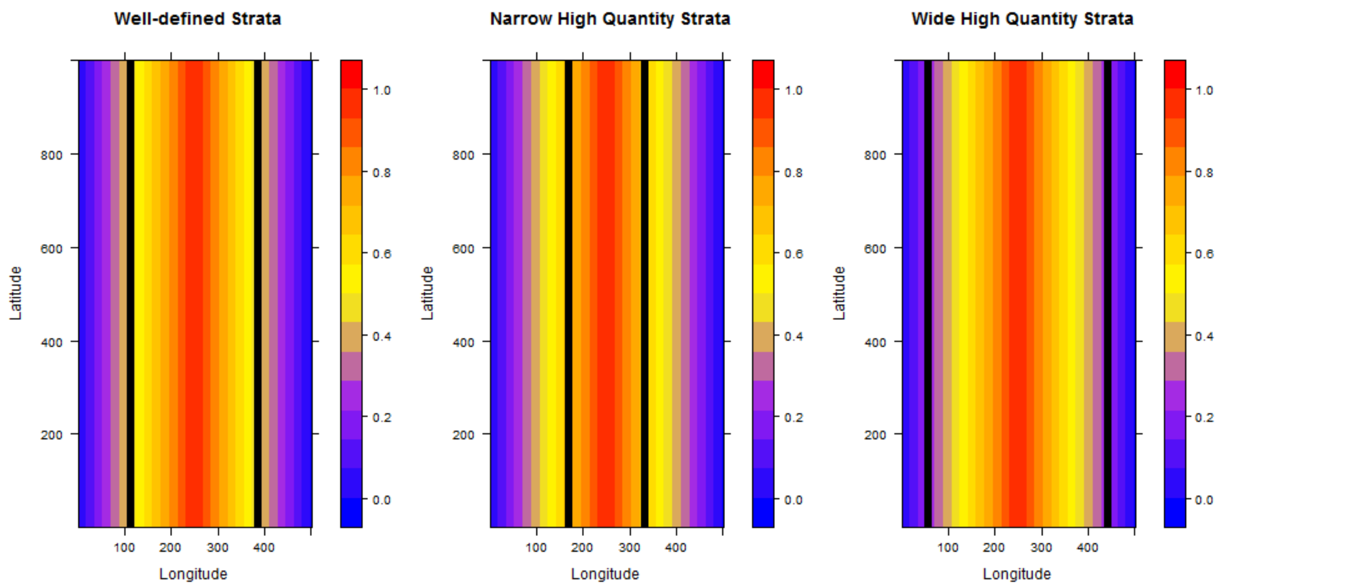

Figure 19: Alternative types of stratifications used for stratified mean estimations.

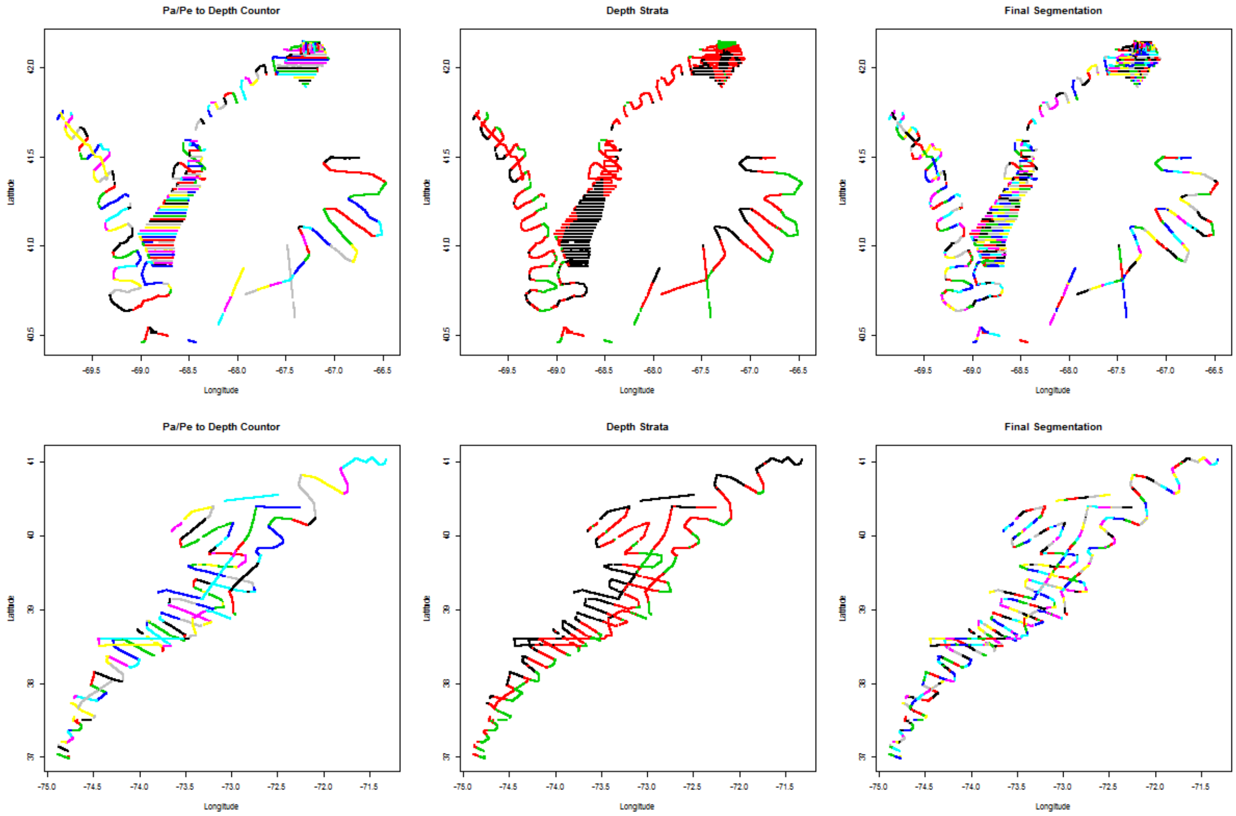

Figure 20: Transect segmentation for stratified mean estimation and the 2013 survey based on orientation to depth contours and distance between points (2 km) (left) and depth strata (center). The final combined segmentation is on the far right for GB (upper panels) and MAB (lower panels).

Appendix B7. Assessment of the sea scallop resource in the Northern Gulf of Maine management area

Samuel B. Truesdell (University of Maine, Orono, ME); Kevin H. Kelly (Maine Department of Marine Resources, W. Boothbay Harbor, ME); and Yong Chen (University of Maine, Orono, ME).

Summary

The sea scallop (*Placopecten magellanicus*) fishery in the Northern Gulf of Maine management area (NGOM) occurs in federal waters and is managed by the New England Fishery Management Council. The NGOM resource and associated fishery are locally important but amount to a small portion of the total stock and less than 0.1% of total landings. The fishery is managed by a TAC independently of the rest of the EEZ sea scallop stock. Management of the NGOM fishery does not involve biological reference points as targets or thresholds.

A cooperative survey was carried out by the Maine Department of Marine Resources and University of Maine during May-June of 2012. Based on survey results, estimated biomass of NGOM sea scallops targeted by the fishery (102+ mm or 4+ in shell height) was approximately 164.19 MT (90% confidence interval from 74.35 to 278.91), an increase from 115.40 MT (66.05 to 173.31) in 2009. These estimates are based on density estimates from the survey assuming a capture efficiency of 43.6%. The previous survey in 2009 noted a large year class of 10-50 mm scallops on Platts Bank; this year class was still evident in 2012 and had grown to approximately 65-90 mm.

Based on these biomass estimates the exploitation rate in weight (landings/stock biomass, assuming harvested scallops greater than 102 mm shell height and a dredge efficiency of 43.6%) during 2012 was 2.1% with a 90% confidence interval from 1.3% to 4.7%.

Several analyses were performed to determine how representative the survey was of the NGOM to determine applicability of survey results to management of the NGOM. The fraction of the NGOM covered by the survey area is 0.11, however using information regarding habitat preferences of scallops, the fraction of the suitable habitat area for the stock within the NGOM covered by the survey is 0.37. The survey extent was designed to ensure coverage of the primary fishing areas, and the fraction of fishing locations within the survey bounds was greater than 50% since 2006 and greater than 70% since 2011. Thus, the survey probably encompasses most of the areas with scallop concentrations high enough to support fishing activity indicating that survey results should be useful information for management of the NGOM scallop stock.

Introduction

The Gulf of Maine scallop fishery that occurs in federal waters is managed by the New England Fishery Management Council. Amendment 11 to the New England Fishery Management Council Sea Scallop Fishery Management Plan (NEFMC 2008) created a separate limited entry program for general category fishing in the Northern Gulf of Maine management area (NGOM; Fig. 1). The area is managed under an annual total allowable catch (TAC; currently 31.75 MT) and a daily possession limit of 90.7 kg (NEFMC 2008). Scallop dredge ring size must be greater than 102 mm, but there are currently no regulations regarding shell size (as in Maine state waters) or meat count.

Landings in the NGOM are low relative to the rest of the scallop stock, averaging just over 7 MT from 2008 to 2013 (total sea scallop landings have been over 20,000 MT in recent years). In 2013 the most landings since the management area's inception in 2008 (over 18 MT) were reported, more than double any other year.

The region has limited fishery-independent data available. There was an offshore survey administered by the Maine Department of Marine Resources in 1974 (Spencer 1974), and in 1983 and 1984 NMFS sampled some areas in this region on their annual survey (Serchuk 1983; Serchuk and Wigley 1984), but no broad-scale surveys were completed between the early 1980s and 2008 when the region was first managed under a TAC. Given the lack of recent fishery-independent data, the initial allowable catch was determined using historical federal Gulf of Maine landings (NEFMC 2008). More recently, Maine Department of Marine Resources/University of Maine scallop surveys in 2009 and 2012, along with UMass Dartmouth video scallop surveys that occasionally sample in this area (e.g., Stokesbury et al. 2010) have offered fishery-independent sources of information to aid in generating the TAC.

The only management area-wide biomass estimate previously available was based on the Maine Department of Marine Resources/University of Maine scallop survey in 2009. This was a point estimate that used swept area to expand the survey results to a subset of the NGOM (this subset is discussed below; Fig. 1). This analysis estimated 103 MT of scallops greater than 102 mm shell height, with a confidence interval that ranged from 53 to 186 MT (Truesdell et al. 2010). This estimate was revised (see Results/Discussion section) during the current analysis and the new estimate for 2009 is 115.40 MT (90% confidence interval from 66.05 to 173.31). The best estimate based on the 2012 survey results indicates that the biomass of NGOM sea scallops over 102 mm shell height was approximately 164.19 MT of meats with a 90% confidence interval ranging from 74.35 to 278.91 MT.

Methods

Survey area identification and delineation

The NGOM management area is bounded by Cape Ann, Massachusetts in the west and the Canadian border in the east (Fig. 1). Prior to 2009 when the first survey was conducted, the NGOM had limited fishery dependent and no recent fishery-independent data available to help design the survey. Scallops are not found uniformly throughout this region so sampling efforts were focused on a subset of areas in the NGOM. To determine this subset, fishing locations from National Marine Fisheries Service vessel trip reports (VTRs) from 2000 to 2008 were reviewed as well as three historical surveys of the region from the 1970s and 1980s (Spencer 1974; Serchuk 1983; Serchuk and Wigley 1984). In addition to the information available, two fishermen with a history of scalloping in the Gulf of Maine were interviewed to help identify current and historical fishing grounds. These sources of information were used qualitatively to determine the five sampling areas: Machias-Seal Island (MSI), Mount Desert Rock (MDR), Platts Bank (PB), Northeast of Cape Ann (NCA) and Northern Stellwagen Bank (NSB; Fig. 1).

To increase sampling precision, the two western strata off the Massachusetts coast (where most fishing occurs), NCA and NSB, were further divided into substrata of expected high, medium and low scallop density.

Survey coverage area

Although the survey is intended to represent the NGOM scallop management area, the entirety of the NGOM was not sampled (Fig. 1); as such it is necessary to document the survey coverage area relative to total stock area, total stock biomass and the area fished. This was accomplished most simply by calculating the ratio of the sampling area (A_{SURVEY}) to the area of the NGOM (A_{NGOM})

$$R_{BASE} = \frac{A_{SURVEY}}{A_{NGOM}} \quad \text{Eqn. 1}$$

where R_{BASE} is the proportion of survey coverage. This baseline ratio is only one approach to estimating the coverage area of the survey, and it assumes that scallops are as likely to be found within the survey area as they are outside. However, the survey was designed specifically to sample the areas where scallops are distributed within the NGOM, so R_{BASE} is likely to be an underestimate of the survey's coverage of the scallop stock. Two additional methods were used to arrive at a more realistic approximation: one based on the depths at which scallops are typically found and another based on fishing effort data.

Sea scallops are typically more abundant at shallower depths (Merrill 1972; Posgay 1979; Serchuk et al. 1979); a depth threshold was employed as one way to estimate the effective coverage proportion of the survey. Serchuk et al. (1979) note that most commercial quantities of scallops are found in depths less than 100 m; this is corroborated by analyses of the NMFS bottom trawl survey from 1982 to 2010 and the NMFS bottom trawl survey from 2010 to 2012.

Employing a depth threshold DTh to determine an effective coverage proportion for the survey can be

$$R_{DTh} = \frac{A^*_{SURVEY}}{A_{DTh}} \quad \text{Eqn. 2}$$

where R_{DTh} stands for the ratio at a particular depth threshold (100 m in this analysis), A^*_{SURVEY} is the survey area shallower than the threshold and A_{DTh} is the area of the NGOM shallower than the depth threshold.

Alternatively, an effective coverage proportion can be estimated using fishing effort data. This assumes that the Gulf of Maine scallop fleet follows an ideal free distribution (Fretwell and Lucas 1969; i.e., fishing activity is directly related to abundance). Vessel monitoring system (VMS) data from 2006 to 2013 were used to determine the effort-based effective coverage proportion R_{VMS} as

$$R_{VMS} = \frac{P_{SURVEY}}{P_{NGOM}} \quad \text{Eqn. 3}$$

where R_{VMS} is the coverage proportion with respect to VMS observations (satellite location records), P_{SURVEY} is the number of VMS observations within the survey areas and P_{NGOM} is the total number of VMS observations within the NGOM.

Two resolutions of VMS data were considered: 1km and 3km. The advantage to the finer resolution is that the locations were more accurate, which is important near the boundaries of the areas. The disadvantage is that for confidentiality reasons less VMS data was available at higher resolutions. At the 1 km resolution 83% of VMS observations were available and at 3 km resolution 91% were available.

Survey design

Surveys were carried out in June and July of 2009 and in May and June of 2012. Dredge tow stations were selected from a grid overlying each stratum. The dimensions of each grid unit were 1 km^2. Each survey followed a two-stage random stratified design in the NCA and NSB strata. Station allocation in the first stage was based on fishing intensity from 2000-2008 vessel trip report (VTR) data and the size of each substratum. Forty stations in each stratum were assigned to the first stage and distributed among substrata according to the formula

$$N_s = 40 * \frac{M(V_s)A_s}{\sum_{s=1}^{S} M(V_s)A_s} \quad \text{Eqn. 4}$$

where N_s (rounded) is the number of stations to be sampled in substratum s, $M()$ is the median function, V_s is the VTR landings from 2000 to 2008 for substratum s and A_s is the area of substratum s. VTR data is assumed to be a proxy for scallop density and so was used to help allocate survey sample size. Such commercial data have also been used in the design of Canadian scallop surveys (Robert and Jamieson 1986; Serchuk and Wigley 1986). Area size was included in the weighting to ensure sufficient effort in the larger substrata.

In the NCA and NSB strata, which were further divided into substrata, a two-stage survey was employed. The approach taken by Francis (1984) was used to allocate tows to the second survey stage. His formula to assign one additional station among strata is:

$$G'_s = \frac{A_s^2 M_s^2}{n_s(n_s+1)} \qquad \text{Eqn. 5}$$

where G'_s is the assumed reduction in variance from adding a single station to a particular substratum s, A_s is the area, M_s is the mean catch rate (when squared, a proxy for the variance suggested by Francis (1984)) and n_s is the number of additional stations. Twenty stations were available for the second stage and were apportioned among the substrata. They were assigned one-by-one (by repeated use of Eqn. 5) according to whichever substratum would gain the most in terms of reduced variance from receiving one additional station. As such, the assignment of the j^{th} station can be written

$$G'_s = \frac{A_s^2 M_s^2}{(n_s+j-1)(n_s+j)} \qquad \text{Eqn. 6}$$

A single stage design was used for the remaining three strata in the eastern GOM.

In 2012 206 stations were sampled using a 2.13 m New Bedford style dredge with 51 mm rings, 4.4 cm head bale, 8.9 cm twine top, 25.4 cm pressure plate and rock chains. This gear was identical to that used in the 2009 survey. The target tow duration in 2009 was 7 minutes at a speed of 6.5km/h (a distance of approximately 750m). This was reduced to 5 minutes and about 540m in 2012, though fixed gear in some locations forced shorter tows.

Data Analyses

Historically, meat count by shell height has been found to vary regionally within the Gulf of Maine (Serchuk and Rak 1983), so separate models predicting meat weight using shell height were employed for each stratum. Depth was included because it has been shown to influence many aspects of scallop life history (Naidu and Robert 2006) and has been used in this type of analysis by Hennen and Hart (2012). These models also included a random effect (as in Hennen and Hart 2012) to account for repeated sampling within a station. The mixed effects models were produced using R (v. 2.15.1, R Core Team 2012) with the package lme4 (Bates et al. 2013). The form of the model within each stratum was

$$\ln(W_{i,t}) = \beta_1 \ln(H_{i,t}) + \beta_2 D_t + R_t + \varepsilon_{i,t} \qquad \text{Eqn. 7}$$

where $W_{i,t}$ is the meat weight of individual i at station t, $H_{i,t}$ is its respective shell height, D_t is the depth, R_t is a random effect term associated with each station, β_1 and β_2, are the coefficients of the explanatory variables, and $\varepsilon_{i,t}$ is the error term for each sample. Depth was important to include as a covariate because although meat weights were sampled whenever possible, the samples were not always evenly distributed throughout the depth range of a stratum, though the results were extrapolated across all depths. PB had a low number of meat weight samples in 2009 so the 2009 samples were combined with those from 2012 for the 2009 PB meat weight model.

Prior to analyzing length frequency distributions, the number of scallops in each 5 mm size class belonging to a particular station was standardized to the mean swept area per station in the relevant stratum or substratum according to the formula:

$$Z_{l,s,c} = \frac{R_{l,s}}{\bar{R}_s} N_{l,s,c} \qquad \text{Eqn. 8}$$

where $Z_{l,s,c}$ is the standardized count for scallops at station l within stratum (or substratum for strata 4 and 5) s in 5 mm size class c, $R_{l,s}$ is the swept area of the station tow, $\bar{R}_s$ is the mean swept area for samples in area s, and $N_{l,s,c}$ is the number of scallops of size class c in tow l of area s. In these analyses the middle of the size bin was always used as the reference size for estimation.

The mean number of scallops within each stratum was estimated and uncertainty was addressed using bootstrapping and percentile confidence limits. Survey sample counts were bootstrapped 50,000 times. Bootstrapping was chosen to estimate confidence bounds because it requires few distributional assumptions (Efron and Tibshirani 1986) and avoids unrealistic confidence bounds that drop outside the range of observation (such as below zero).

To estimate the biomass and confidence limits for each stratum, the predicted meat weights from the mixed effects models at each location (1 km^2) within each stratum were estimated by size class and combined with the (sub)stratum length frequency distribution and the number of scallops per station to calculate the overall biomass per stratum such that

$$B_s = \sum_{g=1}^{G} \sum_{c=1}^{C} W_{s,c,g} P_{s,c} N_s \qquad \text{Eqn. 9}$$

where B_s is the estimated biomass in stratum s, G is the number of 1 km^2 grids in stratum s, c is the number of 5 mm size classes over 102 mm (4 in; assumed to be harvestable size), W is the expected weight per scallop from Eqn. 7, $P_{s,c}$ is the proportion of scallops in stratum s within size class c, and N is the bootstrapped standardized mean count per station in stratum s. The upper and lower confidence limits were estimated by substituting the upper and lower percentile estimates for *N* in each substratum.

The dredge efficiency (vulnerability coefficient) used in this study was 43.6% which was estimated experimentally in Maine state waters (Kelly 2007). The Maine value was used because it was generated near the survey area and is close to other estimates of dredge efficiency (e.g., Gedamke et al. 2004).

Weight-based exploitation rates for the NGOM were estimated for 2009 and 2012 as

$$E = \frac{L}{B} \qquad \text{Eqn. 10}$$

where E is the exploitation rate, L is the landings in weight and B is the total estimated biomass in the NGOM of scallops larger than 102 mm shell height. A 90% confidence interval for the exploitation rate was computed using the 5th and 95th percentiles for biomass, derived from the bootstrapping. Landings were assumed to be error-free.

Results and Discussion

Survey coverage area

The Northern Gulf of Maine management area encompasses a region of 23,470 km^2. Although this entire management area is under the regulations outlined in Amendment 11 to the sea scallop Fishery Management Plan (NEFMC 2008), scallops are not found throughout the region. The survey region (Fig. 1) has an area of 2,652 km^2 and the ratio of this region to the total area in the Northern Gulf of Maine regulatory region is 0.11 (Eqn. 1). While this areal coverage appears low, the effective survey coverage is larger in terms of both potential scallop habitat with respect to depth as well as the realized fishery area according to vessel monitoring system data.

The coverage proportion assuming a depth threshold of 100 m (Eqn. 2) is 0.37 (Fig. 2). This represents an estimate of the survey's coverage of the NGOM stock area, assuming depth is related to the probability of scallop occurrence. Using VMS data to determine the fraction of the fishery that occurs within the survey extent, the effective coverage proportion (Eqn. 3) was greater than 0.9 using either low or high resolution VMS data (Table 1). The proportion of total VMS observations by year (with no data excluded for confidentiality) was calculated by Burton Shank (NMFS NEFSC) for 2006-2013. Since 2009, the first year of the survey, the minimum coverage proportion was 0.69 (in 2010) and in 2013, the most recent year available, it was 0.87 (Table 2).

Scallop demographics within the NGOM

The most heavily fished area within the NGOM is the southwestern part, within survey areas NCA and NSB. In the NCA area, most scallops were found north of Cape Ann near the state waters boundary in both 2009 and 2012 (Fig. 3A). In the NSB area there were some scallops at the northern boundary (especially in 2012) and in both years scallops were found on the northern part of Stellwagen Bank near the southern-central part of the NSB area (Fig. 3A). Both NCA and NSB had wide, multimodal shell height distributions in both years (Figs. 4A-B). NSB was noteworthy in 2012 because it had signs of recent recruitment as well as some of the largest scallops seen on the survey.

In both years scallops were found on the southwest part of PB (Fig. 3B). The growth of the cohort first observed in 2009 was evident (Fig. 4C); the mode shell height grew from 32.5 mm in 2009 to 72.5 mm in 2012. In both years there was a small proportion of scallops that were between 125 and 150 mm. The survey in MDR encountered almost no scallops in both years (Fig. 3C). There were scallops to the south of this area near Mount Desert Rock in both years, but this small region is within Maine state waters and not part of the NGOM. The scallops that were caught in 2009 were mainly less than 100 mm (Fig. 4D). In 2012 only a single scallop was caught in this area. In the MSI area there was no obvious coherence between the spatial distribution of catch in the 2009 and 2012 surveys; scallops within this area appear from these surveys to be fairly evenly distributed relative to the patchiness observed in NCA and NSB to the south (Fig. 3D). The only persistent aggregation was near Machias Seal Island, again within state waters. Little signs of recruitment were seen in this region in either 2009 or 2012 and most scallops were between 110 and 150 mm (Fig. 4E).

The relationship between shell height and meat weight varied by area, as in 2009 (Fig. 5). The best condition meats were in NSB and NCA, while the meats in MSI were clearly smaller for their size. Few samples of larger scallops were taken on PB, but those greater than 100 mm were of similarly poor condition to the scallops sampled in MSI.

Biomass and exploitation rate estimates

Analysis of the surveys produced estimates indicating that the NGOM had overall harvestable biomass in 2009 and 2012 of 115.40 MT (90% confidence interval from 66.05 to

173.31) and 164.19 MT (74.35 to 278.91), respectively. The 2009 estimate was revised slightly since 2010 because of the new meat weight estimates for Platts Bank, a slightly different approach to the bootstrapping (previously a bootstrapping with replacement method was used along with bias corrected confidence intervals; see Truesdell et al. 2010), and the correction of an error that was found in the standardization of length frequencies. The original estimate given was 103 (53 to 186) MT (Truesdell et al. 2010). In addition, the assumed dredge efficiency in 2009 was 40%, but this was changed to 43.6%, which is based on a dredge efficiency study by the Maine Department of Marine Resources.

Harvestable biomass was distributed disproportionally across the areas surveyed. In the eastern half of the NGOM, the MSI area was found to have consistently high biomass (Fig. 6), though the density of biomass was lower than in some regions of the western NGOM (Fig. 7). Further west and offshore, PB was estimated to contain 5.6 harvestable MT in 2009 and 2.1 MT in 2012. However, this assumes that none of the large year class on PB is yet available to fishing since these scallops are under the assumed harvestable size of 102 mm used in this study (Fig. 4C). Given the increased activity on Platts Bank evident in VMS data however, it is likely that some fishermen are targeting this year class though its biomass is not included in the calculations presented here. Still further west in the two strata where most of the fishing currently occurs, NCA and NSB, the mean biomass available for harvest was 17.0 and 43.55 MT in 2009 and 55.6 and 67.2 MT in 2012. Despite their relatively small areas (Fig. 1), the high expected density strata within these regions supported considerable biomass of harvestable scallops in both survey years relative to the other areas surveyed (Fig. 6).

These biomass estimates are dependent on some fixed parameters. Survey dredge efficiency was assumed to be 43.6%, which was determined experimentally in Maine waters. No uncertainty is attached to this estimate however. Gadamke et al. (2004) estimated the efficiency of a dredge with 89 mm rings to be 42.7%, with a potential range based on sensitivity analyses from 35.5 to 52.5%. The gear was different (this study used 51 mm rings), but the mean estimate was similar to the Maine study. The approximate sensitivity range from Gedamke et al.'s study was used as a sensitivity range for the 2012 biomass estimates presented here. If dredge efficiency is assumed to be 35% the 2012 estimate is 207.51 MT (with a 90% confidence interval ranging from 93.35 to 353.29; Table 3). If dredge efficiency is 50% the estimate is 143.14 MT (65.00 to 242.88). No uncertainty was considered for the shell height to meat weight relationships or the length frequency distributions. These sources of uncertainty should be considered in subsequent analyses, though they are probably better estimated than sampling variability which is likely the main source of uncertainty and was quantified by bootstrapping.

Landings were low from 2008 to 2012, though increased notably in 2013 (Fig. 8). To determine the source of this change it would be necessary to examine vessel trip report data; however that information is not currently available to the authors. One possible reason for the higher landings, is the increased fishing effort on PB as the year class first observed in 2009 may have become targeted by the fishery.

The estimated exploitation rate during 2012 was 2.14% (90% confidence interval from 1.26% to 4.72%; Table 3), which is lower than the 6.1% (4.1% to 10.7%) estimated during 2009. The reduced exploitation rate was a function of both a decrease in landings (Figure 8) and the increase in estimated biomass from 115.40 MT in 2009 to 164.16 MT in 2012.

Characterization of scallops in the Gulf of Maine

The Maine Department of Marine Resources mid-1970s survey report (Spencer 1974) noted that most scallops encountered were older and there was no evidence of recent recruitment, leading Spencer to conclude that "only in widely separated years do scallops set in these offshore waters." The report stated that only near-shore fishing was tenable at present, though it was noted that in the 1960s the beds around Jeffereys Ledge were commercially viable. In the early 1980s scallop sets were recorded in the GOM: Serchuk (1984) and Serchuk and Wigley (1984) reported large quantities of small scallops offshore on Fippennies Ledge and Jeffereys Ledge. High densities of commercial size scallops, however, were not found in either of these surveys (Serchuk and Wigley 1984). The Maine Department of Marine Resources/University of Maine 2009 survey identified a large set of scallops on Platts Bank. Another 2009 Gulf of Maine survey corroborated these findings and also observed small scallops on Fippennies Ledge, Jeffreys Ledge and Cashes Ledge (Stokesbury et al. 2010). No such recruitment event was seen in the Maine Department of Marine Resources/University of Maine 2012 survey however.

While the fishery-independent data are not extensive for this region, it is clear that scallop sets in the NGOM are intermittent. In most years recruitment is limited or non-existent, but occasionally large recruitment events do occur. This is supported by the history of the commercial fishery in the region, which is highly variable (Dow 1971; Kelly 2012). The exception may be the western NGOM, in particular the NSB and NCA areas. It is evident from the length frequency distributions (Figs. 4A-B) that recruitment is more stable in this region than to the east where not all size classes are evident. This discrepancy may indicate environmental differences between the eastern and western NGOM, in particular how local oceanography interacts with the early life history of scallops.

Conclusions

Scallops in the NGOM represent a small but locally important fishery. Landings have been low since the inception of the NGOM management area, though they more than doubled in 2013. The best estimates from 2009 and 2012 indicate that scallop biomass increased by about 40% over that period. The exploitation rate in weight (landings/stock harvestable biomass) during 2009 was 6.1% with a 90% confidence interval from 4.1% to 10.7%, and during 2012 was 2.1% (1.3% to 4.7%). Given the region's low biomass relative to the rest of the stock along with its intermittent recruitment in eastern areas, it is probably not necessary to survey the NGOM every year. However, periodic surveys that provide point biomass estimates are likely to be helpful to managers for determining a TAC.

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Table 1: Survey coverage proportion calculated using three methods (See Eqns. 1-3). Num. stands for numerator, den. stands for denominator and prop. stands for proportion.

Type	Num. descriptor	Num. value	Den. descriptor	Den. value	Prop. inside
Area	Total survey area (km^2)	2,652	NGOM	23,470	0.11
Depth thresh.	Survey area < 100m	2,652	NGOM < 100m	7,132	0.37
VMS – low res.	VMS inside survey area	26,661	Total VMS within NGOM	27,217	0.98
VMS – high res.	VMS inside survey area	21,901	Total VMS within NGOM	23,555	0.93

Table 2: Proportion of VMS observations within the NGOM survey area. All VMS observations were included (i.e., none were excluded for confidentiality). Table provided by Burton Shank (NMFS NEFSC).

Year	Proportion inside
2006	0.94
2007	0.94
2008	0.55
2009	0.84
2010	0.69
2011	0.74
2012	0.81
2013	0.87
Overall	0.86

Table 3: Best estimates for 2012 NGOM harvestable biomass (HB) and corresponding exploitation rates (ER) under three assumptions of dredge efficiency.

Assumed Dredge	5^{th} percentile		Mean		95^{th} percentile	
Efficiency	HB	ER %	HB	ER %	HB	ER %
35%	93.35	3.76	207.51	1.69	353.29	0.99
43.6%	74.35	4.72	164.19	2.14	278.91	1.26
50%	65.00	5.40	143.14	2.45	242.88	1.44

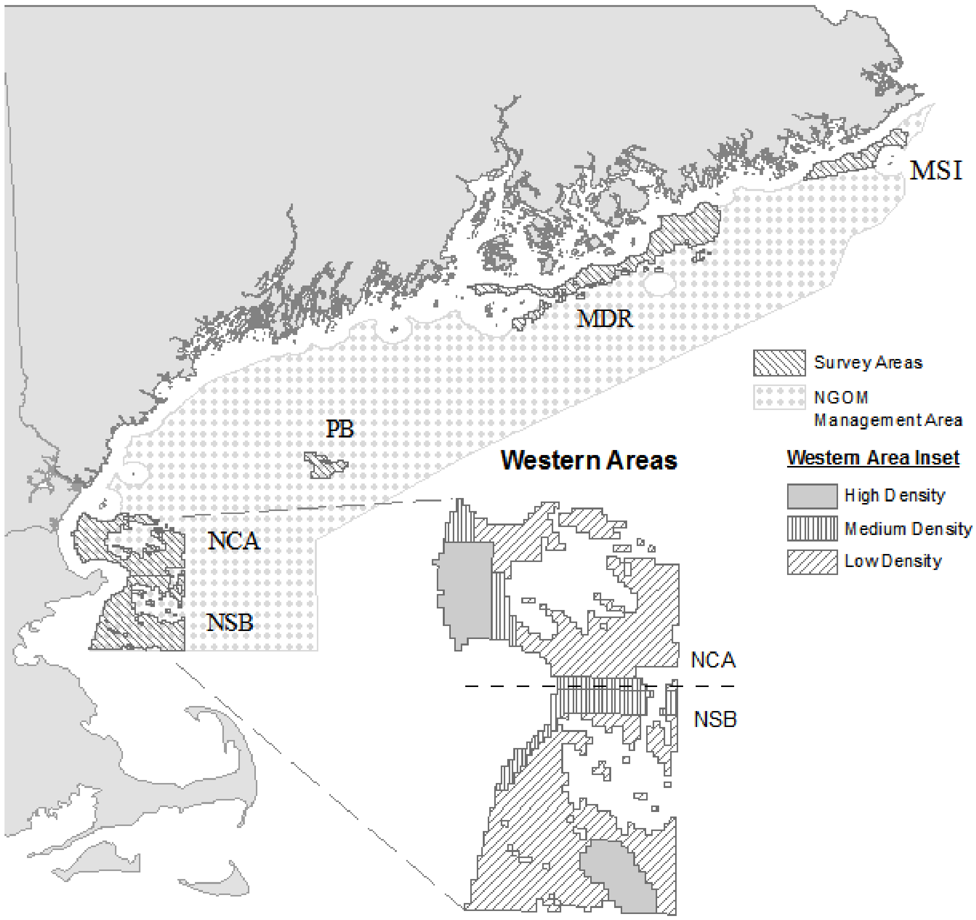

Figure 1: The NGOM and the 5 strata selected for the survey, with substrata of differing expected scallop density appearing in the western areas inset. MSI: Machias-Seal Island; MDR: Mount Desert Rock; PB: Platts Bank; NCA: Northeast of Cape Ann; NSB: Northern Stellwagen Bank.

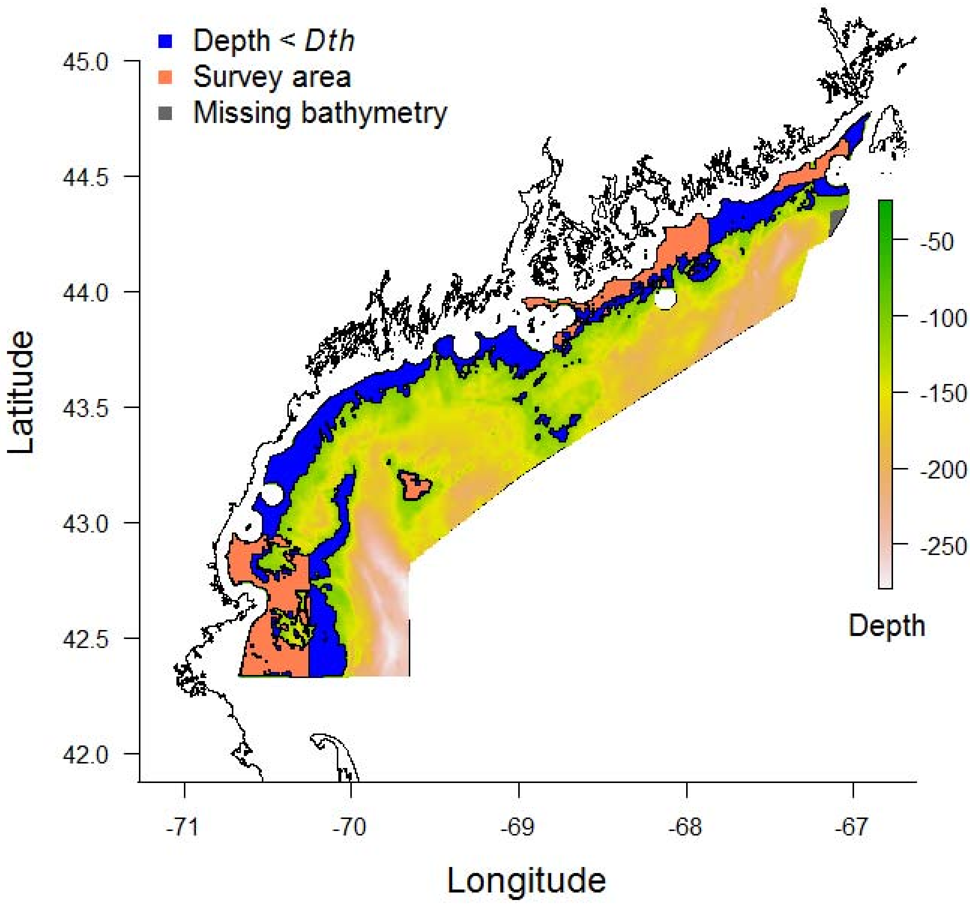

Figure 2: Survey area (pink) relative to the NGOM shallower than 100 m (*Dth*; blue). The survey area accounts for 37% of the NGOM shallower than 100 m.

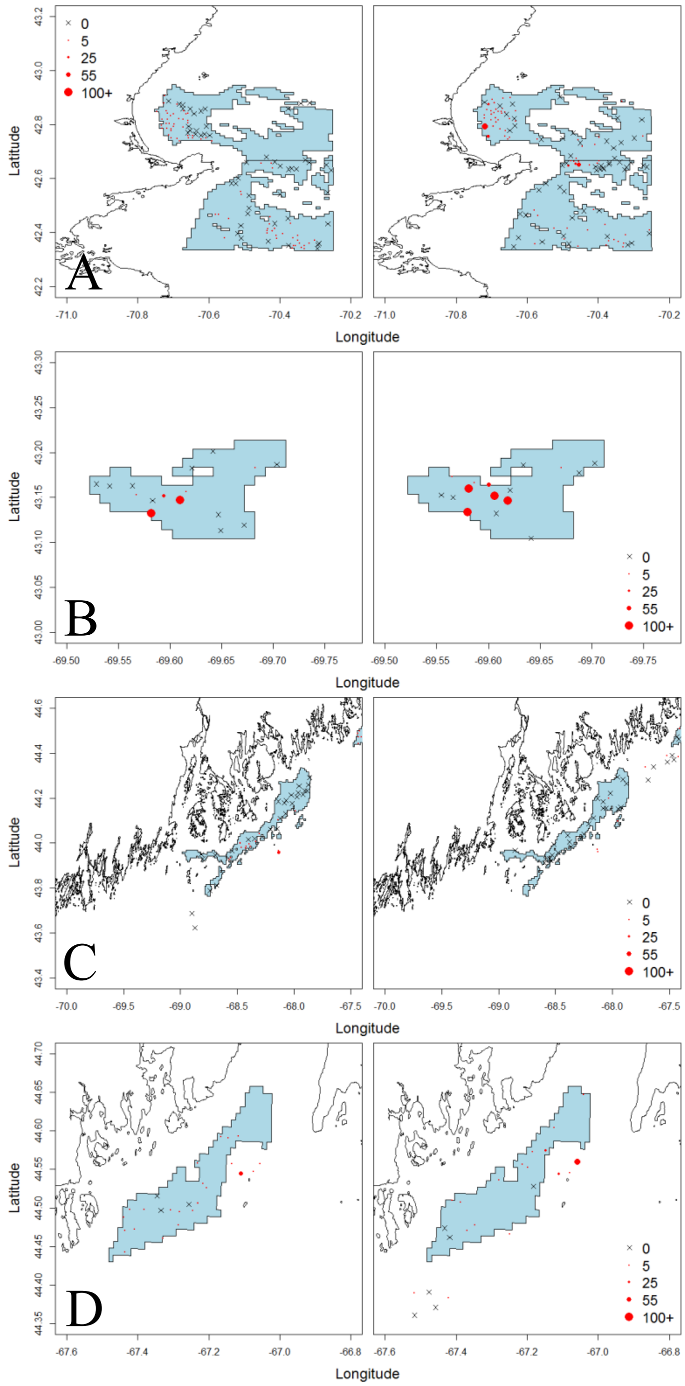

Figure 3: Distribution of survey scallop catch (all sizes) in 2009 (left panels) and 2012 (right panels). A: NCA and NSB; B: PB; C: MDR; and D: MSI.

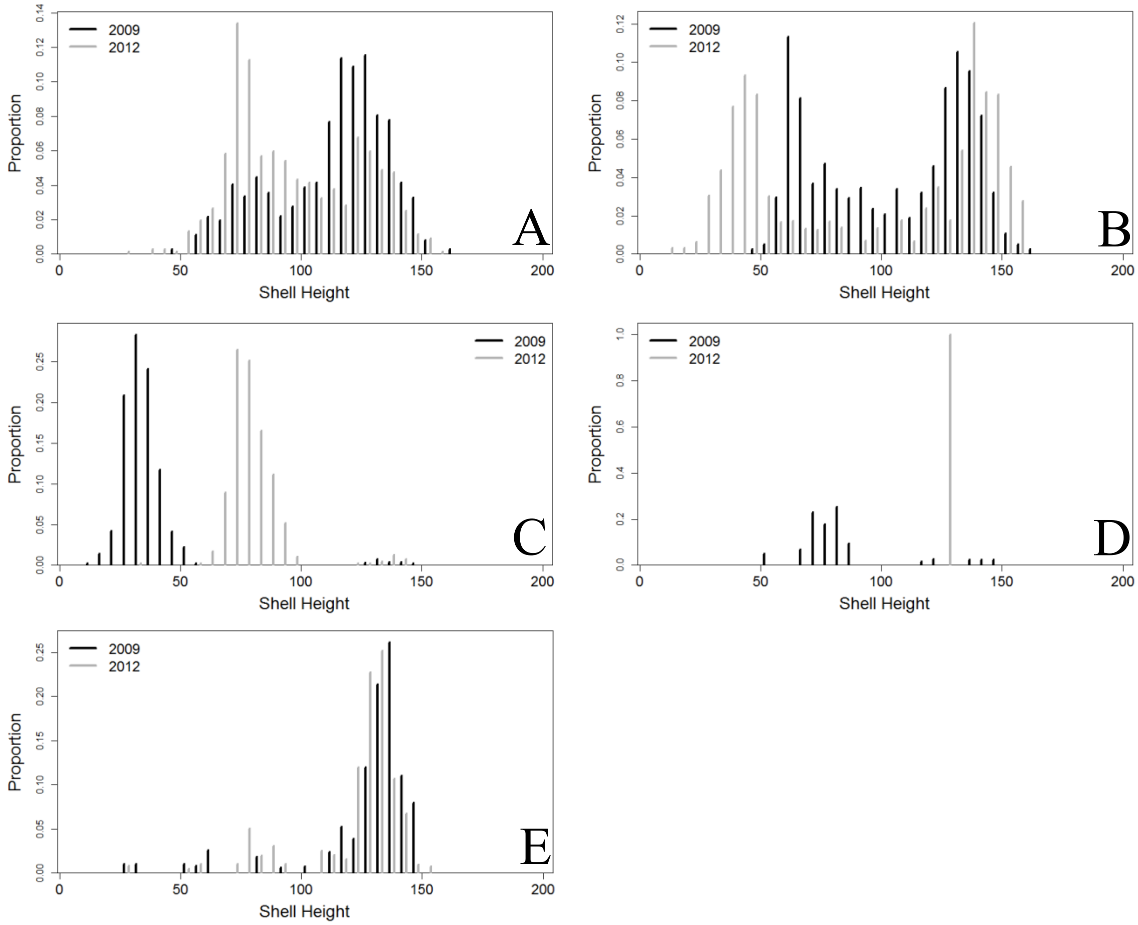

Figure 4: Shell height distribution in mm for each of the areas in 2009 and 2012. A: NCA; B: NSB; C: PB; D: MDR; E: MSI.

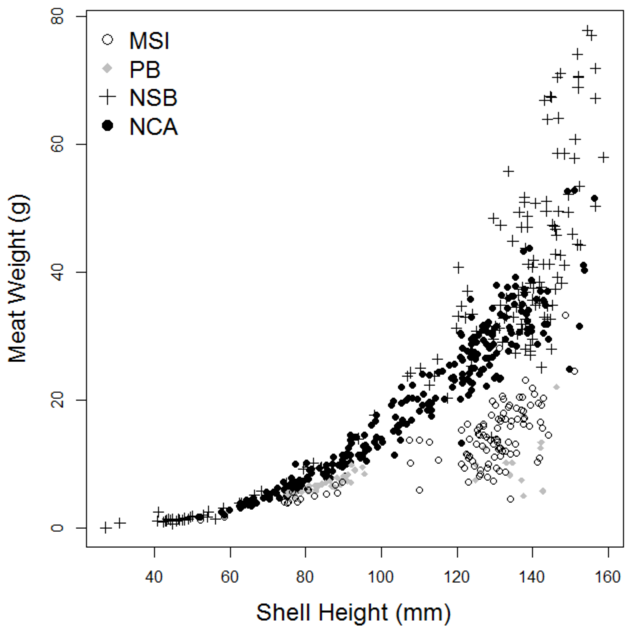

Figure 5: Relationship between shell height and meat weight in 2012 for the survey areas (excluding MDR).

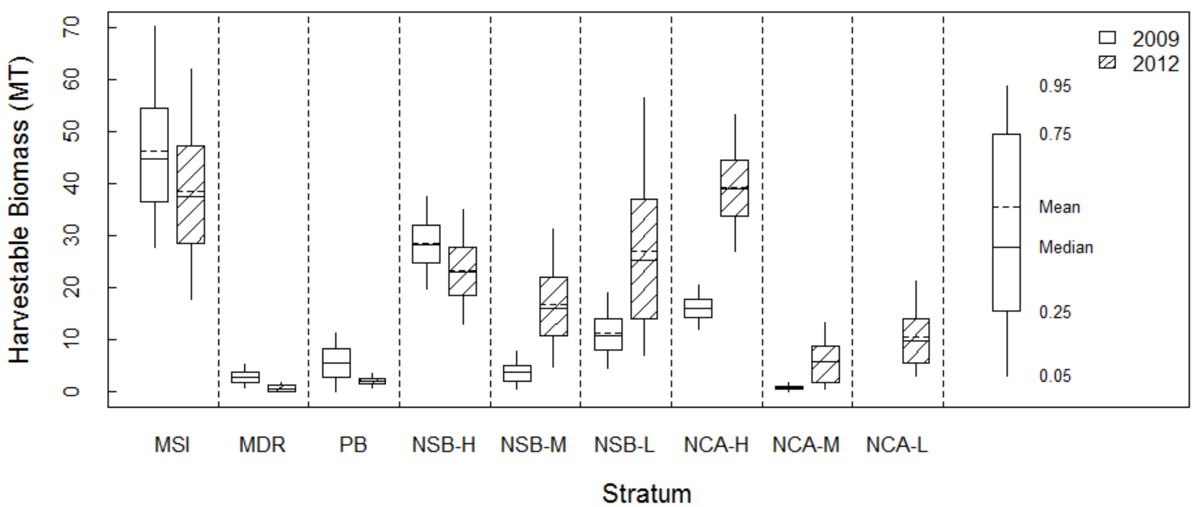

Figure 6: 2009 and 2012 harvestable biomass in NGOM survey strata (and substrata in the western region). H, M and L indicate expected high, medium and low density substrata.

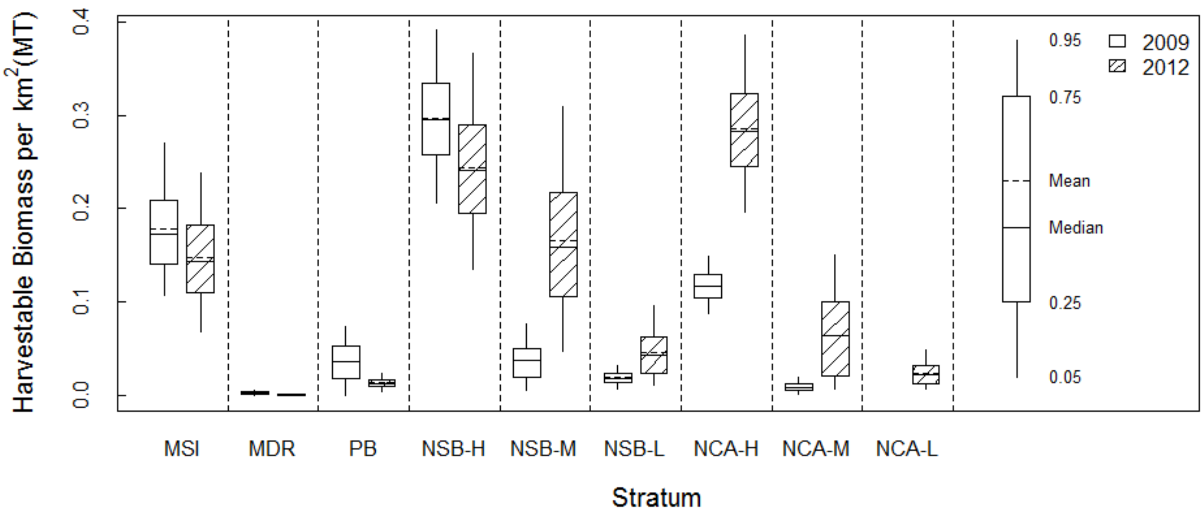

Figure 7: 2009 and 2012 harvestable density (in biomass per km^2) in NGOM survey strata (and substrata in the western region). H, M and L indicate expected high, medium and low density substrata.

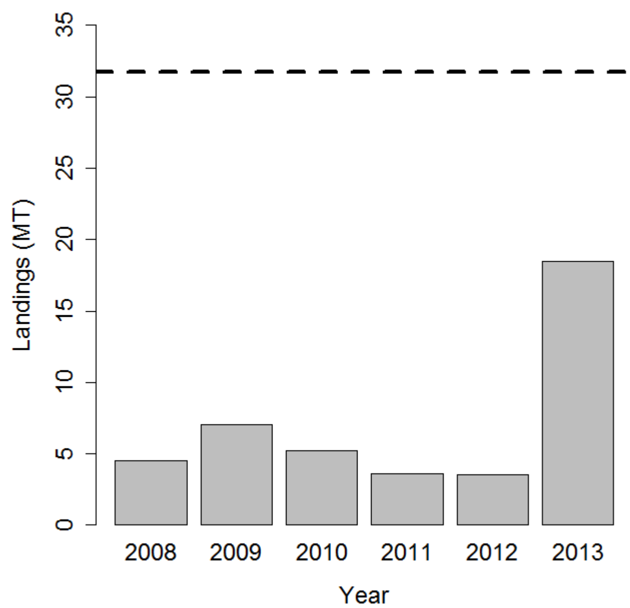

Figure 8: Landings history for the NGOM management area since its inception in 2008. Dashed line is the 31.75 MT quota.

Appendix B8. Relationships between chlorophyll and scallop recruitment potentially useful for stock projections and assessment modeling

Kevin Friedland (NEFSC, Narragansett, RI), Deborah Hart and Burton Shank (NEFSC, Woods Hole, MA)

Summary

Preliminary analyses of remote sensing and scallop dredge data suggest that recruitment to the yearling stage is influenced by summer phytoplankton bloom activity. Blooms in areas likely to influence Middle Atlantic spawning aggregations occur just prior to spring and summer spawning periods with larger bloom levels associated with high yearling settlement. The results of this analysis are encouraging and indicate further work developing techniques for predicting regional recruitment patterns based on chlorophyll concentrations is warranted. Such predictions are at spatial scales of interest to managers (e.g. rotational management areas) and might be used to improve management and profitability of the fishery.

Introduction

This appendix describes an analysis of spring and summer bloom activity and scallop recruitment in the Middle Atlantic Bight during 1998 to 2012. The topic is important because uncertainty about recent and near-term scallop recruitment reduces the accuracy of stock projection analyses use to set harvest levels and to open rotational fishing areas. Recruitment of scallops in the region was represented by two indices based on survey data: i) a yearling index based on the abundance of 1-year old scallops, and ii) a 2-year old index. The two indices generally agree but there are notable disagreements for some year classes, indicating potential measurement errors in the survey data and/or variable survival between age-1 and 2. For the purpose of this summary, we will concentrate on the results of modeling recruitment to the yearling stage.

There the two spawning periods for Middle Atlantic Bight scallops. Spring spawning occurs mostly during May and fall spawning occurs in September. In line with these putative spawning periods, the spring and summer bloom dynamics of the Middle Atlantic Bight were characterized using chlorophyll *a* concentrations based on remote sensing data. The distribution of blooms was evaluated over a 0.5°spatial grid. Chlorophyll *a* concentrations were based on remote-sensing measurements made with the Sea-viewing Wide Field of View (SeaWiFS) and Moderate Resolution Imaging Spectroradiometer (MODIS) sensors. The level-3 processed data, at 9 km and 8-day spatial and temporal resolutions, respectively, were obtained from the from the Ocean Color website (oceancolor.gsfc.nasa.gov). These two sensors provide an overlapping time series of chlorophyll *a* concentrations during the period 1998 to 2013. An analysis restricted to the overlapping period of data from both sensors revealed a systemic and consistent difference (relative bias) between them. We corrected for this bias with simple correction factors applied to MODIS data to approximate the mean levels of the SeaWiFS data. Chlorophyll *a* concentrations (mg m^{-3}) were calculated by taking the average of the constituent pixel elements for each spatial-temporal cell.

The sequential averaging algorithm called STARS or "sequential *t*-test analysis of regime shifts" (Rodionov, 2004, 2006) was used to find the beginning and end of blooms (change points) in the chlorophyll time series. A detected bloom could not exceed nine sample periods (approximately 2.4 months) based on analyses of climatological bloom patterns. Periods bracketed by positive and negative change points exceeding nine 8-day periods were considered to be ecologically different from discrete blooms. This method has been used in previous analyses of Northeast Shelf bloom patterns (Friedland et al., 2008, 2009) and elsewhere (Friedland and Todd, 2012).

We extracted statistics to characterize timing and magnitude of each bloom. Bloom start was defined as the day of initiation, which was the first day of the 8-day bloom period that exhibited bloom conditions. Bloom magnitude was the integral of the chlorophyll concentrations during the bloom period. In some years and locations, no distinct bloom period was detected by the STARS algorithm; when this occurred, bloom magnitude was taken as the integral of chlorophyll concentrations during the climatological (long-term average) bloom period based on average start and end dates for years with blooms.

Results

Yearling scallop recruitment appears to be related to spring and summer phytoplankton blooms in the Middle Atlantic Bight. The area of highest correlation between spring chlorophyll concentrations and yearling recruitment was on the continental shelf off Long Island (Fig. 1a). In contrast, the area of the greatest correlative density between summer chlorophyll concentrations and yearling recruitment was off the New Jersey coast (Fig. 1b). Mean seasonal surface currents suggest that these blooms contributed to both water column chlorophyll and depositional particulate organic carbon in the areas of spawning scallops. These observations are consistent with the hypotheses that blooms either stimulate scallop spawning or support larval survival. Recruitment to age two was not related to the same spring and summer bloom patterns as yearling scallops due primarily to the change in population size of the 2011 year class between year-1 and 2.

Future research

Refine models that predict scallop recruitment based on chlorophyll and predator data to improve estimates from stock assessment and projection models. Investigate statistical approaches to refine yearling recruitment indices. Develop complimentary models of bloom driven settlement and spatio-temporal predation pressure to ultimately stimulate recruitment of scallops to the fishery.

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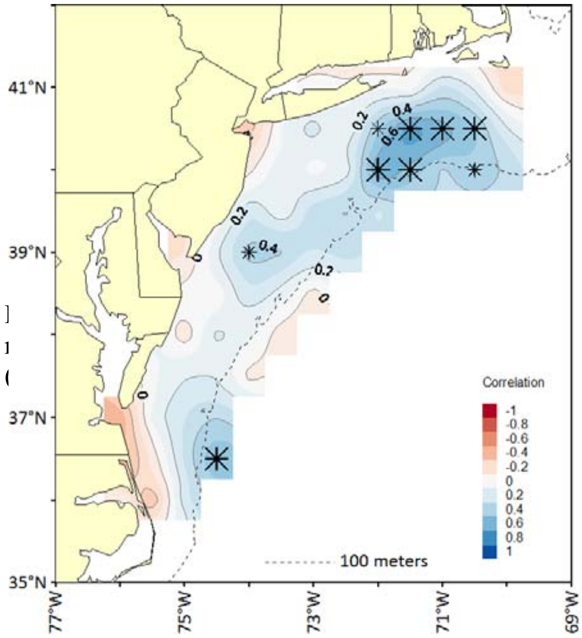
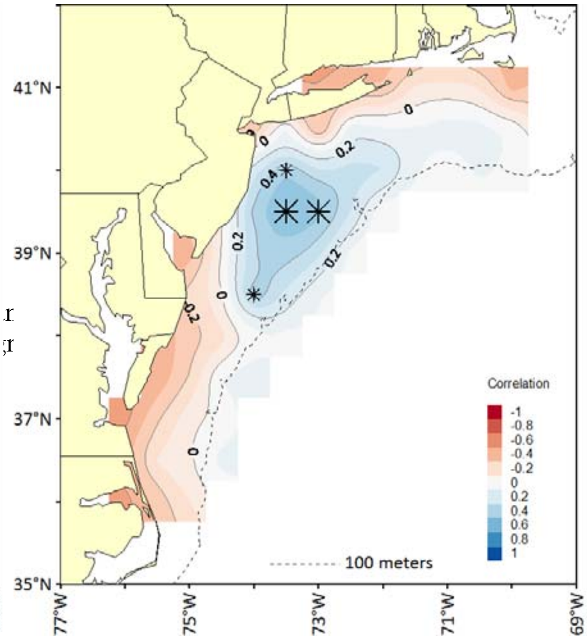
a)


41°N
39°N
37°N
35°N
77°W
75°W
73°W
71°W
69°W
Correlation
-1
-0.8
-0.6
-0.4
-0.2
0
0.2
0.4
0.6
0.8
1
100 meters

b)


41°N
39°N
37°N
35°N
77°W
75°W
73°W
71°W
69°W
Correlation
-1
-0.8
-0.6
-0.4
-0.2
0
0.2
0.4
0.6
0.8
1
100 meters

Appendix B9. Technical documentation for the CASA length structured stock assessment model used in the SARC-59 sea scallop stock assessment.

Larry Jacobson, Northeast Fisheries Science Center, Woods Hole, MA.

[*This technical description is current through CASA version nc246.*]

The stock assessment model described here is based on Sullivan et al.'s (1990) CASA model.5 CASA is entirely length-based with population dynamic calculations in terms of the number of individuals in each length group during each year. Age is almost completely irrelevant in model calculations. Unlike many other length-based stock assessment approaches, CASA is a dynamic, non-equilibrium model based on a forward simulation approach. CASA incorporates a very wide range of data with parameter estimation based on maximum likelihood. CASA can incorporate prior information about parameters such as survey catchability and natural mortality in a quasi-Bayesian fashion and MCMC evaluations are practical. The implementation described here was programmed in AD-Model Builder (Otter Research Ltd.).6

Population dynamics

Time steps in the model are years, which are also used to tabulate catch and other data. Recruitment occurs at the beginning of each time step. All instantaneous rates in model calculations are annual (y^{-1}). The number of years in the model n_y is flexible and can be changed easily (e.g. for retrospective analyses) by making a single change to the input data file. Millimeters are used to measure body size (e.g. sea scallop shell heights). Length-weight relationships should generally convert millimeters to grams. Model input data include a scalar that is used to convert the units for length-weight parameters (e.g. grams) to the units of the biomass estimates and landings data (e.g. mt). The units for catch and biomass are usually metric tons.

The definition of length groups (or length "bins") is a key element in the CASA model and length-structured stock assessment modeling in general. Length bins are identified in CASA output by their lower bound and internally by their ordinal number. Calculations requiring information about length (e.g. length-weight) use the mid-length ℓ_j of each bin. The user specifies the first length (L_{min}) and the size of length bins (L_{bin}). Based on these specifications, the model determines the number of length bins to be used in modeling as $n_L = 1 + \mathrm{int}\left[(L_\infty - L_{\min})/L_{bin}\right]$, where L_∞ is maximum asymptotic size supplied by the user, and int[x] is the integer part of x. The last length bin in the model is always a "plus-group" containing individuals L_∞ and larger. Specifications for length data used in tuning the model are separate (see below).

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5 Original programming in AD-Model Builder by G. Scott Boomer and Patrick J. Sullivan (Cornell University), who bear no responsibility for errors in the current implementation.

6 AD-Model Builder can be used to calculate variances for any estimated or calculated quantity in a stock assessment model, based on the Hessian matrix with "exact" derivatives and the delta method.

Growth

In population dynamics calculations, individuals in each size group grow (or not) at the beginning of the year, based on the annual growth transition matrix $\boldsymbol{P_0}(b,a)$ which measures the probability that a survivor in size bin a at the beginning of the previous year will grow to bin b at the beginning of the current year (columns index initial size and rows index subsequent size).[7] Growth probabilities do not include any adjustments for mortality and are applied to surviving scallops based on their original size in the preceding year.

There are two options for growth transition matrices. Under Option 1, a single annual growth matrix is calculated internally based on raw shell increment data:

$$P_0(b,a) = \frac{n(b \mid a)}{\sum_{j=a}^{n_L} n(j \mid a)}$$

where $n(b|a)$ is the number of individuals that started at size a and grew to size b after one year in the raw size increment data.

Under option 2, the user specifies the number of transition matrices to be supplied in the input file and then assigns one of the matrices to each year in the model. All such growth matrices must have the same number of length groups. The number and size groups in the model and in the growth matrices should be large enough to accommodate the largest maximum size in any year. If growth varies such that maximum size in some time period is lower the maximum value, then the growth transition probabilities for that period of maximum size are set to one along the diagonal. For example, if there were five length groups in the model: [20,25), [25,30), [30,35), [35,40) and [40,45+] mm SH and the maximum size was 34 mm SH in period one and 44 mm SH in period two, the growth transition matrices might look like:

Growth matrix for period 1

Ending size \ Starting size	[20, 25)	[25, 30)	[30, 35)	[35, 40)	[40, 45)
[20,25)	0.7	0	0	0	0
[25, 30)	0.2	0.7	0	0	0
[30, 35)	0.1	0.3	1	0	0
[35, 40)	0	0	0	1	0
[40, 45)	0	0	0	0	1

Growth matrix for period 2

Ending size \ Starting size	[20, 25)	[25, 30)	[30, 35)	[35, 40)	[40, 45)
[20,25)	0.7	0	0	0	0
[25, 30)	0.2	0.7	0	0	0
[30, 35)	0.1	0.2	0.7	0	0
[35, 40)	0	0.1	0.2	0.7	0
[40, 45)	0	0	0.1	0.3	1

Abundance, recruitment and mortality

Population abundance in each length bin during the first year of the model is:

7 For clarity in bookkeeping, mortality and annual growth calculations are always based on the size on January 1.

$$N_{1,L} = N_1 \pi_{1,L}$$

where L is the size bin, and $\pi_{1,L}$ is the initial population length composition expressed as proportions so that $\sum_{L=1}^{n_L} \pi_L = 1$. $N_1 = e^{\eta}$ is total abundance at the beginning of the first modeled year and η is an estimable parameter. It is not necessary to estimate recruitment in the first year because recruitment is implicit in the product of N_1 and π_L. The current implementation of CASA takes the initial population length composition as data supplied by the user, typically based on survey size composition data and a preliminary estimate of survey size-selectivity.

Abundance at length in years after the first is calculated:

$$\vec{N}_{y+1} = P_0\left(\vec{N}_y \circ \vec{S}_y\right) + \vec{R}_{y+1}$$

where $\vec{N}_y$ is a vector (length n_L) of abundance in each length bin during year y, $\boldsymbol{P_0}$ is the matrix (n_L x n_L) of annual growth probabilities $\boldsymbol{P_0(b,a)}$, $\vec{S}_y$ is a vector of length- specific survival fractions for year y, $\circ$ is the operator for an element-wise product , and $\vec{R}_y$ is a vector holding length-specific abundance of new recruits at the beginning of year y.

Survival fractions are:

$$S_{y,L} = e^{-Z_{y,L}} = e^{-\left(M_{y,L} + F_{y,L} + I_{y,L}\right)}$$

where $Z_{y,L}$ is the total instantaneous mortality rate and $M_{y,L}$ is the instantaneous rate for natural mortality (see below). Length-specific fishing mortality rates are $F_{y,L} = F_y s_{y,L}$ where $s_{y,L}$ is the size-specific selectivity8 for fishing in year y (scaled to a maximum of one at fully recruited size groups), F_y is the fishing mortality rate on fully selected individuals. Fully recruited fishing mortality rates are $F_y = e^{\phi + \delta_y}$ where ϕ is an estimable parameter for the log of the geometric mean of fishing mortality in all years, and δ_y is an estimable "dev" parameter.9 The instantaneous rate for "incidental" mortality ($I_{y,L}$) accounts for mortality due to contact with the fishing gear that does not result in any catch on deck (see below).10 The degree of variability in dev parameters for fishing mortality, natural mortality and for other variables can be controlled by specifying variances or likelihood weights $\neq 1$, as described below.

Natural mortality rates are calculated:

$$M_{y,L} = u_L e^{\zeta + \xi_y} + p_L \psi_y g$$

where $\vec{u}$ holds length-specific adjustments to the natural mortality rate for each length group (input by the user and assumed constant over time), ζ is an estimable parameter measuring the mean log natural mortality rate during all years and ξ_y is an estimable year-specific dev parameter. The r.h.s. deals with density-dependent natural mortality which may be important in the population dynamics of small scallops after large recruitment events. In particular, p_L is a

8 In this context, "selectivity" describes the combined effects of all factors that affect length composition of catch or landings. These factors include gear selectivity, spatial overlap of the fishery and population, size-specific targeting, size-specific discard, etc.

9 Dev parameters are a special data type for estimable parameters in AD-Model Builder. Each set of dev parameters (e.g. for all recruitments in the model) is constrained to sum to zero. Because of the constraint, the sums $\phi + \delta_y$ involving n_y+1 terms amount to only n_y parameters.

10 . See the section on per recruit modeling below for formulas used to relate catch, landings and indicental mortality.

descending logistic function based on size (larger size groups experience less density dependent mortality), ψ_y is abundance of sea scallops used to calculate density dependent natural mortality, $g=e^{□}$ is a multiplier that converts from units of abundance to units of instantaneous mortality, and □ is a an estimable scaling parameter. The logisitic function is used to calculate the abundance that controls maximum density dependent mortality while reduceing the importance of large individuals:

$$\psi_y = \sum_l p_L N_{y,L}$$

Where $N_{y,L}$ is on January 1.

The logistic function in density dependent mortality calculations is calculated:

$$p_L = 1 - \frac{1}{1 + e^{-b(L-a)}}$$

were b is the slope parameter and a is the L_{50} parameter. The logistic curve is flat or decreasing with size because $b=e^{□}$ is > 0 where □ is an estimable parameter. The L_{50} parameter is parameterized so that it automatically falls between the first and last sizes in the model:

$$a = L_{min} + (L_{max} - L_{min}) * \frac{e^{\alpha}}{1 + e^{\alpha}}$$

where L_{min} is the size at the bottom of the first size bin in the population model, L_{max} is the top of the last size bin, and □ is an estimable parameter.

Incidental mortality $I_{y,L} = F_y u_L i$ is the product of fully recruited fishing mortality (F_y, a proxy for effective fishing effort, although nominal fishing effort might be a better predictor of incidental mortality), relative incidental mortality at length (u_L) and a scaling parameter i, both of which are supplied by the user and not estimable in the model. Incidental mortality at length is supplied by the user as a vector ($\vec{u}$) containing a value for each length group in the model. The model rescales the relative mortality vector so that the mean of the series is one.

Given abundance in each length group, natural mortality, and fishing mortality, predicted fishery catch-at-length in numbers is:

$$C_{y,L} = \frac{F_{y,L}\left(1 - e^{-Z_{y,L}}\right)N_{L,y}}{Z_{y,L}}$$

Total catch number during each year is $C_y = \sum_{j=1}^{n_L} C_{y,L}$. Catch data (in weight, numbers or as length composition data) are understood to include landings (L_y) and discards (d_y) but to exclude losses to incidental mortality (i.e. $C_y=L_y+d_y$).

Discard data are supplied by the user in the form of discarded biomass in each year or a discard rate for each year (or a combination of biomass levels and rates). In the current model, discards have the same selectivity as landed catch and size composition data for discards are not included in the input file.[11] It is important to remember that discard rates in CASA are defined the ratio of discards to landings (d/L). The user may also specify a mortal discard fraction between zero and one if some discards survive. If the discard fraction is less than one, then the discarded biomass and discard rates in the model are reduced correspondingly. See the section on per recruit modeling below for formulas used to relate catch, landings and incidental mortality.

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11 The model will be modified in future to model discards and landing separately, and to use size composition data for discards.

Recruitment (the sum of new recruits in all length bins) at the beginning of each year after the first is calculated:

$$Ry = e^{\rho+\gamma_y}$$

where ρ is an estimable parameter that measures the geometric mean recruitment and the γ_y are estimable dev parameters that measure inter-annual variability in recruitment. As with natural mortality devs, the user specified variance or likelihood weight ≠ 1 can be used to help estimate recruitment deviations (see below).

Proportions of recruits in each length group are calculated based on a beta distribution *B(w,r)* over the first n_r length bins that is constrained to be concave down.12 Proportions of new recruits in each size group are the same from year to year. Beta distribution coefficients must be larger than one for the shape of the distribution to be unimodal. Therefore, $w=1+e^{\omega}$ and $r=1+e^{\rho}$, where ω and ρ are estimable parameters. It is presumably better to calculate the parameters in this manner than as bounded parameters because there is likely to be less distortion of the Hessian for w and r values close to one and parameter estimation is likely to be more efficient.

Surplus production during each year of the model can be computed approximately from biomass and catch estimates (Jacobson et al., 2002):

$$P_t = B_{t+1} - B_t + C_t$$

In future versions of the CASA model, surplus production will be more calculated more accurately by projecting the population at the beginning of the year forward one year assuming only natural mortality.

Weight at length13

The assumed body weight for size bins except the last is calculated using user-specified length-weight parameters and the middle of the size group. Different length-weight parameters are used for the population and for the commercial fishery. Mean body weight in the last size bin is read from the input file and can vary from year to year. Typically, mean weight in the last size bin for the population would be computed based on survey length composition data for large individuals and the population length –weight relationship. Mean weight in the last size bin for the fishery would be computed in the same manner based on fishery size composition data.

In principle, these calculations could be carried out in the model itself because all of the required information is available. In practice, it seems better to do the calculations externally and supply them to the model as inputs because of decisions that typically have to be made about smoothing the estimates and years with missing data.

Population summary variables

Total abundance at the beginning of the year is the sum of abundance at length $N_{y,L}$ at the beginning of the year. Average annual abundance for a particular length group is:

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12 Standard beta distributions used to describe recruit size distributions and in priors are often constrained to be unimodal in the CASA model. Beta distributions *B(w,r)* with mean $\mu = w/w+r$ and variance $\sigma^2 = wr/\left[(w+r)^2(w+r+1)\right]$ are unimodal when w > 1 and r >1. See http://en.wikipedia.org/wiki/Beta_distribution for more information.

13 Model input data include a scalar that is used to convert the units for length-weight parameters (e.g. grams) to the units of the biomass estimates and landings data (e.g. mt).

$$\overline{N}_{y,L} = N_{y,L} \frac{1-e^{-Z_{y,L}}}{Z_{y,L}}$$

The current implementation of the assessment model assumes different weight-at-length relationships for the stock and the fishery. Average stock biomass is computed using the population weight at length information.

Total stock biomass is:

$$B_y = \sum_{L=1}^{n_L} N_{y,L} w_L$$

where w_L is weight at length for the population on January 1. Total catch weight is:

$$W_y = \sum_{L=1}^{n_L} C_{y,L} w'_L$$

where w'_L is weight at length in the fishery.

F_y estimates for two years are comparable only when the fishery selectivity in the model was the same in both years. A simpler exploitation index is calculated for use when fishery selectivity changes over time:

$$U_y = \frac{C_y}{\sum_{j=x}^{n_L} N_{y,L}}$$

where x is a user-specified length bin (usually at or below the first bin that is fully selected during all fishery selectivity periods). U_y exploitation indices from years with different selectivity patterns may be relatively comparable if x is chosen carefully.

Spawner abundance in each year is (T_y) is computed:

$$T_y = \sum_{L=1}^{n_L} N_{y,L} e^{-\tau Z_y} g_L$$

Where $0 \le \tau \le 1$ is the fraction of the year elapsed before spawning occurs (supplied by the user). Maturity at length (g_L) is from an ascending logistic curve:

$$g_L = \frac{1}{1+e^{a-bL}}$$

with parameters a and b supplied by the user. Spawner biomass is computed using the population length-weight vaoues.

Egg production (S_y) in each year is computed:

$$S_y = \sum_{L=1}^{n_L} N_{y,L} e^{-\tau Z_y} g_L x_L$$

where:

$$x_L = cL^v$$

Where the fecundity parameters (c and v) for fecundity are supplied by the user. Fecundity parameters per se include no adjustments for maturity or survival. They should represent reproductive output for a spawner of given size.

Fishery and survey selectivity

The current implementation of CASA includes six options for calculating fishery and survey selectivity patterns. Fishery selectivity may differ among "fishery periods" defined by the user. Selectivity patterns that depend on length are calculated using lengths at the mid-point of each bin (ℓ). After initial calculations (described below), selectivity curves are rescaled to a maximum value of one.

Option 1 is a flat with s_L=1 for all length bins. Option 2 is an ascending logistic curve:

$$s_{y,\ell} = \frac{1}{1+e^{A_Y - B_Y \ell}}$$

Option 3 is an ascending logistic curve with a minimum asymptotic minimum size for small size bins on the left.

$$s_{y,\ell} = \left(\frac{1}{1+e^{A_Y - B_Y \ell}}\right)\left(1 - D_y\right) + D_y$$

Option 4 is a descending logistic curve:

$$s_{y,\ell} = 1 - \frac{1}{1+e^{A_Y - B_Y \ell}}$$

Option 5 is a descending logistic curve with a minimum asymptotic minimum size for large size bins on the right:

$$s_{y,\ell} = \left(1 - \frac{1}{1+e^{A_Y - B_Y \ell}}\right)\left(1 - D_y\right) + D_y$$

Option 6 is a double logistic curve used to represent "domed-shape" selectivity patterns with highest selectivity on intermediate size groups:

$$s_{y,\ell} = \left(\frac{1}{1+e^{A_Y - B_Y \ell}}\right)\left(1 - \frac{1}{1+e^{D_Y - G_Y \ell}}\right)$$

The coefficients for selectivity curves A_Y, B_Y, D_Y and G_Y carry subscripts for time because they may vary between fishery selectivity periods defined by the user. All options are parameterized so that the coefficients A_Y, B_Y, D_Y and G_Y are positive. Under options 3 and 5, D_y is a proportion that must lie between 0 and 1.

Depending on the option, estimable selectivity parameters may include α, β, δ and γ. For options 2, 4 and 6, $A_Y = e^{\alpha_Y}$, $B_Y = e^{\beta_Y}$, $D_Y = e^{\delta_Y}$ and $G_Y = e^{\gamma_Y}$. Options 3 and 5 use the same conventions for A_Y and B_Y, however, the coefficient D_Y is a proportion estimated as a logit-transformed parameter (i.e. $\delta_Y = \ln[D_Y/(1-D_y)]$) so that:

$$D_Y = \frac{e^{\delta_Y}}{1+e^{\delta_Y}}$$

The user can choose, independently of all other parameters, to either estimate each fishery selectivity parameter or to keep it at its initial value. Under Option 2, for example, the user can estimate the intercept α_Y, while keep the slope β_Y at its initial value.

Per recruit recruit modeling

The per recruit model in CASA uses the same population model as in other model calculations under conditions identical to the last year in the model. It is a standard length-based approach except that discard and incidental mortality are accommodated in all calculations. In per recruit calculations, fishing mortality rates and associated yield estimates are understood to include landings and discard mortality, but to exclude incidental mortality. Thus, landings per recruit L are:

$$L = \frac{C}{(1+\Delta)}$$

where C is total catch (yield) per recruit and Δ is the ratio of discards D to landings in the last year of the model. Discards per recruit are calculated:

$$D = \Delta L$$

Losses due to incidental mortality (G) are calculated:

$$G = \frac{I\left(1-e^{-Z}\right)B}{Z}$$
$$= IK$$

where $I = F\,u$ is the incidental mortality rate, u is a user-specified multiplier (see above) and B is stock biomass per recruit. Note that $C=FK$ so that $K=C/F$. Then,

$$G = \frac{FuC}{F}$$
$$G = uC$$

The model will estimate a wide variety ($F_{\%SBR}$, F_{max} and $F_{0.1}$) of per recruit model reference points as parameters. For example,

$$F_{\%SBR} = e^{\theta_j}$$

where $F_{\%SBR}$ is the fishing mortality reference point that provides a user specified percentage of maximum SBR. θ_j is the model parameter for the j^{th} reference point.

A complete per recruit output table is generated in all model runs that can be used for evaluating the shape of YPR and SBR curves, including the existence of particular reference points.

Per recruit reference points are time consuming to estimate and it is usually better to estimate them after other more important population dynamics parameters are estimated. Phase of estimation can be controlled individually for %SBR, F_{MAX} and $F_{0.1}$ so that per recruit calculations can be delayed as long as possible. If the phase is set to zero or a negative integer, then the reference point will not be estimated. As described below, estimation of F_{max} always entails an additional phase of estimation. For example, if the phase specified for F_{max} is 2, then the parameter will be estimated initially in phase 2 and finalized the last phase (phase >= 3). This is done so that the estimate from phase 2 can be used as an initial value in a slightly different goodness of fit calculation during the latter phase.

Per recruit reference points should have no effect on other model estimates. Residuals (calculated – target) for %SBR, $F_{0.1}$ and F_{max} reference points should always be very close to zero. Problems may arise, however, if reference points (particularly F_{max}) fall on the upper bound for fishing mortality. In such cases, the model will warn the user and advise that the offending reference points should not be estimated. *It is good practice to run CASA with reference point calculations turned on and then off to see if biomass and fishing mortality estimates change*.

The user specifies the number of estimates required and the target %SBR level for each. For example, the target levels for four %SBR reference points might be 0.2, 0.3, 0.4 and 0.5 to estimate $F_{20\%}$, $F_{30\%}$, $F_{40\%}$ and $F_{50\%}$. The user has the option of estimating F_{max} and/or $F_{0.1}$ as model parameters also but it is not necessary to supply target values.

Tuning and goodness of fit

There are two steps in calculating the negative log likelihood (NLL) used to measure how well the model fits each type of data. The first step is to calculate the predicted values for data. The second step is to calculate the NLL of the data given the predicted value. The overall goodness of fit measure for the model is the weighted sum of NLL values for each type of data and each constraint:

$$\Lambda = \sum \lambda_j L_j$$

where λ_j is a weighting factor for data set j (usually λ_j=1, see below), and L_j is the NLL for the data set. The NLL for a particular data is itself is usually a weighted sum:

$$L_j = \sum_{i=1}^{n_j} \psi_{j,i} L_{j,i}$$

where n_j is the number of observations, $\psi_{j,i}$ is an observation-specific weight (usually $\psi_{j,i}$ =1, see below), and $L_{j,i}$ is the NLL for a single observation.

Maximum likelihood approaches reduce the need to specify *ad-h*oc weighting factors (λ and ϕ) for data sets or single observations, because weights can often be taken from the data (e.g. using CVs routinely calculated for bottom trawl survey abundance indices) or estimated internally along with other parameters. In addition, robust maximum likelihood approaches (see below) may be preferable to simply down-weighting an observation or data set. However, despite subjectivity and theoretical arguments against use of *ad-hoc* weights, it is often useful in practical work to manipulate weighting factors, if only for sensitivity analysis or to turn an observation off entirely. Observation specific weighting factors are available for most types of data in the CASA model.

Missing data

Availability of data is an important consideration in deciding how to structure a stock assessment model. The possibility of obtaining reliable estimates will depend on the availability of sufficient data. However, NLL calculations and the general structure of the CASA model are such that missing data can usually be accommodated automatically. With the exception of catch data (which must be supplied for each year, even if catch was zero), the model calculates that NLL for each datum that is available. No NLL calculations are made for data that are not available and missing data do not generally hinder model calculations.

Likelihood kernels

Log likelihood calculations in the current implementation of the CASA model use log likelihood "kernels" or "concentrated likelihoods" that omit constants. The constants can be omitted because they do not affect slope of the NLL surface, final point estimates for parameters or asymptotic variance estimates. For data with normally distributed measurement errors, 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. 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 NLL 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 estimated (implicitly or explicitly) by the model, then one of two equivalent calculations is used. Both approaches assume that all observations have the same variance and standard deviation. The first approach is used when all observations have the same weight in the NLL:

$$L = 0.5N\ln\left[\sum_{i=1}^{N}(x_i-u)^2\right]$$

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 explicitly 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, which are common in stock assessment modeling, but d_f is usually unknown.

If data x have lognormal measurement errors, then ln(x) is normal and L is calculated as above. In some cases it is necessary to correct for bias in converting arithmetic scale means to log scale means (and *vice-versa*) because $\bar{x} = e^{\bar{\chi}+\sigma^2/2}$ where χ=ln(x). It is often convenient to convert arithmetic scale CVs for lognormal variables to log scale standard deviations using $\sigma = \sqrt{\ln(1+CV^2)}$.

For data with multinomial measurement errors, the likelihood kernel is:

$$L = n\sum_{i=1}^{n} p_i \ln(\theta_i) - K$$

where n is the known or assumed number of observations (the "effective" sample size), p_i is the proportion of observations in bin i, and θ_i is the model's estimate of the probability of an observation in the bin. For surveys, θ_i is adjusted for mortality up to the date of the survey and for growth up to the mid-point of the month in which the survey occurs. For fisheries, θ_i accommodates all of the mortality during the current year and is adjusted for growth during January 1 to mid-July. The constant K is used for convenience to make L easier to interpret. It measures the lowest value of L that could be achieved if the data fit matched the model's expectations exactly:

$$K = n\sum_{i=1}^{n} p_i \ln(p_i)$$

For data x that have measurement errors with expected values of zero from a gamma distribution:

$$L = (\gamma - 1)\ln\left(x/\beta \right) - x/\beta - \ln(\beta)$$

where β>0 and γ>0 are gamma distribution parameters in the model. For data that lie between zero and one with measurement errors from a beta distribution:

$$L = (p-1)\ln(x) + (q-1)\ln(1-x)$$

where p>0 and q>0 are parameters in the model.

In CASA model calculations, distributions are usually described in terms of the mean and CV. Normal, gamma and beta distribution parameters can be calculated mean and CV by the method of moments.14 Means, CV's and distributional parameters may, depending on the situation, be estimated in the model or specified by the user.

The NLL for a datum x from gamma distribution is:

$$L = (1-k)*\ln(x) + \frac{x}{\theta} + \ln[\Gamma(k)] + k\ln(\theta)$$

where k is the shape parameter and θ is the scale parameter. The last two terms on the right are constants and can be omitted if k and θ are not estimated. Under these circumstances,

$$L = (1-k)*\ln(x) + \frac{x}{\theta}$$

Robust methods

Goodness of fit for survey data may be calculated using a "robust" maximum likelihood method instead of the standard method that assumes lognormal measurement errors. The robust method may be useful when survey data are noisy or include outliers.

Robust likelihood calculations in CASA assume that measurement errors are from a Student's t distribution with user-specified degrees of freedom d_f. Degrees of freedom are specified independently for each observation so that robust calculations can be carried out for as many (or as few) cases as required. The t distribution is similar to the normal distribution for d_f ≥30. As d_f is reduced, the tails of the t distribution become fatter so that outliers have higher probability and less effect on model estimates. If d_f=0, then measurement errors are assumed in the model to be normally distributed.

The first step in robust NLL calculations is to standardize the measurement error residual $t = (x - \bar{x})/\sigma$ based on the mean and standard deviation. Then:

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14 Parameters for standard beta distributions $B(w,r)$ with mean $\mu = w/w + r$ and variance $\sigma^2 = wr/\left[(w+r)^2(w+r+1)\right]$ are calculated from user-specified means and variances by the method of moments. In particular, $w = \mu\left[\mu(1-\mu)/\sigma^2 - 1\right]$ and $r = (1-\mu)\left[\mu(1-\mu)/\sigma^2 - 1\right]$. Not all combinations of μ and σ^2 are feasible. In general, a beta distribution exists for combinations of μ and σ^2 if $0 < \mu < 1$ and $0 < \sigma^2 < \mu(1-\mu)$. Thus, for a user-specified mean μ between zero and one, the largest feasible variance is $\sigma^2 < \mu(1-\mu)$. These conditions are used in the model to check user-specified values for μ and σ^2. See http://en.wikipedia.org/wiki/Beta_distribution for more information.

$$L = \ln\left(1 + \frac{t^2}{d_f}\right)\left(1 - \frac{1 - d_f}{2}\right) - \frac{\ln(d_f)}{2}$$

Catch weight data

Catch data (landings plus discards) are assumed to have normally distributed measurement errors with a user specified CV. The standard deviation for catch weight in a particular year is $\sigma_Y = \kappa\hat{C}_y$ where "^" indicates that the variable is a model estimate and errors in catch are assumed to be normally distributed. The standardized residual used in computing NLL for a single catch observation and in making residual plots is $r_Y = \left(C_Y - \hat{C}_Y\right)/\sigma_Y$.

Specification of landings, discards, catch

Landings, discard and catch data are in units of weight and are for a single or "composite" fishery in the current version of the CASA model. The estimated fishery selectivity is assumed to apply to the discards so that, in effect, the length composition of catch, landings and discards are the same.

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 ratio of discard and landings (a.k.a. *d/K* ratios) for each year. 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:

$$\hat{L}_t = \frac{\hat{C}_t}{1 + \Delta_t}$$

Estimated discards are:

$$\hat{D}_t = \Delta_t\hat{L}_t.$$

Note that $\hat{C}_t = \hat{L}_t + \hat{D}_t$ as would be expected.

Fishery length composition data

Data describing numbers or relative numbers of individuals at length in catch data (fishery catch-at-length) are modeled as multinomial proportions $c_{y,L}$:

$$c_{y,L} = \frac{C_{y,L}}{\sum_{j=1}^{n_L} C_{y,j}}$$

The NLL for the observed proportions in each year is computed based on the kernel for the multinomial distribution, the model's estimate of proportional catch-at-length $\left(\hat{c}_Y\right)$ and an estimate of effective sample size ${}^{C}N_{Y}$ supplied by the user. Care is required in specifying effective sample sizes, because catch-at-length data typically carry substantially less information than would be expected based on the number of individuals measured. Typical conventions make ${}^{c}N_{Y} \leq 200$ (Fournier and Archibald, 1982) or set ${}^{C}N_{Y}$ equal to the number of trips or tows sampled (Pennington et al., 2002). Effective sample sizes are sometimes chosen based on goodness of fits in preliminary model runs (Methot, 2000; Butler et al., 2003).

Standardized residuals are not used in computing NLL fishery length composition data. However, approximate standardized residuals $r_y = \left(c_{y,L} - \hat{c}_{y,L}\right) / \sigma_{y,L}$ with standard deviations $\sigma_{y,L} = \sqrt{\hat{c}_{y,L}\left(1 - \hat{c}_{y,L}\right) / {}^{c}N_{y}}$ based on the theoretical variance for proportions are computed for use in making residual plots.

Survey index data

In CASA model calculations, "survey indices" are data from any source that reflect relative proportional changes in an underlying population state variable. In the current version, surveys may measure stock abundance at a particular point in time (e.g. when a survey was carried out), stock biomass at a particular point in time, or numbers of animals that dies of natural mortality during a user-specified period. For example, the first option is useful for bottom trawl surveys that record numbers of individuals, the second option is useful for bottom trawl surveys that record total weight, and the third option is useful for survey data that track trends in numbers of animals that died due to natural mortality (e.g. survey data for sea scallop "clappers"). Survey data that measure trends in numbers dead due to natural mortality can be useful in modeling time trends in natural mortality. In principle, the model will estimate model natural mortality and other parameters so that predicted numbers dead and the index data match in either relative or absolute terms.

In the current implementation of the CASA model, survey indices are assumed to be linear indices of abundance or biomass so that changes in the index (apart from measurement error) are assumed due to proportional changes in the population. Nonlinear commercial catch rate data are handled separately (see below). Survey index and fishery length composition data are handled separately from trend data (see below). Survey data may or may not have corresponding length composition information.

In general, survey index data give one number that summarizes some aspect of the population over a wide range of length bins. Selectivity parameters measure the relative contribution of each length bin to the index. Options and procedures for estimating survey selectivity patterns are the same as for fishery selectivity patterns, but survey selectivity patterns are not allowed to change over time.

NLL calculations for survey indices use predicted values calculated:

$$\hat{I}_{k,y} = q_k A_{k,y}$$

where q_k is a scaling factor for survey index k, and $A_{k,y}$ is stock available to the survey. The scaling factor is computed using the maximum likelihood estimator:

$$q_k = e^{\frac{\sum_{i=1}^{N_k}\left[\ln\left(\frac{I_{k,i}}{A_{k,i}}\right)\Big/\sigma^2_{k,i}\right]}{\sum_{j=1}^{N_k}\left(1/\sigma^2_{k,j}\right)}}$$

where N_v and $\sigma^2_{k,j}$ is the log scale variance corresponding to the assumed CV for the survey observation.15

Available stock for surveys measuring trends in abundance or biomass is calculated:

$$A_{k,y} = \sum_{L=1}^{n_L} s_{k,L} N_{y,L} e^{-Z_{y,L}\tau_{k,y}}$$

where $s_{k,L}$ is size-specific selectivity of the survey, $\tau_{k,y}=J_{k,y}/365$, $J_{k,y}$ is the Julian date of the survey in year y, and $e^{-Z_y\tau_{k,y}}$ is a correction for mortality prior to the survey. Available biomass is calculated in the same way except that body weights w_L are included in the product on the right hand side.

Available stock for indices that track numbers dead by natural mortality is:

$$A_{k,y} = \sum_{L=1}^{n_L} s_{k,L} \widetilde{M}_{y,L} \overline{N}_{y,L}$$

where $\overline{N}_{y,L}$ is average abundance during the user-specified period of availability and $\widetilde{M}_{y,L}$ is the instantaneous rate of natural mortality for the period of availability. Average abundance during the period of availability is:

$$\overline{N}_{y,L} = \frac{\widetilde{N}_{y,L}\left(1-e^{-\widetilde{Z}_{y,L}}\right)}{\widetilde{Z}_{y,L}}$$

where $\widetilde{N}_{y,L} = N_{y,L}e^{-Z\Delta}$ is abundance at elapsed time of year $\Delta=\tau_{k,y}-\nu_k$, $\nu_k=j_k/365$, and j_k is the user-specified duration in days for the period of availability. The instantaneous rates for total $\widetilde{Z}_{y,L} = Z_{y,L}\left(\tau_{k,y}-\nu_k\right)$ and natural $\widetilde{M}_{y,L} = M_{y,L}\left(\tau_{k,y}-\nu_k\right)$ mortality are also adjusted to correspond to the period of availability. In using this approach, the user should be aware that the length based selectivity estimated by the model for the dead animal survey ($s_{k,L}$) is conditional on the assumed pattern of length-specific natural mortality ($\vec{u}$) which was specified as data in the input file.

NLL calculations for survey index data assume that log scale measurement errors are either normally distributed (default approach) or from a *t* distribution (robust estimation approach). In either case, log scale measurement errors are assumed to have mean zero and log scale standard errors either estimated internally by the model or calculated from the arithmetic CVs supplied with the survey data.

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15 Scaling factors in previous versions were calculated $q_s = e^{\varpi_s}$ where ϖ_s is an estimable and survey-specific parameter. However, prior distributions were shown to have a strong effect on the parameters such that the relationship $N=qA$ did not hold. The approach in the current model avoids this problem.

The standardized residual used in computing NLL for one survey index observation is $r_{k,y} = \ln\left(I_{k,y} / \hat{I}_{k,y}\right) / \sigma_{k,y}$ where $I_{k,y}$ is the observation. The standard deviations $\sigma_{k,y}$ will vary among surveys and years if CVs are used to specify the variance of measurement errors. Otherwise a single standard deviation is estimated internally for the survey as a whole.

Survey length composition data

Length bins for fishery and survey length composition data are flexible and the flexibility affects goodness of fit calculations in ways that may be important to consider in some applications. The user specifies the starting size (bottom of first bin) and number of bins used for each type of fishery and survey length composition. The input data for each length composition record identifies the first/last length bins to be used and whether they are plus groups that should include all smaller/larger length groups in the data and population model when calculating goodness of fit. Goodness of fit calculations are carried out over the range of lengths specified by the user. Thus length data in the input file may contain large or small size bins that are ignored in goodness of fit calculations. As described above, the starting size and bin size for the population model are specified separately. In the ideal and simplest case, the minimum size and same length bins are used for the population and for all length data. However, as described below, length specifications in data and the population model may differ.

For example, the implicit definitions of plus groups in the model and data may differ. If the first bin used for length data is a plus group, then the first bin will contain the sum of length data from the corresponding and smaller bins of the original length composition record. However, the first bin in the population model is never a plus group. Thus, predicted values for a plus group will contain the sum of the corresponding and smaller bins in the population. The observed and predicted values will not be perfectly comparable if the starting sizes for the data and population model differ. Similarly, if the last bin in the length data is a plus group, it will contain original length composition data for the corresponding and all larger bins. Predicted values for a plus group in the population will be the sum for the corresponding bin and all larger size groups in the population, implicitly including sizes > L_∞.. The two definitions of the plus group will differ and goodness of fit calculation may be impaired if the original length composition data does not include all of the large individuals in samples.

In the current version of the CASA model, the size of length composition bins must be ≥ L_{bin} in the population model (this constraint will be removed in later versions). Ideally, the size of data length bins is the same or a multiple of the size of length bins in the population. However, this is not required and the model will prorate the predicted population composition for each bin into adjacent data bins when calculating goodness of fit. With a 30-34 mm population bin and 22-31and 32-41 mm population bins, for example, the predicted proportion in the population bin would be prorated so that 2/5 was assigned to the first data bin and 3/5 was assigned to the second data bin. This proration approach is problematic when it is used to prorate the plus group in the population model into two data bins because it assumes that abundance is uniform over lengths within the population group. The distribution of lengths in a real population might be far from uniform between the assumed upper and lower bounds of the plus group.

The first bin in each length composition data record must be ≥ L_{min} which is the smallest size group in the population model. If the last data bin is a plus group, then the *lower* bound of the last data bin must be ≤ the upper bound of the last population bin. Otherwise, if the last data bin is not a plus group, the *upper* bound of the last data bin must be ≤ the upper bound of the population bin.

NLL calculations for survey length composition data are similar to calculations for fishery length composition data. <u>Surveys index data may measure trends in stock abundance or biomass but survey length composition data are always for numbers (not weight) of individuals in each length group.</u> Survey length composition data represent a sample from the true stock which is modified by survey selectivity, sampling errors and, if applicable, errors in recording length data. For example, with errors in length measurements, individuals belonging to length bin j, are mistakenly assigned to adjacent length bins j-2, j-1, j+1 or j+2 with some specified probability. Well-tested methods for dealing with errors in length data can be applied if some information about the distribution of the errors is available (e.g. Methot 2000).

Prior to any other calculations, observed survey length composition data are converted to multinomial proportions:

$$i_{k,y,L} = \frac{n_{k,y,L}}{\sum_{j=L_{k,y}^{first}}^{L_{k,y}^{last}} n_{k,y,j}}$$

where $n_{k,y,j}$ is an original datum and $i_{k,y,L}$ is the corresponding proportion. As described above, the user specifies the first $L_{k,y}^{first}$ and last $L_{k,y}^{last}$ length groups to be used in calculating goodness of fit for each length composition and specifies whether the largest and smallest groups should be treated as "plus" groups that contain all smaller or larger individuals.

Using notation for goodness of fit survey index data (see above), predicted length compositions for surveys that track abundance or biomass are calculated:

$$A_{k,y,L} = \frac{s_{k,L} N_{y,L} e^{-Z_{y,j}\tau_{k,y}}}{\sum_{L=L_{k,y}^{first}}^{L_{k,y}^{last}} s_{k,j} N_{y,j} e^{-Z_{y,j}\tau_{k,y}}}$$

Predicted length compositions for surveys that track numbers of individuals killed by natural mortality are calculated:

$$A_{k,y} = \frac{s_{k,L} \tilde{M}_{y,L} \overline{N}_{y,L}}{\sum_{L=L_{k,y}^{first}}^{L_{k,y}^{Last}} s_{k,L} \tilde{M}_{y,L} \overline{N}_{y,L}}$$

Considering the possibility of structured measurement errors, the expected length composition $\vec{A}'_{k,y}$ for survey catches is:

$$\vec{A}'_{k,y} = \vec{A}_{k,y} Ɇ_k$$

where $Ɇ_k$ is an error matrix that simulates errors in collecting length data by mapping true length bins in the model to observed length bins in the data.

The error matrix E_k has n_L rows (one for each true length bin) and n_L columns (one for each possible observed length bin). For example, row k and column j of the error matrix gives the conditional probability P(k|j) of being assigned to bin k, given that an individual actually belongs to bin j. More generally, column j gives the probabilities that an individual actually belonging to length bin j will be recorded as being in length bins j-2, j-1, j, j+1, j+2 and so on. The columns of E_k add to one to account for all possible outcomes in assigning individuals to observed length bins. E_k is the identity matrix if there are no structured measurement errors.

In CASA, the probabilities in the error matrix are computed from a normal distribution with mean zero and $CV = e^{\pi_k}$, where π_k is an estimable parameter. The normal distribution is truncated to cover a user-specified number of observed bins (e.g. 3 bins on either side of the true length bin).

The NLL for observed proportions at length in each survey and year is computed with the kernel for a multinomial distribution, the model's estimate of proportional survey catch-at-length $\left(\hat{i}_{k,y,L}\right)$ and THE effective sample size ${}^{I}N_Y$ supplied by the user. Standardized residuals for residual plots are computed as for fishery length composition data.

Effective sample size for length composition data

Effective sample sizes that are specified by the user are used in goodness of fit calculations for survey and fishery length composition data. A post-hoc estimate of effective sample size can be calculated based on goodness of fit in a model run (Methot 1989). Consider the variance of residuals for a single set of length composition data with N bins used in calculations. The variance of the sum based on the multinomial distribution is:

$$\sigma^2 = \sum_{j=1}^{N}\left[\frac{\hat{p}_j\left(1-\hat{p}_j\right)}{\varphi}\right]$$

where φ is the effective sample size for the multinomial and $\overline{p}_j$ is the predicted proportion in the j^{th} bin from the model run. Solve for φ to get:

$$\varphi = \frac{\sum_{j=1}^{N}\left[\hat{p}_j\left(1-\hat{p}_j\right)\right]}{\sigma^2}$$

The variance of the sum of residuals can also be calculated:

$$\sigma^2 = \sum_{j=1}^{N}\left(p_j - \hat{p}_j\right)^2$$

This formula is approximate because it ignores the traditional correction for bias. Substitute the third expression into the second to get:

$$\varphi = \frac{\sum_{j=1}^{N}\left[\hat{p}_j\left(1-\hat{p}_j\right)\right]}{\sum_{k=1}^{N}\left(p_j - \hat{p}_j\right)^2}$$

which can be calculated based on model outputs. The assumed and effective sample sizes will be similar in a reasonable model when the assumed sample sizes are approximately correct. Effective sample size calculations can be used iteratively to manually adjust input vales to reasonable levels (Methot 1989).

Variance constraints on dev parameters

Variability in dev parameters (e.g. for natural mortality, recruitment or fishing mortality) can be limited using variance constraints that assume the deviations are either independent or that they are autocorrelated and follow a random walk. When a variance constraint for independent deviations is activated, the model calculates the NLL for each log scale residual γ_y / σ_γ, where γ_y is a dev parameter and σ is a log-scale standard deviation. If the user supplies a positive value for the arithmetic scale CV, then the NLL is calculated assuming the variance is known. Otherwise, the user-supplied CV is ignored and the NLL is calculated with the standard deviation estimated internally. Calculations for autocorrelated deviations are the same except that the residuals are $(\gamma_y - \gamma_{y-1}) / \sigma_\gamma$ and the number of residuals is one less than the number of dev parameters.

LPUE data

Commercial landings per unit of fishing effort (LPUE) data are modeled in the current implementation of the CASA model as a linear function of average biomass available to the fishery, and as a nonlinear function of average available abundance. The nonlinear relationship with abundance is meant to reflect limitations in "shucking" capacity for sea scallops.16 Briefly, tows with large numbers of scallops require more time to sort and shuck and therefore reduce LPUE from fishing trips when abundance is high. The effect is exaggerated when the catch is composed of relatively small individuals. In other words, at any given level of stock biomass, LPUE is reduced as the number of individuals in the catch increases or, equivalently, as the mean size of individuals in the catch is reduced.

Average available abundance in LPUE calculations is:

$$^{a}\overline{N}_y = \sum_{L=1}^{n_L} s_{y,L} \overline{N}_{y,L}$$

and average available biomass is:

$$^{a}\overline{B}_y = \sum_{L=1}^{n_L} s_{y,L} w_L^f \overline{N}_{y,L}$$

where the weights at length w_L^f are for the fishery rather than the population. Predicted values for LPUE data are calculated:

$$\hat{L}_y = \frac{^{a}\overline{B}_y \eta}{\sqrt{\varphi^2 + {^{a}\overline{N}_y}^2}}$$

Measurement errors in LPUE data are assumed normally distributed with standard deviations $\sigma_y = CV_y \hat{L}_y$. Standardized residuals are $r_y = \left(L_y - \hat{L}_y\right) / \sigma_y$.

16 D. Hart, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, MA, pers. comm.

Per recruit (SBR and YPR) reference points17

The user specifies a target %SBR value for each reference point that is estimated. Goodness of fit is calculated as the sum of squared differences between the target %SBR and %SBR calculated based on the reference point parameter. Except in pathological situations, it is always possible to estimate %SBR reference point parameters so that the target and calculated %SBR levels match exactly. Reference point parameters should have no effect on other model estimates and the residual (calculated – target %SBR) should always be very close to zero.

Goodness of fit for $F_{0.1}$ estimates is calculated in a manner similar to %SBR reference points. Goodness of fit is calculated as the squared difference between the slope of the yield curve at the estimate and one-tenth of the slope at the origin. Slopes are computed numerically using central differences if possible or one-sided (right hand) differences if necessary.

F_{max} is estimated differently in preliminary and final phases. In preliminary phases, goodness of fit for F_{max} is calculated as $(1/Y)^2$, where Y is yield per recruit at the current estimate of F_{max}. In other words, yield per recruit is maximized by finding the parameter estimate that minimizes it's inverse. This preliminary approach is very robust and will find F_{max} if it exists. However, it involves a non-zero residual ($1/Y$) that interferes with calculation of variances and might affect other model estimates. In final phases, goodness of fit for F_{max} is calculated as (d^2) where d is the slope of the yield per recruit curve at F_{max}. The two approaches give the same estimates of F_{MAX} but the goodness of fit approach used in the final phases has a residual of zero (so that other model estimates are not affected) and gives more reasonable variance estimates. The latter goodness of fit calculation is not used during initial phases because the estimates of F_{MAX} tend to "drift down" the right hand side of the yield curve in the direction of decreasing slope. Thus, the goodness of fit calculation used in final phases works well only when the initial estimate of F_{MAX} is very close to the best estimate.

Per recruit reference points should have little or no effect on other model estimates. Problems may arise, however, if reference points (particularly F_{max}) fall on the upper bound for fishing mortality. In such cases, the model will warn the user and advise that the offending reference points should not be estimated. *It is good practice to run CASA with and without reference point calculations to ensure that reference points do not affect other model estimates including abundance, recruitments and fishing mortality rates.*

Growth data

Growth data in CASA consist of records giving initial length, length after one year of growth, and number of corresponding observations. Growth data may be used to help estimate growth parameters that determine the growth matrix P. The first step is to convert the data for each starting length to proportions:

$$P(b,a) = \frac{n(b,a)}{\sum_{j=n_L-b+1}^{n_L} n(j,a)}$$

where $n(b,a)$ is the number of individuals starting at size *that* grew to size b after one year. The NLL is computed assuming that observed proportions $p(a|b)$ at each starting size are a sample from a multinomial distribution with probabilities given by the corresponding column in the

17 This approach is not currently estimated because of performance problems. The user can, however, estimate per recruit reference point from a detailed table written in the main output file (nc.rep). However, variances are not available in the table.

models estimated growth matrix P. The user must specify an effective sample size ${}^{P}N_j$ based, for example, on the number of observations in each bin or the number of individuals contributing data to each bin. Observations outside bin ranges specified by the user are ignored. Standardized residuals for plotting are computed based on the variance for proportions.

Survey gear efficiency data

Survey gear efficiency for towed trawls and dredges is the probability of capture for individuals anywhere in the water column or sediments along the path swept by the trawl. Ideally, the area surveyed and the distribution of the stock coincides so that:

$$I_{k,y} = q_k B_{k,y}$$

$$q_k = \frac{a_k e_k u_k}{A}$$

$$e_k = \frac{A q_k}{a_k u_k}$$

$$K_t = \frac{A}{a_k u_k}$$

$$e_k = K_t q_t$$

Where $I_{k,y}$ is a survey observation in units equivalent to biomass (or numerical) density (e.g. kg per standard tow), $B_{k,y}$ is the biomass (or abundance) available to the survey, A is the area of the stock, a_k is the area swept during one tow, $0<e_k\leq 1$ is efficiency of the survey gear, and u_k is a constant that adjusts for different units.

Efficiency estimates from studies outside the CASA model may be used as prior information in CASA. The user supplies the mean and CV for the prior estimate of efficiency, along with estimates of A_k, a_k and u_k. At each iteration if the model, the gear efficiency implied by the current estimate of q_k is computed. The model then calculates the NLL of the implied efficiency estimate assuming it was sampled from a unimodal beta distribution with the user-specified mean and CV.

If efficiency estimates are used as prior information (if the likelihood weight $\lambda > 0$), then it is very important to make sure that units and values for the survey data (I), biomass or abundance (B), stock area (A), area per tow (a), and adjustments for units (u) are correct (see Example 1). The units for biomass are generally the same as the units for catch data. In some cases, incorrect specifications will lead to implied efficiency estimates that are ≤ 0 or ≥ 1 which have zero probability based on a standard beta distribution used in the prior. The program will terminate if $e \leq 0$. If $e \geq 1$ during an iteration, then e is set to a value slightly less than one and a penalty is added to the objective function. In some cases, incorrect specifications will generate a cryptic error that may have a substantial impact on estimates.

Implied efficiency estimates are useful as a model diagnostic even if very little prior information is available because some model fits may imply unrealistic levels of implied efficiency. The trick is to down weight the prior information (e.g. $\lambda=1e^{-6}$) so that the implied efficiency estimate has very little effect on model results as long as $0 < e < 1$. Depending on the situation, model runs with e near a bound indicate that estimates may be implausible. In addition, it may be useful to use a beta distribution for the prior that is nearly a uniform distribution by specifying a prior mean of 0.5 and variance slightly less than 1/12=0.083333.

Care should be taken in using prior information from field studies designed to estimate survey gear efficiency. Field studies usually estimate efficiency with respect to individuals on the same ground (e.g. by sampling the same grounds exhaustively or with two types of gear). It seems reasonable to use an independent efficiency estimate and the corresponding survey index to estimate abundance in the area surveyed. However, stock assessment models are usually applied to the entire stock, which is probably distributed over a larger area than the area covered by the survey. Thus the simple abundance calculation based on efficiency and the survey index will be biased low for the stock as a whole. In effect, efficiency estimates from field studies tend to be biased high as estimates of efficiency relative to the entire stock.

Maximum fishing mortality rate

Stock assessment models occasionally estimate absurdly high fishing mortality rates because abundance estimates are too small. The NLL component used to prevent this potential problem is:

$$L = \lambda \sum_{t=0}^{N} \left(d_t^2 + q^2 \right)$$

where:

$$d_t = \begin{vmatrix} Ft - \Phi \; if \; Ft > \Phi \\ 0 \; otherwise \end{vmatrix}$$

and

$$q_t = \begin{vmatrix} \ln(Ft/\Phi) \, if \; Ft > \Phi \\ 0 \; otherwise \end{vmatrix}$$

with the user-specified threshold value Φ set larger than the largest value of F_t that might possibly be expected (e.g. $\Phi=3$). The weighting factor λ is normally set to a large value (e.g. 1000).

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Appendix B10. Forecasting methodology (SAMS model)
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The model presented here is a version of the SAMS (Scallop Area Management Simulator) model used to project sea scallop abundance and landings as an aid to managers since 1999. Subareas were chosen to coincide with current management. In particular, Georges Bank was divided into four open areas (two portions of the South Channel, Northern Edge and Peak, and Southeast Part), the three access portions of the groundfish closures, and the three no access portions of these areas. The Mid-Atlantic was subdivided into seven areas: Virginia Beach, the Delmarva, Elephant Trunk Closed Area and Hudson Canyon South Rotational Areas, New York Bight, Inshore New York Bight, and Long Island.

Methods

The model tracks population vectors $\mathbf{p}(i,t) = (p_1, p_2,..., p_n)$, where $p_j(i,t)$ represents the density of scallops in the jth size class in area i at time t. The model uses a difference equation approach, where time is partitioned into discrete time steps $t_1, t_2,...,$ with a time step of length $\Delta t = t_{k+1} - t_k$. The landings vector $\mathbf{h}(i,t_k)$ represents the catch at each size class in the ith region and kth time step. It is calculated as:

$$h(i,t_k) = [I - \exp(\Delta t H(i,t_k))]p(i,t_k),$$

where I is the identity matrix and H is a diagonal matrix whose jth diagonal entry h_{jj} is given by:

$$h_{jj} = 1/(1+\exp(s_0 - s_1 * s))$$

where s is the shell height of the mid-point of the size-class.

The landings $L(i,t_k)$ for the ith region and kth time step are calculated using the dot product of landings vector $\mathbf{h}(i,t_k)$ with the vector $\mathbf{m}(i)$ representing the vector of meat weights at shell height for the ith region:

$$L(i,t_k) = A_i \, \mathbf{h}(i,t_k) \bullet \mathbf{m}(i)$$

where e_i represents the dredge efficiency in the ith region.

Even in the areas not under special area management, fishing mortalities tend not to be spatially uniform due to the sessile nature of sea scallops (Hart 2001). Fishing mortalities in open areas were determined by a simple "fleet dynamics model" that estimates fishing mortalities in open areas based on area-specific catch rates, and so that the overall DAS or open-area F matches the target. Based on these ideas, the fishing mortality F_i in the ith region is modeled as:

$$\mathbf{F_i = k * f_i * L_i}$$

where L_i is the estimated LPUE (landings per day charged) in the ith region, f_i is an area-specific adjustment factor to take into account preferences for certain fishing grounds (due to lower costs, shorter steam times, ease of fishing, habitual preferences, etc.), and k is a constant adjusted so that the total DAS or fishing mortality meets its target. For these simulations, $f_i = 1$ for all areas.

Scallops of shell height less than a minimum size s_d are assumed to be discarded, and suffer a discard mortality rate of d, taken here, as in the rest of the assessment to be 20%. There is also evidence that some scallops not actually landed may suffer mortality due to incidental damage from the dredge. Let F_L be the landed fishing mortality rate and F_I be the rate of incidental mortality on scallops not caught. For Georges Bank, which is a mix of sandy and hard bottom, we used $F_I = 0.2F_L$. For the Mid-Atlantic (almost all sand), we used $F_I = 0.1F_L$. Incidental mortality for a given shell height bin was then calculated using equations (4.3) and (4.4) of the main document.

Growth in each subarea was specified by a growth transition matrix G, based on area-specific growth increment data from 2001-2012. Recruitment was modeled stochastically, and was assumed to be log-normal in each subarea. The mean, variance and covariance of the recruitment in a subarea was set to be equal to that observed in the historical time-series between 1979-2013. New recruits enter the first size bin at each time step at a rate r_i depending on the subarea i, and stochastically on the year. These simulations assume that recruitment is a stationary process, i.e., no stock-recruitment relationship is assumed. This may underestimate recruitment in the Mid-Atlantic if the recent strong recruitment there are due to a stock-recruit relationship.

The population dynamics of the scallops in the present model can be summarized in the equation:

$$p(i,t_{k+1}) = \rho_i + G\exp(-M\Delta tH)\,p(i,t_k),$$

where ρ_i is a random variable representing recruitment in the ith area. The model was run with 10 time steps per year. The population and harvest vectors are converted into biomass by using the shell-height meat-weight relationship:

$$W = \exp[a + b\ln(s)],$$

where W is the meat weight of a scallop of shell height s. These relationships are subarea-specific; see Appendix B3 for details. For calculating biomass, the shell height of a size class was taken as its midpoint.

Commercial landing rates (LPUE, landed meat weight per day) were estimated using an empirical function based on the observed relationship between annual landing rates, expressed as number caught per day (NLPUE) and survey exploitable numbers per tow. At low biomass levels, NLPUE increases roughly linearly with survey abundance. However, at high abundance levels, the catch rate of the gear will exceed that which can be shucked by a seven-man crew. This is similar to the situation in predator/prey theory, where a predator's consumption rate is limited by the time required to handle and consume its prey (Holling 1959). The original Holling Type-II predator-prey model assumes that handling and foraging occur sequentially. It predicts that the per-capita predation rate R will be a function of prey abundance N according to a Monod functional response:

$$R = \frac{\alpha N}{\beta + N},$$

where α and β are constants. In the scallop fishery, however, some handling (shucking) can occur while foraging (fishing), though at a reduced rate because the captain and one or two crew members need to break off shucking to steer the vessel during towing and to handle the gear during haulback.

The fact that a considerable amount of handling can occur at the same time as foraging means that the functional response of a scallop vessel will saturate quicker than predicted by the above equation. To account for this, a modified Holling Type-II model was used, so that the landings (in numbers of scallops) per unit effort (DAS) L (the predation rate, i.e., NLPUE) will depend on scallop (prey) exploitable numbers N according to the formula:

$$L = \frac{\alpha N}{\sqrt{\beta^2 + N^2}}.$$

The parameters α and β to this model were fit to the observed fleet-wide LPUE vs. exploitable biomass relationship during the years 1994-2012 (previous years were not used because of the change from port interviews to logbook reporting). The number of scallops that can be shucked should be nearly independent of size provided that the scallops being shucked are smaller than about a 20 count. The time to shuck a large scallop will go up modestly with size. To model this, if the mean meat weight of the scallops caught, g, in an area is more than 20 g, the parameters α and β in the above equation are reduced by a factor $\sqrt{20/g}$. This means, for example, that a crew could shuck fewer 10 count scallops per hour than 20 count scallops in terms of numbers, but more in terms of weight.

An estimate of the fishing mortality imposed in an area by a single DAS of fishing in that area can be obtained from the formula $F_{\text{DAS}} = L_a/N_a$, where L_a is the NLPUE in that area obtained as above, and N_a is the exploitable abundance (expressed as absolute numbers of scallops) in that area. This allows for conversion between units of DAS and fishing mortality.

Initial conditions for the population vector **p** *(i,t)* were estimated using the 2013 surveys, with the overall estimates scaled to match the 2013 biomass as estimated by CASA. The 2013 initial conditions were varied depending on the survey standard errors in each subarea, and scaled so that the initial standard error in biomass was about 15,000 mt, a figure that the working group considered a fair measure of the true uncertainty in the initial estimates.

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